

**EFFECT OF SOIL AND WATER CONSERVATION PRACTICES ON THE
DYNAMICS OF SOIL BIOTA AND BIOMASS CARBON IN NORTHERN
HIGHLANDS OF ETHIOPIA**

PhD DISSERTATION

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**EFFECT OF SOIL AND WATER CONSERVATION PRACTICES ON THE DYNAMICS
OF SOIL BIOTA AND BIOMASS CARBON IN NORTHERN HIGHLANDS OF
ETHIOPIA**

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Natural Resources Management and Environmental Sciences**

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DOCTOR OF PHILOSOPHY IN SOIL SCIENCE**

By

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As PhD research advisory committee members, we hereby certify that we have read and evaluated this dissertation, which is titled **“EFFECT OF SOIL AND WATER CONSERVATION PRACTICES ON THE DYNAMICS OF SOIL BIOTA AND BIOMASS CARBON IN NORTHERN HIGHLANDS OF ETHIOPIA”**, and has been prepared under our guidance by Mengistu Welemariam Gebrekirstos. We recommend that the dissertation be submitted as it fulfills the requirements of Haramaya University.

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Dedication

This dissertation is dedicated to my father Welemariam Gebrekirstos and my mother Tiblets Berhe who have invested a lot on me to reach this stage of educational success.

STATEMENT OF THE AUTHOR

First, I declare that this dissertation is my bonafide work and that all sources of materials used for writing it have been duly acknowledged. This dissertation has been submitted in partial fulfillment of the requirements for PhD degree at Haramaya University and deposited at the University library to be made available to borrowers under the rules of the library. I solemnly declare that this dissertation is not submitted to any other institution anywhere for the award of any degree, diploma, or certificate.

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BIOGRAPHICAL SKETCH

The author was born on February 5, 1984 in Shire Endaselassie, Tigray Regional State, Ethiopia to his father Welemariam Gebrekirstos and his mother Tiblets Berhe. He attended Elementary, Junior and Secondary Education at Embadanso (1991-995), Tsehay Junior Secondary School (1996-1997) and Shire Senior Secondary School (1998-2001) in Shire. He joined Debub University, Dilla Campus in 2001 and graduated with B.Ed degree in Biology in 2005. Then, he worked in Sheraro Preparatory School until he joined Mekelle University in 2008 for a postgraduate study where he received his MSc degree in Tropical Land Resources Management in 2010. After graduation, he served as a lecturer and registrar head in College of Agriculture and Natural Resources at Shambu Campus of Wollega University until he joined the School of Graduate Studies of Haramaya University in October 2013 to pursue a study leading to the degree of Doctor of Philosophy in Soil Science.

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LIST OF ACRONYMS AND ABBREVIATIONS

AAOC	Aggregate Associated Organic Carbon
AC	Arbuscular Colonization
AMF	Arbuscular Mycorrhiza Fungi
ANOVA	Analysis of Variance
ASD	Aggregate Size Distribution
Avail. K	Available Potassium
Avail. P	Available Phosphorus
BD	Bulk Density
BoANR	Bureau of Agriculture and Natural Resources
CBSWC	Community Based Soil and Water Conservation
CF	Coarse Fragment
CV	Coefficient of Variation
di	Layer Depth
dm	Dry Matter
EEG	Easily Extractable Glomalin
DMRT	Duncan Multiple Range Test
FAO	Food and Agricultural Organization
FS	Foot Slope
HC	Hayphal Colonization
ISWC	Integrated Soil and Water Conservation
LSD	Least Significant Difference
MBC	Microbial Biomass Carbon
mM	Milli Molar
Mg	Mega gram
MS	Middle Slope
Ms	Mass of Sand
Mt	Mass Total of Soil
MWD	Mean Weight Diameter
PCA	Principal Component Analysis
PSNP	Productive Safety Net Programe
PVLG	Polyvinyl Lactoglycerol
REST	Relief Society of Tigray
SAS	Statistical Analysis System
SEM	Standard Error of Mean
SI	Stability Index
SMB	Soil Microbial Biomass
SMI	Soil Macro Invertebrates
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SQ	Stability Quotient
SW	Soil Weight
SWC	Soil and Water Conservation
TG	Total Glomalin

List of Acronyms and Abbreviations (Cont.....)

TRC	Total Root Colonization
TSBF	Tropical Soil Biology and Fertility
UNDP	United Nations Development Programme
US	Upper Slope
VC	Vesicular Colonization
WFP	World Food Program
WSA	Water Stable Aggregate
WRB	World Reference Base

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EFFECT OF SOIL AND WATER CONSERVATION PRACTICES ON THE DYNAMICS OF SOIL BIOTA AND BIOMASS CARBON IN NORTHERN HIGHLANDS OF ETHIOPIA

ABSTRACT

Soil and water conservation practices in Tigray, northern highlands of Ethiopia are given special emphasis for the restoration of degraded lands. However, information on the effect of SWC practices on the dynamics of soil biota is limited. This study was conducted with the objective of evaluating the effect of two decades long community-based soil and water conservation (CBSWC) practices, viz, stone terraces, enclosure with and without stone terraces which were evaluated against non-conserved communal grazing lands on soil organic carbon (SOC) stock, microbial biomass carbon (MBC), arbuscular mycorrhiza fungi (AMF), soil glomalin and soil macroinvertebrates (SMIs) abundance and diversity. Soil samples were collected from each CBSWC systematically using transects based on slope positions. The selected soil properties were measured using standard procedures. Besides, SMI abundance and diversity were computed. Analysis of variance (ANOVA) was used to analyze data using SAS 9.2. The results revealed that the highest SOC stock (29Mg C ha⁻¹) was found in enclosures with terraces followed by enclosures without terraces (24 Mg C ha⁻¹), and terraces (21 Mg C ha⁻¹), while the lowest (16 Mg C ha⁻¹) was found in non-conserved communal grazing lands. Microbial biomass carbon was the highest (i.e., 640 mg kg⁻¹soil) in enclosures without terraces followed by enclosures with terraces (570 mg kg⁻¹soil); terraces (440 mg kg⁻¹soil) and non-conserved communal grazing lands (370 mg kg⁻¹soil). In this study five major AMF genera including Glomus, Acaulospora, Gigaspora, Scutelospora and Entrphospora were identified. Glomus, the abundant genus accounted for 52% of the total spore density followed by Acaulospora 18%. Enclosures had the highest spore density (60%) followed by stone terraces (23%), while open non-conserved communal grazing lands had the lowest (17%). Total root colonization along the CBSWC measures ranged from 48.6% in open non-conserved communal grazing lands to 68.7% in enclosure with terrace. In addition three forms of AMF root colonization namely hyphae, arbuscules and vesicles were identified. All forms AMF root colonizations were higher in the enclosures followed by terraces. Terraces had significantly higher ($P < 0.05$) total AMF root colonization than open non-conserved communal grazing lands. Easily extractable glomalin (EEG) and total glomalin (TG) were significantly ($P < 0.05$) higher in enclosures as compared to terraces and non-conserved grazing lands. Enclosures had significantly ($P < 0.05$) higher percent of water stable aggregate (WSA) than non-conserved grazing lands. Mean weight diameter (MWD) were also significantly ($P < 0.05$) higher in enclosures. Soil macroinvertebrate diversity was higher (1.21) in the enclosure supported with terraces followed by enclosures without terraces (1.01) and the lowest (0.9) was observed for the non-conserved communal grazing lands. Soil macro invertebrate diversity was also significantly ($P < 0.05$) higher (1.26) in wet than dry (0.70) season. The highest (41%) Sorensen similarity index among SMIs was found between enclosure with terraces and enclosures without terraces during the wet season. The lowest (20%) Sorensen similarity index was found between terraces and enclosures with terraces in dry season. Soil macroinvertebrate abundance was higher in upper (0-10 cm) than lower (10-20 and 20-30 cm) soil depths. It is concluded that building of terraces and enclosures on the degraded highlands improved soil organic carbon stock, microbial biomass carbon, arbuscular

mycorrhiza, Glomalin and soil macroinvertebrate abundance and diversity. Thus, the study recommends scaling out community based soil and water conservation practices as they are proven local technologies so as to regenerate degraded ecosystems sustainably.

Keywords: Exclosure, Terrace, Microbial biomass, Soil organic carbon stock, Arbuscular mycorrhizae fungi, Soil macro invertebrate, Glomalin, Water stable aggregate

CHAPTER 1. GENERAL INTRODUCTION AND DESCRIPTION OF THE STUDY AREA

1.1. General Introduction

Land degradation is a common phenomenon, with more than 20% of all cultivated areas, 30% of forests and 10% of grasslands undergoing degradation in the world (Bai *et al.*, 2011). It is the result of numerous factors, including physical as well as anthropogenic activities (Mahala *et al.*, 2017). It involves the interaction of changes in the physical, chemical and biological properties of the soil and vegetation (Shiferaw *et al.*, 2013).

Ethiopian Highlands cover over 50% of the country and are home to more than 90% of Ethiopia's population of over 80 million people (Hurni *et al.*, 2010). The process of land degradation is accelerated due to high human and livestock population pressure, highly variable and unreliable rainfall and steep topography (Temesgen, 2015). Around 50% of rural highland areas are classified as degraded (Shiferaw *et al.*, 2013). The impact of erosion is more serious in the Tigray highlands, northern Ethiopia (Tamene, 2005). Over exploitation of natural resources for years has been a major cause for the severe land degradation in the region (Gebremeskel *et al.*, 2017). It is estimated that 30-50 percent of soil productivity has been lost for the past 500 years (WFP, 2013). Land degradation decreased the fertility of the soils; diminished grazing land and resulted in an overall loss in agricultural productivity (Nyssen *et al.*, 2007).

The regional government embarked on an economic development strategy, which places greater emphasis on SWC practices to overcome land degradation (BoANR, 2014). Besides, the natural resources conservation program has focused on the full participation of the community as a top priority (Vanmaercke *et al.*, 2010).

The different soil and water conservation practices (SWC) have been started and implemented in Tigray region since the 1970s (Shiferaw *et al.*, 2013; WFP, 2013; Gebremeskel *et al.*, 2017). The World Food Program through food for work scheme has been widely implemented for soil and water conservation activities since 1974. Besides, Relief Society of Tigray (REST), which is a local NGO also started to implement SWC practices in the 1990's as one component of

integrated watershed management through community mobilization (BoANR, 2014). Relief Society of Tigray used two approaches for implementation of SWC activities in Tigray region, viz., free labor of the community and productive safety net program (PSNP) (WFP, 2013; BoANR, 2014). Many of the soil and water conservation structures constructed in Tigray are fully owned by the local communities (BoANR, 2014) and contributed to ensuring their sustainability (Kifle, 2012). The SWC practices include stone faced terraces, enforcement of grazing restrictions and plantation development efforts in degraded areas throughout the region (Nyssen *et al.*, 2007).

Due to steepness of topography of the area, stone terraces are generally quite common and are relatively easy to construct during the dry season (Gebre egziabher *et al.*, 2009). Majority (75%) of the farmers in Tigray are in favor of building stone bund on their land because the local community recognizes this conservation measure as being beneficial (Nyssen *et al.*, 2007). Consequently, stone terraces improved numerous soil properties including soil organic matter, soil fertility and soil aggregation (Rashid *et al.*, 2016).

Another low input rehabilitation measure established on badly eroded land is enclosure (Birhane *et al.*, 2010). It involves protecting the land from cutting of trees and grazing, and enables vegetative regeneration (Mekuria *et al.*, 2007); and increase biomass production and combat land degradation in the area (Descheemaeker *et al.*, 2006). This practice was brought to scale in the early 1990s (WFP, 2013)

Soil management and conservation practices influence soil health through their effect on soil organic carbon, soil macro and microbiota (Corsi *et al.*, 2012; Njira and Nabwami, 2013). Soil microbial biomass carbon is one of the biological early indicators of changes and constitutes an important source and sink of nutrients (Moussa *et al.*, 2007; Xiu-Mei *et al.*, 2008). It regulates the carbon storage, nutrient cycling and climate change mitigation (Rousk *et al.*, 2013) and responds more rapidly to soil disturbance and short-term environmental changes than does SOC (Bergstrom *et al.*, 1998; Haynes, 2008). Recent studies showed that land degradation resulted in significant decline in microbial biomass carbon (Rinnan *et al.*, 2007). Information about soil microbial biomass can help to quantify the extent of degradation and may help to provide the

effective methodology for the restoration in the degraded terrestrial ecosystems (Dwivedi & Soni, 2011).

Soil management reduces the amount of carbon released to the atmosphere and helps to alleviate the problem of global warming and climate change (Chan, 2008). However, land degradation causes depletion of the soil organic carbon and leads to degradation in soil quality and declining biomass productivity (Lal, 2007).

Soil microbial communities are considered as vital factor for the functioning of ecosystems (Oehl *et al.*, 2003; Oruru and Njeru, 2016). Among the soil microbial communities, AMF are key components accounting for about 30% of the whole soil microbial biomass (Rilling *et al.*, 2002). Mycorrhizal formation restores soils of disturbed sites of the semiarid areas by decreasing water scarcity (Barea *et al.*, 2011), regulate abiotic and biotic stresses to plants (Maciel *et al.*, 2014) and enhance land restoration (Allen *et al.*, 2003). They are more common in nutrient limiting environments to mitigate plant stress (Ćirić *et al.*, 2012). Arbuscular mycorrhizal fungi are important drivers of soil microbial communities, both because of their privileged access to plant carbon resources, and because of their extensive hyphal networks (Fester & Sawers, 2011; Aggarwal *et al.*, 2011)

Arbuscular mycorrhizae fungi are also important for soil aggregation through the formation of soil glomalin (Rilling, 2004; Borie *et al.*, 2008; Aggarwal *et al.*, 2011). The name glomalin was given after Glomales, an order of arbuscular mycorrhizal fungi. The fungi use carbon from the plant to grow and make glomalin. In return, the hyphae extend the reach of plant roots. Hyphae function as pipes to funnel more water and nutrients—particularly phosphorus—to the plants (Wright and Updharma, 1996)

Soil glomalin is used for carbon sequestration, improving soil structure and stability against erosion (Ćirić *et al.*, 2012). Glomalin is the most important source of the soil active organic carbon (Rilling *et al.*, 2003) and forms aggregates which add structure to soil and protect stored soil carbon (Wright *et al.*, 2007). Glomalin content is an indication of changes in the environment due to management practice (Loureiro *et al.*, 2010).

The sustainable function of natural and managed ecosystems is also dependent on the contribution of soil fauna (Laveli *et al.*, 2008; Ayuke *et al.*, 2009). The abundance and diversity of soil fauna integrate physical, chemical and biological properties of soil, and reflect general ecological change (Menta *et al.*, 2008; Paolo *et al.*, 2010; Ali *et al.*, 2012). It is one of the most important evaluation criteria of soil ecosystems and resistance to different forms of degradation (Loreau *et al.*, 2002).

Soil fauna constitute 23% of the total diversity of living organisms (Decaens *et al.*, 2006). They are the most conspicuous soil animals that have direct effect on soil properties in the tropics through their feeding and burrowing activities (lal, 1998). They are also essential for the maintenance of healthy productive soils (Begum *et al.*, 2014). Soil degradation in the tropics is related to extreme reductions in activity and diversity of soil fauna (Lal, 1998).

Soil biological resources are the base for an ecosystem to function sustainably. This can be achieved when these resources are conserved using different management practices of which SWC measures are the dominant ones where the region has focused (BoANR, 2014). However, there is lack of quantification and documentation on the benefits of SWC practices on soil biota. This triggered to conduct a reaserch focusing on the evaluation of the change brought by SWC practices on soil biota and microbial biomass carbon. Quantifying the dynamics of soil biota and biomass carbon is important for sustainable soil management and to provide a basis for subsequent measurements (Liu *et al.*, 2012; Xu *et al.*, 2013).

Previously some studies have been conducted in the region on SOC stock in response to land use change (Mekuria *et al.*, 2007; Girmay and Singh, 2012), microbial biomass carbon in response to land uses (Gelaw *et al.*, 2014), on AMF (Birhane *et al.*, 2010, 2017), land use land cover change (Kassa *et al.*, 2012), on erosion control, sediment yield and vegetation diversity (Nyssen *et al.*, 2007; Mekuria *et al.*, 2007). Thus, there is dearth of information concerning the effect of SWC measures on the dynamics of soil biota *viz*; soil organic carbon stock, microbial biomass carbon, arbuscular mycorrhiza fungi, glomalin and soil macroinvertebrates. Therefore, this study was conducted with the following hypothesis:

- H₀₁: Soil and water conservation practices have no significant effect on soil organic carbon stock and microbial biomass carbon
- H₀₂: Soil and water conservation practices have no significant effect on arbuscular mycorrhizal spore density and root colonization
- H₀₃: Soil and water conservation practices have no effect on soil glomalin, aggregate stability and aggregate associated SOC and
- H₀₄: Soil and water conservation practices have no effect on the abundance and diversity of soil macro invertebrates

Therefore to test the above null hypotheses, this study was commenced with the general objective of evaluating effect of soil and water conservation practices on the dynamics of soil biota and biomass carbon in northern highlands of Ethiopia

The specific objectives of the study were:

- To determine SOC stock and MBC distribution in relation to soil and water conservation practices.
- To identify and characterize AMF fungi spore density and root colonization of dominant woody plant species in response to soil and water conservation practices.
- To determine soil glomalin, aggregate stability and SOC associated in soil aggregates as a result of SWC practices
- To evaluate on the abundance and diversity of soil macroinvertebrates in relation to SWC practices

1.2. Description of the Study Area

The study was conducted in Degua Temben district, which is located 50 km west of Mekelle, regional capital of Tigray region, northern Ethiopia. Geographically, it is located at 13°16'23''N to 13°47'44''N Latitude and 39°3'17''E to 39°24'48''E Longitude (see Figure 1.1).

The lithology of the study area comprises Mesozoic sedimentary rocks and Tertiary basalt (Nyssen *et al.*, 2007). Soils are developed from calcium carbonate-rich parent material of the Agula shale formation, which consists mainly of marble and limestone (Beyth, 1972). According to the World Reference Base (WRB, 2006), the major soil types of the study area are *Calcaric Cambisols*, *Vertic Leptosols*, *Vertic Cambisols* and *Lithic Leptosols*.

The area receives annual rainfall in the range of 290 to 900mm year⁻¹ with an average amount of 615 mm year⁻¹. The main rainy season stretches from June to September. The mean monthly minimum and maximum temperatures of the area are 19.5 and 25.5 degree celcius respectively. All the study sites are situated at mid-altitude (i.e., 1800-2200) meters above sea level.

Acacia etbaica, *Carissa edulis*, *Dodonea angustifolia*, *Stereospermum kunthianum*, *Rhus vulgaris* and *Euclea racemosa* are the common woody vegetation species of the area (Birhane *et al.*, 2017). The understory vegetation is also dominated by a diverse assemblage of grass and herbs, which are palatable to livestock.

Mixed crop and livestock farming system is the main means of livelihood in the study area. Cultivated lands, forest land, exclosure and communal grazing lands are the major land uses. Barley (*Hordeum vulgare*), tef (*Eragrostis tef*), wheat (*Triticum aestivum*) and Maize (*Zea mays*) are the major crop types which are widely grown in the study area.

The selected study area was representative of the major community-based soil and water conservation practices by purposive sampling. The CBSWC are mainly terraces and exclosures. They have been accepted and practiced by the community for more than two decades.

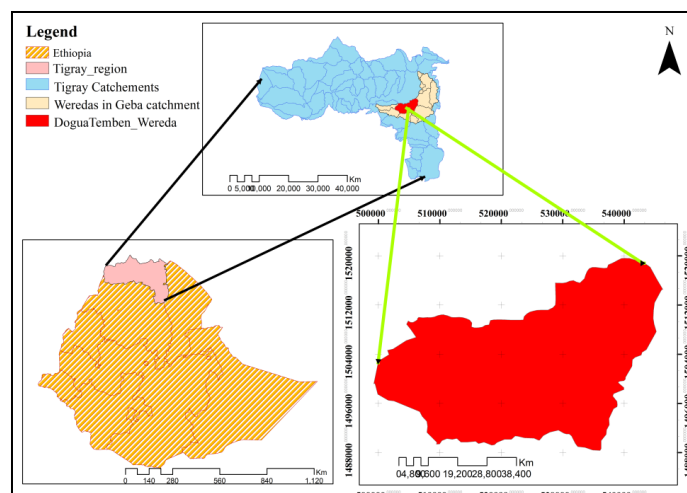


Figure 1.1: Location map of the study area

1.3. Characteristics of the Main Community-based Soil and Water Conservation Measures

The most commonly practiced SWC are terraces and enclosures with and without terraces established since 1997 (BoANR, 2014). These SWC structures are fully constructed and owned by the communities. Before their establishment, the selected conserved grazing lands had comparable initial conditions and history in terms of grazing with the non-conserved free communal grazing lands.

The three selected sites (*Kerano, Tesemat and Alasa*) were each categorized into four management units; terraced grazing lands, enclosure with terrace, enclosure alone and non-conserved free open communal grazing land (Figure 1.2).

Enclosure with stone terrace was restricted from the interference of animals, biological and physical SWC measures. Accumulation of sediment, grasses and litter fall was common on enclosures + terraces. Besides, more woody species were observed as compared to the other SWC measures. These SWC practices had area coverage of 12.74-51.80 ha in the study site.

In the case of enclosures without SWC there was no interference of human and livestock, and no physical structures were built. Trees regenerate naturally and hence showed better vegetation

cover than terraced open grazing lands. Erosion types such as sheet, rill and gully formation in enclosure are relatively less common than terraces and non-conserved free communal grazing lands. Exclosures without SWC cover 14.02-34.70 ha of land in the study site.

The stone terraces in grazing lands were selected as third management unit because they are relatively more stable and durable measures than other physical SWC measures. Terraced grazing lands had more accumulated sediments, organic matter and vegetation cover. Sheet, rill and gully erosion are less common compared to the open communal grazing lands. The area coverage of terrace ranges from 13.87- 24.42 ha.

The open, non-conserved communal grazing land was characterized by low vegetation cover and higher proportions of bare soil with high stone cover. Sheet, rill and gully erosion were common. Area coverage of the non-conserved free communal grazing lands range 11-34.96 ha.

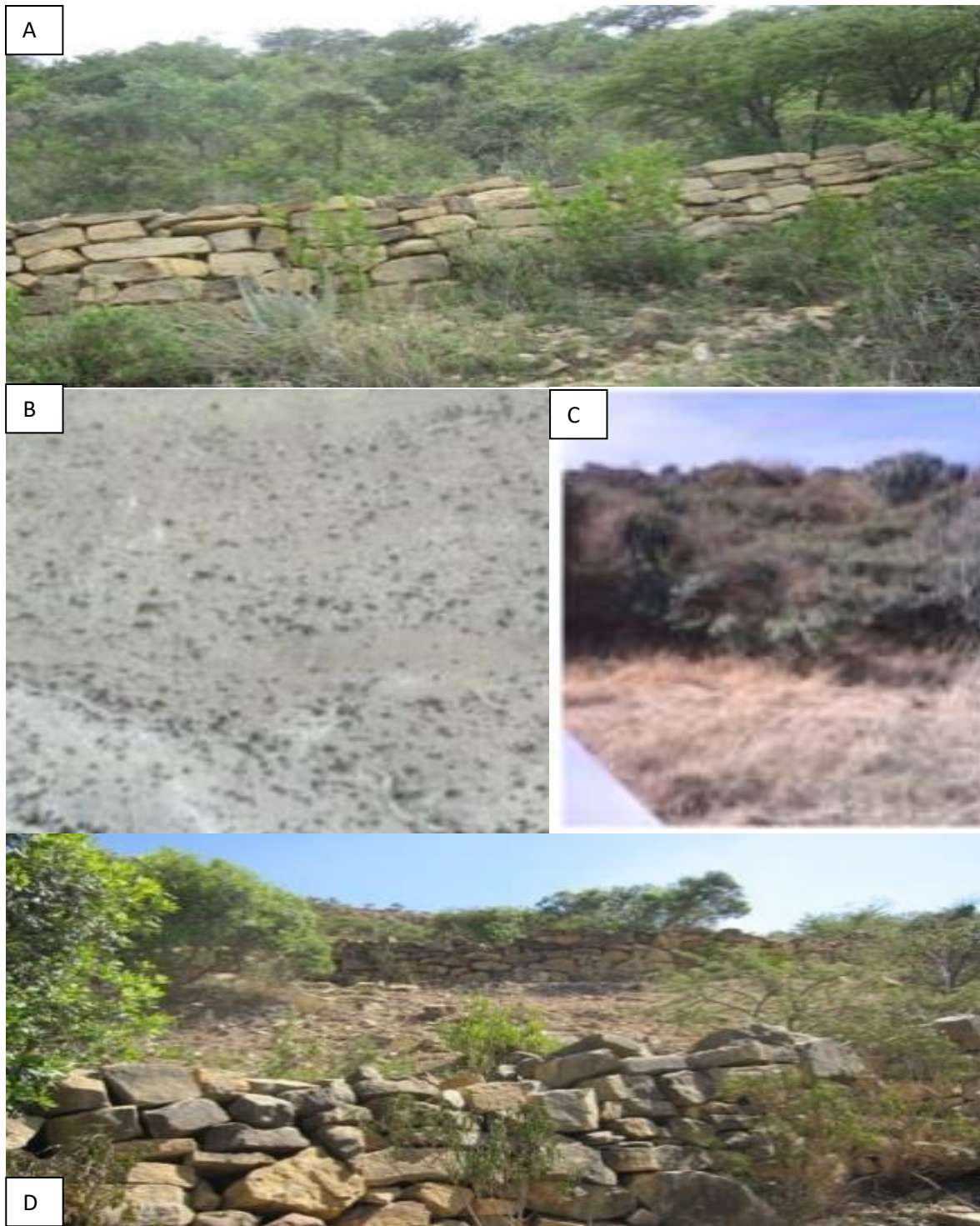


Figure 1.2. Major soil and water conservation measures in one site (A: Exclosures with terrace; B: non-conserved open communal grazing land; C: Exclosure without terrace D: Terraces)

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**EFFECT OF ENCLOSURES BASED COMMUNITY SOIL AND WATER
CONSERVATION PRACTICES ON SOIL ORGANIC CARBON STOCK
AND MICROBIAL BIOMASS CARBON IN THE HIGHLANDS OF
NORTHERN ETHIOPIA**

CHAPTER 2. EFFECT OF ENCLOSURE BASED COMMUNITY SOIL AND WATER CONSERVATION PRACTICES ON SOIL ORGANIC CARBON STOCK AND MICROBIAL BIOMASS CARBON IN THE HIGHLANDS OF NORTHERN ETHIOPIA

Abstract

Land degradation is a threat to natural resources in Tigray highlands of northern Ethiopia, where 30-50 percent of the soil productive capacity has been lost in the past 500 years. Restoration and management of degraded lands through soil and water conservation practices were done to improve soil health through enhancing soil organic carbon (SOC) stock and microbial biomass carbon (MBC). Knowledge on SOC and MBC concentration and distribution is essential to refine soil management, thereby restoring the ecosystem. This research quantified the effect of decades old community based soil and water conservation (SWC) measures, mainly stone terraces, enclosure with and without stone terraces which were evaluated against non-conserved communal grazing lands on the distribution of MBC and SOC stock. Soil samples collection was carried out using systematic sampling design. Transects parallel to each other and to the slope of the landscape were established. In each transect, three landscape positions (i.e., upper, middle and foot slope) were formed. Composite soil samples were taken from four corners and center of 10 m x 10 m plot of each slope positions under the different SWC measures. Analysis of Variance (ANOVA) was used to determine the difference in SOC and MBC using SAS 9.2. Total soil organic carbon concentration was significantly higher in enclosures than terraces and non-conserved free grazing lands. The highest mean value of SOC stock (29 Mg C ha⁻¹) was recorded in enclosures with terraces followed by enclosures without terraces (24 Mg C ha⁻¹) and terraces (21 Mg C ha⁻¹), while the lowest (16 Mg C ha⁻¹) was recorded in non-conserved communal grazing lands. Enclosures with terraces improved SOC stock by 64%, followed by enclosures without terraces by 37%, while terraces improved the SOC stock by 25% compared to non-conserved open communal grazing lands in the last 20 years. Microbial biomass carbon was the highest (640 mg kg⁻¹ soil) in enclosures without terraces, followed by enclosures with terraces, (570 mg kg⁻¹ soil), terraces (440 mg kg⁻¹ soil) and non-conserved communal grazing lands (370 mg kg⁻¹ soil). Enclosures supported with terraces improved and restored the SOC stock and microbial biomass carbon of degraded free grazing lands in the highlands.

Keywords: Land degradation, Rehabilitation, Microbial biomass, Soil Carbon Stock

2.1. Introduction

Soil management and conservation practices influence soil health through their effect on soil microbiota (Corsi *et al.*, 2012; Njira and Nabwami, 2013). Organic carbon and microbial biomass are among the biological indicators describing soil health (Weil, 2004). Soil organic carbon contains two times as much carbon as the atmosphere and 2.5 times higher than that of soil biota (Anikwe, 2010). It plays significant role in soil functioning and the global carbon cycle (Liu *et al.*, 2012) mainly by reducing atmospheric carbon dioxide and helps to alleviate the problem of global warming and climate change (Chan, 2008). It also supports biological activity and dictates the physical properties of soil that determine its resistance to erosion (Alexandra and Bruyn, 1997; Corsi *et al.*, 2012).

The decline of SOC pool leads to soil quality degradation and reduces biomass production. The loss is enhanced by erosion and other degradation processes (Lal *et al.*, 2007). A practical option to increase SOC stock is through SOC sequestration and decreasing soil degradation (Shiferaw *et al.*, 2013) by implementing hillside terraces combined with enrichment plantation in exclosures (Mekuria *et al.*, 2007).

Soils and the microbes that live in them, regulate the carbon storage, nutrient cycling and climate change mitigation (Rousk *et al.*, 2013). Soil microbial biomass (SMB) is responsive to environmental changes (Haynes, 2008). It indicates the soil's capacity to perform ecosystem processes (Loureiro *et al.*, 2010). It also acts as biological early indicator of changes (Moussa *et al.*, 2007; Xiu *et al.*, 2008). It makes up 1–5% of total SOC (Jenkinson *et al.*, 1981) and acts as the most active component in the biochemical process of SOC turnover (Marina *et al.*, 2006). Little change in SMB affects directly ecosystem stability (Moscatelli *et al.*, 2005).

Soil microbial biomass is more sensitive to changes in soil management compared to soil organic matter (Yan *et al.*, 2003). Determination of SMB can help to quantify the extent of land degradation and can provide methods for the restoration of degraded ecosystems (Dwivedi and Soni, 2011). The MBC to SOC ratios are also useful measures to manage soil organic carbon content (Bhatt and banmeru, 2014).

Restoration and management of degraded lands significantly contribute to enhanced SOC (Lal, 2005) and MBC (Haripal and Sahoo, 2014) which are considered as soil health indicators during restoration process (Mureithi *et al.*, 2014). They vary spatially under different management and plant covers (Loureiro *et al.*, 2010) though studies that examine the distribution of these soil properties are limited (Tamene *et al.*, 2011).

Land degradation is the major problem causing loss of SOC and MBC in Tigray highlands (Girmay and Singh, 2012; Gelaw *et al.*, 2014). This is due to overexploitation and mismanagement of these highlands (Gebreegziabher *et al.*, 2009; Shiferaw *et al.*, 2013). Stone faced terraces, enforcement of grazing restrictions and plantation development efforts were implemented as rehabilitation measures since 1970s to tackle land degradation (WFP, 2013; Birhanu, 2014). Area closures (exclosures) were established in the region since 1991 to improve biomass production (Mekuria *et al.*, 2007). They are implemented on biophysically degraded communal grazing lands (Descheemaeker *et al.*, 2006).

Exclosures reduce soil disturbance by decreasing grazing pressure and restricting people's access to uncontrolled cutting of trees and grass for fuel and fodder (Yayneshet *et al.*, 2009). Majority of the farmers (75%) practice terrace construction on their land (Nyssen *et al.*, 2007). Exclosures have also been important rehabilitation measures to increase biomass production (Descheemaeker *et al.*, 2006).

Quantifying and determining the distribution of SOC stock and MBC is important for site-specific sustainable soil management and to provide a valuable basis for subsequent measurements (Liu *et al.*, 2012; Xu *et al.*, 2013). Effective carbon sequestration methods can also be implemented (Li *et al.*, 2013).

Some studies have been conducted previously that examine the SOC stock (Mekuria *et al.*, 2007; Girmay and Singh, 2012), microbial biomass carbon (Gelaw *et al.*, 2014) and on AMF (Birhane *et al.*, 2010, 2017) in response to land use changes and exclosures. However, there is limited information specifically on the effect of community SWC measures on SOC stock and MBC

distribution. Thus, the objective of this study was to evaluate the effect of two decades long community based SWC measures mainly terraces and exclosures on distribution of SOC stock and microbial biomass carbon.

2.2. Materials and Methods

2.2.1. Description of the Study Area

The study was conducted in Degua Temben district, which is located at 50 km west of Mekelle, regional capital of Tigray region, northern Ethiopia. Geographically, it is located at 13^o16'23''N to 13^o47'44''N Latitude and 39^o3'17''E to 39^o24'48''E Longitude (Figure 1.1).

The lithology of the study area comprises Mesozoic sedimentary rocks and Tertiary basalt (Nyssen *et al.*, 2004). Soils of the study sites are developed from calcium carbonate-rich parent material of the Agula shale formation, which consists mainly of marble and limestone (Beyth, 1972). According to world reference base (WRB, 2006) soil classification system, *Calcaric Cambisols*, *Vertic Leptosols*, *Vertic Cambisols* and *Lithic Leptosols* are the dominant soil types.

The area receives annual rainfall of 290 to 900mm year⁻¹ with an average value of 615 mm year⁻¹. The main rainy season is June to September. All the study sites are classified as mid-altitude (1800-2200 meter above sea level) (detail description in section 1.2).

2.2.2. Sampling Technique and Sample Size

The study was conducted in three nearby sites (*Kerano*, *Tesemat* and *Alasa* with in the district) selected by purposive sampling and having all the SWC measures. In each SWC measures, three transects separated at a minimum distance of 75 m were established. Transects were parallel to each other and to the topography of the landscape. In each transect three landscape positions (i.e., upper, middle and foot slope) were established. The upper slope (US) position is the uppermost portion of each study site and it can receive little or no overland flow but may contribute runoff to down slope areas. The middle slope (MS) position receives overland flow from the upper slope and contributes runoff to the foot slope (FS). The FS represents the lowest

part of each study site and receives overland flow from both mid and upper slopes (Mekuria and Aynekulu, 2013).

Soil samples were collected from the top 0-15 and 15-30 cm at four corners and center of a 10 m x 10 m size plot using “X” sampling design from terraces, enclosure with terrace, enclosure without terraces and from non-conserved communal grazing lands from three transects spaced at a minimum distance of 75 m (Mekuria, 2010).

A total of 108 soil samples were collected (i.e., four conservation measures*three slope positions*three samples*three replications) from surface (0-15 cm) for microbial biomass and 216 soil samples (i.e., four conservation measures*three slope positions*three samples*two depths*three replications) from surface (0-15 cm) and subsurface (15-30 cm) for the determination of soil organic carbon stock.

2.3. Analysis and Determination of Soil Properties

2.3.1. Determination of Soil Microbial Biomass Carbon

The soil samples were sieved through a 2 mm mesh to remove stones, roots, and large organic residues, and then sealed in plastic bags and stored at 4°C. Soil microbial biomass carbon was determined using the Substrate Induced Respiration (SIR) method (Anderson and Domsch, 1975). The samples were immediately taken to the laboratory for MBC determination. In the laboratory, 100 g of the field moist soil was thoroughly mixed with 400 mg glucose and 20 g of the mixture was transferred to four nylon bags. Each bag was poured into a 500 ml laboratory bottle mixed with 20 ml 0.1M NaOH solution. The bottles were immediately closed with gas-tight caps, and incubated at 22 °C for 4 hours together with two blanks (without soil samples).

The soil samples were then immediately removed from the bottles to precipitate the CO₂ as barium carbonate by adding 2 ml of 0.5 M barium chloride solution. The remaining sodium hydroxide was titrated against 0.1 M HCl using 3-4 drops of phenolphthalein solution as indicator. The result was calculated according to Schuman *et al.* (2002) (Equation 2.1).

$$mg\ CO_2\ 100g^{-1}h^{-1} = \frac{(B - S) * 2.2 * 100}{4 * SW * dm} \quad [Eq\ 2.1]$$

Where:-

B= mean volume of HCl consumed by blanks (ml)

S= mean volume of HCl consumed by samples (ml)

4=Incubation time (h)

100=conversion factor (100 g dm)

2.2 =conversion factor (1 ml 0.1 M HCl corresponds to 2.2 tolerant mg CO₂)

SW=initial soil weight (g)

Dm=soil dry matter (%)

Assuming a respiratory quotient of 1.1 mg CO₂ 100 g⁻¹ dm h⁻¹ corresponds to 20.6 mg biomass-C.100g⁻¹ dm. Thus, this factor was used to convert the CO₂ into MBC.

2.3.2. Determination of Organic Carbon, Bulk Density and Coarse Fragment

Organic carbon was determined by the wet digestion method using potassium dichromate (Walkley and Black, 1934). The microbial quotient was determined as the ratio of MBC/SOC after converting the percentage of SOC in to mg 100gm⁻¹dry soil. Bulk density was determined by core method. Coarse fragment was determined as percentage weight of soil greater than 2mm in diameter.

2. 3.3. Determination of Soil Organic Carbon Stock

The carbon held in the upper profile is often the most chemically decomposable, and the most directly exposed to natural and anthropogenic disturbances (IPCC, 2003). Therefore, soil organic carbon pool was estimated up to the depth of 30 cm in this study according to Lal *et al.* (2007).

The data for SOC pool was calculated by using the following equation

$$\text{SOC stock} = \text{SOC}_i * \text{BD} * d_i * (1 - \text{CF}_i)/100 \quad [\text{Eq 2.2}]$$

Where,

SOC_i = Soil organic carbon of a given soil depth, Mega gram C ha⁻¹.

BD (Bulk density) = Soil mass per sample volume, Kilo gram soil m⁻³ (equivalent to Kg m⁻³).

d_i = Horizon depth or thickness of soil layer, m

CF_i = % volume of coarse fragments / 100, dimensionless.

The rate of change of total organic carbon and SOC stock over time was computed by subtracting the stocks of free grazing land from that of terraces and exclosures and dividing it by the number of years since the terraces and exclosures were established.

2.4. Data Analysis

The mean, minimum, maximum, standard error and coefficient of variation (CV) were determined using descriptive analysis of the original data variables. The significance of the difference between means of each SOC, MBC and MBC/SOC ratio variable for different SWC measures was carried out by analysis of variance (ANOVA) using SAS 9.2 (SAS, 2004). A Pair wise mean separation was done by Least Significant Difference (LSD). Pearson correlation coefficient was also used for observing any associations between the parameters.

2.5. Results and Discussion

2.5.1. Results

2.5.1.1. Soil bulk density and organic carbon concentration along community-based soil and water conservation measures

There was significant ($P < 0.05$) variation in the content of SOC among the soil and water conservation measures. Specifically both exclosures had significantly higher SOC than grazing lands with and without terraces. The highest mean value of SOC (2.9%) was found on exclosures supported with terraces followed by exclosures without terraces (2.8%) and terraces (2.5%) respectively (Table 2.1). The lowest mean value of SOC (1.9%) was found in non-conserved communal grazing lands. Conversion of free grazing lands to exclosures resulted about 32-34% increase in soil organic carbon. Besides, construction of terraces on free grazing lands resulted 24% increase in soil organic carbon. The SOC content followed the order exclosures > stone terraces > non-conserved open communal grazing lands.

Though no significant variation, free non conserved grazing lands and terraced grazing lands had relatively higher bulk density values than both types of exclosures (Table 2.1). The bulk density of the area was in the order of exclosures without terraces < exclosures with terraces < terraces alone < non-conserved communal grazing lands. The coefficient of variation (CV) for SOC and BD along the different SWC measures was between 17-27% for organic carbon and 10-18% for BD respectively (Appendix Table 1). Non-conserved grazing lands had relatively higher percentage of coarse fragments, while exclosures with terraces had significantly lower percentage of coarse fragments than the other soil and water conservation measures (Table 2.1). The CV values indicate that the variation in SOC and BD are moderate along the SWC measures.

Table 2.1. Effect of community based soil and water conservation measures on soil properties (Mean \pm SEM)

Soil properties	Non-conserved	Terraces	Exclosures	Exclosures	CV (%)
	grazing lands		+terraces	alone	
SOC (%)	1.9 \pm 0.1 ^c	2.5 \pm 0.1 ^b	2.9 \pm 0.1 ^a	2.8 \pm 0.1 ^a	11.56
Bulk density (kg m ⁻³)	1280 \pm 32 ^a	1260 \pm 28 ^a	1240 \pm 21 ^a	1220 \pm 48 ^a	12.37
Coarse fragment (%)	57.6 \pm 0.2 ^b	53.2 \pm 0.2 ^b	47.4 \pm 0.2 ^a	54.4 \pm 0.2 ^b	19.53
SOC stock (Mg ha ⁻¹)	16.0 \pm 1.0 ^c	21.0 \pm 1.3 ^b	29.0 \pm 1.4 ^a	24 \pm 1.3 ^b	27.37

Means of each parameter followed by the same letter across a row do not differ significantly at $P \leq 0.05$

2.5.1.2. Effect of soil depth on soil organic carbon and bulk density

Soil depth resulted significant variation in SOC content (Table 2.2). The upper soil layer (0-15 cm depth) had significantly ($P < 0.05$) and about 21.4% higher SOC than the lower one (15-30 cm depth). Though no significant variation, soil BD was relatively lower in the upper than lower soil depth. Though no significant variation, again coarse fragment was relatively higher on the upper than lower soil depth. Lack of significant variation in bulk density and coarse fragment could be due to similarity in the parent material along a depth. The total SOC and BD in response to soil depth had a coefficient of variation between 16.7-20.9% for organic carbon and 10.9-11.7 for BD (Table 2.2), which is a moderate variation. The soil organic carbon stock in the upper soil layer is significantly ($P < 0.05$) and about 16.7% higher than in the lower soil layer.

Table 2.2. Effect of soil depth on total SOC, bulk density, coarse fragment and SOC stock (Mean \pm SEM)

Soil parameters	Depths (cm)			
	0-15 cm	CV (%)	15-30 cm	CV (%)
Organic carbon (%)	2.8 \pm 0.1 ^a	16.77	2.2 \pm 0.1 ^b	20.90
Bulk density (kg m ⁻³)	1244 \pm 23 ^a	11.70	1258 \pm 25 ^a	10.99
Coarse fragment (%)	55 \pm 1.0 ^a	18.6	52 \pm 2.0 ^a	27.34
Soil carbon stock (Mg ha ⁻¹)	24 \pm 1.0 ^a	31.95	20 \pm 1.0 ^b	32.00

Means of each soil parameter followed by the same letter across a row do not differ significantly at $P \leq 0.05$

2.5.1.3. Distribution of soil organic carbon stock along the community-based soil and water conservation measures

The SOC stock (Mg C ha^{-1}) in exclosures + terraces was significantly higher ($P < 0.05$) than the other SWC measures (Table 2.1). Terraces also had significantly higher SOC stock than free non-conserved grazing lands. Exclosures with terraces had the highest SOC stock (29 Mg ha^{-1}) followed by exclosures without terraces (24 Mg ha^{-1}) and terraces (21 Mg ha^{-1}) respectively, while the lowest (16 Mg ha^{-1}) was observed in non-conserved free grazing lands. Exclosures resulted about 45% increase in SOC stock than non-conserved free grazing lands. Construction of terraces on free grazing lands also resulted about 24% increase in SOC stock. Soil organic carbon stock under the different soil and water conservation measures had a coefficient of variation 27% which lies under the moderate variability category.

2.5.1.4. Distribution of soil microbial biomass carbon along the community-based soil and water conservation measures

Exclosures had significantly higher MBC than terraces and non-conserved communal grazing lands (Table 2.3). The highest value of MBC (640 mg kg^{-1} dry soil) was found in exclosures without terraces followed by exclosures with terraces (570 mg kg^{-1} dry soil) and terraces (440 mg kg^{-1} dry soil) respectively (Table 2.3). The lowest MBC (370 mg kg^{-1} dry soil) was recorded in non-conserved communal grazing lands. The microbial biomass in exclosures was 35-42% higher than that of non conserved free grazing lands. Similarly, the MBC in terraces was 16% higher than that of non conserved free grazing lands. Thus, the microbial biomass was in the order of exclosures > terraces > non-conserved communal grazing lands.

Though the variation was not significant, the microbial quotient was relatively higher in exclosures than terraces and non-terraced communal grazing lands (Table 2.3). It was also higher in exclosures without terraces than exclosures with terraces. This could be due to the presences of more grasses in exclosures alone than exclosures + Terraces, as grasses are easily decomposable and available for microbes than the woody biomass that were dominant in exclosures+terraces.

Table 2.3. Effect of community-based SWC measures on total values of MBC and MBC/SOC ratio (Mean \pm SEM) of surface soil (0-15cm) depth

SWC measures	MBC (mg kg ⁻¹ soil)	MBC/SOC (%)
Non-conserved grazing lands	370 \pm 10 ^b	1.9 \pm 0.1
Terraces	440 \pm 30 ^b	1.8 \pm 0.2
Exclosures with terraces	570 \pm 40 ^a	2.1 \pm 0.1
Exclosures without terraces	640 \pm 50 ^a	2.3 \pm 0.2
CV (%)	44.77	46.09

Means followed by the same letter across a column do not differ significantly at $P \leq 0.05$

2.5.1.5. Effect of slope position on microbial biomass carbon

Foot slope in exclosure without terraces had significantly ($P < 0.05$) higher MBC than the middle and upper slope positions of the non-conserved free grazing lands (Table 2.4). This showed there was more available carbon source for microbes at foot slope in exclosures without terraces. Microbial biomass carbon contents were inconsistent and not significant along the three slope positions of the other SWC measures. However, there is moderate variability in MBC as the CV values for MBC in response to slope position ranged 34.81-43.5% (Table 2.4).

Table 2.4. Effect of slope position on microbial biomass carbon (mg kg⁻¹ soil) (Mean \pm SEM)

SWC measures	Slope positions		
	Foot slope	Middle slope	Upper slope
Non conserved grazing lands	380 \pm 70 ^{bc}	350 \pm 70 ^b	370 \pm 70 ^{bc}
Terraces	460 \pm 70 ^{abc}	450 \pm 70 ^{abc}	400 \pm 70 ^{abc}
Exclosures + terraces	550 \pm 70 ^{acd}	560 \pm 70 ^{ad}	580 \pm 70 ^{ad}
Exclosures without terraces	720 \pm 70 ^d	560 \pm 70 ^{ad}	640 \pm 70 ^d
CV (%)	34.81	42.38	43.35

Means followed by the same letter within a row do not differ significantly at $P \leq 0.05$

2.5.1.6. Relationship between soil properties

Strong and significant ($R^2=0.87$, $P < 0.01$) correlation was found between MBC and MBC/SOC (Table 2.5), which might indicate that MBC has good contribution to SOC concentration and SOC stock. However, SOC was negatively correlated with bulk density. This is in line with a result of a similar study conducted in south Gonder, north western highlands of Ethiopia (Demelash and Stahr, 2010) and in Middle Sulluh Valley, northern Ethiopia (Hishe *et al.*, 2017) where bulk density was negatively correlated with SOC. Soil organic carbon was also negatively and significantly ($R^2=-0.1$, $P < 0.01$) correlated with MBC to SOC ratio. This might indicate that either the soil organic matter source is resistant or poor quality or the microbes are weak to decompose the organic matter source.

Table 2.5. Correlation coefficient between the soil properties (n=108)

	MBC	SOC	MBC/SOC	BD	Coarse fragment
MBC	1				
SOC	0.39**	1			
MBC/SOC	0.87**	-0.1 ^{ns}	1		
BD	-0.11 ^{ns}	-0.19*	-0.02	1	
Coarse fragment	0.21 ^{ns}	0.11 ^{ns}	0.19	-0.03 ^{ns}	1
SOC stock	0.32**	0.56**	0.07 ^{ns}	0.26**	0.76**

*Correlation is significant at $P \leq 0.05$; ** correlation is significant at $P \leq 0.01$ confidence interval; ns: correlation is not significant

2.5.2. Discussion

Exclosures with terraces had significantly higher SOC as compared to the other SWC measures, which could be due to low disturbance, better vegetation cover and retention of soil materials. In agreement with this results, Demelash and Stahr (2010) reported that stone terraces when supported with vegetation increased soil organic matter accumulation compared to non-supported ones in south Gonder. Similar result also reported that returns of biomass and reduced erosion in exclosures enhanced SOC concentration in arid environments (Schuman *et al.*, 2002; Mekuria and Veldkamp, 2005).

The presence of significantly lower SOC concentration in non-conserved grazing lands (Table 2.1) could be due to the fact that grazing reduces exposes the soil to increased erosion losses and reduced carbon inputs (Mekuria, 2010). Grazing lands with terraces had higher SOC than non-conserved free communal grazing lands. This could show that terraces store more organic carbon from the materials they trap during deposition. Similarly, Amdemariam *et al.* (2011) reported high SOC in conserved fields as compared to non-conserved fields in north western part of Ethiopia. Hishe *et al.* (2017) also found higher SOC in conserved land scape than non-conserved landscape in Sulluh valley of northern highlands of Ethiopia.

Though no significant variation, non- conserved grazing lands had relatively higher bulk density than exclosures and terraces, comparable with the result reported by Amdemariam *et al.* (2011). This could be due to low soil organic carbon and high soil compaction by livestock grazing in non-conserved open communal grazing lands. Other studies clearly showed the importance of organic matter for rehabilitating degraded soils by improving other soil properties (Amare *et al.*, 2013). The BD decreased from 1.28 to 1.26 g cm⁻³ due to terrace construction and from 1.28 to 1.22 g cm⁻³ due to exclosure establishment (Table 2.1). Similar study reported comparable result, where mean value for BD varied from 1.12 g cm⁻³ (in exclosure) to 1.34 g cm⁻³ (in grazing land) in near the study site (Girmay and Singh, 2011). Like wise, Mureithi *et al.* (2014) found that on average, exclosures lowered the BD of top soil from 1.48 g cm⁻³ to 1.19 g cm⁻³ in range lands of Kenya.

Soil organic carbon and BD had CV values 11.56% and 12.37% respectively. This indicated that they have moderate variation in response to the SWC measures. Because CV of less than 10% has weak variability, CV between 10% and 100% is considered as moderate variability while with CV higher than 100% is considered as strong variability (Nielsen and Bouma, 1985).

The upper soil depth (0-15 cm) had significantly higher SOC than lower (15-30 cm) soil depth. This could indicate presence of easily decomposable biomass in the upper soil depth. It was found that high SOC at surface soil is due to the input of organic matter from aboveground biomass (Xu *et al.*, 2013). Other studies reported that soil organic carbon is mainly formed by

the decomposition of plant materials in the soil surface. Soils of Bale Mountains in south eastern Ethiopia also showed similar results (Del *et al.*, 2013).

Soil organic carbon stock decreased with increasing depth which could be due to decrease in organic carbon concentration and gravel content down in depth. Gravel content and SOC affects SOC stock (Xu *et al.*, 2013; Alcantra *et al.*, 2015). The SOC stock for the whole 30 cm soil depth was 44 Mg C ha⁻¹. Similarly, a total of 74 Mg C ha⁻¹ SOC stock was reported in the top 40 cm in enclosure in a nearby site (Girmay and Singh, 2012). These results indicate the importance of such restoration measures in addressing SOC depletion in the area.

Conversion of free grazing lands to enclosures and supporting with terraces increased SOC stock by 64%, enclosures without terraces by 37% and terraces by 25% within twenty years. In a similar result it was reported that enclosures increased SOC stock by more than 50% (Mekuria, 2010). Protected grasslands contributed to significant carbon sequestration (Su *et al.*, 2010). Increased SOC stock is related to the restoration of natural vegetation, which enhances above-ground and below-ground litter inputs (Bird *et al.*, 2002; Mekuria *et al.*, 2007). Vegetation restoration and litter accumulation in enclosures significantly increased plant-soil system carbon storage and sequestration (Bird *et al.*, 2002).

The presence of significantly higher MBC in enclosures could be due to fewer disturbances, which encouraged enrichment of the microbial biomass carbon. The existence of better vegetation cover in enclosures protect the soil from erosion and maintain favorable conditions for microbial growth (Yan *et al.*, 2003). High MBC indicated high efficiency of carbon utilization and increase in ecosystem maturity; and vice-versa (Anderson and Domsch, 1975). High MBC can also be used as bio-indicator for the recovery of degraded soils (Yan *et al.*, 2003). Similar study showed that high MBC in enclosures indicate that they had the potential for the restoration of soil quality (Mureithi *et al.*, 2014).

The low MBC in non-conserved communal grazing lands could indicate that there is inefficient utilization of carbon by microbes, which could be due to stress such as grazing and soil compaction by livestock. It was also found that erosion had a more severe disturbance to soil

microbial biomass (Xiaojun *et al.*, 2013). This caused for the loss of carbon in the form of carbon dioxide (Gonzalez *et al.*, 2011). Similarly, Gelaw *et al.* (2015) in a near to the study site reported that environmental disturbances caused for the decrease in microbial biomass carbon.

The MBC/SOC ratio represents the contribution of microbial biomass to organic carbon in soil and is a more useful assessment index for soil health than either MBC or SOC alone (Santos *et al.*, 2012). The MBC/SOC was low in exclosures and terraces than non-conserved communal grazing lands. However, this is in contrast to the report of Anderson and Domsch (1975) who found high MBC/SOC in fields with high total organic carbon. This could be due to woody biomass and litter in exclosures with terraces and terraced grazing lands, which is relatively resistant to microbial decomposition. In a similar result, it was reported that the MBC/SOC ratio at 0–15 cm decreased with increasing woody plant stand suggesting that woody litter may be less suitable as a microbial substrate than grassland litter (Lio and Bouton, 2008). This might indicate decline of MBC/SOC meant the decrease of available organic matter in soils (Zhu *et al.*, 2010). Based on these probable reasons, the contribution of MBC to soil organic carbon was not significant but relatively lower in exclosures with terraces than exclosures without terraces; and in terraced grazing lands than non-conserved communal grazing lands (Table 2.3).

2.5. Conclusion

The different SWC measures had a significant impact on the ability of the soil to sequester SOC and microbial biomass carbon. Exclosures resulted in higher soil organic carbon stock and MBC than terraces and non-conserved open communal grazing lands. Soil depth affected SOC concentrations as it was observed that the upper soil depth had higher SOC than the lower soil depth. Establishment of exclosures and construction of terraces in the open communal grazing lands had a positive impact in terms of SOC stock and microbial biomass carbon storage. This implies restoration of degraded lands with various soil and water conservation measures improves soil health through improving soil properties. Further research is required on the role of plant species under the soil and water conservation on carbon sequestration.

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**EFFECT OF COMMUNITY-BASED SOIL AND WATER
CONSERVATION PRACTICES ON ARBUSCULAR MYCORRHIZA
FUNGI TYPES, SPORE DENSITIES, ROOT COLONIZATION AND SOIL
NUTRIENTS IN NORTHERN HIGHLANDS OF ETHIOPIA**

CHAPTER 3. EFFECT OF COMMUNITY-BASED SOIL AND WATER CONSERVATION PRACTICES ON ARBUSCULAR MYCORRHIZAL FUNGI TYPES, SPORE DENSITIES, ROOT COLONIZATION AND SOIL NUTRIENTS IN THE NORTHERN HIGHLANDS OF ETHIOPIA

Abstract

Land degradation is a major ecological problem in Tigray high lands, Northern Ethiopia where Steep slopes have been cultivated for many centuries. To solve the problem, restoration measures through soil and water conservation practices were undertaken. The objective of this study was to quantify the effect of two decades long community based soil and water conservation practices on arbuscular mycorrhiza fungi spore density, root colonization and soil nutrients. The soil and water conservation measures considered were stone terraces, exclosures + stone terraces, exclosures alone and which were evaluated against the adjacent non conserved open communal grazing lands. Soil and root samples were collected from the rhizosphere of matured woody plant species using a systematic sampling method from 10 m x 10 m plot based on slope positions. Spores were isolated using wet sieving and decanting method, while arbuscular mycorrhiza fungal root colonization was determined using the gridline intersection method. The study revealed that five major genera of AMF, including Glomus, Acaulospora, Gigaspora, Scutellospora and Entrophospora were identified. Glomus was found to be the most abundant genera, which accounted for 52% of the total spore density, followed by Acaulospora (18%). Besides, exclosures had the highest total spore density (60%), stone terraces accounting (23%) whereas the lowest (17%) spore density was recorded in the open communal grazing lands. Total root colonization among the treatments ranged from 48.6% in the open communal grazing lands to 68.7% in the exclosure with terraces. Percentage of hyphal colonization was higher than percentage of arbuscular and vesicular colonization. The total colonization was in the order of exclosure with terraces > exclosures alone > terraces > non-conserved communal grazing lands. Rehabilitating the communal grazing lands with terraces and exclosures is an important approach for restoring AMF and regenerating the degraded lands.

Keywords: Arbuscular mycorrhiza, exclosure, soil properties, terrace

3.1. Introduction

Degradation of communal grazing land is a widespread problem throughout the sub Saharan Africa and its restoration is a challenge for the management of many semi-arid areas (Yaynesht *et al.*, 2009). It is a major ecological problem in Tigray high lands, Northern Ethiopia where Steep slopes have been cultivated for many centuries without effective soil and water conservation measures (Mekuria, 2007; WFP, 2013). This has accelerated the problem of land degradation (Nyssen *et al.*, 2007).

Terraces and exclosures are important components of SWC practices that are implemented to reverse the land degradation process (Nyssen *et al.*, 2004; Descheemaeker *et al.*, 2006) and restore the natural vegetation (Gebrehiwot and Veen, 2014). Exclosures increased biomass, herbaceous cover and vegetative regeneration (Birhane *et al.*, 2010). Exclosures also improved soil nutrient status, and reduced soil erosion (Descheemaeker *et al.*, 2006). Terraces decreased surface erosion and improved soil conditions (Nyssen *et al.*, 2007). Soil quality is improved after installation of terraces (Amdemariam *et al.*, 2011; Amare *et al.*, 2013).

Soil microbiota are considered a vital factor for the functioning of ecosystems (Oehl *et al.*, 2003; Oruru *et al.*, 2016) of which, Arbuscular mycorrhizal fungi (AMF) are key components (Aminifar and Sirousmehr, 2014). AMF are ubiquitous in natural ecosystems and form symbiotic associations with the majority of terrestrial plant roots (Li *et al.*, 2010) with more than 80 % of the terrestrial plant species (Brundrett, 2004). Arbuscular mycorrhizal fungi are fundamental for soil fertility enhancement (Bedini, 2007), and increase organic carbon content, improve soil stability, water relation and increase phosphorus availability (Rilling, 2004). Mycorrhiza play a key role in regulating abiotic and biotic stresses in plants (Maciel *et al.*, 2014).

Degraded lands are characterized by low levels of AMF population density and diversity (Asmelash *et al.*, 2016). Many studies showed that disturbance of semi-arid ecosystems decreased mycorrhizal spore density and root colonization (Muleta *et al.*, 2008; Barea *et al.*, 2011). It was also reported that livestock and human disturbances decreased AMF spore density, root colonization and nutrient availability (Birhane *et al.*, 2017a), and establishment of

exclosures on degraded communal grazing lands significantly improved spore density and root colonization (Birhane *et al.*, 2017b). Nutrient stocks and concentrations of soil organic matter (SOM), total nitrogen (TN) and available phosphorus (AP) were found to be higher in areas with high Arbuscular mycorrhizal fungi (Muleta *et al.*, 2008).

So far some studies were conducted in the region on the effect of land degradation on land use land cover change (Kassa *et al.*, 2012), soil erosion (Nyssen *et al.*, 2004) and in part on the effect of biological SWC measures, specifically exclosures on arbuscular mycorrhiza spore density and root colonization, and vegetation diversity (Birhane *et al.*, 2017). Quantification of AMF spore types, density and root colonization under the different SWC measures is important to know the status of restoration. Besides, spores from well resored SWC measures can be used as inoculant for multiplication and further application in the restoration of other degraded sites. However, knowledge on the role of SWC practices on this area of research is limited. Therefore, notwithstanding the massive undertakings of community-based soil and water conservation practices and the enormous benefits of AMF in soil fertility restoration; there is dearth of information on the effect of SWC practices on AMF in the area. Therefore, this study was conducted with the objective of quantifying the effect of two decades long terraces, exclosures with and without terrace on AMF spore density, root colonization and soil nutrients.

3.2. Materials and Methods

3.2.1. Description of the Study Area

The study was conducted in Degua Temben district, which is located at 50 km west of Mekelle, Regional capital of Tigray region, northern Ethiopia. Geographically, it is located at 13°16'23''N to 13°47'44''N Latitude and 39°3'17''E to 39°24'48''E Longitude (Figure 1.1). The area is characterized by rugged topography with some flat areas.

The lithology of the study area comprises Mesozoic sedimentary rocks and tertiary basalt. Soils were developed from calcium carbonate-rich parent material of the Agula shale formation, which consists mainly of marble and limestone (Beyth, 1972). According to the World Reference Base

soil classification system (WRB, 2006), the soil types are the dominated by *Calcaric Cambisols*, *Vertic Leptosols*, *Vertic Cambisols* and *Lithic Leptosols* in the study area (detail description is given in section 1.2).

3.2.2. Characteristics of the community based soil and water conservation practices

The most commonly accepted and applied soil and water conservation measures (i.e., terraces and exclosures with and without terraces) were established since 1997 by the local community. Before their establishment, the selected SWC measures had similar history in terms of grazing with the non-conserved communal grazing lands (detail description is given in section 1.2) and Table 3.1.

Table 3.1. Characteristics of the community soil and water conservation measures (own assessment)

SWC measures	characteristics
Non-conserved communal grazing land (control)	Low vegetation cover High proportion of bare soil Coarse fragment (57%) Sheet, rill and gully erosion very common Relatively Lower (1.9%) organic carbon (Tekalign, 1991) Moisture content (8.6%) Bulk density (1290 kg m ⁻³) Area (11-34.96ha)
Stone terraces	Relatively more stable and durable than other physical SWC measures Soil organic carbon (2.5%) Moisture content (9.6%) Bulk density (1260 kg m ⁻³) Coarse fragment (53%) High sediment deposit Area (13.87ha-24.42ha)
Exclosure + terrace	Closed from the interference of humans and livestock Enrichment planting common Supported with terraces Medium (2.9%) organic carbon (Tekalign, 1991) Sheet, rill and gully formation are less common Moisture content (10.06%) Bulk density (1240 kg m ⁻³) Coarse fragment (46%) Area (12.74ha-51.80ha)

Exclosures alone	Closed from humans and livestock interference No enrichment planting No physical SWC measures Medium (2.8%) organic carbon (Tekalign, 1991) Moisture content (8.7%) Bulk density (1220 kg m ⁻³) Coarse fragment (51%) Sheet, rill and gully formation are less common Area (14.02ha-34.7ha)
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3.2.3. Sampling Technique and Sample Size

The study was conducted in the three nearby sites (Kerano, Tesemat and Alasa within the district) and having all the SWC measures. In each SWC measures, three transects were established separated at a minimum distance of 75 m. The transect lines were parallel to each other and to the topography of the landscape. In each transect three landscape positions (i.e., upper, middle and foot slope) were established. The upper slope (US) position is the uppermost portion of each study site and it can receive little or no overland flow but may contribute to runoff to down slope areas. The middle slope (MS) position receives overland flow from the upper slope and contributes runoff to the foot slope (FS). The FS represented the lowest part of each study site and received overland flow from both mid and upper slopes.

Three sampling plots or quadrats of 10 m*10 m size (50 m apart from each other) were delineated in each slope position. Soil and root samples were collected from the rhizosphere of matured woody plant species (i.e., *Acacia etbaica*, *Carissa edulis*, *Dodonea angustifolia*, *Stereospermum kunthianum*, *Rhus vulgaris*, *Senna singueana* and *Eucla racemosa*) commonly found in all soil and water conservation measures.

3.2.4. Isolation, Enumeration and Identification of AMF Spores

Spores were extracted from soil samples collected from each corner (i.e., 0-15 cm from the stem) of the same and matured woody plant species in each SWC measures. Spores were extracted from 25 g soils by wet sieving and decanting method in a nest using sieves with different mesh sizes (i.e., 300, 100 and 50 μm) and sucrose density centrifugation (Brundrett, 1996). The

residues on the sieves were washed into beaker with water and filtered through filter papers. Each filter paper was spread on petri dish and spores were counted using a dissection microscope at $40\times$ magnification. A sporocarp was counted as one unit.

3.2.4.1. Spore identification

The identification or characterization was based on morphological characters such as spore size, color, surface ornamentation, wall structure as well as presence and absence of subtending hyphae with reference to the descriptions provided by Schubler and Walker (2010).

3.2.5. Estimation of AMF Root Colonization

The rhizosphere of each matured woody plant species was excavated up to 30 cm depth around all corners (i.e., 0-15 cm distance from the stem of the plant) at the end of rainy season (during October). Only fine and live root samples were collected and put into plastic jar; and filled with 97% ethanol to preserve the roots until processing.

In the laboratory, the root samples were detached from earth in tap water, cut it to approximately 1 cm and cleared in 10 % KOH for 20 min at 120 °C, acidified with 3 % HCl and stained with trypan blue (Brundrett, 1996). The stained root samples were mounted on microscope slides in polyvinyl lactoglycerol (PVLG) and examined for AMF colonization under light microscope. Mycorrhizal colonization with arbuscules, vesicles and hyphae in 100 root segments from each plant species was estimated using the gridline intersection method of McGonigle (1990). Finally, the average AMF root colonization of woody plant species of each plot across slope position within each SWC measure were used to calculate AMF root colonization. Calculation was done as follows

Total intersection points (G) = (p + q+ r + s+ t+ u)

Total root colonization = [(G-p)/G]*100

Hyphal colonization (HC) = [(G-(q+r+s)-p)/G]*100 or (q+r+s+t)/G*100

Arbuscular Colonization (AC) = (q+s)/G*100

Vesicular colonization (VC) = $(r+s)/G*100$

Where;

p: Intersection with no fungal structures

q: Intersection with arbuscules

r: Intersection with mycorrhizal vesicles

s: Intersection with arbuscules and mycorrhizal vesicles

t: Intersection with mycorrhizal hyphae but no arbuscules or mycorrhizal vesicles

u: Intersection where mycorrhizal hyphae not seen to be connected to arbuscules or mycorrhizal vesicles.

3.2.6. Soil Sampling and Analysis

Soil samples were collected from the top (0-30cm) soil depth at a distance of 0 to 15cm from the stem of the woody plants. They were air dried, composited and collected in plastic bags

The soil samples were sieved through 2 mm mesh to remove stones, roots, and large organic residues. Soil texture was determined using a hydrometer method (Bouyoucos, 1962); soil pH by using a combined glass electrode pH meters in 1: 2.5 soil-water (Chopra and Kanwar, 1976); soil organic (OC) by wet combustion (Walkley and Black, 1934); total nitrogen was measured according to Kjeldahl method (Bremmer and mulvaney, 1982) and soil available P was determined by Olsen method (Olsen and Sommers, 1982).

3.3. Data Analysis

The total spore density $100g^{-1}$ soil, percent of root colonization and soil nutrients were tested using ANOVA and comparison of means was carried out using Duncan's multiple range tests. Pearson correlation coefficient was calculated to see for any correlation between the AMF spore density, root colonization and soil nutrients.

3.4. Results and Discussion

3.4.1. Effect of Community Soil and Water Conservation Measures on Arbuscular Mycorrhiza Fungi Spore Density

In this study, five genera of AMF that include *Glomus*, *Acaulospora*, *Gigaspora*, *Scutellospora* and *Entrophospora* were identified. Total spore density was significantly ($P < 0.01$) affected by the community based soil and water conservation measures (Table 3.2). *Glomus* was the most abundant (52%) genus and followed by *Acaulospora* (18%). This could be due to the resistance of these genera to disturbances. Arbuscular mycorrhiza fungi species that belong to the genera *Glomus* and *Acaulospora* are resistant to soil disturbances and altered ecosystems (Muleta *et al.*, 2007; Zangaro *et al.*, 2013; Maciel *et al.*, 2014). They are also not host specific and might be found to be associated with various plants in the same locality (Lee *et al.*, 2013).

Scutellospora accounted only for 13%, *Gigaspora* for 12.7% and *Entrophospora* about 4% of the total spore density (Table 3.2). *Glomus* was the most abundant due to the fact that it is also sporogenous, while *Entrophospora* was found to be the least abundant genus because it is sensitive to disturbances. A study conducted by Birhane *et al.* (2010) showed *Glomus* as abundant genus in *Boswellia papyrifera* dominated woodlands of northern Ethiopia). Likewise, Wubet *et al.* (2003) reported that *Glomus* was abundant in indigenous trees in dry Afromontane forests of Ethiopia.

Basically, AMF spore densities are affected by soil erosion and mechanical disturbance due to livestock grazing. As a result, the lowest spore density which accounted for 16.5% (Table 3.2) was found in non-conserved communal grazing lands. In connection, Muleta *et al.* (2008) found that soil disturbance has reduced AM fungi spore densities. Particularly, AM fungi *Entrophospora* is sensitive to human disturbance and soil erosion. Scarcity of *Entrophospora* in the rhizosphere of woody plant species is attributed to the disruption of the extra-radical hyphae (Muleta *et al.*, 2008).

Total spore density in the exclosures ranged from 1488.9-1518 spores 100g^{-1} of dry soil while stone terraces accounted for 1154 spores 100g^{-1} of dry soil. Both exclosures also had significantly ($P < 0.05$) higher spore density than free grazing lands without SWC structure. This may be due to the presence of favorable environment such as high organic carbon content in exclosures. Similarly, Birhane *et al.* (2017a) found up to 2980 spores 100g^{-1} dry soil in exclosures. Terraces had higher mean total spore density than free grazing land without any SWC structures (Table 3.2). Non-conserved open communal grazing lands had the lowest mean total AMF spore density (825 spores 100g^{-1} dry soils). In this study, it is witnessed that community based soil and water conservation practices caused significant change in spore density. Exclosures resulted in 45.6% increment in spore density as compared to the non-conserved communal grazing lands and 28.5% against the stone terraces. Besides, a stone terrace resulted in 24% increment in spore density as compared to the non-conserved communal grazing lands. Spore density of the different CBSWC measures was in the order of exclosures \geq terraces $>$ non-conserved communal grazing lands.

Table 3.2. Effect of SWC measures on total spore density 100g^{-1} soil (Mean \pm SEM)

Spore type	Non-conserved grazing lands	Terraces	Exclosure s+ terraces	Exclosures alone	p-value	CV (%)
<i>Glomus</i>	436.7 \pm 72 ^b	587.8 \pm 58 ^{ab}	754.4 \pm 86 ^a	815.5 \pm 97 ^a	0.01	37.67
<i>Acaulospora</i>	142.2 \pm 22 ^b	230.0 \pm 42 ^{ab}	276.7 \pm 33 ^a	257.8 \pm 29 ^a	0.03	43.86
<i>Gigaspora</i>	104.4 \pm 18 ^b	152.2 \pm 18 ^{ab}	181.1 \pm 14 ^a	196.7 \pm 30 ^a	0.02	40.81
<i>Scutellospora</i>	125.6 \pm 18 ^a	142.2 \pm 21 ^a	193.3 \pm 27 ^a	188.9 \pm 18 ^a	0.13	44.31
<i>Entrophospora</i>	16.6 \pm 8 ^b	42.2 \pm 9 ^{ab}	83.3 \pm 25 ^a	58.9 \pm 13 ^{ab}	0.03	91.22
Grand total	825.6 \pm 121 ^b	1154.0 \pm 103 ^{ab}	1489.0 \pm 112 ^a	1518.0 \pm 158 ^a	0.001	30.65

Means followed by the same letter across a column do not differ significantly at $P \leq 0.05$

3.4.2. Effect of Slope Position on Total Arbuscular Mycorrhiza Fungi Spore Density

Though no significant, foot slope had higher spore density than the middle and upper slope positions (Table 3.3). Relatively higher spore density (i.e., 1616.6 spores 100g^{-1} dry soil) was recorded at the foot slope in exclosures without terraces whereas relatively lower total spore density (i.e., 723 spores 100g^{-1} dry soil) was found at the upper slope position in non-conserved communal grazing lands (Table 3.3). This slight variation could be due to displacement of spores

by erosion from the upper to down slope position. Variations in spore density between slope positions could also be due to the properties of the soils, host relations, and the differential survival strategies of AMF (Birhane *et al.*, 2017a).

Table 3.3. Effect of slope position on total spore density

Slope position	Soil and water conservation measures				CV (%)
	Non-conserved grazing lands	Terraces	Exclosures + terraces	Exclosures alone	
Foot slope	903.3 ^{ab}	1210.0 ^{ab}	1603.3 ^a	1616.6 ^a	30.86
Middle slope	850.0 ^{ab}	1066.6 ^{ab}	1466.6 ^{ab}	1506.6 ^{ab}	44.34
Upper slope	723.3 ^b	1186.6 ^{ab}	1396.6 ^{ab}	1430.0 ^{ab}	37.3

Means followed by the same letter do not differ significantly at $P \leq 0.05$

3.4.3. Effect of Community-based Soil and Water Conservation Measures on Arbuscular Mycorrhiza Fungi Root Colonization

Root colonization by AMF is characterized by the presence of hyphae, arbuscules and vesicles. In this study, It was found that the exclosures showed a significant ($P < 0.05$) variation in the total AMF root colonization than non-conserved free grazing lands (Table 3.4). The total percent of AMF root colonization was the highest (68.7%) in the exclosures + terraces. The lowest (48.6%) total percent of AMF root colonization was recorded in the non-conserved communal grazing lands (Table 3.4). A similar study conducted by Aggarwal *et al.* (2011) reported that grazing by herbivores decreased AMF root colonization. Other studies also reported that disturbance generally had the greatest impact on biological properties, including symbiotic fungal populations (Borie *et al.*, 2006; Jha, 2014).

There was no significant difference in total percent of root length colonization between the two exclosure types (Table 3.4). However, exclosures had significantly ($P < 0.05$) higher total AMF root colonization than terraces. Besides, terraces had significantly ($P < 0.05$) higher total AMF root colonization than non-conserved communal grazing lands (Table 3.4). Total root colonization by AMF was in the order of exclosures > terraces > non-conserved grazing lands. The presence of high organic carbon and lower disturbance (Table 3.1) could be the reason for high AMF root

colonization in exclosures than terraces and non-conserved grazing lands.

Both exclosures resulted in about 29% increase in total root colonization as compared to non-conserved communal grazing lands and 12% increase as compared to terraces, while terraces resulted in 20% increase in total root colonization as compared to non-conserved communal grazing lands. Percentage of hyphal colonization was the highest in all conservation measures followed by percentage arbuscular colonization but percentage vesicular colonization was the lowest (Table 3.4). In many studies it was also found that root colonization is in the order hyphal > arbuscular > vesicular colonization (Muleta *et al.*, 2008; Birhane *et al.*, 2010; Birhane *et al.*, 2017a).

Table 3.4. Effect of SWC measures on root colonization % (Mean \pm SEM)

Root colonization types	Soil and water conservation measures				CV (%)
	Non-conserved communal grazing lands	Terraces	Exclosure s+ terraces	Exclosures alone	
Total	48.6 \pm 3 ^c	60.5 \pm 2 ^b	68.7 \pm 2 ^a	67.8 \pm 4 ^a	10.0
Hyphal	21.2 \pm 3 ^b	28.2 \pm 4 ^b	45.3 \pm 4 ^a	41.1 \pm 3 ^a	23.44
Arbuscular	7.1 \pm 2 ^b	7.6 \pm 2 ^b	18.5 \pm 1 ^a	15.9 \pm 3 ^a	35.28
Vesicular	4.3 \pm 1 ^b	8.9 \pm 3 ^a	12.4 \pm 1 ^a	12.9 \pm 2 ^a	41.56

Means followed by the same letter across each row do not differ significantly at $P \leq 0.05$

3.4.4. Effect of Community Soil and Water Conservation Measures on Soil Physicochemical Properties

Soil and water conservation had brought significant ($P < 0.05$) variation on some of the rhizosphere soil chemical properties. The rhizosphere soil chemical properties specifically available K and total nitrogen were higher in exclosures than non conserved free grazing lands (Table 3.5). Significant ($P < 0.05$) variation in soil organic carbon content was found between the CBSWC measures. Similarly, Demelash and Stahr *et al.* (2010) reported that soil and water conservation measures increased the soil organic carbon content. The highest organic carbon content was found in the exclosure + terraces followed by the exclosures alone (Table 3.1). The lowest was obtained in the non-conserved open communal grazing lands. Organic matter content

of the CBSWC measures was in the order of enclosure + terrace > enclosure > terrace > non-conserved grazing lands.

The pH content of all the soil and water conservation measures was moderately alkaline (Murphy, 1968). This may indicate that the soils are rich in calcium carbonate. Soil pH was significantly ($P < 0.05$) lower in enclosures than terraced and non-terraced open communal grazing lands (Table 3.5). This could be due to the presence of higher organic matter in enclosures (Table 3.5). Many studies showed that un-grazed lands had high SOC but low pH than grazed sites (Mekuria, 2007; WRB, 2006).

Available phosphorus in terraced and non-terraced open communal grazing land is rated as medium whereas it is classed as high in both enclosures (Murphy, 1968). Available potassium is rated as medium in the terraced and non-terraced open communal grazing lands where as it is rated as high in case of both enclosures (FAO, 2006).

The total nitrogen in terraced and non-terraced open grazing lands is classed as high, while that of both types of enclosures is rated as very high (Murphy, 1968). In conformity, a more significantly higher total soil nitrogen and potassium were found in the enclosure than in the grazed areas (Buyinza and Nabalegwa, 2000) and on terraced than non-terraced lands (Demelash and Stahr, 2010).

Table 3.5. Effect of SWC measures on rhizosphere soil physico-chemical properties

Physico-chemical properties	Soil and water conservation measures				p-value	CV (%)
	Non conserved grazing lands	Terrace	Exclosure + terrace	Exclosure alone		
pH	8.22 ^b	8.21 ^b	8.1 ^a	7.9 ^a	<0.001	2.35
Ec (cS/M)	0.13 ^a	0.12 ^a	0.12 ^a	0.11 ^a	0.2	39.15
TN (%)	0.21 ^c	0.24 ^{bc}	0.28 ^{ab}	0.31 ^a	<0.001	27.03
Avail. P (ppm)	4.13±1 ^a	4.60±1 ^a	5.70±1 ^a	6.99±1 ^a	0.06	69.52
OC (%)	1.9 ^c	2.5 ^b	2.9 ^a	2.8 ^a	<0.001	11.56
Avail. K cmol (+)/Kg	0.49 ^b	0.59 ^b	0.61 ^b	0.8 ^a	0.01	38.81
Sand (%)	36.48±3 ^a	31.43±2 ^b	28.67±2 ^b	32.57±2 ^{ab}	0.05	27.91
Silt (%)	29.81±1 ^a	31.33±1 ^a	29.81±1 ^a	32.38±2 ^a	0.37	18.25
Clay (%)	33.71±3 ^b	37.24±1 ^{ab}	40.29±2 ^a	35.14±2 ^b	0.14	26.1

Means followed by the same letter across each row do not differ significantly at $P \leq 0.05$

Avail. P: available phosphorus; Avail. K: available potassium; OC: organic carbon; TN: total nitrogen; Ec: electrical conductivity

3.4.5. Relationship between AMF Spore Density, Root Colonization and Soil Properties

Correlation of arbuscular mycorrhizal fungal spore density and percent of total root colonization were positive and significant ($P < 0.05$). Spore density and total root colonization (TRC) were positively related with pH, total nitrogen, organic carbon and phosphorus (Table 3.6). Unlike silt and clay fraction, a positive correlation between spore density and total root colonization (TRC); and sand fraction was observed. This could be due to the fact that sand fractions is known to have many macro pores; and this revealed the need of macro pore spaces for increased AMF density. Similarly, Belay *et al.* (2013) found a positive correlation between spore density and sand fraction under acacia species in central highlands of Ethiopia. Other results reported positive relation between spore density and root colonization (Chaudhry *et al.*, 2013; Birhane *et al.*, 2017a). In contrast, a study by Alghamdi and Jais, (2013) in Saudi Arabia showed a negative correlation of AMF with sand proportions but positive and strong correlation of AMF with silt and clay under *Juniperus procera*.

A positive correlation between AMF spore density and available phosphorus was observed. In line with this, Ong *et al.* (2012) found a positive correlation between spore count and soil available P due to the fact that the P concentration in the soil was low, thus allowing the

enhancement of mycorrhizal sporulation. This finding is also in agreement with that of Muleta *et al.* (2008) but in contrast to that of Dobo *et al.* (2016) and Birhane *et al.* (2017a) who found negative correlation between AMF and available phosphorus.

Table 3.6. Pearson correlations between rhizosphere soil properties and AMF spore densities and root colonization

Soil properties	pH	EC	TN	P	K	OC	Sand	Silt	Clay	Total spore	TRC	HC	AC	VC
pH	1													
EC	0.31**	1												
TN	-0.04 ^{ns}	0.40***	1											
Avail. P	-0.05 ^{ns}	0.14 ^{ns}	0.40***	1										
Avail. K	-0.34**	-0.13 ^{ns}	0.46***	0.48***	1									
OC	-0.01 ^{ns}	-0.21*	0.16 ^{ns}	0.04 ^{ns}	0.26*	1								
Sand	-0.15 ^{ns}	0.01 ^{ns}	0.19 ^{ns}	0.07 ^{ns}	0.05 ^{ns}	-0.07 ^{ns}	1							
Silt	0.05 ^{ns}	-0.14 ^{ns}	-0.22*	0.10 ^{ns}	0.04 ^{ns}	-0.06 ^{ns}	-0.26*	1						
Clay	0.10**	0.05 ^{ns}	-0.05 ^{ns}	-0.12 ^{ns}	-0.07 ^{ns}	0.12 ^{ns}	-0.83***	-0.31 ^{ns}	1					
Total spore	0.05 ^{ns}	0.16 ^{ns}	0.46***	0.25*	0.33***	0.21*	0.21*	-0.11 ^{ns}	-0.15 ^{ns}	1				
TRC	0.26*	0.25*	0.23*	0.16 ^{ns}	0.05 ^{ns}	0.38*	-0.09 ^{ns}	0.22*	-0.03 ^{ns}	0.36*	1			
HC	0.21 ^{ns}	0.25*	0.28**	0.12 ^{ns}	0.06 ^{ns}	0.30**	-0.10 ^{ns}	0.13 ^{ns}	0.03 ^{ns}	0.30*	0.86*	1		
AC	0.09 ^{ns}	0.12 ^{ns}	0.21*	0.17 ^{ns}	0.10 ^{ns}	0.11 ^{ns}	-0.01 ^{ns}	0.25*	-0.14 ^{ns}	0.20*	0.51*	0.69 ^{ns}	1	
VC	0.29**	0.18 ^{ns}	0.08 ^{ns}	-0.03 ^{ns}	-0.01 ^{ns}	0.27*	-0.16 ^{ns}	0.09 ^{ns}	0.11 ^{ns}	0.01 ^{ns}	0.67*	0.81 ^{ns}	0.45 ⁿ	1

* Correlation is significant at 0.05 levels (pair wise). ** Correlation is significant at 0.01 levels (pair wise). *** Correlation is significant at 0.001 levels. ^s Correlation is not significant at P < 0.05

3.5. Conclusion

Exclosure management and supporting open grazing lands with terrace enhanced AMF spore density, root colonization and soil nutrients. Therefore, community based soil and water conservation practices are dependable rehabilitation or restoration approaches, which help to boost the arbuscular mycorrhizal fungi and soil nutrients in degraded lands. This implies soil and water conservation measures have a great practical significance to the restoration of the degraded grazing lands through improving arbuscular mycorrhiza spore density and root colonization. Therefore, Re-vegetation strategies in the area should involve use of arbuscular mycorrhiza fungi technologies.

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**EFFECT OF COMMUNITY-BASED SOIL AND WATER
CONSERVATION PRACTICES ON SOIL GLOMALIN,
AGGREGATE STABILITY AND AGGREGATE ASSOCIATED
ORGANIC CARBON CONTENT IN NORTHERN HIGHLANDS OF
ETHIOPIA**

CHAPTER 4. EFFECT OF COMMUNITY-BASED SOIL AND WATER CONSERVATION PRACTICES ON SOIL GLOMALIN, AGGREGATE SIZE DISTRIBUTION, AGGREGATE STABILITY AND AGGREGATE ASSOCIATED ORGANIC CARBON CONTENT IN NORTHERN HIGHLANDS OF ETHIOPIA

Abstract

Land degradation is a major and widespread problem causing losses of ecosystem services in Ethiopia. Extensive utilization of the land resources for centuries resulted in severe land degradation in the Tigray region of northern Ethiopia. To reverse the problem, land restoration activities using soil and water conservation measures have been carried out for decades. This study was initiated to determine the effect of community based soil and water conservation (CBSWC) interventions on soil glomalin, aggregate size distribution (ASD), water stable aggregates (WSA) and aggregate associated organic carbon. Soil samples were collected from exclosures + terraces, exclosures alone, stone terraces and non-conserved grazing lands using systematic sampling based on slope positions. Both easily extractable glomalin (EEG) and total glomalin (TG) were significantly ($P < 0.05$) higher in exclosures compared to terraces and non-conserved grazing lands. The macro-aggregate fraction of all SWC measures ranged 21.91-32.41%, where the lowest was in non-conserved grazing lands, while the highest was in exclosures with terraces. The micro-aggregate fraction ranged 19.9-26.66%, where the lowest was in exclosures, while the highest was in non-conserved grazing lands. The results also indicated that exclosures had significantly ($P < 0.05$) higher percent of WSA compared to terraces and non-conserved grazing lands. Mean weight diameter (MWD) were also significantly ($P < 0.05$) higher in exclosures. The WSA and MWD decreased in the order of exclosures with terraces > exclosures alone > terraces > non-conserved communal grazing lands. Comparison of aggregate associated organic carbon (AAOC) showed relatively higher organic carbon in macro-aggregates than micro-aggregates. Besides, both macro and micro-aggregate associated SOC were higher in exclosures than in terraces and non-conserved grazing lands. The AAOC of both macro and micro-aggregates follows the order exclosures alone > exclosures + terraces > terraces > non-conserved communal grazing lands. Exclosures and terraces are important strategies for rehabilitation of degraded lands through improving glomalin content, aggregate structure and stability, and aggregate associated organic carbon.

Keywords: Mean weight diameter, water stable aggregate, stability index, exclosure, terrace

4.1. Introduction

Land degradation is a major cause of food insecurity in Ethiopia (Shiferaw *et al.*, 2013; WFP, 2013). Due to increasing human and livestock population pressure, large areas of the country, particularly in the northern highlands, have been exposed to land degradation (WFP, 2013). Tigray, the northern part of Ethiopia suffered from extreme land degradation (Mekuria *et al.*, 2009). The rural landscapes of the region severely suffered from a high degree of soil degradation (Gelaw *et al.*, 2014).

Soil and water conservation practices mainly exclosures and stone terraces have been implemented to reverse the land degradation process (Nyssen *et al.*, 2004). Exclosures involve excluding livestock on biophysically degraded communal grazing lands (Descheemaeker *et al.*, 2006) by inhibiting uncontrolled cutting of trees and grass for fuel and fodder (Yayneshet *et al.*, 2009). They are effective in regenerating natural vegetation and controlling soil erosion (Baudron *et al.*, 2015; Gebremeskel *et al.*, 2017).

Stone terraces are another important physical SWC measures in the area to decrease soil erosion and are commonly constructed on steep sloping areas (Gebre-egziabher *et al.*, 2009) with the participation of farmers. Stone terraces reduce nutrient and sediment transport (Sawadogo, 2017). Numerous soil properties including soil organic matter and soil aggregation can be improved through application of stone terraces (Rashid *et al.*, 2016).

Land degradation is one of the major factors affecting soil aggregation. Land management practices including physical and biological conservation measures improve soil stability to erosion and land degradation (Njira and Nabwami, 2013). Exclosures and construction of terraces enhanced the natural vegetation (Gebrehiwot and Veen, 2014; Yimer *et al.*, 2015) through improving physical and chemical properties of soils (Carballas *et al.*, 2004).

Change in aggregate stability is an indicator of organic matter content, biological activity and nutrient cycling in soils (Carballas *et al.*, 2004) which are essential for the functioning of ecosystems (Gelaw *et al.*, 2014).

Aggregate stability is one of the soil physical properties considered during restoration of degraded lands (Boogar *et al.*, 2014). It is a measure of the ability of the soil to resist change due to environmental factors (Bird *et al.*, 2002). Soil organic carbon (SOC) and aggregate stability enhance vegetation growth (An *et al.*, 2013). The stability of soil aggregates influences the water holding capacity of soil and tells the susceptibility of the soil to erosion (An *et al.*, 2013; Hobley *et al.*, 2014; Belaid and Habaieb, 2015).

Management practices are determinants in soil aggregation through glomalin (Borie *et al.*, 2006). Glomalin is a recalcitrant glycoprotein produced by arbuscular mycorrhizal fungi (Rillig, 2004; Wang, 2018). It protects hyphae during transport of nutrients from the plant to the hyphal tip and from soil to the plant (Singh, 2012).

Glomalin forms soil aggregates, improves soil structure and stability against erosion (Wright and Upadhyaya, 1996; Fokom *et al.*, 2013). It is source of active soil organic carbon (Rillig *et al.*, 2003) and contains 30 to 40 percent carbon (Bedini *et al.*, 2007). The SOC associated with various aggregate size fractions reduces the impact of erosive forces (Gelaw *et al.*, 2014) and tells the dynamics of soil organic matter (Singh *et al.*, 2017).

Measurement of glomalin, organic carbon content and aggregate stability enable to assess the risk of soil structural degradation and function (Burri *et al.*, 2009; Boogar *et al.*, 2014). Good understanding of glomalin, ASD and WSA guide the management of soils against erosive and degradative factors (Igwe and Obalum, 2013). Previous studies focused on the effect of land degradation on soil fertility and productivity (Rillig *et al.*, 2003; Gebremeskel *et al.*, 2017). However, studies conducted on the effect of SWC measures on soil glomalin, ASD, WSA and AAOC were limited in the area. Thus, this research was conducted to determine the effect of two decades long CBSWC practices mainly stone terraces and exclosures on soil ASD, glomalin, WSA and organic carbon associated with soil aggregates.

The research questions answered include: did the construction of stone terraces improve ASD glomalin, WSA, and aggregate associated SOC as compared to non- terraced grazing lands?

Could enclosure enhance ASD glomalin, WSA, and AAOC as compared to terraces and non-conserved grazing lands? and finally could enclosures supported with terraces significantly increase soil ASD glomalin, WSA and AAOC as compared terraces, non-terraced enclosures, and non- conserved grazing lands?

4.2. Materials and Methods

4.2.1. Description of the Study Area

The study was conducted in Degua Temben district, located 50 km west of Mekelle, regional capital of Tigray region, northern Ethiopia. Geographically, it is located at 13⁰16'23''N to 13⁰47'44''N Latitude and 39⁰3'17''E to 39⁰24'48''E Longitude (Figure 1.1).

The area receives 290 to 900mm rainfall annually with an average value of 615 mm year⁻¹. The rainy season usually occurs between June and September. The highest rainfall is in July and August. The growing season varies between 90 and 120 days. The maximum temperatures occur in May and June (detail description is given in section 1.2)

4.2.2. Sampling Technique and Sample Size

The study was conducted in three nearby sites (Kerano, Tesemat and Alasa with in the district) and having all the SWC measures. In each SWC measures, three transects separated at a minimum distance of 75 m were established. Transects were parallel to each other and to the topography of the landscape. In each transect three landscape positions (i.e., upper, middle and foot slope) were established. The upper slope (US) position was the uppermost portion of each study site and received little or no overland flow but contributed runoff to down slope areas. The middle slope (MS) position received overland flow from the upper slope and contributes runoff to the foot slope (FS). The FS represented the lowest part of each study site and receives overland flow from both mid and upper slopes (Mekuria and Aynekulu, 2013).

Soil samples were collected from 0-30 cm depth at four corners and center of a 10 m x 10 m size plot using “X” sampling design from terraces, enclosure + terrace, enclosure without terraces and from non-conserved communal grazing lands. A total of 108 soil samples (i.e., four conservation measures*three slope positions*three samples*three replications) were collected for glomalin, ASD, WSA and AAOC determination.

4.2.3. Extraction and Determination of Glomalin Related Soil Proteins

The method described by Wright and Upadhyaya (1998) was used to determine the easily extractable and total glomalin related soil proteins. A one-gram sample of air-dried soil was placed in 8 mL 20mM citrate, pH 7.0 and autoclaved (121 °C) for 30 min to remove the easily-extractable glomalin (EEG). After centrifugation (10,000 x g) and removal of the supernatant, 8 mL 50 mM citrate, pH 8.0 was added to the remaining soil and heated at 121 °C for 60 min to extract total glomalin (TG). Extractions continued with 50 mM citrate until the supernatant became straw in color, indicating that glomalin, a red-brown color, had been removed. One mL of EEG was removed and then the remaining supernatant containing EEG was combined with all of the supernatants from the 50 mM citrate extractions. Bradford dye-binding assay was used to determine protein with bovine serum albumin as the standard (Wright and Upadhyaya, 1996).

4.2.4. Aggregate Stability Determination

The method described by Kemper and Rosenau (1986) was used to determine water stability of air-dried aggregates. 100 gram of air-dried bulk soil that passed through 8mm sieve was sieved using 5, 2, 1, 0.5, 0.25 and 0.053 mm sieves. Each fraction was pre-wetted overnight by capillary action then transferred on the top of a nest of the same sieves immersed in water. The nest of sieves was then vertically tumbled in a column of water for 5 min, at a rate of 50 complete repetitions per minute. The mass of oven-dried particles (105 °C for 24 h) that resisted break down was assessed for each sieve. The respective dry masses were used to compute the mean weight diameter (MWD) and water stable aggregate (WSA) as follows:

$$\text{WSA} = ((M(a + s) - M_s)/(M_t - M_s)) * 100 \quad [\text{Eq 4.1}]$$

Where M (a+s) is the mass of resistant aggregates plus sand (g),

M_s is the mass of sand

M_t is total mass of soil

$$\text{MWD} = \sum X_i W_i \quad [\text{Eq 4.2}]$$

Where;

MWD is the mean weight diameter of water stable aggregates,

X_i is the mean diameter of each sieve fraction (mm), and

W_i is the proportion of the total sample mass in the corresponding size fraction

$$\text{Stability index (SI)} = \frac{1}{(\text{MWD})_d - (\text{MWD})_w} \quad [\text{Eq 4.3}]$$

$$\text{SQ} = \text{SI} * \% \text{ aggregates} > 2 \text{ mm} \quad [\text{Eq 4.4}]$$

Where

SQ= Stability quotient

4.2.5. Determination of Aggregate Associated Organic Carbon

Soil organic carbon content was determined by Walkley and Black (1934) after sieving with 0.25 mm sieve for micro aggregates and 2mm sieve for macro aggregates.

4.3. Data Analysis

Analysis of variance (ANOVA) was used to see for any significant differences of the parameters along the different community soil and water conservation measures using SAS 9.2. Mean comparison were carried out using Duncan's multiple range test (DMRT) and finally correlation analysis was carried out to see for any correlation between ASD, WSA, glomalin related proteins and soil organic carbon associated with macro and micro aggregate fractions.

4.4. Results and Discussion

4.4.1. Effect of Community-based Soil and Water Conservation Measures on Soil Glomalin Content

The content of easily extractable glomalin (EEG) was significantly ($P < 0.05$) higher in exclosures than in non-conserved grazing lands. Easily extractable glomalin in Exclosures is 44.65-55.17% higher than that of non-conserved grazing lands. Generally, EEG was in the order of exclosures > stone terraces > non-conserved grazing lands (Table 4.1). The presence of higher glomalin in exclosures than in non-conserved grazing lands could be due to presence high AMF root colonization. It was reported that glomalin stocks are greater where AMF are more abundant (Treseder and Turner, 2007).

Total glomalin was also significantly ($P < 0.05$) higher in exclosures compared to non-conserved grazing lands. Total glomalin in exclosures was 37-45% higher than in non-conserved grazing lands and 24.55-35.51% higher than that in terraces. Livestock grazing could be the cause for the low content of EEG and TG in non-conserved grazing lands. It was reported that grazing and trampling had negative effect on the amount of glomalin through decreasing vegetation cover (Wright *et al.*, 1999; Bai *et al.*, 2011). Furthermore, as glomalin is produced by AM fungi, trampling may have destroyed the aggregate of the soil and break the hyphae. It was also reported that other factors such as landscape characteristics can affect the amount of glomalin (Sirinikorn *et al.*, 2010).

Table 4.1. Soil glomalin under the different SWC measures (mg g^{-1} soil)

SWC measures	Easily extractable glomalin	Total glomalin
Non-conserved grazing lands	3.21±1.1 ^b	9.84±1.74 ^c
Stone terraces	5.16±1.1 ^{ab}	11.71±1.56 ^{bc}
Exclosures +terraces	7.16±0.93 ^a	15.52±1.9 ^{ab}
Exclosures alone	5.8±1.02 ^a	18.16±2.05 ^a
p-value	0.07	0.05
CV (%)	58.08	41.55

Means followed by the same letter within a column do not differ significantly at $P \leq 0.05$

4.4.2. Effect of Soil and Water Conservation Practices on Dry Aggregate Size Distribution

The result of aggregate size distribution indicated that large macro-aggregates (> 2 mm) were higher than other aggregate sizes in all SWC measures (Table 4.2). This is similar to the result of Xiao *et al.* (2017) who found large macro-aggregates (> 2mm) represented the greatest fraction for all the land uses considered. The effect of different SWC measures on dry aggregate size distribution (Table 4.2) indicated that exclosures + terraces had significantly higher percentage (32.41%) in the > 2 mm fraction. This could be due to the presence of high organic carbon and low disturbances in exclosures. High organic matter from litter-falls and decayed tree roots increased coarser aggregates (Gajic *et al.*, 2006). Besides, it was reported that strong plant root systems are beneficial for soil aggregation (Zhao *et al.*, 2014). This also agreed with the result of Singh *et al.* (2017) who found higher macro-aggregates in vegetated area than bare land.

Non-conserved communal grazing lands had significantly lower (21.91%) in the > 2 mm fraction. Destruction of macro-aggregates by livestock trampling and soil erosion may be the cause for the decrease of large macro-aggregates in non-conserved communal grazing lands. Furthermore, clearing of the natural vegetation could be the cause for the decrease of macro-aggregates due to dispersion and detachment as vegetation cover protects the soil from aggregate break down.

Though not significant, terraces had relatively lower percentage (18.15%) of aggregates in 1-2 mm fraction than the other SWC measures. The decrease could be due to the deposition of fine sediments on terraces. However, exclosures + terraces had significantly higher (22.49%) magnitude of this fraction which could be due to the presence of high organic carbon. Soil aggregation has positive and strong relation with soil organic carbon (Dorji *et al.*, 2015).

Next to the >2 mm size fraction, the highest percentage (26.66%) of dry aggregates was found in the <0.25 mm fraction (micro-aggregates). Non-conserved grazing lands had the highest percent of aggregate in the < 0.25 mm size than exclosures. This could be due to dispersion of

aggregates by disturbance. This could indicate that soil particles are more likely to be detached in non-conserved grazing lands. Corroborating this result, Singh *et al.* (2017) found higher micro-aggregates in bare land. The decreasing order of percentage of micro-aggregates (< 0.25 fraction) was: non-conserved communal grazing lands and stone terraces > exclosures alone > exclosure + terraces.

Table 4.2. Percentage of dry aggregate size distribution under the different SWC measures (Mean \pm SEM)

SWC measures	> 2 mm	1-2 mm	0.5-1 mm	0.25-0.5 mm	< 0.25 mm
Non-conserved grazing lands	21.91 \pm 2.31 ^b	19.60 \pm 1.13 ^b	16.79 \pm 0.78 ^a	14.41 \pm 0.42 ^a	26.66 \pm 3.15 ^a
Terraces	26.33 \pm 1.47 ^{ab}	18.15 \pm 0.74 ^b	16.46 \pm 0.72 ^a	12.30 \pm 0.46 ^b	25.84 \pm 1.32 ^a
Exclosures + terraces	32.41 \pm 3.11 ^a	22.49 \pm 0.73 ^a	17.72 \pm 0.65 ^a	12.61 \pm 0.39 ^{ab}	13.90 \pm 2.04 ^c
Exclosures alone	31.36 \pm 2.28 ^a	20.56 \pm 0.62 ^{ab}	16.59 \pm 0.45 ^a	11.09 \pm 0.66 ^{bc}	19.40 \pm 2.02 ^b
p-value	0.01	0.02	0.53	0.01	0.03
CV (%)	25.39	12.03	11.82	12.85	27.9

Means followed by the same letter with in a column do not differ significantly at $P \leq 0.05$

4.4.3. Effect of CBSWC Measures on Water Stable Aggregates

Wet aggregate stability values varied among the CBSWC measures (Table 4.3). It was significantly higher ($P < 0.05$) in exclosures. Presence of higher soil macroinvertebrates density and mycorrhizal association in exclosures could have improved soil aggregation in exclosures (Welemariam *et al.*, 2018a,b). Especially termites and earthworms are known to enrich the soil with organic materials and improve soil structural stability (Rillig, 2004). Mycorrhizal fungi entangle particles within the hyphae network and cement particles together (Bedini *et al.*, 2007). Besides, the presences of high vegetation cover in exclosures increased supply of organic matter inputs and decreased soil erosion. In line with this, Fokom *et al.* (2013) observed a decrease in WSA as forest is converted to other land uses. This supports the idea that greater stability is associated with organic matter supply (Gajic *et al.*, 2006) because organic matter improves establishment of soil structure through binding and limits soil erosion (Carballas *et al.*, 2004; An *et al.*, 2013). Vegetation has also mulching effect to improve soil aggregation (Dorji *et al.*, 2015; Demenois *et al.*, 2017).

Communal grazing lands had significantly ($P < 0.05$) lower percent of water stable aggregates. This could be due to physical disturbance and low soil organic carbon. Similarly, it was reported that grazing decreased aggregate stability (An *et al.*, 2013) through dispersion of soil aggregates (Carballas *et al.*, 2004).

Conversion of communal grazing lands to exclosures resulted in 20-21% increase in percent of WSA and conserving the open communal grazing lands with terrace resulted in an increase of WSA by 12 percent. The result of stability quotient also indicated that soils in exclosures are 1.23-1.6 times more stable than non-conserved grazing lands. Terraces are also 1.27 times more stable than non-conserved grazing lands (Table 4.3). This may indicate that exclosed areas showed relative recovery from structural degradation. It could also indicate less soil erodibility in exclosures. Vegetations in exclosures protect the soil from structural disturbance (Demenois *et al.*, 2017). However, grazing lands due to their bare surface, they receive few inputs of organic matter and are susceptible to degradation. Grazing breaks the soil apart, exposing the organic matter to microbial decomposition and facilitates soil loss by erosion. It was found that due to trampling effect, bulk density increased in free grazing lands, while aggregate stability decreased (Gurmessa *et al.*, 2015).

Mean weight diameter of aggregates is one measure of aggregate stability (Igwe and Obalum, 2013). Exclosures had significantly ($P < 0.05$) higher MWD than the non-conserved communal grazing lands. This revealed that higher organic matter in exclosures stabilized the soil through aggregation. High values of MWD could indicate lower erodability of soils in exclosures. It would also imply stable aggregates are critical to erosion resistance. It was reported that establishment of exclosures on degraded lands restores aggregate stability (Delelegn *et al.*, 2017). This is because; vegetation cover improves soil structure through improving MWD (Carballas *et al.*, 2004) as MWD is highly responsive to cover (Stavi *et al.*, 2011). It was also reported that organic matter increases both WSA% and MWD through binding (Karami *et al.*, 2012).

Bareness and disturbance by grazing causes for dispersion of soil aggregates (Six and Paustian, 2014) and this might be the cause for the decrease in MWD in non-conserved grazing lands. Similar results reported that the main mechanism of aggregate breakdown is by dispersion through disturbance (Gurmessa *et al.*, 2015). Mean weight diameter was responsive to livestock grazing. A similar result reported that MWDs were lower in grazed grassland area (An *et al.*, 2013).

Table 4.3. Effect of SWC measures on aggregate stability (Mean \pm SEM)

SWC measures	WSA (%)	MWD	SI	SQ
Non-conserved grazing lands	51.8 \pm 3.9 ^b	0.77 \pm 0.04 ^b	1.65 \pm 0.14 ^a	40.73 \pm 3.06 ^a
Terraces	58.5 \pm 2.0 ^{ab}	0.82 \pm 0.04 ^{ab}	1.91 \pm 0.6 ^a	50.24 \pm 3.44 ^{ab}
Exclosure + terraces	65.6 \pm 2.4 ^a	0.93 \pm 0.05 ^a	2.60 \pm 0.57 ^a	64.97 \pm 12.46 ^b
Exclosures alone	64.8 \pm 3.2 ^a	0.97 \pm 0.06 ^a	2.45 \pm 0.44 ^a	52.05 \pm 5.43 ^b
p-value	0.01	0.03	0.46	41.40
CV (%)	14.83	18.04	66.53	0.14

Means followed by the same letter along each column do not differ significantly at $P \leq 0.05$

4.4.4. Effect of SWC Measures on Aggregate Associated Soil Organic Carbon

Exclosures significantly ($P < 0.05$) increased the soil organic carbon associated with macro (> 0.25 mm size) aggregates than non-conserved grazing lands and terraces. However, no significant variation was observed on those associated with micro (< 0.25 mm) aggregates in all soil and water conservation measures (Table 4.4).

Exclosures + terraces had significantly ($P < 0.05$) higher (3.11%) organic carbon associated to macro-aggregates, while non-conserved communal grazing lands had lower (2.26%) macro-aggregate associated carbon. Shrestha *et al.* (2007) found higher amounts of associated SOC concentration under undisturbed sites. Xiao *et al.* (2017) reported the highest SOC in large aggregates under exclosures. In our study, the conversion of communal grazing lands to exclosures resulted in 17-27% increase of SOC in macro-aggregates followed by terrace (9%) as compared to non-conserved grazing lands.

Conversion of communal grazing lands to exclosures resulted in 19-34% increase in SOC in macro-aggregates, while terrace construction resulted in 17% increase in macro-aggregate carbon as compared to non-conserved grazing lands. The low aggregate carbon in non-conserved communal grazing lands could be due to low biomass input caused by livestock grazing and human disturbances.

Soil organic carbon associated with macro-aggregates was relatively higher than SOC associated with micro-aggregates. This indicates that macro-aggregate structures are important in physical protection of soil organic carbon. In line with this, Gelaw *et al.* (2014) in Mandae watershed, northern Ethiopia found higher SOC in macro than micro-aggregates. Many other authors also reported that organic carbon in coarse aggregates was higher than in fine aggregates (Cleide *et al.*, 2011; Boogar *et al.*, 2014; Xiao *et al.*, 2017). This was due to high rate of decomposition in micro-aggregates (John *et al.*, 2005). This suggests that micro-aggregate stability could be a better indicator of potential soil erosion hazards (Singh *et al.*, 2017).

Table 4.4. Aggregate associated soil organic carbon under the different SWC measures (Mean \pm SEM)

SWC measures	SOC in macro-aggregates (%)	SOC in micro-aggregates (%)
Non-conserved grazing lands	2.2 \pm 0.2 ^b	1.8 \pm 0.2 ^b
Terraces	2.5 \pm 0.2 ^b	2.2 \pm 0.2 ^b
Exclosures + terraces	3.1 \pm 0.2 ^a	2.8 \pm 0.2 ^a
Exclosures alone	2.7 \pm 0.1 ^{ab}	2.3 \pm 0.1 ^{ab}
p-value	0.01	0.01
CV (%)	20.04	24.06

Means followed by the same letter along each column do not differ significantly at $P \leq 0.05$

4.4.5. Relationship between Soil Glomalin, Organic Carbon Fractions, Mean Weight Diameter and Aggregate Stability

This study found positive and significant ($P < 0.05$) relationship between easily extractable glomalin and water stable aggregates. Easily extractable glomalin and total glomalin explained about 43% and 40% of the variation in percent of water stable aggregates

respectively (Figure 4.1). The relationship between TG and % WSA was also positive and significant. In line with this, a study by Sirinikorn *et al.* (2010) and Hontoria *et al.* (2009) found positive relationship between soil aggregate stability and glomalin content. Similarly, Wright *et al.* (2007) reported that glomalin related soil protein increased as aggregate size increased. This indicates that both EEG and TG are important for soil aggregation and enhance stability. This could be due to the cementing and recalcitrant properties of glomalin. It was reported that glycoprotein, produced by AMF hyphae has a cementing capacity to maintain soil particles together (Singh, 2012). It also has relatively slow turnover in soil, contributing to lasting effects on aggregation (Rillig, 2004).

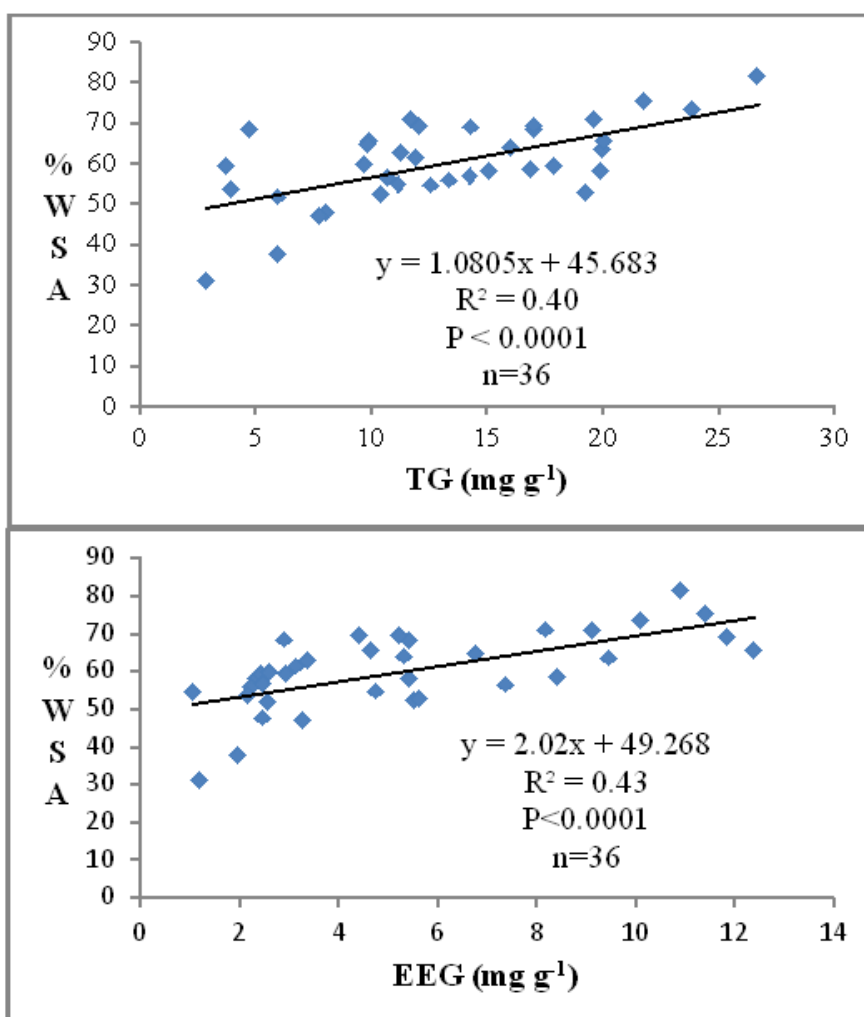


Figure 4.1. Relationship of % water stable aggregate (WSA) with easily extractable (EEG) and total glomalin (TG)

The relationship between EEG and % SOC; and TG with SOC was positive and significant. Easily extractable and TG explained 23 and 21% of the variation in % SOC respectively (Figure 4.2). These positive and significant relationships indicated that EEG and TG contribute to SOC storage. Zhang *et al.* (2017) reported glomalin related soil proteins are used for preserving and accumulating SOC. Glomalin contains approximately 30- 40% of carbon and forms small soil clumps. This granulated material binds carbon in the soil (Shrestha *et al.*, 2007).

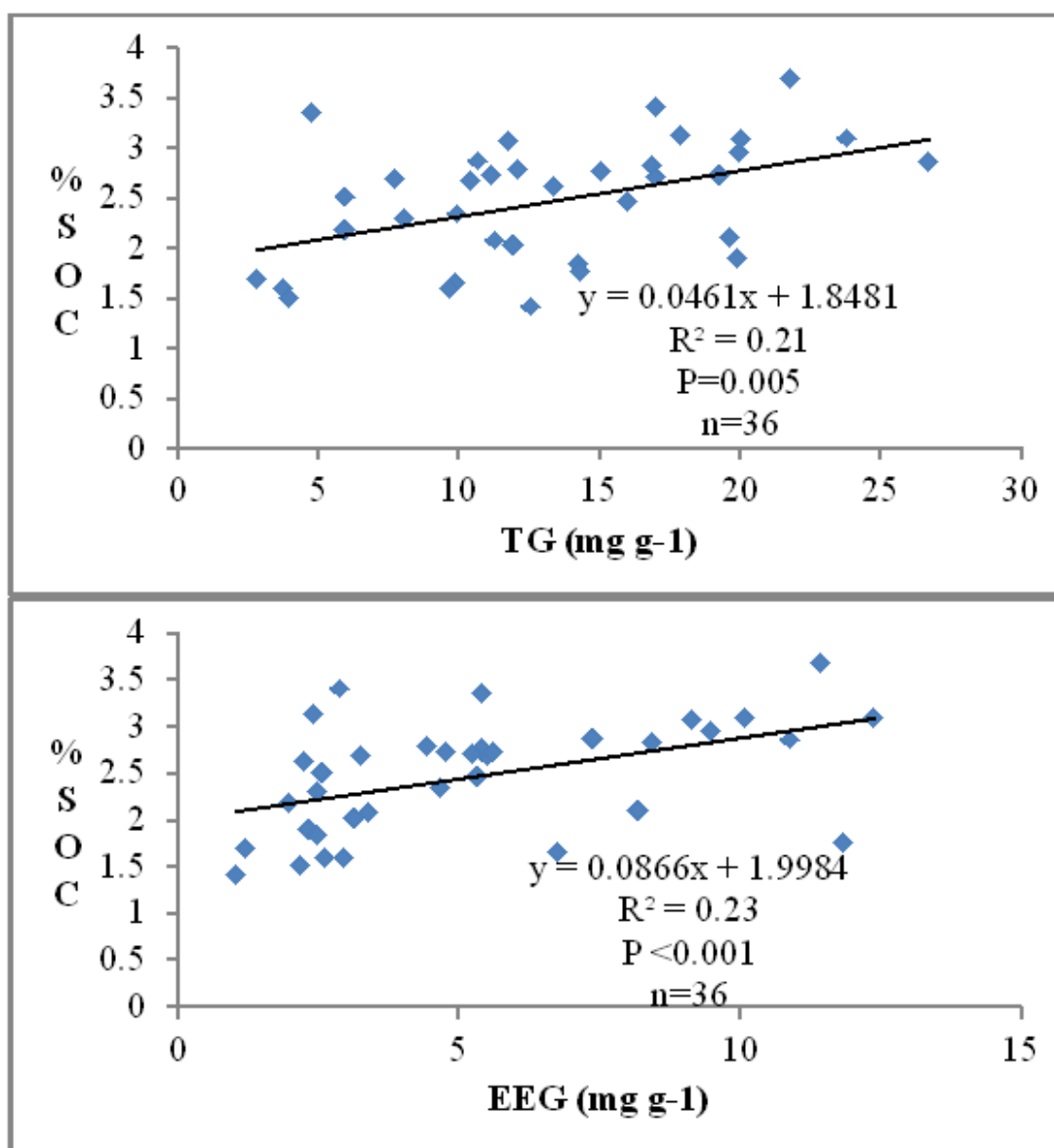


Figure 4.2. Relationship of SOC with with easily extractable (EEG) and total glomalin (TG)

The positive relationship between aggregate associated soil organic carbon and water stable aggregates (Figure 4.3 and 4.4) indicates that SOC may be necessary in soil aggregation. This agrees with the result of Dorji *et al.* (2015) in a montane ecosystem of Bhutan and Gelaw *et al.* (2014) in Mandae Watershed of the northern highlands of Ethiopia. This could be due to the cementing and recalcitrant properties of glomalin which significantly enhance the stability of soil aggregates through SOC sequestration (Rillig, 2004; Wright *et al.*, 2007; Perez, 2012) and slowing its turnover (Bai *et al.*, 2011). It was reported that glomalin has glycoprotein nature and hydrophobic characteristics; therefore it is a very stable biomolecule, with a half-life in soil between 6-42 years and prevents nutrients losses (Wu *et al.*, 2013).

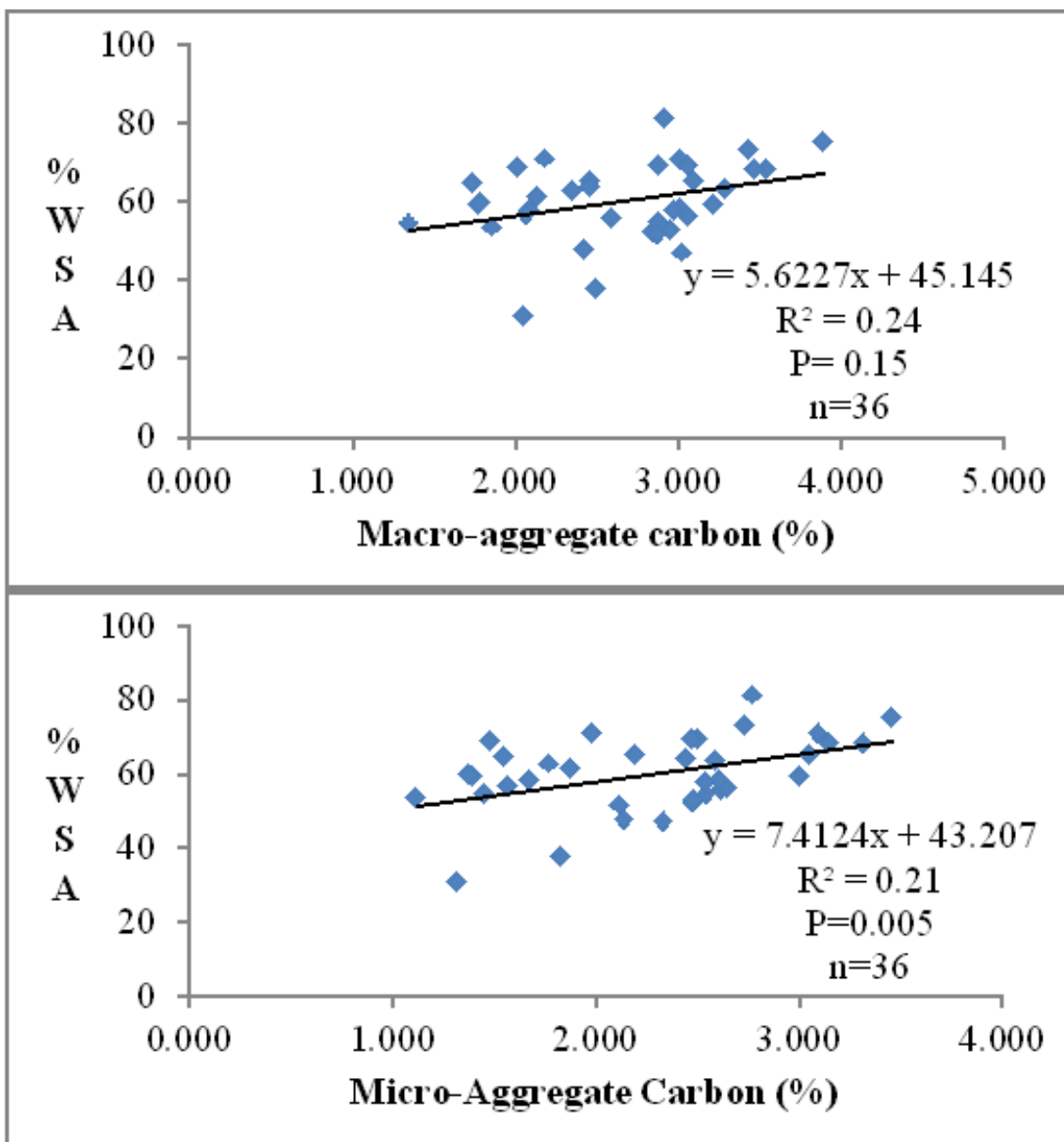


Figure 4.3. Correlation of % water stable aggregates with micro and macro-aggregate carbon fractions

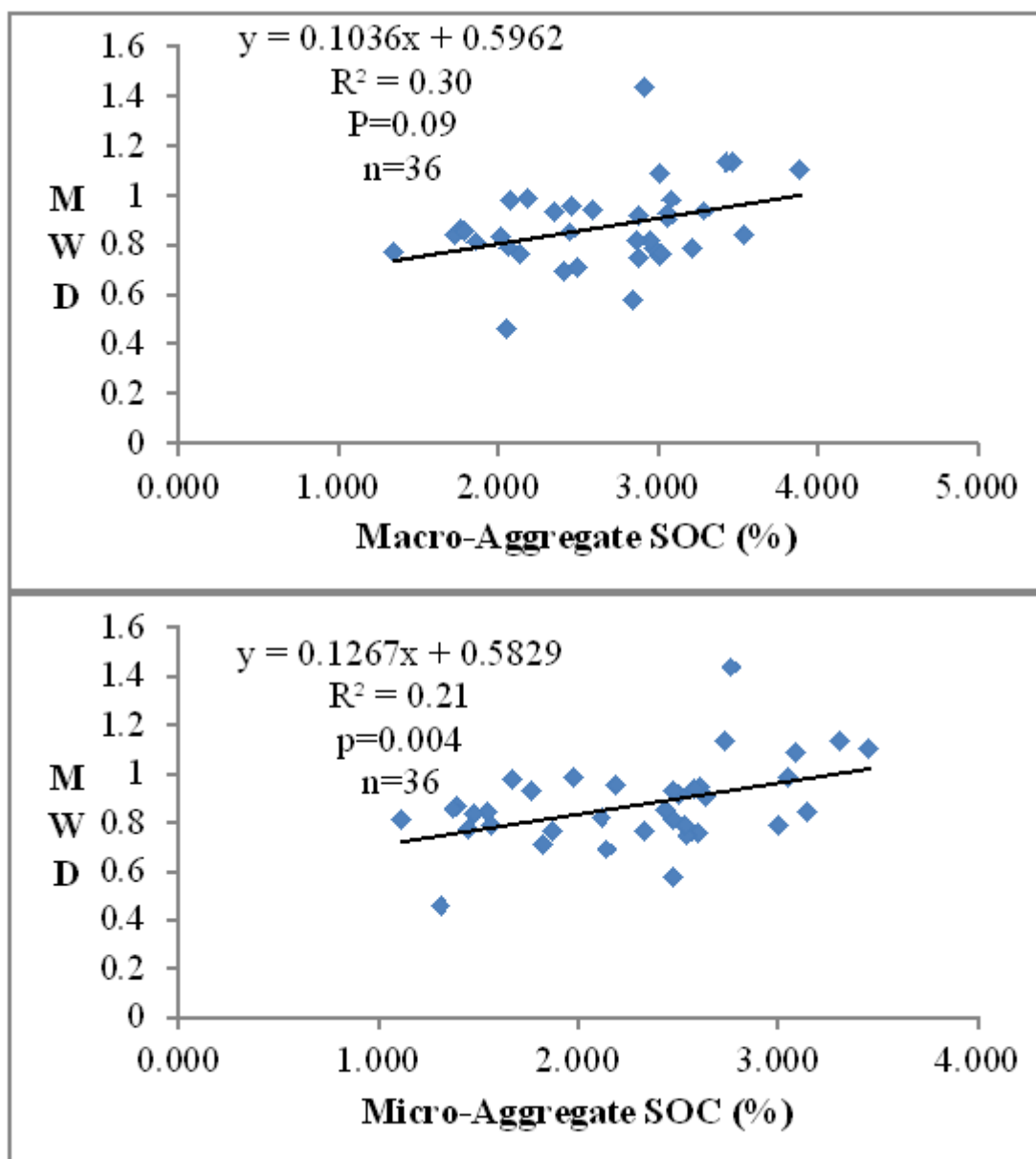


Figure 4.4. Correlation of mean weight diameter (MWD) with micro and macro-aggregate carbon fractions

4.5. Conclusions

Identifying sustainable soil and water conservation practices is necessary to mitigate land degradation. The application of exclosures and terraces increased soil aggregate stability. Soils with high glomalin content had high organic carbon and water stable aggregates. Larger proportions of macro-aggregates were found in exclosures followed by terraces, while large proportions of micro-aggregate fractions were found in non-conserved communal grazing lands. Aggregate stability increased with organic matter content. Macro-aggregates contained higher SOC than micro-aggregates. Significantly higher water stable aggregates were found in exclosures as compared to terraces and non-conserved grazing lands. This implies glomalin content, SOC and aggregate stability have great role in decreasing soil erosion through improving soil structure. Implementation of exclosures and terraces should be expanded to enhance the glomalin content, aggregate stability of the soil and organic carbon associated with aggregates. Further research is also required on the role of glomalin on carbon sequestration.

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**EFFECT OF COMMUNITY-BASED SOIL AND WATER
CONSERVATION PRACTICES ON ABUNDANCE AND DIVERSITY
OF SOIL MACROINVERTEBRATES IN NORTHERN HIGHLANDS OF
ETHIOPIA**

CHAPTER 5. EFFECT OF COMMUNITY-BASED SOIL AND WATER CONSERVATION PRACTICES ON ABUNDANCE AND DIVERSITY OF SOIL MACROINVERTEBRATES IN THE NORTHERN HIGHLANDS OF ETHIOPIA

Abstract

Land rehabilitation programs using soil and water conservation (SWC) practices have been implemented in the northern highlands of Ethiopia for the past three decades. However, there remains a knowledge gap on the effect of SWC practices on abundance and diversity soil macroinvertebrates. The present study determined soil macroinvertebrate (SMI) abundance and diversity in response to spatial conditions i.e., generated by different conservation practices, soil depth, and temporal seasonality with the wet and dry season. The SWC practices considered were enclosure + terrace, enclosure alone, terraces, and were evaluated against non-conserved grazing lands. Each SWC measure was selected in three sites that were considered as replications due to low heterogeneity in terms of human and livestock disturbances and biophysical factors. Soil macroinvertebrates were collected using a monolith according to Tropical Soil Biology and Fertility (TSBF) method. The highest density (55%) of SMI was found in enclosures followed by terraces 26%. Non-conserved communal grazing lands accounted for only 19% of the total. Shannon diversity index indicated significantly ($P < 0.05$) higher (1.21) in the enclosures supported with terraces and the lowest (0.9) was observed in the non-conserved communal grazing lands. Diversity was also significantly ($P < 0.05$) higher (1.26) in wet than dry season (0.70). The highest (41%) Sorensen similarity index among SMI was found between enclosures with terraces and enclosures alone during the wet season. The lowest (20%) Sorensen similarity index was found between terraces alone and enclosures with terraces in dry season. Soil macroinvertebrate abundance was higher in the upper (0–10 cm) soil layer than the lower (10–20 and 20–30 cm) soil depth. Soil macroinvertebrate abundance was positively and strongly correlated with soil moisture ($R^2 = 0.85$) and soil organic carbon stock ($R^2 = 0.95$). However, it was negatively ($R^2 = -0.71$) correlated with bulk density. Generally, the abundance and diversity of SMI increased as enclosures and communal grazing lands were supported with terraces.

Keywords: abundance; diversity; Ethiopia; macroinvertebrate; soil property; Tigray

5.1. Introduction

Land degradation is a common problem in Ethiopian highlands (Nyssen *et al.*, 2007). It is manifested in the form of soil erosion and decline of biodiversity resources in Tigray highlands, northern Ethiopia (Carolyn *et al.*, 2011). It is mainly the result of clearing the natural vegetation and includes deterioration in soil physical, biological, and chemical properties (Nyssen *et al.*, 2007). Grazing on hillsides and other fragile areas is also another common problem in the region. Free grazing contributed to land degradation (Gebregziabher *et al.*, 2009) in the region. To alleviate the problem, the government has mobilized communities and resources for the establishment of physical (terraces) (Mekuria, 2007) and biological (exclosures) soil and water conservation measures (Gebregziabher *et al.*, 2009). Construction of stone terraces on grazing lands combats land degradation (Nyssen *et al.*, 2007).

Terraces enhanced soil properties and vegetation growth (Gebrehiwet and Veen, 2014) and improved land productivity (Amdemariam *et al.*, 2011). Exclosure is practiced in badly eroded lands to protect cutting of trees and restrict grazing to enable vegetative regeneration (Mekuria, 2007; Mastewal *et al.*, 2013). It was found that restoring degraded drylands is technically promising when using area exclosures (Gebregziabher *et al.*, 2009). They are important biological measures to combat land degradation and to increase biomass production (Descheemaeker *et al.*, 2006). Physical conservation structures are designed in fields where biological control practices alone are insufficient to reduce soil erosion (Amdemariam *et al.*, 2011). Thus, exclosures are supported with terraces to enhance their capacity of decreasing soil erosion and improve vegetation growth (BoANR, 2014).

Soil is one of the most essential and diverse natural habitats for biodiversity on earth (Begum *et al.*, 2014). The abundance and diversity of soil organisms integrate physical, chemical, and biological properties of soil (Menta, 2012) and reveal general ecological change (Paolo *et al.*, 2010; Ali *et al.*, 2012). Tropical soil macroinvertebrates (SMI) are highly diverse and provide a number of important ecosystem services. They have direct effect on soil properties through

their feeding and burrowing activities (Ayuke *et al.*, 2009). They contribute to a wide variety of soil processes within the soil system (Ali *et al.*, 2012).

Earthworms, ants, and termites are considered as ecological engineers that mobilize the soil and redistribute the soil organic carbon within the profile (Lal, 1988; Jouquet *et al.*, 2006). They ingest soil particles with their food and contribute to aggregate formation by mixing organic and mineral matter in their gut (Ruiz and Lavelle, 2008). These activities influence soil physical, chemical, and biological properties (Lal, 1988).

Decomposition of organic matter, the formation of humic substances, and nutrient cycling are facilitated by the action of soil macroinvertebrates (Menta, 2012). This has considerable significance on plant growth and maintenance of soil health (Begum *et al.*, 2014). However these beneficial roles of SMI have been ignored and not quantified (Ayuke *et al.*, 2009). Accordingly, SMI abundance and diversity is diminished by land degradation caused by anthropogenic and livestock disturbances (Ayuke *et al.*, 2009; Ali *et al.*, 2014). This reduction of SMI resulted in soil degradation in the tropics (Lal, 1988). Many previous studies conducted in the area focused on the role of terraces and exclosures on erosion control, vegetation diversity, and soil fertility (Nyssen *et al.*, 2007; Mekuria, 2007). However, there is a dearth of information on evaluating and documenting the effect of SWC measures on soil macroinvertebrates abundance and diversity. Therefore, this study was conducted to investigate the effect of community based SWC practices on SMI abundance, diversity, and selected soil properties.

The research questions were: Did the support of free grazing lands with terraces increase abundance and diversity of soil macroinvertebrates? Could protection of free grazing lands through exclosures increase abundance and diversity of soil macroinvertebrates? Could exclosures supported with terraces result in significant increase in abundance and diversity of soil macroinvertebrates? Did soil depth and seasonal variation affect abundance and diversity of soil macroinvertebrates? And finally, could soil properties affect the abundance of soil macroinvertebrates?

5.2. Materials and Methods

5.2.1. Description of the Study Area

The study was conducted in Degua Temben district, which is located 50 km west of Mekelle, regional capital of Tigray region, northern Ethiopia. Geographically, it is located at 13°16'23''N to 13°47'44''N Latitude and 39°3'17''E to 39°24'48'' E Longitude (Figure 1.1). The majority of the area has rugged topography which is typical for the Northern Ethiopian Highlands (Mekuria, 2007). It is also known for extensive SWC measures (BoANR, 2014).

The area receives an annual rainfall of 290 to 900 mm year⁻¹ with an average value of 615 mm year⁻¹. The rainy season usually occurs between June and September. The highest rainfall is in July and August. The growing season varies between 90–120 days. The maximum temperatures occur in May and June (detail description in section 1.2).

5.2.2. Characteristics of the Community-based Soil and Water Conservation Practices

Each of the three selected sites was categorized into four management units described as terraces, enclosures with terraces, enclosures alone and non-conserved open communal grazing lands (Figure 1.2) and their corresponding characteristics are described in Table 5.1 and Table 5.2.

Table 5.1. Characteristics of the community-based SWC measures

SWC measures	characteristics
Non-conserved communal grazing land	Low vegetation cover High proportion of bare soil (personal observation) High stone cover/coarse fragment Sheet, rill and gully erosion very common (personal observation) Low organic carbon (Tekalign, 1991) (Table 5.2) low moisture content (Table 5.2) approximately 10-30% vegetation cover

Stone terraces in grazing lands	relatively more stable and durable than other physical SWC measures higher organic matter than non-terraced fields (Table 5.2) significantly high moisture content than non-conserved grazing lands (Table 5.2) high sediment deposit (personal observation) approximately 10-30% vegetation cover
Exclosure +terrace	closed from the interference of humans and livestock enrichment planting common supported with terraces Medium organic carbon (Tekalign, 1991) (Table 5.2) sheet, rill and gully erosion are less common (personal observation) approximately greater than 80% vegetation cover significantly high moisture content than non-conserved grazing lands (Table 5.2)
Exclosure alone	closed from humans and livestock interference no enrichment planting no physical SWC measures Medium organic carbon (Tekalign, 1991) sheet, rill and gully erosion are less common than non-conserved grazing lands (personal observation) approximately greater than 80% vegetation cover

Table 5.2. Mean values of selected soil properties in response to SWC measures 30 cm depth (mean \pm SEM)

SWC	Soil Moisture (%)	pH	Bulk Density (kg m ⁻³)	SOC (%)	Coarse Fragment (%)	SOC Stock (kg m ⁻²)
Non-conserved grazing lands	8.59 \pm 0.02 ^b	8.22 \pm 0.04 ^b	1290 \pm 31.9 ^a	1.9 \pm 0.1 ^c	57.03 \pm 1.6 ^b	160.7 \pm 11.2 ^c
Terraces in grazing lands	9.61 \pm 0.01 ^a	8.21 \pm 0.04 ^b	1260 \pm 27.8 ^a	2.5 \pm 0.1 ^b	53.2 \pm 2.3 ^b	210 \pm 12.5 ^b
Exclosures + terraces	10.06 \pm 0.01 ^a	8.08 \pm 0.03 ^a	1240 \pm 20.7 ^a	2.9 \pm 0.1 ^a	45.5 \pm 2.0 ^a	288.7 \pm 14.0 ^a
Exclosures alone	8.70 \pm 0.03 ^b	7.9 \pm 0.04 ^a	1220 \pm 47.6 ^a	2.8 \pm 0.1 ^a	54.4 \pm 2.1 ^b	236 \pm 12.6 ^b

Means of each soil property followed by the same letter across each column do not differ significantly at $P \leq 0.05$.

5.2.3. Experimental Design

The three nearby sites selected (i.e., Tesemat, Kerano, and Alasa), considered as replications, were within the same agro ecology in the district. There was no significant difference between spatial heterogeneity in terms of human and livestock disturbance and other biophysical (topography, soil type, and vegetation) among the selected sites. Each of the three selected sites was categorized into four management units described as terraces in grazing lands, enclosures with terraces, enclosures alone, and non-conserved open communal grazing lands (Table 5.1).

Three 40 m long and 5 m meter wide (200 m²) transects were laid along the slope position and sampling monoliths with a dimension of 25 cm long, 25 cm wide, and 30 cm deep were opened. Besides, the 30 cm depth was divided in to three sections at 10 cm interval (Anderson and Ingram, 1993). The transect lines were considered as sampling locations. The minimum distance between the transect lines was 75 m. Within the transects or sampling locations, two sampling monoliths at 10 m distance from each other were opened per season (i.e., 4 monoliths within the 40 m transect for the two seasons). In both seasons, a total of 144 monoliths were opened across the three sites. (i.e., 4 SWC measures x 3 sampling locations/transects x 2 monoliths x 2 season x 3 sites)

5.2.4. Soil MacroInvertebrate Sampling and Identification

The samples were collected during wet (August and September) and dry seasons (April and May) of 2015/16 at 8:00–12:00 a.m. in the morning and 2:00–4:00 p.m. in the afternoon once per month. The soil samples obtained were then carefully hand-sorted on a large plastic tarp and the soil macroinvertebrates (SMI) were preserved in 97% alcohol until identification. Soil macroinvertebrates were identified to family level at Mekelle University Zoology Laboratory and identification was based on morphological characteristics using dissecting microscope and hand lens.

5.2.5. Determination of Diversity Indices of Soil Macroinvertebrates

Shannon's Index of Diversity was used for determining the diversity and its associated evenness (Magurran, 2004). It was calculated as

$$H' = -\sum P_i \ln P_i; i = 1-n \quad [5.1]$$

Where: $P_i = x_i/\sum x_i$

x_i is the number of individuals in the family i and

P_i is the proportion of the total number of individuals belonging to the family i .

Evenness or equitability which indicates the degree of homogeneity in abundance between families is based on the Shannon index of diversity. It assumes a value between 0 and 1, with 1 representing a situation in which all families were equally abundant and evenly distributed, and 0 represents complete absence of the family.

$$E = H/H_{max} = H/\ln S \quad [5.2]$$

Where;

H is the Shannon diversity index and S the number of families

Simpson's index was used for determining dominance of SMI in response to SWC measures.

$$\text{Simpson's index} = \sum (n_i/N)^2 \quad [5.3]$$

Where;

n_i is the number of individuals of a family and

N is the total number of individuals of all families.

The similarity in the composition of the families of SMI within the different SWC measures was determined by using Sorensen's index of similarity (Sorensen, 1984). It is calculated as follows

$$I (\text{Sorensen}) = 2c/(2c + a + b) \quad [5.4]$$

Where a is the family number in one of the SWC measures

b is the number of family in another SWC measure, and

c is the number of family common to both SWC measures.

5.2.6. Soil Sampling and Analysis

Soil pH was measured by using a combined glass electrode pH meter (Chopra and Kanwar, 1976) in 1:2.5 soil-water suspensions, soil bulk density (bd) was measured by the core method; soil organic (oc) by wet oxidation using potassium dichromate (Walkley and Black, 1934). Percent of soil moisture content was calculated as the difference between weight of moist soil and weight of oven dry soil divided by weight of oven dry soil multiplied by 100. Soil organic carbon stock was determined from bulk density, concentration of soil organic carbon, soil thickness, and coarse fragments (Lal *et al.*, 2007).

5.3. Data analysis

The significance level of mean richness, Shannon–Weiner index, Simpson diversity index, evenness and Sorenson similarity index was tested in response to community based SWC measures, soil depth, and season using SAS 9.2. Comparison of means was made based on Duncan's multiple range tests (DMRT). Pearson correlation coefficient was determined to see the relationship between soil properties and abundance of soil macroinvertebrates.

5.4. Results and Discussion

5.4.1. Effect of Soil and Water Conservation Measures on Abundance and Diversity of Soil Macroinvertebrates

Comparison of total soil macroinvertebrate (SMI) contents among the community based SWC measures indicated that 29.63% of the SMI were recorded in exclosures with terraces followed by terraces 26% and exclosures alone 25%. The lowest abundance (19%) of SMI was found in non-conserved communal grazing lands (Table 5.3).

The presence of higher abundance of SMI in exclosures could be due to low human and livestock interference. Besides, exclosures had high vegetation cover and this might have

created a suitable micro habitat for soil macroinvertebrates. Terraces also acted as an important niche for the soil macroinvertebrates. It was reported that soil macroinvertebrate abundance increased with plantation or vegetation cover (Ayuke *et al.*, 2009). The lowest SMI in non-conserved communal grazing lands could be due to relatively low organic matter and moisture content (Table 5.1). Low organic matter causes a decrease in abundance (Wiwatwitaya *et al.*, 2005). Soil compaction through trampling during grazing reduces soil organisms (Ali *et al.*, 2012; Mehraj and Bhat, 2013).

Many of the SMI families were found in all the SWC measures. However, certain families such as family Lumbricidae, Acrididae, Dytiscidae, Staphylinidae, Scutigeridae, Ligiidae, Argasidae and Japygidae were not found on non-conserved open communal grazing lands (Table 5.3). This could be their degree of sensitivity to disturbance and low organic matter in non-conserved grazing lands. The total abundance of SMI was in the order of exclosures with terraces > terraces > exclosures alone > non-conserved grazing lands.

Table 5.3. Effect of SWC on abundance of soil macroinvertebrates (individuals m⁻²)

SMI	Non Conserved Communal Grazing Lands	%	Terrace s in Grazed Lands	%	Exclosure s with Terraces	%	Exclosur e alone	%
Formicidae	745.8	23.8	755.6	24.1	858.7	27.4	780.0	24.8
Termitidae	0.9	0.1	250.7	37.5	241.3	36.1	175.1	26.2
Carabidae	10.4	16.3	3.6	5.6	22.2	34.9	27.6	43.3
Rhinotermitidae	0.0	0.0	8.9	20.8	29.3	68.8	4.4	10.4
Spirostreptidae	9.8	16.7	11.5	19.7	12.4	21.2	24.9	42.4
Gryllidae	12.4	17.9	18.7	26.9	24.0	34.6	14.2	20.5
Lycosidae	6.2	17.5	7.1	20.0	15.1	42.5	7.1	20.0
Scarabaeidae	10.7	26.7	13.3	33.3	11.6	28.9	4.4	11.1
Scorpionidae	2.7	9.1	16.0	54.5	3.6	12.1	7.1	24.2
Cryptopidae	9.8	28.2	2.7	7.7	13.3	38.5	8.9	25.6
Blattidae	2.7	15.0	6.2	34.9	7.1	39.9	1.8	10.0
Lumbricidae	0.0	0.0	4.4	31.3	5.3	37.5	4.4	31.3
Acrididae	0.0	0.0	0.9	9.1	6.2	63.6	2.7	27.3
Dytiscidae	0.0	0.0	0.0	0.0	2.7	100. 0	0.0	0.0
Staphylinidae	0.0	0.0	1.8	25.0	2.7	37.5	2.7	37.5
Scutigeridae	0.0	0.0	2.6	100.0	0.0	0.0	0.0	0.0
Ligiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.9	100. 0
Bostrichidae	0.9	33.3	0.9	33.3	0.0	0.0	0.9	33.3
Argasidae	0.0	0.0	0.9	50.0	0.9	50.0	0.0	0.0
Japygidae	0.0	0.0	0.0	0	0.0	0.0	0.9	100
Total	811.6	19.1	1105.7	26.1	1256.4	29.6	1068.0	25.2

Exclosures with terraces had significantly higher (1.21) Shannon diversity index followed by exclosures alone (1.01) while the lowest (0.9) was found on non-conserved communal grazing lands and terraced grazed lands (Table 5.4). The presence of significantly higher Shannon diversity index in exclosures could be due to high organic matter, moisture content, and low disturbance.

Adeniyi (2009) observed high diversity of soil macro fauna in areas where light penetrates, litters of leaves fall and ground is rocky. Other studies suggested that grazing exclusion is important for the recovery of soil fauna because livestock and human activities are restricted (Mekuria and Aynekulu, 2010; Liu *et al.*, 2012). Similar studies reported that grazing decreases macroinvertebrate diversity (Mills and Adl, 2011; Anagement and State, 2014). This could be due to decrease of plant biomass (Liu *et al.*, 2011).

Richness was significantly higher in exclosures with terraces followed by exclosures alone and terraces in grazed lands. This could be due to the presence of micro patches under trees in exclosures that create habitat heterogeneity and better moisture content on terraces in grazed lands. This is in line with Liu *et al.* (2012), who reported that grazing exclusion has a remarkable effect on richness, where it was significantly higher at the exclosure than at the continuous grazing site. Other studies also showed that reduced grazing pressure can increase species richness (Parfitta *et al.*, 2010; Bugalho *et al.*, 2011). The decreasing order of family richness was as follows: exclosures with terraces > terraces in grazed lands > exclosures alone > non-conserved communal grazing lands.

Simpson's diversity index was relatively higher in exclosures supported with terraces followed by exclosures without terraces (Table 5.4) which could be due to availability of relatively more moisture and organic matter on exclosures soil. Soil organic matter improves soil aeration, temperature, and moisture; and provides a resilient resource base for a wide variety of soil organisms (Lehman *et al.*, 2015). Tree litter contains high diversity and large numbers of macrofauna (Ali *et al.*, 2012).

Though not significant, non-conserved grazing lands and terraces in grazed land had relatively lower Simpson's diversity index which indirectly shows higher dominance of a certain family that can tolerate disturbances by humans and livestock. A study by Mehraj and Bhat (2013) showed that trampling by livestock reduces the diversity of macroinvertebrates. The other reason could be land degradation by erosion, because it was reported that eroded soils have low diversity of soil invertebrates (Barrios, 2007; Balan *et al.*, 2012).

Table 5.4. Effect of SWC measures on abundance and diversity of soil macroinvertebrates (family level) (Mean \pm SEM)

Indices	SWC Measures			
	Non Conserved Communal Grazing Land	Terraces	Exclosures with Terraces	Exclosures alone
Abundance	811.6 \pm 318.9 ^a	1105.7 \pm 318.9 ^{bc}	1256.4 \pm 318.9 ^a	1067.1 \pm 318.9 ^{ab}
Family richness	2.8 \pm 0.32 ^b	3.3 \pm 0.76 ^b	4.4 \pm 0.68 ^a	3.7 \pm 0.61 ^{ab}
Shannon diversity index	0.9 \pm 0.12 ^b	0.9 \pm 0.19 ^b	1.21 \pm 0.16 ^a	1.0 \pm 0.17 ^{ab}
Evenness	0.9 \pm 0.01 ^a	0.9 \pm 0.05 ^a	0.9 \pm 0.01 ^a	0.9 \pm 0.01 ^a
Simpson index (1_D)	0.5 \pm 0.06 ^a	0.4 \pm 0.08 ^a	0.6 \pm 0.07 ^a	0.5 \pm 0.07 ^a
Dominance_D	0.5 \pm 0.06 ^a	0.5 \pm 0.08 ^a	0.4 \pm 0.07 ^a	0.5 \pm 0.07 ^a

Means followed by the same letter with in a row do not differ significantly at $P \leq 0.05$

5.4.2. Effect of Soil Depth on Abundance and Diversity of Soil Macroinvertebrates

Mean total abundance of SMI showed a significant ($P < 0.05$) differences along the three depths where it was higher in 0–10 cm (1852.4 individuals m^{-2}), followed by 10–20 cm (894.7 individuals m^{-2}), and was the lowest in 20–30 cm (434.20 individuals m^{-2}). Thus, SMI in the top soil (0–10 cm) are 51.7 and 76.6% higher than SMI in the 10–20 and 20–30 cm depths respectively (Table 5.5). This shows deeper soil is less preferred by most SMI due to SOC decrease down in depth.

The most abundant SMI families in each soil depth were Formicidae followed by Termitidae (Table 5.5). This shows that Formicidae and Termitidae can inhabit wide ranges of environmental conditions in tropical environments. Lal (1988) also described them as the most dominant SMI in the tropics.

Specifically, except for Lumbricidae, Dytiscidae, Staphylinida, Ligiidae, Scutigeridae, Bostrichidae, Argasidae, and Japygidae, the abundance of the other SMI were significantly ($P < 0.05$) higher in the top soil (0–10 cm) than lower (10–20 cm and 20–30 cm) soil depths (Table 5.5). This revealed that surface soil is more appropriate for these soil

macroinvertebrates. This could indicate their preference to high amount of organic matter. They could also have better tolerance to light and temperature changes. Even though low in number, the presence of surface dwelling (epigeal) SMI such as Gryllidae, Acrididae, Blattoidae, Cryptopidae, Spirostreptidae, and Japygidae below 10 cm and soil dwelling (hypogeal) SMI such as Carabidae, Cryptopidae, Staphylinidae, Formicidae, Termitidae, and Scorpionidae in subsurface soil (20–30 cm depth) could be their temporal relocation in search of moisture or to hide and protect themselves from predators.

Table 5.5. Effect of soil depth on abundance of soil macroinvertebrates (individuals m^{-2}) (Mean \pm SEM)

SMI	Soil Depth		
	0–10 cm	10–20 cm	20–30 cm
Formicidae	1424.4 \pm 300.04 ^a	664.4 \pm 117.33 ^b	266.2 \pm 31.19 ^b
Termitidae	207.3 \pm 156.97 ^a	177.0 \pm 130.51 ^a	116.7 \pm 107.09 ^a
Carabidae	32.0 \pm 15.51 ^a	15.3 \pm 8.14 ^{ab}	1.3 \pm 0.92 ^b
Rhinotermitidae	8.7 \pm 6.16 ^a	0.7 \pm 0.66 ^a	22.7 \pm 19.99 ^a
Spirostreptidae	35.3 \pm 13.44 ^a	8.7 \pm 3.72 ^b	0.0 \pm 0 ^b
Gryllidae	36.7 \pm 11.00 ^a	10.7 \pm 6.22 ^b	4.7 \pm 2.80 ^b
Lycosidae	23.3 \pm 8.28 ^a	2.0 \pm 1.46 ^b	1.3 \pm 0.92 ^b
Scarabaeidae	25.3 \pm 7.69 ^a	1.3 \pm 1.33 ^b	3.3 \pm 2.15 ^b
Scorpionidae	18.7 \pm 6.78 ^a	1.3 \pm 0.92 ^b	2.0 \pm 1.46 ^b
Cryptopidae	14.0 \pm 4.00 ^a	6.0 \pm 2.69 ^a	6.0 \pm 3.02 ^a
Blattoidae	6.0 \pm 2.11 ^a	6.0 \pm 3.01 ^a	1.3 \pm 0.92 ^a
Lumbricidae	6.0 \pm 2.69 ^a	4.7 \pm 3.53 ^a	0.0 \pm 0 ^a
Acrididae	6.0 \pm 2.85 ^a	1.3 \pm 1.33 ^b	0.0 \pm 0.0 ^b
Dytiscidae	2.0 \pm 2.00 ^a	0.0 \pm 0.0 ^a	0.0 \pm 0 ^a
Staphylinidae	2.7 \pm 1.57 ^a	1.3 \pm 1.33 ^a	1.3 \pm 0.92 ^a
Scutigerae	2.0 \pm 1.08 ^a	0.0 \pm 0.0 ^b	0.0 \pm 0 ^b
Ligiidae	0.7 \pm 0.66 ^a	0.0 \pm 0.0 ^a	0.0 \pm 0 ^a
Bostrichidae	1.3 \pm 0.92 ^a	0.7 \pm 0.66 ^a	0.0 \pm 0 ^a
Argasidae	1.3 \pm 0.92 ^a	0.0 \pm 0.0 ^a	0.0 \pm 0 ^a
Japygidae	0.0 \pm 0 ^a	0.7 \pm 0.66 ^a	0.0 \pm 0 ^a
Mean total	1852.4 \pm 350.59 ^a	902.0 \pm 218.58 ^b	326.9 \pm 112.66 ^c

Means followed by the same letter within a row do not differ significantly at $P \leq 0.05$.

Soil SMI richness was significantly ($P < 0.05$) higher at the surface (0–10 cm) depth than the lower (10–20 cm and 20–30 cm) depths. Surface soil (0–10 cm) was 52% and 62% richer in SMI than the 10–20 and 20–30 cm depths respectively. Furthermore, surface soil (0–10 cm) had significantly ($P < 0.05$) higher (1.52) Shannon diversity index than the lower 10–20 cm

(0.83) and 20–30 cm depths (0.58) respectively. Similarly, Simpson index was higher in surface soil than lower depths which show variation in SMI composition decreased down in depth (Figure 5.1). This could be due to the presence of more pore space, good aeration, and more available food in surface soil. Surface soil (0-10 cm) is more appropriate to presence of macroinvertebrates (Kooch and Haghverdi, 2014). A similar study found that most tropical soil fauna live on the top 10 cm soil depth where organic matter is decomposed (Mehraj and Bhat, 2013).

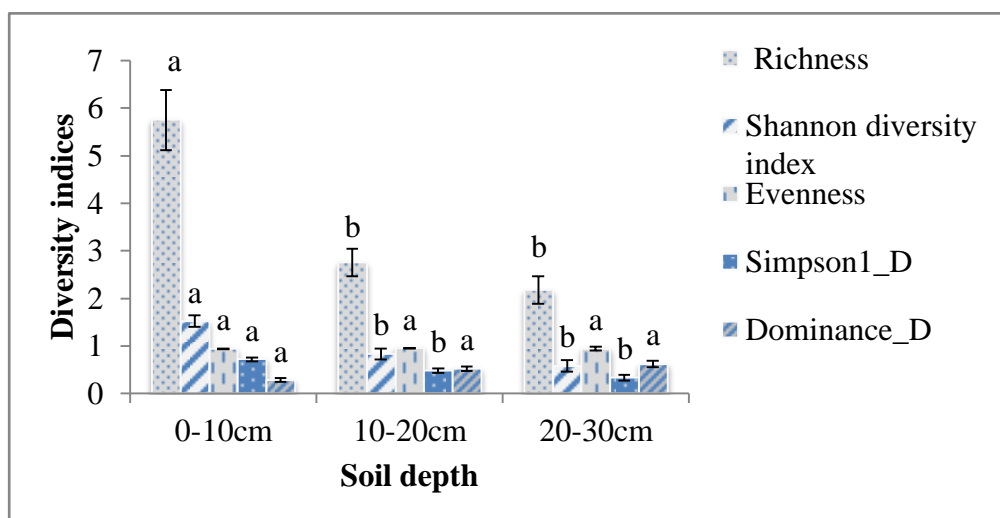


Figure 5.1. Effect of soil depth on SMI diversity. Different letters above the bars indicate significant differences among soil depth. Bars indicate mean \pm SEM.

5.4.3. Effect of Seasonal Changes on Abundance and Diversity of Soil

Macroinvertebrates

Comparison of mean total SMI abundance across seasons indicated significantly ($P < 0.05$) higher (1784.6 individuals m^{-2}) in wet than in dry season (336 individuals m^{-2}). Total SMI in the wet season are 81% higher than in the dry season. This indicated most of the SMI are sensitive to high temperatures of the dry season which might have hindered their reproduction or they might have been hidden under stones as a mechanism to adapt to the temperature changes.

A total of 20 families during the wet season and 13 families during the dry season were identified (Table 5.6). The families commonly found in both seasons were family Formicidae, Termitidae, Gryllidae, Lycosidae, Scarabaeidae, Scorpionidae, Carabidae, Cryptopidae, Blatidae, Acrididae, Bostrichidae, Spirostreptidae, and Argasidae. The SMI families that were common in both seasons decreased their abundance due to seasonal changes as follows: Lycosidae > Termitidae > Carabidae > Scorpionidae > Spirostreptidae > Formicidae > Scarabaeidae > Acrididae > Blattellidae > Cryptopidae.

The other SMI such as Lumbricidae, Staphylinidae, Dytiscidae, Scutigeridae, Japygidae, Ligiidae, and Rhinotermitidae were not found during the dry season (Table 5.6). This indicated that SMI are more sensitive to high temperatures and this might have hindered their reproduction or they might have died. The other reason is they might be hidden under stones as a mechanism to adapt high temperature. This aspect requires further investigation. Many of the families (i.e., Carabidae, Scarabaeidae, Staphylinidae, Dytiscidae and Bostrichidae) in both seasons belong to the order Coleoptera. This could be due to the presence of sclerotized elytra which cover the membranous flight wings and which might protect them from environmental stress and attack by predators. The most abundant SMI in each season were family Formicidae followed by family Termitidae (Table 5.6).

Seasons also brought significant ($P < 0.05$) variation in SMI richness. The highest was found during the wet season while the lowest was found during dry season (Figure 5.2). The decrease in abundance and richness of SMI during the dry season (Table 5.6, Figure 5.2) could be due to lack of resources (such as moisture) and presence of high temperatures.

The highest (1.26) Shannon diversity index was found during the wet season but the lowest (0.70) occurred during the dry season. Similarly, Simpson index was high during the wet season (Figure 5.2). Begum *et al.* (2014) reported that season had greater influence on soil macroinvertebrates. Barros *et al.* (2004) observed high SMI abundance at the rainy season. The decrease in SMI abundance and diversity during the dry season could be due to low moisture content. Besides, diversity is driven by stress such as temperature changes (Decaens,

2010). Similarly, Ali *et al.* (2012) reported that most SMI avoid risk of open space during dry period with high light intensity and temperature.

In both seasons, family Formicidae was the most abundant. This could be due to their ability to adapt to adverse conditions. Ants are found in any type of habitat with greater local diversity and under different types and extent of land disturbances (Folgarait, 1998). However, many SMI are sensitive to changes in their environmental conditions (Ayuke *et al.*, 2009).

Table 5.6. Effect of season on abundance of SMI (individuals m⁻²) (Mean ± SEM)

Soil Macroinvertebrates	Wet Season	Dry Season
Formicidae	1289.3 ± 10.00 ^a	280.7 ± 21.22 ^b
Termitidae	326.4 ± 147.93 ^a	7.6 ± 5.11 ^b
Carabidae	28.0 ± 11.13 ^a	3.7 ± 1.61 ^b
Rhinotermitidae	21.3 ± 13.79 ^a	0.0 ± 0.00 ^a
Spirostreptidae	24.0 ± 9.29 ^a	5.3 ± 2.62 ^a
Gryllidae	16.4 ± 5.96 ^a	18.2 ± 6.98 ^a
Lycosidae	16.9 ± 5.77 ^a	0.9 ± 0.61 ^b
Scarabaeidae	15.6 ± 5.31 ^a	4.4 ± 2.35 ^b
Scorpionidae	12.4 ± 4.73 ^a	2.2 ± 1.13 ^b
Cryptopidae	8.9 ± 2.95 ^a	8.4 ± 2.50 ^a
Blattoidae	6.7 ± 2.24 ^a	2.2 ± 1.13 ^a
Lumbricidae	7.1 ± 2.88 ^a	0.0 ± 0.00 ^b
Acrididae	3.6 ± 1.92 ^a	1.3 ± 0.98 ^a
Dytiscidae	1.3 ± 1.33 ^a	0.0 ± 0.00 ^a
Staphylinidae	3.6 ± 1.44 ^a	0.0 ± 0.00 ^b
Scutigeridae	1.3 ± 0.73 ^a	0.0 ± 0.00 ^b
Ligiidae	0.4 ± 0.44 ^a	0.0 ± 0.00 ^a
Bostrichidae	0.4 ± 0.44 ^a	0.9 ± 0.61 ^a
Argasidae	0.4 ± 0.44 ^a	0.4 ± 0.44 ^a
Japygidae	0.4 ± 0.44 ^a	0.0 ± 0.00 ^a
Mean total	1784.6 ± 263.87 ^a	336.2 ± 26.06 ^b

Means of each SMI followed by the same letter within a row do not differ significantly at $P \leq 0.05$.

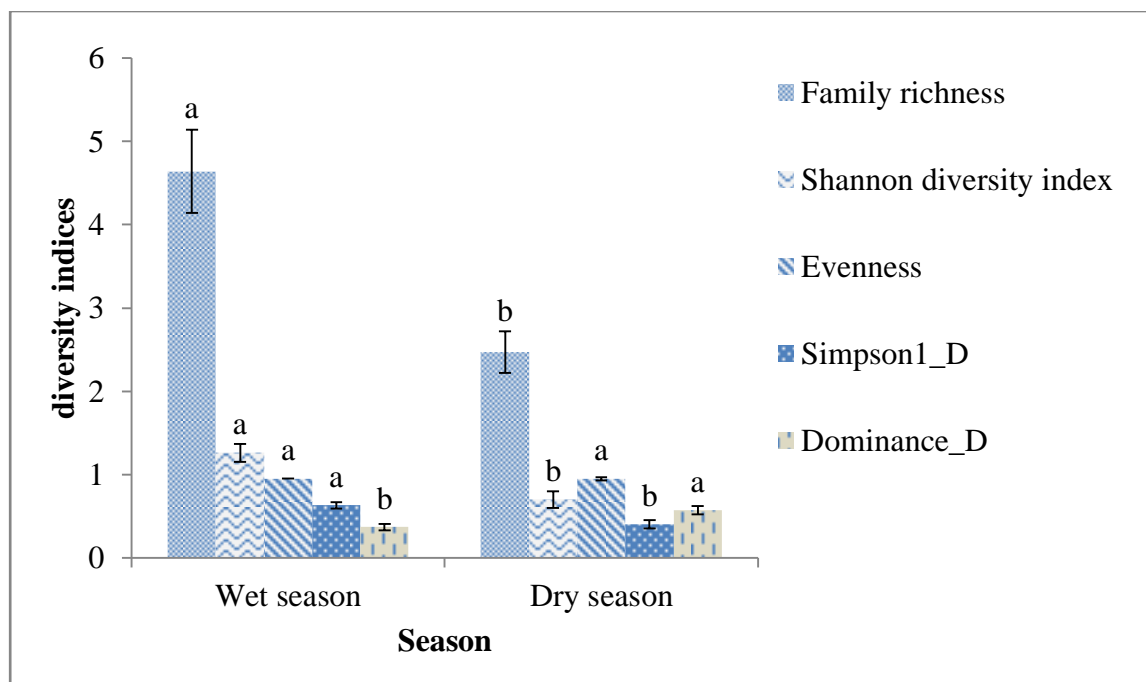


Figure 5.2. Effect of season on SMI diversity. Different letters above the bars indicate significant differences between seasons. Bars indicate mean \pm SEM.

During the wet season, the Sorenson similarity index in macro invertebrate composition between two SWC measures revealed the greatest (41%) between exclosures with terraces and exclosures alone and the lowest (33%) was between exclosures with terraces and non-conserved communal grazing lands (Table 5.7). During the dry season, the highest (39%) Sorenson similarity index among SMIs was found between exclosures alone and non-conserved communal grazing lands, while the lowest (20%) was between terraces and exclosures with terraces. The overall similarity index of SMIs between pairs of SWC measures was not significant (at $P < 0.05$) except the similarity index between exclosures with terraces and terraces in grazed land in dry season.

Table 5.7. Effect of SWC measures on Sorenson's similarity index of soil macroinvertebrates (Mean \pm SEM)

SWC Measures	Wet Season			
	Non-Conserved Grazing Lands	Terraces in Grazed Lands	Exclosures with Terraces	Exclosures alone
Non-conserved grazing land				
Terraces in grazed lands	0.37 \pm 0.04 ^{ab}			
Exclosures with terraces	0.38 \pm 0.02 ^{ab}	0.39 \pm 0.0 _a		
Exclosures without terraces	0.33 \pm 0.05 ^{ab}	0.38 \pm 0.02 ^{ab}	0.41 \pm 0.01 ^a	
Dry Season				
Non-conserved grazing lands				
Terraces	0.32 \pm 0.03 ^{ab}			
Exclosures with terraces	0.36 \pm 0.04 ^{ab}	0.20 \pm 0.0 _c		
Exclosures without terraces	0.39 \pm 0.03 ^{ab}	0.26 \pm 0.07 ^{bc}	0.35 \pm 0.05 ^{ab}	

Means followed by the same letter across each column do not differ significantly at $P \leq 0.05$.

5.4.4. Relative Contribution of Soil Properties to Soil Macroinvertebrates

Soil properties affected the abundance of soil macroinvertebrates. Abundance of SMI was higher in the SWC measures with higher soil moisture content and soil organic carbon stock (i.e., in exclosures and terraces) than non-conserved communal grazing lands, showing that organic matter and moisture are important for SMIs (Figure 5.3). The abundance of SMI was positively and strongly correlated with soil moisture, organic matter (OM) and soil carbon stock (Table 5.8). This result agrees with that of Santorufo *et al.* (2012) who found high abundance of SMI in areas with high soil organic carbon content. Pearson correlation also indicated a positive correlation between abundance, soil moisture, and organic matter. Besides, there is a strong and significant correlation between abundance and SOC stock (Table 5.8).

Soil bulk density, pH, and coarse fragment content were higher in non-conserved communal grazing lands followed by terraces and were lower in exclosures (Figure 5.3). However, SMI abundance was inversely proportional i.e., abundance decreased with the increase of bulk density, pH, and coarse fragment content. The decrease in soil bulk density and soil pH in exclosures is due to the presence of high organic matter. Thus, soils with low bulk density are less compacted and contain enough aeration for soil macroinvertebrates. However, livestock trampling in grazing lands increased soil bulk density and affected the biological soil surface (Khosravi and Mahdieh, 2017).

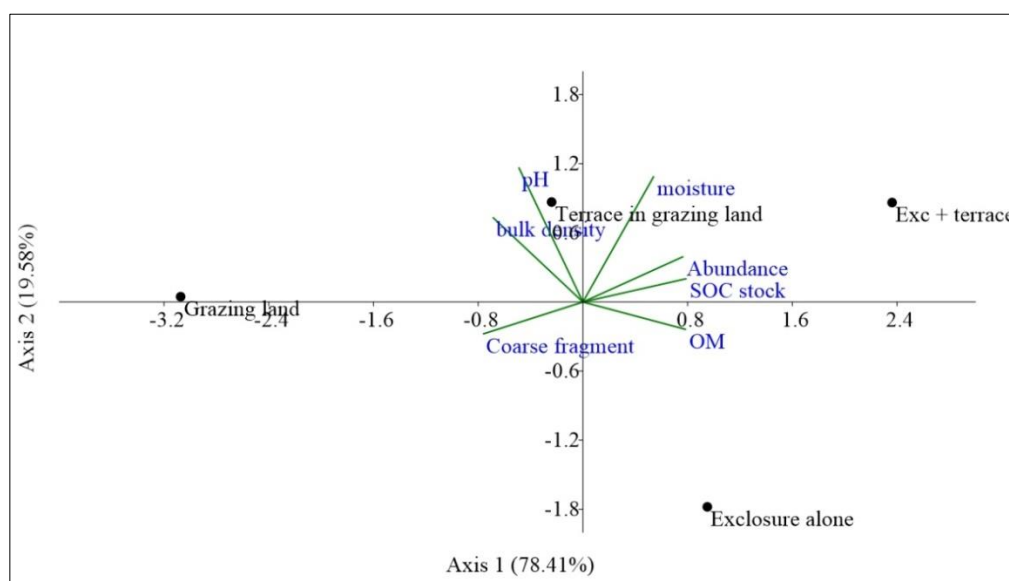


Figure 5.3. PCA bi plot graph of studied soil properties

Table 5.8. Relationship between SMI abundance and soil properties

Parameters	Abundance	Soil Moisture	pH	Bulk Density	Coarse Fragment	SOC	SOC Stock
Abundance	1						
Soil moisture	0.85 ^{ns}	1					
pH	-0.46 ^{ns}	0.85 ^{ns}	1				
Bulk density	-0.71 ^{ns}	-0.22 ^{ns}	0.30 ^{ns}	1			
Coarse fragment	-0.89 ^{ns}	-0.89 ^{ns}	0.58 ^{ns}	0.45 ^{ns}	1		
SOC	0.90 ^{ns}	0.54 ^{ns}	0.03 ^{ns}	-0.93 ^{ns}	-0.72 ^{ns}	1	
SOC stock	0.97 [*]	0.76 ^{ns}	0.31 ^{ns}	-0.77 ^{ns}	-0.92 ^{ns}	0.94 ^{ns}	1

*: Correlation is significant at $P < 0.05$; ns refers to correlation is not significant.

5.5. Conclusions

The abundance and diversity of soil macroinvertebrate was significantly higher during the wet than dry seasons; and on the top soil than in the lower soil depths. Soil nutrients relatively influenced soil macro invertebrate abundance, as it was positively and strongly correlated with soil carbon stocks and soil moisture, while it was negatively correlated with bulk density and coarse fragment content.

Exclosures improved soil macroinvertebrate abundance and diversity. Terraces in grazing lands also improved abundance and diversity of soil macroinvertebrates. Land degradation by grazing and other human interferences decreased diversity and abundance of soil macroinvertebrates. Therefore, communal grazing lands should either be supported with terraces or establishment of exclosures as these measures improved SMI abundance and diversity. This indicates soil and water conservation practices in the northern highlands of Ethiopia have important implications for land restoration and biodiversity recovery; and further research is required on which type of soil macroinvertebrate is best for land restoration and soil health recovery.

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CHAPTER 6. SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

Land degradation is an imperative problem in many tropical developing countries that is undermining development efforts to foster environmental sustainability. It leads to degradation of soil, vegetation cover and loss of soil biological diversity. Due to steep slope, high population density and improper land management, severe land degradation occurred in Ethiopian highlands with more than 85% of the land degraded to various degrees. Thus, management of soil as a natural resource is of utmost importance for overall understanding of soil and allows management of the soil resources. It remains a key development challenge in Tigray highlands northern Ethiopia. Thus, the regional government is trying to reverse the degraded landscape through different SWC measures with the participation of the local communities.

The SWC measures implemented on the hillsides have shown great impact on the improvement of biophysical conditions. This was a good indicator for further application of SWC measures on many degraded hillside landscapes of the region. The CBSWC measures in the study area were mainly stone terraces, enclosure with and without stone terraces; and were evaluated against non-conserved communal grazing lands. The role of SWC measures on crop productivity, vegetation density and diversity, soil physicochemical properties, erosion control and sediment yield had been studied, however there remains a knowledge gap on the effect of CBSWC measures on soil health which is measured mainly in terms of soil biological properties that include: SOC stock, MBC, AMF, glomalin; and SMI abundance and diversity.

Taking this in to consideration, the study had four chapters. The first chapter determined the effect of CBSWC practices on SOC stock and MBC distribution. The second chapter investigated the effect of CBSWC practices on AMF spore density, root colonization and soil nutrients, while the third chapter addressed the knowledge gap on the effect of CBSWC practices on soil glomalin, aggregate stability, aggregate size distribution and SOC associated in soil aggregates. The last chapter evaluated the effect of community based soil and water conservation practices on abundance and diversity of soil macro invertebrates.

Soil sample collection was carried out systematically using transects based on slope positions (i.e, foot, middle and upper slope positions). For the determination of AMF spore density, root colonization and soil nutrients, soil samples were collected from rhizosphere of woody plant species that were commonly found in all SWC measures using quadrats that were laid in the three slope positions. Spores were isolated using wet sieving and decanting method and were classified based on color, presence and absence of subtending hyphae and surface ornamentation according to International Vesicular Arbuscular Mycorrhizae (INVAM) using a compound microscope. Arbuscular mycorrhiza fungi root colonization was done using the gridline intersection method. Soil glomalin was determined using the method of Bradford assay, while for investigation of SMI abundance and diversity, SMIs were collected using a monolith method according to TSBF and they were identified and classified using identification keys with the help of dissecting microscope and a hand lens. Soil macroinvertebrate diversity, abundance and similarity were calculated using Shannon, Simpson and Sorensen indices respectively. The soil physical and chemical properties were analysed using standard procedures.

The results revealed that the highest mean SOC stock was recorded in exclosures with terraces followed by exclosures without terraces and terraces, while the lowest was found in non-conserved communal grazing lands. The upper soil depth had significantly higher SOC stock than the lower soil depth. Microbial biomass carbon was the highest in exclosures without terraces followed by exclosures with terraces, terraces and non-conserved communal grazing lands.

Glomus was the abundant genus among the AMF genera. Both exclosures had the highest spore density of all types of spores followed by stone terraces, while non-conserved communal grazing lands had the lowest. All the plant species were found to be colonized by hyphae, arbuscules and vesicles. All forms of root colonization were high in both exclosures followed by terraces. Terraces had significantly higher total AMF root colonization than non-conserved communal grazing lands. Arbuscular mycorrhiza fungi spore density and root colonization were positive and strongly related with soil nutrients.

Easily extractable glomalin and total glomalin were significantly higher in exclosures compared to terraces and non-conserved grazing lands. The macro-aggregate fraction of all SWC measures was the lowest in non-conserved grazing lands, while the highest was in exclosures with terraces. The micro-aggregate fraction was the lowest in exclosures, while the highest was in non-conserved grazing lands. Exclosures had significantly higher percent of WSA compared to terraces and non-conserved grazing lands. Mean weight diameter (MWD) was also significantly higher in exclosures. Comparison of aggregate associated organic carbon (AAOC) showed relatively higher organic carbon in macro-aggregates than micro-aggregates.

A total of twenty families were identified in all CBSWC measures. The highest density of SMI was found in both exclosures followed by terraces. Non-conserved communal grazing lands accounted for the lowest total soil macroinvertebrates. Shannon diversity index was significantly higher in the exclosure supported with terraces and the lowest was observed in the non-conserved communal grazing lands. It was also significantly higher in wet than dry season. The highest Sorensen similarity among SMI was found between exclosures with terraces and exclosures without terraces during the wet season. The lowest Sorensen similarity index was found between terraces and exclosures with terraces in dry season. Soil macroinvertebrate abundance was higher in the upper than lower soil depth. In both seasons, Formicidae family was the most abundant group followed by Termitidae, while Beetles were found the most diverse families. The correlation between SMI abundance and soil moisture was positive and strong and that of SMI with the soil organic carbon stock was also positive and significant but it was negatively correlated with bulk density.

Based on the results obtained the following conclusions can be drawn:

- Community based soil and water conservation practices improved and restored the SOC stock and microbial biomass carbon of degraded free grazing lands in the highlands.
- Rehabilitating the communal grazing lands with CBSWC measures is an important approach for restoring AMF and regenerating the degraded lands.
- Community based soil and water conservation practices are important strategies for

rehabilitation of degraded lands through improving glomalin content, aggregate structure and stability, and aggregate associated organic carbon.

- The abundance and diversity of SMI increased due to the development of CBSWC measures on open non-conserved communal grazing lands.

The following recommendations can be drawn from the results of this study:

- Scaling up community based SWC practices is necessary as they are proven local technologies so as to regenerate degraded ecosystems sustainably.
- Enrichment plantation on exclosures and grazing lands need to be encouraged.
- Re-vegetation strategies in the area shall involve use of AMF technologies.
- Soil biological properties study need to be included in the policy of management strategies of degraded lands in the region.
- Continuous follow up and protection of the CBSWC practices is necessary in the area.
- Re-vegetation policies in the area should focus on use economically and environmentally viable grass and plant species.

A need to conduct further long term research

- To generate data on the distribution of SMI in response to other factors.
- On AMF at molecular level, at large scale and at different seasons is required.
- To investigate the best group of SMI in ecosystem processes related to soil health.
- To generate data base on SMI trait is needed so as to document, manage and apply them for scientific study on soil health.
- On the impact of different tree species on aggregate stability, glomalin content, SMI and soil carbon sequestration.
- On the role of glomalin on carbon sequestration
- On the intrinsic relationship between soil glomalin, AMF and aggregate stability by including other factors.
- On determination of MBC using other methods, such chloroform fumigation extraction and fumigation incubation by including other environmental factors.

7. APPENDICES

Appendix Table 1. Summary of statistics for SOC, bulk density and SOC stock in response to SWC measures

SWC	Variable	Mean	Median	Minimum	Maximum	Std Error	CV	Skewness	Kurtosis
Non conserved grazing land	Organic carbon (%)	1.9	2.1	0.7	3.2	0.1	27	-0.4	0.2
	Bulk density (kg m ⁻³)	1284	1362	770	1497	25	14	-0.9	0.0
	SOC stock (Mg ha ⁻¹)	16	16	3	31	1.0	45	0.2	-0.7
Terracing	Organic carbon (%)	2.5	2.5	1.3	3.6	0.1	17	0.0	0.5
	Bulk density (kg m ⁻³)	1260	1288	850	1485	21	12	-0.9	0.4
	SOC stock (Mg ha ⁻¹)	21	21	6	40	1.0	37	0.1	0.0
Exclosure + terracing	Organic carbon (%)	2.9	2.9	1.3	3.8	0.1	20	-0.4	0.4
	Bulk density (kg m ⁻³)	1244	1250	1011	1579	18	10	0.3	-0.4
	SOC stock (Mg ha ⁻¹)	29	29	9	43	1.0	27	-0.3	-0.5
Exclosure	Organic carbon (%)	2.8	2.8	1.5	3.8	0.1	18	-0.4	0.1
	Bulk density (kg m ⁻³)	1220	1308	690	1491	30	18	-0.9	-0.1
	SOC stock (Mg ha ⁻¹)	24	23	8	52	1.0	39	0.7	0.6

Appendix Table 2. Summary of statistics for total SOC, bulk density and SOC stock in relation to soil depth

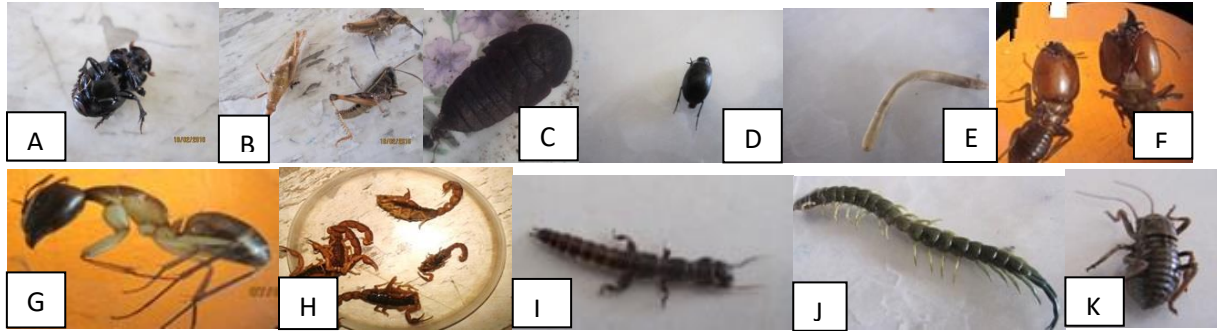
Depth	Variable	Mean	Median	Minimum	Maximum	Std error	CV	Skewness	Kurtosis
0-15 cm	Organic carbon (%)	2.8	2.8	1.3	3.8	0.1	20	0.0	-0.5
	Bulk density (kg m ⁻³)	1258	1275	740	1579	17	14	-1.0	1.0
	SOC stock (Mg ha ⁻¹)	24.2	23.2	5.8	52	0.1	38	0.3	-0.3
15-30 cm	Organic carbon (%)	2.3	2.3	0.7	3.7	0.1	26.2	0.4	0.0
	Bulk density (kg m ⁻³)	1247	1260	690	1483	17	14	-0.7	0.1
	SOC stock (Mg ha ⁻¹)	20	21	3	42	1.0	43	0.2	-0.4

Appendix Table 3. Summary statistics on analysis of MBC (mg kg⁻¹ soil) in response to SWC measures

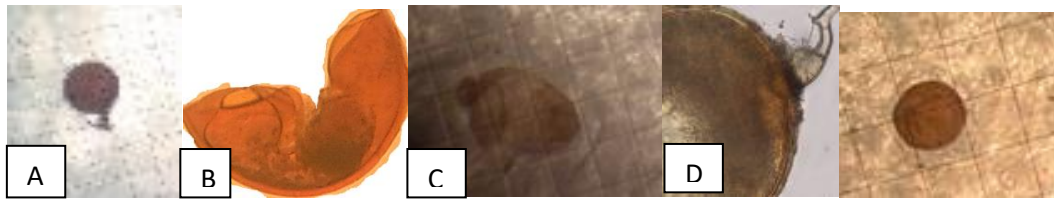
SWC	Mean	Median	Minimum	Maximum	Std Error	CV	Skewness	Kurtosis
Non conserved grazing lands	370	370	210	460	10	15	-1	1
Terraces	440	450	170	790	30	35	0.6	0.8
Exclosures + terraces	560	520	250	1250	40	38	1	2.5
Exclosures without terraces	640	630	210	1250	50	43	0.3	-0.5

Appendix Table 4. Sample analysis of variance (ANOVA) for some soil properties

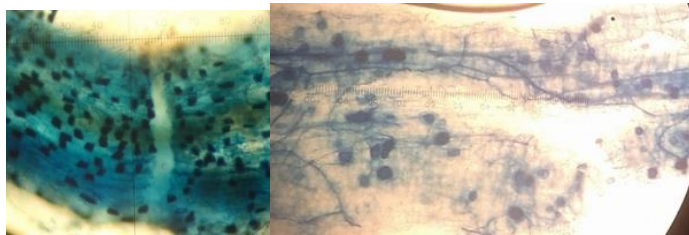
Soil properties	Source	DF	SS	MS	F Value	Pr > F
SOC	depth	1	5.0	5.00	59.72	<.0001
	slope	2	0.18	0.09	1.09	0.34
	SWC	3	9.84	3.28	39.15	<.0001
	SWC *depth	2	0.26	0.13	1.57	0.21
	SWC *depth	3	0.18	0.06	0.73	0.53
	SWC *slope	6	0.57	0.09	1.14	0.35
	SWC *slope*depth	6	0.13	0.02	0.26	0.95
Bulk density	depth	1	3096.77	3096.77	0.13	0.72
	slope	2	1997.95	998.97	0.04	0.95
	SWC	3	43248.34	14416.11	0.60	0.61
	slope*depth	2	3313.02	1656.51	0.07	0.93
	SWC *depth	3	39878.15	13292.71	0.55	0.64
	SWC *slope	6	143929.57	23988.26	1.00	0.43
	SWC *slope*depth	6	31100.11	5183.35	0.22	0.96
SOC stock	depth	1	30786.18	30786.18	8.23	0.0061
	slope	2	609.12	304.56	0.08	0.92
	SWC	3	152533.96	50844.65	13.60	<.0001
	slope*depth	2	2505.42	1252.71	0.34	0.71
	SWC *depth	3	2944.83	981.61	0.26	0.85
	SWC *slope	6	15172.04	2528.67	0.68	0.66
	SWC *slope*depth	6	6823.92	1137.32	0.30	0.93
SMI abundance	depth	2	25128369.33	12564184.67	32.86	<.0001
	season	1	37739224.01	37739224.01	98.69	<.0001
	SWC	3	1844207.60	614735.87	1.61	0.23
	depth*season	2	19405620.78	9702810.39	25.37	<.0001
	SWC *depth	6	6781733.78	1130288.96	2.96	0.05
	SWC *season	3	1905928.04	635309.35	1.66	0.22
	SWC *depth*season	6	6256923.67	1042820.61	2.73	0.06
Simpson (1_D) index	depth	2	0.15	0.07	3.23	0.07
	season	1	0.00	0.001	0.05	0.83
	SWC	3	0.13	0.04	1.88	0.18
	depth*season	2	0.02	0.01	0.45	0.65
	SWC *depth	6	0.15	0.02	1.07	0.43
	SWC *season	3	0.17	0.05	2.42	0.11
	SWC *depth*season	6	0.25	0.04	1.71	0.20
Shannon diversity index	depth	2	1.06	0.53	7.05	0.009
	season	1	0.02	0.02	0.29	0.59
	SWC	3	0.57	0.19	2.53	0.10
	depth*season	2	0.05	0.02	0.36	0.70
	SWC *depth	6	0.54	0.09	1.20	0.36
	SWC *season	3	0.45	0.15	2.01	0.16
	SWC *depth*season	6	0.72	0.12	1.60	0.22



Appendix figure 1. Some common representatives of soil macroinvertebrates (A: Scarabidae, B: Acrididae, C: Ligidae, D: Carabidae, E: Lumbricidae, F: Termitidae, G: Formicidae, H: Scorpionidae, I: Staphilinidae, J: Cryptopidae, K: Gryllidae)



Appendix figure 2. Some representatives of Arbuscular mycorrhiza spores (A: Glomus, B: Acaulospora, C: Gigaspora, D: Scutellospora E: Entrophospora)



Appendix figure 3. Arbuscular mycorrhiza fungi root colonization