The Potential of Carbon Sequestration to mitigate against climate change in forests and agro ecosystems of Zimbabwe

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DEDICATION

To my husband ,Freddy, my mother, Estery and in loving memory of my son Simbarashe.

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Introduction

Chapter 1

1.1 Background

Developing countries are more vulnerable to climate change because a large proportion of their population directly depend on agricultural and natural ecosystems for their livelihoods. Agricultural systems should overcome three simultaneous challenges of: ensuring adequate food for a growing population, increasing small holder farmer's resilience to changing climate and reduction of greenhouse gas emissions. Globally, agro-ecosystems can compensate C emissions by one third through the adoption of mitigation strategies such as addition of biomass to soils, minimal soil disturbance and soil conservation (Cole *et al.*, 1997). Increasing soil C storage has therefore rendered soils a major focal point because of their ability to accumulate significant quantities of organic C (Banger *et al.*, 2009; Food and Agriculture Organisation (FAO), 2010a). Soil management strategies therefore become an important C mitigation approach through mitigation measures involving both CO_2 emissions reduction and increasing C sinks. Global climate change has been linked to increased concentrations of greenhouse gasses (GHGs); carbon dioxide (CO_2), methane (CH4) and nitrous oxide (N_2O) (IPCC, 2007), the most important GHG being CO_2. The concentrations of the Earth's atmospheric CO_2 is controlled by interaction, the structure and dynamics of terrestrial ecosystems, climate, oceans and anthropogenic CO_2 emissions (Foley *et al.*, 2003). The amount of CO_2 in the atmosphere depends on its interactions with the Earth's surface as CO_2 diffuses into oceans or is absorbed by plant photosynthesis. Some of the C absorbed by plants returns to the atmosphere through autotrophic respiration, while the rest is converted to biomass, which eventually becomes litter and converts into soil organic carbon (SOC).

SOC is a heterogeneous mixture of simple and complex organic C compounds derived from biomass and is also linked to soil quality and productivity (Lal, 1986). SOC sequestration is a major sink for atmospheric CO_2 (IPCC., 2000) with the global soil C pool (1550 Pg) being three times greater than the biotic pools (560 Pg) and twice the atmospheric C pool (760 Pg) (Johnson & Curtis, 2001; Lal, 2008). SOC is also slowly returned to the atmosphere through heterotrophic (microbial) respiration. Photosynthesis and respiration processes thus enable terrestrial ecosystems to influence the amount of CO_2 in the atmosphere. During these processes, soils play a key role in global C and nitrogen (N)[1] cycling (Food and Agriculture Organisation (FAO), 2010a) resulting in storage of approximately 75% and 95% of terrestrial C and N respectively (Eswaran *et al.*, 1993; Schlesinger, 1997). CO_2 emissions can thus partly be compensated by creating or promoting carbon (C) sinks, such as increasing biosphere sinks (IPCC, 2007).

Agricultural soils can contribute approximately 89% of GHG mitigation potential through C sequestration with an additional 2% and 9% mitigation potential for N_2O and CH_4 respectively (Smith *et al.*, 2007). This results in estimated emissions reduction of 5-14% over 5 -10 decades (Chan *et al.*, 2008) with the agricultural systems potentially storing about 1400-2900 Mt CO_2eq annually. However, mitigating GHG emissions from agriculture without compromising food security remains a major societal and scientific challenge.

Maximisation of the mitigation potential requires adequate knowledge of land use and management practices associated with improved crop production and reducing degradation (Lal, 1997; Lal *et al.*, 1998; Singh & Lal, 2005). There is need to complement this with increasing knowledge and understanding of how soils respond to changing environmental conditions (Powlson *et al.*, 2010) and land management practices.

The role of planting trees is also interesting, as tree growth sequesters C from the atmosphere into biomass and soil (Harmon *et al.*, 1990). There is, unfortunately insufficient information on changes in soil C storage after establishing plantation forests (Scott *et al.*, 1999) yet the Kyoto Protocol of the United Nations framework Convention on Climate Change (UNFCCC) allows mitigation of GHG emissions through reducing deforestation and stimulating afforestation and reforestation. Afforestation is planting of new forests on land that has not supported forests in the last fifty years whereas reforestation is planting trees on land that supported forests in the past fifty years but that has

[1] Nitrogen is an important nutrient that is essential for plant growth and soil microorganisms affecting photosynthesis, decomposition, C and nutrient cycling .

been converted to non-forested land (Brown *et al.*, 1986). Soil C changes occurring during afforestation, reforestation or deforestation are considered under Article 3.3 of the Kyoto Protocol, while soil C sequestration in croplands, grazing lands, managed forests and land subjected to revegetation are considered under Kyoto Article 3.4. As C is stored under those circumstances, C sequestration processes occur and these processes can be considered as a climate change mitigation strategy. The potential of REDD+ in woodlands and savannahs can be achieved by the recovery of the woodlands after clearing since most of the woodland species have extensive rooting systems that facilitate recovery after cutting (Mistry, 2000). For example, data on primary production and soil carbon storage indicate that miombo woodlands can sequester 900-1600 g m^{-2} yr^{-1} (Frost, 1996). In addition, re-growth stands are highly productive ecosystems with higher growth rate (4.4 - 5.6 mm) than uncut stands (2.3 - 4.8 mm) having high rates of photosynthetic processes and therefore high uptake of carbon dioxide. C emissions as well as offsets through C sequestration in all systems have to be reported to the UNFCCC as part of national greenhouse gas inventories (IPCC, 2006). Furthermore, the contribution of forests to climate change has been recognized as a cornerstone of the post- 2012 climate change agenda with the decision on the reduction of emissions from REDD+ in COP-16 in Cancun. REDD+ includes policy approaches and positive incentives on issues relating to reducing emissions from deforestation and forest degradation in developing countries and recognises the contribution of conservation, sustainable management of forests and enhancement of forest carbon stocks in achieving REDD+ objectives.

The important role of soil as a C source or sink and the role in offsetting atmospheric CO_2 concentrations creates a need for accurate evaluation of the effects of agricultural management practices on soil C and N storage. Furthermore, understanding the soil carbon dynamics under forest plantations (mainly pine) and evaluating the C and N storage potential over an age sequence is important. There is a clear need for research into the consequences of any land-management activities in order to determine and understand their role in global nutrient cycles. This chapter outlines: the importance of C and N in agro ecosystems and the effects of land management practices on C and N storage (section 1.2), soil C and N fractionation and stabilisation (section 1.3), importance of modelling SOC in agro ecosystems (section 1.4), Zimbabwean land use information (section 1.5) and the scope, objectives and outline of the thesis (sections 1.6 and 1.7).

1.2 Carbon and Nitrogen Cycling in Agricultural Systems

Soil organic matter (SOM) comprise all dead organic material within the soil or its surface (Baldock & Skjemstad, 2000) and plays a fundamental role in soil processes. Soil organic matter dynamically determines soil productivity and affects essential nutrient fluxes of C, N, phosphorous (P), sulphur (S) and other nutrients facilitated by microbial decomposition. The quality and quantity of SOM and its position in the soil determines its resistance to microbial decomposition. Next to being beneficial to soil biological, chemical and physical properties, SOM is sensitive and responsive to (changes in) land management activities. Above and belowground plant residues (i.e. dead or decaying biomass and root exudates) constitute the major source of SOM. Soil organic C and N are major components of SOM that affect soil quality very much linked to soil physical structure, water and nutrient holding capacity (Russell, 1973; Lal, 1986; Hu *et al.*, 1997; Jimenez *et al.*, 2002b) and provision of energy for soil organisms (Jimenez *et al*., 2002b). SOC is the C content of SOM, which is approximately 50% and is largest in litter and organic layers.

Generally, soils have a SOC rich topsoil followed by decreasing SOC content with increasing soil depth. SOC is either reported as concentration (i.e. mass of C per unit mass of soil; g kg^{-1}) or as stock or density expressed on area basis (Kg m^{-2} or Mg ha^{-1}). Estimation of C density requires data on soil bulk density, stone content and depth of sampling.

Batjes and Sombroek, (1997) showed that 23% of the plant residues convert into SOC for a native pristine prairie while de Moraes Sà and Séguy (2008) found rates of 15-26% conversion in agricultural systems. In semi-arid areas, the conversion rate of plant residues into SOC is 29% under continuously cropped systems but only 9% for systems that are regularly left under fallow (Campbell *et al.*, 2000). The conversion rate is thus affected by factors such as moisture, temperature, crop species and/type and the amount of residues. Removal of crop residues generally decreases SOC levels in cropping systems (Unger *et al.*, 1990). In agro-ecosystems, the cycles of C and N are linked through pools in crops, crop residues and in SOM (Figure 1.1). The accumulation of plant biomass is in turn

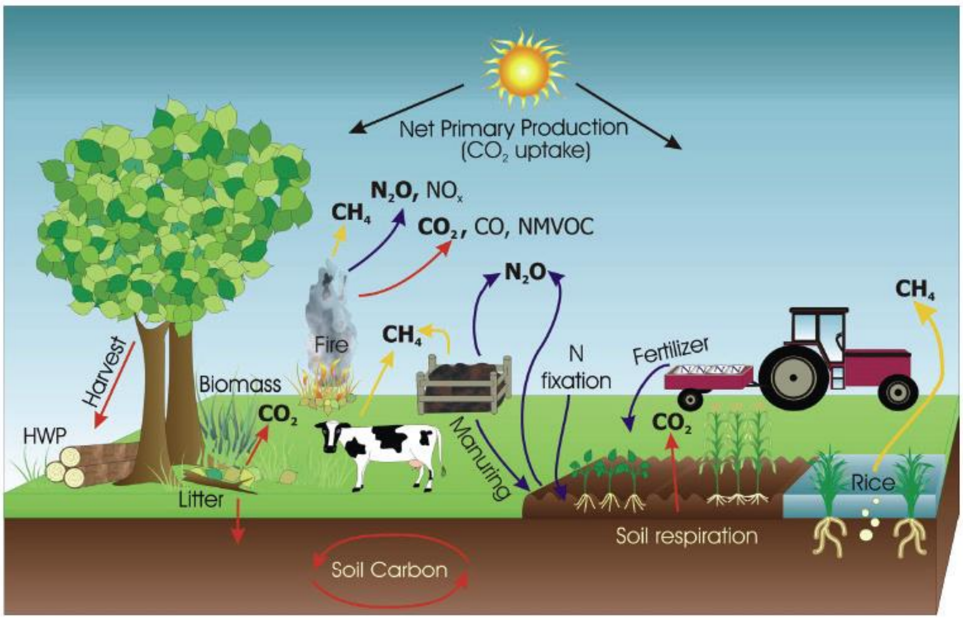

Figure 1.1 Greenhouse gas emission sources/removals and processes in agro ecosystems (IPCC, 2006).

related to the amount of soil available N (Vlek *et al.*, 1981). The dynamics of soil N reserves is continuously maintained by processes of mineralisation and immobilisation. Atmospheric N is found in the form of N_2 that cannot be utilised by most organisms and only enters the soil through processes of dry/wet N deposition, N fixation or through the addition of fertilisers and/or manures. Nitrogen fixation is done by N-fixing bacteria, which either live as free bacteria surviving on their own in the soil carrying out chemical reactions that change N_2 to ammonium (NH_{4+}) (Nitrogen Fixation), or as symbiotic root nodule bacteria that supply N to legumes/pod-producing plants in the family Fabaceae/Leguminosae.

Ammonium and nitrate are forms of N that can be used by most plants and are found in amino acids. However, organically bound N can be converted microbiologically into inorganic mineral forms (mineralization), leading first to formation of ammonium (NH^+_4) and possibly ending up as nitrates (nitrification) (Fowler *et al.*, 2013). Nitrification and denitrification processes in agricultural systems contribute about 0.53 Pg C equivalents of annual nitrous oxide fluxes which accounts for more than 50% of the global anthropogenic N_2O flux (Robertson, 2004). Processes such as ammonia (NH_3) volatilisation and the uptake of N by plants causes depletion of soil N whereas the return of crop residues increases the soil organic N pool (Vlek *et al.*, 1981). In addition, animals also facilitate C and N cycles after they die as their biomass is transformed into ammonium and other constituents, thus continuing the cycle.

The process of SOM transformation occurs after colonisation of dead organic matter by a range of soil organisms, as they obtain nourishment for growth. During decomposition, about half of the C in SOM is mineralised and released as CO_2 (White, 2006) while the other is either stabilised through humification or is complexed with soil minerals (Gregorich & Janzen, 1996). The C and N sequestration capacity of a soil is therefore determined by its ability to protect the SOC and N from degradation (i.e. stability) (Six *et al.*, 2002). The amounts of C stabilised in each soil under specific management conditions is important in determining the extent to which a soil can operate as a C sink. However, the ability of a soil to stabilise the sequestered C varies with physical (e.g. bulk density and clay%) and chemical properties (e.g. pH) (Borken *et al.*, 2002) of the mineral soil matrix and the structure of the SOC (Singh & Lal, 2005; Murillo *et al.*, 2006). Soils can become saturated with C resulting in no further stabilisation even with additional inputs (Six *et al*., 2002).The efflux of soil CO_2 also differs between fine-textured soils and coarse textured soils (Zak *et al.*, 1994). In order for C sequestration to be considered important for climate change mitigation, the net transfer of CO_2 from the atmosphere into the soil or biomass should be in a form that is not immediately reemitted (Powlson *et al.*, 2011).

Soil organic C levels respond linearly to C inputs (Yin *et al.*, 2005; Yin & Cai, 2006; Campbell *et al.*, 2007) with the slopes differing with quality of inputs and soil type. Theoretically, the C equilibrium level can be restored in a soil when management systems remain unaltered and is affected by factors such as soil texture, climate, vegetation type and time. These factors have a bearing on the ability of a soil to reach equilibrium level. Difference between equilibrium SOC levels and the current C stocks indicates the SOC sequestration potential of a system. In some cases, SOC sequestration in excess of the equilibrium level, can occur because of increased productivity owing to the removal of inherent soil constraints limiting plant growth (Russel & Williams, 1982). This demonstrates that changes in SOC storage over time can be manipulated by management practices through regulation of inputs and outputs. Smith *et al*. (2008) reviewed studies on potential of SOC sequestration and estimated average annual mitigation potential of all GHGs, in agricultural systems for warm-dry climates (-0.25 to 0.88 Mg C ha^{-1} yr^{-1}) and warm-moist climates (0.26 to 1.30 Mg C ha^{-1} yr^{-1}) with differences in each zone attributed to initial SOC levels, time periods and site productivity factors.

1.2.1 Effects of land management practices on C and N storage

Some management practices have been identified as suitable for organic C and N sequestration (Dalal & Mayer, 1987; Janzen, 1987; Smith *et al*., 2007) including plant residue management, tillage, manure and fertiliser application and tree planting (FAO, 2004). Several studies have shown that crop rotation and diversity, fertilisation, residue management, conservation tillage, agroforestry and the maintenance or restoration of degraded lands can increase soil C sequestration (Lal, 2004; Smith *et al*., 2007; Purakayastha *et al.*, 2008; Gong *et al.*, 2009). The extent of the influence of each practice on soil C sequestration however, depends on land-use type, crop species, social and ecological dynamics and intensity of the management practices. Biophysical components also affect agricultural systems and their potential for C sequestration (Lal *et al.*, 1999; Whalen & Chang, 2002).

Socio-economic and demographic factors similarly affect decisions on choice of agricultural management practice (Seabrook *et al.*, 2008). In some cases, soil C storage in a particular farmer's field is affected by socio-economic factors such as the farmer's wealth category and/or sources of income (Zingore *et al.*, 2007). In small holder agro ecosystems, CO_2 emissions are not only from decomposition and plant respiration but also indirect emissions come from the production of some agricultural inputs such as fertilisers, pesticides and herbicides. Furthermore, agricultural systems also emit CH_4, through ruminant fermentation (52%), biomass burning (19%) and animal waste treatments (8%) (Robertson, 2004).

Conventional tillage, which is an ancient practice is beneficial in controlling weeds, reducing soil compaction and preparing favourable crop seedbed (Aziz, 2010). On the other hand, although impacts vary with site characteristics, conventional tillage is seen as the major cause of SOM degradation (Buschiazzo *et al.*, 2001). Decreasing soil C stocks in agricultural lands by exposing SOM to microbial decomposition and thus enhancing its decomposition and loss into the atmosphere (Lal *et al*., 1998); (Six *et al.*, 2000; Zotarelli *et al.*, 2005). Tillage also modifies soil physical conditions, such as aeration, porosity, temperature and moisture and this favours decomposition of plant residues and SOC (Drees *et al.*, 1994; Verhulst *et al.*, 2010). In addition, tillage accelerates soil erosion, which promotes transfer of soil nutrients. Furthermore, tillage alters the quality and quantities of soil organic inputs through its effects on crop growth (Doran, 1987).

Conservation tillage (i.e. no, reduced or/ minimum tillage), on the other hand, can enhance soil C sequestration and can reduce C loss and is now considered an important agricultural management practice (Six *et al.*, 2000b; Food and Agriculture Organisation (FAO), 2010a). In comparison with conventional tillage, reduced or no tillage practices in which crop residues are left on the soil surface reduce soil loss due to decreased erosion (Lee *et al.*, 1993), increase surface soil C concentration (Blanco *et al.*, 2009) and improved water use efficiency.

1.2.2 Fertilisation and SOC storage in agro ecosystems

Fertilisers have been successfully used to enhance C sequestration (Lal, 1999) although, the application of N fertilisers can result in nitrous oxide (N_2O) emissions (Robertson *et al.*, 2000) and alters the soil C and N dynamics. The main purpose of fertiliser application is to increase crop production. This is accompanied by subsequent increase in biomass and thus indirectly elevates soil C sequestration. Better results can be obtained using a combination of inorganic fertilisers and organic manure which can maintain and even improve SOC storage (Su *et al.*, 2006). In small holder farming communities, the amount and frequency of inorganic fertiliser application depends on farmer's possibilities. For example, wealthy farmers apply more inorganic fertilisers than the poor farmers who sometimes do not apply any fertilisers (Ncube *et al.*, 2009). Moreover, the education level plays a decisive role in the amount of fertiliser used.

In smallholder farmlands, livestock are an important asset and their dung is mixed with litter and used as the main source of soil amendment (Farmyard Manure (FYM) – referred as manure in this study). Manure is therefore readily available nutrient source although the quantities are often not enough. Application of manure can sequester more C than other fertility treatments over a range of soils and climatic conditions, although factors such as crop species and soil texture are also important (World Bank, 2012). Manure application shows greater C gains in clayey soils than nitrogen fertiliser alone and has the advantage of having both nutritive value being a source of organic C for improving soil tilth. In addition, manure is more resistant to microbial decomposition when compared with plant residues, resulting in greater C storage given the same quantity of C input (Jenkinson *et al.*, 1990). Furthermore, manure application increases development of particulate organic matter (Kapkiyai *et al.*, 1999) and the formation and stabilisation of soil macro-aggregates (Whalen & Chang, 2002).

1.2.3 Chronosequence approach for studying changes in soil C stock

Understanding C and N dynamics in plantation forest soils is needed in order to determine the role of trees in climate change mitigation and the long-term impacts of management activities. Studies of soil C dynamics in plantation forests often involve spatial–temporal substitution methods, such as the chronosequence approach (Covington, 1981). The chronosequence approach facilitates long-term studies of vegetation succession and soil dynamics. The length of time required to make direct investigation of forest succession makes the chronosequence approach more favourable for determination of long-term changes in soil C and N stocks due to afforestation (Covington, 1981; Wallace & Freedman, 1986; Klopatek, 2002). The chronosequence approach assumes that the expected pattern of temporal development that would be exhibited by a particular stand is reflected by the pattern observed among stands of different ages (Wallace & Freedman, 1986). Chronosequence approaches to study soils therefore facilitate the analysis of C and N variation over stand-age using space-for-time substitution and allows investigation of temporal shifts of the C and N amount over stand age.

Forest plantations are either exotic or indigenous forest stands artificially established covering a minimum area of 0.5 ha, with a tree crown cover of at least 10% and total height of mature trees above five metres (FAO, 2001). Conifer and broadleaf tree plantation species have different strategies for belowground allocation of assimilated C (Guo & Gifford, 2002). The storage of C in forest soils is also affected by site quality and land use practice (Lal, 1997). Furthermore, native and exotic trees are able to sequester and stabilise C from the atmosphere and potentially contribute to counteracting the greenhouse effect. Preferably, fast growing trees are recommended as excellent options for mitigating CO_2 emissions through soil and biomass C sequestration (Montagnini & Porras, 1998). Planting the fast growing exotic tree species substitutes requirements for various indigenous wood requirements and can facilitate the regeneration of native species. In addition, trees protect soil by means of the litter layer and leaf canopy, thereby decreasing runoff and erosion and increasing water infiltration rates. There is also a reduction of soil temperatures and improved water holding capacity under the tree canopy (Sanchez *et al.*, 1997). Root activities and organic matter inputs improve soil structure (Gardner *et al.*, 1999).

In forested systems, SOM pools include all the components that can be partitioned in pools of forest floor, above ground biomass, belowground biomass constituents, water soluble organics, light fraction material and stable humus (Stevenson & Cole, 1999). In each forest, the accumulation of C and N varies within different soil depths, having, litter-fall as the main input of C and nutrients to the forest floor, and is important for both the forest-soil system and nutrient cycling (Starr *et al.*, 2005). Forest floor consists of three layers: 1) Litter layer (L), which **c**onsists of fresh and recently fallen, non-decomposed material. This material is identifiable by the naked eye as plant residues. The litter layer usually contains less than 10% fine organic matter; 2) Fragmented layer (F), where organic material is fragmented and partly decomposed with plant residues being macroscopically recognizable; and 3) Humus layer (H), which consists of decomposed organic matter, originating from litter fall from decades ago and root turnover of variable age. The H layer materials have more than 70% is fine organic matter decomposed beyond recognition of their origin. The build-up of a H layer is dependent on the inflow of plant litter and the activity of the decomposer community. In addition the H layer is composed of a the formation and accumulation of recalcitrant or very slowly decomposing litter and microbial residues (Yanai *et al.*, 2003; Keith *et al.*, 2010).

1.3 Soil carbon and nitrogen fractions and their stabilisation

The transformation of SOM gives rise to pools ranging from very active (labile) to stable/inert/passive (non - labile) that are differentiated from each other by their degree of decomposition, recalcitrance, and turnover rate (Gregorich *et al.*, 1994). The constituents of each pool have different functional roles in the dynamics of SOM and nutrient cycling. The labile C pool consists mainly of soil organisms, polysaccharides, celluloses and hemi-celluloses with a turnover time varying from weeks to months, while the recalcitrant pool consists of lignin, lipid polymers, resins, suberins, waxes and fats with turnover time varying from years to decades. The inert pool consists of charcoal and pyrolysed C with turnover time of centuries to millennia. The amount of organic C in each fraction/pool depends on the quality and quantity of the organic matter added to the soil and types of decomposition products from biological activities.

Stability of C in each pool is related to the aggregation of soil particles facilitated by different types of binding agents. Tisdall and Oades (1980) classified binding agents into (a) Transient binding agents - comprised of microbial and plant derived polysaccharides and may be rapidly decomposed by microbes; (b) temporary binding agents - includes roots and fungal hyphae (especially mycorrhizal hyphae) and (c) persistent binding agents - consists of aromatic humic material associated with amorphous Fe and Al compounds, and polyvalent metal cations. Oades (1984) and Six *et al.,* (2000) showed that organic materials within soil aggregates had lower decomposition rates than those outside of aggregates.

The SOM pools can be separated by chemical or physical extraction methods. Chemical methods have, however, proved difficult by affecting the extraction process (Duxbury *et al.*, 1989) whereas, physical separation of SOM reduces chemical alteration of the organic materials (Golchin *et al.*, 1994a; Gregorich & Janzen, 1996; Swanston *et al.*, 2005; Liu *et al.*, 2013). The advantage of physical fractionation techniques is the isolation and detection of SOC storage and sensitivity of soil to land management and climate of a particular site.
Fractionation of SOM by density can be achieved using heavy liquids such as Sodium Polytungstate whose density ranges between 1.6 and 2.0 g cm^{-3}. Density fractionation separates SOM relative to turnover rate, structure and function (Christensen, 2001; Echeverría *et al.*, 2004; Crow *et al.*, 2007). SOM is separated into three basic functional pools of: free light fraction (fLF), occluded light fraction (oLF), and mineral associated fractions (MaHF) (Golchin *et al.*, 1994b; Six *et al.*, 2002; Norris *et al.*, 2011; Sequeira *et al.*, 2011) which vary in C and N concentration, C:N ratios and $^{\delta 13}$C over the soil profile (Figure 1.2). Crow (2011) analysed the mean residence time of the three soil C pools and found them with turnover rates of 3.5 years (fLF), 10 years (oLF), and 714 -2090 years (MaHF).

The light fraction is usually made up of two distinct fractions of fLF and the oLF, with differences between the two largely explained by the positions they occupy in soil aggregates. Though these fractions are different, most studies do not separate them and usually classify them together as the Light fraction (Six *et al*., 2002) especially when comparability is limited by a diversity of methods used to separate the two (Cerli *et al.*, 2012). Compared with the organic C stock of a whole soil sample, the fLF the oLF have higher C and N concentrations and wider C:N ratios (Golchin *et al*., 1994a; Schrumpf *et al.*, 2013).

The light fraction organic C, particulate organic C and microbial biomass C form the basis of the fLF or inter-aggregate organic matter (Yang *et al.*, 2009; Sequeira *et al*., 2011). The fLF consists of partially decomposed plant remains, root fragments, spores, seeds, faecal pellets, fungal hyphae, faunal carcasses, microbes and microbial remains at various stages of decomposition (Poirier *et al.*, 2005; Gregorich *et al.*, 2006). The light fraction serves as an easily decomposable

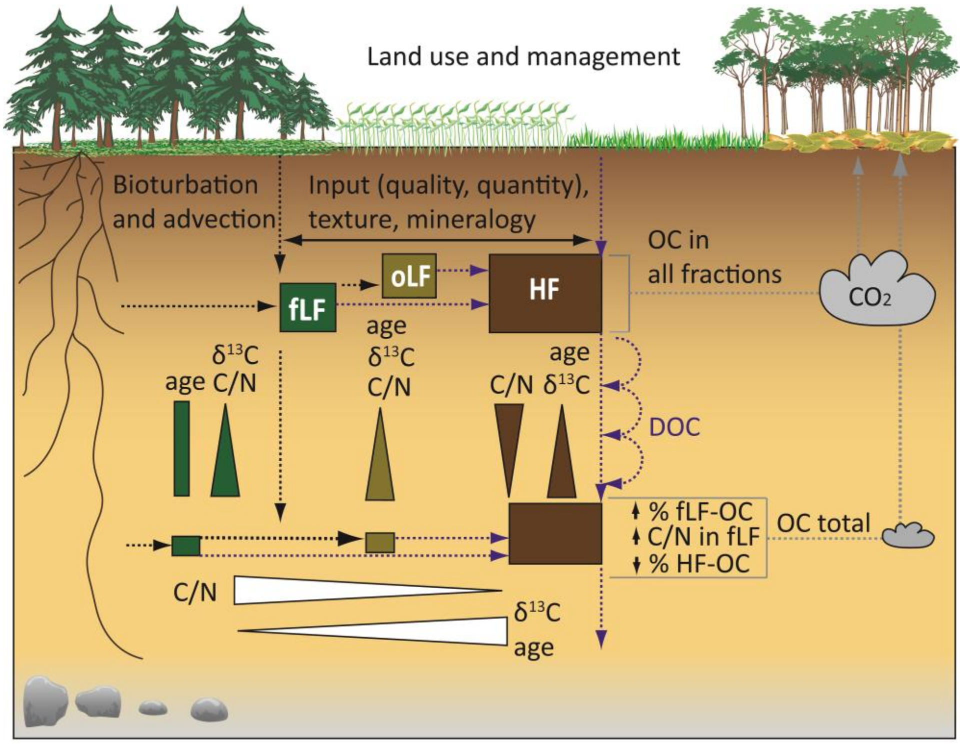

Figure 1.2 Schematic presentation of soil organic matter fractions (Adopted with permission from Schrumpf *et al*. (2013).

substrate for soil micro-organisms and a short-term reservoir of plant nutrients (Gregorich *et al*., 1994), the structure is determined by the quality of organic input and soil type (Hu *et al*., 1997). The fLF therefore, plays a major role in controlling short term ecosystem productivity having a turnover rate of a few days to a few years (Yang *et al*., 2009; Sequeira *et al*., 2011). Other attributes of the fLF apart from the high decomposability include the high sensitivity to management practises and high C/N ratios (Echeverría *et al*., 2004). The fLF is more susceptible to changes in temperature and moisture. In ecosystems that are usually burnt, charcoal can account for a considerable proportion of the fLF (Skjemstad *et al.*, 1990; Cadisch *et al.*, 1996).

Accumulation of light fraction is much higher and faster in upper soil layers and also higher in forests than agricultural systems. The fLF has a significant contribution to total active organic C in the upper soil depths of most soils (Yang *et al*., 2009; Matos *et al.*, 2011) except soils containing black C (von Lützow *et al.*, 2002). In the tropics the light fraction is largely confined to the top 22 cm (Trumbore & Zheng, 1996; Echeverría *et al*., 2004) and the mass, quality and contribution of light fraction to total organic C decreasing with soil depth (Tan *et al.*, 2007). Additions to the light fraction come from plant biomass, excretion of root exudates, root residues and other organic matter accumulating on the soil surface and subsequent mixing through bioturbation (Gale *et al.*, 2000).

In cropping systems, tillage mixes and breaks down soil aggregates exposing previously occluded SOC to decomposers resulting in loss of C and some complexed SOC from intra aggregate light fraction (Post & Kwon, 2000). Turnover of light fraction is sometimes linked to macro-aggregate formation and therefore affected by any form of soil disturbance (Beare *et al.*, 1994; Bremer *et al.*, 1994). Although total SOC declines with tillage, the decline in the amount of labile C fractions indicates instability of a soil's structure (Parton *et al.*, 1987). This fraction is usually ignored in terms of C stabilisation yet it is the most sensitive to disturbance.

The oLF is obtained from soil complexes after application of ultrasonic energy which disrupts the stable soil-aggregates (Golchin *et al*., 1994a; Sequeira *et al*., 2011) to release materials that are more decomposed, trapped and physically protected within soil aggregates (Six *et al*., 2000). This fraction is more biologically resistant and often associated with soil mineral colloids thus distinguishing it from fLF although it is usually found mixed with the fLF. The oLF can be released from large aggregates by microorganisms when the available substrate in the labile pool is depleted (Six *et al.*, 2001). The fraction comprises humified C compounds at various stages of decomposition (Marín-Spiotta *et al.*, 2008) including aromatic structures such as polyphenols which have slow decomposition rates limited by low levels of N, P and S (Lal, 2008). The oLF is therefore less decomposable than the fLF and also has a higher humification index (i.e. higher alkyl and lower O-alkyl C) (Golchin *et al*., 1994a; Gregorich *et al*., 2006; Marín-Spiotta *et al*., 2008). The turnover rate of oLF ranges from a few decades to hundreds of years (Six *et al*., 2000b). Total organic C in the oLF fraction is usually higher in soils that have more clay because of increased aggregate formation. The C:N ratio of oLF is lower than fLF due to both loss of C and incorporation of soil N. In ecosystems, where organic inputs are of comparatively similar quality, the oLF can indicate differences in magnitude, rate and nature of decomposition (Golchin *et al*., 1994b).

The MaHF contains the oldest, most stable and biologically resistant C pool (Kögel-Knabner *et al.*, 2008) consisting of recalcitrant C which is adsorbed onto silt and clay mineral surfaces having turnover time of decades to centuries (Six *et al*., 2002; Gregorich *et al*., 2006; von Lützow *et al.*, 2006; Schrumpf *et al*., 2013). Total organic C in the MaHF is usually higher in soils that have more clay because fine texture induces high adsorption onto charged surfaces of aggregates and makes the C less susceptible to microbial attack (Kölbl & Kögel -Knabner, 2004). The C in this fraction is made of a finer mixture of materials more heterogeneous in nature, whose origin cannot be easily distinguished and has organic C loading of minerals decreasing with soil depth (Schrumpf *et al*., 2013). An understanding of the dynamics of this fraction is important in climate change mitigation due to storage of more decomposed and more recalcitrant C (Alvarez & Alvarez, 2000). Concentration of C in MaHF and its contribution to total SOC increases with soil depth (Tan *et al*., 2007).

1.4 Modelling Soil organic carbon in agro ecosystems

Land management practices, and especially practices that increase SOC, contribute to climate change mitigation. The long-term impacts of such activities can be studied with simulation models validated using local data. Models can be used as a decision support tool for simulating different climate and management scenarios to assess long term impacts of each practice. For example the Climate Rapid Overview And Decision Support simulator (C–ROADS), which provides visual and numerical projections, helps to understand the gap between planned policies and further actions needed for stabilisation of atmospheric GHG concentrations. C–ROADS scenarios show the risks of climate change impacts (Findeling *et al.*, 2003).

Modelling has further quantified different factors controlling SOM dynamics and helped to project the long-term effects of different management activities under changing climate. Models are also useful in predicting loss or storage of SOC under different land-use systems (Lal, 2009). However, most of the information used to understand the impacts of land management on the dynamics of soil organic carbon is obtained from long-term experiments and short-term laboratory experiments. In addition, modelling SOC stabilisation provides a means of assessing land-use and management impacts relative to the effects of climate change (Parton *et al.*, 1994). Models, such as CENTURY and RothC, evaluate SOC dynamics based on environmental and climatic data (Jenkinson *et al.*, 1992; Woomer, 1993; Coleman & Jenkinson, 1996; Falloon & Smith, 2002; Cerri *et al.*, 2007) and help to improve understanding the consequences of soil management. However, only a few of the model-based projections apply to tropical dry lands (FAO, 2004). RothC, for example, has been successfully used in Zambia's woodland and improved fallow systems (Jenkinson *et al.*, 1999; Kaonga & Coleman, 2008a) and testing the model for agricultural and plantation-forest soils is important. Thus, models are important for the determination partitioning of total system C in different land management practices. Other models such as FARMSIM has been used in Zimbabwe to quantify the effects of interactions between different farm activities (Tittonell *et al.*, 2007; Rufino *et al.*, 2011).

1.5 The profile for Zimbabwean ecosystems

Zimbabwe is a landlocked country located in tropical Southern Africa. It falls between latitudes 15˚ 35′ and 22˚ 30′S. The altitude ranges between 162 m and 1,592 m above sea level in the southern and eastern parts respectively. In the east is a series of mountain ranges with peaks up to 1, 592 m above sea level (World Factbook, 2013). The total land area is about 39.0 Mha, of which 17.55 Mha are under fragmented indigenous forests (mainly miombo woodlands and savannahs and about 5.85 Mha being protected forest under national parks, wildlife reserves and forest reserves) and 0.12 Mha is under commercial forest plantations. A further 5.07 Mha is covered by bush lands. About 15.60 M ha is agricultural land while about 0.65 Mha are water bodies and urban settlements (World Bank, 2009). Figure 1.3 shows land tenure distribution in Zimbabwe.

The land area is divided into five natural regions based mainly on rainfall, temperature and soil type (Figure 1.4). Regions I, II, and III have annual rainfall between 650 and 1050 mm and are suitable for intensive animal and crop production whereas regions IV and V are mostly suitable for extensive livestock production because of annual rainfall below 650 mm. Agriculture contributes about 15% to annual GDP.

Zimbabwean soils are inherently infertile (Nyamapfene, 1991). This, when coupled with a long period of cultivation and erosion, contributes to low SOC stocks. Despite the low fertility, the majority of small holder farming activities are dependent on rain fed agriculture which is dominated by conventional tillage and continuous maize cropping. Appropriate soil management is thus key to maintaining agriculture productivity and reducing degradation (Lal *et al*., 1998; Singh & Lal, 2005). Therefore, adoption of conservation farming may play an important role of soil C sequestration in Zimbabwe to boost the low OC levels (<1%) and the generally deficient plant nutrients (Nyamapfene, 1991; Mugwira *et al.*, 1992).

Zimbabwe, like many of the southern African countries, has been experiencing rapid forest decline. Woodlands and forests in communal and resettlement areas are heavily fragmented and degraded due to the clearing for agriculture and harvesting for various wood and non-wood forest products. The major woodland type is the miombo whose crown cover can vary between 20 and 60 % (Walker & Desanker, 2004). Miombo is the vernacular term for the seasonally dry, deciduous woodlands, semi deciduous, semi evergreen or drought deciduous woodlands having some species with pre-rain leaf flush (Frost, 1996).

The woodlands are dominated by *Brachystegia, Julbernardia* and/or *Isoberlinia*, spp. extending across 2.7 million km^2 of some of the world's poorest countries (Campbell *et al.*, 2007). The diverse woodland uses include firewood, timber, poles, watershed protection, provision of soil fertility (leaf-litter), grazing and browsing, edible fruits, mushrooms and caterpillars. The land use change to cultivation has an effect on C stocks (1069 Mt C). The extensive woodland cover makes Zimbabwean forest cover a potential C sink though threatened by deforestation and degradation. The forests/woodlands have potential to act as C sinks provided forest management regimes are properly designed to reduce vulnerability to conversion for agricultural crop production and severe wild fires.

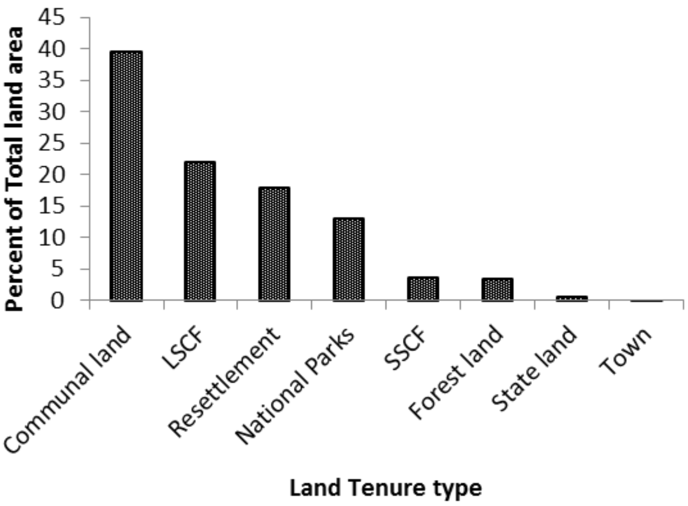

Figure 1.3 Land tenure distribution in Zimbabwe. SSCF = small scale commercial farms, LSCF= large scale commercial farms, Forest land = gazetted forests/woodlands and commercial plantations (Moyo, 2000; Zimbabwe's Fourth National Report to the Convention on Biological Diversity, 2010).

The Zimbabwean land reform programme has shifted >31% of the country's woodlands from private tenure to state control, with usufruct rights for communities and individuals (Matose, 2006). This activity caused forest and woodlands clearance to become the major driver of land use change resulting in an annual loss of about 1.5% (about 300 000 ha) (UNDP, 1997; FAO, 2005). National woodland cover declined from 53% in 1992 to 42.34% in 2008 whereas crop land increased from 27.48% in 1992 to 41.24% in 2008 (Zimbabwe's Fourth National Report to the Convention on Biological Diversity, 2010).

Increased droughts and floods have been cited as the main climate-change effects in Sub-Sahara Africa (FAO 2010). Despite Zimbabwe's relatively tiny contribution to global GHG emissions, droughts and cyclones are being experienced. The 1980s and early 1990s witnessed Zimbabwe's driest periods for the twentieth century. National average rainfall declined by about 5% between 1900 and 2000, notwithstanding the episodes of wetter than average conditions in the 1920s, 1950s, and 1970s (MDG report, 2010). Agricultural seasons have shifted as well, as evidenced by late onset and sometimes late cessation of rainy .season's. Longer-term rainfall predictions for Zimbabwe remain uncertain as climate change causes more climatic extremes. The highest monthly daily maximum temperatures for most parts of the country, for example, are increasing

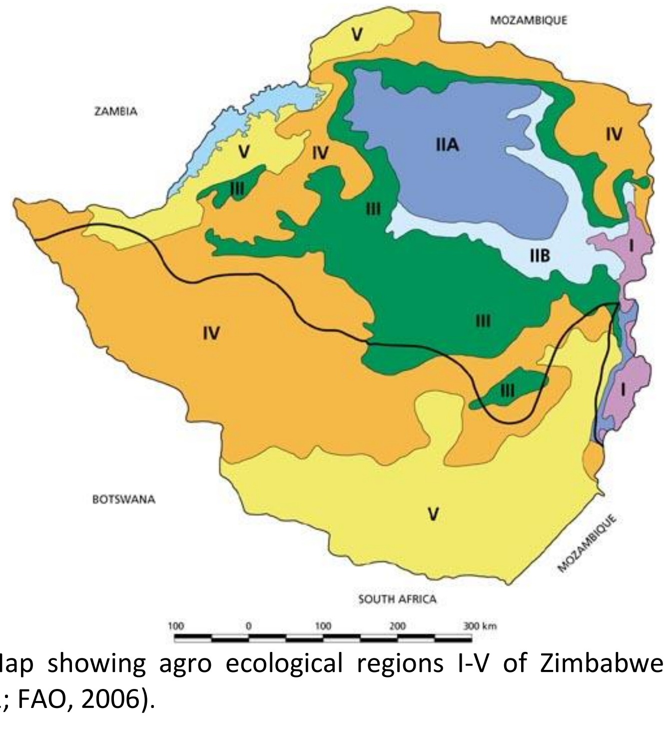

Figure 1.4: Map showing agro ecological regions I-V of Zimbabwe. (Vincent & Thomas, 1961; FAO, 2006).

by approximately 2ºC per century and the percentage of days with low temperatures decreasing at a rate of 15 days per century. Assuming that GHG emissions continue along the projected trajectories, temperatures will rise by between 0.5°C and 2.0°C by 2030, and 1.0°C and 3.5°C by 2070. Various climate models project that rainfall patterns will change and that frequency of drought and flood events will increase as well and projected that an additional 10 to 20% rainfall decline will be experienced in Zimbabwe by 2050 (MDG report, 2010).

In addition to climate change, land degradation has become an important environmental issue in Zimbabwe. There are synergies in simultaneously addressing climate change and this land degradation. Thus, land degradation and climate change will both be mitigated when effective ways to sequester C in soils are implemented. This not only promotes soil restoration but also achieves a win-win scenario (FAO, 2004). There is therefore need for information on appropriate local soil management practices that can be used to achieve this win –win goal in Zimbabwean agro ecosystems.

1.6 Scope and Objectives of this thesis

Considering the high rate of deforestation in African countries, there is potential for management of forests and woodlands for C sequestration through coppice or regeneration management (Chidumayo, 1991; Chidumayo, 1993) while increasing productivity of land through SOC sequestration. However, in most countries (including Zimbabwe) the current policy framework is not sufficient to ensure the reflection of international environmental and social safeguards and standards for sustainable forest resource utilisation e.g. REDD+.

The potential for climate change mitigation through SOC sequestration under agricultural activities in tropical soils is poorly understood. Most agricultural based studies focused on soil fertility and crop yields aiming at ensuring crop productivity but without consideration of the implications of the activities on climate change mitigation. Assessment of SOC dynamics under different land management practices can therefore help to draw meaningful conclusions about SOC's contribution to global C stocks (i.e. as sources or sinks) thus, enabling the assessment of potential roles of agriculture and forestry to mitigate climate change. The study of C storage potential provides important information for shaping policies and incentives for better management of both agricultural and forest systems. Information can be useful when designing projects that provide opportunities for communities to improve their livelihoods in accessing carbon markets. Data will be useful when responding to C demand in future voluntary and regulatory greenhouse gas emissions markets. SOC can become a tradable market commodity under the Kyoto protocol (c.f. Article 3.4) where C credits can be assigned for agricultural C sequestration. The involvement of small holder farmers in soil C sequestration, simultaneously advances food security, mitigate climate change and improve the environment thereby, transforming people from a state of production to that of land resource stewardship.

The participation in C markets requires information on the potential of land-use systems and this study, in turn, facilitates identification of possible C sequestration options. My research will help to clarify some of these issues by analysing SOC and TON and their distribution in density fractions at different depths under selected agricultural practices. Results provide important baseline information for agriculture and forestry operations and enables comparisons between current and projected future soil C sequestration and emission reductions levels.

This research estimates C and N sequestration potential of selected Zimbabwean forests and agro-ecosystems. The focus is C because carbon dioxide is the most abundant greenhouse gas that can be sequestered in agro-ecosystems. The study examines effects of tillage practices and fertilisation on SOC and TON pools in different small holder farming systems relative to native forests. Additionally, the C sequestration potential under tree farming and its impacts on SOM partitioning was studied using a chronosequence approach. The impacts of different tillage and fertility treatments on stabilisation of C and N and the partitioning of SOM fractions as described for cropping and forested systems remains unclear.

The C and N stocks were analysed in bulk soil and in three density separated fractions at different soil depths. Land management practices selected for the study included conservation tillage (i.e. reduced tillage (RP) and no tillage (DS)), conventional tillage (CT), fertility amendments (fertiliser and manure) and plantation forestry practices. The main research hypotheses are:

- Long term fertiliser and manure application will increase SOC in whole soil and eventually in stabilised C fractions;
- Accumulation of different SOC fractions is influenced by the long-term application of N fertiliser alone and the combined application of N fertiliser and manure under continuous maize cropping and conventional tillage;
- The light fraction would be a responsive indicator of short and long term changes in SOC;
- Tillage effects on C and N dynamics vary with soil type;
- Land management and soil properties (texture) affect SOM fraction contents;
- The land use legacy affects partitioning of SOC and N in homefields and outfields; and
- The planting of trees significantly affects soil C storage at different depths over a rotation with higher C sequestration rates in younger stands than older stands.

These hypotheses result in the following specific research objectives:

- To assess the effects of tillage and fertilisation on SOC and TON stabilisation in selected agriculture and forest systems on soils of contrasting texture;
- To assess vertical distribution of SOC and TON and their fractions as affected by different tillage practices (CT, RP and DS) relative natural forests;
- To assess vertical distribution of SOC and TON and their fractions as affected by N fertiliser and a combination of N fertiliser and manure application;
- To quantify C storage in the total soil organic matter and SOM fractions in sandy and clayey soils under different tillage and fertility management;

- To determine organic C and N dynamics in a *Pinus patula* chronosequence through assessment of partitioning of C and N stocks in mineral soils and forest floor relative to native forests; and
- To simulate the changing SOC pool through time and to predict the future pools and impacts under land use change and changing climate scenarios.

1.7 Outline of the thesis

In order to satisfy the hypotheses and objectives outlined in 1.6, this thesis has been structured into five chapters:

Chapter 2 gives an account of the impacts of tillage practices on C and N stocks in bulk soil and in three density separated SOM fractions at 0-10 and 10-30 cm depths. The chapter compares C and N storage in three tillage systems on farmer managed experiments and adjacent natural forests in sandy Haplic Arenosols and clayey Rhodic Ferralsols of Zimbabwe. The C and N storage in tillage systems is compared to baseline measurements taken in 2005 (only at 0-10 cm) for whole soil.

Chapter 3 investigates the effects of long term fertilisation on 1) SOC and TON and 2) density fractions for soil up to 50 cm on Luvisols and Arenosol in Murewa district of Zimbabwe. Soils collected from two farms on fields with contrasting soil types were analysed. The chapter explores the potential of C sequestration under continuous conventionally tilled fields with inorganic fertiliser and cattle manure application relative to distance from homestead. C and N in whole soil and density fractions was determined by dry combustion.

Chapter 4 examines C and N storage in pine plantations located in the eastern highlands of Zimbabwe along an age series. Plantations of exotic pine trees were established after clearing natural forests. These natural forests can either be patches of moist forests or patches of Miombo woodlands depending on soil type and depth. Forests play a major role in regulating the rate of increase of global atmospheric carbon dioxide (CO_2) concentrations creating a need to investigate the ability of exotic plantations to sequester CO_2 from the atmosphere. C and N storage was assessed from randomly selected replicated plots in *Pinus patula* stand ages of 1, 10, 20, 25, and 30 years plus two natural forests. Annual litter fall and C and N in forest floor were assessed. Furthermore, comparisons were made between C and N storage in older pine age classes (25 and 30 years) with the two natural forests (moist forest and Miombo woodland).

Chapter 5 shows the impacts of land management practices on future C stocks using the dynamic RothC model. This model is used to assess the impact of land management on SOC storage and helps to investigate scenarios and hypotheses that are beyond the realm of current assessments. The running of the model requires information on soil type, plant cover and monthly climate input data. In this chapter, experimental data from Chapters 2 and 3 are used to assess performance and ability of RothC to simulate long-term SOM changes in Zimbabwean soils. The results confirm the ability of RothC to simulate short and long-term soil C dynamics. The relationship between conceptual model pools and density separated fractions was determined. Furthermore the equilibrium levels estimated by the RothC model were compared with equilibrium levels by the Langmuir equation.

Chapter 6 summarises the main conclusions derived from the integration of the entire work. It also highlights proposals for management and research to ensure sustainable utilisation of forest and agricultural land resources.

Chapter 1

Effects of tillage on soil organic carbon and nitrogen storage

CHAPTER 2

Land use and management effects on soil organic matter fractions in Rhodic Ferralsols and Haplic Arenosols in Bindura and Shamva districts of Zimbabwe

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Chapter 2

Land use and management effects on soil 0rganic matter fractions in Rhodic Ferralsols and Haplic Arenosols in Bindura and Shamva districts of Zimbabwe

Abstract

Soil organic carbon (SOC) is a major attribute of soil quality that responds to land management activities and is also important in the regulation of global carbon (C) cycling. This study evaluated bulk soil C and nitrogen (N) contents and C and N dynamics in three density separated soil organic matter (SOM) fractions. The study was based on three tillage systems on farmer managed experiments; (conventional tillage (CT), ripping (RP), direct seeding (DS)) and adjacent natural forest (NF) in Haplic Arenosols (sandy) and Rhodic Ferralsols (clayey) of Zimbabwe. Carbon stocks were significantly larger in forests than tillage systems, being significantly lower in sandy soils (15 and 14 Mg C ha^{-1}) than clayey soils (23 and 21 Mg C ha^{-1}) at 0-10 and 10-30 cm respectively. Nitrogen content followed the same trend. At the 0-10 cm depth SOC stocks increased under CT, RP and DS by 0.10, 0.24, 0.36 Mg ha^{-1} yr^{-1} and 0.76, 0.54, 0.10 Mg ha^{-1} yr^{-1} on sandy and clayey soils respectively over a four year period while N stocks decreased by 0.55, 0.40, 0.56 Mg ha^{-1} and 0.63, 0.65, 0.55 Mg ha^{-1} respectively. SOM fractions were dominated by mineral associated heavy fraction (MaHF) which accounted for 86-93% and 94-98% on sandy and clayey soils respectively. Tillage systems on sandy soils had smallest average free light fraction (fLF) and occluded light fraction (oLF) C stocks (25.3± 1.3 g m^{-2} and 7.3 ± 1.2 g m^{-2}) at 0–30 cm when compared with corresponding NF (58.4± 4 g m^{2} and 18.5 ±1.0 g m^{-2}). Clayey soils, had the opposite, having all fLF C and N in tillage systems being higher (80.9±12 g C m^{-2} and 2.7±0.4 g N m^{-2}) than NF (57.4± 2.0 g C m^{-2} and 2.4 ± 0.3 g N m^{-2}). Results suggest that oLF and MaHF C and N are better protected under DS and RP where they are less vulnerable to mineralisation while fLF contributes more in CT. Thus, DS and RP can be important in maintaining and improving soil quality although their practicability can be hampered by unsupportive institutional frameworks. Under prevailing climatic and management conditions, improvement of residue retention could be a major factor that can distinguish potential of different

management practices for C sequestration. The exploitation of the benefits of RP or DS and corresponding sustainability of systems need support for surface cover retention which should also be extended to conventional tillage.

Key words: Soil density fractionation; soil management; soil organic matter; soil carbon sequestration; soil nitrogen; tillage.

2.1 Introduction

Soil organic matter (SOM) represents a large, dynamic and complex terrestrial reservoir of carbon (C) in the form of organic compounds derived from plant, animal or microbial biomass (Baldock, 2009). Land use practices in agro ecosystems affect the storage of soil organic carbon (SOC) and nitrogen (N). The rates of C release from the soil varies with land use type, climate and the soil matrix. Several studies have shown that significant amounts of C were lost from soil as carbon dioxide (CO_2) when forests were converted to agriculture (IPCC., 2000; Food and Agriculture Organisation (FAO), 2010a), through the release of physically protected soil C (Six *et al.*, 1999; McConkey *et al.*, 2003; Denef *et al.*, 2007). Despite this loss, agriculture is inevitably required to enhance food security in the developing countries which are threatened by food shortages (FAO, 1996).

In this regard, conservation agriculture (CA) has been recommended as a means of C accumulation and soil quality enhancement (Ken & Johnson, 1993; Chivenge *et al.*, 2007b; Álvaro-Fuentes *et al.*, 2008; Dercon *et al.*, 2010) and has been proposed as a means of sustainable land use management (Food and Agriculture Organisation (FAO), 2010a). In southern Africa, the reasons for the success of conservation farming have been clearly outlined (Marongwe *et al.*, 2011; Andersson & Giller, 2012). However, the success of some conservation farming practices such as no-tillage in C storage depends on quality and quantity of organic residue inputs and the degree of soil disturbance (de Moraes Sá *et al.*, 2011).

Chapter 2

No tillage or minimal tillage practices might affect SOC stocks, although some studies have shown that no-tillage increases SOC concentrations only in the upper layers of some soils having no significant differences with conventional tillage over the whole profile (Blanco-Canqui & Lal, 2008; Poirier *et al.*, 2009). Soil organic C may be present in the soil as either 1) relatively fresh (labile) SOM not protected by the soil matrix, as 2) SOM physically protected in aggregates (occluded), or as 3) SOM adsorbed onto mineral surfaces (chemically stabilised) (Dalal & Mayer, 1986a). Although land management systems such as tillage may not have an effect on bulk SOC and N, they may have an effect on individual SOM fractions (Von Lützow & Kögel-Knabner, 2009). Moreover, changes in SOM fractions may provide insight into the effects of tillage practices on SOC stabilisation (von Lützow *et al.*, 2007 ; Marín-Spiotta *et al*., 2008).

Isolation of these functional pools can be done by density or size separation. Density fractionation is often coupled with ultrasonic dispersion to give three mutually exclusive fractions: free light fraction (fLF) extracted before the breakdown of aggregates; occluded light fraction (oLF) isolated after ultrasonic disruption and mineral associated fraction (MaHF) recovered in the remaining heavy precipitate which is considered as stable (Poirier *et al*., 2005; Gregorich *et al*., 2006). Evidence of differences in residence time of fLF and oLF has been shown (Golchin *et al*., 1994b; Gregorich *et al*., 2006) and confirmed by radio C dating (Baisden *et al.*, 2002; Swanston *et al*., 2005). The mineral-associated heavy fraction contains more processed material with a slower turnover rate and a higher degree of chemical protection (Hassink, 1995) than oLF.

Soil organic C and N in density fractions are important attributes of the quality of a soil and associated impacts of land management systems. The applicability and feasibility of density fractionation has not been fully exploited in Zimbabwean soils. Some studies in Zimbabwe focused on crop production and soil organic matter fractions in aggregate sizes and have revealed that physically separated SOM fractions change with aggregate size (Chivenge *et al*., 2007b; Nyamadzawo *et al.*, 2009). Despite the importance of SOC storage in the light fraction, there is little information about size, composition and stability of free and occluded light fractions in sandy and clayey soils of Zimbabwe. In addition, research on experimental stations has not provided the required information on C sequestration potential since it does not represent farmer's conditions (Giller *et al.*, 2011).

In this study, active collaboration with farmers in the research plots provided a true picture of small holder farmers' condition which can result in the transfer of technologies to their own fields. The native forests in the area are mainly miombo woodlands which are subjected to annual wild fires and are utilised by local communities for timber and non -timber forest products. We therefore aim to evaluate the effects of land management systems on: 1) bulk soil C and N contents and 2) the C and N dynamics of three fractions, (i.e. the free light fraction (fLF), occluded light fraction (oLF) and mineral associated heavy fraction (MaHF)) on farmer managed tillage experiments on sandy and clayey soils. In order to assess the effect of agricultural land use, irrespective of treatment, on SOC and N contents, samples were also collected from adjacent natural forests to show the C benefits of not clearing more land for agriculture. Agricultural land is mostly a product of deforestation of these native forests.

2.2 Material and methods

2.2.1 Study site

This research was carried out at two experimental locations; Hereford in Bindura district (17˚42′ S; 31˚44′ E) and Nyarukunda in Shamva district (17˚00′S; 31˚ 43′E) (Figure 2.1) which were established in farmers' fields in 2005. The two areas transcend a zone with altitudinal ranges between 1000 and 1800 m.a.s.l. and annual unimodal rainfall of 750-1000 mm. Sandy soils, mostly derived from coarse granite cover nearly 70% of Zimbabwe (Thompson & Purves, 1981) mostly of the Kaolinitic order, Fersiallitic group under the Zimbabwean soil classification, corresponding to Alfisols in soil Taxonomy (Nyamapfene, 1991).

Chapter 2

The dominant soils in the Nyarukunda area can be classified as coarse grained sandy to sandy loam soils, corresponding to Ferric Luvisols (Thompson & Purves, 1981; Scoones, 2001; FAO, 2006) but the IPCC default classes derived from the harmonised world soils database (Batjes, 2010) classified them as Arenosols, (>70% sand and <8% clay) and are broadly referred as sandy soils. Hereford soils are red clays varying from silty clay loam to clay, with characteristics corresponding to Rhodic Ferralsols (Nyamapfene, 1991; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) and falling into the category of low activity clays (Batjes, 2010).

The two areas have the same climate (Region II) and vegetation type of miombo woodland dominated by *Brachystegia spiciformis* and *Julbernardia globiflora* to mixed woodland (dominated by A*cacia polyacantha*) and bush land. They, however, differ in land use history and soil characteristics. The forest on clayey soils was part of a commercial farming area and was not as intensively utilised as the one on sandy soils and most of the agricultural land was created after deforestation of natural forest areas. Shamva's Nyarukunda ward, is a communal area where farmers have been practicing conservation farming on small pieces of land for environmental and socio-economic gains (Belder *et al.*, 2007; Thompson, 2010). Major crops grown in the area are: maize (*Zea mays L.*), cow-peas *(Vigna unguiculata* L. Walp), sugar beans (*Phaseolus vulgaris L.*), groundnuts (*Arachis hypogaea L.*), sorghum (*Sorghum bicolor, L.*) and sunflower (*Helianthus annuus, L.*). Additionally, farmers in Hereford grow commercial crops such as soybeans (*Glycine max. L.* Merr*)* and tobacco (*Nicotiana tabacum L.).* In Hereford mechanised tillage with disc and harrow was used before the farm was designated as a new resettlement area in 2000.

Within each area, four farmer's field were chosen, each being 0.3 ha, subdivided into three equal portions (0.1 ha) where the three main treatments were established: 1) conventional farming (CT) – consists of an ox drawn plough to a depth of 15-20 cm once before planting. Residues were removed and the remaining biomass incorporated into the soil during ploughing in the next season; 2) minimum tillage with an ox drawn ripper (RP) 15-20 cm followed by manual planting and fertiliser application. Maize stover was supplemented by thatching grass in the first season. In subsequent years, crop residues were to be retained in the field after harvesting. Ground cover of 2.5-3.0 Mg ha^{-1} was required in each RP plot; and 3) no tillage using an ox drawn direct seeder (DS) – synchronised seeding and fertiliser application. Residues were also retained or supplemented to achieve the 2.5-3.0 Mg ha^{-1} ground cover.

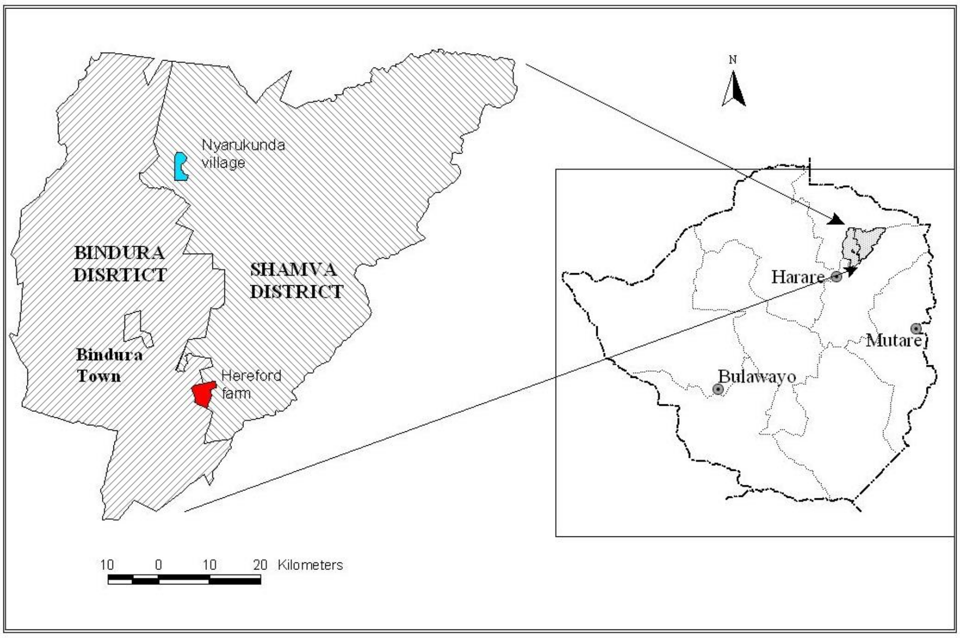

Figure 2.1: Map of Zimbabwe showing location of Shamva and Bindura districts.

Each tillage system was divided in half, where during the first two years, maize (*Z. maize*. L) was grown on one side of the field and cowpeas *(V. unguiculata* L. Walp) at Nyarukunda or soy bean (*G. max* L. Merr) at Hereford grown on the other. During the next two years the sides were switched. Each treatment received annual basal fertiliser of 165 kg ha^{-1} compound D (i.e. 11 kg ha^{-1} N, 10 kg ha^{-1} P, 10 kg ha^{-1} K), which was followed by 69 kg ha^{-1} N applied as ammonium nitrate in splits at 4 and 7 weeks after germination. The soy bean received no top dressing but was inoculated with a commercial rhizobium before sowing. Weed control was by herbicide in RP and DS while in CT it was done conventionally by cultivator or plough. All fields in each soil type were planted with similar maize and legume varieties each year. In addition to the agricultural sites in each area, an indigenous forest site in close proximity and on the same soil type was included as they are a target for agricultural expansion and can be a useful reference for C storage capacity.

Initial C and N stocks as recorded during the 2005/2006 season (International Maize and Wheat Improvement Centre (CIMMYT), unpublished results -2005/2006) had an average C concentration ranging between 3 and 5 g kg^{-1} on sandy soils compared to 13 and 15 g kg^{-1} for the clayey soils. Mean C stocks at 0-10 cm under CT, RP and DS were 4.79, 6.38, 5.45 Mg ha^{-1} and 6.08, 16.08, 17.64 Mg ha^{-1} on sandy and clayey soils respectively (Table 2.1). Nitrogen stocks were 0.93, 0.93, 1.06 and 1.92, 2.04, 1.92 Mg ha^{-1} on sandy and clayey soils respectively. The pH was higher in clayey soils than in sandy soils. All values of soil characteristics were higher on clayey soils than sandy soils except the amount of course and medium sand which were obviously larger in sandy soils.

2.2.2 Surface litter and soil sampling

At each sample point, surface litter was collected by inserting a 30 cm diameter ring into the litter on top of the mineral soil and the litter was collected from inside the ring (g litter per surface area). Three composited soil samples (each based on 4 sub-samples) were collected from each field, (i.e. per treatment, for two depth intervals of 0-10 and 10-30 cm). A total of 144 soil samples were taken from the agricultural plots, (i.e. 2 areas × 4 sites × 3 treatments × 2 depths × 3 replicates). The four sites in each soil type were treated as replicates relative to tillage systems. Forest soil samples totalled 24 soil samples (2 areas × 6 sample points × 2 depths) plus six litter samples. An additional soil sample was collected, at each depth by inserting a metal ring of 100 cm^3 into the soil for bulk density measurements. The soils were air-dried and sieved (< 2 mm) before C and N analysis and physical fractionation.

Table 2.1: Bulk soil chemical and physical soil properties at 0–10 cm depth before establishment of three tillage systems at sites selected for the study at Nyarukunda and Hereford study areas.

Characteristic	**Shamva – Nyarukunda (Sandy soils)**			**Bindura- Hereford (Clayey soils)**		
Tillage	**CT**	**RP**	**DS**	**CT**	**RP**	**DS**
pH (Ca Cl_2)	5.2	4.9	5.0	5.4	5.5	5.6
BD (g cm^{-3})	1.33	1.33	1.33	1.2	1.2	1.2
SOC (Mg ha^{-1})	4.8	5.4	5.5	16.1	16.1	17.6
TON (Mg ha^{-1})	0.9	0.9	1.1	1.9	2.0	1.9
C:N ratio	5.3	7.0	5.0	8.4	8.0	9.3
P (mg kg^{-1})	16.3	15.8	13.5	26.7	30.5	31.0
ex K (cmol/kg)	0.2	0.3	0.2	0.5	0.6	0.6
ex Ca (cmol/kg	1.3	1.0	1.0	8.5	8.9	8.6
ex Mg (cmol/kg	0.7	0.3	0.2	4.6	3.7	5.0
TEB (cmol/kg	2.5	1.7	1.6	14.0	13.6	14.7
BASE SAT %	70	60	35	64	63	75
Clay %	4	5	4	22	23	26
Silt %	7	7	8	20	20	20
sand %	89	87	89	58	57	54

CT = conventional tillage, RP= ripping, DS= direct seeding.
Source: (International Maize and Wheat Improvement Centre (CIMMYT), unpublished results -2005/2006).

2.2.3 Soil density fractionation

Soil samples were subjected to physical fractionation to obtain three organic matter fractions (free light fraction (fLF), occluded light fraction (oLF) and mineral associated heavy fraction (MaHF)) using the method described by Roscoe *et al*. (2000). The fractions were recovered by density separation using sodium polytungstate ($3Na_2WO_4.9WO_3.xH_2O$; SPT). Briefly, 10g air dried soil <2 mm was shaken in a centrifuge tube with 50 ml SPT solution of density 1.6 g cm^{-3} and gently mixed by inverting 5 times and allowed to stand for at least 1 hour to fully wet the sample. Next, the samples were centrifuged for 23 minutes at 4500 rpm. The supernatant was poured over a Büchner funnel with a 0.45 µm Whatman GF/F glass filter using a Millipore vacuum filtration unit (Millipore, Bedford, MA). The light fraction (LF) also referred to as the “free light” (fLF) or “labile” fraction was rinsed off the filter into a glass beaker and oven dried at 40 ºC.

SPT solution with density 1.6 g cm^{-3} was added to the residual soil material and shaken by hand to bring the precipitate into solution. An ultrasonic probe was used for 5 minutes to disperse the occluded light fraction after which the suspension was left to stand for 30 minutes. The suspension was centrifuged for 23 minutes at 4500 rpm. The centrifuged tubes were decanted over a Büchner funnel with a 0.45 µm Whatman GF/F glass filter. The “occluded light” (oLF) or “physically protected light” fraction was rinsed off the filter into a glass beaker and oven dried at 40 ºC.

The remaining precipitate, the mineral-associated (MaHF) fraction, needed to be rinsed thoroughly,(i.e. at least 5 times for clayey soils and three times for sandy soils, or until conductivity of the filtrate was <50µS cm^{-1}) in order to remove all STP. Total C and N contents of fLF, oLF and MaHF and whole soil were determined by dry combustion using a CN analyser. Bulk soil C and N were estimated using equation [1] and the amount of SOC in fractions was expressed as concentration (g C kg^{-1} of soil) and multiplied by depth and bulk density to obtain stocks of each fraction.

$$\text{Organic C (Mg ha}^{-1}\text{)} = \text{Depth (cm)} \times \text{Bulk density (g/cm}^{3}\text{)} \times \text{Carbon content (\%)} \qquad [1]$$

Data were tested for homoscedasticity using Levenes’s test and it showed that error variances of dependent variables were not equal across groups except C:N ratios. Statistical comparisons of C and N among land management systems were therefore done using multivariate analysis of variance (MANOVA) which is a generalisation of analysis of variance that permits testing for mean differences on several dependent measures simultaneously. Wilk’s Lambda was used as the multivariate measure. Carbon and N were the dependent variables whilst soil type, management system and depth were the fixed factors. Tukey's HSD test was used for post-hoc analysis of C and N in bulk soil and the three fractions in each soil type, management system and soil depth. Statistical analyses were performed with software package SPSS 19.0 for windows. Differences were considered significant at $p < 0.05$.

2.3 Results

2.3.1 Surface litter in land management systems

Mean C of litter for tillage systems was 0.31 and 0.22 Mg ha^{-1} on sandy and clayey soils respectively compared to 0.64 and 0.60 Mg ha^{-1} in respective forests (Figure 2.2).

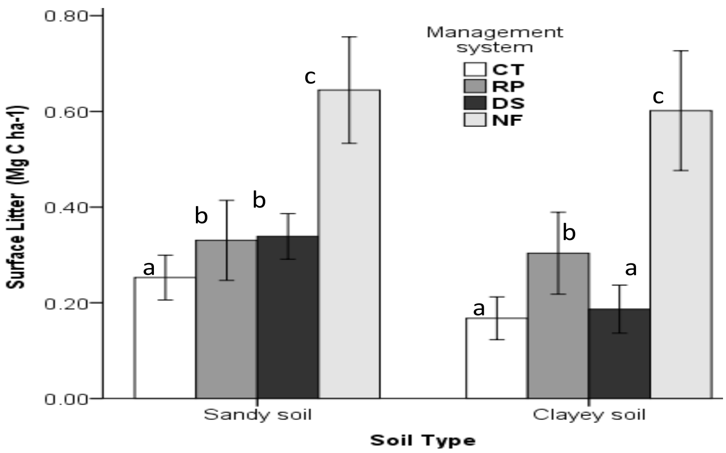

Figure 2.2: Amount of litter persisting in fields before the cropping season. CT = conventional tillage; RP = ripping; DS = direct seeding and NF = natural forest. Different letters show significant differences at p = 0.05. Error bars ± 1 SE.

There was no significant difference in surface litter accumulation among tillage systems on sandy soils. The amount of surface residue among tillage systems in each soil type was 34% and 46% lower than respective NF on sandy and clayey soils. On Clayey soils RP had highest surface litter accumulation which was half of the litter on forest floor. In all fields, termites seemed to play an important role as they were present and active in all plots and they may have a significant effect on the degradation of organic inputs and promotion of the release of nutrients to crops.

2.3.2 Soil bulk density, pH, C and N in land management systems

Despite some human influence at the forested sites, (e.g. fire wood collection and fire) the soils at the Nyarukunda and Hereford woodlands were considered to represent natural soil conditions for each location. There was a stronger relationship between bulk density and total organic C on clayey (R^2=0.987) than sandy soils (R^2= 0.016). Tillage neither affected bulk density, soil pH (p>0.1) nor C:N ratios although higher bulk densities were more associated with CT and least with RP being higher on sandy soils (1.32 g cm^{-3}) than clayey soils (1.20 g cm^{-3}) (Table 2.2).

Soil organic C stocks varied significantly (p<0.01) between tillage systems and natural forests being significantly higher on clayey soils (F = 187.69; p< 0.01) than on sandy soils in all land management systems and at all depth levels (Figure 2.3). Mean C stocks in the land use systems up to 30 cm depth ranged from 7.97 to 29.25 Mg ha^{-1} and 30.6 to 43.9 Mg ha^{-1} on sandy and clayey soils respectively. CT, RP and DS had 36%, 32%, 32% and 14%, 15% 14% lower C than their respective forests on sandy and clayey soils respectively. On Sandy soils C was higher under DS (10.9 Mg ha^{-1}) than RP (10.7 Mg ha^{-1}) and CT (8.0 Mg ha^{-1}) at 0-30 cm whereas on clayey soils, C was higher under RP (32.3 Mg ha^{-1}) than CT (31.2 Mg ha^{-1}) and DS (30.6 Mg ha^{-1}) though not significantly. Depth distribution showed significantly higher (F= 22.98; p<0.01) C stocks at 0-10 cm than at 10-30 cm in all cropping systems (Table 2.2).

On sandy soils, at 0-10 cm depth C was higher under DS (6.9 Mg ha^{-1}) than RP (6.4 Mg ha^{-1}) and CT (5.2 Mg ha^{-1}) whilst on clayey soils there was more C under CT (19.1 Mg ha^{-1}) than RP and DS although the differences were not significant.

Table 2.2: Soil characteristics, C and N stocks in three tillage systems and natural forests on sandy and clayey soils (n=12).

Soil	Management	pH	BD(g cm^3)	Total organic C (Mg ha^{-1}) 0-10 cm	10-30 cm	0-30cm	Total organic N (Mg ha^{-1}) 0-10	10-30	C:N
Sandy	CT	4.7(0.1)	1.37(0.02)	5.16(0.77)^a	2.81 (0.76)^a	7.97	0.38(0.07)^a	0.22(0.07)^a	14
	RP	5.2(0.1)	1.32(0.03)	6.38(.76)^a	3.90(0.76)^b	10.28	0.53(0.07)^b	0.34(0.07)^b	13
	DS	4.7(0.1)	1.33(0.03)	6.91(1.76)^a	4.46(0.77)^b	11.37	0.50(0.05)^b	0.37(0.06)^b	13
	NF	5.1(0.1)	1.27(0.3)	14.96(1.09)^b	14.29(1.09)^c	29.25	1.20(0.10)^c	1.09(0.10)^c	13
Clayey	CT	5.8(0.3)	1.22(0.06)	19.13(0.76)^a	12.04(0.77)^a	31.17	1.29(0.07)^a	0.96(0.07)^a	14
	RP	5.8(0.3)	1.21(0.05)	18.22(0.78)^a	14.05(0.76)^b	32.27	1.39(0.07)^b	1.20(0.07)^b	13
	DS	6.0(0.3)	1.16(0.04)	18.03(0.76)^a	12.59(0.76)^a	30.62	1.37(0.06)^b	0.99(0.07)^a	13
	NF	5.7(0.3)	1.21(0.2)	22.77(1.09)^b	21.11(1.08)^c	43.88	2.47(0.11)^c	1.46(0.10)^c	14

CT =conventional tillage, RP = minimum tillage with a ripper, DS= no tillage using direct seeder, BD = bulk density. Standard error of the mean shown in parenthesis. Means followed by different letters are significantly different at *p*=0.05. Tukey's HSD test.

A comparison with the baseline C stocks (Table 2.1) showed that CF, RP and DS increased C stocks at 0-10 cm depth by 4%, 4%, 8% and 8%, 6%, 2% on sandy and clayey soils respectively (Table 2.1 and 2.2). The SOC stocks increased under CT, RP and DS by 0.10, 0.24, 0.36 Mg ha^{-1} yr^{-1} and 0.76, 0.54, 0.10 Mg ha^{-1} yr^{-1} on sandy and clayey soils respectively.

Over the four year period N stocks were reduced by 0.55, 0.40, 0.56 Mg ha^{-1} and 0.63, 0.65, 0.55 Mg ha^{-1} on sandy and clayey soils respectively. There were significant differences in N content (p=0.015) between sandy (0.5±0.3 Mg ha^{-1}) and clayey soils (1.2±0.3 Mg ha^{-1}) which did not have a corresponding effect on the C:N ratio. The mean C:N ratios ranged from 9-20 and 10-24 on sandy and clayey soils respectively. The C:N ratio and total organic N were not significantly different by depth (p=0.012).

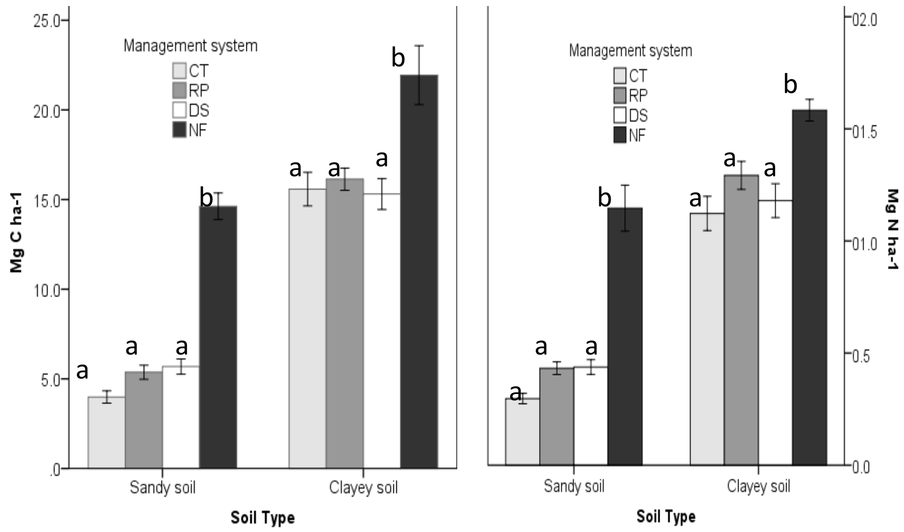

Figure 2.3: Soil organic carbon (SOC) and total organic nitrogen (TON) stocks in three tillage treatments (CT = conventional tillage, RP = ripping and DS = direct seeding) and natural forest (NF) in sandy and clayey soils at 0–30 cm depth.

2.3.3 Carbon and N in density fractions

The light fraction only accounted for ±1% of the soil mass. However, the organic C and N concentration of the density fractions were inversely related to their masses. On sandy soils mean C and N concentration of fLF was 33±8% and 1.5±0.5% respectively while the oLF C and N were 26±8% and 1.2±0.3%. The mean MaHF C and N were 0.7±0.3% and 0.05±0.02% respectively. On clayey soils, mean C and N concentration of fLF was 40±6% and 1.5±0.3% respectively while in the oLF C and N were at 41±4% and 1.8±0.2%. The MaHF C and N were 2.8±1% and 0.19±0.07% respectively. Despite the small proportion of the two light fractions to the soil mass, they contributed to 6±2% of the bulk SOC and 1.7±1.3% of the total N stocks (Table 2.3).

Table 2.3. Distribution of soil organic matter fractions and their relationship to SOC and TON as affected by different land management system at 0–30 cm depth.

	Management	fLF + oLF C	fLF C/ SOC	fLF N/TON	oLF C/SOC	oLF N/TON	MaHF/SOC	MaHF/TON	C:N Ratios		
		(g C m^{-2})	%						fLF	oLF	MaHF
Sandy	CT	24.1(5.3)^{a}	4.7	2.5	1.5	1.0	93.8	96.6	25.5	21.4	13.1
Soil	RP	37.8(5.3)^{ab}	5.9	3.0	1.5	0.9	92.6	96.1	24.8	20.8	12.1
	DS	35.7(5.3)^{a}	5.0	2.7	1.5	0.9	93.4	96.4	24.9	21.8	12.8
	NF	76.9(7.6)^{b}	4.1	2.5	1.3	0.7	94.6	96.8	23.0	22.6	13.1
	HSD	8.4	6.8	2.3	0.5	0.2	7.4	2.5	15.6	2.9	7.2
Clayey	CT	97.3(5.6)	5.4^{a}	3.0^{a}	1.2	0.74	93.4^{a}	96.2^{a}	25.8^{ab}	21.4	14.1
Soil	RP	109.9(5.3)	5.9^{a}	2.7^{ab}	1.2	0.69	92.9^{a}	96.6^{ab}	26.9^{ab}	20.8	12.5
	DS	89.9(5.3)	4.9^{a}	2.3^{b}	1.3	0.70	93.8^{a}	96.9^{ab}	27.8^{b}	21.8	13.0
	NF	85.7(7.5)	2.7^{b}	1.5^{c}	1.5	0.79	95.8^{b}	97.7^{b}	24.5^{a}	22.6	13.6
	HSD	19.2	4.8	1.4	0.4	0.1	5.5	1.8	11.4	1.9	11.2

CT =conventional tillage, RP = minimum tillage with a ripper, DS= no tillage using direct seeder, NF= natural forest, BD = bulk density. fLF = free light fraction, oLF = occluded light fraction, MaHF = mineral associated heavy fraction. Standard error of the mean shown in parenthesis. Means followed by different letters are significantly different in each soil type at p=0.05. HSD = Honest Significant Difference.

The amount of organic C stored in density fractions and their depth distribution differed between land use types (Figure 2.4). The fLF C and N were significantly larger in clayey soils than in sandy soils and significantly larger in the top layer than the lower layer (P<0.01). Tillage systems on sandy soils had smallest average fLF and oLF C stocks (25.3 ± 1.3 g m^{-2} and 7.3 ± 1.2 g m^{-2}) at 0–30 cm when compared with corresponding NF (58.4 ± 4 g m^2 and 18.5 ±1.0 g m^{-2}). The two light fractions were not significantly different between RP and DS on sandy soils. Clayey soils, had the opposite, having all fLF C and N in tillage systems being higher (80. ± 12 g C m^{-2} and 2.7 ± 0.4 g N m^{-2}) than NF (57.4 ± 2.0 g C m^{-2} and 2.4 ± 0.3 g N m^{-2}). The fLF N was significantly lower under CT than DS and RP on sandy soils whereas on clayey soils fLF N was lowest in NF.

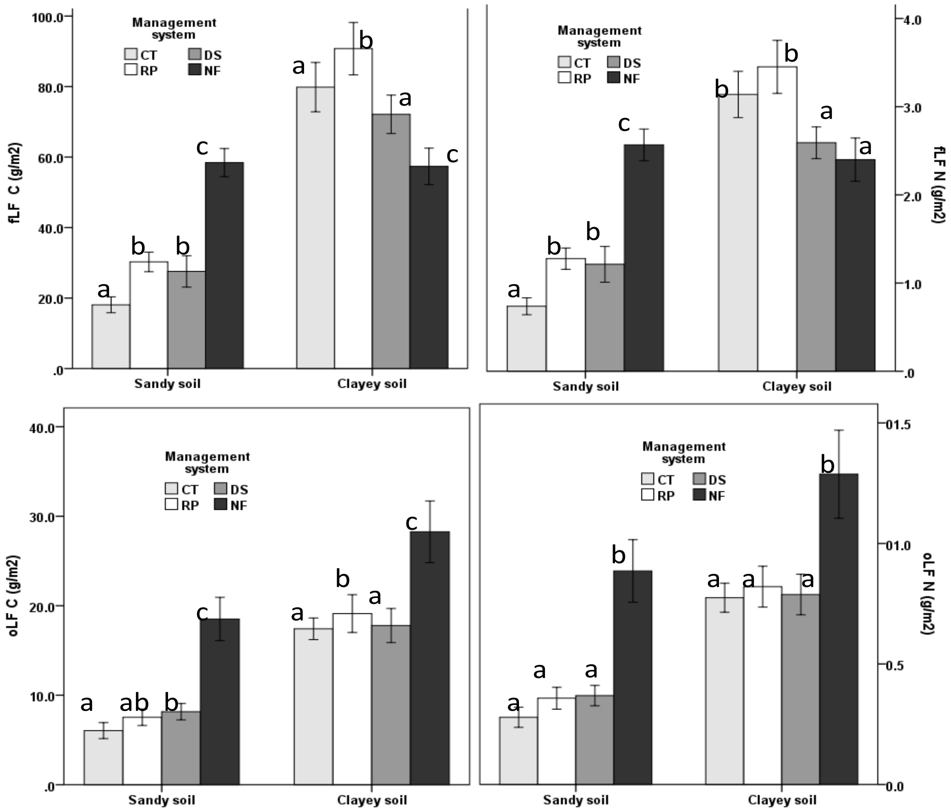

Figure 2.4: Distribution of carbon and nitrogen in free light fraction (fLF) and occluded light fraction (oLF) in three tillage systems (CT –conventional tillage, RP-Ripping and DS – Direct seeding) and natural forest(NF) in sandy and clayey soils up to a depth of 30 cm. Different letters show significant differences of the mean at *p*=0.05.

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The amount of organic C stored in density fractions and their depth distribution differed between land use types (Figure 2.5 and 2.6). The fLF C and N were significantly larger in clayey soils than in sandy soils and significantly larger in the top layer than the lower layer (P<0.01). Tillage systems on sandy soils had smallest average fLF and oLF C stocks (25.3 ± 1.3 g m^{-2} and 7.3 ± 1.2 g m^{-2}) at 0–30 cm when compared with corresponding NF (58.4 ± 4 g m^{2} and 18.5 ±1.0 g m^{-2}).

The two light fractions were not significantly different between RP and DS on sandy soils. Clayey soils, had the opposite, having all fLF C and N in tillage systems being higher (80.9 ± 12 g C m^{-2} and 2.7 ± 0.4 g N m^{-2}) than NF (57.4 ± 2.0 g C m^{-2} and 2.4 ± 0.3 g N m^{-2}). The fLF N was significantly lower under CT than DS and RP on sandy soils whereas on clayey soils fLF N was lowest in NF.

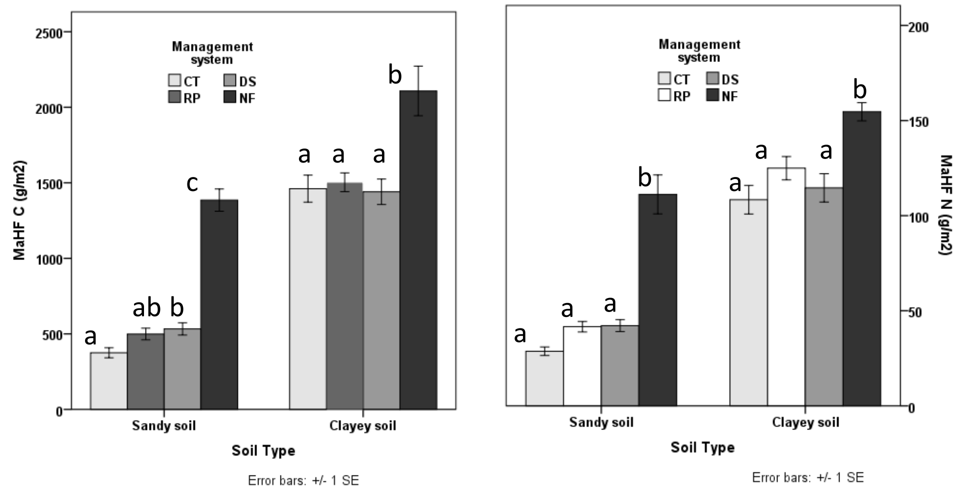

Figure 2.5: Distribution of carbon and nitrogen in mineral associated heavy fraction (MaHF) in three tillage systems (CT –conventional tillage, RP- Ripping and DS – Direct seeding) and natural forest(NF) in sandy and clayey soils up to a depth of 30 cm. Different letters show significant differences of the mean at *p*=0.05.

The oLF C and N fractions were lowest under CT and highest under DS on sandy soils. On clayey soils RP significantly increased oLF C and N at 0-10 when compared with DS and CT. At 10-30 cm DS had more C and N than RP and CT although they were all lower than NF. The largest proportion of soil C and N was found in the MaHF (C: 93±6%; N: 96±2%) and (C: 94±4%; N: 97±1%) on sandy and clayey soils respectively (Table 2.3). Thus, MaHF represented a consistent trend relative to land management system (Figure 2.6) and was significantly correlated with total SOC (R^2 = 0.998; *p* <0.01; n=168; SE =0.29) although the range was 87-98% of total SOC and TON. The MaHF C and N were higher in top than lower soil depth in all management systems. Interaction of depth and texture significantly affected MaHF and oLF C and N contents.

The C:N ratios for MaHF were smaller than fLF and oLF ranging from 9-21 compared to 20-30 for fLF and 19-25 for oLF. The mean C:N ratios decreased in the order fLF (23±6) > oLF (20±3) > HF (13±3) across sites on sandy soils with similar trends on clayey soils of fLF (26±4) > oLF (23±2) > HF (13±3). The C:N ratios of oLF decreased with soil depth for most sites while C:N ratios of fLF remained constant.

2.4 Discussion

2.4.1 Surface litter retention in agricultural and forest systems

Forests had more C and N in litter than agriculture systems (Figure 2.2) although their potential is retarded by the removal of forest litter often used as soil amendments in agricultural fields (Kowero, 2003; Giller *et al.*, 2006) affecting soil organic matter input. Regeneration of forest ecosystems can be seriously affected by factors such as seed fall, seed bank, nutrient availability and microclimate (Holl, 1999). In addition to soil texture, size and composition of above ground biomass, history of utilisation also affects the accumulation of organic matter (Sleutel *et al.*, 2011). Thus, in areas from which a large part of the surface layer of the soil is removed or disturbed, as in this study and most severely in the Nyarukunda (sandy) area, restoration of the SOC and N levels can be slow. Despite this, the forests remain important in C storage. Inputs through litter accumulation in miombo woodlands is also be affected by frequent fires.

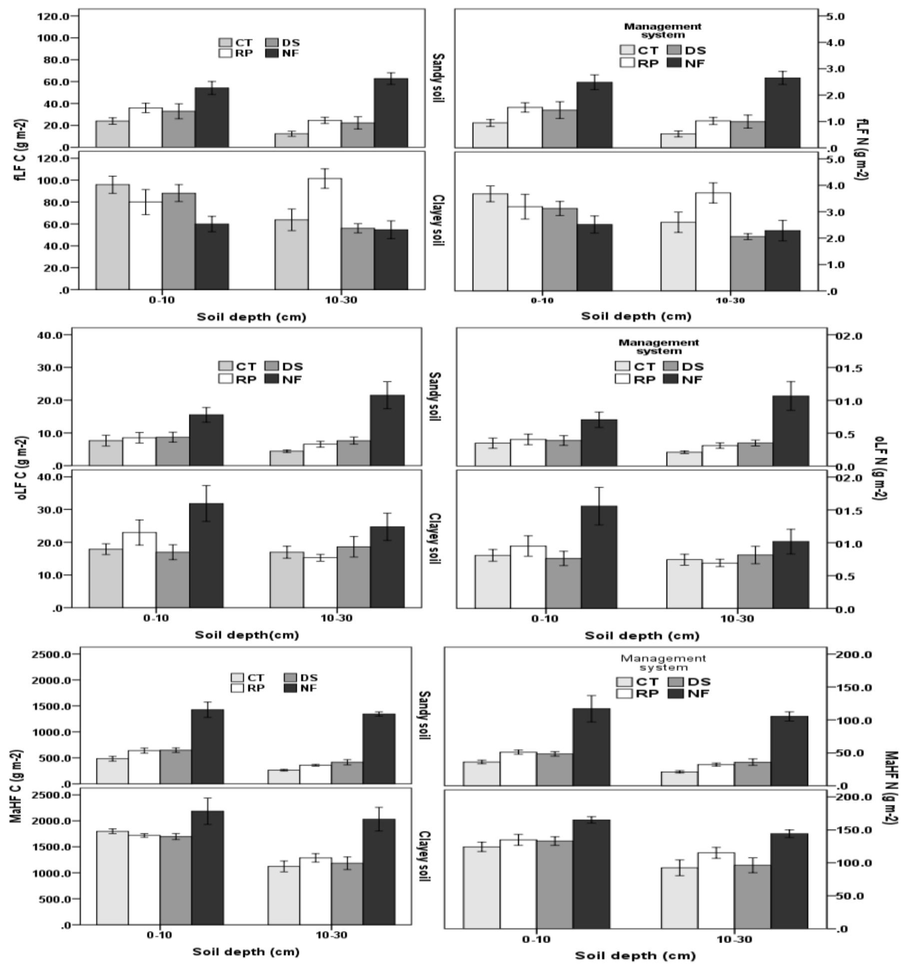

Figure 2.6: Distribution of organic C and N in density fractions (fLF = free light fraction, oLF = occluded light fraction and MaHF = mineral associated fraction) in three tillage systems (CT—conventional tillage, RP—ripping and DS—direct seeding) and natural forest (NF) at two depth intervals.

Conservation tillage systems (RP and DS) should receive annual organic matter inputs of 2.5 – 3.0 Mg ha^{-1} which is normally supplemented by thatch grass material which is easily degraded and only persists till end of the growing season after which the crop residues take over. The effects of residue retention may even be larger than other factors such as soil texture or rainfall as the residue retention was more applicable in the initial years when the plots were protected from free ranging livestock resulting in DS plots having more C than CT and RP (Thierfelder & Wall, 2012). At the time of the study all fields were no longer protected. Crop residues are sometimes collected to feed livestock in the dry season or the livestock feed *in situ* resulting in very low organic inputs into the soil. This has an effect on the amount of C a system can store and may result in low C accumulation under conservation agriculture (CA) systems. The attainment of complete cover in CA systems may not be practical in Zimbabwean rural systems under existing local institutional arrangements. In addition, the thatch grass used to supplement residues is sometimes threatened by annual wildfires strengthening the need for community fire management and fire prevention policies.

However, climatic condition, including the long dry spell from April to November, is a major limiting factor to the preservation of organic matter in all tillage systems as reflected by the low amounts of surface litter. During this period the sites are equally disturbed by being subjected to trampling by animals, excessive dryness, termite degradation of organic residues and sometimes wind erosion (mostly in August and September).

2.4.2 Soil bulk density, C and N in different land management systems

Bulk densities found in the study at 0-10 cm (Table 2.2) are comparable to those found by King and Campbell (1994) and Walker and Desanker (2004) who compared miombo woodlands and arable lands in the southern African region. Sandy soils tend to have higher bulk densities by virtue of their higher sand content which makes them more vulnerable to trampling by grazing animals and humans especially during the dry season.

Soil organic C stocks differed greatly between agriculture and forested land with the two forests having higher C and N contents as compared to agriculture lands (Table 2.2; Figure 2.3). Our results (46.11 Mg ha^{-1} and 53.9 Mg ha^{-1}) are comparable to C stocks found in other woodlands of southern Africa ranging from 32 -133 Mg ha^{-1} (Woomer, 1993 ; Walker & Desanker, 2004; Zingore *et al.*, 2005; Ryan *et al.*, 2011). In this study, soil texture was important in determining the amounts of C and N storage in soils since the two sites receive equivalent amounts of rainfall. Clayey soils have greater protection allowing slow decomposition and depletion of organic matter while on sandy soils organic matter is more exposed and vulnerable to rapid decomposition in the presence of favourable environmental conditions. The clay and silt size particles have larger surface areas which allow better stabilisation of organic matter than sandy soils (Powlson *et al.*, 2013).

Although the current stocks were not significantly different among tillage practices. A comparison of C stocks at 0-10 cm (Tables 2.2 and 2.3) on sandy soils shows an increase of 0.38, 0.93 and 1.46 Mg ha^{-1} over the four year period for CT, RP and DS respectively. Increase of C stocks in top layers has been observed elsewhere (Gulde *et al.*, 2008; Anikwe, 2010; Dercon *et al.*, 2010; Blanco-Moure *et al.*, 2011). Brahim *et al.* (2009) also found DS having greater C stocks in 0-10 cm and the 10-20 cm layers with higher C stocks under CT at 20-30 cm. Contrary to this study, Anikwe (2010) and other scholars (Tan et al., 2007; Jagadamma & Lal, 2010) found significant differences between C and N stocks of CT and no tillage (DS).

Crop rotation using leguminous crops has been done in all tillage systems and this practice is known to enhance microbial activity (Dinesh *et al.*, 2004) and may have an impact on the insignificance of C stored in the three tillage system although usually little residue was left in the field to last until the next cropping season. The C stocks under CT in this study are higher than reported by Thierfelder *et al.* (2012) who worked in the same study area and reported a stock of 3.2 Mg C ha^{-1} at 0-10 cm while at 10-30 cm their stocks were 7.8 Mg ha^{-1} compared to 7.97 Mg ha^{-1} in this study. Their study was however, based on fields initiated in 2004 whereas in this study the results are the mean of four farmers' fields established in 2005/2006 season. In their DS fields, C stocks at 0-10 cm were less than half of those reported in this study while at 10-30 they found 10 Mg C ha^{-1} compared to 4.46 Mg C ha^{-1} in this study.

C stocks in clayey soils under CT, RP and DS increased at by 3.05, 2.14 and 0.38 Mg ha^{-1} respectively 0-10 cm when compared with the baseline data (Table 2.1 & 2.2). The CT management system had the lowest C stocks at 10-30 cm where RP was highest resulting in the RP having highest stocks at 0-30 cm although the difference between CT and DS was not significant. All tillage systems had significantly lower C and N when compared to NF. In Zambia, Stroosnijder and Hoogmoed (2004) found C sequestration rates of 0.3 – 3.0 Mg ha^{-1} yr^{-1} under conservation tillage systems over a nine year period. Increases of total organic C in the short term have been reported after three years (McCarty *et al.*, 1998) although the increases can be insignificant (Franzluebbers & Arshad, 1996; Liang *et al.*, 2007).

Similarly, Thierfelder and Wall (2012) also analysed C stock changes at some of the fields in Hereford and found increases of 19%, 21% and 38% under CF, RP and DS from 2004 to 2008 at a depth of 0-20 cm. They showed increases of 6.9, 7.6 and 10.2 Mg ha^{-1} respectively (i.e. 1.7, 1.9, 2.6 Mg ha^{-1} yr^{-1} respectively). They found highest increases under DS while CT and RP were not significantly different. Differences with our results mainly under DS systems, could be attributed to the lack of continuous cover in the designated fields by the year 2010 as the fences were broken. This shows the importance of residue retention to gain greater positive impacts under RP and DS in both sandy and clayey soils. Despite spatial heterogeneity and the differences in amounts of C and N, CT and RP systems may have benefited from homogenisation of soil profile with the plough (CT) or ripper (RP) and organic matter accumulation which however, resulted in a decrease in the concentration gradient of C with soil depth when compared with NF on sandy soils. Clayey soils tend to benefit more from minimum tillage while sandy soils could be more sensitive to this disturbance.

These results suggest that RP could be an effective soil management technique for increasing soil C storage on clayey soils.

Studies which have shown major differences between CT and DS had continuous cover under the no till system (Brahim *et al*., 2009), which was not the case in this study. Our results agree with Poirier *et al.* (2009) who found no significant differences between tillage systems although they considered the profile up to 60 cm. They found significant differences in the top 20 cm which is similar to this study at 10-30 cm. Studies on similar soils show a decline of C and N through CT (Chivenge *et al.*, 2007a) which is not evident in this study. In this study the ox-drawn plough may not be going as deep as 30 cm leaving the soil below 30 cm depth relatively unaffected.

Contrary to most tillage studies, de Rouw *et al*. (2010) found significant differences between DS and CT with highest C (+590g C m^{-2}) in CT and a loss of (-133 g C m^{-2}) under DS over a five year period. The loss of C was mainly attributed to the slow process of biomass decomposition in no till systems (DS). They concluded that tillage helps to incorporate residues that may never be captured for decomposition under no till. In addition, the mouldboard plough helps to prepare a weed-free seedbed and facilitates percolation of water into the soil profile (Riches *et al.*, 1997). This could be the same case in our study area were variable amounts of residue are not preserved in place in CA systems after harvesting and equal amounts of fertiliser are added to all treatments annually. Other studies on sandy soils have shown that the incorporation of organic matter into the mineral soil by cultivation facilitates interactions between organic residues and inorganic colloids thus physically stabilising the residues and reducing decomposition rates resulting in preservation of soil C (VandenBygaart *et al.*, 2002).

The high C:N ratios, reflect less microbial activity and also shows the inherent N deficiency in these soils (Nyamangara *et al.*, 2000) supporting the need for supplementing with fertiliser and/or manure to increase crop production (Giller, 2002; Zingore *et al*., 2007) and C sequestration. The observed increase in C stocks in all tillage systems could be a result of better nutrition and improved management of the fields compared to the time prior to the start of the experiments and the addition of fertiliser has been shown to increase soil C stocks in continuously tilled fields after nine years (Mujuru *et al.*, Submitted).

2.4.3 Soil C and N dynamics in density fractions

We observed contrasting effects of tillage on the fLF C and N concentrations in the sandy and clayey soils (0-10 cm), (i.e. in the sandy soils increasing fLF C present under respectively CF, DS, RP and NF systems, while for the clayey soils we observed decreasing fLF C under respectively RP, CT, DS and NF (Figure 2.4). The significant difference between fLF C and N on sandy and clayey soils shows the fraction's vulnerability to management activities on soils of different texture when compared with the amount of light fraction in natural forests (Figure 2.4). This shows the sensitivity of this fraction to changes in soil management (Sharifi *et al.*, 2008; Yoo & Wander, 2008; Sequeira *et al*., 2011; Xu *et al.*, 2011) proving that the light fraction maybe most affected by land management practices (Tan et al., 2007) on both sandy and clayey soils. For sandy soils, the less favourable microbial living conditions may induce larger turnover time of C in the unprotected fLF when compared to clayey soils. Conventional tillage may have enhanced the incorporation of above ground litter into the mineral soil and therefore contributed to the initial phase of soil C stabilisation. This mixing effect turned out to be more effective in the well-structured clayey soils than in the less-structured sandy soils.

Tan *et al*. (2007) found significantly higher fLF C under DS than CT which is consistent with our results on sandy soils. The fLF C was also reduced with depth in all land use systems. Studies by Balesdent *et al*. (2000) suggested that tillage affected fLF through increasing the rate of aggregate destruction which causes a decrease in organo- mineral complexes within aggregates associated with the fLF. The fLF C is considered to be more labile than oLF C (Hassink, 1995). The significantly higher fLF C and N in tillage systems than NF on clayey soils could be a result of persistent annual fires that retard addition of fLF to forest soils.

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The oLF was higher in NF than tillage systems. Among tillage systems the oLF C was higher in DS and RP than CT. However, the oLF C and N fractions were marginally small in the sandy soil, showing that SOM stabilisation by physical protections is nearly absent in cropping systems whereas on clayey soils, the physically protected oLF C and N was considerable compared to fLF. Favourable moisture and temperatures during the cropping season promotes greater mineralisation of light fraction organic matter under RP and DS resulting in less accumulation of fLF C and N.

Differences in C:N ratios between fLF and oLF were probably due to a slightly advanced form of decomposition in oLF (Hassink, 1995). The fLF C:N ratios were similar to those reported by Gregorich *et al*. (2006) which were between 17 and 22. The N contents which were close to the detection limit mainly in the light fractions may contribute to greater variability of C:N ratios at the two depths in all land management systems. In addition, a larger C utilisation in the clayey top soil might be a result of (micro-) biological activity mainly during the cropping season. The utilisation of fresh litter (substrate) in a soil involves processes where, microbes respire C as CO_2 and assimilate C and N resulting in a decrease of the average C:N ratio of the fLF.

In contrast to amounts of C and N in light fractions, the MaHF contributed most of the C an N with values of 86-97% and 90-98% on sandy and clayey soils respectively. Thus MaHF represented a consistent trend relative to land use system (Figure 2.5) increasing under RP and DS on clayey and sandy soils respectively. In each soil type this fraction was not affected by tillage as shown by no significant differences. The total SOM retention capacity of the clayey soils based on physical protection (oLF) and adsorption onto mineral surfaces (MaHF) was twice as much as on sandy soils.

The MaHF accounted for the largest proportion of C and N in all tillage and forest systems suggesting that C loss after conversion from forest to cropland caused a reduction in both light and heavy fraction of organic matter (Figure 2.4 and Figure 2.5) mainly in the top 10 cm of the clayey soils and the 10-30 cm layer of the sandy soil. Thus the higher the clay content, the larger the difference between LF and MaHF.

2.5 Conclusion

The present C and N stocks were lower in tillage systems than NF in the two soil types. More C and N were stored at 0-10 cm than 10-30 cm depth. Although the total C and N stocks were not significantly different, the amount of C addition to the soil differed significantly between CT and DS on sandy soils and was highest under CT and lowest under DS on clayey soils. Over a four year period, the three tillage systems under CT, RP and DS had positive impacts on SOC stocks at 0-10 cm with magnitudes of 0.38, 0.93, 1.46 Mg ha^{-1} and 3.05, 2.14, 0.38 Mg ha^{-1} on sandy and clayey soils respectively. Our results do not support the hypothesis that losses of C and N from the soil are associated with conventional farming and less with no-till practices, thus conflicting with other studies, which found decreases of both C and N stocks in conventionally cultivated lands.

The SOM fractions were dominated by MaHF C and N which accounted for 86-93% and 94-98% on sandy and clayey soils respectively. In these semi-arid areas, the protection of organic C and N in MaHF seems to be a main cause for soil enrichment by organic matter. Results also show that clayey soils can store more C and N after mild disruptions (RP) although difference with CT were not significant. The MaHF in the sandy soils indicates that SOM adsorption onto mineral surfaces is by far the most effective stabilisation mechanism although, when compared to the clayey soils, this total SOM stabilisation capacity is rather limited. The C:N ratios for MaHF were smaller than fLF and oLF ranging from 11-17 compared to 20-30 for fLF and 17-22 for oLF. The separation of SOM into fLF, oLF and MaHF allowed us to assess the capacity of tillage systems to enhance the physical protection of organic C and N within sandy and clayey soils.

Results from this study may also suggest that crop residues are important for successful RP and DS in both sandy and clayey soils and the effects of residue may even be larger than other factors. Therefore in these soils, under the prevailing climatic and management conditions, the issue of tillage alone cannot help in C sequestration unless institutional arrangements for residue retention are improved. The exploitation of the benefits of CA and corresponding sustainability of systems need support for surface cover retention (de Moraes Sá, *et al*. 2011) which can also be extended to conventional tillage.

Chapter 2

CHAPTER 3

Effects of nitrogen fertiliser and manure application on soil organic carbon and nitrogen storage under continuous maize cropping on Arenosols and Luvisols of Zimbabwe

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Chapter 3

Effects of nitrogen fertiliser and manure application on soil organic carbon and nitrogen storage

Abstract

The maintenance of soil organic carbon (SOC) pool in agricultural soils is important for long term crop productivity and represents a potential sink for atmospheric carbon dioxide. Data from a nine year continuous maize experiment were analysed to assess the effects of fertilisation on SOC and total organic nitrogen (TON) in clayey (Luvisols) and sandy (Arenosol) soils of Murewa district, Zimbabwe. Density fractionation was used to assess the distribution of SOC and TON in three soil organic matter (SOM) fractions and their sensitivity to fertilisation on two fertility gradients. The relationship between light and heavy fraction organic C were analysed to determine equilibrium levels. Treatments included: unfertilised control (control), nitrogen fertiliser (N Fert) and nitrogen fertiliser plus cattle manure (N Fert + manure). Results showed significantly higher SOC and TON stocks under N Fert and N Fert + manure than control at all depths. In clayey soil, C and N stocks in N Fert were significantly higher than control and significantly less than N Fert + manure. On sandy outfields, N Fert had highest C and N than control and N Fert + manure at all depths. On clayey soil, mean total organic C at 0-50 cm depth was: control (17.98±1.46 Mg ha^{-1})<N Fert (26.73±1.58 Mg ha^{-1})< N Fert + manure (31.99±1.42 Mg ha^{-1}) while on sandy soil it was: control (5.98±1.42 Mg ha^{-1})< N Fert + manure (10.71±1.41 Mg ha^{-1})<N Fert (11.76±1.34 Mg ha^{-1}). Nitrogen followed similar trends. Compared with control, N Fert and N Fert + manure enhanced fLF C on homefields and outfields by 19%, 24% and 9%, 22% on clayey soil and 17%, 26% and 26%, 26% respectively on sandy soil. Homefields on clayey soil, under N Fert and N Fert + manure had similar equilibrium levels, being 2.5 times more than control. The equilibrium levels for MaHF C under N Fert and N Fert + manure treatments were higher than current stocks in both homefields and outfields. Carbon stocks under control in outfields were similar to theoretical equilibrium levels of 16.1 Mg ha^{-1} whereas in homefields equilibrium levels were only 3 units higher than current stocks. Results suggest that the application of manure can be a low cost alternative for enhancing soil quality and promoting soil C sequestration under conventionally tilled continuous maize cropping in Zimbabwe.

Effects of fertilisation on organic C and N storage

Key words: manure, fertiliser, sequestration, light fraction, heavy fraction, equilibrium level.

3.1 Introduction

The importance of agricultural soil for climate change mitigation through soil C sequestration has dominated current global climate change debates (Food and Agriculture Organisation (FAO), 2010a) and important for sustainable soil productivity (Lal 2004). Soil organic carbon (SOC modifies soil physical, chemical and biological properties, eventually affecting nutrient provision, water retention and microbial nourishment (Jimenez *et al.*, 2002a). The amount and stability of SOC in agricultural soils determines their productivity and contribution to sustaining environmental quality (Lal, 2004). Agriculture practices therefore, influence the amounts of C stocks in small holder farming systems including a potential role in sequestering C to diminish GHG concentrations and global warming.

Practices that have been identified for enhancing SOC sequestration in agricultural soils include i) application of manure and inorganic fertiliser (Liu, 2004; Gulde *et al*., 2008; Gong *et al*., 2009), ii) conservation tillage (Six *et al*., 2000; Chivenge *et al*., 2007b; Marongwe *et al*., 2011), iii) crop rotation (Havlin *et al.*, 1990; Paustian *et al.*, 1998), iv) improved fallowing, v) mulching (Mafongoya & Dzwowela, 1999) and vi) intercropping. A combination of manure and inorganic fertiliser was superior for improving SOM contents and its fractions eventually enhancing C sequestration (Manna *et al.*, 2005; Rudrappa *et al.*, 2006; Purakayastha *et al*., 2008; Li *et al.*, 2010) mainly from increased dry matter production. Contrary to these positive results, Wu et al (2004) found decreasing SOC after application of fertility amendments while others found no effect of fertility amendments on SOC storage (Šimon, 2008).

Most arable fields in smallholder farming areas of eastern and southern Africa are characterised by gradients of decreasing soil fertility with increasing distance from homesteads and the fertility gradients are a result of preferential application of manure, household waste, and inorganic fertilisers on fields closest to homesteads (Tittonell *et al.*, 2005; Zingore *et al.*, 2007b). These resource allocation strategies cause significant increases in the quantity of SOC after addition of more farmyard manure and inorganic fertilisers to fields close to homesteads than fields away from homestead on both clayey and sandy soils (Zingore *et al.*, 2008; Dunjana *et al.*, 2012). Increased SOC and TON were closely related to increases in crop yields. Progressive studies were done in Murewa district, Zimbabwe at 3 years (Zingore *et al*., 2008) and at 7 years (Dunjana *et al*., 2012) and confirmed decreasing C and N with distance from homesteads. Similar results have also been reported elsewhere (Mapfumo & Giller, 2001; Tittonell *et al*., 2005; Zingore *et al*., 2008; Masvaya *et al.*, 2010). However, Haileslassie (2007) found contradictory results with greater soil fertility in fields away from homesteads while working in the east African highlands of Ethiopia.

The losses or gains of SOC and TON over short and medium term are difficult to detect due to spatial and temporal variability in bulk soil and may not be a sensitive indicator of short and medium term changes in soil quality (Hassink, 1997). Labile organic C fractions such as free light fraction (fLF) and dissolved organic carbon can be used as early and sensitive indicators of changes in SOC (Haynes, 2000). Light fraction organic C is important in reflecting short term turnover of nutrients (Hassink, 1997) with greater concentrations of N than the mineral associated heavy fraction (Compton & Boone, 2000). Methods such as density fractionation, size separates or a combination of the two divide SOM into three basic levels of functional, structural complexity and different turnover rates (von Lützow *et al*., 2006). The light fraction is easily decomposed, has a greater turnover rate and is smaller than the mineral associated fraction (MaHF) and can be used to characterise impacts of different soil amendments.

The importance of the free light fraction (fLF), occluded light fraction (oLF) and mineral associated fraction (MaHF) is widely recognised in studies outside Southern Africa (e.g. (Dalal & Mayer, 1986b; Janzen *et al.*, 1992; Liu *et al.*, 2005; Swanston et al., 2005) but the relationship between the three fractions is poorly understood in African soils. Yin *et al*. (2005) and Yin and Cai (2006) studied this relationship by using a linearized form of the Langmuir equation and found a positive correlation between light fraction organic matter (LFOM) and the ratio of LFOM and heavy fraction organic matter (HFOM) (i.e. LFOM/HFOM ratio) under organic and inorganic fertiliser treatments in China. They calculated the potential increase and the maximum equilibrium value of HFOM of soils from three long-term experimental fields in China. Their LFOM/HFOM ratios ranged between 0.01 and 0.50 with higher values under organic fertiliser amendments. Knowledge of the equilibrium levels allows evaluation of the potential of different soil types and agricultural practices for more soil C sequestration (Yin & Cai, 2006). Maintaining soil and crop productivity is a major challenge especially in rain fed arid and semi - arid areas characterised by long annual dry spells. The objectives of this study were to determine how nitrogen fertiliser and cattle manure application in continuously tilled maize cropping systems affect: 1) total SOC and total soil organic nitrogen (SON) relative to distance from homestead and soil depth; 2) C and N storage in three density separated SOM fractions and their sensitivity to fertilisation and 3) the relationship between light and heavy fraction organic C. We hypothesised that long term fertiliser and manure application will increase SOC in whole soil and eventually in stabilised C fractions; accumulation of SOC and TON in SOC fractions is influenced by the long-term application of N fertiliser and manure in conventionally tilled continuous maize cropping systems; the light fraction would be a responsive indicator of short and long term changes in SOC. We also hypothesised that there is a stronger relationship between LF and MaHF in clayey soils than in sandy soils, because of the larger adsorption capacity of clay particles for organic molecules.

3.2 Materials and methods

3.2.1 The study site

The soils were collected from Murewa district of Zimbabwe which is located ~80 km east of Harare, latitude 17° 39' 13" S and longitude 31° 48' 30" E. The area receives unimodal rainfall of between 750-1000 mm annually, with a mean maximum temperature of 26°C and mean minimum temperature of 14°C. The soils in the area are predominantly granitic sandy soils (Haplic Arenosols)

(FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) strongly weathered having, low levels of available nutrients and low nutrient reserves. The sandy soils are interspaced with pockets of dolerite intrusions that give rise to small patches of more fertile clay soils (Chromic Luvisols) (Nyamapfene, 1991; FAO, 2006).

Farming practices are based on integrated crop/livestock systems with maize (*Zea mays* L.) as the dominant staple crop. In addition, other crops such as sun flower (*Helianthus annuus* L.), groundnut (*Arachis hypogaea* L.), sweet potatoes (*Ipomoea batatas* L.) and assorted vegetables are also grown. Cattle are the main livestock, communally grazed in open access areas and in arable fields during the dry season. They are corralled close to homesteads at night. In most cases crop residues are used to feed cattle over the dry period and organic fertiliser is used to fertilise crops, with a small proportion of farmers using residues as mulch in fields where conservation agriculture (CA) is practised. Zingore *et al*. (2007b) showed that available P and SOC decreased significantly with distance from homesteads, particularly where farmers have cattle due to preferential application of manure and mineral fertilisers on fields closest to the homesteads. Cattle provide draught power for conventional tillage and for ferrying fertility inputs to the fields.

3.2.2 Characterisation of experimental sites

Soils were collected from experimental positions after nine years of continuous sole maize (*Z. mays* L.) cultivation, based on soil fertility gradients and treatments described by Rusinamhodzi *et al*. (2013). Two farms in the medium wealth category were selected on fields with contrasting soil types: one on Arenosol (sandy soil) and another on red clay soil which is a Luvisol (clayey soil). Each field represented typical homefields (<50 m from homestead) and outfields (100–500 m from homestead) as described by Zingore *et al*. (2007). Mean SOC concentrations given by Zingore *et al*. (2007) for homefields and outfields on sandy soil were 5 and 3 g C kg^{-1} whereas homefields and outfields on clayey soils had 14 and 7 g C kg^{-1} respectively. The TON concentrations in homefields and outfields were 0.4, 0.3 g kg^{-1} and 0.9, 0.5 g kg^{-1} on Arenosols (sandy soil) and

Luvisols (clayey soil) respectively. The C:N ratios were 13, 10 and 16, 14 respectively. Biomass yields generally increased with time in treatments under N fertiliser and Manure (Table 3.1). Residue retention was minimal and less than 0.5 Mg ha^{-1} due to competing uses and the free range grazing in the off season period.

Experimental design in each soil type was a split plot design within randomised complete block. This paper reports the results of three treatments in each soil type and field position selected on basis of applicability to smallholder farmers: 1) unfertilised control (control), 2) Nitrogen Fertiliser (N Fert) = 100 kg N ha^{-1} split applied as ammonium nitrate in equal amounts at about 3 and 6 weeks after plant emergence and 3) Cattle manure (0.9% N) + ammonium nitrate (100 kg N ha^{-1}) (N Fert + manure) where cattle manure was applied at 5 Mg ha^{-1} (equivalent of 10 kg P ha^{-1}), prior to each cropping season.

Table 3.1: Mean maize grain yields for the period 2003-2011

Soil type	position	Treatment	% change in grain yield	Increase/decrease in grain yield (Mg ha^{-1})
Clayey	Homefield	Control	-60	-1.5
Clayey	Homefield	N Fert	-12	-0.6
Clayey	Homefield	N Fert+ manure	12	0.7
Clayey	Outfield	Control	-12	-0.1
Clayey	Outfield	N Fert	56	1.5
Clayey	Outfield	N Fert+ manure	2	0.1
Sandy	Homefield	Control	-34	-0.5
Sandy	Homefield	N Fert	-20	-0.5
Sandy	Homefield	N Fert+ manure	4	0.3
Sandy	Outfield	Control	-34	-0.1
Sandy	Outfield	N Fert	24	0.2
Sandy	Outfield	N Fert+ manure	66	1.1

Source: Rusinamhodzi *et al* 2011.

3.2.3 Soil sampling and analysis

Four representative soil samples were collected from each soil type and field position (i.e. homefields and outfields) at four incremental depths up to 50 cm. Samples were composited for each depth and replication, air dried and passed through a 2-mm sieve before analysis of C and total N concentrations by dry combustion in a EA1108 CHN Elemental analyser (Fisons Instruments). Additional samples were taken from all four depths using a metal ring (volume 100 g m^3) for bulk density measurement and results were expressed on oven-dry-basis. Bulk density was expressed as the ratio of dry weight of soil core and volume of metal core. Total SOC stock for each of the four depths (0–10, 10– 20, 20–30 and 30-50 cm) was computed separately for each treatment, field position and depth using equation [1].

$$\text{SOC stock (Mg ha}^{-1}\text{)} = \text{SOC concentration (\%)} * \text{BD} * \text{Depth (cm)} \qquad [1]$$

Where BD is bulk density (g/cm^3).

The SOC and TON in amended treatments was used to estimate the sequestration rate for each treatment relative to the unfertilised control.

3.2.4 Soil density fractionation

Sodium polytungstate (SPT) ($3Na_2WO_4.9WO_3.xH_2O$) was used to separate soil organic matter by density into three fractions, (fLF, oLF and MaHF) following the method described by Roscoe *et al.*(2000). A brief description of the method is given in Mujuru *et al.,* (2013) . Briefly, ten (10g) of air dried soil <2 mm were dispersed in sodium polytungstate solution (1.60 g cm-3), centrifuged at 4500 rpm, filtered and dried at 40 °C and weighed. Material left in the supernatant was considered to be fLF (mostly partially decomposed plant residues), whereas that in the sediment was a mixture of oLF and MaHF (more fully-decomposed residues and mineral material). SPT solution with density 1.6 g cm^{-3} was added to the residual soil material and an ultrasonic probe was used to disperse the occluded light fraction and centrifuged for 23 minutes, decanted over a Büchner funnel, oven dried at 40ºC. The remaining precipitate, the MaHF, needed to be rinsed thoroughly, (i.e. at least 5 times for clayey soils and three times for sandy soils, or until conductivity of the filtrate was <50µS cm^{-1}) in order to remove all SPT. Total C and N contents of fLF, oLF and MaHF were determined after grinding by dry combustion using a CN analyser.

3.2.5 Soil C sequestration potential

The Langmuir equation was used to evaluate the adsorption of fLF onto mineral surfaces, where the adsorbed organic matter, i.e. MaHF, is regarded as sequestered OM (Yin *et al.* (2005). We assumed that light fraction (LF) C is decomposed and adsorbed onto mineral soil particles as MaHF C over time and that soil minerals can randomly adsorb LF C until the MaHF has reached C saturation. Therefore, interaction between LFC and soil minerals was regarded as adsorption and desorption process which can be described using the Langmuir equation. The following linearisation was used to fit the data (Yin & Cai, 2006; Bolster & Hornberger, 2007).

$$LFC / MaHFC = LFC / MaHFC_{max} + 1 / (k\ MaHFC) \quad [2]$$

Where $MaHFC_{max}$ is the maximum adsorption capacity for organic C (equilibrium value for soil organic C in the MaHF) and k is the equilibrium constant.

LFC/MaHFC versus LFC yields a linear relationship with slope 1 / ($MaHFC_{max}$) and intercept 1 / (k MaHFC). Equation [2] was used to determine the relationship between light fraction (fLF +oLF) C and its adsorption onto mineral surfaces. The goodness of fit was checked through the coefficient of determination R^2 and the p value.

3.2.6 Statistical analysis

Data on C and N contents were tested for normality and homogeneity of variance using Kolmogorov-Smirnoff and Levene's test respectively. Effects of N fertiliser and cattle manure application on SOC and TON and density separated fractions within each depth were analysed using one way ANOVA. Differences were considered significant at $p \leq 0.05$. Tukey's HSD test was used to compare the mean differences between fertility treatments and depth for bulk soil and the three fractions (fLF, oLF and MaHF) in the two soil types. Linear regression was used to evaluate the relationship between light fraction and mineral associated heavy fraction carbon. All statistical analysis was performed with SPSS for windows version 19 and GENSTAT 15.

3.3 Results

3.3.1 Soil carbon and nitrogen contents

For each soil type, bulk density in homefields was not significantly different from bulk density in outfields ranging from 1.1 to 1.4 g cm^{-3}. In addition, bulk densities were not statistically different by depth in each field position. SOC and TON stocks of the 0-50 cm depth increment were significantly (p=0.000) higher (26.3 Mg C ha^{-1} and 1.6 Mg N ha^{-1} respectively) on clayey soil than sandy soils (12 Mg C ha^{-1} and 0.6 Mg N ha^{-1}) and also higher in homefields than outfields (Figure 3.1).

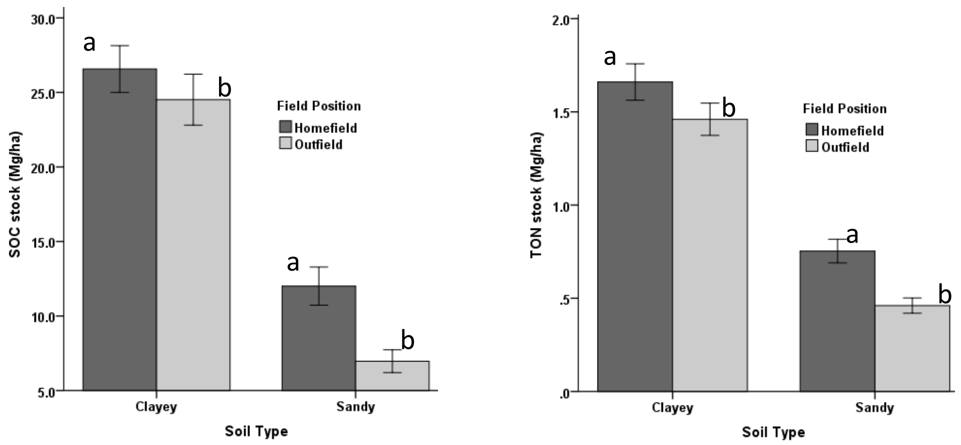

Figure 3.1: Soil C and N contents (0 – 50 cm) of homefields and outfields of clayey and sandy soils.

Effects of fertilisation on organic C and N storage

The storage of C and N was significantly affected by fertility amendment (Figure 3.2) when compared to the unfertilised control. Mean SOC on clayey homefields was significantly higher ($p < 0.015$) than outfields and followed the order: control (19.8±1.46 Mg ha^{-1}) < N Fert (28.8±1.58 Mg ha^{-1}) < N Fert + manure (31.9±1.42 Mg ha^{-1}), while on sandy soil it was control (8.2±1.42 Mg ha^{-1})< N Fert + manure (14.1±1.41 Mg ha^{-1}) ≈ N Fert (13.8±1.34 Mg ha^{-1}). Outfields followed similar trends for control, N Fert and N Fert + manure having 16.1, 24.6, 32.1 Mg C ha^{-1} and 3.4, 9.5, 7.7 Mg C ha^{-1} on clayey and sandy soils respectively. Therefore, application of N Fert and N Fert + manure increased C stocks on clayey soils by 1.0, 1.3 Mg C ha^{-1}yr^{-1} and 0.9, 1.8 Mg C ha^{-1}yr^{-1} in homefields and outfields respectively when compared to the control. On sandy soils, C stocks increased by 0.65, 0.62 Mg ha^{-1}yr^{-1} and 0.68, 0.48 Mg ha^{-1}yr^{-1} in homefields and outfields respectively relative to control.

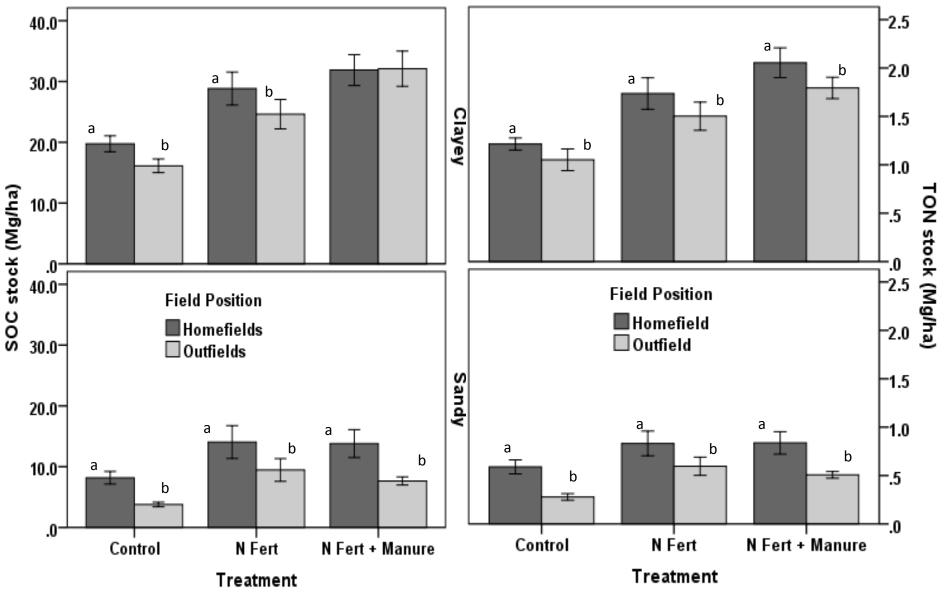

Figure 3.2: Effects of nitrogen fertiliser (N Fert) and nitrogen fertiliser plus cattle manure (N Fert + manure) application on SOC and TON stocks in homefields and outfields of clayey and sandy soils. Different letters show significant differences in each soil type and field position at p = 0.05. Error bars show ±1 standard error of the mean.

TON followed similar trends to those of SOC in all treatments except N Fert + manure in homefields. The nitrogen stocks on clay soils were also higher in homefields (1.7 Mg N ha^{-1}) than outfields (1.5 Mg N ha^{-1}) being significantly higher (p=0.000) under N Fert + manure (2.1 Mg ha^{-1}) than control (1.1 Mg ha^{-1}) whilst on sandy soil, nitrogen stocks were also greater in homefields (0.73 Mg N ha^{-1}) than outfields (0.37 Mg N ha^{-1}) and significantly higher (p= 0.000) in fertility treatments than control.

At all sampling depths, fertility amendments enhanced SOC and TON stocks better than the unfertilised control irrespective of field position (Figures 3.3 and 3.4). Application of N fertiliser plus cattle manure significantly increased the SOC stocks in soil compared to application of N Fert alone at all depths on clayey soils. On sandy soil, application of N Fert resulted in greater SOC than N Fert + manure and control at all depths except the 10−20 cm depth on homefields and 20−30 cm depth on outfields (Figure 3.3).

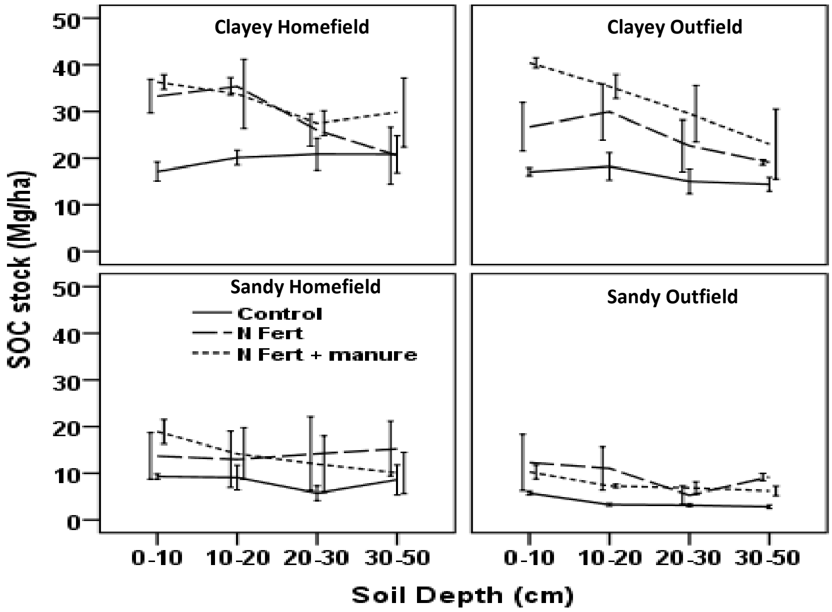

Figure 3.3. Depth distribution of SOC stocks under different fertility treatments on clayey and sandy soils. N Fert = nitrogen fertiliser, N Fert + manure = nitrogen fertiliser plus cattle manure.

Effects of fertilisation on organic C and N storage

Application of N Fert and N Fert + manure resulted in significantly higher (p=0.012) accumulation of SOC (33.3 and 36.4 Mg ha^{-1} respectively) than the control (17.1 Mg C ha^{-1}) at 0-10 cm and TON followed similar trends being higher under N Fert (2.1 Mg ha^{-1}) and N Fert + manure (2.3 Mg ha^{-1}) than control (1.1 Mg N ha^{-1}) (Figure 3.4). At 0-10 cm, application of N Fert + manure increased SOC by 15% while N Fert increased SOC by 11% relative to control treatment. Although the C and N decreased with increasing depth in all except N Fert in sandy homefields, there were no significant differences between the 30−50 cm depths of N Fert and control in homefields.

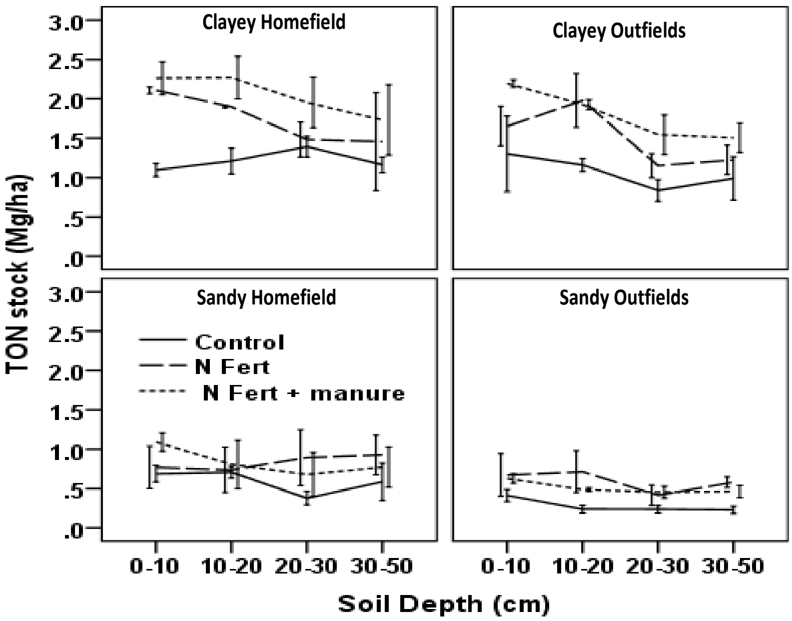

Figure 3.4 Depth distribution of TON stocks under different fertility treatments on clayey and sandy soils. N Fert = nitrogen fertiliser, N Fert + manure = nitrogen fertiliser plus cattle manure.

In clayey soils, TON was significantly higher under N Fert + manure than control in both homefields and outfields except 10-20 cm depth in outfields where N Fert and N Fert + manure were similar. On sandy soils, significant differences between three treatments were only at 20-30 cm in homefields and 10-20 cm in outfields.

The C:N ratios were higher in clayey (1:17) than sandy soils (1:15) although within each soil type there were no significant differences. The C:N ratios of homefields under control, N Fert and N Fert + manure were 17, 17, 15 and 14, 18, 17 on clayey and sandy soils respectively whereas in outfields the C: N ratios were 15,16,18 and 11,16,15 for clayey and sandy soils respectively.

3.3.2 Carbon and nitrogen in density fractions

Similar to total SOC and TON stocks, density fractions also showed significant difference between soil types $p<0.001$), field position ($p= 0.007$), fertility treatments ($p<0.001$) soil depth ($p<0.01$), soil type x treatment ($p<0.001$) and soil type x field position x treatment (p<0.001). The C and N stocks in the three density fractions were generally in the order oLF< fLF< MaHF (Table 3.2).

The addition of nitrogen fertiliser and cattle manure had variable influence on the three fractions resulting in significant increases of fLF C and MaHF C compared to control treatments. On clayey soils, the proportions of SOC in fLF, oLF and MaHF were similar under control and N Fert + manure (3%, 2%, 95%) while the proportions under N Fert were 3%, 1% and 96% respectively. On sandy soils, the proportions of fLF, oLF and MaHF were different between control (4%, 8%, 88%), N Fert (4%,1%, 95%) and N Fert + manure (5%, 4%, 91%) respectively.

The fLF C and N were 0.78 Mg C ha^{-1} and 0.04 Mg N ha^{-1} on clayey soils whereas on sandy soils they were 0.41 Mg C ha^{-1} and 0.02 Mg N ha^{-1}. The oLF followed similar trends being 0.32 Mg C ha^{-1}; 0.02 Mg N ha^{-1} and 0.12 Mg C ha^{-1}; 0.01 Mg N ha^{-1} on clayey and sandy soils respectively (Figure 3.5).

The oLF N was statistically similar in the two soil types. Field position showed greater fLF C in homefields than outfields but with no significant effect on oLF N. N Fert and N Fert + manure treatments enhanced fLF C in homefields and outfields by 19%, 24% and 9%, 22% respectively, when compared with the control. In sandy soils, N Fert and N Fert + manure treatments increased C storage by 17%, 26% and 26%, 26%, when compared with control on homefields and outfields respectively. Thus, on sandy outfields, N Fert and N Fert + manure had similar impacts on fLF C storage. N Fert had more oLF C on clayey homefields. On sandy soils, N Fert and N Fert + manure had less oLF C than control in homefields whereas in outfields, N Fert enhanced oLF C stocks by 15% and N Fert + manure was 8% lower than control.

Table 3.2: Total SOC, TON (Mg ha^{-1}) and C:N ratios for density fractions as affected by fertiliser and manure applications on clayey and sandy soil under continuous maize cropping.

Soils	**Position**		**SOC**			**TON**			**C:N Ratio**		
		Treatment	**fLF**	**oLF**	**MaHF**	**fLF**	**oLF**	**MaHF**	**fLF**	**oLF**	**MaHF**
Clayey	Home Field	Control	0.55	0.14	19.8	0.02	0.007	1.19	28(1)	20(2)	16(2)
		Fertiliser	0.83	0.56	28.8	0.04	0.034	1.66	21(1)	16(2)	17(2)
		Manure	1.10	0.22	32.0	0.05	0.007	2.00	22(1)	19(2)	15(2)
	Outfield	Control	0.39	0.37	16.1	0.02	0.019	1.02	20(1)	19(2)	15(1)
		Fertiliser	0.76	0.15	24.6	0.04	0.008	1.46	19(1)	19(2)	16(2)
		Manure	0.92	0.22	32.1	0.05	0.011	1.73	18(1)	20(2)	18(2)
		Mean	0.78	0.32	25.6	0.04	0.014	1.51	22	19	17
Sandy	Home Field	Control	0.30	0.19	8.2	0.01	0.010	0.57	30(1)	19(2)	13(2)
		Fertiliser	0.55	0.11	14.0	0.02	0.006	0.80	28(1)	18(2)	17(2)
		Manure	0.69	0.18	12.9	0.04	0.009	0.78	17(1)	20(2)	16(2)
	Outfield	Control	0.14	0.08	3.6	0.01	0.004	0.27	14(1)	20(2)	14(2)
		Fertiliser	0.38	0.12	9.0	0.02	0.007	0.57	19(1)	17(2)	16(2)
		Manure	0.39	0.06	7.7	0.04	0.003	0.49	20(1)	20(2)	15(2)
		Mean	0.41	0.12	9.2	0.02	0.007	0.58	23	20	15
		MSE	0.17	0.03	4.3	<0.01	<0.01	0.13	16	11	10

Different letters show significant difference per soil type and field position at α = 0.05. fLF = free light fraction, oLF = occluded light fraction, MaHF = mineral associated heavy fraction. Fertiliser = N Fert 100 kg ha^{-1},Manure = 100 kg N ha^{-1} + 5 Mg ha^{-1} cattle manure ~0.9% N.

The MaHF dominated the SOM accounting for greater than 98% of the total soil mass and 95-97% and 84-95% of SOC on clayey and sandy soils respectively. The MaHF N accounted for 95-96% of TON in the two soil types. Mineral associated heavy fraction C concentration ranged from 12 to 27 g kg^{-1} on clayey soils and from 4 to 12 g kg^{-1} on sandy soils. The resulting C stocks in MaHF were also significantly different between clayey homefields and outfields (25.7 and 22.7 Mg ha^{-1}) and sandy homefields and outfields (11.3 and 6.6 Mg C ha^{-1}) (Figure 3.6). There were significant differences in MaHF C stocks were significantly different between treatments; control, N Fert and N Fert + manure having 17.3, 25.6, 30.8 Mg C ha^{-1} and 5.6, 11.2, 10.1 Mg C ha^{-1} on clayey and sandy soils respectively. On clayey soils, the N Fert + manure and N Fert application increased MaHF C by 21%, 16% and 6%, 16% in homefields and outfields respectively whereas on sandy soils increases under N Fert + manure and N Fert treatments were 16%, 18%, and 19%, 27%, in homefields and outfields respectively. MaHF N was 1.11, 1,56, 1.86 Mg N ha^{-1} and 0.42, 0.69, 0.64 Mg N ha^{-1} for control, N Fert and N Fert + manure treatments on clayey and sandy soils respectively.

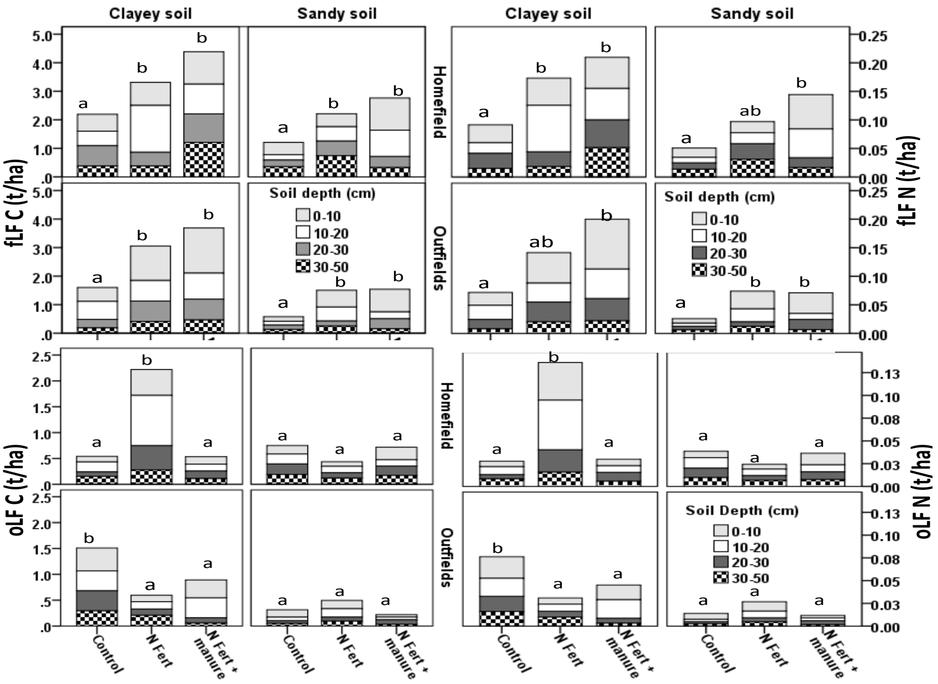

Figure 3.5: Soil C and N stocks in free light fraction (fLF) and occluded light fraction (oLF) under different fertility treatments at four depths layers on clayey and sandy soils separated into homefields and outfields. N Fert = nitrogen fertiliser, N Fert + manure = nitrogen fertiliser plus cattle manure. Different letters show significant differences in each soil type and field position at p = 0.05.

Vertical distribution of C and N in density fractions showed that control treatments on clayey soils had the lowest fLF C and N at all depth levels with the oLF N being significantly higher in control (p=0.013) than N Fert + manure at all depths except 20-30 cm depth on sandy soil.

The MaHF C and N decreased with increasing soil depth having higher magnitudes on clayey than sandy soils and was strongly related to bulk soil organic C (R^2=

0.996, p<0.01 and R^2= 0.998, p<0.01) for sandy and clayey soils respectively. MaHF N had similar relationships (Figure 3.6).

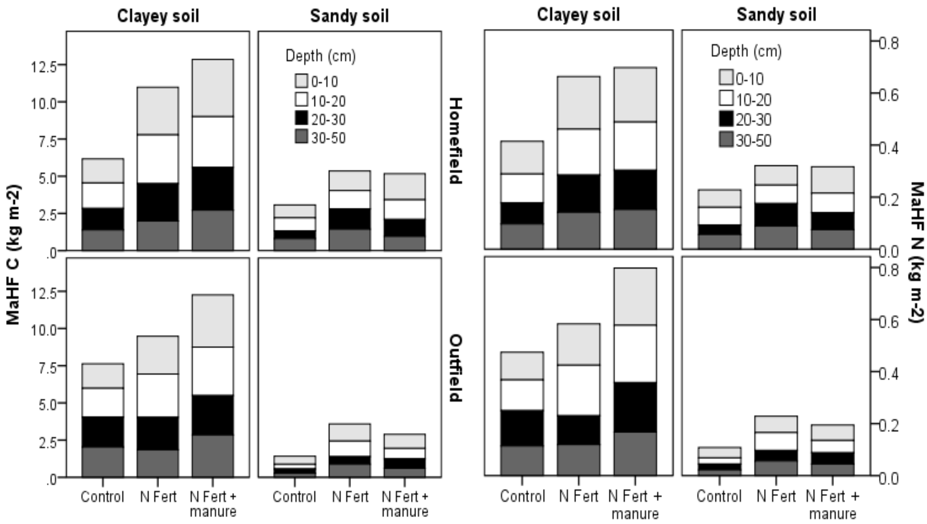

Figure 3.6: Depth distribution of mineral associated heavy fraction (MaHF) C and N in homefields and outfields of clayey and sandy soils as affected by fertility treatments. N Fert = nitrogen fertiliser, N Fert + manure = nitrogen fertiliser + cattle manure.

The C:N ratios for MaHF were smaller than fLF and oLF ranging from 13-18 compared to 16-20 for oLF and 14-30 for fLF (Table 3.2). The MaHF C:N ratios were significantly higher in clayey than sandy soils (*p*=0.007). The fLF (*p*=0.001) and oLF (*p*=0.024) C:N ratios were significant higher in control than N Fert + manure treatments.

3.3.3 Soil C sequestration potential

There was a linear relationship between light fraction C and the ratio of LF C/MaHF C with all fertility treatments on clayey soils being highly significant (p<0.01) i.e. control (R^2 = 0.863 and 0.903), N Fert (R^2= 0.725 and 0.780) and N Fert + manure (R^2 = 0.731 and 0.615) in homefields and outfields respectively) (Table 3.3).

Table 3.3: Regression equations for fertility treatments in homefields and outfields on clayey soils and associated theoretical maximum adsorption capacity of mineral associated heavy fraction (MaHF).

Field Position	Treatment	Regression equation	df	R^2	SE	Theoratical maximum adsorption capacity (Mg ha^{-1})
Homefield	Control	LFC/MaHFC = 0.0583LFC + 0.005	9	0.863*	0.005	17.15
	N Fert	LFC/MaHFC = 0.0227LFC + 0.01934	6	0.725*	0.002	44.05
	N Fert + manure	LFC/MaHFC = 0.0237LFC + 0.0115	11	0.731*	0.01	42.19
Outfield	Control	LFC/MaHFC = 0.0622LFC - 0.0046	10	0.903*	0.004	16.08
	N Fert	LFC/MaHFC = 0.0502LFC - 0.0038	10	0.780*	0.004	19.92
	N Fert + manure	LFC/MaHFC = 0.0221LFC + 0.0138	11	0.615*	0.01	45.25

LFC = light fraction carbon, N Fert = nitrogen fertiliser, N Fert + manure = nitrogen fertiliser +, N Fert = nitrogen fertiliser, N Fert + manure = nitrogen fertiliser + cattle manure. *Significant at p= 0.05, SE = Mean Standard Error.

On sandy soils, the three treatments had weak R^2 values ($R^2 < 0.30$) and were excluded from further analysis using regression. Thus, LFC data failed to fit an appropriate linear or non-linear regression model as was shown by the low R^2 values despite sandy soils having significant gains in C and N under N Fert and N Fert + manure treatment when compared with the control.

The regression equations were used to estimate the equilibrium level and results showed that the control treatments had lowest equilibrium values (Table 3.3; Figure 3.7). The slopes of the regression equations were steepest under control (5.8% and 6.2%) on homefields and outfields respectively. The equilibrium levels for MaHF C in homefields under control (17.15 Mg ha^{-1}), N Fert (44.05 Mg ha^{-1}) and N Fert + manure (42.19 Mg ha^{-1}) treatments were higher than current stocks in Figure 3.1. In outfields only the control treatment had current stocks lower than the equilibrium value of 16.08 Mg ha^{-1}.

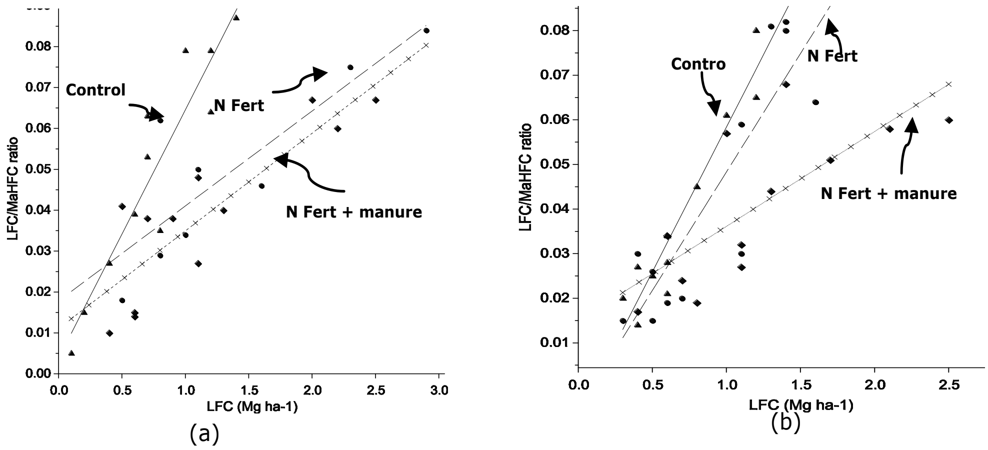

Figure 3.7: The relationship between light fraction C and mineral associated heavy fraction (MaHF) C in (a) homefields and (b) outfields under control, nitrogen fertiliser (N Fert) and nitrogen fertiliser plus manure (N Fert + manure) treatments on clayey soils. Unfertilised control =▲, nitrogen fertiliser (NF)= ●, nitrogen fertiliser plus cattle manure (N Fert + manure) = ♦

3.4 Discussion

3.4.1 Carbon and nitrogen in bulk soil

The higher C and N stocks in clayey soils (Figure 3.1) confirm the importance of texture in SOC dynamics. Clay soils have greater capacity to protect organic matter from decomposition as the finer texture may result in presence of occluded and adsorbed stable humus. Management activities however, could also be an important factor influencing the capacity of a soil to store SOC. In this study, C stocks in N Fert + manure were not significantly affected by distance from homestead on clayey homefields (Figure 3.2).

Homefields had higher organic C and N than outfields supporting the findings of Zingore *et al.* (2007) and shows that the fertility gradients have been maintained over the nine year period. Nitrogen followed similar trends except for N Fert treatment on homefields. Benefits of inorganic and organic fertiliser amendments are supported by Matsumoto *et al.* (2008) and Gregorich and Janzen

(1996). Nitrogen fertiliser is known to enhance growth of crop roots and shoots and promotes C inputs into the soil although it can also negatively affect local micro-environment thus controlling decomposition thereby causing no effect on C accumulation (Campbell *et al.*, 1997). Bremer *et al.* (1994) and Eghball and Ginting (2003) found the application of manure and compost to have significant C sequestration effects while chemical fertilisers had no effect.

Application of N fertiliser alone had less effect on SOC and TON storage than a combination of N fertiliser and cattle manure on clayey soil whereas on sandy soils a similar observation was made only at 0−10 cm depth (Figures 3.3 and 3.4). Thus, increase of SOC contents under N Fert and the N Fert + manure in both clayey and sandy soils relative to the control treatments could be attributed to 9 years of continuous addition of C via manure, root biomass and residues associated with higher crop yields (Liu *et al*., 2005; Banger *et al*., 2009; de Rouw *et al*.; Ding *et al.*, 2012) in both homefields and outfields. Return of crop residues to fields is threatened by competing uses as some farmers collect and store the crop residues for livestock feed over the dry season. If left in the field, existing policies allow free range livestock grazing over the dry period, threatening the persistence of residues in the fields. Therefore, if the crop residues are grazed *in-situ,* input of organic matter is most likely a result of varying amounts of root biomass, remnants of crop residues and the organic manure. These directly add C substrate and nutrients to the soil (since manure contains significant amounts of C as a major constituent). Therefore, variations in C and N stocks are likely a result of different C inputs from biomass leftovers (Purakayastha *et al.*, 2007). Rusinamhodzi *et al.,* (2011) reported increased crop yields under fertility treatments and decreased crop yields in the control treatments over the nine year period. The 0.9% difference in N content between N Fert and N Fert + manure treatments resulted in no significant gain in maize grain yields in homefields.

The lack of significant differences between N Fert and N Fert + Manure on sandy soil could be a result of enhanced decomposition under N Fert + manure and low C inputs under N Fert. The organic component of manure can easily be mineralised on sandy soils unlike on clayey soils where it can be complexed with clay mineral surfaces. On clayey soils, N Fert + manure treatment had more SOC and TON than N Fert alone thus agreeing with Wu *et al.* (2004) who worked on Chinese soils and found organic additions more effective in C accumulation than inorganic fertiliser alone. Stroosnijder and Hoogmoed (2004), while working in Burkina Faso, found an increase of 1.8 g C kg^{-1} soil after 10 years of annual application of 10 Mg ha^{-1} of manure and they concluded that a combination of either hand hoeing or conventional tillage with manure can increase SOC and crop yields.

Differences in C and N accumulation under each treatment could also be influenced by initial C and N stocks. The magnitude of differences among treatments was greater at 0-10 cm and 10−20 cm than lower depths (Figures 3.3 and 3.4). Similarly, Brar *et al.* (2013) reported a greater build-up of SOC and TON with fertilisation. In support of the results on clayey soils, their studies also found a combination of manure and inorganic fertiliser having significant cumulative benefits of both increased C stocks and crop yield (Berzsenyi *et al.*, 2000; Hao *et al.*, 2002) than inorganic fertiliser alone at 0−15 cm (Liu *et al*., 2005). In this study, beneficial effects of organic and inorganic amendments were possibly not fully maximised in surface layers on sandy soil as a result of a possibility of leaching of nutrients to lower layers resulting in significantly higher accumulation of C and N at 30-50 cm depths under the N Fert treatment (Figures 3.3 and 3.4). This is because sandy soils are low in clay and silt and have reduced capacity to protect SOC. Sandy soils are also prone to leaching losses of both C and N resulting in accumulation at lower depths facilitated by faster infiltration rates. In clayey soils, macro and micro aggregation protects SOC from degradation through binding with clay and silt size particles (Six *et al* 2002).

Addition of manure and other organic fertilisers improves nutrient efficiency and enhances biomass yields (Nyamangara *et al.*, 2003). Inorganic fertilisers may also cause accelerated decomposition of SOM on clayey soils when compared with organic inputs resulting in less C accumulation. Soil conditions in the study area are prejudiced by reduced cover and continuous surface disturbance during the dry period. In most cases, the unfenced smallholder farms are subjected to depletion of above ground crop residues consumed by free ranging livestock during the dry season. Some farmers, however, store the crop residues for personal livestock feed over the dry period or they use the residues as bedding material in livestock pens.

In most parts of southern Africa, smallholder farmers use conventional tillage with small patches of land dedicated to conservation tillage practices despite the poor soil fertility (Nyamapfene, 1991). In most cases, large amounts of chemical and/or organic fertiliser are required to improve crop productivity. Given these conditions, N Fert and N Fert + manure treatments had limited impact on C:N ratios within each soil types as shown by the small variations in C:N ratios.

3.4.2 Carbon and nitrogen in density fractions

The largest amounts of light fraction were found under N Fert + manure treatment reflecting the presence of inherent labile materials. The proportion of fLF and oLF in the soils was 1−2% of initial soil mass which is similar to the proportions obtained by Swanston *et al.* (2005) although it was less than that obtained from elsewhere. For example, Yin *et al.* (2005) worked on three soil types in China and found the light fraction accounting for 1−27% of bulk soil organic matter and Tan *et al.* (2007) found a range between 5% and 12% while working on tillage experiments in the USA.

The addition of manure plus nitrogen fertiliser and addition of nitrogen fertiliser alone showed greater partitioning of fLF C in homefields than outfields. Fertility treatments had less effect on oLF C and N (Table 3.2). This could be an indication of the importance of land use history with fLF reflecting changes caused by land management e.g. effect of fertility gradients which decrease with increasing distance from homestead (Zingore *et al*., 2007). In this case, the analysis of SOC fractions were used to show the influence of land use activities on organic matter quality because the fractions have different stabilities and turnover rates and are extracted from different positions in the soil matrix (Golchin *et al*., 1994b). The fLF is the most dynamic and sensitive to fraction known to be easily influenced by management practices (Janzen *et al*., 1992) and can be affected by quality and quantity of materials added to the soil (i.e. inorganic or organic inputs). The amount of fLF in each treatment might also be attributed to the quantities of crop root biomass which is proportional quantity of better yields (Banger *et al.*, 2010). Similar to this study, Bremer *et al.*(1994) found more light fraction under inorganic fertiliser plots than the other treatments, while Mirsky *et al*. (2008) found the fLF unaffected by fertility treatments. The oLF, which consists of processed organic material and partially degraded plant materials associated with mineral particles and aggregates (Golchin *et al*., 1994b) was however, less in manure than control. N Fert + manure had greater beneficial effects on SOC in fLF and to a greater extent MaHF whereas it did not enhance the oLF indicating that aggregation was not affected.

Across the four sampling depths, significant differences were observed mostly between N Fert + manure and control treatments showing the importance of the organic component of manure in contributing to labile fraction. The addition of manure and inorganic fertiliser was observed to affect distribution of fLF in surface layers decreasing with increasing soil depth, depending on tillage practice (Rudrappa et al., 2006; Brar *et al.*, 2013). The N Fert and N Fert + manure treatments significantly increased C and N in fLF in the top 20 cm as compared to the lower 20-30 and 30-50 cm depths. This could be attributed to the fact that organic amendments induce rapid and conspicuous alterations to the function and structure of soil microbial communities thus affecting soil organic matter fractions. The form and application rate of manure and associated C content have been reported to influence the comparative reaction of soil microorganisms (Paul & Beauchamp, 1996). In support of these results, Banger *et al.* (2010) found higher LFC in surface than the lower layers under inorganic fertiliser and organic fertiliser application. Smallholder farms assessed in this study had less or no residue input strategies leaving root biomass as the main source of LF organic matter. The remnant residue left after grazing by livestock and root biomass had different impacts fLF deposition on clayey and sandy soils with the sandy soils accumulating more fLF than clayey soils.

The fLF C may be used as an indicator of change in C accumulation following application of manure and inorganic fertiliser on both clayey and sandy soils. This more available labile C pool (i.e. fLF) increases microbial activity, subsequently increasing aggregation, and SOC stabilisation through occlusion mainly in clayey soils. In addition, increased microbial turnover also enhances production of dissolved organic carbon (DOC) which in turn increases stabilisation of SOC by adsorption. Furthermore, the quality of the incoming above and below ground plant residues and organic amendments determine the quality and degradability of organic matter by soil microorganisms (Gregorich & Ellert, 1993). Although the LF was considerably smaller than the heavy fraction (Figures 3.5 and 3.6), it plays an important role in soil biological processes as an energy source for the soil microorganisms (Gong *et al*., 2009).

The MaHF was the most abundant density fraction on both clayey and sandy soils and is considered a more stable C pool with a long turnover time (>100 years) (von Lützow *et al.*, 2008) and high amounts of protected C. The proportion of MaHF C to total SOC was 96% and 92% for clayey and sandy soils while N was 96−97%. The results suggest that on clayey soils, addition of N Fert + manure was possibly more effective in stabilising C and N in the MaHF (Figure 3.6), when compared to N Fert alone. The interaction between SOC and mineral surfaces can result in increased protection of SOC from microbial degradation, and clay particles can encapsulate particles or patches of the SOM. Furthermore, the presence of clay particles in soil provides greater surface area onto which organic material can be adsorbed (Baldock & Skjemstad, 2000).The results suggest that some of the manure added annually to the soil becomes stabilised in the soil to form the MaHF C and N. On the other hand, N Fert can stimulate microbial activity and enhance C turnover resulting in increased adsorption of decomposition products onto mineral surfaces mainly on clayey soil.

The increase in recalcitrant C after addition of a combination of N Fert + manure is normally attributed to higher lignin content of the manure when compared with nitrogen fertiliser alone and some of the lignin is assumed to enter directly into the slow pool (Parton *et al.*, 1987). However, a study by Hofmann *et al*. (2009) showed that lignin may not be an important factor in SOM decomposition. Therefore, the significant increase of C and N in MaHF and associated fractions after manure addition suggests the importance of manure in stabilisation of SOC and N agricultural soils. The potential may not have been fully exploited since the soils studied are subjected to conventional tillage known to cause increased mineralisation of labile LFC without significant effect on MaHF.

Larger C:N ratios of the light fraction (Table 3.2) can be attributed to decreased net N mineralisation and N immobilisation to meet metabolic demands for microbial decomposers (Swanston *et al.*, 2004). In another study Mujuru *et al.* (2013) analysed C:N ratios in density fractions from similar soils under different tillage systems and found higher C:N ratios in fLF (17−32) than those in this study.

3.4.3 Soil C sequestration potential

Sandy soils showed no statistically significant linear dependence of the LFC/MaHFC ratio and LFC although there were positive linear relationships between SOC or TON and associated MaHF. Yin *et al*. (2005) also found that Chinese fluvo-aquic and reddish paddy soils had no linear relationship between LF and the LF/MaHF ratio. The relationship of LFC and LFC/MaHF C ratio on clayey soils (Table 3.3) showed that application of both N Fert + manure and N Fert alone on homefields enabled greater LFC and MaHFC over time and can be maximised when the management practices remain unaltered.

Potential for more C storage can be realised when C stocks are below equilibrium. In outfields, only the control had reached equilibrium whereas the rest had potential to store more C. The differences in slopes caused corresponding effects on the equilibrium levels in each field position and treatment. The slope was greatest under control showing a greater amplitude in the loss of LF over the nine year period and the intercept was also highest under control showing that the amount of C mineralisation may be greater than the other treatments and field positions. The lowest intercept was in inorganic fertiliser of outfields. Similar regression was used to determine equilibrium levels of organic matter in heavy fraction (Yin *et al*., 2005; Yin & Cai, 2006) under organic and inorganic fertiliser application in Chinese soils. Their results showed that C stocks under organic fertilisers were below equilibrium levels while inorganic fertilisers were above equilibrium levels.

Results showed that in homefields, the equilibrium values under N Fert + manure and N Fert were similar but 2.5 times more than in the control. This could be attributed to better management in homefields than outfields and control treatments. In outfields however, the equilibrium value of control treatments was lowest whilst in current stocks in homefields under control treatments were 3 Mg ha^{-1} lower than equilibrium levels. Outfields have inherent infertility (Zingore *et al*., 2007) and the nitrogen fertiliser inputs could be utilised by crops leaving the soil with insufficient organic inputs. Yin and Cai (2006) found matching equilibrium levels under organic and inorganic fertilisers while working on soils from a long term experiment in paddy soils of China whereas in fluvo-aquic soils, organic fertilisers had twice as much heavy fraction as in inorganic fertilisers.

The long term balance between addition and loss of organic matter under control could result in a balance between the crop uptake and inputs from residues and crop roots. In this case, land use history and management become important in attainment of equilibrium levels of a soil. The adsorption of organic matter onto mineral surfaces mostly depends on the amount of organic matter input and its quality (Rasmussen *et al.*, 1980). Control treatments and outfields reflect a long term balance between C inputs and outputs and show that the soils have approached a new lower steady state. When MaHF reaches equilibrium levels, it can no longer increase whereas the LF can increase with more inputs striving to attain a new equilibrium level (Yin & Cai, 2006). The mineral associated heavy fraction C can increase over time in soils that are below equilibrium level provided the same land use management practice is maintained or improved. Results suggest a potential C and N sink on clayey soils as indicated by the differences between actual and estimated equilibrium levels.

Although cattle manure has greater potential to enhance soil fertility and soil C storage, smallholder farmers do not have large herd of livestock to supply adequate manure to their fields. Only a few can afford to apply manure annually to some of their fields while the majority use inorganic fertilisers (which are also beyond the reach of many) (Rusinamhodzi *et al*., 2013). This suggest a need for promotion of cheaper alternatives and assistance for inorganic fertiliser inputs for small holder farmers to enhance soil C sequestration. Alternatives such as use of compost, legume crops, reduced tillage and improved fallows should be some of the extension options for promoting soil C sequestration. Currently, residue retention is a challenge for small holder farmers to maintain residues *in situ* because of unfavourable policies which allow free range grazing during the dry season (6-8 months). There is therefore need for strategic management of crop residues so that farmers can make decisions of whether to keep their residues or to use them for other alternative uses. The use of abundant grass biomass to improve quantity of SOM in agro ecosystems can be more beneficial and reduces emissions from wildfires.

3.5 Conclusions

The results have shown that C and N stocks can be enhanced in home fields and outfields on clayey and sandy soils using N Fert and a combination of cattle manure + nitrogen fertiliser respectively. The higher amounts of C and N in N Fert and N Fert + manure treatments than control shows the importance of application of soil fertility amendments for the maintenance or improvement of C and N stocks in agricultural systems. Carbon and nitrogen in 0-10 cm depth increased considerably under N Fert and N Fert + manure treatments compared to the control and supported the hypothesis that SOC and TON decreased with increasing soil depth. Despite being under conventional tillage, fertility amendments mitigated the decrease of total SOC and TON in both homefields and outfields but at different magnitudes.

Nitrogen fertiliser alone or in combination with cattle manure consequently increased the accumulation of C and N in fLF, oLF, MaHF after nine years of continuous cropping except that manure treatment had limited capacity to increase oLF C and N. The distribution of C and N in the three fractions was greater in homefields than outfields with greater stabilisation of SOC under N Fert + manure treatment on clayey soils whereas N Fert had greatest impacts on sandy soils.

On clayey soil, estimation of the equilibrium level enabled us to assess the relationship between SOC fractions and determine the potential of fertilisation to sequester additional C in clayey soils. Assessment of the maximum SOC protective capacity suggested a potential for additional SOC storage in the clayey homefields and outfields under N Fert and N Fert + manure treatments. In outfields, the combination of manure and nitrogen fertiliser showed highest potential to sequester more C in MaHF. Sustaining SOC levels in these soils can be challenged

Chapter 3

by the low crop yields which give rise to low biomass residues coupled with a long annual dry and fallow season.

Soil organic carbon and nitrogen storage in a Pinus patula chronosequence

CHAPTER 4

Soil carbon and nitrogen sequestration over an age sequence of *Pinus patula* plantations in Zimbabwean Eastern Highlands

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Chapter 4

Soil carbon and nitrogen sequestration over an age sequence of *Pinus patula* plantations in Zimbabwean eastern highlands

ABSTRACT

Forests play a major role in regulating the rate of increase of global atmospheric carbon dioxide (CO_2) concentrations creating a need to investigate the ability of exotic plantations to sequester atmospheric CO_2. This study examined pine plantations located in the eastern highlands of Zimbabwe relative to carbon (C) and nitrogen (N) storage along an age series. Samples of stand characteristics, forest floor (L, F and H layers) and 0 −10, 10−30 and 30−60 cm soil depths were randomly taken from replicated stands in *Pinus patula* Schiede & Deppe stand ages of 1, 10, 20, 25, and 30 years plus two natural forests. Sodium polytungstate (density 1.6 g cm^{-3}) was used to isolate organic matter into free light fraction (fLF), occluded light fraction (oLF) and mineral associated heavy fraction (MaHF). In both natural and planted forests, above ground tree biomass was the major ecosystem C pool followed by forest floor's humus (H) layer. In addition, the mineral soil pool had 45%, 31% and 24% of SOC stored at the 0–10, 10–30 and 30–60 cm soil depths respectively. Stand age caused significant differences in total organic C and N stocks. Carbon and N declined initially soon after establishment but recovered rapidly at 10 years, after which it declined following silvicultural operations (thinning and pruning) and recovered again by 25 years. Soil C and N stocks were highest in moist forest (18.3 kg of C m^{-2} and 0.66 kg of N m^{-2}) and lowest in the miombo (8.5 kg of C m^{-2} and 0.22 kg of N m^{-2}). Average soil C among the *Pinus* stands was 11.4 kg m^{-2} of C m^{-2}, being highest at 10 years (13.7 kg of C m^{-2}) and lowest at 1 year (9.9 kg of C m^{-2}). Some inputs of charcoal through bioturbation over the 25 year period contributed to stabilisation of soil organic carbon (SOC) and its depth distribution compared to the one year old stands. Nitrogen was highest at 10 years (0.85 kg of N m^{-2}) and least at 30 years (0.22 kg of N m^{-2}). Carbon and N in density fractions showed the 20 year old stand having similar proportions of fLF and oLF while the rest had significantly higher fLF than oLF. The contribution of fLF C, oLF C and MaHF C to SOC was 8–13%, 1–7% and 90–91% respectively. Carbon and N in all fractions decreased with depth. The mineral associated C was significantly affected by stand age whilst the fLF and oLF were not. Conversion of depleted miombo woodlands to pine plantations yield better C gains in the short and long run whilst moist forests (MF) provide both carbon and biodiversity. Our results highlight the importance of considering forestry age based C pools in estimating C sink potential over a rotation and the possibility of considering conservation of existing natural forests as part of future REDD + projects.

Key words: carbon, Soil organic matter fractions, Forest floor, C:N ratios, *Pinus* spp., Plantation forestry.

4.1 Introduction

Changes in soil organic matter (SOM) can result in significant contributions to emissions or uptake of greenhouse gases from forests and other land use systems. Forests govern C transfers directly through photosynthesis and respiration and indirectly by influencing the structure and size of plant-leaf development (Eliasson, 2007; Van Minnen, 2008). They represent an important C pool (Brown, 2002) that favour sequestration of C due to their increased woody biomass, extensive roots, and abundant litter (Sharrow & Ismail, 2004). The extensive rooting system of forest species influence soil microbial biomass thus control the cycle of C between the atmosphere and the soil (Brown, 2002). In general, tropical forests contain less C in soils than their biomass C, storing about 60% C aboveground and 40% belowground (Dixon *et al.*, 1994). However, especially in these forests, roots go deeper and thus, root turnover may add to C sequestration in deeper horizons due to slow C turnover (Jobbagy & Jackson, 2000).

The accumulation of soil C and N varies within different soil horizons and depths. Some forest sub-soils hold about 45% of total SOC bound to the clay particles to form microaggregates. This complexion of SOC in the forest sub soils is essential for long-term stabilisation (von Lützow *et al*., 2006). The extent of this stabilisation is determined by organo-mineral interactions, micropores, type and nature of clay surfaces, and C location within the microaggregates.

Next to climate and soil type, the sequestration of C depends on forest species and management (Lal, 2003; Lamlom & Savidge, 2003) having a compromise between ecosystem C storage capacity and timber extraction. Long-term differences in SOC storage among three tree species have been studied by Seely *et al.* (2002) who concluded that all tree species are important C pools although they have different C storage capacities. Vesterdal *et al*. (2002) compared soils under Norway spruce (*Picea abies* L) and oak (*Quercus robur* L.) and showed them to sequester 0.9 kg m^{-2} of C and 0.2 kg m^{-2} of C respectively after 29 years and the SOC being mostly concentrated in the upper soil horizons. In Hawaii, Kaye *et al*. (2000) reported storage of C by 17-year old *Eucalyptus* and *Albizia lebbeck* trees and reported that *Albizia* had 2 kg m^{-2} more soil C and 0.23 kg m^{-2} more soil N. The greatest potential for above ground biomass C storage in coniferous plantations (e.g. pines) is found in tree biomass (Peichl & Arain, 2006) with additional amounts from forest floor and mineral soil C (Taylor *et al.*, 2007; Noh *et al.*, 2010). Net rate of C uptake is greatest when forests are young, and slows with time. Old forests continue to sequester C at a decreased rate with decreased rate of respiration (Marris, 2008). When forests are cut, C is returned quickly to the atmosphere if the woody tissue is burned or converted to products that are short-lived (Ecological Society of America, 2000). Depending on harvesting practices, most of soil C remains in the soil and become part of the C stock of growing forest or a subsequent cycle in a plantation system. In addition to type of tree species, stand age is also critical in determining the amount of C in an ecosystem influencing the quality and quantity of C inputs released into an ecosystem (Matos *et al.*, 2010; Penne *et al.*, 2010). Some studies have shown that conversion of native forests to conifers can cause up to 15% losses of SOC depending on period following conversion while others estimated up to 20% SOC reductions over periods below 40 years (Guo & Gifford, 2002).

Soil organic carbon and nitrogen storage in a Pinus patula chronosequence

The general impacts of plantation forests have been outlined by region (Nilsson & Schopfhauser, 1995) and the IPCC (2003) suggested that the real C stock estimates might be much lower than indicated as some of the C has not been accounted for. Some studies have indicated relative increases in surface soil C stocks in plantations (Schwertmann *et al.*, 1986) while other studies found limited capacity for soil C accumulation (Richter *et al.*, 1999; Liao *et al.*, 2012). In this study we will not only look at quantities of C but also at its stability. Next to quantity, type and degree of stabilization is also important for the assessment of C sequestration.

Most studies on soil C and N dynamics over stand age are mainly from other regions either from experimental stations reflecting site specific conditions e.g. (Covington, 1981; Rita *et al.*, 2011) or model estimations on a local or regional scale e.g. (Peltoniemi *et al.*, 2004). Forest systems of Zimbabwe include rainforest, indigenous woodlands, plantations and bushland/grasslands covering 0.1, 65.9, 0.4% and 1.5% of the land area respectively. Plantation forests consist of *Pinus* spp.*(*68%), *Eucalyptus (20%) Acacia mearnsii* (11%) and *Poplar* spp.(1%) (Forestry Commission Zimbabwe 1996). The relatively extensive woodland cover makes Zimbabwean forests a potential C sink, but the sink is threatened by agricultural expansion and demand for wood.

The characterisation of C in the above ground biomass of forests is well advanced, but the below ground C dynamics is poorly understood causing a need for correlating the below ground biomass to the above ground biomass to predict C storage in forest soils (Brown, 2002). Determination of the flux of global C cycle needs substantial research which can link patterns and long term effects of C and N accumulation in the soil relative to forest age. The role of forests in the global C cycle has therefore initiated great interest in exploring the capacity of forest ecosystems to increase C uptake by means of afforestation and sustainable forest management through initiatives such as reduced emissions form forest degradation and deforestation (REDD+). There are few studies quantifying the potential for soil C accumulation and stabilisation under natural and exotic plantations in Zimbabwe and this creates a need for studies on the soil C sink potential of forest plantations. Reliable knowledge of the C and N dynamics in forest soils is therefore fundamental to understanding sustainable forest management practices and their role in climate change mitigation

In this study we measured forest floor and soil C and N pools induced by plantation forestry at different stages during a rotation cycle. Our aim was to describe the distribution, accumulation and stability of forest floor and soil C and N pools and their temporal shifts over time. We hypothesise that 1) forest floor and soil C and N pools under plantation forestry are lower than natural forest, 2) more C and N is stored in the forest floor and soil pools with increasing stand age within a cycle and 3) soil C stabilisation increases with stand age.

4.2 Study site and methodology

The study was carried out within the Nyanga Pine Division of Wattle Company P/L in Eastern Zimbabwe at Mutarazi Estate situated at S19 01.032 E32 35.810, lying at the extreme South of the Nyanga District (Figure 4.1). The altitude ranges between 1 020 m and 1 920 m above sea level. The terrain is characterised by relatively moderate slopes, and forms part of the Eastern escarpment of the Nyanga mountain range. It is drained mainly by the perennial Mutarazi river. The total plantation area is 3,806, 990 ha of which 2,548 ha was re-planted as at September 2010 (WATCO., 2010).

Mutarazi Estate falls into Natural Region (NR) I of the Zimbabwe agro ecological classification system (WATCO., 2010) with annual rainfall estimated at around 1500 mm and the seasonality follows the same general pattern as the rest of the country (bulk of rainfall being confined to the months of November to March). Small amounts of winter precipitation in the form of mist, fog and rainfall do occur on areas of high elevation. Average maximum temperature is 28˚C with a minimum of 0˚C. Lowest temperatures occur between May and August while the highest are from October to February. Relative humidity varies between mean 58% in September to a mean of 86% in January/ February. The prevailing wind is easterly blowing dominantly during the months of November to May.

The soils are orthoferrallitic within the Kaolinitic order (Zimbabwean classification) which corresponds to Rhodic ferralsols in FAO classification (FAO, 2006). The soils are characterised by good depth, permeability and structural stability exhibiting a high degree of resistance to erosion. They have extremely poor chemical characteristics, with particularly high levels of acidity and low weathering rates (WATCO., 2010).

A wide variety of broadleaf, large tree species occur in natural forests including *Macaranga mellifera*, *Ilex mitis*, *Schrebera alata*, *Rapanea melanophloeos*, *Olea capensis* and *Schefflera umbellifera*. The understorey of these forests is usually dominated by extensive banks of ferns comprising mainly of *Asplenium* and *Cyathea* spp. Widespread stands of *Psychotria zombamontana* also occur. Forest fringes are dense dominated by species including *Hypericum*

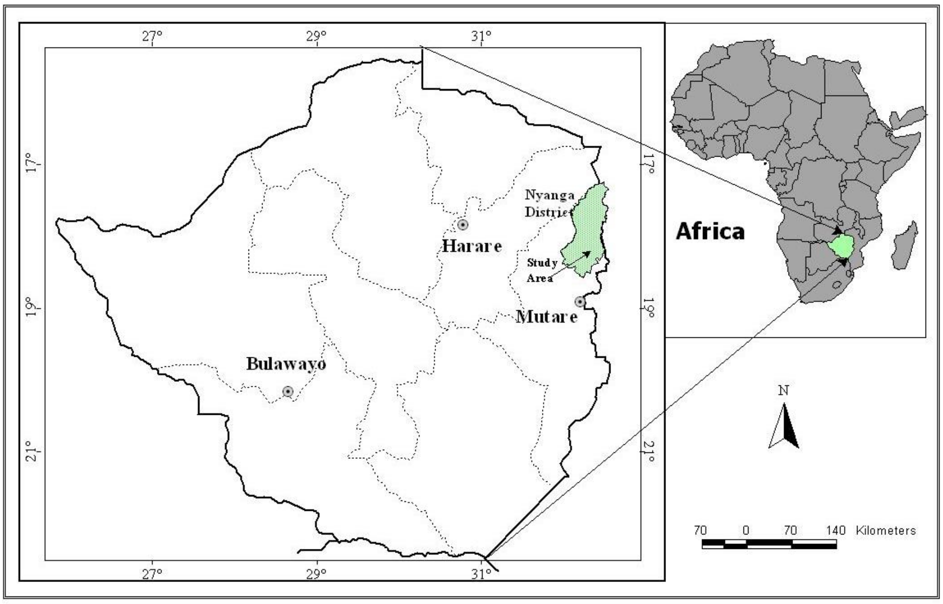


Figure 4.1: Map of Zimbabwe showing the location of Nyanga pine's Mutarazi forest.

revolutum, P*teridium, Rubus* and *Smilax anceps.* These forests are usually protected from invasions by commercial pine species and from fires by wide fire guards (20−50 m) and inside fire traces. Individual compartment records showed that natural forests were cleared for establishment of pine plantations in all stands considered for the study (WATCO., 2010).

Mutarazi estate is made up of 14 blocks with a total of 206 compartments of pine trees at different stand ages in rotation. Total land under *Pinus* species is 2 548 ha covering about 70% of total land area. The average age of the whole estate as at September 2010 was 10.9 years. The *Pinus* species planted in the estate include *P. elliottii* (Slash Pine), *P. taeda* (Loblolly Pine), *P. patula* (Patula Pine), and *P. tecunumanii* (Tecun Uman Pine). *P. patula,* Schiede & Deppe is the most dominant comprising more than 95% of the planted area. The compartments are established with 1 100 stems per ha (s.p.h.a), thinning twice at 4 years to 650 s.p.h.a and at 12 years to 400 s.p.h.a before clear felling at 25 years. Cleaning or weeding is carried out twice in the first year, and one slash weeding in the second year (WATCO., 2010). Before the establishment of pines, native forests were cleared and burnt before marking and pitting for initial planting.

4.2.1 Experimental design and data collection

Plots were selected to represent pine age classes of 1, 10, 20, 25, and 30 years. Also, a moist broad leaf forest and a miombo woodland were included in the study to represent soils prior to clearing for pine plantation. The miombo woodland is frequently accessed by neighbouring communities whereas the moist forest is less accessible. Among the pine stands, two stands were selected from different management blocks and in each stand, three sampling plots (0.04 ha each) were randomly selected. All pine stands were in their first rotation except the 1 year old which is now entering a second rotation. At each sampling plot, geo-location and altitude were recorded using a Garmin GPS device. Slope was recorded using a clinometer. Aspect, undergrowth species and ground cover were observed and noted. At each sampling point tree measurements, forest floor and soil samples were collected as described below.

At each plot centre, forest floor was sampled from inside a metal ring of 30 cm diameter. The forest floor was stratified into three layers: (1) litter layer (L) − consisting of fresh and recently fallen, non-decomposed material. The material is identifiable by the naked eye as plant residues. This layer usually contains less than 10% fine organic matter, (2) fragmented layer (F) − organic material is fragmented and partly decomposed with plant residues being macroscopically recognizable and (3) the humus layer (H) − consists of decomposed organic matter, originating from litter fall from decades ago and root turnover. The materials are decomposed and their origins are no longer distinct from each other with more than 70% being fine organic matter (Currie, 1999; Schulp *et al.*, 2008; Keith *et al*., 2010).

Next, pits were dug to a depth of 60 cm. At each depth increment, i.e. 0 – 10, 10 – 30 and 30 – 60 cm, a composited 300 – 400 g bulk sample was taken by sampling the four sides of the pit. In addition, bulk density (BD) samples were taken using a 100 cm^3 metal ring sampler at each depth increment centre. All soil samples were put into labelled air tight plastic bags and stored in a cool dry place. A total of 108 bulk soil samples and 108 bulk density samples were collected (5 pine stand ages × 2 stands each × 3 pits × 3 depths plus 2 natural forests x 3 pits each x 3 depths). At each sampling point, a 1 m^2 × 1 m^2 area was cleared to trap litter to assess annual litter fall in each age class.

Diameter at breast height (DBH, at 1.3 m) and tree height were recorded for every tree within a radius of 11.28 m. Tree height was measured for every third tree using a Suunto hypsometer. Stand stem volume (V) for pine stands was calculated from stand basal area ($BA=\pi\ D^2/4$) and mean tree height (H) using the standard biometric equation:

$$V = BA \times H \times f \qquad [1]$$

The equation includes a standard stem form factor (f) of 0.4 (Cannell, 1984). Basic wood density was obtained from Muneri and Balodis (1998) and a biomass expansion factor of 1.3 (FAO, 1997) were used to convert stem wood volume to biomass.

For the natural forests, generalised allometric equations intended for all species types in broad forest types and ecological zones were used to determine the forest C stocks using equations 2 (Brown *et al.*, 1989) and equation 3 (Malimbwi *et al.*, 1994) and an average of the two was used.

$$34.47 - 8.067DBH + 0.659DBH^2 \qquad [2]$$

$$\text{Exp } 2.516 \ln(DBH) - 2.642 \qquad [3]$$

The equations use DBH to explain variation in aboveground forest C stocks. Based on this we assumed that we will be able to generate reliable estimates of C stocks based on trees with DBH ≥5 cm in the natural forests without using species-specific allometric equations. The mean of two equations was taken as the biomass of each forest. Carbon stocks were calculated using a factor of 0.5 to obtain C stocks (as 50% of biomass is C).

4.2.2 Laboratory analysis

The field moisture content was determined gravimetrically by drying each bulk density sample in an oven at 105˚C for 48 hours. Samples were weighed before and after drying and percentage field moisture and the BD grams (g) of dry soil/100 cm^3 (volume of ring) were calculated. Soil pH was measured with a pH meter (Orion 701A) in a 1 M KCl solution suspension for each stand age and soil depth.

The bulk soil samples (BS) were passed through a 2.00 mm Retsch sieve after which > 2.00 mm particles were discarded. These particles included roots, large charcoal and rock material in most cases. A sample of 10 g of the dry soil sieved to 2.00 mm (BS) was put into a moisture free hard graphite container with a metal ball inside. The container was put on a Retsch mill and span at 85 rpm for 5 minutes. The resultant ground soil was sealed in a glass container. A 15−20 mg subsample of the ground soil was weighed into a tarred 5 x 5 mm Aluminium foil, sealed and analysed for total C and N by dry combustion in a EA1108 CHN Elemental analyser (Fisons Instruments). The total C in forest floor and SOM were used to obtain total organic carbon (TOC, kg m^{-2} of C) for each age class thus quantifying the relative contribution of each forest or plantation age. Forest floor C stocks were calculated by multiplying C concentration with sample mass and dividing this by the area of the sample. Soil organic C stocks were calculated by multiplying C concentration with bulk density and thickness of the soil layer with a correction for stone content following equations 4 and 5:

$$\text{Carbon stock} = d \times BD \times SOC \times CF_{st} \qquad [4]$$

where : Carbon stock (kg/m^2), d = depth of horizon (m), BD = bulk density (kg/m^3) of the soil layer. SOC = SOC concentration expressed as weight based percentage and CF_{st} = correction factor for stone and gravel content:

CF_{st} = 1 – (%stone +%gravel) /100 [5]

4.2.3 Soil organic matter fractions and characterisation

Soil Organic Matter fractions were obtained from bulk soil following the method by Golchin *et al*, (1994b) but as described by Roscoe *et al.* (2000) with three states of physical protection for soil organic C: free light (fLF) (non-protected and extractable without sonication), occluded (oLF) (extractable by sonication) and protected (MaHF) (remained in the residue after sonication). Sodium Polytungstate (SPT) solution with a density of 1.6 g cm^{-3} was used to separate the fractions at 4 500 rpm in a Mistral 6000 centrifuge. Ultrasonic energy at 90%, 30 W output for 5 minutes from a Vibracell (Sonic Materials) was applied to the sample after removing the first fraction, to separate the occluded fraction from the mineralised fraction. The free and occluded fractions were extracted by vacuum filtration, filtering through a Whatman 0.5µm glass fibre filter, using a vacuum filtering unit, decanted into a tarred beaker, washed with distilled water to remove excess SPT and dried at 40˚C. The mineralised fraction was obtained by totalling the dry weights of the first two fractions and subtracting from 10.0 g which was the original weight of the soil sample. The obtained fractions were prepared and analysed for total C and N by dry combustion.

4.2.4 Statistical and Data Analysis

Data was analysed after testing for normality (Kolmogorov−Smirnov test) and homogeneity of variance (Levene’s test). One way analyses of variance (ANOVA) in SPSS v.18 (SPSS Inc., Chicago, Illinois, USA) was used to assess the effects of age and depth on the forest floor (C and N) soil pH, bulk density, whole soil C and N contents, soil organic C and N storage in density fractions and the associated C:N ratios. A separate analysis was done to assess differences between older pine age classes (25 and 30 years) and natural forests. Tukey's HSD tests were used to test significant effects at $p \leq 0.05$.

4.3 Results

4.3.1 Forest floor C and N distribution

The L, F and H layers of the forest floor were distinct in all natural and plantation forests except the 1 year old plantation stands which had no L layer but had abundant ground cover dominated by pioneer species. Mean thickness of the three forest floor layers (L, F, H) for 1, 10, 20, 25 and 30 years were 0.0, 3.0, 3.5 cm; 1.3, 2.5, 3.7cm; 1.3, 1.9, 2.9 cm; 2.2,3.0,1.8 cm and 1.7, 5.2, 3, 2 cm respectively. In the MW each layer was 0.5 cm whereas in the MF thickness of L, F and H layers was 0.5, 1.0 and 1.0 cm respectively (Figure 4.2).

Among the pine stands, total forest floor C was lowest in the 1 year old and highest in the 30 year old. All pine stands had significantly higher total forest floor C and N than natural forests except the 1 year old which had total N stocks statistically similar to MF (Figure 4.3). The amount of C and N in the forest floor was highest at 30 years (5.4 kg of C m^{-2} and 0.22 kg of N m^{-2}). Miombo woodlands had significantly low ($p < 0.01$) C and N stocks in all three forest floor layers with 1.1 kg m^{-2} of C and 0.05 kg m^{-2} of N while the moist forest had 2.2 kg of C m^{-2} and 0.12 kg N m^{-2}. Despite the absence of the L layer in the one year old stand, there were no significant differences in cumulative forest floor C and N with the 10 and 20 year old stands.

Among the *Pinus* stands the C in the L layer of forest floor increased under respectively the 10, 20 and 25 year old stands and at 30 years it was lower than at 25 years. Nitrogen followed similar trends with a decrease after 25 years. The increase between 10 and 25 years old stands was supported by increasing mean annual litter fall of 0.304, 0.741 and 0.932 kg m^{-2} dry mass for 10, 20 and 25 years respectively (Table 4.1). The mean annual litter fall of the 30 year stands was slightly higher with 0.989 kg m^{-2}. The annual C additions to the L layer were therefore , 0.15, 0.37, 0.47 and 0.49 g C m^{-2} yr^{-1} for the 10, 20 25 and 30 year old stands with significant differences ($p < 0.001$) between successive years before age 25.

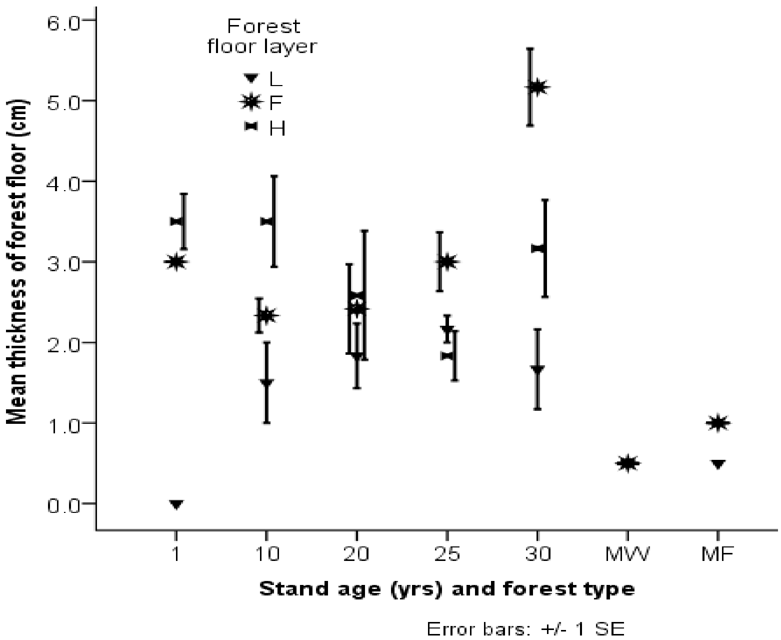
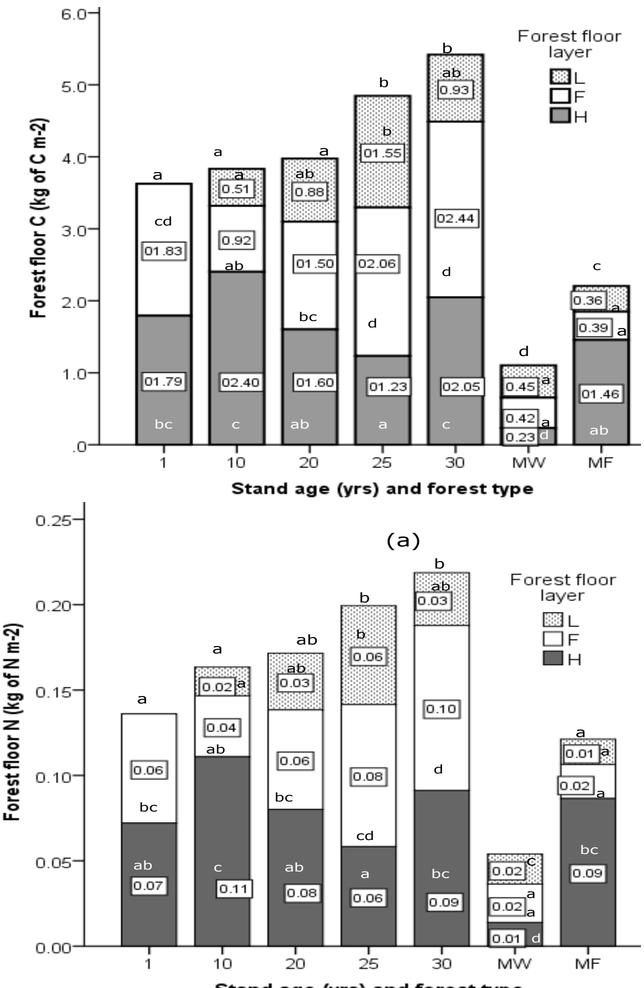


Figure 4.2: Depth of forest floor litter (cm) as a function of stand age and forest type.

Carbon and N content of the F layer decreased significantly (p=0.024) from 1 year to 10 years and the increased significantly in subsequent years from 10 to 20 years (p= 0.008) and from 20 to 30 years (p = 0.022) of stand age. The L and F layer C and N were significantly higher (p < 0.01) in pine stand ages of 20, 25 and 30 years than natural forest. The 25 and 30 year old stands had significantly higher (p<0.01) C in the F layer than the natural forests.

The H layer, C and N contents were significantly higher (p = 0.010) under the 10 year old stands (2.4 kg m^{-2} of C and 0.11 kg m^{-2} of N) than the 1, 20 and 25 year old stands pine stands. The C in H layer of the 10 year old and MF was significantly higher (p<0.01) than in the F layer. The C:N ratios of the plantation stands and the MF showed similar trends with C:N ratio decreasing from L to F to H whereas, in MW the H layer C:N ratio was higher than the F layer (Table 4.2).



(b)

Figure 4.3: The partitioning of a) organic carbon and b) organic nitrogen in forest floor of plantations (1, 10, 25, 30 years), miombo woodlands (MW) and moist forests (MF). Different letters show significant differences in each stand and forest floor layers at p ≤ 0.05.

Table 4.1: Mean stand characteristics of *Pinus patula* stands of different ages and natural forests.

Stand Characteristics	1 year	10 years	20 years	25 years	30 years	Miombo woodland	Moist forest
Elevation (m)	1864	1895	1808	1875	1897	1512	1871
Site Index	20	26	26	24	26	na	na
Stocking (SPH)	1100 (0)	650 (0)	395 (2)	397(2)	398(1)	308 (15)	712 (21)
Mean dbh (cm)	nd	23.1(0.2)	32.1(0.6)	36.7(0.5)	38.1(0.1)	9.2(2.5)	29.6(5.1)
Mean Ht (m)	0.74(0.1)	18.3(0.2)	22.6(1.0)	29.4(0.5)	29.9(0.9)	6.3 (2.2)	14.9(4.2)
BA (m^2 ha^{-1})	0.45(0.30)	27.2(0.5)	32.4(1.2)	42.1(1.1)	45.4(0.4)	8.2(3.3)	70.93(5.1)
Volume (m^3 ha^{-1})	0.13(0.07)	199.3(4.9)	292.8(18.3)	494.1(19.2)	543.3(19.1)	nd	nd
litter fall (kg $m^{-2}yr^{-1}$)	nd	0.304	0.741	0.932	0.989	nd	nd
Biomass C ($Mgha^{-1}$)	0.02(0.01)	23.0(0.6)^{ab}	33.8(2.1)^{b}	57.1(2.2)^{c}	62.8(2.2)^{c}	10.7(3.0)^{a}	103.1(11.6)

nd = not determined, SPH =Stems per hectare, BA = basal area, Ht = height, dbh = diameter at breast height (1.3 m above ground)

Table 4.2: C:N ratios of three forest floor litter layers in pine stand ages and natural forests (MW and MF) (stand mean ± S.D).

Stand / Litter layer	1	10	20	25	30	MW	MF
L	-	26(2)^{ab}	29(4)^{bc}	26(1)^{ab}	31(2)^{c}	23(2)^{a}	26(1)
F	31(3)^{a}	23(1)^{bc}	25(2)^{c}	26(2)^{c}	24(2)^{c}	21(4)^{b}	20(1)
H	26(4)^{a}	22(3)^{b}	20(3)^{b}	21(1)^{b}	23(3)^{ab}	23(1)^{ab}	16(1)
Overall mean	28(4)^{a}	26(4)^{ab}	24(5)^{b}	24(3)^{b}	26(4)^{ab}	20(4)^{c}	20(3)

Means followed by different superscripts in a row represent significant difference at $p< 0.05$. Tukey's HSD.

4.3.2 Forest stand and soil characteristics

Generally moist forests had significantly higher moisture content than the rest followed by the 1 year old pine stand (Table 4.1). Bulk density in MF was significantly higher than the 25 year old stand but significantly lower than all stands except the 10 year old. Among the pines, the 25 year old stand had significantly lower bulk density than all. Depth had no significant effect on soil pH (Table 4.3) and therefore only mean pH is recorded (Table 4.1) and it ranged from

4.2 to 5.1 with the MF having significantly higher (p < 0.01) pH than all stands. The 25 year old stands had significantly lower pH than MW, 1, 10, 20 and 30 year old stands and MF.

Table 4. 3: Results of the ANOVA on effects of forest stand age and soil depth on soil pH, soil organic carbon and nitrogen in bulk soil and density fraction.

Characteristic	Age		Depth		Age * Depth		R^2
	F value	*P* value	F value	*P* value	F value	*P* value	
Soil pH	25.70	**<0.001**	0.393	<0.676	0.627	0.814	0.654
SOC (kg m^{-2})	22.22	**<0.001**	271.57	**<0.001**	22.98	**<0.001**	0.945
TON (kg m^{-2})	7.22	**<0.001**	34.16	**<0.001**	7.17	**<0.001**	0.862
fLF C (Mg ha^{-1})	7.37	**<0.001**	17.63	**<0.001**	0.71	0.680	0.484
fLF N (Mg ha^{-1})	9.39	**<0.001**	15.93	**<0.001**	1.02	0.429	0.508
oLF C (Mg ha^{-1})	6.04	**<0.001**	6.55	**0.002**	1.35	0.232	0.391
oLF N (Mg ha^{-1})	5.42	**0.001**	6.60	**0.002**	1.27	0.270	0.375
MaHF C (Mg ha^{-1})	18.15	**<0.001**	191.47	**<0.001**	21.33	**<0.001**	0.942
MaHF N (Mg ha^{-1})	5.32	**0.008**	24.32	**<0.001**	6.92	**<0.001**	0.855
C:N whole soil	5.14	**0.001**	19.76	**<0.001**	4.36	**<0.001**	0.559
C:N fLF	5.64	**0.001**	0.625	0.538	0.92	0.504	0.294
C:N oLF	5.59	**0.001**	0.557	0.575	0.32	0.957	0.258
C:N MaHF	4.56	**0.002**	20.19	**0.001**	4.45	**<0.001**	0.658

SOC= soil organic carbon, TON =total organic nitrogen, fLF= free light fraction, oLF=occluded light fraction, MaHF=mineral associated heavy fraction

Mean diameter at breast height (dbh), height, basal area and stand volume increased with increased stand age having a higher rate of increase from 1 to 10 and 20 years but increased at a decreasing rate from 25 years to 30 years (Table 4.1). Biomass C was 0.02, 23.0, 33.8, 57.1 and 62.6 Mg C ha^{-1} for the 1, 10, 20, 25 and 30 year old stands respectively. In MW and MF the biomass C stocks were 11 and 103 Mg ha^{-1} respectively.

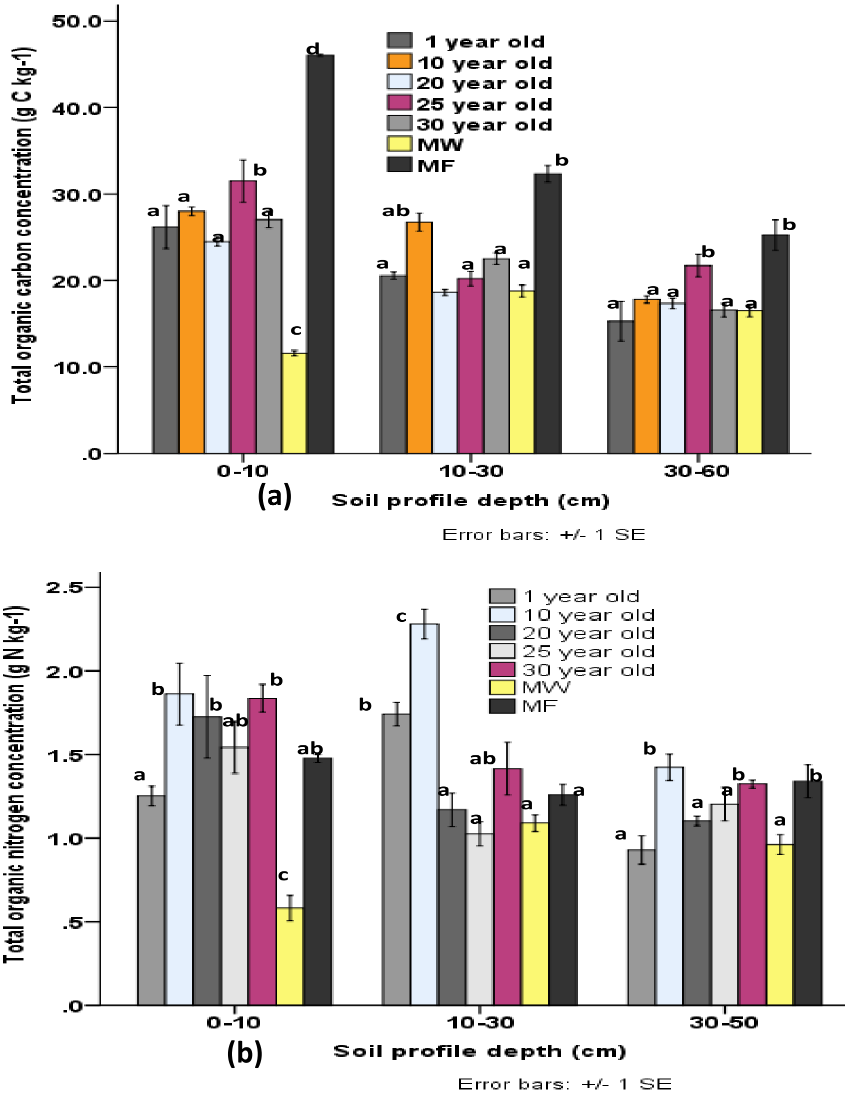
The cumulative total C and N stocks of 0 – 60 cm depth were largest under moist forest (18.3 kg C m^{-2}) and lowest in MW (8.5 kg C m^{-2}) (Table 4.4). Among the plantation stands, highest stocks were found under 10-year old stands (13.7 kg C m^{-2}) and lowest in the one year old stands (9.9 kg C m^{-2}). Among the pines there was an increase in C and N from 1 to 10 years followed by a decrease at 20 years after which there was an increase at a decreasing rate (see Table 4.4). The concentration of C and N was significantly different (p < 0.01) (Table 4.3) between the three soil layers with highest C percentages in the 0−10 cm depth except for the MW. At the 0 – 10 cm depth increment, soil C and N concentration were lowest under MW and highest under MF (Figure 4.4). Soil C concentration decreased for each stand age and natural forest with increasing soil depth from a mean of 36.9 (± 0.5) g C kg^{-1} at 0−10 cm to about 19.4 g kg^{-1} (± 0.5) at 30−60 cm depth although there were deviations in the one year old stands. Nitrogen followed similar trends from 2.1 g C kg^{-1} (± 0.1) at 0−10 cm to 1.3 g C kg^{-1} (±0.1) at 30−60 cm.

Table 4.4: depth distribution of bulk SOC and TON in *Pinus patula* stands of different ages and natural forests.

Soils Characteristics	1 year	10 years	20 years	25 years	30 years	Miombo woodland	Moist forest
Moisture (%)	$12.5(0.3)^{a}$	$9.1(0.1)^{b}$	$9.7(0.1)^{b}$	$11.1(0.1)^{ab}$	$9.2(0.0)^{b}$	$9.2(0.4)^{b}$	$18.0(0.1)^{c}$
BD (g cm^{-3})	$1.68\ (0.02)^{a}$	$1.54\ (0.01)^{ab}$	$1.69\ (0.01)^{a}$	$1.43\ (0.02)^{b}$	$1.64(0.01)^{a}$	$1.93\ (0.04)^{c}$	$1.45\ (0.01)^{b}$
Mean pH	$4.5(0.6)^{a}$	$4.7(0.1)^{b}$	$4.6(0.0)^{ab}$	$4.2(0.1)^{ab}$	$4.5(0.1)^{a}$	$4.6\ (0.1)^{b}$	$5.1(0.1)^{c}$
Bulk soil C (kg m^{-2})							
0-10 cm	4.0(1.9)	7.2(1.9)	4.3(1.8)	6.6(1.9)	4.6(1.9)	2.2(0.3)	8.4(0.3)
10-30 cm	3.5(1.9)	4.0(1.6)	3.2(0.18)	2.5(1.9)	3.8(1.9)	3.4(0.2)	5.8(0.2)
30-60 cm	2.4(1.9)	2.5(0.19)	2.6(0.19)	3.0(1.9)	2.8(1.9)	2.9(0.3)	4.1(0.3)
Total (0-60 cm)	$9.9(2.0)^{a}$	$13.7(1.2)^{b}$	$10.1(0.1)^{ab}$	$12.1(1.1)^{b}$	$11.2(2.1)^{ab}$	$8.5(0.2)^{a}$	$18.3(1.2)^{c}$
Bulk soil N (kg m^{-2})							
0-10 cm	0.20(0.02)	0.36(0.02	0.28(0.02	0.35(0.02	0.31(0.02	0.13 (0.03)	0.53(0.02)
10-30 cm	0.30(0.02	0.32(0.02	0.21(0.02	0.14(0.02	0.23(0.02	0.20 (0.03)	0.33(0.03)
30-60 cm	0.18(0.02	0.21(0.02	0.17(0.02	0.18(0.02	0.21(0.02	0.17 (0.03)	0.20(0.03)
Total (0-60 cm)	$0.66(0.01)^{b}$	$0.85(0.01)^{b}$	$0.66(0.01)^{a}$	$0.67(0.01)^{a}$	$0.22(0.01)^{ab}$	$0.22(0.02)^{a}$	$0.25(0.02))^{ab}$
C:N Ratio	15(2)	16(1)	15(2)	19(3)	17(2)	$13(3)^{b}$	$24(4)^{c}$

Different superscripts show significant difference at $p < 0.05$. Tukey's HSD test. Figures in parenthesis show standard error of the mean.

The C:N ratios for whole soil increased with stand age from 10 years to 25 years and decreased again at 30 years. Depth distribution of mineral soil C:N ratios decreased with increasing depth in all pine stands with increasing age to a maximum at 25 years after which it decreased at 30 years as a result of decreased C and increased N content (data not shown). Natural forests (MW and MF) showed no trend with depth having highest C:N ratios in MF at 10−30 cm.



4.4: Soil organic (a) Carbon and (b) Nitrogen concentration up to a depth of 60 cm in pine stands (1, 10, 20, 25 and 30 years) and natural forests (MW and MF). Different letters show significant differences for each depth at $p \leq 0.05$. Error bars show standard error of the mean.

4.3.3 Carbon and nitrogen in density fractions

The distribution of fLF C and N was significantly higher than oLF in all stands. Among the pines the fLF and oLF were highest at 10 years and 20 years respectively (Figure 4.5 a and b). Carbon and N contents of soil fractions were in the order MaHF > fLF > oLF in all forest types and ages at the three depths levels. The fLF C contributed between 8% and 13% to total organic C whilst the oLF C contributed the least (1 − 7%) and MaHF C the most (90 −91%) to total SOC. The amount of fLF and oLF C and N decreased with increasing soil depth. Nitrogen followed similar trends.

The C:N ratios of SOM fractions showed a difference between the light fractions and the mineral associated fraction. There was a general decrease in C:N ratios in each age class and natural forest in the order; fLF>oLF>MaHF (Figure 4.6). The 25 year old stands had highest C:N ratios while the 30 year old showed; MaHF <fLF< oLF. Significant differences in oLF and MaHF C:N ratios were mostly between natural forests and older pine stands.

The amount of fLF and oLF C and N decreased with increasing soil depth. The 10 year old fLF C was significantly higher than the 20 and 25 year old and the MW. Free light fraction N was significantly lower in the 20 year old stand than all except MW. Vertically there was no significant difference in fLF C although there was a decrease with increasing depth in all stands. The fLF N at 0–10 cm was significantly higher than the lower layers ($P<0.01$) (Figure 4.7).

The oLF C in the 25 year old stands was significantly lower than all stands. Natural forests (MW and MF) had significantly higher C than the 25 year old. Vertically, the oLF C at 30−60 cm was significantly lower than the 0–10 cm and 10–30 cm depths. The oLF N at 10–30 was significantly lower than the 0–10 cm depth ($p = 0.042$).

Among the pine stands MaHF C was significantly different between successive years being significantly lower than MF ($p< 0.02$) except the 10 year old stands. The 25 year old stand was not significantly different from the 20 and the 30 year old stands. The MaHF C content of MW and MF was significantly different

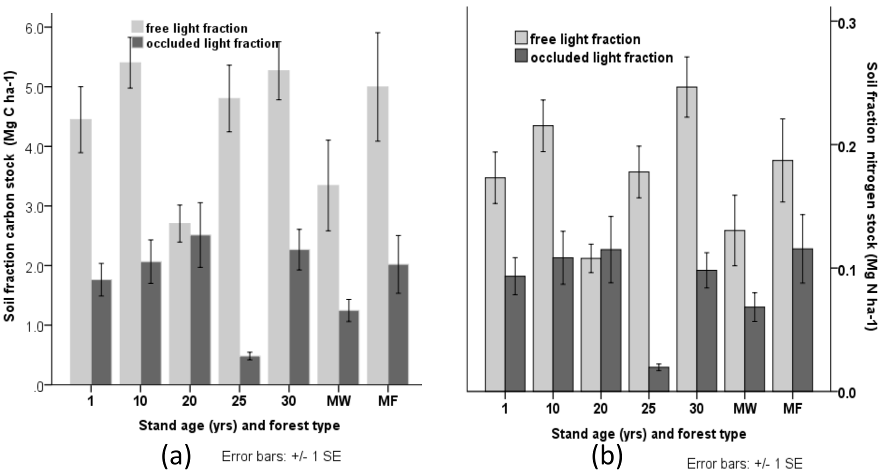


Figure 4.5:Distribution of soil organic (a) Carbon and (b) nitrogen in free light fraction (fLF) and occluded light fraction(oLF) in pine stands and natural forests.

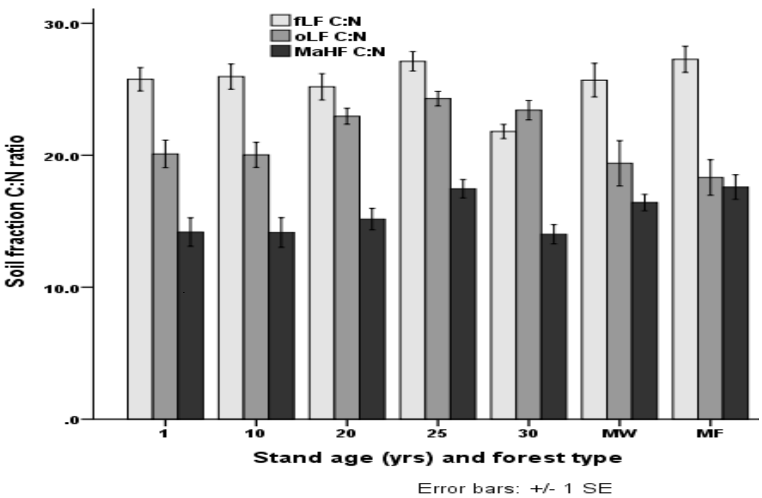


Figure 4.6: C:N ratios of three density fractions (free light fraction (fLF), occluded light fraction (oLF) and mineral associated heavy fraction (MaHF)) in pine stands and natural forests.

from the pine stands being significantly lower in MW and higher in MF. Significant differences by depth were shown between all depths with the 0−10 cm layer being significantly higher than the 10−30 cm and 30−60 cm depths ($p < 0.02$). The C and N in MaHF decreased with increasing depth except MW.

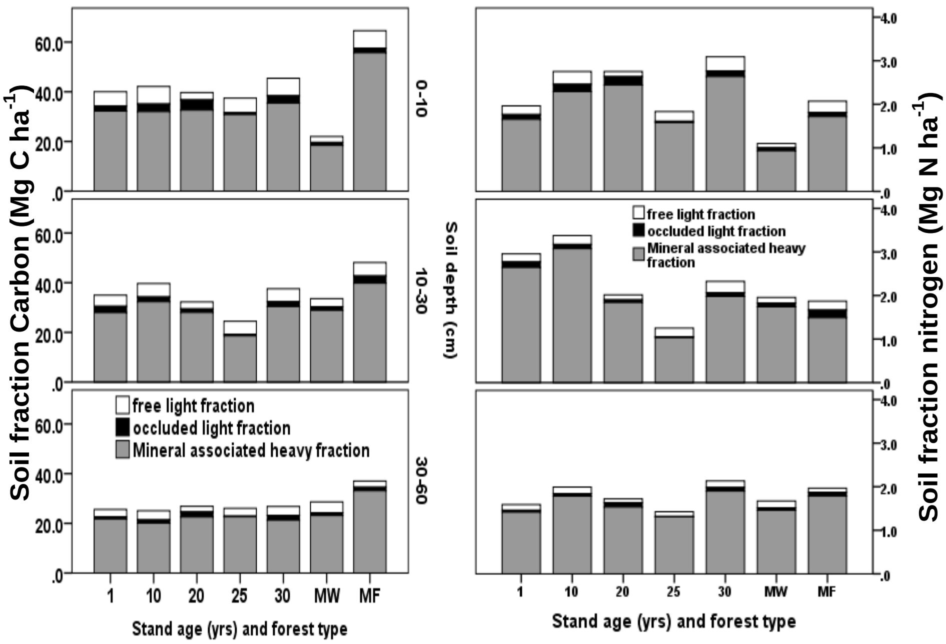


Figure 4.7: Depth distribution of carbon and nitrogen in density fractions in pine stands of 1, 10, 20, 25 and 30 years and natural forests (MW and MF). fLF = free light fraction; oLF = occluded light fraction and MaHF = mineral associated heavy fraction.

Correlations between forest floor C and C in density fractions showed a stronger relationship (67%) between forest floor and MaHF followed by fLF (60%). There was also a positive correlation between fLF and MaHF C (64%). The oLF had weak relationships with the other two fractions and with forest floor fractions.

4.4 Discussion

4.4.1 Forest floor C and N

The thickness of forest floor layers in the 30 year old stand was significantly higher than all stands (Figure 4.2). The thickness of the forest floor layers did not have corresponding effects on the amounts of C in pines. The MW had lowest forest floor thickness and soil C and N stocks. The importance of forest type and management in determining C and N stocks in forest ecosystems was demonstrated by the differences in C and N in L, F and H forest floor layers in pine stands and natural forests. These differences in C and N stocks may in turn have an impact on mechanisms of nutrient cycling (Kim *et al.*, 2010). In this ecosystem, fire was used as part of management tool to prepare the 1-year old sites and this had an effect on the amounts of litter thus impacting forest floor pools as demonstrated by the lack of the L layer in the 1- year old stands (Figure 4. 3). Czimczik *et.al.* (2003) also attributed low forest floor litter dry masses to effects of fire while working in Scots pine forests. At year one, there was no L layer but the F and H layers were similar and higher than the 10 and 20 –year olds. By the age of 10, there was an increase in H layer and a decrease in F with additions onto the L layer. There is possibility that some of the F material was transformed into H while some of the H material might have been incorporated into mineral soil by the age of 10 years and beyond.

The period shortly after establishing a new rotation by planting seedlings, shows higher decomposition than accumulation of organic material on forest floor. As the young trees grow older, a higher amount of biomass is accumulated leading to higher litter-fall. By the age of 10, there was more H layer C and N from the decomposition of accumulated organic matter that survived the fires during land preparation coupled with the accumulation from decaying pioneer species. The net C input in the initial years is not only from litter fall, but also from residue decomposition after conversion and also decay of pioneer species including grass species which dominate the forest floor before canopy closure. The relationship between age and C conforms to the Covington's curve only for the L layer where the layer starts to develop with time up to a maximum level. Covington (1981) also found a general decrease in forest floor organic matter in the first 15 years after harvesting of Northern hardwoods. In a rotation, litter-fall becomes important for nutrient cycling in the forest-soil, being the largest inflow of C and nutrients to the forest floor (Starr *et al*., 2005). After year 10 there was a reduction in H with increases in F and L layers.

At 20 years L, F and H were not significantly different while at 25 more L had accumulated on the forest floor and part of F and H incorporated into the soil. There was a reduction of H layer and an increase in F and L layers. More mineralisation and increased breakdown of L to F constituents continued up to the age of 30 years with reduced quantities of fresh litter. This cycle substantiates the importance of the current 25 year rotation age of pines where thickness of the forest floor layers and subsequently the C they contain are associated with an increase in stand age in pine tree forests (Dames *et al.*, 1998; Bens *et al.*, 2006; Matos *et al*., 2010; Penne *et al*., 2010; Shrestha & Chen, 2010). *Pinus* plantations are known to culminate in volume and biomass production at the age of 25 years which is much earlier than natural forests (Augustin *et al.*, 2007). Although total C in forest floor increased between the 10 and 30 year old stands, our results suggests that forest floor C increased at a decreasing rate beyond the age of 25 years. The results agree with the findings of Li *et al.* (2011) who found no further increase beyond the age of 35 years while working in Korean pine stands.

There was a decrease in C:N ratios in all except MW from the L to F to H layers (Table 4.2), suggesting an increase in humification with depth. Although C:N ratios in forest floor and soil generally remain stable (Yang & Luo, 2011), there were significant differences between overall C:N ratios of the one year old stand and the 20 and 25 year old stands (Table 4.2). The C:N ratios of forest floor F and H layers were lower under MF than the other stands indicating a better quality of F and H materials.

Fine root decomposition can also add C to these layers (Hoosbeek *et al.*, 2011). The influence of fine roots and charcoal on C:N ratios is also reported by Golchin *et al.*(1994b). Burning causes short term increases in soil available N which results in a stimulation of growth of pioneer species in the one year old stands. In addition, fire reduces quality of the substrate resulting in larger C:N ratio in one year old stands.

4.4.2 Forest stand and soil characteristics

Among the pines, the 1 year old stands had more moisture despite having less shade. The thick accumulation of F and H forest floor material after the process of burning for land preparation provides ground cover thus reducing soil and water loss (van Bodegom *et al.*, 2008). In addition, one year old trees take up less water than trees of 10 or more years of age. Higher soil moisture content and temperature during the early years of the rotation accelerated litter decomposition and increased the transformation of F into H litter which was manifested in the 10 year old stands. This may also have caused an increase of dissolved organic carbon (DOC) production and leaching of the DOC into mineral soil, thus subsequently adding more C into the soil.

There were significant differences in pH among pine ages which followed no trend and between MF and all the other stands (Table 4.1). The pH was significantly lower in pine stands than MF thus supporting the idea of pines acidifying soils (Parfitt & Ross, 2011; Kundhlande *et al.*, 2012). Liao *et al.* (2012) however, found no significant differences in pH of natural and planted forests while working on other coniferous species.

The bulk density in plantation stands was lower than MW but significantly higher than MF being significantly different between the one year old and the 25 year old stand (Table 4.1). There was no trend in bulk density with age although Liao *et al.* (2012) found significant bulk density increases in plantation stands with age. In this study, storage of C remained high in MF despite the low bulk density and was also lowest in MW despite having the highest bulk density. We therefore attribute C storage in the different stands with differences in C concentration (Figure 4.4).

Total aboveground tree C increased from 1 to 25 years after which it increased at a decreasing rate, demonstrating a rapid increase from the 1 year to the 10 year old stand, and from the 10 to the 25year old stand (Table 4.1). Carbon storage estimated for the MW in this study (10.7 Mg ha^{-1}) is higher than biomass C stocks of coastal miombos in Tanzania (Malimbwi *et al*., 1994) but lower than miombo woodlands in Mozambique (21.2 Mg ha^{-1}) and Tanzania (23.3 Mg ha^{-1}) (Shirima *et al.*, 2011). The estimated storage of C in MF (103 Mg ha^{-1}) is lower than estimates from tropical rainforests of Africa (202 Mg C ha $^{-1}$) (Lewis *et al.*, 2009).

The species diverse, natural moist forest which is assumed to be an old and more stable ecosystem contained more biomass and soil C and N compared to the homogenous 25 and 30 year old pine forests and the MW (Tables 4.1 and 4.4). Studies in Mozambique showed 7.6 kg C m^{-2} in MW soils (Ryan *et al*., 2011), a figure which is lower than the C in the MW in this study area and within the range of other MW in Southern Africa (3 – 13.3 kg m^{-2}). The conversion from a moist forest to a plantation, can however result in depleted C and N stocks (Hudiburg *et al.*, 2009; Gonzalez-Benecke *et al.*, 2010; Wendling *et al.*, 2010; Liao *et al*., 2012) whilst C benefits may be realised when MW is converted to pine plantations beyond 25 years. Similar results of differences between broad leaved forests and pine plantations were reported by Jandl *et al.,* (2007) and Wendling *et al.,* (2010), who showed the former ecosystems to contain more SOC than shallow rooted pine plantations. In addition, pine plantations have poorly developed rooting systems which make them less efficient at trapping nutrients when compared to natural moist forests (van Bodegom *et al*., 2008). Nevertheless, Brown *et al.* (1986) stated that plantations can sequester more C with time as they develop and grow into older age classes and supports the results of this study.

A rotation has a set of programmed silvicultural operations such as thinning (4 years and 14 years) and pruning (3 years and 12 years) and weeding which can continually add C inputs into forest floor and soil C pools and this could be the reason why the one year old stand was not severely depleted. Although the forest was under pines for 25 years, the SOC stocks were not restored to the levels of MF though part of the C is stored as root biomass. In addition, the rotation system contains charcoal produced during burning for land preparation causing redistribution of C and N at depths of 10-30 cm at 10 years and even deeper after several years. There were no significant differences between MW and the older age classes of *Pinus* stands of 20, 25 and 30 years at a depth of 30-60 cm. The lower biomass C in MW at 0−10 cm and 20−30 cm could be attributed to frequent disturbance as neighbouring communities utilise the woodland for timber and non- timber forest products and this results in a higher degradation pressure accompanied with lower C storage than their potential (Brown, 1997).

A diverse undergrowth can produce litter exposed to decomposers following establishment of a new rotation and hence the 10 year old stand had the highest C and N concentrations at a time when all pioneer species are gone which could be attributed to incorporation of F and H material into mineral soil. This high accumulation of C could be associated with litter input from the non-tree vegetation (Black *et al.*, 2009). In this way the process of succession and decomposition of pioneer species enhanced more fine roots and litter in the upper soil thereby increasing C concentrations in the early years of establishment (Hoosbeek *et al*., 2011). There was also an increases in C inputs into upper soil layers despite the burning for land preparation where fine roots are burnt and charcoal is added into the soil. Black *et al.* (2009) studied *Picea sitchensis,* Bong. Carr. and found highest sequestration rates at 10 years, which subsequently declined after canopy closure in older and thinned stands. There could also be other factors affecting C and N dynamics including management activities such as pruning, thinning and other tending operations. In this study, the C and N were lowest soon after establishment but recovered rapidly by the age of 10 years, after which it declined possibly as a result of silvicultural operations such as thinning and pruning after which was a recovery again by the age of 25 years. This shows that plantations can be used efficiently to create C sinks because of the rapid growth rates soon after establishment and additions that come as the trees develop and grow into older age classes.

There was a small decrease in soil C:N ratios before 10 years of stand development which is similar to the findings of Georgiadis (2011) who was working in temperate forests. This was supported by Olsson (1996) who showed decreased C:N ratios during the initial 8 to 15 years of stand development. The highest C:N ratio was in the 25 year old stands showing a limitation in the rate of decomposition which may be affected by the low pH values.

4.4.3 Carbon and nitrogen in soil organic matter fractions

There were no outstanding differences in mass distribution of the MaHF among stand ages and forest types. Differences were distinct in fLF and oLF showing highest proportion of fLF in moist forest whilst oLF was not outstanding but only lowest at 25 years (Figure 4.7). The pattern of C and N storage in SOM fractions was consistent with bulk soil C. Carbon and N in different forests showed greater quantities of fLF C in moist forest due to incorporation of above ground litter into the mineral soil by bioturbation. Under pine stands, bioturbation is negligible leaving the above ground litter on top of the mineral soil. Carbon and N in fLF under pines is mostly the result of root turnover.

The fLF C showed a decrease with increasing depth (Figure 4.7) in all pine stand ages except the 1 year old and natural forests which is in line with the observed root distribution. Tropical soils, such as the ones in the study area, are reported to contain high fLF C in the top 30 cm of the soil (Trumbore *et al.*, 1996) decreasing with depth due to less inputs (litter and fine roots become fewer with increase in depth). However, this trend was not evident in the one-year old stand, MW and MF. This deviation could be attributed to the influence of fire before land preparation and additions through root decomposition in the soil layers after subsequent harvesting in pine stands and root decay after land clearing. A combination of the effects of preparatory fire, management (harvest removal and pruning) and decomposition could result in addition of more free light organic matter to deeper soil layers without any pattern.

Total C and N in soil and in occluded and mineral associated fractions generally decreased with increase in depth in all forest types and ages which is similar to other studies (Tan *et al.*, 2007; Usuga *et al.*, 2010; Jiménez *et al.*, 2011). Age of a forest plays an important role in determining SOC quantities as shown by significant differences in SOC fractions. The MaHF is important for containing recalcitrant C and thus contributes more to the long term stabilised SOC pool (Tan et al., 2007). Increasing stable C with successive rotations were noted by Zhang *et al.* (2009).

The C:N ratios among the density fractions were different between the light fractions and the mineral associated fractions. A wider variation of C:N ratios among density fractions were also shown by Golchin *et al.* (1994). The observed decrease of C:N ratios with increasing SOC stabilisation, except for MW where anthropogenic effects may be dominant, is due to increased humification of SOM and the accumulation of large N-rich organic molecules originating from microbial biomass (Chan *et al*., 2008). Differences between C:N ratios of fLF, oLF and MaHF are probably due to differences in decomposition states where oLF is slightly more decomposed whilst MaHF is more advanced (Hassink, 1995).

4.5 Conclusion

We investigated the C storage potential of a Pinus patula age sequence (1, 10, 20, 25 and 30 years) and two natural forests. Cumulative C and N storage in stem biomass, forest floor and soil increased with stand age in pine stands. Although we may not have a perfect reference forest, assumption of stable equilibrium condition and C storage potential can be assessed using existing forest fragments (MF and MW). In this regard the conversion from a moist forest to a plantation forest, results in depletion of C and N stocks but conversion of a miombo woodland to a pine plantation can be beneficial in the long run. Forest floor C and N peaked at 30 years and this may be related to additions from fine root biomass and litter fall. Stem biomass increased from 1 to 10 years and from 10 to 25 years and increased at a decreasing rate thereafter. As stem biomass and forest floor C increased at 30 years, SOC decreased. Pine plantations store significantly more C and N in the forest floor than natural forests. Carbon in the forest floor decreased from L to F to H while in mineral soil it decreased with increasing soil depth. Tree biomass increased with increasing age in this pine age sequence and corresponded with increasing forest floor C. Soil organic C and N concentration however decreased for each stand age and natural forest with increasing soil depth from a mean of 36.9 g of C kg^{-1} at 0–10 cm to about 19.4 g of C kg^{-1} at 30–60 cm depth although there were deviations in the MW. Nitrogen followed similar trends from 2.1 g of N kg^{-1} at 0–10 cm to 1.3 g of N kg^{-1} at 30–60 cm.

Soil organic carbon and nitrogen storage in a Pinus patula chronosequence

In pine plantations soil C storage is maximised at 10 years and declines after thinning and gradually increases again towards the rotation age of 25 years. Thickness of litter layer did not have corresponding effect on the amounts of C stored in forest soils although there was positive correlation between litter layer and MaHF and fLF. Total C and N in bulk soil and density fractions generally decreased with increase in soil depth for all forest types and ages.

The contribution of fLF C, oLF C and MaHF C to SOC was 8–13%, 1–7% and 90–91% respectively. The C:N ratios of SOM fractions decreased as: fLF > oLF > MaHF. Plantations can therefore be an efficient means of creating C sinks owing to the rapid development rates soon after establishment and additions that come as the trees develop and grow into older age classes. Thus, the period of stand maturing between the age of 20 and 25 may be considered most important for C sequestration due to an increase in both the above and below ground C and N storage. If plantation forestry is to benefit from global arrangements such as REDD+, mitigation should aim at reducing disturbances such as fire and other forms of C emissions. The focus should be on afforestation and enrichment planting to increase and maintain the area of forest land coupled with proper monitoring and silvicultural practices that increase C sequestration. Maintenance of high conservation value moist forests has greatest benefits of both C and biodiversity conservation and these should be conserved and if possible be considered as part of future REDD+ projects. Additional studies are needed on biomass C partitioning.

Chapter 4

CHAPTER 5

Modelling soil carbon from agriculture and forest areas of Zimbabwe

Chapter to be submitted as Mujuru L. Mureva A and Hoosbeek MR (2014)

Chapter 5

Modelling soil carbon from agriculture and forest areas of Zimbabwe

ABSTRACT

Assessment of the potential for soil organic carbon (C) sequestration based on land management, soil type and climate conditions is important for selecting agricultural practices that can be used to mitigate greenhouse gas emissions. The Rothamsted carbon model (RothC) was used to predict soil organic carbon (SOC) changes in response to land management practices and climate change. Agricultural and natural forest soils on Luvisols and Arenosols under tillage and fertility treatments were assessed. Density separated soil organic matter fractionations were compared with conceptual pools of the RothC model. Modelled SOC stocks were comparable to measured stocks of 2010. There was good correlation between the density fractions and modelled values of Humified Organic Matter (HUM) + Inert Organic Matter (IOM) and Resistant Plant Material (RPM) with poor correlation for the decomposed plant material (DPM). Microbial biomass was part of RPM + DPM. Results showed strong positive relationships between measured MaHF and HUM + IOM (R^2 = 0.98). All treatments showed rapid increase in the initial years with slow increase thereafter except for the control which showed a decline in C stocks with time. The results suggest greater increase of SOC stock in clayey soils and natural forests than cropping systems on sandy soils. Sandy soils have less capacity to store more C unless there are supplementary organic inputs and/or integration of practices such as agroforestry. Results have shown that linking RothC model with measured soil data, can be

useful for estimating the potential C sequestration resulting from land management practices in Zimbabwean forest and cropping systems.

Key words: RothC, soil carbon pools, modelling, climate change, tillage, fertilisation

5.1 Introduction

Sequestration of C in agricultural and forest soils is seen as a way of decreasing atmospheric carbon dioxide concentrations and mitigating climate change. The capacity of a soil to store C depends on soil type, land management practice and climatic conditions. Human activities continue to increase greenhouse gas emissions and statistics have shown that atmospheric concentrations of carbon dioxide (CO_2) and other heat-trapping gases increased from 1.7 ppm between 1993 and 2003 to 2.1 ppm over the 2003-2012 decade. By March 2013 the concentration of atmospheric C had risen to 393.31 ppm (National Climate Data Center (NOAA), 2013).

Land use change has had a significant impact on global C stocks with cultivation reported to cause significant depletion of organic matter and releasing carbon dioxide (CO_2) into the atmosphere (IPCC., 2000). Some of the major causes of CO_2 release from the earth to the atmosphere are deforestation and degradation which are driven mostly by agricultural expansion and shifting cultivation (Williams *et al.*, 2008), production of charcoal and fuel wood (Chidumayo, 1991), legal and illegal timber logging (Sunseri, 2009), construction and wild fires. Agricultural activities that release C from the soil into the atmosphere include tillage and other forms of soil disturbances that facilitate gaseous exchange between the soil and the atmosphere. Soil disturbances also enable the incorporation of plant materials into the soil (Pretty *et al.*, 2002).

Although agricultural activities have been identified as major sources of CO_2 emissions, it is also possible to have agricultural activities that are adapted to reverse these negative effects and promote soil C sequestration in addition to other benefits of food security and ecosystem sustainability (Food and Agriculture Organisation (FAO), 2010a). Soil C sequestration can offer a valuable offset for greenhouse gas emissions in agriculture, forestry and other land uses (AFLOU) (IPCC, 2006) and benefit from existing C markets. Lal (1999) stated that developing countries have the greatest potential for soil C sequestration since most of the soils are highly degraded and therefore below C saturation. Agricultural practices that reduce emissions and promote C storage can be designed to achieve sustainable land use but the need for simple, rapid monitoring methods cannot be overemphasised. A number of scientific evidence gaps are linked to the accuracy of C accounting, ascribed to a lack of data and uncertainties related to C storage and C flux models (Stringer *et al.*, 2011).

Soil carbon models have been developed for prediction and provision of information on the rate of soil C sequestration or loss. Several models have been designed and reviewed by Smith *et al*. (1997). Among these are RothC, DNDC, CENTURY and DAISY all based on conceptual carbon pools with different turnover rates. The RothC model is among the models which have been identified by FAO (2004) as a widely applicable easy to use model. The model has been applied to estimate SOC changes in response to land use or climate change in arable and non- arable soils in many parts of the world (Jenkinson *et al.*, 1987; Coleman & Jenkinson, 1996; Kaonga & Coleman, 2008b; Yokozawa *et al.*, 2010) although with limited applicability in Southern Africa.

Changes in the rates of soil C sequestration due to changing environmental or management factors, can take several years to become apparent (Pretty *et al.*, 2006). Therefore, future impact of agriculture activities and associated land use change can only be predicted by the use of observations in combination with models which can provide a means of evaluating the changing practices in the future. Currently, there is limited information on the potential for future C sequestration in sandy and clayey soils of Zimbabwe especially on smallholder farmers’ fields despite the importance of SOC in soil biological, physical and chemical processes. Equilibrium levels are important in determining the potential of a soil to store more C and several mathematical models are used. In this study we compare the output of the Langmuir equation with the equilibrium levels estimated using RothC model.

There is lack of data for monitoring long-term SOC in Zimbabwean soils making it imperative to use the measured soil data as validation dataset for testing the RothC model outputs for the time period of 2010–2050 and to investigate long term effects of rising temperatures on SOC storage. The objectives of the study were to: (1) assess the effects of land management practices on future soil C storage (2) test the relationship between density separated SOC pools and modelled pools required in RothC and compare estimates of soil C changes between measured and simulated C. (3) assess the sensitivity of Roth C model to temperature rise and (4) compare equilibrium levels from RothC with equilibrium levels estimated using the Langmuir equation.

5.2 Materials and Methods

5.2.1 Study sites

Modelling of tillage impacts was done on soils representing clayey and sandy soils from Hereford in Bindura district (17˚42΄ S; 31˚44΄ E), Nyarukunda in Shamva district (17˚00΄S; 31˚ 43΄E) and Murewa district latitude (17° 39' 13" S and longitude 31° 48' 30" E). A description of the sites is given by Mujuru *et al*. (2013) and Mujuru *et al*. (Submitted). Briefly, Hereford soils are red clays varying from silty clay loam to clay, with characteristics corresponding to Rhodic Ferralsols (Nyamapfene, 1991; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) and falling into the category of low activity clays (Batjes, 2010). Sandy soils, are derived from coarse granite covering almost 70% of Zimbabwe (Thompson & Purves, 1981) and are classified as the Kaolinitic order, Fersiallitic group under the Zimbabwean soil classification, which corresponds to Ferric Luvisols (Thompson & Purves, 1981; Scoones, 2001; FAO, 2006) but using IPCC default classes derived from the harmonised world soils database (Batjes, 2010) they can be classified them as Arenosols, (>70% sand and <8% clay) and are broadly referred to as sandy soils. Murewa soils are granitic sands (Haplic Arenosols) (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) which are strongly weathered having low levels of available nutrients and low nutrient reserves. These are interspaced with pockets of dolerite intrusions that give rise to small patches of relatively fertile clays (Chromic Luvisols) (Nyamapfene, 1991; FAO, 2006). Bindura represented clayey soils whereas Shamva represented sandy soils for tillage assessments whereas Murewa represented both clayey and sandy soils under fertility treatments.

Seven land management practices (A-G) were selected with consideration to the current farming systems and examined using density separated SOC pools for soils at 0-10 and 10-30 cm depths: [A] - conventional tillage (CF) with maize legume rotation– consists of an ox drawn plough to a depth of 15-20 cm once before planting, residues were removed and the remaining biomass incorporated into the soil during ploughing in the next season,

[B] - Ripping (RP) - minimum tillage with an ox drawn ripper to a depth of 15-20 cm with maize- legume rotation, crop residues were to be retained in the field after harvesting, ground cover of 2.5-3.0 Mg ha^{-1} was required in RP,

[C] - Direct seeding (DS) - no tillage using an ox drawn direct seeder with synchronised seeding and fertiliser application with maize-legume rotation, residues were also retained or supplemented to achieve the 2.5-3.0 Mg ha^{-1} ground cover and cropping in each tillage system was maize (*Zea mays*. L) /cowpeas *(Vigna unguiculata* L. Walp) rotation at Nyarukunda or soy bean (*Glycine max* L. Merr) at Hereford, each treatment received annual basal fertiliser of 165 kg ha^{-1} compound D (i.e. 11kg ha^{-1} N, 10 kg ha^{-1} P, 10 kg ha^{-1} K), which was followed by 69 kg ha^{-1} N applied as ammonium nitrate in splits at 4 and 7 weeks after germination (Thierfelder et al., 2012; Thierfelder & Wall, 2012),

[D] - Natural forest (NF),

[E] - Conventional tillage with continuous maize cropping and no fertility amendments (control),

[F] - Conventional tillage under continuous maize cropping with annual addition of nitrogen fertiliser (N Fert),

[G] - Conventional tillage with continuous maize cropping with a combination of nitrogen fertiliser and cattle manure (N Fert + manure) where ammonium nitrate supplies 100 kg N ha^{-1} and cattle manure applied at 5 Mg ha^{-1} supplied an equivalent 10 kg P ha^{-1} and 0.9 % N each cropping season. Each scenario was run for sandy and for clayey soil under current and changing temperature conditions. The SOC stock predicted by RothC model were compared with the SOC data for 2010 (Mujuru *et al* 2013; Mujuru *et al* submitted).

5.2.2 Modelling of carbon sequestration potential

Data for SOC for whole soil and density fractions under tillage and fertility treatments were obtained from Mujuru *et al*. (2013) and Mujuru *et al*. (Submitted) respectively (Table 5.1). The RothC model used in this study is based on monthly time step calculations that simulate SOC turnover over periods ranging from a few years to a few centuries (Jenkinson & Rayner, 1977; Jenkinson *et al*., 1990). The model inputs were (a) climatic data (monthly rainfall (mm), monthly evapotranspiration (mm), average monthly mean air temperature (°C)), Climate data were obtained from world climate database collection of meteorological data (2013). The ETo calculator (Raes, 2009) was used to estimate potential evapotranspiration based on temperature and rainfall of each specific location (b) Soil data (clay%, initial soil organic carbon (SOC) stock (Mg C ha^{-1}), depth of the soil layer (30 cm), inert organic matter (IOM) (Table 5.1) approximated using equation [1] proposed by Falloon *et al*, (1998) because the radiocarbon content was not known and because we did not do any chemical fractionation to separate this chemically resistant pool. In this study, the IOM was assumed to be part of the mineral associated heavy fraction (MaHF).

$$IOM = 0.049TOC^{1.139} \quad [1]$$

Where: *TOC* is Total organic carbon, Mg C ha^{-1}

IOM is Inert organic matter, Mg C ha^{-1}

and (c) land use and land management data (soil cover, monthly input of plant residues (Mg ha^{-1}), monthly input of farmyard manure (FYM) (Mg C ha^{-1}), residue quality factor (decomposable plant material (DPM)/resistant plant material (RPM) ratio) (Coleman & Jenkinson, 1999). Soil cover was based on whether the soil is bare or vegetated in a particular month and is indicated as either covered or fallow (Coleman & Jenkinson, 1999).

SOC is split into four active pools DPM, RPM, microbial biomass (BIO) and humified organic matter (HUM) which decompose by a first-order process, each with its own characteristic rate, and an amount of inert organic matter (IOM) resistant to decomposition (Figure 5.1). The plant materials in RothC are subdivided into DPM and RPM, whereas plant debris in soils is found as free light

Table 5.1: Inert organic matter (IOM), clay% and SOC stocks in whole soil and density fractions in agricultural lands and natural forest systems (Mean (SD)) to a depth of 30 cm.

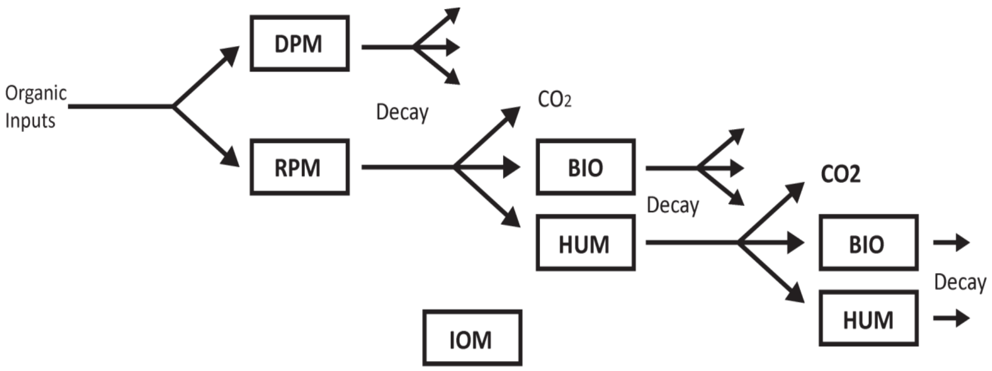
Soil type	Land management	SOC	LFC	MaHF	IOM	Clay content
		(Mg ha^{-1})				(%)
Clayey	CF	31.17(4.57)	2.18(0.36)	28.99(4.38)	1.13	22
Clayey	RP	32.27(3.04)	2.39(0.37)	29.88(3.05)	1.17	23
Clayey	DS	30.62(4.23)	2.02(0.27)	28.60(4.13)	1.10	26
Clayey	NF	43.88(5.70)	2.37(0.20)	41.51(5.17)	1.65	25
Clayey	Control	17.48(1.66)	0.67(0.10)	16.81(4.58)	1.33	54
Clayey	N Fert	24.74(1.80)	0.95(0.66)	23.79(7.08)	2.24	52
Clayey	N Fert + manure	31.12(1.61)	1.18(0.09)	29.94(7.13)	2.72	54
Sandy	CF	7.97(1.69)	0.49(0.12)	7.48(1.62)	0.24	4
Sandy	RP	10.28(1.93)	0.76(0.15)	9.52(1.37)	0.34	4
Sandy	DS	11.37(2.08)	0.75(0.25)	10.62(1.99)	0.36	5
Sandy	NF	29.25(2.57)	1.58(0.20)	27.67(2.55)	1.04	5
Sandy	Control	5.92(1.15)	0.38(0.10)	5.54(1.54)	0.38	12
Sandy	N Fert	11.66(1.60)	0.56(0.08)	11.10(1.04)	0.84	12
Sandy	N Fert + manure	10.72(1.61)	0.65(0.07)	10.07(1.52)	0.75	12

CF = conventional farming, RP = Ripping, DS = direct seeding, N Fert = Nitrogen fertiliser, N Fert + manure = nitrogen fertiliser + cattle manure.

fraction (fLF), occluded light fraction (oLF) and forms of dissolved organic carbon (DOC). The three measured fractions are particulate organic matter found outside of aggregates (fLF), POM found within aggregates (oLF), and a mineral-associated fraction (MaHF). In the model, plant C inputs are assumed to exclusively enter the DPM and RPM in proportions, determined by the source of the plant materials (Figure 5.1). The process algorithms in RothC are affected by the three climatic factors (soil moisture, temperature and plant cover) with each pool decomposing by first order kinetics having characteristic decomposition rates.

The model apportions plant litter input between DPM and RPM depending upon the vegetation type. For most agricultural crops and improved grassland, a DPM/RPM ratio of 1.44 was used, i.e. 59% DPM and 41% RPM whereas DPM/RPM ratio of 0.25 was used for deciduous or tropical woodland i.e. 20% as DPM and 80% RPM. In the model, the proportion that goes to CO_2, BIO and HUM is determined by the amount of clay in the soil (Coleman & Jenkinson, 1999).

Although RPM estimated using the RothC model correlated well with measured LFC in sandy soils and natural forests, LFC in the cropland clayey soils neither correlated with RPM nor DPM in croplands. Because of this discrepancy the model default decomposition rate constants for active compartments were initially used (i.e. RPM (0.3), DPM (10.0) BIO (0.66) and HUM (0.02)) (Coleman & Jenkinson, 1999).



RPM: Resistant Plant Material
DPM: Decomposable Plant Material **HUM: Humified Organic Matter**
BIO: Microbial Biomass **IOM: Inert Organic Matter**

Figure 5.1: Structure of the Rothamsted Carbon Model (Coleman and Jenkinson, 1999).

The measured SOC content was used for running the RothC model to equilibrium under constant environmental conditions. The constant climatic conditions were taken as the average of the climatic data from 2000–2050. For each land management practice, RothC was initially run to equilibrium (10 000 years), iteratively fitting carbon inputs to match the initial SOC stock and the corresponding distribution in compartments (DPM, RPM, BIO,HUM) with different decomposition rates. Data of C and radiocarbon ages in all the compartments received in equilibrium mode (initial soil state, initial radiocarbon ages) were used to run the model in short term mode (from 2000- 2050). The assumption implied in this inverse simulation procedure, was that RothC could simulate the dynamic changes in SOC under the specified conditions. Model estimated pools of RPM + DPM + BIO were compared with LF C (fLF +oLF) whilst HUM + IOM were compared with MaHF for each land management system. The RPM, DPM, BIO, LF represent amounts of C input.

5.2.3 Comparison of equilibrium SOC levels from RothC and Langmuir equation

SOC equilibrium levels were estimated using the Langmuir equation and compared with equilibrium levels estimated by RothC model. The Langmuir equation can be used to evaluate the adsorption of light fraction C (LF C) onto mineral surfaces, and becoming mineral associated heavy fraction C (MaHF C) and is regarded as sequestered C (Yin et al. (2005). We assumed that over time, the LF C decomposes and in part becomes adsorbed onto mineral soil particles as the MaHF C. In addition, soil minerals can randomly adsorb LF C until the MaHF has reached C saturation. Therefore, interaction between LF C and soil minerals follows adsorption and desorption processes that can be described using the Langmuir equation. Equation 2 shows the linearization used to fit the data following Yin and Cai (2005) and Bolter and Hornberger (2007).

$$\text{LF C / MaHF C} = \text{LF C / MaHF C}_{max} + 1 / (k \text{ MaHF C}) \qquad [2]$$

Where $MaHFC_{max}$ is the maximum adsorption capacity for organic C (equilibrium value for soil organic C in the MaHF) and k is the equilibrium constant. LFC/MaHFC versus LFC yields a linear relationship with slope $1/(MaHFC_{max})$ and intercept 1 / (k MaHFC). The equilibrium level estimated using Langmuir equation was compared with the equilibrium level obtained from RothC model.

5.2.4 Data analysis

Relationships between mean values (± SD) for modelled and measured SOC and equilibrium C levels in each land management system were analysed using linear regression. The goodness of fit was tested using correlation coefficient (r) and root mean square error (RMSE) in addition to standard deviation. RMSE shows the percentage term for the total difference between predicted and observed values. The bias value was calculated as Yi –Xi where Yi = measured SOC or equilibrium level and Xi = modelled SOC or equilibrium level. A t-test was conducted to compare measured with modelled C. Significant differences were tested at p≤ 0.05.

5.3 Results and Discussion

5.3.1 Relationship between measured and modelled SOC stocks

The RothC model is designed to simulate soil organic C turnover using user estimates of the C inputs making the evaluation of the soil C turnover components easier. If input data is not available, inverse simulation techniques are used to determine inputs needed to match the observed SOC in a particular year. Running the RothC model in reverse mode estimated the inputs required to attain the SOC stocks in 2010 for the seven land management practices in clayey and sandy soils. The simulated amounts of the SOC in the RothC was initially used to calculate the annual plant inputs to soils using the mode of known total SOC content.

Soil organic matter concentration at any time is the balance of the C addition to the soil pool and the carbon lost from it through decomposition and other loss mechanisms. One may argue that the lack of simulation of plant production, hence C input into soil could be a disadvantage if not done carefully. Inverse simulation techniques allow the determination of input needs to match the observed SOC. With the default setting of the decomposition rate constant for resistant plant material, there was good agreement between the simulated SOC and measured SOC (2010) in all treatments (Table 5.2). However, the largest discrepancies were found in sandy soils. On the other hand, the model appeared to under-estimate SOC in all tillage systems mainly on clayey soils. In sandy soils, SOC in CF, DS and N Fert + manure were overestimated.

Carbon Stocks on clayey soils with continuous conventional tillage, under control, N Fert and N Fert + manure had higher C than in sandy soils although they had similar inputs. Differences are most likely a result of clay content and the root biomass input corresponding to higher crop yields. Highest measured SOC was found in natural forests whilst lowest was in control treatments of both tillage and fertility treatments. Measured SOC on clayey soil was NF > RP > CF > N Fert + manure > DS > N Fert > Control whereas the modelled SOC was: NF > N Fert + manure > DS > RP > CF > N Fert > Control. In sandy soils the trend for measured SOC was NF > N Fert > N Fert + manure > RP > control > CF whilst the modelled was NF > N Fert + manure > N Fert > DS > RP > CF > control. The higher SOC in NF reflects the changes associated with conversion of forests to croplands.

Table 5.2: Soil organic carbon (SOC) measured and predicted using site specific soil input values for the 0-30 cm depth in seven land management practices on clayey and sandy soils.

Soil type	Treatment	Measured	Modelled	Difference	%
			Mg ha^{-1}		Difference
Clayey	CF	31.17(0.77)	28.45	-2.72	4.56
Clayey	RP	32.27(076)	29.38	-2.89	4.69
Clayey	DS	30.62(0.77)	30.47	-0.15	0.25
Clayey	NF	43.88(1.08)	41.18	-2.7	3.17
Clayey	Control	17.48(1.58)	19.09	1.61	4.40
Clayey	N Fert	24.74(1.46)	27.06	2.32	4.48
Clayey	N Fert + manure	31.12(1.42)	34.77	3.65	5.54
Sandy	CF	7.97(0.74)	8.42	0.45	2.75
Sandy	RP	10.28(0.77)	8.91	-1.37	7.14
Sandy	DS	11.37(1.27)	11.84	0.47	2.02
Sandy	NF	29.25(1.09)	25.84	-3.41	6.19
Sandy	Control	8.92(1.42)	7.76	-1.84	13.45
Sandy	N Fert	11.66(1.41)	13.27	1.61	6.46
Sandy	N Fert + manure	10.72(1.34)	14.43	3.71	14.75

CT = conventional tillage, RP = ripping, DS = direct seeding, NF = natural forest, N Fert = Nitrogen fertiliser, N Fert + manure = nitrogen fertiliser + cattle manure.

Under the current climate scenario, there is potential for additional carbon storage in Compared with measured SOC, RothC underestimated all SOC values in all natural forests and tillage treatments on clayey soils whereas on sandy soils the model overestimated the three fertility treatments and DS (Table 5.2). Clayey soils had lower deviations (0.25−5.54%) than sandy soils (2.02 14.75%). The regression model combining both clayey and sandy soils performed better than separated analysis y = 0.9125x + 0.7656 having significant positive correlation between modelled and measured values on both sandy and clayey soils ($p<0.001$, R^2 =0.978; SE = 1.62) Figure 5.2.

Higher estimates of C than measured could be caused by misrepresentation of clay content coupled with a soil depth which was higher than the model's 23 cm. In addition, RothC estimates lower than measured values could be a reflection of a potential insensitivity of the model to tillage. Although the model was developed in Rothamsted, UK, based on the farming systems where soil was managed by ploughing and stubble incorporation soon after crop harvest, the simulated SOC matched the observed SOC, suggesting that large fraction of the suitability of the model to predict changes in SOC in semi-arid soils.

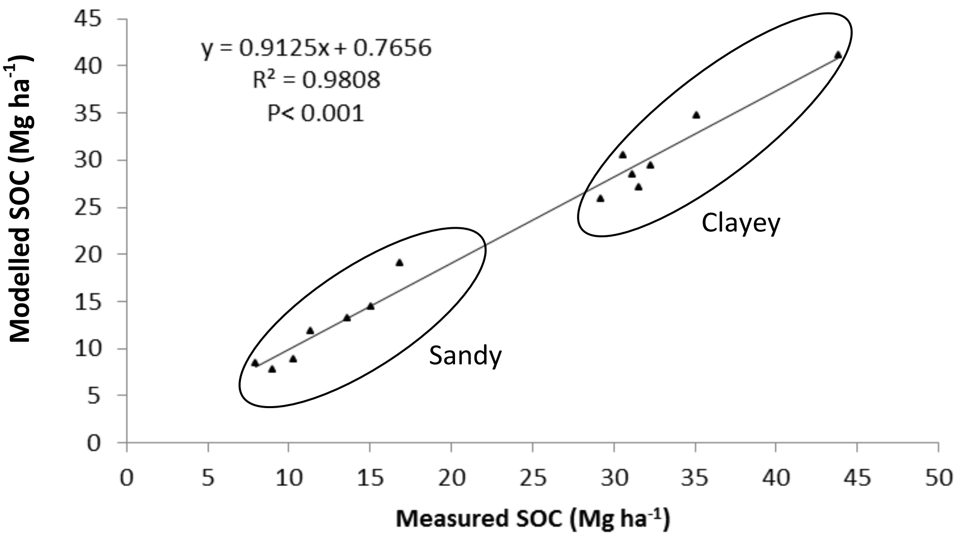


Figure 5.2: The relationship between observed C stocks and modelled C stocks from RothC carbon model.

5.3.2 Relationship between measured SOC pools and conceptual RothC SOC pools

Density fractionations were compared with model outputs of RPM+DPM and HUM+ IOM. Estimated resistant plant material carbon pool (RPM) in the RothC model and measured light fraction carbon (LFC) content were compared. The conceptual pools estimated using the RothC model for each site are shown in Table 5.3.

Table 5.3: Conceptual SOC pools in the RothC model estimated for each site

Soil type	Land management practice	RPM	DPM	BIO	HUM	Total
Clayey	CF	2.53	0.20	0.36	13.57	16.66
Clayey	RP	2.27	0.18	0.33	12.31	15.27
Clayey	DS	3.04	0.2	0.45	15.65	19.51
Clayey	NF	5.68	0.07	0.42	15.41	21.65
Clayey	Control	4.84	0.29	0.73	24.95	31.1
Clayey	N Fert	3.95	0.39	0.65	24.02	29.4
Clayey	N Fert +Manure	4.39	0.43	0.72	26.66	32.63
Sandy	CF	1.84	0.3	0.26	9.02	11.72
Sandy	RP	1.26	0.07	0.19	6.75	8.34
Sandy	DS	1.34	0.08	0.2	7.15	8.85
Sandy	NF	3.78	0.06	0.28	10.67	14.85
Sandy	Control	1.3	0.09	0.16	5.83	7.47
Sandy	N Fert	2.19	0.15	0.28	9.82	12.59
Sandy	N Fert +Manure	2.41	0.16	0.3	10.81	13.84

CT = conventional tillage, RP = ripping, DS = direct seeding, NF = natural forest, N Fert = Nitrogen fertiliser, N Fert + manure = nitrogen fertiliser plus cattle manure.

Although there was a good relationship between light fraction (fLF +oLF) C and RPM (R^2 = 0.674, p < 0.001, SD= 0.185, SS = 1.25) (Figure 5.3a) and between

MaHF C and HUM (R^2 = 0.797, p<0.01, SD=3.74, SS= 824.71), they did not match perfectly for substitution in the model but MaHF C and HUM + IOM had a perfect match (R^2 = 0.98, p< 0.01) (Figure 5.3b).

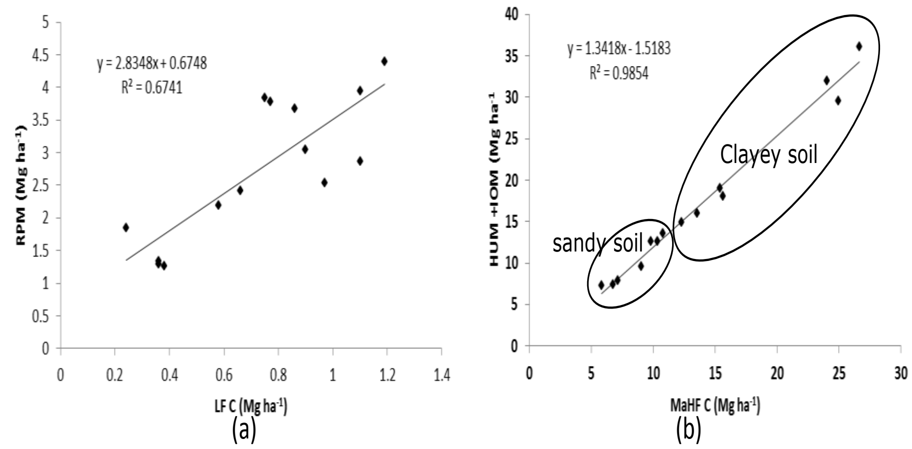


Figure 5.3: Relationship between (a) measured LF C and model RPM and (b) MaHF C vs modelled HUM + IOM.

5.3.3 Long term prediction of C storage in different systems

The average values of SOC stock estimated for a 40 year modelling period based on 2010 measured C values, showed greater increases in C stock in clayey than sandy soils. Additional storage capacity varied with land management and ranged between -29% and 36% (-12.88 to +33.96 Mg ha^{-1}). This shows that practices such as conventional farming without fertility amendments cause loss of C from the soil (29%) (Figures 5.4).

In cropping systems the conservation tillage practices reached more than 70% of their potential to store C. However, the overall assessment shows that cropping systems are only able to sequester as much C as Natural forests with addition of large quantities of organic inputs as reflected in RP and DS. Other studies confirm that the cropping systems are characterised by lower SOC than natural forests on similar soils (Guo & Gifford, 2002). The lower C in control and CF systems also support the findings of Scholes and Hall (1996) who showed that

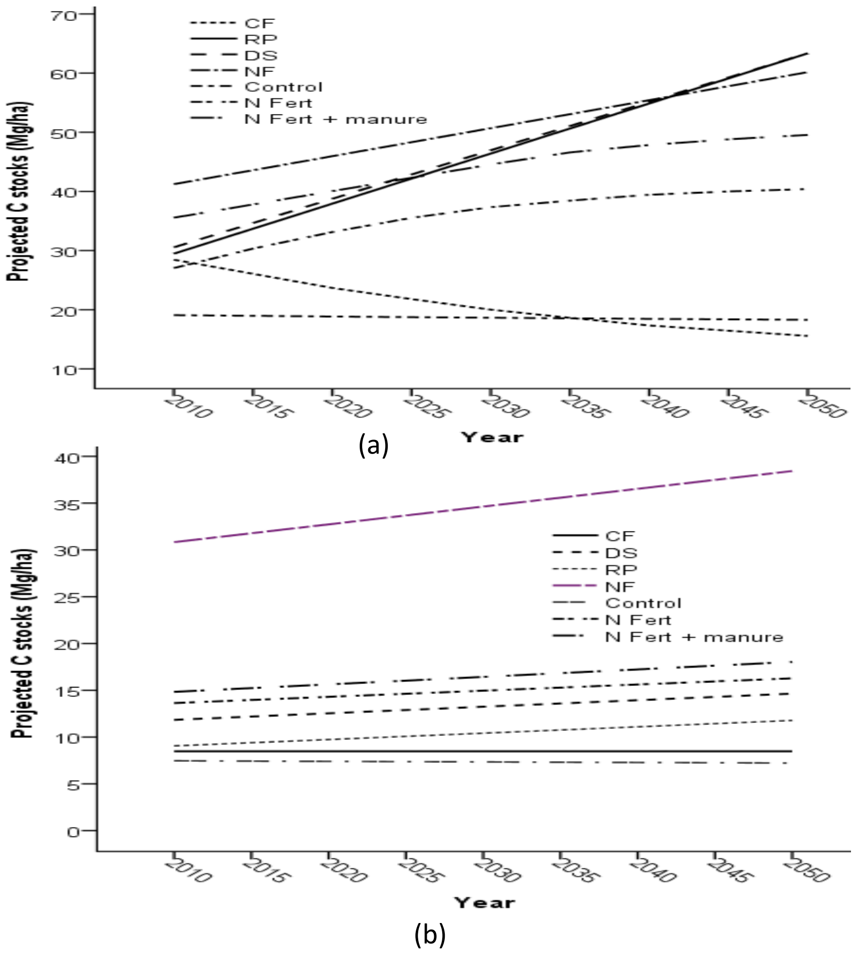


Figure 5.4: Predicted SOC in seven land management practices under current climate on (a) clayey and (b) sandy soils. CF = conventional farming with maize legume rotation, RP = Minimum tillage with maize legume rotation, DS = no tillage with maize legume rotation, NF = Natural forest, Control = Conventional tillage under continuous maize cropping (no fertility amendments), N Fert = conventional tillage under continuous maize cropping (nitrogen fertiliser 100 kg ha^{-1}) and N Fert + manure = Conventional tillage under continuous maize cropping (nitrogen fertiliser + cattle manure).

approximately 50% of SOC is lost in the first 20 years following conversion of tropical woodland, grassland or savannah. Such losses can be evident even after 5 years (Lal, 1999). Although tillage is considered a major cause of SOC loss in agricultural systems (Pretty *et al*., 2002), the model did not predict a large difference between tilled and untilled sites on clayey soils. Instead, soil fertility amendments, mainly a combination of N fertiliser and manure, can be more beneficial for SOC sequestration.

The modelling of C to 2050 showed that most sites have a theoretical capacity to store more C ranging from 0.06 t0 33.96 Mg ha^{-1} over the 40 year period. The resultant annual rates of C sequestration are in the range of 0.002-0.31 Mg ha^{-1} and 0.46 – 0.85 Mg ha^{-1} yr^{-1} in sandy and clayey soils respectively depending on practice. The rate of C sequestration in sandy soils is less than the range for dry lands (0.1-0.2 Mg ha^{-1} yr^{-1} proposed for conservation tillage in dry lands by Lal (1999) but clayey soils showed higher values than Lal *et al* (1999). Though the increase is small, the model showed higher rate of increase in the early years followed by an increase at a decreasing rate in later years. This confirms the fact that SOC storage can only be possible up to a limit beyond which no sequestration occurs because SOC will only continue to increase up to equilibrium. Clayey soils have greater potential than sandy soils whose increase is marginal in all treatments except NF. Later, mainly after 2030, a decrease of the rate of C increase is noticed in soils under N Fert and N Fert + manure on clayey soils. The rate of SOC stock increase in the initial years of the modelling period can be affected by changing input factors or incomplete initialisation.

Increased clay content supports higher SOC storage potential although it can be affected by soil depth. The aim of any land management system should be to increase plant productivity while maintaining the soil health. Results suggest that sandy soils are almost saturated and have limited capacity to sequester more C. Productivity and C sequestration can only be achieved through soil fertility amendments i.e. eliminating the constraints to plant growth in order to achieve reasonable changes in future C stocks. Thierfelder *et al.,* (2010), Thierfelder and Wall (2012) and Rusinamhodzi *et al.,* (2011) showed that average crop yields for smallholder farming systems yield about 2.5 Mg ha^{-1} grain yield and this yields about 7 Mg ha^{-1} of total above ground biomass. Thus about 3.36 Mg C ha^{-1} is produced annually as biomass C assuming plant C content of 48%.

Chapter 5

The RothC model was also run under past and future changing climate given temperature rise of 1.5˚ C rise and a baseline of 1850. A 1.5˚ C rise in temperature shows different responses in clayey and sandy soils with the clayey soils having greater C accumulation whereas in sandy soils, modelled C stocks are below current stocks. Clayey soils benefit from increased temperatures whereas sandy soils tend to decline. Under the changing climate scenario the potential for additional carbon storage is limited in all land management practices on sandy soils. Clayey soils showed similar trends for NF but greater potential for DS than the two fertility treatments (N Fert and N Fert + manure) whereas on sandy soils, fertility treatments had higher accumulation than DS and RP but lower than NF (Figure 5.5). An increase in temperature can result in increased C stocks.

Although the N fertiliser sites accumulated more C than no till sites (DS) other studies found inorganic fertilisers detrimental to C storage e.g. Farage *et al*. (2007) apart from the costs associated with its acquisition. Increased temperatures cause increased plant production resulting in higher biomass and C. However, soil respiration also increase with increasing temperature but the balance between these two processes will make the difference. In this assessment, modelling with RothC suggests that increased biomass production may be larger than increased decomposition resulting in increased C stocks.

In all cases, modelling of SOC in clayey and sandy soils of Zimbabwe showed that a new steady state will be reached if the current practices are maintained, and so subsequent declines in SOC become relatively small with time. The modelling has shown that under current climatic conditions all systems except the natural forest on clayey soils have reached steady state whereas a 1.5˚ C rise in temperature causes some of the systems on clayey soils to sequester more C. The results also show that when holding all the other factors constant, the model is sufficiently sensitive to a rise in global temperatures with sandy soils reaching an equilibrium much earlier than clayey soils.

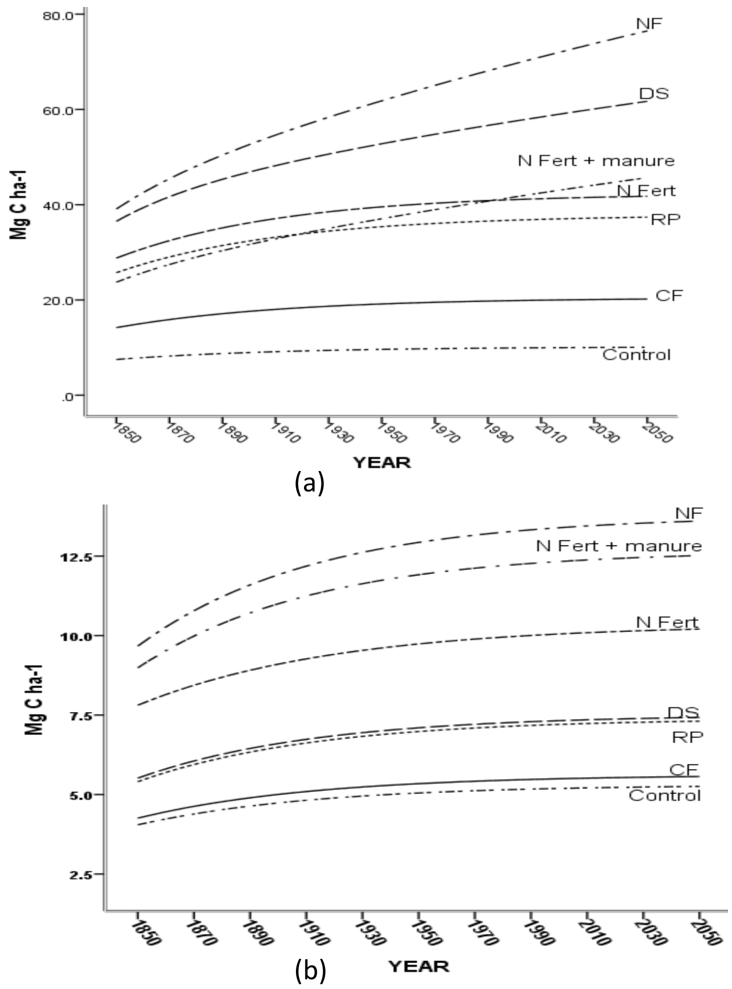


Figure 5.5: Fitted and projected SOC stocks on (a) clayey and (b) sandy soil up to year 2050 under a temperature rise of 1.5˚ C. CF = conventional farming with maize legume rotation, RP = Minimum tillage with maize legume rotation, DS = no tillage with maize legume rotation, NF = Natural forest, Control = Conventional tillage under continuous maize cropping (no fertility amendments), N Fert = conventional tillage under continuous maize cropping (nitrogen fertiliser 100 kg ha^{-1}) and N Fert + manure = Conventional tillage under continuous maize cropping (nitrogen fertiliser + 5 Mg ha^{-1} yr^{-1}cattle manure).

5.3.4 Comparison of equilibrium level estimation by Langmuir equation vs RothC model

The Langmuir equation could not estimate equilibrium levels for the three fertility systems (control, N fertiliser and N fertiliser + manure) on sandy soils due to poor model fit (Table 5.4) having R^2 values below 0.50. Information on equilibrium levels is important in determining the amount of additional C a soil can add. This is normally achieved using equations such as the Langmuir equation whose limitations have been pointed out (Bolster & Hornberger, 2007).

Table 5.4: Equilibrium C levels estimated by Langmuir equation for land management practices on clayey and sandy soils .

Soil type	**Practice**	**Equilibrium C (Mg ha^{-1})**	***R^2**	***SE**	***P value**
Clayey	CF	38.76	0.43	0.022	< 0.001
Clayey	RP	13.84	0.49	0.020	< 0.001
Clayey	DS	33.67	0.61	0.025	< 0.001
Clayey	NF	16.81	0.63	0.011	0.002
Clayey	Control	17.15	0.86	0.005	<0.010
Clayey	N Fert	44.05	0.73	0.002	<0.010
Clayey	manure	42.19	0.73	0.022	0.031
Sandy	CF	6.36	0.09	0.01	Ns
Sandy	RP	7.45	0.61	0.022	<0.001
Sandy	DS	8.02	0.17	0.019	0.050
Sandy	NF	14.22	0.49	0.013	0.011
Sandy	Control	ND	0.86	0.009	<0.001
Sandy	N Fert	ND	0.62	0.016	<0.001
Sandy	manure	ND	0.75	0.009	<0.001

CT = conventional tillage, RP = ripping, DS = direct seeding, N Fert = Nitrogen fertiliser, Manure = nitrogen fertiliser + cattle manure, Ns = not significant, ND= not determined, SE = Standard error.* from regression of the Langmuir equation

A comparison of equilibrium levels estimated using the RothC model and Langmuir equation shows positive correlation between the two methods on clay soils (R^2 = 0,87, P<0.01, SE = 4.86) (Figure 5.6). In clayey soils, measured SOC in CF, DS, N Fert and N Fert + manure were below equilibrium levels whereas RP, control and natural forest reached equilibrium. In sandy soils all systems were below equilibrium levels estimated by RothC as the Langmuir could not be applied to sandy soils due to poor model fit.

It is important to note that uncertainties of up to 20% can be found due to insufficient information about soil and climate (Poussart *et al.*, 2004). Therefore, since the data used in this study was obtained from short term experiments (4-9 years), Sources of uncertainties between the measured and modelled data can be recognised and based on the input data. For example, the values of initial carbon stock. Another uncertainty is that the data of SOC stock were modelled using 30 cm soil depth whilst the model is based on 23 cm depth. The SOC stock is calculated from bulk density and depth.

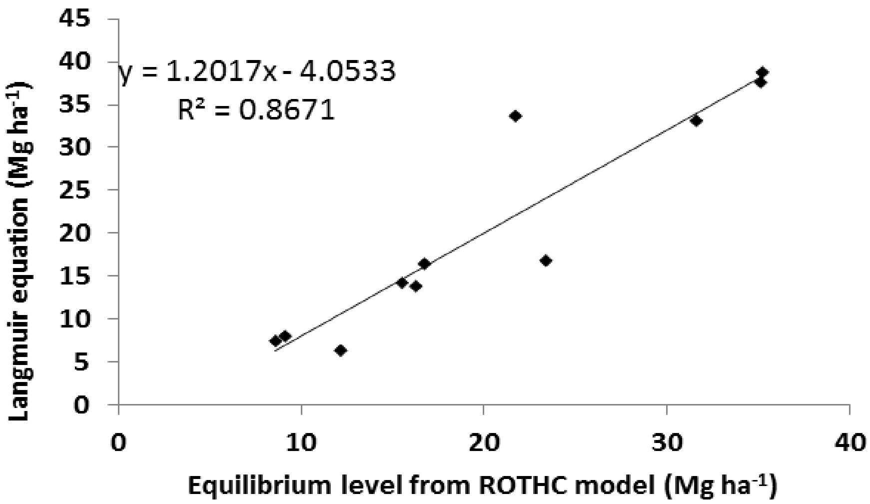


Figure 5.6: Relationship between equilibrium levels estimated by RothC model and the Langmuir equation.

5.4 *Conclusion*

RothC model is one of the most widely used models for the estimation and prediction of SOC stock on agricultural and forest land due, based on successful past evaluations and the generally good availability of required input data. The RothC model is simple and uses readily available input data to estimate SOC stock on any management systems in non-waterlogged soils. The modelling approach represented an important and promising method for the estimating C stock changes and provided some level of confidence for future soil C scenarios for cropping and forest systems on sandy and clayey soils of Zimbabwe. On the basis of our results, it can be concluded that RothC 26.3 model is able to estimate SOC stock changes on Zimbabwean sandy and clayey soils although it cannot distinguish effects of soils disturbance (such tillage intensity). Practices that had more organic inputs such as the conservation tillage practices and the N Fert + manure showed greater potential for future C storage.

There was also good relationship between equilibrium levels estimated by RothC model and those estimated using the Langmuir equation. The model also showed that under current climatic conditions all systems on sandy soils were approaching steady state. Therefore, the RothC model equilibrium output can be used for assessing the capacity of a soil to store additional C based on measured values. The model's conceptual pools of RPM and HUM showed a good correlation with measured LFC and measured MaHF in both sandy and clayey soils. Holding other factors constant, a 1.5˚ C rise in temperature causes some of the systems on clayey soils to sequester more C than the current. A higher temperature has a positive effect on the stock of soil organic C in natural forests and in soils with N fertiliser and manure inputs. The model is therefore sufficiently sensitive to a rise in global temperatures with sandy soils reaching an equilibrium much earlier than clayey soils. The modelling approach represents one of the most promising methods for the estimation of SOC stock changes and allowed us to evaluate the changes in SOC in the past and future periods on the basis of measured data.

Modelling SOC in agricultural and forested systems

Chapter 5

CHAPTER 6

SYNTHESIS

6.1 Introduction

Agricultural and forestry systems of sub-Saharan Africa (SSA), are more vulnerable to the changing climatic conditions caused by increased concentrations of greenhouses in the atmosphere (Food and Agriculture Organisation (FAO), 2010a). Projections showed several Sub-Sahara African countries becoming increasingly more prone to severe droughts (Rojas *et al.*, 2011). The models also suggested major effects on agricultural production due to disturbances in rainfall patterns and water availability regime, thus, increasing Africa's vulnerabilities to food deficits. Such challenges can be overcome by choosing land management practices that increase productivity while maintaining environmental integrity. Agricultural systems account for a large share of total land use in Zimbabwe, making them a prime target in any strategy aimed at slowing, halting, or reversing the emission of carbon into the atmosphere. Generally, there is no compensation for ecosystem services including benefits of environmental goods such as C sequestration, a factor likely contributing to the historically observed reduction in some ecosystem services manifested in form of land degradation in agricultural land and deforestation in natural forests.

Soil organic carbon is a constituent of soil organic matter important for maintaining soil fertility, soil moisture, soil structure and energy for soil biota. The dynamic processes that influence SOM quality and quantity are complex, operating through time at different locations and situations (Baldock & Skjemstad, 1999) resulting in SOM being both a source of C release (e.g. land degradation) and a sink for C sequestration. Managing land for C sequestration is a sustainable strategy that can be implemented in both agricultural and forest systems for productivity. The C storage potential of a soil is a function of existing organic C levels and biomass input relative to other factors (e.g. the practice, climate, soil mineral composition, soil biota, and position in the landscape) facilitating the continuous exchange of C between the earth and the atmosphere. In this way the, rate of sequestration or release of C is strongly influenced by these interactions with isolation of the influence of each individual factor often complicated (Lal, 2007). In a meta –analysis, Guo and Gifford (2002) showed the importance of land use on C storage by revealing increased soil C stocks following land use change from crop to pasture while the reversed land use change led to decreased C stocks. This supports the idea that the feasibility of increasing soil C storage depends on the management practices, type of soil and fertilisation (This thesis). However, C (air) is a common property, where any one individual, community, organisation or nation takes only a small share of the cost of the C they add to the atmosphere. In this regard, the private costs that motivate decisions to sequester C may fall short of social costs, resulting in too much carbon dioxide accumulating in the atmosphere and affecting all. Despite this fact, land management practices that minimise C losses by creating positive ecosystem C budget and enhancing C storage need to be identified and promoted. Such identification improves our understanding of the extent to which each land management practice affects greenhouse gas reduction efforts and mitigation of human induced increases in atmospheric carbon dioxide.

Alternative livelihood development strategies have been suggested as a means of enabling rural people to shift from subsistence livelihoods through projects, such as fisheries and beekeeping. Such projects help to decrease the amount of deforestation and forest degradation in line with REDD+ objectives by shifting local economies away from activities that damage forests, such as clearing more land for cropping, unsustainable charcoal production and firewood vending. Initiatives with such a focus have met with varying success e.g. in Kenya Wild life Works have established a clothing factory and in Tanzania they have started bee-keeping projects as part of REDD+.

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The land management options should focus on increasing net primary production (NPP) and reducing C loss from soils (e.g. combating land degradation and deforestation). In the quest to promote soil C sequestration to mitigate climate change, it is important to realise that the C sequestration process is based on the fact that: the process of soil C storage is reversible; soils have finite capacity to store C and is linked to fluxes of other greenhouse gases, such as nitrous oxide (N_2O) and methane (Powlson et al. (2011).

Changes in SOC associated with land use and management occur slowly and can be explained by the way C and N are allocated in different SOM fractions. Density fractionation can separates SOM that is residing outside aggregates (fLF) from that inside aggregates (oLF) and organic matter bound to minerals (MaHF). The pools/fractions have different turnover and they can show impacts of land management practices on SOM (this thesis) which are normally not evident in whole soil. Thus, SOM fractions serve as good indicators of both short term and future changes in total C and N stocks and provide evidence of stability of each fraction. However, the storage of SOC is suggested to be influenced by selective preservation of recalcitrant compounds, physical protection against decomposition and interactions with mineral surfaces (von Lützow *et al.*, 2008).

6.2 Potential of tillage management practices for SOC sequestration

Agriculture activities have been one of the key drivers of deforestation in most of the tropical countries and agriculture is likely to benefit from agroforestry practices that have potential to increase soil fertility at minimal costs. The use of agroforestry can be part of the afforestation/reforestation or it can be a special option by itself to contribute to emissions reductions through additional sequestration and/or avoided emissions. When land is managed using a combination of agricultural and forestry approaches, there are benefits of sequestration of additional C in trees and/or soil thus, reducing C emissions when compared to business-as-usual agricultural practices. Agroforestry simultaneously increases plant cover more than contained in natural woodlands. Planting of trees in agro ecosystems can be in the form of protection of existing trees in agricultural land, creation of agro-forestry parks, soil fertility enhancement and soil erosion control, other on farm tree planting activities, fruit tree planting or livestock management. Trees can either be native or exotic species. In this regard, planting trees reduces the need to open up more natural forests for crop production and thus, help to strengthen small holder forest management initiatives while mitigating climate change since trees not only have the potential to sequester and store additional C in above ground biomass, but even more so significantly increase SOC. Natural woodlands on sandy soil had lower C and N stocks than woodlands on clayey soils (Figure 2.3).

Although there are no estimates of total GHG emissions from deforestation and degradation of African woodlands (Bond *et al.*, 2010) the socio-economic relations between the state, private sector, and local people, coupled with the persistent poverty have hastened the rate of deforestation. Poverty, hunger and increasing demand for agricultural land have chiefly driven local communities to over exploit forest resources for their livelihoods. Greater loss of C into the atmosphere is attributed to the conversion of native land to agriculture and much of the newly cropped land is unsuitable for agriculture and degrades quickly, thereby forcing the farmer to convert even more land to agriculture (Walker & Desanker, 2004). There is however a relationship between SOC content and soil texture as shown by the greater difference between native forests and croplands on sandy soils than clayey soils (Table 2.1). Such differences are likely a result of harvesting of major above ground components in croplands without substantial returns to the soils. There are competing uses for residues (Livestock feed and fuel). Aboveground biomass only enters the soil labile carbon pool via roots, root exudates and litter input leading to increased C and N storage at deeper layers of the soil. It is estimated that about 1.14 Pg C may be annually emitted into the atmosphere through erosion induced processes (Janzen, 2006). In addition, accelerated erosion and other degradation processes cause significant loses of topsoil (Lal, 2012) although the impacts of erosion on C losses depends on the fate of the C after the soil is deposited.

Smallholder farmers have traditionally used the hand hoe for tilling and weeding land. The hoe only ploughed the upper surface leaving a hard pan at lower levels. A development from the hoe led to the use of the mouldboard plough going deeper and breaking the hard pan thus allowing rain water permeation. The plough exposed soils to greater bioturbation. Tillage is known to affect distribution of organic C in the soil profile, mixing soil materials in the plough layer and destroying soil structure. Ploughing exacerbates depletion of SOC depending on climatic, edaphic and management factors. To date, several tillage practices are used to improve soil moisture and fertility in croplands. Cultivation and tillage however, affect the distribution of SOC depending on the quantity and quality of SOM. Depletion of SOC in arable or other highly disturbed systems, can also occur due to lower plant litter input (Figure 2.2), faster SOM breakdown (as it is more accessible in soil aggregate disruption during ploughing) and increased displacement of C-rich surface soil through erosion. The SOC can be preserved using maize/legume rotations with magnitudes varying with soil texture and tillage intensity (Chapter 2) and can also be increased using agroforestry technologies.

Furthermore, the inherent infertility of Zimbabwean soils (Nyamapfene, 1991) coupled with regular tillage operations result in lower C levels in croplands than native forests. Depletion of the SOC pool in tillage systems demonstrates potential impacts of conversion of natural forests to croplands although other interacting factors such as variable temperature regimes, low biomass C inputs, higher decomposition rates and alterations in soil moisture determine the pace. Although the tillage studies (Chapter 2), were short term observations, the analysis took into consideration both the bulk/whole soil analysis and density fractions. Density fractions, mainly the light fraction are a good indicators of the impacts of short term changes in land management (Marín-Spiotta *et al*., 2008). Despite the short period of time in this assessment, results suggest that there were short term increases in whole soil C and N stocks over a four year period with magnitudes varying with soil type (Chapter 2). In this study, SOC and TON were greater under minimum and no tillage activities at 0-10 cm depth in both sandy and clayey soils when compared with initial stocks. Contrary to this, clayey soils showed conventional tillage being comparable to ripping (minimum tillage) and direct seeding (no till) (Table 2.2). Sandy soils are however more sensitive to tillage and tend to benefit more from no tillage practices and this supports other scholars who also found C storage benefits under no tillage (DS). Although the methodologies used to analyse the initial C stocks could be different, a comparison of conservation tillage practices with conventional tillage showed similar trends to those obtained by comparing with the baseline data.

In this study the magnitude of SOC and TON depletion was minimised by ripping and direct seeding (Chapter 2) when compared with conventional tillage on sandy soils (Figure 2.3). Results suggests that tillage and other forms of soil disturbance can release C from the soil enhancing gaseous exchange between the soil and the atmosphere but can also facilitate the incorporation of plant materials into the soil (Pretty *et al*., 2002). Better management, more C inputs and the root biomass incorporation into the soil during ploughing contributed to a an increase in SOC under conventional tillage. In tillage systems, plant roots are the major source of organic matter input to the soil whereas in forest systems the major component is above ground biomass retained after litter fall in addition to below ground parts.

Soils with low C stocks have less capacity for SOC accumulation as shown by sandy soils having less capacity to stabilise more C and N than clayey soils (This thesis) although the extent of C and N storage is modified by land management practice and other environmental factors. Results show the likelihood of strong C decomposition regime facilitating increased sequestration of labile C during the cropping season. Although sandy soils showed low greenhouse gas mitigation potential (shown by low potential of C storage) when compared with clayey soils, results suggest that the no tillage system (DS) is the best management for C storage on sandy soil whilst the conventional tillage (CT) system is the worst. Sandy soils are hence more sensitive to disturbance.

Currently tillage practices are confined to farmer managed experimental plots with potential for up scaling on small portions of land. Inputs and equipment are supplied by the project where a farmers share the direct seeder and the ripper. The experimental plots have provided some information on technical performance of each tillage practice. The relevance of the practices is however hampered by two major challenges: 1) Policy that does not allow residue retention because livestock graze freely in the dry season. Residues either have to be removed or left in the field for livestock to graze.2). Even if left in the field, residues are consumed by termites and other small organisms during the long dry period lasting five to six months. Given this scenario, all conservation tillage practices get supplementary residues (grass) at the beginning of each cropping season. In sandy soils, the practice of conservation tillage increases soil C and N in the top soil layers when compared to conventional tillage. Other factors such as climatic conditions and possible incorporation of residues under conventional tillage did not lead to more soil C loss when compared with no tillage (DS) at 0−10 cm. In this case conservation tillage practices did not show expected results of greater C and N possibly due to competing needs for crop the residues (mainly as animal feed).

In clayey soils, SOC and TON stocks in conservation tillage practices (RP and DS) were not significantly different from conventional tillage but were lower than in natural forests. Clayey soils seem to be more affected by various stress factors including a long dry spell and trampling by both small and large livestock causing possible mixing of soils in the upper soil layers and reduction of residues left *in situ.* Ideally, no till and reduced tillage plots should have continuous surface cover and this is usually violated because of institutional and socio-economic factors. Farmers are not able to fence off their fields over the dry period subjecting all fields to open access grazing. Only small portions on experimental plots and home gardens are protected. Fencing does not make the fields inaccessible to small livestock e.g. goats. To compound the matter, the erected fences are usually vandalised as is the case in the study area.

Although conventional tillage practices are often known to have a greater rate of SOC loss when compared to conservation tillage, there was a relative SOC gain under conventional tillage at 0−10 cm when compared to reduced tillage (RP) and no tillage (DS) practices on clayey soils whereas on sandy soils no tillage (DS) had better C storage than RP and CT. Some studies have shown that conservation tillage increases C storage in the 0−10 cm depth while conventional tillage decreases C stocks at the same depth level. The lack of significantly different C and N gains under conservation tillage systems could be due to the limited residue cover which makes the soils even more vulnerable to agents such as wind erosion than ploughed soils, where the transient roughness created by tillage may reduce wind and water erosion (Blanco *et al.*, 2009). Thus, the inadequate amounts of crop residues coupled with climatic conditions (with a long dry spell) could have direct influence on soil water conservation, soil erosion control and nourishment of microbial communities in each system.

Despite having lower C and N stocks than NF, tillage systems increased C stocks at 0-10 cm. Interestingly, the rate of C increase in CT on sandy soil was similar to the rate of C increase under DS on clayey soils although the actual values were higher in clayey soils than sandy soils. Increase in C and N stocks in cropping systems has been attributed to better nutrition and management intensity compared to the time prior to the start of the experiment. For example, fertiliser application is limited in most smallholder farms as inorganic fertiliser is usually beyond the reach of most farmers. As a result, most fields are highly depleted creating a potential for C sequestration. Sanderman *et al*. (2010) reported between 0.2 and 0.3 Mg C $ha^{-1}yr^{-1}$ increases in C stocks under improved land management of croplands (including enhanced rotation and no-tillage) than under conventional management across a range of Australian soils. They found greatest largest gains within the first 5 to 10 years with the rate of change decreasing to almost zero after 40 years and they attributed absolute declines to continuous responses of soils to the initial disturbance of the native soils. Although Lal *et al* (1997) suggested that conservation tillage may provide even less opportunity to increase SOC in the tropics, results suggested that conservation tillage can lead to high SOC near the surface compared to conventional tillage only in sandy soils. The hypothesis that soils under no tillage management accumulate more SOC in surface soil layers than conventional tillage (Jagadamma & Lal, 2010) can only be applicable to sandy soil in this study since on clay soils CF accumulated more C in surface soil than RP and DS. However, at lower depths, RP and DS had more C and N (Table 2.2) which is contrary to most studies which associate CF with distribution of soil organic C and N in the profile. VandenBygaart *et al*. (2002) found that at 0-30 cm, DS accumulated more C and N than CF and RP. This was however contrary to clayey soils where the highest amounts at 0−30 cm were under RP and the lowest C stock were under DS. There was a demonstration of differences in C storage as affected by texture, mainly the clay content. Clay content is known to exert a major control on amounts of organic C a soil can store (Schimel *et al.*, 1994). Biotic factors influence the storage of C and N in the soil although they were not part of this study.

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There is need for continuous maintenance of environmental benefits resulting from increased SOC content. Fundamental soil biological processes, which govern soil nutrient cycles are based on key processes of soil C storage and can be achieved by either reduced decomposition or improved management, either by fertility amendments, or by adoption of reduced tillage. Exposure of organic material to microbial degradation results in less fLF. The SOC pool receive fresh organic matter through litter input whereby soil microbial activity drives the process of conversion of litter into stable humus and is also related to bioturbation (e. g. termite and earthworm activities), which affect both aeration and SOM incorporation into clay minerals at deeper soil layers. In this study, the activity of arthropods was important for mixing organic matter at different soil layers. The transformation of litter to SOC can be illustrated as in Figure 6.1.

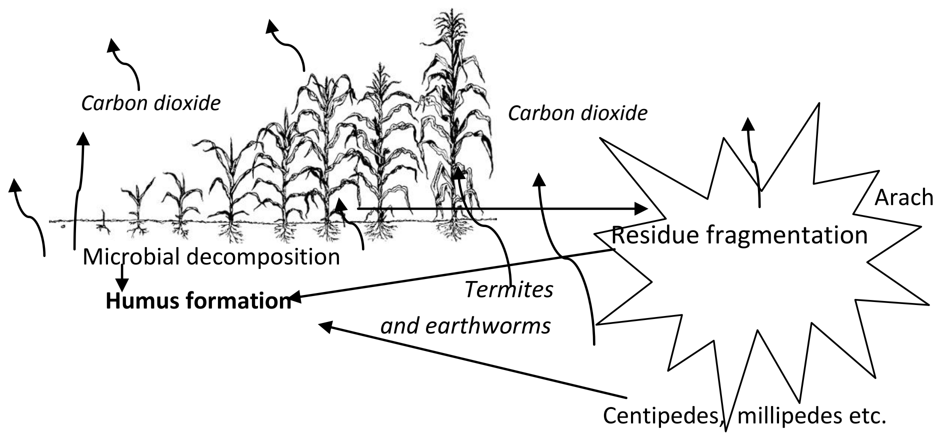

Figure 6.1 Process of litter transformation in litter decomposition in agricultural systems

Accordingly, the sampling time of the study (spring) might also affect the amounts of C in the different SOC pools in both croplands and forests. Shortage of organic matter for soil fauna is often demonstrated by the consumption of crops in the field (mostly by termites) mainly towards the end of the growing season. Litter and the resulting humic substances are decomposed, mainly by the bulk soil micro-organisms that comprise bacteria, fungi and soil meso- and macro-fauna, resulting in respirations and subsequent soil CO_2 efflux and alterations to the soil chemical and physical properties. Seneviratine (2003) suggested inoculation of soil fauna into litter to enhance decomposition, thus improving nutrient storage. Activities of inoculation are however, not necessary in smallholder areas under study as the litter is always inadequate for the decomposers mainly during the cropping season when climatic conditions are favourable.

The natural forests are often subjected to annual fires making a possibility of charcoal fragments floating with the fLF and oLF since they are within the same densities. It is therefore possible that charcoal derived from annual burnings contributed to fLF and oLF C in all sites. The overlapping densities make it impossible to separate the charcoal from plant materials in light fractions. The fLF and oLF reflect the extent of alterations imposed on the soil by management activities and can thus be used as sensitive indicators of the quality of a soil. The light fraction C accounted for the lowest proportion of the total soil C in all tillage (6.6%) and forest systems (4.2%) on sandy soils, but the proportion increased and stabilised in the clayey soils (Table 2.3).

Although most studies of SOC are based on long term assessments, the light fraction C is thought to be an early indicator of soil quality improvement and C sequestration because it is more sensitive to land-use and management practices than total SOC (Six *et al*., 2002). In support of this, Soon *et al*. (2007) showed light fraction C significantly responded to tillage after 4 years, whereas the tillage effects on total SOC was not apparent until the 12th year. In the present study, I found that the response of the light fraction C storage was not more sensitive than the response of total soil C. The clayey and sandy soils are in areas with a long dry period characterised by slow decomposition resulting greater fLF. The greater amounts of fLF in sandy soils show importance of fLF in soils of limited sorption capacity.

The lack of baseline data on SOC pools makes it difficult to gauge the effects of land management activities on SOC pools. However, if we consider the natural forest (chapter 2) as the natural state, we observe that conservation tillage practices and fertility amendments increase SOC and TON in MaHF. Ripping and direct seeding show an increase of 5% each on sandy soils while they increase C by 5 and 7% respectively in clayey soils when compared with conventional tillage.

Furthermore, DS on clayey soils has a difference of 1% with the adjacent natural forest while on sandy soils differences with natural forest are up to 9%.

The main way of enhancing stable SOM pool is to increase litter incorporation and humus formation rates (de Moraes Sà & Séguy, 2008). The maximum level of C stabilisation under conventional tillage is likely to be lower than that under RP and DS because of faster decomposition and lower soil aggregation (Reicosky & Archer, 2007). The stability of SOM is demonstrated when organic matter resists further transformation or degradation (Sollins *et al.*, 1996). The MaHF controlled the behaviour of all soils. The lack of significant differences between CT, RP and DS on clayey soils may be attributed to relatively sufficient C and N supply by the crops during the growing season, the amounts of clay and silt and possible existence of Fe and Al oxi-hydroxides (although these were not assessed in the study). The cropping season is characterised by rapid depletion of organic matter likely to have corresponding effects on the increase in MaHF C and N. The accumulation of greater quantities of MaHF C and N in both sandy and clayey soils could be a result of the transformation of fragmented litter into humus by earthworms and termites. Once plant residues enter the soil mineral horizon, they go through microbial degradation and, are simultaneously stabilised through interactions with soil mineral particles.

The management options that increase SOC should increase productivity, profitability and promote sustainability of a land area. Although residue retention is critical for the success of conservation tillage practices, local institutional arrangements during the long dry period are not favourable (fields are communally grazed by animals and left overs are consumed by termites and other micro fauna. For this reason residues are supplemented by grass at the beginning of the cropping season to maintained the recommended 2.5-3.0 Mg ha^{-1}. Furthermore, conservation tillage practices have lower short term gains despite the insignificant SOC stocks among the tillage systems (especially on clayey soils), conservation tillage practices have lower short term gains in crop yields than conventional tillage (Thierfelder *et al.*, 2012; Thierfelder & Wall, 2012) with better gains at three years i.e. after accumulation of residues and consequently SOC. The low production makes it difficult for farmers to understand the benefits of conservation tillage practices if associated with some loss in the initial years.

Smallholder farmers grow crops for subsistence and cash income and they may not be willing to loose production for a reason. Therefore, to facilitate faster adoption of reduced and no tillage practices, there is need for subsidies in the initial years to compensate farmers for opportunity costs foregone. The future is also uncertain given the possibilities of climate change effects with no guarantee for a good harvest in the next season. Increased C stocks and better crop yields make conservation tillage practices superior only after the initial years. In sandy soils, better moisture retention and better management make DS more superior than the other two practices. The reduction in crop yields in initial years under conservation tillage may reduce the chances of farmers shifting from conventional to conservation tillage. In order to match yields in conventional tillage practices, conservation tillage practices need more N fertiliser in the initial years (Rusinamhodzi *et al*., 2011).

Furthermore, since SOC is an environmental good/service, there may be need for incentives, either in the form of direct government subsidies or credits from an emissions trading market to stimulate positive uptake of technologies that increase SOC stocks. Such regulatory measures will prepare the ground for adoption of other practices which in the same way as CA can be hampered by unfavourable institutional arrangements. In developing countries, the strong theoretical basis for SOC sequestration is partially supported by a limited number of field studies. In addition, a general lack of research in this area is currently preventing a more quantitative assessment of the potential of soil C sequestration in both agricultural and forest soils. Smallholder farmers can only benefit from CA practices in the presence of government support (e.g. for conservation tillage equipment and supportive policies) and this will result in achievement of the goals of C sequestration, soil conservation and eventually better crop production (FAO, 2010).

6.3 Effects of fertilisation on SOC and TON storage

Poor soil fertility coupled with erratic rainfall constrain food production and sustainability of small holder farming systems. The inherently infertile soils have been identified to have potential for enhancing SOC sequestration with an estimated global sequestration potential of 0.4–1.2 Gt C yr^{-1} with corresponding positive impacts on crop yields (Lal, 2004). There is however, need for continuous replenishment of the labile C pools through practices that increase C inputs and stabilise labile C pools (e.g. the return of crop residues and the application of manure and fertilisers). Techniques that have been proposed for improving soil fertility include legume intercropping, green manuring, agroforestry (e.g. improved fallows), inorganic and organic fertilisers (Vanlauwe *et al.*, 2010). Addition of manure and inorganic fertilisers has often been recommended to increase SOM and enhance soil C sequestration (Gulde *et al*., 2008). Several scholars have highlighted the benefits of combining manure with inorganic fertiliser for improving SOM and its fractions (Rudrappa *et al*., 2006; Purakayastha *et al*., 2008).

Application of N fertiliser and cattle manure reduces depletion of SOC and TON stocks in cropping systems (Chapter 3) and show that amendments reduce depletion of SOC stocks. Mann (1986) showed that conversion of native vegetation to croplands had impacts ranging from a loss of 70% to gains of up to 200% in SOC stocks depending on soil type, fertilisation tillage system, cropping history and sampling depth with the greatest changes occurring in the first 20 years. When conventional tillage is the only available option, the application of nitrogen fertiliser (chapter 3) is more beneficial for increasing C stock in sandy soils. Nitrogen fertilizer can increase soil organic matter in soils that have nitrogen deficiency but emissions from CO_2 released from fossil fuel combustion during the production and transport can reduce the net amount of carbon sequestered in fertilised systems. The fertilisers can also be lost through run off into nearby streams and water bodies where it may have significant negative ecological effects. In some cases the use of N fertiliser without organic amendments leads to mineralisation of SOM and loss of SOC (Manley *et al*. (2002). This was however true for clayey soils where the application of manure plus N fertiliser was more favourable than N fertiliser alone.

Factors such as soil moisture and soil texture are also important in facilitating changes in micro climate resulting in more rapid mineralisation of C in soils that have a high initial C stock (Mann, 1986). Blanchart *et al*. (2007) showed that on sandy soils, fertiliser application causes an initial nitrogen flush early in the cropping season (about 20 days), with net N mineralisation accompanied by net nitrification favouring loss of N early in the cropping season (about 20 days). The N flush is accompanied by increased microbial biomass at the onset of the rainy season which is followed by rapid decay of organic matter in soils that have limited capacity to protect their organic matter. The accelerated decomposition during the cropping season consequently results in more soil CO_2 releases into the atmosphere. However, increased decomposition rates may stimulate greater soil N availability, leading to higher net primary production (NPP) with potential to increase C inputs into the soil through rhizodeposition and litter fall thus offsetting increased soil C loss (Gelman *et al.*, 2013). Soils might then gain more C than they lose due to such favourable environmental conditions in a N-rich environment. The higher N content might also stimulate initial litter decomposition but also suppresses humus decay in some stages, leading to stabilisation of SOM in mineral-associated fractions. Results support the hypothesis that application of N fertiliser increases N availability causing increased NPP and subsequently litter production and thus soil C inputs (Figure 3.2).

Rudrappa *et al*. (2006) and Brar *et al*. (2013) found considerable build-up of SOM fractions under fertiliser being greatest under a combination of manure and N fertiliser. Favourable moisture and temperatures during the cropping season promotes greater mineralisation of fLF organic matter resulting in less accumulation of LFC and N in fertiliser and manure treatments on clayey soils. The fLF C and N is governed by the degree to which temperature and moisture conditions constrain decomposition of accumulated SOM (Biederbeck *et al.*, 1994). The addition of N fertiliser and manure can significantly improve fLF C at 0−10 cm and at 10−20 cm depths when compared with the control and N fertiliser only. In clayey soils, the control treatment had the lowest fLF C at all depths levels due to low productivity capacity, inputs coming from better crop growth after addition of fertiliser and manure. The oLF was more under fertiliser at 0− 10 and 10−20 cm in clayey soils while in sandy soils there was no statistical difference at the three depths and also among treatments. Increased fLF and oLF N under N fertiliser may be a result of priming effect of fertiliser on fresh organic inputs, which stimulates microbial activity and eventually decomposition.

However, the use of both fertiliser and cattle manure may limit the number of farmers participating in carbon sequestration activities, mainly because of the prohibitive costs of inorganic fertilisers and the lack of draught power. The poor will not be able to contribute unless some enabling economic and institutional policy strategies are designed. Existing carbon markets are likely to have only modest impacts on the poor, even if relatively high carbon prices are offered. It is highly unlikely that poor farmers will participate in carbon sequestration projects as they are likely to be constrained by the same attitudinal, economic and institutional factors that have inhibited their socio-economic improvement in the past.

There was strong positive linear relationship between LF C and MaHF C on clayey soils whereas sandy soils had no linear relationship. It is possible that fLF and oLF in both sandy and clayey soils may be exhausted and need to be replaced through continuous cover. The linear relationship on clayey soils showed a decreasing intercept representing a decline of labile SOC with the remaining levels being largely recalcitrant C (Figure 3.5 and Table 3.3). The inherent infertility of outfields caused the soils to reach a new low equilibrium mainly in the control treatments whereas application of N fertiliser on outfields indicated a state close to equilibrium level. The added fertiliser can facilitate rapid decomposition of available SOM which is in turn taken up by the crops. There is always a limit to sequestration potential of a soil, such that once a particular amount of C has been sequestered, the soils have limited capacity to serve as C sinks. It is for this reason that degraded soils and ecosystems are thought to have the highest potential for C sequestration (Lal, 2004). Similarly, Powlson *et al*. (2012) analysed data from long term experiments at Rothamsted research in UK and found that soils under control treatment remained stable over the experimental period. They attributed this to the attainment of a new low equilibrium which is maintained by the cropping regimes. Soils with the same properties could exhibit different levels of maximum C sequestration potential even under similar management practices. In such cases, the capacity for soil C sequestration, varies considerably among sites on same soil type due to initial levels of labile SOC and the ability of management practices to stabilise greater amounts of organic inputs (Mann, 1986). In support of this, Chan *et al*. (2008) showed that soil C stocks may increase over periods of

50-100 years only until the equilibrium level is achieved and where management practices are not altered but rather improved. Even under such conditions, there is a finite limit to C sequestration potential beyond which the soils are less able to function as C sinks.

The MaHF is based on sorption of organic C to mineral surfaces and clayey soils had more MaHF than LF. The greater MaHF was due to the protection of MaHF caused by stronger sorption and reduced desorption of organic C to the mineral surfaces (Kögel-Knabner, *et al.,* 2008). In this case, the MaHF becomes less bioavailable as the fLF becomes a more available energy/nutrient source.

6.4 C:N ratios in different land management practices

Carbon to nitrogen ratio is an important indicator of soil quality. Results showed sandy soils having higher C:N ratio than clayey soil (Chapter 2) under different tillage treatments. Fertility treatments using inorganic fertiliser and manure showed a different trend with clayey soils having higher C:N ratios than sandy soils. A similar trends was reported by Zingore *et al,*. (2007) while reporting on the initial soil conditions of the Murewa study area. Other studies have reported higher C:N ratios in sandy soils than loamy soils and clayey soils. Highest C:N ratios were found in forest floor L layer (Table 4.1). The C:N ratios of the top 10 cm was higher than the lower depths in conventionally tilled treatments than reduced and no tillage treatments. Tillage treatments had significant short term effects on C:N ratios whereas the nine years of cultivation under different fertility treatments did not cause any significant differences in C:N ratios within each soil type. The differences in C:N ratios by depth were not evident in fertility treatments (Chapter 3) which had C:N ratios between 10 and 23.

Density separation studies have shown higher C:N ratio in fLF than oLF and MaHF with differences in C:N of up to seven units in tillage and fertility treatments (Table 2.2 and 4.1). On average, the C:N ratio of fLF was 24 (standard deviation [SD] 4) and that of oLF was 21 (SD 3), indicating higher mean and variance in fLF. The MaHF had consistently lower C:N ratios (mean 14, SD 3). Other studies, on agricultural sites (Poirier *et al.,* 2005), calcareous forest soils (Rovira & Vallejo, 2003), lowland tropical forest and pastures Marin-Spiotta *et al.,*

(2008) showed a distinct decline in C:N from fLF to oLF, in accord with current models.

The C:N ratios were however higher for the fLF under N fertiliser which could be a result of increased N in microbial biomass. Higher C:N ratios in the fLF are possibly a result of existence of higher amounts of carbohydrates (Dalal & Henry, 1988; Golchin *et al*., 1994a) mainly in form of unprocessed plant residues and sugars (Oades, 1972). The results support current models showing declining C:N ratio from fLF to oLF then MaHF as organic C is oxidised by heterotrophic microbes during decomposition, whilst N is somewhat preserved in the microbial biomass because of microbial N demand (their biomass has a much lower C:N ratio than plant biomass). The C:N ratio of oLF is intermediate between that of MaHF and that of fLF. Results support that MaHF is dominated by microbially processed compounds and/or N-rich compounds, such as peptides, with little inclusion of high C:N plant detritus (Chan *et al*., 2008).

Pine plantation soils tended to have higher C:N ratios in fLF than agricultural soils and natural forests, perhaps because the conifers contain source materials with wider ranges of chemical compositions than agricultural soils (e.g. (Kölbl & Kögel -Knabner, 2004). In addition to the preferential degradation of C-rich, labile substrates, selective preservation of high C:N compounds seems necessary to account for the 10–20 unit higher C:N ratios in fLF in these soils.

6.5 Potential of plantation forests to mitigate climate change

Agriculture and overgrazing cause major losses of woody vegetation cover (IPCC, 2007) as people expand cropping areas to meet the demand for food in addition to their need for wood energy. As natural forest shrink, there is need to conserve the remaining forest areas and focus on fast growing plantations species to relieve pressure through promotion of private planting of fast growing species mainly on degraded areas under favourable climatic zones. In forest ecosystems, C storage is mainly a result of higher accumulation of biomass in woody tissue corresponding to above ground biomass production which is faster than growth and turnover rates of the belowground biomass. Atmospheric C retained in plant biomass increases the carbon reservoir of vegetation (Schulze, 2006).

Synthesis

Forests and woodlands provide essential materials for local consumption (wood fuel being the primary energy source), trade and export. In addition, they provide environmental and cultural services. Forests and woodlands are often important sacred and burial sites in Africa. The use of wood fuel is likely to continue in both rural and urban areas as it remains the most reliable, affordable and accessible source of energy for poor households. Furthermore, the frequent power cuts make the use of woodfuel unavoidable. If the value of above and below ground C sequestration is added to the above values, the value of forests increases and can adequately support the livelihoods of forest dependent communities. The benefits of C sequestration in forests is well documented and their role in biodiversity conservation is an added advantage. Therefore, investing in forest plantations and/or conservation can allow communities to benefit from carbon trading. In forest ecosystems, C is stored in biomass, forest floor and soils.

The forest floor layer is mainly composed of woody tissue (e.g. twigs, bark) leaves, flowers, fruits, mosses, lichens and fungi whereas below-ground inputs are from dead roots and their associated mycorrhizal hyphae and root exudates. Some studies showed that roots and mycorrhiza were also more important elements for the accumulation of below ground SOM than above-ground leaf litter. Exotic monocultures (including pine forests) exhibit superior ability in sequestering C above and below ground (Shan *et al.*, 2001). The build- up of SOM in forested ecosystems depends on the amount of litter, its chemical composition, the rates and mechanisms of decomposition, and climatic factors (Berg *et al.*, 1995). The fate of all forest floor material is eventually to become below-ground SOM input and this can be facilitated by bioturbation and cultivation.

Although the living fine root biomass constitutes only a small fraction of the total stand biomass, the contribution of fine roots to total stand biomass, and total soil C inputs can be substantial. When the same type of vegetation grows on the same site for a long time, an equilibrium state is attained through litter input and its associated rate of decomposition. Ultimately, an organic layer that is typical of a particular site is eventually formed (Evans *et al.*, 2001).

Wild fires are a threat to forests and woodlands, causing enormous destruction to both flora and fauna. However, fire plays an important role in determining the distribution and composition of some vegetation types and is responsible for the extensive occurrence of grasslands in Southern Africa. In tropical savannahs (woodlands, bush lands and grasslands) fire is a common tool used to convert forestland to agriculture and forest plantations. During the burning, C is lost from the forest floor, whereas losses from the mineral soil layers depends on fire intensity, frequency and thickness of litter layers. In all cases, fire can become a source of CO_2 emission to the atmosphere and alter net assimilation rates of the standing vegetation causing a decrease in the supply of organic matter and root-derived substrates to the soil (Duguy *et al.*, 2007). Not only is CO_2 lost but other soil nutrients are also either volatilised or transformed into ashes resulting in a net loss of nutrients through wind blow, erosion or leaching. In other natural systems, fires of low intensity are used to consume the under storey and part of the forest floor layers to prevent fuel build-up (Ferran *et al.*, 2005). The fire events consume all forest floor material the severity of which depends on the intensity of the fire. High intensity fires can extend deeper into the mineral soil layers (Duguy *et al*., 2007) causing some alteration of soil physico-chemical properties (Neary *et al.*, 1999) and affecting soil nutrient fluxes and consequently C stocks.

During and after a fire, in the short term soil fertility usually increases due to higher nitrogen and phosphorous availability in the soil solution. Increased pH and concentration of base cations temporarily enhance soil respiration causing higher nitrogen mineralization and nitrification. The processes cause some areas as in the one year old stand not to show drastic decline in soil C and N stocks (Chapter 4). The magnitude of the nutrient flush depends on the temperature and duration of the fire and the amount of organic matter burned.

In the 0–60 cm depth, the fLF, oLF and MaHF C and N were significantly lower in the miombo woodland and pine stands than in the moist forest. Generally, LF, oLF and MaHF C and N stocks declined consistently with increasing soil depth. Significant differences in LF C, oLF C and MaHF C were between the moist forest and the one year old pine stand at 0–20 cm depth, especially the top 10 cm, whereas there was less change below 30 cm, indicating that labile fraction losses due to forest transition mainly occurred in the surface soils. This suggests that conversion of miombo woodlands and moist forest to pine plantations significantly reduces fLF C, which may be attributed to a combination of factors including quantity of litter materials, microbial activity and management disturbances, which would change greatly with the forest conversion.

Results suggest that net C storage is likely to occur in forested land and the ultimate fate of the wood products determines the final assessment of C sequestration potential. For example, if the rotation age is extended or if the forest is not harvested, the C is sequestered for longer periods of time. Sequestration can also be long lived if trees are processed to sawn timber and used in construction of houses and these can last for decades. On the other hand, sequestration is short lived if trees are burned for firewood, or processed into paper. Tree residues can also be recycled into organic amendments for use in cropping systems.

6.6 Modelling soil C stocks in Zimbabwean agro- ecosystems

Future impact of land management practices can effectively be predicted by the use of observations in combination with models providing a means of evaluating the changing practices in the future. In the RothC soil C model decomposition processes are represented in three arbitrary SOM pools, which only vaguely relate to measurable SOC fraction, with specific turnover rates and an additional clay decomposition modification factor. Simulation modelling using the RothC model, showed good agreement between the simulated and observed SOC for 2010 in all treatments on sandy soils except the natural forest where the C was underestimated by 30%. On clayey soils, fertility treatments showed better agreement with small deviations than tillage treatments and natural forest. Highest measured SOC was found in natural forests on clayey soils whilst lowest was in control treatments of both tillage and fertility treatments. There may be many weaknesses and limitations of SOM models, since most were parameterised under particular management or climatic regions. Ideally SOM is affected by complex interactions which models do not account for such as parent material, pH, time, litter quality and biota management. Despite this shortfall, the modelling approach gives a guideline of C dynamics under a particular management system. The failure of RothC to account for some of these factors may affect the accuracy of predictions. Furthermore, the theoretical compartments/pools of RothC model are difficult to match with measurable pools making it difficult to initialize the models and validate model-calculated results for the individual pools (Kutsch *et al.*, 2009). Despite this disparity, there was good relationship between measured mineral associated heavy fraction and modelled HUM + IOM whereas the LF had a weak relationship with RPM.

Under dry land conditions on the sandy soils of West-Africa, simulation model predictions using the CENTURY and RothC models suggested that conversion to no-tillage will result in small increases in soil C contents (0.1–0.2 t ha^{-1} $year^{-1}$) (Farage *et al*., 2007). Only practices that entailed an increased input of organic matter, for example through agroforestry or manure, were predicted to result in large increases in soil C. Although it is often difficult to separate the effects, increases in modelled SOC are probably due to increased biomass production and retention in conservation tillage systems (RP and DS) rather than reduced or no-tillage supporting the arguments of Corbeels *et al*. (2006).

6.7 Conclusion

Any technology or practice that increases the photosynthetic input of carbon and/or slows the return of stored carbon to CO_2 through respiration, erosion or fire has potential to store carbon and can be a potential C sink. Significant amounts of SOC can be stored in this way, through a range of practices, depending on local environmental conditions. Substantial amounts of vegetative C can also be stored in perennial plantings including agro-forestry systems. The use of conservation tillage, and use of manure and nitrogen fertiliser contribute to improving soil C status in cropping systems although the benefits of N fertiliser can offset by higher N_2O emissions from soils and CO_2 from production of fertiliser. Results of this study indicate that C storage varies with tillage and fertility amendments ranging from 15.3 to 32.0 and 4.0 -14.6 Mg ha^{-1} on clayey and sandy soil respectively. The difference in SOC content between native forests and croplands (50%) on sandy soils was less than the difference between native forests and croplands in clayey soils (16%). Although sandy soils showed a weaker greenhouse gas mitigation potential (shown by low potential of C storage) when compared with clayey soils, results suggest that the no tillage system (DS) is the best management for C storage on sandy soil whilst the conventional tillage (CT) system is the worst. When conventional tillage is the only available option, the application of nitrogen fertiliser (Chapter 2) is more beneficial for increasing C and N stock. Significant differences in C and N storage in plantation stands were shown between the different age classes and natural forest stands at 0−10 cm and 10−30 cm. However, when sampling deeper in the profile (below 30 cm), differences were not evident because of the increasing magnitude of random variation associated with a greater soil depth.

Separation of SOM into discrete fractions has been successfully used to isolate changes in the structure and function of SOM pools in response to land management activities. The application of density fractionation has been limited in tropical soils of Africa. Three distinct SOM pools (free light fraction (fLF), occluded light fraction (oLF) and mineral associated heavy fraction (MaHF)) were isolated by density fractionation procedure for sandy and clayey soils in smallholder farming systems and on a *Pinus patula* age series. In all treatments the stocks of C and N were in the order oLF<fLF<MaHF. Each fractional component was higher under RP and DS than CT and higher under N Fert + manure than N fertiliser and control on clayey soil except oLF which was higher under control than manure. On sandy soils the three fractions were higher under DS than RP and CT and higher under N fertiliser than N Fert + manure and control.

The fLF contributed less to total soil mass, has a high C concentration and wider C:N ratios relative to the oLF and MaHF, a characteristics consistent with general trends in other findings. In conceptual models of SOM pools, the MaHF is assumed to be a more stable pool. Results show that, in cropping systems, planted forests and natural forests, soils can be isolated into distinct SOM fractions that are distinguished by their contributions to total soil mass, C and N storage. The fLF is of recent origin, while the MaHF is a more recalcitrant pool of SOM and the oLF is intermediate. Relative to the fLF, the oLF was the smallest fraction with respect to total soil mass and having a higher C:N ratios than MaHF but also having high C and N concentrations. This shows the extent of physical protection within aggregates and suggests that stability is a function of the position of SOC within the soil matrix. If well protected, the oLF is less altered by microbial processing when compared to the fLF. We can infer from these patterns that mostly, the MaHF and to a lesser degree, the fLF and oLF all drive soil properties in the upper soil layers because of their contribution to soil mass and C concentrations, whereas at lower depths, the bulk soil was most similar to the dominating MaHF.

The vertical distribution of fLF and oLF showed similar depth trends in soil mass, C concentration, C content, and C:N ratios. The C and N concentration and contribution to total soil mass decreased with increasing soil depth whereas there was no general depth trend in the C:N ratio. Distribution of fLF and oLF was consistent with their conceptual and functional roles of being relatively labile pools of SOM that are free and occluded within the soil matrix and that are poorly decomposed. While the fLF and oLF are critically important in the context of organic matter cycling and site productivity, their contribution to total C and N storage is small (average 6%). These pools are also much easier to manipulate using selected management practices than the MaHF. It is much easier to alter the rate and nature of inputs and controls on outputs in the upper soil layers than manipulating deeper soil layers. Cases where there is less SOC and TON in light fractions, suggest that the light fraction inputs were transformed rapidly into MaHF leading to more physically protected C and N in MaHF. Although fLF and oLF are often more sensitive to land management practices than TOC, in some soils there may be no significant difference between MaHF due to accruals in MaHF (Jastrow, 1996) resulting in rapid response of MaHF to management.

In the 1, 10 and 25 year old plantation stands, the C:N ratio decreased with depth, while in the 20 and 30 year old stands, the C:N ratio decreased in both the free and occluded light fractions. Factors such as the preferential leaching and desorption of C from deep soil along with other processes may also play a role and account for some of the variability in C and N over the age sequence.

Differences in the deeper mineral soil layers, in particular the 30–50 cm depth interval, are mainly driven by the MaHF fraction. The MaHF accounted for over 98% of the soil mass at 0-30 cm and over 85% of SOC below 30 cm. This shows that the contribution of the MaHF to total C and N storage also occurs below 20 cm and yet depths beyond 20 cm are often overlooked in most studies examining changes in C and N storage in response to land management.

There is also a need to obtain more data on long-term effects of different tillage systems on C and N dynamics for various agro activities in order to gain a full understanding of the C and N cycling. Understanding soil quality can help in the design of crop and soil systems for agricultural sustainability and should focus on further investigation under a more holistic approach.

Charcoal and wood fuel use, logging, poor agricultural and land use practices will ultimately, continue to threaten forests unless alternative energy sources are utilised, alternative livelihoods are sought, and sustainable agricultural methods are employed on farmlands. Urban and peri-urban energy demand increases fuelwood and charcoal prices and demand eventually leading to deforestation unless alternative energy supply is provided.

Finally, the discussions in his thesis suggest that cultivation, tree planting and forest conservation for carbon sequestration can become profitable if harmonised with other services for soil conservation, water quality, wildlife habitat, biodiversity and the environment. Although a hectare of forest may sequester more C than a hectare of crop land, tree plantations cannot be markedly expanded on cropland or woodlands without generating some costs to society. Opportunities for expanding carbon sequestration in tree plantations exists in marginal lands where rainfall is adequate. In Africa, REDD+ pilot projects have demonstrated that climate change mitigation through forest carbon payments can enhance the incomes of the rural poor, as well as increasing opportunities for adaptation and growth. Generally, payments for REDD+ and other ecosystem services have great potential in light of the diversity of schemes that are likely to emerge and the diversity of services likely to be obtained including potential positive impact on the environment. REDD+ has ability to save public and private sector funds by promoting a diversity of benefits, improving people's livelihoods and having potential to reduce conflicts. Since C sequestration is best viewed as part of environmental benefits or costs, activities normally accrue to society as externalities. The focus on the monetary valuation and payment for environmental services can contribute to the attraction of political support for soil conservation. Developing countries therefore, need to formulate enabling economic and institutional land management policies that have positive impacts on poverty alleviation, food security and environmental sustainability. The current initiatives for GHG mitigation show the need for synergy between sustainable development, climate change and environmental integrity as a means for realisation of the mitigation potential of the agricultural sector.

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SUMMARY

Climate change adversely affects human livelihoods and the environment through alteration of temperatures, rainfall patterns, sea level rise and ecosystem productivity. Developing countries are more vulnerable to climate change because they directly depend on agriculture and natural ecosystem products for their livelihoods. Mitigation of climate change impacts includes practices that can store carbon (C) in soil and biomass thus, reducing concentrations of atmospheric carbon dioxide (CO_2) and other greenhouse gasses. In addition, planted and natural forests that store large amounts of C, can become key resources for mitigating and reducing vulnerability to climate change, whilst infertile agricultural soils require large amounts of chemical and/or organic fertilisers to improve productivity. Increasing awareness about climate change mitigation has led to realisation of a need for sustainable land management practices and promoting soil C sequestration to reduce the greenhouse effects.

The C storage potential of agricultural soils is compounded by conventional tillage practices, covering large areas with only small portions of fields dedicated to conservation farming practices. Maintaining soil and crop productivity under these agricultural systems becomes a major challenge especially in rain-fed arid and semi-arid regions, characterised by long annual dry spells. Conservation tillage practices, such as no-till and reduced tillage, have been reported to increase soil organic carbon (SOC) stocks in agricultural systems as they reduce soil disturbance, whereas conventional tillage has been criticised for causing soil C losses, accelerating soil erosion and displacing of soil nutrients, despite benefits, such as reduced soil compaction, weed control and preparation of favourable seedbed, which have been reported under conventional tillage. The identification of appropriate agricultural management practices is critical for realisation of the benefits of Soil C sequestration and reducing emissions from agricultural activities.

This thesis was planned to improve our understanding on how tillage, fertilisation, tree planting or natural forest conservation can enhance C

Summary

sequestration and thus mitigate climate change. The main goal was to quantify the influence of tillage, fertilisation and plantation forestry practices on C and N dynamics in bulk soil and density separated soil organic matter (SOM) fractions relative natural forest. Tillage treatments under reduced tillage (RP), no tillage (DS) and conventional tillage (CT) were compared with natural forests (NF) in sandy Haplic Arenosols and clayey Rhodic Ferralsols. Impacts of fertilisation were assessed from three fertility treatments; unfertilised control (control), nitrogen fertiliser (N Fert) and nitrogen fertiliser plus cattle manure (N Fert + manure) in conventionally tilled fields on Arenosols (sandy soil) and Luvisols (clayey soil) along two soil fertility gradients. Similarly, C and N storage in tree farming was studied using a Pinus patula chronosequence. Soil sampling followed randomised complete block design with four replications in agricultural systems and two replicates in each plantation age stands and natural forest. Sodium polytungstate (density 1.6 g cm^{-3}) was used to isolate organic matter into free light fraction (fLF), occluded light fraction (oLF) and mineral associated heavy fraction (MaHF). Carbon an N were analysed by dry combustion and C and N stocks calculated using bulk density, depth and C and N concentration. The RothC model was used to match density separated fractions with conceptual model pools for agricultural and natural forest soils.

Findings from tillage studies showed significantly larger C and N stocks in natural forests than tillage systems despite the open access use of the natural forests. The C and N stocks were significantly lower in sandy than clayey soils. At 0–10 cm depth, SOC stocks increased under CT, RP and DS by 0.10, 0.24, 0.36 Mg $ha^{-1}yr^{-1}$ and 0.76, 0.54, 0.10 Mg $ha^{-1}yr^{-1}$ on sandy and clayey soils respectively over a four year period while N stocks decreased by 0.55, 0.40, 0.56 Mg $ha^{-1}yr^{-1}$ and 0.63, 0.65, 0.55 Mg $ha^{-1}yr^{-1}$ respectively. Under prevailing climatic and management conditions, improvement of residue retention could be a major factor that can distinguish the potential of different management practices for C sequestration.

Among the fertility treatments, there were significantly higher SOC and TON stocks under N Fert and N Fert + manure at 0-10 cm soil depth in Luvisols. Although this effect was not significant at 20-30 cm and 30-50 cm depth. On Arenosols, N Fert had highest C and N at all depths except at 0-10 cm. The storage

of C and N on Luvisols, followed: control < N Fert < N Fert + manure whereas Arenosols had control < N Fert + manure < N Fert. Compared with control, N Fert and N Fert + manure enhanced fLF C on homefields and outfields by 19%, 24% and 9%, 22% on Luvisols and 17%, 26% and 26%, 26% respectively on Arenosols. Homefields on Luvisols, under N Fert and N Fert + manure had similar equilibrium levels, which were 2.5 times more than control.

Forests play a major role in regulating the rate of increase of global atmospheric CO_2 storing C in soil and biomass although the C storage potential varies with forest type and plant species composition. In this research, storage of C and N were highest in moist forest and lowest in the Miombo woodland. In both natural and planted forests, above ground tree biomass was the major ecosystem C pool followed by forest floor's humus (H) layer. The mineral soil had 45%, 31% and 24% of SOC stored at the 0–10, 10–30 and 30–60 cm soil depths respectively. Stand age affected C and N storage significantly having an initial decline after establishment recovering rapidly up to 10 years, after which it declined and increased again by 25 years. Average soil C among the *Pinus* compartments was 12 kg m^{-2}, being highest at 10 years and lowest in the 1 year old stands. Organic N was also highest at 10 years and least at 25 years. The proportional mass of fLF and oLF in Miombo woodlands was similar while the other stands had higher fLF than oLF. The highest LF was in the moist forest. In the *Pinus patula* stands the fLF C contributed between 22−25%, the oLF C contributed 8−16% and MaHF C contributed between 60−70% to total SOC. Carbon in MaHF and oLF increased with depth while the fLF decreased with depth in all except the 1 and the 10 year old stands. Conversion of depleted Miombo woodlands to pine plantations can yield better C gains in the short and long run whilst moist forests provide both carbon and biodiversity. Where possible moist forests should be conserved and enrichment planting done in degraded areas to sustain them and if possible the forests can be considered as part of future projects on reduced emission from deforestation and degradation (REDD+). It is believed that REDD+ can promote both conservation and socio economic welfare, including poverty alleviation by bringing together the development of the forest and climate change link in African forests and woodlands. The focus on the monetary valuation and payment for environmental services can contribute to the attraction of political support for soil

conservation. Developing countries therefore, need to formulate enabling economic and institutional land management policies that have positive impacts on poverty alleviation, food security and environmental sustainability.

Soil C models are used to predict impacts of land management on C storage. The RothC 2.63 model was used for estimating SOC stock under selected land management practices on the clayey and sandy soils of Zimbabwe. There is greater potential to store more C in clayey soils than sandy soils and in practices that receive more organic inputs. Results show that the RothC model pool of HUM + IOM is related to the measured MaHF from density fractionation and that the model can be used to estimate SOC stock changes on Zimbabwean agricultural and forest soils. The relationship between equilibrium levels estimated by the RothC model and those estimated using the Langmuir equation was good. A 1.5˚ C rise in temperature was found to cause the A and B systems on clayey soils to sequester more C. The results also show that, when holding all the other factors constant, the model is sufficiently sensitive to a rise in temperatures with sandy soils reaching an equilibrium much earlier than clayey soils. The modelling approach represents one of the most promising methods for the estimation of SOC stock changes and allowed us to evaluate the changes in SOC in the past period on the basis of measured data. However, since the data were obtained from short term experiments (4−9 years), further ground validation can be hampered by the lack of long-term experimental trials in the southern African region. The deficiency of adequate experimental sites also limits further work on model uncertainties. The understanding soil quality and dynamics however, helps to design sustainable agricultural systems, while achieving the urgently needed win-win situation in enhancing productivity and sequestering C.

SAMENVATTING

Klimaatverandering heeft een negatief effect op het welzijn van mensen en op het milieu door veranderingen van temperatuur, neerslagpatronen, zeespiegelstijging en ecosysteem productiviteit. Ontwikkelingslanden zijn kwetsbaarder voor klimaatverandering omdat ze meer afhankelijk zijn van landbouw en natuurlijke ecosysteem producten voor levensonderhoud. Tegengaan van de gevolgen van klimaatverandering omvatten maatregelen zoals het opslaan van koolstof (C) in biomassa en bodem waarmee de concentraties van atmosferische CO_2 en andere broeikasgassen verlaagd worden. Bovendien, plantages en bossen die grote hoeveelheden C opslaan kunnen belangrijke mogelijkheden bieden om de kwetsbaarheid door klimaatverandering te verminderen. Daarnaast vereist onvruchtbare landbouwgrond grote hoeveelheden anorganische of organische meststoffen om de productiviteit te verbeteren. Een toenemende bewustwording m.b.t. klimaatverandering heeft geleid tot de realisatie dat er behoefte is aan duurzame land managementpraktijken en de bevordering van opslag van C in de bodem om de effecten van broeikasgassen te verminderen.

Het C-opslag potentieel van landbouwgrond is afgenomen door conventionele grondbewerkingsmethoden die betrekking hebben op grote gebieden. Alleen op kleine arealen worden duurzame landbouwmethoden toegepast. Behoud van bodem en gewas productiviteit onder conventionele landbouwsystemen wordt een grote uitdaging met name in de regen-gevoede droge gebieden die gekenmerkt worden door lange jaarlijkse droge perioden. Duurzame grondbewerkingsmethoden, zoals “zonder-ploegen” en gereduceerde bodembewerking kunnen de bodem organische koolstof (SOC) voorraden verhogen in omdat ze de verstoring van de bodem verminderen. Conventionele grondbewerking wordt bekritiseerd door het veroorzaken van bodem C verliezen, het versnellen van de bodemerosie en het verlies van nutriënten ondanks de voordelen zoals verminderde bodemverdichting, onkruidbestrijding en toename van nutriëntbeschikbaarheid aan het begin van het groeiseizoen. De vaststelling van goede landbouwmethoden is cruciaal voor de verwezenlijking van de voordelen van de bodem C opslag en daarmee de vermindering van emissies als gevolg van landbouwactiviteiten.

Dit proefschrift heeft als doel om onze kennis te verbeteren m.b.t. hoe grondbewerking, bemesting, het planten van bomen en bosmanagement kunnen bijdragen aan de verbetering van C opslag in de bodem. Het voornaamste doel was het kwantificeren van de invloed van grondbewerking, bemesting en plantage bosbouwpraktijken op C en N dynamiek in de bodem en op afzonderlijke bodem organische stof fracties. Verschillende grondbewerkingsmethoden, zoals “verminderde grondbewerking” (RP), geen grondbewerking (DS) en conventionele grondbewerking (CT) werden vergeleken met natuurlijke bossen (NF) in Haplic Arenosols en Rhodic Ferralsols. Effecten van bemesting werden beoordeeld door middel van drie behandelingen; onbemest (controle), stikstof toediening (N Fert) en stikstof toediening plus organische (vee) mest (N Fert + mest) in conventioneel bewerkte velden op Arenosols (zandige grond) en Luvisols (kleiachtige bodem) langs twee bodemvruchtbaarheidsgradiënten. Koolstof en N opslag in plantages werd ook bestudeerd met behulp van *Pinus patula* opstanden met oplopende ouderdom. Grondmonstername was gebaseerd op een gerandomiseerde volledig blokontwerp met vier herhalingen in de landbouwsystemen en twee herhalingen per plantage opstandleeftijd en in de oorspronkelijke bossen. Natrium polytungstate (dichtheid 1.6 g cm^{-3}) werd gebruikt voor het isoleren van bodem organisch materiaal in de vrije lichte fractie (fLF), beschermde lichte fractie (oLF) en de mineraalgebonden fractie (MaHF). Koolstof en N werden geanalyseerd door droge verbranding in een “CN analyser” en C en N voorraden berekend aan de hand van bodemdichtheid, diepte en C en N concentratie. Het RothC model werd gebruikt om gemeten bodem organische stoffracties te vergelijken met conceptuele bodemkoolstofvoorraden in landbouw- en bosbodems.

Bevindingen uit grondbewerkingsstudies lieten aanzienlijk grotere C en N voorraden zien in natuurlijke bossen dan in landbouwsystemen ondanks de open toegang en het gebruik van natuurlijke bossen. De C en N voorraden waren beduidend lager in zandige dan in kleiige bodem. Op 0–10 cm (diepte), SOC voorraden stegen onder CT, RP en DS met 0.10, 0.24, 0,36 Mg $ha^{-1}yr^{-1}$ en 0.76, 0.54, 0,10 Mg $ha^{-1}yr^{-1}$ op zandige en kleiige bodems respectievelijk over een periode van vier jaar terwijl N voorraden respectievelijk daalden met 0,55, 0.40, 0.56 Mg $ha^{-1}yr^{-1}$ 0.63 en 0,65, 0.55 Mg $ha^{-1}yr^{-1}$. Onder het huidige klimaat en bodembeheer zou verbetering van het behoud van gewasresten een belangrijke factor zijn die het potentieel van verschillende C-opslagmethoden zou kunnen verbeteren.

Bemestingsproeven liet aanzienlijk hogere SOC en TON voorraden zien onder N Fert en N Fert + mest op 0-10 cm (diepte) in Luvisols. Hoewel dit effect niet significant was op 20-30 cm en 30-50 cm (diepte). In Arenosols, N Fert gaf de hoogste C en N waarden op alle diepten behalve op 0-10 cm. De opslag van C en N in Luvisols volgde van laag naar hoog: control < N Fert < N Fert + mest. Terwijl in Arenosols de volgorde was: controle < N Fert + mest < N Fert. In vergelijking met controle, N Fert en N Fert + mest vergrootte fLF C op homefields en outfields met 19%, 24% en 9%, 22% op Luvisols en 17%, 26% en 26%, 26% respectievelijk op Arenosols. Homefields op Luvisols, onder N Fert en N Fert + mest hadden vergelijkbare FLF niveaus die 2,5 keer groter waren dan in de controle velden.

Bossen spelen een belangrijke rol bij het reguleren van de stijging van mondiale atmosferische CO_2 door het opslaan van C in bodem en biomassa, hoewel het C-opslag potentieel per bos type varieert. In dit onderzoek, opslag van C en N waren het hoogst in vochtige bossen en het laagste in het Miombo-bos. In zowel natuurlijke als aangeplante bossen, de bovengrondse biomassa bevatte de grootste C voorraad gevolgd door de humus (H) laag van de bosbodem. De minerale bodem had 45%, 31% en 24% SOC opgeslagen op 0–10, 10–30 en 30–60 cm bodemdiepte. Bosopstand leeftijd beïnvloed aanzienlijk. Na een aanvankelijke daling na initiatie volgde een toename van C en N opslag rond de leeftijd van 10 jaar met een verdere stijging tot 25 jaar. Gemiddelde bodem C onder de *Pinus* percelen was 12 kg m^{-2} na tien jaar met de laagste C waarden na 1 jaar. Organische N was ook het hoogste na 10 jaar en het minste na 25 jaar. De hoeveelheid C in fLF en oLF was ongeveer gelijk in Miombo-bos, terwijl in de andere bosopstanden meer fLF dan oLF aanwezig was. De hoogste LF was in het vochtige bos. In de *Pinus patula* opstand, fLF bevatte tussen 22−25 %, de oLF 8−16% en MaHF 60−70% tot het totaal SOC. Koolstof in MaHF en oLF steeg met diepte, terwijl de fLF met diepte afnam behalve in de 1- en 10-jarige opstanden. Conversie van verarmd Miombo-bos naar Pinus plantages kan betere C-opslag op de korte en lange termijn opleveren terwijl vochtige bossen zowel meer koolstof opslag en biodiversiteit bieden. Waar mogelijk moeten vochtige bossen worden behouden en aangetaste gebieden zouden deels aangeplant kunnen worden en indien mogelijk zouden de bossen kunnen worden beschouwd als onderdeel van toekomstige projecten die gericht zijn op verminderde uitstoot door het voorkomen van ontbossing en degradatie (REDD+). In het algemeen kan gesteld worden dat REDD+ zowel de kwaliteit van het milieu en sociaal economische welvaart kan bevorderen, met inbegrip van armoedebestrijding, door het samenbrengen van de ontwikkeling van het bosgebieden en het tegengaan van klimaatverandering. Aandacht voor monetaire waardering en de betaling voor ecosysteemdiensten kan bijdragen aan de toename van politieke steun voor bodembescherming. Ontwikkelingslanden moeten daarom economische en institutionele beleidsregels opstellen voor landgebruik met het doel positieve effecten te hebben op armoedebestrijding, voedselzekerheid en ecologische duurzaamheid.

Bodem C modellen worden gebruikt om de gevolgen van veranderend landgebruik op C-opslag in de bodem te voorspellen. Het RothC 2.63 model werd gebruikt voor het schatten van SOC voorraad onder geselecteerde vormen van landgebruik op kleiige en zandige bodems van Zimbabwe. Er is een groter potentieel om C op te slaan in kleiige bodems dan in zandige bodems en onder landgebruik waarmee meer organische stof wordt ingebracht. Resultaten tonen aan dat de in het RothC model gebruikte C voorraden “HUM + IOM” gerelateerd zijn aan de gemeten MaHF voorraad, en dat het model gebruikt kan worden om veranderingen in de SOC voorraad in Zimbabwaanse landbouw- en bosbodems te schatten. De relaties tussen evenwichtsniveaus geschat door het RothC-model en die geschat met behulp van de vergelijking Langmuir waren goed. Een gesimuleerde stijging van de temperatuur met 1.5˚ C veroorzaakte in systemen op kleiige bodems een toename van C-opslag. De resultaten toonden ook aan dat wanneer alle andere factoren constant gehouden worden, het model voldoende gevoelig is t.a.v. een stijging van de temperatuur waarbij met zandige bodems een nieuw evenwicht eerder bereikt wordt dan met kleiige bodems. Het gebruik van modellen vertegenwoordigt een van de meest veelbelovende methoden voor het schatten van veranderingen in SOC en geeft de mogelijkheid om veranderingen in SOC voorraad te evalueren op basis van gemeten data. Echter, aangezien de bodem C data werden verkregen uit korte termijn experimenten (4−9 jaar) is lagere termijn voorspelling beperkt door het ontbreken van lange termijn meetreeksen in de regio zuidelijk Afrika. Ook het tekort aan geschikte experimentele proefvelden beperkt verdere werkzaamheden m.b.t. modelonzekerheden. Meer begrip van de kwaliteit en dynamiek van de bodem is van belang bij het ontwerpen van duurzame landbouwsystemen waarbij zowel het verbeteren van productiviteit en het vastleggen van bodem C een "win-win" situatie oplevert.

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Lizzie Mujuru (nee Choruma) was born on 13 May 1965 in Wedza, Zimbabwe. She did her primary education at Mkondwa primary and Chemhanza primary schools in Wedza. She proceeded to secondary education at Sandringham high school in Norton, Zimbabwe. In 1986 she became one of the first two women to receive a diploma in forestry from the Zimbabwe College of Forestry. Before moving on to university education, Lizzie spent seven years as a forestry extension officer under the Zimbabwean Forestry Commission. In 1996, she received her Bachelor in Forestry degree at Sokoine University of Agriculture in Tanzania funded by a scholarship from DANIDA. Upon returning to Zimbabwe in 1996, she joined the Forestry Extension Services as a provincial technical officer for a few months before moving to lecture at Zimbabwe College of Forestry. In 2003, she received her MSc degree in Tropical Resource Ecology from the University of Zimbabwe with a study focussing on environmental determinants of small scale vegetation structure and composition in a native moist forest in Zimbabwe. In 2005, she joined the Bindura University of Science Education lecturing staff and chaired the Department of Environmental Sciences from 2009 to 2012. In 2008, she joined the Watson International Scholars of the Environment Programme at Brown University, USA and was awarded a certificate in land-use science. The training sessions created a desire to do PhD studies focusing on climate change and land use. She started her PhD research in 2010 under Earth System Science group at Wageningen University funded by the Netherlands fellowship programme. During this PhD period she also received some research funds from the Climate Food and Farming (CLIFF) network and the Research Council of Zimbabwe.

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Some Publications And Conference Papers

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The Netherlands Research School for the
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born on 13 May 1965 in Wedza, Zimbabwe

has successfully fulfilled all requirements of the
Educational Programme of SENSE.

Wageningen, 20 May 2014

the Chairman of the SENSE board
Prof.dr.ir. Huub Rijnaarts

the SENSE Director of Education
Dr. Ad van Dommelen

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- Multivariate Analysis (2010)
- Field Training- Earth System Science (2010)
- Information Literacy and EndNote Introduction (2011)
- Soil Ecology- taking global issues underground (2011)
- REDD+SCIENCE+GOVERNANCE: Opportunities and Challenges (2012)
- Techniques of Scientific Writing (2012)

Oral Presentations

- *Density fraction and its effect on organic carbon and nitrogen in sandy and clayey soils.* Research and Post graduate center at Bindura University. 26 August 2011. Bindura University.
- *Effects of tillage on soil organic carbon and nitrogen storage*. Research and intellectual expo, 6 September 2012, Harare, Zimbabwe.
- *Carbon and nitrogen in agro ecosystems.* Climate food and farming network (CLIFF) 23-27 February 2013, Copenhagen, Denmark.

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