Soil Quality and Carbon Footprint of Different Land Uses by Smallholder Farmers in Ethiopia
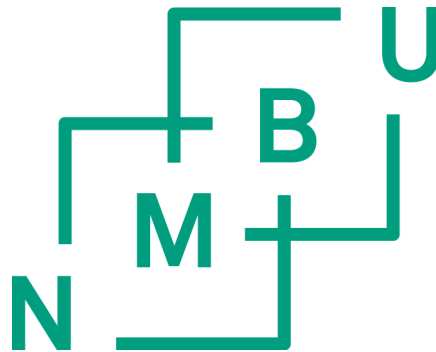
Jordkvalitet -og karbon fotspor av forskjellige bruksområder på småbruk i Etiopia

Philosophiae Doctor (PhD) Thesis

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Abstract

In Ethiopia, deforestation of natural forests and extensive use of agricultural lands have resulted in soil degradation. Despite recent massive restoration measures implemented on degraded landscapes, nowhere in the country the problem is more manifest than in Tigray, Northern Ethiopia. Most soils in this part of the country are already exhausted by several decades of over exploitation and mismanagement. There are different types of land uses in the region but quantitative information is lacking about the impacts of these land uses on soil organic carbon (SOC) and total nitrogen (TN) storage capacities and on soil quality. Therefore, this study assessed effects of different land uses on soil organic carbon (SOC) and total nitrogen (TN) stocks, on associations of SOC and TN with soil aggregates and primary particles, and on soil quality. Data for papers I, II, III and V have been collected from the following five land uses within Mandae watershed in eastern Tigray: (1) tree-less rainfed cultivation (RF) (2) *Faidherbia albida* based agroforestry (AF), (3) open pasture (OP), (4) irrigation based Guava fruit production (IR) and (5) *Faidherbia albida* based silvopasture (SP). The objectives of this study include: (i) measuring SOC and TN stocks and concentrations in soils under the five land uses (AF, RF, OP, IR and SP) and four depths (0-5, 5-10, 10-20 and 20-30 cm) (Paper I), and soils under four land uses (AF, RF, OP and SP) and three depths (0-15, 15-30 and 30-50 cm) (Paper V), (ii) determining magnitudes of SOC and TN associated with soil aggregates and primary particles under the five land uses and two depths (0-10 and 10-20 cm) (Paper II), and (iii) compare the effects of only three agricultural land uses (AF, RF and IR) in 0-15 cm depth on selected physical, chemical and biological soil quality indicators, and on an overall integrated soil quality index (SQI) of the soils under these land uses (Paper III). In addition, C-use efficiency, and C-sustainability index of the smallholder crop-livestock mixed production systems in the whole country, Ethiopia, was assessed using C footprint analysis on data obtained from the abstracts of the central statistics agency (CSA) of Ethiopia and FAO databases (Paper IV).

Soil OC and TN concentrations differed significantly among different land uses and across depths. Both SOC and TN were higher in OP and SP than in other land uses. The highest SOC concentration in 0-5 cm was measured in OP (25.4 g kg^{-1}) followed by that in SP (16.0 g kg^{-1}), and the lowest was in RF (2.29 g kg^{-1}). In 5- 10 and 10-20 cm depths, SOC concentration followed the same trend except that the amount of SOC in OP and SP land use systems decreased by about 50% compared with that in the top 0-5 cm depth. Total N concentration followed similar trends. Further, SOC and TN concentrations were highly correlated among land uses and depths. Total stocks in 0-30 cm layer were 25.8, 16.1, 52.6, 24.4 and 39.1 Mg ha^{-1} for SOC, and 2.7, 1.6, 4.9, 1.9 and 3.5 Mg ha^{-1} for TN in AF, RF, OP, IR and SP land uses, respectively. With RF as the baseline and taking the duration of 50 years since land use conversion, the average rate of accumulation was 0.73, 0.46, and 0.19 Mg C ha^{-1}yr^{-1} and 0.065, 0.038, and 0.022 Mg N ha^{-1}yr^{-1} for OP, SP and AF, respectively. Soils under IR also accumulated 0.56 Mg C ha^{-1} yr^{-1} and 0.019 Mg TN ha^{-1}yr^{-1} in the 0-30 cm layer and in comparison with the RF land use system on an average of 15 years. Similar trends were also observed for both SOC and TN stocks in 0-50 cm depth soils under OP, SP and AF land uses in comparison with RF.

Open pasture had the highest WSA >2 mm (88.7 %) and SOC associated with macroaggregates (20.0 g kg^{-1}) which were significantly higher (P < 0.0001; P < 0.01 for WSA and SOC, respectively) than that in other land uses in 0-10 cm depth. SOC associated with both macro- and microaggregates decreased with depth. Macroaggregates contained higher SOC than microaggregates in both layers under all land uses. AF had the highest SOC associated with microaggregates (2.6 g kg^{-1}) followed by that in SP (2.3 g kg^{-1}), indicating its potential to stabilize SOC more than other land uses. TN associated with macroaggregates followed a trend similar to that of SOC. Similarly, OP had significantly higher SOC (P <0.001) and TN (P <0.001) associated with sand particles than RF, AF and IR. Sand-associated SOC and TN were the highest in uncultivated systems. Moreover, the higher SOC associated with clay particles in soils under OP, SP and AF showed that grass and tree based systems are rich in stable SOC as clay-associated SOC has higher residence time than that associated with sand or silt fractions.

Among the three agricultural land uses, AF had significantly higher values (P <0.05) than RF for all soil functions except for soil's resistance against degradation (RD). For the overall SQI, the values for the three land uses were in the order: 0.58 (AF) >0.51 (IR) >0.47 (RF). Thus, AF scored significantly higher SQI (P <0.01) than that of RF. Major driving soil properties for the integrated SQI were soil organic carbon (26.4 %), water stable aggregation (20.0 %), total porosity (16.0 %), total nitrogen (11.2 %), microbial biomass carbon (6.4 %) and cation exchange capacity (6.4 %). These six parameters together contributed more than 80 % of the overall SQI.

Carbon-based inputs increased 2-fold from the lowest (0.32 Tg Ceq y^{-1}) in 1994 to the highest (0.62 Tg Ceq y^{-1}) in 2010. Similarly, total C-output increased linearly from the lowest (5 Tg Ceq y^{-1}) in 1994 to the highest (17 Tg Ceq y^{-1}) in 2011. Further, the average rate of increase in C-output from 1994 to 1999 was marginal at 0.3 Tg Ceq y^{-1}, but the 11 years average rate of increase from 2000 to 2011 was relatively higher at 0.8 Tg Ceq y^{-1}. The relationship between annual total C-based input and output was strong (R^2 =0.86; P <0.001). The CSI of the smallholder

agricultural production systems in Ethiopia was comparable with other more intensive systems in other regions of the world with the 18-year average value of ~22.

In conclusion, results from the case study in Tigray showed significant decline in SOC and TN contents and their association with aggregates and primary particles by land use change from grazing lands and silvopastures to agricultural lands. Agroforestry and irrigation land uses also showed improvements in many soil quality indicators than that of the control, RF. On the other hand, the study on C footprint analysis for the whole country, Ethiopia, showed a recent nationwide significant expansion in area of cultivated land encroaching to the remaining grazing lands and forest areas and this trend raises questions about the sustainability of the process. Therefore, improvement of crop yields via intensification on land already under cultivation and conservation of the remaining grazing lands and forests should be prominent among a portfolio of agricultural development strategies both at regional and national levels.

Keywords: Land use; C-sequestration; Soil Organic Carbon or Total Nitrogen; Soil quality; C-footprint; Sustainability; Ethiopia

Sammendrag

I Etiopia har avskoging av naturskog og overdreven bruk av landbruksjord resultert i jorddegradering. Til tross for større restaureringstiltak rettet mot degradert jord, er problemet mest omfattende i Tigray, Nord Etiopia. Jorda i regionen er utarmet etter overdreven utnyttelse og vanskjøtselen. Det finnes flere bruksområder i regionen, men kvantitativ informasjon relatert til virkningen av forskjellige bruksområder på karbon- og nitrogenlagring, samt andre jordkvalitetsparametre er mangelfull. Denne studien har som mål å avdekke effekten av forskjellige bruksområder på karbon (SOC) og nitrogen (TN) mengde i jord, samt effekten av karbon og nitrogen på jordaggregater og primære jordpartikler (sand, silt og leire) samt jordkvalitet.

Data for de første tre studiene ble samlet fra følgende fem bruksområder i Mandaes nedbørsfelt i øst Tigray: (1) Regnbasert dyrking uten trær (RF), (2) *Faidherbia albida* basert agroskogbruk (AF), (3) Åpent beite (OP), (4) Vanningsbasert Guava-produksjon(IR), og (5) *Faidherbia albida*-basert beite. Formålene med studiet var: (i) Måling av konsentrasjoner og mengder av karbon og nitrogen i jord under fem bruksområder (AF, RF, OP, IR and SP) og fire jorddybder (0-5, 5-10, 10-20 and 20-30 cm), (ii) Bestemme mengde av SOC og TN bundet til jordaggregater og primære jordpartikler i de samme bruksområdene og i to jorddybder ((0-10 and 10-20 cm), og (iii) Sammenligne virkningen av tre bruksområder (AF, RF og IR) i 0-15 cm jorddybde på utvalgte fysiske, kjemiske –og biologiske jordkvalitetsindikatorer og samlet integrert jordkvalitetsindeks (SQI) under disse bruksområdene. I tillegg ble karbonutnyttelsesgraden og bærekraftindeksen for karbon undersøkt på små bruk med husdyrproduksjon ved bruk av karbonsporingsanalyse på data samlet fra Central Statistics Agency (CSA) i Etiopia and FAO databaser (Formål IV)

Konsentrasjoner av SOC og TN var signifikant forskjellig mellom bruksområder og jorddyp. Både SOC og TN var høyere i OP og SP enn andre bruksområder. Den høyeste konsentrasjonen av SOC i 0-5 cm ble målt i OP (25.4 g kg^{-1}) etterfulgt av SP (16.0 g kg^{-1}), men laveste i RF (2.29 g kg^{-1}). SOC konsentrasjonen i dybdene 5-10 og 10-20 cm fulgte samme trend, bortsett fra at mengde SOC i OP og SP bruksområdene ble redusert med 50% sammenlignet med 0-5 cm dyp. Konsentrasjonene av SOC og TN var sterkt korrelert med bruksområder og jorddybder. Totalmengdene av SOC i 0-30 cm var 25.8, 16.1, 52.6, 24.4 og 39.1 Mg ha^{-1} sammenlignet med TN som var 2.7, 1.6, 4.9, 1.9 and 3.5 Mg ha^{-1} for henholdsvis AF, RF, OP, IR, og SP bruksområder. Med RF som referanse med 50 års løpetid etter bruksendringer var akkumuleringsratene 0.73, 0.46, og 0.19 Mg C ha^{-1}år^{-1} for SOC. For TN var ratene henholdsvis 0.065, 0.038, and 0.022 Mg N ha^{-1}år^{-1} i OP, SP, og AF. Jord under IR akkumulerte også 0.56 Mg C ha^{-1} år^{-1} og 0.019 Mg TN ha^{-1}år^{-1} i 0-30 cm jorddybde i forhold til RF i en 15 års periode.

Åpne beiter hadde høyest WSA> 2mm (88.7) og SOC bundet til makroaggregater (20.0 g kg^{-1}) og signifikant høyere (P < 0.0001; P = 0.0017 henholdsvis for WSA and SOC) enn i andre bruksområdene i 0-10 cm jorddybde. SOC bundet til både makro -og mikroaggregater var redusert med jorddybde. Makroaggregater inneholdt høyere SOC enn mikroaggregater i begge dyp uansett bruksområde. AF hadde høyeste mengde av SOC bundet til mikroaggregater (2.6 g kg^{-1}) etterfulgt av SP (2.3 g kg^{-1}). Dette viser at AF har høyere potensiale for stabilisering av SOC enn andre bruksområder. TN i makroaggregater fulgte samme trend som SOC. På samme måte hadde OP signifikant høyere SOC (P <0.001) og TN (P <0.001) bundet til sandpartikler enn RF, AF, og IR. SOC og TN bundet til sand var høyest i udyrkede områder. Videre viser høyere mengde av SOC bundet til leirpartikler i OP, SP, og AF at gras og områder dekket av skog er rik på stabil SOC. Dette skyldes at SOC bundet til leirpartikler har høyere oppholdstid enn SOC bundet til silt- eller sandpartikler.

Ved å sammenligne de tre landbruksområder, hadde AF signifikant høyere verdier for alle jordfunksjoner (P<0.05) enn RF tatt med i undersøkelsen bortsett fra jordas motstand mot degradering (RD). Sett under ett avtok SQI verdier for tre bruksområder i følgende rekkefølge: 0.58 (AF) >0.51 (IR) >0.47 (RF). Dermed hadde AF signifikant høyere SQI (P<0.01) enn i RF. De styrende jordegenskapene som virket inn på integrert SQI var SOC (26.4 %), WSA (20.0 %), total porøsitet (16.0 %), total nitrogen (11.2 %), mikrobial biomassekarbon (6.4 %), og CEC (6.4 %). Disse 6 parameterne kontrollerte samlet mer enn 80 % av den totale SQI.

Karbontilførselen fordoblet innholdet fra det laveste (0.32 Tg Ceq år^{-1}) i 1994 til det høyeste (0.62 Tg Ceq år^{-1}) i 2010. På samme måte, økte total ytelse lineært fra det laveste (5 Tg Ceq år^{-1}) i 1994 til det høyeste 17 Tg Ceq år^{-1}) i 2011. Videre var den gjennomsnittlige rate av C- ytelse fra 1994 til 1999 marginalt på 0.3 Tg Ceq år^{-1}. Over 11 år var den gjennomsnittlige raten relativt høyere. Den var 0.8 Tg Ceq år^{-1}. Korrelasjonen mellom årlig tilførsel og ytelse av C var sterk (R^2 =0.86; P <0.001). CSI for landbruksproduksjon på små bruk i Etiopia var i samsvar med andre regioner av verden med en gjennomsnittsverdi på 22 over18 år.

Som en konklusjon viser de første tre studiene at bruksendringer fra både beite og husdyrbeite påvirket SOC og TN innholdet i jord og bindingen til aggregater og primære jordpartikler, samt andre jordkvalitetsindekser. Sammenlignet med kontrollområdet (RF) ble alle jordegenskapene forbedret både i agroskogbruk og systemer med vanning. På den andre side viser studiet om C fotsporanalysene, en signifikant utvidelse av areal under dyrket mark som legger beslag på gjenværende beite- og skogsområder. Denne trenden setter spørsmåltegn ved bærekraftigheten av slike systemendringer. De økte avlingene som følge av intensivt jordbruk på eksisterende arealer må sees i lys av verdien det ligger i å bevare gjenværende beite og skogsarealer når framtidens landbruksutviklingsstrategier skal konkretiseres regionalt- og nasjonalt.

Nøkkelord: Bruksområder; karbonlagring; jord organisk karbon (SOC); totalt nitrogen(TN); jordkvalitet; C-fotspor; bærekraftighet; Etiopia

Lists of symbols and acronyms

ADLI   Agriculture Development-Led Industrialization
AF   Agroforestry
ANOVA   Analysis of variance
ATA   Agricultural Transformation Agency
AVK   Available potassium
AVP   Available phosphorous
AWC   Available water capacity
BD   Bulk density
C   Carbon
CEC   Cation exchange capacity
Ceq   Carbon equivalent
CH_4 Methane
Ci   Carbon input
cm   centimeter
C-MASC   Carbon Sequestration and Management Center
Co   Carbon output
CO_2 Carbon dioxide
CS   Carbon sequestration
CSA   Central Statistical Agency
CSI   Carbon sustainability index
CV   Coefficient of variation
EC   Electrical conductivity
FAO   Food and Agriculture Organization
FDRE   Federal Democratic Republic of Ethiopia
Fig.   Figure
GDP   Gross Domestic Product
GHG   Greenhouse gas
Gg   Gigagram
GMD   Geometric mean diameter
GTP   Growth and Transformation Plan
ha   hectare
HCl   Hydrochloric acid
HSD   Tukey's studentized test
km   kilometer
IR   Irrigation
LU   Land use
m.a.s.l.   meters above sea level
MBC   Microbial biomass carbon
MDS   Minimum dataset
Mg   Mega gram
mg   milligram
mm   millimeter
MWD   Mean weight diameter
N_2O Nitrous oxide
NS   Not significant
OC   Organic carbon
OP   Open pasture
P   Precipitation
Pg   Pentagram
PNS   Plant nutrient supply
R^2 Correlation coefficient
RD   Resistance to degradation
RF   Rainfed
RMPS   Recommended management practices
SAS   Statistical Analysis Software

SCS	Soil carbon sequestration
SIC	Soil inorganic carbon
SOC	Soil organic carbon
SOM	Soil organic matter
SP	Silvopasture
SQI	Soil quality index
Std.error	Standard error
T	thickness of soil layer
Tg	Teragram
TN	Total nitrogen
TP	Total porosity
UDSA	United State Department of Agriculture
USA	United States of America
WSA	Water stable aggregation
WE	Water entry
WMA	Water movement and availability
Y	year
%	percentage
µm	micrometer
ρb	bulk density
ρp	particle density

1. Introduction

Ethiopia with an area of 1.13 million sq.km (Kidanu, 2004, Girmay, 2009) and an estimated population of about 90 million in mid-2013 (Population Reference Bureau, 2013) is the second most populous country in Africa. The climate is tropical monsoon with wide topographic-induced variations (Girma, 2001). The elevation varies from 125 m below sea level to 4620 m above sea level; the terrain is of high plateaus with a central mountain range divided by the Great Rift Valley. The country's major natural renewable resources consist of land, water and natural vegetation that comprise enormous biodiversity (Mulugeta, 2004). Ethiopian economy is an agrarian one, the most important objective of which is achieving food security (ATA, 2013). Agriculture accounts for 46 percent of gross domestic product (GDP) and 85 percent of total employment. Smallholder farmers with an average holding of less than one hectare account for over 90 % of the agricultural area under crop production (CSA, 2011). Although GDP growth is high and the economy has been growing steadily for the past ten years, per capita income remains among the lowest in the world (FDRE, 2011). The agriculture system in Ethiopia is predominantly rain-fed, where the performance of the sector is highly dependent on the timely onset, duration, amount and distribution of rainfall that makes the sector highly vulnerable to drought and other natural calamities. Due to increasing human and livestock population pressure on arable land and forest resources, large areas of the country, particularly in the northern and central highlands, have been exposed to loss of soil quality, degradation and ecological imbalances (Mulugeta, 2004). Furthermore, Ethiopia is experiencing the effects of climate change such as an increase in average temperature and a change in rainfall patterns (FDRE, 2011).

Considering the available human and land resources, and the contribution of agriculture to the national economy, employment and the problem of food insecurity, Ethiopia has put agricultural development as a policy priority since the early 1990s. The government of Ethiopia has set a long-term economic policy of 'Agriculture Development-Led Industrialization (ADLI)' with the primary objective of achieving rapid and sustainable economic growth by improving the productivity of the agriculture sector. Another recent policy document of the government, the national "Growth and Transformation Plan (GTP)," also reaffirmed the importance of increasing agricultural productivity for strengthening the industrial base and fostering export growth of the country (FDRE, 2011). Further, all policy documents of the country envisage agriculture as a driver of the economic development to industrialization and modernization. Hence, achieving agriculture-led sustainable economic development requires sustainable land management, including soil, water and biodiversity conservation, on all lands.

1.1.   Land use change and soil degradation in Ethiopia

Alterations in the land surface caused by humans to produce food and fodder through agricultural activities are known for centuries. Severe and widespread soil degradation can be observed in the north and central highlands of Ethiopia, where agriculture has a long history, and farmers have practiced crop cultivation for many centuries and even millennia. Now, the rural landscape in those areas has suffered from a high degree of soil degradation due to agricultural practices that do not consider soil care measures. Deforestation and subsequent conversion into permanent cultivation are the primary cause for a number of impacts in Ethiopia.

In most tropical regions, the common agricultural land use system is a smallholder farming system with agricultural production in small parcels for subsistence purposes with no or little external inputs. Further, the small farm sizes are insufficient to provide for ever-increasing human populations (Shiferaw and Holden, 2000). In response to the increasing demands for food production, agricultural lands are expanding at the expense of natural vegetation and grasslands (Lambin et al., 2000; Hartemink et al., 2008). These changes in land use (LU) systems have great impacts, among others, on soil degradation and sustainability of agricultural productions (Lambin et al., 2003). In Ethiopia, sustainable development of the agricultural sector is challenged by increasing population growth and gradual decline of natural resources. To satisfy the increasing demand for food, more grazing and forestlands have been brought into arable lands. As a result, forest cover of the country estimated to be around 40 % at the beginning of the 20th century has declined to less than 5 % and with estimated annual rate of deforestation of 0.15 to 0.2 million hectares (Girmay, 2009).

Deforestation and conversion to cropland and other managed ecosystems are widely pronounced in Ethiopia. The decrease in vegetation cover and disturbance of the natural ecosystem have caused widespread soil degradation, with an attendant decline in concentrations of soil organic matter (SOM) and available nitrogen (N) pools (Mulugeta et al., 2005; Girmay et al., 2008; Gelaw et al., 2013, 2014). Although the impact of land use and management on soil characteristics vary among soils and eco-regions, it is generally recognized that such changes have exacerbated the problem of soil erosion and degradation.

Land use change and subsequent soil degradation due to soil erosion have posed a tremendous challenge to increasing agricultural productivity and economic growth in Ethiopia (Amare et al., 2005; Hengsdijk et al., 2005; Girmay et al., 2008). Soil fertility depletion on smallholder farms is one of the most important consequences of soil

degradation that causes declines in crop production. As a consequence of continuously low crop yields, the economic situations of several farming communities in the country have been adversely affected, leading to a cyclic poverty. Further, the process of soil fertility depletion is aggravated by removal of crop residue and animal dung for domestic use, either as household fuel or as animal feed (Amare et al., 2005; Girmay et al., 2008). Few studies conducted in different parts of the country showed negative nutrient balances. For example, field-level studies by Eyasu (2002) and Amare et al. (2006) reported -102-and -72-kg ha^{-1} N budgets in soils of Southern and Central highlands of Ethiopia, respectively. Similarly, a study by Zenebe (2007) indicated that the use of manure as fuel instead of as organic fertilizer estimated to reduce Ethiopia's agricultural GDP by 7 %.

1.1.1. Land use change and its effects on soil organic carbon stocks

Land use change can cause a change in land cover and associated carbon stocks (Bolin and Sukumar, 2000). The change from one ecosystem to another could occur naturally or be the result of human activities such as food and timber production. Each soil has a carbon carrying capacity depending on the nature of vegetation, and climatic conditions (Guo and Gifford, 2002).

Expansion of agriculture caused a significant conversion from natural vegetation cover to cropland, with significant losses in SOM mainly during the first few years of cultivation (Mulugeta et al., 2005). A study by Solomon et al. (2002) showed that a conversion of a humid tropical forest to maize (*Zea mays*) cultivation in Southern Ethiopia resulted in 55–63 % reduction in SOC concentration. Accurate information lacks on the time of deforestation and conversion to agriculture especially in Northern and Central highlands of Ethiopia because crop cultivation in these parts of the country has been practiced for many centuries. However, given the longer years of cultivation and poor practices of recycling crop residues, it is likely that the magnitude of depletion in SOM stock is higher than that observed in other parts of the country. The relative loss of SOC derived from the natural vegetation in a continuous cultivated field also varies with particle size distribution, higher losses for sand and lower losses for clay soils. Furthermore, climate variability influences soil C and N stocks through its effect on vegetation type and consequently on the composition, quantity, and degree of SOM decomposition. For instance, Spacini et al. (2006) in their study conducted in five separate areas located in different geographical regions of Ethiopia found the highest SOC concentration under rainforest vegetation in Jima, Southwest Ethiopia, and the lowest under a Guinea savannah vegetation in Sirinka, Northeast Ethiopia.

Land use change from forest to grazing land showed a decline in SOM. A study by Woldeamlak and Stroonsnijder, (2003) at Chemoga watershed, Blue Nile basin, central Ethiopian highlands and another study by Abule et al. (2005) in the Middle Awash Valley, Eastern Ethiopia, reported decline in SOM concentrations after conversion of forest to open grazing. The result by Woldeamlak and Stroonsnijder, (2003) showed that a conversion of forest to open grazing land reduced SOM by 14.8–47.5%, but the SOM concentration in the open grazing land was slightly higher than that in cropland. On the other hand, area enclosures, degraded lands protected from animal and human interference, which are famous in Tigray, northern Ethiopia, have shown positive effects in improving SOM either in the form of sediment trapped from run-on or through addition of biomass C to the soil (Mekuria and Veldkamp, 2005; Descheemaeker et al., 2006; Mekuria et al., 2009). However, magnitudes of SOM trapped or produced showed differences across different ages of enclosures, diversity and richness of species and other site-specific differences in soil-type and microclimate (Girmay et al., 2008).

Moreover, change in SOC stocks after land use change differs across depths. The SOM in topsoil is more labile and prone to changes upon deforestation than that in subsoil (Veldkamp et al., 2003). Gelaw et al. (2014) in Northern Ethiopia, and Solomon et al. (2002) and Mulugeta et al. (2005) both in Southern Ethiopia observed higher changes in SOM in the topsoil layer than the subsoil layer after land use conversion. These trends may be attributed to the low-decomposition rate and accumulation of translocated SOM from the topsoil through leaching (Girmay et al., 2008).

1.2. Building climate resilient agriculture through soil carbon sequestration

World soils constitute the largest carbon (C) reservoir of the terrestrial biosphere (Batjes, 1996). The C pool in the soil comprises two distinct components: (i) SOC estimated at 1550 Pg, and (ii) soil inorganic carbon (SIC) pool estimated at 950 Pg, both to 1-m depth. The total soil C pool of 2500 Pg is 3.3 times the atmospheric pool and 4.0 times the biotic pool (Batjes, 1996). However, soils of the managed ecosystems have lost much of the original SOC pool (Lal, 2000). Conversion of natural to managed ecosystems depletes SOC pool for two major reasons: (1) carbon input into agricultural ecosystems is lower than that in natural ecosystems, and (2) the sum of losses from agricultural ecosystems due to erosion, mineralization and leaching is higher than that in natural ecosystems. Further, the magnitude of SOC depletion is high in soils prone to erosion and those managed by low-input or extractive farming practices. Loss of the SOC pool is also high in soils of coarse texture and those with high initial pool. Most agricultural soils have lost 20 to 40 Mg C ha^{-1} due to historic land use and management (Lal, 2000). The maximum soil C-sink

capacity, amount of C that can be stored in it, approximately equals the historic C loss. In other words, most agricultural soils now contain lower SOC pool than their capacity because of the historic loss. Much of the historic C loss (about 66-90 Pg C) from the soil can be restored through C sequestration (CS) in 25-50 years (Lal, 2004) with appropriate land management. Indeed, soil has possessed a promising potential for C sequestration and storage. Further, Soil OM contributes to plant available nutrients; buffers environmental stress, improve water-holding capacity, and reduce erosion (Lal, 2000; Wang et al., 2010). Thus, apart from removing CO_2 from the atmosphere, restoration of SOC through CS in agroecosystems can benefit food production and improve agricultural sustainability. The IPCC fourth-assessment report identified agriculture as among the economic sectors having the greatest near-term climate change mitigation potential, largely through CS (van Wesemael et al., 2010).

Adoption of recommended management practices (RMPs) such as no-till enhances SOC sequestration and improves soil structure and other qualities (Lal and Kimble, 1997). Soil OM increases plant available water capacity (AWC), water infiltration rate, and decrease surface runoff. In return, improvements in these soil hydrological properties are important to reducing susceptibility of agroecosystems to pedological/agronomic droughts (Lal, 2012). For the rain-fed agriculture in Ethiopia, water availability is the primary factor controlling crop productivity, so any soil and crop management practice that can enhance soil water storage and availability are likely to increase yield and overall productivity. Adoption of no-till and crop residue retention have shown to retain more water in semi-arid soils, not only because of reduced evaporation, but also results in the development of new and more extensive pore systems that enhance soil water holding capacity (Bescansa et al., 2006). On the other hand, tillage disrupts soil aggregates; compact the subsoil and disturb plant and animal communities resulting in a decrease in soil organic matter, and microbial and faunal activities (Plante and McGill, 2002; Bronick and Lal, 2005).

To sum up, soil mismanagement can cause depletion of SOC stock with an attendant emission of CO_2 into the atmosphere (Reicosky et al., 1997; Lal 2004a; Chen et al., 2009). Whereas, an appropriate land use and soil management with RMPs can increase SOC stock thereby reducing net emission of CO_2 to the atmosphere (Paustian et al., 2000; Sampson and Scholes, 2000); increase sustainability of farming systems and contribute to reducing farmers' vulnerability to climate variability (Verchot et al., 2007).

1.2.1.   Soil quality indices as indicators of agricultural sustainability

The concept of soil quality has grown out of concern about the sustainability of agriculture (Parr et al., 1992; Warkentin, 1995; Wander and Drinkwater, 2000). Many definitions of soil quality emphasized the concept of a soil's fitness to perform functions (Larson and Pierce, 1991; Warkentin, 1995; Karlen et al., 1997). The widely accepted definition of soil quality is "the ability of soil to function within ecosystem boundaries to support healthy plants and animals, maintain or enhance air and water quality, and support human health and habitation" (Karlen et al., 1997). These functions are impacted by multiple soil attributes. Accordingly, soil scientists identified a minimum data set (MDS) of soil parameters that could be used to quantify soil quality (Larson and Pierce, 1991; Arshad and Coen, 1992; Doran and Parkin, 1994). The selection of MDS parameters has been based upon a wealth of soil management research that relates soil attributes to soil function and ideally relates management practices to soil attributes. Soil quality functions proposed by Larson and Pierce (1991, 1994) and Karlen and Stott (1994) are examples of theoretical frameworks that combine physical, chemical and biological measures to assess soil conditions. Since many of the issues of sustainability are related to soil quality, assessment of soil quality and the direction of change with time is a primary indicator of whether agriculture is sustainable (Karlen et al., 1997; Masto et al., 2007).

Sustainability of agricultural systems should be an important issue in many developing countries including Ethiopia. In Ethiopia, increasing demographic pressure on finite land resource bases is posing a major problem on the food security and enhanced quality of life of both the current and future generations. Arable soils in the country are amongst the oldest in Africa and are highly degraded. The long-term and widespread use of extractive farming practices are among important causes of low and declining agronomic production in the country. The magnitude of nutrient mining because of crop harvests is huge. For example, farmers remove all crop biomass from the fields and use less farmyard manure (Amare et al., 2005; Hengsdijk et al., 2005; Zenebe, 2007; Girmay et al., 2008). Consequently, field-level studies in different parts of the country showed negative nutrient balances (Eyasu, 2002; Amare et al., 2006).

It is important to recognize that agroecosystems are sustainable in the long term only if the outputs of all components harvested are balanced by inputs into the system (Lal, 2009), and the negative nutrient and carbon budgets are changed to positive balances, in order to restore the soils and thereby maximize their productivity and ecosystem services. As soil quality is a combination of soil physical, chemical and biological properties that are able to change readily in response to variations in soil conditions (Brejda et al., 2000), it may be affected by land use type and agricultural management practices. Land use type and agricultural management practices cause alterations in soil

properties, which in turn result in a change in land productivity for better or worse (Islam and Weil, 2000; Sanchez-Maranon et al., 2002).

Integrated soil quality indices based on a combination of soil properties provide a better indication of soil quality than individual parameters. Karlen and Stott (1994) developed a soil quality index (SQI) based on four-soil functions, namely the ability of the soil to: (1) accommodate water entry, (2) facilitate water movement and absorption, (3) resist surface degradation, and (4) sustain plant growth. Each soil function is explained by a set of indicators. Several authors among them Glover et al. (2000), Masto et al. (2007) and Fernandes et al. (2011) used a similar framework. The SQI helps to assess the soil quality of a given site or ecosystem and enables comparisons between the conditions at the plot, field or watershed levels under different land use and management practices. Soil quality research in Ethiopia and in other SSA countries is almost non-existent. Therefore, it should be a priority for a comprehensive understanding of the effects of different land uses and soil management strategies to develop environmental-friendly management plans in the country and in the wider region.

1.3. Carbon footprint of agricultural production systems

Ethiopia is experiencing the effects of climate change such as an increase in average temperature and a change in rainfall patterns (FDRE, 2011). The agricultural sector is the most sensitive sector to climate change. Because it is the most important sector in the country, whatever happens to the agricultural sector can significantly affect the entire economy. Although Ethiopia's GHG emissions are attributable to its agricultural sector (FDRE, 2011), its contribution to the global increase in GHG emissions has been practically negligible. In contrast, currently degraded soils under different land uses in the country have large SCS potentials if RMPs are adopted (Girmay et al., 2008).

Managing soils create positive soil and ecosystem C budgets and offset emissions of CO_2 from fossil fuel combustion, in the context of the Kyoto protocol (Schlesinger, 2000). Further, the rate of accumulation of soil organic carbon is often higher on fertilized and irrigated fields. However, adoption of RMPS especially inorganic fertilizers and irrigation carries carbon "costs" in the form of CO_2 emissions during production, transportation and application of fertilizers, and in terms of the energy used to pump irrigation water (Schlesinger, 2000). Thus, a quantitative approach of assessing the agronomic productivity and ecological contributions or costs of management practices is relevant to smallholder farmers of the tropics (Lal, 2010). Evaluating the sustainability of farming systems through application of carbon footprint analyses, measuring impacts of agricultural activities on the environment in terms of the amounts of GHGs produced in CO_2 equivalent, (Dubey and Lal, 2009) is an important tool to measure eco-efficiency of a range of agroecosystems (Lal, 2010). Eco-efficient technologies are those, which minimize the adverse environmental impact, and maximize agronomic production (Lal, 2010).

Adoption of improved systems of soil, crop and water management technologies that enhance eco-efficiency, increase the SOC pool, improve soil quality, conserve water in the root zone, and create positive C and nutrient budgets (Lal, 2010). In view of the increasing population and scarcity of natural resources in Ethiopia and other developing countries in SSA and elsewhere, there are options for meeting the growing demand for food, fuel, fodder, and other agricultural products (Lal, 2010). The following are among the available options: (1) replacing extractive farming practices with scientifically proven technologies, (2) managing soil and other natural resources to enhance their resilience to climate variability (Walker and Salt, 2006), and (3) increasing production from agroecosystems on the basis of per unit area and input of external inputs (fertilizers, irrigation, energy). For instance, reduction in emissions of GHGs from application of mineral or organic fertilizers and improvement in their use efficiencies in agricultural systems can be achieved by better matching fertilizer supply to crop demand (van der Velde et al., 2013). More closely integrating animal waste and crop residue management with crop production systems (Lal, 2007) can also be used to achieve these objectives.

1.4. Rationale and objectives of the study

Ethiopia is one of the most environmentally troubled countries in SSA (Hagos et al., 1999). Tigray, the northernmost region of Ethiopia, experienced the worst land degradation in the Ethiopian highlands. Despite recent efforts to restore degraded landscapes, the problem is still prevalent in the region. Now, there are different types of land uses in the region but little quantitative information is available on the effects of these land uses on biophysical resources such as SOM, which is important for agricultural production and other ecosystem services. Most of the research conducted on land degradation in the region has focused on soil erosion by water. Similarly, the policy responses to land degradation have focused on promoting adoption of few physical structures such as terraces and bunds. However, farmers in the region identify moisture stress and declining soil fertility, both directly related to loss in SOM, as the most limiting factors (Hagos et al., 1999; Hengsdijk et al., 2005; Girmay et al., 2008). Therefore, it is important to investigate soil organic matter stock and its stability, and the status of soil quality in relation to land use type in the region.

Land use is an important factor controlling soil organic matter content since it affects amount and quality of litter input, litter decomposition rates and processes of organic matter stabilization in soils (Römkens et al., 1999). As soil is the largest terrestrial pool of organic carbon, small changes in its stock could result in significant impacts on the atmospheric C concentration. Precise estimates of SOC storage are thus important in studies to detect the potential for C sequestration or emission induced by land use change (Wiesmeier et al., 2012). Many studies have reported decline of SOC stocks because of deterioration of natural ecosystems or the conversion of natural or extensively used areas into intensively used croplands (Lal, 1996; Ashagrie et al., 2007; Berhongaray et al., 2013). Others have examined the amount of SOC sequestered through planting trees, implementing conservation tillage, re-establishing grasslands, or controlling desertification (Lal, 2001; DeGryze et al., 2004; Cantarello et al., 2011; Li et al., 2013; Zhang et al., 2013). However, a high degree of uncertainty is associated with the estimates of C sequestration and emission rates for different land use types because variations in C stocks depend on many regional factors including climate, topography, soil type, and other ecosystem properties (Bolliger et al., 2008; Cantarello et al., 2011). In addition to C, soil nitrogen (N) also controls the overall soil turnover and functioning of C and N in SOM (Batlle-Aguilar et al., 2011). Although N is essential for life, its bio-available forms are sufficiently low that they can constrain plant growth and N cycle in many ecosystems. Further, C uptake in terrestrial ecosystems depends on availability of nutrients such as N to support growth of new biomass (Thornton et al., 2009).

Soil structure and its dynamics including formation, stabilization, and destabilization of aggregates exert important controls on soil carbon dynamics (Christensen, 2001). Land use also has a significant effect on aggregate size distribution and stability (Saha et al., 2011). The presence of higher proportions of macro-aggregates in forest than in cultivated soils indicates the effect of tillage on soil aggregate turnover. Increased return of residues and reduced disturbance of soils under pasture also results in better soil aggregation and sequestration of more carbon than in intensively tilled croplands (Percival et al., 2000). Thus, the role of land use systems in stabilizing CO_2 levels and increasing carbon (C) sink potentials of soils has attracted considerable scientific attention in the recent past (Kumar and Nair, 2011; Murthy et al., 2013). However, such data is very limited in Ethiopia in general and in Tigray in particular. Thus, it is imperative to investigate SOC and TN storage capacities of land use systems in the region and to generate relevant information for land managers and policy makers to help them for their land management decisions.

In addition to its significance for production of food and fiber, soil is also a critical component of the earth's biosphere for the maintenance of local, regional and global environmental quality (Doran and Parkin. 1994). To preserve soil and its functions, it is necessary to understand conditions and processes occurring in it through determination of soil quality. Soil quality is a combination of soil physical, chemical and biological properties that are able to change easily in response to variations in soil conditions, which may be affected by land use type and agricultural management practices. Integrated soil quality indices (SQI) based on a combination of soil properties provide a better indication of soil quality than individual parameters. Soil quality index helps to assess soil quality of a given site or ecosystem and enables comparisons between soil conditions at different scales under different land use and management practices (Karlen and Stott, 1994; Glover et al., 2000; Masto et al., 2007; Fernandes et al., 2011). Soil quality research in Ethiopia is at its infant stage. Therefore, research is needed for a comprehensive understanding of the effects of different land uses and soil management strategies on overall soil quality to develop environmental-friendly management plans in the region.

Managing degraded agricultural soils by adopting RMPs creates positive ecosystem nutrient and C budgets, and hence provides dual benefits of offsetting emissions of CO_2 from fossil fuel combustion and improving soil quality. However, adoption of RMPs for agriculture involves off-farm or external inputs, which are C-based operations and products (Pimentel, 1992). Thus, it is important to evaluate the sustainability of farming systems through application of carbon footprint analyses (Lal, 2004b; Dubey and Lal, 2009). However, no attempt has been made hitherto to evaluate the C footprints of agricultural productions in the region. Therefore, assessment of agricultural C-emissions of the crop-livestock mixed-farming system in Ethiopia, and evaluating its C-use efficiency and relative sustainability determined by its C-footprints will help subsistence farmers avoid losses, reduce environmental damages and increase benefits through productivity increments and payments for ecosystem services.

Based on the background described above, the study was carried out with the following specific objectives:

- Measuring and comparing SOC and TN concentrations and stocks of soils in five-land use systems: tree-less rainfed cultivation (RF), *F. albida* based agroforestry (AF), irrigation based Guava fruit production (IR), open pasture (OP), and *F. albida* based silvopasture (SP), each at four-soil depths (0-5, 5-10, 10-20 and 20-30 cm) (Paper I).
- Determining the magnitudes of SOC and TN concentrations associated with aggregates and primary particles under five-land use systems: treeless rain-fed cultivation (RF), *F. albida* based agroforestry (AF), irrigation

based Guava fruit production (IR), open pasture (OP), and *F. albida* based silvopasture (SP), each at two-soil depths (0-10 and 10-20 cm) (Paper II).

- Comparing effects of three agricultural land use systems: tree-less rainfed cultivation (RF) *F. albida* based agroforestry (AF), irrigation based Guava fruit production (IR), on selected physical, chemical and biological soil quality indicators, and on an overall integrated SQI (Paper III).
- Assessing C-emissions, C-use efficiency, and C-sustainability index of the smallholder crop-livestock mixed production systems in Ethiopia (Paper IV).
- Measuring and comparing SOC and TN concentrations and stocks of soils in four-land use systems: tree-less rainfed cultivation (RF), *F. albida* based agroforestry (AF), open pasture (OP), and *F. albida* based silvopasture (SP), each at three soil-depths (0–15, 15-30 and 30-50 cm) (Paper V).

2.   Materials and Methods

2.1.   Study site

Mandae watershed is located in Tigray regional state, Northern Ethiopia. Geographically, it is located between 13°83'00 N to 13°85'00 N latitude and 39°50'00E to 39°53'00E longitude having an area of about 10 km^2, and elevation of 1960 to 2000 meters above sea level (m. a. s. l.). The average daily air temperature of the area ranges between 15℃ and 30℃ in winter and summer, respectively. The mean annual rainfall of the area is about 558 mm, with a large inter-annual variation. Soils in the watershed are classified as Arenosols, and association of Arenosols with Regosols according to the World Reference Base for soil resources (WRB, 2006). These soils are developed from alluvial deposits and Adigrat sandstones. Textures of these soils were dominated by sand, loamy-sand and sandy-loam fractions, and pH ranged from 6.8 to 7.9 (Rabia et al., 2013). Major land uses of the watershed include *F. albida* based agroforestry (28 ha), rainfed crop production (12 ha), open pasture (23 ha), irrigation based guava fruit production (11 ha) and *F. albida* based silvopasture (12 ha) (Figure 1).

Mandae watershed

Figure 1Map of the Study Area in Relation to Ethiopia

2.2. Soil sampling techniques

For the first and the last studies on SOC and TN concentrations and stocks, 100 (four-depths, five-land uses, and five replications) (Paper I) and 60 (three-depths, four-land uses, and five replications) (Paper V) evenly distributed soil samples were collected using a soil auger. For paper I, soil samples were taken to 30-cm depth and were separated into depth increments of 0–5, 5-10, 10-20 and 20-30 cm. Similarly, samples were taken to 50-cm depth and were separated into increments of 0–15, 15-30 and 30-50 cm depths for Paper V. For the second study (Paper II) on SOC and TN concentrations associated with aggregate sizes and primary particles, another 100-soil samples (50 aggregate and 50 composite soil samples) were obtained from the surface (0-10 cm) and subsurface (10-20 cm) layers of five sites randomly chosen at different locations for each land use system. For the third study on soil quality (Paper III), 24-soil samples (9 samples for microbial biomass C (MBC) determination and 15 for determining other basic soil parameters) were sampled from the surface (0-15 cm) layer at three and five replications for MBC and other parameters, respectively. Samples were collected at randomly chosen locations for each of the three agricultural land use system (AF, RF and IR). For all of the above four case studies, the summit position of the watershed was excluded to minimize confounding effects of slope and erosion. Individual farms of different sizes in AF, RF and IR land uses were used as replicates. In OP and SP, which were communal lands, adjacent plots to sampled fields in other land uses were used as replicates. For all composite samples, soil cores within each replicate were collected randomly from eight points within a 64 m^2 area at each sampling site and were well mixed and combined to a composite sample by depth. Thus, a minimum of 40-point samples were represented in computing the average values of each soil parameter. Additionally, soil bulk density (ρ_b) samples were taken for the same depth intervals as other soil samples at each replicate by the core method (Blake and Hartge, 1986).

2.3. Data collection for the carbon footprint analysis

Data on C-based inputs into the soil and outputs from predominant cereal crops grown by smallholder private peasant farmers were obtained from annual agricultural abstracts of the Central Statistics Agency (CSA) of Ethiopia (CSA, 1994-2011) and the FAO database (Paper IV).

2.4. Soil analyses Methods

Soil samples were air-dried, gently ground and passed through 2-mm sieve prior to chemical analyses. Further, identifiable crop residues, root material, and stones were removed during sieving. Soil samples for C and N analyses were additionally ground using a ball-mill grinder. Because soils did not show carbonates when testing with 10-% HCl, it was assumed that the total C obtained in the analyses closely estimates the SOC concentration. Concentrations of SOC and TN (% w/w) in the composite sample and those associated with aggregates and primary particles were determined at the Carbon Sequestration and Management Center (C-MASC) laboratory (The Ohio State University, USA) using auto CN analyzer (Vario Max CN Macro Elemental Analyser, Elementar Analysensysteme GmbH, Hanau, Germany) by dry combustion method (Nelson and Sommers, 1996). Size fractionation using sodium hexametaphosphate method (Cambardella and Elliott, 1992, 1993) and water stable aggregation (WSA) by the wet sieving method (Yoder, 1936) were also measured at C-MASC laboratory. Available P (Olsen) (Olsen et al., 1954) and Cation exchange capacity (CEC) by ammonium distillation method (Chapman, 1965), pH by pH meter (1:1.25, soil:water ratio), electric conductivity by EC meter (1:5, soil:water ratio) were all analyzed at the Mekelle University (MU) soil laboratory, Ethiopia. Microbial biomass carbon (MBC) was determined following the fumigation-extraction method (Brookes et al., 1985; Vance et al., 1987) at the Department of Environmental Science (IMV) soil laboratory, Norwegian University of Life Sciences (NMBU), Ås, Norway.

2.5. Computational methods
2.5.1. Soil carbon and Total nitrogen stocks

Soil OC and TN stocks (Mg ha^{-1}) were calculated using the model developed by Ellert and Bettany (1995) as follows:

$$\text{SOC (or TN) Stock} = \text{Conc} * \rho b * T * 10\,000\ \text{m}^2\text{ha}^{-1} * 0.001\ \text{Mg kg}^{-1} \qquad \text{(Eq. 1)}$$

Where: SOC (or TN) Stock = Soil Organic Carbon or Total Nitrogen Stock (Mg ha^{-1}), Conc. = Soil Organic Carbon or Total Nitrogen Concentration (kg Mg^{-1}), ρ_b = Dry bulk density (Mg m^{-3}) and T = Thickness of soil layer (m).

2.5.2. Water stable aggregates, mean weight diameter and geometric mean diameter

Water stable aggregates (WSA) (Kemper and Rosenau, 1986), mean weight diameter (MWD) and geometric mean diameter (GMD) (Castro-Filho et al., 2002; Loss et al., 2011) were calculated according to the following formulae:

$$\%WSA = (\frac{mi}{mj}) * 100 \qquad \text{(Eq. 2)}$$

Where, mi is the mass of aggregates retained in a specific size class of average diameter (g), and mj is total mass of aggregates (g).

$$MWD = \sum_{j=1}^{n} xjmj \qquad \text{(Eq. 3)}$$

Where, j=1-to- n, and n is the number of aggregate ranges, mj is the proportion of each size class to the total sample and xj is mean diameter of the size classes (mm).

$$GMD = \exp\left(\frac{\sum_{i=1}^{n} mi\,ln\,xi}{\sum_{i=1}^{n} mi}\right) \qquad \text{(Eq. 4)}$$

Where, n is the number of aggregate ranges, mj is the weight of the aggregates in each size class(g) and lnxi is the natural logarithm of the mean diameter of the size classes (mm).

2.5.3. Microbial biomass carbon (MBC)

Nine field-moist soil samples (40 g each) from the surface 0-15 cm depth were collected in three replications from three agricultural land uses (AF, IR and RF) in May 2012 for determination of microbial biomass carbon (MBC) by the fumigation-extraction method (Brookes et al. 1985; Vance et al. 1987). For each plot, one out of the three subsamples (each 10.0 g fresh soil) was fumigated with ethanol-free chloroform for 24 h at 25°C in an evacuated extractor. From the remaining two subsamples, one was used for moisture determination, and the other treated as control for each plot. Fumigated and non-fumigated soils were extracted with 40 ml 0.5-mol l^{-1} K_2SO_4 (1:4 soil:extractant) and shaken for 1 h on a reciprocal shaker. The extracts were filtered using Whatman No.42 filter paper of seven-cm diameter and stored frozen at -15°C prior to analysis. Total organic carbon in the extracts was measured using Total Organic Carbon Analyzer (SHIMADZU).

Microbial Biomass Carbon (MBC) was calculated as follows:

$$MBC = \frac{E_C}{KE_C} \qquad \text{(Eq. 5)}$$

Where E_C = (organic C extracted from fumigated soils) - (organic C extracted from non-fumigated soils) and $KE_C = 0.45$ (Wu et al., 1990).

2.5.4. Soil quality index (SQI)

Ten-soil quality indicators grouped into three: (1) physical (BD, WSA and TP), (2) chemical (CEC, pH, TN, AVP, and AVK), and (3) biological (SOC and MBC) were used to undertake SQI evaluation. Threshold values for each soil quality indicator was set based on the range of values measured in natural ecosystems (the adjacent grass pasture in this case), and on critical values in the literature. After finalizing the thresholds, the soil property values recorded under the three agricultural land use systems were transformed into unit-less scores (between 0 and 1), using the following equation (Masto et al., 2007):

$$\text{Non-linear score}(Y) = \frac{1}{(1+e^{-b(x-A)})} \qquad \text{(Eq. 6)}$$

Where, x is the soil property value, A the baseline or value of the soil property where the score equals 0.5 and b is the slope of the tangent to the curve at the baseline.

After transformation of each soil quality indicator, integration of the normalized soil property values into an overall integrated SQI was performed using the framework suggested by Karlen and Stott (1994) as follows:

$$SQI = WE(wt) + WMA(wt) + RD(wt) + PNS(wt) \qquad \text{(Eq. 7)}$$

Where, SQI = overall soil quality index, WE = soil's ability to accommodate water entry, WMA = soil's ability to facilitate water movement and availability, RD = soil's ability to resist degradation, PNS = soil's ability to supply nutrients for plant growth, and wt = a numerical weighting for each soil function.

The numerical weights were assigned to each soil function according to their importance in fulfilling the overall goals of maintaining soil quality under specific conditions. For this study, weight values of 0.2, 0.2, 0.2 and 0.4 were assigned to WE, WMA, RD, and PNS, respectively. More value was assigned to PNS because the use of both organic and inorganic fertilizers was minimal in the study area, and hence nutrient supply was considered the most important

production constraint. Further, sustaining crop production is the major goal of soil management strategies in most developing countries including Ethiopia.

2.5.5. Carbon sustainability index (CSI)

There are various ways to assess sustainability of a production system. For instance, economists use productivity or total factor productivity (Herdt and Steiner, 1995), soil scientists use soil quality (Doran and Parkin, 1994), ecologists use energy coefficients (Odum, 1998) and engineers assess the energy use efficiency (Stout, 1984). However, in the context of the global climate change and anthropogenic emissions of GHGs into the atmosphere, sustainability of a production system can be assessed by evaluating temporal changes through calculating carbon sustainability index (CSI) (Lal, 2004b). Accordingly, CSI was calculated by dividing the difference between total C output and C input by C input (Lal, 2004b) as follows:

$$\text{CSI} = \frac{(C_O - C_i)}{C_i}$$
(Eq. 8)

Where, CSI is C sustainability index, Co is C output, and Ci is C input

2.6. Statistical analyses

Effects of different land use systems and soil depth on SOC and TN and other soil parameters were subjected to one-way ANOVA. Correlation analysis was used to evaluate the relationships among SOC, TN and WSA. The respective correlation coefficients for each land use system were calculated from the average of the whole soils for all depths and land uses. Differences between means of treatments were considered significant at the 0.05 level using the Tukey's studentized (HSD) test. The data were analyzed using SAS version 9.2 software package (SAS, 2007) for papers I, II and V, and R version 3.02 software package (R core Team, 2012) was used for papers III and IV. Additionally, excel spreadsheet was used for computing different parameters estimated using different models.

3. Results and Discussion

3.1.1. Effects of land use and depth on soil organic carbon and total nitrogen storage capacities of soils

Soil OC and TN concentrations differed significantly among different land uses and across depths. Both SOC and TN were higher in OP and SP land uses than other land uses. Additionally, SOC and TN were higher in the top than the deeper soil layers under those land uses. Thus, the highest SOC concentration in 0-5 cm was measured in OP (25.4 g kg^{-1}) followed by that in SP (16.0 g kg^{-1}), and the lowest was in RF (2.29 g kg^{-1}). In 5- 10 and 10-20 cm depths, SOC concentration followed the same trend except that the amount of SOC in OP and SP land use systems decreased by about 50% compared with that in the top 0-5 cm depth (Table 1; Paper I). Total N followed a similar trend. Likewise, SOC and TN concentrations decreased from 0-15 cm to 15-30 and 30-50 cm depths in OP and SP land uses (Paper V). This result was in agreement with a report by Aticho (2013) who found a negative correlation between SOC content and sampling depth in Kafa, Southwest Ethiopia. In contrast, soils under AF and RF land use systems showed different trends in SOC and TN concentrations across depths probably because of the mixing effects of tillage (Beare et al., 1997; Chen et al., 2009; Gelaw et al., 2013) (Table 1; Paper I).

Table 1 Comparison of soil organic carbon and total nitrogen concentrations among different depths within each land use.

Depth (cm)	Land Use				
	AF	RF	OP	IR	SP
Soil Organic Carbon Concentration (g kg^{-1})					
0-5	6.28(0.46)[ab]	2.96(1.46)	25.4(9.25)[a]	6.66(3.15)	16.0 (7.39)
5-10	6.56(0.60)[a]	3.00(1.42)	12.6 (5.48)[b]	5.68(2.68)	8.66(5.15)
10-20	5.67(0.44)[bc]	3.27(1.52)	10.4(4.77)[b]	5.18(2.44)	7.63(4.67)
20-30	4.93(0.43)[c]	4.00(0.87)	9.02(3.69)[b]	4.43(2.22)	7.33(4.86)
		NS		NS	NS
Total Nitrogen Concentration (g kg^{-1})					
0-5	0.68(0.05)[a]	0.31(0.16)	2.24(0.92)[a]	0.53(0.24)	1.62(0.76)[a]
5-10	0.69(0.11)[a]	0.32(0.12)	1.15(0.60)[ab]	0.43(0.20)	0.77(0.50)[ab]
10-20	0.61(0.07)[ab]	0.35(0.15)	0.99(0.56)[b]	0.39(0.20)	0.65(0.44)[ab]
20-30	0.50(0.05)[b]	0.37(0.07)	0.86(0.43)[b]	0.35(0.17)	0.62(0.41)[b]
		NS		NS	

RF, Dryland crop production; AF, F. albida based agroforestry; OP, communal open grazing/pasture; IR, irrigation based fruit production; SP, F. albida based silvopasture.

± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, P = 0.05)

Similarly, SOC and TN stocks are higher in the topsoil layers under tree-and grass-based land use systems (OP, SP, and AF) compared with those in RF indicating the potentials of tree-and grass-based land use systems for soil carbon sequestration (SCS) in the region. Girmay et al. (2008) estimated rates of SCS potentials of currently degraded soils in Ethiopia under rangeland, irrigation, and rain-fed cropping land uses over the next 50 years with widespread adoption of soil-specific restoration measures in the order: 0.3–0.5, 0.06–0.2, and 0.06–0.15 Mg C ha−1 yr−1, respectively. Likewise, Mekuria et al.(2009) reported 36-50 % increase in mean SOC stock through conversion of degraded grazing lands to exclosures, areas closed from human and animal interference to promote natural regeneration of plants on formerly degraded communal grazing lands, in Tigray, Northern Ethiopia. On the other hand, the results of this study indicate the risks of CO_2 release if these land uses are converted to croplands. In agreement with this assertion, Fantaw et al. (2006) in their study in southeastern Ethiopia reported an average of 40 – 45 % SOC stock held in the top 30-cm of 1-m depth mineral soils, indicating the risks of large amounts of CO_2 release following deforestation and subsequent cultivation. Furthermore, accumulations of SOC and TN stocks calculated for 0-30 cm soil layer taking RF as a baseline were in the order: OP >SP >AF >IR (Paper I) due to differences in organic inputs. Similar results were also found when SOC and TN stocks were calculated for OP, SP and AF land uses in comparison with that in RF in 0-50 cm soil layer and an assumption of 50 years duration since conversion (Table 2; Paper V).

Table 2 Magnitude and rates of Soil Organic Carbon and Total Nitrogen Stocks Accumulation in Four Different Land Uses in 0-50 cm depth in 50 Years.

Land use	SOC Stock	SOC Accumulation (Stock-RF)	Rate of SOC Accumulation	TN Stock	TN Accumulation (Stock-RF)	Rate of TN accumulation
	(Mg C ha^{-1})	(Mg C ha^{-1})	(Mg C ha^{-1}yr^{-1})	(Mg C ha^{-1})	(Mg C ha^{-1})	(Mg C ha^{-1}yr^{-1})
RF	29.4(3.5)[b]	-	-	2.8(0.2)[b]	-	-
AF	36.6(1.5)[ab]	7.2(4.7)	0.14(0.1)	3.9(0.2)[ab]	1.1(0.2)	0.02(0.00)
OP	64.2(11.8)[a]	34.8(11.0)	0.70(0.2)	6.0(1.5)[a]	3.2(1.4)	0.07(0.03)
SP	49.4(14.9)[ab]	20.0(14.0)	0.40(0.3)	3.8(1.3)[ab]	1.0(0.6)	0.02(0.03)
	NS	NS		NS	NS	

RF: Dryland crop production; AF: *Faidherbia albida* based agroforestry; OP: communal open grazing/pasture; SP: *Faidherbia albida* based silvopasture.

± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, P = 0.05).

3.1.2. Soil organic carbon and total nitrogen concentrations associated with aggregate sizes and primary particles under different land uses

Land use had a significant effect on aggregate size distribution in surface soils. The highest WSA in 0- to 10-cm depth was measured in OP followed by that in SP land use. Both land uses received more biomass input from grass and tree residues than that added in cultivated soils. Because residues of crops are harvested for fuel and feed in Tigray in particular (Hengsdijk et al., 2005; Girmay, 2009) and in Ethiopia in general (Amare et al., 2005; Girmay et al., 2008), SOM and WSA are adversely affected in agricultural land uses. Further, soils under OP and SP land use systems had more macroaggregates, whereas cultivated soils had a higher proportion of microaggregates because tillage has been found to induce loss of C-rich macroaggregates and a gain of C-depleted microaggregates (Six et al., 2000) (Paper II).

Moreover, land use had significant effects on SOC and TN concentrations associated both with aggregate sizes and with primary particles. The highest concentration of SOC associated with macroaggregates was measured in OP followed by that in SP and AF in both surface (0-10 cm) and subsurface (10-20 cm) soil layers (Table 3). Besides, SOC and TN concentrations associated with macro- and microaggregates decreased with depth in these land uses. In contrast, SOC and TN concentrations associated with macroaggregates in RF showed an opposite trend due to the adverse effects of tillage on the surface 0-10 cm depth (Table 3; Paper II). This result is in agreement with Ashagrie et al. (2005) who indicated that SOM that binds microaggregates into macroaggregates is a labile fraction and is highly sensitive to land use change and cultivation.

Table 3 Land use effects on water stable aggregation (%) and soil organic carbon and total nitrogen concentrations (g kg^{-1}) associated with micro- and macroaggregates

Land use	Water Stable Aggregates	Soil Organic Carbon (g kg^{-1})		Total Nitrogen (g kg^{-1})	
	(%)	**Macro**	**Micro**	**Macro**	**Micro**
		Soil depth 0-10 cm			
AF	56.8(10.8)[bc]	7.20(5.07)[b]	2.56(0.44)	0.64(0.47)[b]	0.26(0.04)[a]
RF	37.5(10.4)[c]	2.12(0.90)[b]	1.89(0.99)	0.16(0.08)[b]	0.18(0.08)[ab]
Op	88.7(6.4)[a]	19.95(11.15)[a]	1.30(0.21)	1.71(1.03)[a]	0.11(0.03)[b]
IR	52.3(9.9)[bc]	6.64(3.14)b	2.25(1.08)	0.46(0.30)b	0.20(0.12)ab
SP	69.3(12.5)[b]	7.97(3.31)b	2.29(0.34)	0.73(0.36)ab	0.22(0.04)ab
			NS		
		Soil depth 10-20 cm			
AF	57.3(11.0)	4.83(3.02)	2.59(1.46)	0.45(0.31)	0.23(0.09)
RF	44.2(6.9)	2.28(0.60)	2.13(0.82)	0.18(0.08)	0.19(0.10)
Op	63.6(32.4)	10.61(9.49)	1.46(0.36)	0.94(0.90)	0.13(0.04)
IR	59.5(18.1)	3.56(2.02)	1.32(0.48)	0.30(0.21)	0.10(0.04)
SP	69.1(14.2)	5.40(3.38)	1.46(0.47)	0.43(0.30)	0.12(0.04)
	NS	NS	NS	NS	NS

RF, Dryland crop production; AF, F. albida based agroforestry; OP, communal open grazing/pasture; IR, irrigation based fruit production; SP, F. albida based silvopasture.

± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, P = 0.05).

Macroaggregates contained higher SOC concentrations than microaggregates in both depths and all land uses. This result was in agreement with Tisdall and Oades (1980) who advocated the idea that the presence of decomposing roots and hyphae within macroaggregates increased SOC concentrations than in microaggregates. Elliott (1986) also suggested that macroaggregates have elevated SOC concentrations because of the organic matter binding microaggregates into macroaggregates. The magnitude of soil disturbance and the amount of residue incorporated into the soil impacted water stable aggregates (WSA) and the associated SOC and TN (Blanco-Canqui and Lal, 2004; Srinivasarao et al., 2011). Therefore, WSA was strongly correlated with SOC ($R^2 = 0.85$; Fig. 1a) and TN ($R^2 = 0.79$; Fig. 1b) concentrations (Paper II).

(a)

(b)
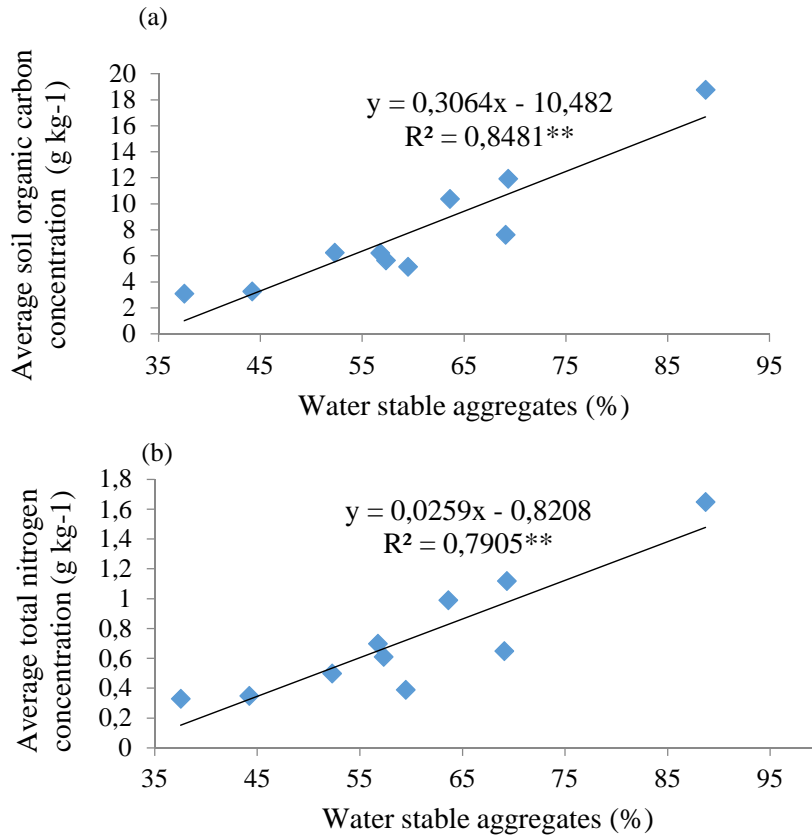
Figure 2 Correlations among Soil Organic Carbon, Total Nitrogen and Water Stable Aggregation Under Different Land Uses.

Similarly, the presence of higher SOC and TN concentrations associated with sand, silt and clay particles in soils under OP and SP than other land uses further indicated the detrimental effects of cultivation on SCS potentials of soils (Tiessen and Stewart, 1983; Dalal and Mayer, 1986; Wu et al., 2006) (Table 4). Moreover, the higher SOC concentrations associated with clay particles in soils under OP, SP and AF indicated that grass- and tree-based land use systems are rich in stable SOC as clay-associated SOC has higher residence time than sand or silt-associated SOC fractions (Ashagrie et al., 2005) (Paper II).

Table 4 Land use effects on soil organic carbon and total nitrogen concentrations (g kg^{-1}) associated with primary particles

	Soil Organic Carbon (g kg^{-1})			Total Nitrogen (g kg^{-1})		
Land use	Sand	Silt	Clay	Sand	Silt	Clay
	Soil depth 0-10 cm					
AF	2.8(1.0)[bc]	0.39(0.05)[b]	1.7(0.3)[b]	0.23(0.10)[bc]	0.04(0.01)[b]	0.22(0.04)[bc]
RF	0.80(0.2)[c]	0.37(0.25)[b]	1.3(0.8)[b]	0.07(0.02)[c]	0.04(0.03)[b]	0.15(0.08)[c]
OP	8.5(2.9)[a]	5.9(3.0)[a]	3.7(0.8)[a]	0.62(0.22)[a]	0.55(0.30)[a]	0.48(0.12)[a]
IR	2.2(1.3)[c]	1.1(1.1)[b]	1.8(0.7)[b]	0.18(0.09)[c]	0.10(0.08)[b]	0.20(0.09)[bc]
SP	6.8(3.9)[ab]	2.2(2.1)[b]	2.6(1.0)[ab]	0.58(0.34)[ab]	0.21(0.21)[b]	0.35(0.15)[ab]
	Soil depth 10-20 cm					
AF	1.4(0.4)[b]	0.32(0.16)[b]	2.1(0.5)[ab]	0.12(0.04)[b]	0.04(0.02)[b]	0.24(0.06)[ab]
RF	0.80(0.1)[b]	0.36(0.29)[b]	1.5(0.5)[b]	0.07(0.02)[b]	0.04(0.04)[b]	0.17(0.06)[b]
OP	4.1(1.8)[a]	4.0(2.9)[a]	3.3(0.8)[a]	0.30(0.15)[a]	0.37(0.28)[a]	0.39(0.12)[a]
IR	1.2(0.3)[b]	0.89(0.77)[b]	1.7(0.6)[b]	0.16(0.08)[ab]	0.08(0.07)[b]	0.18(0.07)[b]
SP	1.5(0.6)[b]	1.2(1.5)[ab]	2.1(1.1)[ab]	0.13(0.04)[b]	0.10(0.03)[b]	0.23(0.13)[ab]

RF, Dryland crop production; AF, F. albida based agroforestry; OP, communal open grazing/pasture; IR, irrigation based fruit production; SP, F. albida based silvopasture.

± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, P = 0.05)

3.2. Soil quality indices for evaluation of smallholder Agricultural land use systems

Land use had significantly affected majority of the individual soil functions considered for this study and the overall integrated soil quality index (SQI) (Table 5; Paper III). Thus, AF had significantly higher (P <0.05) score than RF for three out of the four soil functions studied. The three soil functions affected by land use system were those to: (1) accommodate water entry, (2) facilitate water movement and availability, and (3) supply plant nutrients. The relatively higher values of soil physical, chemical and biological quality indicators under AF land use were responsible for the improvements. On the other hand, soils under all the three agricultural land use systems showed low resistance to surface degradation revealing the detrimental effects of tillage on soil quality (Karlen et al., 1994; Glover et al., 2000).

When all the soil quality functions were integrated using the framework suggested by Karlen and Stott (1994), SQI values were in the order: 0.58 (AF) >0.51 (IR) >0.47 (RF). However, it was only between AF and RF land use systems that showed a significant difference (P <0.01) in SQI value. Nevertheless, integrated SQI values for soils under all the three land uses were very small compared with that of an ideal soil (SQI=1.00) (Table 5). Two major causes could be ascribed to this result: (1) Arenosols in the region are inherently infertile soils (Hartemink and Huting, 2008; Rabia et al., 2013), and (2) nutrient mining as a result of crop harvests and complete removal of crop residues for feed and fuel are common practices in the region (Hengsdijk et al., 2005; Girmay, 2009). This result was also in agreement with findings of several authors (Amare et al., 2005; Hengsdijk et al., 2005; Girmay, 2009; Kebede and Yamoah, 2009) who reported that low organic matter and nutrient stocks are typical characteristics of soils in Tigray.

On the other hand, the results of this study confirmed findings by several researchers in Ethiopia and elsewhere that *F. albida* agroforestry trees enhance soil fertility and increase crop yields (Poschen, 1986; Kamara and Haque, 1992; Sanchez, 1994; Palm, 1995; Asfaw and Ågren, 2007; Hadgu et al., 2008). One fundamental principle of sustainability is to return to the soil the nutrients removed through harvests and other loss pathways (Sanchez, 1994), and one of the main tenets of agroforestry is that trees enhance soil fertility (Sanchez, 1994; Palm, 1995). Thus, adoption of *F. albida* based agroforestry systems combined with other recommended management practices such as no-tillage, residue and manure management are needed to build up soil structure, organic matter and moisture holding capacity of soils. This will help to build climate resilient agriculture in the region through efficient and sustainable use of the soil resources.

Table 5 Effects of Land Use Type on Four Soil Functions and on an Integrated Soil Quality Index in 0-15 cm depth

Soil Function	Land use			
	RF	AF	IR	
Accommodate Water Entry (0.20)	0.09 (0.00) [b]	0.11 (0.002) [a]	0.10 (0.004) [ab]	*
Facilitate Water Entry and Availability (0.20)	0.10 (0.004) [b]	0.12 (0.004) [a]	0.11 (0.004) [ab]	*
Resist Surface Degradation (0.20)	0.09 (0.003)	0.11 (0.002)	0.09 (0.005)	NS
Source of Plant Nutrients (0.40)	0.19 (0.01) [b]	0.24 (0.004) [a]	0.21 (0.015) [ab]	*
Integrated Soil Quality Index (1.00)	0.47 (0.01) [b]	0.58 (0.01) [a]	0.51 (0.02) [ab]	**

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; IR, irrigation-based *P. guajava* fruit production

± Mean values followed by standard errors in the parentheses; values with different letters are significantly different. .* = P < 0.05; ** = P < 0.01; NS = not significant

(Tukey's test, P = 0.05).

3.4. Carbon footprint and sustainability of smallholder cereal production systems in Ethiopia

The trend for 18-years C-based inputs for Ethiopian smallholder cereal production systems started with a low value at 0.32 Tg Ceq y^{-1} in 1994 and increased to reach the highest at 0.62 Tg Ceq y^{-1} in 2010. Likewise, total C-output increased linearly from the lowest at 5 Tg Ceq y^{-1} in 1994 to the highest at 17 Tg Ceq y^{-1} in 2011. Further, the carbon sustainability indices (CSI) of the farming systems during the study period (1994 to 2011) were in the range of 15 to 28, which were comparable with those of other intensive systems in other regions of the world (Figure 3; Paper IV). For example, the CSI of agriculture in the state of Ohio, USA, between the 1950s and 1980s was in the range of 20 to 27, and that of Punjab, India, between the 1970s and 1980s was in the range of 15 to 30 (Dubey and Lal, 2009).
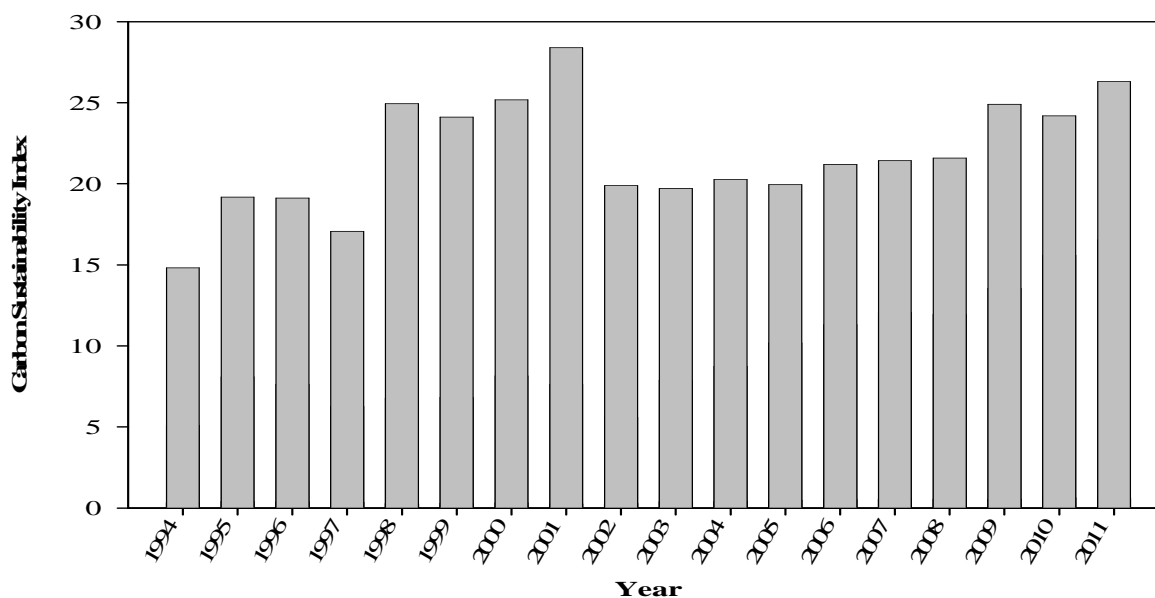
Figure 3 trends in carbon sustainability index in Ethiopian smallholder cereal production systems between 1994 and 2011.

In the last two decades, total area cultivated under cereal production increased 1.5-folds from 6.42 to 9.63 million hectares; total nitrogen and phosphorus fertilizers used increased almost 3-folds from 120,000 to 350,000 tons, and total cereal grain produced has increased a little more than 3-folds from 5.77 to 18.68 Tg. Similarly, national average cereal yield increased from <1Mg ha^{-1} in 1994 to ~2Mg ha^{-1} in 2011, which was higher than that of the SSA

average at 1.3Mg ha^{-1} (World Bank, 2014). However, it was less than the average cereal yield for other developing countries; e.g., 2.4Mg ha^{-1} for Middle East and North Africa, 3.1Mg ha^{-1} for Europe and central Asia, and 3.8Mg ha^{-1} for Latin America and Caribbean (World Bank, 2014). Further, fertilizer-use efficiency (FUE) of cereal production (Tg grain/Tg fertilizer) was between 40 and 60, except the extremely high values during the years 1998-2002 with the least average fertilizer applications. The stagnated FUE indicated that further increase in the applications of N and P fertilizers were not effective in increasing yields. Results elsewhere indicated that high returns from modern inputs are obtained if fertilizer, improved seeds and other recommended practices are all applied together. The supply of improved seeds in Ethiopia has lagged behind at 5% and irrigation use at <1% (Dercon and Hill, 2009). Thus, the recent increase in total cereal production in the country may be attributed mainly to the increase in cultivated land during the same period, which raises questions about the quality of the land brought under cultivation and the sustainability of the process.

4.   Conclusions and Recommendations

According to the results of the study at Mandae watershed, SOC and TN concentrations generally decreased with depth in grass-and tree-based land uses. However, soils in intensively cultivated land uses showed different patterns in SOC and TN concentrations with depth may be because of the mixing effects of tillage. Most of the SOC and TN in grass-and tree-based land uses (OP and SP) concentrated in the top 0-5 and 0-15 cm layers, indicating the risks of large amounts of CO_2 to be released from the surface soil if these systems are converted into arable lands. Further, soils under agroforestry and fruit tree-based agricultural land uses (AF and IR) also had higher SOC and TN concentrations than that under sole-cropping (RF) system, indicating a large potential of adopting agroforestry and irrigation-based fruit production practices to sequester SOC and TN in these soils. Furthermore, the significant improvements in total SOC and TN stocks measured under grass-and tree-based land use systems in both the 0-30 and 0-50 cm depths than those measured under continuous sole cropping system (RF) revealed the carbon sequestration (CS) potentials of adopting these grass-and tree-based land uses systems in the region.

Moreover, the presence of higher proportions of water stable aggregates in the surface soil layers under OP and SP land uses compared to cultivated soils showed the amount of grass and tree residues added to these soils and their degree of decomposition, which are vital factors in the formation and stabilization of aggregates. Soils under OP and SP land use systems followed by that under AF also had greater amounts of macroaggregates, while soils under RF had a higher amount of microaggregates. Tillage practices under the three agricultural land use systems (RF, AF and IR) may have been responsible for the loss of C-rich macroaggregates. Accordingly, the highest concentrations of SOC and TN associated with macroaggregates were measured in grass-and tree-based (OP, SP and AF) land uses in both 0-10 and 10-20 cm depths. This study also indicated that sand, silt and clay-associated SOC and TN concentrations were higher in soils under uncultivated/undisturbed systems, supporting the hypothesis that SOC and TN distribution in different size fractions were influenced by soil cultivation.

Soil quality assessment and comparison among the three agricultural land uses (AF, RF and IR) showed relatively higher WSA, TN and SOC concentrations in soils under AF land use. Thus, the result further indicated improvements in water entry, movement and availability functions of the soil under AF than under IR and RF land uses. The soil's ability to supplying plant nutrients was also improved under AF land use than RF largely due to higher levels of AVK, CEC, SOC, TN and AVP in the rooting zones of AF. Further, when selected physical, chemical, and biological soil quality indicators were integrated into an overall SQI, AF land use received a higher rating (0.58) than that of RF (0.47). The result also indicated that SQI for AF would likely have been higher if not for the continuous tillage practices, which disrupted structures of the soil.

Carbon footprint assessment of the smallholder cropping systems in Ethiopia showed a strong relationship between annual total C-based input and total annual C-output. The result also indicated that the annual C-sustainability indices (CSI) of the farming systems were comparable with other more intensive systems in other regions. However, national average cereal-grain yield gaps are still larger when compared with other developing regions and some managed experimental plots in the country. Moreover, fertilizer use efficiency (FUE) showed no major improvements with further increase in applications of N and P fertilizers. As demonstrated by studies in several places elsewhere, higher returns from modern inputs are obtained if fertilizer, improved seeds and other recommended practices are all applied together. However, supply of improved seeds and use of irrigation in Ethiopia are lagging behind at <5 % and <1 %, respectively. Thus, horizontal expansion in area may have been the major source of recent production increases in the country, which according to some studies is not sustainable.

In conclusion, the result of this study, both from the case study in Tigray and the C-footprint assessment for the whole country, highlighted the need for Ethiopian smallholder agricultural production systems to shift to sustainable intensification via adoption of restorative land uses and recommended management practices (RMPs). Improvement of crop yields by intensification on land already under cultivation and conservation of the remaining

grazing lands and forests should be prominent among a portfolio of agricultural development strategies both at regional and national levels for adaptation and mitigation of climate change and advancement of food security.

5.   Future Perspectives

1. The results of this study showed the potentials of incorporating *F. albida* trees into farms and conserving grass- and tree-based land use systems for soil carbon sequestration within a dry land watershed in Eastern Tigray, Northern Ethiopia. In different parts of Ethiopia, there are numerous types of tree- and grass-based land use systems of different species of trees and grasses and under different climatic and soil conditions. Therefore, it is very important to study the impacts of these land use systems on carbon sequestration both in the form of biomass and SOC. Moreover, currently there are massive efforts going on throughout the country to restore degraded lands using different physical and biological measures. It will be paramount importance to collect baseline data at this initial time to evaluate the impacts of these efforts in terms of carbon sequestration, agricultural productivity and other ecosystem services.
2. Soil quality indexing for soils under three agricultural land use systems showed a significant improvement in the quality of soils under canopies of *F. albidia* trees. However, the study was conducted with limited data for financial constraints. Therefore, it will be necessary to repeat this research with more data and land use systems. It will also be important to correlate the soil quality indices with crop yields and other ecosystem services at different climatic and soil conditions.
3. Carbon footprint assessment was conducted only for the smallholder cereal-production system, which is of course the dominant system in the midlands and highlands of Ethiopia. However, recently commercial farms mainly by foreign investors using more carbon-based operations and products is expanding in the lowlands of the country, where the remaining forestland is located. Therefore, it is high time to evaluate this system's sustainability and correct before much damage is made on the environment.

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Paper I

Gelaw AM, Singh BR, Lal R. 2014. Soil organic carbon and total nitrogen stocks under different land uses in a semi-arid watershed in Tigray, Northern Ethiopia. **Agriculture Ecosystems and Environment, 188:256-263.**

Soil organic carbon and total nitrogen stocks under different land uses in a semi-arid watershed in Tigray, Northern Ethiopia

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ARTICLE INFO

ABSTRACT

In Ethiopia, massive deforestation of natural forests and extensive use of agricultural lands have resulted in soil degradation. Soil organic carbon (SOC) quantity and quality are crucial to soil quality. However, knowledge on the effects of land use change on soil carbon storage in semi-arid northern Ethiopia is very limited. To address this problem, a study was undertaken within a semi-arid watershed in eastern Tigray, Northern Ethiopia, to estimate SOC and total nitrogen (TN) concentrations and stocks in 0–5, 5–10, 10–20 and 20–30 cm soil layers for five land uses: rainfed crop production (RF), agroforestry based crop production (AF), open communal pasture (OP), silvopasture (SP) and irrigation based fruit production (IR) each with five replications. Generally, both magnitude and difference in SOC and TN concentrations showed a decreasing trend with depth within and among most land uses. SOC and TN concentrations were highly correlated in all land uses and depths. Total stocks in 0–30 cm layer were 25.8, 16.1, 52.6, 24.4 and 39.1 Mg ha^{-1} for SOC compared with 2.7, 1.6, 4.9, 1.9 and 3.5 Mg ha^{-1} for TN in AF, RF, OP, IR and SP land uses, respectively. With RF as baseline and the duration of 50 years since land use conversion, the average rate of accumulation was 0.73, 0.46, and 0.19 Mg C ha^{-1} yr^{-1} in comparison with 0.065, 0.038, and 0.022 Mg N ha^{-1} yr^{-1} for OP, SP and AF, respectively. Soils under IR also accumulated 0.56 Mg C ha^{-1} yr^{-1} and 0.019 Mg TN ha^{-1} yr^{-1} in the 0–30 cm layer and in comparison with the RF land use system on an average of 15 years. The results of this study revealed that conversion of croplands to grasslands or integration of appropriate agroforestry trees in cropping fields in the region has large technical potential of SOC and TN sequestrations.

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1. Introduction

Ethiopia is one of the largest countries in Africa both in terms of land mass and population. The vast majority of the population (>80%) depends on agriculture which contributes around 42% of its national gross domestic product (GDP) (Barrow et al., 2011). Despite its strategic importance for the country's economic development, the agricultural sector suffers from low efficiency, population pressure, ineffective land management, and unfavorable land use practices leading to widespread land degradation and loss of environmental quality (Ashagrie et al., 2005; Amare et al., 2013). As a result of this human interference on natural forest and grazing lands, soil organic matter (SOM) has declined to low level especially in cultivated soils of the highlands (Chibsa and Ta, 2009).

Global climate change caused by rising levels of carbon dioxide (CO_2) and other greenhouse gases is recognized as a serious environmental issue of the twenty-first century (Kumar and Nair, 2011). Agriculture is the human enterprise that is most vulnerable to climate change. Tropical agriculture, particularly subsistence agriculture is particularly vulnerable, as smallholder farmers do not have adequate resources to adapt to climate change (Verchot et al., 2007). The role of land use systems in stabilizing CO_2 levels and increasing carbon (C) sink potentials of soils have attracted considerable scientific attention in the recent past (Kumar and Nair, 2011). Soil organic matter (SOM), which contains more reactive SOC than any other single terrestrial pool, plays a major role in determining C storage in ecosystems and moderating atmospheric concentrations of CO_2 (Post et al., 1982). Soil C sequestration is the process of transferring CO_2 from the atmosphere into the soil in a form that is not immediately reemitted, and this process is being considered as a strategy for mitigating climate change (Sundermeier et al., 2005; Lal et al., 2007; Chen et al., 2009). It is a natural, cost-effective, and environment-friendly process (Lal, 2004). Once sequestered, C remains in the soil as long as restorative land use, no-till farming,

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and other recommended management practices are developed (Lal, 2004).

Agricultural soils can be a source or a sink of atmospheric CO_2 (Al-Kaisi et al., 2005; Chen et al., 2009). Land misuse and soil mismanagement can cause depletion of soil organic carbon (SOC) stock with an attendant emission of CO_2 into the atmosphere (Reicosky et al., 1997; Lal, 2004; Chen et al., 2009). In contrast, an appropriate land use and soil management with RMPs can increase SOC stock thereby reducing net emission of CO_2 to the atmosphere (Paustian et al., 2000; Sampson and Scholes, 2000); increase sustainability of farming systems and contribute to reducing farmers' vulnerability to climate variability (Verchot et al., 2007).

Conversion of cropland to grassland is one of the most effective strategies for C sequestration (Lal et al., 1999; Smith et al., 2000; Chen et al., 2009). Grasslands generally have higher SOC levels than cropland. For example, Omonode and Vyn (2006) estimated that soils under warm season native grasses sequestered on average 2.1 Mg C ha^{-1} y^{-1} more than corn (Zea mays L.)-Soybean (Glycine max L.) crop sequence in the upper 2.5 cm layer soil in west-central Indiana, USA. Chibsa and Ta (2009) also found higher SOC content in grassland soils than in fallow lands and cultivated land use systems in Bale, Southeast Ethiopia.

Agroforestry (AF), the practice of growing trees and crops in interacting combinations on the same unit of land, primarily by resource-poor smallholder farmers in developing countries, is an important strategy for soil carbon sequestration (SCS) (Nair et al., 2009). Several studies under various agroforestry systems (AFS) in diverse ecological conditions indicated that tree-based agricultural systems, compared to treeless systems, store more C in soils under comparable conditions (Roshetko et al., 2002; Mutuo et al., 2005; Verchot et al., 2007; Takimoto et al., 2008; Gupta et al., 2009; Kumar and Nair, 2011; Murthy et al., 2013). Some studies conducted in different parts of Ethiopia also showed higher levels of SOC under canopies of different agroforestry tree species than those outside the tree canopies. For instance, Abebe (1998) reported 35- and 43% increase in SOC contents in the top soil layer under canopies of *Cordia africana* trees in rangeland and cropland ecosystems, respectively, than those in their immediate surface soils in Western Ethiopia. Similarly, Enideg (2008) estimated 46.69 and 23.01% increase in SOC content in the surface and subsurface soils, respectively under canopy zones of *Ficus thonningii* trees as compared to those beyond the canopy zones in Gondar district, Northwestern Ethiopia. Berhe et al. (2013) also observed a significant decrease in SOC with increasing distance from *Ficus Thonningii* tree trunks in central Tigray, Northern Ethiopia. Similar results were also reported by Tadesse et al. (2000) in Southern Ethiopia under *Millettia ferruginea* agroforestry trees.

In Tigray, Northern Ethiopia, large part of the area was once covered with acacia woodland (Eweg et al., 1998). The early 1960s marked the disappearance of much of the woodland under pressure from the rapidly growing population (Eweg et al., 1998). Nowadays, in the region, farmers take care of naturally growing *Faiderbhia albida* ((Del.) A Chev.) trees in and around their farm and grazing lands in order to improve soil fertility and increase crop and pasture yields (Hadgu et al., 2009). Several authors have shown the positive effects of *F. albida* parkland agroforestry trees on yields of different crops in traditional smallholder farming systems in different parts of Ethiopia (Poschen, 1986; Kamara and Haque, 1992; Asfaw and Ågren, 2007; Hadgu et al., 2009). However, no information was available in these studies on the effects of *F. albida* park land agroforestry trees on SOC sequestration of soils under their canopies particularly in this part of the country, Tigray.

In arid and semi-arid regions, irrigation is essential for crop production in order to increase water availability in the soil (Feng et al., 2005) which in turn enhances biomass production, increases the amount of above ground and root biomass returned to the soil and improves SOC concentration in the soil (Lal, 2004). In the semi-arid parts of Ethiopia, high rainfall variability contributes to low crop yields and associated low incomes of farmers (Moges et al., 2011). One option to improve the performance of agriculture in this part of the country is by introducing rainwater harvesting practices (Moges et al., 2011; Dile et al., 2013). Therefore, towards enhancing agricultural development, rainwater harvesting has been widely adopted in many parts of the country over the last few decades (Moges et al., 2011). Further, field studies from Northern Ethiopia on *in situ* water harvesting systems showed improvement in soil water content in the root zone (McHugh et al., 2007; Araya and Stroosnijder, 2010) and increase in crop yield (Araya and Stroosnijder, 2010). Water harvesting systems function as climate change adaptation strategy in dryland agricultural systems by strengthening the resilience of these systems against shocks by securing adequate water availability, plant water uptake capacity, and nutrient availability (Falkenmark and Rockström, 2008). Moreover, water harvesting systems can sequester carbon in vegetation and soil, contributing to mitigation of climate change effects (Lal et al., 1998; Conant et al., 2001; Liniger et al., 2011). Therefore, most of the future growth in crop production in developing countries including Ethiopia is likely to come from intensification, with irrigation playing an increasingly strategic role (Postel, 2003; FAO, 2011).

Type of land use system is an important factor controlling soil organic matter content since it affects amount and quality of litter input, litter decomposition rates and processes of organic matter stabilization in soils (Römkens et al., 1999). The role of land use systems in stabilizing CO_2 levels and increasing carbon (C) sink potentials of soils has attracted considerable scientific attention in the recent past (Kumar and Nair, 2011; Murthy et al., 2013). In Ethiopia very few studies have been conducted on SOC and TN storage capacities of soils under different land uses. The purposes of this study was therefore to determine SOC and TN storage capacities of five land use systems in northern Ethiopia and to generate relevant information for land managers and policy makers to help them for their land management decisions. To achieve these goals SOC and TN concentrations and stocks of soils in five land use systems: treeless rain-fed cultivation (RF) *F. albida* based agroforestry (AF), irrigation based Guava fruit production (IR), open pasture (OP) and *F. albida* based silvopasture (SP) were measured each at four soil depths (0–5, 5–10, 10–20 and 20–30 cm) and compared.

2. Materials and methods

2.1. Study location

Mandae watershed is located in Tigray Regional State, northern Ethiopia. Geographically, the study site is located between 15°26′00N to 15°32′00N latitude and 55°00′00E to 55°60′00E longitude, having an area of about 10 km^2, and an elevation of 1960–2000 m above sea level (m.a.s.l.). The average daily air temperature of the area ranges between 15 °C and 30 °C in winter and summer, respectively. The mean annual rainfall of the area is about 558 mm, with a large inter-annual variation. Soils in the watershed are classified as Arenosols, and association of Arenosols with Regosols according to the World Reference Base for soil resources (WRB, 2006). These soils are developed from alluvial deposits and Adigrat sandstones. Textures of these soils were dominated by sand, loamy sand and sandy loam fractions and pH ranged from 6.8 to 7.9 (Rabia et al., 2013). Major land uses of the watershed include *F. albida* based agroforestry (27.7 ha), rainfed crop production (11.9 ha), open pasture (23.2 ha), irrigation based guava fruit production (11.3 ha) and *F. albida* based silvopasture (11.7 ha) (Fig. 1). Agricultural rotation in the selected farms is usually maize
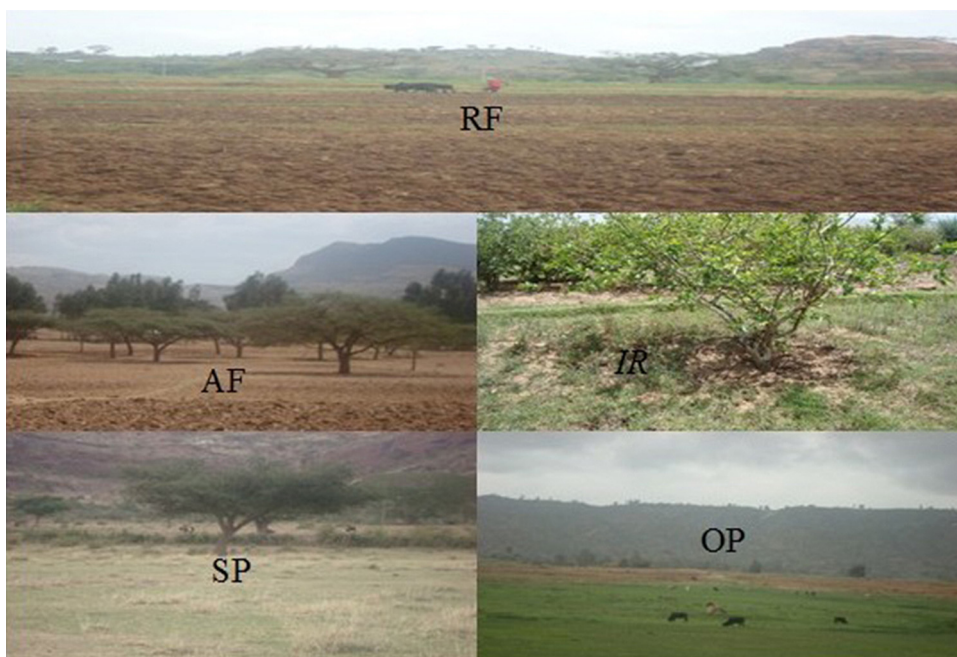
Fig. 1. Different land use systems in the area with rainfaed crop production (RF), F. albida based agroforestry (AF), F. albida based silvopasture (SP), open pasture (OP), and irrigation based P. guajava fruit production (IR).

(*Zea mays*)-teff (*Eragrostis tef*)-field beans (*Vicia faba*)-finger millet (*Eleusine coracana*) in the agroforestry and rainfed cultivation land use systems. Fallow is not practiced in the area due to population pressure and scarcity of farmlands. Use of chemical fertilizers is minimal and land is prepared for cultivation by using a wooden plow with oxen. Farming in the area is a typical extractive type. Crop residues and manures are used for animal feed and fuel, respectively. No pesticides and other agricultural chemicals are used in the area. Irrigation from shallow wells started in the area in late 1990s and most of the irrigated areas are covered by guava fruits with some vegetables such as tomatoes and onions. Guava fruit production is dominant in the irrigated sites because of its lower demand for water and external inputs such as fertilizer (personal communication with farmers).

Faidherbia albida (Del.) (A Chev.) trees in and around farm and grazing lands are remnants from the original woodland in the region. These trees are selectively left by farmers to improve soil fertility and increase crop and pasture yields (Hadgu et al., 2009). Grazing lands/open pastures and silvopastures are commonly owned and managed by communities. Trees are integrated into the pasture systems, the silvopasture systems, on the peripheral marginal parts where fertility of the soils was low (Gelaw et al., 2013). Further, no external inputs such as fertilizers are used and no rotational grazing or other kinds of management techniques are practiced in both the open pasture and the silvopasture systems. However, grazing pressure will be higher in these systems only during cropping seasons, June to October; otherwise animals graze freely all over the different land use systems except the irrigation land use system in the area. Mixed crop-livestock, smallholder farming is a typical farming system of the region.

2.2. Soil sampling and analysis

A total of 100 evenly distributed soil samples (four depths, five land uses, and five replications) were collected using a soil auger. Soil samples were taken to 30 cm-depth (for assessing C and N stocks as per the IPCC guidelines (IPCC, 2006)), and separated into increments of 0–5, 5–10, 10–20 and 20–30 cm depths.

The summit position of the watershed was excluded to minimize the confounding effects of slope and erosion. Individual farms of different sizes in AF, RF and IR land uses were used as replicates. In OP and SP which are communal lands, adjacent plots to sampled fields in other land uses were used as replicates. Soil cores within each replicate were collected randomly from eight points within a 64 m^2 area at each sampling site/replicate and were well mixed and combined to a composite sample by depth. Thus, a minimum of 40 point samples were represented in computing the average values of each soil parameter. Samples were air-dried, gently ground and passed through a 2-mm sieve. Identifiable crop residues, root material, and stones were removed during sieving. Soil samples for C and N analyses were additionally ground using a ball-mill grinder. Concentrations of SOC and TN (% w/w) were determined at the Carbon Sequestration and Management Center Laboratory (The Ohio State University, USA) using auto CN analyzer (Vario Max CN Macro Elemental Analyser, Elementar Analysensysteme GmbH, Hanau, Germany) by dry combustion method (Nelson and Sommers, 1996). Soil bulk density (ρ_b) samples were taken for the same depth intervals as other soil samples for each replicate/plot by the core method (Blake and Hartge, 1986). Core samples were collected from all depth intervals using 100 cm^3 volume stainless steel tubes (5 cm diameter and 5.1 cm height). The initial weight of soil core from each layer was measured in the laboratory immediately after collection. Simultaneously, soil moisture content was determined gravimetrically by oven drying the whole soil at 105 °C for 24 h to calculate the dry ρ_b. No adjustment was made for rock volume because it was rather minimal. Soil ρ_b value for each depth interval was used to calculate the SOC and TN stocks (Mg ha^{-1}) using the model by Ellert and Bettany (1995):

$$\text{SOC (or TN) Stock} = \text{Conc.} \rho b \cdot T \cdot 10\,000\,\text{m}^2\,\text{ha}^{-1} \cdot 0.001\,\text{Mg kg}^{-1} \quad (1)$$

where

SOC (or TN) Stock = Soil Organic Carbon or Total Nitrogen Stock (Mg ha^{-1}).

Conc. = Soil Organic Carbon or Total Nitrogen Concentration (kg Mg^{-1}).

Table 1
SOC and TN concentrations, SOC and TN stocks, C:N ratio and BD of soils under five different land uses and four soil depths.

Land use	Soil organic Carbon		Total Nitrogen		C:N	Bulk density
	g kg^{-1}	Mg ha^{-1}	g kg^{-1}	Mg ha$^-$)	–	Mg m^{-3}
Soil Depth 0–5 cm						
AF	6.28 (0.46)[bc]	4.57 (0.45)[bc]	0.68 (0.05)[bc]	0.49 (0.06)[bc]	9.26 (0.26)[b]	1.46 (0.11)[ab]
RF	2.96 (1.46)[c]	2.28 (1.15)[c]	0.31 (0.16)[c]	0.24 (0.13)[c]	9.63 (0.83)[b]	1.53 (0.05)[a]
OP	25.4 (9.25)[a]	16.1 (5.79)[a]	2.24 (0.92)[a]	1.42 (0.56)[a]	11.6 (1.13)[a]	1.28 (0.16)[b]
IR	6.66 (3.15)[bc]	5.20 (2.65)[bc]	0.53 (0.24)[c]	0.42 (0.20)[c]	12.5 (1.30)[a]	1.54 (0.06)[a]
SP	16.0 (7.39)[ab]	11.1 (4.86)[ab]	1.62 (0.76)[ab]	1.13 (0.50)[ab]	9.92 (0.44)[b]	1.42 (0.09)[ab]
Soil Depth 5–10 cm						
AF	6.56 (0.60)[ab]	5.01 (0.63)[ab]	0.69 (0.11)[ab]	0.52 (0.09)[ab]	9.69 (1.25)[b]	1.53 (0.12)
RF	3.00 (1.42)[b]	2.34 (1.21)[b]	0.32 (0.12)[b]	0.25 (0.11)[b]	9.26 (1.21)[b]	1.53 (0.07)
OP	12.6 (5.48)[a]	8.61 (4.11)[a]	1.15 (0.60)[a]	0.78 (0.43)[a]	11.7 (2.00)[ab]	1.35 (0.15)
IR	5.68 (2.68)[b]	4.37 (1.98)[ab]	0.43 (0.20)[b]	0.33 (0.15)[ab]	13.8 (3.19)[a]	1.55 (0.06)
SP	8.66 (5.15)[ab]	6.52 (3.70)[ab]	0.77 (0.50)[ab]	0.58 (0.36)[ab]	12.1 (2.36)[ab]	1.53 (0.09) NS
Soil Depth 10–20 cm						
AF	5.67 (0.44)[ab]	8.53 (1.14)	0.61 (0.07)	0.92 (0.14)	9.32 (0.64)[b]	1.50 (0.09)
RF	3.27 (1.52)[b]	5.18 (2.64)	0.35 (0.15)	0.55 (0.26)	9.31 (0.64)[b]	1.56 (0.08)
OP	10.4 (4.77)[a]	14.6 (6.96)	0.99 (0.56)	1.38 (0.77)	11.5 (2.66)[ab]	1.40 (0.11)
IR	5.18 (2.44)[ab]	7.90 (3.74)	0.39 (0.20)	0.60 (0.32)	13.5 (1.27)[a]	1.53 (0.06)
SP	7.63 (4.67)[ab] NS	11.1 (6.75) NS	0.65 (0.44) NS	0.95 (0.63)	12.6 (2.29)[a]	1.47 (0.07) NS
Soil Depth 20–30 cm						
AF	4.93 (0.43)	7.68 (0.56)	0.50 (0.05)	0.78 (0.09)	9.96 (0.94)	1.56 (0.03)
RF	4.00 (0.87)	6.26 (1.53)	0.37 (0.07)	0.58 (0.12)	10.7 (0.68)	1.56 (0.14)
OP	9.02 (3.69)	13.3 (5.53)	0.86 (0.43)	1.28 (0.67)	11.3 (2.63)	1.47 (0.08)
IR	4.43 (2.22)	6.93 (3.70)	0.35 (0.17)	0.55 (0.28)	12.5 (1.07)	1.54 (0.07)
SP	7.33 (4.86) NS	10.3 (6.68) NS	0.62 (0.41) NS	0.87 (0.56) NS	11.9 (0.92) NS	1.42 (0.05) NS

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation based fruit production; SP, *Faidherbia albida* based silvopasture.
± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, *P* = 0.05).

ρ_b = Dry bulk density (Mg m^{-3}).

T = Thickness of soil layer (m).

The SOC (or TN) stock in the 30 cm depth for each land use was calculated by summing SOC (or TN) stocks in the 0–5, 5–10, 10–20 and 20–30 cm depth intervals. Accumulation of SOC (or TN) stock in the same soil depth (30 cm) for each land use was estimated by calculating the difference in SOC (or TN) stock between each one of the four land uses (AF, OP, IR and SP) and the control (RF) land use. Because of the variable durations of the different land uses, the rate of accumulation of SOC (or TN) stock for 0–30 cm layer for each of the four land uses (AF, OP, IR and SP) was estimated by dividing accumulation values by the assumed duration of each land use (Puget and Lal, 2005). Based on the survey conducted with the farmers, an average duration of 50 years was taken as the age for OP, AF and SP land uses for adopting since early 1960s (Eweg et al., 1998). However, irrigation was started in 1995 (15 years).

2.3. Statistical analysis

Soil parameters under different land uses and depths were subjected to one-way ANOVA. Differences between means of treatments were considered significant at the 0.05 level using the Tukey's studentized (HSD) test. The data were analyzed using SAS version 9.2 software package (SAS, 2007).

3. Results

3.1. Soil organic carbon and total nitrogen concentrations and stocks in different land uses across depths

Soil Organic Carbon concentration in 0–5 cm depth differed significantly (*P* < 0.001) among land uses. The highest SOC concentration was measured in OP (25.4 g kg^{-1}) followed by that in SP (16.0 g kg^{-1}), IR (6.66 g kg^{-1}), AF (6.28 g kg^{-1}) and RF (2.96 g kg^{-1}) land uses. The lowest SOC concentration measured in RF (2.29 g kg^{-1})

was significantly lower (*P* < 0.001) than that in OP and SP treatments (Table 1). In 5–10 and 10–20 cm depths, SOC concentration followed the same trend except that the amount of SOC in OP and SP land use systems decreased by about 50% compared with that in the top 0–5 cm depth but remained the same with slight changes in other depths. In the two lower depths, SOC concentration in OP was significantly higher (*P* < 0.01 and *P* < 0.05, respectively) than that in RF land use (Table 1). Soil OC concentration in 20–30 cm depth did not differ among land uses. Similarly, the highest SOC stock in 0–5 cm depth was measured in OP land use system (16.1 Mg ha^{-1}) followed by that in SP (11.1 Mg ha^{-1}) and the lowest SOC stock (2.28 Mg ha^{-1}) was measured in RF (control) land use (Table 1). SOC stock in OP was significantly higher (*P* < 0.001) than that in AF, IR and RF land use systems. However, SOC stocks under IR and AF land use systems did not differ from that under RF in the same 0–5 cm layer (Table 1). SOC stock under OP was significantly higher (*P* < 0.05) than that under RF in 5–10 cm depth. However, SOC stocks under other land use systems (SP, AF and IR) did not differ from that in RF and among each other (Table 1). Moreover, no significant difference in SOC stock was measured across land uses in 10–20 and 20–30 cm layers.

Concentration of TN in 0–5 cm depth followed a trend similar to that of SOC. Accordingly, the highest TN concentration in 0–5 cm depth was measured in OP (2.24 g kg^{-1}) followed by that in SP (1.62 g kg^{-1}) and AF (0.68 g kg^{-1}) land uses (Table 1). In 0–5 cm depth, TN concentration in OP differed significantly (*P* < 0.001) from those in RF, IR and AF land uses; that in SP differed significantly (*P* < 0.001) from those in RF and IR land uses (Table 1). In 5–10 cm depth, concentration of TN in OP differed significantly (*P* < 0.05) from those in RF and IR land uses, but it was not significantly different from those in other land uses (Table 1). Concentration of TN did not differ among land uses in 10–20 and 20–30 cm depths. Total N stock among land uses and across depths also followed a trend similar to that of SOC stock (Table 1). In 0–5 cm depth, OP land use system had a significantly higher (*P* < 0.001) TN stock (1.42 Mg ha^{-1}) than those under AF, RF and IR land use systems (Table 1). TN stock in SP (1.13 Mg ha^{-1}) also differed significantly from those in IR and RF land use systems. Total N stock differed significantly (*P* < 0.05) only between OP (0.78 Mg ha^{-1}) and RF (0.25 Mg ha^{-1}) in the 5–10 cm depth. In 10–20 and 20–30 cm depths, TN stock did not differ among all land use systems (Table 1).

In the top 0–5 cm depth, C:N ratios were significantly higher (*P* < 0.001) in OP and IR than those in other land uses. In 5–10 cm depth, C:N ratios generally increased from those in 0–5 cm depth in most land uses except RF which showed slight decrease (Table 1). However, C:N ratio in 5–10 cm depth in IR was significantly higher (*P* < 0.05) than those in RF and AF land uses. A similar trend was observed in 10–20 cm depth, and C:N ratios in IR and SP were significantly higher (*P* < 0.01) than

Table 2
Comparison of soil organic carbon and total nitrogen concentrations and soil organic carbon and total nitrogen stocks among different depths within each land use.

Depth (cm)	Land use									
	AF		RF		OP		IR		SP	
	g kg^{-1}	Mg ha^{-1}	g kg^{-1}	Mg ha^{-1}	g kg^{-1}	Mg ha^{-1}	g kg^{-1}	Mg ha^{-1}	g kg^{-1}	Mg ha^{-1}
Soil Organic Carbon										
0–5	6.28 (0.46)[ab]	4.57 (0.45)[b]	2.96 (1.46)	2.28 (1.15)[b]	25.4 (9.25)[a]	16.1 (5.79)	6.66 (3.15)	5.20 (2.65)	16.0 (7.39)	11.1 (4.86)
5–10	6.56 (0.60)[a]	5.01 (0.63)[b]	3.00 (1.42)	2.34 (1.21)[b]	12.6 (5.48)[b]	8.61 (4.11)	5.68 (2.68)	4.37 (1.98)	8.66 (5.15)	6.52 (3.70)
10–20	5.67 (0.44)[bc]	8.53 (1.14)[a]	3.27 (1.52)	5.18 (2.64)[ab]	10.4 (4.77)[b]	14.6 (6.96)	5.18 (2.44)	7.90 (3.74)	7.63 (4.67)	11.1 (6.75)
20–30	4.93 (0.43)[c]	7.68 (0.56)[a]	4.00 (0.87)	6.26 (1.53)[a]	9.02 (3.69)[b]	13.3 (5.53)	4.43 (2.22)	6.93 (3.70)	7.33 (4.86)	10.3 (6.68)
			NS			NS	NS	NS	NS	NS
Total Nitrogen										
0–5	0.68 (0.05)[a]	0.49 (0.06)[c]	0.31 (0.16)	0.24 (0.13)[c]	2.24 (0.92)[a]	1.42 (0.56)	0.53 (0.24)	0.42 (0.20)	1.62 (0.76)[a]	1.13 (0.50)
5–10	0.69 (0.11)[a]	0.52 (0.09)[bc]	0.32 (0.12)	0.25 (0.11)[bc]	1.15 (0.60)[ab]	0.78 (0.43)	0.43 (0.20)	0.33 (0.15)	0.77 (0.50)[ab]	0.58 (0.36)
10–20	0.61 (0.07)[ab]	0.92 (0.14)[a]	0.35 (0.15)	0.55 (0.26)[ab]	0.99 (0.56)[b]	1.38 (0.77)	0.39 (0.20)	0.60 (0.32)	0.65 (0.44)[ab]	0.95 (0.63)
20–30	0.50 (0.05)[b]	0.78 (0.09)[a]	0.37 (0.07)	0.58 (0.12)[a]	0.86 (0.43)[b]	1.28 (0.67)	0.35 (0.17)	0.55 (0.28)	0.62 (0.41)[b]	0.87 (0.56)
			NS			NS	NS	NS	NS	NS

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation based fruit production; SP, *Faidherbia albida* based silvopasture.
± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, $P = 0.05$).

those in RF and AF land uses. In 20–30 cm depth, there was no significant difference in C:N ratio among land uses (Table 1). Further, there was no single clear trend in C:N ratio with depth across land use systems.

The lowest bulk density (ρ_b) value of 1.28 Mg m^{-3} in 0–5 cm soil depth was measured in OP land use system, and it was significantly lower ($P < 0.01$) than that of 1.54 Mg m^{-3} in IR land use and that of 1.53 Mg m^{-3} in RF land use (Table 1). Bulk density generally showed an increasing trend in most land uses. However, it did not differ among all land use systems in lower depths (Table 1).

3.2. Soil organic carbon and total nitrogen concentrations and stocks across depths in each land use system

Soil OC concentration (g kg^{-1}) decreased with depth in all land uses except in RF (control), which was uniformly low in all depths (Table 2). Soil OC concentration in 0–5 cm depth in OP land use system was significantly higher ($P < 0.01$) than those in the lower three depths. Soil OC concentration in AF land use system differed significantly among different depths ($P < 0.001$) with a higher SOC concentration in 5–10 cm depth than the lower 10–20 and 20–30 cm depths. No significant difference in SOC concentration was observed with depth in RF, IR and SP land uses (Table 2). On the other hand, SOC stock (Mg ha^{-1}) did not show any trend with depth among land use systems (Table 2). The trend in SOC stock (Mg ha^{-1}) differed from that of SOC concentration (g kg^{-1}), due to differences in ρ_b with depth. Depth had significant effects on SOC stocks only in AF and RF land uses ($P < 0.001$ and $P < 0.01$ for AF and RF, respectively). Soil OC stocks in the lower, 10–20 and 20–30 cm, layers were significantly higher than those in the top 0–5 and 5–10 cm layers (Table 2). Soil OC stock in RF land use also followed a similar trend. Therefore, SOC stock in 20–30 cm layer was significantly higher than those in 0–5 and 5–10 cm layers (Table 2). However, SOC stock did not differ with depth in OP, IR and SP land use systems (Table 2).

Depth distribution of TN concentration (g kg^{-1}) in each land use followed a trend similar to that of SOC concentration (Table 2). Thus, significantly higher ($P < 0.01$) TN concentrations were measured in 0–5 and 5–10 cm depths than that in 20–30 cm depth in AF land use. There was also a significantly higher ($P < 0.05$) TN concentration in 0–5 cm depth than those in the lower 10–20 and 20–30 cm depths in OP land use (Table 2). Concentration of TN also differed significantly among depths in SP land use system (Table 2) and, TN concentration in 0–5 cm depth was significantly higher ($P < 0.05$) than that in 20–30 cm depth. Similar to SOC stocks, TN stocks significantly differed with depth in AF and RF land use systems (Table 2). Total N stocks in 10–20 and 20–30 cm depths were significantly higher ($P < 0.001$) than those in 0–5 and 5–10 cm depths in AF land use. Total N stock in 20–30 cm depth was also significantly higher ($P < 0.01$) than those in 0–5 and 5–10 cm depths in RF land use (Table 2). Neither SOC nor TN stocks did differ with depth in soils under OP, IR and SP land use systems (Table 2).

3.3. Magnitude and rates of soil organic carbon and total nitrogen accumulations under different land uses

Total SOC stock in 0–30 cm depth under OP (52.6 Mg ha^{-1}) was significantly higher ($P < 0.01$) than that in RF (16.1 Mg ha^{-1}). However, it did not differ from other land uses (Table 3). Similarly, TN stock in the same depth under OP land use (4.87 Mg ha^{-1}) was significantly higher ($P < 0.05$) than those in RF (1.62 Mg ha^{-1}) and IR (1.90 Mg ha^{-1}) land uses. No significant difference in total TN stock was observed among other land uses (Table 3).

Among the four grass- and tree-based land use systems (OP, SP, AF and IR), OP accumulated the highest SOC stock (36.5 Mg ha^{-1}) followed by those in SP (23.0 Mg ha^{-1}) and AF (9.7 Mg ha^{-1}) land uses. The lowest accumulation of SOC stock

(8.3 Mg ha^{-1}) was measured in IR land use (Table 3). The amount of SOC stock accumulated in OP land use system was significantly higher ($P < 0.05$) than those in IR and AF land uses (Table 3). Similarly, the highest accumulation of TN stock was measured in OP (3.25 Mg ha^{-1}) followed by those in SP (1.91 Mg ha^{-1}) and AF (1.10 Mg ha^{-1}) land uses. The lowest TN stock accumulation (0.28 Mg ha^{-1}) was measured in IR land use (Table 3). However, the only statistically significant ($P < 0.05$) difference in TN stock accumulation was measured between OP and SP land uses (Table 3).

The highest rate of SOC stock accumulation was measured in OP land use (0.73 Mg C ha^{-1} yr^{-1}) followed by those in IR (0.56 Mg C ha^{-1} yr^{-1}) and SP (0.46 Mg C ha^{-1} yr^{-1}) land uses. The lowest rate was also measured in AF (0.19 Mg C ha^{-1} yr^{-1}) land use (Table 3). However, the effect of land use change on rate of SOC stock accumulation did not statistically differ among land uses (Table 3). Similarly, no significant difference was observed in the rate of TN stock accumulation among land uses. However, the highest rate of TN stock accumulation was also measured in OP land use (0.07 Mg N ha^{-1} yr^{-1}) followed by those in SP (0.04 Mg N ha^{-1} yr^{-1}) and AF (0.02 Mg N ha^{-1} yr^{-1}) land uses. The lowest rate of TN stock accumulation (0.02 Mg N ha^{-1} yr^{-1}) was measured in IR land use (Table 3).

4. Discussion

4.1. Soil organic carbon and total nitrogen concentrations in different land uses and depths

Concentrations of SOC and TN decreased with depth in OP, SP and IR land uses. It was in agreement with results by Trujillo et al. (1997) who observed a diminishing trend in SOC content with depth in their study in the Colombian Savannas. Haile et al. (2008) also observed a declining trend in SOC concentration with depth in silvopastural systems in Florida. Recently, Aticho (2013) in his study in Kafa, Southwest Ethiopia, also reported a negative correlation between SOC content and sampling depth. However, in this study soils under AF and RF land use systems showed different trends in SOC and TN concentrations across depths because of the mixing effects of tillage (Beare et al., 1997; Chen et al., 2009; Gelaw et al., 2013).

Distribution of SOC and TN concentrations with depth showed that high proportions of these in OP and SP land uses were concentrated in the top 0–5 cm depth, indicating the risks of large amounts of CO$_2$ release from the surface soil when these land uses are converted into an arable land use. A study conducted by Aticho (2013) in Southwestern Ethiopia showed a loss of 32.98–36.63% SOC content due to conversation of forest lands to continuous cultivation. The relative distribution between the surface and sub-surface soils obtained in this study was comparable with results for similar soils of tropical and subtropical regions elsewhere (Sombroek et al., 1993; Batjes, 1996; Fantaw et al., 2006).

The relatively higher SOC and TN concentrations under OP and SP than the other land use systems indicated that SOC and TN

Table 3
Magnitude and rate of soil organic carbon and total nitrogen stocks accumulation in different land uses in 0–30 cm depth.

Land Use	Assumed duration since conversion (Year)	SOC stock (Mg C ha^{-1})	SOC Accumulation (Stock-RF) (Mg C ha^{-1})	Rate of SOC accumulation (Mg C ha^{-1} yr^{-1})	TN Stock (Mg N ha^{-1})	TN Accumulation (Stock-RF) (Mg N ha^{-1})	Rate of TN accumulation (Mg N ha^{-1} yr^{-1})
RF	–	16.1 (6.5)[b]	–	–	1.62 (0.6)[b]	–	–
AF	50	25.8 (1.9)[ab]	9.7 (7.17)[b]	0.19 (0.14)	2.71 (0.3)[ab]	1.10 (0.64)[ab]	0.022 (0.013)
OP	50	52.6 (21.8)[a]	36.5 (17.0)[a]	0.73 (0.34)	4.87 (2.36)[a]	3.25 (1.96)[a]	0.065 (0.039)
IR	15	24.4 (11.5)[ab]	8.3 (8.73)[b]	0.56 (0.58)	1.90 (0.9)[b]	0.28 (0.55)[b]	0.019 (0.037)
SP	50	39.1 (21.5)[ab]	23.0 (19.7)[ab]	0.46 (0.39) NS	3.52 (2.01)[ab]	1.91 (1.86)[ab]	0.038 (0.037) NS

AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation based fruit production; SP, *Faidherbia albida* based silvopasture.
± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, $P = 0.05$).

concentrations in soils of the region can be increased by converting arable lands to managed grasslands and silvopastures or adopting no-till and reduced tillage practices (Girmay et al., 2008; Mekuria et al., 2009). Concentrations of SOC and TN in AF and IR land uses were also higher than those in the control or RF, indicating that adoption of agroforestry could enhance SOC and TN concentrations. This is in agreement with Albrecht and Kandji (2003) who indicated that with adequate management of trees in arable and grazing lands, a significant fraction of the atmospheric CO_2 could be captured and stored both in plant biomass and soils. Takimoto et al. (2008) in their study on carbon sequestration potentials of traditional and improved agroforestry systems in West African Sahel also found that a major portion of the total amount of C in the system is stored in the soil. Further, rainfall appears to be the major factor governing organic matter and nitrogen contents of East African soils (Birch and Friend, 1956). This was confirmed by the result that SOC and TN concentrations in IR land use were higher than those in RF. It was in agreement with Lueking and Schepers (1985) who reported an increase in dry matter production and increase in SOC and TN concentrations in Nebraska sand-hill soils due to adoption of irrigation. Further, in dry land areas, irrigation is one primary means for increasing plant production (Lal et al., 1999), which offers increased scope for CS. Field studies from Northern Ethiopia on in situ water harvesting systems also showed improvement in soil water content in the root zone (McHugh et al., 2007; Araya and Stroosnijder, 2010) and increase in barley yield by 44% (Araya and Stroosnijder, 2010). Therefore, there exists a large technical potential for SCS in the crop lands of the region by adopting irrigation (Lal et al., 1998; Conant et al., 2001).

4.2. Effects of land use changes on soil organic carbon and total nitrogen stocks accumulation

The increase in SOC and TN stocks both in the top 0–5 cm and 5–10 cm layers, and the total SOC and TN stocks in the 0–30 cm depth under the tree-and grass-based land use systems (OP, SP, and AF) (Table 3) compared with those in RF indicated the vast potentials of SCS in those land uses in the region. These results were in agreement with Lal (2002) who observed that, other factors remaining the same, grazing land soils have more SOC than cropland soils because of: (1) low soil disturbance due to lack of plowing, and (2) more root biomass and residue returned on the surface. The results were also in agreement with the findings of Girmay et al. (2008) who reviewed C sequestration potentials of Ethiopian soils with adoption of restorative measures to be in the range from 0.066 to 2.2 Tg C y^{-1} on rain fed croplands, and from 4.2 to 10.5 Tg C y^{-1} on range lands. Mekuria et al. (2009) also reported 36–50% increase in mean SOC stock through conversion of degraded grazing lands to exclosures, areas closed from human and animal interference to promote natural regeneration of plants on formerly degraded communal grazing lands, in Tigray, Northern Ethiopia. Similarly, Omonode and Vyn (2006) reported higher TN and SOC

stocks in grasslands than croplands in west-central Indiana, USA. Similar results were also reported by Mclauchlan et al. (2006), when agricultural lands of the northern Great Plains depleted in SOM by decades of cultivation were changed to grasslands through federal conservation programs. On the other hand, this result indicated the risks of large amounts of CO_2 release if these land uses are converted to croplands. In agreement with this assertion, Fantaw et al. (2006) in their study in southeastern Ethiopia indicated that on average 40–45% of SOC stock in 1 m depth of mineral soils was held in the top 30 cm, indicating the risks of large amounts of CO_2 release following deforestation and conversion into Agroecosystems. Further, Sombroek et al. (1993) reported that most soils of the tropics and sub-tropics have lost 20–50% of SOC following forest clearance and conversion to farmland.

4.3. Magnitude and Rates of SOC and TN stocks accumulation across land uses

Taking RF as a baseline, accumulations of SOC and TN stocks for the other land uses were found in the order: OP > SP > AF > IR due to differences in organic inputs. This result was in agreement with Zeleke et al. (2004) who observed 41-, 67- and 107% SOC contents following three years application of: (1) fertilizer alone (F), (2) maize residue alone (R), and (3) maize residue + fertilizer (RF) treatments, respectively, compared with a control on an Andosol in South Central Rift Valley of Ethiopia. Similarity in trends of accumulation of SOC and TN stocks were attributed to the fact that most N forms part of the SOM as soil carbon levels increase, microbes shift to immobilize N rather than mineralize it and therefore decrease net N mineralization rates (Mclauchlan et al., 2006). Ganuza and Almendros (2003) reported that SOC and TN reached the maximum average in soils under pasture in the Basque Country (Spain) with r between topsoil SOC and TN contents: 0.94; $P < 0.001$.

The average SOC accumulation rate in OP land use (0.73 Mg C ha^{-1} yr^{-1}) over 50 years was similar to 0.6–0.8 Mg C ha^{-1} yr^{-1} reported by Mensah et al. (2003) for a restored grassland in the top 0–15 cm in east central Saskatchewan, Canada, after 5–15 years of establishment. Girmay et al. (2008) using assumptions from Lal (2001) estimated rates of SCS potentials of currently degraded soils in Ethiopia under rangeland, irrigation, and rain fed cropping land uses over the next 50 years with widespread adoption of soil-specific restoration measures in the order: 0.3–0.5, 0.06–0.2, and 0.06–0.15 Mg C ha^{-1} yr^{-1}, respectively.

Compared with open (treeless) pasture systems, silvopastural agroforestry systems that integrate trees in to pasture production are likely to enhance SOC sequestration, especially in deeper soil layers (Haile et al., 2008). However, the results of the present study indicated that OP land use had higher SOC accumulation rates than the SP systems because: (1) trees are integrated into the pasture system in the study area only on the peripheral marginal parts where fertility of the soils was low, and (2) soil samples were taken

to 30 cm depth. These results are in agreement with the findings of Puget and Lal (2005) who observed that pasture soil had more than 1.5 times SOC stock than that in forest soils in 0–5 cm depth of a Mollisol in central Ohio, reflecting the larger grass root density in the top layer. In addition, the amount of accumulation of SOC stock in AF land use was greater than that in IR land use but the rate of accumulation of SOC stock was vice versa. These differences may be attributed to the differences in time of 50 years vs. 15 years for AF and IR land uses, respectively. Smith (2004) observed that soil C sinks increase most rapidly soon after a C-enhancing change in land management has been implemented, and the sink strength in soil becomes smaller as time goes on and SOC stock approaches a new equilibrium.

The lowest rate of accumulation of TN stock in IR land use may be attributed to N loss through leaching by irrigation water or may be because of poor TN content in the litter fall from the guava trees. Hengsdijk et al. (2005) reported that the use of crop residue did not specifically improve TN stock at farm level in Hawezin, Northern Ethiopia, because of poor TN content in the crop residue. Conversely, the improvement of the rate of TN accumulation in AF over that in IR land use may be ascribed to the N-fixing capacity of *F. albida* trees and incorporation of some of the fixed N in to the soil through litter fall and decomposition (Poschen, 1986; Kamara and Haque, 1992; Asfaw and Ågren, 2007; Hadgu et al., 2009).

5. Conclusion

SOC and TN concentrations generally decreased with increase in depth in OP, SP, and IR land uses. However, soils in AF and RF land use systems showed different patterns in SOC and TN concentrations with depth because of the mixing effects of tillage. Most of the SOC and TN concentrations in OP and SP land uses were concentrated in the top 0–5 cm depth, indicating the risks of large amounts of CO_2 to be released from the surface soil if these systems are converted into arable lands. Soils in AF and IR land uses also had higher SOC and TN concentrations than those in RF system, indicating a large potential of adopting these practices to sequester SOC and TN in these soils. Furthermore, substantial improvements in total SOC and TN stocks both in the 0–30 cm depth and in each depth in grass and tree based land use systems than those in the continuous sole cropping system (RF) revealed the opportunities of adopting these systems and at the same time the risks of large amounts of CO_2 emissions from the soils if these systems are converted into RF. Generally, the result of this study supported the hypothesis that land use change from dry land crop cultivation to tree and grass based systems improved SOC and TN concentrations and stocks of soils. Therefore, adoption of restorative land uses, conversion of croplands to grasslands or integration of appropriate agroforestry trees in agricultural lands, has large technical potential for SOC and TN sequestrations in the region.

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Paper II

Gelaw AM, Singh B R, Lal R. 2013. Organic carbon and nitrogen associated with soil aggregates and particle sizes under different land uses in Tigray, Northern Ethiopia. **Land Degradation & Development. DOI: 10.1002/ldr.2261.**

ORGANIC CARBON AND NITROGEN ASSOCIATED WITH SOIL AGGREGATES AND PARTICLE SIZES UNDER DIFFERENT LAND USES IN TIGRAY, NORTHERN ETHIOPIA

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ABSTRACT

Soil aggregation serves as storage and cycling of soil organic carbon (SOC) and total nitrogen (TN), which are essential for the functioning of terrestrial ecosystems. Thus, aggregate and particle associated SOC and TN in 0- to 10-cm and 10- to 20-cm layers were estimated for five land uses: rainfed cultivation (RF), agroforestry (AF), open pasture (OP), silvopasture (SP), and irrigation (IR) each with five replications within a watershed in Ethiopia. OP had the highest $WSA > 2$ mm (88·7%) and SOC associated with macroaggregates (20·0 g kg^{-1}), which were significantly higher ($P < 0·0001$; $P = 0·0017$ for WSA and SOC, respectively) than that in other land uses in 0- to 10-cm depth. SOC associated with both macroaggregates and microaggregates decreased with depth. Macroaggregates contained higher SOC than microaggregates in both layers under all land uses. AF had the highest SOC associated with microaggregates (2·6 g kg^{-1}) followed by that in SP (2·3 g kg^{-1}), indicating its potential to stabilize SOC more than other land uses. TN associated with macroaggregates followed a trend similar to that of SOC. OP had also significantly higher SOC ($P = 0·0001$) and TN ($P = 0·0004$) associated with sand particles than RF, AF, and IR. Sand-associated SOC and TN were the highest in uncultivated systems. Moreover, the higher SOC concentrations associated with clay particles in soils under OP, SP, and AF showed that grass-based and tree-based systems are rich in stable SOC as clay-associated SOC has higher residence time than that associated with sand or silt fractions. Copyright © 2013 John Wiley & Sons, Ltd.

INTRODUCTION

Aggregates are secondary particles formed through the combination of mineral particles with organic and inorganic substances (Bronick & Lal, 2005). Soil aggregation and aggregate stability are influenced by land use, management, lithology, and local climate (Cerdà, 1996; 2000). Aggregates occur in a variety of manners and sizes. These are often grouped by size: macroaggregates ($>0·25$ mm) and microaggregates ($<0·25$ mm) with these groups being further divided by size (Tisdall & Oades, 1982). Different size groups differ in properties such as binding agents and carbon and nitrogen distribution. Soil organic carbon (SOC) and total nitrogen (TN) retention in the soil can be characterized by short-term storage in macroaggregates or by long-term sequestration in microaggregates (Carter, 1996; Sainju, 2006; Haile *et al.*, 2008; Sainju *et al.*, 2009).

Land use and soil management affect soil aggregate size distribution and stability, which are important indicators of soil physical quality (Boix-Fayos *et al.*, 2001; Herrick *et al.*, 2001; Barthés & Roose, 2002). Clay content and SOC associated with aggregates are principal determinants of water stable aggregation (Boix-Fayos *et al.*, 2001). Important soil physical properties such as bulk density (ρ_b) and water stable aggregation (WSA) are affected by soil organic matter (SOM) and texture (Young, 1988). Conversion from forest to other land uses results in increase in soil ρ_b and decrease in hydraulic conductivity (Ks), thereby increase in soil susceptibility to erosion (Spaans *et al.*, 1989; Lal, 2003). SOC concentration and soil physical properties such as soil ρ_b and aggregate stability are also affected by farming practices (Hao *et al.*, 2001). Tillage disrupts soil aggregates, compact the subsoil and disturb plant and animal communities resulting in a decrease in soil organic matter, and microbial and faunal activities (Plante & McGill, 2002; Bronick & Lal, 2005). On the other hand, adoption of no-till enhances SOC sequestration and improves soil structure (Lal & Kimble, 1997).

The balance between C in residues returned to soil and that released in to the atmosphere controls soil organic carbon sequestration in crop lands (Blanco-Canqui & Lal, 2004). Soil aggregation and SOC pool are increased by cropping and tillage systems that promote return of residues to soil (Gale & Cambardella, 2000). Land use also has significant effect on aggregate size distribution (Saha *et al.*, 2011). The presence of higher proportions of macroaggregates in forest than in cultivated soils indicates the effect of tillage on macroaggregate turnover. Increased return of residues and reduced disturbance of soils under pasture also results in better soil aggregation and sequestration of more carbon than in intensively tilled crop lands (Percival *et al.*, 2000). Generally, decrease in cultivation increases in soil

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structural stability and SOC content (Six *et al.*, 2000; Eynard *et al.*, 2004). Furthermore, irrigation is important to enhancing agricultural production and increasing biomass return in to soils in arid and semi-arid regions (Lal, 2004). Increased SOM content in irrigated soils can increase aggregation. Under long-term management, there are more water stable aggregates in irrigated than in dry land soils (Blanco-Canqui & Lal, 2004).

The arrangement of soil aggregates and the pore spaces among them determines soil structure (Golchin *et al.*, 1997; Christensen, 2001; Lal & Shukla, 2004). Soil structure and SOM are two of the most dynamic properties that are sensitive to land use and soil management (Blanco-Canqui & Lal, 2004). Several authors have reported that soil susceptibility to runoff and water erosion is linked to aggregate stability (Reichert & Norton, 1994; Amezketa *et al.*, 1996; Barthés & Roose, 2002; Shrestha *et al.*, 2007). Evaluation of soil susceptibility to runoff and water erosion in the field is often expensive or time consuming (Barthés & Roose, 2002). Thus, laboratory determination of WSA on small soil samples, which is less expensive and time consuming, has received much attention. Surface flows of water and raindrop impact are primary sources of energy causing disruption of soil aggregates in the field, and thus the attendant increase in erosion. However, aggregate-associated C provides strength and stability and counters the impact of destructive forces. Soil organic carbon associated with macroaggregates is usually lost more rapidly than those associated with microaggregates due to a lower protective effect of biophysical and chemical processes (Jastrow & Miller, 1998).

Soil texture moderates several important processes, including SOM dynamics and C sequestration, and is a very important determinant of soil quality (Kettler *et al.*, 2001). Thus, the degree of association of SOC with particle sizes is most frequently reported as an indicator of the impact of land use and soil management on soil quality (Saha *et al.*, 2011). Physical fractionation of soil according to the size and density of particles is achieved by applying various degrees of dispersion to break bonds between the elements of soil structure. It allows the separation of uncomplexed SOM and of variously sized organomineral complexes and can provide information on the importance of interactions among organic and inorganic components and the turnover of SOM (Christensen, 2001). SOC associated with the coarse fraction is commonly less decomposed material and has a higher C:N ratio than that associated with the fine fraction (Christensen, 2001; Shrestha *et al.*, 2007). It is more labile and therefore, the first to be affected by changes in land use and soil management (Ashagrie *et al.*, 2005). In contrast, SOC associated with clay has a lower C:N ratio than that associated with the coarse fraction due to its aliphatic and humified nature (Christensen, 2001; Ashagrie *et al.*, 2005; Shrestha *et al.*, 2007). It is more stable and is altered more by physical and chemical processes than by land use changes (Khanna *et al.*, 2001).

In Ethiopia, agricultural clearing and overgrazing mainly due to increasing population density have resulted in soil and environmental degradation (EFAP, 1994; Ashagrie

et al., 2005). According to a survey with older farmers in the area, large parts of Tigray, Northern Ethiopia were once covered with acacia woodland (*Acacia etbaica* (Sch.) and *Acacia albida* now renamed as *Faiderbhia albida* (Del.) A Chev.) (Eweg *et al.*, 1998; IAO, 2009 unpublished). The early 1960s marked the disappearance of much woodland under pressure from the rapidly growing population (Eweg *et al.*, 1998). Nowadays, in the region, farmers take care of naturally growing *Faiderbhia albida* trees in and around their farms and grazing lands in order to improve soil fertility and increase crop and pasture yields (Hadgu *et al.*, 2009).

In arid regions, irrigation is essential to crop production, because it increases soil available water capacity (Zhao-Zhong *et al.*, 2005). The potentially irrigable land area in sub-Saharan Africa is estimated at 39 Mha (Hillel, 1997). Most irrigation is in areas with low levels of SOC in natural soils. Irrigation in a drought prone soil can enhance biomass production, increase the amount of above ground and root biomass returned to the soil and improve SOC concentration (Lal, 2004). Therefore, there is a large potential for C sequestration by the use of irrigation (Lal *et al.*, 1998; Conant *et al.*, 2001). For enhancing agriculture development, rainwater harvesting is also important in Ethiopia, and it has been widely adopted since 2000 as a protection against drought and to promote production of high value cash crops by small land holders (Ibrahim, 2004 unpublished).

Therefore, the overall goals of this study were to assess the effects of land use on soil aggregate formation and to quantify association of SOC and TN with size fractions of aggregate and primary particles. Principal objectives of this study were to determine the magnitude and stability of SOC and TN associated with aggregates and primary particles. The study was conducted to test the hypothesis that land use change from dry land rainfed cultivation (RF) to irrigation (IR), agroforestry (AF), open pasture (OP), and silvopasture (SP) land use systems improves soil structure and enhances SOC and TN pools associated with soil aggregates and primary particles.

MATERIALS AND METHODS

Mandae watershed is located in Tigray Regional State, northern Ethiopia. Geographically, the study site is located between 15°26'N to 15°32'N latitude and 55°00'E to 55°60'E longitude, having an area of about 10 km^2, and an elevation of 1960 to 2000 m.a.s.l. The average daily air temperature of the area ranges between 15°C and 30°C in winter and summer, respectively. The mean annual rainfall of the area is 558 mm, with a large inter-annual variation. Soils in the watershed are classified as Arenosols and association of Arenosols with Regosols according to the World Reference Base for soil resources (WRB 2006). These soils are developed from alluvial deposits and Adigrat sandstones. Textures of these soils are dominated by sand, loamy sand and sandy loam fractions, and pH ranged from 6·8 to 7·9 (IAO 2009 unpublished). Major land uses of the watershed include *Faidherbia albida* based agroforestry

(27·7 ha), rainfed crop production (11·9 ha), open pasture (23·2 ha), irrigation-based guava fruit production (11·3 ha), and *Faidherbia albida* based silvopasture (11·7 ha). Agricultural rotation is usually maize-teff-beans-millet in the agroforestry and rainfed cultivation land use systems. Fallow is not practiced in the area due to population growth and scarcity of farmlands. Use of chemical fertilizers is minimal and land is prepared for cultivation by using a wooden plow with oxen. Crop residues and manures are used for animal feed and fuel, respectively. No pesticides and other agricultural chemicals are used in the area. Irrigation from shallow wells started in the area in late 1990s and most of the irrigated areas are covered by guava fruits.

Faidherbia albida (Del.) A Chev. trees in and around farm lands and grazing lands are remnants from the original woodland in the region. These trees are selectively left by farmers to improve soil fertility and increase crop and pasture yields (Hadgu *et al.* 2009). Grazing lands/open pastures and silvopastures are commonly owned and managed by communities. Trees are integrated into the pasture systems, silvopastures, on the peripheral marginal parts where fertility of the soils is low based on performance of grazing lands as perceived by farmers (personal communication with farmers). No external inputs such as fertilizer are used and no rotational grazing or other kinds of management are practiced in both the open pasture and silvopasture systems. However, grazing pressure will be higher in these systems only during cropping seasons, June to October; otherwise animals graze freely all over the different land use systems except the irrigation land use system in the area. Mixed crop-livestock, smallholder farming is a typical farming system of the region. On the basis of a survey conducted with farmers in the area, an average duration of 50 years was taken as the age for OP, AF, and SP land uses for adopting since early 1960s (Eweg *et al.* 1998). However, irrigation was started in 1995 (15 years).

A total of 100 soil samples (50 aggregate and 50 composite soil samples) were obtained in May 2010 from the surface (0–10 cm) and subsurface (10–20 cm) layers of five sites randomly chosen at different locations for each land use system. The summit position of the watershed was excluded to minimize the confounding effects of slope and soil erosion. The samples for aggregate analysis were taken by using a technique similar to that described by Srzednicki & Keller (1984). Soil profiles of 1 m by 1·5 m were excavated, and soils were sampled using a small gardening spade, transported to Mekelle University Laboratory, Ethiopia, and air-dried. Soil samples for bulk density (ρ_b) measurements for each depth were also obtained from the same profiles using 100-cm³ volume stainless steel tubes (5-cm diameter and 5·1-cm height). Composite soil samples were collected randomly from eight points at each sampling site and depth using auger samplers and mixed to analyze other soil parameters. Thus, 40 point samples were represented in computing the average values of each soil parameter.

Soil ρ_b was determined by the core method (Blake & Hartge, 1986). The SOC and TN concentrations for composite samples were determined at the Carbon Sequestration and Management Center Laboratory (The Ohio State University, USA) using auto CN analyzer (Vario Max CN Macro Elemental Analyser, Elementar Analysensysteme GmbH, Hanau, Germany) by dry combustion method (Nelson & Sommers, 1996).

The size distribution of aggregates was measured by the wet sieving method (Yoder, 1936). Bulk soil samples were passed through 8-mm sieves and retained on 4·75-mm sieves to collect soil aggregates between 4·75 and 8-mm size. Fifty grams of aggregates retained on the 4·75-mm sieve size were placed on the top sieve of a nest of sieves of 4·75-, 2·0-, 1·0-, 0·5-, and 0·25-mm openings and gradually pre-wetted under tension for 30 min. Then mechanical sieving using the Yoder apparatus was carried out for 30 min. Aggregates retained in each sieve were transferred to beakers. The weight of each aggregate fraction was recorded after oven-drying at 40°C for 72 h. WSA (Kemper & Rosenau, 1986), the geometric mean diameter and the mean weight diameter (Castro Filho *et al.*, 2002; Loss *et al.*, 2011) were calculated according to the following formulae:

$$\%WSA = \left(\frac{mi}{mj}\right)*100 \qquad (1)$$

Where: *mi* is the mass of aggregates retained in a specific size class of average diameter (g) and *mj* is total mass of aggregates (g)

$$MWD = \sum_{j=1}^{n} xjmj \qquad (2)$$

Where: $j = $ 1-to-n and *n* is the number of aggregate ranges, mj is the proportion of each size class to the total sample and xj is mean diameter of the size classes (mm)

$$GMD = \exp\left(\frac{\sum_{i=1}^{n} milnxi}{\sum_{i=1}^{n} mi}\right) \qquad (3)$$

Where: *n* is the number of aggregate ranges, *mj* is the weight of the aggregates in each size class(g), and *lnxi* is the natural logarithm of the mean diameter of the size classes (mm).

Both SOC and TN associated with different aggregate size fractions were also determined by the dry combustion method (Nelson & Sommers, 1996).

Composite samples were prepared for surface and subsurface layers pooling equal amounts (by weight) of air-dry, sieved (2-mm mesh) bulk soil samples for each depth and land use system. Size fractionation was conducted using a method described by Cambardella & Elliott (1992, 1993). Briefly, 20 g of soil, on a dry weight basis, was dispersed with 60 ml of 0·5 g l⁻¹ sodium hexametaphosphate in an end-to-end shaker for 16 h. After 16 h of shaking, the soil was wet-sieved through a 53 µm sieve. The SOC concentration in the >53 µm fraction is the one associated with sand and is referred to as particulate organic matter (Cambardella & Elliott, 1992; Barrios *et al.*, 1996). The soil fraction, which passed through 53 µm sieve, was separated into silt and clay fractions by repeated centrifugation (400 rpm for 30 min).

Finally, all the sand, silt, and clay particles obtained were dried at 40°C for 72 h and ground with a mortar and pestle for analysis of SOC and TN concentrations using the dry combustion method (Nelson & Sommers, 1996).

Aggregates were categorized as macro (>0·25 mm) and micro (<0·25 mm), and their relative proportions in the sample were expressed as a percentage of the total sample. The SOC (or TN) concentrations associated with particle size and aggregate size classes were calculated by multiplying the SOC (or TN) concentrations (g kg^{-1} of soil in fraction size) by the percent weight of the fraction. Concentration of SOC and TN associated with the aggregates and particle size fractions were computed as g kg^{-1} of fraction, and the data were analyzed using SAS version 9·2 software package (SAS institute Inc., Cary, North Carolina, USA) (SAS, 2007). Soil parameters under different land uses were subjected to one-way ANOVA. Correlation analysis was used to evaluate the relationships among SOC, TN, and other soil physical and chemical parameters. The respective correlation coefficients for each land use system were calculated from the average of the whole soils for all depths and land uses. Treatment differences were compared for significance at 0·05 level of probability.

RESULTS

There was no clear trend in soil bulk density (ρ_b) with regard to depth. Soil ρ_b had an increasing trend with depth in some land use systems (i.e., RF and OP), but a decreasing trend with depth in other land uses (Table I). In 0- to 10-cm depth, the least ρ_b (1·32 Mg m^{-3}) measured under OP land use system was significantly lower ($P = 0·0081$) than that in other land use systems. No significant difference in ρ_b was observed among other land use systems (Table I). Soil ρ_b in 10- to 20-cm depth did not differ among land use systems.

Both mean weight diameter and geometric mean diameter followed the same trend within and across depth. Both

indices were significantly higher ($P < 0·0001$) in OP than in other land uses in 0- to 10-cm depth, but no such a difference was observed in 10- to 20-cm depth.

The highest WSA of 88·7% in 0- to 10-cm depth was measured in OP, and it was significantly different ($P < 0·0001$) from that in other land use systems. The second highest WSA of 69·6% was measured in SP followed by AF and IR (Table I) and the least being in RF (Table I). In 10- to 20-cm depth, the highest WSA of 69·1% was measured in SP followed by that in OP (63·6%), IR (59·5%), AF (57·3%), and RF (44·2%), without any significant difference among land use systems (Table I). In general, WSA increased with depth with minor exceptions (Table I).

Open pasture land use system had the highest fraction (80·7%) of 2- to 4·75-mm size, and it was significantly higher ($P < 0·0001$) than that in other land uses in 0- to 10-cm depth. The second and third highest fractions in this size category were measured in SP and IR land use systems with 34·8% and 13·9%, respectively. In 10- to 20-cm depth, no significant difference was observed among land use systems (Table II). On the other hand, OP had the smallest fraction (2·4%) of 1- to 2-mm size fraction, which was significantly lower ($P = 0·02$) than that in AF (7·6%) in 0- to 10-cm depth. However, no significant difference was observed among other four land use systems, either in 0- to 10-cm or 10- to 20-cm depth (Table II). AF land use showed the highest percentage (18·8%) in 0·5- to 1-mm size fraction, which was significantly higher ($P = 0·0056$) than that in OP (1·8%). In the 0·25- to 0·5-mm size fraction, IR land use system showed the highest proportion (25·5%) followed by AF (23·7%) in 0- to 10-cm depth, which was significantly higher than that in OP ($P = 0·01$) (Table II). In 10- to 20-cm depth, OP showed significantly lower ($P = 0·0027$) proportion of aggregates in this size category than AF, SP, and IR land use systems. Whereas the highest proportion of micro-aggregates in size fractions <0·25 mm (microaggregates)

Table I. Land use effects on soil chemical and physical properties

Land use	Water stable aggregates (%)	Soil organic carbon (g kg^{-1})	Total nitrogen (g kg^{-1})	Bulk density (Mg m^{-3})	Geometric mean diameter (mm)	Mean weight diameter (mm)
	Soil depth 0–10 cm					
AF	56·8(10·8)^{bc}	6·2(0·9)^{b}	0·70(0·11)^{b}	1·53(0·07)^{a}	0·38(0·09)^{b}	0·68(0·15)^{b}
RF	37·5(10·4)^{c}	3·1(1·5)^{b}	0·33(0·12)^{b}	1·44(0·07)^{ab}	0·26(0·06)^{b}	0·51(0·12)^{b}
OP	88·7(6·4)^{a}	18·8(7·8)^{a}	1·65(0·78)^{a}	1·32(0·11)^{b}	3·80(1·86)^{a}	4·68(1·17)^{a}
IR	52·3(9·9)^{bc}	6·3(3·1)^{b}	0·50(0·23)^{b}	1·54(0·06)^{a}	0·42(0·20)^{b}	0·97(0·76)^{b}
SP	69·3(12·5)^{b}	11·9(6·7)^{ab}	1·12(0·67)^{ab}	1·50(0·13)^{a}	0·96(0·77)^{b}	1·93(1·21)^{b}
	Soil depth 10–20 cm					
AF	57·3(11·0)	5·7(0·4)^{ab}	0·61(0·07)^{ab}	1·50(0·09)	0·36(0·09)	0·63(0·15)
RF	44·2(6·9)	3·3(1·5)^{b}	0·35(0·15)^{b}	1·56(0·08)	0·31(0·04)	0·65(0·06)
OP	63·6(32·4)	10·4(4·8)^{a}	0·99(0·56)^{a}	1·40(0·11)	2·14(2·36)	2·80(2·32)
IR	59·5(18·1)	5·2(2·4)^{ab}	0·39(0·20)^{ab}	1·53(0·06)	0·72(0·59)	1·43(1·29)
SP	69·1(14·2)	7·6(4·7)^{ab}	0·65(0·44)^{ab}	1·47(0·07)	0·67(0·38)	1·18(0·68)
	NS			NS	NS	NS

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation-based fruit production; SP, *Faidherbia albida* based silvopasture·
±Mean values followed by standard errors in the parentheses; values with different letters are significantly different· NS = not significant (Tukey's test, $P = 0.05$)·

Table II. Aggregate size distribution under different land uses

Land use	Aggregate sizes				
	2·00–4·75 mm	1·00–2·00 mm	0·50–1·00 mm	0·25–0·50 mm	<0·25 mm
Soil depth 0–10 cm					
AF	8·2(4·8)[c]	7·6(3·0)[a]	18·8(8·6)[a]	23·7(5·4)[a]	39·7(10·24)[ab]
RF	6·1(2·5)[c]	5·9(2·0)[ab]	10·6(4·5)[ab]	16·7(4·5)[ab]	58·5(9·97)[a]
OP	80·7(15·2)[a]	2·4(2·6)[b]	1·8(2·2)[b]	3·8(4·7)[b]	8·3(7·5)[c]
IR	13·9(15·6)[bc]	4·2(1·9)[ab]	9·1(3·4)[ab]	25·5(12·5)[a]	45·0(10·0)[ab]
SP	34·8(21·7)[b]	6·1(1·8)[ab]	10·1(8·3)[ab]	20·1(13·8)[ab]	28·6(12·2)[b]
Soil depth 10–20 cm					
AF	6·7(3·7)	6·7(2·4)	15·5(7·8)	29·3(9·5)[a]	39·4(10·7)
RF	8·1(2·0)	6·9(1·9)	11·1(2·6)	19·1(3·3)[ab]	52·0(6·8)
OP	48·0(38·6)	4·4(2·8)	4·3(3·4)	7·2(6·8)[b]	32·9(33·1)
IR	21·7(26·0)	4·2(1·6)	11·4(12·9)	22·8(8·9)[a]	36·5(19·5)
SP	21·0(17·4)	6·6(4·3)	18·3(10·6)	24·4(8·2)[a]	28·7(14·9)
	NS	NS	NS		NS

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation-based fruit production; SP, *Faidherbia albida* based silvopasture.
±Mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, $P = 0.05$).

was observed in the control (RF) followed by that in IR and AF land uses, clearly reflecting the effects of tillage induced disturbance on aggregate formation and stability. In 0- to 10-cm depth, RF land use had significantly higher ($P < 0.0001$) proportion of microaggregates (58·5%) than that in SP (28·6%) and OP (8·3 %). However, no significant difference in proportion of microaggregates was observed among land uses in 10- to 20-cm depth (Table II).

The highest SOC concentration (18·8 g kg^{-1}) in 0- to 10-cm depth was measured in OP land use system, and it was significantly higher ($P = 0.0004$) than that in other land use systems except SP (Table I). In 10- to 20-cm depth, SOC concentration was significantly higher in OP than in the control (RF). SOC concentration did not differ significantly among other land uses (Table I).

Concentration of TN both in surface and subsurface depths followed a trend similar to that of SOC (Table I). In both depths, concentration of TN in OP differed significantly ($P = 0.002$) from that in the control (RF).

Concentrations of SOC and TN associated with macroaggregates (>0·25 mm) and microaggregates (<0·25 mm) and for three primary particles are shown in Table III. OP land use showed the highest SOC concentration associated with macroaggregates (20·0 g kg^{-1}), and it was significantly different ($P = 0.0017$) from that in other land uses in 0- to 10-cm depth. In this layer, SP and AF land uses contained the second and the third highest SOC concentrations in macroaggregates with 8·0 and 7·2 g kg^{-1}, respectively (Table III), but there was no significant difference between them and among other land uses. SOC associated with macroaggregates in the 10- to 20-cm depth followed the same trend as that in 0- to 10-cm depth.

AF land use had the highest SOC associated with microaggregates followed by that in SP and IR in 0- to 10-cm depth, but they were not significantly different (Table III). Generally, the same trend of SOC association with microaggregates was observed in 10- to 20-cm depth (Table III).

In 0- to 10-cm depth, SOC associated with sand particles differed significantly ($P = 0.0001$) among land uses. The highest SOC associated with sand particles was measured in OP land use (8·5 g kg^{-1}) followed by that in SP (6·8 g kg^{-1}), and the least in RF (control) (0·8 g kg^{-1}), and it was significantly different in OP than in other land uses. In 10- to 20-cm layer, OP land use also contained significantly higher ($P < 0.0001$) SOC associated with sand particles (4·1 g kg^{-1}) than in other land uses (Table III).

The highest SOC associated with silt and clay particles was also observed in OP in both soil depths. SOC associated with silt was significantly higher in OP than that in other land use systems except in SP in 10- to 20-cm layer (Table III). SOC associated with clay was also significantly higher in OP than AF, RF, and IR land uses in 0- to 10-cm depth but only significantly higher than that in RF and IR land uses in 10- to 20-cm layer. Furthermore, SOC associated with primary particles in OP was in the order sand silt > clay in both depths (Table III).

Concentration of TN associated with macroaggregates followed a trend similar to that of SOC (Table IV). Accordingly, OP contained the highest TN associated with macroaggregates in both depths. In 0- to 10-cm depth, TN associated with macroaggregates in OP land use was significantly higher ($P = 0.0035$) than RF, IR, and AF land uses. However, TN associated with macroaggregates in 10- to 20-cm depth was not statistically significant among all land uses (Table IV). TN associated with microaggregates in AF was significantly higher ($P = 0.042$) than that in OP land use system only in 0- to 10-cm depth (Table IV).

In 0- to 10-cm depth, the highest TN concentration associated with sand particles was measured in OP followed by that in SP, and it was significantly higher ($P = 0.0004$) than that in AF, RF, and IR (Table IV). In 10- to 20-cm depth, OP again contained the highest TN associated with sand particles followed by that in IR (Table IV).

Table III. Soil Organic Carbon (g kg^{-1}) associated with aggregates and primary particles

Land use	Macro	Micro	Sand	Silt	Clay
	Soil depth 0–10 cm				
AF	7·2(5·1)[b]	2·6(0·4)	2·8(1·0)[bc]	0·39(0·05)[b]	1·7(0·3)[b]
RF	2·1(0·9)[b]	1·9(1·0)	0·80(0·2)[c]	0·37(0·25)[b]	1·3(0·8)[b]
OP	20·0(11·2)[a]	1·3(0·2)	8·5(2·9)[a]	5·9(3·0)[a]	3·7(0·8)[a]
IR	6·6(3·1)[b]	2·3(1·1)	2·2(1·3)[c]	1·1(1·1)[b]	1·8(0·7)[b]
SP	8·0(3·3)[b]	2·3(0·3) NS	6·8(3·9)[ab]	2·2(2·1)[b]	2·6(1·0)[ab]
	Soil depth 10–20 cm				
AF	4·8(3·0)	2·6(1·5)	1·4(0·4)[b]	0·32(0·16)[b]	2·1(0·5)[ab]
RF	2·3(0·6)	2·1(0·8)	0·80(0·1)[b]	0·36(0·29)[b]	1·5(0·5)[b]
OP	10·6(9·5)	1·5(0·4)	4·1(1·8)[a]	4·0(2·90)[a]	3·3(0·8)[a]
IR	3·6(2·0)	1·3(0·5)	1·2(0·3)[b]	0·89(0·77)[b]	1·7(0·6)[b]
SP	5·4(3·4) NS	1·5(0·5) NS	1·5(0·6)[b]	1·2(1·5)[ab]	2·1(1·1)[ab]

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation-based fruit production; SP, *Faidherbia albida* based silvopasture.
±Mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, $P = 0.05$).

Total N associated with silt was lower than that associated with sand particles in all land uses in both depths except in OP in 10- to 20-cm depth where silt-associated TN was slightly higher than that associated with sand (Table IV). OP land use showed significantly higher TN associated with silt particles ($P = 0.0005$ and $P = 0.0065$ in 0- to 10-cm and 10- to 20-cm layers, respectively) than in other land uses.

In 0- to 10-cm depth, both OP and SP showed significantly higher ($P = 0.0003$) TN associated with clay particles than that in RF (control), but in 10- to 20-cm depth, only OP showed a significantly higher TN associated with clay than that in RF and IR land uses (Table IV).

C:N ratio followed a decreasing trend from macroaggregates to microaggregates, and from sand to silt and clay particles (Table V). In 0- to 10-cm depth, IR had the highest C:N ratio (16·3) associated with macroaggregates, and it was significantly higher ($P = 0.02$) than under AF and SP (Table V). However, no significant difference was observed

in C:N ratio associated with macroaggregates in all land uses in 10- to 20-cm depth. Similarly, C:N ratio associated with microaggregates, silt and clay particles in both layers, did not differ significantly (Table V). OP land use contained the highest C:N ratio associated with sand particles (13·8), and it was significantly higher ($P = 0.012$) than that in RF in 0- to 10-cm depth (Table V).

Correlation analysis showed that WSA was the most important parameter associated with SOC ($R^2 = 0.85$) and TN ($R^2 = 0.79$) concentrations for the land uses studied (Figure 1a and 1b). The highest correlation ($R^2 = 0.97$) was obtained between SOC and TN (Figure 1c).

DISCUSSION

Soil bulk density (ρ_b) decreased with depth. It was in agreement with results by other studies (Hajabbasi *et al.*, 1997; Sahani & Behera, 2001; Shrestha *et al.*, 2007) where they

Table IV. Total nitrogen (g kg^{-1}) associated with aggregates and primary particles

Land use	Macro	Micro	Sand	Silt	Clay
	Soil depth 0–10 cm				
AF	0·64(0·47)[b]	0·26(0·04)[a]	0·23(0·10)[bc]	0·04(0·01)[b]	0·22(0·04)[bc]
RF	0·16(0·08)[b]	0·18(0·08)[ab]	0·07(0·02)[c]	0·04(0·03)[b]	0·15(0·08)[c]
OP	1·7(1·0)[a]	0·11(0·03)[b]	0·62(0·22)[a]	0·55(0·30)[a]	0·48(0·12)[a]
IR	0·46(0·30)[b]	0·20(0·12)[ab]	0·18(0·09)[c]	0·10(0·08)[b]	0·20(0·09)[bc]
SP	0·73(0·36)[ab]	0·22(0·04)[ab]	0·58(0·34)[ab]	0·21(0·21)[b]	0·35(0·15)[ab]
	Soil depth 10–20 cm				
AF	0·45(0·31)	0·23(0·09)	0·12(0·04)[b]	0·04(0·02)[b]	0·24(0·06)[ab]
RF	0·18(0·08)	0·19(0·10)	0·07(0·02)[b]	0·04(0·04)[b]	0·17(0·06)[b]
OP	0·94(0·90)	0·13(0·04)	0·30(0·15)[a]	0·37(0·28)[a]	0·39(0·12)[a]
IR	0·30(0·21)	0·10(0·04)	0·16(0·08)[ab]	0·08(0·07)[b]	0·18(0·07)[b]
SP	0·43(0·30) NS	0·12(0·04) NS	0·13(0·04)[b]	0·10(0·03)[b]	0·23(0·13)[ab]

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation-based fruit production; SP, *Faidherbia albida* based silvopasture.
±Mean values followed by standard errors in the parentheses; values with different letters are significantly different· NS = not significant (Tukey's test, $P = 0.05$).

Table V. C:N associated with aggregates and primary particles

Land use	Macro	Micro	Sand	Silt	Clay
	Soil depth 0–10 cm				
AF	11·4(0·4)[b]	9·9(0·5)	12·3(0·8)[ab]	9·0(0·4)	7·7(0·3)
RF	13·3(1·2)[ab]	10·2(1·2)	10·6(1·4)[b]	9·2(0·7)	8·9(1·3)
OP	12·0(1·51)[ab]	12·1(1·6)	13·8(0·9)[a]	11·0(0·8)	7·8(0·5)
IR	16·3(4·1)[a]	13·0(4·5)	11·5(2·2)[ab]	9·9(1·9)	8·7(0·7)
SP	11·7(2·7)b	10·7(1·9) NS	11·8(0·6)[ab]	10·5(1·1) NS	7·6(0·8) NS
	Soil depth 10–20 cm				
AF	11·1(1·7)	11·1(2·0)	12·0(1·5)	8·9(0·4)	8·5(0·3)
RF	13·1(2·1)	11·8(2·5)	11·5(5·6)	8·9(2·3)	8·7(0·3)
OP	12·7(2·7)	11·6(2·2)	14·6(2·8)	11·3(1·3)	8·6(0·6)
IR	12·8(1·9)	13·1(2·8)	8·7(2·9)	12·1(2·8)	9·4(0·5)
SP	14·6(4·6) NS	12·2(1·5) NS	11·0(1·9) NS	10·4(1·7) NS	9·3(0·7) NS

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; OP, communal open grazing/pasture; IR, irrigation-based fruit production; SP, *Faidherbia albida* based silvopasture.
±Mean values followed by standard errors in the parentheses; values with different letters are significantly different· NS = not significant (Tukey's test, $P = 0.05$).

found higher ρ_b in surface layers than in subsurface layers in cultivated soils as surface layers can be compacted by plowing and other mechanical operations.

The highest WSA in 0- to 10-cm depth was measured in OP followed by SP land use (Table I). Both land uses received more biomass input from grass and tree residues than that added in cultivated soils. Hairiah *et al.* (2006) reported high WSA in forest soils in west Lampung, which received more litter biomass than cultivated soils. The amount of plant residues and the degree of SOM decomposition are vital factors in the formation and stabilization of aggregates (Blanco-Canqui & Lal, 2004). In a recent study by García-Orenes *et al.* (2012) in eastern Spain revealed that application of oat residues for 5 years improved soil structure and aggregate stability to the level of that under natural vegetation cover. Among cultivated soils, WSA was generally higher in AF than those in RF and IR land uses, due to OM input from litter fall from *F. alibida* trees (Hadgu *et al.*, 2009), but the difference was not statistically significant because SOC concentrations and soil texture may have influenced WSA. Similarly, increase in carbon storages and paddy rice yields were reported in a long-term experiment using compost in lowland sandy soil in eastern Thailand as a result of application of organic fertilizer, which improve soil productivity and carbon storage as it is able to maintain proper soil aggregates and promote soil microbial activity (Jaiarree *et al.*, 2011). The magnitude of soil disturbance and the amount of residue incorporated into the soil impacted WSA and the associated SOC (Blanco-Canqui & Lal, 2004; García-Orenes *et al.*, 2009; Srinivasarao *et al.*, 2011). Therefore, WSA was strongly correlated with SOC ($R^2 = 0.85$; Figure 1a) and TN ($R^2 = 0.79$; Figure 1b) concentrations.

Land use had a significant effect on aggregate size distribution in surface soils. Soils under OP and SP land use systems had more macroaggregates, whereas cultivated soils had a higher proportion of microaggregates because tillage has been found to induce loss of C-rich macroaggregates

and a gain of C-depleted microaggregates (Six *et al.*, 2000). In comparison, AF had a higher proportion of intermediate sized aggregates (0·25–1·0 mm) indicating the positive effect of SOM derived from tree biomass (leaf litter and root exudates) on WSA (Table II). In the microaggregate size category, RF followed by IR and AF had significantly higher amounts of microaggregates than that in OP land use system (Table II). The mulch from grass and leaf litter in OP and SP land use systems restored SOM, created habitats for soil mesofauna and microfauna and flora, and enhanced soil aggregation (Blanco-Canqui & Lal, 2004). Because residues of crops are harvested for fuel and feed in the study area, SOM and WSA are adversely affected in RF, IR, and AF land uses. Furthermore, SOM that binds microaggregates into macroaggregates is a labile fraction and is highly sensitive to land use change and cultivation (Ashagrie *et al.*, 2005). Shrestha *et al.* (2007) also reported that cultivated soils in a Nepalese watershed contained a higher amount of microaggregates than macroaggregates.

Higher SOC and TN concentrations were measured in OP and SP than the other land uses in both 0- to 10-cm and 10- to 20-cm layers. Likewise, Barua & Haque (2013) found significantly higher soil organic carbon concentration at 0- to 10-cm depth under vegetated land uses than that in the adjacent degraded hill areas in Chittagong district, Bangladesh. Further, a recent study by Batjes (2012) in the upper Tana river catchment, Kenya, indicated that an average of 44–50% of the SOC content in soil is stored in the upper 30 cm, the layer most vulnerable to changes in land use or management. García-Orenes *et al.* (2010) in eastern Spain also found high SOC content in soils under natural cover due to the higher input of root exudates and plant residues as well as their long-term accumulation. Both SOC and TN concentrations decreased with depth in all land uses except RF, which showed slightly the reverse trend (Table I). Trujillo *et al.* (1997) also observed that SOC generally diminishes with depth regardless of vegetation, soil texture, and clay
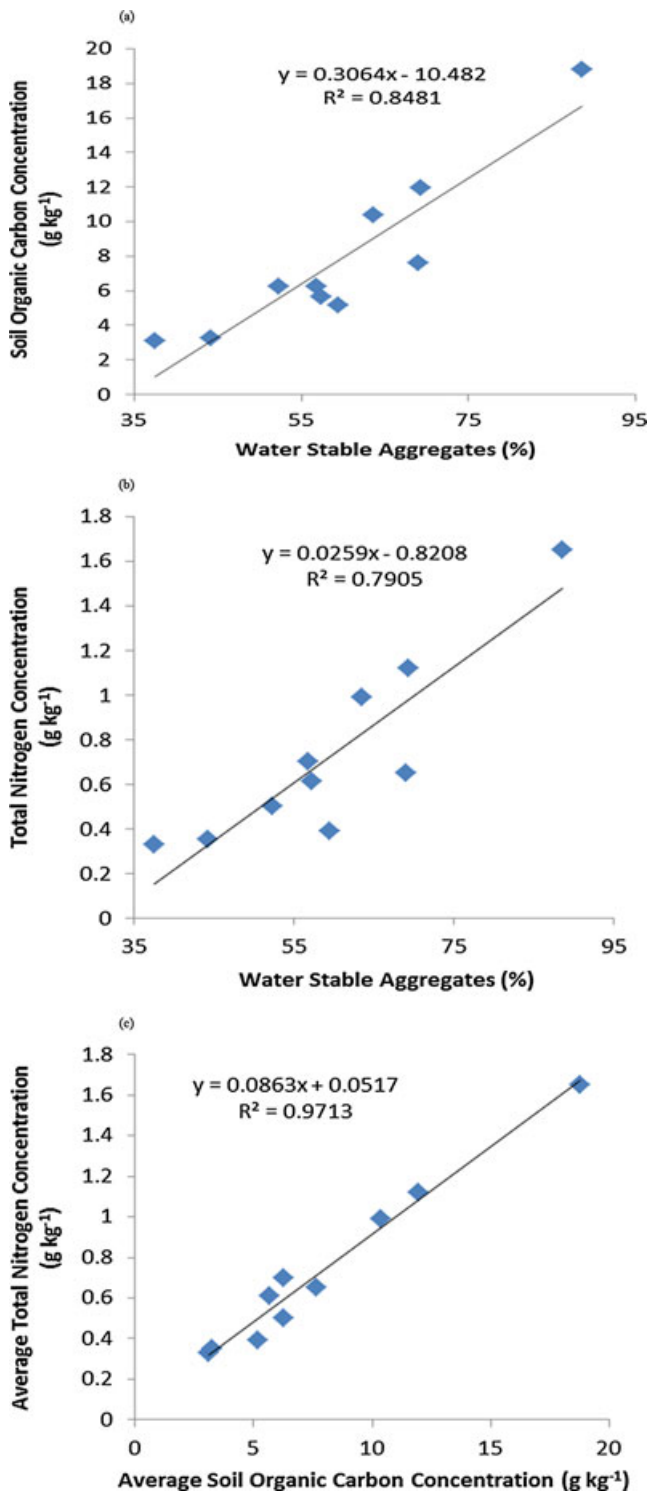
Figure 1. Correlations among soil organic carbon, total nitrogen and water stable aggregation under different land uses.

size fraction. Haile *et al.* (2008) reported that SOC concentration declined with depth in silvopastural systems in Florida. Similar trends were also reported by Shrestha *et al.* (2007) who measured a decrease in SOC concentrations with depth in soils under different land uses in a watershed in Nepal. However, soils under RF slightly increased with depth showing the effects of tillage practices in altering the depth

distribution of SOC because of mixing (Beare *et al.*, 1997; Chen *et al.*, 2009).

The highest concentration of SOC associated with macroaggregates was measured in OP followed by SP and AF in both 0- to 10-cim and 10- to 20-cm depths (Table III). Further, SOC concentrations associated with both macroaggregates and microaggregates decreased with depth except in RF, which showed slightly the reverse due to the adverse effect of tillage on the surface 0- to 10-cm depth. Macroaggregates contained higher SOC concentrations than microaggregates in both depths and land uses. The presence of decomposing roots and hyphae within macroaggregates increased SOC concentrations than in microaggregates (Tisdall & Oades, 1980). Elliott (1986) also suggested that macroaggregates have elevated SOC concentrations because of the organic matter binding microaggregates into macroaggregates. Similar results were reported by Shrestha *et al.* (2007). The highest SOC concentration associated with microaggregates in 0- to 10-cm layer in AF land use was followed by those in SP and IR (Table III). The higher SOC concentration in microaggregates under AF indicates its high potential to stabilize SOC.

Concentrations of TN associated with macroaggregates followed trends similar to that of SOC because of a strong association of SOC with N (Ganuza & Almendros, 2003). As soil carbon levels increase, microbes shift to immobilize N rather than mineralize it, which result in decrease in net N mineralization rates (Mclauchlan *et al.*, 2006). Therefore, OP had the maximum TN concentrations associated with macroaggregates in both depths (Table IV).

Open pasture and SP land uses also had higher SOC and TN concentrations associated with sand particles in 0- to 10-cm depth than other land uses. The data indicated that sand-associated SOC and TN concentrations were higher in soils under uncultivated/undisturbed natural systems. SOC distribution in different size fractions was found to be influenced by soil cultivation (Tiessen & Stewart, 1983; Dalal & Mayer, 1986; Wu *et al.*, 2006). The highest SOC and TN concentrations associated with both silt and clay particles were obtained in OP land use system in both depths followed by that in SP (Tables III and IV). The higher SOC concentrations associated with clay particles in soils under OP, SP, and AF indicated that grass-based and tree-based land use systems are rich in stable SOC as clay-associated SOC has higher residence time than sand or silt-associated SOC fractions (Ashagrie *et al.*, 2005).

C:N ratio was characterized by a decreasing trend from macroaggregates to microaggregates and from sand to silt and clay particles in all land uses for both depths except in IR, which showed a mixed result. In general, the coarser fractions contained undecomposed or partly decomposed soil organic matter with a higher C:N ratio than the finer fractions. Elliott (1986) suggested that the increase in macroaggregate associated C:N with time because cultivation is due to the fact that the accumulating organic matter is less highly processed. Light fraction materials associated with macroaggregates and sand sized fractions usually

consist of relatively recent detritus and consequently have elevated C:N compared with the heavy fractions associated with microaggregates and finer, silt and clay, particles (Spycher *et al.*, 1983; Sollins *et al.*, 1984; Jastrow, 1996).

CONCLUSION

Higher proportions of water stable aggregates were measured in the surface soil layers of OP and SP land uses showing that the amount of grass and tree residues added to these soils and the degree of decomposition are vital factors in the formation and stabilization of aggregates compared with cultivated soils. The magnitude of soil disturbance and the amount of residue incorporated into the soil impacted WSA and the associated SOC and TN. Therefore, WSA was strongly correlated with SOC ($R^2 = 0·85$) and TN ($R^2 = 0·79$) concentrations. Soils under OP and SP land use systems also had greater amounts of macroaggregates, whereas cultivated soils had a higher amount of microaggregates. RF followed by IR and AF land uses had higher amounts of microaggregates than OP land use system because tillage has been found to induce a loss of C-rich macroaggregates and a gain of C-depleted microaggregates. Accordingly, the highest concentration of SOC and TN associated with macroaggregates was measured in grass-based and tree-based (OP, SP, and AF) land uses in both 0- to 10-cm and 10- to 20-cm depths. This study also indicated that sand, silt, and clay-associated SOC and TN concentrations were higher in soils under uncultivated/undisturbed OP and SP systems, confirming that SOC and TN distribution in different size fractions were found to be influenced by soil cultivation. The higher SOC concentrations associated with clay particles in soils under OP, SP, and AF land uses also indicated that grass-based and tree-based land use systems are rich in stable SOC as clay-associated SOC has higher residence time than sand or silt-associated fractions. Generally, the result of this study supported the hypothesis that land use change from dry land crop cultivation to tree-based and grass-based systems improved magnitude and stability of SOC and TN associated with aggregates and primary particles. Thus, adoption of restorative land uses, no-till farming, and other recommended less intensive cultivation practices are needed for soil carbon stabilization and sustainable use of soil resources in the region as illustrated in this study.

ACKNOWLEDGEMENTS

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Paper III

Gelaw AM, Singh B R, Lal R. Soil quality indices for evaluation of tree-based agricultural land uses in a semi-arid watershed in Tigray, Northern Ethiopia (**under review in Agroforestry Systems**)

Soil quality indices for evaluation of tree-based agricultural land uses in a semi-arid watershed in Tigray, Northern Ethiopia

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Abstract

In Ethiopia, population growth and increasing resource demands are placing ever more pressure on the agricultural landscape. Most soils in the country are already exhausted by several decades of over exploitation and mismanagement. Since many of the issues of agricultural sustainability are related to soil quality, its assessment is very important. Therefore, in 0-15 cm depth, integrated soil quality indices (SQI) were estimated for three agricultural land uses: rain fed cultivation (RF), agroforestry (AF) and irrigation (IR) each with five replications within a semi-arid watershed in eastern Tigray, Northern Ethiopia. Using the framework suggested by Karlen and Stott (1994), four soil functions regarding soil's ability to: (1) accommodate water entry (WE), (2) facilitate water movement and availability (WMA), (3) resist degradation (RD), and (4) supply nutrients for plant growth (PNS) were estimated for each land use. Accordingly, AF land use had significantly higher values ($P < 0.05$) than RF for all soil functions except for RD. Finally, the four soil quality functions were integrated into an overall SQI, and the values for the three land uses were in the order: 0.58 (AF) > 0.51 (IR) > 0.47 (RF). Thus, AF scored significantly higher SQI ($P < 0.01$) than that of RF. Major driving soil properties for the integrated SQI were soil organic carbon (26.4%), water stable aggregation (20.0%), total porosity (16.0%), total nitrogen (11.2%), microbial biomass carbon (6.4%) and cation exchange capacity (6.4%). These six parameters together contributed more than 80 % of the overall SQI.

Keywords: Soil quality, soil functions, land degradation, land use, Ethiopia

1. Introduction

Land degradation and declining soil fertility are critical problems affecting agricultural productivity and human welfare in Sub-Saharan Africa (Sanchez et al. 1997). The main soil-environmental concern in the region is nutrient depletion as well as the loss of soil organic matter (SOM) and its related functions (Smaling 1993; Sanchez et al. 1997). In Ethiopia, total cultivated land has reached around 12-million hectares in mid-2013, but most of the soils are highly degraded (ATA 2013). Further, population growth and agricultural production are not growing *at par*. As a result, expansion to marginal lands and protected areas has become a common practice.

Tigray, the northernmost region in Ethiopia, is most known for its serious land degradation problems. Much of the woodland in Tigray started to disappear in the early 1960s under pressure from the rapidly growing population (Eweg et al. 1998). Hengsdijk et al. (2005) wrote their observations as follows: "perhaps nowhere in the world land degradation and soil nutrient depletion are more evident than in the marginal highlands of Tigray." In the region, a short and variable rainy season in combination with degraded soils resulted in low soil productivity and frequent crop failures. As a result, the local population are structurally dependent on food aid (Belete, 2002). If unattended to, land degradation and soil nutrient depletion would further reduce agricultural productivity and increase pressure on marginal environments, adversely affecting food security and livelihoods of smallholder farmers in the region (Belete, 2002).

Indeed, Tigray is not only known for its severe land degradation, but also for its vast environmental rehabilitation efforts in the last two decades (Girmay, 2009). Among the recent efforts towards enhancing agricultural development in the region, rainwater harvesting has been widely adopted (Moges et al. 2011) because supplementary irrigation is essential for crop production in arid regions as it increases soil water availability during dry spells (Feng et al. 2005). Further, farmers in Tigray have a culture of selectively taking care of trees, which are remnants of the original woodlands. *Acacia albida* Del. (Syn. *Faiderbhia albida* (Del.) A Chev.) trees are among the most selected ones in the region. Nowadays, farmers grow these trees in and around their farmlands in order to improve soil fertility and increase crop yields (Hadgu et al. 2008).

Sustainability of agricultural systems is an important issue in Ethiopia including Tigray. Many of the issues of agricultural sustainability are related to soil quality. Thus, its assessment and the direction of change with time is a primary indicator of whether agriculture is sustainable (Karlen et al. 1997; Masto et al. 2007). Soil quality is a combination of soil physical, chemical and biological properties that are able to change readily in response to variations in soil conditions (Brejda et al. 2000). It may be affected by land use type and agricultural management practices because these may cause alterations in soil's physical, chemical and biological properties, which in turn results in change in land productivity (Islam and Weil 2000; Sanchez-Maranon et al. 2002). Integrated soil quality indices based on a combination of soil properties provide a better indication of soil quality than individual parameters.

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Karlen and Stott (1994) developed a soil quality index (SQI) based on four soil functions, namely the ability of the soil to: (1) accommodate water entry, (2) facilitate water movement, and absorption, (3) resist surface degradation and (4) sustain plant growth. Each soil function was explained by a set of indicators. Several authors among them Glover et al. (2000), Masto et al. (2007) and Fernandes et al. (2011) used a similar framework.

A soil quality index (SQI) helps to assess the soil quality of a given site or ecosystem and enables comparisons between conditions at plot, field or watershed level under different land uses and management practices. Soil quality research is vital for a comprehensive understanding of effects of different land uses and soil management strategies on potentials of soils for agriculture and other ecosystem functions. However, research on soil quality is almost non-existent in Ethiopia. Therefore, this study was conducted at a typical semi-arid agricultural watershed in Eastern Tigray, Northern Ethiopia, with the following objectives:

 (1) To evaluate effects of *F. albida* based agroforestry (AF), irrigation based *Psidium guajava* fruit production (IR) and a tree-less row-crop management (RF) (Figure 1) on selected physical, chemical and biological soil quality indicators and,

 (2) To compute an overall integrated soil quality index (SQI) for each land use system and compare among indices. The study was conducted to test the hypothesis that land use change from dry land rainfed cultivation (RF) to *F. albida* agroforestry (AF) and irrigation based *P. guajava* fruit production (IR) systems improves selected physical, chemical, and biological soil quality indicators and the overall integrated soil quality index of soils.

2.   Materials and Methods
2.1 Descriptions of the Study Site
Mandae watershed is located in Eastern Tigray, Northern Ethiopia. Geographically, it is located between 13°83'00 N to 13°85'00 N latitude and 39°50'00E to 39°53'00E longitude, with an area of about 10 km^2, and an elevation of 1960 to 2000 m. a. s. l. Average daily air temperature of the area ranges between 15°C and 30°C in winter and summer, respectively. Mean annual rainfall of the area is 558 mm, with a large inter-annual variation. Soils are classified as Arenosols, and associations of Arenosols with Regosols according to the World Reference Base for soil resources (WRB 2006). These soils are developed from alluvial deposits and Adigrat sandstones. Their textures are dominated by sand, loamy sand and sandy loam fractions (Rabia et al. 2013). Major land uses of the watershed include *Faidherbia albida* based agroforestry (27.7 ha), rainfed crop production (11.9 ha), open pasture (23.2 ha), and irrigation-based guava (*P. guajava)* fruit production (11.3 ha). Agricultural rotation in the area is usually maize (*Zea mays*)–teff (*Eragrostis tef*)-field beans (*Vicia faba*)-finger millet (*Eleusine coracana*) in the agroforestry and rainfed cultivation land use systems. Fallowing is not practiced in the area due to population pressure and scarcity of farmlands. Use of chemical fertilizers is minimal and land is prepared for cultivation by using a wooden plow with oxen. Crop residues and manures are used for animal feed and household fuel, respectively. No pesticides and other agricultural inputs are used in the area. Irrigation from shallow wells started in the area in late 1990s and guava fruits cover most of the irrigated areas. Mixed crop-livestock smallholder farming is a typical farming system of the region.

Figure 1 The three agricultural land use systems in the area with dryland crop production (RF), Faidherbia *albida* based agroforestry (AF) and irrigation-based *P. guajava* fruit production (IR).

2.2 Soil Sampling and Analysis

Fifteen soil samples were collected in May 2010 from the surface (0-15 cm) layer of five sites randomly chosen at different locations from three agricultural land uses (AF, IR and RF). The summit position of the watershed was excluded to minimize confounding effects of slope and soil erosion. The samples were air-dried, mixed, ground, and passed through a 2-mm sieve for chemical analyses. Core samples were also collected from the same depth using 100 cm^3 volume stainless steel tubes (5-cm diameter and 5.1-cm height). Initial weights of the soil cores were measured in the laboratory immediately after collection. Simultaneously, soil moisture content was determined gravimetrically by oven drying the whole soil at 105°C for 24 hours to compute dry bulk density (ρ_b) (Blake and Hartge 1986). No adjustment was made for rock volume because it was rather minimal. The major parts of the soil analyses were carried out at Mekelle University soil laboratory, Ethiopia. Soil organic carbon (SOC) and total nitrogen (TN) were analyzed at the Carbon Sequestration and Management Center (C-MASC) Laboratory (The Ohio State University, USA) using auto CN analyzer (Vario Max CN Macro Elemental Analyser, Elementar Analysensysteme GmbH, Hanau, Germany) by the dry combustion method (Nelson and Sommers 1996). Similarly, water stable aggregation (WSA) was measured at C-MASC soil physics laboratory by the wet sieving method (Yoder 1936). Because soils did not show carbonates when tested with 10 % HCl, it was assumed that the total C obtained in the analysis closely estimates soil organic carbon (SOC) concentration. Available P (Olsen) was analyzed using a standard Olsen method (Olsen et al. 1954). Cation exchangeable capacity (CEC) was estimated titrimetrically by ammonium distillation method (Chapman 1965). Lastly, total porosity was calculated using the equation:

$$\text{Total Porosity (TP)(V\%)} = \left[1-(\rho b/\rho s)\right]*100 \tag{Eq. 1}$$

Where, ρs = particle density (2.65 g/cm^3), ρb = bulk density

2.2.1 Microbial Biomass Carbon (MBC)

Another set of nine field-moist soil samples (40 g each) from the surface (0-15 cm) depth were collected in three replications from the three agricultural land uses (AF, IR and RF) in May 2012 for the determination of microbial biomass carbon (MBC). The samples were transported in an icebox to the Norwegian University of Life Sciences soil laboratory, Ås, Norway. The MBC analysis was carried out following the fumigation-extraction method (Brookes et al. 1985; Vance et al. 1987). At first, each sample was divided in to three subsamples, and one out of the three (10.0 g) was fumigated with ethanol-free chloroform for 24 h at 25°C in an evacuated extractor. Afterwards, from the remaining two subsamples, one was used for moisture determination and the other treated as control for each plot. Fumigated and non-fumigated soils were extracted with 40-ml 0.5-mol l^{-1} K_2SO_4 (1:4 soil:extractant) and shaken for 1-h on a reciprocal shaker. The extracts were filtered using Whatman No.42 filter paper of 7-cm diameter and stored frozen at -15°C prior to analysis. Finally, total organic carbon in the extracts was measured using Total Organic Carbon Analyzer (SHIMADZU) at NMBU laboratory, Ås, Norway. Microbial Biomass Carbon (MBC) was calculated as follows:

$$\text{MBC} = E_C/KE_C \tag{Eq. 2}$$

Where E_C = (organic C extracted from fumigated soils) - (organic C extracted from non-fumigated soils) and KE_C = 0.45 (Wu et al., 1990).

2.3 Soil Quality Assessment

Effects of land use on soil quality was assessed following the framework suggested by Karlen and Stott (1994). We followed this framework because of its flexibility, ease of use and its potential for interactive use. The approach uses selected soil functions, which are weighted and integrated according to the following expression:

$$\text{SQI} = \text{WE(wt)} + \text{WMA(wt)} + \text{RD(wt)} + \text{PNS(wt)} \tag{Eq. 3}$$

Where, SQI = overall soil quality index, WE = soil's ability to accommodate water entry, WMA = soil's ability to facilitate water movement and availability, RD = soil's ability to resist degradation, PNS = soil's ability to supply nutrients for plant growth, and wt = a numerical weighting for each soil function.

These numerical weights were assigned to each soil function according to their importance in fulfilling the overall goals of maintaining soil quality under specific conditions of this study. According to Karlen and Stott (1994), the sum of weights for all soil functions must equal 1.0. Karlen and Stott (1994) assigned equal weight to each soil function. However, different weight values of 0.2, 0.2, 0.2 and 0.4 were assigned for this study for WE, WMA, RD, and PNS, respectively (Table 1). For this study, the soil's ability to supply nutrients for plant growth (PNS) was assigned with more value than other functions, because use of chemical fertilizers was minimal in the area and hence

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nutrient supply was considered the most important production constraint. Further, sustaining crop production is the major goal of soil management strategies in most developing countries including Ethiopia. The PNS function was further divided into three second-level functions viz. nutrient storage, nutrient cycling and nutrient availability (Table 1).

Table 1 Soil quality-indexing framework adapted from Glover et al., 2000.

Function	Weight	Indicator level 1	Weight	Indicator level 2	Weight	Source for indicators/ weights
Accommodate Water Entry	0.20	WSA	0.40			Karlen et al., 1994a; Glover et al., 2000
		BD	0.20			Glover et al., 2000
		POR	0.20			Masto et al., 2007
		SOC	0.20			Masto et al., 2007
Facilitate Water Movement and Availability	0.20	POR	0.60			Karlen et al., 1994a; Glover et al., 2000; Masto et al., 2007
		SOC	0.40			Karlen et al., 1994a; Glover et al., 2000
Resist Surface Degradation	0.20	WSA	0.60			Karlen et al., 1994a; Glover et al., 2000
		Microbial Processes	0.40	MBC	0.60	Karlen et al., 1994a; Glover et al., 2000; Masto et al., 2007
				SOC	0.20	Karlen et al., 1994a; Glover et al., 2000; Masto et al., 2007
				TN	0.20	Karlen et al., 1994a; Masto et al., 2007
Supply Plant Nutrient	0.40	Nutrient Storage	0.40	CEC	0.40	Masto et al., 2007
				SOC	0.40	Masto et al., 2007
				TN	0.20	Masto et al., 2007
		Nutrient Cycling	0.20	SOC	0.40	Karlen et al., 1994a; Masto et al., 2007
				MBC	0.20	Karlen et al., 1994a; Masto et al., 2007
				TN	0.40	Karlen et al., 1994a
		Nutrient Availability	0.40	SOC	0.20	Masto et al., 2007
				pH	0.20	Karlen et al., 1994a
				TN	0.20	Masto et al., 2007
				AVP	0.20	Masto et al., 2007
				AVK	0.20	Masto et al., 2007

An ideal soil would fulfill all the functions considered important, and would have an integrated SQI of 1.0 under the proposed framework. However, as a soil fails to meet the ideal criteria, its SQI would fall, with zero being the lowest rating. Associated with each soil function are soil quality indicators that influence, to varying degrees, the specific soil function. Threshold values for each soil quality indicator were set based on the range of values measured in natural ecosystems (the adjacent grass pasture in our case) and on critical values in the literature (Table 2). Glover et al. (2000) also used adjacent grass pasture areas to determine critical values for a study conducted in Washington State, USA. As with each soil function, numerical weights assigned to soil quality indicators contributing to the specific function must also sum to 1.0. After finalizing the thresholds, the soil property values recorded under the three agricultural land use systems were transformed into unit-less scores (between 0 and 1), using the following equation (Masto et al. 2007):

$$\text{Non-linear score(Y)} = \frac{1}{(1+e^{-b(x-A)})} \qquad \text{(Eq. 4)}$$

Where, x is the soil property value, A the baseline or value of the soil property where the score equals 0.5 and b is the slope of the tangent to the curve at the baseline.

The score for each indicator was calculated after establishing the baseline, the lower, and the upper threshold values (Table 2). Threshold values are soil property values where the score equals one (upper threshold) when the measured soil property is at the most favorable level; or equals zero (lower threshold) when the soil property is at an unacceptable level. Baseline values are generally regarded as minimum target values (Masto et al. 2007). There are two baselines for 'Optimum' curves, lower base line and upper base line, which corresponds to 0.5 score of the growth and death curves, respectively (Masto et al. 2007).

Table 2 Scoring function values and references for evaluating soil quality from Masto et al., 2007.

Indicator	Scoring curve	Depth (cm)	LT	UT	LB	UB	OPT	Slope	Source of threshold/ baseline values
Physical properties									
BD (Mg m^{-3})	Less is better	0-15 cm	1.0	2.0	1.5	-	-	-2.0832	Hussain et al., 1999; Grass pasture
WSA (>0.5 mm)	More is better	0-10 cm	0.0	40.0	20.0	-	-	0.0339	Adjacent grass pasture
TP (V%)	Optimum	0-15 cm	20.0	80.0	40.0	60.0	50.0	0.0644	Karlen et al., 1994; Masto et al., 2007; Adjacent grass pasture
Chemical Properties									
CEC (cmol (+) kg^{-1})	More is better	0-15 cm	0.0	18.0	9.0	-	-	0.0757	Masto et al., 2007; Adjacent grass pasture
pH (1:2.5)	Optimum	0-15 cm	3.0	9.0	5.0	8.0	7.0	0.5332; -0.496	Fernandes *et al.*, 2011
TN (kg ha^{-1})	More is better	0-15 cm	0.0	2000.0	1000.0	-	-	0.0007	Masto et al., 2007; Adjacent grass pasture
AVP (kg ha^{-1})	More is better	0-15 cm	0.0	50.0	25.0	-	-	0.0226	Masto et al., 2007
AVK (kg ha^{-1})	More is better	0-15 cm	0.0	400.0	200.0	-	-	0.0036	Masto et al., 2007

Biological Properties									
SOC (g kg^{-1})	More is better	0-15 cm	0.0	10.0	5.0	-	-	0.134	Masto et al., 2007 Adjacent grass pasture
MBC (mg kg^{-1})	More is better	0-15 cm	0.0	300.0	150.0	-	-	0.0042	Masto et al., 2007

Using this non-linear scoring curve equation, three types of standardized scoring functions typically used for soil quality assessments were generated: (1) 'More is better'; (2) 'Less is better', and (3) 'Optimum' as per earlier studies (Karlen and Stott 1994; Karlen et al. 1994a, b; Glover et al. 2000; Masto et al. 2007; Fernandes et al. 2011). The equation defines a 'More is better' scoring curve for positive slopes, a 'Less is better' curve for negative slopes, and an 'Optimum' curve is defined by the combination of both positive and negative slopes. These scoring curves are presented in detail by many authors (Wymore 1993; Karlen et al. 1994a; Hussain et al. 1999; Glover et al. 2000; Fernandes et al. 2011).

Table 3 Relative importance of selected soil properties for the soil quality index

Soil quality indicator	Weight	Soil function
Soil organic carbon	0.264	Accommodate water entry Facilitate Water movement and availability Resist Surface structure degradation Supply plant nutrients
Aggregate Stability	0.200	Accommodate water entry Facilitate Water movement and availability Resist surface structure degradation
Bulk density	0.040	Accommodate water entry
Porosity	0.160	Accommodate water entry Facilitate water movement and availability
Microbial biomass carbon	0.064	Resist surface structure degradation Supply plant nutrients
Cation exchange capacity	0.064	Supply plant nutrients
Total Nitrogen	0.112	Supply plant nutrients Resist surface structure degradation
Available phosphorus	0.032	Supply plant nutrients
Available Potassium	0.032	Supply plant nutrients
pH	0.032	Supply plant nutrients
Total	1.0	

2.4 Statistical Analyses
Effects of different land use systems on soil quality indicators, functions and integrated quality indices were subjected to one-way ANOVA. Excel spreadsheet was used for transforming soil quality indicator values into unit-less scores. Differences between means of parameters were considered significant at the 0.05 level using the Tukey's studentized (HSD) test. The data were analyzed using R version 3.02 software package (R core Team, 2012).

3. Results and Discussion

3.1 Soil Physical Quality indicators

Bulk density ranged from 1.48 (Mg m^{-3}) in AF to 1.57 (Mg m^{-3}) in both IR and RF land use systems (Table 4). However, there was no significant difference in BD among land uses. Although soils under AF land use contained SOC concentration twice more than that under RF, the detrimental effects of tillage may have offset the beneficial effects of SOC on BD (Karlen et al. 1994b; Glover et al. 2000). Soils under AF land use also had the highest percentage of WSA (17.3%) but it was not significantly higher than that under IR and RF land uses. Addition of more organic matter from leaf and root litters from the *F. albida* trees in AF than the other land uses likely explains the improved WSA in AF (Tisdall and Oades 1982). Similarly, Gelaw et al. (2013) reported that soils under natural grazing lands adjacent to cultivated lands were well structured, and contained higher SOC concentrations. Total porosity (TP) ranged from 35.5 % in RF to 43.5 % and 44.9 % in AF and IR land uses, respectively. However, the difference among land uses was not statistically significant. Similar to BD and WSA, the detrimental effects of tillage may have offset the beneficial effects of SOC on TP (Karlen et al. 1994b; Glover et al. 2000; Gelaw et al. 2013).

3.2 Soil Chemical Quality Indicators

The CEC of soils in this study ranged from the highest in AF (11.5 cmol (p+) kg^{-1}) to the lowest in IR (4.8 cmol (p+) kg^{-1}). It was significantly higher ($P < 0.01$) under AF than that under IR and RF land uses (Table 4). Generally, CEC of these soils was low, with an exception of some improvements under AF land use. Rabia et al. (2013) also reported similar results for the same area. According to their study, up to 90% of soil samples from this area had extremely-low-to-low CEC values. Electrical conductivity (EC) values of the soils were much lower than the salinity hazard level for most crops (FAO, 1976) (Table 3). Rabia et al. (2013) also reported low EC values and less salinity problems in all soils under all the land uses in the area. Arenosols generally have neutral pH values (Hartemink and Huting 2008). However, soils under IR land use showed a significantly higher ($P < 0.001$) pH value than that under other land uses, and it was slightly alkaline. The source of slight alkalinity development in the soil under IR land use could be from the supplemental irrigation. Rabia et al. (2013) also reported similar results.

Soils under AF contained the highest total nitrogen (TN) stock (1568.6 kg ha^{-1}), and it was significantly higher ($P < 0.05$) than that in IR and RF land uses (Table 4). Hadgu et al. (2008) reported similar results in their study in central Tigray, Northern Ethiopia, which compared soils under canopies of *F. albida* and eucalyptus trees with that in tree-less fields. Similarly, available potassium (AVK) was significantly higher ($P < 0.001$) in AF than that in other land uses (Table 4). In contrast, available phosphorus (AVP) contents did not differ among land uses. The higher AVK in AF than that in other land uses could be related to the recycling of nutrients in the aboveground biomass, root biomass or through the recycling of depositions by cattle, which gather for shade under tree-canopies during sunny days (Arevalo et al. 1998). Depommier et al. (1992) also reported a significant increase in soil K content and an increase in sorghum (*Sorghum bicolar*) yield on soils under canopy of *F. albida* trees compared with soils 15-m away from canopies of those trees in two parklands in Burkina Faso. Results presented here are also in accord with reports by Nair (1993) that microsite enrichment qualities of trees such as *F. albida* in West Africa and *P. cineraria* in India have long been recognized in many traditional farming systems.

3.3 Soil Biological Quality Indicators

Both SOC and MBC are among principal soil parameters, which affect biological processes and soil quality. The highest SOC concentration was measured in AF (6.4 g kg^{-1}) followed by that in IR (5.9 g kg^{-1}), and the lowest was in RF (3.2 g kg^{-1}) (Table 4). Thus, SOC was significantly higher ($P < 0.05$) in AF than that in RF land use. However, it did not statistically differ between AF and IR, and between IR and RF land uses (Table 4). On the other hand, MBC was slightly higher in soils under IR (100.1 mg kg^{-1}) than that under AF and RF, but the differences were not statistically significant (Table 4). Higher MBC values under IR than that under AF and RF may be explained by less disturbance of soils under IR than those under the other intensively tilled land uses. The intensity of tillage in IR was less than that under AF and RF land uses. Besides, irrigation farms under guava fruits were not convenient for oxen plowing. Weed control and irrigation in IR land use are practiced by hand. The SOC in intensively cultivated soils has less physical protection than that in less cultivated soils because tillage disrupts macroaggregates and exposes previously protected SOM microbial processes (Islam and Weil 2000; Gelaw et al. 2013). Similarly, Franchini et al. (2007) reported an increase in MBC under no-till systems (NT) than that under conventional tillage systems (CT) receiving more plant residues in Southern Brazil. The lower MBC regardless of more plant residue addition under CT was due to higher CO_2-emission, which implies little conversion of carbon from plant residues into MBC (Franchini et al. 2007). Indeed, parameters associated with soil microbiological activities are sensitive, considered rapid indicators of effects of soil management, and are useful as indicators of soil quality (Franchini et al. 2007).

Table 4 Effects of land use systems on selected soil physical, chemical and biological quality indicators at a watershed in eastern Tigray, north Ethiopia

Soil Quality Indicator	Land use			
	RF	AF	IR	
Physical				
BD (Mg m^{-3})	1.57(0.03)	1.48(0.05)	1.57(0.02)	NS
WSA (>0.5 mm)	11.3(1.8)	17.3(2.5)	13.6(3.6)	NS
TP (V%)	35.4(3.6)	43.5(2.0)	44.9(2.7)	NS
Chemical				
CEC (cmol(p+) kg^{-1})	5.4(1.0)[b]	11.5(0.8)[a]	4.8(1.8)[b]	**
pH	6.6(0.3)[b]	6.4(0.2)[b]	8.0(0.03)[a]	***
TN (kg ha^{-1})	809.7(134.6)[b]	1568.6(85.4)[a]	1042.7(244.6)[ab]	*
AVP (kg ha^{-1})	24.4(10.7)	39.1(4.3)	39.8(4.7)	NS
AVK (kg ha^{-1})	216.5(56.9)[b]	1019.1(161.0)[a]	297.7(71.8)[b]	***
Biological				
SOC (g kg^{-1})	3.2(0.7)[b]	6.4(0.3)[a]	5.9(1.1)[ab]	*
MBC (mg kg^{-1})	75.5(24.1)	95.9(10.3)	100.1(31.3)	NS

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; IR, irrigation based fruit production; ± Mean values followed by standard errors in the parentheses; values with different letters are significantly different.* = P < 0.05; ** = P < 0.01; NS = not significant (Tukey's test, P = 0.05).

3.4 Soil Quality Indicators Integration and Assessment

For this study, four soil functions contributed to the overall soil quality index (SQI). They were weighted according to their relative importance in fulfilling the goals of maintaining soil quality in the area. Thus, the major driving soil parameters for the integrated SQI were SOC (26.4 %), WSA (20.0 %), TP (16.0 %), TN (11.2 %), MBC (6.4 %) and CEC (6.4 %). These six soil quality indicators together contributed more than 80 % of the variability in overall SQI (Table 3). Further, BD contributed 4.0 % followed by AVP, AVK and pH each contributing 3.2 % to the overall SQI. Regarding the soil's function for plant nutrient supply, SOC, TN, and CEC contributed 32-, 24-, and 16 % of the PNS function, respectively. Available P, AVK and pH each contributed 8 % of the soil's function for plant nutrient supply. The soil's MBC contribution to this function was minimal (4 %). Overall, SOC alone contributed more than 25 % and 30 % of SQI and PNS, respectively.

Integration of the soil property values into SQI using the framework resulted in a significantly higher (P < 0.05) score in AF than in RF land use system for its ability to accommodate water entry (Table 5). The relatively higher WSA, TP and SOC values of the soil in AF land use than those in RF were largely responsible for the improvement in the soil's ability to accommodate water entry in AF (Table 4). Glover et al. (2000) also reported higher scores for soil's ability to accommodate water entry because of higher WSA and lower BD under integrated and organic management systems than those under a conventional system in Washington State, USA. Regarding the soil's ability to facilitate water movement and availability, AF also scored significantly higher (P < 0.05) value than that of RF because of the relatively higher TP and SOC values in AF (Table 5). These results indicated that AF land use system

improved the soil's ability to hold and release water mainly due to its higher SOC content (Table 4). However, land use had no significant effect on soil's resistance to surface degradation and all are uniformly low (Table 5).

Table 5 Soil Quality Ratings for Different Land uses

Soil Function	Land use			
	RF	AF	IR	
Accommodate Water Entry (0.20)	0.09 (0.00)[b]	0.11 (0.002)[a]	0.10 (0.004)[ab]	*
Facilitate Water Entry and Availability (0.20)	0.10 (0.004)[b]	0.12 (0.004)[a]	0.11 (0.004)[ab]	*
Resist Surface Degradation (0.20)	0.09 (0.003)	0.11 (0.002)	0.09 (0.005)	NS
Source of Plant Nutrients (0.40)	0.19 (0.01)[b]	0.24 (0.004)[a]	0.21 (0.015)[ab]	*
Integrated Soil Quality Index (1.00)	0.47 (0.01)[b]	0.58 (0.01)[a]	0.51 (0.02)[ab]	**

RF, Dryland crop production; AF, *Faidherbia albida* based agroforestry; IR, irrigation based fruit production; ± Mean values followed by standard errors in the parentheses; values with different letters are significantly different. .* = P < 0.05; ** = P < 0.01; NS = not significant (Tukey's test, P = 0.05).

This clearly showed the detrimental effects of tillage on soil structure (Karlen et al., 1994b; Glover et al., 2000; Gelaw et al. 2013). In contrast, AF scored significantly higher (P < 0.05) value for soil's ability to supply plant nutrients than RF land use system largely due to higher levels of AVK, CEC, SOC, TN and AVP in the rooting zones of AF land use system (Table 5). The score for soils in IR land use was not significantly different from that in RF (Table 5). Further, the score for nutrient storage capacity of soils in AF land use was significantly higher (P < 0.05) than that in RF, but it was not significantly different from that in IR (Figure 2). However, nutrient cycling was not significantly affected by land use regardless of some improvements in AF. Trees in agroforestry systems can improve nutrient cycling and increase soil chemical fertility through bringing up nutrients from deeper layers and minimizing leaching hazards (Nair 1993). However, nutrient availability was affected by land use. Thus, AF scored significantly higher (P < 0.01) value for its capacity for nutrient availability than that in other land uses (Figure 2).
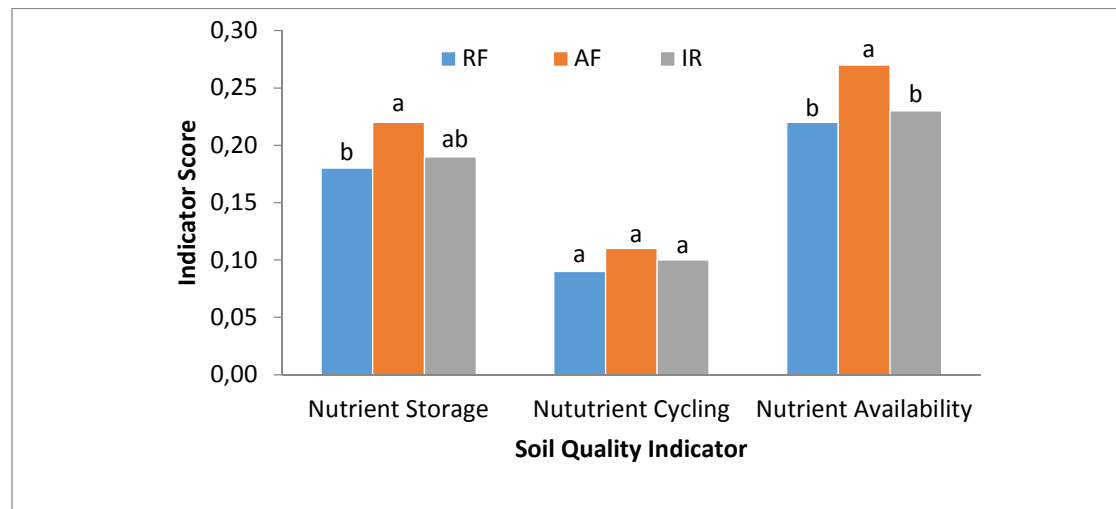

Figure 2 Effects of three agricultural land use systems (RF, AF and IR) on nutrient supplying capacities of soils at a semi-arid watershed in Tigray, Northern Ethiopia.

Finally, integrated SQI calculated for the land uses using the framework by Karlen and Stott (1994) were in the following order: 0.58 (AF) > 0.51 (IR) > 0.47 (RF) (Table 5). Soil quality index differed significantly ($P < 0.01$) between AF and RF land use systems (Table 5). Similarly, Karlen et al. (1994a) reported an improvement in soil quality rating from 0.45 to 0.86 in over ten-year period by retention or addition of crop residues on a no-till (NT) continuous corn in Wisconsin, USA. In another study, Karlen et al. (1994b) reported a significant improvement in SQI ratings from 0.48 and 0.49 in plow and chisel, respectively, to 0.68 in NT using selected physical, chemical and biological soil quality indicators on Rozetta and Palsgrove silt loam soils in Wisconsin, USA.

Regardless of a significant improvement in AF than that in RF land use, SQI ratings in all the three land use systems were very small compared with an ideal soil (Table 5). This result was in agreement with findings from other authors (Hengsdijk et al. 2005; Girmay et al. 2008) who reported that low organic matter and nutrient stocks are typical characteristics of soils in Tigray mainly due to nutrient mining because of crop harvests and complete removal of crop residues for feed and fuel. One fundamental principle of sustainability is to return to the soil the nutrients removed through harvests and other loss pathways (Sanchez 1994), and one of the main tenets of agroforestry is that trees enhance soil fertility (Sanchez 1994; Palm 1995). This is supported by observations of higher crop yields near trees of *F. albida* in Ethiopia (Poschen 1986; Kamara and Haque 1992; Asfaw and Ågren 2007; Hadgu et al. 2008) and elsewhere (Kwesiga and Coe 1994; Sanchez and Palm 1996), which showed the potentials of agroforestry systems for improving soil quality and productivity of smallholder farms in Ethiopia and the wider region.

4.   Conclusion

Relatively higher WSA, TN and SOC concentrations measured in soils under AF land use resulted in improved water entry, movement and availability than those under IR and RF. Soil's ability to supplying plant nutrients was also improved under AF than under RF land use largely due to higher levels of AVK, CEC, SOC, TN and AVP in the rooting zones of AF land use. However, there was no significant improvement in the soil's resistance to surface degradation in all land uses, which may be because of the detrimental effects of tillage. Further, when selected physical, chemical, and biological soil quality indicators were integrated into an overall SQI, AF land use received a higher soil quality rating (0.58) than that of RF (0.47). The result also indicated that SQI for AF would likely have been higher, if not for the continuous tillage practices, which disrupted structures of the soil. Further, the result highlighted potentials of *F. albida* based AF systems for improving soil quality and productivity of smallholder farms. Thus, adoption of *F. albida* based agroforestry systems in combination with other recommended management practices (RMPs) such as no-till, residue and manure management are recomended to build up soil structure, soil organic matter and soil moisture capacity of soils for a more efficient and sustainable use of soil resources in the country.

5.   References

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Paper IV

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Carbon Footprint and Sustainability of the Smallholder Cereal Production Systems in Ethiopia

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Abstract

Sustainability of an agricultural system depends on its carbon (C) footprint. Thus, major C-inputs and C-outputs of the smallholder cropping systems in Ethiopia were collated and their C-equivalence and sustainability indices (CSI) were computed. C-based inputs increased 2-fold from the lowest (0.32TgCeq y^{-1}) in 1994 to the highest (0.62TgCeq y^{-1}) in 2010. Similarly, total C-output increased linearly from the lowest (5TgCeq y^{-1}) in 1994 to the highest (17TgCeq y^{-1}) in 2011. Further, the average rate of increase in C-output from 1994 to 1999 was marginal, i.e., 0.3TgCeq y^{-1}, but the 11-years average rate of increase from 2000 to 2011 was relatively higher, i.e., 0.8TgCeq y^{-1}. The relationship between annual total C-based input and output was strong (r^2 = 0.88; P < 0.001). The CSI of the smallholder agricultural production systems in Ethiopia was comparable to more intensive systems in other regions, with 18-years average value of ~22.

Key words: carbon input/output; climate change; fertilizers; sustainable intensification.

Introduction

Ethiopia is a country with an agrarian economy, the most important objective of which is to achieve food security (ATA, 2013). The predominant farming system in Ethiopia is rain-fed agriculture, and its performance is highly dependent on the timely onset, duration, amount and distribution of rainfall. This makes the sector highly vulnerable to drought and other natural calamities. Because of increasing pressure of human and livestock populations on arable land and forest resources, large areas of the country, particularly in the northern and central highlands, have become vulnerable to loss of fertility, degradation and ecological imbalances. Furthermore, Ethiopia is experiencing the effects of climate change, such as an increase in average temperature and a change in rainfall patterns (FDRE, 2011). Hence, climate change adaptation and mitigation, which can go hand in hand in the agricultural sector (Kang and Banga, 2013) should be considered as a critical component of the overall developmental goals of the country.

Earth's climate is rapidly changing mainly because of increased anthropogenic greenhouse gas (GHG) emissions (Ruddiman, 2003; IPCC, 2006; Gan et al., 2011). Emissions are caused by a range of human activities, and farming is one of them (Janzen et al., 2006). Globally, along with fossil fuel combustion, agricultural practices have had a major impact on the global carbon (C) cycle leading to an increase in the global temperature during the 20th century by 0.6 ± 0.2^0C at an average rate of increase of 0.17^0C per decade since 1950 (Lal, 2004a; IPCC, 2007). The emissions of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) from agriculture and land-use change together account for approximately one-third of the annual increase in radiative forcing of climate change (Cole et al., 1997). Between 1970 and 1990, direct emissions from agriculture increased by 27% (IPCC, 2007). Ethiopia's GHG emissions are attributable to livestock and crops (FGRE, 2011). However, contribution of Ethiopian agriculture to the global increase in GHG emissions since the industrial revolution has been practically negligible.

The large effect of land use-changes is attributable to emissions of CO_2. By contrast, CH_4 and N_2O are the major contributors to agricultural impacts, as the agricultural sector produces about 50 and 70%, respectively, of the total anthropogenic emissions of these two gases (Cole et al., 1997). Options to mitigate CO_2 emissions from agriculture and land-use changes include the reduction of emissions from present sources and the creation and strengthening of C-sinks. Increasing the role of agricultural land as a sink for CO_2 includes C-storage in managed soils and C sequestration (CS) after reversion of surplus farm lands to natural ecosystems (Follett, 2001). There is a strong link between food security and the carbon pool in terrestrial ecosystems, especially the soil organic carbon pool (Lal, 2013). Thus, agricultural transformation to ensure food security in the face of changing climate with minimum ecological cost will help mitigate the negative effects of climate change (Kang and Banga, 2013).

Enteric fermentation in ruminants (Johnson et al., 1992), flooded rice fields, anaerobic animal waste processing (Gerber et al., 2013) and biomass burning (Cole et al., 1997) are major sources of global CH_4 emissions. On the other hand, the primary sinks for CH_4 are oxidation with hydroxyl radicals in the troposphere (Crutzen, 1981) and aerobic soils that provide an additional sink of 10-20 % of annual CH_4 emissions (Reeburgh et al., 1993).

Nitrous oxide emissions from application of mineral or organic N fertilizers in agricultural systems can be avoided by improving N fertilizer-use efficiencies of agricultural management practices. It can be done by better matching N supply to crop demand (van der Velde et al., 2013), and by more closely integrating animal waste and crop-residue management with crop production (Lal, 2007). Further improvements in farm technology, such as the use of controlled-release fertilizers (van der Velde et al., 2013), use of nitrification inhibitors, and water management (Mueller et al., 2012) lead to improvements in N-use efficiency and further limit N_2O production.

1

In a nutshell, more productive and climate-resilient agriculture requires better management of natural resources, such as land, water, soil, and germplasm, through practices such as conservation agriculture, precision agriculture, and integrated pest management (Kang and Banga, 2013).

In Ethiopia, over-exploitation and mismanagement of the soil resources across several decades has exhausted the extractive agricultural production systems (ATA 2013). Therefore, judicious use of recommended management practices (RMPs) is important to reverse the situation and increase productivity. Adoption of RMPs for agriculture involves off-farm or external input, which are C-based operations and products (Pimentel, 1992). There has been considerable increase in use of nitrogen (N) and phosphorus (P) fertilizers in Ethiopia in the last two decades.

Evaluating sustainability of farming systems through application of carbon footprint analysis and measuring impacts of agricultural activities on the environment in terms of the amounts of GHGs emitted in C-equivalent is very important (Dubey and Lal, 2009). Therefore, this study was planned with the objective of evaluating carbon footprint of the smallholder cropping systems in Ethiopia. The specific objectives were to: (1) assess C-emissions from the farming systems, (2) evaluate C-use efficiencies of the farming systems, and (3) compute the annual C-sustainability indices of these systems. The study was designed to test the following hypotheses: (i) crop yields increase with increased C-based inputs; (ii) sustainability of a cropping system increases with increased use efficiency of C-based inputs.

Materials and Methods
Carbon Inputs
Data on C-based inputs into the soil for predominant cereal crops: teff (*Eragrostis tef*), barley (*Hordeum vulgare*), maize (*Zea mays*), wheat (*Triticum aestivum*), sorghum (*Sorghum bicolor)* and finger millet (*Eleusine coracana*) were collected from annual abstracts of the Central Statistics Agency (CSA) of Ethiopia and from FAO database (FAO, 2014). The C-based inputs considered in this study were: (1) annual rates of N and P fertilizers used for each crop and, (2) amount of human labor and draft animals (oxen). These data were used to calculate C-equivalent of inputs and outputs, and sustainability indices of the smallholder cropping systems in Ethiopia from 1994 to 2011. Carbon equivalent (CE) is a term coined to describe different greenhouse gases in a common unit (Lal, 2004b). The advantage of using CE rather than other energy units lies in its direct application to the rate of enrichment of atmospheric CO_2, which was a major global issue at the dawn of the 21st century (Lal, 2004b).

Fertilizers
Manufacture, packaging, transport and application of fertilizers have hidden C-costs (Dubey and Lal, 2009). On the basis of numerous studies, estimates of hidden C-costs of fertilizers for production, packaging, storage, and distribution range from 0.9 to 1.8 kg CE kg^{-1} N, to 0.1 to 0.3 kg CE kg^{-1} P. Only N and P-fertilizers are used in Ethiopia. Ethiopian soils are traditionally believed to be rich in potash, so that no K fertilizer is used. For this study, the highest CE values, 1.8 and 0.3 for N and P, respectively, were used (Table 1) because Ethiopia imports fertilizers from Europe, USA and other regions. Hence, the hidden costs are considered high. Further, insecticides and pesticides are not commonly used by smallholders in Ethiopia. Irrigation is also negligible, so it is not included in these calculations.

Tillage and Harvesting
The other important component of C-input, i.e., direct emissions attributable to fuel use for tillage and harvesting, was not considered in this study because farmers in Ethiopia use neither machinery nor fuel for these farming operations. Rather, manual labor and draft animals (oxen) are used for plowing, seedbed preparation, weeding, harvesting and transportation. Therefore, hidden C costs for production by manual labor (33.046 kg CE ha^{-1}) and for draft animals (0.0377 kg CE ha^{-1}) (Table 1) were calculated based on the assumptions from Pimentel (1992) for labor and from Esmay (1992) for draft animals, respectively. The total input for labor and equipment (1.64 GJ ha^{-1}) for manual production of maize was taken from a study by Pimentel (1992) and the hidden C-cost for oxen was assumed as 0.52 KW ha^{-1} based on the energy use per hectare of rice (*Oryza sativa*) production in developing countries from Esmay (1992).

Table 1 Carbon equivalent (CE) conversion factors for different C-based inputs

Input	Equivalent C emission (kg CE kg^{-1})	Source
(a) Fertilizer		
N	1.8	Dubey and Lal (2009)
P_2O_5	0.3	Dubey and Lal (2009)
(b) Units of energy used		
Gigajoule (GJ)	20.15	Pimentel (1992)
Kilowatt hour (kW h)	7.25 x 10^{-2}	Esmay (1992)

Carbon Outputs

Three components of crop-based C-outputs were grain yield, and straw and root biomass yields (Table 2).Annual productions of teff, barley, maize, wheat, sorghum and finger millet were obtained from statistical abstracts of the Central Statistics Agency (CSA) of Ethiopia (1994 to 2011). Straw biomass yield for all crops was computed by using values of grain:straw ratios from published literature(Table 2) (Duke et al., 1983; Dubey and Lal, 2009). Similarly, below-ground root biomass for the crops was also calculated from published values of shoot:root ratios (Dubey and Lal, 2009; Gupta et al., 2011; Delelegn and Fassil, 2011; Sher et al., 2013). Finally, total C produced in biomass was assumed to contain 40% C (Dubey and Lal, 2009) and the net total C-output was calculated by adding up all outputs present in the total biomass of crops (grain, straw, and root).

Table 2 Grain to straw, and shoot to root ratios of cereal crops

Crop species	Grain:Straw	Shoot:Root	Source
Teff	1.10	4.1	Duke et al. (1983); Delelegn and Fassil (2011)
Wheat	1.23	7.4	Dubey and Lal (2009); Duke et al. (1983)
Barley	1.45	7.1	Dubey and Lal (2009); Duke et al. (1983)
Maize	1.10	6.3	Dubey and Lal (2009); Duke et al. (1983)
Sorghum	1.10	5.9	Duke et al. (1983); Sher et al. (2013)
Millet	1.10	2.8	Duke et al. (1983); Gupta et al. (2011)

Carbon Sustainability Index (CSI)

There are various ways to assess sustainability of a production system. In the context of the global climate change and anthropogenic emissions of GHGs into the atmosphere, sustainability of a production system can be assessed by evaluating temporal changes through calculating carbon sustainability index (CSI) (Lal, 2004b). Accordingly, CSI of the cropping systems in this study were calculated by dividing the difference between total C output and C input by C input (Lal, 2004b) as follows:

$$CSI = (Co - Ci)/Ci \qquad (Eq\ 1)$$

Where, CSI is C sustainability index, Co is C output, and Ci is C input

Statistical Analysis

Excel spreadsheet was used for descriptive analyses of the data. Correlation analysis was used to evaluate the relationship between C input and C output. Data were analyzed using R version 3.02 software package (R core Team, 2012).

Results and Discussion

Cereal crops constitute more than 80 % of the annual total crop production, and more than 96 % of the total grain production in Ethiopia comes from private peasant holders (CSA, 2011). Hence, these crops are highly important for advancing food security of smallholder framers in the country. In recent years, overall nationwide cereal grain production has considerably increased. However, yields of major food crops are still lower than that in other developing regions and some managed experimental plots in the country. On the other hand, land cultivated for cereal production increased 1.5-fold from 1994 to 2010, from 6.42 to 9.63 million hectares (Mha) (Figure 1). Thus, the increases in total production may have been achieved from the increased area of cultivated land (Dercon and Hill, 2009). In fact, the increment in area coverage was not linear as there were decreases in late 1990s and early 2000s, probably because of the border war with Eritrea (Kalewongel, 2008) and drought (Di Falco et al., 2011) (Figure 1). Similarly, application of N and P fertilizers for cereals increased 2.9-fold from 120,000 tons to 350,000 tons during 1994-2011 (Figure 1).However, fertilizer application also decreased to 50,000 tons in late 1990s and remained in the

range of 60,000 to 80,000 tons in early 2000s.Moreover, these data showed the worst consequences of war for a poor country such as Ethiopia, diverting resources from production to destruction. According to Kalewongel (2008), only in 1999, Ethiopian and Eritrean governments purchased arms worth 430 and 306 million dollars, respectively.

Generally, average fertilizer application in Ethiopian smallholder cereal-production systems has been low compared with international standards (Figure1). Further, the border war with Eritrea from 1998 to 2002 significantly affected the already meager resource allocation to the sector and the average fertilizer application was merely 3 to 4 kg ha^{-1} (Figure1). However, the average fertilizer application during the study period almost doubled, increasing from 19 kg ha^{-1} to 36 kg ha^{-1}. Nonetheless, fertilizer-use efficiency (FUE) of the system (Tg grain/Tg fertilizer) remained between 40 and 60, except the extremely high values during the years 1998-2002 with the least average fertilizer applications (Figure1). The stagnated FUE indicated that further increase in N and P fertilizers application was not effective in increasing yields. High returns from modern inputs are obtained if fertilizer, improved seeds and other recommended practices are all applied together. The supply of improved seeds in Ethiopia has lagged behind at 5% and irrigation use at <1% (Dercon and Hill, 2009). The findings of this study are also in accord with that of Dercon and Hill (2009), who reported that much of the increase in total production of major food crops in Ethiopia has thus far come from the increase in cultivated land.

This study further showed that cereal yield in Ethiopia was one of the lowest in the world. In1994, the average cereal yield was <1Mg ha^{-1} and it slowly increased to ~1Mg ha^{-1} until it reached ~1.5Mg ha^{-1} in 2006 and the highest in 2011 at an average yield of ~2Mg ha^{-1} (Figure1).The maximum average yield obtained in 2011 (~2Mg ha^{-1}) was higher than that of the Sub-Saharan Africa (SSA) average at 1.3Mg ha^{-1} (World Bank, 2014). Nonetheless, it was less than the average cereal yield for other developing countries; e.g., 2.4Mg ha^{-1} for Middle East and North Africa, 3.1Mg ha^{-1} for Europe and central Asia, and 3.8Mg ha^{-1} for Latin America and Caribbean (World Bank, 2014).
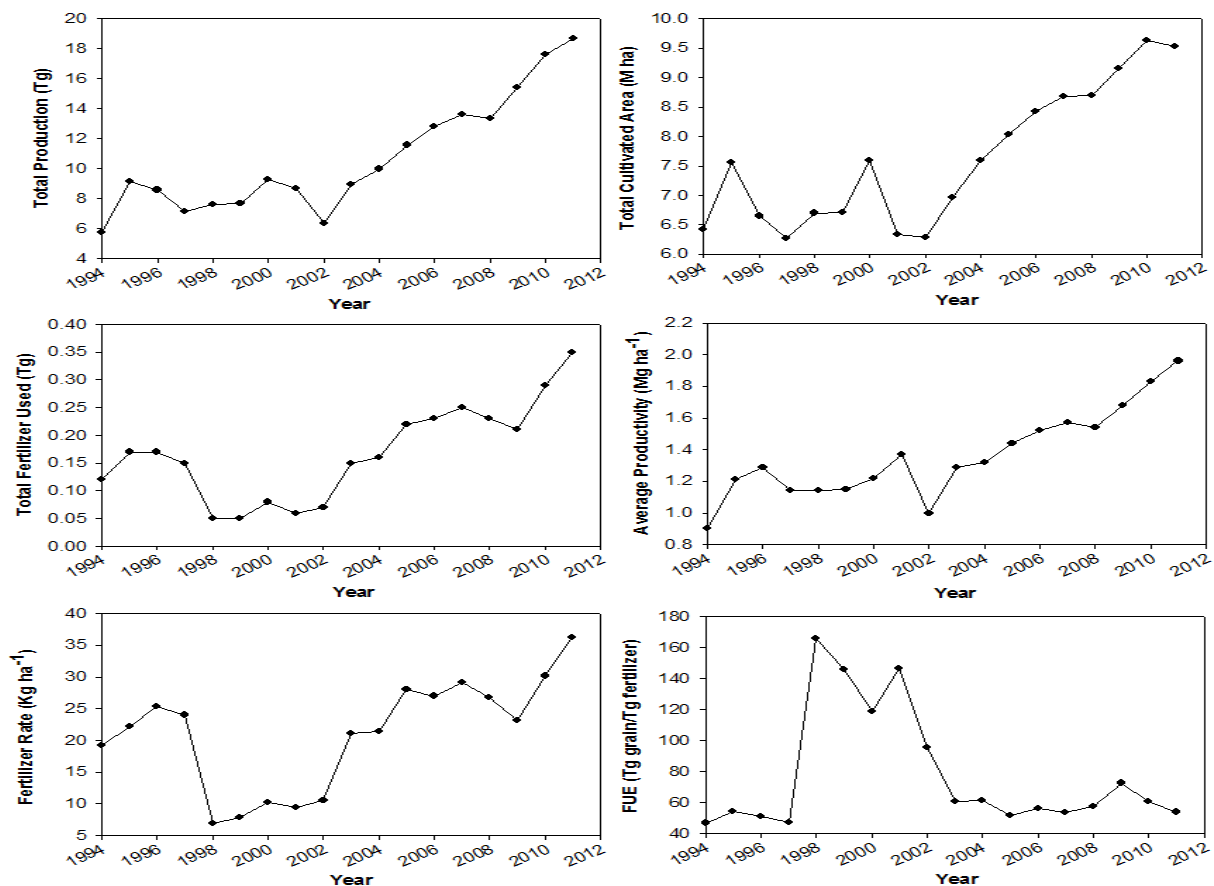

Figure 1 Trends in total cultivated land, total fertilizer use, average fertilizer application rate, and their effects on total production, average productivity, and fertilizer use efficiency of smallholder cereal-production systems in Ethiopia from 1994 to 2011.

Carbon Equivalence of Inputs

The 18-years C-based inputs for Ethiopian smallholder cereal-production systems from 1994 to 2011 included hidden C-costs for fertilizer, labor and draft animals (oxen). It started with a low input of 0.32TgCeq y[-1] in 1994 (Figure 2). The C-based input increased slightly during the following three years (1995 to 1997) before it decreased to 0.26TgCeq y[-1] in 1998 and then stagnated for the following five years up to 2002. However, starting from 2003, C-based inputs increased linearly to the highest value at 0.62TgCeq y[-1] in 2010. The maximum C-based input in 2010 was almost twice that of 1994 and 2.4-times the minimum inputs recorded in 1998 and 2001 (Figure 2). The increase in C-based inputs was rather small in comparison with the 39-fold increase in C-based inputs recorded in the pioneer Green Revolution state of Punjab in India from 1960 to 2000 (Dubey and Lal, 2009). The average rate of increase in C-based input for this study was -0.003TgCeq y[-1] from 1994 to 1999 and 0.03TgCeq y[-1] from 2000 to 2011 (Figure 2). The negative trend in average rate of C-based inputs in the late 1990s was an indication of the impacts of regional instability on investment in production inputs, as this was the result of the border war with Eritrea during the same period (Kalewongel, 2008). However, the trend became successively positive starting from 2003, as there was relative stability in the country and hence investment in agricultural inputs increased (Figure 2).
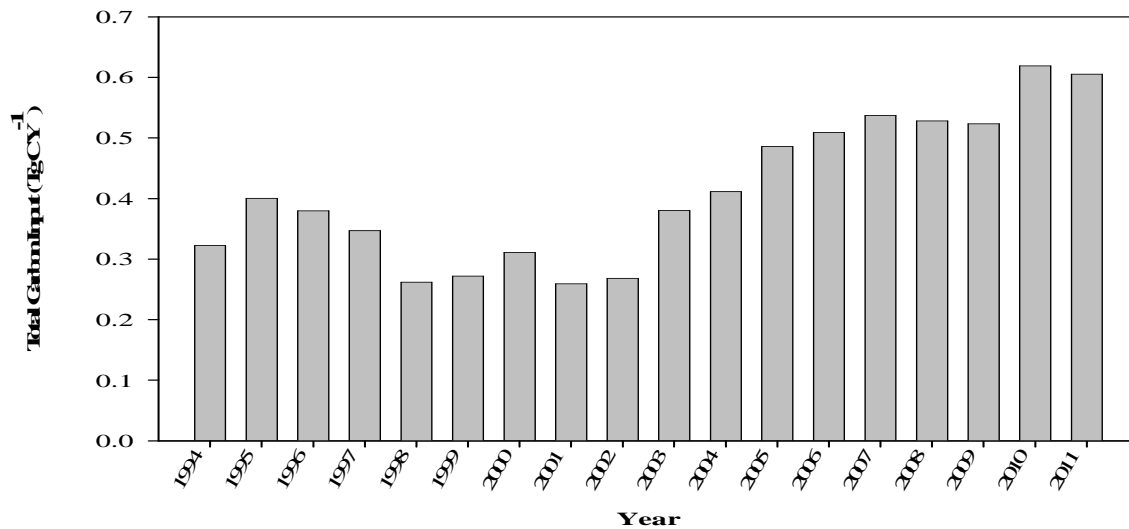

Figure 2 Trends in total C-based inputs to major cereal crops under smallholder production systems in Ethiopia from 1994 to 2011.

Carbon Equivalence of Outputs

Carbon-based output for this study included an aggregation of C present in grains, roots and straw biomass of the six predominant cereal crops (teff, wheat, barley, maize, sorghum and finger millet). These staple cereal crops are grown in Ethiopia in varying proportions according to soil type, altitude, and the prevailing climatic and market conditions (FAO, 2006). Total C output increased linearly from the lowest 5TgCeq y[-1] in 1994 to the highest 17TgCeq y[-1] in 2011 (Figure 3). The relatively lower C-output (Figure 1) during the late 1990s and early 2000s (1998-2002) were linked with the lowest C-based inputs to the system during those years (Figure 1). The 6-year average rate of increase in C-output from 1994 to 1999 was marginal, i.e., 0.3TgCeq y[-1]. However, its 11-years (2000 to 2011) average rate of increase was relatively higher at 0.8TgCeq y[-1].
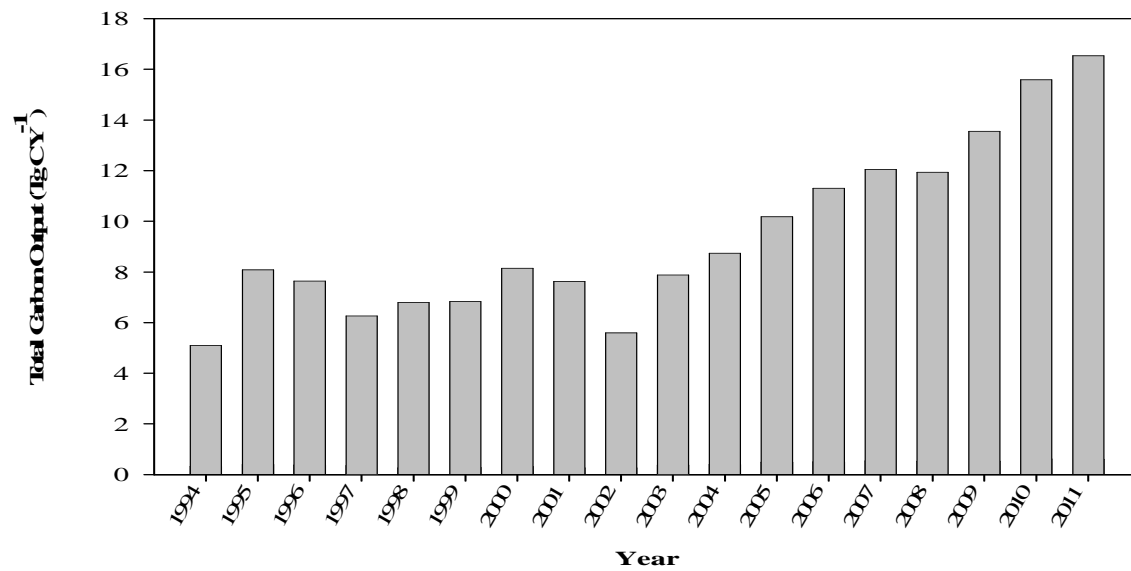
Figure 3 Temporal Changes in outputs of major cereal crops under smallholder production systems in Ethiopia from 1994 to 2011, expressed in C-equivalents

Carbon Sustainability Index (CSI)
The CSI was relatively low during the first four years, the lowest at 15 in 1994 (Figure 4). Its increase from 1998 to 2001 with the highest value at 28 in 2001, indicated the minimum usage of inputs (e.g., fertilizer) (Figure 1). However, CSI decreased between 2002 and 2008 from the preceding years and remained in the range of 20 to 22 (Figure 4). The CSI increased again in the last three years of the study period to 26 in 2011. Generally, CSI of the smallholder cropping system in Ethiopia is comparable with other systems in other regions, with an 18-year range of 15-28. For example, CSI of the agricultural system in the state of Ohio, USA, between the 1950s and 1980s was in the range of 20 to 27, and that of Punjab, India, during the 1970s and 1980s was in the range of 15 to 30 (Dubey and Lal, 2009).
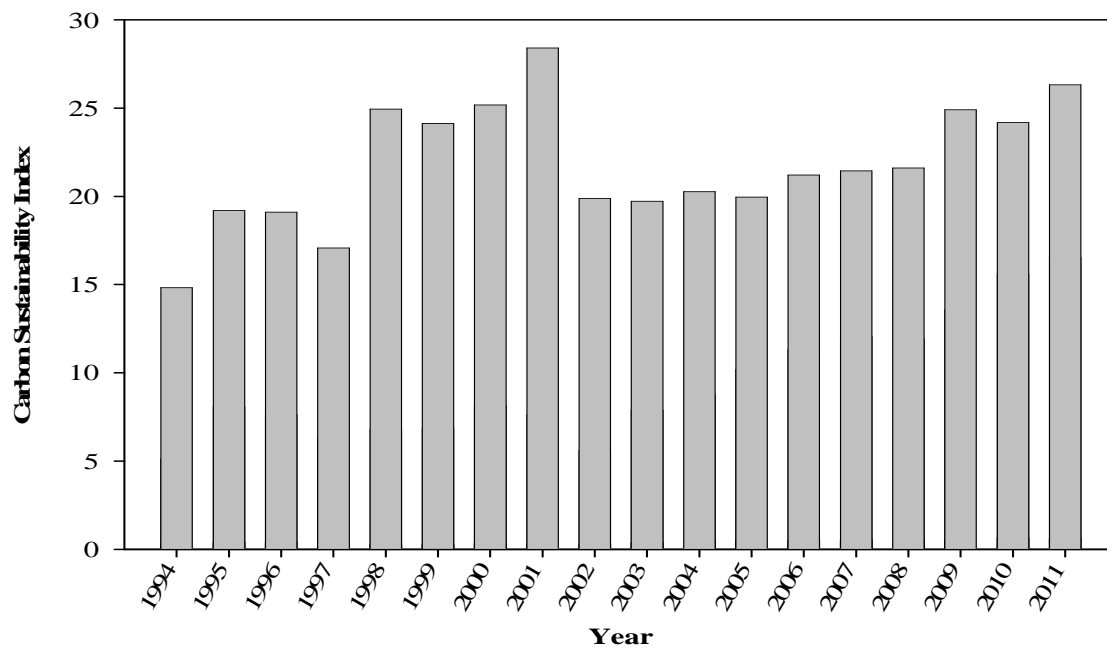

Figure 4 Trends in Carbon Sustainability Indices (CSI) of smallholder cereal-production systems in Ethiopia between 1994 and 2011.

Carbon Output vs. Carbon Input

The relationship between annual total C-based input and total annual C-output was strong ($P < 0.001$) with r^2-value of 0.85 (Figure 5). This positive relationship can also be clearly seen above (Figures 2 and 3); both, inputs and outputs, showed increases in the first few years of the study period. Similarly, decreasing trends were observed in both, inputs and outputs, in late 1990s and early 2000s. Further, they showed steady increases throughout the study period after 2003 (Figures 2 and 3). This showed that there exists a scope for further improvement in yields of crops with increased use of inputs.
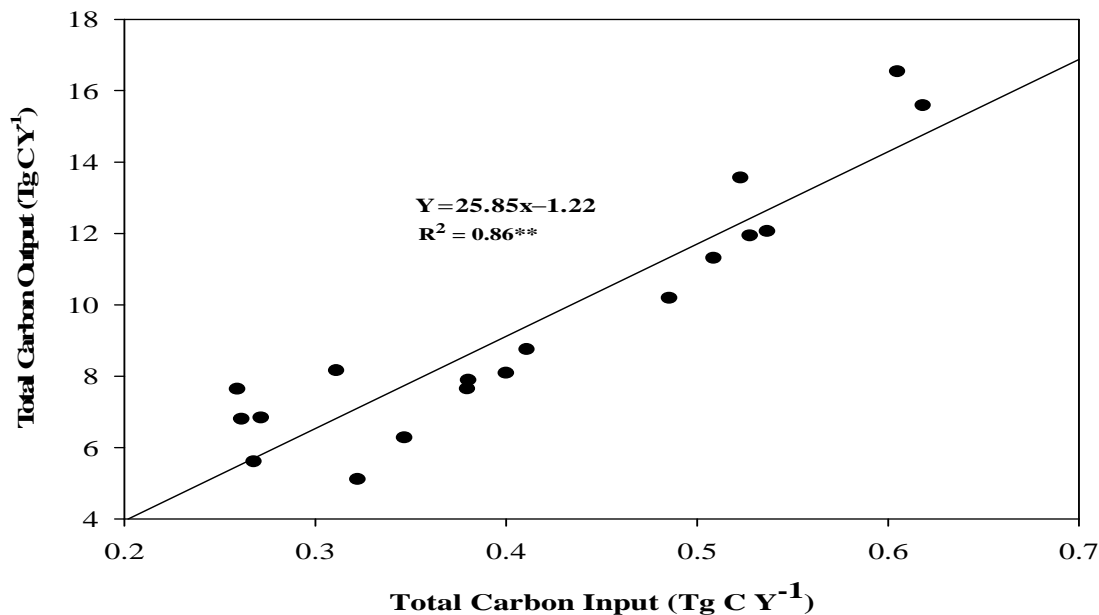
Figure 5 Relationship between total C-input and total C-output in Ethiopian smallholder cereal-production systems from 1994 to 2011.

Conclusion and Recommendations

The relationship between annual total C-based input and total annual C-output was strong. The result also indicated that C-sustainability index of the farming systems in Ethiopia was comparable with intensive systems in other regions. However, national average cereal grain yield gaps are still large compared with other developing regions and some managed experimental plots in the country. Moreover, fertilizer-use efficiency showed no major improvements with further increase in applications of N and P fertilizers. As demonstrated by studies reported elsewhere, higher returns from modern inputs are obtained if fertilizer, improved seeds and other recommended practices are all applied together. However, supply of improved seeds and use of irrigation have lagged behind at <5% and <1%, respectively. Thus, horizontal expansion in area has been the major source of recent production increases in the country. However, this trend raises questions about the quality of the land brought under cultivation and the sustainability of the process. Therefore, to narrow the yield gaps and to increase total production without increasing the area, Ethiopian farmers will need to use more fertilizer, improved seeds and irrigation. On the other hand, given the very high costs of modern inputs (e.g., fertilizer and improved seeds) for most smallholder farmers in the country, the transition to more intensive farming may initially be restricted to high-potential areas where the yield returns are the highest. Otherwise, subsistence farmers can be motivated through payments for ecosystem services.

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Land use impact on soil organic carbon and total nitrogen storage in a typical dry land district in Tigray, northern Ethiopia

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Abstract
Soil serves to store and cycle soil organic carbon (SOC) and total nitrogen (TN), which are essential for functioning terrestrial ecosystems. We measured soil organic carbon (SOC) and total nitrogen (TN) concentrations and stocks in three soil depths (0–15, 15–30, and 30–50 cm) for four different land uses, namely, rainfed cultivation (RF), agroforestry (AF), open pasture (OP), and silvopasture (SP), with five replications within a watershed in Ethiopia. OP land use showed higher SOC concentration in the 0–15 cm layer. The highest SOC concentration (12.6 g kg^{-1}) in 0–15 cm depth was found in OP land use system. Except for SP (8.6 g kg^{-1}), it was significantly higher ($p < 0.001$) than those in other land use systems. The concentration of TN across land uses in different depths followed a trend similar to that of SOC. Thus, the highest TN concentration in 0–15 cm layer in OP (1.1 g kg^{-1}) was significantly higher ($p < 0.01$) than that in RF land use. OP also had significantly higher ($P < 0.05$) SOC and TN stocks in the 0–50 cm depth than those in RF. The results of this study suggest that conversion of RF into grass and tree-based land uses has large technical potential for SOC and TN sequestration.

Keywords: soil organic carbon, total nitrogen, carbon sequestration, land use, Ethiopia

Introduction
Climate change has the potential to modify existing ecosystem functions in diverse ways, including both the enhancement and reduction of crop yields and production. These impacts are potentially profound in the areas of the world that are the most vulnerable. Sub-Saharan Africa contains some of these vulnerable systems (Vagen et al. 2005; Tieszen et al. 2004). Africa's major role in the global carbon cycle can be attributed to the substantial release of carbon associated with land use conversions from forest or woodlands to agriculture (Smith 2008). Land management following conversion also impacts carbon status, soil fertility, and agricultural sustainability (Ringius 2002); Tieszen et al. 2004; Lal 2006). Soils often continue to lose carbon following land conversion to agriculture (Woomer et al. 2004; Tschakert et al. 2004; Liu et al. 2004). Further, carbon in soil is closely coupled to soil nitrogen and the continued mining of soil for crops or fuel without replenishment of nutrients results in decreased productivity and impacts food security. However, carbon and nitrogen stocks can be replenished with combinations of residue retention, manure addition, nitrogen fertilization, agroforestry, and conservation practices (Lal 2006).

Several decades of massive deforestation of natural forests and extensive use of agricultural lands in Ethiopia have resulted in soil and environmental degradation (Ashagrie et al. 2005). In Tigray, northern Ethiopia, large parts of the area were once covered with acacia woodland *(Acacia etbaica* (Sch.) and *Faiderbhia albida* (Del.) (Eweg et al. 1998). The early 1960s marked the disappearance of much woodland under pressure from the rapidly growing population (Eweg et al. 1998). Nowadays, farmers selectively take care of naturally growing *F. albida* trees in and around their farms and grazing lands in order to improve soil fertility and increase crop and pasture yields (Hadgu et al. 2009; Gelaw et al. 2013). Communal grazing lands are important land use systems in the region, as livestock is a vital source of livelihoods for poor farmers. Conversion of cropland to grassland is one of the most effective strategies for carbon sequestration (Lal et al. 1999; Smith et al. 2000). Soil retention of soil organic carbon (SOC) and total nitrogen (TN) can be characterized by short-term storage in macroaggregates or by long-term sequestration in microaggregates.

Land use type affects amount and quality of litter input, their rates of decomposition and processes of organic matter stabilization in soils (Römkens et al. 1999). The role of land use in stabilizing CO_2 levels and increasing carbon (C) sink potentials of soils has attracted considerable scientific attention in the recent past (Kumar and Nair 2011; Murthy et al. 2013). In Ethiopia, very few studies have been conducted on SOC and TN storage capacities of soils with different land uses. Therefore, the principal objective of this study was to assess the effects of different land uses on soil organic carbon (SOC) and total nitrogen (TN) retention potentials of soils. In doing so, we determined the SOC and TN concentrations and stocks in soils under four different land uses, rainfed cultivation (RF), agroforestry (AF), open pasture (OP), and silvopasture (SP).

2. Materials and Methods

The study was conducted in the *Abraha-Atsbaha* district in eastern Tigray, northern Ethiopia. Geographically, it is located between 13°83'00 N to 13°85'00 N latitude and 39°50'00E to 39°53'00E to 55°60'00E longitude. The study covered an area of about 10 km^2 and is found at an elevation of 1960 to 2000 M.A.S.L. The average daily air temperature of the area ranges from around 15°C in winter to 30°C in summer. The mean annual rainfall of for this region is 558 mm, but the inter-annual variation is substantial. Soils in the watershed are classified as Arenosols and an association of Arenosols with Regosols, according to the World Reference Base for soil resources (WRB 2006). These soils developed from alluvial deposits and Adigrat sandstones. The texture of these soils are dominated by sand, loamy sand, and sandy loam fractions, and their pH ranges from 6.8 to 7.9 (Rabia et al. 2013). Major land uses in the watershed include *F. albida* based agroforestry (27.7 ha), rainfed crop production (11.9 ha), open pasture (23.2 ha), and *F. albida*-based silvopasture (11.7 ha). The most commonly practiced agricultural rotation in the agroforestry and rainfed cultivation land use systems is maize (*Zea mays*)-teff (*Eragrostis tef*)-field beans (*Vicia faba*)-finger millet (*Eleusine coracana*). Fallow is not practiced in the area due to population pressure and scarcity of farmlands. Use of chemical fertilizers is minimal and land is prepared for cultivation by using a wooden plow with oxen. Crop residues and manures are used for animal feed and fuel respectively. No pesticides and other agricultural chemicals are used in the area.

Faidherbia albida trees in and around farmlands and grazing lands are remnants from the original woodland in the region. These trees are selectively left by farmers to improve soil fertility and increase crop and pasture yields (Hadgu et al. 2009). Grazing lands/open pastures and silvopastures are commonly owned and managed by communities (Figure 1). Trees are integrated into the pasture systems (silvopastures) on the peripheral marginal areas where soil fertility levels are estimated to be low, based on performance of grazing lands as perceived by farmers (personal communication with farmers). No external inputs such as fertilizer are used and no rotational grazing or other kinds of management are practiced in both the open pasture and silvopasture systems. However, grazing pressure will be higher in these systems only during cropping seasons which are from June to October. Otherwise animals graze freely all over the different land use systems. Mixed crop-livestock smallholder farming is a typical farming system of the region.
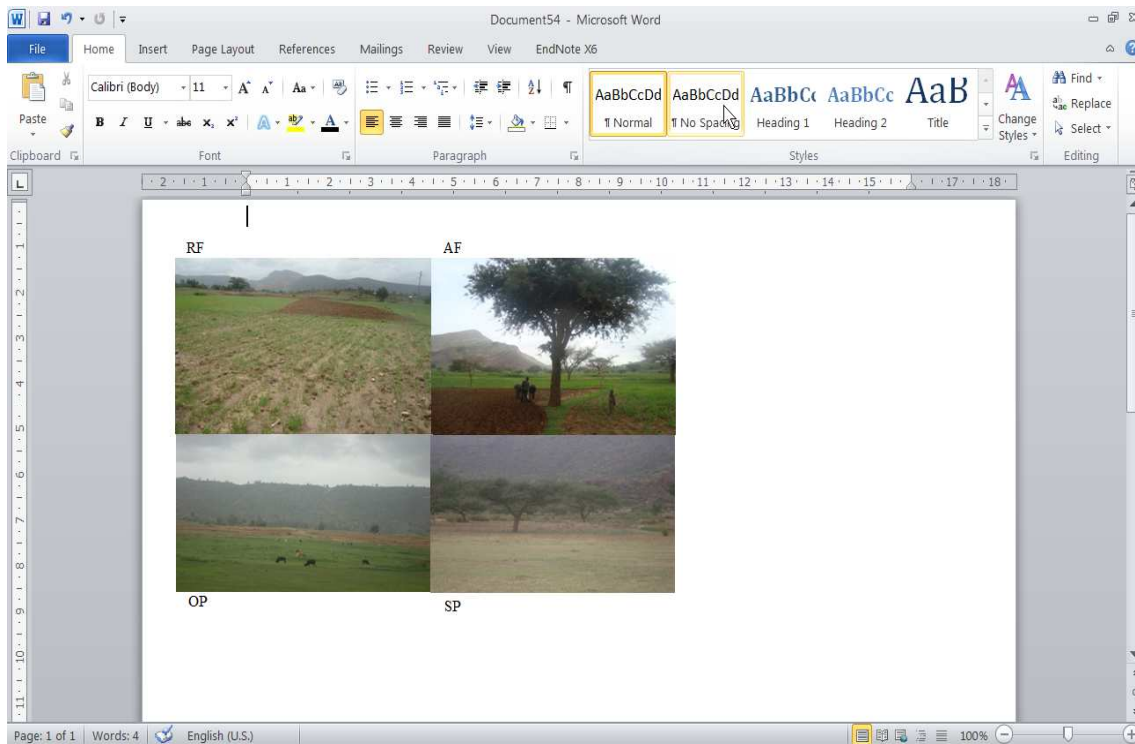

Figure 1 Different land use systems in the district with tree-less rainfed crop production (RF), *F. albida* based agroforestry (AF), open communal grazing/pasture land (OP) and *F. albida* based silvopasture (SP).

2.1 Soil sampling and analysis

A total of 60 composite soil samples were collected for SOC and TN measurements in three soil depths (0–15, 15–30, and 30–50 cm). Soil samples within each replicate were collected randomly from eight points within a 64 m^2 area at each sampling site/replicate and were well mixed and combined into a composite sample by depth. Thus, a minimum of 40 point samples were represented in computing the average values of each soil parameter. Samples were air-dried, gently ground, and passed through a 2 mm sieve. Identifiable crop residues, root material, and stones were removed during sieving. Soil samples for carbon (C) and nitrogen (N) analyses were also pulverized using a ball-mill grinder.

Finally, SOC and TN (% w/w) concentrations in composite samples were determined by the dry combustion method (Nelson and Sommers 1996). Soil bulk density (ρ_b) samples were taken for the same depth intervals as other soil samples for each replicate/plot using the core method (Blake and Hartge 1986). Core samples were collected from all depth intervals using 100 cm^3 volume stainless steel tubes (5 cm diameter and 5.1 cm height). The initial weight of soil core from each layer was measured in the laboratory immediately after collection. Simultaneously, soil moisture content was determined gravimetrically by oven drying the whole soil at 105^oC for 24 hours to calculate the dry ρ_b. No adjustment was made for rock volume because it was rather minimal. Soil ρ_b value was used to calculate the SOC and TN stocks (Mg ha^{-1}) using the model by Ellert and Bettany (1995):

$$\text{SOC (or TN) Stock} = \text{Conc} * \rho b * T * 10\ 000\ m^2ha^{-1} * 0.001\ \text{Mg kg}^{-1}$$

Where, SOC (or TN) stock = soil organic carbon or total nitrogen stock (Mg ha^{-1}); Conc. = soil organic carbon or total nitrogen concentration (kg Mg^{-1}); ρ_b = dry bulk density (Mg m^{-3}), and T = thickness of soil layer (m)

The SOC (or TN) stock in the 50 cm depth for each land use was calculated by summing SOC (or TN) stocks in the 0–15, 15–30 and 30–50 cm depth intervals. Accumulation of SOC (or TN) stock in the same soil depth (50 cm) for each land use was estimated by calculating the difference in SOC (or TN) stock between each one of the three land uses (AF, OP, and SP) and the control (RF) land use. Rate of accumulation of SOC (or TN) stock for 0–50 cm layer for each of the three land uses (AF, OP, and SP) was estimated by dividing accumulation values by the assumed duration of each land use (Puget and Lal 2005). Based on the survey of farmers conducted, an average duration of 50 years was taken as the age for OP, AF, and SP land use adoption since the early 1960s (Eweg et al. 1998).

2.2 Statistical analysis

The effects of different land use systems on SOC, TN, and WSA were subjected to one-way ANOVA. Differences between means of treatments were considered significant at $p < 0.05$ in the Tukey studentized (HSD) test. The statistical analysis system (SAS) software package was used for statistical analyses (SAS 2007).

1. **Results and Discussion**

3.1 Effects of land use on soil organic carbon and total nitrogen concentrations

The highest SOC (12.6 g kg^{-1}) concentration in the 0–15 cm layer was measured in the OP land use system, and, except for SP, it was significantly higher ($P < 0.001$) than that in other land use systems (Table 1). OP land use system also showed the highest TN concentration in the 0–15 cm layer, but it was significantly higher ($P < 0.01$) than that in the RF land use system (Table 1). However, SOC and TN concentrations in 15–30 and 30–50 cm layers did not differ among land uses. These results indicated that SOC and TN concentrations in soils of the region can be increased by converting arable lands to grasslands and silvopastures or adopting no-till and reduced tillage practices called conservation tillage. These conclusions are also in agreement with Lal (2002) who observed that with other factors remaining the same, grazing land soils have more SOC than cropland soils because of (1) low soil disturbance due to lack of plowing, (2) more root biomass and residue returned, and (3) return of cattle dung and manure. Moreover, concentrations of SOC and TN in AF were slightly higher than those in RF, indicating that the adoption of agroforestry systems could enhance SOC and TN concentrations. With adequate management of trees in arable and grazing lands, a significant fraction of the atmospheric CO_2 could be captured and stored both in plant biomass and soils (Albrecht and Kandji 2003).

Relatively higher concentrations of SOC and TN were measured in upper soil layers than in lower ones in all land uses except in RF (control), which was uniformly low across depths except a sign of slight increase with depth (Table 1) because of the tillage effect. Tillage practices can alter the depth distribution of SOC due to mixing (Beare et al. 1997; Chen et al. 2009). This agrees with the results of Trujilo et al. (1997), who reported that SOC generally diminishes with depth regardless of vegetation, soil texture, and clay size fraction. Haile et al. (2008) also observed a declining trend in SOC concentration with depth in silvopastural systems in Florida. Higher concentrations of SOC and TN in upper soil layers in OP and SP land uses indicated a risk of large amounts of CO_2 release from the surface soil when these land uses are converted into an arable land use.

Table 1. Soil Organic Carbon (SOC) and Total Nitrogen (TN) Concentrations across Different Land Uses

Land Use	Soil Organic Carbon Concentration (g kg^{-1})			Total Nitrogen Concentration (g kg^{-1})		
	0-15 cm	15-30 cm	30-50 cm	0-15 cm	15-30 cm	30-50 cm
RF	3.2(0.7)[c]	3.9(0.4)	4.1(0.4)	0.3(0.10)[b]	0.4(0.03)	0.4(0.01)
AF	6.4(0.3)[bc]	4.8(0.3)	3.8(0.3)	0.7(0.02)[ab]	0.5(0.03)	0.4(0.02)
OP	12.6(1.2)[a]	8.8(2.7)	6.3(1.9)	1.1(0.14)[a]	0.9(0.3)	0.6(0.2)
SP	8.6(2.0)[ab]	6.6(2.3)	5.2(2.1)	0.8(0.2)[ab]	0.5(0.2)	0.3(0.2)
		NS	NS		NS	NS

RF: Dryland crop production; AF: *Faidherbia albida* based agroforestry; OP: communal open grazing/pasture; SP: *Faidherbia albida* based silvopasture.

± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, P = 0.05).

3.2 Amount and rates of soil organic carbon and total nitrogen accumulation under different land uses

Total SOC stock in 0–50 cm depth under OP (64.2 Mg ha^{-1}) was significantly higher (P < 0.05) than that in RF (29.4 Mg ha^{-1}). However, it did not differ from other land uses (Table 2). Similarly, TN stock in the same depth under OP land use (6.0 Mg ha^{-1}) was significantly higher (P < 0.05) than that in RF (2.8 Mg ha^{-1}) land use. No significant difference in total TN stock was observed among other land uses (Table 2). The results were in agreement with the findings of Mekuria et al. (2009) who reported 36–50% increase in mean SOC stock through conversion of degraded grazing lands to exclosures, areas closed from human and animal interference to promote natural regeneration of plants on formerly degraded communal grazing lands, in Tigray, Northern Ethiopia. Similarly, Omonode and Vyn (2006) reported higher TN and SOC stocks in grasslands than croplands in west-central Indiana, USA. Mclauchlan et al. (2006) also reported similar results, when agricultural lands of the northern Great Plains depleted in soil organic matter (SOM) by decades of cultivation were changed to grasslands through federal conservation programs.

Among the three grass- and tree-based land use systems (OP, SP, and AF), OP accumulated the highest SOC stock (34.8 Mg ha^{-1}) followed by SP (20.0 Mg ha^{-1}), and AF (7.2 Mg ha^{-1}) land uses (Table 2). Similarly, the highest accumulation of TN stock was measured in OP (3.2 Mg ha^{-1}) followed by those in AF (1.1 Mg ha^{-1}) and SP (1.0 Mg ha^{-1}) land uses. However, there were no statistically significant differences in SOC and TN stock accumulations among land uses (Table 2). Compared with open (treeless) pasture systems, silvopastural agroforestry systems that integrate trees in to pasture production are likely to enhance SOC sequestration, especially in deeper soil layers (Haile et al. 2008). However, the results of the present study indicated that OP land use had higher SOC accumulation rates than the SP systems because those trees were integrated into the pasture system in the peripheral areas where fertility of the soil was low. These results are also in agreement with the findings of Puget and Lal (2005), who observed that pasture soil had more SOC stock than that in forest soils in the top layer of Mollisols in central Ohio, reflecting the larger grass root density in the layer.

The highest rate of SOC stock accumulation was measured in OP land use (0.70 Mg C ha^{-1} yr^{-1}) followed by SP (0.40 Mg C ha^{-1} yr^{-1}). The lowest rate of SOC stock accumulation was measured in AF (0.14 Mg C ha^{-1} yr^{-1}) land use (Table 2). However, the effect of land use change on rate of SOC stock accumulation did not statistically differ among land uses (Table 2). Similarly, the highest rate of TN stock accumulation was also measured in OP land use (0.07 Mg N ha^{-1} yr^{-1}) followed by SP (0.02 Mg N ha^{-1} yr^{-1}) and AF (0.02 Mg N ha^{-1} yr^{-1}) land uses. However, no significant difference was observed in the rate of TN stock accumulation among land uses. The result of this study was comparable to the findings of Girmay et al. (2008). They estimated rates of soil carbon sequestration (SCS) potential in currently degraded soils in Ethiopia under rangeland, irrigation, and rain fed cropping land uses over the next 50 years if there is widespread adoption of soil-specific restoration measures in the order: 0.3–0.5, 0.06–0.2, and 0.06–0.15 Mg C ha^{-1}y^{-1}, respectively. On the other hand, the higher amounts of SOC stocks in soils under grasslands and silvopastures indicates a risk of large amounts of CO_2 release if these land uses are converted to croplands. In agreement with this assertion, Fantaw et al. (2006) in their study on southeastern Ethiopia, indicated that on average 40–45 % of SOC stock in 1 m depth of mineral soils was held in the top 30 cm, indicating the risks of large amounts of CO_2 release following deforestation and conversion into Agroecosystems.

Table 2. Magnitude and Rates of Soil Organic Carbon and Total Nitrogen Stocks Accumulation in Four Different Land Uses in 0–50 Cm Depth in 50 Years

Land use	SOC Stock	SOC Accumulation (Stock-RF)	Rate of SOC Accumulation	TN Stock	TN Accumulation (Stock-RF)	Rate of TN accumulation
	(Mg C ha^{-1})	(Mg C ha^{-1})	(Mg C ha^{-1}yr^{-1})	(Mg C ha^{-1})	(Mg C ha^{-1})	(Mg C ha^{-1}yr^{-1})
RF	29.4(3.5)[b]	-	-	2.8(0.2)[b]	-	-
AF	36.6(1.5)[ab]	7.2(4.7)	0.14(0.1)	3.9(0.2)[ab]	1.1(0.2)	0.02(0.00)
OP	64.2(11.8)[a]	34.8(11.0)	0.70(0.2)	6.0(1.5)[a]	3.2(1.4)	0.07(0.03)
SP	49.4(14.9)[ab]	20.0(14.0)	0.40(0.3)	3.8(1.3)[ab]	1.0(0.6)	0.02(0.03)
		NS	NS		NS	NS

RF: Dryland crop production; AF: *Faidherbia albida* based agroforestry; OP: communal open grazing/pasture; SP: *Faidherbia albida* based silvopasture.

± Column mean values followed by standard errors in the parentheses; values with different letters are significantly different. NS = not significant (Tukey's test, P = 0.05).

4. Conclusion

Higher SOC and TN concentrations were measured in the top layers of soils under grass- and tree-based land use systems, OP, and SP. Furthermore, soils under these land use systems had higher SOC stocks than that in RF in the 0–50 cm depth. This is an indication of that both OP and SP land use systems received more biomass inputs from grass and tree residues than the cultivated soils under AF and RF, as the amount of plant residues and the degree of SOM decomposition are vital factors in the formation and stabilization of organo-mineral complexes, which are called aggregates. Thus, the adoption of tree- and grass-based and other restorative land uses such as no-till farming and other recommended less intensive cultivation practices can result in carbon accumulation, stabilization, and sustainable use of soil resources in the region.

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