

Institute of Plant production and Agroecology in the Tropics and Subtropics

University of Hohenheim

Agroecology in the Tropics and Subtropics

PD.Dr. Anna Treydte

The influence of land use and cover changes on the pastoral rangeland systems of southern Ethiopia - How much woody cover is enough?

Dissertation

Submitted in fulfillment of the requirements for the degree

“Doktor der Agrarwissenschaften” (Dr.sc.agr.)

To the Faculty of Agricultural Sciences.

Presented by

Hasen Yusuf Mohammed, MSc.

Stuttgart 2013

Date of oral examination: 07.03.2014
Examination committee:
Supervisor and reviewer: Pd Dr. Anna Treydte
Co-reviewer: Jun.-Prof. Dr. Uta Dickhöfer
Additional examiner: Prof. Dr. rer. nat. Andreas Fangmeier
Vice-dean of studies and head of the committee: Prof. Dr. Dr. h.c. Rainer Mosenthin

Statement of uniqueness

Author's Declaration

I, Hasen Yusuf, hereby affirm that I have written this thesis entitled “The influence of land use and cover changes on the pastoral rangeland systems of southern Ethiopia - How much woody cover is enough?” independently as my original work as part of my dissertation at the Faculty of Agricultural Sciences at Hohenheim University.

All authors in the quoted or mentioned publications in this manuscript have been accredited. No piece of work by any person has been included without the author being cited, nor have I enlisted the assistance of commercial promotion agencies. This thesis has not been presented at any other board for examination.

Hasen Yusuf

Stuttgart 05.12.2013

Acknowledgement of the author

The research work was funded by the Dr. Hermann Eiselen Ph.D. Grant from the Foundation fiat panis and the Food Security Center of the University of Hohenheim, while the PhD stipend was covered by DAAD (German Academic Exchange Service) program.

First of all I would like to thank Professor Dr.Joachim Sauerborn for giving me an opportunity to join his working group and providing me unreserved support and cooperation through out my study period.

This thesis would not have been achieved without the expert guidance of my supervisor, PD. Dr. Anna Treydte. I want to express my deep gratitude to her for starting as my supervisor, helping in setting up the tracks of the study, continuous follow up and encouragement throughout my study period. She was not only readily available for me, but also read and responded to the drafts of each manuscripts originated from my research work more quickly than I could have imagined. Thank you especially for your time, endurance and efforts you have put into my work throughout the study period and even after leaving Hohenheim.

I want to thank Dr. Abule Ebro for providing me his expert advice and assistance during the execution and selection of research sites in Ethiopia, and reviewing and proof reading of the manuscripts produced from my research work.

I would like to express my heartfelt gratitude to all staff and colleagues in the Institute of Agroecology (380b) for all such friendly support and cooperation during my study period, especially Eva Schmidt, Michael Yongha Bo, Baset Garboni, Lenard Mounde. Samuel Addie Apukiya, Samuel Tufa, Marc Cotter, Inga Häuser, Resa Golbon, and Professor Konrad Martin, Sabine Baumgartner, Regina Belz, and Despina Savvidou, Thanks for providing me an intellectually stimulating and lovely working environment. I am especial thankful to the secretary Eva Shmidt for her extraordinary assistance in facilitating the housing condition throughout my stay in Hohenheim.

I am deeply thankful to all staff of Food Security Center (FSC) of University of Hohenheim, especially Dr. Detlef Virchow, Dr. Brigitte Kranz, Helene Stauss, Dr. Andrea Jost for all support and cooperation especially in handling all the administrative matters related to my study and offering me stimulatory excursions and training workshops throughout the study period.

I would like to thank all staff of forage gene bank team at International Livestock Research Institute (ILRI) in Addis Ababa Ethiopia. My especial thank goes to Dr. Alexandra Jorge and Yonas Asmare for collaboration to use their laboratory and assisting me during forage quality analysis.

I would like to thank the cooperation of Wollega University for allowing me to pursue my PhD study. I am especially grateful to Professor Fekadu Beyene, the president of University for his encouragement and unreserved support in various ways during my study period.

I am also thankful to staff and friends at the Yabello Dryland Agricultural Research Center (YDARC) for moral and logistics support rendered me during my field research work. Especial thanks goes to my field assistants Gebeyew Elias and Bediru Roba. Acknowledgement is also due to Samuel Tufa, the center manager for hosting me in their guest house during my field work period.

I am also particularly thankful to my German family I lived with them for two years Harald, Martina, Lisa, Kael, and Inga for their extraordinary assistance and encouragement in various ways during my course of studies.

I thank my parents for their unconditioned support, prayers, and encouragement during this long stay abroad. Last but certainly not least, a very personal and especial thanks goes to my wife, Sada Mamude for being with me and a moral support rendered me in times of doubt when I certainly need help.

Overview of publications

In order to comply with regulations for a cumulative PhD thesis at the Faculty of Agricultural Sciences, three publications have been included into this work. As these publications have been prepared to fit the regulations of the different peer-reviewed scientific journals, the style, citations and the layout of the reference section may vary between chapters.

Chapter 2

Hasen Yusuf M, Treydte, A.C., Abule, E., and Sauerborn J. (2013). Predicting aboveground biomass of encroacher woody species in semi-arid rangelands, Ethiopia. *Journal of Arid Environments* 96, 64 – 72.

Chapter 3

Hasen Yusuf, M., Treydte, A.C. E. Abule, and J. Sauerborn (in review). Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on aboveground C stock density submitted to *Journal of Carbon Balance and Management* on 19.11.2013 (publisher's acknowledgement of receipt enclosed)

Chapter 4

Hasen Yusuf, M., A.C. Treydte, and J. Sauerborn (in review). Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on soil carbon and nitrogen, submitted to *PLOS ONE* (publisher's acknowledgement of receipt enclosed)

List of abbreviations and acronyms

AGB	Aboveground Biomass
AGC	Aboveground Carbon
ANOVA	Analysis of Variance
BECVOL	Biomass Estimates From Canopy Volume
BGC	Belowground Carbon
C	Carbon
CEC	Cation Exchange Capacity
CO ₂	Carbon dioxide
CV	Coefficient of Variation
CA	Crown area
DBH	Diameter at Breast Height
DM	Dry matter
EC	Electrical Conductivity
ENVI	Environment for Visualizing Images (ENVI)
ETM+	Enhanced Thematic Mapper plus
FAO	Food and Agriculture Organization
GDP	Gross Domestic Product
GLM	General Linear Model
GPS	Geographical Positioning System
ha	Hectare
H	Height
HE	High encroachment
LE	Low encroachment
TLU	Tropical Livestock Unit
ILRI	International Livestock Research Institute
IPCC	International Panel on Climate Changes
m.a.s.l	meters above sea level
MEA	Millennium Ecosystem Assessment
ME	Moderate encroachment

MSE	Mean square of error
MSS	Multi Spectral Scanner
N	Nitrogen
NPP	Net primary productivity
OC	Organic Carbon
P	Phosphorus
pH	Power of Hydrogen ion concentration
r^2	Coefficient of determination
% RCA	Percent relative canopy cover
SAS	Statistical Analysis System
SB	Stem base
SE	Severe encroachment
SOM	Soil Organic Matter
SOC	Soil Organic Carbon
SON	Soil Organic Nitrogen
SE	Standard error
TSN	Total Soil Nitrogen

Table of Contents

Acknowledgement of the author	1
Overview of publications	3
List of abbreviations and acronyms	4
1 General Introduction.....	9
1.1 Importance of pastoral rangelands	9
1.2 Major threats to Borana rangelands	10
1.2.1 Intensification of grazing and ecosystem C and N stocks.....	11
1.2.2 Woody encroachment and rangeland productivity	13
1.3 Specific objectives	18
1.4 Outline of the thesis	18
2 Predicting aboveground biomass of encroacher woody species in semi-arid rangelands, Ethiopia..	27
2.1 Introduction.....	29
2.2. Material and methods.....	31
2.2.1. Study area.....	31
2.2.2 The species investigated.....	31
2.2.3. Samples	32
2.2.4. Statistical analyses	35
2.3. Results.....	36
2.4. Discussion	43
2.5. Conclusion	48
2.6. References.....	49
3 Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on aboveground C stock density.....	56
3.1 Introduction.....	58
3.2 Methods.....	60
3.2.1 Site description.....	60
3.2.2 Land cover change analysis	62
3.2.3 Field sampling design	66
3.2.4 Data collection	68
3.3 Results.....	70

3.3.1	Land cover changes.....	70
3.3.2	Floristic composition.....	72
3.3.3	Woody vegetation structure	73
3.3.4	Aboveground carbon stocks.....	78
3.4	Discussion	81
3.5	Conclusion	84
3.6	References.....	86
4	Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on soil carbon and nitrogen	91
4.1	Introduction.....	93
4.2	Material and Methods	95
4.2.1	Study area.....	95
4.2.2	Land use and grazing patterns in Borana	97
4.2.3	Sampling design.....	98
4.2.4	Sampling and laboratory analysis	101
4.2.5	Statistical analyses	102
4.3	Results.....	103
4.3.1	Carbon stock in the herbaceous vegetation.....	103
4.3.2	Soil organic carbon and nitrogen	104
4.3.3	Soil bulk density.....	109
4.3.4	Factors related to SOC retention	109
4.4	Discussion	110
4.4.1	Effects of grazing on SOC and TSN stocks	110
4.4.2	Effect of woody encroachment on SOC and TSN stocks	112
4.5	Conclusion	114
4.6	References.....	115
5	General Discussion.....	120
5.1	Trends of expansion of woody encroachment	120
5.2	Allometric functions	123
5.3	Woody encroachment and aboveground biomass C stock.....	124
5.4	Woody encroachment, soil organic carbon and total soil nitrogen	126
5.5	Grazing, vegetation structure and aboveground biomass carbon.....	128

5.6	Grazing, herbaceous root biomass carbon, soil organic carbon and total soil nitrogen	128
5.7	Ecosystem level carbon stocks.....	130
5.8	Limitations and conclusion of the study	131
6	Summary	133
7	Zusammenfassung	136
8	Curriculum vitae.....	139

1 General Introduction

1.1 Importance of pastoral rangelands

Drylands (arid and semiarid areas) cover 41% of Earth's land surface and hosting approximately 2.7 billion humans (Bai et al., 2008). Sixty-five percent of global drylands host rangelands, a base for livestock production that contributes to foods and livelihoods security of 800 million people (Mortimore, 2009). In Africa, drylands constitute about 43% of inhabited surface and are mainly used for pastoral and agro-pastoral activities to support 268 million people (IIED and SOS Sahel, 2010). In East Africa, grassland or savanna ecosystems cover extensive areas of the dryland surface, which account more than 60% of the total land area of the region (Neely and Bunning, 2008) and are a basis for livestock industry. In Ethiopia, pastoral and agro-pastoral areas support about 40% of the cattle, 50% of the small ruminants, and almost all camels (Hogg, 1997). Livestock production largely carried out in dry areas provides foods and incomes for an estimated 12 - 15% of the Ethiopia's pastoral and agro-pastoral population and also constitutes 20% of total growth domestic product (GDP) of Ethiopia (Aklilu, 2002).

Dry rangelands, because of their vast nature, hold huge potential to be one of the terrestrial sinks for carbon (C), globally accounting for 36% of the total C stock of terrestrial ecosystems despite its low C density compared to forest ecosystems (Lal, 2004a). Their potential as C sinks could be an important ecosystem service to mitigate global warming (FAO, 2010; Tennigkeit and Wilkes, 2008; IPCC, 2007; Neely and Bunning, 2007). Reid et al. (2005) suggested that the payment for C sequestration could diversify livelihood options and potentially increase adaptation potentials of the resource-poor pastoral people to climate change. While the potential for C sequestration in the dry rangelands was highlighted in the previous studies (e.g., Tennigkeit and Wilkes, 2008; Neely and Bunning, 2007), to date no empirical research work has been done to estimate C stocks in vegetation and soils of the southern Ethiopian rangelands. Management of rangeland aboveground vegetation (herbaceous and woody species) can have significant effect on the ecosystem C stocks. Similarly, C stocks in a soil particularly in degraded rangelands may be increased for C credit purposes through adoption of better grazing management practices that may include moderate stocking rates, sustainable grazing systems such as rotational grazing and seasonal use (Garnaut, 2008). However, the response of vegetation and soil C stock to such grazing management changes has not been investigated and it has not been quantified the way the reduction in grazing pressure could increase C stocks in the Borana rangelands.

1.2 Major threats to Borana rangelands

Historically, the Borana pastoralists in southern Ethiopia (Fig.1.1) , i.e., whose gross incomes from livestock keeping is at least 50% (Swift, 1988), specialized on extensive cattle keeping and their land use system was largely characterized by a sustainable exploitation of rangeland resources based on seasonal herd mobility in connection with flexible stocking densities (Oba and Kotile, 2001).

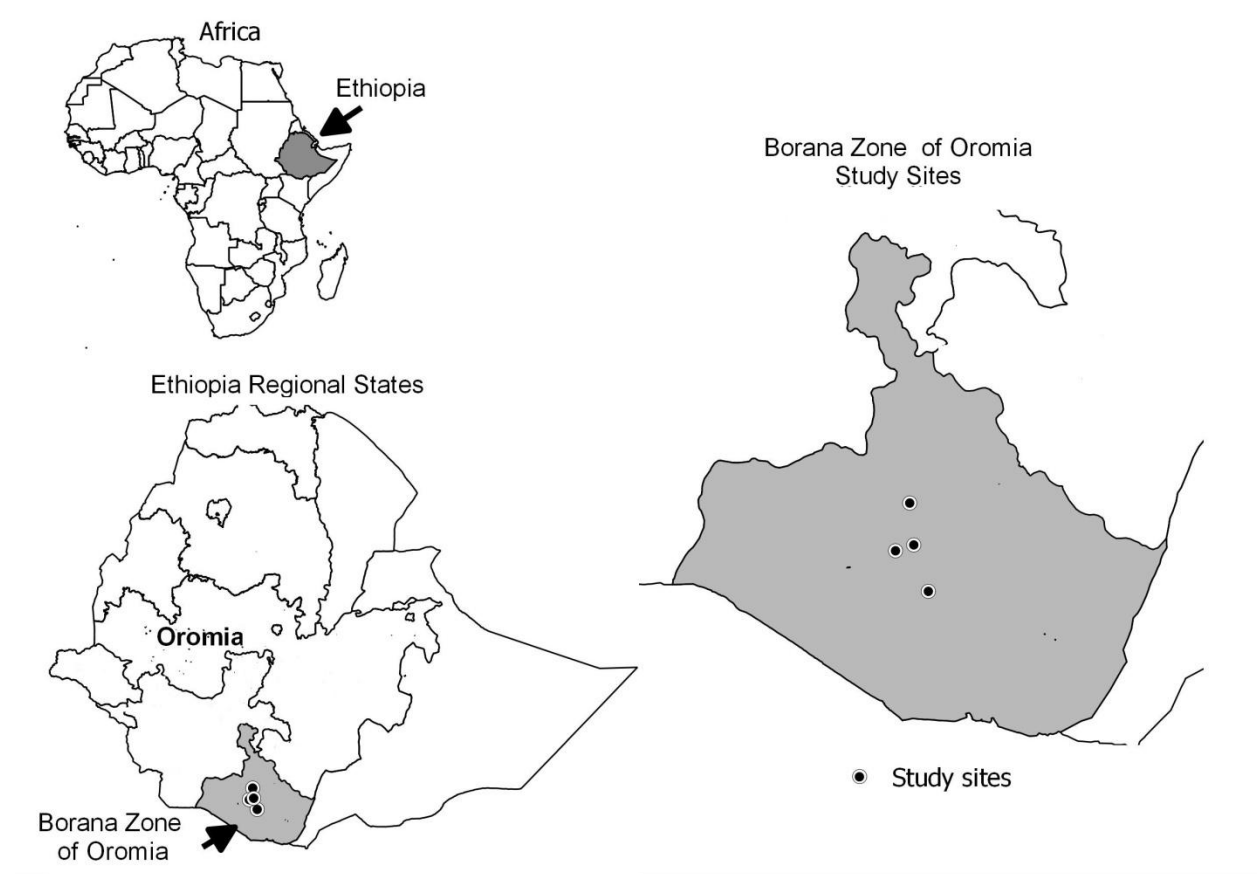


Fig.1.1. Map of study area (all country and regional boundaries shown on the Map are not considered authoritative)

Movement patterns corresponded with local rainfall and according rangelands productivity, shifting towards dry areas in the wet season and to more humid areas in dry seasons (Swift, 1995). The pastoralist's rangeland management system also involved periodic burning of the rangelands (Coppock, 1994). Amongst the East African rangelands the Borana pastoral system in southern Ethiopia was regarded to be especially productive until the 1980s (Cossins and Upton, 1987). These authors indicated that the Borana pastoral system had higher net primary productivity and returns of energy and protein per hectare compared to industrialized ranching

systems in North Australia which only realize 16% of the energy and 30% of the protein per hectare compared to the Borana system. However, increasing human and livestock populations, changes in fire regimes, expansion of crop production and changing demographics and traditional institutional conditions have forced pastoralists to intensify grazing, which have resulted in deterioration of the rangelands (Homann et al., 2008a; Homann et al., 2008b; Watson, 2003; Oba et al., 2000b; Cossins and Upton, 1987). Climate variability including an increased frequency of extreme weather events and prolonged droughts exacerbated the degradation of rangelands in dry parts of Africa (Neely and Bunning, 2008).

The term rangeland degradation/deterioration refers to both soil and vegetation and is generally defined as the reduction of the economic or biological productivity of lands (FAO, 2011). Loss of plant cover, undesirable change in herbaceous species composition (e.g. annual grasses replacing perennials), soil erosion of various types associated with intensification of grazing and woody encroachment have been dominant features in the Borana rangelands (Fig. 1.2) which could have different implications for pastoral productivity and rangelands C storage potentials (Conant and Paustian, 2002).



Fig.1.2. Rangeland degradation, i.e., loss in herbaceous plant cover, in semiarid Borana rangelands (pictures taken in dry season of 2011 by Hasen Yusuf).

1.2.1 Intensification of grazing and ecosystem C and N stocks

Grazing influences soil carbon (C) and nitrogen (N) stocks, however, studies so far have shown mixed results of grazing impacts on soil C and N; studies showed increasing (Reeder and Schuman, 2002), neutral (Shrestha and Stahl, 2008) or even decreasing effects of grazing (Pei et al., 2008). Grazers influence soil organic C (SOC) and soil organic N (SON) through

mechanisms that change C and N input and output in the soil (Baisden and Amundson, 2003). Grazers (i.) decrease primary productivity by reducing photosynthetic surface areas, (ii.) change plant C and N belowground allocation (Semmartin et al., 2010; Reeder et al., 2004), and (iii.) affect litter quality, decomposition and mineralization rates (Semmartin et al., 2010). Grazers can also affect legume species abundance and, hence, N fixation rates, which may reduce N inputs to the soil (Allard et al., 2007). Ruminant enteric fermentation as well as C and N emissions from animal wastes through volatilization and leaching impact SOC and SON stocks in the rangeland soils (Pineiro et al., 2009). Loss of soil C and N associated with grazers arise mainly from changes in soil organic matter decomposition and mineralization rates (Wang et al., 2011) or increased erosion under heavy grazing (Savadogo et al., 2007).

Grazing decreases plant cover and, thus, may increase soil organic matter mineralization rates because of higher soil temperature and moisture variability and by increasing desertification (Wang et al., 2011). Livestock affect soil condition through an excessive removal of plant tissue, and physical compaction and break-up of the top soils; this is followed by loss of top soils by wind and water erosion, and overall depletion of soil physical, chemical and biological properties (Savadogo et al., 2007; Neff et al., 2005; Mwendera and Saleem, 1997). Intensification of grazing exacerbated rangeland degradation, particularly in dry areas because of their soil's coarse texture, low organic matter content, low water and nutrient retention capacities, and low inherent soil fertility (Lal, 2004b). Therefore, decreasing the grazing pressure should increase C storage by the ecosystem, thereby removing CO₂ from the atmosphere. However, the effect of grazing intensity on SOC and SON stocks depends on climate (precipitation and temperature), soil properties, and vegetation types (C3 vs. C4) (McSherry and Ritchie, 2013; Pineiro et al., 2010). The overall consequences of grazing on herbaceous productivity, SOC and SON accumulation vary along gradients of these variables, however, only few studies have been conducted on quantifying the effects of grazing exclusion on SOC in dry rangelands of east African pastoral grazing systems (Verdoodt et al., 2009). Savanna ecosystems are complex and in a continuous state of change naturally, and also due to unsustainable land use practices (Walker and Abel, 2002). One of such dynamic changes in savanna rangelands is the increasing trend of woody encroachment in the grazed ecosystem and under livestock exclusion (Angassa and Oba, 2008, 2007), which could influence the effect of grazing on SOC and SON storage.

1.2.2 Woody encroachment and rangeland productivity

1.2.2.1 Causes of woody encroachment

The rapid expansion of woody encroachment in southern Ethiopia (Fig 1.3 and Fig. 1.4) has been widely reported as a common form of rangeland degradation (Angassa and Oba, 2008; Solomon et al., 2007; Gemedo et al., 2006; Oba and Kotile, 2001). Woody encroachment is an increase in cover, density and biomass of indigenous woody plant species (Van Auken, 2009), and has been reported globally for drylands and savannas in the recent decades (Eldridge et al., 2011). The causes of woody encroachment include overgrazing, reduced fire frequency and intensity (Higgins et al., 2007), increasing atmospheric CO₂ concentration, and long-term climate variability especially increased precipitation intensity (Kulmatiski and Beard, 2013).

Rangeland burning practices by pastoralists is among the determinant factors for the occurrence of African savannas (Sankaran et al., 2005). Intense and frequent fires often suppress or kill woody plant seedlings, thus preventing the establishment of a continuous canopy cover which would limit further herbaceous plants growth (Smit et al., 2010; Higgins et al., 2007). When fires are less intense, woody plants saplings can grow and escape the fire flame zone and becomes fire-resistant (Brown and Archer, 1999). For example, prior to suppression of fires by government policy from Borana pastoral system in the 1970s rangeland management systems in the area, including fire, reduce woody species proliferation and may have maintained open rangelands for centuries (Angassa and Oba, 2008). According to Angassa and Oba (2008), the complete suppression of fire is the main cause of an increased woody encroachment in the Borana rangelands.

Intensification of grazing may also contribute to woody encroachment in several ways: (i.) Herbaceous species compete with woody plants for resources (e.g., water) in the topsoil and reduction of herbaceous growth by grazing reduces the competitive vigor of herbaceous species, potentially enhances woody plants growth (ii.) Heavy grazing may weaken the intensity of fires by reducing the herbaceous biomass, fostering woody encroachment. (iii.) Animals may act as dispersal agents for seeds of woody species (Riginos et al., 2009; Riginos and Young, 2007; Brown and Archer, 1999). A tendency towards an increase in woody encroachment despite efforts of mechanical removal of woody plants, however, suggests that increases in atmospheric CO₂ and long-term climate variability might contribute to woody encroachment in savanna ecosystems (Kulmatiski and Beard, 2013; D'Odorico et al., 2010; Polley et al., 1994). The

increases in global CO₂ concentration may also have benefited C₃ woody species at the expense of C₄ grasses in arid and semiarid tropical ecosystems (Polley et al., 1994).



Fig.1.3. Rangelands severely encroached by *Acacia reficiens* and with low herbaceous growth potential (pictures taken by Hasen Yusuf, 2011)



Fig.1.4. Rangelands severely encroached by *Comiphora* species (pictures taken by Hasen Yusuf, 2011)

1.2.2.2 Effects of woody encroachment on pastoral productivity and ecosystem C stocks

Compared to original savanna/grasslands, woody encroachment is often associated with reduced herbaceous productivity, species richness and diversity (van Auken, 2009). The Millennium Ecosystem Assessment (2005) described it as an ecological symptom of degradation and desertification of the savannas/grasslands (MEA 2005). In east Africa, a small increase in woody encroachment in dry savannas resulted in strong reductions in pastoral production (Oba et al., 2000a). Van Wijngaarden (1985) reported that in East African dry savanna ecosystems increases in woody cover by 10% reduced herbaceous production by 7%, while grazing potential was eliminated when woody cover reached 90%. This phenomenon encourages many pastoralists to control woody encroachment mechanically (Angassa and Oba 2008). However, this approach has shown unsuccessful for long-term woody encroachment control as the reductions in woody plant cover achieved are usually temporary, and woody plant abundance typically recovers

remarkably in the decade following treatment application (Grant et al., 1999; Scifres et al., 1985). Although woody encroachment has been recognized as a major rangeland management issue in the Borana rangelands (Fig.1.5), neither the rate nor the spatial extent have yet been adequately quantified on a landscape level.



Fig.1.5. The influence of woody encroachment, i.e., canopy density and cover, on understory vegetation and soils in semiarid Borana rangelands (both pictures were taken during the wet season by Hasen Yusuf in 2011)

While woody encroachment is often regarded as severe rangeland degradation in pastoral communities, particularly within the context of grazing (Oba et al., 2000a), it has been shown to trigger a significant increase in C sequestration potential in tropical American savanna and grasslands (Knapp et al., 2008), Australia (Daryanto et al., 2013), and South Africa (Shackleton and Scholes, 2011). This C accumulation appears to be a function of enhanced below- and aboveground net woody vegetation primary productivity (NPP), low decomposition rates beneath trees, biochemical recalcitrance of woody plants litter, and organic matter stabilization in protected soil aggregates (Knapp et al., 2008). Woody plants in a grassland system tend to increase C uptake by extending the growing season, expanding the depth of the niches from which soil nutrients and water are drawn and, in the case of nitrogen fixing trees, enhancing soil fertility (Nair et al., 2009). When woody plants occur in suitable soils, C is sequestered in the biomass and soil as well (Jose and Bardhan, 2012).

Management by tree clearing and re-clearing after woody regrowth (Daryanto et al., 2013; Gifford and Howden, 2001) could have big effects on the total rangeland ecosystem C stocks, mainly through the amount of woody biomass. On the other hand, a shift in vegetation

cover may also affect belowground carbon (BGC) allocation patterns of the herbaceous vegetation, ultimately reducing belowground C storage and perhaps offsetting gains in aboveground carbon (AGC) stocks (Jackson et al., 2002). Nonetheless, the influence of woody encroachment on SOC seems to be precipitation-dependent. For example, the drier sites in the Chihuahuan desert in USA (< 280 mm rainfall) gained soil C and N with encroachment while more wet sites (400 - 800 mm rainfall) lost C and N with encroachment (Jackson et al., 2002).

The influence of woody encroachment on grassland SOC also seems to be determined by soil characteristics, woody species composition, structure, litter quality and quantity and other traits of encroacher woody species (Eldridge et al., 2011; Hudak et al., 2003; Jobbagy and Jackson, 2000). Given that southern Ethiopian rangelands occupy a large area (> 95 000 km²) and woody encroachment within tropical grasslands and savanna is a worldwide phenomenon (Van Auken, 2009; Eldridge et al., 2011), an understanding of how woody encroachment affects annual production and ecosystem C stocks is essential for quantifying and balancing rangeland ecosystem services and the global C budget (Daryanto et al., 2013; Gifford and Howden, 2001). Carbon storage in the vegetation (woody plants and grasses) and soils of the rangelands through change in livestock and rangeland management practices may hold potential to diversify pastoral livelihoods if linked to payment for sequestration or avoidance of C emissions. However, it is unclear how much C is currently stored in the vegetation biomass and soils of these rangeland ecosystems. It has further never been quantified by the way in which reduced grazing intensity and increased woody encroachment can enhance soil or ecosystem C stocks in this semiarid rangelands. Therefore, this study aimed at providing answers to the following questions:

1. How strongly has woody encroachment expanded in the Borana rangelands over the last four decades and how much C is currently and could potentially be stored in the aboveground biomass (AGB) in different grazing regimes?
2. How much carbon is currently stored in the soils of semi-arid Borana rangelands and how are the soils organic C (SOC) and total soil nitrogen (TSN) stocks influenced by woody encroachment and grazing intensity?

We expect that the area covered by woody plants have increased over the last 40 years. This could have led to a decline in open savanna areas, particularly valuable to pastoralists, but could have significantly increased the above-andbelowground C and N stocks, which could be positive from a global warming mitigation perspective. We also expect that the changes in the ecosystem

C stocks in different ecosystem components depends on the pastoral grazing land management i.e., grazing, browsing and woody plant utilization. We further would expect that long-term grazing relaxation (enclosures) increases ecosystem C and N stocks. However, the response of herbaceous vegetation to grazing will also be influenced by the amount of woody vegetation, which might highlight a joint effect on the C stocks in the herbaceous component of the ecosystem.

1.3 Specific objectives

The main objective of this thesis was to develop reliable tools to estimate AGC stocks in woody vegetation species. Further objectives were

1. To analyze the changes in vegetation cover in Yabello district rangelands from 1976 to 2012 at a landscape scale
2. To analyse vegetation structure under different grazing regimes and various woody encroachment levels
3. To determine the AGC stocks in woody and herbaceous vegetation under different grazing regimes and at various woody encroachment levels
4. To determine SOC and TSN stocks in different grazing regimes at various levels of woody encroachment

1.4 Outline of the thesis

This research focused on the assessment of the rangeland ecosystem C stocks and how these stocks have been affected by woody encroachment and intensified grazing. The main body of the research consists of the analysis of the long-term trends of woody species encroachment expansion into savanna ecosystem and development and application of allometric tree/shrub biomass models for the quantification of aboveground biomass (AGB) at the plot level. It also presents an assessment of the effects of woody encroachment and intensified grazing on vegetation structure (objective 2), The newly developed allometric biomass models facilitated the quantitative estimation of woody AGC stock at the plot level and also enabled the assessment of the influence of woody encroachment and grazing on AGC pools in study areas. Overall, the research outcomes presented in chapters 2 and 3 provide extensive answers for objectives 1, 2, 3 and 4 of this study.

Objective 5 is concerned with the assessment of the influences of the woody encroachment and grazing on SOC and TSN stocks. Examining the influence of woody encroachment and grazing on SOC and TSN leads to a better understanding of the overall ecosystem organic C and N dynamics. Hence, the aim was to determine the gain in SOC as a result of woody encroachment and reduction of grazing intensity.

Taken together, the chapters of this thesis presents long-term changes in woody encroachment and the impacts of the woody encroachment expansion on rangeland vegetation structure as well as plot and site-based estimates of above-and belowground C pools (Fig. 1.6). Although each chapter provides new and important findings, it is only the combination of each part that can help understand the effects of grazing and vegetation cover dynamics on the rangeland ecosystem C storage. This information is necessary to develop sound rangeland management policies that link the C storage potential of the rangelands to global climate change mitigation and adaptation strategies through establishing viable mechanism of payment for ecosystem services.

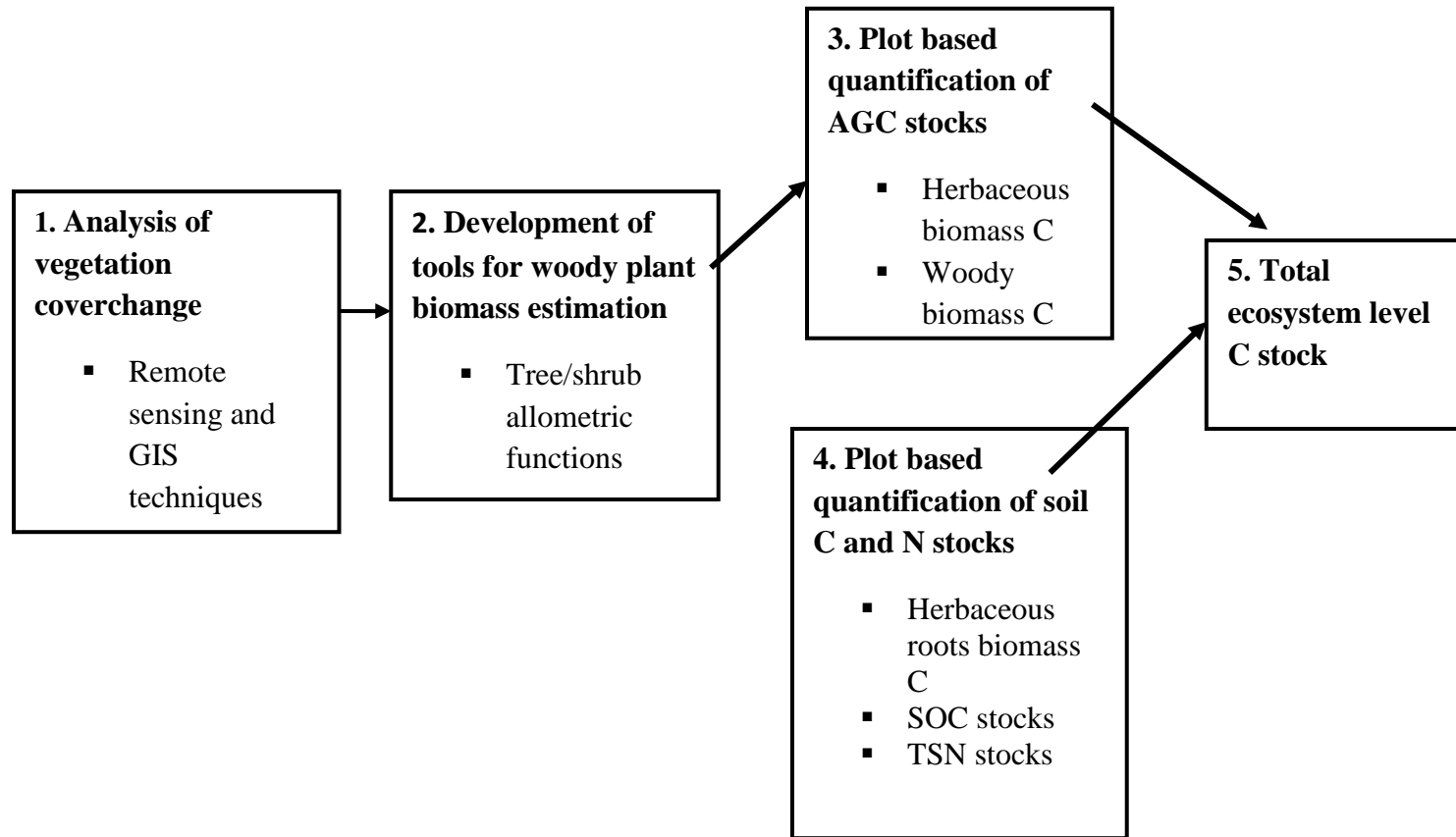


Fig.1.6. Schematic representation of different aims and methodologies used in the study

References

- Aklilu, Y. (2002). An Audit of the Livestock Marketing Status in Kenya, Ethiopia and Sudan. OAU/Interafrican Bureau for Animal Resources, I and II. Available at <http://www.eldis.org/vfile/upload/1/document/0708/DOC11340.pdf>
- Allard, V., Soussana, J.F., Falcimagne, R., Berbigier, P., Bonnefond, J.M., Ceschia, E., D'hour, P., Hénault, C., Laville, P., Martin, C., and Pinarès-Patino, C. (2007). The role of grazing management for the net biome productivity and greenhouse gas budget (CO₂, N₂O and CH₄) of semi-natural grassland. *Agriculture, Ecosystems and Environment*, 121: 47-58.
- Angassa, A., and Oba, G. (2007). Effects of management and time on mechanisms of bush encroachment in southern Ethiopia. *African Journal of Ecology* 46, 186–196.
- Angassa, A. and Oba, G. (2008). Herder perceptions on impacts of range enclosures, crop farming, fire ban and bush encroachment on the rangelands of Borana, Southern Ethiopia. *Human Ecology*, 36: 201-215.
- Bai, Z.G., Dent, D.L., Olsson, L. and Schaepman, M.E. (2008). Proxy global assessment of land degradation. *Soil Use and Management*, 24: 223-234.
- Bond, W.J. and Midgley, G.F. (2012). Carbon dioxide and the uneasy interactions of trees and savannah grasses. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 367: 601-612.
- Brown, J.R. and Archer, S. (1999). Shrub invasion of grassland: recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology*, 80: 2385-2396.
- Coppock, D. L. (1994). The Borana plateau of southern Ethiopia: Synthesis of pastoral research, development and change, 1980-91. *International Livestock Research Institute* (ILRI), Addis Ababa, Ethiopia.
- Conant, R.T. and Paustian, K. (2002). Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global Biogeochemical Cycles*, 16: 1143.
- Cossins, N.J. and Upton, M. (1987). The Borana pastoral system of Southern Ethiopia. *Agricultural Systems*, 25: 199-218.
- Cossins, N.J. and Upton, M. (1988b). Options for improvement of the Borana Pastoral System. *Agricultural Systems*, 27: 251-278.
- Cossins, N.J. and Upton, M. (1988a). The impact of climatic variation on the Borana pastoral system. *Agricultural Systems*, 27: 117-135.

D'Odorico,P., Fuentes,J.D., Pockman,W.T., Collins,S.L., He,Y., Medeiros,J.S., DeWekker,S. and Litvak,M.E. (2010). Positive feedback between microclimate and shrub encroachment in the northern Chihuahuan desert. *Ecosphere*, 1: art 17.

Daryanto, S., Eldridge,D.J. and Throop,H.L. (2013). Managing semi-arid woodlands for carbon storage: Grazing and shrub effects on above- and belowground carbon. *Agriculture, Ecosystems and Environment*, 169: 1-11.

Eldridge, D.J., Bowker, M.A., Maestre, F.T., Roger, E., Reynolds, J.F. and Whitford,W.G. (2011). Impacts of shrub encroachment on ecosystem structure and functioning: Towards a global synthesis. *Ecology Letters*, 14: 709-722.

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

Gemedo, D., Maass, B.L. and Isselstein, J. (2006). Rangeland condition and trend in the semi-arid Borana lowlands, southern Oromia, Ethiopia. *African Journal of Range and Forage Science*, 23: 49-58.

Gifford, R.M. and Howden, M. (2001). Vegetation thickening in an ecological perspective: Significance to national greenhouse gas inventories. *Environmental Science and Policy*, 4: 59-72.

Grant, W.E., Hamilton, W.T.,and Quintanilla, E. (1999). Sustainability of agroecosystems in semi-arid grasslands: simulated management of woody vegetation in the Rio Grande Plains of southern Texas and northeastern Mexico. *Ecological Modelling* 124: 29–42.

Garnaut, R. (2008). *The Garnaut Climate Change Report*. Cambridge University Press. 634p.

Higgins, S.I., Bond,W.J., February, E.C., Bronn, A., Euston-Brown, D.I.W., Enslin, B., Govender, N., Rademan, L., O'Regan, S., Potgieter, A.L.F., Scheiter, S., Sowry, R., Trollope, L. and Trollope,W.S.W. (2007). Effects of four decades of fire manipulation on woody vegetation structure in savanna. *Ecology*, 88: 1119-1125.

Hogg, R. (1997). *Pastoralists, ethnicity, and the state in Ethiopia*. Haan Publishing, London.

Homann, S., Rischkowsky, B. and Steinbach, J. (2008a). The effect of development interventions on the use of indigenous range management strategies in the Borana Lowlands in Ethiopia. *Land Degradation and Development*, 19: 368-387.

Homann, S., Rischkowsky, B., Steinbach, J., Kirk, M. and Mathias, E. (2008b). Towards endogenous livestock development: Borana pastoralists' responses to environmental and institutional changes. *Human Ecology*, 36: 503-520.

Hudak, A.T.,Wessman,C.A. and Seastedt,T.R. (2003). Woody overstorey effects on soil carbon and nitrogen pools in South African savanna. *Austral Ecology*, 28: 173-181.

- IIED and SOS Sahel UK 2010. *Modern and mobile* The future of livestock production in Africa's drylands, available at <http://pubs.iied.org/pdfs/12565IIED.pdf>
- Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, W.T. and Wall, D.H. (2002). Ecosystem carbon loss with woody plant invasion of grasslands. *Nature*, 418: 623-626.
- Jobbágy, E.G. and Jackson, R.B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, 10: 423-436.
- Jose, S. and Bardhan, S. (2012). Agroforestry for biomass production and carbon sequestration: an overview. *Agroforestry Systems*, 86: 105-111.
- Köchy, M. and Wilson, S.D. (2001). Nitrogen Deposition and Forest Expansion in the Northern Great Plains. *Journal of Ecology*, 89: 807-817.
- Knapp, A.K., Briggs, J.M., Collins, S.L., Archer, S.R., Bret-Harte, M.S., Ewers, B.E., Peters, D.P., Young, D.R., Shaver, G.R., Pendall, E. and Cleary, M.B. (2008a). Shrub encroachment in North American grasslands: Shifts in growth form dominance rapidly alters control of ecosystem carbon inputs. *Global Change Biology*, 14: 615-623.
- Kulmatiski, A. and Beard, K.H. (2013). Woody plant encroachment facilitated by increased precipitation intensity. *Nature Climate Change*, 3: 833-837.
- Lal, R. (2004a). Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. *Science*, 304: 1623-1627.
- Lal, R. (2004b). Carbon Sequestration in Dryland Ecosystems. *Environmental Management*, 33: 528-544.
- McSherry, M.E. and Ritchie, M.E. (2013). Effects of grazing on grassland soil carbon: a global review. *Global Change Biology*, 19: 1347-1357.
- MEA. (2005). Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Desertification Synthesis. World Resources Institute; Washington D.C.
- Mortimore, M. (2009). *Dryland Opportunities: New paradigm for people, ecosystems and development*. New York/Gland, Switzerland/London: UNDP/IUCN/IIED. Available at: <http://pubs.iied.org/pdfs/G02572.pdf>.
- Mwendera, E.J. and Saleem, M.A.M. (1997). Infiltration rates, surface runoff, and soil loss as influenced by grazing pressure in the Ethiopian highlands. *Soil Use and Management*, 13: 29-35.
- Neely, C. and S. Bunning. (2008). Review of Evidence on Dryland Pastoral Systems and Climate Change: Implications and opportunities for mitigation and adaptation. FAO – NRL Working Paper. Rome, Italy.

Neff, J.C., Reynolds, R.L., Belnap, J. and Lamothe, P. (2005). Multi-decadal impacts of grazing on soil physical and biogeochemical properties in southeast Utah. *Ecological Applications*, 15: 87-95.

Oba, G. and Kotile, D.G. (2001). Assessments of landscape level degradation in southern Ethiopia: pastoralists versus ecologists. *Land Degradation and Development*, 12: 461-475.

Oba, G., Post, E., Syvertsen, P.O. and Stenseth, N.C. (2000a). Bush cover and range condition assessments in relation to landscape and grazing in southern Ethiopia. *Landscape Ecology*, 15: 535-546.

Oba, G., Stenseth, N.C. and Lusigi, W.J. (2000b). New Perspectives on Sustainable Grazing Management in Arid Zones of Sub-Saharan Africa. *BioScience*, 50: 35-51.

Pei, S., Fu, H. and Wan, C. (2008). Changes in soil properties and vegetation following enclosure and grazing in degraded Alxa desert steppe of Inner Mongolia, China. *Agriculture, Ecosystems and Environment*, 124: 33-39.

Pineiro, G., Paruelo, J.M., Jobbágy, E.G., Jackson, R.B. and Oesterheld, M. (2009). Grazing effects on belowground C and N stocks along a network of cattle enclosures in temperate and subtropical grasslands of South America. *Global Biogeochemical Cycles*, 23.

Pineiro, G., Paruelo, J.M., Oesterheld, M. and Jobbágy, E.G. (2010). Pathways of grazing effects on soil organic carbon and nitrogen. *Rangeland Ecology and Management*, 63: 109-119.

Polley, H.W., Johnson, H.B. and Mayeux, H.S. (1994). Increasing CO₂: Comparative Responses of the C₄ Grass *Schizachyrium* and Grassland Invader *Prosopis*. *Ecology*, 75: 976-988.

Ramachandran Nair, P.K., Mohan Kumar, B. and Nair, V.D. (2009). Agroforestry as a strategy for carbon sequestration. *Journal of Plant Nutrition and Soil Science*, 172: 10-23.

Reeder, J.D. and Schuman, G.E. (2002). Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environmental Pollution*, 116: 457-463.

Reeder, J.D., Schuman, G.E., Morgan, J.A. and Lecain, D.R. (2004). Response of organic and inorganic carbon and nitrogen to long-term grazing of the shortgrass steppe. *Environmental Management*, 33: 485-495.

Reid, R., Thornton, P., McCrabb, G., Kruska, R., Atieno, F. and Jones, P. (2004). Is it possible to mitigate greenhouse gas emissions in pastoral ecosystems of the tropics? *Environment, Development and Sustainability*, 6: 91-109.

Riginos, C., Grace, J.B., Augustine, D.J. and Young, T.P. (2009). Local versus landscape-scale effects of savanna trees on grasses. *Journal of Ecology*, 97: 1337-1345.

Riginos, C. and Young, T. (2007). Positive and negative effects of grass, cattle, and wild herbivores on *Acacia* saplings in an East African savanna. *Oecologia*, 153: 985-995.

- Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D.J., Cade, B.S., Gignoux, J., Higgins, S.I., Le Roux, X., Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G., Caylor, K.K., Coughenour, M.B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C., Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K.L., Prins, H.H.T., Ringrose, S., Sea, W., Tews, J., Worden, J. and Zambatis, N. (2005). Determinants of woody cover in African savannas. *Nature*, 438: 846-849.
- Savadogo, P., Sawadogo, L. and Tiveau, D. (2007). Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. *Agriculture, Ecosystems and Environment*, 118: 80-92.
- Scifres, C.J., Hamilton, W.T., Conner, J.R. *et al.* (1985). Integrated brush management systems for South Texas: development and implementation. Texas Agricultural Experiment Station, College Station.
- Semmartin, M., Di Bella, C. and de Salamone, I.G. (2010). Grazing-induced changes in plant species composition affect plant and soil properties of grassland mesocosms. *Plant and Soil*, 328: 471-481.
- Shackleton, C.M. and Scholes, R.J. (2011). Above ground woody community attributes, biomass and carbon stocks along a rainfall gradient in the savannas of the central lowveld, South Africa. *South African Journal of Botany*, 77: 184-192.
- Shrestha, G. and Stahl, P.D. (2008). Carbon accumulation and storage in semi-arid sagebrush steppe: Effects of long-term grazing exclusion. *Agriculture, Ecosystems and Environment*, 125: 173-181.
- Smit, I.P.J., Asner, G.P., Govender, N., Kennedy-Bowdoin, T., Knapp, D.E. and Jacobson, J. (2010). Effects of fire on woody vegetation structure in African savanna. *Ecological Applications*, 20: 1865-1875.
- Solomon, T.B., Snyman, H.A. and Smit, G.N. (2007). Cattle-rangeland management practices and perceptions of pastoralists towards rangeland degradation in the Borana zone of southern Ethiopia. *Journal of Environmental Management*, 82: 481-494.
- Sternberg, M., Gutman, M., Perevolotsky, A., Ungar, E.D. and Kigel, J. (2000). Vegetation response to grazing management in a Mediterranean herbaceous community: a functional group approach. *Journal of Applied Ecology*, 37: 224-237.
- Swift, J. (1998). Les Grands Themes du Development Pastoral et le cas de Quelques Pays Africains. FAO/ESH Working Papers on Pastoral and Agro-pastoral Societies, Rome.
- Swift J. (1995). Dynamic ecological systems and the administration of pastoral development. In: Living with uncertainty (ed I. Scoones) pp. 153-173. Intermediate Technology Publications Ltd, London, UK.
- Troy Baisden, W. and Amundson, R. (2003). An analytical approach to ecosystem biogeochemistry modeling. *Ecological Applications*, 13: 649-663.

Verdoodt, A., Mureithi, S.M., Ye, L. and Van Ranst, E. (2009). Chronosequence analysis of two enclosure management strategies in degraded rangeland of semi-arid Kenya. *Agriculture, Ecosystems and Environment*, 129: 332-339.

Van Auken, O.W. (2009). Causes and consequences of woody plant encroachment into western North American grasslands. *Journal of Environmental Management*, 90: 2931-2942.

Walker, B, Abel, N. (2002) Resilient rangelands – Adaptation in complex systems. Pp293-313 in *Panarchy: Understanding transformations in human and natural systems*. Eds LH Gunderson and CS Holling. Publ. Island Press.

Wang, S., Wilkes, A., Zhang, Z., Chang, X., Lang, R., Wang, Y. and Niu, H. (2011). Management and land use change effects on soil carbon in northern China's grasslands: A synthesis. *Agriculture, Ecosystems and Environment*, 142: 329-340.

Watson, E.E. (2003). Examining the Potential of Indigenous Institutions for Development: A Perspective from Borana, Ethiopia. *Development and Change*, 34: 287-310.

**2 Predicting aboveground biomass of encroacher woody species in semi-arid rangelands,
Ethiopia**

Published in:

Journal of Arid Environments, year 2013, volume96, pages 64-72

Predicting aboveground biomass of encroacher woody species in semi-arid rangelands, Ethiopia

M. Hasan-Yusuf^{a*}, A.C. Treydte^a, E. Abule^b, and J. Sauerborn^a

^a Agroecology in the Tropics and Subtropics, University of Hohenheim, 70599 Stuttgart, Germany

^b Adami Tulu Research Center, P.O. Box 35, Zeway, Ethiopia

*corresponding author: hasanyusuf12@gmail.com

Abstract

Species-specific allometric models were developed to predict aboveground biomass (AGB) of eight woody species in the Borana rangelands, Ethiopia. The 23 equations developed (8 species; three biomass components: total aboveground, stem and branches) fit the data well to predict total AGB and by components for each of the species ($r^2 > 0.7$; $p < 0.001$). The AGB of tree shaped species (e.g., *Acacia bussei* and *Acacia etabaica*) were significantly predicted from a single predictor (circumference of the stem at ankle height), with a high coefficient of determination ($r^2 > 0.95$; $p < 0.001$). In contrast, the AGB of bushy shrubs (e.g., *Acacia oerfota*) was more effectively predicted by using the canopy volume ($r^2 = 0.84$; $p < 0.001$). Shrubs with a tall stem and an umbrella-like canopy structure (e.g., *Acacia mellifera*) were most accurately predicted by a combination of both circumference of the stem at ankle height and canopy volume ($r^2 = 0.95$; $p < 0.001$). Hence, our species-specific allometric models could accurately estimate their woody AGB in a semi-arid savanna ecosystem of southern Ethiopia. These equations will help in future carbon-trade discussions in times of climate change and CO₂ emission concerns and mitigation strategies.

Keywords: aboveground biomass, *Acacia* species, allometric equations, bush, carbon sequestration, rangeland, regression models

2.1 Introduction

Woody plants have strongly increased in both cover and density in grasslands and savanna systems worldwide over the past century, which has been particularly visible in African savannas (Munyati et al., 2010; Sankaran et al., 2005; Bassett and Zuéli, 2001; Oba et al., 2000). These increases might trigger biome shifts from grassland to shrubland (Briggs et al., 2005). This is of concern to pastoralists and their grazing livestock, since the thickening woody vegetation competes with the herbaceous forage and reduces livestock carrying capacity (Angassa and Oba, 2008; Abule et al., 2007; Oba et al., 2000), whereas higher tree densities in grass dominated or mixed tree-grass systems will, on the other hand, significantly increase the carbon (C) storage capacity of the grassland systems (Hibbard et al., 2003; Hughes et al., 2006). While some authors have estimated the net gain in C sequestration through woody plant encroachment for Australian (Gifford and Howden, 2001) and tropical American savannas and grasslands (Asner et al., 2003; Lett et al., 2004; San Jose et al., 1998), to our knowledge there are no similar publications for East Africa. To quantify the net gain in C stock due to woody plant proliferation in grassland ecosystems reliable estimates of the standing biomass are required. These biomass estimates are also useful for making informed land management decisions including assessment of fuel loads for burning activities (Bird and Shepherd, 1989; Tietema, 1993a), estimating forage quantity and quality (Bellefontaine et al., 2000) or monitoring shrub invasion (Lett et al., 2004).

Biomass can be estimated using direct or indirect methods. Direct measurement of tree AGB involves felling a proper number of trees/shrubs and estimating their field and oven-dry weights, a method that can be costly and impractical, especially when dealing with numerous species and large sample areas (Willebrand et al., 1993). An indirect method is to use allometric regression equations based on easily accessible and measurable woody plant dimensions, rather than performing this so-called “destructive sampling” in the field. These equations can help predict the biomass component based on some easily measurable predictor variables such as stem diameter / circumference, shoot height or crown diameter, which can be measured non-destructively (Whittaker and Woodwell, 1968). Therefore, the most accurate method is to calculate regression equations from destructively sampled trees that are in the size range of

interest and to apply these equations to every tree in the stand for verification purposes (Abola et al., 2005).

The use of the available generalized and species-specific biomass equations across wider ecological zones can lead to a bias in estimating biomass for particular species and sites (Henry et al., 2011), because there are variations among species in wood-specific gravity, tree sizes, growth stages, and since some geographic areas have not been covered by the equations (Navar et al., 2002). Consequently, developers of the equations often caution against extrapolation beyond their study area (Grundy, 1995; Navar et al., 2002). In addition, the accuracy of biomass estimations can be affected by several factors and are known to vary with soil, climate, disturbance regime, succession status, topographic conditions (Ketterings et al., 2001; Litton and Kauffman, 2008). Therefore, species-specific allometric biomass equations tailored to estimate biomass of a particular species in a given biome are essential for providing more accurate estimates (Litton and Kauffman, 2008).

However, for many parts of east African dry savannas and shrublands either species-specific or generalized allometric biomass equations, which could be used to assess biomass, C stocks and changes in these stocks, are very scarce (Henry et al., 2011). As this region, however, is strongly affected by climate change and since pastoralists have been reported to suffer due to declining grazing land resources caused by increasing woody cover (Angassa and Oba, 2008; Gemedo-Dale et al., 2006), species-specific allometric equations for dominant woody species should be developed and applied to estimate C stocks and C sequestration in standing vegetation in East Africa. While this information is of scientific interest to control for the uncertainties and biases involved in using inaccurate biomass equations it can also help to obtain financial rewards for sequestered C or for CO₂ emission reductions through appropriate management of terrestrial biomass (Henry et al., 2011). We expect that for dryland woody species, the biomass predictor dendrometric variables vary depend on the woody species growth form and architecture. We also assume that for some of woody plant species, a combination of measurable dendrometric variables will provide the best estimates while only one single dimension is not as reliable. We further hypothesize that bushy-shaped woody species is less accurately predictable than tree-shaped ones.

Our objective was to develop site- and species-specific allometric functions through destructive harvest of woody vegetation to predict branch, stem, and total AGB for eight most

dominant woody species, based on their dendrometric measurements. The purpose of the study was to contribute to the accurate estimation of the woody vegetation biomass and Cstocks in the Borana rangelands, southern Ethiopia.

2.2. Material and methods

2.2.1. Study area

The study area was located in the Borana rangelands at Masade plain (46° 83' 19'' N, 38° 19' 50'' E) in Yabelo district, Borana Zone, at an altitude of 1485 m.a.s.l. The Borana rangelands cover about 95 000 km² (Coppock, 1994) of the Southern Oromia Regional State of Ethiopia. The rangelands are characterized by an arid and semi-arid climate with pockets of sub-humid zones and an average annual rainfall between 238 mm and 896 mm, with a high coefficient of variation ranging from 18% to 69% across years (Angassa and Oba, 2008). Rainfall is bimodal, with 60% of the annual rainfall occurring in March - May, followed by a minor peak in September - November. Long-term variability in the quantity and distribution of rainfall often results in frequent recurrent droughts (Angassa and Oba, 2008). The mean annual maximum and minimum temperatures are 30°C and 13°C in the in the long warm dry and short cool season, respectively. The soil at this particular study site is categorized as Chromic-cambisols on coarse grained acid plutonic and/or crystalline foliated metamorphic rock and fine-grained acid (extrusive) organic sedimentary rocks. It is deep, well drained, moderately fine textured, characterized by few surface stone and few rocks outcrops and contains low levels of organic carbon and available phosphorus (OPADC, 2011). The vegetation is characterized by a dense tree / shrub savanna, the herbaceous layer being composed of perennial grasses and forbs. Woody vegetation is dominated by Fabaceae and Burseraceae families, the main woody species being *Acacia mellifera* (Vahl.) Benth., *Commiphora africana* (A. Rich.) Engl., *Commiphora habessinica* (Berg) Engl., *Acacia senegal* (L.) Willd., *Grewia bicolor* Juss., *Acacia bussei* Harms ex Sjoestedt., *Acacia tortilis* (Forssk.) Hayne., *Acacia oerfota* (Forssk) Schweinf. , *Acacia reficiens* Wawra., and *Acacia drepanolobium* Harms ex Sjoest.

2.2.2 The species investigated

For this study eight woody species namely, *A. mellifera*, *A. senegal*, *A. bussei*, *A. drepanolobium*, *A. etabaica* Schweinf., *A. oerfota*, *A. reficiens* and *C. africanawere* sampled. The species sampled were found to be most relevant for our research because: (1) they accounted for about

49 % of the woody vegetation cover in the study area (Gemedo-Dalle et al., 2006); (2) they are widely distributed with relatively higher cover abundance across the Borana rangelands (Gemedo-Dalle et al., 2006); (3) they are identified as the major encroacher woody species affecting rangeland productivity (Angassa and Oba, 2008; Coppock, 1994; Oba et al., 2000); (4) the species are natives to tropical savanna ecosystems and of socio-economic importance, for example, due to their fuel value, forage and medicinal values (Gemedo-Dalle et al., 2005).

2.2.3. Samples

The destructive sampling of the woody species was conducted in an open natural stand of vegetation of the Masade plain. This site was selected because it represented one of the most woody species encroached area in the Borana rangelands and also where individual plant of various size category i.e., from the smallest to the tallest size individuals of each species (data not shown) can easily be obtained for sampling purpose. Twenty individual woody plants of the species of interest were identified and tagged for destructive sampling. The woody plants varied in height from < 1m to the largest specimen typical to an area, this was > 5 – 10 m (Gemedo-Dalle et al., 2006). A sample of woody plants included individuals with heights of 0.5, 1, 1.5, 2, 2.5, 3, 3.5, 4, 4.5, 5, 5.5, 6, 6.5, 7, 7.5, 8, 8.5, 9, 9.5, and 10 m (Payton et al., 2004). Sampling of the woody species was undertaken during the end of long dry season, in April 2011, when the woody species had a minimal amount of leaves, to ensure that biomass increments would remain rather constant at this period of the year (Bellefontaine et al., 2000).

Prior to harvest, woody plant canopy volume (radius of length (a) x width (b) x height (H)), circumference of the stem at ankle height (5 - 10 cm aboveground) and total plant height dimensions were measured to calculate predictor variable values for our allometric equations. An *elliptic crown area* $CA = \pi \times a \times b$ (Eq. (1)) and an *ellipsoid crown volume* $CV = 4/3 \times \pi \times a \times b \times H$ (Eq. (2)) were calculated since elliptic formulae best approximate the shape of shrubs and provide the highest flexibility for multiple canopy dimensions (Vora, 1988). The two longest canopy radii perpendicular to each other and parallel to the ground were measured and used to calculate CA , using Eq. (1) (Vora, 1988). For multi-stemmed shrubby species (e.g., *A. oerfota*), shrubby canopy clump width radii and height were used to calculate CA and CV . For trees and single-stem tall shrubs, the circumference of the stem at ankle height (5 - 10 cm aboveground) (SB) above the ground was measured (Tietema, 1993a). The CA width measurements with vertical height at the highest foliage were used to calculate CV using Eq. (2) (Murray and

Jacobson, 1982). In total, 160 woody plants, i.e., twenty individual plants for each species were harvested and dendrometrically measured (see Table 2. 1).

Table 2. 1. Range of woody plant dendrometric variables and dry weight biomass component of stem (Ws), branch (Wb) and total dry aboveground biomass (Wt) for the sample woody plants. SB = circumference at stem base, H = height, CV = crown volume, CA = crown area, Mc=moisture content, nm= not measured. Values are sample mean \pm standard error over 20 samples per species.

Species	SB (cm)	H (m)	CV (m ³)	CA (m ²)	Wb(Kg)	Ws(Kg)	Wt(kg)	MC (%)
<i>Acacia mellifera</i>	33.0 \pm 4.7	4.7 \pm 0.4	145.8 \pm 69.1	32.6 \pm 11.0	84.52 \pm 36.09	15.40 \pm 6.02	99.92 \pm 42.02	37.70 \pm 1.45
<i>Acacia bussei</i>	39.7 \pm 5.4	4.5 \pm 0.3	102.2 \pm 35.1	26.3 \pm 6.4	77.00 \pm 30.95	17.89 \pm 5.98	94.89 \pm 36.65	33.34 \pm 1.35
<i>Acacia etabaica</i>	40.9 \pm 4.5	4.6 \pm 0.3	54.0 \pm 15.1	14.2 \pm 3.2	37.87 \pm 9.87	20.41 \pm 7.19	58.29 \pm 16.43	33.36 \pm 1.37
<i>Acacia senegal</i>	29.4 \pm 2.9	4.3 \pm 4.3	52.6 \pm 9.5	17.6 \pm 2.4	26.25 \pm 6.77	7.95 \pm 1.58	34.20 \pm 8.24	38.28 \pm 1.48
<i>Acacia drepanolobium</i>	34.0 \pm 3.5	5.1 \pm 0.36	70.9 \pm 19.3	17.3 \pm 3.2	30.45 \pm 8.87	9.56 \pm 1.77	40.02 \pm 10.38	44.27 \pm 3.76
<i>Acacia oerfota</i>	nm	4.4 \pm 0.3	83.6 \pm 14.4	26.6 \pm 3.6	30.71 \pm 7.23	12.10 \pm 2.99	42.81 \pm 10.12	43.44 \pm 0.96
<i>Acacia reficiens</i>	nm	4.4 \pm 0.6	55.8 \pm 22.1	14.9 \pm 0.6	50.21 \pm 14.02	32.16 \pm 13.47	82.38 \pm 26.55	31.94 \pm 1.05
<i>Commiphora africana</i>	25.7 \pm 1.7	2.0 \pm 0.1	9.5 \pm 1.6	6.3 \pm 0.8	2.3 \pm 0.48	1.9 \pm 0.32	4.2 \pm 0.60	66.6 \pm 0.81

After dendrometric measurements the AGB of each woody plant was harvested by cutting it at ground level and separating the biomass into live and dead gross 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” AGB (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 (Navar et al., 2004).

The stem was de-branched, and stem and branches were cut into sections to facilitate weighing in the field. Each component of the plant was weighed to the nearest 100 g using a spring balance, and the fresh weight was recorded in the field. A fresh biomass subsample of 250 - 500 g for each stem, gross branch and thin branch component was brought to the laboratory for dry biomass determination, dried to constant mass for 72 hours at 60°C (Payton et al., 2004) and then weighed to the nearest 10 g. Dry to fresh weight ratios were determined for each sample per component and multiplied by the fresh weight of the biomass component to calculate dry biomass. The moisture content of each tree/shrub component was further determined by subtracting the dry from the wet biomass. *SB*, *H*, *CA* and *CV* data vs component biomass were used to develop allometric equations through multiple regression models.

2.2.4. Statistical analyses

Natural log (ln) linear regression techniques were used to develop allometric models from *SB* (in cm), *H* (in m), *CA* (in m²), *CV* (in m³) and their interactions to predict individual woody plants branches biomass, stem biomass and total AGB separately for the selected eight woody species. We included all woody plant dimensional measurements in the allometric equation for biomass prediction to compensate for highly variable architecture and branching patterns of trees and shrubs in the dryland ecosystem, which often makes biomass prediction problematic (Cole and Ewel, 2006). The natural log (ln) transformed linear fit in form of $\ln(y) = \alpha \ln(x) + \beta$ (Eq.(3)) (Vacher, 1999) was used, with *y* representing the total AGB (in kg dry weight for every individual plant of each species), *x* representing the predictor variable (e.g., *SB* for each species under study), and α and β as the scaling coefficient (or allometric constant) and intercept term, respectively, derived from the regression fitted to the empirical data.

We also explored the use of a nonlinear power function, $y = \beta x^\alpha$ (Eq.(4)) models for estimating biomass (Gayon, 2000). While natural log (ln) linear regression models have become

conventional practice to fit the empirical data to linear regression models (Vacher, 1999), the power function in Eq. (4) has also been a rather common mathematical model (e.g., Liton and Kauffman, 2008). In both cases, the final model choice was based on the analyses of residuals. We decided to use natural log (ln) transformed linear model because: (1) all relationships examined showed a better fit to the log- linear function compared to the nonlinear power function (Eq. (4)); (2) for our data natural log (ln) transformed linear models always resulted in a more robust model fit compared to direct power transformed nonlinear models, based on the goodness-of-fit parameters and analysis of residuals; (3) a systematic bias introduced during log transformation of the data can easily be corrected using the back-transformation of the values (Sprugel, 1983).

Generalized linear models (GLM) (SAS, 2002) were used for the analysis of the goodness of fit for all regression equations. The goodness of fit was determined by examining P -values, the mean square of the error (MSE), the coefficient of determination (r^2), the coefficient of variation (CV), and the degree of homoscedasticity of residual plots by regressing the residuals (observed minus predicted values) against predictor variables and the predicted versus observed values of each model. The difference between the chosen fit and the predictors and their interactions for each plant dry biomass was regarded significant if the 95% confidence intervals (CI) did not overlap. The assumption of multicollinearity in multiple regressions was considered as not violated since we systematically omitted the predictor variable that showed a strong correlation coefficient (≥ 0.9) with another predictor variable while being less strongly associated with a particular response variable (Tabachnick and Fidell, 1996). Accordingly, CA was completely omitted from all models as it was strongly correlated with CV for all species.

2.3. Results

The goodness of fit analysis for the models across all species indicated that the species-specific regression models relating biomass with selected predictor woody species dimensions were highly significant (Appendix 2.1). *Acacia bussei*, *A. mellifera*, and *A. etabaica* had a comparatively higher goodness of fit for total dry AGB compared to *A. oerfeta*, *A. reficiens* and *C. africana* (see Table 2.2). In general, the results showed that the total dry AGB was accurately predictable ($adjR^2 \geq 0.84$) and that all models allowed for sound estimates of the AGB of the studied species based on their dendrometric measurements (see Appendix 2.1). 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 (Table 2.2, Appendix 2.1). The simplest model, based on only one single predictor variable, was obtained for *A. etabaica*, *A. bussei*, and *A. oerfota*, which showed a good fit to the data (Appendix 2.1). The remaining species were explained by models requiring a combination of two predictor variables to obtain a good fit (Appendix 2.1).

According to our hypothesis, the AGB of tall trees with open canopy architecture (e.g., *A. bussei* and *A. etabaica*) was significantly well predicted by a single predictor (SB) only, leading to a high coefficient of determination ($adj\ r^2 > 0.95$; $p < 0.001$). In contrast, the AGB of bushy shrubs (e.g., *A. oerfota*) with a clumps-forming canopy structure was predicted well using canopy measurements (CV; $adj\ r^2 = 0.84$; $p < 0.001$). Shrubs with a tall stem and an umbrella-like canopy structure (e.g., *A. mellifera*) were most accurately predicted by a combination of both stem circumference at the base and canopy volume (SB and CV, respectively; $adj\ r^2 = 0.95$; $p < 0.001$). Each of the species-specific allometric regression models developed is described below:

Table 2.2. Allometric functions of the best fits for the different tree dendrometric variables: Height (H, in cm), circumference at of the stem ankle height (SB, in cm), crown volume (CV, in m³) and dry weight biomass components of branches + leafy shoots (Wb, in Kg), stem (Ws, in Kg) and total aboveground dry biomass (Wt, in Kg) of 20 individuals each for eight woody species of semi-arid Borana rangelands.

Woody species		Allometric equation	adj r^2	F	P
<i>Acacia bussei</i>	Wt	$\ln(Wt) = -5.6308 + 2.5838 \ln(SB)$	0.96	435.39	p<0.001
	Ws	$\ln(Ws) = -6.5502 + 2.4056 \ln(SB)$	0.93	221.81	P<0.001
	Wb	$\ln(Wb) = -6.1253 + 2.6524 \ln(SB)$	0.95	395.53	P<0.001
<i>Acacia mellifera</i>	Wt	$\ln(Wt) = -2.7777 + 0.963 \ln(SB) + 0.7503 \ln(CV)$	0.96	203.89	p<0.001
	Ws	$\ln(Ws) = -3.596 + 0.7085 \ln(SB) + 0.7463 \ln(CV)$	0.92	117.82	p<0.001
	Wb	$\ln(Wb) = -3.1639 + 0.9991 \ln(SB) + 0.7625 \ln(CV)$	0.96	203.22	p<0.001
<i>Acacia etabaica</i>	Wt	$\ln(Wt) = -7.0822 + 2.877 \ln(SB)$	0.96	425.24	p<0.001
	Ws	$\ln(Ws) = -4.8249 + 1.4757 \ln(SB)$	0.94	282.41	p<0.001
	Wb	$\ln(Wb) = -7.3611 + 2.8475 \ln(SB)$	0.94	287.09	p<0.001
<i>Acacia senegal</i>	Wt	$\ln(Wt) = -1.5515 + 0.5579 \ln(SB) + 0.5922 \ln(CV)$	0.82	44.60	p<0.001
	Ws	$\ln(Ws) = -2.4616 + 0.7157 \ln(SB) + 0.3349 \ln(CV)$	0.90	87.04	p<0.001
	Wb	$\ln(Wb) = -3.4604 + 1.0684 \ln(SB) + 0.6967 \ln(CV)$	0.76	31.63	p<0.001
<i>Acacia drepanolobium</i>	Wt	$\ln(Wt) = -3.3387 + 1.1296 \ln(SB) + 1.7012 \ln(H)$	0.84	45.78	p<0.001
	Ws	$\ln(Ws) = -2.6987 + 0.8113 \ln(SB) + 1.2066 \ln(H)$	0.72	22.64	p<0.001
	Wb	$\ln(Wb) = -4.6227 + 1.3193 \ln(SB) + 1.8487 \ln(H)$	0.82	40.99	p<0.001
<i>Acacia oerfota</i>	Wt	$\ln(Wt) = -1.32 + 1.1084 \ln(CV)$	0.85	101.70	p<0.001
	Ws	$\ln(Ws) = -3.4682 + 1.2825 \ln(CV)$	0.77	63.63	p<0.001
	Wb	$\ln(Wb) = -1.4773 + 1.0726 \ln(CV)$	0.84	102.92	p<0.001
<i>Acacia reficiens</i>	Wt	$\ln(Wt) = 0.1774 + 0.872 \ln(CV)$	0.88	143.00	p<0.001
	Ws	$\ln(Ws) = -0.1968 + 2.0115 \ln(H)$	0.70	45.45	p<0.001
	Wb	$\ln(Wb) = -0.2346 + 1.0438 \ln(CV)$	0.83	93.22	p<0.001
<i>Commiphora africana</i>	Wt	$\ln(Wt) = -2.7882 + 1.1324 \ln(SB) + 0.3163 \ln(CV)$	0.93	133.5	p<0.001
	Ws	$\ln(Ws) = -2.7882 + 1.1324 \ln(SB)$	0.59	27.99	p<0.001

Acacia bussei. Stem circumference at the base (*SB*) alone was an effective predictor variable for estimating total dry AGB, stem biomass and branch biomass (Table 2.2). Models were highly significant for all tree AGB components ($p < 0.001$), with $adj r^2$ values of 0.934–0.958 (Appendix 2.1). Model fit was comparable for all components of AGB, with minimal bias across the entire range of *SB*. In addition, regressing each plant dendrometric measurement alone against total dry AGB was fairly suitable for the indirect determination of dry AGB. Height *H* had the poorest fit ($adj r^2 = 0.819$) compared to canopy parameters *CV* ($adj r^2 = 0.922$), *CA* ($adj r^2 = 0.903$) and the stem circumference *SB* ($adj r^2 = 0.958$); Fig. 2.1).

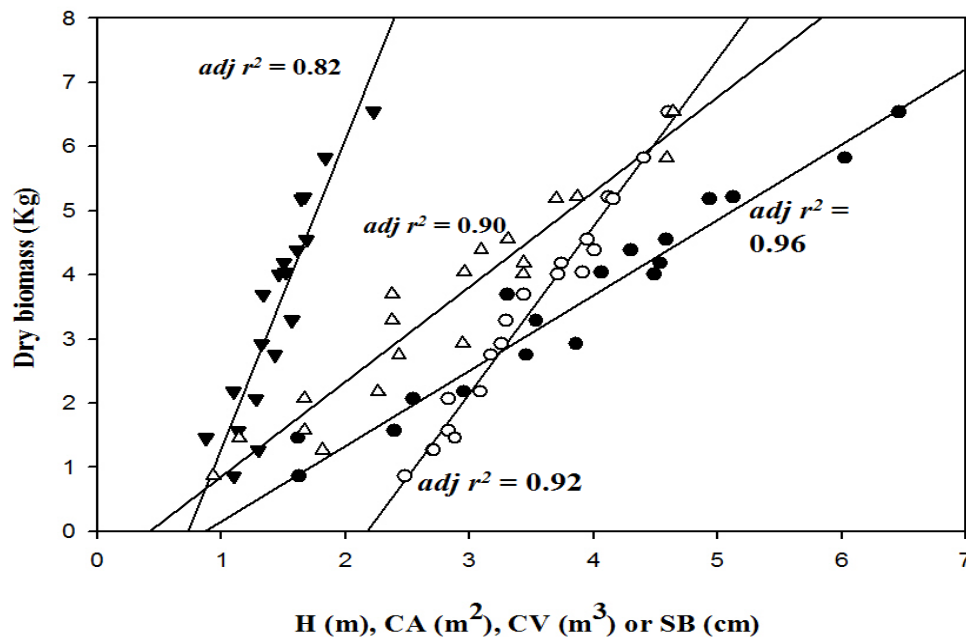


Fig 2.1. Natural log (ln) linear regression lines of total dry biomass versus height (*H*, filled triangles), crown area (*CA*, open triangles), crown volume (*CV*, open circles) and circumference of the stem at ankle height (*SB*, filled circles) for *Acacia bussei*.

Acacia mellifera. A combination of *SB* and *CV* was the most effective predictor for all AGB components (Table 2.2). Models were highly significant for the different AGB components, with $adj r^2$ values of 0.924–0.955 (Appendix 2.1). The fitted multiple regression model using both *SB* and *CV* as predictor variables significantly improved the accuracy of the predictability of the AGB compared to simple linear models obtained from *SB* ($adj r^2 = 0.881$), *CA* ($adj r^2 = 0.892$), *CV* ($adj r^2 = 0.936$) and *H* ($adj r^2 = 0.703$) only. Model fit was better for “branches” and “total” AGB than for “stem” biomass (Appendix 2.1). Further, residual plots (data not shown)

demonstrated that there was little bias towards over –or under estimation of biomass for both smaller and larger predictor values.

***Acacia etabaica*.** The circumference of the stem at ankle height (5-10cm aboveground) height (*SB*) accurately predicted all components of AGB with $adj\ r^2$ values ranging from 0.94 to 0.96 ($p < 0.01$; Table 2.2) while regressions of other predictors achieved lower fits, i.e. *H* ($adj\ r^2 = 0.867$), *CV* ($adj\ r^2 = 0.855$) and *CA* ($adj\ r^2 = 0.797$; Fig. 2. 2). Residual plots also demonstrated that there was no large or systematic bias toward over-or underestimation of biomass at any *SB* within the range used to develop the models.

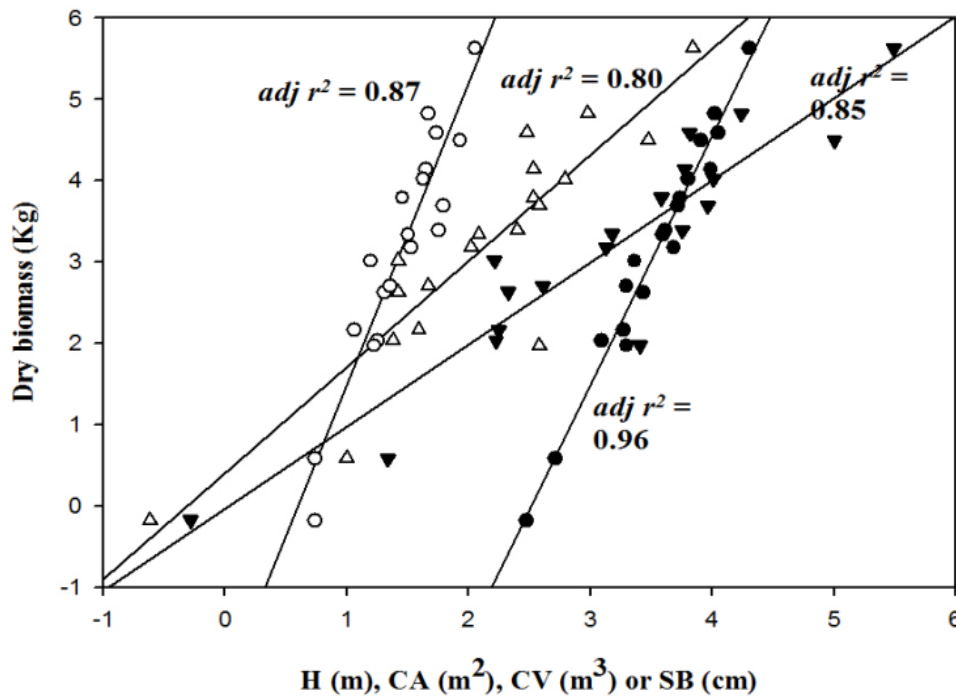


Fig 2.2. Natural log (ln) linear regression lines of total dry biomass versus height (*H*, open circles), crown area (*CA*, open triangles), crown volume (*CV*, filled triangles) and stem base circumference (*SB*, filled circles) for *Acacia etabaica*

***Acacia senegal*.** The combination of *SB* and *CV* as predictors for the different AGB components was with $adj\ r^2 = 0.821$ higher than using the single predictors of *SB* ($adj\ r^2 = 0.746$), *CV* ($adj\ r^2 = 0.706$) or *CA* ($adj\ r^2 = 0.741$). Height *H* was unsuitable for prediction of total AGB ($adj\ r^2 = 0.106$). The regression models were highly significant for all AGB components ($p < 0.001$), with $adj\ r^2$ values of 0.76 to 0.90. Model fit was better for stem and total AGB than for branch biomass (Appendix 2.1).

***Acacia drepanolobium*.** The combination of *SB* and *H* explained most of the variability for total AGB, stem and branch biomass (Table 2.2). Models were significant for all AGB categories ($p < 0.01$), with $adj\ r^2$ values of 0.75–0.84 (Table 2.2). The regression of total AGB against a combination of *H* and *SB* showed higher $adj\ r^2$ values (0.84) compared to any $adj\ r^2$ values of a single use of *H* ($adj\ r^2 = 0.799$), *CV* ($adj\ r^2 = 0.775$), *CA* ($adj\ r^2 = 0.677$) and *SB* ($adj\ r^2 = 0.75$). Model fit was better for total AGB and branch biomass than for stem biomass (Appendix 2.1).

***Acacia oerfota*.** Canopy volume (*CV*) alone was a good predictor variable for estimating AGB, stem and branches in *A. oerfota* (Table 2.2) with $adj\ r^2$ values ranging from 0.77 to 0.85 ($p < 0.001$ for all models; Table 2.2). The regression of total AGB against *CV* explained the maximum variation in total AGB as shown by higher $adj\ r^2$ values (0.84) compared to *CA* ($adj\ r^2 = 0.69$) and *H* ($adj\ r^2 = 0.322$; Fig. 2.3).

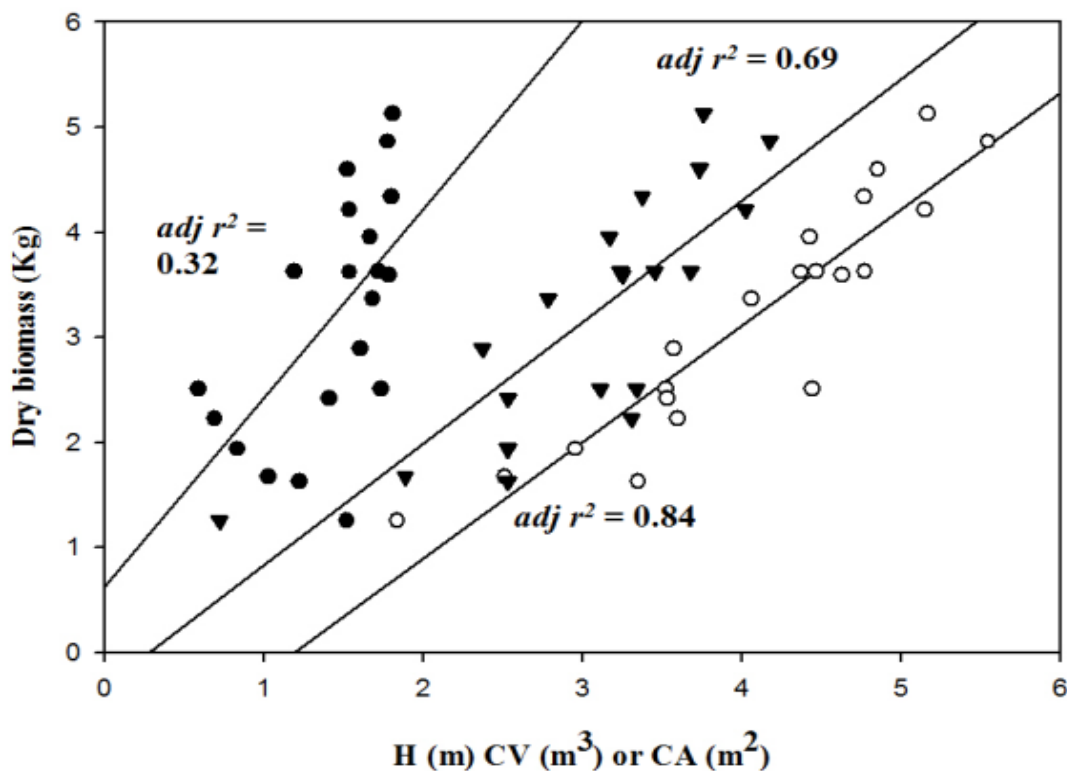


Fig 2.3. Natural log (ln) regression lines of total dry biomass versus crown volume (CV, open circles), crown area (CA, filled triangles) and height (H, filled circles) for *A. oerfota*

Acacia reficiens. Canopy volume (CV) accurately predicted branches and total AGB in *A. reficiens* with $adj\ r^2$ values ranging from 0.83 to 0.88 ($p < 0.01$ for all models; Table 2.2). Total AGB and of the stem was more reliably predicted compared to other components, with minimal bias across the entire range of CV and a better goodness of fit (Appendix 2.1). The variation in total AGB was better explained by CV as shown by higher $adj\ r^2$ values (0.88) compared to CA ($adj\ r^2 = 0.66$) and H ($adj\ r^2 = 0.556$; Fig. 2.4).

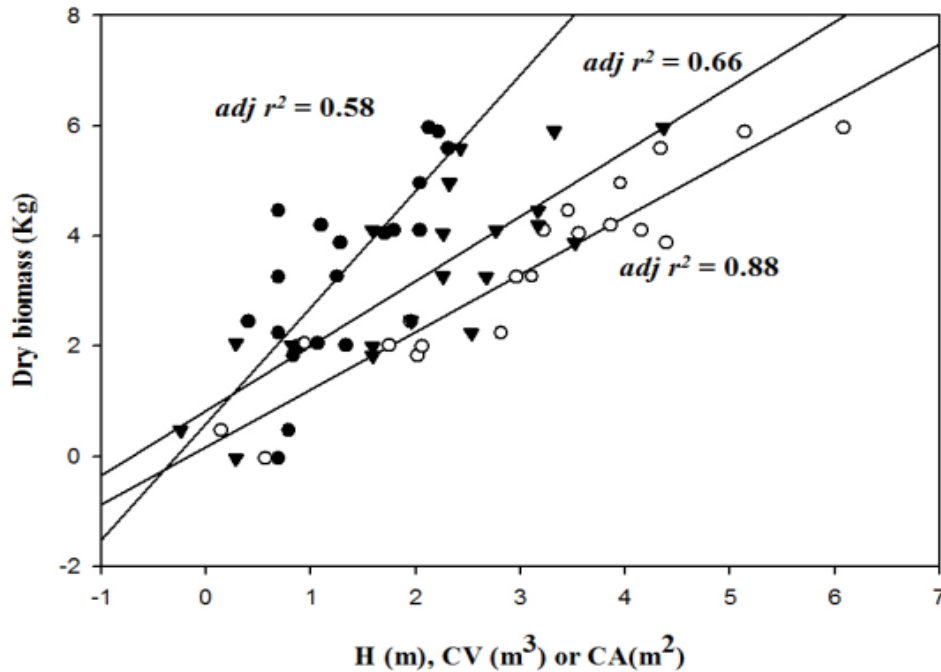


Fig 2.4. Natural log (ln) regression lines of total dry biomass versus crown volume (CV, open circles), crown area (CA, filled triangles) and height (H, filled circles) for *A. reficiens*

Commiphora africana. A combination of CV and SB was most effective for estimating total dry AGB ($adj\ r^2 = 0.924$) compared to a single use of H ($adj\ r^2 = 0.579$), CV ($adj\ r^2 = 0.579$), CA ($adj\ r^2 = 0.579$) and SB ($adj\ r^2 = 0.579$). Biomass of branches and stem of *C. africana* was less predictable compared to total AGB. Models relating branches and stem to any of the measured predictor variables presented lower goodness of fit compared to total AGB. Further, the predicted model fitted less well when the branch biomass was estimated, suggesting that this component of biomass was less accurately predictable (Appendix 2.1).

2.4. Discussion

A systematic screening of the regression models for estimating AGB and woody plant components showed that SB, CV, H and a combination of these variables could be used as predictor variables. The allometric equations were relatively satisfactory for predicting total AGB since total variation explained by the relationship was above 80% (Appendix 2.1). Actual model performance, expressed as a goodness of fit ($adj\ r^2$) depended on both species involved and the biomass component to be estimated (Appendix 2.1). The relationship was much stronger for larger woody species such as *A. mellifera*, *A. bussei*, and *A. etabaica* (Table 2. 1; Appendix 2.1). Allometric equations being strongly different for different tree species within the same climatic zones have been reported previously and mainly attributed to differences in specific wood gravity (weight per volume) of the species, the floristic composition and growth strategies of the species (Abola et al., 2005). In general, there is variability of basic wood density among species, individual of the same species, among geographical location and with age (Abola et al., 2005; Nygard and Elfying, 2000). Moreover, wood specific gravity of the species also differs among woody plant sections: it is higher at the base of the stem than the top of the stem and branches (Nygard and Elfying, 2000). Even though the models we presented here were based on harvested dry AGB of woody plants, precluding the need for estimates of specific wood gravity of each plant and its parts (Liton and Kauffman, 2008), in our study, we have shown that the biomass of branches and total AGB were predicted very well with relatively good precision compared with the stem biomass in most of the species studied except for *C. africana* (Appendix 2.1). This is in agreement with Navar et al., (2004) that the ability to predict the biomass of large woody components such as branches, constituting about 2/3 of total dry AGB in semi-arid or arid tree or shrub species, was as accurate as stem biomass. While species-specific and its components predictions were well fitted through our allometric equations, the growth form and location of a species needs to be taken into account (as seen by Henry et al., 2011), e.g. whether neighbouring trees compete for light and, thus, may reduce branch growth.

The allometric models we present for predicting total AGB, biomass in stem and branches of different species here rely on different predictor variable or variables combination (Appendix 2.1). Generally, our models indicated that the most important single or set of predictor variable/s were represented by be stem basal circumference/diameters for tall trees with more or less open canopy structure (Table 2.2). In contrast, pairs of canopy volume and stem

basal circumferences were more reliable for predictions of tall trees with closed and umbrella-like canopy structures (Appendix 2.1). We also showed that, even if canopy volume only is used as a predictor variable, biomass can still be significantly well predicted for shrubs whose growth form comprises discrete canopy clumps with multiple stems (e.g., *A. oerfota*; Fig. 2.3 and 2.4). This concurs with Hofstad (2005) observation that for shrubland woody plant species, a combination of stem and canopy related variables could improve the accuracy of AGB predictions substantially compared to diameter and height measures alone that have traditionally been used in the wet and humid ecosystems. In contrast, most of earlier models published in the tropical shrublands required the estimates of circumference or cross-sectional area of the stem at the base alone (Henry et al., 2011; Okello et al., 200; Poupon, 1979; Tietema, 1993a), stem diameter at breast height and total height of the woody plant (Cissé, 1980; Sawadogo et al., 2010) while only few authors (e.g., Tietema, 1993b) combined stem and canopy related measurements as predictor variables. However, we observed that the biomass of woody plant species was significantly well predicted from dendrometric variable/s which is directly related with the parts of the plant which constitutes relatively larger proportion of dry biomass which in turn governed by the inherent growth form and plant architecture of the species (e.g., *A. bussei* and *A. etabaica* from *SB*; *A. oerfota* from *CV*; and *A. mellifera* and *A. senegal* from *CV* and *SB*). This shows that the species which had larger proportions of their biomass allocated in their canopies, allowing the inclusion of crown volume, width or height as reliable predictor variable in the model, which might propose that the inclusion of canopy measurements will provide a more accurate predictability in the model of bushy species (e.g., *A. oerfota*, *A. mellifera* and *A. senegal*). We also emphasize here that developing a generalized allometric model for a group of woody species with similar growth forms and functional types could overcome the difficulty of species-specific model requirements for biomass and C stock estimations. This consideration is particularly crucial in tropical ecosystems with diverse species, growth forms and functional types as it integrates part of the variability related to the plant architecture and wood gravity (Henry et al., 2011).

Our study provides a unique data set on biomass estimates for lowland woody species of eastern Africa. When comparing our results to studies conducted in Western and Southern Africa, the species-specific allometric equations for *A. mellifera* and generalized shrubland equations by (Tietema, 1993a, 1993b) provide by 32% and 48% lower total dry biomass values,

respectively. When using only stem base circumference (Tietema, 1993a), predictions were less accurate than when canopy measurements were included (Tietema, 1993b) to predict the biomass of *A. mellifera*. This implies the importance of canopy related measurements for precise and accurate estimation of biomass and C stocks in this species. Similarly, the total dry AGB we predicted for *A. senegal* individuals differed from predictions by Smektala et al. (2002), Poupon(1979) and Tietema (1993a) in western Africa by 42%, 43%, and 21%, respectively. Henry et al. (2011) also reported that allometric equations developed for *A. senegal* in Somalia (Bird and Shepherd, 1989) provided by about 90% and 21% higher estimates of the total AGB predicted using allometric equations developed by Smektala et al., (2002), and Poupon (1979), respectively. This shows how strongly biomass estimates can differ regionally and, accordingly, influence the the C-stock estimation attempts. Hence, our estimates contribute to the thus far unknown biomass estimates of the eastern African varieties of Acacia species (Table 2. 1). *A. drepanolobium* allometric equations in Kenya (Okello et al., 2001), using stem basal area as predictor variable, underestimated the AGB by 32% compared to our study, which could be due to the difference in the predictor variables and tree age and size used. Our predictor variable being *SB and H* and this species tree size range particularly for these predictor variables used as seen in (Appendix 2.1). The existing allometric equation of *C. africana* (Poupon, 1979), conducted in Cameroon, in west Africa showed a biomass estimate that was 42 % higher compared to our equation. This difference could be caused by size and age characteristics of the *C. africana* population structure. Our sampled *C. africana* individuals were rather young and small in size (the average height of the individual in the population of *C. africana* in the study area was less than 1.5 m; full data not shown), i.e., in its early phase of woody species succession in the grassland, indicating that our allometric functions for *C. africana* could not be applied to systems dominated by older and larger *C. africana* individuals.

Besides using different predictor variables and variations of wood gravity in the different woody species and/or growth forms and its different parts within and across ecological zones, a significant part of the observed differences across modeled equations can also be explained by the variability of ecological and human impacts that determine the growth characteristics of a woody plant. For example, it has been shown that reduced water stress can allow for taller plant growth and biomass production (Brown et al., 1997). Moreover, soil properties also have been reported to affect woody plant biomass, e.g. lower biomass of the same tree species were

obtained in highly acidic and less deep soils in the humid tropics compared to less acidic and deep soils of the wet tropic climatic zone (Brown et al., 1997). Therefore, to predict woody plant biomass reliably, species- and site specific equations are of high importance, if carbon and nutrient storage or wood yields need to be assessed.

Traditionally, species-specific models once developed can be used across sites with varying climatic characteristics (Abola et al., 2005). However, our study and the model comparisons indicated that care should be taken in applying one single model to estimate biomass across the entire climatic gradients in which the species are found. In addition, extrapolating beyond the data range of the predictor variables (see Appendix 2.1) used in these models development may cause bias when estimating, for instance, the biomass for tall trees. This is problematic because the tallest trees at a given site can account for most of the biomass (Brown and Lugo, 1984). Very importantly, the closet conformity of our models to other models in Eastern Africa (e.g., Bird and Shepherd, 1989) in contrast to models from West and South Africa indicates that our biomass predictions are adequate and can potentially be used across arid and semi-arid climatic zones of the East African ecosystem. We suggest that the most reliable way to determine whether the models can appropriately be used at a given site is to analyze the size structure for randomly selected individuals and then compare these values to the ones we used for our model development (Appendix 2.1). Therefore, our model could be applied to woody plants of the same species within the range of the structural characteristics measured.

Rangelands usually carry natural or semi-natural vegetation that provides a habitat suitable for herds of a wide variety of wild or domestic ungulates (Homewood, 2004). The vegetation of Eastern African rangelands is largely composed of desert and semi-desert (26 % of the land surface), bushland (33 %), woodland (21 %), and pure grassland (7 %) (White, 1983). Although pure grassland is found only in Central and South-Eastern Sudan, Northern and Western Tanzania and Northwest Kenya, the herbaceous layer of semi-deserts, bushlands, and woodlands are dominated by grasses and forbs. The pure grasslands or grasslands areas with varying amounts of woody vegetation constituted 75% of eastern African land surface (Reid et al., 2005). Naturally, these woody components of the vegetation are dominated by *Acacia* species which are an integral feature of Eastern African vegetation, mainly of the arid and semiarid savanna regions (White, 1983). Furthermore, some of the native woody species, including the genus *Acacia*, have been progressively increasing in many parts of African

savannas. For example, in Ethiopia, *A. drepanolobium*, *A. mellifera*, *A. senegal*, *A. reficiens*, *A. oerfeta*, and *Commiphora* species have been identified as heavy encroachers (Angassa and Oba, 2008; Abule et al., 2007; Gemedo-Dalle et al., 2006; Oba et al., 2000; Coppock, 1994). In Kenya, *A. drepanolobium* (Okello et al., 2001), in Tanzania, *A. auriculiformis* (Kotiluoto et al., 2009), in Botswana, *A. tortilis*, *A. erubescens*, *A. mellifera*, *Dichrostachys cinerea*, *Grewia flava*, and *Terminalia sericea* (Moleele et al., 2002) and in Namibia, *A. mellifera* and *A. reficiens* are among the main encroacher species (Joubert et al., 2008). The reason for this woody plant encroachment in the grassland systems is equivocally established, but possible factors contributing to woody plant encroachment include changing land use practices such as high stocking rates and associated heavy grazing (Bassett and Zuéli, 2001), altered burning practices (Bassett and Zuéli, 2001), exclusion from fire and grazing (Oba et al., 2000), changing climate and rainfall (Sankaran et al., 2005), atmospheric nitrogen deposition (Brown and Archer, 1989) and elevated Carbon dioxide (CO₂) (Wigley et al., 2010).

While this generally has a negative effect on the available grass resources in rangelands (Angassa and Oba, 2008; Abule et al., 2007; Gemedo-Dalle et al., 2006; Oba et al., 2000; Coppock, 1994) the emergence of markets for climate change mitigation on the other hand presents new opportunities for increasing the economic and ecological returns to rangelands (IPCC, 2007). The fourth International Panel on Climate Changes indicates that rangelands have the second highest technical potential for mitigating C emission from agricultural management changes, at over 1400 Mt CO₂ equivalents per year by 2030 (IPCC, 2007). One means of reducing net C emissions is to increase the rate of carbon sequestered in terrestrial ecosystems, including in soils, below- and aboveground biomass (IPCC, 2007). Accordingly, several authors showed the potential significant net increase in C stocks and C sequestration potential due to woody plant proliferations in the rangelands and savanna ecosystem, emphasizing the importance of integrating this potential to sustainable rangeland management practices and livestock production systems (Hibbard et al., 2003; Houghton, 2003; Gifford and Howden, 2001; San Jose et al., 1998). Allometric equations have become a common tool needed to quantify carbon stocks and changes in woody components of different vegetation types including the net gains in carbon sequestration potentials due to recent woody species encroachment in the rangelands (Henry et al., 2011; Asner et al., 2003).

The current established allometric equations may allow a rapid estimate of available potential biomass and C stock in the woody component of the Borana rangelands and aid in a sustainable management of the rangelands for both economic and environmental services. Circumference or the stem diameter is the most common predictor in many biomass allometric models (Henry et al., 2011) in tropical shrublands, but adding crown volume or a combination of the components of crown volume (i.e. widths or circumference, and height) variables can significantly improve the prediction power (Vora, 1988). In addition, the fact that these variables can easily be measured in open savannas and rangelands could avoid practical problems and ensure future precise biomass estimations. Furthermore, Asner et al. (2003) demonstrated that the equations developed from crown dimensions can easily be integrated with a high resolution remote sensing analyses to estimate biomass and C-stock over large spatial and temporal scale in the open forest or shrublands.

2.5. Conclusion

The species-specific allometric equations we present for the eight most widespread woody plant species in the Borana rangelands will significantly improve our capacity to accurately estimate biomass and corresponding carbon stocks in East African savanna systems. In particular, we highlight several dendrometric variables based on the architecture and growth form of each woody plant species that are essential predictors and simple tools for accurate equation development and for biomass and carbon stock estimates. In addition, our models can be used to predict AGB in branches and stems separately. Estimating biomass by component is essential for studies of carbon sequestration and fuelwood dynamics, as branches and stem wood have different carbon allocation and flux patterns as well as fuel wood characteristics (Litton and Kauffman, 2008). While our research provides unique data on important dominant woody plant encroacher species, further validation of the models developed here across different sites and land use systems within arid and semi-arid ecosystems is important for a wider application of our equations to biomass and carbon sequestration studies. In addition, care should be taken in applying the allometric equations developed in this study to other sites without knowledge of site characteristic and woody plants size structures. We recommend that the individual woody plant size structure of each species in the area of interest be evaluated and compared to the values presented in this study to determine how appropriate the allometric equations are for the study sites in focus.

2.6. References

- Abola, J. R., Arvalo, J. R., Fernandez, A. 2005. Allometric relationships of different tree species and stand aboveground biomass in the Gomera laurel forest (Canary Islands). *Flora* 2000, 264 - 274.
- Abule, E., Snyman, H. A., Smit, G. N. 2007. Rangeland evaluation in the middle Awash valley Ethiopia: II. Woody Vegetation. *Journal of Arid Environment* 70, 272 – 292.
- Angassa, A., Oba, G., 2008. Herder perceptions on impacts of range enclosures, crop farming, fire ban and bush encroachment on the rangelands of Borana, southern Ethiopia. *Human Ecology* 36, 201–215.
- Asner, G.P., Archer. S.R., Hughes, R.F., *et al.*, 2003. Net changes in regional woody vegetation cover and carbon storage in Texas drylands, 1937–1999. *Global Change Biology* 9, 316–335.
- Bassett, T.J., Zuéli, K.B., 2000. Environmental discourses and the Ivorian Savanna. *Annals of the Association of American Geographers* 90, 67–95.
- Bellefontaine, R., Gaston, A., Petrucci, Y., 2000. Management of natural forests of dry tropical zones. Food and Agricultural organization of the United Nations, Rome, Italy.
- Bird, N.M., Shepherd, G., 1989. Charcoal in Somalia: a woodfuel inventory in the Bay region of Somalia. British Forestry Project Somalia Overseas Development Administration, National Range Agency, Ministry of Livestock, Forestry and Range.
- Briggs, J.M., Knapp, A.K., Blair, J.M., Hilster, J.L., Hoch, G.A., 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. *BioScience* 55, 243–254.
- Brown, J.R., Archer, S., 1989. Woody plant invasion of grasslands: establishment of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) on sites differing in herbaceous biomass and grazing history. *Oecologia* 80, 19–26.
- Brown, S., Schroeder, P., Birdsey, R., 1997. Aboveground biomass distribution of US eastern hardwood forests and the use of large trees as an indicator of forest development. *Forest Ecology and Management* 96, 37–47.
- Brown, S., Lugo, A.E., 1984. Biomass of tropical forests: A new estimate based on forest volumes. *Science* 223, 1290–1293.
- Cissé, M.I., 1980. The browse production of some trees of the Sahel: relationships between, maximum foliage biomass and various physical parameters. In: Le Houérou, H.N. (Ed.), *Browse in Africa*. International Livestock Center for Africa, ILCA. Addis Ababa, pp. 205e210.
- Cole, T.G., Ewel, J.J., 2006. Allometric equations for four valuable tropical tree species. *Forest Ecology and Management* 229, 351-360.

Coppock, D.L., 1994. The Borana Plateau of Southern Ethiopia: Synthesis of Pastoral Research, Development and Changes 1980–1990. System study No. 5. International Livestock Centre for Africa, Addis Ababa, 374pp.

Gayon, J., 2000. History of the Concept of Allometry. *American Zoologist* 40, 748.

Gemedo-Dalle, Y., Maass, B.L., Isselstein, J., 2006. Encroachment of woody plants and its impact on pastoral livestock production in the Borana lowlands, Southern Oromia, Ethiopia. *African Journal of Ecology* 44, 113–299.

Gemedo-Dalle, Y., Brigitte L. M., Johannes I., 2005. Plant biodiversity and ethnobotany of Borana pastoralists in Southern Oromia, Ethiopia. *Economic Botany* 59, 43-65.

Gifford, R.M., Howden, S.M., 2001. Vegetation thickening in an ecological perspective: significance to national greenhouse gas inventories and mitigation policies. *Environmental Science and Policy* 4, 59–72.

Grundy, I.M., 1995. Woody biomass estimation in dry biombo woodland in Zimbabwe. *Forest Ecology and Management* 72, 109-117.

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

Hibbard, K. A., Schimel, D.S, Archer, S. R., Ojima, D.S., Parton, W., 2003. Grassland to woodland transitions: integrating changes in landscape structure and biogeochemistry. *Ecological Applications* 13, 911–926.

Hofstad, O., 2005. Review of biomass and volume functions for individual trees and shrubs in southeast Africa. *Journal of Tropical Forest Science* 17, 151–162.

Homewood, K. 2004. Policy, environment and development in African rangelands. *Environmental Science and Policy* 7, 125–143.

Houghton, R. A., 2003. Why are estimates of the terrestrial carbon balance so different? *Global Change Biology* 9, 500–509.

Hughes, R. F., Archer, S. R., Asner, G. P., Wessman, C. A., Mcmurtry, C., Nelson, J., Ansley, R. J., 2006. Changes in aboveground primary production and carbon and nitrogen pools accompanying woody plant encroachment in a temperate savanna. *Global Change Biology* 12, 1733–1747.

(IPCC) Intergovernmental Panel on Climate Change 2007. Agriculture. In: Climate change 2007. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental

Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. pp. 499–540.

Joubert, D.F., Rothauge, A., Smit, G.N., 2008. A conceptual model of vegetation dynamics in the semiarid highland savanna of Namibia, with particular reference to bush thickening by *Acacia mellifera*. *Journal of Arid Environment* 72, 2201-2210.

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

Kotiluoto, R., Ruokolainen, K., Kettunen, M., 2009. Invasive *Acacia auriculiformis* Benth. in different habitats in Unguja, Zanzibar. *African Journal Ecology* 47, 77-86.

Lett, M.S., Knapp, A.K., Briggs, J.M., Blair, J.M., 2004. Influence of shrub encroachment on aboveground net primary productivity and carbon and nitrogen pools in a mesic grassland. *Canadian Journal Botany* 82, 1363–1370.

Litton, C.M., Kauffman, J.B., 2008. Allometric models for predicting above-ground biomass in two widespread woody plants in Hawaii. *Biotropica* 40, 313-320.

Moleele, N.M., Ringrose, S., Matheson, W., Vanderpost, C., 2002. More woody plants? The status of bush encroachment in Botswana's grazing areas. *Journal of Environmental Management* 64, 3-11.

Munyati, C., Shaker, P., Phasha, M.G., 2010. Using remotely sensed imagery to monitor savanna rangeland deterioration through woody plant proliferation: a case study from communal and biodiversity conservation rangeland sites in Mokopane, South Africa. *Environ Monit Assess* 10, 1583–1584.

Murray, R.B., Jacobson, M.Q., 1982. An evaluation of dimension analysis for predicting shrub biomass. *Journal of Range Management* 35, 451–454.

Nàvar, J., Gonzalez, N., Maldonado, D., Graciano, J., Dale, V., Parresol, B., 2004. Biomass equations for shrub species of Tamaulipan thornscrub of North-eastern Mexico. *Journal of Arid Environments* 59, 657–674.

Navar, J., Nàjera, J., Jurado, E., 2002. Biomass estimation equations in the Tamaulipan thornscrub of north-eastern Mexico. *Journal of Arid Environment* 52, 167.

Nygaard, R., Elfyng, B., 2000. Stem basic density and bark proportion of 45 woody species in Young savanna coppice forests in Burkina Faso. *Annals of Forestry Science* 57, 143-153.

Okello, B.D., O'connor, T.G., Young, T.P., 2001. Growth, biomass estimates, and charcoal production of *Acacia drepanolobium* in Laikipia, Kenya. *Forest Ecology and Management* 142, 143-153.

Oba, G., Post, E., Syvertsen, P.O., Stenseth, N.C., 2000. Bush cover and range condition assessments in relation to landscape and grazing in southern Ethiopia. *Landscape Ecology* 15, 535–546.

(OPADC) Oromiya Pastoral area Development Commission 2011. Borana land use study project soil survey report for Dawa sub-basin.

Poupon, H., 1979. Structure et dynamique de la strate ligneuse d'une steppe sahélienne au nord du Sénégal. Univ. Paris-Sud, Centre d'Orsay et Ed. ORSTOM, Paris.

Payton, I. J., Newell, C.L., Beets, P.N., 2004. New Zealand Carbon Monitoring System indigenous forest and shrubland data collection manual. Caxton Press, Christchurch, New Zealand.

Reid, R. S., Serneels, S., Nyabenge, M., and Hanson, J., 2005. The changing faces of pastoral systems in grass-dominated ecosystem of eastern Africa, in: Suttie, J. M., Reynold S. G., Batello, C. (eds.), Grasslands of the world. FAO, Rome, pp. 19-76.

Sankaran, M., Hanan, N.P., Scholes, R.J., Ratnam, J., Augustine, D. J., Cade, B. S., Higgins, G. J., Roux, X. L., Ludwig, F., Ardo, J., Banyikwa, F., Bronn, A., Bucini, G., Caylor, K.K., Coughenour, M. B., Diouf, A., Ekaya, W., Feral, C.J., February, E.C., Frost, P.G.H., Hiernaux, P., Hrabar, H., Metzger, K.L., Prins, H.H.T., Ringrose, S., Sea, W., Tews, T., Worden, J., Zambatis, N., 2005. Determinants of woody cover in African savannas. *Nature* 438, 846–849.

San Jose, J.J., Montes, R.A., Farinas, M.R., 1998. Carbon stocks and fluxes in a temporal scaling from a savanna to a semi-deciduous forest. *Forest Ecology and Management* 105, 251–262.

Sawadogo, L., Savadogo, P., Tiveau, D., Dayamba, S. D., Zida, D., Nouvellet, Y., Oden, P.C., Guinko, S., 2010. Allometric prediction of aboveground biomass of eleven woody tree species in Sudanian savanna woodland of West Africa. *Journal of Forestry Research* 21, 475-481

Smektala, G. Hautdidier, B., Gautier, D., Peltier, R., Njemoun, A., 2002. Constitution de Tariffs de biomasse pour l'évaluation de la disponibilité ligneuse de savanes au nord Cameroun Actes du Colloque, 271-31 mai 2002, Garulla Cameroun

Sprugel, D.G., 1983. Correcting for bias in log-transformed allometric equations. *Ecology* 64, 209–210.

Tabachnick, B.G., Fidell, L. S., 1996. Using Multivariate statistics. Newyork, Haper Collins College Publishers.

Tietema, T., 1993a. Biomass determination of fuelwood trees and Bushes of Botswana, South Africa. *Forest Ecology and Management* 60, 257-269.

Tietema, T., 1993b. Possibilities for the management of indigenous woodlands in Southern Africa: A case study from Botswana, in: Pierce, G.D., Gumbo, D.J., (Eds.), *The Ecology and Management of Indigenous Forest in Southern Africa*. Zimbabwe Forestry Commission and SAREC, Harare, pp. 134-142

Vacher, H.L., 1999. Computational geology 8: the power function. *Journal of Geoscience* Edition 47, 473-481.

Vora, R.S., 1988. Predicting biomass of 5 shrub species in northeastern California. *Journal of Range Management* 41, 63-65.

White, F., 1983. *The vegetation of Africa : a descriptive memoir to accompany the Unesco/AETFAT/UNSO vegetation map of Africa*. Paris, UNESCO.

Whittaker, R.H., Woodwell, G.M., 1968. Dimension and production relations of trees and shrubs in the Brookhaven Forest, New York. *Journal of Ecology* 56, 1-25.

Wigley, B.J., Bond, J., Hoffman, M.T., 2010. Thicket expansion in a South African savanna under divergent land use: local vs. global drivers? *Global Change Biology* 16, 964-976.

Willebrand, E., Ledin, S., Verwijst, T., 1993. Willow coppice systems in short rotation forestry: effects of plant spacing, rotation length and clonal composition on biomass production. *Biomass and Energy* 4, 323-331.

Appendix 2.1. The aboveground biomass regression equations for eight woody tree/shrub species. Total aboveground, stem, and branches + leafy shoots dry biomasses were estimated based on destructive sampling in the semi-arid Borana rangelands. Note: *CV* = the ellipsoid crown volume, *SB* = the circumference at base of the stem, α = the slope parameter, β = the intercept parameter, MSE = the mean standard error of the estimate, CF = Correction factor, r^2 = the coefficient of determination, p = the p-value ($\alpha=0.0001$) nf = no fit possible. Only the best predictors and/or their combination are shown.

Species	Dry biomass component	Predictor Variable	Predictor range	α (SE)	β (SE)	MSE	CF	Adj r^2	p
<i>Acacia bussei</i>	Total	SB	11.9 - 99.8 cm	2.5838(0.1238)	-5.6305(0.4410)	0.1063	1.89	0.96	<0.0001
	Stem	SB	11.9 - 99.8 cm	2.4056(0.1465)	-6.5502(0.5218)	0.1488	1.86	0.93	<0.0001
	Branches	SB	11.9 - 99.8 cm	2.6523(0.1333)	-6.1253(0.4750)	0.1233	1.91	0.95	<0.0001
<i>Acacia mellifera</i>	Total	SB	11.0 - 86.4 cm	0.9629(0.3263)	-2.7777(0.6410)	0.121	1.82	0.96	<0.0001
		CV	1.8 - 1408.6 m ³	0.7502(0.1353)					
	Stem	SB	11.0 - 86.4 cm	0.7084(0.3892)	-3.5960(0.7646)	0.1721	1.85	0.92	<0.0001
		CV	1.8 - 1408.6 m ³	0.7462(0.1614)					
	Branch	SB	11.0 - 86.4cm	0.9991(0.3345)	-3.1639(0.6572)	0.1272	1.82	0.96	<0.0001
		CV	1.8 -1408.6 m ³	0.7625(0.1387)					
<i>Acacia etabaica</i>	Total	SB	11.9 - 97.1 cm	2.8769(0.1395)	-7.0821(0.5066)	0.0942	1.83	0.96	<0.0001
	Stem	SB	11.9 - 97.1 cm	2.9514(0.1756)	-8.5593(0.6378)	0.1493	1.80	0.94	<0.0001
	Branch	SB	11.9 - 97.1 cm	2.8475(0.1680)	-7.3611(0.6103)	0.1367	1.84	0.94	<0.0001
<i>Acacia senegal</i>	Total	SB	4.1 -52.1 cm	1.1599(0.3277)	-3.0190(0.6967)	0.3017	1.71	0.82	<0.0001
		CV	5.1 - 154.7 m ³	0.5922(0.2052)					
	Stem	SB	4.1 - 52.1 cm	1.4314(0.2219)	-4.2728(0.4717)	0.1383	1.80	0.90	<0.0001
		CV	5.1 - 154.7 m ³	0.3349(0.1389)					
	Branch	SB	4.1 - 52.1 cm	1.0684(0.4028)	-3.4604(0.8564)	0.4558	1.69	0.76	<0.0001
		CV	5.1 - 154.7 m ³	0.6967(0.2522)					
<i>Acacia drepanolobium</i>	Total	SB	7.9 - 68.2 cm	1.1295(0.3133)	-3.3387(0.7125)	0.1716	1.88	0.84	<0.0001
		H	3.4 - 9.5 cm	1.7011(0.5553)					
	Stem	SB	7.9 - 68.2 cm	0.8113(0.3182)	-2.6987(0.7236)	0.1770	1.84	0.72	<0.0001
		H	3.4 - 9.5 cm	1.2066(0.5640)					
	Branch	SB	7.9 - 68.2 cm	1.3193(0.3745)	-4.6227(0.8517)	0.2452	1.89	0.82	<0.0001
		H	3.4 - 9.5 cm	1.8487(0.6639)					
<i>Acacia oerfota</i>	Total	CV	6.3 - 255.8 m ³	1.1084(0.1099)	-1.320(0.4596)	0.2077	1.71	0.85	<0.0001
	Stem	CV	6.3 - 255.8 m ³	1.2825(0.1607)	-3.4689(0.6723)	0.4445	1.80	0.77	<0.0001
	Branch	CV	6.3 - 255.8 m ³	1.0726(0.1057)	-1.4773(0.4421)	0.1922	1.69	0.84	<0.0001

<i>Acacia reficiens</i>	Total	CV	1.2 - 439.8 m ³	0.0872(0.0872)	0.1774(0.0872)	0.3369	1.71	0.88	<0.0001
	Stem	H	1.5 - 10.1 cm	2.0115(0.2983)	-0.1968(0.4259)	0.6315	1.80	0.70	<0.0001
	Branch	CV	1.2 - 439.8 m ³	1.0438(0.1081)	-0.2356(0.3644)	0.7189	1.69	0.84	<0.0001
<i>Commiphora africana</i>	Total	SB	11.9 - 39.9 cm	1.1323(0.1563)	-2.7881(0.4079)	0.0255	1.71	0.93	<0.0001
		CV	1.0 - 22.6 m ³	0.3163(0.0683)					
	Stem	SB	11.9 - 39.9 cm	1.1023(0.2083)	-2.5611(0.6691)	0.098	1.80	0.59	<0.0001
	Branches	nf		nf	nf	nf	nf	nf	nf

3 Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on aboveground C stock density

Submitted to: Journal of Carbon Balance and Management on 19.11.2013

Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on aboveground C stock density

M. Hasen-Yusuf^{a*}, A.C. Treydte^a, and J. Sauerborn^a

^a Agroecology in the Tropics and Subtropics, University of Hohenheim, 70599 Stuttgart, Germany

Telephone: 49-711-459 23602

*corresponding author email: hasanyusuf12@gmail.com

Abstract

Background

The widespread phenomenon of woody encroachment in savannas can strongly alter carbon (C) sequestration potentials in the long term, which could have regional and even global climate change mitigation implications. Carbon storage could represent an interesting ecosystem service for mitigation of global climate change and thereby payment for C sequestration could potentially diversify the livelihood options and increase the adaptation potentials of the resource poor pastoral people to climate changes. While the study so far highlighted the potential of the dry savannas for C sequestration, to date very few empirical research work has been carried out to estimate the vegetation cover change processes contribution to this ecosystem C stocks. In this study we analysed long term vegetation cover changes and assessed the structure and C stocks of aboveground biomass at different woody encroachment levels and grazing regimes.

Results

Areas covered by shrub and tree savanna (open savanna types) declined from 45% to 9%, while heavily encroached areas (bushland thickets and bushed savanna) increased from 22% to 61% from the year 1976 to 2012. The total and the regenerative woody plants (< 2m height) were higher in low- to high woody encroachment levels but greatly reduced at severe woody encroachment. Estimated total aboveground C stocks did not significantly vary between grazing regimes but varied significantly among the woody encroachment levels, with total aboveground C stocks ranging from 2.3 Mg ha⁻¹ in the low encroachment level (LE) to 9 Mg ha⁻¹ in the severe encroachment level (SE). Enclosures contained 50% more herbaceous aboveground C stocks and had 47% higher woody plants density than openly accessible grazing land

Conclusion

Our results suggest that woody encroachment is an imminent and ongoing process in Borana rangelands. The observed large increases in aboveground C stocks that occurred through woody encroachment on landscape and regional scales will foster future C-trade discussions with respect to climate change mitigation and adaptation strategies. However, conserving woody biomass C stocks will threaten the sustainability of livestock production, particularly of those depending on grass rangelands, and this tradeoff must be quantified and considered in the future

Keywords: Borana; Ethiopia; savanna; climate change; enclosure; carbon stock; land cover changes; communal grazing land

3.1 Introduction

Savannas occupy 10 to 15% of terrestrial ecosystems (Scholes and Hall, 1996), supporting over 30% of the world's human population and hosting most of its rangeland, livestock and wild animal's biomass (Reid et al., 2005; Solbrig et al., 1991). Approximately 60% of east Africa is covered by savannas, comprising plant communities with a continuous herbaceous layer and a discontinuous woody stratum (Reid et al., 2005). Over the last century, African savannas have been encroached by woody species despite differing climate and management practices such as variable domestic herbivore stocking rates and fire regime (Wigley et al., 2010). Woody encroachment, which refers to an increase in cover, density and biomass of indigenous woody plant species, has been reported over much of the world's arid and semi-arid environments ('drylands') in the recent decades (Van Auken, 2009). Similar to other African savannas, density and cover of woody plants has increased substantially within the southern Ethiopian savanna rangelands (Angassa and Oba, 2008; Solomon et al., 2007; Dalle et al., 2006). The woody cover in the Borana rangelands, southern Ethiopia, for example, was less than 40% in the 1980s (Coppock, 1994) but has increased to 52% in the early 2000s (Dalle et al., 2006).

Factors contributing to this woody encroachment in Borana include the governmental fire prohibiting policy of the 1970s and high livestock grazing pressure (Angassa and Oba, 2008; Homann et al., 2008; Solomon et al., 2007; Dalle et al., 2006). Intensified grazing can decrease the vigor of grasses while favouring seed germination, establishment, and survival of woody plants (Brown and Archer, 1999). Secondly, animals may serve as dispersal agents for seeds of woody plants (Riginos and Young, 2007). Thirdly, intensive grazing may weaken the intensity of fires by reducing the herbaceous plants biomass burning fuel load, which further thrive the woody species (Riginos and Young, 2007; Scholes and Archer, 1997). The complete suppression of fire accelerated these encroachment processes and has allowed the woody species to thrive in the Borana rangelands (Angassa and Oba, 2008) whereas intense and frequent fires regularly maintain the woody species in a young state by 'top-killing' seedlings (Smit et al., 2010). When fires are rare events, woody plants can grow to a fire-resistant size (Brown and Archer, 1999).

Once woody plants form a dense stand with a canopy cover of more than 40 % and/or 2400 plants ha⁻¹ (Roques et al., 2001) their encroachment starts threatening the sustainability of pastoral subsistence (Angassa and Oba, 2008; Dalle et al., 2006; Oba et al., 2000) by affecting herbaceous productivity and cattle carrying capacity. Prior studies have documented strong negative impacts of encroaching woody plants on herbaceous species diversity and

productivity in the southern Ethiopian rangelands (Angassa and Oba, 2010; Oba et al., 2001; Oba et al., 2000). Although woody encroachment has been recognized as a major rangeland management issue in Borana rangelands, neither the rate nor the spatial extent have yet been systematically quantified on a landscape level.

Despite its negative effect on herbaceous productivity, a net increase in carbon (C) sequestration potentials was reported from many savanna and tropical grassland ecosystems undergoing woody encroachment (Daryanto et al., 2013; Briggs et al., 2005; Hibbard et al., 2003). While some authors have estimated the C stock and/or the net gain in C sequestration potentials through woody encroachment for South -and West African savanna systems (Woollen et al., 2012; Shackleton and Scholes, 2011), Australian (Daryanto et al., 2013) and tropical American savannas and grasslands (Knapp et al., 2008; Asner et al., 2003), to our knowledge, there is no similar quantitative estimate for southern Ethiopia rangelands. Plant biomass is a critical component within the C cycle that can provide both short- and long-term C sequestration. The biomass of woody plants in the system is influenced by age, species composition and size structure, and productivity of the entire vegetation stand (Litton et al., 2006; Briggs et al., 2005), which is further largely impacted by herbivores and humans utilization (Wessels et al., 2011). Therefore, understanding the effect of woody encroachment and land use (grazing and browsing by domestic livestock) on different components of ecosystem biomass (herbaceous, trees, and shrub plants) represents a first crucial step towards identifying the potential impacts that land use and vegetation cover changes have on C cycling. This information is also of high scientific significance globally as C stocks need to be quantified for currently developed global models on C sequestration and emissions (Shackleton and Scholes, 2011). Further, potential financial rewards have been suggested for sequestered C or for CO₂ emission reductions through appropriate management of terrestrial biomass in the savanna ecosystem (Daryanto et al., 2013; Woollen et al., 2012; Shackleton and Scholes, 2011).

Thus, in our study we wanted to quantify the expansion of woody encroachment in the Borana rangelands over the last four decades. We expected that expansion will positively impact the aboveground C (AGC) stocks but that the changes in the AGC stock across different ecosystem components depend upon the pastoral grazing land management i.e., grazing, browsing and woody plant utilization. Therefore, the objectives of this study were (i) to analyze the changes in vegetation cover in Borana rangelands from 1976 to 2012 at a landscape scale and (2) to compare woody plant population structure and C stocks in AGB in two grazing land management systems (communal grazing vs. enclosure) among four woody

encroachment levels. We used Landsat imagery to examine changes in woody encroachment and undertook field based research to analyse woody vegetation structure and derive site-based estimates of C pools. Woody plant biomass at each site was quantified using allometric biomass equations that had been developed for the most dominant woody species in the area by destructive sampling (Hasen-Yusuf et al., 2013).

3.2 Methods

3.2.1 Site description

Study sites were located in a semi-arid pastoral system in Yabello and Dire Districts, Borana, of southern Ethiopia, predominantly used for livestock (cattle, camel, goat and sheep) production (Angassa and Oba, 2008). The sites represent with Chromic Cambisol (FAO/UNESCO, unpublished) with similar climatic conditions and livestock population density but vary considerably in vegetation cover (Table 3.1). On the study sites, < 40% of the shrubs had been established before the 1970s (Coppock, 1994) and the most extensive woody encroachment occurred after the 1980s because of fire suppression (Angassa and Oba, 2008)

Mean annual precipitation was 550 mm in the region, with a 66% coefficient of variation across years. Rainfall is bimodal, with 55% of the annual precipitation occurring in March - May, followed by 30% in September - November. Mean annual air temperature is 20°C, with a mean monthly maximum of 21°C in February and a mean monthly minimum of 18.5°C in July (Ethiopian Meteorological Agency).

The entire study area is dominated by savanna grasses, interspersed by trees and shrubs of different density and cover (Coppock 1994). The perennial grass species comprise *Cenchrus ciliaris*, *Chrysopogon plumulosus*, *Cynodon dactylon*, *Sporobolus pyramidalis*, *Digitaria milanjiana*, and *Panicum repens*. Shrubs such as *Commiphora spp.*, *Acacia mellifera*, *Acacia Senegal*, *Acacia reficiens*, *Acalypha fruticosa*, and *Grewia spp.*, have expanded throughout the study area. Main tree species are *Acacia tortilis*, *Acacia bussei*, *Delonix baccal* and *Sterculia stencarpa*.

Table 3.1. Woody encroachment levels, age of enclosure, soil and livestock population density characteristics of the research sites in the Yabello and Dire districts of Borana zone. Geological information was summarized from the Borana land use study project soil survey report (unpublished data). Woody encroachment levels: Low encroachment (LE), moderate encroachment (ME), high encroachment (HE), and severe encroachment site (SE).

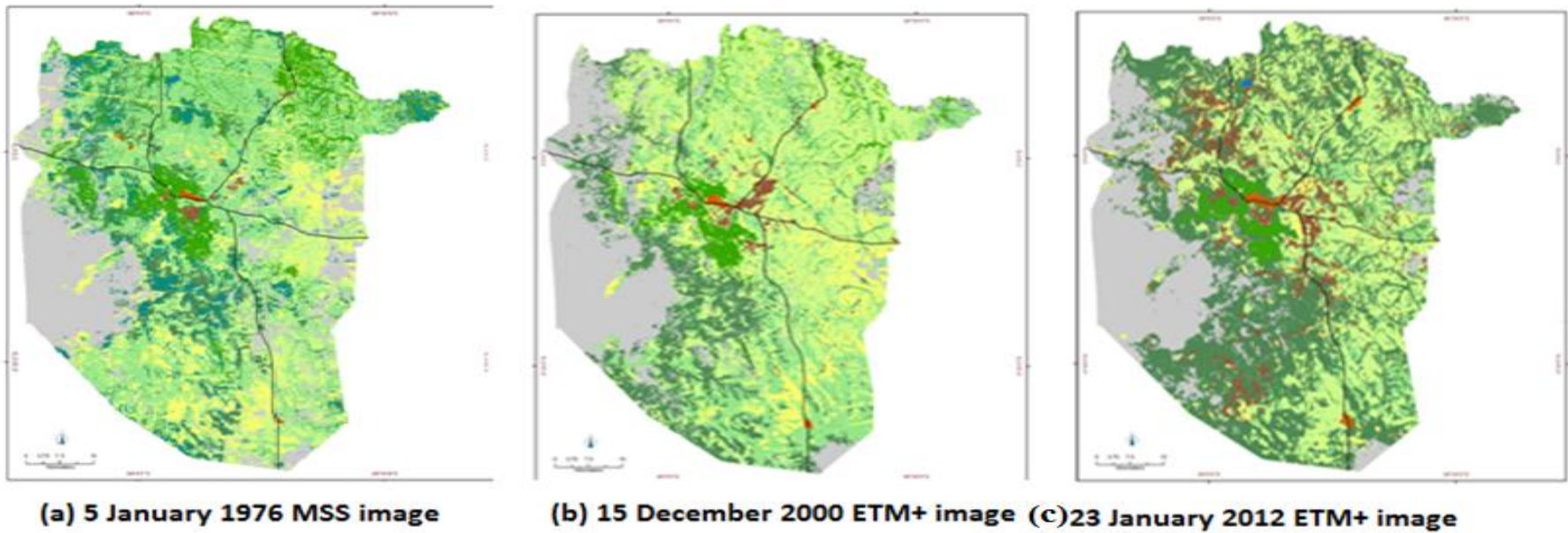
Encroachment level	Location (latitude, longitude)	Average elevation (masl)	Geology	Enclosure age	Soil texture (%)			Textural class	Livestock (head km ⁻²)					*TLU km ⁻²
					Sand	Silt	Clay		Cattle	Camel	Goat	Sheep	Equines	
LE	04°56'33" 38°10'12"	1550	Quartz-feldspathic gneiss and alluvium (sand silt and clay)	35	46	10	44	Sandy Clay	30	1.05	11	3	0.7	23.9
ME	04°24'02" 38°17'03"	1480	Alluvium: sand, silt and clay	30	67	18	15	Sandy Loam	31	0.7	9	10	2	25.6
HE	04°38'58" 38°04'58"	1290	Plateau basalt: alkaline basalt and trachyte	12	64	19	17	Sandy Loam	25	2.6	18	12	0.5	23.3
SE	04°41'08" 38°11'41"	1470	Quartz-feldspathic gneiss	8	65	20	15	Sandy Loam	25	2.6	18	12	0.5	23.3

*. Tropical Livestock Unit was calculated using the livestock population data obtained for each site from local Agricultural Offices following (Jahnke, 1982). 1 TLU=250 kg live weight.

3.2.2 Land cover change analysis

Environment for Visualizing Images (ENVI) 4.5 software was utilized for image processing and mapping was undertaken using ArcGIS 9. A handheld GPS (Garmin eTrex®) with 5 m accuracy was used for position location in the field. A Landsat Multi Spectral Scanner satellite image (hereafter MSS image) from the year 1976 and an Enhanced Thematic Mapper plus (ETM+ image) from the years 2000 and 2012, respectively, were selected for the study. All images were obtained from the United States Geological Survey (USGS) Denver, CO. The image dates spanned across the late short rainy season to the early dry season (December-January); during this time, woody vegetation was largely still in leaf while grasses had wilted, which enabled a precise mapping of woody cover.

Because images were historical, meteorological condition data (e.g., aerosol composition) were not available and, consequently, no atmospheric correction was undertaken for the multi-temporal remote sensing images. However, the training data for the classification were separately derived from the respectively classified images (Song et al., 2001) and image normalization techniques (Schott et al., 1988) were used to minimize non-earth surface target feature differences between the images. The images were geometrically registered to a common projection using geometric correction function in ENVI software and the spatial registration error was kept to subpixel (< 30m) level to avoid errors in the interpretation of feature changes in multi-temporal image sets (Lillesand et al., 2008). As high resolution was preferred, the lower spatial resolution (60-m) image was re-sampled to the 30-m resolution of the 23 January 2012 landsat image using the image sharpening and color normalized (Brovey) nearest neighbor resampling method within ENVI. The technique performs an image fusion operation with higher resolution panchromatic data. A shape file of Yabello and parts of Dire district were then used in sub-setting from each of the multi-spectral images. The delineated area of each respective image of the study area was estimated to be 7,554 km² (Fig.3.1A-C).



Vegetation cover class

- Settlement
- Cultivated land
- Shrub savanna
- Bare land
- Tree savanna
- Bushed savanna
- Bushland thickets
- Dense shrubland
- Forest

Fig. 3.1. Land use/cover in Yabello and Dire district, Borana rangelands, in 1976 (a), 2000 (b) and 2012 (c)

For ground verification purpose, four woody encroachment levels were identified in the field, namely low encroachment (here after named 'LE'), moderate encroachment (here after 'ME'), high encroachment (here after named 'HE'), and severe encroachment (here after 'SE'). The encroachment levels were differentiated mainly on the basis of the stage of woody encroachment determined through personal interviews with local people and rangeland managers but also based on woody plants density and structural variables of height and canopy cover. Canopy cover is the proportion of the ground area covered by the vertical projection of the tree/shrub canopy.

The number of woody plants, their height, canopy cover and canopy height were collected from 8 plots of 100 m x 100 m size, representing 4 woody encroachment levels and two grazing management levels, respectively (Table 3.2). This field work was conducted in May – June, 2011, during the late growing season. The GPS coordinates of the center of the plots were recorded to enable the location of the plots in the images. The data from the plots were used as training data in identifying and mapping the different levels of woody encroachment on the images. The training areas were three pixels (90 m) on the imagery utilized, selected in the field. In addition, GPS coordinates on historical and current vegetation cover maps were collected from 34 additional sites (including forest, bare ground and grassland, settlements, and cropland locations). Vegetation cover of these additional sites were visually assessed and categorized to represent severe (> 70% canopy cover), high (50 - 69% canopy cover), moderate (20 - 49% canopy cover) and low (< 20% canopy cover) woody vegetation cover. The data from these additional field sites, together with the data from the field plots (Table 3.2) aided in the location and mapping of woody encroached areas.

Table 3.2. Woody encroachment site data from field plots of 100 x 100 m size at the study area, the total number of woody plants (total), proportion of plants taller than 3 m (> 3m), the average canopy cover in % (canopy) and the canopy cover in % of trees taller than 3 m (canopy > 3m) as well as the dominant woody species.

Land cover class	Plot No	Total	> 3m	canopy	canopy > 3m	vegetation description
Severe encroachment (SE)	1	1065	13	67	58	Contains mostly fully matured tree and shrub stands with a woody canopy cover of 62%, almost bare of any herbaceous vegetation Dominant species: <i>A. bussei</i> , <i>A. etbabica</i> , <i>A. mellifera</i> , <i>Grewia</i> spp, and <i>Commiphora</i> spp
	2	1331	5	57	41	
High encroachment (HE)	3	2065	3	79	34	Comprise small to medium-sized shrubs and trees that formed an almost impenetrable thicket, with a woody canopy cover of 72% Dominant species: <i>A. reficiens</i> , <i>A. senegal</i> , <i>A. mellifera</i> , and <i>Commiphora</i> spp
	4	2109	2	67	30	
Moderate encroachment (ME)	5	1843	4	60	38	Comprise of dwarf shrubs and thick perennial grass-dominated stands with average woody canopy cover of 56% Dominant species: <i>Commiphora africana</i> , <i>C. habessinica</i> , <i>C. schimperi</i>
	6	2400	2	57	30	
Low encroachment(SE)	7	616	1	9	7	Comprise mosaic of tree and shrub patches in a perennial herbaceous species stand, with an average total woody canopy cover of 27% Dominant species: <i>Boscia mossambicensis</i> , <i>A.tortilis</i> , <i>Grewia</i> spp, and <i>Rhus natalensis</i>
	8	1606	5	42	22	

At training field plots, the number and canopy cover of woody plants > 3m height was high at severe -and high encroached sites (Table 3.2). The different levels of woody encroachment (SE, HE, ME and LE) were delineated on the respective image using supervised maximum likelihood classification techniques (Richards, 1994). Additionally, forest, settlements, cultivated area, bare lands and rock outcrops, i.e., a total of eight land cover classes were delineated (Fig.3.1). Local knowledge derived from herders and elders in the Borana region minimized spectral confusion to avoid classification inaccuracy.

Classification accuracy was assessed using the most recent image of the year 2012 according to Munyati et al. (2010). Two hundred fifty random sample pixel locations were generated from the classified 2012 images of the study site and visited in the field in May 2013 to judge the accuracy of their classification, which reached above 70%. As similar classification methods were employed for the 1976 and 2000 images, accuracy was judged to be satisfactory. After selectively combining classes, classified images were sieved, clumped and filtered with a 3×3 median filter to smoothen the classified images in ENVI (Version 4.2, Research Systems Inc., CO, USA). The area covered by each woody encroachment and land cover class shapefile, converted from raster imagery, was calculated with the Arc View GIS (Version 3.1, ESRI, Redlands, USA).

3.2.3 Field sampling design

Field based observation complemented woody vegetation structure and helped deriving C pool estimates along the four wood encroachment levels (LE, ME, HE, and SE, Fig.3.2) identified during the land cover change analysis (see section 2.2, Table 3.2, Fig.3.1). The other land cover classes (e.g., forests, bare lands) were not included in our study due to time and financial constraints.

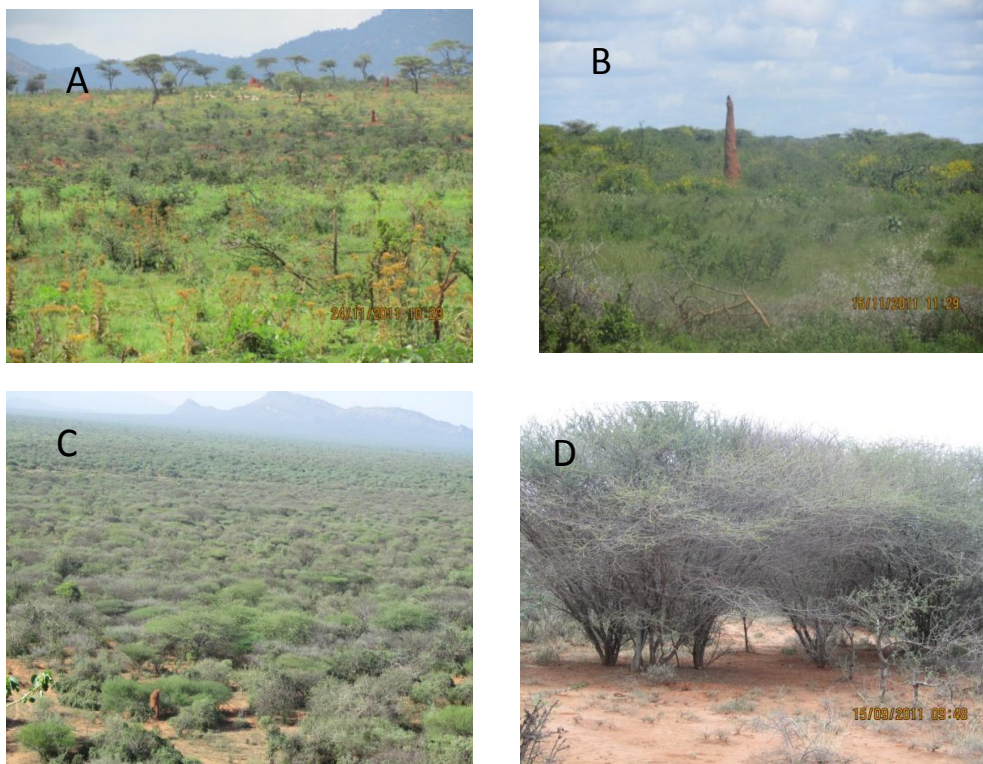


Fig. 3.2. Study sites: low encroachment site (A), moderate encroachment site (B), severe encroachment site (C), high encroachment site (D) in the semi-arid rangelands of Yabello and Dire districts, Borana, southern Ethiopia. Pictures taken by HasenYusuf in 2011.

Each woody encroachment level was divided into two grazing regimes, namely, rangeland enclosure (hereafter named ‘enclosure’) and communal grazing land (hereafter named ‘open’). The open grazing land represents the most common land use system in the Borana rangelands and is defined as communal rangelands belonging to the communities whose members have equal access rights (Oba, 1998). Enclosures in this study represent a shrub-fenced area that excludes grazers during the wet season, although some grazing may be allowed in the late dry season and in drought years when the forage is extremely scarce (Napier and Desta, 2012).

We randomly selected four replicates of enclosures within the same age group at each encroachment level, and 10 - 25 ha in size and 1 - 2 km apart (aerial distance, measured using Garmin GPS 72 (Garmin International Inc., USA) and adjacent open grazing lands in each of the four encroachment levels to examine the influence of grazing exclusion across the woody encroachment gradients (Fig3.2). To measure soil and vegetation attributes within each grazing regime we established four belt-transects (10 x 500 m), 300 – 500 m apart from

each other. Along each transect, four (10 x 10 m) plots were established at a 100 m intervals (4 woody encroachment level × 2 grazing regime × 4 replicate of grazing regimes × 5 plots). Previous studies in the present study area have shown that plot sizes = 100 m² were effectively used for sampling shrub-dominated vegetation (Angassa and Oba, 2010). To assess herbaceous species composition, biomass, and diversity inside and outside the enclosures, caged subplots of (1 × 1 m) were randomly nested within the larger 100 m² plots used for woody species.

3.2.4 Data collection

3.2.4.1 Vegetation species composition and structure

Percentage basal cover by species of herbaceous plants rooted within the caged 1 x 1 m² subplot was recorded during the peak season of primary production each at the end of the long rainy season (March-June) and the short rainy seasons (September-November). Percentage bare ground and rock cover (hereafter named 'bare ground') was recorded where no vegetation was encountered in each plot. Relative basal cover abundance (% RCA) by species was calculated from total herbaceous basal cover across each grazing regime excluding bare ground.

The species composition of woody species rooted within 10 x 10 m² plots were assessed using the plot count method (t'Mannetje and Jones, 2000). To analyze woody vegetation structure canopy diameter, crown rise, and height of the woody plants were measured using a 5 m long graduated wooden pole. Relative canopy area cover (in %) by species was calculated from total woody species canopy area cover. Canopy area of each woody plant with a canopy starting at ≥ 1 m height was calculated from longest canopy diameter and the canopy diameter perpendicular to the longest diameter. The distance between the ground and the bottom of the canopy (crown rise) were also measured to classify the total canopy area cover for each woody plant height class. Woody canopy area cover per height class was then compared across each bush encroachment level and grazing management (Wessels et al., 2011). These canopy height classes were also used to relate to the influence of herbivory and human utilization on herbaceous and woody plant structure (Wessels et al., 2011).

All woody plants rooted in the 10 x 10 m² plot, were counted and total density and canopy area cover were computed as the number of individual woody plants and total canopy area cover, respectively, divided by the area sampled and converted to ha (Kent and Coker 1992). Plant height was measured as the total height of the plant stem from the ground level to

the highest foliage. Total woody plants rooted within the 100 m² plots were further divided into six functional height classes (< 1 m, 1 - 2 m, 2 - 3 m, 3 - 4 m, 4 - 6 m and > 6 m, Wessels et al., 2011) to calculate the height-specific density for woody plant structural population analysis.

3.2.4.2 Aboveground biomass C stock

Peak herbaceous species biomass rooted within 0.5 x 0.5 m² were harvested to the ground level (t'Mannetje and Jones, 2000) in May and December 2011, the end of the long and short rainy season, respectively ($n = 20$ caged subplots per grazing management, total 160 subplots). Harvested material was oven dried (at 60°C for 48 h) and weighted. To nondestructively estimate woody biomass, (i) basal circumference at 10 cm AGB, (ii) maximum tree/shrub height, (iii) longest canopy diameter, (iv) canopy diameter perpendicular to longest diameter, and (v) crown rise were used. Woody canopy area and volume was calculated from these dimension measurements (Smit 1996) and biomass was quantified using species-specific allometric equations developed for the most dominant species (Hasen-Yusuf et al., 2013, see also Henry et al., 2011). For the less dominant trees and shrubby growth forms we developed two additional allometric equations (Hasen-Yusuf et al., 2013). The woody vegetation was classified into trees (> 3 m in height) and shrubs (< 3 m in height) in order to separately approximate the biomass of tree and shrubs in this rangeland (Levick et al., 2009). The leaf dry biomass (leaf dry matter (DM) yield of the woody vegetation was estimated using the Biomass Estimates From Canopy Volume Model (BECVOL-model; Smit, 1996), based on the relationship between tree canopy volume and its leaf DM for microphyllous ($y = -3.880 + 0.7080 x$) and broad-leaved species ($y = -5.45 + 0.079 x$) separately (Smit, 1996). Here y = total leaves dry mass per tree (in g); x = leaves spatial volume (m³) per tree. Leaf DM yield for each woody plant was summed to estimate leaf DM yield by species and total woody vegetation leaf DM yield per plot. Carbon concentrations of the total dry AGB (woody and leaves) (kg ha⁻¹) were estimated by multiplying dry biomass by 0.48 (Schlesinger, 1977).

3.2.4.3 Statistical analyses

The influences of grazing, woody encroachment and their interactions on woody plant density, canopy cover, and height distribution, tree biomass C pool, shrub biomass C pool, and herbaceous biomass C pool were analyzed using the General Linear Model Procedure in SAS version 9 (SAS Institute, 2002). Differences in all response variables were evaluated by treating woody encroachment level as main effect; grazing management regime was nested

within woody encroachment level. Mean comparisons were made using Tukey's test ($p < 0.05$). All values reported are means (\pm Standard Error).

3.3 Results

3.3.1 Land cover changes

The most striking change in vegetation cover between 1976 and 2012 was the continuous decline in open shrub and tree savanna areas from 45% in the 1976 to 9% in the 2012 (-36%) and the noticeable increase in the amount of moderate and severe woody encroached areas (bushed savanna and dense bushland thicket, respectively) from 22% in 1976 to 61% in 2012 (+39%) (Fig3.1, Table 3.3). This pattern indicated a continuous trend of new woody encroachment and thickening of the already existing tree/shrub savanna and woodlands/shrublands (Table 3.3). The bare land, and settlement areas remained relatively stable while cropland increased slightly and forest land declined by 6% from 1976 to 2012. The cropland was expanded into the wetter and most valuable grazing tree savanna or/and shrub savanna areas. Dense shrubland in 1976 fully developed into dense bushland thickets in 2000 and 2012 (Table 3.3).

Table 3.3. Change in area of vegetation cover class (in km² and % of total area) indicative of woody encroachment for the study district.

Cover class	1976		2000		2012		Percent land cover change		
	km ²	%	km ²	%	km ²	%	1976-2000	2000-2012	1976-2012
Settlement	20	0.3	40	0.5	56	0.7	0.3	0.2	0.5
Cultivated land	22	0.3	117	2	550	7	1.3	5.7	7.0
Bare land	1394	19	1565	21	1500	20	2.3	-0.9	1.4
Shrubsavanna	721	10	612	8	172	2	-1.5	-5.8	-7.3
Tree savanna	2641	35	1620	21	486	6	-13.5	-15.0	-28.5
Bushed savanna	958	13	2191	29	1912	25	16.3	-3.7	12.6
Bushland thicket	688	9	1045	14	2720	36	4.7	22.2	26.9
Dense shrubland	485	6	0	0.0	0	0.0	0.0	0.0	0.0
Forest	623	8	365	5	157	2	-3.0	-3.0	-6.0
Total area	7554		7554		7554				

3.3.2 Floristic composition

There were considerable variations in the species composition among the four encroachment levels (Table 3.4 and Table 3.5). At the Low encroachment sites the three woody species that contributed most to the woody AGC stocks were *Commiphora africana*, *C. habessinica*, and *C. schimperi* (47% combined). Dominating grass species at the LE site were *Chrysopogon plumulosus*, *Cenchrus ciliaris*, *Achyranthes aspera*, contributing 27% to RCA of the herbaceous species. The top two woody species in terms of AGC stock at the moderate encroachment site were *A. tortilis* and *Rhus natalensis* which when combined contributed to 57% of the total woody AGC stocks. The top three herbaceous species at ME site in terms of relative basal cover abundance (RCA) were *Chrysopogon plumulosus*, *Xerophyta humilis* and *Barleria argentea* when combined contributed to 16% of RCA. At the HE site the three woody species that contributed more than half (59%) to the woody AGC stock were *A. senegal*, *A. mellifera*, and *A. bussei*. The herbaceous layer of HE site was dominated by *Cynodon dactylon*, *Sporobolus pyramidalis*, *Helichrysum glumaceum*, which when combined contributed to 34% to RCA of the herbaceous community of this site. At the SE site almost half (48%) of total AGC stock was contributed by *A. mellifera* followed by *A. bussei* (14%), and *A. etabaica* (11%). The dominant herbaceous species of this site (RCA = 35% when combined) included *C. plumulosus*, *C. ciliaris*, and *H. glumaceum*.

Table 3.4. The top five woody plant species contributing greater aboveground (AGC) stocks in each encroachment level sorted by decreasing C stocks. LE = low encroachment, ME = moderate encroachment, HE = high encroachment, SE = severe encroachment. Density ha⁻¹ = total number of individual plants of each species per 10m x 10m plots. Canopy cover (m² ha⁻¹) = the sum total of individual canopy of each species per 10m x 10m plots.

Encroachment levels	Species	Density ha ⁻¹	Canopy cover (m ² ha ⁻¹)	AGC stock kg ha ⁻¹	%
LE	<i>Commiphora africana</i>	1220	1854	853	24
	<i>Commiphora schimperi</i>	270	675	493	14
	<i>Commiphora habessinica</i>	267	669	295	8
	<i>Acacia tortilis</i>	97	244	274	8
	<i>Lannea rivae</i>	167	419	238	7
ME	<i>Acacia tortilis</i>	130	325	653	33
	<i>Rhus natalensis</i>	67	168	481	24
	<i>Grewia bicolor</i>	112	281	94	5
	<i>Ormocarpum trichocarpum</i>	145	362	91	5
	<i>Acacia drepanolobium</i>	110	275	61	3
HE	<i>Acacia senegal</i>	597	1494	3204	41
	<i>Acacia mellifera</i>	135	337	703	9
	<i>Acacia bussei</i>	15	37	685	9

	<i>Acacia reficiens</i>	45	112	577	7
	<i>Grewia bicolor</i>	172	431	431	6
SE	<i>Acacia mellifera</i>	332	831	3730	49
	<i>Acacia bussei</i>	35	87	1044	14
	<i>Acacia etabaica</i>	62	156	817	11
	<i>Grewia bicolor</i>	297	743	505	7
	<i>Commiphora habessinica</i>	77	193	394	5

Table 3.5. The dominant top five herbaceous species based on relative percent cover abundance (%RCA) in each woody encroachment level. LE =low encroachment, ME = moderate encroachment, HE = high encroachment, SE = severe encroachment. RCA = relative percent basal cover abundance and RF = relative frequency.

Encroachment level	Species	RCA (%)	RF (%)
LE	<i>Chrysopogon plumulosus</i>	16	10
	<i>Achyranthes aspera</i>	10	8
	<i>Sporobolus confinis</i>	9	8
	<i>Cenchrus ciliaris</i>	8	7
	<i>Indigofera schimperi</i>	7	6
ME	<i>Xerophyta humilis</i>	6	3
	<i>Barleria argentea</i>	5	5
	<i>Chrysopogon plumulosus</i>	5	9
	<i>Cenchrus ciliaris</i>	4	6
	<i>Entropogon macrostachyus</i>	3	8
HE	<i>Cynodon dactylon</i>	14	7
	<i>Helichrysum glumaceum</i>	13	1
	<i>Sporobolus pyramidalis</i>	7	6
	<i>Sporobolus festivus</i>	5	4
	<i>Digitaria milanijana</i>	4	5
SE	<i>Helichrysum glumaceum</i>	13	11
	<i>Chrysopogon plumulosus</i>	13	9
	<i>Cenchrus ciliaris</i>	9	5
	<i>Indigofera spinosa</i>	7	5
	<i>Digitaria milanijana</i>	4	6

3.3.3 Woody vegetation structure

There was a gradient of increasing woody plants density and canopy area cover in > 6m height class from low encroachment site to severe encroachment, though the height of woody plants at moderately encroached site was only limited to < 6 m (Fig. 3.4 and Fig.3.5). The size-class distribution profile of LE, ME, and HE differed from that of the SE; the woody plants at these sites were mostly in the smaller size class (< 2 m), with only few woody plants in the larger size classes (Fig.3.4).

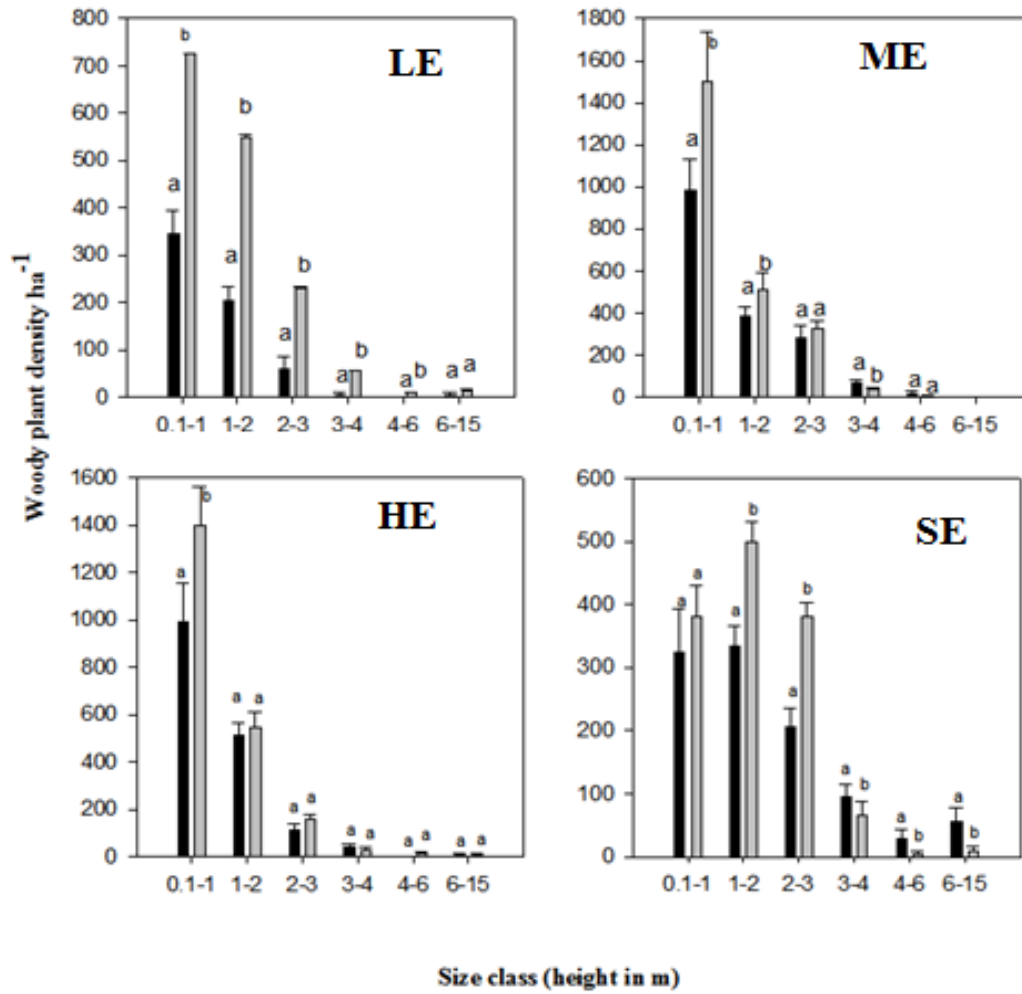


Fig.3.4. Size class frequency distributions for woody plants in communal open grazing lands (black bar) and enclosures (grey bar) in LE, low encroachment, ME, moderate encroachment, HE, high encroachment, and SE, severe encroachment sites in semiarid rangelands of Southern Ethiopia. Lower letters represent statistical differences between grazing regimes. Means not represented by similar letter are different.

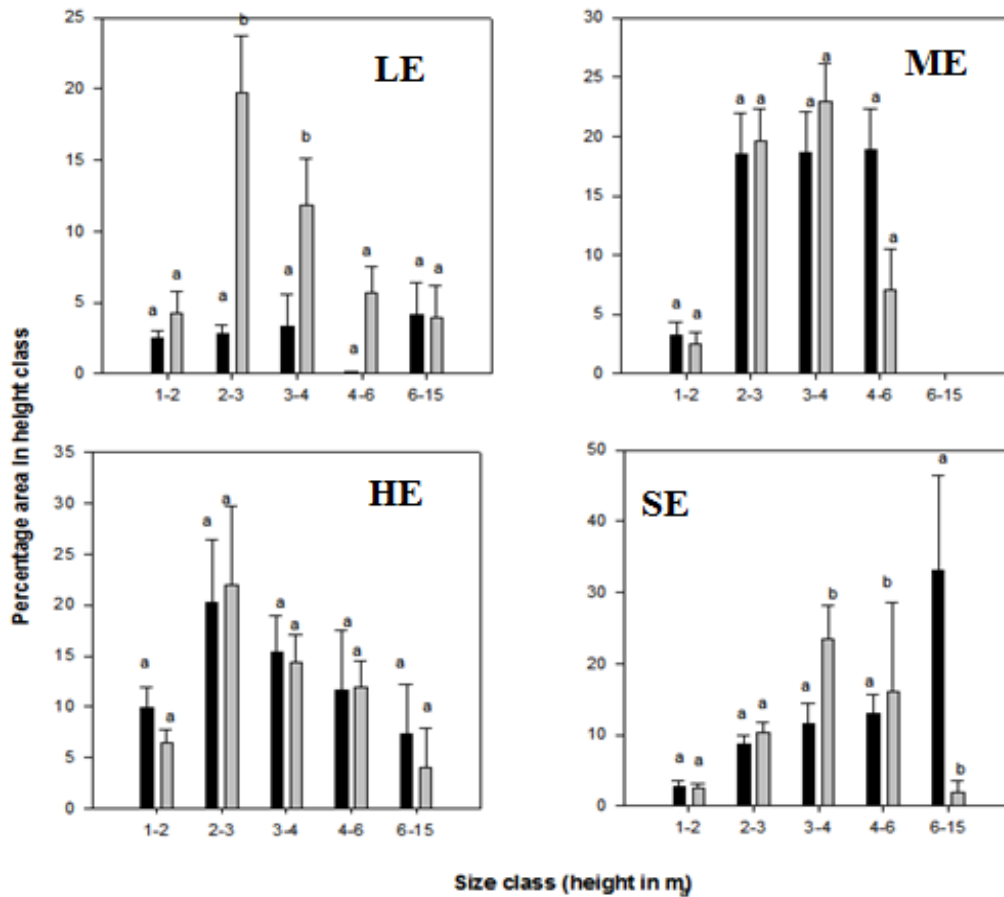


Fig.3.5. Percent area in each height classes covered by woody plants canopy cover in communal open grazing (black bar) and enclosure (grey bar) in LE, low encroachment, ME, moderate encroachment, HE, high encroachment, and SE, severe encroachment in semiarid rangelands of Southern Ethiopia. Lower letters represent statistical differences between grazing regimes. Means not represented by similar letter are different.

Regenerative woody plant density (< 1 m height) was significantly affected by woody encroachment levels and grazing regime and levels of woody encroachment interactions (Table 3. 6). Moderate to high woody encroached sites (ME and HE) had 50-100% significantly higher density of small woody plants in the enclosures, while at the SE site it was similar to the adjacent open grazing area (Table 3. 7). The medium-sized woody plant (2 - 3m, 3 - 4 m height) density was significantly affected by grazing, woody encroachment levels and their interactions (Table 3. 6). However, the mean densities of woody plants in this height classes (2 - 3m, 3 – 4 m) did not differ significantly among the levels encroachment and grazing regimes except at LE site (Table 3. 7, Fig. 3.4). The density of tall trees (> 6m height) was significantly affected by encroachment levels and encroachment and grazing regime interaction (Table 3. 6). High to severely encroached sites tended to have higher densities of tall woody plants (6 - 15 m height) in the open grazing regime, but only significantly so at the SE site (Table 3. 7). Levels of encroachment significantly influenced

total woody plant density independently of grazing regimes and vice versa (Table 3. 6). Mean density of total woody plants increased through the moderate and high woody encroachment levels (ME and HE) but declined at severely encroached site. However, the mean total woody plant density of the ME, HE, and SE site were statistically the same (Table 3. 7).The total woody plant density inside the enclosures was with 6 – 130% higher than communal grazing land ,and highly variable, but also significantly higher than that of communal grazing land across all encroachment levels (Table 3. 7).

Table 3.6. Results from analysis of variance of the effects of woody encroachment and grazing regime on woody plants density (number of plants ha⁻¹) in each height class.

Variable	Source	df	F	p
woody plant density				
< 1m	Encroachment	3	21	**
	Grazing	1	2	ns
	Encroachment x grazing	5	4	*
1 – 2m	Encroachment	3	4	*
	Grazing	1	17	**
	Encroachment x grazing	5	5	**
2 – 3m	Encroachment	3	3	*
	Grazing	1	18	**
	Encroachment x grazing	5	7	**
3 – 4m	Encroachment	3	4	*
	Grazing	1	0.01	ns
	Encroachment x grazing	5	3	*
5 – 6m	Encroachment	3	1	ns
	Grazing	1	6	*
	Encroachment x grazing	5	3	*
6 – 15m	Encroachment	3	5	**
	Grazing	1	1	ns
	Encroachment x grazing	5	5	**
Total	Encroachment	3	16	**
	Grazing	1	11	**
	Encroachment x grazing	5	2	ns

** . highly significant at $p < 0.01$

*. significant at $p < 0.05$

ns. not significant at $p=0.05$

Table 3. 7. Mean (\pm SE) woody density (number of plants ha⁻¹) in each height class across two grazing regimes along four woody plants encroachment levels (LE, low encroachment, ME, moderate encroachment, HE, high encroachment, and SE, severe encroachment) in semiarid rangelands of Southern Ethiopia. Different lowercase letters represent statistical differences determined by the grazing regime and woody encroachment interaction term in the General Linear Model and Turkey's means comparisons ($P < 0.05$). N, replication of grazing regime per encroachment levels.

Encroachment level	Grazing regime	N	Density of woody plants (No. of plants ha ⁻¹) in height class						Total
			< 1 m	1 – 2 m	2 – 3 m	3 – 4 m	5 – 6 m	6 – 15 m	
LE	Open	4	345±49a	205±29a	60±25a	5±5a	0a	5±5a	745±126a
	Enclosure	4	725± 2b	550±4b	230±30b	55±12b	10±2a	15±2.3a	1690±6b
ME	Open	4	985±149b	385±47ab	280±58b	65±21b	20±12b	0a	1905±22b
	Enclosure	4	1500±237c	510±80b	325±40b	35±13b	5±5a	0a	2460±224bc
HE	Open	4	990±163b	510±55b	115±25c	40±15b	0a	10±7a	1665±57b
	Enclosure	4	1400±165c	545± 64b	155±24c	25±12b	15±8.2bc	8±4a	2148±194b
SE	Open	4	325±67a	335±32a	205±30bc	95±20c	30±13bc	55±22b	1395±163b
	Enclosure	4	380±49a	500±31b	380±24b	65±23bc	5±5a	10±6.9a	1485±85b

** . highly significant at $p < 0.01$

* . significant at $p < 0.05$

ns. not significant at $p = 0.05$

3.3.4 Aboveground carbon stocks

The mean annual herbaceous AGC stock (summed over two growing seasons) was $0.7 (\pm 0.06)$ Mg ha⁻¹ (derived from 1.5 Mg dry matter ha⁻¹), representing 12% of total vegetation AGC stock (Table 3.9). The low encroachment (LE), ME, HE, and SE sites comprised $1.3 (\pm 0.2)$, $0.5 (\pm 0.04)$, $0.7 (\pm 0.09)$, $0.2 \pm (0.03)$ Mg C ha⁻¹, respectively. The mean AGC stock in communal grazing land and enclosures was $0.5 (\pm 0.06)$ and $0.8 (\pm 0.1)$ Mg C ha⁻¹, respectively. The herbaceous AGC stocks in the communal grazing lands were up to 50% as low as that of enclosures, depending on the woody encroachment levels (Table 3.9) but differences were only significant at LE and HE sites.

A mean total aboveground carbon (AGC) stocks of 5.9 Mg ha⁻¹ (1.6 Mg ha⁻¹ – 9.8 Mg ha⁻¹) were estimated summing all woody vegetation (trees and shrubs) and herbaceous layer C pools at our study site (Table 3.9). The mean AGC of our LE, ME, HE, and SE sites constituted $2.3 (\pm 0.4)$, $3.9 (\pm 0.4)$, $7.8 (\pm 1.7)$, and $8.6 (\pm 1.4)$ Mg ha⁻¹, respectively. The mean AGC stock across all encroachment levels in communal grazing lands and enclosures was $6.4 (\pm 1.1)$ and $6.1 (\pm 0.6)$ Mg ha⁻¹ respectively. The mean total AGC, shrub AGC, and tree AGC stocks were significantly increased with an increasing gradient of woody encroachment levels, independent of grazing regime (Table 3.8, Table 3.9). The mean total AGC at heavily encroached sites HE and SE sites were more than 2-3 times higher than that of ME and LE (Table 3.9). Mean shrub AGC stock was by 68% significantly higher in ME compared to HE and SE whereas AGC stocks of trees were more than five times higher in HE and SE compared to ME (Table 3.9).

Table 3.8. Results from analysis of variance of the effects of woody encroachment and grazing regime on aboveground Carbon (AGC) stocks in tree, shrub and herbaceous vegetation biomass.

Variable	Source	<i>df</i>	F	p
Carbon stocks(Mg ha ⁻¹)				
Shrub wood AGC	Encroachment	3	42	**
	Grazing	1	2	NS
	Encroachment x grazing	5	2	NS
Tree wood AGC	Encroachment	3	10	**
	Grazing	1	4	NS
	Encroachment x grazing	5	3	NS
Herbaceous AGC	Encroachment	3	21	**
	Grazing	1	11	**
	Encroachment x grazing	5	5	**
Woody leaves carbon	Encroachment	3	16	**
	Grazing	1	3	NS
	Encroachment x grazing	5	5	**
Total AGC stocks	Encroachment	3	9	**
	Grazing	1	4	NS
	Encroachment x grazing	5	3	NS

Table 3.9. Means (+SE) of grazing management regime and woody encroachment levels for calculated AGC stocks in tree, shrub and herbaceous vegetation. Different lowercase letters represent statistical differences determined by the grazing regime and woody encroachment interaction term in the General Linear Model and Tukey's means comparisons ($p < 0.05$) Abbreviation of encroachment level see Table 3. 7.

Encroachment levels	Grazing regime	Shrub wood AGC (Mg ha ⁻¹)	Tree wood AGC (Mg ha ⁻¹)	Herbaceous AGC (Mg ha ⁻¹)	Woody vegetation leaves C (Mg ha ⁻¹)	Total AGC (Mg ha ⁻¹)
LE	Open	0.1±0.05a	0.5±0.4a	1.0± 0.1b	0.01±0.4a	1.6±2.4a
	Enclosure	0.5 ±0.07a	2.3±0.6b	1.7±0.2c	0.5±0.04b	5.0±1.3b
ME	Open	2.0±0.30b	1.1±0.50ab	0.4±0.03a	0.8±0.1b	4.3±0.6b
	Enclosure	1.6 ±0.2b	0.9 ± 0.50a	0.3±0.03a	0.7±0.05b	3.5±0.5b
HE	Open	0.7±0.0a	6.4±0.2c	0.5±0.1a	0.9±0.2b	9.0±2.4c
	Enclosure	0.4±0.0a	6.3±1.2c	1.0±0.1a	0.7±0.1b	8.2±1.3c
SE	Open	0.7±0.2 a	8.1±2.7d	0.03±0.01d	1.2±0.1b	9.8±2.9c
	Enclosure	0.5±0.09a	4.3±0.2e	0.2±0.01a	0.6±0.1b	5.6±1.6b

**, highly significant at $p < 0.01$

*, significant at $p < 0.05$

Ns. not significant at $p = 0.05$

3.4 Discussion

Our multi-temporal satellite imagery analysis and quantitative studies demonstrated extensive encroachment of woody plants in the Borana rangelands over the last four decades following fire suppression. The most striking change in rangeland vegetation cover was the dramatic drop in the open vegetation types such as shrub savanna and tree savanna and the significant area increase in dense vegetation types such as dense shrubland thickets and bushed savannas. This woody vegetation increase is consistent with results from other savanna systems worldwide (Hughes et al., 2006; Briggs et al., 2005; Hibbard et al., 2003; Asner et al., 2003; Gifford and Howden, 2001). Savannas may, therefore, switch from grass-dominated ecosystems to dense shrub or woodland ecosystems once fire is suppressed (Hibbard et al., 2003). Significant changes in woody cover seemed to have occurred across our study sites due to human influences such as changes in fire and grazing management, and climate variability (Angassa and Oba, 2008; Dalle et al., 2006; Oba et al., 2001; Oba et al., 2000).

All our study sites encompassed woody vegetation cover of less than 40% in the 1970s (Coppock, 1994). Trees and shrubs likely expanded out from small patches to form larger patches, similarly to other studies (Wiegand et al., 2006; Gillson, 2005). Furthermore, the local informants noticed that woody plant encroachment processes seemed accelerated over the last decades. This was also reflected by our woody density, canopy cover, and population size assessment, which indicates that woody plant populations in low, moderate, and high encroachment sites are still expanding and will remain increasing in density and cover while the most severely encroached site (SE) contained mainly old woody vegetation without many seedlings, and, thus, might be in a climax woody state. Given the complete suppression of fire and increasing trends of livestock population in the Borana rangelands, these woody encroachment trends are likely to continue. On the contrary, the lack of regeneration as shown by the low seedling and sapling densities (< 1 m tall) at the severely encroached site may gradually lead to an open savanna as trees die off over time. This agrees with findings of (Belay and Moe, 2012; Wiegand et al., 2006) who indicated that the woody patch density declined in savannas as tree/shrub patches get older because of 'self thinning'. The high number of seedlings and saplings in enclosures across all encroachment levels indicates that besides other factors (e.g., fire suppression, moisture, competition with grasses) livestock exclusion will play a major role in the establishment, encroachment, and growth of woody species.

Floristically, the four woody encroachment levels are typical of the broader vegetation types of the semiarid Borana rangelands, each with the vegetation community composition

differentiated largely on the basis of topo-edaphic gradients (Coppock, 1994). However, the density and canopy area cover of the taller woody plants (< 6 m in height) increased gradually with woody encroachment levels, with SE being dominated by tall trees. Generally, the woody plant species composition and structure in the savanna ecosystem are influenced by edaphic factors (Venter and Govender, 2012), age of patch establishment (Wiegand et al., 2006), impacts through herbivores, and human utilization of woody plants (Wessels et al., 2011; Augustine and Mcnaughton, 2004; Roques et al., 2001).

In our study, enclosures had higher total woody density, which concurs with findings of (Scogings et al., 2012). Enclosures at LE, ME, and HE had more than double the woody plant densities, particularly in the 1 - 2 m height classes, inside enclosures compared to communal open grazing sites, indicating that herbivore exclusion strongly impacts woody plants in their early ages (Scogings et al., 2012; Roques et al., 2001). Enclosures are currently expanding in the Borana rangelands, which might foster young woody plant establishment and growth. The lower density of woody plants of < 3 m in the communal grazing sites on the other hand indicates the significance of browsers such as goats in reducing the rate of bush encroachment by lowering the regeneration potential of encroaching woody plants in savanna systems (Levick et al., 2009; Augustine and Mcnaughton, 2004). The lack of significant differences in this higher height classes between enclosures and communal open grazing lands reflects human utilization impact on tall trees at both sites (Wessels et al., 2011). Woody plant density and cover values were by 60-100% uniquely higher in the enclosures than communal grazing areas for all height levels at the low encroached site. This might reflect heavy browsing and selective harvesting of woody vegetation in the communal grazing land as this area borders fully sedentary pastoral village.

Our estimates of AGC stocks of 5.9 Mg C ha^{-1} ($1.6 \text{ Mg C ha}^{-1} - 9.8 \text{ Mg C ha}^{-1}$) are higher than, e.g., those of a Sudanese woodland savanna (1.1 Mg C ha^{-1} , Alam et al., 2013) but lower than those of savanna woodlands ($9.5 \text{ Mg C ha}^{-1} - 26 \text{ Mg C ha}^{-1}$) in West and South Africa (Woollen et al., 2012; Shackleton and Scholes, 2011; Williams et al., 2008) highlighting the importance of vegetation composition, climatic regimes, soils and impacts of land uses (Woollen et al., 2012).

Our comparisons of different encroachment levels indicated that the expansion of woody plants into relatively open savanna, and subsequent conversion to bushland/shrublands substantially increased the AGC stocks. Our estimate of AGC stocks in highly encroached sites (SE, HE) were up to three times greater than that of the less encroached sites (LE, ME). Similar trends of AGC stock increase were reported when open savannas are encroached by

woody plants (Lett et al., 2004; Asner et al., 2003). The significance of additional C stocks in woody species is further emphasized by the low understory (herbaceous species) C stock values contributing only 12% to the total AGC stock of this ecosystem. Hence, the dramatic shift in the AGC stock with woody plant encroachment is likely to increase overall AGC stock within the ecosystem, overriding the herbaceous C stock component of the ecosystem. However, the differences in the AGC stock across encroachment level seemed to be influenced by woody species composition and size structure rather than by total canopy cover or density alone. The abundance of tall trees (> 6 m) with complex canopy structure e.g., *A. mellifera*, *A. bussei*, *A. senegal*, *A. tortilis* in HE and SE help to store more than twice as much C as compared to the dominant shrubs at ME, though these sites were comparable in terms of total woody density and cover. This is in agreement with the findings of (Litton et al., 2006) who reported that a large proportion of the total AGC stock in dry forests and savanna ecosystems was comprised of tall trees with large canopy cover. In addition, C storage is influenced by the wood gravity of the dominant trees/shrubs in the area. For example, LE site was predominantly (60%) covered by *Commiphora species*, which have a low wood gravity (with $\approx 67\%$ water content in fresh wood biomass) compared to other woody species (e.g., *A. senegal* $\approx 38\%$, *A. mellifera* $\approx 37\%$, *A. bussei* $\approx 33\%$) found at SE and HE (Hasen-Yusuf et al., 2013).

The impact of long-term enclosures was more clearly visible in the herbaceous AGC stocks than in the woody AGC stocks across all encroachment levels. Higher herbaceous biomass in enclosures compared to open grazing sites have been reported from grazing systems worldwide such as the Borana rangelands (Angassa and Oba, 2010), northern and central Ethiopia (Yayneshet et al., 2009), South Africa (Eccard et al., 2000) and New Zealand (McIntosh et al., 1997).

Our data set has demonstrated that a long-term livestock exclusion may not necessarily increase the herbaceous biomass in the savanna ecosystem due to the suppressive effect of woody plants on herbaceous biomass (Smit, 2001; Harrington and Johns, 1990). Therefore, the herbaceous biomass difference was not more pronounced between enclosures and communal open grazing sites at high and severe encroachment areas. This also highlights the tradeoffs between C sequestration in the woody vegetation vs. pastoral production in rangeland systems.

A complete understanding of the effect of vegetation cover changes (woody encroachment) and grazing pressure on ecosystem C stock should also take belowground dynamics into account. Belowground tree/shrub root biomass also is assumed to also increase

over time following woody encroachment in the savanna system (Jobbagy and Jackson, 2000). However, woody encroachment could also decrease the belowground C stock by affecting the large amount captured by herbaceous biomass to be allocated in the belowground through roots and soil organic C in wet regions (Jackson et al., 2002). In this semiarid rangeland, soil organic carbon (SOC) stocks did not show similar patterns of AGC stock increase along the gradients of woody encroachment but stocks were rather largely influenced by variations in soil characteristics (HasenYusuf et al., in press).

Our current estimates of an AGC stock of 6 Mg C ha^{-1} was largely derived from woody plants that thrived after long-term fire suppression from the rangelands. This estimate enhanced is probably even higher taking belowground C stocks into account. The mean AGC stocks estimated by this study could also be increased under appropriate management as demonstrated in the benchmark site in an arid and semi-arid woodland vegetation of South Africa (Shackleton and Scholes, 2011). What constitutes appropriate management would have to be determined in consultation with local pastoralists (Shackleton and Scholes, 2011). However, pastoralists are currently unaware of the potential of C credits via appropriate management of rangelands including woody plants in their rangeland systems (Lipper et al., 2010). To increase C sequestration in vegetation and soils could, though, represent a highly valuable alternative livelihood income source if appropriate and efficient financing systems would be established (Follett and Reed, 2010; Lipper et al., 2010). Co-management of the rangelands for environmental services (C sequestration and reduction of emissions from livestock systems) through an environmental payment service scheme could also be considered to help the pastoralists to diversify and sustain their livelihood in the long run (Neely et al., 2009). Implementation of a sustainable rangeland management practices that includes moderate stocking rates and sustainable grazing systems, for example, rotational grazing and seasonal land use, will promote the sustainability of both economic (livestock production) and environmental services (C sequestration).

3.5 Conclusion

Historical and on-going woody encroachment in the savanna ecosystem has a significant implication for the global C-cycle. Assessment of its contribution to the global C budget has practical importance for C accounting and global C-credit / offset programs. Our data indicate that the wide-spread woody encroachment in semi-arid savanna of Borana rangelands substantially increased the C stock in the aboveground biomass but was largely determined by species composition, size structure, grazing management and site characteristics. The woody population structure suggested high recruitment except at the oldest encroached site,

indicating the increase in AGC stocks is expected. Further investigation of the belowground C stock associated with woody encroachment is necessary. Our result also shows that woody encroachment had an overriding influence on the impact of grazing management on herbaceous biomass C stocks with significant reduction in herbaceous biomass in heavily encroached sites compared to less encroached sites. Our C stock estimates can then be compared to other studies and used in calibrating the AGC stocks in the future model of land use and cover changes. The observed large increases in AGC stocks that occurred through woody encroachment on landscape and regional scales will foster future C-trade discussions with respect to climate change mitigation strategies. However, conserving woody biomass C stocks will threaten the sustainability of livestock production, and this tradeoff must be quantified and considered in the future.

3.6 References

- Alam, S.A., Starr, M., Clark, B.J.F., 2013. Tree biomass and soil organic carbon densities across the Sudanese woodland savannah: A regional carbon sequestration study. *J. Arid Environ.* 39, 67-76.
- Angassa, A., Oba, G., 2008. Herder perceptions on impacts of range enclosures, crop farming, fire ban and bush encroachment on the rangelands of Borana, Southern Ethiopia. *Hum. Ecol.* 36, 201-215.
- Angassa, A., Oba, G., 2010. Effects of grazing pressure, age of enclosures and seasonality on bush cover dynamics and vegetation composition in southern Ethiopia. *J. Arid. Environ.* 74, 111-120.
- Asner, G.P., Archer, S., Hughes, R.F., Ansley, R.J., Wessman, C.A., 2003. Net changes in regional woody vegetation cover and carbon storage in Texas drylands, 1937-1999. *Glob. Chang. Biol.* 9, 316-335.
- Augustine, D. J., Mcnaughton, S. J., 2004. Regulation of shrub dynamics by native browsing ungulates on East African rangeland. *J. Appl. Ecol.* 41, 45 – 58.
- Belay, T. A., Moe, S. R., 2012. Woody dominance in a semi-arid savanna rangeland - Evidence for competitive self-thinning. *Acta. Oecol.* 45, 98-105.
- Bond, W. J., Midgley, G. F., 2012. Carbon dioxide and the uneasy interactions of trees and savannah grasses. *Philos Trans R Soc Lond B Biol Sci.* 367, 601-612.
- Briggs, J.M., Knapp, A.K., Blair, J.M., Heisler, J.L., Hoch, G.A., Lett, M.S., McCarron, J.K., 2005. An ecosystem in transition: Causes and consequences of the conversion of mesic grassland to shrubland. *BioScience* 55, 243 – 254.
- Brown, J.R., Archer, S., 1999. Shrub invasion of grassland: Recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology* 80, 2385 – 2396.
- Conant, R.T., Paustian, K., 2002. Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global Biogeochem Cycles* 16, 90 – 99.
- Coppock, D.L., 1994. The Borana Plateau of Southern Ethiopia: Synthesis of Pastoral Research, Development and Changes 1980–1990. System study No. 5. International Livestock Centre for Africa, Addis Ababa, 374pp.
- Dalle, G., Maass, B.L., Isselstein, J., 2006. Encroachment of woody plants and its impact on pastoral livestock production in the Borana lowlands, southern Oromia, Ethiopia. *Afr. J. Ecol.* 44, 237-246.
- Daryanto, S., Eldridge, D.J., Throop, H.L., 2013. Managing semi-arid woodlands for carbon storage: Grazing and shrub effects on above- and belowground carbon. *Agric. Ecosyst. Environ.* 169, 1-11.
- Eccard, J.A., Walther, R.B., Milton, S.J., 2000. How livestock grazing affects vegetation structures and small mammal distribution in the semi-arid Karoo. *J. Arid. Environ.* 46, 103-106.

- Follett, R.F., Reed, D.A., 2010. Soil carbon sequestration in grazing lands: Societal benefits and policy implications. *Rangeland Ecol. Manage.* 63, 4-15.
- Gifford, R.M., Howden, M., 2001. Vegetation thickening in an ecological perspective: Significance to national greenhouse gas inventories. *Environ. Sci. Policy.* 4, 59-72.
- Gillson, L., 2005. Evidence of Hierarchical Patch Dynamics in an east African savanna? *Landsc. Ecol.* 19, 883-894.
- Harrington, G.N., Johns, G.G., 1990. Herbaceous biomass in a Eucalyptus savanna woodland after removing trees and/or shrubs. *J Appl. Ecol.* 27, 775-787.
- Hasen-Yusuf, M., Treydte, A.C., Abule, E., Sauerborn J., 2013. Predicting aboveground biomass of encroacher woody species in semi-arid rangelands, Ethiopia. *J. Arid. Environ.* 96, 64 – 72.
- Hasen-Yusuf, M., Treydte, A.C., Sauerborn J, Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on soil carbon and nitrogen. *PlosOne* (in press).
- Henry, M., Picard, N., Trotta, C., Manlay, R.J., Valentini, R., Bernoux, M., Saint-André, L., 2011. Estimating tree biomass of sub-Saharan African forests: a review of available allometric equations. *Silva. Fenn.* 45, 477–569.
- Hibbard, K. A., Schimel, D.S, Archer, S. R., Ojima, D.S., Parton, W., 2003. Grassland to woodland transitions: integrating changes in landscape structure and biogeochemistry. *Ecol. Appl.* 13, 911–926.
- Homann, S., Rischkowsky, B., Steinbach, J., 2008. The effect of development interventions on the use of indigenous range management strategies in the Borana Lowlands in Ethiopia. *Land. Degrad. Devlop.* 19, 368-387.
- Hughes, R. F., Archer, S. R., Asner, G. P., Wessman, C. A., Mcmurtry, C., Nelson, J., Ansley, R. J., 2006. Changes in aboveground primary production and carbon and nitrogen pools accompanying woody plant encroachment in a temperate savanna. *Glob. Chang. Biol.* 12, 1733–1747.
- Jackson, R.B., Banner, J.L., Jobbágy, E.G., Pockman, W.T., Wall, D.H., 2002. Ecosystem carbon loss with woody plant invasion of grasslands. *Nature* 418, 623 - 626.
- Jahnke, H.E, 1982. Livestock production systems and livestock development in tropical Africa. *Kieler Wissenschaftsverlag Vauk, Kiel, Germany.*
- Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol. Appl.* 10, 423 – 436.
- Kent, M., and Coker, P., 1992. *Vegetation Description and Analysis: A Practical Approach.* Belhaven Press, London.
- Knapp, A.K., Briggs, J.M., Collins, S.L., Archer, S.R., Bret-Harte, M.S., Ewers, B.E., Peters, D.P., Young, D.R., Shaver, G.R., Pendall, E., Cleary, M.B., 2008. Shrub encroachment in North American grasslands: Shifts in growth form dominance rapidly alters control of ecosystem carbon inputs. *Glob. Chang. Biol.* 14, 615-623.

- Lett, M.S., Knapp, A.K., Briggs, J.M., Blair, J.M., 2004. Influence of shrub encroachment on aboveground net primary productivity and carbon and nitrogen pools in a mesic grassland. *Can. J. Bot.* 82, 1363–1370.
- Levick, S.R., Asner, G.P., Kennedy-Bowdoin, T. Knapp, D.E., 2009. The relative influence of fire and herbivory on savanna three-dimensional vegetation structure. *Biol. Conserv.* 142, 1693-1700.
- Lillesand, T. M., Kiefer, R. W., Chipman, J. W., 2008. Remote sensing and image interpretation, 6th ed.. Wiley, New York.
- Lipper, L., Dutilly-Diane, C., McCarthy, N., 2010. Supplying carbon sequestration from West African rangelands: Opportunities and barriers. *Rangeland EcolManage* 63, 155-166.
- Litton, C.M., Sandquist, D.R., Cordell, S., 2006. Effects of non-native grass invasion on aboveground carbon pools and tree population structure in a tropical dry forest of Hawaii. *For. Ecol. Manage.* 231, 105-113.
- McIntosh, P.D., Allen, R.B., Scott, N., 1997. Effects of exclosure and management on biomass and soil nutrient pools in seasonally dry high country, New Zealand. *J. Environ. Manage.* 51, 169-186.
- Munyati, C., Shaker, P., Phasha, M.G., 2010. Using remotely sensed imagery to monitor savanna rangeland deterioration through woody plant proliferation: a case study from communal and biodiversity conservation rangeland sites in Mokopane, South Africa. *Environ. Monit. Assess.* 10, 1583–1584.
- Napier, A, Desta S., 2012 Review of Pastoral Rangeland Enclosures in Ethiopia, <http://sites.tufts.edu/feinstein/2012/> date 20, 06, 2013.
- Neely, C., Bunning, S., Wilkes, A., 2009. Review of evidence on drylands pastoral systems and climate change. Implications and opportunities for mitigation and adaptation, in land and water discussion paper 8. FAO, Rome, P.39.
- Oba, G., Kotile, D.G., 2001. Assessments of landscape level degradation in southern Ethiopia: pastoralists versus ecologists. *Land. Degrad. Develop.* 12, 461–475.
- Oba, G., Post, E., Syvertsen, P.O., Stenseth, N.C., 2000. Bush cover and range condition assessments in relation to landscape and grazing in southern Ethiopia. *Landsc. Ecol.* 15, 535–546.
- Oba, G. 1998. Assessment of indigenous range management knowledge of the Borana pastoralists of southern Ethiopia. Commissioned by GTZ-Borana Lowland Pastoral Development Program in collaboration with the Oromiya Regional Bureau for Agricultural Development, Negelle/Borana Ethiopia.
- Reid, R. S., Serneels, S., Nyabenge, M., Hanson, J., 2005. The changing faces of pastoral systems in grass-dominated ecosystem of eastern Africa, in: Suttie, J. M., Reynold S. G., Batello, C. (eds.), *Grasslands of the world*. FAO, Rome, pp. 19-76.
- Richards J.A., 1994. Remote Sensing Digital Image Analysis: an Introduction. Second, Revised and Enlarged Edition. Springer-Verlag, Berlin.

- Riginos, C., Young, T.P., 2007. Positive and negative effects of grass, cattle, and wild herbivores on Acacia saplings in an East African savanna. *Oecologia* 153, 985-995.
- Roques, K.G., O'Connor, T.G., Watkinson, A.R., 2001. Dynamics of shrub encroachment in an African savanna: Relative influences of fire, herbivory, rainfall and density dependence. *J Appl. Ecol.* 38, 268-280.
- Scholes, R.J., Archer, S.R., 1997. Tree-grass interactions in Savannas. *Annu. Rev. Ecol. Syst.* 28, 517-544.
- Scholes, R.J., Hall, D.O., 1996. The carbon budget of tropical savannas, woodlands and grasslands. in: Breymer, A.I., Hall, D.O., Melillo, J.M., Agren, G.I. (Eds.), *Global Change: Effects on Coniferous Forests and Grasslands*. John Wiley and Sons, chichester, UK, pp.69-99.
- Scogings, P.F., Johansson, T., Hjätén, J., Kruger, J., 2012. Responses of woody vegetation to exclusion of large herbivores in semi-arid savannas. *Aust. Ecol.* 37, 56-66.
- Shackleton, C.M., Scholes, R.J., 2011. Aboveground woody community attributes biomass and carbon stocks along a rainfall gradient in the savannas of the central lowveld, South Africa. *S Afr J Bot* 77, 184-192.
- Schlesinger, W.H., 1977. Carbon balance in terrestrial detritus. *Annu Rev Ecol and Syst* 8, 51 – 58.
- Schott, J., Salvaggio, C., and Volchok, W., 1988. Radiometric scene normalisation using pseudo invariant features. *Remote Sens. Environ.* 26, 1–16.
- Smit, G.N., 1996. Biomass estimation from canopy volume (version 2)—users guide. Unpublished Report, University of the Free State, Bloemfontein, South Africa.
- Smit, G.N., 2001. The influence of tree thinning on the vegetative growth and browse production of *Colophospermum mopane*. *S Afr. J. Wild. Res.* 31, 99-114.
- Smit, I.P.J., Asner, G.P., Govender, N., 2010. Effects of fire on woody vegetation structure in African savanna. *Ecol. Appl.* 20, 1865-1875.
- Solbrig, O.T., Menaut, J.C., Mentis, M., Shugart, H.H., Stott, P., Wigston, D. (Eds.), 1991. *Savanna modelling for global changes: Biology International*, 24.
- Solomon, T.B., Snyman, H.A Smit, G.N., 2007. Cattle-rangeland management practices and perceptions of pastoralists towards rangeland degradation in the Borana zone of southern Ethiopia. *J Environ. Manage.* 82, 481-494.
- Song, C., Woodcock, C. E., Seto, K. C., Lenney, M. P., Macomber, S. A., 2001. Classification and change detection using Landsat TM data: When and how to correct atmospheric effects? *Remote Sens. Environ.* 75, 230–244.
- t'Mannetje, L., Jones, R.M., 2000. *Field and Laboratory Methods for Grassland and Animal Production Research*. CABI

- Van Auken, O.W., 2009. Causes and consequences of woody plant encroachment into western North American grasslands. *J Environ. Manage.* 90; 2931-2942.
- Van Der Maarel, E., 1979. Transformation of cover- abundance values in phytosociology and its effect on community similarity. *Vegetation* 9, 97–114.
- Venter, F.J., Govender, N., 2012. A geomorphic and soil description of the long-term fire experiment in the Kruger national park, South Africa. *Koedoe* 54.
- Wessels, K. J., Mathieu, R., Erasmus, B.F.N., 2011. Impact of communal land use and conservation on woody vegetation structure in the Lowveld savannas of South Africa. *For. Ecol. Manage.* 261, 19-29.
- Wiegand, K., Saltz, D., Ward, D., 2006. A patch-dynamics approach to savanna dynamics and woody plant encroachment - Insights from an arid savanna. *Perspect. Plant Ecol. Evol. Syst.* 7, 229-242.
- Wigley, B.J., Bond, W.J., Hoffman, M.T., 2010. Thicket expansion in a South African savanna under divergent land use: Local vs. global drivers? *Glob. Chang. Biol.* 16, 964-976.
- Williams, M., Ryan, C.M., Rees, R.M., 2008. Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique. *For. Ecol. Manage.* 254, 145-155.
- Woollen, E., Ryan, C.M., Williams, M., 2012. Carbon Stocks in an African Woodland Landscape: Spatial Distributions and Scales of Variation. *Ecosystems* 15, 804-818.
- Yayneshet, T., Eik, L.O., Moe, S.R., 2009. The effects of exclosures in restoring degraded semi-arid vegetation in communal grazing lands in northern Ethiopia. *J. Arid. Environ.* 73, 542-549.

4 Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on soil carbon and nitrogen

Submitted to Plos One on 23.10.2013

Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on soil carbon and nitrogen

M. Hasen-Yusuf ^{a*}, A.C. Treydte ^a, and J. Sauerborn ^a

^a Institute of Plant Production and Agroecology in the Tropics and Subtropics, University of Hohenheim, 70593 Stuttgart, Germany

Telephone: +49-(0)711-459 23602

*corresponding author email: hasanyusuf12@gmail.com

Abstract

Intensified grazing and wide-spread woody encroachment may strongly alter soil carbon (C) and nitrogen (N) pools. However, the direction and quantity of these changes have rarely been quantified in East African savanna ecosystem. As shifts in soil C and N pools might further potentially influence climate change mitigation we quantified and compared soil organic carbon (SOC) and total soil nitrogen (TSN) content in enclosures (“enclosure”) and communal grazing lands (“open”) across varying woody cover (woody encroachment levels). Estimated mean SOC and TSN stocks at 0 - 40 cm depth varied across grazing regimes and among woody encroachment levels. The open grazing land at the heavily encroached site on sandy loam soil contained the least SOC ($30 \pm 2.1 \text{ Mg ha}^{-1}$) and TSN ($5 \pm 0.57 \text{ Mg ha}^{-1}$) while the enclosure at the least encroached site on sandy clay soil comprised the greatest mean SOC ($81.0 \pm 10.6 \text{ Mg ha}^{-1}$) and TSN ($9.2 \pm 1.48 \text{ Mg ha}^{-1}$). Soil OC and TSN did not differ under enclosure at heavily encroached sites but were twice as high in enclosure compared to open grazing soils at low encroached sites. Mean SOC and TSN in soils of 0 - 20 cm depth were up to 120% higher than that of the 21 - 40 cm soil layer. Soil OC was positively related to TSN, cation exchange capacity (CEC), but negatively to sand content. The results show that SOC and TSN stocks were partly affected by woody encroachment and grazing management but more importantly by inherent soil characteristic such as sand content. On the sites of sandy loam texture soils, maintaining high woody cover may not increase SOC and TSN stocks, or reduce their loss. Hence improving the herbaceous layer cover through a reduction in livestock grazing and woody encroachment restriction are the key strategies to maintain SOC and TSN stocks or reduce their losses and, thereby, for climate change mitigation in semi-arid rangelands.

Key words: Borana rangelands, carbon stock, climate change mitigation, drylands, enclosure, exclosure, grasslands, grazing management, land use change, livestock, savanna, SOC, TSN

4.1 Introduction

Soil is the largest terrestrial reservoir of carbon (C) and nitrogen (N) [1] and can store about three times as much C and N than the atmosphere [2], sequestered mainly in decomposed plant litter and residues. Recent rapid losses of soil C and N due to intensive livestock or agricultural uses and changes of fire regimes have been reported for tropical savannas, which cover ca. 10 to 15% of all terrestrial ecosystems [3 - 4]. These ecosystems, if well managed, may have a high potential to store an appreciable fraction of atmospheric CO₂ as organic carbon (OC) in the soil [4]. Given the vast area cover of savanna systems, enhanced C and N fluxes from these systems linked to land use and cover changes could greatly influence the global C and N cycle, with direct impact on potential climate change mitigation and adaptation strategies [2]. Nevertheless, the ways in which to steer soil C and N stocks in savanna ecosystems are complex and poorly understood as the impacts of land use and associated vegetation cover changes, climate and soils are complex and vary spatially and temporally.

Previous studies have shown mixed results of grazing effects on soil organic carbon (SOC) and soil organic nitrogen (SON), with studies showing positive [5], neutral [6] or negative effects of grazing [7]. Grazers affect SOC and SON by mechanisms that alter C and N cycles in the soil [8]: i. grazers reduce primary productivity [9], ii. alter plant C and N belowground allocation [10], and iii. affect litter quality and mineralization rates [10]. Further, grazers can affect legume abundance and hence N fixation rates, which may alter N inputs to the soil [11]. Ruminant metabolism, C and N emissions from animal wastes through volatilization and leaching impact SOC and TSN stock in the soil [12]. Changes in soil C and N outputs associated with grazers arise mainly from changes in soil organic matter decomposition and mineralization rates [13] or increased erosion under intense grazing [14]. Grazing generally decreases herbaceous plant cover and thus may increase soil organic matter mineralization rates because of greater soil temperature and/or soil moisture fluctuations and by increasing desertification [13]. The effect of grazing on SOC and SON stocks depends on precipitation, soil properties, plant species composition, and grazing intensity [5-14]. Hence, the overall consequences of grazing on SOC and SON accumulation may vary along gradients of these variables and so far only few studies have been conducted on quantifying these effects in semi-arid rangelands of east African pastoral grazing systems.

Over the last century, African savannas have been encroached by woody species despite differing climate and management practices such as variable domestic herbivore stocking rates and fire regime [15]. Woody encroachment, which refers to an increase in

cover, density and biomass of indigenous woody plant species, has been reported over much of the world's arid and semi-arid environments ('drylands') in the recent decades [16]. The causes of woody encroachment include grazing intensification, changes in fire frequency and intensity [17], changes in N deposition [18], increasing atmospheric CO₂ concentration and climate change [19]. Similarly, the structure and composition of semi-arid southern Ethiopian rangeland vegetation has changed dramatically, mainly due to the Ethiopian government fire prohibiting policy and grazing intensification since the 1970s [20]. Large areas of southern Ethiopian rangelands have become encroached by woody plant species, resulting in a substantial reduction of the herbaceous layer and, as a result, pastoral production [21]. While the woody encroachment is often regarded as severe rangeland degradation, particularly in the context of cattle grazing or pastoral production [21], it was also shown to trigger a large increase in C sequestration potential in tropical America [22], Australia [23], and South African [24] savanna systems. This C accumulation appears to be a function of enhanced below- and aboveground net primary productivity (NPP), low decomposition rates beneath shrubs, biochemical recalcitrance of shrub litter, and organic matter stabilization in protected soil aggregates [23]. However, this seems to be precipitation-dependent, i.e., the drier sites in the Chihuahuan desert in USA (< 280 mm rainfall) gained soil C and N with encroachment while wetter sites (> 600 mm rainfall) lost C and N with encroachment [25]. So far, little quantitative information is available on soil C and N stocks capacities and the influence of woody encroachment on these stocks in east African semi-arid savanna systems.

A recent assessment of AGB biomass and woodycover data has indicated an increasing potential for AGC stocks by encroaching woody plant species in a semi-arid Ethiopian rangeland (26). However, it is not yet clear how the influence of this woody encroachment has affected soil C sink and the soil C influx that arises when grazing pressure is relaxed. This information is critically important since more than 70% of an ecosystem C pool is located in the soil [2] and could potentially be influenced by intensification of grazing and vegetation cover changes.

The main aim of this study was to investigate whether soil C and N stock will increase with woody encroachment and under rangeland enclosure. If so, we would expect that severely woody encroached sites will contain the most SOC and total soil nitrogen (TSN) stocks. We further would expect that long-term grazing relaxation (rangeland enclosure) will increase SOC and TSN stocks. Thirdly, we hypothesize that woody encroachment and grazing exclusion will interact and that, thus, severely encroached sites from which grazing has been

excluded for long time would have greater SOC and TSN than sites that are less encroached and grazed.

4.2 Material and Methods

4.2.1 Study area

Study sites were located in a semi-arid pastoral system within approximately 10 - 70 km geographic range in Yabello and Dire Districts, Borana, southern Ethiopia. This semi-arid rangeland is used predominantly for livestock (cattle, camel, goat and sheep) production [27]. The sites represent similar soil types, climatic conditions and livestock population density (Table 4.1). The soils in the study sites comprise a Chromic Cambisol according to FAO/UNESCO system (unpublished data). The study sites are representatives of the recent woody encroachment phenomenon, in which < 40% of the shrubs established before 1970s [28], and the most extensive woody encroachment occurred after the 1980s because of grazing pressure and fire suppression [20].

Fifty years (1957-2012) of climate data (Ethiopian Meteorological Agency) indicated a long-term mean annual precipitation of 550 mm in the region, with a 66% coefficient of variation across years and 85% falling during two growing seasons. Rainfall is bimodal, with 55% of the annual precipitation occurring in March - May, followed by 30% in September - November. Mean annual air temperature is 20°C, with a mean monthly maximum of 21 °C in February and a mean monthly minimum of 18.5°C in July.

Table 4.1. Woody encroachment levels, age of enclosure, soil and livestock population density characteristics of the research sites in the Yabello and Dire districts of Borana zone. Geological information was summarized from the Borana land use study project soil survey report (unpublished data). Woody encroachment levels: Low woody encroachment site (LE), moderate woody encroachment site (ME), severe woody encroachment site (SE), highest woody encroachment (HE).

Encroachment level	Location (latitude, longitude)	Elevation (masl)	Geology	Soil type	Enclosure age	Soil texture (%)			Textural class	Livestock (head km ⁻²)					*TLU km ⁻²
						Sand	Silt	Clay		Cattle	Camel	Goat	Sheep	equines	
LE	04° 56' 33" 38° 10' 12"	1542-1564	Quartz-feldspathic gneiss and alluvium (sand silt and clay)	Cambisols	35	46	10	44	Sandy Clay	30	1.05	11	3	0.7	23.9
ME	04° 24' 02" 38° 17' 03"	1439-1514	Alluvium: sand, silt and clay	Cambisols	30	67	18	15	Sandy Loam	31	0.7	9	10	2	25.6
SE	04° 41' 08" 38° 11' 41"	1436-1518	Quartz-feldspathic gneiss	Cambisols	8	65	20	15	Sandy Loam	25	2.6	18	12	0.5	23.3
HE	04° 38' 58" 38° 04' 58"	1247-1323	Plateau basalt: alkaline basalt and trachyte	Cambisols	12	64	19	17	Sandy Loam	25	2.6	18	12	0.5	23.3

*. Tropical Livestock Unit was calculated using the livestock population data obtained for each site from local Agricultural Offices following [28].

1 TLU=250 kg live weight.

4.2.2 Land use and grazing patterns in Borana

Historically, the land use system in the Borana pastoral system was largely characterized by sustainable exploitation of rangeland resources based on herd mobility in connection with flexible stocking densities [30]. Movement patterns corresponded with local rainfall and associated natural resource productivity, shifting towards dry areas in the wet season and more humid areas in dry seasons [28]. The land use also involved periodic burning of the rangelands [28]. Following the 1970s drought period in the area, several ponds or deep wells were established in some parts of the rangelands and the pastoralists shifted to use the areas near these ponds or deep-wells (permanent water points) for grazing in the dry season and drought years whereas the other parts of the landscape were utilized during the wet season [30].

However, this extensive, the season based rotational grazing system has changed to a semi-sedentary year-round intensive grazing system since the 1980s because of increasing human and livestock populations, water points, roads and market infrastructure development, settlement programs and frequent drought events [20]. The Borana pastoral community was estimated to be 480,000 people in 1980s, increasing with an annual population growth rate of about 2.5 - 3% [31]. The livestock density in 1982, measured by aerial observation, was 14.3 and 11.9 Tropical Livestock Unit (TLU) km⁻² (1 TLU = 250 kg live weight) [29] domestic herbivore stocking rates in the wet and the dry-season, respectively [32]. By 2000, a household based survey provided stocking densities of 45 - 153 TLU km⁻² [33]. Similarly, Homann et al. [2008] estimated 105 and 43 TLU km⁻² during and after the 1999/2000 drought year, respectively [31]. Though the livestock population increase is often dampened by frequent drought events (occurring every 5 to 6 years), a rise in the net livestock density beyond stocking carrying capacity has been reported in the Borana rangelands [31, 33].

More exclusive forms of land ownership have been induced since the mid-1960s by the establishment of traditional rangeland enclosures and government ranches [33]. The former is a small section of grazing land put aside during the wet season by individual pastoral households or the community at large to conserve pasture for calves, heifers, and sick animals during the dry season. Fires have been completely suppressed by pastoralists in the rangelands since the 1970s because of government regulations and because the standing biomass was rather used for forage, to feed high cattle densities [28].

By 1980s, with the expansion of ponds, boreholes and shallow wells and government settlement programs, crop cultivation has drastically expanded into wetter and most valuable grazing areas [31]. By 2000, more than 16% of the total grazing area had been converted to

crop cultivation [33]. Year-round intensive grazing combined with suppression of fire and other climatic factors led to the conversion of grass into shrub-dominated savanna/woodlands [20].

4.2.3 Sampling design

The study was conducted along a gradient of woody plants encroachment representing four levels of woody encroachment in southern Ethiopia pastoral rangelands. The levels were based on the stage of woody encroachment determined through personal interviews with local people and district agricultural office managers, and also supported by ground quantification of the woody plants canopy cover and density (Table 4.2). Woody vegetation structure was quantified by measuring tree/shrub density; canopy diameters, canopy height, and stem height of the woody species using an 8-m long graduated wooden pole. Canopy cover was calculated using the average of the two longest canopy diameters perpendicular to each other and parallel to the ground. Stem height was measured as the total height of the plant stem from the ground level to the highest foliage. These data were used to compute tree and shrub densities and canopy cover per hectare for the grazing regimes and encroachment levels (Table 4.2). The woody encroachment levels were arranged from low to highest encroachment. The site of low encroachment (LE) has a mosaic of tree and shrub patches in a perennial herbaceous species stand, with an average total woody canopy cover of 27%. The site with moderate encroachment (ME) is composed of dwarf shrubs and thick perennial grass dominated stands with an average total woody canopy cover of 56%. The site with highest level of encroachment (HE) has small to medium-sized shrubs and trees that form an almost impenetrable thicket with a canopy cover 72% (Table 4.2, Fig.4.1). The site with severe encroachment (SE) contains fully matured tree and shrub stands with a woody canopy cover of 62% of the ground, in which herbaceous plants have been almost eliminated (Table 4.2, Fig.4.1).

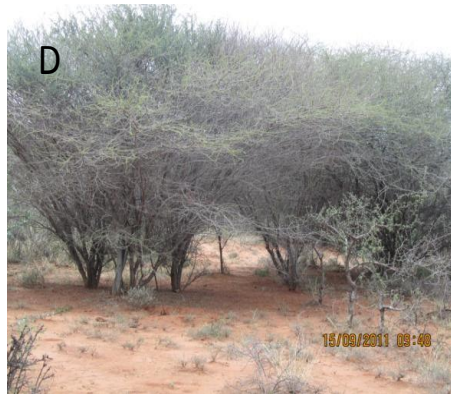
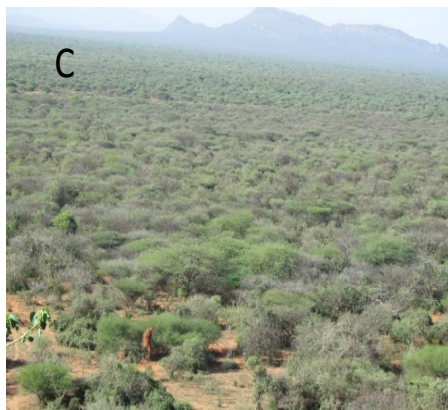
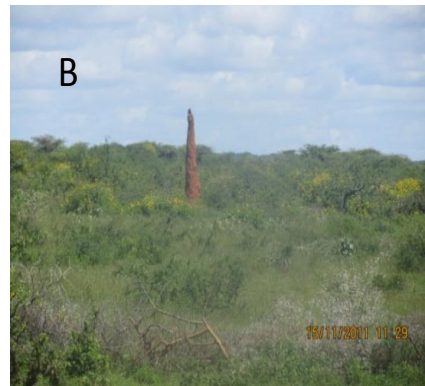


Fig. 4.1. Study sites: low woody encroachment site (A), moderate woody encroachment site (B), severe woody encroachment site (C), high woody encroachment site (D) in the semi-arid rangelands of Yabello and Dire districts, Borana, southern Ethiopia. Pictures taken by HasenYusuf in 2011.

Table 4.2. Average values of vegetation characteristics of the grazing regimes for each encroachment level in the Yabello and Dire districts of Borana, southern Ethiopia. Low woody encroachment site (LE), moderate woody encroachment site (ME), severe woody encroachment site (SE), highest woody encroachment site (HE). Open = open access grazing land, enclosure = areas of livestock enclosure, reserved for heifers and calves only in the dry season. Canopy cover and woody density were assessed from 160 plots of 10m × 10m, 20 plots from each grazing regime.

Variables	LE			ME			HE			SE		
	Open	Enclosure	Average	Open	Enclosure	Average	Open	Enclosure	Average	Open	Enclosure	Average
Tree canopy cover* (%)	0	10	5	19	7	13	16	20	18	46	18	32
Shrub canopy cover (%)	9	34	22	40	45	43	48	60	54	23	36	30
Total woody canopy cover (%)	9	44	27	59	52	56	64	80	72	69	54	62
Herbs canopy cover	70	62	66	72	71	72	51	67	59	24	62	43
Tree density ha ⁻¹	0	74	36	111	32	71	170	214	192	200	116	158
Shrubs density ha ⁻¹	616	1532	1074	1732	2368	2050	1425	1895	1660	865	1215	1040
Total density	616	1606	1110	1843	2400	2121	2065	2109	1852	1065	1331	1198

* Canopy cover is the proportion of the ground area covered by the vertical projection of the tree/shrub/herb canopy

Each woody encroachment level was divided into two grazing regimes namely, rangeland enclosure (hereafter named 'enclosure') and communal grazing land (hereafter named 'open') (Fig.4.2). The open grazing land represents the most common land use system in the Borana rangelands and is defined as the communal rangelands that are not privately owned, yet belonging to the communities whose members have equal access rights to the communal resources [34]. Enclosures in this study represent a shrub fenced area that covers 10 - 25 ha grazing land and protects from grazing during the wet season, while the adjacent openly grazed rangelands are utilized, although some grazing may occur in the enclosure in the late dry season and in drought years when the forage is extremely scarce [35].

We randomly selected four replicate of enclosures within the same age group and 10 - 25 ha in size and 1-2 km apart (aerial distance, measured using Garmin GPS 72 (Garmin International Inc., USA) and adjacent open grazing lands in each of the four encroachment levels to examine the influence of grazing exclusion across the gradients of woody encroachment (Fig.4.1). The replicates in each site were located on similar lithology, soils, topography and slope. To measure soil and vegetation attribute within each grazing regime we established three belt-transects (10 m width \times 1000 m length) at 300 - 500m apart. Along each transect, four (10 \times 10 m) plots were established at a 200 m interval along the linear belt transect using meter tape, GPS and compass (4 woody encroachment level \times 2 grazing regime \times 4 replicate grazing regimes \times 5 plots). Previous studies in the present study area have shown that plot sizes $< 100 \text{ m}^2$ were effectively used for sampling shrub dominated vegetation [36]. To assess herbaceous species composition, biomass, and diversity inside and outside the enclosures, caged subplots of (1m x 1m = 1m^2) were randomly nested within the larger 100 m^2 plots used for woody species.

4.2.4 Sampling and laboratory analysis

Vegetation sampling was done at the end of the long and short rainy seasons (end of May and mid December, respectively), soil samples were collected at the end of the long rainy season. Herbaceous AGB was destructively quantified [37]; grass and forb species rooted within the caged 1 m^2 subplot were harvested to ground level, oven dried (at 60°C for 48 h) and weighted using a 0.1 g scale. Four soil core samples at 0 - 20 and 21 - 40 cm depth were collected from each corner of the subplots using intact soil core sampler of 6.5 cm diameter and using the intact soil core sampling method [38]. Samples of the same depth were mixed thoroughly in a large bucket in order to obtain one composite soil sample per depth increment per plot [38]. The soil samples to the depth increments excavated in pits were removed from the hole and extracted before the core was augered to the next depth increment to minimize

compaction of each depth increment. The device also allowed estimating the bulk density of each soil depth increment from intact soil core samples [38, 39]. Soil cores were placed in plastic lined paper bags and oven dried (70°C) until constant weight [40]. Cores were sieved through a 2 mm sieve, and fine soil (< 2 mm), coarse roots (> 2 mm), and gravel /rocks (> 2 mm) were separated and weighed to the nearest 0.01 g. Coarse herbaceous roots and other belowground organic material were separated both visually and by floatation methods from soils, rocks, and gravel and oven dried (60°C) to constant weight [39]. The carbon (C) content of the above- and belowground vegetation biomass estimated as 47% of the dry mass [41]. The fine soil (< 2 mm) fraction was then ground using an analytical mill (IKA[®], Model A10) [39]. A fraction of a soil samples were treated with 0.1 M HCl before analysis to test for inorganic carbon. Samples which tested positive for inorganic C were completely digested with 0.1M HCl to remove inorganic C [39].

Standard analytical procedures of the Ethiopian National Soil Testing Center were used for all chemical and physical analyses. SOC was determined using the Walkley–Black method [42] and TSN was determined using Kjeldahl [43]. Ammonium and sodium acetate extracts were used to determine exchangeable cations (EC) and cation exchange capacity (CEC) [44], pH and electric conductivity (EC) were determined using a suspension of 1:5 soil:water. Particle size analyses were determined using the Hydrometer method [45]. Bulk density (g m^{-3}) was calculated as the mass of the fine soil (< 2 mm) fraction divided by the volume of the entire core to avoid overestimating the mass of the soil when stones and gravels were present [46]. Percent SOC and TSN were multiplied by each sample fraction mass to obtain total SOC and TSN per core sample [39]:

$$SOC (\text{kg ha}^{-1}) = \text{Mass} < 2\text{mm soil (kg)} / \text{Volume of core (cm}^3) * d * cf * C \quad (1)$$

$$TSN (\text{kg ha}^{-1}) = \text{Mass} < 2\text{mm soil (kg)} / \text{Volume of core (cm}^3) * d * cf * N \quad (2)$$

Where d = depth (cm), C = organic carbon concentration, N = total nitrogen concentration and cf is the conversion factor = $(\text{kg cm}^{-3}) * (10,000 \text{ cm}^2 \text{ m}^{-2}) * (10,000 \text{ m}^2 \text{ ha}^{-1})$.

4.2.5 Statistical analyses

The influences of grazing, woody encroachment and soil depth and their interactions on SOC % and SOC stock, herbaceous root OC content, TSN % and total TSN stock, SOC:TSN ratio, and soil bulk density were evaluated using SAS version 9.1 mixed model procedures (Proc MIXED). Differences in all response variables were evaluated by treating woody encroachment level as main effect, grazing management regime and soil depth was nested within woody encroachment level, age of enclosures was considered as random effect. Mean

comparisons were made using Tukey's test ($p < 0.05$). All values reported are means (\pm SE). Linear regressions were used to determine the relationship between SOC and TSN concentration, soil texture, cation exchange capacity (CEC), pH, and soil bulk density.

4.3 Results

4.3.1 Carbon stock in the herbaceous vegetation

Significant differences in herbaceous AGC and root stocks (HRC) were found between open and enclosure area with the exception of the LE site which had lower stocks in a comparison to the other sites (Table 4.3 and Table 4.4). However, mean herbaceous aboveground -and root C stocks did not show a consistent trend of decrease or increases with the increase in the levels of woody encroachment. More than 85% of herbaceous root biomass C storage was found in the top 20 cm soil depth and its vertical distribution in the 21 - 40 cm soil depths was not significantly affected by grazing management and woody encroachment (Table 4.3, Table 4.4).

Table. 4.3. Results from analysis of variance of the effects of woody encroachment and grazing regime and soil depth on herbaceous root biomass carbon (HRC) and herbaceous aboveground C (HAGC) stocks.

Variables	Sources	Num df	F	p
HRC	Encroachment	3	2.5	NS
	Grazing	1	0.2	NS
	Soil depth	1	34.9	**
	Encroachment \times depth	3	1.6	NS
	Encroachment \times grazing	3	4.5	*
	Grazing \times depth	1	0.1	NS
HAGC	Encroachment	3	21	**
	Grazing	1	11	**
	Encroachment \times grazing	3	5	**

** highly significant at $p < 0.01$

*significant at $p < 0.05$

NS not significant at $p < 0.05$

Table 4.4. Mean (\pm SE) herbaceous root C (HRC) and herbaceous aboveground C (HAGC) stocks in enclosures and open grazing land across four levels of woody encroachment. Encroachment levels: Low woody encroachment site (LE), moderate woody encroachment site (ME), highest woody encroachment site (HE), severe woody encroachment site (SE). N, replication of grazing regimes per site. Different lowercase letters represent statistical differences determined by the interaction of grazing regime and woody encroachment terms in the mixed Model and Tukey's means comparisons ($P < 0.05$)

Encroachment level	Grazing regime	N	HRC (Mg ha ⁻¹)			HAGC (Mg ha ⁻¹)
			0 – 20cm	21 – 40cm	0 - 40 cm	
LE	Open	4	1.01 \pm 0.3	0.04 \pm 0.04	1.06 \pm 0.4 ^b	0.97 \pm 0.06 ^b
	Enclosure	4	1.53 \pm 0.4	0.08 \pm 0.05	1.61 \pm 0.4 ^c	1.65 \pm 0.15 ^c
ME	Open	4	0.49 \pm 0.1	0.03 \pm 0.02	0.52 \pm 0.3 ^a	0.37 \pm 0.03 ^a
	Enclosure	4	0.20 \pm 0.2	0 \pm 00	0.29 \pm 0.7 ^a	0.29 \pm 0.03 ^a
HE	Open	4	0.50 \pm 0.5	0 \pm 0	0.50 \pm 0.3 ^a	0.55 \pm 0.06 ^a
	Enclosure	4	1.84 \pm 0.5	0.12 \pm 0.07	1.96 \pm 0.4 ^{cd}	0.90 \pm 0.06 ^a
SE	Open	4	0.47 \pm 0.3	0.08 \pm 0.08	0.54 \pm 0.1 ^a	0.03 \pm 0.01 ^d
	Enclosure	4	1.11 \pm 0.9	0.26 \pm 0.13	1.37 \pm 0.5 ^d	0.17 \pm 0.01 ^a

4.3.2 Soil organic carbon and nitrogen

The mean total soil organic carbon stock (SOC stock) and total soil nitrogen stock (TSN stock) in 0 - 40 cm soil depth ranged from 30 ± 2.1 and 5 ± 0.57 Mg ha⁻¹ respectively, in the open grazing soils at the HE site to 81.0 ± 10.6 and 9.2 ± 1.48 Mg ha⁻¹ respectively, in the enclosure soils at the LE site (Table 4), with low variances except at LE (Fig. 4.2 A and B).

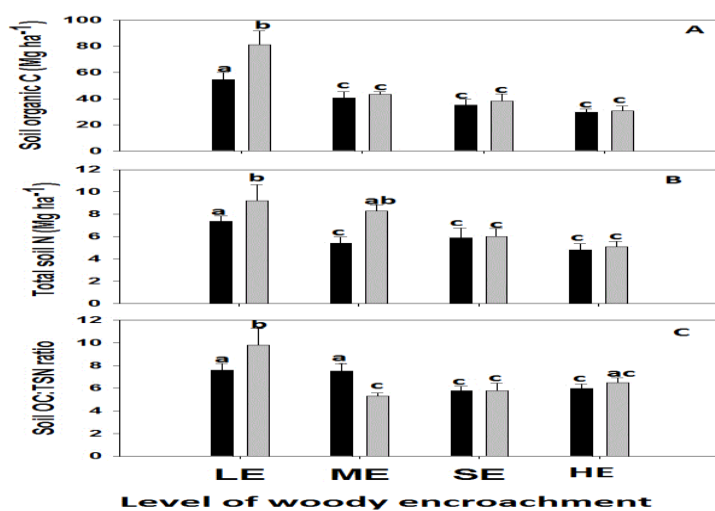


Fig. 4.2. Means and standard errors by levels of woody encroachment and grazing regime (grey bars = enclosure, black bars = open grazing land) for soil organic carbon (A), total soil nitrogen content (B), and soil organic carbon to total soil nitrogen ratio (C). Low woody encroachment site (LE), moderate woody encroachment site (ME), highest woody encroachment site (HE), severe woody encroachment site (SE). Different lowercase letters represent statistical differences determined by woody encroachment by grazing interaction term in the mixed model and Tukey's means comparisons ($P < 0.05$).

Soil OC stock for 0-40 cm had been significantly affected by grazing regime, woody encroachment levels and their interactions (Table 4.4). Total SN stocks for 0-40cm had significantly affected by woody encroachment levels and encroachment level by grazing regime interaction (Table 4.4), suggesting that increasing the level of woody encroachment affected the response of SOC and TSN stocks to grazing regime. The LE site tended to have higher SOC and TSN stock in the enclosures while the mean SOC and TSN stocks in the enclosure at ME, SE, HE was statistically the same as the adjacent open grazing area (Table 4.5, Fig. 4.2)

Mean SOC and TSN stocks tended to be higher in the 0-20 cm soil layer, independently of the level of woody encroachment and grazing regime (Table 4.5). The 0 – 20 cm soil layer tended to have twice as much SOC and TSN stock in the enclosures at low woody encroachment level while the mean SOC and TSN stocks in the 0-20cm at all other treatment was not statistically different from the 21-40cm soil layer (Table 4.5).

SOC:TSN ratio had been significantly affected by woody encroachment levels, soil depth and interaction of woody encroachment and grazing regime (Table 4.4) with enclosures showing both higher and lower ratios at the LE and ME sites but remaining relatively unchanged at severely SE and HE sites, despite an increase in percent SOC and percent TSN concentrations (Table 4.5).

Table 4.4. Results from analysis of variance of the effects of woody encroachment and grazing regime and soil depth on soil bulk density (BD), percent soil organic carbon (SOC %), soil organic carbon (SOC) stocks, percent total soil nitrogen (TSN %), total soil nitrogen (TSN) stock.

Variables	Sources	Num df	F	p
SOC (%)	Encroachment	3	29	**
	Grazing	1	2	**
	Depth	1	43	**
	Encroachment x depth	3	0,8	NS
	Encroachment x grazing	3	6	*
	Grazing x depth	3	1	NS
SOC (Mg ha ⁻¹)	Encroachment	3	9	**
	Grazing	1	9	**
	Depth	1	41	**
	Encroachment x depth	3	0.2	NS
	Encroachment x grazing	3	4	*
	Grazing x depth	3	0.1	NS
TSN (%)	Encroachment	3	2	**
	Grazing	1	0.1	NS
	Depth	1	45	**
	Encroachment x depth	3	4	*
	Encroachment x grazing	3	0.4	NS
	Grazing x depth	3	0.1	NS

TSN (Mg ha ⁻¹)	Encroachment	3	11	**
	Grazing	1	0.5	NS
	Depth	1	36	**
	Encroachment x depth	3	4	*
	Encroachment x grazing	3	0.7	NS
	Grazing x depth	3	0.2	NS
SOC:TSN	Encroachment	3	5	**
	Grazing	1	2	*
	Depth	1	4	**
	Encroachment x depth	3	2	*
	Encroachment x grazing	3	3	*
	Grazing x depth	3	0.4	NS
Soil bulk density	Encroachment	3	46	**
	Grazing	1	22	**
	Depth	1	4	*
	Encroachment x depth	3	04	NS
	Encroachment x grazing	3	17	**
	Grazing x depth	1	0.7	NS

** highly significant at $p < 0.01$

*significant at $p < 0.05$

NS not significant at $p = 0.05$

Table 4.5. Mean (\pm SE) soil bulk density (BD), percent soil organic carbon (SOC %), soil organic carbon (SOC) stocks, percent total soil nitrogen (TSN %), total soil nitrogen (TSN) stock for two soil depths in enclosures and open grazing land across four levels of woody encroachment (for characteristics see also Table 1). Encroachment levels: Low woody encroachment site (LE), moderate woody encroachment site (ME), highest woody encroachment site (HE), severe woody encroachment site (SE). N, replication of grazing regimes per site. Different lowercase letters represent statistical differences determined by the grazing regime and woody encroachment interaction term in the mixed Model and Tukey's means comparisons ($P < 0.05$)

Encroachment level	Grazing regime	Depth (cm)	N	BD (g cm^{-3})	TSN (%)	SOC (%)	SOC:TN	TSN (Mg ha^{-1})	SOC (Mg ha^{-1})
LE	Open	0 - 20	4	1.3 \pm 0.10	0.13 \pm 0.01	1.00 \pm 0.10	7.6 \pm 0.60	3.9 \pm 0.27	29.5 \pm 2.98
		21 - 40	4	1.5 \pm 0.08	0.13 \pm 0.01	0.90 \pm 0.09	6.2 \pm 0.60	3.5 \pm 0.21	24.9 \pm 2.53
		Total						7.4\pm0.47^a	54.4\pm6.51^a
	Enclosure	0 - 20	4	1.0 \pm 0.04	0.21 \pm 0.03	1.85 \pm 0.24	9.8 \pm 1.50	6.1 \pm 1.05	55.6 \pm 8.18
		21 - 40	4	1.1 \pm 0.05	0.11 \pm 0.02	0.94 \pm 0.10	9.1 \pm 1.40	3.1 \pm 0.43	25.4 \pm 2.42
		Total						9.2\pm1.48^b	81.0\pm12.60^b
ME	Open	0 - 20	4	1.3 \pm 0.07	0.13 \pm 0.01	0.97 \pm 0.09	7.5 \pm 0.63	3.2 \pm 0.36	23.1 \pm 2.20
		21 - 40	4	1.3 \pm 0.05	0.09 \pm 0.01	0.72 \pm 0.11	7.9 \pm 0.72	2.2 \pm 0.23	17.3 \pm 2.74
		Total						5.4\pm0.59^c	40.4\pm3.94^c
	Enclosure	0 - 20	4	1.3 \pm 0.05	0.23 \pm 0.01	1.2 \pm 0.07	5.3 \pm 0.30	5.0 \pm 0.33	26.1 \pm 1.45
		21 - 40	4	1.4 \pm 0.04	0.15 \pm 0.01	0.77 \pm 0.02	5.4 \pm 0.37	3.3 \pm 0.23	17.2 \pm 0.69
		Total						8.3\pm0.56^{ab}	43.3\pm3.14^c
HE	Open	0 - 20	4	1.2 \pm 0.03	0.13 \pm 0.01	0.73 \pm 0.05	6.0 \pm 0.40	3.1 \pm 0.39	17.3 \pm 1.22
		21 - 40	4	1.4 \pm 0.04	0.08 \pm 0.01	0.48 \pm 0.03	6.3 \pm 0.32	1.2 \pm 0.17	12.5 \pm 0.93
		Total						4.8\pm0.57^c	29.8\pm3.70^c
	Enclosure	0 - 20	4	1.3 \pm 0.05	0.13 \pm 0.02	0.83 \pm 0.11	6.48 \pm 0.46	3.0 \pm 0.29	19.6 \pm 2.21
		21 - 40	4	1.1 \pm 0.06	0.07 \pm 0.01	0.45 \pm 0.04	6.82 \pm 0.56	2.0 \pm 0.19	11.1 \pm 1.36
		Total						5.1\pm0.48^c	30.7\pm4.90^c
SE	Open	0 - 20		1.7 \pm 0.04	0.12 \pm 0.02	0.70 \pm 0.11	6.0 \pm 0.36	3.5 \pm 0.55	20.9 \pm 3.15
		21 - 40		1.5 \pm 0.04	0.08 \pm 0.01	0.47 \pm 0.05	5.8 \pm 0.24	2.3 \pm 0.34	14.1 \pm 1.27
		Total						5.9\pm0.88^c	35.0\pm4.42^c
	Enclosure	0 - 20		1.6 \pm 0.03	0.12 \pm 0.02	0.70 \pm 0.09	6.5 \pm 0.64	3.6 \pm 0.50	21.4 \pm 2.49

21 - 40	1.8±0.05	0.07±0.01	0.46±0.06	7.6±0.72	2.4±0.23	17.0±2,77
Total					6.0±0.74^c	38.3±5.26^c

4.3.3 Soil bulk density

Significant differences in soil bulk density was not found between open and enclosure area with the exception of the LE site which had lower soil bulk density in the enclosures compared to openly grazed area (Table 4.4 and Table 4.5). The bulk density tended to be lower in 0 - 20 than in 21 - 40 cm soil depths in less woody encroached sites (LE and ME sites), while it was not consistent for severely encroached sites (SE and HE) sites (Table 4.5). The bulk density in the 0 - 20 cm soil ranged from $1.0 \pm 0.04 \text{ gm}^{-3}$ (in LE, enclosure) to about 70% higher values in the enclosure and open grazing land of SE site (Table 4). The deeper soil (21 - 40 cm) bulk density ranged from $1.1 \pm 0.05 \text{ gm}^{-3}$ (in LE, enclosure) to $1.8 \pm 0.05 \text{ gm}^{-3}$ (in SE, enclosure).

4.3.4 Factors related to SOC retention

Percent SOC was linked to the %TSN (0 - 20 cm soil depth) for all study areas (Fig 4.4A). Soil OC concentration (% SOC) were significantly but weakly negatively related to % soil sand contents and bulk density (Fig. 4.4 C and D).

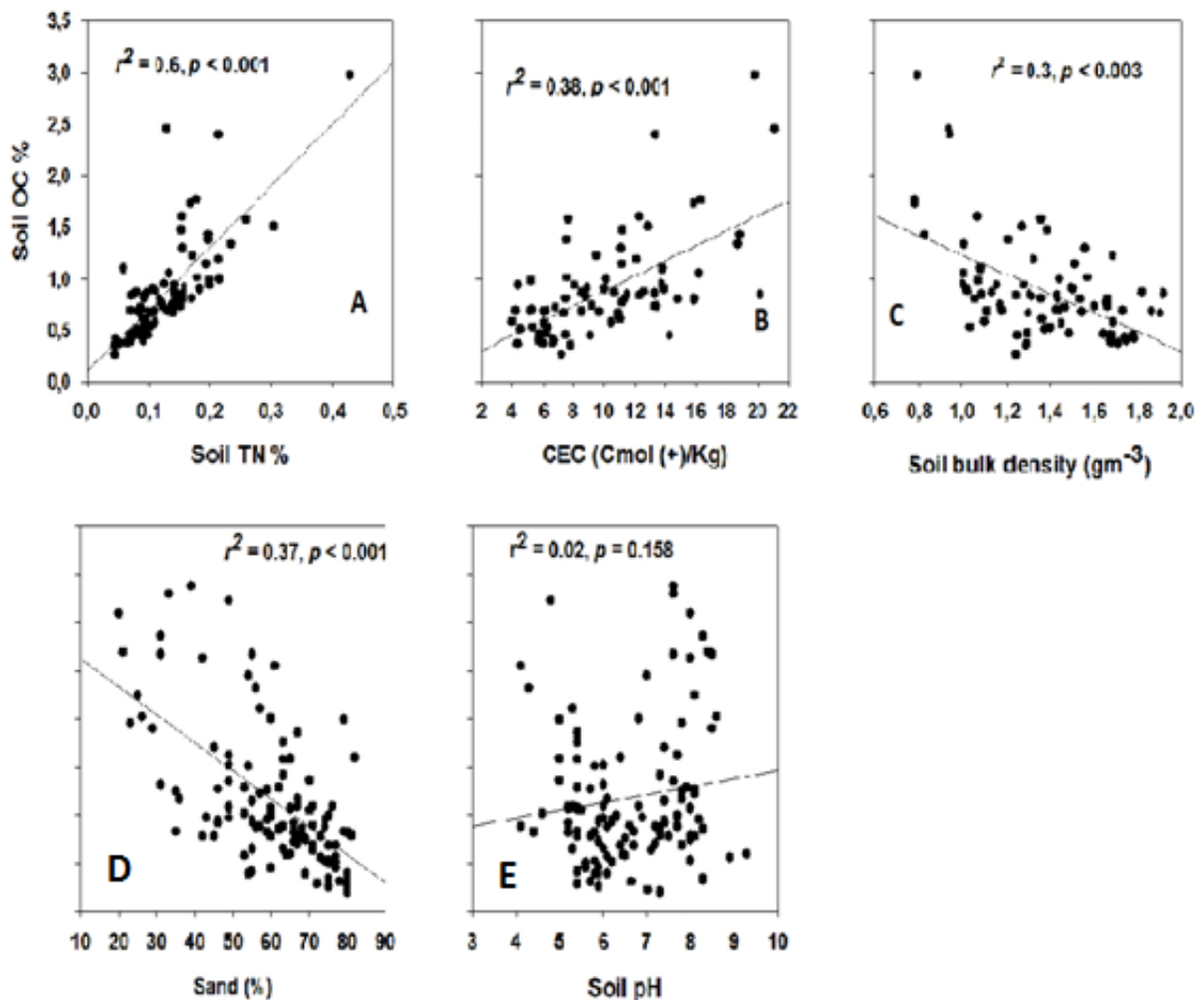


Fig. 4.4. Regression of soilorganic C concentration (%SOC) against total soil nitrogen(%TSN,A), cation exchange capacity (CEC; B), soil bulk density (C), sand fraction (% sand; E) and soil pH (F) within a depth of 0 - 20 cm. r^2 and P values are given for each plate.

4.4 Discussion

4.4.1 Effects of grazing on SOC and TSN stocks

Our results showed that the response of herbaceous above- and belowground biomass C stocks to grazing was strongly influenced by the woody encroachment site characteristics. The pattern of the herbaceous above- and belowground biomass C stocks response to grazing in most of the woody encroachment sites (e.g., LE, HE and SE sites) agree with a herbaceous biomass decrease observed in other semi-arid environments [9, 38]. Angassa and Oba, (2010) reported an increase of about 64% in mean herbaceous AGB in the enclosure compared to surrounding open grazing lands after 15 - 25 years of livestock exclusion in Borana [36]. A more than 200% increase in herbaceous AGB within the enclosure was also reported from 5 - 15 years enclosures in northern Ethiopian rangelands [47]. Bagchie and Ritchie (2010) reported a 32 – 33% increase in AGC stocks and a 21 – 63% increase in root biomass C in enclosures compared to open grazing land in the Trans-Himalayas pastoral system [48]. Schuman et al.(1999) observed a 20 – 52% and 7–16% increase of C and 15 – 30% and 18 – 52% increase of N in AGB and roots (0 – 60 cm depth), respectively, after 12 years of enclosure on a native mixed grassland in Wyoming, USA [49].

In our study, however, we found that the effects of grazing on herbaceous above- and belowground biomass C stock were significantly influenced by woody encroachment. In low encroached site, grazing influences on the herbaceous above- and belowground biomass C stocks were clearly visible. However, the lack of significant grazing impacts on herbaceous above- and belowground biomass C stocks at moderate woody encroachment site (LE) may be associated with the relatively higher woody density and canopy cover in the enclosure compared to adjacent open grazing land at this site (Table 3.2) that indicates woody encroachment could have a significant effect on the expression of grazing in the dynamics of herbaceous above- and belowground biomass C stocks. These results are consistent with other studies that have shown the significant decrease of the understory herbaceous vegetation productivity with an increase in woody density and cover [21, 49]. This decrease may be linked to the competitive advantage of encroacher woody species for soil moisture through their deep root systems and rainfall interception by shrub/tree canopies which could reduce

available soil moisture in arid and semi-arid environments where rain falls mostly occurs as small events, a response that may increase competitive effects under tree/shrub canopies [50]. Generally, though influenced by woody encroachment, the greater accumulations of herbaceous biomass C in most of our enclosures studied suggest that there is potential to store more C in the rangelands by reducing grazing intensity.

Our result demonstrated that the response of SOC and TSN to grazing interacted strongly with woody encroachment sites suggesting increasing the level of woody encroachment and other related site variation (e.g., soil texture) may affect the response of SOC and TSN stocks to grazing. Soil OC and TSN stocks were significantly higher in the enclosures than in the open grazed in low woody encroachment level on sandy clay soils, but the difference in SOC and TSN was statistically the same in moderate to highest encroachment levels on sandy loam soils. The increase may be related to increased vegetation (woody and herbaceous) production, litter quality and nutrient cycling [52], and the ability of the soil to retain the extra N after exclusion of herbivory [53]. Higher grazing intensity is generally thought to decrease soil C and N by direct removal of AGB, i.e., reduction of potential CO₂ fixation in photosynthetic tissue and reduction in belowground C inputs through lower root production and higher root litter turnover [9]. Our result is in agreement with Mekuria (2013) who reported increased soil organic matter (SOM) and TSN after grazing exclusion for 5-10 years in northern Ethiopia [54]. Similarly, studies from Central Asia, found a significant decrease of SOC and TSN due to intensive grazing in semi-arid environments [7, 55]. Cumulative root biomass not only increases soil C inputs but also N retention within the soil [57] because both organic N and C changes are closely linked in the SOM [13]. Hence, the allocation of N in root biomass and tight cycling within the soil has been suggested as a mechanism that can reduce N loss [57]. The higher N concentration and TSN stocks in our enclosures soils might be a result of lower N losses via volatilisation of ammonia and nitrate through animal urine and dung patches and, thereby, an enhanced in N availability for SOM formation and storage [53]. Higher N losses will decrease N concentration in the soil and limit SOM formation and SOC storage in the open grazed system [54].

High SOC and TSN stocks in the enclosures can also potentially improve soil physical properties such as soil structure and bulk density, which in turn may increase water infiltration rates into the soil [54, 58]. In our study, soil bulk density was lower in the enclosures compared to the open grazing land and the difference was particularly high at LE site, which may be linked to the fine, sandy clay textured soils of this site. The effect of grazing intensity on bulk density is especially remarkable in wet and fine textured soils [14] as it is susceptible

to soil compaction caused by trampling through livestock [14, 58]. Soil compaction potentially can reduce water infiltration and increase runoff which often resulted in decreasing water availability for plant growth. In addition this can lead to loss of fertile top soil and nutrients especially under heavy grazing conditions [14, 54]. As a result this can reduce plant productivity and SOC and TSN storage as observed in most of our open grazing lands. Increased soil erosion due to a decrease in vegetation cover associated with continuous, heavy grazing was reported as the main causes for the loss of soil OM in many parts of African and Central Asian grasslands [7, 14, 55]. The lack of significant differences in SOC and TSN between the grazing regimes at severely encroached sites (e.g., SE and HE sites) may be the result of both labile and minerals associated OM loss in the top soil due to livestock trampling induced soil erosion, which increase the negative effects of heavy grazing on herbaceous productivity and C and N inputs [59]. Our findings suggest that intensified grazing decreases SOC and TSN stocks, and the losses from the top soil layer may not effectively be restored by short period (< 15 years) grazing exclusion at severely woody encroached sites, particularly on coarse, sandy loam textured soils, which are less resistant to rainfall (e.g., at HE and SE sites).

4.4.2 Effect of woody encroachment on SOC and TSN stocks

On average, our SOC stocks of about 44 Mg ha⁻¹ fall within the range reported by earlier studies for tropical woodland and savanna ecosystems, i.e., 20 - 80 Mg ha⁻¹ [60, 61]. Our results demonstrated that SOC and TSN stocks have declined with the increase of woody encroachment level. Several qualitative and quantitative indicator data that we collected at HE and SE sites included (i) high bare soil cover, i.e., 40% and 57%, respectively, (ii) low herbaceous (grasses and herbs) ground cover, i.e., 60 % and 40 %, respectively, (iii) exposed tree roots (pers. obs.), (iv) a similar SOC and TSN stocks-soil depth relationship of upper and lower soil layers, and (v) similar soil bulk density within the top soil (< 20 cm soil) of HE and SE sites with the sub soils (> 20 cm soil) at LE and ME sites (Table 4.5), indicate the likely loss of organic matter with the top soils by erosion at the severely shrub encroached sites suggesting woody encroachment management is as important as that of grazing to restore herbaceous aboveground -and root C, SOC and TSN stocks in this semiarid rangelands. Previous study from the same region had shown a 30 - 61% lower OM in the top 10 cm soil layer in woody encroached sites compared to open grasslands [62]. The finding also concurs with Guo and Gifford (2002) who showed SOC stock losses when grassland was converted to plantations in New Zealand rangeland systems [67]. Jackson et al., (2002) also reported a decrease in SOC stocks in semiarid grasslands experiencing woody encroachment and

associated the reduction in SOC stocks to the incorporation of soil N to aboveground woody plant parts [25]. Hudak et al., (2003) linked the reduction of SOC and TSN stocks at severely woody encroached site in South African dry savanna to a reduction of herbaceous root production caused by woody encroachment [63]. Woody plant species once established often outcompete herbaceous species, reduce the herbaceous above- and belowground biomass [51] that might further expose top soil to livestock trampling and rain. Schlesinger et al. (1990) similarly showed that the bare inter-space between woody plants experiences higher temperatures leading to higher evapotranspiration, resulted to a slow organic N incorporation, ammonia volatilization and increased soil erosion (6 4).

However, several studies have shown that many other biotic and abiotic factors can determine SOC and TSN stocks (Jobbagy and Jackson, 2000). Soil properties can influence SOC concentrations and the occurrence of woody encroachment itself (Sankaran et al 2005, Archer et al., 2001). For example, Archer et al. (2001) indicated that soil texture strongly influences where *Prosopis* can establish in a southern Texas savanna rangeland [68]. Similarly Vågen and Winowiecki, (2013) has shown inherent high soil sand fraction strongly limits SOC stocks in East African savanna and woodlands, independently of climatic factors and vegetation type differences [60]. Similarly, our study suggested that soil texture could played an important role for the low SOC and TSN stocks observed in the heavily encroached sites as the sand content independently explained 37% of the variations in % SOC and % TSN across the sites. High soil sand content often associated with less adsorption and stabilization of organic matter [63]. In contrast, soils with higher clay content form tight aggregates that protect SOC from microbes [64]. The historical land use pattern and disturbance, including soil erosion, condition and productivity of the sites before the occurrence of woody encroachment may also influence the variation in SOC and TSN stocks across the sites [69]. Hence, the initial causes of top soil losses due to erosion in severely woody encroached sites may stem from long term overgrazing and livestock trampling rather than woody encroachment given the common notion that the latter has been considered a symptom of grazing pressure induced rangeland degradation [28].

Further, semi-arid ecosystems generally have extreme rainfall events that can be highly erosive [61]. Therefore, the low SOC and TSN in the severely shrub encroached sites of our study area might be linked to a high prevalence of soil erosion caused by confounding effects of long term grazing, i.e., livestock trampling in addition to the impacts of high shrub cover on understory herbaceous vegetation productivity. Lack of sufficient replications in our experimental design limits us to isolate the role of woody encroachment for SOC and TSN

storage decline at heavily encroached sites in the presence of uncontrolled these many potential factors (e.,g., soil texture) that could determine the SOC stocks in these semiarid rangelands. On the basis of our data presented here SOC and TSN stocks tended to decrease as result of the expansion of woody encroachment into semiarid savanna ecosystem. However, as savanna soils, vegetation structure and climate are highly variable and the SOC and TSN storage can be determine by these factors further field studies will be needed to evaluate the large scale net effects of woody plants encroachment and site characteristics on SOC storage in the Borana rangeland ecosystem.

4.5 Conclusion

The results show that SOC and TSN stocks were partly affected by woody encroachment and grazing management but more importantly by inherent soil characteristic such as sand content. On the sites of sandy loam texture soils, maintaining high woody cover may not increase SOC and TSN stocks, or reduce their loss. Hence improving the herbaceous layer cover through a reduction in livestock grazing and woody encroachment restriction are the key strategies to maintain SOC and TSN stocks or reduce their losses and, thereby, for climate change mitigation in semi-arid rangelands.

4.6 References

1. Chapin FS, McFarland J, McGuire AD, Euskirchen ES, Ruess RW, et al. (2009) The changing global carbon cycle: linking plant-soil carbon dynamics to global consequences. *J Ecol* 97: 840–850.
2. Lal R (2004) Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. *Science* 304: 1623-1627.
3. Reid RS, Thornton PK, McCrabb GJ, Kruska RL, Atieno F, et al. (2004) Is it possible to mitigate greenhouse gas emissions in pastoral ecosystems of the tropics? *Environ Dev Sustain* 6: 91–109.
4. Scholes RJ, Hall DO (1996) The carbon budget of tropical savannas, woodlands and grasslands. in: Breymer AI, Hall DO, Melillo JM, Agren GI (Eds.) *Global Change: Effects on Coniferous Forests and Grasslands*. John Wiley and Sons, chichester, UK, pp.69-99.
5. Reeder JD, Schuman GE (2002) Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environ Pollut* 116: 457-463.
6. Shrestha G, Stahl PD (2008) Carbon accumulation and storage in semi-arid sagebrush steppe: effects of long-term grazing exclusion. *Agric Ecosyst Environ* 125: 173–181.
7. Pei SF, Fu H, Wan CG (2008) Changes in soil properties and vegetation following enclosure and grazing in degraded Alxa desert steppe of Inner Mongolia, China. *Agric Ecosyst Environ* 124: 33–39.
8. Baisden WT, Amundson R (2003) An analytical approach to ecosystem biogeochemistry modeling. *Ecol Appl* 13: 649–663.
9. Reeder J D, Schuman J E, Morgan J A, Lecain D R, (2004) Response of organic and inorganic carbon and nitrogen to long-term grazing of the shortgrass steppe. *J Environ Manage* 33: 485– 495.
10. Semmartin M, Di Bella C, de Salamone IG (2010) Grazing-induced changes in plant species composition affect plant and soil properties of grassland mesocosms. *Plant Soil* 328: 471-481.
11. Allard VP, Newton CD, Liefvering M, Clark H, Matthew Cet al. (2003) Nitrogen cycling in grazed pastures at elevated CO₂: N returns by ruminants, *Glob Chang Biol* 9: 1731 – 1742.
12. Piñeiro G, Paruelo JM, Oesterheld M, (2006) Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of Southern South America. *Glob Chang Biol* 12:1267-1284.
13. Wang C, Wan S, Xing X, Zhang L, Han X (2006) Temperature and soil moisture interactively affected soil net N mineralization in temperate grassland in Northern China. *Soil Biol Biochem* 38, 1101-1110

14. Savadogo P., Sawadogo L, Tiveau D, (2007) Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. *Agric Ecosyst Environ* 118: 80-92.
15. Wigley BJ, Bond WJ, Hoffman MT (2010) Thicket expansion in a South African savanna under divergent land use: Local vs. global drivers? *Glob Chang Biol* 16: 964-976.
16. Van Auken OW (2009) Causes and consequences of woody plant encroachment into western North American grasslands. *J Environ Manage* 90: 2931-2942.
17. Higgins SI, Bond WJ, February EC, Bronn A, Euston-Brown DIW (2007). Effects of four decades of fire manipulation on woody vegetation structure in savanna. *J Ecol* 88:1119-1125.
18. Köchy M, Wilson SD (2001) Nitrogen Deposition and Forest Expansion in the Northern Great Plains. *J Ecol* 89: 807-817.
19. Bond WJ, Midgley GF (2012) Carbon dioxide and the uneasy interactions of trees and savannah grasses. *Philos Trans R Soc Lond B Biol Sci* 367:601-612.
20. Angassa A, Oba G (2008) Herder perceptions on impacts of range enclosures, crop farming, fire ban and bush encroachment on the rangelands of Borana, Southern Ethiopia. *J Hum Eco* 136: 201-215.
21. Oba G, Post E, Syvertsen PO, Stenseth NC (2000) Bush cover and range condition assessments in relation to landscape and grazing in southern Ethiopia. *Landsc Ecol* 15:535-546.
22. Knapp A K, Briggs JM, Collins SL, Archer SR, Bret-Harte MS (2008) Shrub encroachment in North American grasslands: Shifts in growth form dominance rapidly alters control of ecosystem carbon inputs. *Glob Chang Biol* 14: 615-623.
23. Daryanto S, Eldridge DJ, Throop HL (2013) Managing semi-arid woodlands for carbon storage: Grazing and shrub effects on above- and belowground carbon. *Agric Ecosyst Environ* 169: 1-11.
24. Shackleton CM, Scholes RJ (2011) Aboveground woody community attributes biomass and carbon stocks along a rainfall gradient in the savannas of the central lowveld, South Africa *S Afr J Bot* 77: 184-192.
25. Jackson RB, Banner JL, Jobbägy EG, Pockman WT, Wall DH (2002) Ecosystem carbon loss with woody plant invasion of grasslands. *Nature* 418: 623-626.
26. Hasen-Yusuf M., Treydte, AC., Abule E., Sauerborn J. (2013) Predicting aboveground biomass of woody encroacher species in semi-arid rangelands, Ethiopia. *J Arid Environ* 96: 64-72.

27. Tolera A, Abebe A (2007) Livestock production in pastoral and agro-pastoral production systems of southern Ethiopia. *Livestock Research for Rural Development*, 19.
28. Coppock DL, (1994) The Borana Plateau of Southern Ethiopia: Synthesis of Pastoral Research, Development and Changes 1980–1990. System study No. 5. International Livestock Centre for Africa, Addis Ababa, 374pp.
29. Jahnke HE (1982) Livestock production systems and livestock development in tropical Africa. Kieler Wissenschaftsverlag Vauk, Kiel, Germany.
30. Oba G, Kotile DG (2001) Assessments of landscape level degradation in southern Ethiopia: Pastoralists versus ecologists. *Land Degrad Develop* 12: 461-475.
31. Homann S, Rischkowsky B, Steinbach J (2008) The effect of development interventions on the use of indigenous range management strategies in the Borana Lowlands in Ethiopia. *Land Degrad Develop* 19: 368-387.
32. Cossins NJ, Upton M (1988) Options for improvement of the Borana pastoral system. *Agric Syst* 27:251-278.
33. Kamara A (2001) Property rights, risk, and livestock development in southern Ethiopia, socio-economic studies on rural development edn. Wissenschaftsverlag Vauk, Kiel, Germany. Dissertation.
34. Oba G (1998) Assessment of indigenous range management knowledge of the Borana pastoralists of southern Ethiopia. Commissioned by GTZ-Borana Lowland Pastoral Development Program in collaboration with the Oromiya Regional Bureau for Agricultural Development, Negelle/Borana Ethiopia.
35. Napier A, Desta S (2012) Review of Pastoral Rangeland Enclosures in Ethiopia, <http://sites.tufts.edu/feinstein/2012/> date 20, 06, 2013.
36. Angassa A, Oba G (2010) Effects of grazing pressure, age of enclosures and seasonality on bush cover dynamics and vegetation composition in southern Ethiopia. *J Arid Environ* 74:111-120.
37. t'Mannetje L, Jones RM, (2000) Field and Laboratory Methods for Grassland and Animal Production Research. CABI.
38. Qiu L, Wei X, Zhang X, Cheng J, Gale W, et al. (2012) Soil organic carbon losses due to land use change in a semi-arid grassland. *Plant Soil* 355:299–309.
39. Rau BM, Johnson DW, Blank RR, Tausch RJ, Roundy BA(2011) Woodland expansion's influence on belowground carbon and nitrogen in the Great Basin U.S. *J Arid Environ* 75:827-835.
40. Blake GR, Hartge KH (1986) Bulk density. In: Klute, A. (Ed.), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*. ASA Monograph 9: 363–375.

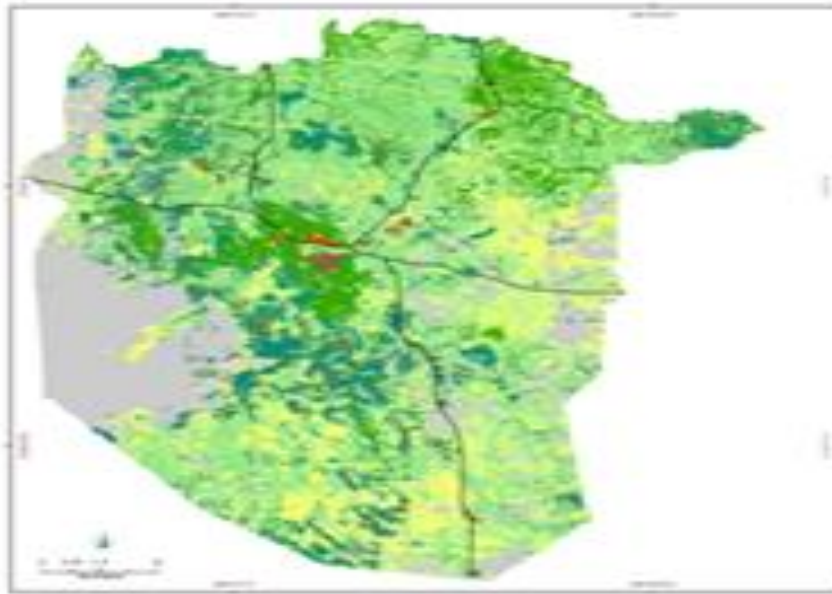
41. Schlesinger W (1977) Carbon Balance in Terrestrial Detritus. *Annu Rev Ecol Evol Syst* 8: 51-81.
42. Walkley A (1947) A critical examination of a rapid method for determining organic carbon in soils - effect of variations in digestion conditions and of inorganic soil constituents. *Soil Sci* 63:251-264.
43. Bremner JM, Mulvaney CS (1982) Nitrogen total. In: Page, AL, et al., (Eds.), *Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties*. ASA Monograph 9: 595–624.
44. Thomas GW (1982) Exchangeable cations. In: Page, A.L., et al., (Eds.), *Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties*. ASA Monograph 9: 159–165.
45. Gee GW, Bauder JW, (1982) Particle size analysis. In: Klute, A. (Ed.), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*. ASA Monograph 9: 383–411.
46. Throop H, Reichmann L, Sala O, Archer S (2012) Response of dominant grass and shrub species to water manipulation: an ecophysiological basis for shrub invasion in a Chihuahuan Desert Grassland. *Oecologia* 169:373-383.
47. Yayneshet T, Eik LO, Moe, SR (2009) The effects of exclosures in restoring degraded semi-arid vegetation in communal grazing lands in northern Ethiopia. *J Arid Environ* 73:542–549.
48. Bagchi S, Ritchie ME (2010) Introduced grazers can restrict potential soil carbon sequestration through impacts on plant community composition. *Ecol Lett* 13:959–968
49. Schuman GE, Reeder JD, Manley JT, Hart RH, Manley WA (1999) Impact of grazing management on the carbon and nitrogen balance of a mixed-grass rangeland. *Ecol Appl* 9:65–71.
50. Roques KG, O'Connor TG, Watkinson AR (2001) Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. *J Appl Ecol* 38: 268-280.
51. Gibbens RP, Lenz JM, (2001) Root systems of some Chihuahuan Desert plants. *J Arid Environ* 49:221–263.
52. Austin AT, Vivanco LA (2006) Plant litter decomposition in a semi-arid ecosystem controlled by photodegradation. *Nature* 442:555-558.
53. Piñeiro G, Paruelo JM, Jobbagy EG, Jackson RB, Oesterheld M (2009) Grazing effects on belowground C and N stocks along a network of cattle exclosures in temperate and subtropical grasslands of South America. *Global Biogeochem Cycles*, 23.
54. Mekuria W, (2013) Conversion of Communal Grazing Lands into Exclosures Restored Soil Properties in the Semi-Arid Lowlands of Northern Ethiopia. *Arid Land Research and Management* 27, 153-166

55. Su YZ, Li YL, Cui HY., Zhao WZ (2005) Influences of continuous grazing and livestock exclusion on soil properties in a degraded sandy grassland, Inner Mongolia, northern China. *Catena* 59:267–278
56. DellCJ, Williams MA, Rice CW (2005) Partitioning of nitrogen over five growing seasons in tallgrass prairie. *J Ecol* 86:1280 - 1287.
57. Stewart A, Frank D (2008) Short sampling intervals reveal very rapid root turnover in a temperate grassland. *Oecologia* 157:453-458.
58. Mwendera EJ, Saleem MAM (1997) Hydrologic response to cattle grazing in the Ethiopian highlands. *Agr Ecosyst Environ* 64:33–41.
59. Steffens M, Kolbl A, Totsche KU., Kogel-Knabner I (2008) Grazing effects on soil chemical and physical properties in a semi-arid steppe of Inner Mongolia (PR China). *Geoderma* 143: 63–72.
60. Vågen TG, Winowiecki LA (2013) Mapping of soil organic carbon stocks for spatially explicit assessments of climate change mitigation potential. *Environ Res Lett* 8:015011.
61. Alam SA, Starr M, ClarkBJF (2013) Tree biomass and soil organic carbon densities across the Sudanese woodland savannah: A regional carbon sequestration study. *J Arid Environ* 39:67-76.
62. Mesele, S., Gebrekidan, H., Gizachew, L., Coppock, D.L., 2006. Changes in land cover and soil conditions for the Yabelo District of the Borana Plateau, 1973-2003. Research Brief 06-06-PARIMA. Global Livestock Collaborative Research Support Program. University of California, Davis. 4 pp.
63. Hudak AT, Wessman CA, Seastedt TR (2003) Woody overstorey effects on soil carbon and nitrogen pools in South African savanna. *Austral Ecol* 28:173-181.
64. Schlesinger WH, Reynolds JF, Cunningham GL, Huenneke LF, Jarrell WM, (1990). Biological feedbacks in global desertification. *Science* 247:1043–1048.
65. Jobbagy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol Appl.* 10: 423-436.
66. Bronick CJ, Lal R (2005) Soil structure and management: a review. *Geoderma* 124:3–22.
67. Guo, L. B. & Gifford, R.M. Soil carbon stocks and land use change: a meta analysis. *Glob Change Biol* 8, 345–360 (2002).
68. Archer, S., Boutton, T.W. & Hibbard, K. A. in *Global Biogeochemical Cycles in the Climate System* (eds Schulze, E.-D. et al.) 115–137 (Academic, San Diego, 2001).
69. Burke, I. C. et al. Texture, climate, and cultivation effects on soil organic matter content in U.S. grassland soils. *Soil Sci Soc Am J* 53, 800–805 (1989).

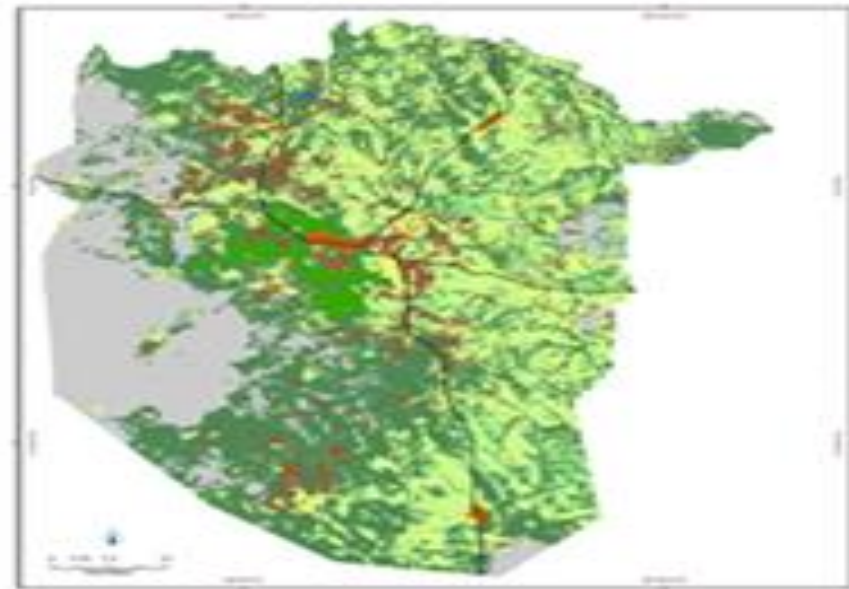
5 General Discussion

5.1 Trends of expansion of woody encroachment

Our multi-temporal satellite imagery analysis and quantitative vegetation studies demonstrated extensive expansion of woody encroachment in the Borana rangelands over the last four decades following fire suppression. Low woody encroached areas (shrub savanna and tree savanna) declined from 45% to 9%, while heavily woody encroached areas (bushland thickets and bushed savanna) increased from 22% to 61% in the years from 1976 to 2012 (chapter 3, fig. 3.1 and Table 3.3). This result suggests a continuous trend of new woody encroachment into the existing tree/shrub savanna and a thickening of the existing woodlands/shrublands (Table 3.3). A similar trend was shown by Mesele et al. (2006) for the Yabello District of Borana for the period between 1973 and 2003; here, the total grassland cover decreased by 86% while bushland and bushed grassland increased by 46% and 48% respectively. Our research area represented an example of the trend of vegetation cover changes in many savanna ecosystems, mainly due to suppression of fire from the system (see Angassa and Oba, 2008; Roques et al., 2001). Though fire was suppressed by the government policy around the same time period across the entire study area, our study indicates that the woody species structure across the landscape are in different stages and varied strongly (see chapter 3), which implies a nonlinear pattern of woody encroachment across the landscape. Variations in topo-edaphic factors (see Venter and Govender, 2012), and impacts through herbivores, and human utilization of woody plants across the landscape could have contributed to the nonlinear pattern of these changes (see also Roques et al., 2001; Augustine and McNaughton, 2004; Wessels et al., 2011). The rate and pattern of vegetation dynamics in the dry savanna ecosystem could also be determined by several other ecological processes, including episodic periods of drought or favorable precipitation, altered fire regimes, and severe soil erosion (see Westoby et al., 1989).



(a) 5 January 1976 MSS image



23 January 2012 ETM+ image

Vegetation cover class

- Settlement
- Cultivated land
- Shrub savanna
- Bare land
- Tree savanna
- Bushed savanna
- Bushland thickets
- Dense shrubland
- Forest

Fig. 5.1. Land cover in Yabello district, Borana rangelands, in 1976 and 2012

Floristically, the four woody encroachment levels studied are typical of the broader vegetation types of the Borana semi-arid savannas and woodlands, each with plant community compositions differing largely on the basis of topo-edaphic gradients (Coppock 1994, chapter 3). However, several changes were evident along the gradient of increasing levels of woody encroachment. Density, height of the canopy and canopy area cover of large woody plants (> 6 m height class) increased with the increase in woody encroachment level/stage (chapter 3). However, total woody plant density and the density of regenerative woody plants (< 1 m height class) increased in a curvilinear fashion along the gradients of woody encroachment levels. Higher total woody plant density and a greater proportion of woody plants in small size classes (<1 m height) were found in the moderate and highly woody encroached sites (chapter 3, Table 3.6 and Fig 3.4). The greater proportion of regenerative woody plant (< 1 m) density shown in recent and low encroached sites suggests an ongoing active expansion of woody encroachment into relatively open grasslands, while a greater proportion of old and larger woody vegetation without many seedlings in the highest level of woody encroachment site suggests that a woody vegetation climax state might have been reached. Given the complete suppression of fire in the Borana rangelands, these woody encroachment trends are likely to continue, thus the encroaching woody species, whether occurring in isolated patches or as extensive shrublands/woodlands, can cause an irreversible state transition from historical savanna grasses dominance to an alternative, but stable, woody vegetation state (chapter 3 see also Briske et al., 2005). On the contrary, though further information on woody species soil seed bank still required, low recruitment as shown by the low regenerative woody plants (< 1 m height woody plants) at severely encroached site could gradually lead to an open woodland as larger trees die off over time due to “self thinning” (chapter 3, see also Belay and Moe, 2012; Wiegand et al., 2006).



a) Grassy state (cleared of shrubs)

b) Woody state

Fig. 5. 2. A shift in a vegetation state from open savanna state (a) to woody state (b) (Picture taken from the study sites HasenYusuf, 2011)

5.2 Allometric functions

Biomass estimates of single species were successfully derived and the performance of the allometric models used depended on the species and biomass components to be estimated (see chapter 2). The allometric models were highly accurate for larger tree species such as *A. mellifera*, *A. bussei*, and *A. etabaica* (Table 2.2). The allometric models we present for predicting total AGB in stem and branches of different species here rely on the combination of different predictor variable/s (see chapter 2). The models indicated that the most important single or set of predictor variables were represented by be stem basal circumference for tall shrubs with a more or less simple canopy structure (see chapter 2). In contrast, pairs of canopy volume and stem basal circumferences were more reliable for predictions of tall shrubs with closed and umbrella- like canopy structures (see chapter 2). We also showed that, even if canopy volume only is used as a predictor variable, biomass can still be well predicted for shrubs whose growth form comprises discrete canopy clumps with multiple stems (e.g., *A. oerfota*, chapter 2). This concurs with observations by (Hofstad, 2005) that for shrubby woody plant species, a combination of stem and canopy related variables could improve the accuracy of AGB predictions substantially compared to diameter and height measures alone that have traditionally been used in the wet and humid forest ecosystems (Henry et al., 2011, see chapter 1). The indicators of the goodness of fit of the species-specific models such as r^2 , p , MSE , and cv have shown the strong relationship between total tree/shrub or components dry biomass and selected dendrometric variable or variables. The accuracy of the woody AGB estimate is vastly enhanced when a unique set of equations locally derived for individual species is used (see also Henry et al., 2011). Similarly, Hofstad (2005) and Tietema (1993)

recommended that species-specific allometric equations be used for the dominant species, and a general summary equation for the less abundant species. The low comparability of our allometric equation results with others in different ecoregions within African savanna for the same species (chapter 2, see also Henry et al., 2011) questions the general applicability of a single equation across different ecoregions. However, rapid assessment of tree/shrub aboveground biomass could be possible if a single equation can be applicable for a group of species or functional types of closely related shapes, but can lead to significant compromises of the biomass prediction accuracy as shown by lower models goodness of fit indicators. For example, two separate equations for shrubs and larger trees can be used for general estimates of standing AGB for a given ecoregion. Generally, the selection of appropriate dendrometric predictor variables (CV vs, stem or height), and species-specific or generalized (shape dependent) tree/shrub allometric functions depend on the purpose of the study and the accuracy level required.

5.3 Woody encroachment and aboveground biomass C stock

Herbaceous biomass C stocks are maximized at the lowest levels of woody encroachment with the study sites in this study holding 1.3, 0.5, 0.7, and 0.2 Mg C ha⁻¹ herbaceous AGC stocks at LE, ME, HE, and SE sites respectively. The decrease in herbaceous biomass with increases in the levels of woody encroachments shows non-linear pattern. This non-linear relationship suggests that species-specific traits (e.g. canopy shape, plant height, and rooting pattern) in the plant community across the woody encroachment levels might have influenced the outcome of tree density and/or tree canopy cover and herbaceous biomass C stock relationships. The lower herbaceous biomass C stock at ME compared to HE levels might be linked to variation in woody species composition and associated species traits between the sites (see chapter 3, Table 3.4). The average herbaceous AGC stocks across sites (0.7 Mg C stock ha⁻¹ \approx 1.5 Mg dry mass ha⁻¹) was within the range of 1.5 - 2.7 Mg dry mass ha⁻¹ reported by Cossins and Upton (1987), but lower than 5 Mg dry matter ha⁻¹ reported by Angassa and Oba (2010) from Borana rangeland system. However, our herbaceous biomass data provide only a snapshot of two seasons during a rather dry year and, hence, comparisons with other studies have to be taken with care. In addition, the high and severely woody encroached sites which significantly lowered our mean herbaceous biomass estimations of the rangelands were not included by Angassa and Oba (2010). Adding woody plants AGC stocks at each site rapidly increases total aboveground biomass C stocks of the ecosystem and shifts the balance of biomass C stocks contribution from herbaceous to woody plants. The estimated AGC stocks in the woody components of the

ecosystem at this study site increased by more than 300% from low woody encroachment levels to the highest encroachment levels (see chapter 3). This emphasizes that woody plants expansion into the savanna grasslands can dramatically increase the amount of C stored on a site. The significance of additional C stocks in woody species is further emphasized by the low understory (herbaceous species) C stock values contributing only 12% to the total AGC stock of this ecosystem. Similar trends of woody AGC stock increases were reported when open savanna or grasslands were encroached by woody plants in other ecosystems (see also Lett et al., 2004; Asner et al., 2003, chapter 3).

The AGC stock of the woody vegetation stand of this study area has shown general increasing patterns with an increase in the encroachment levels, but was not linearly related to total woody plant density and/or canopy cover. For example, at the SE site, AGC stock was more than double that of the ME site, though the tree/shrub density and cover of SE was lower or comparable to ME level indicating clear influence of species composition and traits, and size structure of the vegetation community (see chapter 3). The latter is often governed by the genetic make up of each species in the community and environmental factors. This finding is in agreement with Litton et al. (2006) that the total biomass of the vegetation stand was largely determined by larger trees (see chapter 2). This may suggest that keeping a low density of larger trees species of high C storing potential like *Acacia tortilis*, *Acacia senegal*, *Acacia etabaica*, *Acacia mellifera*, *Acacia bussei* and *Acacia reficiens* (see chapter 3) in the landscape can increase the amount of biomass and C stored in the trees and herbaceous vegetation; thereby, also livestock production will not greatly be compromised. However, the question how much woody plants density/cover is enough is difficult to predict as the influence of woody plants on understory herbaceous production and total ecosystem C stock including woody C pool is not linear. It is rather largely determined by functional traits such as plant height, canopy height, canopy shape, rooting pattern, allelopathy, and wood density of each woody plant species in the community. Management of woody species with a facilitative role on understory herbaceous production and overall ecosystem C storage in this ecosystem requires further work on evaluation of morphological, phenological, and ecophysiological traits of encroaching woody species in the plant community. Further, more information on their impacts on ecosystem structural (e.g., herbaceous diversity) and functional processes (e.g., biomass production, N, C, and water cycling) needs to be gathered additionally.

The plot level woody cover estimates in this study suggested that rangelands sites reached cover values of > 55% (see Chapter 3) which is slightly above the 50% woody cover

in 2004 reported by Dalle, et al. (2006) for Borana rangelands. The landscape scale analysis using satellite imagery has shown that heavily encroached areas (bushland thickets and bushed savanna) increased from 22% to 61% in the years from 1976 to 2012 (see chapter 3). If the woody encroachment expansion in the Borana rangelands continues at their present rate the remaining open savanna would likely be totally covered with woody vegetation within a few decades. The woody canopy cover ranged from 30 to 70 % in the study sites, and the corresponding C stocks ranged from 2 - 9 Mg ha⁻¹ respectively (see chapter 3). There are relatively few data from eastern Africa savanna vegetation types against which we can compare our AGC stocks. However, our lowest mean estimate of 2.3 Mg AGC stock ha⁻¹ at low woody encroachment with 30% woody cover is higher than the average value of 1 Mg AGC stock ha⁻¹ reported in a Sudanese *Acacia* woodland savanna region (Allan et al., 2013). The authors have linked this low value to rapidly declining woody cover in their region. Though larger scale studies will be required for the Borana rangeland systems, our results thus far indicate that the high woody cover increases observed was probably due to a fire suppression that has resulted in very high woody AGC stock values compared to other East African savannas. Therefore, the Borana savanna case may highlight the importance of managing woody plants; by avoiding fire and tree/shrub clearing a significant increase in East African savanna total ecosystem C stocks, mainly through the amount of woody biomass, is expected.

5.4 Woody encroachment, soil organic carbon and total soil nitrogen

In contrast to the AGC stocks, the value of SOC stocks was lower in the heavily encroached sites compared to the low encroachment site, but the difference among woody encroachment levels was not statistically significant on similar soil texture (see chapter 4). The result also showed that high SOC and TSN stocks were found at high herbaceous cover (yield) and vice versa. However, the relationship between herbaceous biomass yield and SOC and TSN seemed additionally influenced by soil characteristics. Several studies also showed that though biomass accumulation in the soil through plant litter or root is the first step to increase SOC and TSN stocks in the soil the long-term SOC sequestration can only be achieved when the organic molecules from the biomass form tight associations with mineral soil particles (Jobbagy and Jackson, 2000). Clay particles have larger space and greater affinity to form such tight associations as compared to sand particles (Jobbagy and Jackson, 2000). Therefore, the higher SOC and TSN obtained at low encroachment on clay loam soil could be a function of both high accumulation of herbaceous biomass and greater clay particles. A trend of reduced soil organic matter with a shift of grassland/savanna vegetation to woodland

vegetation was reported from the same region by Mesele et al. (2006) and in agreement with Guo and Gifford (2002) who showed SOC losses when grassland was converted to plantations in New Zealand rangeland systems. Jackson et al. (2002) also reported a decrease in SOC in semiarid grasslands experiencing woody encroachment and associated the reduction in SOC to the incorporation of soil N to aboveground woody plant parts. Hudak et al. (2003) linked the reduction of SOC and TSN at severely woody encroached site in South African dry savanna to a reduction of herbaceous root production caused by woody encroachment. Schlesinger et al. (1990) similarly showed that the bare inter-space between shrubs experiences higher temperatures and evapo-transpiration, leading to a slow organic N incorporation, ammonia volatilization and increased soil erosion.

However, several studies have shown that many other biotic and abiotic factors can determine SOC stocks (Jobbagy and Jackson, 2000). Soil properties can influence SOC concentrations and the occurrence of woody encroachment itself (Sankaran et al., 2005, Archer et al., 2001). For example, Archer et al. (2001) indicated that soil texture strongly influences where *Prosopis* can establish in a southern Texas savanna rangeland. Similarly Vågen and Winowiecki (2013) has shown inherent high soil sand fraction strongly limits SOC stocks in East African savanna and woodlands, independently of climatic factors and vegetation type differences (see chapter 4). The historical land use pattern and disturbance, including soil erosion, condition and productivity of the sites before the occurrence of woody encroachment may also influence the variation in SOC and TSN stocks across the sites (Burke et al., 1989). Lack of sufficient replications in our experimental design limits us to isolate the role of woody encroachment for SOC and TSN storage decline at heavily encroached sites in the presence of uncontrolled many potential factors (e.g., soil texture) that could determine the SOC stocks in these semiarid rangelands

The estimated mean SOC stocks of about 44 Mg ha⁻¹ in the current study (see chapter 4) falls within the 20 - 80 Mg ha⁻¹ range reported by earlier studies for east African woodland and savanna ecosystems (see Alam et al., 2013; Vågen and Winowiecki, 2013). The understanding of belowground C dynamics due to woody encroachment is also important (Jobbagy and Jackson, 2000; Conant and Paustian, 2002) as soils can store about three times more C than the vegetation AGC stocks in grassland/savanna ecosystems (Lal, 2004). On the basis of the data on AGC and SOC analysis presented here, SOC stocks tended to decrease while AGC stocks increases as result of the expansion of woody encroachment into semiarid savanna ecosystem. However, as savanna soils and vegetation compositions and structure are highly variable and the SOC storage can be determined by these factors further field studies

will be needed to evaluate the large scale net effects of woody plants encroachment and site characteristics on SOC storage in the Borana rangeland ecosystem.

5.5 Grazing, vegetation structure and aboveground biomass carbon

At all levels of woody encroachment sites the presence of grazers and browsers in the open grazing land led to a significant reduction of total woody plant density, particularly of plants < 1m in height (see chapter 3). The increased density of total woody plants in enclosures was shown in the previous studies (see Wessels et al., 2011; Angassa and Oba, 2010). As reported in Chapter 3 protection from browsing and grazing led to increased woody plant recruitment in the enclosures, i.e., the areas protected from herbivory in agreement with the finding of Angassa and Oba (2010) and Wessels et al. (2011). Medium-sized browsers such as goats can severely limit woody species recruitment through seedling predation (see Moe et al., 2009; Allred et al., 2012). In this thesis it was also shown that herbivory is important to limit the expansion of woody encroachment in the semiarid savanna rangelands.

The impact of long-term livestock exclusion during the plant growing season on C stocks was only clearly visible for herbaceous AGC stocks. Enclosure at low encroachment sites increased herbaceous AGC stocks by 0.7 Mg ha^{-1} . Although grazing is less likely at severe encroachment site with a woody cover exceeding 60 % and herbaceous cover is less than 25% (see chapter 4, Table 4.2). Enclosure at severe encroachment site increased only 0.2 Mg ha^{-1} herbaceous AGC stocks, nearly five times lower than that gained by enclosure at low encroached site. In general, an increase of herbaceous AGC stock by enclosure at our severely encroached site was insignificant and suggested the contribution of herbaceous biomass C to the whole rangeland ecosystem C stock was overridden by the gains in woody plants biomass C. Further, the response of herbaceous AGC stocks to grazing was also strongly influenced by the woody encroachment characteristics including woody density, canopy cover, species composition and their specific traits and other site characteristics (see chapter 3). These results are consistent with other studies that showed a significant decrease of the herbaceous biomass yield with persistent heavy grazing in the communal rangelands (Angassa and Oba, 2010; Yayneshet et al., 2009), and with an increase in woody encroachment (Oba et al, 2000).

5.6 Grazing, herbaceous root biomass carbon, soil organic carbon and total soil nitrogen

In most of woody encroachment levels, our data showed that long-term enclosure resulted in an increase in herbaceous root C stocks (see chapter 4). This pattern of lower herbaceous root C stocks under grazing agrees with observations in other semi-arid grasslands (400 to 850 mm

of mean annual precipitation) (Reeder et al., 2004, Qiu et al., 2012). However, in our savanna ecosystem, unlike in other semiarid grassland systems, the increase in herbaceous root due to enclosure might have been limited by woody encroachment (see Hudak, et al., 2003; Scholes and Archer, 1997).

Long-term enclosure at the low encroachment site increased SOC and TSN stocks by 50% and 25% due to the increased C concentration mainly in the top 20 cm soil depth (see chapter 4). As the woody encroachment levels increased to moderate- high -and severe encroachment levels, enclosure resulted in only 7%, 3%, 9% increase in SOC stocks respectively. Similar trends were observed for TSN stocks. This suggests that the increase in SOC and TSN by enclosure were maximized in the low encroachment site, and decreased with the increasing woody encroachment levels (see chapter 4). The SOC increase in the enclosures might be linked to an increased herbaceous above- and belowground production and nutrient deposition (Austin and Vivanco, 2006). Higher grazing intensity is generally thought to decrease soil C and N by direct removal of aboveground herbaceous biomass, i.e., a reduction of potential CO₂ fixation in photosynthetic tissue and in belowground C inputs through lower root production and higher root litter turnover (Reeder et al., 2004). The increase of the cumulative root biomass by exclusion of herbivory not only increases soil C inputs but also N storage within the soil (Piñeiro et al., 2009; Stewart and Frank, 2008). As both organic N and C dynamics are closely linked in the soil organic matter (SOM) (Wang et al., 2006) the storage of N in root biomass and cycling within the soil has been suggested as a mechanism that can reduce N loss (Stewart and Frank, 2008). The N allocated to herbaceous AGB is not fully recycled into the soil because of grazing, surface respiration by decomposers and photo-oxidation (Austin and Vivanco, 2006). Livestock exclusion can reduce the loss of N by volatilisation of ammonia and nitrate through animal urine and dung patches and, thereby, the resulting N availability will enhance SOM formation and SOC storage (Piñeiro et al., 2009). The decrease of herbaceous plant cover in open grazing land may also increase SOM mineralization rates because of high soil temperature fluctuations and soil moisture variability and, hence, a decrease in SOC storage (Wang et al., 2006). The increase of SOC and TSN associated with enclosure at the low encroachment site was exceptionally strong and could be related to deeper and clay dominated soils compared to other sites. Piñeiro et al., (2009) suggested SOC and TSN storage increased under enclosure, determined by the ability of the soil to retain the extra N after exclusion of livestock. Jobbagy and Jackson, (2000) also indicated that SOC storage capacity is largely linked to soil texture and soil depth since the adsorption of organic matter to clay and silt particles determines SOM stabilization. Our

dataset also showed an overall negative correlation between SOC stocks and soil sand fraction. This makes measuring changes in SOC stocks in soils at larger scale very difficult in spatially highly variable savanna ecosystem, as more data will be needed from multiple sites and grazing regimes to make definite conclusion about the response of SOC stocks to grazing management.

Only few studies have attempted to explain the mechanism behind SOC reduction due to grazing pressure in African savanna systems. The available studies highlighted increased soil erosion associated with soil compaction caused by trampling through livestock as the main causes for the loss of top soil OM in many parts of African and Central Asian grasslands (Mekuria, 2013; Savadogo et al., 2007; Su et al., 2005; Mwendera and Saleem et al., 1997). Soil compaction potentially can reduce water infiltration and increase runoff which often results in decreasing water availability for plant growth. In addition, this can lead to a loss of top soil and nutrients especially under intense grazing conditions (Savadogo et al., 2007; Mwendera and Saleem et al., 1997).

5.7 Ecosystem level carbon stocks

The expansion of woody encroachment into the rangelands increased the AGC pool through woody vegetation biomass. Our result showed that on average > 95% of estimated rangeland ecosystem C stocks at the low encroachment site measured to a depth of 40 cm was stored in soils, but at heavy woody encroachment level aboveground woody biomass C stocks accounted to 30% of ecosystem C stocks. These AGC stocks in woody biomass which constituted 30% of total ecosystem C (even more if root biomass carbon was included) at highly encroached sandy soil sites can be considered as addition to the grassland ecosystem C storage potential. However, this can only be considered as a gain to the system if the lower SOC found at these high woody encroachment sites were caused by the inherent low SOC storage potential of sandy soils or historical disturbances (overgrazing and associated soil erosion). Hence, woody encroachment on sandy soils will give an overall benefit from a global warming perspective.

However, the SOC stocks at the heavy encroachment sites were by 50% lower when compared to the low encroachment site, but it is unclear at this point whether this low value in soil C and N stocks at the heavily encroached site is associated to woody encroachment or inherent sandy dominated soils of the sites. It is also unclear if original savanna grasslands can be restored, particularly with respect to soils, aiming at an increase in SOC stocks. This is currently beyond the scope of our study but an important consideration for future studies.

Long term enclosure at low woody encroachment (< 30 % woody cover) effectively increased above- and belowground herbaceous biomass C, SOC and TSN (see chapter 4), and may have positive impacts on overall ecosystem C pools due to herbaceous above- and belowground biomass increases being incorporated into the soil. Enclosures at higher encroachment sites (> 30% woody cover) showed no significantly increased AGC stocks of woody components as the large trees or shrubs which constituted greater proportion of ecosystem C pools were harvested by humans to foster herbaceous production in the enclosures. Increases of ecosystem C stocks through enclosures were also not consistent along woody encroachment levels and seemed dependent of the interactions between herbaceous plants and woody species and soil properties. Eventually, the total net gain of rangeland ecosystem C stocks by woody encroachment and grazing will be determined by the stage of woody encroachment, functional traits of woody species and soil characteristics.

5.8 Limitations and conclusion of the study

Expansion of woody encroachment and intensification of grazing present difficulties on how best to manage the semiarid Borana rangelands to encompass the tradeoff of livestock production and C sequestration. The empirically derived biomass models and C stocks we presented here constitute a first attempt to estimate C pools in encroaching woody vegetations of southern Ethiopian savanna rangelands. The biomass models are applicable in similar ecoregions of east African savannas and woodlands. This study offers insights into our knowledge gap on soil C and herbaceous root biomass dynamics in relation to woody encroachment and intensification of grazing. The C stocks estimated across woody encroachment levels only represent a small fraction of the landscape within the vast and highly variable Borana savanna rangelands. However, the estimates provide baseline data and framework for future research works. Further work needs to be carried out to determine other biophysical factors that may determine the influence of grazing and woody encroachment on ecosystem C dynamics. Hence, care must be taken on extrapolating our results over a wider range of Borana rangelands. The result represents plot level estimates, and may not be applicable for landscape or regional scale analysis. Therefore, upscaling the plot level estimate to landscape or regional scale estimate using fine resolution satellite image is important in future works. The total C storage potential of this rangeland ecosystem to 40 cm soil depth (not including aboveground litter biomass C stock, woody plant root biomass C stock, and SOC in the below 40 cm soil depth) ranged between 40 Mg C ha⁻¹ to 83 Mg C ha⁻¹. It was shown that 95% of the estimated ecosystem C stocks on our plots were stored in soils up to 40 cm depth at lowest encroachment level, but at heavy woody encroachment levels

AGC stocks accounted for over 30% of our estimated ecosystem C stocks. Though woody encroachment tends to increase woody biomass C stock in the grassland system, the stability of the woody biomass C stock gained via woody biomass in the pastoral grazing system is highly dependent on its intended use (pastoralists land use preference). Because of a preference for grasses, woody encroachment is likely to result in a degradation of the land use value with respect to cattle and sheep grazing, but in a positive effect on land use value for goat and camel browsing, given the preference of goats for woody browse. In the Borana rangeland system many pastoralists have considered few alternative land uses for highly woody encroached sites apart from limited goat and camel browsing (Angassa and Oba 2008). Carbon sequestration services could be potentially an alternative land use option for highly encroached sites, however, many uncertainties (e.g. lack of C credit mechanisms for savannas or/and shrublands, low C price of $< \$10 \text{ tone}^{-1}\text{C}$, quantification belowground C dynamics) are likely to influence practical applicability and the profitability C production in the rangelands. Therefore, a careful introduction of higher domestic browser numbers (i.e., camel and goat feeding on woody vegetation) will enable the pastoral community to benefit from and adapt to the vegetation changes at least in the short term. Management of woody plants, through e.g., browsers or selective thinning may be just as critical as grazing management for sustainable livestock production. Co-management of the rangelands for broader environmental services (biodiversity conservation, C sequestration and reduction of emissions from livestock systems) through an environmental payment service scheme (PES) could also help the pastoralists to diversify and sustain their livelihood in the future.

6 Summary

The Borana rangelands in southern Ethiopia are facing deterioration caused by intensification of grazing and woody plant encroachment, resulting in marked reductions in pastoral production. This process affects the food security and livelihoods of the Borana pastoral people negatively. Woody plant encroachment might result in an increase in carbon (C) storage in these rangelands, which represents an important aspect for climate change mitigation potentials. However, it is unclear how much C is currently stored in the above- and belowground vegetation biomass and in the soils of these rangeland ecosystems and how grazing intensity and woody cover influence soil or ecosystem C-stocks.

The research work presented in this thesis aimed at developing tools to estimate the aboveground woody biomass C stocks. It describes the structure of semiarid savanna vegetation in different grazing regimes at various levels of woody encroachment, examines changes in woody plant encroachment, and provides field-based quantification methods and tools to derive site-based estimates of above- and belowground C pools. The thesis also aimed at assessing the influence of grazing on herbaceous above- and belowground biomass C stocks, soil organic carbon (SOC) and total soil nitrogen (TSN) to estimate possible increases in ecosystem C stocks by long term reduction of grazing intensity (e.g., low livestock density and seasonal grazing) at various levels of woody plant encroachment.

A long-term temporal satellite imagery over the last 37 years and GIS mapping aided by ground truthing was used to investigate vegetation cover changes. In the field, data was collected to analyze vegetation attributes such as composition and structure under different grazing regimes and woody encroachment sites. Herbaceous species were destructively harvested to quantify the biomass and C stocks in the herbaceous vegetation community. Allometric tree biomass models were developed by destructively harvesting eight woody species to indirectly quantify the woody biomass and C stocks. Total soil nitrogen and SOC stocks in the different grazing management systems and woody encroachment levels were assessed from soil cores collected within 0- 40 cm soil depth.

The performance of allometric biomass models as expressed as a goodness of fit ($adj\ r^2$) depended on the species and biomass components estimated. The allometric models were highly accurate for large woody species such as *A. mellifera*, *A. bussei*, and *A. etabaica*. The most important single models predictor variable identified was stem basal circumference for tall shrubs with more or less open canopy structure. Meanwhile, for tall shrubs with closed and umbrella-like canopy structures, pairs of canopy volume and stem basal circumferences

were more reliable predictors. It was further shown that, by using canopy volume as a stand-alone predictor variable, biomass can still be accurately predicted for shrubs whose growth form comprise discrete canopy clumps with multiple stems (e.g., *A. oerfota*).

Vegetation cover analysis using temporal Landsat imageries from 1976 to 2012 revealed that areas covered by shrub and tree savanna (open savanna types) in the 1970s declined from 45% to 9%, while heavily encroached areas (bushland thickets and bushed savanna) increased from 22% to 61% during this time interval. The abundance of total and the regenerative woody plants (< 1 m height) were high in lower woody encroachment sites but significantly reduced at heavily woody encroachment sites. At all levels of woody encroachment enclosures significantly increased total woody plant density, especially the proportion of woody plants in < 1 m height size class compared to the open rangelands.

Estimated total aboveground biomass C stocks varied significantly between woody encroachments levels, with total aboveground biomass C stocks ranging from 2 Mg ha⁻¹ in the low encroachment site to 9 Mg ha⁻¹ in heavy encroachment sites. Enclosures significantly raised the herbaceous biomass C stocks, with enclosures containing 50% more herbaceous aboveground biomass C stocks than openly grazed land. However, the response of herbaceous aboveground biomass C stocks to grazing was also strongly influenced by the woody encroachment characteristics including woody density, canopy cover, species composition and other specific traits of woody species.

Mean total SOC stock in the 0 - 40 cm soil depth ranged from 30 Mg ha⁻¹ in the openly grazed soils at the high woody encroachment site to 81 Mg ha⁻¹ in the enclosure soils at the low encroachment site ha⁻¹. Soil OC and TSN did not differ in the enclosure at heavily encroached sites but were two times as high in enclosures compared to openly grazed soils at low encroached sites. Soil OC was positively related to TSN and soil cation exchange capacity (CEC), but negatively to sand content. Contrary to expectations, SOC stocks did not uniformly follow the pattern of increasing aboveground biomass C stocks with increasing woody encroachment. Rather, it seemed to be influenced by variations in soil characteristics across the Borana rangelands.

The study highlights the influence of woody encroachment and reduction of grazing pressure on ecosystem C stocks. The allometric models developed by this study can serve as a tool for future biomass and C sequestration studies in semiarid regions of east Africa. The information presented on the ecosystem C stocks by this thesis could help integrate the effects of grazing

and vegetation cover dynamics on the rangeland C storage. An understanding of these interactions are deemed necessary to develop a sound rangeland management policy that can link the C storage potential of the rangelands to global climate change mitigation and adaptation strategies through establishing a viable mechanism of payment for ecosystem services.

Einfluss von Änderungen in der Landnutzung und -bedeckung auf Weideland in Südäthiopien – Wie viel Waldbedeckung ist genug?

7 Zusammenfassung

Das Borana Weideland in Südäthiopien steht vor einer Verschlechterung, die durch intensivierete Beweidung und die Ausbreitung von Gehölzpflanzen verursacht wird. Dies führt zu einer deutlichen Abnahme der pastoralen Produktion und hat auch einen negativen Einfluss auf die Nahrungssicherheit und den Lebensunterhalt der Borana Bevölkerung. Auf der anderen Seite resultiert die Ausbreitung von Gehölzpflanzen möglicherweise in einer erhöhten Kohlenstoffspeicherung, ein wichtiger Aspekt bezüglich des Potentials zur Abmilderung des Klimawandels. Jedoch ist unklar wie viel Kohlenstoff (C) tatsächlich in der ober- und unterirdischen Biomasse und im Boden dieses Weidelandes gespeichert ist und wie Beweidungsintensität und die zunehmende Bedeckung von Gehölzpflanzen die Kohlenstoffvorräte des Ökosystems beeinflussen.

Die vorliegende Arbeit hat zum Ziel, Methoden zur Schätzung der oberirdischen Gehölzbiomasse bzw. der C-Vorräte zu entwickeln. Sie beschreibt die Struktur der semi-ariden Savannenvegetation unter verschiedenen Beweidungsformen, unter zunehmendem Deckungsgrad an Gehölzpflanzen und präsentiert feldbasierte Methoden zur Quantifizierung und standort-basierten Schätzung der ober- und unterirdischen C-Vorräte. Außerdem bezweckt diese Arbeit, den Einfluss von Beweidung auf die ober- und unterirdischen Biomasse bzw. C-Vorräte des Unterwuchses (Krautschicht) zu bewerten sowie den Effekt auf Bodenkohlenstoff (Soil organic carbon – SOC) und Gesamtstickstoff (total soil nitrogen – TSN) abzuschätzen. So kann eine mögliche Zunahme des Ökosystemkohlenstoffvorrats durch langfristige Reduktion der Beweidung (z.B. niedrigere Viehdichte und saisonale Beweidung) unter variierender Gehölzbedeckung eingeschätzt werden.

Mittels Satellitenbildern der letzten 37 Jahre, eines Geoinformationssystems (GIS) und Kartierung wurden langfristige und groß-skalige Änderungen der Vegetationsbedeckung untersucht. Im Feld wurden Daten gesammelt, um Vegetationseigenschaften wie Artzusammensetzung und -struktur in Abhängigkeit von Beweidung und Bedeckung zu analysieren. Krautige Arten wurden destruktiv geerntet, um die Biomasse und C-Vorräte zu quantifizieren. Durch destruktives Ernten von acht Baumarten wurden allometrische Modelle entwickelt, um indirekt die hölzerne Biomasse und C-Vorräte zu quantifizieren.

Gesamtstickstoff des Bodens und organischer Kohlenstoff (SOC) wurden mittels Bohrkernen von 0-40 cm Tiefe untersucht.

Die Güte der allometrischen Biomassemodelle ($adj\ r^2$) hing von der geschätzten Baumart und Biomassekomponente ab. Bei Baumarten mit hoher Gesamtbiomasse, z.B. *A. mellifera*, *A. bussei* und *A. etabaica*, trafen die allometrischen Modelle sehr akkurat zu. Die wichtigste Prädiktorvariable der Modelle bei hohen Stüchern mit mehr oder weniger offener Krone war der Basalumfang, während Kronenvolumen und Basalumfang des Stammes die zuverlässigsten Prädiktoren bei Arten mit geschlossener schirmförmiger Kronenstruktur waren. Weiterhin wurde gezeigt, dass die Biomasse von Sträuchern mit büschelartiger Kronenstruktur und mehreren Stämmen (e.g. *A. oerfota*) auch mittels des Kronenvolumens als alleinigem Prädiktor akkurat bestimmt werden konnte.

Die Analyse der Vegetationsbedeckung mittels Landsat Bildern von 1976 bis 2012 zeigte, dass die mit Sträuchern und Baumsavanne bedeckte Fläche seit 1970 von 45% auf 9% abgenommen hat, während die Fläche mit hoher Dichte an Gehölzpflanzen von 22% auf 61% zunahm. Die Häufigkeit von regenerativen Gehölzpflanzen (<1 m Höhe) war hoch in den Gebieten mit niedriger Deckung an Gehölzpflanzen, aber signifikant geringer in Gebieten mit hoher Deckung an Gehölzpflanzen. In eingezäunten Flächen erhöhte sich die Gesamtdichte an Gehölzen im Vergleich zu offenem Weideland unabhängig vom Deckungsgrad, besonders der Anteil an Nachwuchs (<1 m Höhe).

Die oberirdische Biomasse bzw. C-Vorräte unterschieden sich signifikant in den Gebieten mit unterschiedlich starker Gehölzausbreitung, wobei die C-Vorräte von 2 Mg ha⁻¹ in Gebieten mit niedriger Ausbreitung bis zu 9 Mg ha⁻¹ in Gebieten mit hoher Ausbreitung reichten. In den eingezäunten Flächen war der C-Vorrat in der Krautbiomasse um 50% höher als im offenen Weideland. Allerdings war dieser Beweidungseffekt auf die oberirdische Biomasse stark von den Charaktereigenschaften der Gehölzpflanzen beeinflusst, beispielsweise von der Dichte an Gehölzen, deren Kronenbedeckung, Artenzusammensetzung und weiteren Merkmalen.

Der SOC-Vorrat in 0-40 cm Bodentiefe reichte von 30 Mg ha⁻¹ im offenen Weideland und hoher Gehölzausbreitung bis zu 81 Mg ha⁻¹ in eingezäunten Flächen unter niedrigem Ausbreitungsniveau. Soil OC und TSN unterschieden sich nicht im eingezäunten Gebiet bei hoher Gehölzausbreitung, waren aber zwei mal so hoch in eingezäunten Flächen im Vergleich zum offenen Weideland bei geringer Gehölzdichte. Soil OC korrelierte positiv mit TSN sowie der Kationaustauschkapazität (CEC) des Bodens, aber negativ mit dem Sandgehalt. Entgegen der Erwartung folgte der SOC-Vorrat nicht dem Muster ansteigender oberirdischer Biomasse

und zunehmender Gehölzausbreitung sondern schien stattdessen von den variierenden Bodeneigenschaften beeinflusst zu sein.

Diese Studie hebt den Einfluß der Ausbreitung von Gehölzpflanzen und die Abnahme des Beweidungsdrucks auf den Kohlenstoffvorrat des Ökosystems hervor. Die hier entwickelten allometrischen Modelle können als Werkzeug für zukünftige Biomasse und Kohlenstoff-Sequestrierungsstudien in semiariden Regionen Ostafrikas dienen. Die in dieser Arbeit präsentierten Informationen über Ökosystemkohlenstoffvorräte können dabei helfen, Beweidungseffekte und Vegetationsdynamik zu integrieren. Das Verständnis dieser Wechselwirkungen ist notwendig, um ein geeignetes Weidemanagement und eine Politik zu entwickeln, die das C-Speicherpotenzial des Weidelandes als Klimaschutzbeitrag und Anpassungsstrategien bzw. Mechanismen wie „Zahlung für Ökosystemdienstleistungen“ miteinander verknüpfen kann.

8 Curriculum vitae

I. PERSONAL INFORMATION

Name: Hasen Yusuf Mohammed
Nationality: Ethiopian
Place of birth: Wollega
Birth date: 07/07/1979
Address in Ethiopia: Wollega University, College of Agriculture and Rural Development, P.O.Box-315, Nekemte, Ethiopia

II. EDUCATIONAL QUALIFICATION

University degree	qualification	Certificate	Date awarded
1 st degree	Plant sciences	BSc	July 6 2002
2 nd degree	Botanical sciences	MSc	July 2008

III. WORK EXPERIENCES

Position	Institution	Duration
Director of research and community service	Wollega University	April 2009 - September 2010
Lecturer in Plant Science department	Wollega University	November 2007 -March 2009
Junior researcher in Agronomy and Crop physiology	Oromia Agricultural Research Institute (OARI)	February 2003 - November 2007

Publication

A). Publications with peer review process

1. **Hasen Yusuf**, Treydte, A.C., Abule, E., and Sauerborn J. (2013). Predicting aboveground biomass of encroacher woody species in semi-arid rangelands, Ethiopia. *Journal of Arid Environments* 96, 64 – 72.
2. **Hasan Yusuf**, Trydete A.C, Sebsebe Demissew and Zerihun Woldu,2011. Assessment of woody species encroachment in grasslands in Nechsar National Park, Ethiopia, *African Journal of Ecology* 39, 397-409.
3. Tolera Abera, Daba Fayisa, **Hasan Yusuf**, Olani Nikus and A.R. Al-Tawaha (2005). Grain yield of maize as affected by biogas slurry and N-P fertilizer rate at Bako, Western Oromia, Ethiopia, *Bioscience Research* 2(1), 31-37.

4. Tolera Abera, Girma W/Tsadik, Daba Feyisa, **Hasan Yusuf** and Gemechu Keneni (2005). Evaluation of improved wheat varieties under different management practices in Eastern Wollega highlands, Pakistan Journal of Biological Sciences, 8(6), 849-854.
5. Tolera Abera, Daba Feyisa, **Hasan Yusuf** and Tesfaye G/giorgis 2005. Influence of Precursor crops on organic and inorganic fertilizer response of maize at Bako, western Ethiopia, Pakistan Journal of Biological Sciences, 8(12), 1678- 1684.

B). Submitted publications with peer review process

1. **M. Hasen Yusuf**, A.C. Treydte , and J. Sauerborn: Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on soil carbon and nitrogen, submitted to *PLOS ONE* (publisher's acknowledgement of receipt enclosed)
2. M. Hasen Yusuf, A.C. Treydte , E. Abule , and J. Sauerborn: Managing semi-arid rangelands for carbon storage: Grazing and woody encroachment effects on aboveground C stock density (publisher's acknowledgement of receipt enclosed)

C). Publications without peer review process

1. **M. Hasen Yusuf**, Treydte , A.C., Sauerborn, J. (2013). How can grazing management and woody cover can alter herbaceous species diversity, forage quantity, and carbon stocks in a semi-arid rangeland of Ethiopia? Proceeding of *UNCCD 2nd Scientific conference*, Bonn, Germany, 09-12, 2013.
2. **M-Hasen Yusuf** , Angassa, A., Baumgartner, S., Sauerborn, J., Treydte, A.C. (2012). Sustainable rangeland management: what have we learned about livestock and wildlife herbivory? Proceeding of 4th International Ecosummit Conference, Ohio, USA, 29 September - 06 October, 2012.
2. Tolera Abera, Daba Fayisa, **Hasan Yusuf** 2005. Effects of inorganic and organic fertilizer on Maize – Climbing Bean Intercropping in soil fertility in Western Oromia. Proceeding of the Global Food and Product Chain Dynamics, Innovations, Conflicts and Strategies, Deutscher Tropontag, October 11-13, 2005. Stuttgart-Hohenheim.
3. Tolera Abera, Daba Feyisa, Girma W/Tsadik, **Hasan Yusuf** and Gemechu Keneni 2005. Evaluation of improved Wheat varieties under different management practices in Eastern Wollega highlands. Proceedings of the 12th Regional Wheat Workshop for Eastern Central and Southern Africa pp. 134-139, Nakuru, Kenya.

Signature

Date