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Above- and belowground carbon stocks in semi-arid land-use systems under integrated watershed management in Gergera watershed, Ethiopia

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Dedicated to
My parents who are the reason for who I am today!

ABSTRACT

Enhanced carbon stocks help to improve the productivity and resilience of farming systems, especially in smallholder communities relying on subsistent agriculture. This study investigated the total terrestrial stock of organic carbon and its controlling factors in prevalent land-use systems in the Gergera watershed, northern Ethiopia, as part of the impact assessment of the integrated watershed management (IWM) program introduced in the region.

Land-use and land-cover change (LULC) over 20 years (1994-2014) were analyzed using Landsat remote sensing imagery and a random forest algorithm. Above- and belowground biomass and soil were sampled from four major land-use systems, i.e. exclosures, croplands, rangelands and bare land. The soil samples collected at four slope positions, i.e. ridge, backslope, footslope and valley bottom and from four depth intervals (0-15, 15-30, 30-60 and 60-100 cm) were analyzed for organic carbon (SOC), bulk density, rock fragment, and other physical and chemical parameters. Soil cesium-137(^{137}Cs) was analyzed to trace the pattern of SOC distribution in the watershed.

The LULC change analysis indicates an improved vegetation cover since the adoption of IWM due to conversion from cropland to forest land and from bare land to rangeland on 3.3% and 6.3 % of the watershed area, respectively. Reduced vegetation cover is also observed due to changes of cropland to bare land and forest land to rangeland on 3.5% 5.7% of the area.

The field survey revealed significantly higher aboveground carbon stock in the plant biomass of exclosures (9.08(±1.44) Mg C ha^{-1}) followed by croplands and rangelands with 3.16(±0.24) Mg ha^{-1} and 1.49(±0.18) Mg ha^{-1}, respectively. The belowground biomass carbon content is particularly low in the croplands (0.76(±0.09) Mg ha^{-1}), exceeded by that in the exclosures and rangelands where values average 3.67(±0.06) Mg ha^{-1} and 3.16 (±0.39) Mg ha^{-1}, respectively. The total terrestrial carbon stocks differ according to the land use systems in the ranked order of exclosures (55.6.11(±4.89) Mg ha^{-1}) $\approx$ rangelands (53.77(±4.4) Mg ha^{-1}) > croplands (31.69(±3.99) Mg ha^{-1}) $\approx$ and bare land (35.52(±6.47) Mg ha^{-1}). Besides the land use type, the SOC stock in the examined land-use systems is found negatively related to the content of coarse fragments and bulk density of the soil, which both measured the highest values in croplands and exclosures. Topsoils had greater SOC in all land-use systems but the deeper soils (30-100cm) still contained 36 % of the SOC stock.

The pattern of ^{137}Cs distribution in the watershed generally indicates the presence of erosion, mostly on backslopes of exclosures and rangelands. However, the positive significant correlation between ^{137}Cs and SOC distribution in exclosures points at a build-up of SOC.

The overall results of the study highlight that more efforts in application of improved soil management practices are still required to enhance the current status of the SOC pool, particularly in the croplands, and thereby sustain the land productivity.

Ober- und unterirdischer Kohlenstoffbestand in halbtrockenen Landnutzungssystemen bei integriertem Management im Wassereinzugsgebiet Gergera, Äthiopien

Kurzfassung

Erhöhte Kohlenstoffbestande verbessern die Produktivität und Belastbarkeit der landwirtschaftlichen Systeme insbesondere in kleinbäuerliche Gemeinschaften, die von der Subsistenzwirtschaft abhängen. Diese Studie untersucht den terrestrischen Gesamtkohlenstoffbestand und die beeinflussenden Faktoren in den vorherrschenden Landnutzungssystemen im Wassereinzugsgebiet Gergera, Nordäthiopien als Teil einer Bewertung der Effektivität des in der Region eingeführten integrierten Managementprogramms (IWM).

Hierzu wurden die Veränderungen der Landnutzung und Landbedeckung (LULC) über 20 Jahre (1994-2014) mit Hilfe von Landsat Satellitenaufnahmen und eines Random Forest Algorithmus analysiert. Über- und unterirdische Biomassen- sowie Bodenproben wurden in den vier wichtigsten Landnutzungssystemen genommen, d.h. in von Landnutzung ausgeschlossenen Flächen (exclosures), auf landwirtschaftlichen Anbauflächen (cropland), offenen Weideflächen (rangelands) und vegetationsfreiem Land (bare land) an jeweils vier Hangpositionen, d.h. Bergrücken (ridge), Mittelhang (backslope), Hangfuß (footslope) und Talboden in jeweils vier Bodentiefen (0-15, 15-30, 30-60 und 60-100 cm). Die Bodenproben wurden auf organischem Kohlenstoff (SOC), Bodendichte, Mengen an groben Bestandteilen und weiterer Parameter analysiert. Außerdem wurde in einer Boden-Cäsium-137 (^{137}Cs)-Analyse die Verteilung des SOC im Einzugsgebiet bestimmt.

Die LULC-Analyse identifiziert eine Verbesserung in der Vegetationsbedeckung seit der Anwendung des IWM Programms durch Umwandlung von Anbauflächen in Waldflächen bzw. vegetationsfreie Flächen in offene Weideflächen auf 3.3% bzw. 6.3% des Einzugsgebiets. Eine Abnahme der Vegetationsbedeckung von Anbauflächen zu vegetationsfreien Flächen auf 3.5% sowie von Forstflächen zu offenen Weideflächen auf 5.7% des Einzugsgebietes wurde außerdem beobachtet.

Die statistische Analyse zeigte einen signifikant höheren oberirdischen Kohlenstoffbestand in den Exclosures (9.08(±1.44) Mg C ha^{-1}) gefolgt von Anbauflüchen (3.16(±0.24) Mg ha^{-1}) und offenen Weideflächen (1.49(±0.18) Mg ha^{-1}). Der Kohlenstoffbestand in der unterirdischen Biomasse in den Anbauflächen war besonders niedrig (0.76(±0.09) Mg ha^{-1}). Die durchschnittlichen Werte waren höher in den Exclosures (3.67(±0.06) Mg ha^{-1}) bzw. offenen Weideflächen (3.16 (±0.39) Mg ha^{-1}). Der Gesamtkohlenstoffbestand unterschied sich abhängig von Landnutzung in der Reihenfolge Exclosures (55.6. 11(±4.89) Mg ha^{-1}) ≈ Weideflächen (53.77(±4.4) Mg ha^{-1}) ≈ vegetationsfreie Flächen (35.04(±6.47) Mg ha^{-1}) > Anbauflächen 31.69(±3.99) Mg ha^{-1}. Neben Landnutzung besteht ein Zusammenhang zwischen SOC-Bestand und groben Bestandteilen bzw. Bodendichte. Beide Werte waren am höchsten in den Anbauflächen und Exclosures. Der SOC-Bestand korrelierte negativ mit der Menge an groben Bestandteilen. Der Oberboden wies höhere SOC-Werte in allen Landnutzungssystemen aus aber die tieferen Schichten (30-100 cm) enthielten immer noch 36% des SOC.

Das Muster der ^{137}Cs-Verteilung weist im Allgemeinen darauf hin, dass das Einzugsgebiet von Erosion betroffen ist, meistens auf dem Mittelhang der Exclosures und offenen Weideflächen. Jedoch ist die Korrelation zwischen ^{137}Cs-Verteilung in den Exclosures ein Hinweis auf SOC-Aufbau.

Die Ergebnisse dieser Studie zeigen, dass größere Anstrengungen bei der Umsetzung von Bodenmanagementmaßnahmen erforderlich sind, um den aktuellen Status des SOC-Pools zu verbessern, insbesondere auf den Anbauflächen, und dadurch die Flächenproduktivität aufrecht zu erhalten.

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LIST OF ABBREVIATIONS AND ACRONYMS

AGB	Aboveground biomass
ANOVA	Analysis of variance
BD	Bulk density
BGB	Belowground biomass
CD	Crown diameter
CEC	Cation exchange capacity
CO_2	Carbon dioxide
CV	Coefficient of variation
DBH	Diameter at breast height
IWM	Integrated Watershed Management
LLD	Lower limit of detection
LME	Linear Mixed Effect
LULCC	Land use and land cover change
MSE	Mean square error
R:S ratio	Root to shoot ratio
SOC	Soil organic carbon
SOM	Soil organic matter
UNCCD	United Nations Convention to Combat Desertification
UNEP	United Nations Environment Program
WB method	Walkley Black method

1 INTRODUCTION

1.1 Background

1.1.1 Dryland degradation

Different definitions of land degradation exist depending on the context of a problem. Considering a widely accepted definition by the United Nations Convention to Combat Desertification (UNCCD, 1996):

> *"Land degradation means reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as: (i) soil erosion caused by wind and/or water; (ii) deterioration of the physical, chemical and biological or economic properties of soil; and (iii) long-term loss of natural vegetation".*

Degradation of drylands (i.e. desertification) is an important concern because they cover 47% of the world's terrestrial surface. These areas are characterized by a shortage of rainfall, which is also erratic in nature, significantly affecting plant productivity and vegetation cover in agricultural and forest ecosystems (Sharma *et al.,* 2012).

Land-use and land-cover (LULC) change due to deforestation, subsequent expansion of agricultural lands and settlements (Hurni *et al.,* 2010; Yin *et al.,* 2014) accounts for 33% of the increase in the CO_2 concentration in the atmosphere (Sharma *et al.,* 2012) due to the deterioration of the vegetation cover (Biro *et al.,* 2011; Tolo *et al.,* 2012; Hurni *et al.,* 2010) and consequent loss of organic matter (FAO, 2004). Overall, vegetation loss and land degradation comprises 10–15 Pg of carbon emissions from the vegetation pool (Lal *et al.,* 1999).

The largest carbon pool in terrestrial ecosystems is found in soils, where the carbon stock is more than three times higher than that in all the biotic components (Zhang *et al.,* 2013). Land degradation due to soil erosion and vegetation loss, especially in drylands, has resulted in the loss of soil organic carbon (SOC), which is the basis of land fertility (Collins *et al.,* 2001; UNEP*,* 2008; Runólfsson and Arnalds, 2004). Land degradation in the drylands has been aggravated by their low plant productivity coupled with high temperatures, which caused a historic global carbon loss of 20–30 Pg C (Lal, 2003). Of this, the SOC loss was estimated to be 9–14 Pg (Lal *et al.,* 1999), leading to plant nutrient exhaustion, biodiversity loss and reduction in plant water availability (Lal, 2009).

Unsustainable cropping practices are responsible for the low SOC content in the croplands (Fang *et al.,* 2012; Oades, 1998; Bationo *et al.*, 2006; Gabathuler *et al.,* 2009 and Gelaw *et al.,* 2014). Generally, degraded croplands in dryland areas contain less than 1% SOC (Gabathuler *et al.,* 2009). However, despite their level of degradation, the agricultural areas in drylands provide much of the world's grain and livestock production and are expected to increase productivity in the coming decades to satisfy the ever increasing demand for food and shifts in diets (Sharma *et al.,* 2012). Indigenous intensification of the smallholder farming practices such as short or no fallowing periods, intensive weeding and use of marginal lands for crop production have become common to sustain food security in these areas (FAO, 2004). Further improvements in cropland management are needed to increase productivity without further degrading the land (Branca *et al.,* 2011) and to rehabilitate already degraded cropland soils through carbon sequestration.

The majority of arid areas is used for grazing (Lal, 2004) which is thus even more important factor of dryland degradation. Excessive and uncontrolled grazing practice has resulted in desertification through soil fertility mining (Lal, 2003). Alkemade *et al.* (2013) also observed biodiversity loss due to the intense utilization of drylands for livestock production and due to the conversion to croplands. Improved grazing land management, however, was proved to enhance grazing land productivity in the drylands. The practices included introduction of legume fodder species, controlled grazing, low stocking rate, erosion control and agroforestry (Lal, 2003 and 2009), as well as the introduction of rotationary grazing system into pastures that are otherwise continuously grazed (Kotze *et al.*, 2014). All these measures can enhance carbon sequestration and climate change mitigation (Dean *et al.,* 2015).

1.1.2 Dryland degradation in Ethiopia

In Ethiopia, dryland degradation is one of the major causes of food insecurity and rural poverty (Zeleke, 2010). The main causes are complex, and although the opinions regarding the reasons for the degradation differ (Mitiku *et al.,* 2006), deforestation, overgrazing and cropland expansion at the expense of forest lands are likely of greatest importance (Jolejole-Foreman *et al.,* 2012). Shiferaw *et al*. (2013) summarized the very diverse and complex causes of land degradation in Ethiopia into five main categories. These include (i) the rugged topographic features and steep slopes in a large part of the country, and torrential rainfalls which occur within a very short period of the year; (ii) poor farming practices such as mono-cropping, removal of crop residue, low agricultural input; (iii) dependence on subsistence agriculture; (iv) insecure land tenure system; and (v) limited scientific knowledge and agricultural extension services.

The long history of droughts is one of the reasons that exacerbated the land degradation and reduced the buffer capacity of the vegetation to overcome the drought periods (Haregeweyn *et al*., 2007). Soil erosion through water and wind is also a deep rooted and complex problem in Ethiopia. It is triggered by the geomorphologic features of the landscape (Hancock *et al.,* 2010), which is characterized by steep slopes, and also by erosive rainfall and human impacts through misuse of natural resources (Nyssen *et al.,* 2004). Biodiversity loss, accelerated erosion and loss of fertile soils also have been observed following deforestation of steep slopes (Birhanu, 2014; Hurni, 1988). The rugged topographic features coupled with erosive rainfall increase the chance of SOC losses and redistribution (Hancock *et al.,* 2010). Nyssen *et al*. (2004) stressed that terrain deformation or mass movements were among the main land degradation processes induced by water erosion. The resulting loss of fertility due to sheet, rill and gully erosion has led to a 3.4% loss of land productivity (Hawando, 1997). However, Nyssen *et al.* (2001) indicated that despite the steepness of the slopes, the lands in Ethiopia show resistance to gully erosion due to deep soils with considerably high amounts of rock fragments.

Water erosion of cropland soils resulted in an average soil loss of 42 t ha^{-1} yr^{-1} mainly of the fertile topsoil (Nyssen *et. al*., 2003a) with values ranging from 18.8 t ha^{-}

^{1} yr^{-1} to 136 t ha^{-1} yr^{-1} (Hurni, 1990). Besides, this soil erosion by water is also accountable for the loss of about 30 kg ha^{-1} yr^{-1} of nitrogen, 200 kg ha^{-1} yr^{-1} of SOM and 15-75 kg ha^{-1} yr^{-1} of phosphorus (Hawando, 1997), all essential for plant production. Soil erosion does not only result in on-site soil degradation, but also leads to off-site problems related to downstream sedimentation as well as surface and groundwater pollution. The extent of the damage depends on the severity of the erosive nature of the rainfall especially in areas with rugged topographic features (Hancock *et al.,* 2010).

Mulugeta and Fisseha (2004) studied the impact of LULC change and found a reduction in SOC stocks as a result of deforestation and subsequent cultivation of lands. Nyssen *et al*. (2004) and Haregeweyn *et al*. (2007) also stated deforestation and agricultural land expansion as determinants of land degradation in Ethiopia. There have been numerous reports regarding the rate of deforestation and the extent of remaining forest cover in the country. Mitiku *et al.* (2006) estimated that deforestation and forest degradation resulted in a loss of 62,000 ha yr^{-1} of forests and woodlands, leaving only 2.2% out of the original 65% of forest cover in Ethiopia. Hawando (1997) reported a higher deforestation rate of 150,000–200,000 ha yr^{-1} and Sutcliffe (2006) estimated even higher loss of 233,340 ha yr^{-1} of forest, woodlands and shrublands. Berry (2003) has reported that the current forest cover of the country is only 3% in contrast to a 40% forest cover a century ago. However, Moges *et al*. (2010) has indicated that about 50% of Ethiopia is covered by woody vegetation that includes dense forests. The most recent review, by Shiferaw *et al*. (2013), has suggested that overall 50 Mha of Ethiopian land, which is 45% of the total land mass in the country, is degraded and depleted in SOC at a rate of 0.02-0.97 Mg ha^{-1} yr^{-1}. The loss of biomass carbon poses a huge constraint to agricultural production and food security (Amde *et al.,* 2001; Gashaw *et al.,* 2014).

In developing countries such as Ethiopia, that has the largest livestock population in Africa, communal grazing lands have been the most important sources of livestock feed (Benina and Pender, 2002; Gebremedhin *et al.,* 2000). The Ethiopian highlands cover about 44% of the total area of the country (Hawando, 1997) and support about 80% of the tropical livestock units or 75% of the livestock population and the whole poultry population (Birhanu, 2014; Hawando, 1997). The unrestricted access to the very limited communal lands has been resulting in overexploitation of this land-use system and desertification (Benina and Pender, 2002). Moreover, despite the shortage of feed for the livestock, the interest of the farmers to increase the size of their herds has led to increased overgrazing and is a challenge for the grazing land management activities (Birhanu, 2014).

Unsustainable management of cultivated lands and short fallow periods were also found to reduce soil aggregate stability and lower the soil fertility status (Assefa and Keulen, 2009; Tesfahunegn, 2011; Tilahun and Asefa, 2009). Compared to agroforestry-based crop production, open communal pasture, silvopasture and irrigation-based fruit production, the soils under rainfed agriculture had the lowest SOC stocks. This could be due to the extractive nature of crop cultivation (Gelaw *et al.,* 2014). Besides, 30% of the agricultural lands are not fertilized (Shiferaw *et al.,* 2013), and other significant areas of cultivated land receive fertilizers and other agricultural inputs far below the recommended rates. Haile *et al*. (2014) observed high bulk density and low SOC in cereal-cropped lands, presumably due to the destruction of the soil aggregates and exposure of the SOM to decomposition during soil tillage (Fontaine *et al.,* 2007). Gelaw *et al*. (2014) also mentioned heavy dependence on the natural resources in the low-input subsistence agriculture, which is the main livelihood strategy and the land tenure insecurity as socio-economic causes of land degradation.

1.1.3 IWM activities in Ethiopia

Aiming to reverse land degradation resulting from water erosion and to contribute to household food security in the Tigray region in the northern part of Ethiopia, the regional government along with NGOs has been implementing physical soil and water conservation structures such as hillslope terraces and soil bunds for more than three decades, albeit with a variable success. Mitiku *et al*. (2006) has stated that the top-down approach in the agricultural extension services, insufficient attention to awareness creation, and the main focus on the physical conservation measures rather than on social, cultural and economic circumstances, are the main reasons for the ineffectiveness of these conventional physical soil and water conservation measures.

To improve the effectiveness of the physical soil and water conservation measures, the integrated watershed management (IWM) approach was introduced in 1991 (Mekuria *et al.,* 2007). IWM is the formulation and application of an integrated series of actions designed to secure sustainable development throughout a catchment (Fernández, 2008), and is among the approaches to reclaim degraded drylands. In contrast to the conventional measures, the IWM activities have been addressing the whole series of challenges occurring in a specific watershed by incorporating various rehabilitation activities to tackle problems of land degradation and thus food insecurity (Ann *et al.,* 2006; Fikir *et al.,* 2008; Fernández, 2008; Förch, 2009).

IWM strategies included the establishment of exclosures on steep, eroded slopes mainly of forest lands and in other areas that were considered as a source of sediments (Descheemaeker *et al.,* 2006; Haregeweyn *et al.,* 2007). With the aim of natural vegetation restoration, the exclosure areas are guarded to prevent deforestation and grazing (Haregeweyn *et al.,* 2007). The rehabilitation activities implemented in all the land-use systems also include gully rehabilitation using gully-side stabilization with gabion walls, gully check dams and gully plantation, farm ponds for rainwater harvesting for agricultural production and household consumption, application of hillside terracing, contour soil and stone bunds, deep trenches, and bund planting to control soil and water erosion (Hadush, 2015; Kumasi and Asenso-Okyere, 2011; Haregeweyn *et al.,* 2007). Diversification of economic activities (Gashaw, 2015) such as beekeeping and poultry production, a cut-and-carry system on rangelands and in-situ moisture conservation on croplands are also part of the versatile IWM activities (Fikir, 2005). The adoption of such activities in a multi-sectoral approach tackled problems related to water, soil, and energy and vegetation resources in such a way that balanced the human and environmental needs (Förch, 2009). The success of these IWM practices is dependent on the participation of the local communities throughout the different stages of the activities and requires trust building (Descheemaeker *et al.,* 2010; Chisholm and Woldehanna, 2012).

In the study area, the Gergera watershed, one of the most degraded areas in the Tigray region of northern Ethiopia, IWM activities have been conducted since 1998

(Fikir *et al.,* 2008). Following the interventions, a number of studies assessed the effectiveness of the IWM approach with respect to mitigating land degradation and improving water productivity in the region (Fikir *et al.,* 2008; Descheemaeker *et al.;* 2010; Förch, 2009; Zeleke*,* 2010). The impacts have been categorized into improvements in water resource availability, soil quality, vegetation cover and socio-economic status (Förch, 2009). Depending on the age of the exclosures, a visual estimation of the vegetation ground and canopy cover showed a clear difference in the cover type composition (Damene *et al.,* 2013). Mekuria *et al.* (2007) also reported natural vegetation restoration in exclosures due to increased soil organic matter (SOM), total nitrogen (TN) and available phosphorus (AP) in the upper 0-15 cm depth. Giday *et al.* (2008) concluded the success of vegetation restoration in exclosures based on an increased supply of fuelwood by 2.3 Mg ha^{-1} yr^{-1}. A LULC analysis for 1994-2005 by Fikir (2005) showed a positive impact of the exclosures judged by the increased area coverage of the forest lands and increased vegetation cover in different parts of the watershed. However, LULC analysis should be performed with a higher temporal resolution in order to capture the changes in vegetation dynamics that result from changes in land management over the evaluation period.

Increased recharge in the subsurface water also occurred, which created the opportunity for irrigation practices thus contributing to poverty alleviation and improved livelihoods (Förch, 2009; Fikir *et al.,* 2008). Tesfahunegn *et al.* (2011) revealed that despite the sloping terrain, IWM practices such as terracing, land-use redesign, grassed waterways and gully stabilization structures on cropland areas improved the physical and chemical soil qualities including soil texture, resistance to erosion and soil fertility, which consequently led to improved crop yields. Positive effects were also reflected by gully and river bank stabilization and rehabilitation of degraded rangelands. These improvements in the natural resources of the different intervention areas such as gullies, degraded hill slopes, rangelands and croplands greatly contributed to the improved livelihood status of the local communities through alternative income sources, improved productivity, and increased production portfolio (Fikir *et al.,* 2008). However, these studies often indicating the success of IWM measures in Tigray, were generally conducted for specific land-use systems, or using either SOC or vegetation cover as sole indicators. Moreover, the studies on SOC only focused on topsoils rather than the whole soil profile. To-date, no study has investigated the belowground biomass carbon stocks in the region.

1.1.4 Carbon sequestration for land rehabilitation

Despite carbon losses due to natural and anthropogenic processes, there is still a huge potential for carbon sequestration through degraded land restoration by restoring depleted organic carbon (Thuille and Schulze, 2006; Lal, 2003) in agricultural and forest areas. Agriculture is the main source of livelihood in the semi-arid regions of Africa, where rainfed crop production and pastoralism are the main practices (Perez *et al.,* 2007). The rainfed agricultural lands are the most carbon-devoid areas due to poor biomass productivity, resulting from the low-input, subsistence farming. In these areas, improved productivity and carbon sequestration relies on agroforestry practices (Albrecht and Kandji, 2003), as trees contribute to crop nutrition and act as a carbon sink above- and belowground (Montagnini and Nair, 2004). Moreover, agroforestry indirectly contributes to enhanced carbon pools through reduced pressure on the natural forests which would have been otherwise harvested for timber and fuelwood. Beyene *et al.* (2013) assessed the suitability of community-managed forests under the REDD initiative as very promising for CO_2 emission reduction in Ethiopia. Reduced tillage practices and improved management of rangelands such as controlled grazing and limited stocking rate were found to enhance SOC due to reduced soil disturbance (Liu *et al.,* 2011). In general, degraded land restoration, conversion to an appropriate land use and application of more sustainable management practices for cropland and rangelands, such as crop rotation, integrated nutrient management and conservation tillage in the dryland areas, has a carbon sequestration potential of 1 Pg C yr^{-1} (Lal, 2004).

The importance of SOC sequestration is not only in its contribution to climate change mitigation but also in its impact on soil fertility. SOC is the most important indicator of soil quality especially in agricultural soils because of its influence on the major soil physical and biological properties that can affect soil productivity (Chan, 2010). In particular, enhancing SOC stocks in drylands where soils have weak structures and low vegetation cover (Lal, 2009) can simultaneously help to improve the land productivity, to achieve the goal of alleviating rural poverty and thus food security and improve the livelihoods of the communities (FAO, 2004; Sharma *et al*, 2012).

However, the lack of information and empirical data on SOC sequestration has posed a challenge for accurate predictions of these impacts (Stringer *et al.,* 2012). Watson *et al*. (2013) indicated huge uncertainties regarding the amounts of carbon sequestered due to inaccurate data that was based on biome-scale generalizations. This study revealed an underestimation of about 63% and 58% of carbon stocks in the Ethiopian Bale mountain humid and dry forests, respectively. Strengthening the availability of accurate and site-specific database development in the country would benefit national natural resources management planning and enhance the country's participation in international carbon trading projects while improving the land productivity.

1.1.5 Factors impacting SOC dynamics

Understanding the global carbon cycle requires understanding the dynamics of each carbon pool. However, the size of global carbon pools and sources are uncertain due to the complex nature of the factors that can affect them. Baccini *et al*. (2012) has indicated that the uncertainties arise from lack of adequate data on the global aboveground biomass of forests and on regional deforestation rates. The use of different sources of data and incomplete inventory of carbon at regional and global scale has significantly contributed to the poor information quality (Fang *et al.,* 2012).

Biological processes in terrestrial ecosystems (Heimann & Reichstein, 2008) largely control the release and absorption of the most important greenhouse gases, carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), which have a profound effect in the global climate change. Carbon dioxide, with the largest atmospheric concentration among the greenhouse gases, is stored in or emitted from the vegetation and soils depending on the input-output balance (Schlesinger, 1990; Liu *et al.,* 2011) determined by the biological processes of addition of organic matter (OM) or its decomposition, respectively. The carbon stored both in the biomass and soils have different levels of stability (Lal R, 2005; Bohre *et al.,* 2012), as soils are considered to be a relatively stable carbon pool while the plant biomass is the main source of OM (Grimm *et al.,* 2008; Silver *et al.,* 2000; Bationo *et al.,* 2006; Oades, 1998).

The main factors that control the soil and biomass carbon storage differ according to (agro)-climatic zones and characteristic vegetation type and land management practice. For example, in semi-arid areas of China, the total SOC and biomass carbon storage of forest lands was found to be mainly dictated by the stand age due to biomass increases in growing stands (Wang *et al.,* 2015). Zhao et al. (2014) also found that young forest biomass of 62.4 Mg C ha^{-1} increased to 177 Mg C ha^{-1} in a mature forest. Studies are less conclusive regarding the root biomass stocks and consequent belowground C inputs, but Wang *et al*. (2015) found that the root biomass increased with stand age, resulting in carbon accumulation. Hulvey *et al*. (2013) indicated that multi-species forests had the potential to sequester more carbon than single-species forests, and that tree species composition was more important than the species richness for carbon sequestration.

Soil temperature, soil fauna (particularly termite activity), and soil clay content were named the main factors that control the turnover rate of OM in soils of cultivated lands in West African agro-ecosystems (Bationo *et al*., 2006). In particular, higher soil temperature and activities of termites increased the soil organic matter (SOM) turnover and clay content, resulting in the loss of SOC (Bationo *et al.,* 2006). However, based on incubation experiments on grassland soils, Fontaine *et al*. (2007) regarded temperature as a less important factor for the SOC dynamics in deep soil layers because soil disturbance and addition of fresh OM to the deeper soil horizon overpowered the impact of temperature on SOC turnover. Oades (1998) also revealed that in well-drained, deep grassland soils the SOC turnover was modified by fresh OM addition along the soil profile, but soil temperature and interaction of carbon with clay particles were still observed as the main controlling factors of SOC turnover. The addition of fresh OM to deeper soils may result in activation of the microbial activity through supply of oxygen and energy to the microorganisms, which in turn results in a higher decomposition of the SOC pool in deep soil which has been stored for a longer time (Oades, 1998; Fontaine *et al.,* 2007; Schmidt *et al.,* 2011). In other words, the land-use practices that increase soil disturbance such as crop cultivation can result in addition of fresh OM to the deeper soils and may facilitate the loss of SOC (Liu *et al.,* 2011; Schulp and Verburg, 2009). Studies are less consistent regarding the impacts of soil clay content on SOC build-up. A review by Oades (1998) revealed that higher clay content was very important for retaining the SOC that can be otherwise removed from the soil. In same vein, Bationo *et al.* (2006) indicated that low clay content resulted in the loss of SOC. In contrast, McLauchlan (2006) observed no relationship between SOC and clay content of the soil on grassland abandoned from crop cultivation, despite a significant positive correlation between clay and soil aggregate size. Wells *et al.* (2012) made same observation on undisturbed soils of grassland. Total nitrogen and bulk density were found to be strongly related to SOC although the relationship differed from site to site depending on the type of land management (Wang *et al.,* 2001; Grimm *et al.,* 2008).

The content of coarse fragments, i.e. soil particles larger than 2 mm in diameter, is among the factors that can affect the SOC stock (Throop *et al.,* 2012). The availability of rock fragments on soil surface was found favorable for water retention and for reduction in raindrop erosion and evaporation (Nyssen *et al.,* 2002; Novak and Surda, 2010). Based on laboratory simulations, Zavala *et al.* (2010) found that the higher the rock fragment fraction the lower soil erosion and runoff generation. Poesen and Lavee (1994) also concluded that rock fragments could positively impact on soil productivity directly through the modification of soil properties, such as soil erodibility and soil temperature by enhancing soil cover and water holding capacity. Therefore, all other factors being constant, soils with high rock fragment content are able to support more biomass and higher crop productivity particularly in drylands characterized by moisture shortages and high water erosion. However, this is a reverse situation in humid climates where higher productivity was observed in fine-textured, non-rocky soils (Poesen and Lavee, 1994).

Considering the influence of the topography, some studies showed that SOC was distributed following the catena definition of soil distribution (Grimm *et al.,* 2008).

However, Martinez (2010) showed that the distribution of SOC in the landscape depended on the level of soil disturbance, i.e. in areas with no soil disturbance no trend was observed in SOC distribution within the landscape.

1.1.6 Informational needs for the assessment of carbon stocks

In the assessment of IWM impacts, an accurate quantification of the above- and belowground biomass particularly of perennial plants is essential, as this pool secures long- and short-term carbon storage globally (Litton and Kauffman, 2008) and plays an important role in reducing erosion and nutrient leaching (Silver *et al.,* 2000). Generally, the level of organic carbon in various land-use areas is sensitive to multiple factors which cannot be generalized based on studies in similar contexts. However, due to the high diversity of woody species, most biomass estimations have relied on generalized allometric functions. Moreover, the belowground biomass carbon has been overlooked due to the considerable efforts involved in acquiring primary data through time-consuming and laborious excavation of trees and shrubs and perennial root systems of grasses. Root biomass is frequently estimated based on the general root to shoot ratio, an approach that lacks rigor (Nair *et al.,* 2009). Hence, accurate up-to-date methods and site-specific carbon accounting are crucial for proper planning of land restoration measures (Perez *et al.*, 2007; Alam *et al.,* 2013). Moreover, assessment methods that consider spatial and temporal dynamics of the biotic carbon pool (Stringer *et al.,* 2012) since the introduction of IWM are required, such as application of tools developed in rapidly growing domain of remote sensing. Monitoring LULC changes using satellite images helps to investigate the terrestrial carbon dynamics of a specific area. In particular, spatio-temporal LULC analyses that are supplemented with field-observed biophysical information such as soil properties (Biro *et al.,* 2011) can render a more precise evaluation of dryland degradation (FAO, 2004) and help to identify priority areas for rehabilitation and biodiversity conservation (Mochizuki and Murakami, 2012).

With regard to SOC status, many studies dealing with various LULC classes in Ethiopia and specifically in Tigray region only focused on the 30-cm topsoil (Gelaw *et al.,* 2014; Damene *et al.,* 2013; Mekuria *et al*, 2007, Tesfahunegn *et al.,* 2011; Limeneh *et* *al.,* 2004). The SOC stored below 30 cm is recognized as the most stable carbon and is estimated to account for 30-70% of the total SOC (Batjes, 2001; Guo *et al*., 2006a). Hence, ignoring the SOC stored deeper in the soils may lead to underestimation of the ongoing processes of degradation and rehabilitation. Therefore, the SOC assessments should be extended to a deeper depth for a more complete understanding of the ecosystem functioning and impacts of the soil and water conservation activities.

Information on the redistribution of SOC due to soil erosion and deposition (Hanckok *et al.,* 2010) in the watershed in different land-use types is needed for an assessment of IWM impacts at the landscape scale in Tigray. Knowing the effect of soil erosion in redistribution of SOC helps to gain empirical knowledge and devise relevant management activities that consider topographic variability in different land-use systems (Bot and Benites, 2005).

Various conventional techniques, ranging from a long-term erosion monitoring and measurement plot establishment to the use of erosion pins, have been used to quantify soil erosion (Collins *et al.,* 2001). In Ethiopia, monitoring of soil erosion processes have been undertaken by the Soil Conservation Research Program (SCRP) that started in 1981 (Hurni, 1982; Mitiku *et al.,* 2006). The long-term monitoring of erosin in the SCRP includes establishment on-farm test plots and hydrometric stations for river discharge and sediment yield (Mitiku *et al.,* 2006). These methods are time consuming and costly, and measurements may be subject to personal precision and operational challenges (Afshar *et al.,* 2010). The measurements of the radioactive isotope ^{137}Cs provide some advantages over the conventional methods (Mabit *et al.,* 2008). In particular, the isotopic method relies on one-off soil sampling for each sampling point unlike the conventional methods requiring continuous measurements. ^{137}Cs measurements enable assessing a medium-term erosion of up to 50 years (Mabit *et al.,* 2008; 2014). Determination of SOC and ^{137}Cs and their distribution along the soil depth may not always show similar patterns due to effects of land use and soil erosion processes which modify the distribution (Mabit *et al.,* 2008). Many studies also indicated that movement patterns of ^{137}Cs and SOC in the landscape (e.g., Ritchie and McCarty, 2003; Wells *et al.,* 2012) correlated strongly with high soil disturbance (Martinez, 2010), particularly in cultivated lands (Ritchie and McCarty, 2003; Wells *et al.,* 2012). This important relation makes it possible to assess the effects of soil erosion on SOC distribution along the topographic position. Such relationship is not observed in virginal lands with undisturbed soils. The SOC distribution in uncultivated soils is dictated by biological factors such as vegetative biomass and carbon oxidation and mineralization rather than by erosion (Hancock *et al.,* 2010). Schulp and Verburg (2009) also found the relative topographic position rather than the absolute elevation to affect SOC variability, and that soil type and SOC had the strongest association among the site factors. Hence, careful consideration is needed when applying ^{137}Cs data to predict SOC redistribution patterns in the landscape.

1.2 Research hypotheses

The general hypothesis of the study is that the introduction of the IWM program contributes to LULC change resulting in an increased forest cover after 1998, the year of the IWM introduction. In particular, higher carbon stocks in exclosures are expected given the higher woody biomass above- and belowground, and higher litter inputs. Significant SOC stocks should be found below the depth of 30 cm regardless of LULC class due to the stable nature of the subsoil SOC. Furthermore, higher SOC is expected in the footslope and valley bottom positions due to the effect of erosion.

1.3 Research objectives

This study aims to comprehensively investigate the total terrestrial organic carbon stock in prevalent land-use systems in the Gergera watershed, Tigray, Ethiopia, as part of the impact assessment of the effectiveness of the soil and water conservation program introduced in the region. Specifically this study aims to evaluate:

- The land-use and land-cover change following the introduction of the IWM program;
- The carbon concentration and stock above- and belowground, i.e. aboveground biomass carbon, root biomass carbon, and soil organic carbon pools in major land-use systems;

- The SOC distribution in the soil profile;
- The SOC redistribution along a topographic gradient due to soil erosion and deposition.

2 METHODOLOGY

2.1 Study site description

2.1.1 Location

Tigray is a regional state in northern Ethiopia covering an area of 53,386 km^2. The region is one of the most food-insecure areas in the country (WFP-Ethiopia, 2009) relying on agriculture as the main economic activity. About 20% of the total landmass of the region is suitable for agriculture of which 93% is already under cultivation.

The study was conducted in the Gergera watershed in Atsbi-Womberta district in the eastern part of Tigray. The watershed is located at $13^0$43’30” to $13^0$55’0” N and $39^0$38’30” to $39^0$47”30” E, covering an area of 143.9 km^2 (Figure 2-1). The Atsbi-Womberta district is among the most highly populated and food-insecure districts of the region. To tackle the food insecurity and land degradation problems, IWM activities have been implemented in the study area since 1998 (Fikir *et al.,* 2008).

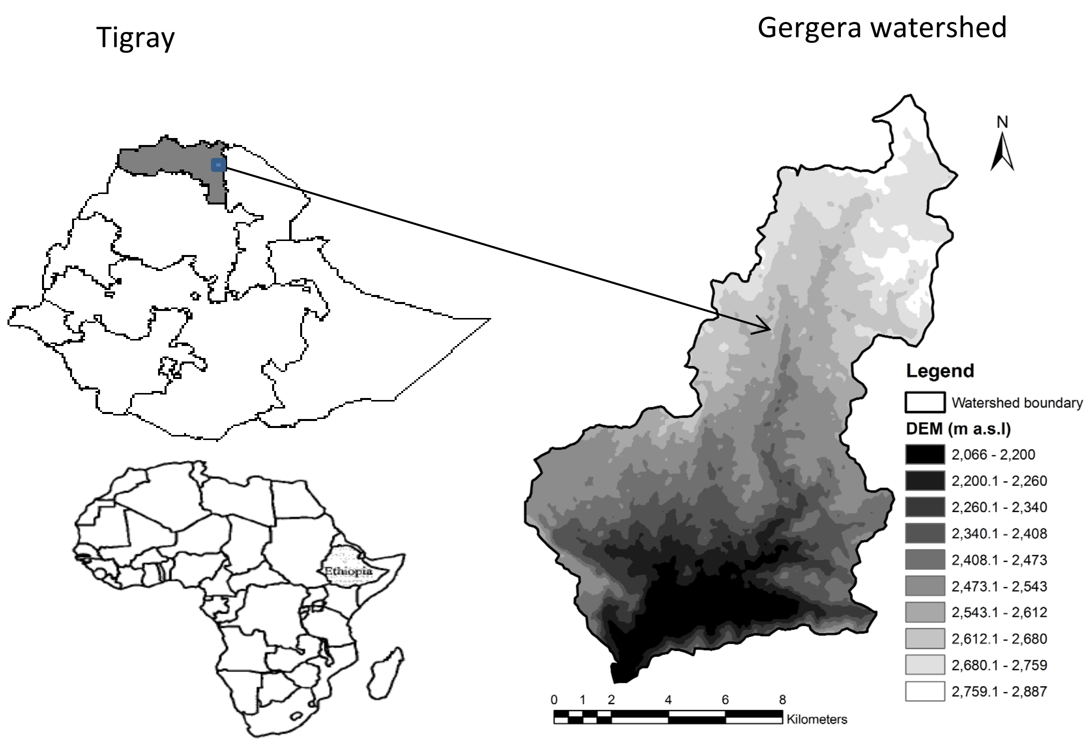

Figure 2-1 Location, boundary and digital elevation model (DEM) of the Gergera watershed

2.1.2 Climate

The area is characterized by a bimodal rainfall regime with peak rainfall from late June to early September and a short and less intense rainy season between March and April. The latter is usually used for cropland preparation. The peak rainfall season is the period when the crop cultivation takes place (Rahel, 2008). The average annual rainfall of the area is 610 mm (Berhane, 2008). Annual temperatures range between 15 and 20°C, averaging 17.4°C (Figure 2-2).

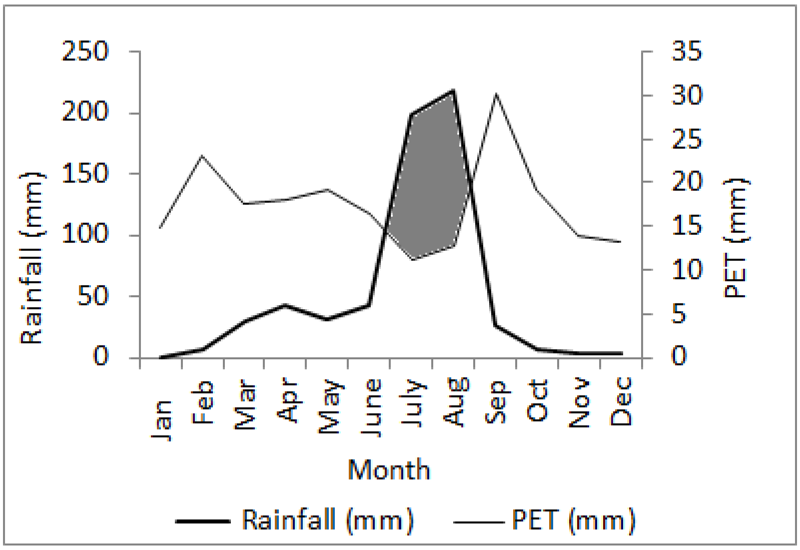

Figure 2-2 Mean monthly rainfall (RF) distribution (1992-2006) and potential evapotranspiration (PET) in the Gergera watershed, Tigray, Ethiopia. Shaded area indicates the rainy season. Data source: Berhane, 2008.

The area has an altitude ranging from 2066 to 2887 m a.s.l. and is classified as the agro-ecological zone between the middle altitude (Woina Dega) with heights ranging 1500-2300 m, and the highest altitude (Dega) above 2300 m (Flint *et al.,* 2009).

2.1.3 Geology and soil

The main geologic formations in the area are dominated by Adigrat and Enticho sandstones, but also include Paleozoic sedimentary rocks, Edaga Arbi Tillite, and recent alluvial sediments (Nata and Bheemalingeswara, 2010). The white rock outcrop Enticho sandstone is found interfingered with Edaga Arbi Tillite (Gebreyohannes *et al.,* 2010). The geologic formations gave rise to the dominance of the loamy textured soils in the area. Morphologically, the area is characterized by an undulating terrain with rugged topographic features (Woldewahid *et al.,* 2012), which is the result of a quick uplift of geologic formations during the Pliocene and Pleistocene periods (Nyssen *et al.,* 2004).

The dominant soil types identified through soil sampling in the area were Leptosols, Regosols, Cambisols and Fluvisols. Leptosols and Regosols are the most abundant and found in the ridge and backslopes. Regosols and Cambisols are commonly found in the footslopes, while Fluvisols are common along the valley bottom. The soil texture is dominated by silt loam followed by loam. Most of the samples examined had neutral pH and very few were slightly alkaline.

2.1.4 Land use and vegetation

Generally fragmented landholdings exist in this area (Figure 2-3), where the landholdings of the poor farmers are less than 0.5 ha, and even the size of landholdings of wealthy farmers is less than 1 ha (Fikir, 2005). This fragmentation gave rise to a mixed crop/animal production system, which enabled the farmers to maximize their benefits from the limited resources and thus ensure food security (Woldewahid *et al.,* 2010).

Figure 2-3 Major land-use systems in the Gergera watershed: A) Protected forest land (exclosure); B) Rangeland; C) Cropland in different slope positions; D) Bare land.

The main crops grown in the watershed include cereals, i.e. *Hordeum vulgare* (barley), *Triticum aestivum* (wheat), *Eragrostis teff* (teff), *Zea mays* (maize) and *Sorghum bicolor* (sorghum); and pulses, i.e. *Phaseolus vulgaris* (beans), *Pisum sativum subsp. arvense* (field pea) and *Lens culinaris* (lentil). Teff is produced in southern part of the watershed, where the altitude below 2600 m is suitable (Rahel, 2008). Next to teff production, apiculture is important for rural livelihoods in the central to southern part of the watershed, where suitable bee forages such as 'Gribiya' (*Hypostus ariculata*) and 'Tebeb' (*Basium clandiforbium*) are widely available due to the altitudinal suitability (Meaza, 2010). Following the IWM interventions, market-oriented high-value crops are also produced on the valley bottom areas around the watershed outlet, which is characterized by available groundwater and relatively fertile soil (Woldewahid *et al.,* 2011). The irrigated production includes vegetables, such as tomato, onion, cabbage and spices. The exclosures are used for controlled fodder grass harvesting and integrated beekeeping activities as a means of income generation (Meaza, 2010). Dominant plant species found in the area include trees, i.e. *Juniperus procera* Hochst. ex Endl. ('Tsihdi'), *Acacia abyssinica* Hochst. ex Benth ('Chea'), *Olea European* subsp. *cuspidata* ('Auli'e'*),* *Dodonea angustifolia* L. and *Eucalyptus globulus* Labill. ('TsaedaBahrzaf'). Shrubs include *Dodonia angustifolia* L. ('Tahses'), *Euclea racemose* Murr. subsp. *schimperi* (A.DC.) F. Whit ('Keleaw') and *Beciumgrandiflorum* ('Tebeb'). Dominant grass species include *Cynodon dactylon* ('Tahay') and *Hyperrhenia hirta*. ('Goiti ebab'). The livestock production in the area involves cattle, sheep, goats, horses, and chicken.

2.2 Data collection and analyses

2.2.1 Land-use and land-cover change

Multi-temporal imagery acquired for different seasons was used to detect the LULC changes. The use of images from different seasons makes it easier to distinguish between multiple land-cover classes based on spectral-phenological variations (Guerschman *et al.,* 2003; Wolter *et al.,* 1995). Specifically, cloud-free Landsat imagery acquired in the dry and wet seasons of the year 1994, 1995 and 2014 were employed (Table 2-1). Precision terrain-corrected (L1T) format Landsat imagery for footprint 168/51 was used to this end. To ensure radiometric consistency, the Landsat Ecosystem Disturbance Adaptive Processing System (LEDAPS) was used to convert raw digital numbers to surface reflectance and correct atmospheric influence for Landsat TM data (Masek *et al.,* 2006). For Landsat 8 Operational Land Imager (OLI) archive, surface reflectance data were generated from the L8SR algorithm (Roy *et al.,* 2014).

The auxiliary information including elevation and topographic slope was also used for the LULC change detection. A digital elevation model (DEM) was collected from the regional databases of the Tigray region, which was originally extracted from Aster images and resampled to match the spatial resolution of the Landsat imagery. Finally, all spectral bands and topographic information as input variables for model prediction were stacked.

The Random Forest model (Breiman, 2001) implemented in the statistical software CRAN R (Development Core Team, 2012) was used to predict stable LULC conversions. The number of variables randomly sampled as candidates at each split (mtry) was set to the square root of the number of input variables, the minimum size of the terminal nodes and the number of the trees was set to 10 and 500, respectively. Focal classes were: stable cropland, stable forest, stable grassland, settlement areas and bare land. These LULC classes are defined according to Weber (2010) and Lusch and Goodwin (2012) as follows:

1. Forest lands: Land with woody vegetation with at least 25% canopy cover by trees of all sizes. These areas include the natural forests, managed forest lands and unmanaged woodlands. The exclosures are former forest lands located on steep slopes which have been degraded due to fuelwood collection, animal grazing and browsing and heavy erosion and flooding. Consequently, these areas were managed for natural regeneration and excluded/ restricted from animal and human interference.
2. Rangelands: Land with permanent grass and/or herbaceous plant cover. These lands had been continuously used (for at least 5 years) for fodder production. The proportion of woody vegetation cover is <10%.
3. Croplands: Land used for cultivation of annual crops, such as cereals, pulses, vegetables, fodder crops, and root crops. Permanent pastures are not included in this class. Only rainfed lands are considered with no permanent irrigation structure observed.
4. Bare land/ settlement areas: This class includes settlement areas, bare land with very little or no vegetation cover and rock outcrop areas.

Because of the lack of high-resolution imagery for the 1990s, several resources were used to collect samples for the model training. These included personal communication with farmers, LULC map from literature (Fikir, 2005) and the raw Landsat images. A total of 382 sampling points was used for the LULC classification.

Table 2-1 Landsat imagery used for land-use and land-cover change mapping.

Year	Imagery DOY	Season	Sensor	Spectral band
1994	347	Dry	TM	1-5, 7
1995	30	Dry	TM	1-5, 7
2014	226	Wet	OLI	1-7
2014	338	Dry	OLI	1-7

Source: Data downloaded from USGS (U.S. Geological Survey) Landsat archive service: http://landsat. usgs.gov/

2.2.2 Above- and belowground biomass carbon

Soil and vegetation sampling was conducted in 2012 from the end of September till December. For the study of above- and belowground biomass carbon stocks, four land-use systems (identified as focal LULC classes) with different vegetation characteristics were considered: exclosure, cultivated land (rainfed croplands), rangeland and bare land.

Sampling was conducted during the time of no rainfall, as this could have affected the samples. Stems, branches, and foliage of living trees as well as herbaceous undergrowth were sampled in the exclosures for estimating the aboveground biomass (AGB) carbon stocks. Biomass sampling in croplands included crop as well as weed vegetation. Grasses were sampled in rangelands. The bare land was not sampled for biomass due to the absence of a vegetation cover.

A systematic sampling method was used to locate the transect lines and main plots throughout the study site in such a way that they fell within the different topographic positions (ridge, backslope and footslope positions). The transect lines had different lengths depending on the length of the slopes and crossed from the top of the slopes down to the valley bottoms. For biomass collection in the exclosures, 17 main sampling plots of 40 m x 5 m were established along the transect lines. Within each main plot, 6 sub-plots of 0.5 m x 0.5 m (total 102 sub-plots) were sampled (Figure 2-4) for estimation of the understory vegetation biomass and litter (Hairiah *et al.,* 2010).

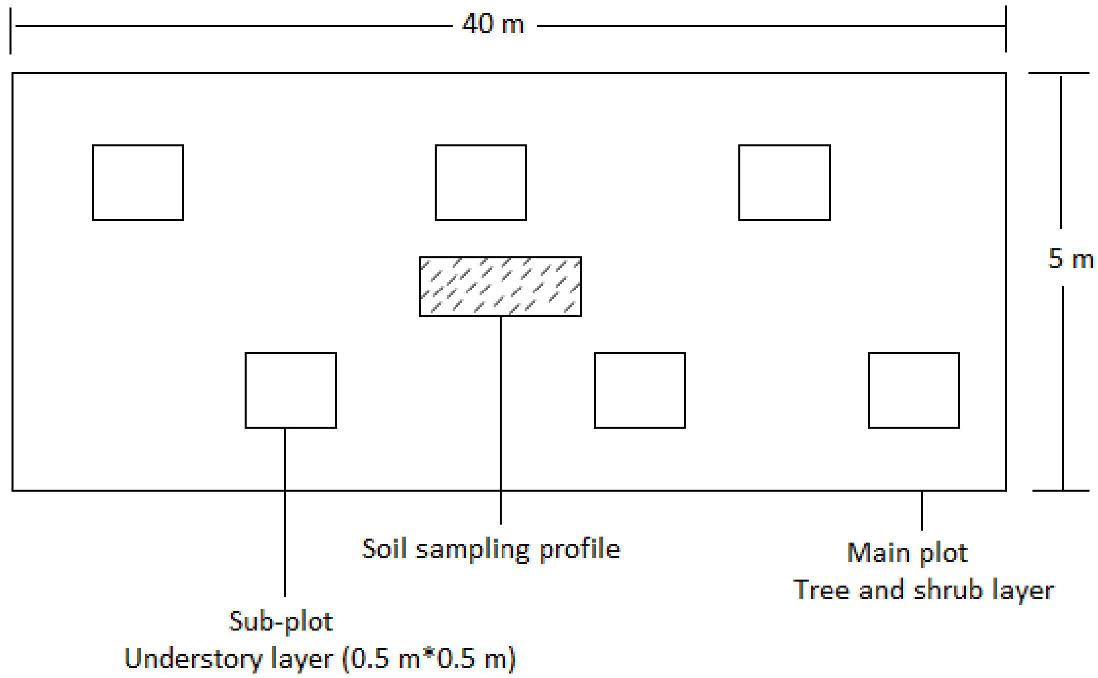

Figure 2-4 Plot layout for biomass and soil sampling in exclosures, crop- and rangelands.

From the 17 main plots in the exclosures (Figure 2-4), 494 trees were measured for height, diameter at breast height (DBH) (Hairiah *et al.,* 2010) and crown diameter (Table 2-2). DBH was determined at 1.3 m using a graduated stick and clinometer. The DBH of trees with <30cm diameter was measured using a caliper. In cases when the diameter exceeded 30 cm, the circumference of the tree stem was measured using a tape meter and this value was then converted into diameter. The crown diameter measurement was based on the average width of the east-west and north-south tree crown measurement using a measuring tape. For trees with ramified stems below 1.3 m height, the DBH was calculated according to Getachew and Stahl (1998):

$$DBHc = \sqrt{\sum_{i=1}^{n}((DBHi)^2)} \tag{2-1}$$

Where DBHc is the computed diameter at breast height (cm) and DBHi is the diameter at breast height of individual stems extending from the same root, and *i* represents the number of ramified stems measured.

Table 2-2 Mean values of stem height, crown diameter and DBH of all trees measured in the main sampling plots of exclosures. Values in parenthesis are standard errors of mean.

Acacia abyssinica, N= 38			*Juniperus procera,* N=318		Other species, N= 103	
	Mean	Range	Mean	Range	Mean	Range
Height(m)	2.6(±0.2)	0.8-8.4	3.99(±0.09)	0.8-9.7	1.9(±0.2)	0.6-12.9
Crown diameter (m)	2.5(±0.2)	0.9-4.5	2.97(±0.08)	0.65-8.7	1.64(±0.09)	0.5-7.0
DBH(cm)	5.6(±0.4)	1.8-10.5	8.0(±0.4)	1.5-95.5	2.8(±0.2)	1-23

The measured trees were classified according to DBH classes, i.e. 2.5-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, 40-60 cm and >60 cm (Hairiah *et al.,* 2010). Stratified random sampling was used to select 1-2 trees per each DBH class to harvest for biomass measurements. However, as more than 97% of the trees had a diameter <16 cm, no tree was felled within the larger diameter classes. Besides, felling the trees was restricted in the area as the land was a designated for the rehabilitation. Therefore, 2-3 trees per plot, totaling 46 trees, were felled. These trees were measured for fallen height, and fresh total weights of stems, branches, and foliage. Next, 8 cm long branch and stem samples were randomly selected, one from each the thick and thin parts of the tree stem and branches. A foliage sample of 250 to 350 g was taken randomly from the canopy. The samples were stored in closed plastic bags to prevent moisture loss until they reached the laboratory for further analysis. After recording the fresh weight, the samples were oven-dried at 65° C to constant weight, and weighed to compute the total dry biomass of each tree.

The relationship of total AGB with crown diameter, height and DBH was tested with linear, exponential and logarithmic functions in order to identify parameters that best predict the biomass. Besides, as the linear regression model using log transformation introduces a systematic bias (Sprugel, 1983), a nonlinear function was used to predict the total biomass (Litton and Kauffman, 2008). The following equations were tested:

$$ln\, B = ln\alpha + \beta ln\,(X) \quad (2\text{-}2)$$

$$B = \alpha(X)^{\beta} \quad (2\text{-}3)$$

Where B is the total biomass, X is the parameter tested for fitness, a and b are slope and scaling coefficients, respectively. In parameters were transformed with a natural log function to linearize the power function. Biomass estimation of non-dominant woody species was done based on the functions developed for the dominant tree species (Giday *et al.,* 2013). The equation developed for *A. abyssinica* was used for the non-dominant woody species because these were too few for developing species-specific biomass functions. Besides, application of the biomass functions of the dominant species for the biomass estimation of the co-occurring non-dominant species is assumed a legitimate compromise to using other generalized functions reported from different locations in Ethiopia and elsewhere.

Litter samples were collected from the sub-plots selected for herbaceous plant sampling in the exclosures. All coarse and fine litter components of the dead plant material were sampled (Hairiah *et al.,* 2001). Care was taken not to include soil material in the samples. The weight of the samples was measured in the field. The samples were sieved to separate the litter from soil. Next, all the samples belonging to same main plot were thoroughly mixed. Sub-samples of 250-300 g were then taken for dry biomass analysis.

The choice of the sampling period is essential in the estimation of the non-permanent AGB stock in croplands and grasslands as opposed to exclosures where woody evergreen vegetation is the main constituent of the AGB carbon stock. The biomass dynamics of the cultivated lands is season dependent There is no crop cover in the period from December to May and in June when land preparation for cropping ends, resulting in an absence of AGB. Therefore, in general, the croplands remain without cover for most of the year after the farmers harvest crops in October-November.

Rangelands have a maximum grass biomass during the rainy season from June to September, which is the period when they regenerate and before grasses are harvested. A minimum rangeland biomass is observed after the grass harvesting period.

Considering the above, the sampling period corresponded with the time of peak biomass. Above- and belowground biomass (crops, weeds and grasses) was sampled from three 0.5 m x 0.5 m sub-plots (Shukla *et al.,* 2011; Maraseni and Cockfield, 2011) randomly established in main plots of about 0.25 ha (average landholding size) in the croplands and rangelands. The samples were weighed in the field and processed following the same procedure as for the exclosure biomass samples. A total of 75 sub-plots on croplands and 42 sub-plots on rangelands were sampled.

The selection of trees for root excavations aimed to obtain representative root samples. To this end, sampled trees were selected based on the DBH class, crown diameter and height class that were mainly found in the measured tree samples (Table 2-3). Due to logistical restrictions on excavations of tree roots, only 9 trees of 2 species, i.e. 5 of *J. procera* and 4 of *A. abyssinica* were manually uprooted. The root biomass included root stump, coarse roots (∅>10mm), medium-size roots (∅-10mm-2mm) and fine roots (∅<2mm) (Razakamanarivo *et al.,* 2012). Based on these measurements, the root to shoot ratio was calculated to estimate the belowground biomass of the trees that were not excavated.

Table 2-3 Mean values of stem height, crown diameter and DBH of trees excavated for root sampling. Values in parenthesis are standard errors of mean.

	Acacia abyssinica, N=4			*Juniperus procera,* N=5		
	Mean	SD	Range	Mean	SD	Range
Height(m)	2.5(±0.48)	0.95	1.69-3.66	3.93(±0.15)	0.34	3.5- 4.45
Crown diameter(m)	2.29(±0.18)	0.36	1.77-2.63	3.1(±0.29)	0.42	2.15- 3.93
DBH(cm)	5.23(±0.82)	1.64	3.7-7.1	8.5(±0.61)	1.37	6.8-10.5

Carbon storage in woody biomass was calculated assuming 46% carbon in wood (Hairiah *et al.,* 2010). For above- and belowground biomass of crops and grasses and tree litter same conservative values of 46% carbon in the dry matter was assumed in accordance to Hairiah *et al.* (2010) although a broad range of values is reported in other literature (MacDicken, 1997; Alam *et al.,* 2013; Dean *et al.,* 2015).

2.2.3 Soil sampling for chemical analyses

Composite samples of disturbed soil (MacDicken, 1997) from four depths of the soil profiles (0-15, 15–30, 30-60 and 60-100 cm) were collected for the SOC assessment (Table 2-3). A total of 61 soil profile pits of 1.5 m length and 1 m width were opened in the center of the main plots (Figure 2-4); 195 composite soil samples (Table 2-4) were collected from the four sides of the profile pits. At the same time, undisturbed soil samples were collected from each soil depth from the soil profile walls using a core sampler of 100 cm^3 volume for further analysis of bulk density. The samples were put in plastic bags and labeled in the field. Some grassland and cropland plots were sampled down to 140 cm (Table 2-3). Deeper soils were found in these locations due to the direct relation of the chemical recalcitrance and turnover of OM to the soil depth (da Silva *et al.,* 2008). Auger-sampling was not possible due to the soil compaction, which did not allow sufficient penetration into the soil.

Table 2-4 Number of soil samples by land use and sampling depth

Soil layer (cm)	Exclosure	Rangeland	Cropland	Bare land
0-15	17	14	20	8
15-30	17	14	19	6
30-60	12	12	15	5
60-100	8	10	11	3
100-140	0	3	1	0
Total	54	53	66	22

2.2.4 Cesium carbon (^{137}Cs) assessment

Soil samples were also collected for a fallout radionuclide ^{137}Cs investigation in order to assess the effect of soil erosion on SOC redistribution in the study area (Table 2-5). Composite soil samples were collected from the same pits mentioned in the above section, but in thinner layers from the topsoil only, i.e. 0-5, 5–10, 10-20 and 20-30 cm (Wang *et al.,* 2011). The samples were collected from exclosure, cropland, rangeland and bare land locations at different slope positions in the watershed, i.e. ridge, backslope, footslope and valley bottom. To have a benchmark for soil erosion assessment, reference sites were selected on a hilltop (Figure 2-5 and Figure 2-6) where no significant soil erosion/deposition processes were expected (Graffenried and Shepherd, 2009). This is because a determination of ^{137}Cs flux requires a relatively undisturbed reference site (Graffenried and Shepherd, 2009; Afshar *et al.,* 2010, Martinez *et al.,* 2009, Hancock *et al.,* 2010) to be able to categorize the areas as depositional or erosional. Besides, according to Mabite *et al.* (2008) representative reference sites should show an exponential reduction in the distribution of ^{137}Cs along the soil depth. Hence, to develop the distribution function of the ^{137}Cs along the soil depth and investigate the representativeness of the reference sites, the soils there were sampled from deeper depths, i.e. 30-50 cm and 50-90 cm.

Table 2-5 Number of soil samples used in ^{137}Cs analyses by land-use class and sampling depth in Gergera watershed

Soil layer (cm)	**Exclosure**	**Rangeland**	**Cropland**	**Bare land**	**Reference sites**
0-5	16	15	21	8	3
5-10	15	15	21	7	3
10-20	15	13	20	5	3
20-30	14	17	19	4	3
30-50	-	-	-	-	3
50-90	-	-	-	-	2
Total	60	60	81	24	17

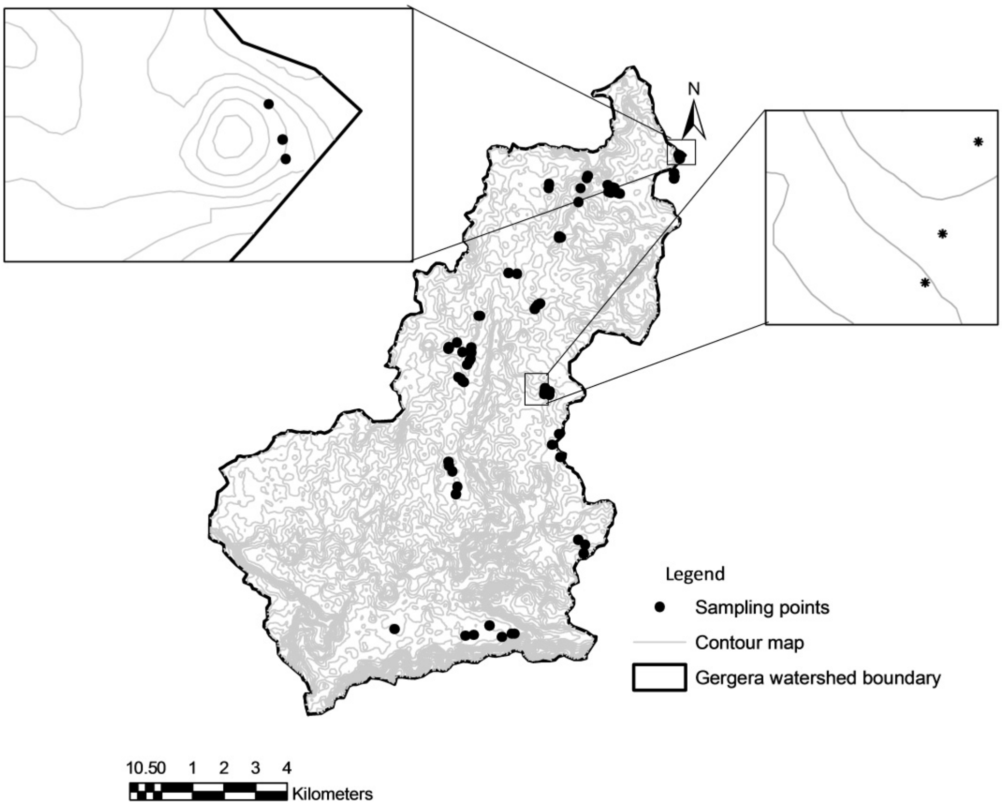

Figure 2-5 Location of reference sites and sampling points in Gergera watershed.

Although grasslands also have been stated as a suitable reference for cesium-based analysis of soil erosion impacts (Mabit et al., 2008), the grasslands in the Gergera watershed were highly disturbed and located in areas susceptible to erosion and deposition processes. Consequently, the selected reference sites were located in forest area (Table 2-6) at one edge of the watershed due to the absence of other suitable areas. The difficulty of locating a suitable reference site was also stated as one of the major limitations of erosion studies using the cesium method in the review by Porêba (2006).

Table 2-6 Characteristics of sampling points in the reference sites

Reference site	Coordinates		Elevation, (m)	Slope, (%)	Vegetation cover, (%)	Soil texture
	X	Y				
1	584988	1535389	2808	3	50-60	Sandy loam
2	585018	1535311	2796	2	50-60	Silt clay loam
3	585025	1535268	2808	2	50-60	Sandy loam

Figure 2-6. Location of the sites used in ^{137}Cs study: (a) Reference site characterized by minimum human and animal interference; (b) One of the sampling sites in forest land.

In total, 245 soil samples were collected (Table 2-5). However, as most of the soil samples from the 10-20 cm depth had ^{137}Cs levels below the detection level, the samples from the 20-30cm depth were excluded and the number of samples analyzed was thus reduced to a total of 189 samples. The ^{137}Cs radionuclide measurement was conducted using a Gamma Spectrometry kit in the Institute of Geology, University of Bonn. The ^{137}Cs radionuclide was detected at 662 keV peak with a counting time of more than 21,600 seconds for 100g of soil. All results of ^{137}Cs counting were then normalized for November 1, 2012 as a generalized sampling date, irrespective of any +/- of real sampling dates which ranged between mid of September to end of December, to calculate the cesium concentration in each sample. The measurements of ^{137}Cs concentration in Bq kg^{-1} were then converted to ^{137}Cs activity in Bq ha^{-1} using the following formula:

$$^{137}Cs_{stock}(Bq\ ha^{-1}) = \quad ^{137}Cs_{conc}(Bq\ kg^{-1}) * BD\ (kg\ m^{-3}) * SD(m) \qquad (2\text{-}4)$$

Where $^{137}Cs_{stock}$ is cesium stock (Bq ha^{-1}), $^{137}Cs_{conc}$ is cesium concentration (Bq kg^{-1}), BD is the bulk density of the fine earth (kg m^{-3}) and SD is soil depth (cm). Interpretation of the results was then done based on the principle that the strong bonding of ^{137}Cs with the soil particles resulted in its movement to be strongly associated with the movement of soil particles by water and wind erosion or tillage (Mabit *et al.*, 2008). Hence through comparing the results of the ^{137}Cs activity of each sampling point to the reference site the deposition and erosion areas were identified.

2.2.5 Soil and biomass sample analysis

The soil samples were taken to the laboratory and air-dried at room temperature for 48 hours. Some of the samples taken from the rangelands that had a higher amount of moisture were exposed for 72 hrs. The samples were then ground with a mortar and pestle to pass through a 2-mm sieve. Until the soil samples could be analyzed, they were stored in a dry and cool place.

SOC concentration of the samples was measured using the Walkley-Black (WB) method (Walkley and Black, 1934) in the Soil Chemistry and Fertility Laboratory, Department of Land Resources Management and Environmental Protection, Mekelle University, Ethiopia. A correction factor of 1.32 was applied to the SOC results, as a complete digestion of OM cannot be achieved due to the extent of oxidation by dichromate. In addition, 80 soil samples were randomly selected for further analysis of SOC by a CN analyzer. Analysis of the latter samples was conducted in the laboratory of the Institute of Crop Science and Resource Conservation, Department of Plant Nutrition, University of Bonn, Germany.

The undisturbed core soil samples were weighed for fresh weight and oven-dried at 105°C for 24 hours. The presence of soil rock fragments complicates determination of soil bulk density, leading to over- or underestimation of the SOC stock (Throop *et al.,* 2012). Different bulk density calculation methods result in variable estimates of SOC stocks (Grimm *et al.,* 2008, Throop *et al.,* 2012). In particular, Mehler *et al*. (2014) found that considering the variation in the rock fragment content in soils resulted in 37% lower SOC stock estimates compared to the calculations using a constant value of bulk density. This requires accurate estimation of the amount of rock fragments for SOC stock calculation (Mehler *et al.,* 2014; Bornemann *et al*, 2011). Therefore, the oven-dried soil samples were ground with a mortar and pestle and passed through a 2-mm sieve to separate rock fragments >2 mm in size. Bulk density was then calculated following Grimm *et al.,* (2008), Lorenz and Lal (2007), and Throop *et al.,* (2012):

$$BD = {}^{Mass\ of\ soil}/_{Volume\ of\ core} \quad (2\text{-}5)$$

Where mass of soil (g) is the mass of the fine earth fractions <2 mm and volume of core (100 cm^3) includes the fine earth and rock fragments.

SOC stocks for given areas were computed employing the following equation (Grimm, *et al.,* 2008; Throop *et al.,* 2012):

$$SOC_{stock}(Mg\ ha^{-1}) = BD\ (g\ cm^{-3}) * C_{conc}\ (\%) * SD(cm) * 1000 \quad (2\text{-}6)$$

Where 1000 is the unit conversion factor, BD is the bulk density of the fine earth (g cm^{-3}), C_{conc} is carbon concentration (%), SD is soil depth (cm) and SOC_{stock} is soil organic carbon stock (Mg ha^{-1}).

Finally, the total terrestrial carbon was computed as:

$$TC_{stock}(Mg\ ha^{-1}) = AGBC_{stock} + BGBC_{stock} + SOC_{stock} \quad (2\text{-}7)$$

Where TC_{stock} is total carbon stock (Mg ha^{-1}), $AGBC_{stock}$ is aboveground biomass carbon stock (Mg ha^{-1}), $BGBC_{stock}$ is belowground biomass carbon stock (Mg ha^{-1}) and SOC_{stock} is soil organic carbon stock (Mg ha^{-1})

The SOC included in the total carbon stock computation is as was determined for the upper 30 cm soil depth, according to the standard depth definition of the Kyoto Protocol guideline.

Next to SOC, other soil parameters were analyzed to assess the soil nutrient status that can be related to SOC. Cation exchange capacity (CEC) and exchangeable cations (Ca^{2+}, Na^+, Mg^{2+} and K^+) were analyzed using the flame photometry method. Total nitrogen (TN) was determined with the Kjeldahl method (Bremner and Mulvaney, 1982), and available phosphorus (P) using the Olsen method. Electrical conductivity (EC1:2.5) and pH (H_2O) were analyzed using an EC meter and pH meter, respectively. These results were interpreted based on criteria specified in Appendix A1.

2.3 Statistical data analysis

Comparison of the two SOC analysis methods, i.e. WB and CN analyzer method, was conducted employing Pearson correlation and a linear regression. In addition, the Bland and Altman (1986) method of analysis was applied to assess the agreement of the results. Pearson correlation analysis was conducted to determine the significance of correlations between SOC and the other soil parameters.

Scatter plots of linear regression were used to depict the relationship of DBH, crown diameter and height with the AGB. Linear regression analysis with natural log transformed data and nonlinear regression analysis of a power function were applied for developing a function for AGB estimation. Identification of the best function was done based on the goodness of fit of the regression equation determined by a lower P-value with a high coefficient of determination (R^2) and low mean square error (MSE) (Litton and Kauffman, 2008).

Factorial three-way analysis of variance (ANOVA) was used to evaluate the effect of land-use system, soil depth and topographic position on SOC concentration and stock. Due to the highly unbalanced sampling design and the repeated measurements of the soil depth in similar land-use systems, Linear Mixed Effect (LME) model was applied. The dependent variables (SOC concentration and SOC stock) were checked for homoscedasticity and log transformed when appropriate. Among the independent variables, land use system was considered as a categorical random effect variable, and all the other variables, including soil depth and slope position, were considered as continuous fixed-effect variable.

For the soil samples with a ^{137}Cs value below the detection limit, the lower limit of detection value (LLD), which is 1 Bq kg^{-1} based on the results of the laboratory analysis in this study, was used (Mabit *et al.,* 2010). ^{137}Cs distribution along soil depth, slope position and land-use system was analyzed using the LME modeling procedure. The SOC stock and ^{137}Cs distribution relationship was assessed using regression. The statistical analysis was carried out using Stata software version 13.

3 RESULTS AND DISCUSSION

3.1 Land-use and land-cover change in Gergera

The LULC change analysis for the period from 1994 to 2014 indicates the occurrence of land-use changes that might signify an improvement in vegetation cover, such as a conversion of cropland to forest land and change of bare land to rangeland (Figure 3-1). A reduction in land vegetation cover was also observed in the class changes from cropland to bare land, rangeland to cropland, and forest land to rangeland over the examined 20 years.

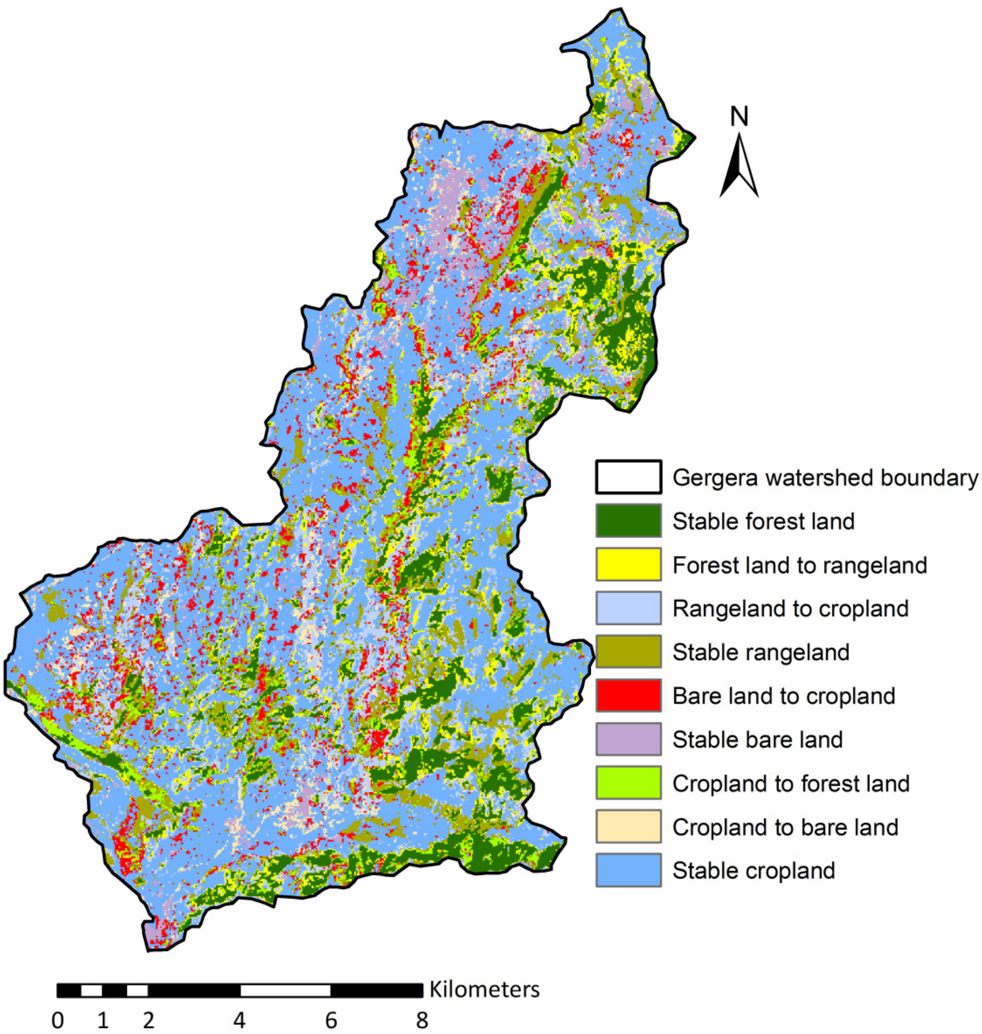

Figure 3-1 Map of land-use and land-cover change (LULC) between 1994 and 2014 in the Gergera watershed of the Tigray region of Ethiopia.

Some 30% of the study area underwent LULC changes while 70% was found stable during the investigation period. The main stable LULC classes were forest land, rangelands, bare land, and croplands, which also constituted the largest LULC class overall in the land area (Figure 3-2). The forest lands were commonly found on the escarpments that were far from settlement areas and around churches, and are considered as remnant natural forests. The managed forests in the exclosures, where the degraded hillslope measures have been implemented, were commonly found on the steep slopes in the mountains, and the woodlands were found in different slope positions. Not only the rangelands but also these forest lands were used for grazing which left them overgrazed and degraded (Biryahwaho et al., 2012). The settlement areas in the northern and southern part of the watershed and the rock outcrop areas of the mountains comprised the main coverage of the bare land. However, bare land was also observed adjacent to the croplands and rangelands. Being the dominant LULC class (45% of the study area), croplands were found throughout the watershed (Figure 3-1 and Figure 3-2). Rangelands occupied the areas with high soil moisture and gently sloping and flat areas of the watershed adjacent to the croplands and woodland forests.

The accuracy assessment confirmed the good performance of LULC change classification with an overall accuracy of 80%. However, user accuracies varied between 89% for the stable forest cover and 60% for the class of conversion from cropland to bare land. The producer accuracy was found highest for the stable forest class (89%) and lowest for that of conversion from cropland to bare land (45%).

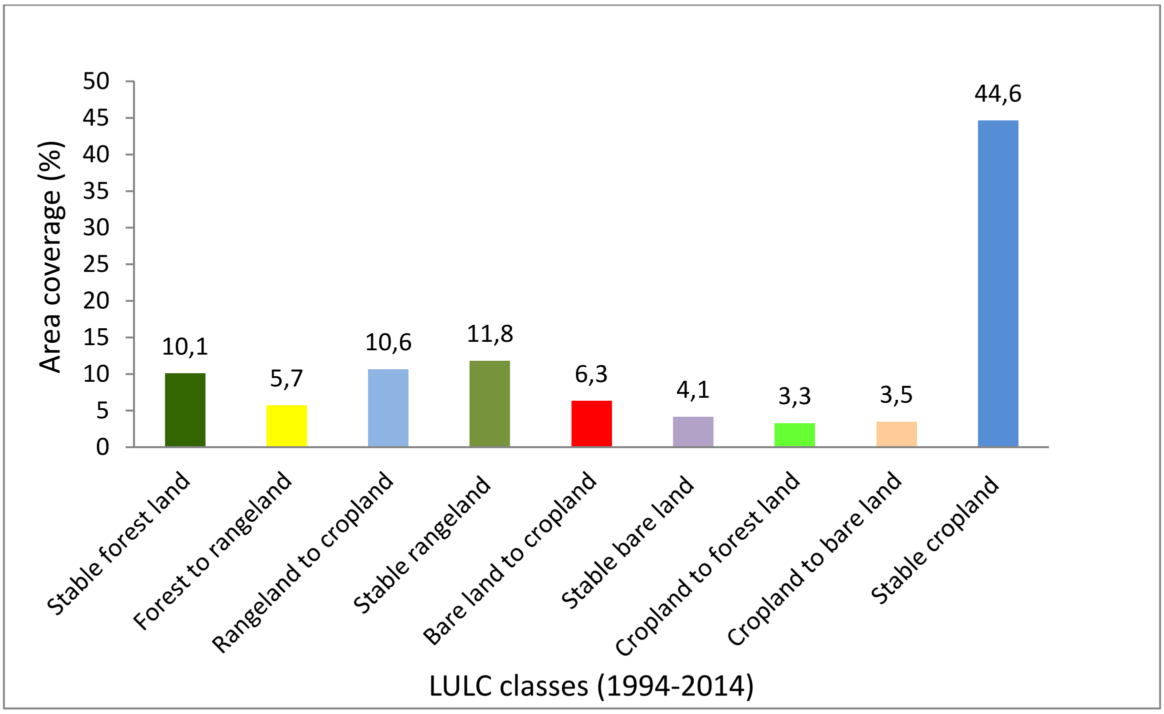

Figure 3-2 Area of the Land-Use and Land-Cover (LULC) classes of the study area during 1994 and 2014 in the Gergera watershed.

Some confusion that occurred in the forest to rangeland change cover classes might be due to high intra-class variability in the forest cover. The inter-class similarities between the croplands and bare lands within the heterogeneous landscapes (Ghimire *et al.,* 2010) could be a reason for the confusion between these classes. An increased understory grass cover of the forest lands, especially in areas under improved forest management (exclosures), introduced a phenological change in the forest class that impacted the model performance. The low accuracy for the conversion class from cropland to bare land may be attributed to the similarity and variation in reflectance of both the cropland and the bare soil in the wet and dry season, respectively. Clark *et al.* (2010) reported low user accuracy for pine and eucalyptus plantations as well as low producer accuracy for herbaceous vegetation where high variations exist. The highest user and producer accuracies in this study were recorded for woody vegetation and water body where less seasonal variability occurred. Besides, the distinct seasonality of the rainfall in dryland areas that determines the temporal variability of vegetation growth has been a challenge in distinguishing sparsely covered rangelands from bare lands (Yin *et al.,* 2014; Hüttich *et al.,* 2011).

Visual interpretation suggests that modification within the same LULC class might have occurred during the investigation period. For example, land cover within the 'stable forest' class became visibly denser (Figure 3-3). The increased vegetation cover might be attributed to the effect of the land rehabilitation activity through IWM, which included the introduction of exclosures. However, monitoring of land modifications within the same LULC class requires frequent observations (e.g. at annual intervals), which poses challenges for this study area due to the data scarcity for Landsat imagery of the Gergera watershed.

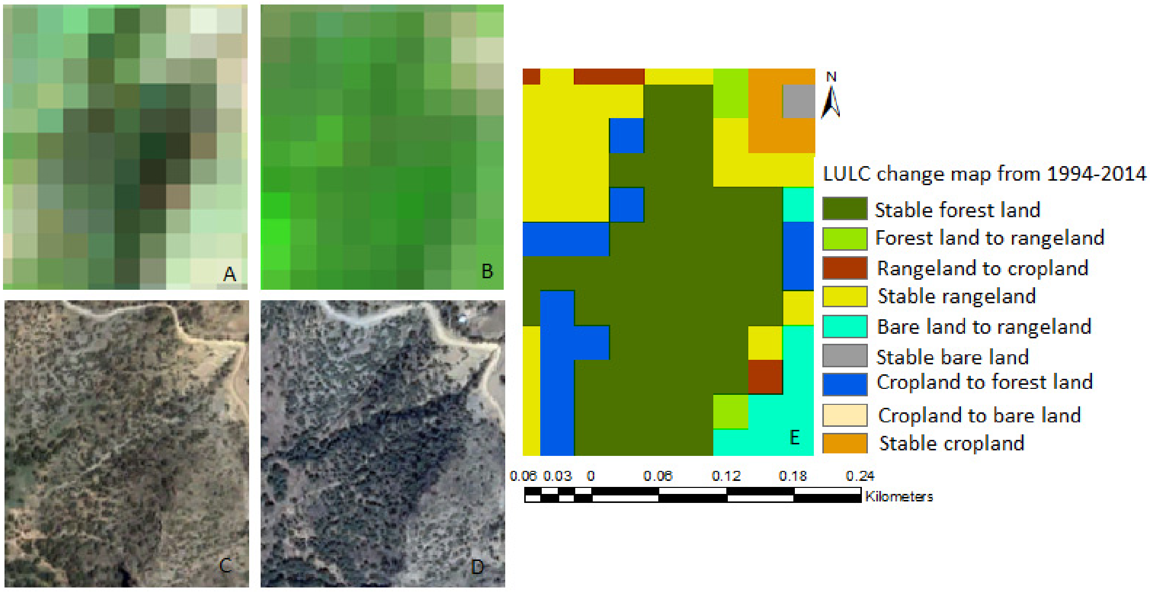

Figure 3-3 Landsat composite images of forest exclosures (A), 1994 (RGB: 5,4,3); (B), 2014 (RGB: 7,5,3), and Google earth images of the status of the exclosure c after 10 years (2007), and (D) after 17 years (2014); (E) is the LULC class of the forest exclosure with stable forest in green.

The knowledge of the LULC change in the Gergera watershed was an important guideline for the above- and belowground carbon survey and for understanding how soil erosion and deposition might have taken place in the different LULC classes. The biomass and soil samples were collected from the stable LULC classes where the carbon stock status was likely attributed to the change of land management under the IWM rather than to the shifts of one land-use class to another. Therefore, the study of the ecosystem carbon stock concerned only these four major land-use systems in the area, i.e. exclosures ("stable forest land"), rangelands, croplands, and bare lands.

3.2 Biomass carbon stocks

3.2.1 Biomass of dominant woody species in exclosures

The dominant tree species, *J. procera*, on average accounted for 79% of the trees measured in all the plots established in the exclosures. The second dominant tree species, *A. abyssinica*, accounted for 11%. All the other woody plant species (section 2.1.4) together made up 10%. Trees with small diameters prevailed for all species, where 48% of the trees had DBH of 2.5-5 cm, 38% were in the DBH range of 5.1-10 cm, and those with 10.1-15 cm DBH comprised 11%. Only about 4% of the trees were found within the large DBH class of >15 cm. The average tree density was 1,047 (±163) trees ha^{-1}. The mean values of the parameters determined by the destructive sampling of trees are shown in Table 3-1.

Table 3-1 Characteristics of the dominant tree species in exclosures.

Plant parameters	*A. abyssinica*			*J. procera*		
	N	Mean(±SE)	Range	N	Mean(±SE)	Range
Aboveground biomass (Mg ha^{-1})	15	1.6(±0.4)	0.1-4.3	20	12.0(±2.4)	1.2-30.7
Height (m)	14	2.7(±0.3)	1.5-3.9	21	3.8(±0.2)	1.0-6.1
Crown diameter (m)	14	2.4(±0.2)	0.9-3.7	21	2.8(±0.2)	1.1-5.3
DBH (cm)	14	5.9(±0.6)	3.1-10.5	21	7.5(±0.6)	2-11.8
Belowground biomass (Mg ha^{-1})	15	0.9(±0.2)	0.1 2.3	20	5.0(±1.0)	0.1- 21
R:S ratio	4	0.4(±0.1)	-	5	0.5(±0.1)	-

Significant positive linear relationships were observed between the aboveground woody biomass and DBH as well as log transformed values of crown diameter and DBH in *J. procera* (Figure 3-4). In *A. abyssinica,* the DBH and the natural log transformed value of DBH were tightly related with the biomass.

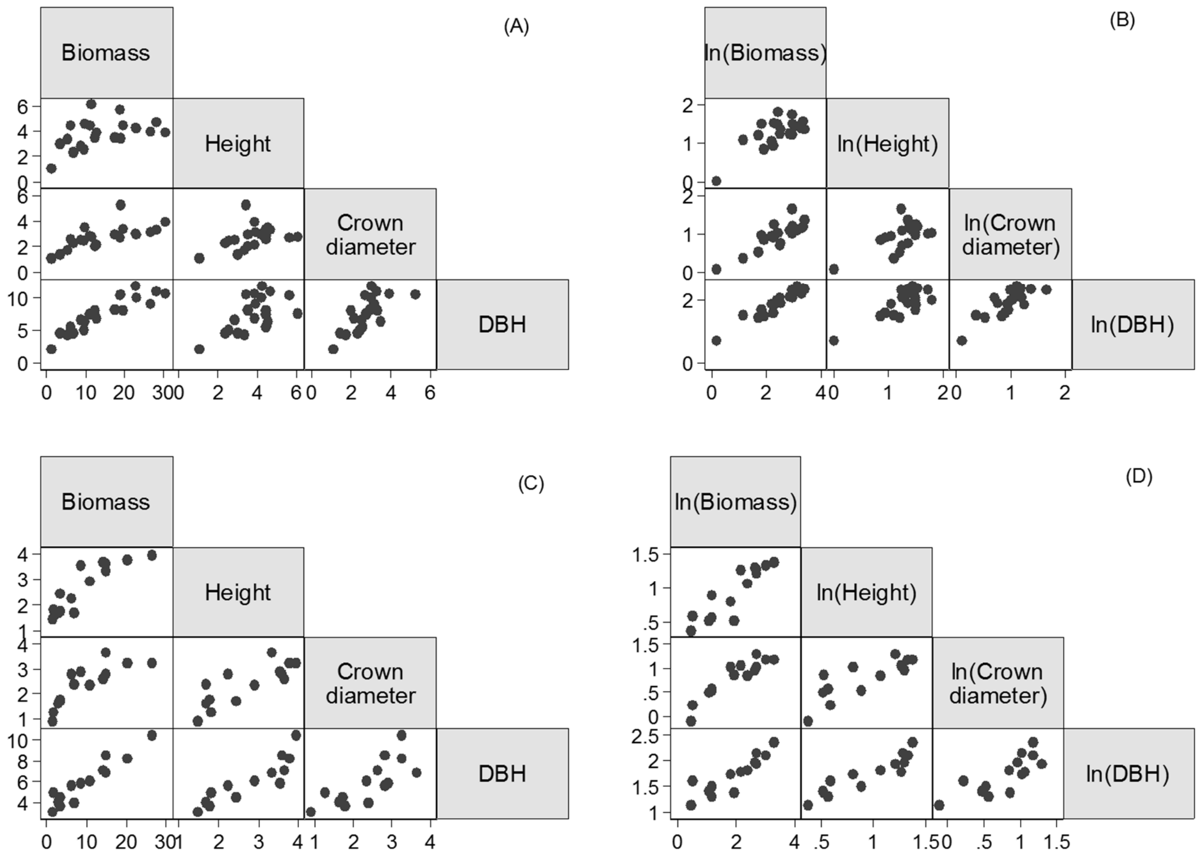

Figure 3-4 Scatter plots of relationships between the aboveground woody biomass (per plant) and DBH (cm), stem height (m), and crown diameter (m) before and after the natural log transformation for (A&B) *Juniperus procera* and (C&D) for *Acacia abyssinica*; $P<0.05$.

The non-linear biomass-DBH relationships (Equation 3.1 and Equation 3.4 in Table 3-2) were found to be the best predictors of the aboveground woody biomass of both tree species.

Table 3-2. Aboveground biomass functions developed for *Juniperus procera* and *Acacia abyssinica*. Rows in bold indicate the equations used for the biomass prediction of both tree species.

Species	Equation	A	β	Adj R^2	Root MSE	CV (%)	P	Equation
J. procera	**$B = \alpha(DBH)^{\beta}$**	**0.5514**	**1.6005**	**0.95**	**3.7**	**25.6**	**<0.001**	**3-1**
	$lnB = ln\alpha + \beta ln(DBH)$	1.004	1.7794	0.90	0.3	10.1	<0.001	3-2
	$lnB = ln\alpha + \beta ln(CD)$	0.6126	1.9143	0.68	0.5	18.2	<0.001	3-3
A. abyssinica	**$B = \alpha(DBH)^{\beta}$**	**0.2429**	**2.0118**	**0.96**	**2.5**	**25.6**	**<0.001**	**3-4**
	$lnB = ln\alpha + \beta ln(DBH)$	-2.062	2.3135	0.77	0.4	22.9	<0.001	3-5

B is aboveground woody biomass, DBH is diameter at breast height, CD is crown diameter and CV is coefficient of variation.

Not all of the equations shown in Table 3-2 were effective in predicting the root biomass. The relationship between DBH and root biomass of *J. procera* and *A. abyssinica*, for instance, had adj-R^2 values of 0.02 (P=0.09) and 0.19 (P=0.3) only. The root: shoot ratio (R:S ratio) was, therefore, found to be a more reliable predictor of the root biomass (Figure 3-5). In allometric study of *Jatropha curcas* in arid Burkina Faso, Baumert & Khamzina (2015) also found the estimation of belowground biomass from the root to shoot ratio to be a valid approach, provided that R:S ratios are specified for different growt stages, as RSR decreases with increasing plant size.

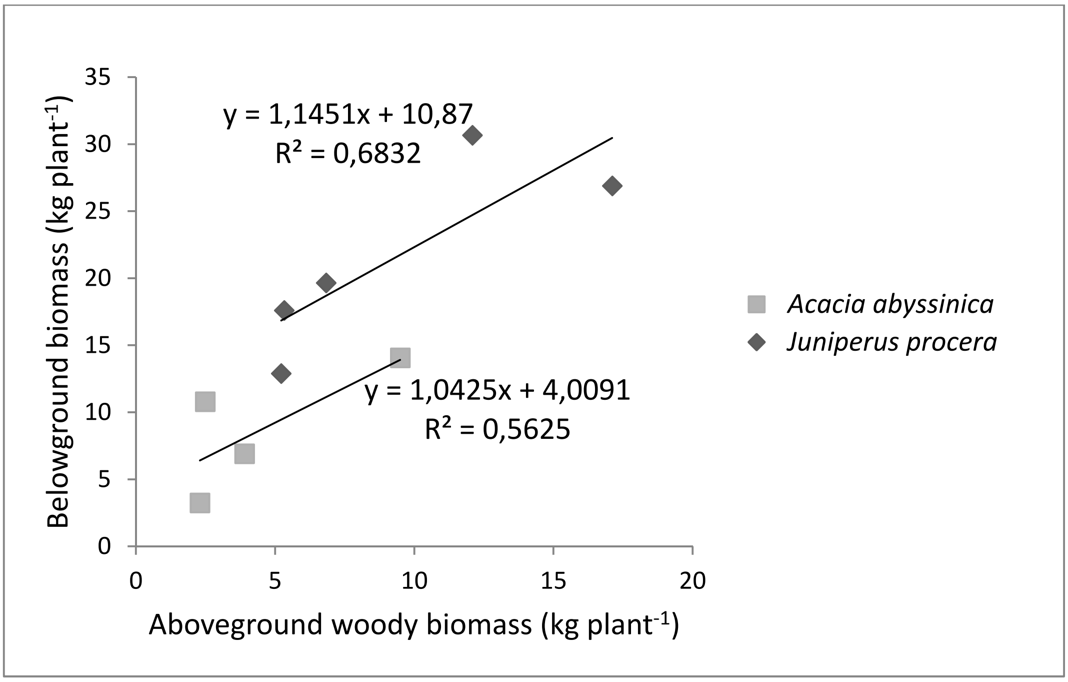

Figure 3-5 Above- vs. belowground biomass relationship of the dominant tree species, *J. procera* and *A. abyssinica*, in exclosures of the Gergera watershed

The R:S ratio measured 0.42 (±0.06) in *J. procera* and 0.54 (±0.11) in *A. abyssinica* in examined stands which were dominated by younger trees. These values exceed the average tropical/sub-tropical dry forest R:S ratio of 0.24 and the global average ratio 0.3 reported by IPCC (2003) for woody species. However, the R:S ratio of the main exclosures' species, *J. procera* is close to the values respectively reported for dense (0.51) and open vegetation (0.49) in regenerated forests of semi-arid Brazil (Costa *et al.,* 2014). More specifically, this review study found that the R:S ratio differed depending on plant age, soil type and vegetation density, and varied from 0.67 for 60-year-old forests to 0.22 in areas with Litholic Neosols (corresponding with Lithic Leptosols in the Gergera). In general, studies on woody plants reported a R:S ratio varying from 0.20 (Santantonio *et al.,* 1977) to 0.67 (Costa *et al.,* 2014), depending on the climate, soil texture, altitude, temperature and soil nutrient content that all affected the plant growth.

3.2.2 Biomass of non-woody species

The non-woody plants including perennial grasses and other herbaceous species contributed a relatively marginal amount of biomass, about 7%, in the exclosures (Figure 3-6). However, litter biomass composed of both tree and grass residures accounted for more than 20% of the total biomass in the exclosures. This accumulation of litter stock might signify the importance of natural regeneration of the understory vegetation with the absence of animal stocking in these land-use systems (Haregeweyn *et al.,* 2007), indicated by the LULCC study results (Figure 3-3) but might be also due to a poor decomposability of the litter (Khamzina et al. 2016).

In contrast, the biomass of the rangelands and croplands was entirelycomposed by the non-woody, annual and perennial plant species. The R:S ratio in the croplands and rangelands was found to be 0.25 (±0.03) and 2.62 (±0.37), respectively. Williams *et al.* (2013) determined the R:S ratio for dryland crops ranging between 0.13 and 0.17 in the top 30 cm, which was just above the ratio 0.1 in areas with very low soil fertility (Hairiah *et al.,* 2001). In water-limited ecosystems, like the Gergera watershed, an optimal root-shoot allocation strategy is mainly determined by rainfall availability, as plant water competition modifies the depth distribution of the roots in the soil (van Wijk, 2011). This might at least partly explain why in contrast to the aboveground biomass (best estimated with the non-linear relationship to DBH), the belowground biomass was most tightly related with the R:S ratio.

3.2.3 Aboveground carbon stock

Carbon content in the aboveground biomass of exclosures was significantly higher with 9.08(±1.44) Mg C ha^{-1} than that in the rangelands, croplands and bare lands (Figure 3-6). A review by Silver et al. (2000) also reported greater biomass carbon stock in exclosures compared to that in other land use systems, because of both managed and natural reforestation in the exclosed areas. A woody AGB carbon stock of 3 Mg ha^{-1} with an annual increase of 0.3 Mg ha^{-1} was reported in a 10-year-old exclosure in drylands of northern Ethiopia (Aynekulu *et al.,* 2008). This value is much lower than the value obtained in the Gergera, where the maximum age of the exclosure did not exceed 15 years.

The woody plants comprised the largest proportion of the biomass carbon in the exclosures (Figure 3-6).. More than 60% of C was stored in *J. procera* trees. Bazezew *et al.* (2015) also reported on a dominance of *J. procera* in the biomass carbon contribution in dry evergreen montane forests in southern Ethiopia.

Rangelands, with 1.49(±0.18) Mg ha^{-1} of AGB carbon stock, in this respect were not statistically different from croplands with 3.16(±0.24) Mg ha^{-1} even though the measurements were conducted at the peak biomass season. However, the plant cover of the rangelands and croplands is season dependent and in particular the croplands are bare during the dry season and hence contain no aboveground biomass carbon.

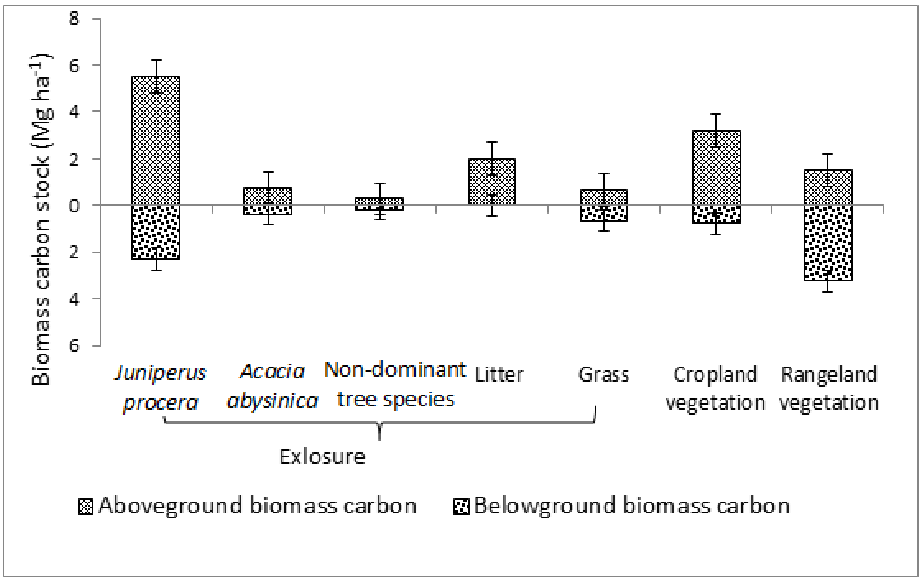

Figure 3-6 Above- and belowground biomass carbon stock contributed by different plant species in the major land-use systems of Gergera watershed.

3.2.4 Belowground biomass carbon stock

Belowground biomass carbon stocks differed significantly among the land-use systems and, in the exclosures, varied according to the tree species. The greatest carbon content, 3.67±0.06 Mg ha^{-1} was found in the exclosures. Rangelands contained only slightly lower amount of carbon (3.16 (±0.39) Mg ha^{-1}) although exclosures included biomass of tree and grass roots whereas belowground biomass (BGB) of rangelands originated from grasses only. This observation can generally be associated with the presence of deeper soils in the rangelands than in the exclosures. Deeper soils allowed a larger rooting volume in rangelands where the root biomass was twice as much as the AGB. Silver *et al*. (2000), IPCC (2003) and Snyman (2005) also reported the root biomass of grasses being higher or at least equal to the AGB.

The BGB carbon stock of croplands, averaging 0.76(±0.09) Mg ha^{-1}, was significantly lower than that in exclosures and rangelands. In the review of the global distribution of belowground biomass in terrestrial biomes, Jackson *et al.* (1996) likewise reported the lowest root biomass (0.15 kg m^{-2}) in croplands compared to tundra (1.2 kg m^{-2}), shrublands (4.8 kg m^{-2}) and grasslands (1.4 kg m^{-2}). Gregory *et al.* (1997) stated that the annual crops sequestered a higher amount of carbon in their roots especially during the early stage of development when more than a half of the total assimilated carbon was stored belowground. Moreover, Bolinder *et al.* (1997) reported that the carbon input from cereal crops, i.e. wheat, barley and oats, was in the range between 279 g C m^{-2} (2.79 Mg ha^{-1}) and 114 g C m^{-2} (1.14 Mg ha^{-1}). This variation depends on the type of cultivar and on removal or addition of the crop residue to the soil. Hence, considering the short-lived annual crop biomass is important especially in low-input agricultural systems where they might be the only organic carbon inputs to build the SOC.

3.3 SOC concentrations and stocks

3.3.1 Agreement of Walkley-Black and elemental analyzer methods

3.3.1.1 Method comparison by correlation analysis

The correlation analysis revealed that the SOC results obtained from the CN analyzer method correlated closely (r=0.91) with both the original and corrected values of the WB method. The linear regression analysis showed a tight positive relation between the two results (R^2=0.82; Figure 3-7). This was also true in the study conducted on an organic-rich sedimentary rock site in Israel (Gelman et al., 2012). There, the authors recommended a recovery factor when using the WB method and found a correlation of r^2=0.92. Our results with carbon-poor soils show that recovery factor does improve the comparability of WB and CN analyser derived results as far as the correlation coefficient alone is concerned (Figure 3-7a). However, with corrected values of WB method, the trendline virtually overlaps with the 1:1 line, indicating the improved agreement between the plotted values (Figure 3-7b).

Wang et al. (2012) conducted a study on calcareous soils and found the modified WB method as accurate as the CN analyzer, except for soils with very low SOC concentrations. In the latter case, the results were over-estimated. This could probably be due to the calcareous nature of the soil, which affects the $K_2Cr_2O_7$ reactions with the inorganic soil components (Walkley and Black, 1934). However, in this study, although the SOC concentration is very low except for few observations, the results of SOC analyses by the WB method were neither over- nor under-estimated. Yet, the carbon-poor soils in our study did not show the presence of carbonates which likely was of advantage for accurate determination of very low SOC concentration levels.

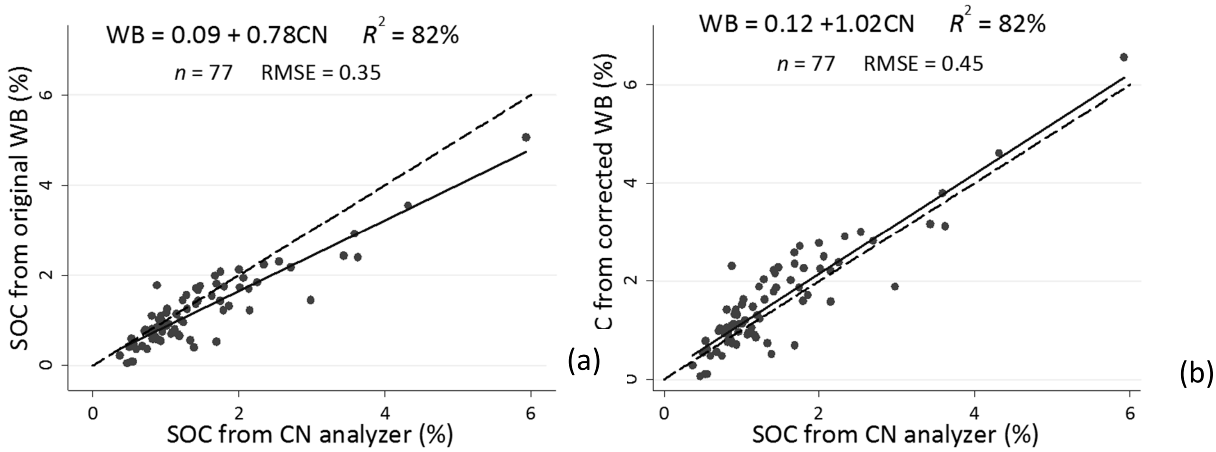

Figure 3-7 Correlation of soil organic carbon (SOC) values measured with Walkley and Black (WB) method vs. CN analyzer method. (a) Original values from WB method; (b) WB method values with correction factor of 1.32. Solid line indicates best fit values and dashed line indicates 1:1 values.

3.3.1.2 Method comparison by Bland and Altman analysis

According to Bland and Altman (1986), correlation analysis does not show agreement of two methods correctly, as it only indicates how strong the relation between two variables is. Hence, to eliminate this bias, the analysis was done using the Bland and Altman method of agreement. Even though this method is mainly used in clinical research (Bland and Altman, 1986; González-Agüero et al., 2013; Griffiths and Murrells, 2010), in this study it is applied to the soil carbon data to reveal the agreement with results of the conventional method

The Bland and Altman analysis (Figure 3-) confirmed the high correlation of the CN analyzer results with both the original and corrected values of the WB method (Pearson’s correlation coefficient =0.81). Furthermore, this analysis revealed that the measurement differences of more than 96% of the original values and more than 97% of the corrected values of the WB method compared to the CN analyzer method were within the Bland and Altman limit of agreement (-1.26, 0.95 and -1.10, 1.55, respectively). This limit of agreement is set based on the concept inter-methods difference ± 1.96 SD. However, there is room for setting different thresholds for maximum acceptable levels of SOC difference between methods.

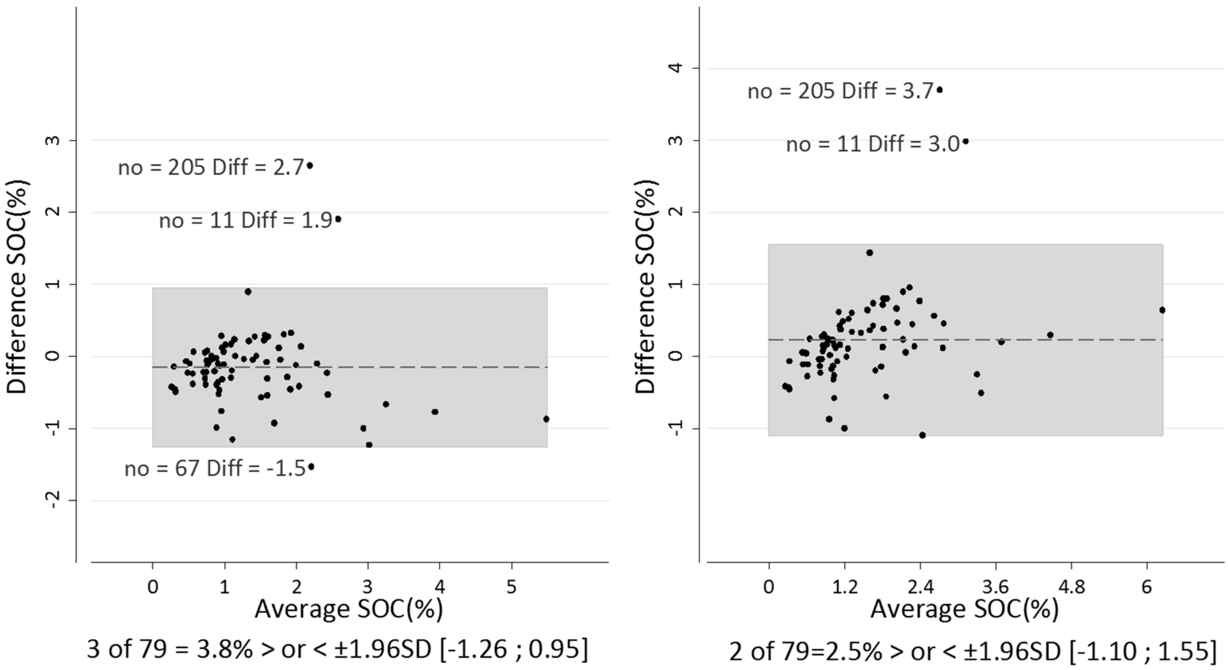

Figure 3-8 Bland-Altman comparison of SOC concentrations obtained with Walkley and Black (WB) and CN analyzer methods. Central line represents the inter-methods difference (bias of a. -0.156 and b. 0.23). Upper and lower limit of shaded area is 95% limit of agreement (inter-methods difference ± 1.96SD); (a) Original values for WB method; (b) WB method values with a correction factor of 1.32.

Furthermore, the analysis shows highly significant concordance of the CN analyzer method with both the original and the corrected WB method (concordance correlation coefficient =0.000) with correlation values between the difference and mean of methods of -0.092 and 0.33, respectively. No trend was observed between the means and the differences.

3.3.2 Soil parameters

3.3.2.1 Soil pH and clay

The pH of the soils ranged between 5.45 and 8.51 with both the minimum and maximum values observed in the rangelands (Figure 3-9). A neutral soil pH in the range of 6.6-7.3 was predominant, found in 68% of the samples, while 24% were slightly alkaline. Strongly alkaline and moderately acidic soils each covered only 0.5% of the area. Therefore, most soils in the examined land-use systems can be categorized as pH-neutral Table A 1).

A significantly lower clay content was observed in the exclosures than in the croplands and rangelands (Figure 3-9). No significant differences were observed in clay content of rangelands, croplands and bare land.

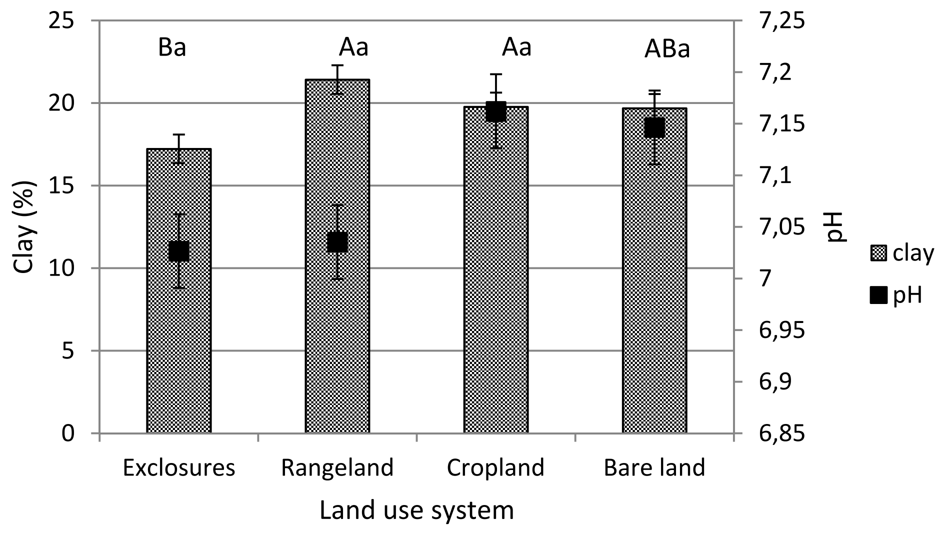

Figure 3-9 Mean clay and pH levels in the different land-use systems. Same capital and small letters above bars indicate insignificant differences in clay content and pH levels, respectively. P = 0.05 based on LSD test. Error bars show 95% confidence interval.

3.3.2.2 Soil macro- and micronutrients

A majority of the soils had a low to very low soil macronutrient concentrations averaging 0.10 (±0.005) % for total N (TN), 0.098 (±0.01) cmol kg^{-1} for exchangeable K (exc. K), and 8.88 (±0.89) mg kg^{-1} for available P (av. P). The soil fertility is considered to be poor (Table A 1), as the soils that have < 0.15 % of TN, <0.2 cmol kg^{-1} of exchanheable K and 5-10 mg kg^{-1} of available P are categorized as such (Hazelton and Murphy, 2007). Many previous studies also revealed a high deficiency of NPK, which hindered the productive capacity of the soil in Ethiopia (Abegaz and Keulen 2009; Stoorvogel and Smaling, 1990).

In addition to the generally high deficiency of these nutirients, variations among the land-use systems were observed (Table A 3). Significantly lower TN was found in the croplands and bare lands than in the other land-use systems. On the other hand, significantly higher exchangeable K values were observed in the croplands and rangelands compared to the exclosures and bare lands (Figure 3-10).

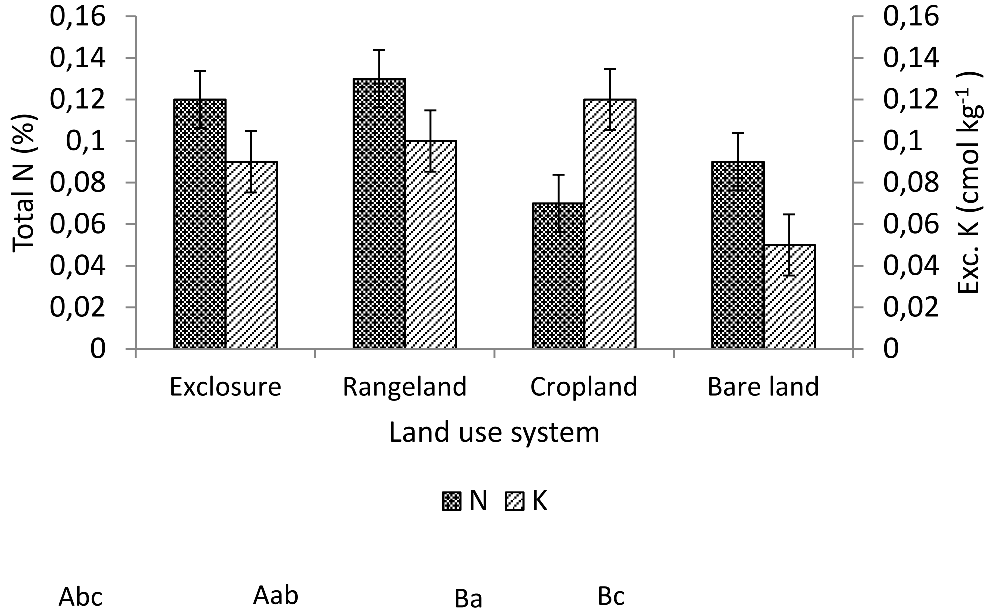

Figure 3-10. Mean total N and exchangeable K in the different land-use systems. Same capital and small letters above bars indicate insignificant differences in total Nitrogen content and exchangeable K levels, respectively. P = 0.05 based on LSD test. Error bars show 95% confidence interval.

The available P values were also significantly higher in croplands (14.6 ±1.4 mg kg^{-1} soil) compared to the other land-use systems which did not significantly differ from each other. The CEC in rangelands (11.3 ±0.4 cmol kg^{-1}) was significantly higher than in the other systems (Figure 3-11).

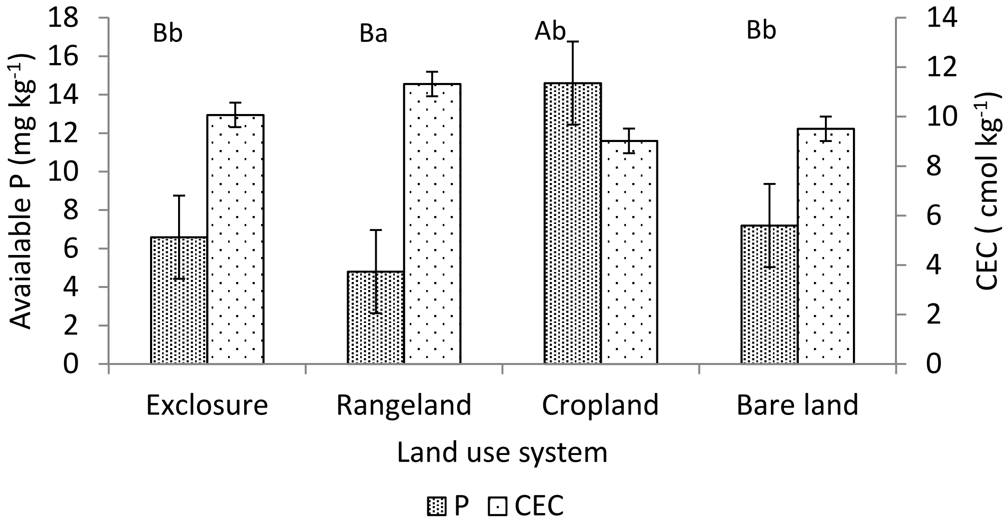

Figure 3-11 Mean available P and cation exchange capacity (CEC) in the different land-use systems. Same capital and small letters above bars indicate insignificant differences in available P content and CEC level, respectively. P = 0.05 based on LSD test. Error bars show 95% confidence interval.

3.3.2.3 Rock fragments and bulk density

Rock fragment content of the soils significantly differed with the land-use type. The largest proportion (28±3%) was observed in exclosures followed by croplands (21±1.4%) and bare lands (14±1.5%). Rangelands had the least proportion of rocks in the soil (8±0.8%) (Figure 3-12). Stone and gravel cover is commonly used as a mulch to reduce soil erosion and maintain soil moisture in croplands (Sharma *et al.,* 2012), explaining the higher amount of rock fragments there. The rock fragment content was rather homogeneously distributed along the entire soil profile. In contrast, the study by Mehler *et al.* (2014) in the forest sites of western part of Thuringia, Germany, reported significantly larger rock fragment content in deeper soils, specifically the values increasing from 4% in the top layer to 34% in 90 cm depth.

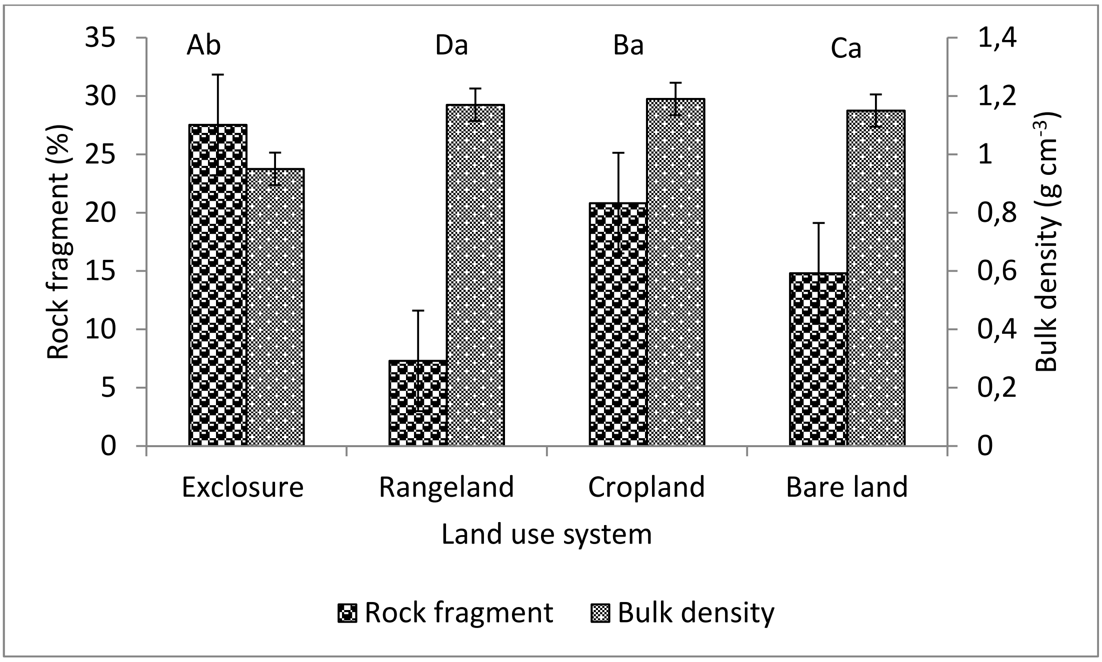

Figure 3-12 Mean rock fragment content and bulk density of the < 2mm soil particles in different land use systems. Same capital and small letters above bars indicate insignificant differences in rock fragment content and bulk density, respectively. P = 0.05 based on LSD test. Error bars show 95% confidence interval.

The bulk density of the soil excluding coarse fragments (soil particles <2mm) in exclosures (0.95±0.03 g cm^{-3}) was significantly lower than in rangelands (1.17±0.03 g cm^{-3}), croplands (1.19±0.03 g cm^{-3}), and bare lands (1.15±0.04 g cm^{-3}), but the latter three classes did not differ significantly among each other (Figure 3-12). The lower bulk density in exclosures corresponds with the largest rock fragment content in these soils. Besides, the core sampling method for bulk density analysis in areas with high rock fragment content within the given core volume can introduce sampling errors. This is because the soil corer may hinder sampling of large rock fragments due to the limited size of the core, resulting in samples which might not be representative for the area (Throop *et al.,* 2012).

3.4 SOC concentration

Considering the SOC concentration independently from SOC stocks is important when comparing the different land-use systems as the SOC values were estimated without the influence of other parameters (Schrumpf *et al.,* 2011), such as bulk density and rock content (Table 3-3). There was no difference in SOC concentrations in the soil profiles of exclosures and rangelands but both values significantly exceeded those in the croplands and bare lands. Most land-use systems exhibited higher SOC concentrations in topsoils than in the rest of the soil profile. In contrast, bare lands lacking vegetative cover showed homogeneous SOC distribution along the soil depth.

Table 3-3. SOC concentration within soil profiles in major land-use systems in Gergera watershed, Tigray region, northern Ethiopia. Values in parenthesis are standard errors. Values followed by the same superscripts are not significantly different for land-use systems at P = 0.05 based on LSD test

Soil depth	Exclosures		Rangelands		Croplands		Bare lands	
	N	Mean SOC (%)	N	Mean SOC (%)	N	Mean SOC (%)	N	Mean SOC (%)
0-15 cm	16	$1.94(\pm0.18)^{a}$	15	$1.47(\pm0.23)^{ab}$	21	$0.91(\pm0.09)^{a}$	8	$0.88(\pm0.17)^{a}$
15-30 cm	16	$1.16(\pm0.11)^{b}$	15	$1.66(\pm0.3)^{a}$	19	$0.74(\pm0.06)^{ab}$	6	$1.06(\pm0.15)^{a}$
30-60 cm	10	$0.82(\pm0.13)^{b}$	12	$1.11(\pm0.26)^{b}$	15	$0.42(\pm0.07)^{b}$	5	$0.72(\pm0.25)^{a}$
60-100 cm	7	$0.76(\pm0.2)^{b}$	11	$1.0(\pm0.14)^{b}$	11	$0.46(\pm0.06)^{ab}$	3	$0.71(\pm0.04)^{a}$
100-140 cm		-	3	$0.84(\pm0.5)^{b}$	1	0.57^{ab}		-
Overall weighted mean	49	$1.08(\pm0.10)^{A}$	56	$1.20(\pm0.12)^{A}$	67	$0.58(\pm0.04)^{B}$	22	$0.83(\pm0.09)^{B}$

3.5 SOC stock

3.5.1 SOC stock variations along the slope position

Within the study area, most of the exclosures were located on steep slopes (Decheemaeker *et al.,* 2006), back slopes, and ridge positions, where relatively higher erosion occurred. In contrast, most of the rangelands were in the valleys or the footslope positions, where greater soil deposition takes place. The bare land was also mostly located on backslopes and footslopes. Due to their topographic positions, the exclosures had very shallow soils or soils with higher amounts of rock fragments, which strongly affected the soil bulk density and SOC stock (Throop *et al.,* 2012).

The LME model analysis of slope position on SOC stock indicates that, in general, the relationship between slope position and SOC stock was significant (Table

3-4). Specifically, this was due to the effect of slope position in the rangelands and exclosures (Table A 4). A similar finding was also reported by Breuer *et al. (*2006); Rhanor (2011) and Grimm *et al.* (2008) who observed statistically significant differences in SOC stock along the slope position, with a higher SOC found in the footslope and toeslope than on the summit, shoulder and backslope. However, Parras-Alcántara *et al*. (2015) observed no effect of slope position on the distribution of SOC down the hillslope. This was due to the presence of different soil development conditions along the hillslope, where older soils were found to have higher SOC compared to the younger soils.

Table 3-4 Results of linear mixed-effect model analysis of effects of soil depth, land-use type, slope position and interactions of these factors on SOC stock.

Term	DF	F-ratio	P-value
Land-use system	3	5.03	0.0017**
Soil depth	4	9.61	0.0000**
Land-use system X soil depth	10	2.06	0.0238*
Landscape position	20	4.693	0.0000**
Land-use system X landscape position	2	1.76	0.1720

*N=194, No. of groups= 60, DF: degree of freedom Note: *=significant at p<0.05 and **=significant at p<0.01*

3.5.2 SOC stock distribution along the soil depth

Generally across the land-use systems, the SOC stock peaked in the topsoil, 0-15 and 15-30 cm layers (Table 3-5). These results are in line with those by Chibsa and Ta'a (2009) who observed a consistent decrease in SOC concentrations across the soil profile down to 60 cm depth in different land-use types in southern Ethiopia. The exception to this pattern in Gergera were bare lands and rangelands where although SOC stocks in the 0-30 cm were larger than in any deeper layer, the differences along the soil profile were not statistically significant. In contrast, the SOC stocks in exclosures and croplands were found decreasing with the depth from the soil surface down to 100 cm (Table 3-5). This might be explained by a translocation of clay particles and organic matter from the surface soil (Dou *et al.,* 2007 & Lorenz and Lal, 2005), as well as by redisposition of wind-blown degraded soil due to revegetation of exclosures.

Table 3-5. SOC stock (Mg ha^{-1}) along soil depth in major land-use systems in Gergera watershed. Values followed by the same capital letters within the same land-use system along the soil depths and by small letters in different soil depths in the same land-use system are not significantly different at P = 0.05 based on LSD test

Land-use system	Soil depth	N	Mean(±SE)
Exclosures	0-15	16	26.71(±3.09)^{a}
	15-30	16	16.44(±1.96)^{b}
	30-60	10	12.08(±1.95)^{b}
	60-100	7	12.19(±2.65)^{b}
	Average	49	21.24 ±2.75^{A}
Rangelands	0-15	15	23.19(±3.4)^{a}
	15-30	15	25.93(±3.94)^{a}
	30-60	12	19.01(±4.3)^{a}
	60-100	11	17.62(±2.46)^{a}
	100-140	3	15.39(±8.41)^{a}
	Average	56	22.03(±2.6)^{A}
Croplands	0-15	21	14.16(±1.51)^{a}
	15-30	19	13.55(±1.21)^{a}
	30-60	15	8.01(±1.33)^{b}
	60-100	11	8.32(±1.25)^{b}
	100-140	1	12.87^{ab}
	Average	67	11.3 (±0.90)^{B}
Bare lands	0-15	8	14.76(±2.84)^{a}
	15-30	6	19.04(±3.2)^{a}
	30-60	5	12.15(±4.09)^{a}
	60-100	3	12.22(±0.76)^{a}
	Average	22	10.49(±2.58)^{B}

Particularly in croplands, where the lowest SOC stock was observed, the 0-30 cm soil layer was significantly more C-dense than the deeper soil layers, possibly due to the cultivation of shallow-rooted crops.

The results regarding the SOC stock in the different depth intervals indicate the importance of the consideration of depths below the topsoil. On average, 36% of the total SOC stock was found in the soil horizon below 30 cm, with 45% (±7.56%) in the subsoil of rangelands (Figure 3-13).

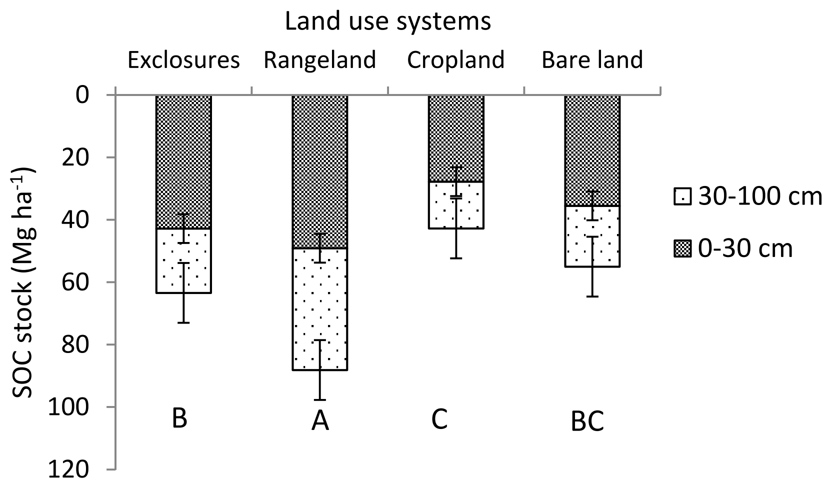

Figure 3-13 SOC stock (Mg ha^{-1}) in surface and sub-surface soils in major land-use systems in Gergera watershed. Different letters below bars indicate significant differences in 0-100cm SOC stocks among the land-use systems. P = 0.05 based on LSD test

3.5.3 SOC stock variations with land-use systems

Among the land-use systems, exclosures were implemented on degraded communal lands (Mekuria and Aynekulu, 2011) unsuitable for farming due to shallow soils and rugged relief. Nevertheless, the pairwise comparison of the marginal means indicates that exclosures had consistently higher SOC stocks (21.2 ±2.7 Mg ha^{-1}) than the croplands (11.3 ±0.9 Mg ha^{-1} and bare lands (10.5±2.6 Mg ha^{-1}) P > 0.05; Table 3-5). The exclosures stored SOC stocks similar to those of the rangelands (22.0±2.6 Mg ha^{-1}) although the latter were characterized by a more favorable topography for SOC storage. Mekuria *et al*. (2007) showed that exclosures had a high potential to rehabilitate degraded and abandoned grazing lands given the higher SOC recovery than that observed in open grazing land. Besides, rangelands have slower SOM decomposition rates than open grazing lands, resulting in relatively higher SOC accumulation (Sharma *et al.,* 2012).

All in all, reduced soil disturbance in rangelands and exclosures seems to be associated with higher SOC stocks (Theirfelder & Wall 2012). Fontaine *et al*. (2007) likewise found that rangelands with undisturbed soils were characterized by a reduced oxidation of the sub-soils, which could create a favorable environment for the microbial organisms and facilitate the decomposition of SOM. On the other hand, the topographic location of the exclosures could hinder the SOC stock accumulation due to slope steepness, shallowness of the soil, and high content of rock fragments, explaining why SOC stock in 15-year-old exclosures did not yet exceed that of rangelands. A non-significant difference in the SOC concentration of the exclosures and rangelands in Tigray was previously observed by Mekuria *et al*. (2014) who explained this finding by favorable conditions created through improved land management in the exclosures that facilitated plant growth, plant nutrient uptake, and decomposition and turnover rate of SOM. In contrast, the studies by Chen *et al*. (2012) measured 3.5 times as much SOM in exclosures than in grazing lands on a sand dune area after 25 years of grazing control in semi-arid areas of China. Given these contrasting findings, it is important to consider area-specific factors for SOC accumulation and decomposition, such as rainfall and other climatic conditions (Mekuria *et al.,* 2014). Besides, age of the exclosures should be considered when comparing them with other systems, as the SOC accumulation increases with time (Damene *et al.,* 2013; Chen *et al.,* 2012).

The cultivated lands had low SOC stocks, which were not statistically different from those of the bare land (Figure 3-13). The study by Fang *et al.* (2012) in a semi-arid loess plateau in China and a review by Lal (2010) also showed a lower SOC in cultivated lands than in any other land-use system under consideration. Similar results were reported by Bationo *et al*. (2006) in West African agro-ecosystems and Gelaw *et al*. (2014) in Ethiopia. The low SOC stock detected in croplands despite their location on deeper soils might be attributed to the poor, tillage-based management on these lands (Sharma *et al.,* 2012). Bationo *et al.* (2006) also mentioned that the erratic rainfall, inherent poor soil fertility, high soil and water temperatures, and low water holding capacity that meant the soil water depletion during longer dry spells, resulted in low SOC and productivity of these agricultural soils. Oades (1998) considered the soil structure deterioration and increased mineralization due to agricultural disturbance to be the main cause of SOC losses from cultivated lands. Moreover, Fontaine *et al*. (2007) found that a high disturbance of soil in croplands enhanced the process of mixing the surface OM into the deeper soils, resulting in the loss of SOC.

The statistical assessment of SOC stocks revealed that bare lands were significantly lower in SOC stock as compared with the rangelands and exclosures, whereas no significant differences were observed between the bare land and croplands. The similarity in SOC stock between these two land classes could probably be due to animal wastes and the crop residues deposited in the bare land areas for cattle. Besides, the bare land was located in significantly less stony areas in contrast to the croplands (Figure 3-13). However, the very low to non-productive status of the bare land still caused these lands to have low SOC stock.

3.5.4 Relationship of SOC to soil plant nutrients

The SOC status is an essential factor for soil quality, while other variables, i.e. soil water holding capacity, nutrient retention ability, aggregate stability, which help to develop erosion-resistant soil, affect the soil quality and are mainly affected by the SOC level in the soil (Lal, 2010). Therefore, it is important to identify the relationship of SOC concentrations with soil properties, total nitrogen (TN), available P, clay, cation exchange capacity (CEC), pH and exchangeable cations (Mg, Ca, K and Na) in the different land-use systems, soil depths and slope positions.

The SOC concentration showed positive and highly significant correlation ($r > 0.95$; at $p>0.05$) with TN in all land-use systems (Table A-2). In exclosures and rangelands, SOC concentration significantly correlated with CEC at $r= 0.29$ and 0.42 at $p>0.05$, respectively (Table 3-6). Moreover, the pH values had a low but statistically significant negative correlation with SOC in the exclosures ($r= -29$; at $p<0.05$).

No correlation between SOC and available P was observed in the exclosures and rangelands but it was statistically significant in the croplands and bare lands at $r= 0.25$ and 0.60; at $p>0.05$, respectively.

SOC concentration and clay content of the rangeland soils were positively but weakly correlated ($r=0.29$ at $p>0.05$) (Table 3-6). No correlation at all was observed in the other land-use systems. The reason could be the low level of clay in the soils because, according to Scholes & Hall (1996), the low-activity clay type found in the dryland areas is not capable of stabilizing large amounts of carbon. The low level of SOC especially in cultivated lands (Lal, 2010) could also explain the low capacity of the soil to retain the plant nutrients, which in turn results in the loss of SOC.

Table 3-6. Pearson's correlation matrix of SOC concentration with soil plant nutrients in exclosures (a), rangelands (b), croplands (c) and barelands (d). Insignificant correlations are not included. *= significant at p<0.05 and **=significant at p<0.01

(a) Exclosures

	OC	Total N	Exc. K	pH	EC	CEC
OC	1					
Total N	0.93**	1				
Exchangeable K	0.34*	0.13	1			
pH	-0.29*	-0.29*	0.07	1		
EC	0.31*	0.29*	0.17	-	1	
CEC	0.28*	0.21	0.32*	-	0.34*	1

(b) Rangelands

	OC	Total N	Clay	CEC
OC	1			
Total N	0.96**	1		
Clay	0.29*	0.29*	1	
CEC	0.4205**	0.4061**	0.35**	1

(c) Croplands

	OC	Total N	Available P
OC	1		
Total N	0.95**	1	
Available P	0.25*	0.20	1

(d) Bare lands

	OC	Total N	Available P
OC	1		
Total N	0.94**	1	
Available P	0.59**	0.48*	1

3.5.5 Relationship of SOC to rock fragment

Both SOC stock and concentration exhibited a negative although statistically low correlation with the amount of rock fragments in the soil (Figure 3-14 A&B). The correlations were most noticeable in the exclosures which also contained more of the soil coarse fraction (Figure 3-14 C&D). Bornemann *et al*. (2011) reported that SOC variations were proportional to the amount of rock fragments (and thus inversely to the amount of fine earth). At elevated58 contents of rock fragments in a soil there is a disproportionally higher organic input to the available fine earth material, which in turn could result in saturation of the soil with organic carbon (Bornemann *et al.,* 2011). This is because a limited amount of soil particles of <2mm in size can also only sequester a limited amount of SOC. Thus, through a more rapid saturation any excess biomass input can only be accumulated in a labile form that has a relative fast turnover time (Gulde *et al.,* 2008). Besides, the reduced solum may lead to a limited supply of available nutrients and limited space for plant growth (Poesen and Lavee, 1994), which could decrease the plant biomass that is an input for SOC development.

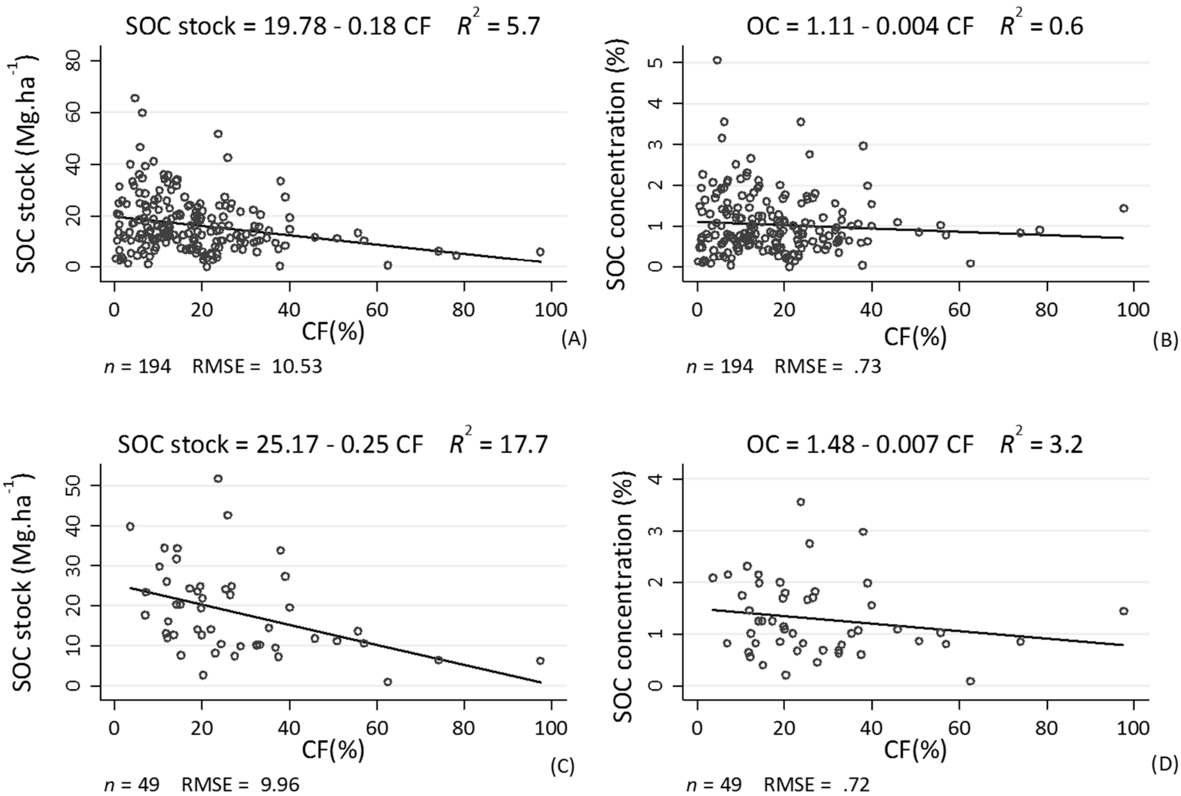

Figure 3-14 Fitted scatter plots of rock fragment and (A) SOC stock in all land-use systems (B) SOC concentration in all land-use systems; (C) SOC stock in exclosures and (D) SOC concentration in exclosures. CF=coarse fraction.

On the other hand, rock fragments also help to increase the infiltration rate through increased surface roughness and irregularity, and also to increase raindrop interception

(Zavala *et al.*, 2010). This in turn helps to conserve moisture and improve the biomass production (Danalatos *et al*., 1995) and thereby SOC accrual in semi-arid areas despite the soil volume lost to the presence of stones.

3.6 Terrestrial carbon stocks

The total stock of terrestrial organic carbon was found in this ranked order according to the examined land-use systems: exclosures (54±5 Mg ha^{-1}) ≈ rangelands (54±4 Mg ha^{-1}) > croplands (30±4 Mg ha^{-1}) ≈ bare lands (29±6 Mg ha^{-1}) (Figure 3-15). Bare lands did not store any biomass carbon but were as significant as croplands in terms of SOC stock. The higher SOC stock in rangelands than in exclosures with the presence of trees might be because of the longer time of SOM turnover in grasslands, which is twice as long as that in forest areas (Oades, 1998). However, in the exclosures with SOC originating predominantly from residues of the conifer species, a slow decomposition of the recalcitrant needle-leaf litter was likely responsible for the modest soil carbon deposits but significant litter stocks (Laganière et al., 2010).

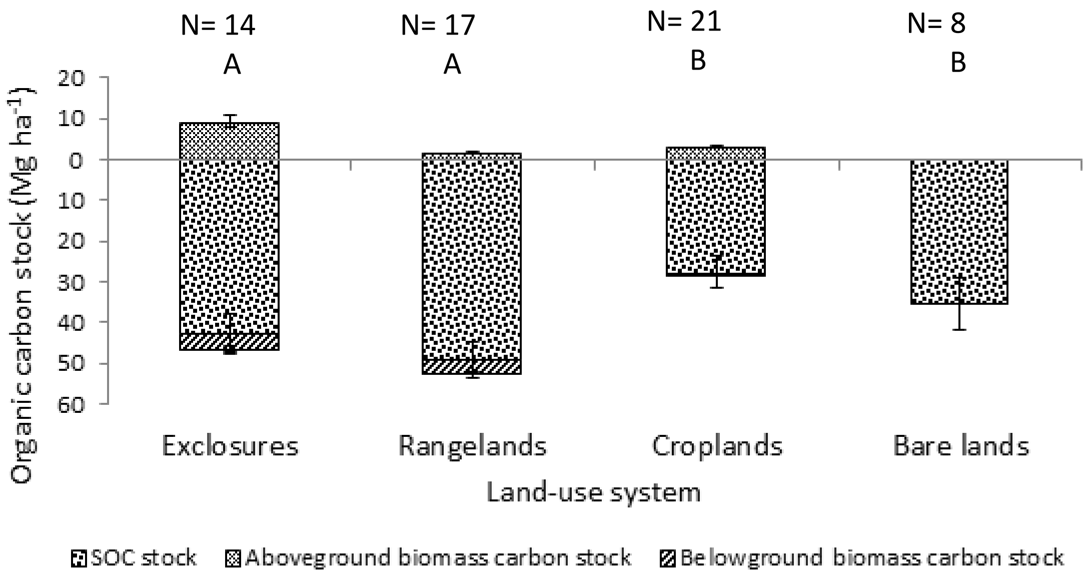

Figure 3-15. Mean values of terrestrial carbon stock in major land-use systems in the Gergera watershed, northern Ethiopia. Values indicated by the same letters are not significantly different at P = 0.05 based on LSD test.

In contrast, the addition of OM in the grasslands was mainly through depositions from the extensive root systems of perennial grasses. These residues are well incorporated into the soil and not readily accessible to the soil organisms, helping to slow down the decomposition process (Oades, 1998).

It is generally known that croplands under continuous cereal monoculture lose SOC, especially in semi-arid areas (Oades, 1998; Lobe et al., 2001). The severity of fertility loss in croplands is indicated by the SOC stock, which did not significantly differ from that of bare lands, highly degraded and abandoned areas.

Despite a poor soil quality in the exclosures, the result of poor natural fertility and erosion reflected by shallow soils and high rock fragment content, the biomass and SOC stock tended to be similar to that observed in the better quality soils in the highlands of central Tigray (Descheemaeker *et al.,* 2006). The significant carbon stocks in the exclosures of Gergera regardless their soil shallowness and unfavorable topographic location might be an indication of the positive impact of the IWM on SOC accrual. However, there is still a need to develop a comprehensive guideline for the management of exclosures in order to insure their long-term sustainability and improve the plant regeneration rate and biodiversity (Haregeweyn *et al.,* 2007). Particularly managing the species composition may help to increase the availability of wood and of animal feeds to satisfy the local demand while maintaining the main objective of soil rehabilitation.

3.7 SOC redistribution by erosion

3.7.1 Reference sites

The results of the LULC analysis (section 3.1) show that the reference sites selected for the ^{137}Cs study are located within the stable forest LULC class with a vegetation cover of 50-60% (Figure 2-5). This finding was also confirmed by a personal communication with local farmers in 2012.

The cesium values for the reference sites ranged between 0.93 and 0.1 Bq ha^{-1} (13.3 and < 1 Bq kg^{-1}) with 1 Bq kg^{-1} cesium being the lowest limit of detection (LLD). The highest cesium content was recorded in the surface soil (0-5 cm) (Table 3-7). The exponential reduction in cesium with depth (Figure 3-16) indicated the absence of or minimum erosion, which is the main criterion a reference site, should fulfill (Mabite et al., 2008, 2014). Owens et al. (1996) and Wallbrink et al. (1999) also reported cesium concentrated in the upper soil layers with no cesium beyond 30 cm depth in a typical reference site. The coefficient of variation (CV) of the cesium content of the reference sites in the study area averaged 16% (Table 3-7), thus above the 10% threshold for a suitable reference site (Mabit et al., 2014). This higher CV could probably be due to the heavier soil texture in reference Site 2, as clay particles can show a positive correlation with cesium (Theocharopoulos *et al* 2003). However, an even larger coefficient of variation (34%) was reported by Theocharopoulos *et al.* (2003), highlighting the challenge in identifying suitable reference areas with undisturbed soils, i.e. soils showing no or minimum soil erosion, particularly in cultivated fields.

Table 3-7 Descriptive statistics of the cesium distribution in different soil depths and average total cesium of the reference sites in the Gergera watershed

Statistical parameters	Cesium activity (Bq ha^{-1})						
	0-5cm	5-10 cm	10-20 cm	20-30cm	30-50cm	50-90cm	Total
Observations	3	3	3	3	3	2	3
Mean cesium (±SE)	0.86 (±0.06)	0.52 (±0.6)	0.17 (±0.06)	0.15 (±0.03)	0.13 (±0.01)	0.14 (±0.01)	2.35 (±0.22)
CV (%)	12.3	18.1	59.7	29.7	-	-	16.5
Maximum	0.93	0.6	0.28	0.2	-	-	2.69
Minimum	0.74	0.42	0.11	0.11	-	-	1.93

The cesium values in the reference sites in this study were higher than those measured in nearby countries. For example, Fernius (2002, cited in Norén and Spörndly, 2009) reported only 2-6 Bq kg^{-1} in the reference sites in Kenya and Rwanda, and Norén and Spörndly (2009) found a maximum value of 3.32 Bq kg^{-1}. Mekuria *et al*. (2012) reported a cesium content of 2026 Bq m^{-2} in reference sites in southern Ethiopia, which is also lower than the values in the Gergera watershed. Norén and Spörndly (2009) associated the low cesium in these areas to the ending half-life of cesium (30 years), beyond which cesium is expected to decay and become difficult to detect.

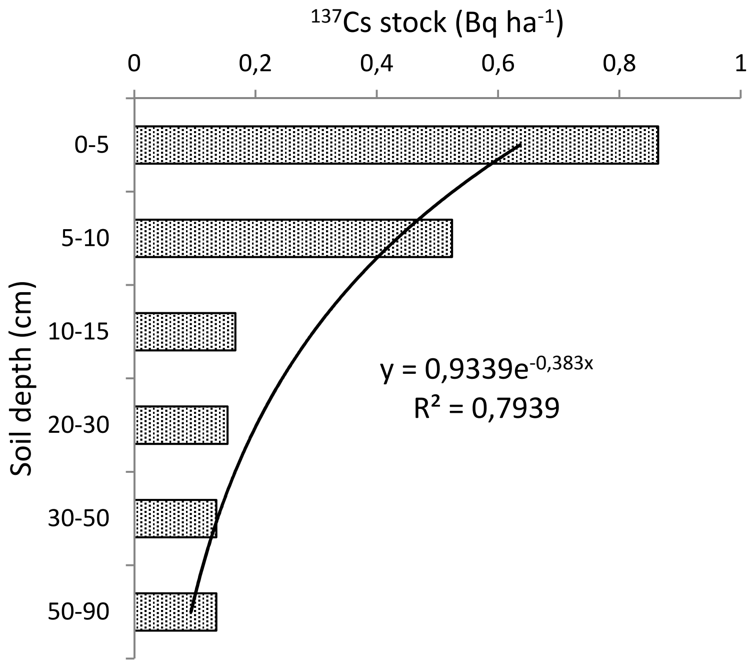

Figure 3-16 Exponential function of average cesium stock in reference sites in the Gergera watershed.

3.7.2 Cesium activity in sampling points

About 30% out of the total 171 samples from the different land-use systems showed cesium activity below the LLD. The largest share of soil samples (Table 3-8) that contained cesium below this limit was collected in the exclosures. This is characteristic of areas affected by soil erosion (Porêba 2006; He and Walling, 2000; Mabit *et al.*, 2014) and confirms the findings by Fikir (2005), who indicated that exclosures had been in general highly affected by erosion in the Gergera watershed prior to IWM establishment. Fewer samples with cesium activity below LLD were found in croplands, which could be because of the mechanical disturbance of the cropland soil, resulting in uniform distribution of cesium throughout the tilled layer (Li *et al.*, 2006).

Table 3-8. Proportion of samples per land-use system with cesium below the lower limit of detection (LLD)

Land-use type	N	Samples with cesium below LLD (<1Bq kg^{-1})	% of total samples with cesium below LLD
Exclosure	46	20	43.5
Cropland	62	10	16.1
Rangeland	43	10	23.3
Bare land	20	6	30.0

The 23% of the soil samples with cesium below the LLD in rangelands included one profile pit at the backslope with no detectable cesium at all, and two pits at the valley bottom that had no detectable cesium in the 0-5 cm and 10-15 cm depths. However, these two pits contained 0.67 and 0.18 Bq ha^{-1} of cesium in the intermediate 5-10 cm depth. Such cesium distribution patterns are possible in areas where soil with high cesium values is covered by sediment with no detectable cesium (Norén and Spörndly, 2009; Navas & Walling, 1992).

Only 13% of the values measured along the slope in the different land-use systems were higher than the cesium concentration of the reference sites, i.e. 26.07 (±2.43) Bq kg^{-1} (Table 3-9). Moreover, the cesium stock above 2.35(±0.22) Bq ha^{-1} (recorded at the reference site) was detected at only one site in the exclosures, indicating that most of the watershed area was affected by erosion (Martinez *et al*., 2009). The majority of the samples with a higher cesium activity than at the reference sites were found in the rangelands (Table 3-9), which are the areas considered as depositional areas (Martinez *et al*., 2009). Bare lands had no samples with a cesium concentration exceeding that of the reference value. However, it is important to note that the distribution of cesium might not be affected only by the direct erosion and redistribution process of sediments. Navas and Walling (1991) indicated that in semi-arid areas, the micro- and meso-topographic variation resulting in local vegetation cover differences could be one of the main reasons for highly localized variations in the cesium distribution.

Table 3-9. Proportion of sampling plots with total cesium activity higher than that in the reference sites (26.07 (±2.43) Bq kg^{-1}) in the land-use systems of the Gergera watershed

Land-use type	N	Sampling plots with cesium concentration above that at reference sites	% of sampling plots with cesium concentration above that at reference sites
Exclosure	14	2	14.3
Cropland	21	2	9.5
Rangeland	17	4	23.5
Bare land	8	0	0

3.7.3 ^{137}Cs distribution along soil depth and slope position in land-use systems

The LME model results show that soil depth, slope position and interaction of land-use system and slope position had a significant effect on the cesium distribution in the watershed (Table 3-10). The impact of land-use system and its interaction with soil depth was not statistically significant. It was not possible to identify the effect of the interaction of all three factors without overfitting the model.

Table 3-10. Results of linear mixed-effect model analysis of effects of soil depth, land-use type, slope position and interactions of these factors on cesium distribution in the Gergera watershed.

Factor	DF	F	P
Land-use system	3	1.26	0.29
Soil depth	2	6.74	0.0012**
Slope position	20	3.48	0.0000**
Land-use system X soil depth	6	1.69	0.1176
Land-use system X slope position	2	0.29	0.75**

N= 173, Note:=significant at p<0.05 and **=significant at p<0.01*

The mean cesium stock of the specific depths and slope positions (Table 3-11 and Figure 3-17) at the sampling sites indicates that the average cesium stocks were lower compared to those of the reference site, putting all the land-use systems regardless the slope positions in the same category of erosional area (Martinez *et al*., 2009). Further comparison of the cesium stock in the different soil depths and slope positions indicates the soil movement pattern. A significantly lower cesium content was observed in the lower depth than the middle depth and topsoil of the exclosures. Mabit et al. (2008) associated a higher cesium presence in the middle of a profile than in the upper part with the development of organic matter on the surface soil, which could reduce the cesium activity at the surface. A decreasing value with increasing sampling depth was observed in the backslope and particularly foot slope positions of exclosures, which is a common observation in uncultivated soils (He and Walling, 2000). Due to only few sampling plots at the ridge of the rangelands that hardly occurred at the ridge slope, these results are inconclusive.

Considering the distribution along the slope positions, the total cesium in exclosures was on average significantly higher in the ridge positions and footslopes than on the backslopes, which showed the lowest cesium values (Table 3-11). Similarly, Navas and Walling (1991) found middle slopes to be the eroded areas in uncultivated soils of semi-arid Australia. This cesium distribution patterns in exclosures of Gergera could be an indicator of the soil movement from the backslope downwards, resulting in higher soil sedimentation on the footslopes (Wang et al. 2011).

The average total cesium in the rangelands was highest in the footslopes and lowest in the backslopes. Akin to exclosures, rangeland average total cesium at the valley bottom and ridge was statistically similar to that at the ridge (Table 3-11). Nearly homogeneous distribution of cesium occurred in the ridge, backslopes and footslopes of the croplands (Figure 3-17 and Table 3-11), which is probably due to continuous soil tillage (Mabit et al., 2008; He and Walling 2000). All slope positions where the bare lands were sampled were statistically non-significantly different from each other (Table 3-11).

Table 3-11 Pairwise comparison of mean total cesium distribution (mean ±SE) along slope positions in different land-use systems in 0-20 cm soil depth in the Gergera watershed

Land-use system	Slope position	N	Mean cesium concentration (±SE) (Bq kg^{-1})	Range of cesium concentration (Bq kg^{-1})	Mean cesium stock (Bq ha^{-1})	Range of cesium stock (Bq ha^{-1})
Exclosure	Ridge	3	25.4(±11.3)^{a}	10.3 – 47.8	1.27(±0.25)^{a}	0.05-1.1
	Backslope	6	5.9(±1.4)^{b}	3.0 – 11.6	0.21(±0.15)^{b}	0.03-0.28
	Footslope	5	15(±3.1)^{a}	5.8 – 27.5	0.77(±0.18)^{a}	0.03-1.24
	Overall mean	14	**12.96(±2.9)** ^{A}	**3 – 47.8**	**0.59**(±0.15)^{A}	**0.13-2.41**
Rangeland	Ridge	2	21.1(±5.4)^{ab}	15.7 -26.5	1.26(±0.31)^{ab}	0.29-0.57
	Backslope	3	8(±5)^{b}	3.0 -13.0	0.48(±0.31)^{b}	0.04-0.43
	Footslope	4	30.2(±8.1)^{a}	6.3 -38.3	1.37(±0.31)^{a}	0.19-1.1
	Valley bottom	8	12.1(±2.9)^{b}	3.0- 28.6	0.64(±0.13)^{b}	0.04-0.95
	Overall mean	17	**15.2(±2.7)** ^{A}	**3.0- 38.3**	**0.78(±0.12)** ^{A}	**0.15-1.68**
Cropland	Ridge	6	14.9(±1.5)^{a}	10.1 -31.6	0.82(±0.19)^{a}	0.06-0.45
	Backslope	8	15.9(±2.9)^{a}	3.0 – 24.1	0.78(±0.17)^{a}	0.12-0.65
	Footslope	3	12.6(±5.9)^{a}	4.4 – 35.3	0.75(±0.19)^{a}	0.05-0.95
	Valley bottom	4	10.3(±2.4)^{a}	3.8 – 14.7	0.71(±0.19)^{a}	0.07-0.83
	Overall mean	21	**13.8(±1.8)** ^{A}	**3.0-35.3**	**0.76(±0.09)** ^{A}	**0.16-2.2**
Bare land	Ridge	2	9.6(±6.6)^{a}	2.0 – 16.1	0.52(±0.31)^{a}	0.05-0.53
	Backslope	3	8.5(±2.5)^{a}	4.9 – 11.5	0.499(±0.25)^{a}	0.06-0.43
	Footslope	3	5.9(±2.6)^{a}	2.7 – 11.0	0.27(±0.22)^{a}	0.05-0.48
	Overall mean	8	**7.8(±1.8)** ^{A}	**2.0-16.1**	0.40(±0.09)^{A}	**0.16-0.89**

**Means followed by the same letters within a column for the same land-use system are not significantly different at P = 0.05.*

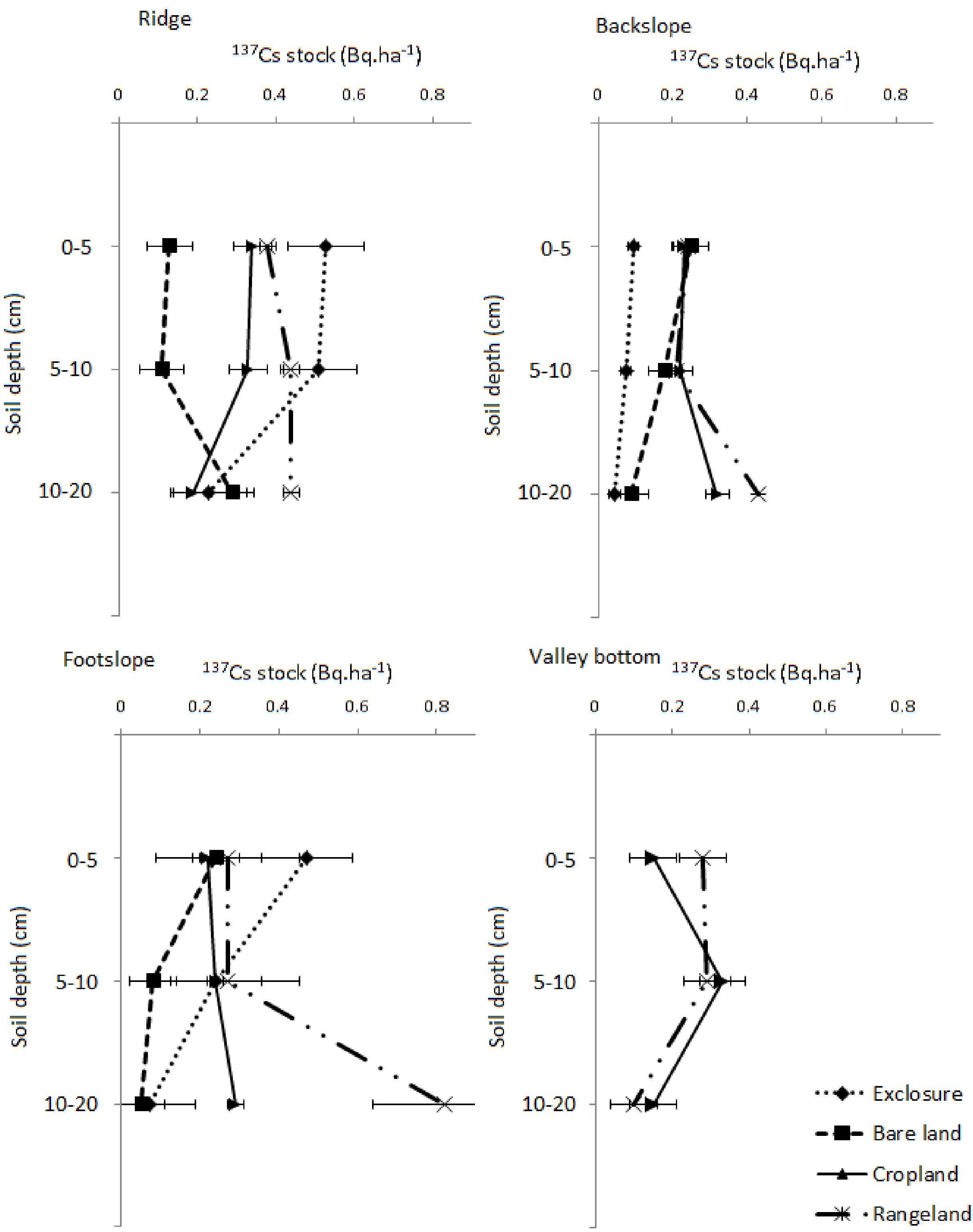

Figure 3-17 Distribution of cesium stock in 0-5, 5-10 and 10-20 cm soil depth in different slope positions of major and-use systems in the Gergera watershed.

3.7.4 Relationship of cesium and SOC distribution

The correlation analyses of SOC stock and the cesium difference compared to the reference site for each land-use system indicated no relationship between these two variables in the rangelands, croplands and bare lands (Figure 3-18). The results of the rangelands agree with the findings in other studies (Hancock *et al*., 2010; Martinz *et al*., 2010) reporting no relationship between SOC and cesium in undisturbed soils. On the other hand, in highly disturbed soils such as in cultivated lands, the erosion of SOC and cesium usually follows the catena, and higher cesium is associated with high SOC content due to the redistribution of both by erosion and deposition (Mabit *et al*., 2008 and 2014; Martinez *et al*., 2010). The non-significant correlation in the croplands in this study could probably be due to the very low SOC content, which masked its relation to the cesium content. Sedimentation of soils with very low SOC could also show no relation of SOC to cesium. Land use and land management in the local area has also been indicated as influencing soil erosion and sedimentation rates more strongly than other factors, such as lithology and soil type (FAO/IAEA, 2001).

In contrast to all other land-use systems examined, a significant correlation between the cesium difference and total SOC stock was observed in the exclosures (R^2=0.47, $p<0.05$). This is in line with the finding by Wang *et al*. (2011), who revealed a significant relation between SOC and cesium after re-vegetation in semi-arid areas of China. The authors explained the strong correlation between SOC and cesium by a decrease in erosion and build-up of SOC, as the transport of both the SOC and cesium was in the same direction towards lower slope position (Guoxiao *et al.*, 2008). Erosion areas commonly show lower levels of cesium and SOC than the depositional areas with high cesium values because the latter areas have a higher chance to accumulate organic matter and hence high SOC (Mabit et al., 2014).

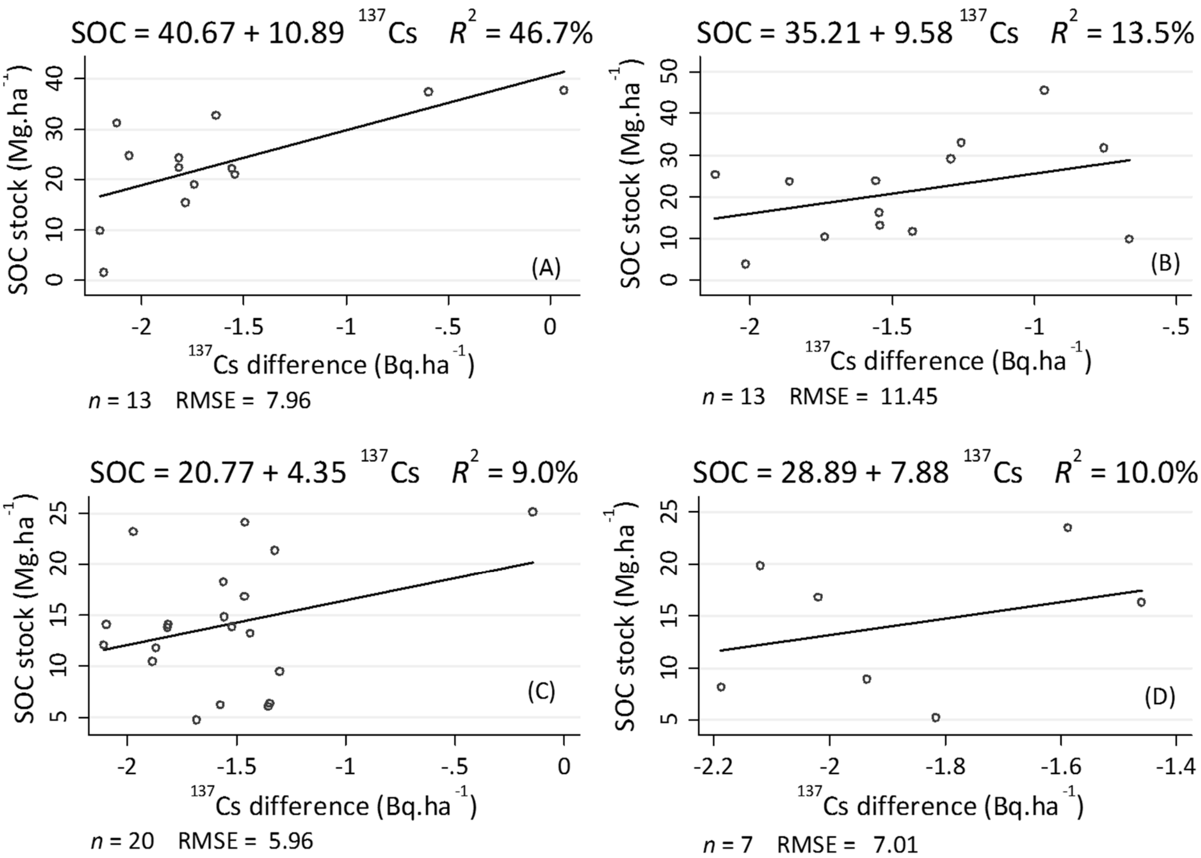

Figure 3-18 Total SOC stock (0-20 cm soil depth) plotted against cesium difference between reference site and sampling sites in (A) Exclosure, (B) Rangeland, (C) Cropland and (D) Bare land

Intriguingly, within the different land-use systems (section 4.3) except for the rangelands, slope position had no significant effect on the SOC stock in all systems. On the other hand, the cesium analysis indicates a significant effect of slope position for the exclosures and rangelands (Table 3-11). This significant difference in the cesium content but a non-significant difference in SOC in the exclosures along the slope positions can be due to a considerable soil erosion prior to the establishment of exclosures under the IWM (Mabit *et al.*, 2008). This is particularly true for the backslopes, most eroded slope positions. The effective treatment of degraded hillslope has been found to reduce soil erosion and runoff generation (Descheemaeker *et al.*, 2006), providing suitable conditions for SOC development. Hence, knowing the distribution of SOC along the hillslope is important to better understand the SOC distribution in a catchment and the response of the landscape to different management options (Hancock *et al*. 2010).

The lack of relationship of cesium distribution to slope position in croplands and bare lands (Table 3-11) was also reported by Sutherland (1991) who studied various land-use systems where however the cesium distribution was at random rather than a result of erosion and sediment redistribution. In contrast, Hancock *et al*. (2010) found a strong correlation between slope position and SOC concentration, indicating the strong influence of geomorphology and hydrology on the SOC concentration.

In general, the cesium analysis in Gergera indicates that even though the watershed underwent severe erosion which reduced the cesium stock there considerably below that of the reference site, it still is possible to identify soil movement when detectable levels of cesium are present in the area. This is evident by the findings in the exclosures and rangelands, where significant variations can be observed in the cesium content in different slope positions and soil depths.

4 GENERAL DISCUSSION

4.1 Land-use land-cover change since the IWM introduction

The LULC change analysis based on Landsat imagery before and after the introduction of IWM practices in the Gergera watershed revealed some improvements in the vegetative cover of the degraded area over the 20 years. These included changes associated (i) with land conversions, such as from cropland to forest land and bare land to rangeland (Figure 3-1) as well as (ii) with an increased land cover within same land class, such as due to forest regrowth in the exclosures. Fikir *et al.* (2008) also indicated increased forest cover, from 32.45 ha in 1994 to 98 ha in 2005, in the Gergera sub-watershed in southern part of the study area. To approximately same spatial extent (about 30%) negative land transformations persisted, such as conversions from cropland to bare land, rangeland to cropland, and forest land to rangeland, resulting in reduced land cover in the area.

A LULC change analysis is crucial in addressing the land quality in relation to climate change monitoring and for understanding the land-cover degradation and regeneration (De Sherbinin, 2002; Mochizuki and Murakami, 2012). Based on remote sensing, LULC provides seamless and periodic observations of the Earth and especially in semi-arid areas such as the Gergera watershed, where the extent, degree and nature of land-use change remains uncertain, LULC analyses can yield vital information and might be the only viable source e.g. of forest area estimates (Scholes and Hall, 1996). Information on LULC change with the use of the medium resolution images of Landsat that capture more spatially detailed information is particularly useful for land monitoring in sparsely vegetated, heterogeneous drylands.

Due to the unbiased estimation of the generalization error of the random forest (RF) algorithm (Rodriguez-Galiano *et al.,* 2012), the classification result is robust and the accuracy assessment is conducted by the model itself. The RF method performed the classification with a good accuracy. It also helped to reduce the errors that could have been incurred by the separate classification of the images for detecting the change classes. Mochizuki and Murakami (2012) recommended the method for multi-dimensional image classification purposes among other tested methods due to its high accuracy. However, some factors remained that affected the accuracy of the method, such as the terrain complexity and distinct temporal rainfall pattern in arid and semi-arid areas, which result in temporal variations of the phenology patterns (Hüttich *et al.,* 2011). Yin *et al*. (2014) also mentioned that the type of image used to classify the LULC changes was a determining factor where the use of high resolution images was found to yield high-accuracy change-detection maps.

The Landsat based map developed in this study for the Gergera watershed data is an important indicator for changes in soil parameters, such as SOC stocks (Nosetto *et al*., 2006) and aided in selection of the suitable sampling sites in the delineated LULC classes and therefore in overall spatial framework for the field assessment of the aboveground, belowground and soil-inherent carbon stocks in major land-use systems of Gergera.

4.2 Biomass carbon stocks

Several studies on biomass and carbon estimation for *J. procera,* the main forest tree species in exclosures, have been done in Ethiopia (Bazezew *et al*., 2015; Bazezew *et al*., 2015; Teklay and Gebreslassi, 2014). However, these estimations have been based on general allometric functions, as often the case due to time- and resource constraints for the field biomass sampling (Ketterings *et al*., 2001). Moreover, for the same reason there was no biomass function specifically developed for *J. procera* growing in areas with similar agro-climatic conditions in Ethiopia or elsewhere. In this respect, the biomass functions developed based on empirical data collected in 15-year-old exclosures under semi-arid conditions of northern Ethiopia makes an important input to the knowledge on biomass and carbon stock accrual in *J. procera* and *A. abyssinica* dominated forests. Moreover, these allometric equations might be of use for studies on ecosystem processes, such as organic matter dynamics in regenerating forests under similar agro-ecological conditions in Sub-Saharan Africa, subject to local calibrations (Baumert & Khamzina, 2015).

The species-specific R:S ratios and the belowground biomass data that are even less frequently reported given that few studies attempted time consuming, laborious root excavations, are nevertheless important in estimating the ecosystem biomass and carbon stocks. As the study in Gergera has shown, the roots contribute a large share in tree biomass stock, ranging from 42% in *J. procera* to 56% in *A. abyssinica*. Therefore, neglecting this belowground contribution would result in a significant underestimation of the ecosystem carbon pools and stocks.

Another important carbon pool was identified in the litter layer, generally reported as considerable in reforestation and afforestation systems (Laganière et al., 2010). This is particlarly so under arid conditions where the litter decomposition is constrained due to moisture limitations and is therefore conserved in a relatively long-term carbon pool (Khamzina *et al*., 2016).

There is a common argument that afforestation/reforestation of semi-arid areas results in a higher carbon sequestration, but at the same time jeopardizes the groundwater recharge due to the high evapotranspiration by trees (Nosetto *et al.*, 2006). However, in the Gergera watershed this might not always be true for several reasons. First, the rugged relief does not allow the rainwater to easily infiltrate into the soil (Huang *et al*., 2012). Next, the slope steepness coupled with low vegetation cover was the cause of soil erosion and run-off that destroyed the crops in cultivated lands and resulted in floods downstream (Fikir, 2005). Under such conditions, increasing vegetation cover could significantly increase the depth of the wetting zone in the soil by intercepting the run-off (Huang *et al*., 2012). Consequently, Fikir *et al*., 2008 observed both the increased woody vegetation cover and the groundwater recharge under the IWM interventions in the study watershed. This, therefore, indicates that both objectives can be achieved through the forestation activities with a proper choice of species on suitable locations. Mekuria *et al.* (2009) however warned that despite the positive impacts, expansion of exclosures might create pressure on the remaining grazing lands, leading to their overgrazing and degradation. Although the evidence of significant carbon stocks measured in Gergera rangelands in 2012 is not in support of this argument, further expansion of the exclosure areas should be recommended with caution, considering the possible trade-off with the rangeland management.

Improving the vegetation cover in cultivated areas enhances organic carbon stocks, positively impacting on soil fertility and agronomic production (Lal, 2003). The review by Lal (2006a) indicated that SOC improvement in the root zone estimated at 1 Mg C ha^{-1} resulted in a 20 to 40 kg ha^{-1} Mg^{-1} of SOC increase in wheat, 30 to 60 kg ha^{-1} Mg^{-1} SOC in bean and 200 to 300 kg ha^{-1} Mg^{-1} SOC in maize production. In turn, crop types with prolific and deep rooting systems, such as pearl millet (*Pennisetum glaucum*), Lablab bean (*Dolichos lablab)* and sorghum *(Sorghum bicolar)* (Creswell and Martin, 1998) generally have a significant impact on SOC improvement on agricultural lands (Lal, 2003). However, root biomass has usually been neglected as an organic carbon input in the agro-ecosystems often due to the methodological difficulties in its determination (Hairiah *et al*., 2001).

Among the SOC sequestration measures that may be more or less applicable under different environmental conditions, zero tillage coupled with crop residue application, manuring, legume-based rotations and integrated nutrient management were found to be effective in drylands (Lal, 2010). Besides, agroforestry practices in farming communities with small landholding size, as in the study area, increased land productivity while achieving the carbon sequestration target (Seeberg-Elverfeldt *et al.,* 2007). Thus the integration of these two land-use strategies would play an important role in the climate change mitigation and at the same time address the food security issue, critical in semi-arid Ethiopia.

4.3 Variation of SOC among the land-use systems

The results of this study evaluated SOC stocks throughout the 100-cm soil profile in the four land-use systems in Gergera (Figure 3-13). On average, 36% of the SOC stock was found in the sub-soil, between 30 and 100 cm depth, ranging from 35.2% (±7.0%) in the croplands to 45% (±7.6%) in the rangelands. Batjes (2001) also reported that at the global scale the upper 30 cm contained only about 50% of the SOC. Next to the significant SOC level, also the stability of the organic carbon in the deeper soil, that can provide a longer storage than that the surface soil (Rumpel and Knabner, 2011), adds to the importance of considering the deeper SOC in the carbon accounting.

When comparing carbon stocks among the different land-use systems, it is important to note that the SOC saturation might not be detected through the SOC stock values. SOC saturation could occur in stony soils because of the limited amount of fine earth material due to the dominance of coarse fractions (Bornemann *et al.,* 2011; Schlesinger, 1990) but the SOC stock would nevertheless be low as observed in the case of Gergera exclosures where considerable rock fragment content was detected. Specifically, the SOC concentration of exclosures (1.94±0.18%) was significantly higher than that of rangelands (1.47±0.23%) in the 0-15cm depth (Table 3-3), whereas the difference was insignificant for SOC stocks in these land-use systems (21.24±2.75 Mg ha^{-1} in exclosures vs. 22.03(±2.6) Mg ha^{-1} in rangelands (Table 3-5). This discrepancy shows that reporting SOC concentrations next to SOC stocks is imperative for soils with considerable amounts of coarse fragments. This information will not only help understand the issue of SOC saturation but be of use for guiding the management efforts to increase SOC levels in stony soils.

The rangelands and exclosures, which were protected from direct human and livestock interference in Gergera, showed larger SOC stocks than the other systems. However, the adverse effect of soil shallowness and presence of rock fragments on the SOC storage in the exclosures was indicated by the comparison of the SOC concentration of the exclosures and the bare lands. The considerable SOC presence in the rangelands could be due to the dominance of deeper alluvial soils and a higher soil moisture, which can increase SOC sequestration by deep-rooted plants, slowing down the release of soil carbon (Sharma *et al*., 2012). Nosetto *et al.* (2006) indicated that exclosures required a longer time for SOC build-up, hence the IWM intervention period was found to be important for detecting any change in SOC stock. Therefore, soil rehabilitation in Gergera exclosures is likely to become more evident over time due to continuous vegetation (re)growth there.

The SOC concentration in Gergera croplands averaged 0.67(±0.05), which is below 1%, indicating the severely low fertility status of the soils (Van De Wauw, 2005).

According to Lal (2015), a threshold SOC concentration of at least 1.5% should be maintained in order to sustain the cropland productivity. The inherently low fertility of the soil, erratic rainfall resulting in heavy erosion and loss of soils, and high temperatures have been among the main natural causes of the low SOC content and poor fertility of the croplands. Besides, the low input and extractive cropping practice due to the high cost and low availability of mineral fertilizers (Assefa, 2015) accelerated the loss of SOC and nutrients in these lands. Numerous studies showed that crop residue removal from the soil for different purposes aggravated the nutrient extraction and exhausted the soil fertility (e.g., Ashagrie *et al*., 2007; Fantaw *et al*., 2007). These current cultivation practices would need to be dramatically altered to improve the cropland productivity in Ethiopia.

Various improved soil management practices are recommended for enhancing cropland productivity and SOC sequestration. Sharma *et al.* (2012) indicated that minimum soil disturbance and balanced fertilizer application can restore the SOC lost through human activities. Conservation agriculture, a set of practices that integrates (i) a continuous, minimum mechanical disturbance of the soil, (ii) permanent soil cover and (iii) crop rotation (Theirfelder and Wall 2012; Theirfelder *et al*., 2013) and agroforestry, the integration of annual crop, woody plant and animal farming in some spatial or sequential arrangement (e.g. Nair *et al.,* 2009; Seeberg-Elverfeldt *et al*., 2007), have been often recommended for enhancing soil fertility, SOC sequestration, land productivity and rural livelihoods in sub-Saharan Africa.

4.4 Effect of erosion on SOC redistribution

The transect sampling was found to be appropriate in soil sampling for the assessment of the ^{137}Cs and SOC distribution in the Gergera watershed, covering 144 km^2. Mabit *et al.* (2014) also stated that the conventional grid sampling method might not be suitable for such large areas. This confirms that soil sampling for the cesium assessment could not be a ‘one-size-fit’ approach, but rather should be adjusted based on the size of the area under investigation (FAO/IAEA, 2001).

In all the samples collected from the different land-use systems in Gergera, the cesium values below the limit of detection indicate the persistence of soil erosion processes in the area. However, the insignificant relationship of SOC and cesium contents could also be an evidence of reduced soil erosion hence SOC redistribution and development. In this respect, the consideration of specific slope position is important when examining the SOC erosion and deposition processes by the trace ^{137}Cs isotope analysis (Wang *et al.,* 2011; Guoxiao *et al.,* 2008; Mabit *et al.,* 2014). The significance of SOC-cesium relationship is likely higher in severely disturbed soils and is nil in undisturbed soil, as was observed e.g. in rangelands by Martinez *et al*. (2010). This pattern held true in Gergera watershed for the undisturbed rangelands and bare land. However, there was also no relationship in the land-use systems with highly disturbed soils of croplands, which could probably be due to the tillage practice resulting in mixing up of soil horizons. The soil development level at each sampling point could be even more important in explaining SOC content than the slope position (Parras-Alcántara *et al.,* 2015).

5 OVERALL CONCLUSIONS AND RECOMMENDATIONS

This study contributes to the impact evaluation of IWM strategies introduced in Ethiopia in the 90-s, by using biomass and carbon stocks as indicators of ecosystem rehabilitation. In particular, the study focused on LULC change in the Gergera watershed in semi-arid Ethiopia and assessed the above- and belowground biomass and carbon pools in major land-use systems, i.e. forest exclosures, rangelands, croplands and bare lands after the IWM interventions. In absence of reference data before the introduction of IWM practices, the evaluation of changes in ecosystem carbon stock is a challenging task. To this end, various methodologies were employed to uncover the possible trends and assess the current land state in the study watershed. The main results can be summarized and conclusions drawn as follow:

- The LULC change analysis using Landsat satellite imagery acquired in 1994, 1995, and 2014 reveals that LULC conversions have occurred on almost 30% of the watershed area, slightly more so towards an increased vegetation cover. The latter might be associated with the IWM interventions, particularly in degraded forest areas where exclosures were established to regenerate the tree cover.
- Among the examined LULC classes, significantly higher aboveground biomass and carbon stocks were measured in the exclosures, while their stocks belowground were comparable with those in the rangelands, characterized by deeper soils and vegetated with deep-rooted perennial grasses.
- The findings of the considerable biomass stocks belowground, in tree roots in the exclosures and grass roots in rangelands, point out that the belowground biomass pools need to be estimated and included for a more precise carbon accounting, particularly in semi-arid ecosystems.
- Significant variations in SOC stocks were observed depending on the land-use system, soil depth and slope position examined. Total SOC stock was found to be highest in rangelands and exclosures. As the exclosures had been established primarily on degraded hillslopes, these results are another indication of the SOC

restoration after the IWM intervention. Long-term management strategies for exclosure areas need to be developed following their rehabilitation.

- The croplands exhibited the lowest SOC stocks overall, to the point that no significant differences with SOC contents of the bare land were observed. Such low status of cropland soil fertility is alarming, requiring appropriate management interventions.
- The examination of SOC distribution over the entire soil profile showed that soils below 30-cm depth contained 36% of total SOC, revealing the importance of frequently neglected sub-soil carbon stocks in carbon accounting.
- The significant correlation of SOC and trace cesium isotope in the exclosures could be attributed to the build-up of SOC there, corroborating the results of LULC assessment and SOC field survey.
- The overall comparisons of cesium status in the undisturbed reference site and examined land-use systems suggests that soil erosion processes persist in the watershed, particularly on the backslopes. The significantly higher cesium stock at the footslopes indicates deposition of soil eroded from the up slopes. The use of cesium analysis for revealing erosion and deposition processes was not effective in the croplands, probably due to the soil tillage practices.

Based on the above-stated findings and conclusions, the following methodological recommendations can be made for considering them in further studies on ecosystem carbon stocks in northern Ethiopia and in similar semi-arid regions:

- Based on the outcomes of the LULC change analyses, the land monitoring with a high temporal resolution should be conducted in Gergera to clearly reveal the changes in vegetation cover at regular intervals. This would help to monitor the success of IWM program over time and support the spatial guidance of IWM strategies e.g. by revealing the most vulnerable areas, still requiring interventions.
- Determining species-specific allometric relationships and root: shoot ratios for estimating above- and, in particular belowground biomass stocks are needed for a more precise estimation of carbon stocks in the biotic pool. The functions

developed for *J. procera* and *A. abyssinica* growing in the Gergera watershed can be of use for studies elsewhere, subject to local calibrations.

- For the determination of SOC concentrations, the conventional WB method has proved to be as reliable as that based on costly elemental analyses and can be therefore universally applicable for carbon-poor, non-calcareous soils common in semi-arid regions. Development of a site-specific correction factor for the original WB would help to contribute to an internationally accepted accuracy of quantification of SOC.
- In SOC investigations in shallow soils with a large content of rock fragments, it is important to consider both, carbon concentration and stock, to reduce misinterpretations of the soil fertility status.
- Assessments of SOC should always include the subsurface soil as it contains significant amounts of carbon, which are also more stable than those found in the topsoil.
- Erosion assessment using cesium distribution as an indicator can be conducted even when a small amount of cesium is found in an area compared to the reference site. This is because significant variations within the area can still be indicative of local erosion and deposition sites at a given level of cesium in the area.
- The results of this study provide information on the current conditions and, in part, dynamics of carbon in major land-use systems in the Gergera watershed after the IWM interventions. Subsequent monitoring and analyses should be performed to capture further impacts of the interventions and to respond to changes by adjusting the management strategies in the watershed.

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7 APPENDICES

7.1 Appendices A

Table A 1: Soil nutrient level classification according to Hazelton and Murphy (2007)

Soil chemical property	**Categorization of soil plant nutrient levels in the soil**					
	Extremely low	**Very low**	**Low**	**Moderate**	**High**	**Very high**
SOC (%)	<0.4	0.4-0.6	0.6-1.0	1.0-1.8	1.8-3.0	>3.0
Cation exchange capacity (CEC, cmol kg^{-1})	-	<6	6–12	12–25	25–40	>40
Total N (%)		<0.05	0.05-0.15	0.15-0.25	0.25-0.5	>0.50
Available P (Av. P, mg kg^{-1})		<5.0	5.0-10	10-17	17-25	>25
Exchangeable Na (Exc. Na, cmol kg^{-1})		0-0.1	0.1-0.3	0.3-0.7	0.7-2.0	>2.0
Exchangeable Mg (Exc. Mg, cmol kg^{-1})		0-0.3	0.3-1.0	1.0-3.0	3.0-8.0	>8.0
Exchangeable Ca (Exc. Ca, cmol kg^{-1})		0-2.0	2.0-5.0	5.0-10	10-20	>20
Exchangeable K (Exc. K, cmol kg^{-1})		0-0.2	0.2-0.3	0.3-0.7	0.7-2.0	>2.0
Electrical conductivity (EC, dS m^{-1})		**Non-saline** <2	**Slightly saline** 2-4	**Moderately saline** 4-8	**Highly saline** 8-16	**Extremely saline** >16
pH	**Strongly alkaline** 9.0–8.5	**Moderately alkaline** 6.0–5.6	**Slightly alkaline** 6.5–6.1	**Neutral** 7.3–6.6	**Slightly acidic** 6.5–6.1	**Moderately acidic** 6.0–5.6

Table A 2: Pearson's correlation matrix of soil plant nutrients across the land-use systems

	SOC	TN	Av. P	Exc.K	Clay	Exc.Ca	Exc.Mg	Exc.Na	pH	CEC
SOC	1.00									
TN	0.95**	1.00								
Av. P	-0.01	-0.06	1.00							
Exc. K	0.05	0.03	0.43**	1.00						
Clay	0.15*	0.16*	-0.14	0.03	1.00					
Exc. Ca	0.08	0.11	0.07	-0.06	0.32**	1.00				
Exc. Mg	0.09*	0.11*	-0.15*	0.13	0.32**	0.53**	1.00			
Exc. Na	0.14*	0.16*	-0.15*	0.17*	0.23**	0.25**	0.38**	1.00		
pH	-0.23**	-0.24**	0.23**	0.08	-0.14	-0.13	-0.22**	-0.11	1.00	
EC	0.12	0.11	-0.01	0.06	0.13	0.29**	0.24**	0.63**	-0.22**	1.00
CEC	0.37**	0.35**	-0.24**	0.29**	0.35**	0.09	0.44**	0.42**	-0.26**	0.26**

Note: *Significant at p<0.05 and ** significant at p<0.01, refer to Table A 1 for description of abbreviations

Table A 3 Pairwise comparison of mean values of soil plant nutrients in the different land-use systems. Values in parenthesis are standard errors. Values followed by the same letters within each column are not significantly different at P = 0.05 based on least significant difference test

Land use system	N	Clay	pH	N	P	K	CEC
Exclosure	49	17.22(0.64)^{B}	7.03(0.05)^{A}	0.12(0.01)^{A}	6.59(1.4)^{B}	0.09(0.01)^{BC}	10.07(0.45)^{B}
Rangeland	56	21.42(1.10)^{A}	7.04(0.07)^{A}	0.13(0.01)^{A}	4.8(0.55)^{B}	0.1(0.01)^{AB}	11.32(0.5)^{A}
Cropland	67	19.77(0.98)^{A}	7.16(0.04)^{A}	0.07(0.01)^{B}	14.6(2.1)^{A}	0.12(0.02)^{A}	9.02(0.29)^{B}
Bareland	22	19.68(0.98)^{AB}	7.15(0.09)^{A}	0.09(0.01)^{B}	7.2(1.3)^{B}	0.05(0.01)^{C}	9.51(0.49)^{B}

Table A 4: Pairwise comparison of mean SOC stock along slope position in different land-use systems. Values followed by the same letters in different slope positions within the same land-use system are not significantly different at P = 0.05 based on the least significant difference test

Land-use system	Slope position	N	Mean SOC stock (Mg ha^{-1})	SE
Exclosure	Ridge	7	18.9[a]	3.63
	Back slope	21	17.2[a]	2.09
	Foot slope	21	19.1[a]	2.09
Rangeland	Ridge	6	19.8[b]	3.92
	Back slope	6	40.9[a]	3.93
	Foot slope	12	20.9[b]	2.77
	Valley bottom	32	18.4[b]	1.69
Cropland	Ridge	13	11.2[a]	2.66
	Back slope	26	11.8[a]	1.88
	Foot slope	12	12.7[a]	2.77
	Valley bottom	16	10.8[a]	2.39
Bare land	Ridge	4	10.9[a]	4.79
	Back slope	12	16.9[a]	2.77
	Foot slope	6	13.8[a]	3.92

7.2 Appendices B. Photos and descriptions of selected sample soil profiles examined during the field survey in the Gergera watershed in September to December 2012.

Classification: Regosols

Location: X 0579762 Y 1520135 Z 2195

Position: backslope

Parent material: slates

Drainage: well drained

Surface stoniness: 90%

Land use: exclosure

Horizon symbol	Depth	pH	EC	Clay	Silt	Sand	Profile description
A1	0-15	7.79	73.7	22.5	52.5	25	Gray (10YR 6/1, dry); silt loam; weak fine crump structure; many fresh gravels and stones; common fine roots; diffused smooth boundary to
A2	15-34			14	28	58	Gray (10YR 6/1, dry); sandy loam; weak fine crump structure; many fresh gravels and stones; commonly fine roots; abrupt smooth boundary to C-horizon

Classification: Haplic Cambisol
Location: X= 0581866 Y= 1534324 Z 2639
Position: valley bottom
Parent material: Granite
Drainage: slightly drained
Surface stoniness: 5%
Land use: rangeland

Horizon symbol	depth	Clay	Silt	Sand	Profile description
A	0-10	23.5	49.5	27	Light yellowish brown (10YR 6/4, dry); Silt loam; strong fine sub angular blocky structure; few angular, fresh gravels; many course roots; diffused smooth boundary to
B1	10-30	13	29	58	Light yellowish brown (10YR 6/4, dry); Sandy loam; weak fine granular structure; few angular, fresh gravels; many moderate roots; clear smooth boundary to
B2	30-58	14.5	42.5	43	Pale brown (10YR 6/3, dry); loam; moderate medium sub angular blocky structure clear smooth boundary to
	58-100+	19.5	51.5	29	Pale brown (10YR 6/3, dry); Silt loam; strong course sub angular blocky structure; few angular, fresh gravels; few fine roots.