ESTIMATION OF ABOVE AND BELOW GROUND CARBON STOCKS IN SELECTED LANDUSE PATTERNS IN MT. MARSABIT ECOSYSTEM.

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DECLARATION

This thesis is my original work and has not been presented for a degree in any other university.

Signed..

Date ....1/08/2006..........................

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This thesis has been submitted for examination with my approval as the supervisor.

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DEDICATION

This thesis is dedicated to my late father Henry Barrack Ouko, whose inspiration and love for education predicated and ensured that I would travel this road, and come this far.

He would indeed have been proud.

And also to my husband Agola Gregory Opiyo, who never fails to remind me to live as if every other day were my most precious.

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ABSTRACT

Intensified agricultural practices lead to a reduction in ecosystems carbon stocks. This is mainly due to removal of aboveground biomass as harvest with subsequent burning and/or decomposition and loss of soil carbon as carbon dioxide and soil through erosion. The effects of forest conversion and subsequent cultivation on carbon stocks and soil properties were monitored in demarcated land use types along transects in Mt. Marsabit ecosystem. The main objective of this study was to develop a practical understanding of the impact of deforestation and the mitigation measures being put in place on soil properties with emphasis on carbon stocks.

Near infrared reflectance spectroscopy (NIRS) and 12 chemical and physical parameters were studied on soil samples collected from three different land use systems namely forest, cropped and pasture land. Soil properties were calibrated to spectral reflectance using partial least square (PLS) regression. Two hundred and twenty two soil samples had been augured from the 0-20 cm and 20-50 cm depths. Seventy four samples from the reflectance data were chosen for prediction. There were four hundred and twenty seven soil samples obtained using core rings in soil profiles laid in each land use type, to a depth of 150 cm at 5 cm intervals. The soil samples were used to estimate the belowground carbon. The collection of litter was done at 90 points using a 0.5×0.5 m quadrant. The litter was dried and weighed as part of the aboveground carbon. The diameter at breast height (DBH) was measured from 161 trees selected randomly in the forest for use in calculating the standing aboveground carbon.

Total carbon ranging between 0.98 – 8.1 gkg^{-1}, r^2 was 0.97, bias was 0.021 gkg^{-1} with root mean standard error of prediction (RMSE) of 0.78 gkg^{-1}. Predicted and measured values for the other soil properties were: total nitrogen $r^2 = 0.95$, bias was 0.001 gkg^{-1} with RMSE of 0.14 gkg^{-1}; pH $r^2 = 0.95$, bias was 0.0003 $cmolkg^{-1}$ with RMSE of 0.01 $cmolkg^{-1}$; exchangeable magnesium $r^2 = 0.94$, bias was 0.0009 $cmolkg^{-1}$ with RMSE of 0.18 $cmolkg^{-1}$; exchangeable calcium $r^2 = 0.76$ bias was 0.0007 $cmolkg^{-1}$ with RMSE of 0.16 $cmolkg^{-1}$ and CEC $r^2 = 0.76$ bias was 0.0008 $cmolkg^{-1}$ with RMSE of 0.09 $cmolkg^{-1}$. These soil properties were significantly different ($P \leq 0.001$) (appendix 1) in cropped and pasture compared to forest and their variation was successfully predicted ($r^2 > 0.76$) using NIRS. The mean spectral reflectance was significantly different among the three land use systems ($P \leq 0.001$). The total (above and belowground) carbon stocks were significantly different ($P \leq 0.001$) in cropped and pasture compared to forest areas. The carbon stocks declined by 47.4% and 45.6% in cropped and pastureland sites relative to the forest site.

Belowground carbon tended to be almost constant at 100 cm forest, 80 cm in pasture land and 60 cm in cropped land probably because soil carbon contents generally decrease with depth, as organic inputs are primarily deposited on the soil surface or occur in the topsoil where most of the turnover of fine roots occurs. In general, however, decomposition processes were slower down the soil profile and the carbon stocks that existed below the topsoil were better protected from physical disturbance by vegetation roots and they were likely to change more slowly after land use change. Also, the rooting system of the trees in the forest was dense compared to the bushes in the pastureland and the crops in the cropped land use systems.

1.0 INTRODUCTION

Most changes in land use affect the amount of carbon held in vegetation and soil, thereby, either releasing carbon dioxide (a greenhouse gas) to, or removing it from the atmosphere. The greatest fluxes of carbon result from conversion of forests to open lands and vice versa. The model-based estimates of these fluxes are attributed to land-use change and are highly variable (Goodale *et al.*, 2002) largely due to the uncertainties in the areas annually affected by different types of land-use change. Uncertain rates of tropical deforestation, for instance, account for more than half of the range in estimates of the global carbon flux (Houghton, 1994). Three other factors which account for the uncertainties include: (i) the initial stocks of carbon in ecosystems affected by land-use change, (ii) per hectare changes in carbon stocks in response to different types of land-use change, and (iii) legacy effects; the time taken for carbon stocks to equilibrate following a change in land use (Goodale *et al.*, 2002). In regard to the tropics, recent satellite-based estimates of deforestation are lower than previous estimates (Houghton, 1994).

In Africa, forests and rangelands are under threat from human population pressures and systems of land use (FAO, 1990). Projected population growth in the tropics will require as much as 60% increase in food production during the first quarter of the 21st century (Moss, 1993). This implies that the expansion of agricultural production will be necessary by either increasing yield per unit area in existing agricultural land through high input farming systems or taking more land for production through conversion of forests, grasslands and wetlands. The conversion of tropical forest to smallholder cropland is one of the most common patterns of land use change in Africa (Brown, *et al.*, 1996; Moss, 1993). Agricultural expansion in the tropics, through deforestation has major environmental implications with potential causal links to degradation of the natural resource base. At the global level, this invariably leads to loss of biological diversity, raises the specter of climate change, and disrupts hydrological cycles (Young, 1997). At the local scale it leads to soil fertility decline, crop production loses, and decline in water quality from increased sediment loads arising from runoff, and soil erosion being the major effects (Sanchez, 1976; Nair, 1990; Moss, 1993).

Deforestation and tillage in conjunction with crop residue removal induces a lower equilibrium level of soil organic carbon (Woomer, *et al.*, 1994). Total belowground carbon content decline and humus composition changes in close relation with decreasing, soil porosity, infiltration capacity, structural stability and fertility (Sanchez, 1976; Lal *et al.*, 1998; Six *et al.*, 2000). Soil organic carbon loss by overland flow and soil erosion processes is also triggered by canopy removal, destruction of surface root mat and loss of leaf litter (Moss, 1993). Erosion tends to selectively displace soil organic carbon enriched components of the surface soils. This significantly reduces soil organic carbon stocks at a particular location (IPCC, 2000). The manifestations of soil degradation (erosion, surface compaction, loss of chemical nutrients) therefore, derive primarily from progressive loss of soil organic matter (Nair, 1990).

Tropical soils degrade rapidly when converted from natural vegetation to continuous cultivation. The resulting degradation sequences are in large part due to the disruption of the closed nutrient cycle (Nye and Greenland 1960; Moss, 1993; Mc Donagh *et al.*, 2001). The nutrients are stored in the biomass and topsoil through a constant cycle of transfer and biological processes of litter fall, root decomposition and plant uptake. Soil degradation occurs, in rangelands (Smith, *et al.*, 2003) where the depth of the A horizon has been found to decline under grazing due to soil compaction and compression under livestock trampling.

About 80 per cent of the land area in Kenya is arid and semi-arid (ASAL). The development and protection of ASAL from land degradation is highlighted in the five years National Development Plan of 2002-2008. According to the current National Development Plan, the focus is on water resources; improved road access; increased production in livestock and crop sub-sector; improved educational facilities; conservation of biodiversity; reduction of land degradation; institutionalization of effective drought management measures; contingency planning and mitigation; and strengthening of local institutions, including user groups, to manage community-based resources.

In Marsabit district with a very poor infrastructure and a nomadic population, there is an almost exclusive reliance on livestock resource for its wealth which poses problems. Mount Marsabit and its environs is a unique ecological system in Eastern Africa with the most developed and extensive upland forest on an extinct volcano within an arid setting. This upland forest has over thousands of years developed a distinct plant association endemic to this area. It is the only source of water for the surrounding desert region. The mountain is home to various rare herbivore species. In addition, the mountain harbors two crater lakes – Paradise and the Elephant Pool. The lakes recharge comes from mist condensate on species of saprophytic moss plants living on indigenous forest trees, and not from rainfall in this arid area (National Development Plan, 2002-2008). Currently, the Marsabit Mountain forest occupies an area of 15 km^2 (About 15,280 ha) but it is disappearing at the rate of 1.6 ha per year. The national closed-canopy forest cover now stands at 2-3% (FHI, 2001). If conservation action is not taken immediately, this valuable and unique ecosystem could be lost forever. This would be an ecological disaster that would spell doom for a large pastoral population in Northern Kenya.

The Marsabit forest ecosystem is threatened by many factors. These includes the increasing demand for firewood and building materials, encroachment by human settlement, farming, overgrazing, fodder and forest fires. Encroachment on the forest is as a result of expansion of the Marsabit town southwards into the forest. There is also increased conversion of the forest areas into farmlands outside as well as within the forest reserve. The main areas of concern currently are Gabra Settlement Scheme, Mura Dambi, Badassa, Songa and Kituruni. Marsabit forest offers good grazing land for livestock that is the main livelihood of the pastoral communities. The forests are overgrazed during the dry periods. Most of the firewood sold in Marsabit town is collected from the forest. In addition, charcoal production along the forest edge is also posing a threat to the integrity of the forest ecosystem. Some of the driving forces of these threats are poverty, competition for pasture, water and land, lack of clear policies on land use and land tenure, and lack of clear boundaries. There are also the issues of institutional responsibilities between the various actors such as the forest department, the Kenya wildlife service and the Marsabit County Council. Lack of a sense of ownership towards the forest and its resources by local communities (KFWG, 2001; AGREF, 2002) is also a problem.

The above occurrences call for immediate action in order to save the forest. The mitigation measures require a shift from mining and harvesting of soil and biomass resources to actively designing and creating agro ecosystems that closely relate to the natural ecosystems in form and function. Natural ecosystems in the tropics maintain their high productivity through above and belowground biological processes that sequester soil organic carbon, maintain internal nutrient cycling and efficient water retention (Mutuo *et al.,* 2003).

The objective of this study was to assess the impact of forest conversion to cultivation and pasture lands on soil productivity (macronutrients N, P, K, and CEC), carbon stocks, and soil physical properties (bulk density, texture). The approach was to compare soil samples from cropped fields, and pasture lands with samples from virgin forests. The differences between these samples were directly ascribed to the difference in the land use systems or conversion.

1.1 Justification

The challenges facing scientists today are losses of carbon stocks from terrestrial ecosystems and increasing concentration of greenhouse gasses in the atmosphere (Mutuo *et al.*, 2003). Population increase and lack of adequate farm lands have aggravated the problem with many people turning to conversion from forest to agriculture for their livelihoods. Marsabit forest ecosystem, for example, is threatened by many factors such as increasing demand for firewood and building materials, encroachment as a result of expansion of the Marsabit town southwards, overgrazing, and forest fires. These activities have led to forest degradation. There is also increased conversion of the forested areas into farmlands outside and within the forest reserve. Widespread deforestation for fuel wood and other domestic uses has accentuated the impact of harsh dry environments (Boahene, 1998).

Conversion of forest land to cultivated and pasture land changes the quantity and quality of organic matter inputs into the soil. The decline in primary production potential of the soil may be caused by aggregate disruption as well as enhanced organic matter mineralization, nutrient removal through biomass exports, and leaching. In addition to changes in fertility with conversion, Lal (1979) reported parallel declines in moisture retention and surface water infiltration rates, both symptoms of soil degradation. There has been limited quantification to substantiate the degree, extent, trends and risks of soil degradation vis-à-vis different land use or land management that are applicable despite the perceived enormity of the soil productivity decline with conversion from forestlands, particularly in sub Saharan Africa.

The conventional assessment methods to determine soil degradation include: plot level, laboratory or experiment base. These methods are expensive, time consuming and very specific. A number of different tests are typically required to provide case-specific diagnoses. The results obtained at one site cannot be replicated at another site with relevant outcomes, owing to high spatial and temporal variation in environmental and decision situations. Large numbers of samples are often required to detect significant changes. This precludes the use of most soil testing procedures for monitoring, evaluation, and impact assessment of land management.

There is no scientific consensus on the definitions of appropriate endpoints or indicators for assessing the impacts of land use systems on soil productivity decline at the landscape or farm level (Shepherd and Walsh, 2000). The assessment of diverse effects of land use and land use change on soil productivity requires the integration of physical, chemical and biological attributes of the soil. The methods that can provide rapid and integrated assessments of soil productivity are urgently needed. These methods should be sensitive to the physical and chemical attributes of soil, as affected by land use and land management factors at the plot and landscape level. Developments in laboratory and field based reflectance spectrometry present a unique capability for rapid, cheap, integrated assessments and routine monitoring of soil productivity status (Janik *et al.*, 1998; Shepherd and Walsh, 2000; Chang *et al.*, 2001; Confalonieri, 2001). However, there has been little focus on the application of reflectance spectrometry in the assessment of different soil functions and their response to fertility management.

Both soil and vegetation act as carbon sinks, reducing the amount of carbon dioxide in the atmosphere (USDA NRCS, 2000). According to Palm (2002) agroforestry has a great potential for carbon sequestration and mitigation of greenhouse gas emissions while increasing food security in degraded areas. Afforestation is an important strategy of reducing such impact of drought. Agroforestry could play a very important role in reducing the atmospheric concentration of CO_2 by (i) storing C in tree biomass and in soil and; (ii) helping protect natural C sinks through the improvement of land productivity and the provision of forest products such as firewood and timber on agricultural land. Expansion of food production and harvest of tree products are often cited among the major drivers for deforestation (Kotto-Same *et al.*, 1997; Mutuo *et al.*, 2003). If these problems can be addressed through agroforestry, then vast C pools which are increasingly under threat can be protected. Despite widespread recognition of the potential of agroforestry for C sequestration and other CO_2 mitigating effects, there are very few quantitative data on specific systems (Palm *et al.*, 2002). Kapkiyai *et al.* (1998) recommended that carbon sequestration among smallholdings in the central Kenyan highlands require that additional mechanisms other than soil carbon storage through biomass of perennial plants be explored. There is need to explore how various soil management practices impact on carbon stocks and sequestration of carbon from soil as well as in soils.

1.2 Objectives

The broad objective of this study was to obtain an understanding of the impact of land use conversions on carbon stocks and related soil properties in Mt. Marsabit forest. This was to be achieved through the following specific objectives;

1. To investigate effect of land-use change on soil chemical attributes
2. To evaluate the use of near infrared spectroscopy for non-destructive characterization and prediction of management sensitive soil properties under different land use systems.
3. To quantify carbon stocks in the forest, cropped and pasture land systems of Mt. Marsabit ecosystem.

2.0. LITERATURE REVIEW

2.1 Land use change and carbon dynamics.

The emergence of life on earth has led to the conversion of carbon dioxide (CO_2) that was in the atmosphere and oceans, into innumerable inorganic and organic compounds on land and in the sea (USDA NRCS, 2000). The development of life on earth in different ecosystems over million of years has established patterns of carbon flow through the global environmental system (USDA NRCS, 2000). Natural exchanges of carbon between the atmosphere, the oceans and terrestrial ecosystems are now being modified by human activities and changing the land use patterns. Human activity for instance, has led to a steady addition of CO_2 to the atmosphere leading to an increase in concentration from 285 parts per million on a volume basis (ppmv) before the industrial revolution of the 19th century to 336 ppmv in 1998 (IPCC, 2000).

The major factors leading to losses in carbon stocks and tropical deforestation are forest conversion and land use change and this contributes as much as 25% of the net annual CO_2 emissions (IPCC, 2000). Little is known of the importance of CO_2 emissions from tropical soils (Albrecht and Kandji, 2003). Land use changes can alter land cover biomass and cause associated adjustment in carbon stocks (Houghton, 1999). Conversion of natural to agricultural ecosystems cause depletion of the soil organic carbon pool by as much as 60% in soils of temperate regions and 75% or more in cultivated soils of the tropics (Lal, 2004). Severe depletion of the soil organic carbon pool degrades soil quality, reduces biomass production, and adversely affects water quality. The depletion may be exacerbated by projected global warming (Batjes and Sombroek, 1997). The contribution of soils to the global greenhouse gas budget is increasing due to intensification of agricultural practices and conversion of natural to agricultural soil ecosystems in developing countries (Fearnside, 1992). Land use change in tropical countries is mediated by slash and burn practices, deforestation of primary or secondary forests, and ploughing of grasslands (Davidson and Ackerman, 1993). All these processes remove the competition for nutrients in soil, change soil structure and increase soil mineralization rates. The available information on the soil carbon pool and fluxes in dry areas, and the historic loss of carbon due to past management is sketchy (Fearnside and Laurance, 2003).

Each soil has a carbon carrying capacity depending on the nature of vegetation, precipitation and temperature (Houghton, 1994). Little is known on the effects of land use change on the composition of stabilized soil carbon pools. It is extremely important to understand the impact of different land use on soil organic matter content and composition. Soils may provide an immediate sink for atmospheric carbon dioxide with proper management (Gregorich *et al.*, 1998). Research has shown that the process of soil carbon depletion with cropped land use can be reversed by utilization of pasture or reforestation (Six *et al.*, 2000). Vegetation degradation has led to historic carbon loss of 4 to 6 Mg C ha^{-1} (Lal *et al.*, 1998). The carbon loss, both in above and belowground biomass, resulting from disappearance of the Marsabit forest would be enormous. About two-thirds of the carbon lost can be resequestered through soil and vegetation restoration. The potential of carbon sequestration through desertification control is considerable (IPCC, 1995). Deforestation has led to denudation of the landscape and exacerbated risks of erosion and desertification. Establishing fuel wood plantations (woodlots) is critical to desertification control and restoration of degraded soils. The Kenya Forest Working Group (KFWG) (2001) recommended the development of a management plan for Mt. Marsabit and its forest that should take into consideration among other things, land use planning for the whole mountain, forest zoning for utilization and preservation, additional/alternative sources of fodder during dry periods, firewood (woodlots, agroforestry) and water outside the forest, and monitoring and correction of sustainable utilization levels.

The equilibrium between carbon inflows and outflows in soil is disturbed by land use change until a new one is eventually reached in the new ecosystem (Guo and Gifford, 2002). The equilibrium dynamics of carbon in different land use options are a function of the plant residues returned to the soil, the litter carbon content, the amount of soluble versus non soluble carbon components, the placement of above ground versus belowground inputs and the degree of soil aggregate disturbance (Post and Kwon, 2000). African soils are typically highly weathered thus contain very low organic matter. The role of these soils to the greenhouse gas budget is unknown and fails to provide accurate emission and sink inventories (Dessai and Schipper, 2003).

Soil organic carbon (SOC) dynamics through its influence on soil chemical and physical properties as well as plant nutrient supply is associated with decline in soil functional capacity following land use change and subsequent cultivation (Paustian *et al.*, 1997; Woomer and Swift, 1993; Young, 1997, Hill and Schutt, 2000). Simultaneously, soil organic carbon is a source and sinks for plant nutrients for example N, P and K. The charge properties of SOC, makes it a site for ion exchange (Young, 1997). Soil organic carbon influences characteristics of soil with regard to soil erodibility, hydraulic conductivity, and aggregate stability (Hill and Schutt, 2000). Soil organic carbon serves as a source of energy for soil biota, influencing biologically mediated processes such as nitrogen fixation, litter decomposition, microbial immobilization and mineralization processes (Woomer and Swift, 1993; Moss, 1993).

Agricultural production potential in low input cropping systems of the tropics is based on the nutrients mineralized from labile SOC. These are typically accumulated under "natural" vegetation cover (van Noordwijk *et al.*, 1997). It has been shown that transition from native forest to continuous cultivation leads to a reduction in ecosystem carbon storage and overall reduction in SOC in the topsoil (Woomer *et al.*, 1994; Moss, 1993; van Noordwijk *et al.*, 1997). Changes in SOC content may be interpreted in terms of modified rates of mobilization, mineralization and decomposition as well as changes in the degree of protection of various SOC pools (Post and Kwon, 2000).

The most obvious effect of tree clearance is the removal of litter input which translates into a dramatic loss of C inputs from in situ cycling. Litter fall in tropical forests can range from 3.6 to 12.6 t ha yr^{1}, providing plant nutrients amounting to 28-224 kg N ha^{-1} yr^{1}, 1-15 kg P ha^{-1} yr^{1} and 6-68 kg K ha^{-1} yr^{1} (Greenland *et al.*, 1992). Different modes of forest clearance result in varying magnitudes of SOC and nutrient depletion. Deforestation by burning and bulldozing removes topsoil and deplete soil surface of litter and SOC. Large declines of C:N ratio are observed after burning, presumably due to high volatilization of N compared to C. In addition to fast mineralization rates, there is possibly NO_3-N leaching losses (Moss, 1993). The total below ground C contents also decline, and humus composition changes in association with decreasing soil porosity, structural stability and fertility (Dudal and Deckers, 1993).

The elevated soil organic carbon loss from logged forest is partly due to canopy removal and partly to disruption of surface litter. Destruction of surface root mat and loss of leaf litter trigger SOC loss by overland flow and soil erosion processes (Moss, 1993). Erosion tends to selectively displace SOC enriched components of the surface soil and hence significantly reduce SOC stocks at a particular location (IPCC, 2000). Erosion processes may also stimulate soil organic matter decomposition rates by dispersing soil aggregates and releasing the physically protected organic matter (Lal *et al.*, 1998).

The rapid decline in soil organic carbon associated with conversion of native forest to cropland is attributed to increased decomposition of crop residues and cultivation effects that decrease levels of physical protection to decomposition due to a certain extent to a lower fraction of insoluble material. Tillage breaks macro-aggregates and exposes organo-mineral surfaces that are otherwise inaccessible to decomposers (Post and Kwon, 2000). Cultivation also changes, temperature, moisture and aeration status of the soil resulting in elevated litter decomposition rates (Elliot and Cambardella, 1991). Tillage increases organic matter oxidation by disrupting aggregates, exposing new surfaces to microbial attack, and by changing the redox conditions at lower depth in the profile (Anderson *et al.*, 1990). In a recent study of SOC under maize in southern Tanzania, McDonagh *et al.* (2001) reported steep decline with only one third of the original levels of SOC remaining after twenty five years of cultivation. This loss was equivalent to 300-350 kg C ha^{-1} yr^{1}.

2.2 Effects of land-use change on the carbon balance

Most changes in land use affect the vegetation and soil of an ecosystem and thus change the amount of carbon held on a unit of land. These changes may be large, for example, with the conversion of forest to cropland or the reforestation of cleared lands. The changes may also be negligible, for example, with the replacement of bison by cattle on natural grasslands (Solomon *et al.*, 2002). Attempts to document the effects of various types of land use change on carbon stocks have led to a number of results. Conversion of forest and pastures for example to continuous cropping systems has consistently led to declines in soil C stocks (van Noordwijk *et al.*, 1997; Guo and Gifford, 2002). The C losses from converting natural forests to logged forests in Brazil, Indonesia and Cameroon ranged from 100 Mg C ha^{-1} to 150 Mg C ha^{-1} (Palm *et al.*, 2000; Hairiah *et al.*, 2001). These data suggested that the largest quantity of C was lost in the vegetation after forest conversion, with little losses from the soil organic matter pool. Conversion to continuous cropping or pasture systems led to loss of almost all aboveground C stocks and about 25 Mg C ha^{-1} from the soil organic matter pool in the surface 20 cm. Soil C stocks declined by 22.5 t C ha^{-1} after 14 years of continuous maize cropping in the Peruvian Amazon (Palm *et al.*, 2000). Conversion of logged forests to tree-based agroforestry systems led to losses of aboveground C ranging from 70 to 140 Mg C ha^{-1} and less than 10 Mg C ha^{-1} from the surface soil organic matter pool in Sumatra and Indonesia (Hairiah *et al.*, 2001)

As a result of the growing interest in carbon accounting, the definition of land-use change might better be expanded to include all direct human effects on terrestrial carbon storage. These are the various forms of forest and agricultural management as well as harvests and land conversion (Eve *et al.*, 2002). However, proper management activities, such as forest thinning, low-impact logging, fertilization, selection of species or varieties, and tilling practices, though they affect carbon stocks, have not always been explicitly considered in analyzing the sources and sinks of carbon from land-use change (Goodale *et al.*, 2002).

2.3 Effects of land-use change on the carbon balance of terrestrial ecosystems

Rates of land-use change of forest converted per year to croplands have received little attention (De Fries *et al.*, 2002). Nevertheless, it is important to recognize that lack of knowledge associated with rates of land-use change contribute more to uncertainty of carbon fluxes than uncertainties in biophysical variables. Uncertain rates of tropical deforestation account for more than half of the total range of flux estimates for the globe (Guo and Gifford, 2002).

The 25–30% loss of organic carbon from the top one meter of soil with cultivation is better known from studies than either the initial or ending stocks (Osbert *et al.*, 2004). Absolute loss of carbon resulting from cultivation is uncertain because of the spatial heterogeneity of initial stooks of soil carbon and the distribution of biomass including their management for soil improvement. As a result of this inherent property of ecosystems, the initial carbon stocks of lands cleared, harvested, or otherwise modified are not generally documented and are thus poorly known (Osbert *et al.*, 2004). Carbon stocks may be well known for areas of a hectare or less where measurements exist which are then extrapolated to larger scales (Goodale *et al.*, 2002) rather than determined. Consequently, despite precise estimates of regional biomass from forest inventories in northern mid-latitude countries, fine-scale spatial variations are less well characterized. In the tropics, the scarcity of forest inventories limits understanding of the spatial distribution of forest biomass (Goodale *et al.*, 2002). A comparison of seven approaches to mapping aboveground biomass in Amazonian forests, showed estimates that varied by more than a factor of two for the total biomass. More importantly, the estimates disagreed as to the distribution of high and low biomass regions (Houghton *et al.*, 2001). The biomass of forests, according to these seven estimates, varied from 25% larger to 32% smaller than the average biomass for the region. The reasons for different estimates of biomass for tropical forests have been discussed (Brown and Lugo, 1984, 1992; Fearnside, 1992; and Fearnside and Laurance, 2003). These reasons include the fact that the absolute loss of carbon resulting from cultivation is uncertain because of the spatial heterogeneity of initial stocks of soil carbon and the distribution of biomass. The magnitude of a carbon sink or source depends on both the initial and final stocks (Houghton *et al.*, 1999). The spatial and temporal heterogeneity of carbon contributes uncertainty to rates of decay and regrowth. The processes that determine the duration it takes a changed system to equilibrate (Moss, 1993).

2.4 Changes in carbon stocks as a result of net changes in cropland area

The largest estimated net flux of carbon from land-use change is from conversion of natural ecosystems to cropland (Goodale *et al.*, 2002). These are among the changes in carbon stocks best documented (Guo and Gifford, 2002). When all the initial vegetation is replaced by crops and its biomass are known, it is possible to calculate the net loss of carbon associated with clearing. Given that forests hold so much more carbon per unit area than grasslands, the loss of carbon associated with cropland expansion depends primarily on whether the lands were claimed from forests or open lands (Guo and Gifford, 2002). The variation in carbon stocks of different crop types is relatively small as long as tree (permanent) crops are differentiated from herbaceous crops (Post and Kwon, 2000; Guo and Gifford, 2002). Some uncertainty results from the lands surrounding and interspersed with croplands (such as, hedgerows, buildings, roads, and others). These uncertainties are small in relation to other factors such as spatial heterogeneity (Post and Kwon, 2000; Murty *et al.*, 2002). Uncertainties also result from estimating the time it takes for the release of carbon to occur. This includes quantities of biomass that is burnt during clearing, the rates of decay of stumps and roots as well as quantities of woody materials that are harvested from the site (wood products) and therefore, not returned to the soil (Guo and Gifford, 2002; Murty *et al.*, 2002). The average soil carbon in the upper meter of soil profile is reduced by 25–30% as a result of cultivation (Mann, 1985; Detwiler, 1986; Schlesinger, 1986; Johnson, 2000).

Another uncertainty with respect to changes in soil carbon stocks relates to amount released to the atmosphere, and those moved to other locations through erosion process, and perhaps buried in anoxic environments and thereby sequestered (Post and Kwon, 2000). Comparison of erosion rates with the amount of organic carbon in freshwater sediments suggests that much of the carbon lost through erosion may accumulate in riverbeds, lakes, and reservoirs (Stallard, 1998; Smith *et al.*, 2001). When cropland is abandoned, carbon reaccumulates in vegetation as the land reverts to the natural ecosystem (Harrison *et al.*, 1995). The greater the biomass of the returning ecosystem, the greater the long-term carbon sink associated with recovery. In the short term, however, the magnitude of the annual sink for a particular parcel of land will vary with rate of recovery. This may be affected by the intensity of previous land use or by biophysical factors such as distance from seed source, herbivory, soil fertility, or climate

(Uhl *et al.*, 1988; Kozlowski, 2002). The rate of recovery of vegetation can also depend on the climate conditions, growing season length and soil type (Johnson *et al.*, 2000). Soil carbon may also reaccumulate after abandoning of cultivation, although the rates in mineral soil are rather modest (Post and Kwon, 2000) compared to the much faster rates of carbon accumulation in vegetation, surface litter, or woody debris (Harrison *et al.*, 1995; Hooker and Compton, 2003). Globally, carbon accumulation in mineral soils recovering from past tillage is likely to amount to less than 0.1 Pg C yr^{1} (Post and Kwon, 2000). The carbon stocks maintained in aboveground biomass, however, do differ between forest and cropped field, and the rate of litter fall, leading to differences in soil organic matter (SOM) in soils. Thus, a better understanding of the relationships between C stocks and effects of land use practices is required in the context of the global C balance for effective and sustainable restoration of C in terrestrial ecosystems.

The C content of agricultural soils has generally been depleted in repeated cropping periods by 20-50 % relative to their original condition. This removal of C from the soil causes severe degradation of soil fertility (Hairiah *et al.*, 2001). Most C enters ecosystems through leaves and C accumulation is most obvious when it occurs in above-ground biomass. However, more than half of the assimilated C is eventually transported below-ground through root growth and turnover, and oxidation of organic matter. Net accumulation of organic matter occurs through practices that increase the amount of plant-fixed C that is returned to soils in the form of residues (leaves, stems, branches and roots) (Hairiah *et al.*, 2001). Roots make a relatively large contribution to organic matter content due to their location in the soil, and linkage to soil particles (van Noordwiijk *et al.*, 1997). The soil organic matter is composed of decomposing residues: by-products formed by biota responsible for decomposition of the residues, the micro-organisms themselves, and the more resistant soil humates (Hairiah *et al.*, 2001).

2.4.1 Soil organic matter dynamics

The amount of organic matter at any one time in a soil is the result of the rates of addition and loss of the organic matter (Dean *et al.,* 2004). Under natural forest vegetation, the rate of input is high and the equilibrium is also higher though the rate of decomposition is accelerated (Feller, 1993). In opposition, cultivation of tropical soils results in rapid decline in soil organic carbon (Nye and Greenland, 1960). It appears that resistant C fractions which make up a large part of carbon in temperate grassland soils, contribute a much smaller part in tropical grasslands (Scholes *et al.,* 1992). Fresh organic material breaks down rapidly after addition to the soil than mummified material releasing nutrients through mineralization (Lal and Sanchez, 1992). The external factors influencing the rate of soil organic matter mineralization includes: temperature, rainfall, vegetation, biological activity, organic matter composition and soil disturbance. According to Mejiboom *et al.* (1995) coarse sand fraction should be used for density fractionation and recovery of light fraction soil organic matter instead of whole soil. Barrios *et al.* (1996) concluded that while the light fraction was a more labile fraction of the organic matter than more humified material, both fractions are involved in releasing nutrients through mineralization. Polysaccharide component of soil organic matter plays a role in interparticle bonding and aggregate stabilization in conjunction with fungal hyphae and other materials (Lavelle *et al.*, 1992).

The lack of a suitable measure of soil organic matter as an indicator of soil quality has limited the predictive understanding of SOM dynamics (Woomer *et al.*, 1994). The physical fractionation of soil is to isolate and quantify functional pools of SOM (Christensen, 1992). When not replenished, soil organic matter functions as a non-renewable resource and some (migrant) farmers may be tempted to follow or create new forest margins and leave a zone of depleted soil behind. In the humid tropics of Asia, such lands are generally occupied by grasses such as *Imperata cylindrica* that may partly restore the soil organic matter, or at least prevent further degradation (Kotto-Same *et al.*, 1997).

Organic matter levels in virgin soils are usually in a state of dynamic equilibrium where losses are balanced by additional inputs. Cultivation of soils, however, alters this equilibrium by increasing SOM losses (Sanchez *et al.*, 1989). Land use practices such as, conventional tillage leads to exposure of aggregates protected organic matter to microbial decomposition (Elliot *et al.*, 1993). Considerable attention has been directed towards the identification of land use activities that would maintain adequate SOM levels. The overriding control on the decomposition rate of plant residues in tropical grasslands and savannahs is water availability. The chemical composition of the litter controls its decay rate when it is moist. Grass litter decomposes faster than tree litter provided it is in contact with the soil surface (Houghton *et al.*, 1994). However, grass litter frequently remains attached to the main plant, where it decays gradually until burned, grazed or trampled (Scholes and Archer, 1997). Soil fauna such as earthworms and termites move surface litter into the soil and fragment it, thereby speeding up decomposition (Lavelle *et al.*, 1992)

2.5 Changes in carbon stocks as a result of net changes in pasture and rangeland area

Changes in the area of pastures, especially rangelands, are not as well documented as changes in croplands, since both pastures are not easily distinguished from natural grasslands (Hooker and Compton, 2003). Fortunately, the conversion of grasslands to pastures, and vice versa, probably does not involve much change in carbon stocks unless the lands are overgrazed (Post and Kwon, 2000). Temporary pastures are considered to be part of crop rotation, and as such, classified under croplands (FAO, 2001). Both increases and decreases of carbon stocks may occur after conversion from one system to the other (Post and Kwon, 2000; Guo and Gifford, 2002; Osher *et al.*, 2004). For example, pasture soils cleared from forests in the Brazilian Amazon have been shown to lose carbon in some cases and gain it in others depending on the vegetation cover (Davidson, *et al.*, 2000).

The change in carbon stocks is determined by rainfall, soil fertility, management practices, species of grass and other planted vegetation, or other factors that govern the quantity and quality of productivity at a site. Guo and Gifford (2002) using data from 16 countries with most data from Australia, Brazil, New Zealand and USA, observed a modest mean increase in soil carbon (about 10%) in upper soil layers (<100 cm) when forests were converted to pastures. However, some sites had large carbon gains while others had huge losses. Similarly, abandoned pastures return to the ecosystems they were derived from, accumulating carbon in vegetation and, perhaps, soil over time (Post and Kwon, 2000).

2.6 Carbon stocks in established tree plantations

The rate of accumulation of carbon aboveground is well documented for plantations. The largest increases in plantations were in China and India, although the spatial heterogeneity is not readily available for large regions (FAO, 1995). For example, plantations may be established for timber, shelter belts, orchards, or fuelwood, and the stocks of carbon in biomass consequently vary. Whether plantations are established on non-forest lands or on recently cleared forests also affect the net changes in biomass and soil carbon. Guo and Gifford (2002) found that the establishment of plantations on forest lands or pastures generally decreased soil carbon stocks, while establishment on croplands increased C stocks. Parton *et al.* (2000) found that plantations established on agricultural lands (both croplands and pastures) lost soil carbon during the first 5–10 years but gained it over periods longer than 30 years.

2.7 Carbon stocks under different agricultural management practices

Many studies have addressed the potential for management to sequester carbon. Recent analyses from the U.S. suggest a current sink of 0.015 penta gigatons (Pg) C yr^{1} in croplands (Eve *et al.*, 2002), while for Europe a net source of 0.300 Pg C yr^{1}. The differences could be because of reduced application of organic manure to cropland (Janssens *et al.*, 2003). In Canada the flux of carbon from cropland management is thought to be changing from a net source to a net sink, with a current flux almost zero (Smith *et al.*, 2000). Globally, the current flux is uncertain. Houghton, (1994) suggests that degraded lands are expanding, especially in Africa and Asia, due to unsustainable agricultural practices thus further reducing the net sink for carbon stocks.

Davidson and Ackerman (1993) reported that on average, cultivation resulted in 25 and 30% loss of SOC. Most of the losses occurred in the first 2-5 years. The main carbon sequestration strategy should therefore be directed towards minimizing losses. Juo and Lal (1979) reported nearly double SOC contents in the top 10 cm in no till versus ploughed treatments in an experiment that evaluated soil changes following six years of continuous maize cropping. They attributed much of the difference to lower erosion under no-till.

Most agricultural management practices are aimed at increasing crop or pasture productivity and thus, directly increase organic input levels. Although soil depth is fixed, it is not easily changed by management. In many soils of the sub-humid and humid tropics, the effective rooting depth is constrained by lack of nutrients or increasing acidity. Where native grasses have low productivity, management can help improve carbon inputs by selecting adapted, deep-rooted grass species (Cerri *et al.*, 1991; Fisher *et al.*, 1994). Removal of forest vegetation to plant deep-rooted pasture grasses resulted in massive losses of carbon from the forest biomass (Davidson *et al.*, 2000). Agroforestry- based management options involving fast growing nitrogen-fixing tree species are aimed at rehabilitating degraded lands. Lal *et al.* (1979) reported that fallows planted with several different species of grasses or legumes increased SOC by between 18 and 30% after 2 years on an eroded Alfisol in Nigeria. Kotto-Same *et al.* (1997) have shown how this can be realized in a slash and burn system in Cameroon in West Africa. Woomer *et al.* (2000) further gives results from a multi-location experiment throughout the tropics. When considering management option, we should appreciate whether they are aimed at SOM-increasing or SOM-protecting measures or both. Kotto-Same *et al.* (1997) reported that effective management must consider both above and belowground compartments. The change (positive or negative) in soil C stocks that can be linked to management of the land is generally less than 20 Mg C ha^{-1} (Houghton, 1999). This amount is smaller than the changes that can be achieved in aboveground C storage when woody vegetation is reintroduced or the typical lifespan of trees in the system is increased. This assessment, however, may change if more knowledge is obtained on C storage in deeper layers of the soil and the way this C storage depends on vegetation or land use.

2.8 Methods of measuring carbon stocks

There are two methods used to measure losses or accumulation of C on land and they include those for C stocks and for C fluxes. The emphases in this section are on measuring stocks of C in the vegetation of forest, cropped, and pasture land systems. Soils may continue to lose carbon for several decades following initial cultivation, while forests may continue to accumulate carbon for centuries after harvest. Thus changes in land use have a legacy that lasts long after the initial change (Goodale *et al.*, 2002). The losses and accumulations of carbon for different ecosystems and different types of land use are calculated from the annual net flux of carbon attributable to land-use change. Those lands not known to have been directly affected by human activity are assumed to be in steady state (that is, neither accumulating nor losing carbon) and are used as the baseline (Hairiah *et al.*, 2001). Changes in climate, CO_2, or other environmental variables, and natural disturbances, are ignored so that the calculated flux is attributable to land-use change alone. Data on C-stocks can be used: to directly assess the current C stocks in above and belowground pools in plots that represent a certain land cover, as part of a land use system; to extrapolate to the 'time-averaged C stock' of a land use system (Hairiah *et al.*, 2001). Carbon stock data can also be used to initialize simulation models that can explore C dynamics with respect to land use change and the subsequent effect on the global climate. Furthermore, biodiversity and profitability assessments can be compared with C stock data to study trade-offs among global environmental benefits and private incentives to the farmer (Hairiah *et al.*, 2001).

2.8.1 Assessment of aboveground Carbon: through allometric relations for trees

Table 1. Allometric relation for estimating biomass from tree diameter and height.

Life zone (rainfall, mm/yr)	EQUATION (W = tree biomass, kg/tree; D = dbh, cm; H = height, m; ρ = wood density, g/cm³)	Range, cm	Number of trees	R^2
Dry (<1500)	$W = 0.139\ D^{2.32}$ (Brown, 1997)	5-40	28	0.89
Moist (1500-4000)	$W = 0.118\ D^{2.53}$ (Brown, 1997) $W = 0.049\ \rho\ D^2\ H$ (Brown *et al.*, 1993) $W = 0.11\ \rho\ D^{2+c}$ with c (default 0.62) $H = a\ D^c$ based on (Ketterings *et al.*, 2001)	5 - 148	170	0.9
Wet (>4000)	$W = 0.037\ D^{1.89}\ H$ (Brown, 1997)	4 - 112	160	0.9

Source; Hairiah *et al.*, 2001

A major proportion of the C and nutrients in terrestrial ecosystems is found in the tree component. To reduce the need for destructive sampling, biomass can be estimated from an easily measured property such as stem diameter at a specified height, using allometric equations (Brown, 1997). Such equations exist for many forest types and a small number are species specific.

Destructive measurement of trees (cutting down and weighing) to generate allometric equations that has high precision needs a lot of labor and time, but when it is done it can be applied to other tree species in the same forest area. A substantial number of allometric equations have been developed for various climatic zones, forest types and tree species (Brown, 1997). Use of these equations for a new situation is associated with a difficult choice among the various equations; the calculated estimates may vary by over a factor of 2 between equations which are applied to one specific data set (Ketterings *et al.,* 2001). Dead wood, both lying and standing, is an important C pool in forest and should be measured for an accurate representation of C stocks. However, to estimate their biomass from their calculated volume, samples need to be taken, to measure the wood density. This is because the dead wood left in the forest is sometimes still solid, but is often found partly decayed.

2.8.2 Belowground Carbon: root biomass

Root derived carbon forms the basis of energy channel for belowground food web (Moore, 1988). The position of roots partly in the soil matrix and partly in larger aggregates or cracks depends on the land use system. Data from standing root biomass in forests in the tropics are scarce (van Noordwijk *et al.*, 1994). However, there are various methods which have been used such as, ^{14}C pulse labeling techniques to measure belowground C outputs from plants (Swinnen, 1994). The peak and trough method by Anderson and Ingram (1993) is such that, measuring root biomass requires manually picking or washing roots out of a known volume of soil, and then separating living from dead by examining each root individually. This task is tedious, especially because many samples must be analyzed to account for large spatial variation. In addition, the belowground biomass can be estimated from the proximal roots at the stem base using allometric equations. The other method is based on sequential analysis of images taken in minirhizotrons (van Noordwijk *et al.*, 1994) This study however, did not account for root biomass of the trees in forests or the other land use systems, because of the methodological complexity to measure them accurately.

2.8.3 Belowground C: Soil Organic Matter (SOM)

Accumulation of soil organic matter is a function of the amount of plant, animal and microbial inputs the soil has received in the past (Brady, 1996) and the rate at which it decays (Johnson *et al.*, 2000). The decay rate in the carbon cycle is related to the quantity and quality of the plant residue returned to the soil, soil type, and amount of clay mineral content and their management (Martens, 2000). Soil organic matter content is as a result of organic inputs and the past rates of decomposition, as determined by inherent properties of the soil and the vegetation or land use system of the site. There are large differences in C storage capacity of soil which are determined by; soil texture, landscape, degree of drainage, mineralogy and physical disturbance (Van Noordwijk *et al.*, 1997).

Due to the large inherent variability of soil carbon content and the relatively slow responses to change, it is not easy to assess the influence of land use or changes in land use on soil C stocks from 'survey' type data, especially where land use practices tend to occur on specific soils or landscape positions (Paustian *et al.*, 1997). Current methods for inventory of soil C stocks at national scale (Houghton, 1997) are based on an estimate of the soil C stocks under natural vegetation and the relative changes due to aspects of human land use. These include soil tillage, drainage and a reduction in organic inputs relative to the natural vegetation. Many of the existing data sets in tropical countries have not been analyzed to their full potential and thus could be used to obtain location-specific estimates of the reference C content. The difference between current and potential C storage can then be expressed as a C saturation deficit (Van Noordwijk *et al.*, 1997).

2.9 Diffuse Reflectance Spectrometry (DRS) Techniques

Recent developments in laboratory and field based reflectance spectrometry, may have emerged as promising tools for assessing soil functional capacity (Shepherd and Walsh, 2000). Diffuse reflectance spectrometry (DRS) is a rapid non-destructive analytical technique for studying interactions between incident light and material surfaces. Soil reflectance is a function of the inherent spectral behavior of heterogeneous combinations of mineral, organic and fluid matter that comprise soils (Stoner *et al.*, 1981). DRS methods are sensitive to both organic and inorganic phases of soil and the actual mineralogical detection is a function of spectral coverage. Resolution and signal to noise ratio of the sensor, the abundance of material and strength of absorption features in the measured region (Clark, 2000). Organic matter, moisture, mineral oxides, texture and surface conditions have unique influences on soil reflectance (Stoner *et al.*, 1981).

The fundamental molecular frequencies associated with soil organic matter generally occur in mid-infrared (MIR) (2.5-25 μm) and have overtones and combination modes both in the near infrared (NIR) (0.7-1.0 μm) and the short wave infrared (SIR) (1.0-2.5 μm). Hydroxyl ions (OH^-), Sulfates (SO_4^{2-}) and carbonates (CO_3^{2-}) produce particular diagnostic absorption features in the SIR (Ben-Dor, 1999). These features are important for distinguishing different types of clay. There is a strong relationship between soil color and visible reflectance (0.4-0.7 μm). Organic matter, water, iron oxides and chemical composition of transition metals in clay minerals are important determinants of soil color (Ben-Dor, 1999).

Reflectance spectrometry can provide rapid prediction of soil physical, chemical and biological properties. Soil spectra commonly depict broad and shallow absorption bands due to iron oxide phases. The narrow and distinct absorption features at 1.4, 1.9, and 2.2 μm relate to clay minerals (Galvao *et al.*, 2001). Organic carbon lowers soil albedo (relative reflectance averaged across the entire spectrum) and causes the spectral continuum to become more concave from the visible to the SIR (Hill and Schutt, 2000). Although albedo is the primary source of variability, absorption features play a significant role in relation to specific chemical characteristics (Palacios *et al.*, 1998). Stoner *et al.* (1981) reported that reflectance measurements in the NIR (0.72-1.3 μm) and MIR (1.3-3.0 μm) spectral regions often reveal textural, structural and mineralogical differences. Mathew (1976) found that cation exchange capacity (CEC), silt, clay, organic matter and iron content were highly correlated with soil spectral reflectance. Dalal and Henry (1986) found that within a narrow range of soil color and moderate amounts of organic matter (0.3-2.5%), the NIR reflectance technique provides a rapid and simultaneous measurement of moisture, organic C and total N in soils.

Despite the evidence of the potential for reflectance spectrometry in soil testing, there has been little focus on its potential as a rapid integrated method for sensing soil functional capacity for plant production and precision soil management (Shepherd and Walsh, 2000). Studies have shown that overall soil reflectance increases under conditions of soil physicochemical degradation, while intact non degraded soils tend to be associated with low overall soil reflectance (Hill and Schutt, 2000; Placious and Ustin, 1998). Organic carbon significantly influences the albedo and shape of the soil spectral continuum between 0.35 μm and 1.4 μm.

3.0 MATERIALS AND METHODS

3.1 Farming Systems and Land Use Management on Mount Marsabit

Traditionally the mountain area was used as a reserve grazing area but significant crop production started in 1945. As recently as 1960, the area under crops was less than 700 hectares. Currently, the major forms of land use competing for space include crop and livestock production, human settlement, wildlife and forest conservation. According to Warui (2000), most settled farmers result to destructive forms of land use (such as charcoal production) because the farming systems they employ fail to meet their livelihood requirements. The agricultural development and the impact this has had on land use around the mountain is presented in table 2.

Table 2. Historical perspective of agricultural development on Marsabit Mountain.

YEAR	ACTIVITY
	Marsabit mountain is not inhabited but is a fall back grazing area to Maasai related tribes (the Laikipia) and possibly the Samburu and Rendille.
1897	Emperor Menelik expels Boran speaking tribes (Borana, Gabbra, and Shekuye) from Ethiopia and they enter Kenya. No Boran was on Marsabit Mountain then.
1922	Colonialists bring 20 Burji farmers to Marsabit mountain to start farming as a policy decision to encourage farming on Marsabit Mountain.
1945	Borans begin to cultivate crops. Small scale, around Sagante and town.
1950	First demarcation of farming plots done (town, Karantina, Hula hula, Majengo and Sagante) total of 350 ha.
1958	Grazing control on Marsabit mountain instituted. Farmers allowed only 12 animals per household 8 cows, 2 oxen and 2 shoats.
1970's	Songa and Badassa settlement schemes opened and Rendille who lost animals during drought are settled while refugees from Ethiopia are settled in Badassa.

YEAR	ACTIVITY
1980	Gabra scheme is opened and Gabra pastoralists who have lost animals are settled.
1983	Marsabit Mountain area is declared an adjudication area and this prompts land grabbing even in Marginal areas.
1989	Cereal Board opened up to assist farmers export maize and beans. This led to more land converted to crop production especially maize
1995	Tribal clashes (Boran vs. Burji) led to displacement from farms and destruction of permanent crops

Source: Sobania, (1970).

3.2 Site description

3.2.1 Geographical location and size

Marsabit district is one of the thirteen districts that form Eastern province. It borders Ethiopia and Moyale to the north, Turkana district to the west, Samburu district to the south and Wajir and Isiolo districts to the east. The district lies between latitude 01^0 15’ North and 04^0 27’ North and longitude 36^0 03’ East and 38^0 59’ East. The district covers an area of 69,430 km^2 including 4,125 km^2 covered by Lake Turkana. It is the second largest district in the country accounting for 11 per cent of the total area of the country. The district is divided into six divisions (figure 1a) namely – Central, Gadamoji, Maikona, North Horr, Laisamis and Loiyangalani. The divisions are sub-divided into 28 locations and 65 sub-locations. There is only one Local Authority – Marsabit County Council which has 20 wards (National Development Plan, 2002-2008).

3.2.2 Population

The district is home to a number of ethnic groups, the major ones being Boran, Gabra, Rendille, Burji, Turkana and Ariaal (1999 National Population Census). The total population is 121,478 with an estimated annual growth rate of 2.1% (National Development Plan, 2002-2008).

Figure 1a: Mt Marsabit Forest and its environs

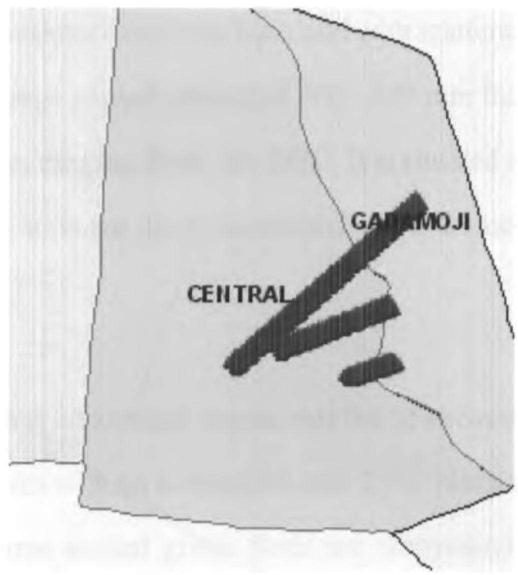

Figure 1b: Sampling transects across the three land use systems

3.2.3 Topography; Geology and Climate

The district receives between 200 mm to 1,000 mm of rainfall per annum for the lowest to the highest elevation respectively. The rainfall displays both temporal and spatial variation and is bimodal. The district occupies the driest region in the country. Low rainfall combined with high temperature result in high evapo-transpiration rates that exceed annual precipitation leading to a marked moisture deficiency (National Development Plan, 2002-2008). The area around Mt. Marsabit can be classified into 4 different agro-climatic zones. These are Sub-humid, Semi-arid, Arid, and Very arid zones.

The sub-humid area covers an area of about 200 km^2. This area includes the forest and areas surrounding it. It has an average annual rainfall of about 1000 mm, temperatures of between 14-23°C, relative humidity of 50 - 90% (with misty mornings), an altitude of between 1200 – 1500 m above sea level (a.s.l.) The soils are deep, well drained, volcanic, high in organic matter, reddish - brown clay to clay loam (FHI, 2001).

The semi-arid area covers about 500 km^2 and consists of medium bushland with scattered shrubs and dense perennial grass. It has an average annual rainfall of 300 - 800 mm that is unreliable. The temperatures are cool to warm ranging from 16 - 26°C. It is situated at an altitude of between 600 - 1200 m a.s.l. The soils are deep, excessively well drained sandy loams (FHI, 2001).

The arid area covers about 600 km^2. The area has an average annual rainfall of between 100-200 mm (very erratic) and high temperatures with an average of over 25°C. Natural vegetation is short scattered shrubs with sparse annual grass. Soils are stony-sandy loams and are generally shallow. Winds are consistent and fast in this area (FHI, 2001). The extremely arid area covers over 700 km^2 with an average annual rainfall less than 100 mm (very erratic) with very high temperature average of about 30°C. The altitude is less than 300 m a.s.l. The natural vegetation consists of very sparse annual grass with sandy stony soils high in salinity levels (FHI, 2001).

3.2.4 Selection of study site and ground description

Maps (figure 1a) and land use databases and their interpretation at field scale was applied to select the relevant land use patterns representative of the area. Three transects were laid down cutting across the chosen land use patterns. These were the forest, cropped and pastureland using global positioning system (GPS) (figure 1b). Transect A cut across the three land use systems. Transect B was mainly forest while transect C was cropped land with a little portion of transitional forest. Intact primary forest plots were paired with cultivated fields in proximity to one another (< 100 m apart). The current land use, signs of physical degradation and dominant vegetation components were recorded to control for potential local differences in major soil types and landscape positions for each plot (case and control approach). Slope, terrain profile and aspect measurements were obtained using a clinometer. Ground and woody vegetation canopy cover and surface rock/stone cover were assessed using Braun-Blanquet cover ratings (Shepherd *et al.,* 2002).

3.3 Field methods

3.3.1. Soil sampling and plant biomass collection

The case and control approach as described by Shepherd *et al.* (2002) was used. Assessment of carbon stocks were aimed at specific area that is, vegetation or land-use types. Stratification to obtain a clear, operational definition of the unit of analysis was done. A sampling strategy was implemented using a cluster survey design as described by Shepherd *et al.* (2002). A cluster comprised of 13 plots measuring 30 by 30 m each systematically laid out. One cluster was randomly located in each landuse within a transect in a quadrant within transects A, B and C. All plots were geo-referenced at their center-points (15 m) using a survey grade differential global positioning system (GPS). The starting point and direction of the quadrant's central axis were randomly selected. Stakes were inserted into the ground at the two ends of the central axis. A 30 m tape measure was tied between the stakes at approximately 30 cm height. Quadrant sides were identified using 2 m poles, one for each side of the central tape. The continuous identification narrow quadrant boundaries with short poles were designed to reduce errors in the width of sampling areas (Brown *et al.*, 1996). Quadrant locations sides were also marked for later plot identification. For aboveground vegetation and stone / rock cover, ratings were obtained with the assistance of a 1 x 1 m plot frame and then averaged in instances where the distribution of cover values within the plot were judged to be clumped. For each plot, 3 top soil (0-20 cm) and 3 subsoil (20-50 cm) samples were augured at the 5, 15, and 25 m positions of the centre-line of the plot and in direction of the dominant slope gradient (figure 2). In some instances, subsurface restrictions prevented recovery of subsoil samples. In these cases, the restriction type and depth were recorded.

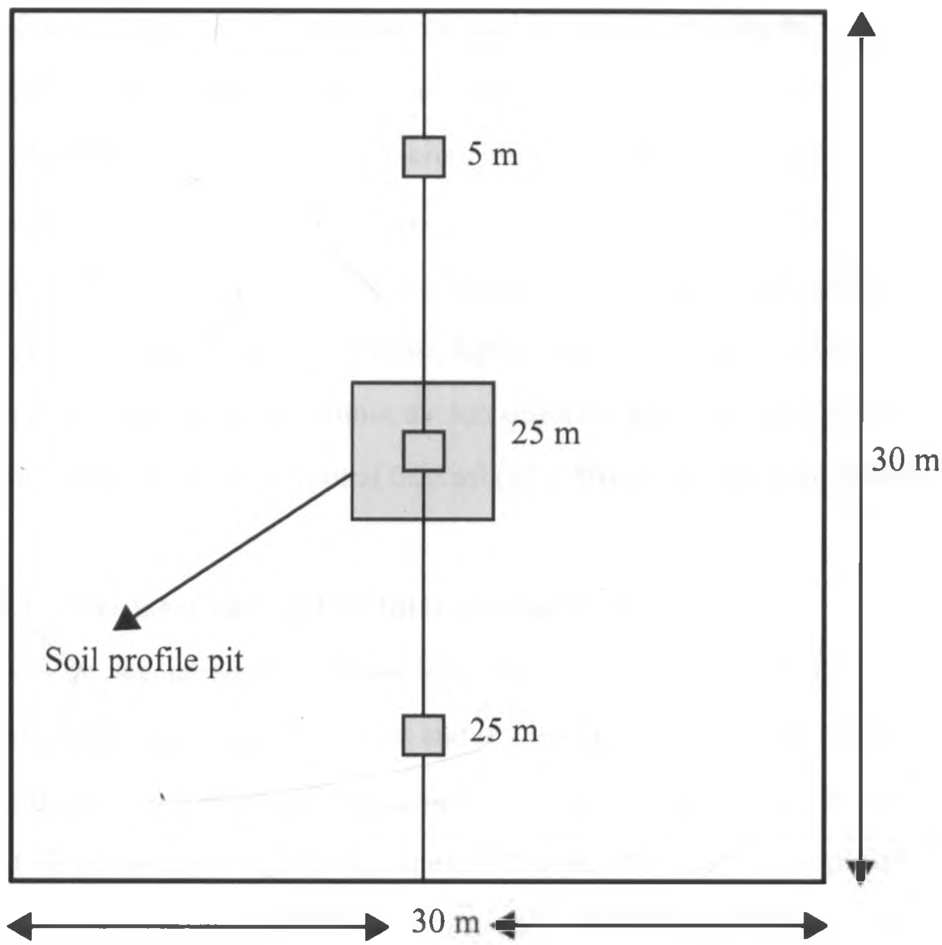

Figure 2. Sampling points within a 30 × 30m plot

3.3.1.1 Estimation of carbon stock in tree above ground biomass

The relative proportions of system carbon stocks were sampled in Mt. Marsabit forest within different landuse patterns according to the method described by Woomer *et al.* (1998). Diameters at breast height (DBH) were measured using dial calipers and diameter tapes and recorded. Where possible, tree names were identified and recorded as well. Five quadrants of 5 m × 20 m, assigned at random were established within each landuse. Quadrants were not allowed to 'cross-over' one another. The diameter at breast height (DBH) was recorded for all trees with diameters greater than 2.5 cm falling within the quadrant. The diameters of smaller trees (2.5-15 cm) were measured with a dial caliper and larger ones (> 15 cm) with a diameter tape, which converts circumference to diameter. Correction was made for tree buttressing by either calculating the mean from maximum and minimum diameter (D) or, if within reach, the diameter above the buttress. Only trees with more than 50% of their diameter falling within the quadrant were recorded. Tree biomass estimates were assigned using the allometric approach of Brown *et al.* (1989) using the equation for Dry life zones based on diameter ($W=0.139D^{2.32}$) where W is tree biomass, kg/tree and D is diameter at breast height, cm. Loose green materials falling within the sub-quadrate were collected by hand. Surface litter was collected from the central 0.25 m^2 (50 × 50 cm) of each sub-quadrant.

3.3.1.2 Under-storey and surface litter measurements

Under-storey vegetation and surface litter was measured at the 5, 15, and 25 meter positions of the centerline of the plot and in direction of the dominant slope gradient using a 0.5 m × 0.5 m frame. Under-storey biomass included trees with DBH < 2.5 cm and all herbaceous vegetation, vines and lianas. Plant tissues originating outside the quadrant but falling within it were cut and stored in sample bags whereas those originating within the quadrant but falling outside were cut and discarded. The remaining under storey vegetation falling within the quadrant was cut at ground level and collected according to Kotto-Same *et al.* (1997). Surface litter, within a 50 cm × 50 cm frame at the 5, 15, and 25 meter positions of the center line of the plot were raked and collected. These were taken to the laboratory dried and weighed to form part of aboveground carbon. All woody necromass <10 cm in diameter within the frame were cut using secateurs and collected. Samples were weighed, sub-sampled, oven dried at 65°C to constant weight and corrected for moisture content. 'Live vegetation was assumed to contain 45% C on a dry weight basis' (Woomer *et al.*, 1998). The surface litter was ground and taken to the laboratory for total organic carbon analyses following procedure described by Anderson and Ingram (1993). During sampling of the quadrants for aboveground carbon stocks, field notes were recorded on species composition of different land uses. The most abundant trees falling within each quadrant was identified using binoculars and/or based on bark characteristics. Unknown species were collected, pressed and transported to the Kenya National Museum for identification.

3.3.1.3 Soil and root sampling

Soils were collected in mid January to February 2005, just before the onset of long rains. Soil and litter samples were recovered from a 900 m^2 quadrant. Using an auger, soil samples were collected at two depths 0-20 and 20-50 cm, where soil depth allowed. Coarse roots were recovered during excavation using a hand saw. Soil samples were taken at the 5, 15, and 25 meter positions of the center line of the plot and in direction of the dominant slope gradient. Soil profile pits (figure 2) at 1.5 m depths were dug in three plots within a cluster. Soils from pit walls were collected at 10 cm intervals up to a depth of 150 cm using core rings (5 cm diameter and 5 cm height) to ascertain carbon changes with depth. All samples were air-dried, crushed and passed through a 2-mm sieve. Core rings of known volumes were driven into the soil (at two depths 10 cm and 35 cm) until their tops were at the same level with the ground surface to obtain samples for bulk density determination. The buried cylinders were then dug out carefully slicing off the bottom but not disturbing the soil core. The soils were placed in sample bags for moisture and bulk density determination. Soil and root samples were then transported to the laboratory for spectral and chemical analyses. The soil samples were air-dried, ground and passed through a 2 mm sieve for spectral and chemical analyses but a portion of the ground sample was passed through a 0.5 mm sieve for total carbon and nitrogen determination.

3.3.2. Soil characterization procedures

3.3.2.1. Soil organic carbon

Organic carbon in the soils was determined using the Walkley - Black method (Blake, 1965). A known amount of soil passed through a 0.5 mm screen was put in a 250 ml Erlenmeyer flask and 10 ml of potassium dichromate solution (oxidizing agent) added with a pipette, swirling the flask gently. 20 ml of concentrated sulphuric acid was added rapidly, directing the stream into the soil-dichromate mixture, and the flasks were allowed to stand to cool for 30 minutes. Five milliliters of orthophosphoric acid and 5 ml of barium diphenylamine sulphonate indicator - a carbon indicator were added and the solution titrated with 0.5 N ferrous sulphate. The phosphoric acid gives the environment necessary to obtain a good end - point when titrating with ferrous sulphate. A blank titration was done in the same manner but without soil to standardize the dichromate. The amount of organic carbon in the samples was determined by the amount of chromic ion (Cr^{3+}) produced in the reaction below.

$$2Cr_2O_7^{2-} + 3CO + 16H^+ = 4Cr^{3+} + 3CO_2 + 8H_2O$$

Dichromate + carbonmonoxide + hydrogen = chromic ion + carbondioxide + water

Total organic carbon (%) was calculated as:

$$\%C = \frac{(\text{m.e. dichromate} - \text{m.e. } FeSO_4) \times 0.3 \times 1.3}{\text{Weight of soil in grams}}$$

3.3.2.2 Determination of Cation Exchange Capacity (CEC) and Exchangeable Bases

Cation exchange capacity and exchangeable bases were determined using procedures outlined by Blake (1965). First a known amount of soil passed through a 2 mm sieve was leached with 100 ml I N ammonium acetate pH 7.0 to replace all naturally occurring exchangeable cations with ammonium ions. The leachate obtained was used for the analysis of exchangeable bases (Na, Mg, K and Ca). Secondly, the soils were washed with alcohol to remove excess ammonium acetate and were then leached with 100 ml of 1 N KCl.

3.3.2.3 Cation Exchange Capacity (CEC)

Using the 1N KCl leachate in section 3.3.2.2, ammonium nitrogen was determined by the use of a Markham still steam distiller followed by titration. The indicator in this method is the one required by the LABEX method to determine CEC. Twenty milliliters of boric acid and 2 drops of the phenothelein indicator were placed in a 250 ml conical flask and placed under the condenser. Ten milliliters of the sample, 300 ml of distilled water and 10 ml of sodium hydroxide were put into a Kjeldahl flask respectively and placed on the distiller to distil until the sample collected on the conical flask was 100 ml. The chemical reaction was as follows:

$$\underset{\text{(pink)}}{NH_3\,(g) + H_3BO_3} \rightarrow \underset{\text{(green)}}{NH_4H_2BO_3 + HCl} \rightarrow \underset{\text{(pink)}}{NH_3\,(g) + H_3BO_3}$$

The total number of negative charges in a soil that are balanced by exchangeable cations is the CEC. Thus the distillate was titrated with 0.01N hydrochloric acid. The end point was when the distillate changed from green to pink. CEC was calculated as follows:

$$\text{CEC (Cmol/Kg)} = \frac{\text{Titre} \times \text{Normality} \times 100 \text{ ml} \times 100 \text{ g}}{10 \text{ ml} \times \text{weight of soil in grams}}$$

3.3.2.4 Exchangeable Calcium and Magnesium

The ammonium acetate leachate was used for calcium and magnesium determination. For dilution, 1 ml of the leachate in section 3.3.2.2 and 9 ml of deionized water were placed into a 50 ml volumetric flask and mixed thoroughly. A mixed stock standard solution (1200 Ca + 240 mg Mg/l) was used to prepare the working standards which were mixed in ammonium acetate extracting solution. Absorbance and concentration readings of the standards and samples were then taken using the Atomic Absorption Spectrophotometer (AAS) model 210 VGP.

Calculations:

$$\text{Ex Ca (Cmol/Kg)} = \frac{\text{AAS reading (ppm)} \times 250 \times 100 \times \text{dilution}}{\text{Equivalent weight} \times 1000 \times \text{weight of soil}}$$

$$\text{Ex Mg (Cmol/Kg)} = \frac{\text{AAS reading (ppm)} \times 250 \times 100 \times \text{dilution}}{\text{Equivalent weight} \times 1000 \times \text{weight of soil}}$$

3.3.2.5 Exchangeable Potassium and Sodium

The ammonium acetate leachate in section 3.3.2.2 was used for potassium and sodium determination. A mixed stock standard solution (1200 K + 240 Na Mg/l) was used to prepare the working standards which were mixed in ammonium acetate extracting solution. The flame photometer was used. The flame photometry works in that certain metallic ions when ignited in a flame emit visible light of characteristic wavelength. The amount of light of that wavelength is a measure of the concentration of that particular ion. Na filter for Na determination and a K filter for K determination were used. Absorbance readings of the standards and samples were then taken from the flame photometer.

Calculations:

$$\text{Ex K (Cmol/Kg)} = \frac{\text{Graph reading} \times 250 \times \text{dilution}}{\text{Weight of soil}}$$

$$\text{Ex Na (Cmol/Kg)} = \frac{\text{Graph reading} \times 250 \times \text{dilution}}{\text{Weight of soil}}$$

3.3.2.6 Determination of total nitrogen.

The Kjeldahl wet digestion method by Bremner was used in which the sample was digested with sulphuric acid so that all the nitrogen is converted to ammonium (Page *et al.*, 1982). One gram of soil was weighed and put into a digestion tube. Ten grams of mixed catalyst was added then 20 ml of concentrated (36 N) sulphuric acid gently added. Digestion started at low temperature until the mixture cleared (30 minutes). The temperature was then raised up to the end of digestion (90 minutes). The mixture was allowed to cool, but before cooling completely, a small amount of distilled water was added. This was to prevent solidification of the digest. The digest was then transferred into a distillation flask and 40 mls of 10 N NaOH added. This was done quickly to avoid escape of ammonia gas before actual distillation started. The NH_3 released during distillation was collected in 1% boric acid (20 mls) with a mixed indicator. Titration was with 0.01N hydrochloric acid. The end point was when the distillate changed from green to pink. Total nitrogen was calculated as follows:

$$\text{(\%) N} = \frac{\text{Titre} \times \text{Normality} \times 14 \times 100 \times \text{dilution}}{1000}$$

3.3.2.7 Determination of extractable Phosphorus.

The Mehlich method of using double acid (DA) which is 0.05 N HCl in 0.025 N H_2SO_4 was used (Page *et al.*, 1982). Five grams of soil were weighed into a 125 ml extracting bottle, 50 ml of double acid reagent was then added accurately. The bottles were stoppered tightly and placed horizontally in a rack on a mechanical, reciprocating shaker and shaken for 30 minutes. The soils were then filtered through Whatman no. 2 filter paper. Standards stock solution was made using monobasic potassium phosphate and toluene and made to 100 ppm P. Working standards were prepared from this stock. Different amounts of aliquots namely 1, 5 and 10 ml were pipetted from the extract into a 50 ml volumetric flask. Approximately 25 ml of distilled water followed by 8 ml of reagent B and distilled water was then added to mark and mixed thoroughly. This was allowed to stand for 15 minutes. Colorimetric determination was done by ascorbic acid method. Ascorbic acid helps form phosphomolybdate blue complex and the colour was then measured from the SP 500 Spectrophotometer.

Calculations:

$$\text{P (Cmol/Kg)} = \frac{\text{Graph reading} \times 50 \times \text{dilution}}{\text{Weight of soil in grams}}$$

3.3.2.8 Soil pH in water

Soil pH was determined following the procedure outlined by Blake (1965). Five grams of air-dry soil (passed through a 2 mm sieve) was weighed into a plastic specimen bottle and 12.5 ml of distilled water was added. The bottle was then tightly stoppered and shaken for 30 minutes in a mechanical reciprocating shaker. The suspension was allowed to stand for 10 minutes and then the soil pH was measured by inserting electrode of a pH meter into the partly settled suspension. The pH was then reported as “soil pH measured in water”.

3.3.2.9 Determination of Soil Bulk Density

The bulk density (Pb) was determined by the core-ring method (Blake, 1965) using metal core-rings of 5 cm diameter and a height of 5.1 cm. Three samples at each depth were sampled and dried at 105 °C for 24 hours to constant weight.

Calculations:

$$\text{Bulk density } (Mg/M^3) = \frac{\text{Oven dry weight of the soil in Mg}}{\text{Volume of the soil in } M^3}$$

3.3.2.10 Mechanical Analysis of Soil

Particle size distribution was determined using the hydrometer method as outlined by Blake (1965). Fifty-one grams of air-dry soil that had been ground to pass through a 2 mm sieve were transferred to a "milk shake" mix cup. This soil was free of organic matter, having been previously destroyed by adding small portions of 30 % hydrogen peroxide solution. Five percent sodium hexametaphosphate was added to raise the electrokinetic potential of the soil, and then mechanical shaking for six hours dispersed the soil. The suspension was transferred to a 1000 ml glass cylinder and topped to the mark with distilled water. The cylinder was stirred for five minutes until all soil was in suspension and immediately the soil hydrometer was placed into the suspension. The first reading measured the percentage of silt and clay, while the second reading indicated the percentage of total clay in the suspension. Before each reading, temperature of the soil suspension was recorded. Greater temperatures result in reduced viscosity due to liquid expansion and a more rapid descent of falling particles. The first reading was taken after 40 seconds and temperature was recorded. The second reading was taken after 3 hours. Temperature was recorded.

Calculations:

$$\text{sand} = 100 - [H1 + 0.2\,(T1 - 68) - 2.0]\,2$$
$$\text{clay} = [H2 + 0.2\,(T2 - 68) - 2.0]\,2$$
$$\text{silt} = 100 - [\%\ \text{sand} + \%\ \text{clay}]$$

where: H1 was hydrometer reading at 40 seconds.
T1 was temperature reading at 40 seconds
H2 was hydrometer reading after 3 hours
T2 was temperature reading after 3 hours

The results were corrected to a temperature of 68 Fahrenheit. For every degree over 68 F, 0.2 was added to the hydrometer reading before computation and every degree under 68 F, 0.2 was subtracted from the hydrometer reading. 2.0 was subtracted from the hydrometer reading to compensate for the added dispersing agent.

3.3.2.11 Near Infrared Reflectance Spectroscopy Measurements

Air dried soil was ground and passed through a 2 mm sieve. The soil was then packed in 12 mm deep, 55 mm diameter Duran Petri dishes. Reflectance spectra were recorded for each soil sample using a FieldSpec™ FR spectroradiometer (Analytical Spectral Devices Inc, Boulder Colorado) at wavelengths from 0.35 μm to 2.5 μm with a spectral sampling interval of 0.01 μm. The samples were scanned through the bottom of the Petri dishes using a high intensity source probe (Analytical Spectral Devices, Boulder, CO). The probe illuminated the sample (4.5 W halogen lamp WelchAllyn, Skaneatles Falls, NY) giving a correlated colour temperature of 3000 K. This also collected the reflected light from 3.5 cm diameter sapphire window through a fibre optic cable. Reflectance spectra were recorded at two positions, successively rotating the sample dish through 90° between readings to sample within dish variation. The average of 25 spectra (manufacturers default value) was recorded using calibrated spectralon (Labsphere®, Sutton, NH) placed in a glass Petri dish as a reference. Reflectance readings of each wavelength band were expressed relative to the average of the white reference readings.

3.3.2.12 Near Infrared Reflectance Spectroscopy

Spectroscopic transformation (CAMO Inc., 1998) was applied to convert spectral reflectance to absorbance (logarithmic transformation of the inverse of reflectance). Another data treatment applied was first derivative processing (differentiation with 2nd order polynomial smoothing with a window width of 0.02 µm) using a Savitzky-Golay filter as described by Fearn (2000). Derivative transformations minimize variation among samples caused by variations in grinding and optical set-up (Marten and Naes, 1989).

In this study, a calibration set of 74 samples which made a third of the total number of augured samples (222) were selected. This calibration set was selected based on the samples contributing most to the different spectra and the shortest squared Euclidean distance from the centre sample in principle component space. Principle component analysis was implemented in Unscrambler version 7.5 (CAMO Inc, 1998). Individual soil variables were calibrated against 214 (0.36-2.49 µm) reflectance bands using Partial Least Squares (PLS) regression (Martens and Martens, 2000). Cross validation was applied to determine the optimum number of factors (latent variables) needed to describe all the variation in the data to guard against over fitting. Cross validation was done by successively removing sub-sets of calibration samples from the model estimation and using them as temporary, local test samples (Martens and Martens, 2000). In essence, sample one in the calibration set was deleted and calibration performed on the rest of the samples. The procedure was repeated by deleting sample two until all samples had been deleted from the model at least once (Naes *et al.*, 2002).

The calibrations were tested by predicting the respective soil properties on the validation data set comprising the remaining one third of 74 soil samples. The coefficient of determination (r^2), root mean standard error of prediction (RMSE) and bias were used to evaluate the prediction ability of the near infrared PLS technique. The analyses were performed using the software Unscrambler version 7.5 (CAMO Inc, 1998).

3.3.2.13 Data compilation and analysis

Data was entered into a computer spreadsheet with sites occupying rows and soil parameters as columns. The spreadsheet was imported into a statistical package and summary statistics, t-tests of soil parameters contained in adjacent land uses, Analysis of variance (ANOVA) for selected soil parameters and linear regressions conducted. The data was analyzed by standard ANOVA procedure for completely randomized experimental design using genstat. Main effects were separated by least significance difference (LSD) at the $p = 0.05$ level. Land use systems (LUS) were the plots with treatments being the measured soil parameters C, N, P, K, Mg, Ca, CEC and pH. For the spectral data, multivariate statistical approach was engaged. Graphical linear mixed (Edwards, 2000) and survival time (Box and Cox, 1995) models were used to test impacts of forest conversion to agriculture and range land. Plot variables such as slope, landscape position were used as covariates or baseline conditions when analyzing the treatment effects on response variables (C, N, P, K, Mg, Ca, CEC and pH). Principal component (PCR) and partial least squares (PLS) regression analysis was used to relate reflectance to measured soil properties (C, N, P, CEC, K, Mg, Ca and pH) from the study.

4.0 RESULTS AND DISCUSSION

4.1 Estimation of above and belowground carbon stocks in Mt Marsabit Ecosystem

4.1.1 Existing vegetation cover

Forest land use system

The forests consist of well-spaced, large trees with little understorey and thick layers of surface litter. The forest has a variety of vegetation cover varying from thick tropical forest, woodland, perennial grassland, evergreen to semi-deciduous bushland, deciduous bushland, shrubland, dwarf shrub to grassland. The tropical forest was formed by trees of over 40 meter heights, with closed canopy (KFWG, 2001). In some parts of the forest there was evidence of succession. Particular species dominated the forests hence combination of species at one particular quadrant was not possible and one tends to be biased.

Figure 3a. Existing forest coverage

Figure 3b. Forest showing destruction at Mt. Marsabit

The most common species of large trees on Mt. Marsabit included: *Apodytes dimidiata, Cassipourea molasana, Croton megalocarpus, Diospyros abyssinica, Ekebargia capensis, Juniperus procera, Olea africana, Olea capensins, Premna maxima, Strombosia schefflei, Teclea norbilis, and Teclea simplicifolia.* (KFWG, 2001).

Pasture land use system

Livestock production was the most important aspect of livelihood of the people of Marsabit. In Central and Gadamoji division, the most important species cattle, and this was mainly the local Boran breed though some improved dairy crosses could be found. Local sheep breeds and goats (Galla) were common in the two divisions (Central and

Gadamoji). Less numerous but still important to the farmers were donkeys and chicken. Only a few farmers practiced bee keeping. Donkeys were used for farm transport. The chicken provided eggs and meat for the family and could also be sold. Farmers practicing apiculture sold the honey. Female adult cows (milking cows) provided milk for the family and also for sale. Bulls were used as draught - animals for ploughing and weeding, while sheep and goats provided the family with meat and could also be sold wherever a need arose. Animals kept on farm also supplied manure. This is an input for crop production. However, only a small percentage of farmers used manure.

Figure 4a. Vegetation cover in pastureland

According to Mburu, (1996), most of the farmers interviewed said that their animals were at Shurra, a government holding ground about 40 km east of Marsabit Town. The Rendille and Samburu farmers had their animals around their place of origin, in these cases at Korr, Laisamis, Loglogo and Karare. Other common grazing areas were Badassa and Gof-choba. About half of the farmers' respondents had animals both on and off the farm. The sizes of the herds varied alot. On average the herds were between

5-21 animals. The animals were mostly 'off-farm' during the wet season as water was easily available. During the dry season, the animals were moved to the highlands (figure 4a).

Figure 4b. Pastureland showing quadrant used for recovering surface litter.

Cropped land use system

Agricultural activities were concentrated around the mountain and cover about a third of the total land area (Mburu, 1996). The main crops grown were maize, beans, sorghum, millet, teff and cowpeas. The rate of adoption of appropriate farming practices and technologies was low despite the high budget spent on their promotion (Mburu, 1996). Most of the farmers practiced subsistence farming. Crop production in Central and Gadamoji divisions was a major socio economic activity for at least 42% of the population. About 20 years ago crop production was centered on the sub-humid and higher areas of the semi-arid area but with population pressure; lower areas within the semi-arid land have been converted to crop production.

Figure 5. Vegetation cover in cropped land

4.2 Effects of land-use change on soil chemical attributes

Table 3 shows the chemical and physical properties measured in the three land use systems. Data from the soils sampled at 0-50 cm, showed that soils under forest, cropped and pasture differed significantly in a number of chemical attributes (table 3). The mean C and N contents were significantly different ($p \leq 0.001$) in soils under forest compared to those in cropped and pasture (appendix 1). Carbon content declined by 32.1% and 42.8% in cropped and pasture land respectively relative to the forest soils. The nitrogen contents showed a similar trend and declined by 53.6% and 43.2% in cropped and pasture land as compared to the forest soils. The trends suggest that N losses from converted sites are higher than C losses. This may account for the high C: N ratio in the pasture lands and forest relative to the cropped lands and could be due to the forest's higher root biomass. Sanchez *et al.* (1983) noted that high C:N ratio plant

Table 3. Chemical and physical properties in the three land use types

Soil property	Forest				Cropped				Pastureland			
	Range	Mean	SE	n	Range	Mean	SE	n	Range	Mean	SE	n
pH	5.5 – 7.6	6.7	0.01	32	6.7 – 8.2	7.3	0.09	23	6.9 – 7.5	7.2	0.03	19
Exch Ca, cmol/kg	3.25 – 16.5	11.1	0.64	32	5 – 10.25	7.07	0.31	23	5.38 – 12.13	8.92	0.46	19
Exch K cmol/kg	2.5 - 21	11.58	1.02	32	5 – 24.5	11.46	0.8	23	5 - 15	9.26	0.53	19
CEC cmol/kg	17.6 – 47.9	36.89	1.26	32	23.7 – 38.6	29.18	0.78	23	23.9 – 35.5	29.96	0.73	19
Exch Mg, cmol/kg	3.75 – 10.63	7.43	0.26	32	5.83 – 12.5	8.2	0.38	23	6.25 – 12.5	9.49	0.46	19
Carbon g/kg	1.99 – 17.74	6.54	0.66	32	1.17 – 3.08	2.11	0.12	23	1.62 – 4.77	2.79	0.23	19
Nitrogen g/kg	0.55 – 2.12	1.25	0.08	32	0.28 – 1.02	0.67	0.05	23	0.28 – 0.86	0.54	0.03	19
Ext P cmol/kg	5.5 – 52.5	21.83	2.18	32	5 - 350	91.85	17.4	23	2.5 - 60	22.64	3.97	19
Clay g/kg	2 - 50	26.5	2.21	22	31 - 58	46.4	1.6	23	32 - 58	44.4	1.7	18
Silt g/kg	23 - 66	36.8	1.84	22	19 - 43	28.7	1.4	23	19 - 41	31.2	1.57	18
Sand g/kg	26 - 54	36.7	1.62	22	22 - 26	24.9	0.28	23	21 - 31	24.4	0.56	18

residues produce a strong demand for N by heterotrophic soil microbes leaving less N available for nitrification, leading to low N supply in the soil. Soil organic matter having a C: N ratio of less than 30 is easily mineralized and releases soluble inorganic N whereas SOM having a C: N ratio greater than 30 is more resistant to mineralization (Juo and Franzluebbers, 2003).

Exchangeable Ca and Mg were significantly different ($p \leq 0.001$). They were higher in forest compared to pasture and cropped areas. This could be due to a high rate of removal through leaching and uptake by crops compared to trees. Sanchez (1976) attributed this decline in exchangeable Ca and Mg to leaching and crop uptake and removal.

Exchangeable potassium was not significantly different ($p \leq 0.402$) among the LUS probably due to its luxury consumption and leaching as weathering advances. The CEC was lower in cropped and pasture lands compared to forest. Converting forests to crop production or grasslands depletes SOM and hence soil carbon content which may also result in decrease in base cation retention (Sanchez, 1976; Mann, 1986; Juo and Franzluebbers, 2003). The high amounts of P in the cropped lands could be due to additions of phosphorus rich manure by farmers compared to no addition in forests and pastureland. The soil texture ranged from loam in forest to clay in cropped and pastureland. This is due to the topography such that forest is undulating compared to the cropped and pasturelands eluviations and erosion could be the reason for higher sand and silt in the forest compared to cultivated and range lands.

Decline in SOM and other soil attributes namely total N, exchangeable Ca, CEC and particle size distribution, and increase of others namely pH and Mg in the study area have resulted in decline in agricultural productivity. The effects of forest conversion and subsequent cultivation on soil properties is such that there is a rapid decline on the soil properties as opposed to the forest system. Forest is a closed system so everything recycle within. Cropland and pastureland are open systems where nutrients are mined through crop removal unlike the forest.

4.3 Near Infrared Spectroscopy (NIRS) For Non-Destructive Characterization and Prediction of Management Sensitive Soil Properties of Soils under Different Land Use Systems

4.3.1 Near infrared reflectance spectroscopy readings

Mean relative reflectance varied among the three LUS (Figure 6). However, the mean soil spectral reflectance from the three LUS exhibited similar pattern indicating similar mineralogy. Relative reflectance averaged across the entire spectrum (albedo) of all the soils ranged from 0.025 to 0.28.

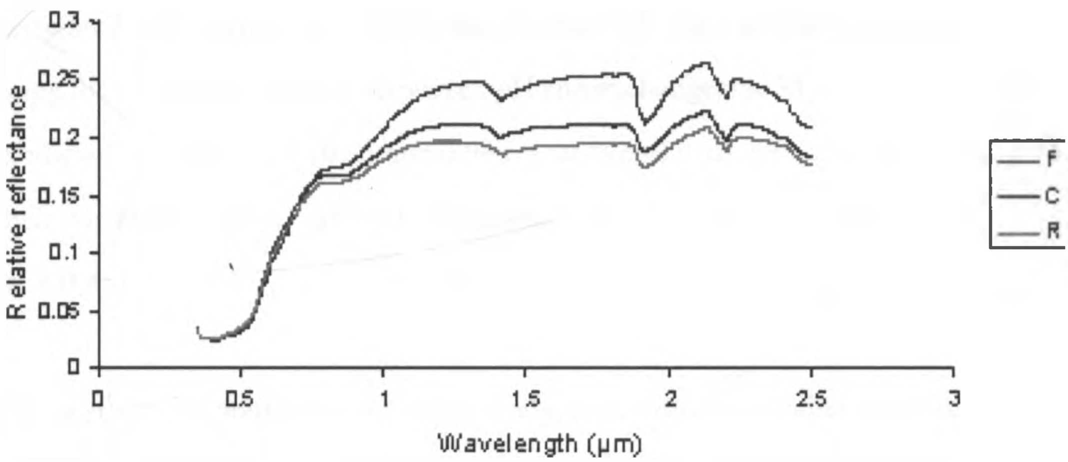

Figure 6. Near Infrared Reflectance Spectroscopy of Soil Samples, where F = forest; C = cultivated and R = range land.

Spectral reflectance patterns among the three LUS were low in the visible region and absorption bands at 1.4, 1.9, and 2.2 μm wavelength region as described by other researchers (Ben-Dor *et al.*, 1999; Shepherd and Walsh, 2002). Differences in albedo are related to soil organic matter contents (Baumgardner *et al.,* 1985; Ben-Dor *et al.*,

1999) with absorption features occurring at wavelengths 1.4, 1.9, and 2.2 μm. These absorption features were associated with clay minerals, the hydroxyl (OH) ions occur at 1.4 and 1.9μm and the clay lattice (OH) ions at 1.4 and 2.2 μm (Hurrt, 2002). Variation in the visible range and at the 0.9 μm are commonly associated with Fe^{2+} and Fe^{3+} (Hurrt, 2002), but could also be influenced by organic matter contents (Ben-Dor *et al.*, 1999).

4.3.2 NIRS Prediction of Soil Properties using Partial Least Squares Regression

Figure 7 shows regression of soil properties measured by standard laboratory procedures and predicted by NIRS - PLS technique. It was observed that major soil characteristics such as, carbon, nitrogen, pH, exchangeable magnesium and calcium and CEC were reliably predicted ($r^2 > 0.76$) by NIRS PLS (fig. 7). Generally, the cross validation models with high regression (r^2) values such as those obtained with N, CEC and Ex Ca also yielded large validation r^2 values ($r^2 > 0.76$). That these properties were highly correlated with carbon ($r^2 = 0.97$), may explain this high prediction accuracy attained using the validation data set. However, pH and exchangeable Mg were more accurately predicted by NIRS-PLS (0.95 and 0.94 respectively) than would be expected based on their correlations with carbon. Correlation coefficients of extractable K was very low ($r^2 = 0.35$).

The accuracy of prediction for total carbon, total nitrogen, exchangeable calcium and CEC are comparable to those already reported (Chang *et al.,* 2001; Shepherd and Walsh, 2002). For instance, for 62 validation samples of total carbon ranging between 9 – 73 gkg^{-1}, r^2 was 0.91, bias was -0.7 gkg^{-1} with RMSE of 3.77 gkg^{-1}. Shepherd and Walsh (2002) reported r^2 of 0.8, bias was -0.1 gkg^{-1} and RMSE of 3.1 gkg^{-1} for total carbon values between 2.3 -55 for a sample size of 337 soils. Similarly, predicted and measured values for total N $r^2 = 0.9$; bias = -0.01 gkg^{-1}; RMSE = 0.49 gkg^{-1}.

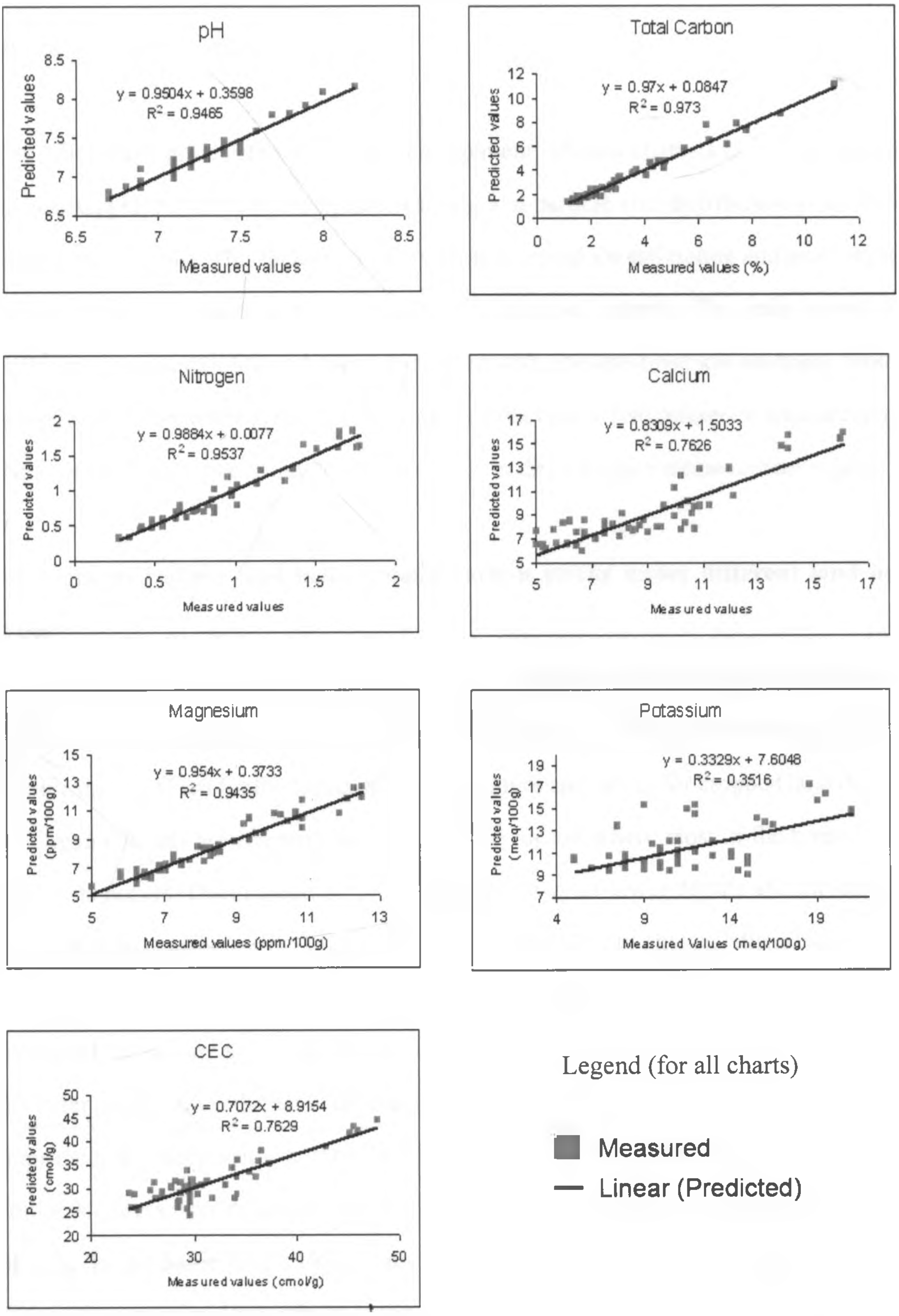

Figure 7. Regression of soil properties measured by standard laboratory procedures and predicted by NIRS – PLS technique.

Confalonieri *et al.* (2001) reported that NIRS can be used to determine soil nitrogen and carbon content accurately.

The NIRS gave good estimates of the management induced changes in soil properties namely C, pH, N, CEC, exchangeable Ca, Mg and particle size distribution as a result of land conversions. This technique offers great potential for estimating and monitoring variations in soil constituents under different land use systems. The main merits of NIRS are its repeatability and speed compared with conventional soil analyses. NIRS is reputed to give better precision and accuracy than the actual reference measurement (Naes *et al.*, 2002), particularly when the error in the reference measurement is large.

4.4 Estimated above and belowground carbon stocks under different land use types.

Figure 8 shows the relationship of belowground carbon stocks with depth. The carbon stocks declined with depth with high levels obtained in 0 – 20 cm depth followed by a decline up to 100 cm for the forest, 80 cm for pasture and 60 cm for cropped land (figure 8). Organic inputs are primarily deposited in the topsoil where most of the turnover of fine roots occurs. Decomposition processes are slower at lower depths and the carbon stocks that do exist below the topsoil are better protected from physical disturbance such as soil tillage, and are likely to change more slowly after land use change. Even though the carbon stocks are similar in the top and subsoil layers, the potential for C storage in the latter greatly exceeds that in the former. However, the assessment of land use change impacts on the deep soil C storage is difficult due to the large inherent variability of C contents, unless well-designed experiments are conducted for a sufficiently long period of time (Kotto-Same *et al.*, 1997). This could be due to a lot of microbial activities in the top soil and very low activity in the deeper horizons.

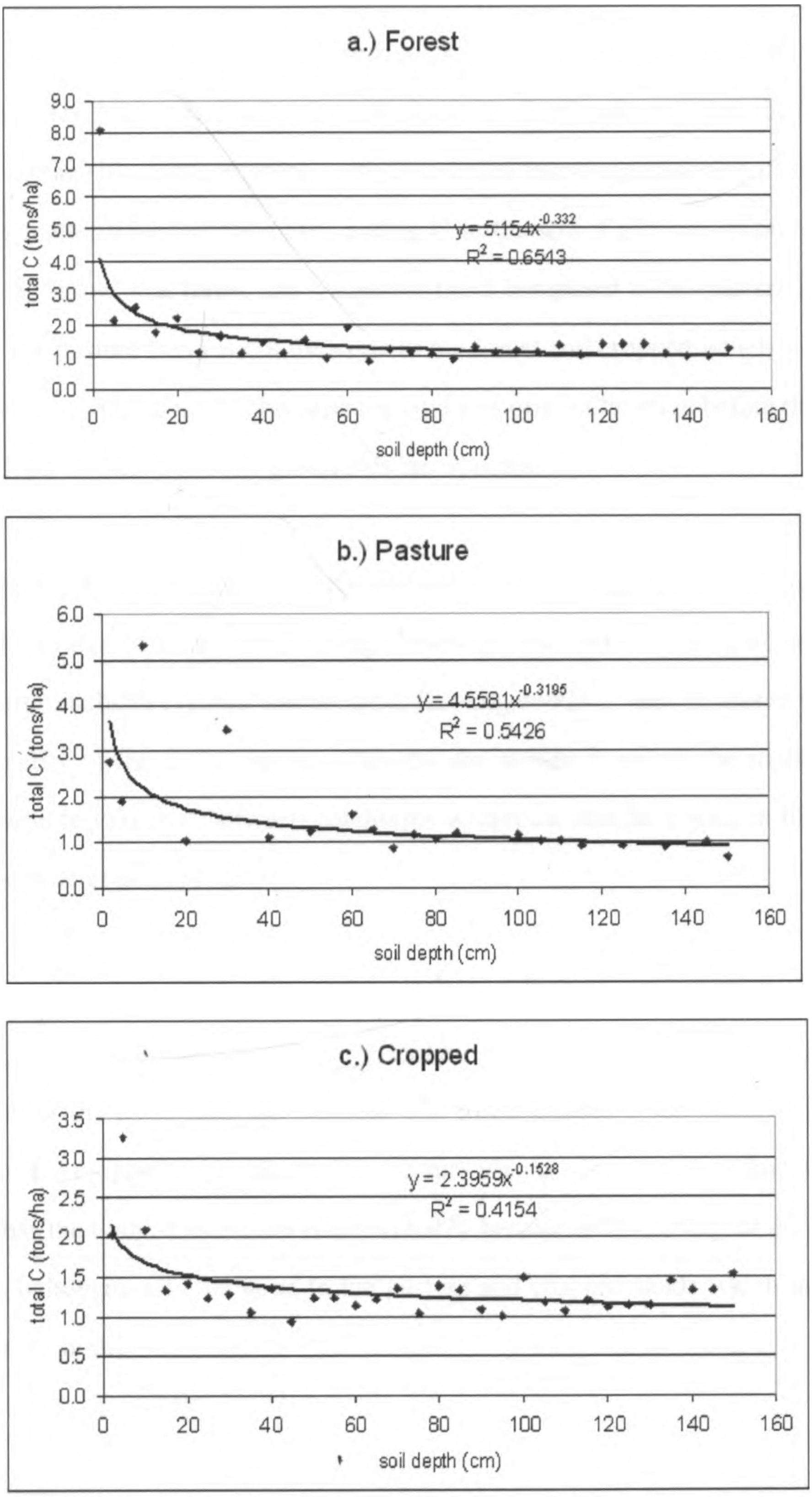

Figure 8. Regression analyses of belowground C stocks and depth.

The forest soils had the highest amounts of carbon in the surface layer (0-20cm) which was 8.1 gkg^{-1} followed by the pasture layer 5.3 gkg^{-1} and the cultivated layer contained 3.2 gkg^{-1}. The high values of carbon stocks under the forest may have been due to the higher surface litter and decaying roots from felled forest vegetation contributing to soil organic matter which decomposes adding C to the soil. Carbon distribution by depth (figure 8) shows that forest had a superior trend compared to the pasture and cropped lands. The pasture trend is not the same as the forest and cropped which tend to have a similar shape. This could be because the land was originally range before the forest was introduced then cropped and grasslands incorporated.

Soil depth varies considerably between sites. The rooting depth of trees may have been the source of most soil C storage below ground (besides the gradual downward transport of soluble organic compounds). Rooting depth of trees is related to the length and severity of the dry season as when the dry season is severe the roots are deeper. Considerable variation between conditions within the humid tropics is likely to exist (Kotto-Same *et al.*, 1997).

Figure 9 shows the very first decline of carbon stocks with depth (the exponential phase). The relationship between the carbon stocks with depth has a unique pattern for each of the three land use systems. The forest exudes a very sharp initial decline whereas the cropped is gradual and the pasture is so gradual that it is almost stable. The forest has the highest inflection point probably because of the ability of the forest to fix carbon belowground compared to the pasture and cropped lands which have shallow inflection points.

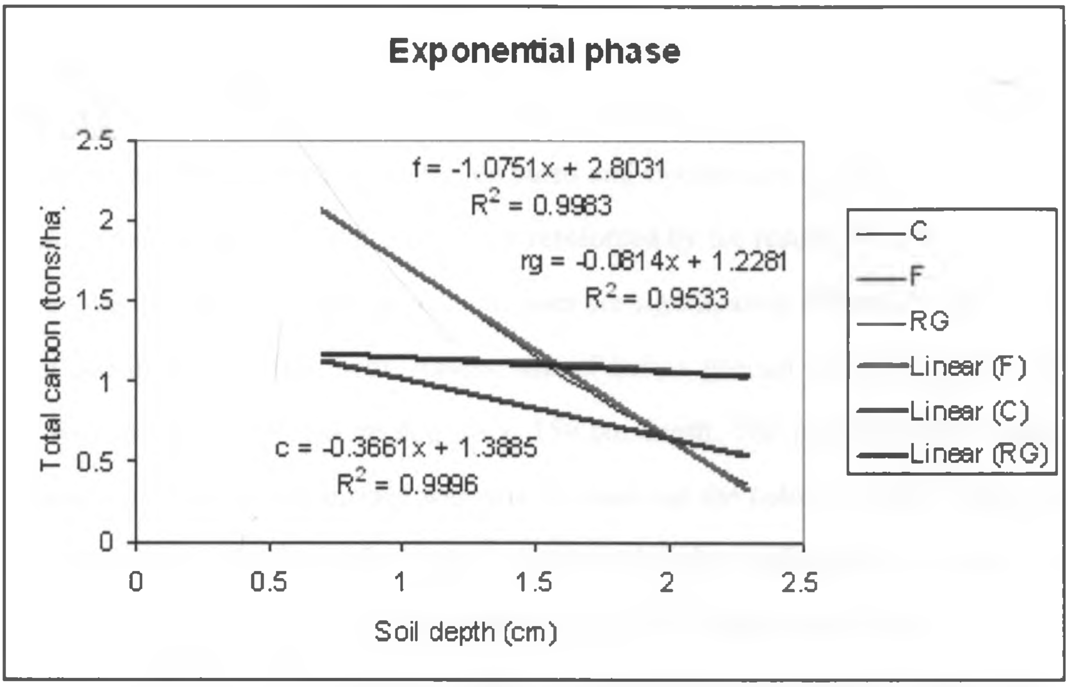

Figure 9 Regression analyses between the carbon stocks with depth where c=cultivated, f=forest and rg=range.

The forest has a regression (r^2) of 0.99, the cropped has an r^2 of 0.99 and the pasture has an r^2 of 0.95. This could be attributed to the fact that there is very low carbon input relative to the starting point carbon at the surface. There is a lot of carbon on the surface but at depth, the carbon additions are not very apparent. This is especially so in the forest where the surface litter is a lot compared to the pasture and cropped land use systems. Forest has a closed nutrient cycle system with fewer losses compared to the open systems of pasture and cropped lands. Within this cycle (Nair, 1990), nutrients are stored in the biomass and topsoil through a constant cycle of transfer of nutrients and biological processes of litter fall, root decomposition and plant uptake.

The cumulative carbon is highest in the forest (48.4 g kg^{-1}) land use system compared to the cropped (41.8 g kg^{-1}) and pasture (31.7 g kg^{-1}) lands (appendix vi). This could be because the pasture land soil profiles were relatively shallow with most of the profile pits going up to 90 cm as opposed to 150 cm in both the forest and cropped land use systems. The gentleness of the accumulation of carbon stocks as one goes down the profile is due to the conversions from forest to agriculture (Kotto-Same *et al.*, 1997). Deforestation results in massive loss of forest system carbon. However, simple soil conservation practices such as controlled burns and erosion control can result in greater system carbon storage. This observation is reinforced by the results presented in table 4 where total C contents of the various land uses are significantly affected by the amount of surface litter covering them. Assessment of below ground carbon dynamics was such that soil profile pits were dug up to 150 cm depth. The profiles had a common characteristic that the soil surface was dark in color and the color became lighter as the profile went deep. The forest profile had a larger and darker surface then the pasture and the cropped land use system. This could be due to the thick surface litter on the forest floor compared to the very low or no litter on the pasture and cropped land use systems. The rooting system of trees in the forest was also dense compared to the bushes in the pasture and the crops in the cultivated land use systems

Total estimated above and below ground carbon stocks

Table 4 shows the total estimated carbon stocks in forested, cropped and pastureland systems. The total carbon stocks were significantly different ($P \leq 0.001$) in cropped and pasture compared to forest system (appendix iii). It was higher in the forest than the cropped and the pasture land. A decline in carbon stocks of 45.6% and 47.4% in the pasture land and cropped land respectively were observed relative to the forest. The original organic carbon in the soil lost due to conversion to agricultural management could be attributed to reduced inputs of crop plant residue, increased mineralization rates of crop plant residue returned, and tillage effects that decrease the amount of physical protection for soil organic carbon (Davidson and Ackerman, 1993; Post and Mann, 1990). Unfortunately, land use systems which minimize system carbon loss such as conventional tillage, agroforestry to mention but two, are no longer favored by farmers.

Table 4. Total estimated above and below ground carbon stocks under forested, cropped and pastureland systems

Carbon stocks	Forest		Cropped		Pastureland	
	Mean	SE	Mean	SE	Mean	SE
Above C (tons/ha)	4.3×10^9	24.5	1.5×10^9	20.3	1.7×10^9	19.2
Below C (tons/ha)	1.4×10^9	44.6	1.2×10^9	11.4	0.9×10^9	11.5
Total	5.7×10^9		2.7×10^9		2.6×10^9	

The carbon pool most vulnerable to loss when forests are converted to agriculture is aboveground biomass. Kauffman *et al.* (1995) in Brazil, Kotto-Same *et al.* (1997) in the Amazonian forest, Saldarriaga *et al.* (1988) in the upper Rio Negro of South America, Ramakrishnan (1983) in North eastern India and Ewel *et al.* (1981) in Costa Rica all agreed that the most vulnerable carbon pool is the aboveground biomass. The susceptibility of tree biomass to destruction is inferred in the latter two studies because these commenced not with the original forest and subsequent cropland but at the time of land abandonment following slash-and-burn. The loss of soil carbon is small by comparison with aboveground biomass but not unimportant (Nye and Greenland, 1960). Decaying roots from felled forest vegetation may contribute to soil organic matter following conversion to agriculture and subsequent fallows. Andriesse and Schelhaas (1987) reported soil carbon losses following slash-and-burn in the top 75 cm of a soil. Woomer *et al.* (1997) estimated carbon losses of 8 t ha $^{-1}$ yr^{1} due to slash-and-burn at coastal sand dune forests in Mozambique.

5.0. CONCLUSION AND RECOMMENDATIONS

5.1 Conclusion

This study has shown that conversion of forests to agricultural use affects the soil properties. The average total soil C and N were higher under forest than in cropped or pasture systems. The CEC was lower in cropped and pasture lands compared to the forest thus the exchangeable Ca and Mg were higher in forest compared to pasture and cropped lands.

Near infrared reflectance spectroscopy was sensitive and gave good estimates of management induced changes in soil properties. Therefore it offers great potential for estimating and monitoring variations in these constituents under different land use and management practices, and could be used as a rapid a technique for estimating chemical characteristics of soils.

The carbon stocks were especially affected in that the carbon sinks were reduced in the converted land use systems and a decline in carbon stocks of 45.6% and 47.4% in the pasture and cropped land was observed. This study showed that the forest had the most total carbon content compared to the pastureland and cropped land use systems. Soil carbon contents generally decreased with depth.

Recommendations and guidelines for further research.

The strategies to increase the soil carbon pool should include soil restoration and woodland regeneration, no-till farming, use of cover crops, nutrient management, manuring application, water conservation and harvesting, and agroforestry practices. In the case of Mt. Marsabit, agroforestry is the most appropriate in that other combination of trees and crops and forages may be beneficial to sustainable use of soil water resources and carbon sequestration in site specific situations.

Developments of cheaper, portable spectrometers with flexible software and easier calibration methods should be encouraged and are expected. The technology should be used increasingly in a wide range of soil studies and surveys. The spectrometers should become standard equipment in the laboratories. Spectral libraries should be widely used to study the effects of land use changes on soil quality and to establish baselines for carbon offset projects.

Further research that needs to be carried out should include the greenhouse gas (such as carbondioxide, carbonmonoxide, methane, nitrous oxide) emissions, and the potential of improved vegetation cover improvement on carbon sequestration in Mt. Marsabit ecosystem.

6.0. REFERENCES

AGREF (2002). Development and implementation of a sustainable resource management plan for Marsabit Mountain and its associated watershed.

Albrecht, A. and Kandji S.T. (2003). Carbon sequestration in tropical agroforestry systems. Agriculture, Ecosystems & Environment.

Analytical Spectral Devices Inc. (1997). FieldSpecTM User's guide. Analytical Spectral Devices Inc., Boulder CO.

Anderson, J.M. and Ingram, J.S.I. (Eds). (1990). Tropical soil biology and fertility. - A handbook of methods. CAB international.

Andriesse, J.P., Schelhaas, R.M. (1987). A monitoring study on nutrient cycles in soils used for shifting cultivation under various climatic conditions in tropical Asia. III. The effects of land clearing through burning on fertility level. Agric. Ecosyst. Environ. 19: 311-332.

Batjes N.H. and Sombroek W.G. (1997). Possibilities for carbon sequestration in tropical and subtropical soils. Global Change Biol. 3: 161–173.

Ben-Dor, E., Irons, J.R. and Epema, G.F. (1999). Soil reflectance. pp. 11-188. In Rencz (ed.) Remote sensing for earth sciences: Manual of remote sensing. Vol 3. John Wiley & Sons, New York.

Blake, G.R. (1965). Bulk density. Black, C.A. (ed.) In: Methods of soil analysis pp 374 – 390.

Boahene, K. (1998). The challenge of deforestation in tropical Africa reflections on its principle causes, consequences and solutions. Land degradation and development. 9: 247-258.

Box, G.E., and Cox, D.R. (1995). An analysis of transformation with discussion. J.R. Stat. Soc. B26: 211-250

Brady, N.C. (1996). Alternatives to slash and burn: a global perspective. Agriculture Ecosystems and Environments 58: 3-11.

Brown, S. (1997). Estimating biomass and biomass change of tropical forests, a primer. FAO Forestry paper 134, FAO, Rome.

Brown, S., and Lugo, A. E. (1984). Biomass of tropical forests: A new estimate based on volumes. Science 223: 1290–1293.

Brown, S., and Lugo, A. E. (1992). Aboveground biomass estimates for tropical moist forests of the Brazilian Amazon. Interciencia 17: 8–18.

Brown, S., and Lugo, A. E. (1993). Management of tropical soils as sinks or sources of atmospheric carbon. Plant and Soil 149: 27–41.

Brown, S., Sathaye, J., Cannel, M. and Kauppi, P.E. (1996). Mitigation of carbon emissions to the atmosphere by forest management. Commonwealth Forestry Review 75: 80-91.

CAMO Inc. (1998). The Unscrambler User Manual. CAMO Inc, Corvallis, OR.

Cerri, C.C., Volkov, B., Andreaux, F. (1991). Nature and behaviour of organic matter in soils under natural forest and after deforestation, burning and cultivation near Manaus. Forestry Ecol Mgmt 38: 247-257.

Chang, C.W., Laird, D.A., Mausbach, M.J. and Hurburgh, Jr C.R. (2001). Near-Infrared Reflectance Spectroscopy-Principal Components Regression Analysis of Soil Properties. Soil Science Society of America Journal 65:480-490.

Christensen, B.T. (1992). Physical fractionation of soil and organic matter in primary size and density separates. Advances in Soil Sciences 20: 1-90.

Clark, D. (2002). Are tropical forests an important carbon sink? Reanalysis of the long-term plot data. Ecological Applications 12: 3–7.

Confalonieri, M., Fornasier, F., Ursino, A., Boccardi, F., Pintus, B. and Odoardi, M. (2001). The potential of near infrared reflectance spectroscopy as a tool for the agricultural soils. Journal of Near Infrared Spectroscopy 9: 123-131.

Dalal, R.C. and Henry, R.J. 1986. Simultaneous determination of moisture, organic carbon and total nitrogen by near infrared reflectance spectrometry. Soil science society of America journal, 50:120-123.

Davidson E.A., Keller M., Erickson H.E., Verchot L.V. and Veldkamp E. (2000). Testing a conceptual model of soil emissions of nitrous and nitric oxide. BioScience 50: 667–680.

Davidson, E. A., and Ackerman, I. L. (1993). Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20: 161–193.

Dean, A. M., Thomas, E. R. and David, T. L. (2004). Soil organic carbon content and composition of 130-year crop, pasture and forest land use managements. Global Change Biology 10:2: 170-181.

De Fries, R. S., Houghton, R. A., Hansen, M. C., Field, C. B., Skole, D. and Townshend, J. (2002). Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 90s. Proc. Nat. Acad. Sci. USA 99: 14256–14261.

Dessai, S. and Schipper, E.L. (2003). The Marrakech Accords to the Kyoto Protocol: analysis and future prospects. Global Environmental Change pp. 178 – 200.

Detwiler, R.P. (1986). Land use change and the global carbon cycle: The role of tropical soil. Biogeochemistry 2: 67–93.

Dudal, R. and Deckers, J. (1993). Soil organic matter in relation to soil productivity. In (K. Mulongoy and R. Merckx (eds.) Soil organic matter dynamics and sustainability of tropical agriculture. John Wiley and sons. Chichester.

Edwards, D. (2000). Introduction to graphical modeling. Springer-Verlag, Newyork.

Elliot, T.E. and Cambardella, C.A. (1991). Physical separation of soil organic matter. Agro. Ecosyst. Environ. 34: 407-419.

Elliot, T.E., Cambardella, C.A. and Cole, C.V. (1993). Modification of ecosystems processes by management and mediation of soil organic matter dynamics. In: (K.

Mulongoy and R. Merckx (Eds). Soil organic matter dynamics and sustainability of tropical agriculture. John Wiley and sons. Chichester, U.K. pp. 257-268.

Eve, M. D., Sperow, M., Paustian, K. and Follett, R. F. (2002). National-scale estimation of changes in soil carbon stocks on agricultural lands. Environmental Pollution 116: 431–438.

Ewel, J., Buresh, C., Brown, B., Price, N., Raich, J. (1981). Slash and burn impacts on a Costa Rican wet forest site. Ecology 62: 816-829.

Fearn, T. (2000). Savitzky-Golay filters. Near Infrared Reflectance (NIR) News. 11(6): 14-15.

Fearnside, P. M., and Laurance, W. F. (2003). Comment on Determination of deforestation rates of the world's humid tropical forests. Science 299: 1015a–1015b.

Fearnside, P. M. (1992). Forest biomass in Brazilian Amazonia: Comments on the estimate by Brown and Lugo. Interciencia 17:19–27.

Feller C. (1993). Organic inputs, soil organic matter and functional soil organic compartments in low-activity clay soils in tropical zones. In: (Mulongoy K. and Merckx R. eds.) Soil Organic Matter Dynamics and Sustainability of Tropical Agriculture. IITA/KU Leuven, John Wiley, New York, USA, pp. 77–88.

Fisher M.J., Rao I.M., Ayarza M.A., Lascano C.E., Sanz J.I., Thomas R.J. and Vera R.R. (1994). Carbon storage by introduced deep-rooted grasses in the South American Savannas. Nature 371: 236–238.

Food and agriculture organization (FAO), (2001). Carbon sequestration in soils: Proposals for land management in arid and tropical area. FAO, Rome, Italy.

Food and agriculture organization (FAO), Forest Resources Assessment (1990). Global Synthesis, FAO Forestry Paper 124, FAO, Rome, Italy, 1995.

Food for the hungry international, Kenya (FHI). (2001). Cooperating to save Marsabit Mountain. Report of an FHI sponsored workshop to develop strategies for saving Marsabit Mountain through close inter-agency collaboration.

Galvao, L.S., Pizzaro, M.A. and Epiphanio, J.C.N. (2001). Variations in reflectance of tropical soils: Spectral-chemical composition relationships from AVIRIS data. Remote Sensing Environment 75:245-255.

Goodale, C. L., Apps, M. J., Birdsey, R. A., Field, C. B., Heath, L. S., Houghton, R. A., Jenkins, J. C., Kohlmaier, G. H., Kurz, W., Liu, S., Nabuurs, G.J., Nilsson, S. and Shvidenko, A. Z. (2002). Forest carbon sinks in the Northern Hemisphere. Ecological Applications 12: 891–899.

Government of Kenya (GoK). (2001). National development plan, 2002 – 2008. Nairobi, Ministry of Finance and Planning.

Greenland, D.J., Wold, A. and Adams, D. (1992). Organic matter dynamics in soils of the tropics-From myth to complex reality. In: Myths and Science of soils in the Tropics (Eds. Lal, R. and P.A. Sanchez), pp. 17-33. SSSA Special Publication no. 29. American Society of Agronomy, Madison.

Gregorich E.G., Greer K.J., Anderson D.W. and Liang B.C. (1998). Carbon distribution and losses: erosion and deposition effects. Soil Tillage Res. 47: 291–302.

Guo, L.B. and Gifford, R.M. (2002). Soil carbon stocks and land use change: a meta analysis. Global Change Biol. 8: 345–360.

Hairiah K., Sitompul S.M., van Noordwijk M. and Palm C.A. (2001). Carbon stocks of tropical land use systems as part of the global carbon balance: effects of forest conversion and options for clean development activities. Alternatives to slash-and-burn _ASB_ Lecture Note 4. ICRAF, Bogor, Indonesia.

Harrison, K. G., Post, W. M. and Richter, D. D. (1995). Soil carbon turnover in a recovering temperate forest. Global Biogeochemical Cycles 9: 449–4354.

Hill, J. and Schutt. (2000). Mapping complex patterns of erosion and stability in dry Mediterranean ecosystems. Remote Sensing Environment 75:557-569.

Hooker, T. D. and Compton, J. E. (2003). Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. Ecological Applications 13: 299–313.

Houghton, R. A. (1994). The worldwide extent of land-use change. Bio- Science 44: 305–313.

Houghton, R. A., Hackler, J. L. and Lawrence, K. T. (1999). The U.S. carbon budget: contributions from land-use change. Science 285: 574–578.

Houghton, R. A., Hackler, J. L., Lawrence, K. T. and Brown, S. (2001). The spatial distribution of forest biomass in the Brazilian Amazon: A comparison of estimates. Global Change Biology 7: 731–746.

Houghton, R.A. (1999). The annual net flux of carbon to the atmosphere from changes in land use. Tellus 51: 298-313.

Hurtt, G. C., Pacala, S. W., Moorcroft, P. R., Caspersen, J., Shevlaikova, E., Houghton, R. A. and Moore III, B. (2002). Projecting the future of the U.S. Carbon Sink. Proc. Nat. Acad. Sci. USA 99: 1389–1394.

Intergovernmental Panel on Climate Change (IPCC) (1995). Guidelines for national greenhouse gas inventories. Reference Manual vol. 4. IPCC, Bracknell, UK.

Intergovernmental Panel on Climate Change (IPCC). (2000). Land use, land-use change, and forestry. Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Intergovernmental Panel on Climate Change (IPCC). (2002). Jackson, R. B., J. L. Banner, E. G. Jobbágy, W. T. Pockman, and D. H. Wall. Ecosystem carbon loss with woody plant invasion of grasslands. Nature 418: 623–626.

Janik, L.J., Merry, R.H. and Skjemstad, J.O. (1998). Can mid infrared diffuse reflectance analysis replace soil extractions? Australian Journal of Experimental Agriculture 38:681-696.

Janssens, I. A., Freibauer, A., Ciais, P., Smith, P., Nabuurs, G.J., Folberth, G., Schlamadinger, B., Hutjes, R. W. A., Ceulemans, R., Schulze, E.D., Valentini, R., and Dolman, A. J. (2003). Europe's terrestrial biosphere absorbs 7–12% of European anthropogenic CO_2 emissions. Science 300: 1538–1542. John Wiley & Sons. 21

Johnson, C. M., Zarin, D. J. and Johnson, A. H. (2000). Post-disturbance above-ground biomass accumulation in global secondary forests. Ecology 81: 1395–1401.

Juo, A.S.R. and Franzluebbers, K. (2003). Tropical soils properties and management for sustainable agriculture. Oxford University Press.

Juo, A.S.R., Lal, R. (1979). Nutrient profile in a tropical alfisol under conventional no tillage systems. Soil Sci. 127: 168-217.

Kauffman, J.B., Cummings, D.L., Ward, D.C., Babbitt, R. (1995). Fire in the Brazilian Amazon: 1. Biomass, nutrient pools, and losses in slashed forests. Oecologia 104: 397-408.

Kapkiyai, J.J., Karanja, N.K., Woomer, P.L. and Quereshi, J.N. (1998). Soil organic carbon fractions in a long-term experiment and the potential for their use as a diagnostic assay in highland farming systems in central Kenya. African Journal of Crop Science 6: 19-28.

Kenya forests working group (KFWG). (2001). Mt. Marsabit status report pp.87-109

Ketterings, Q.M., Coe, R., van Noordwijk, M., Ambagau, Y. and Palm, C.A. (2001). Reducing uncertainty'in the use of allometric biomass equations for predicting above-ground tree biomass in mixed secondary forests. Forest Ecology and Management 120: 199-209.

Kotto-Same, J., Woomer, P.L., Appolinaire, M. and Louis, Z. (1997). Carbon dynamics in slash-and-burn agriculture and land use alternatives of the humid forest zone in Cameroon. Agriculture, Ecosystems & Environment 65:245- 256.

Kozlowski, T. T. (2002). Physiological ecology of natural regeneration of harvested and disturbed forest stands: implications for forest management. Forest Ecology and Management 158: 195–221.

Kumada, K., Ohta, S., Arai, S., Kitamura, M., Imai, K. (1985). Changes in soil organic matter and nitrogen fertility during slash-and-burn and cropping phases in experimental shifting cultivation. Soil Sci. Plant Nutr. 31: 611-623.

Lal R. (2004). Soil Quality and Soil Erosion. CRC Press, Boca Rason, Florida, USA, 352 pp. 309-319.

Lal, R, Kimble, J.M., Follet, R.F., Stewart, B.A. (Eds.). (1998). Soil processes and the carbon cycle. CRC Press, Boca Raton, FL, 609 pp. 523-560.

Lal, R. and Sanchez, P.A. (1992). Myths and Science of Soils in the Tropics. SSSA Special Publication no. 29. American Society of Agronomy, Madison.

Lal, R., Wilson, G.F., Okigbo, B.N. (1979). Changes in properties of an alfisol produced by various crop covers. Soil Sci. 127 (6): 377-382.

Lavelle, P., Blanchart, E., Martin, A., Spain, A.V. and Martin, S. (1992). Impact of soil fauna on the properties of soils in the humid tropics. In: Myths and Science of Soils of the Tropics. SSS A Special Publication 29, pp. 157-185. Soil Sci. Soc. Am., Madison

Mann, L. K. (1985). A regional comparison of carbon in cultivated and uncultivated alfisols and mollisols in the central United States. Geoderma 36: 241–253.

Mann, L. K. (1986). Changes in soil carbon storage after cultivation. Soil Science 142: 279–288.

Martens, H., and Naes, T. (1989). Multivariate calibration. Wiley, Chichester, UK.

Mathew, J.A. (1976). Multivariate statistical classification of soil spectra. Remote Sensing of Environment 57:108-118.

Mburu M. (1996). MDP/GTZ Farming systems survey for Central division –Marsabit District.

McDonagh, J.F, Thomsen, T.B. and Magid, J. (2001). Soil Organic matter decline and compositional change associated with cereal cropping in southern Tanzania. Land Degradation and Development 12: 13-26.

Mejiboom, F.W., Hassink, J. and Van Noordwijk M. (1995). Density fractionation of soil macro organic matter using silica suspensions. Soil Biology and Biochemistry 27: 1109-1111. Methane, Nitrogen Oxides and Halomethanes (Eds. J.E. Rogers and W.B. Whitman.) American Society for Microbiology, Washington, pp. 219-235.

Moore, J.C., Walter, D.E. and Hunt, H.W. (1988). Arthropod regulation of micro and meso biota in belowground detrital food webs. Annu., Rev., Entomol. 33: 419 – 439.

Moss, S.M. (1993). Organic matter in tropical soils: current conditions, concerns and prospects for conservation. Progress in Physical Geography 17(3):265-305.

Murty, D., Kirschbaum, M. F., McMurtrie, R. E. and McGilvray, H. (2002). Does conversion of forest to agricultural land change soil carbon and nitrogen? A review of the literature. Global Change Biology 8: 105–123.

Mutuo P.K. (2003). Potential of improved tropical legume fallows and zero tillage practices for soil organic carbon sequestration. Ph.D. Thesis, University of London.

Naes, T., Isacksson, T., Fearn, T. and Davies, T. (2002). A user-friendly guide to Multivariate Calibration and Classification. NIR Publications Chichester, UK.

Nair, P.K.R. (1990). Prospects for agroforestry in the tropics. World Bank Technical Paper No. 131, World Bank, Washington, DC, USA.

Nye, P. H and Greenland, D.J. (1960). Changes in soil after clearing tropical forest. Plant and Soil 21, no. 1.

Okalebo, J.R., Gathua, K.W. and Woomer, P.L. (2002). Laboratory methods of soil and plant analysis: A working manual. Second edition.

Osbert, J. S., John, C., Beverly, E. L. and Vernon W. (2004). Dynamics of carbon stocks in soils and detritus across chronosequences of different forest types in the Pacific Northwest, USA. Global Change Biology 10: 1470-1481.

Osher, L. J., Matson, P. A. and Amundson, R. (2003). Effect of land use change on soil carbon in Hawaii. Biogeochemistry 65: 213–232.

Page, A.I., Miiller, R.H. and Keeney, D.R. (Eds.). (1982). Methods of soil analysis.

Palacios-Oureta, A. and Ustin, S.L. (1998). Remote sensing of soil properties in santa Monica mountains: I. spectral analysis. Remote Sensing Environment 65: 170-183.

Palm, C.A., Woomer P.L., Alegre J.C., Arevalo L., Castilla C., and Cordeiro D.G. (2000). Carbon sequestration and trace gas emissions in slash-and-burn and alternative land-uses in the humid tropics. ASB Climate Change Working Group Final Report; Phase II. Nairobi, Kenya.

Palm, C. A., Swift, M. J.and Woomer, P. L. (1996). Soil biological dynamics in slash-and-burn agriculture. Agric. Ecosyst. Environ. 58: 61-74.

Palm, C.A., Alegre, J.C., Arevalo, L., Mutuo, P.K., Mosier, A.R. and Coe, R. (2002). Nitrous oxide and methane fluxes in six different land use systems in the Peruvian Amazon. Global Biogeochem. Cycles 16: 107316: 1073.

Parton, R. Kelly, M. Sykes, R. Neilson, and B. Rizzo. (2000). Contribution of increasing CO_2 and climate to carbon storage by ecosystems in the United States. Science 287: 2004–2006.

Paustian K., Andr'en O., Janzen H.H., Lal R., Smith P., and Tian G. (1997). Agricultural soils as a sink to mitigate CO_2 emissions. Soil Use Manage. 13: 1–5.

Post, W.M and Kwon, K.C. (2000). Soil carbon sequestration and land -use change: processes and potential. Global Change Biology 6: 317-327

Ramakrishnan, P.S., Toky, O.P. (1981). Soil nutrient status of hill agro-ecosystems and recovery pattern after slash and burn agriculture (jhum) in north-east India. Plant and Soil 60: 41-64.

Saldarriaga, J.G., West, D.C., Tharp, M.L., Uhl, C. (1988). Long term chronosequence of forest succession in the upper Rio Negro of Columbia and Venezuela. J. Ecol. 76: 938-958.

Sanchez, P.A. (1976). Properties and Management of Soils in the Tropics. New York, 20.

Sanchez, P.A. (2000). Linking climate change research with food security and poverty reduction in the tropics. Agriculture, Ecosystems & Environment 82:371-383.

Sanchez, P.A., Villachica, J.H. and Bandy, D. E. (1983). Soil fertility dynamics after clearing a tropical rainforest in Peru. Soil Science Society of America Journal 47: 1171-1178.

Sanchez. P.A., Palm, C.A., Szott, L.T., Cuevas, E. and Lal, R. (1989). Organic input management in tropical agro ecosystems. In dynamics of soil organic matter in tropical ecosystems (D.C. Coleman, J.M. Oades, and G. Uehara, Eds.) pp. 125-152, University of Hawaii press Honolulu.

SAS user's guide (2000). SAS/STAT User guide Version 8.02, Cary NC.SAS Institute Inc.

Schlesinger, W. H. (1986). Changes in soil carbon storage and associated properties with disturbance and recovery in The Changing Carbon Cycle: A Global Analysis, (edited by J. R. Trabalka and D. E. Reichle), pp. 194–220, Springer-Verlag, New York.

Scholes, M.C., Martin, R., Scholes, R.J., Parsons, D. and Winstead, E. (1997). NO and N_2O emissions from savanna soils following the first simulated rains of the season. Nutrient Cycling in Agroecosystems 48:115-122.

Scholes, R. J., and Archer, S. R. (1997). Tree-grass interactions in savannas. Ann. Rev. Ecol. Syst. 28: 517–544.

Shepherd, K.D. and Walsh, M.G. (2000). Diffuse reflectance spectrometry in the NARL long term experiments-preliminary results. Natural resource problems, priorities and policies programme working paper 2000-3. International Centre for Research in Agroforestry, Nairobi, Kenya.

Shepherd, K.D. and Walsh, M.G. (2002). Development of reflectance spectral libraries for characterization of soil properties. Soil Science Society of America Journal 66: 988-998.

Six, J., Merckx, R., Kimpe, K., Paustian, K., Elliot, E.T. (2000). A re-evaluation of the enriched labile soil organic matter fraction. Eur. J. Soil Sci. 51(2): 283-293.

Smith, S. V., Renwick, W. H., Buddemeier, R. W. and Crossland, C. J. (2001). Budgets of soil erosion and deposition for sediments and sedimentary organic carbon across the conterminous United States. Global Biogeochemical Cycles 15: 697–707.

Smith, W. N., Desjardins, R. L. and Pattey, E. (2000). The net flux of carbon from agricultural soils in Canada 1970–2010. Global Change Biology 6: 557–568.

Sobania, N.W. (1979). Background history of the MT. Kulal region of Kenya (IPAL Technical report number A-2).

Solomon, D., Fritzsche, F., Tekalign, M., Lehmann, J. and Zech, W. (2002). Soil organic matter composition in the subhumid Ethiopian highlands as influenced by deforestation and agricultural management. Soil Science Society of America Journal 66:68-82.

Stallard, R. F. (1998). Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. Global Biogeochemical Cycles 12: 231–257.

Stoner, E.R. and Baumgardner, M.F. (1981). Characteristic variations in reflectance of surface soils. Soil Science Society of America Journal 45:1161-1165.

Swinnen, J. (1994). Rhizodeposition and turnover of root-derived material in barley and wheat under conventional and integrated management. Agric. Ecosys. Environ. 51: 115 – 128.

Uhl, C., Buschbacher, R. and Serrao, E. A. S. (1988). Abandoned pastures in eastern Amazonia. I. Patterns of plant succession. Journal of Ecology 76: 663–681.

United States Department of Agriculture Natural Resources Conservation Service (USDA-NRCS). (2000). Growing carbon: A new crop that helps agricultural producers and the climate too. SSSA 64: 60-75.

Van Noordwjik, M., Cerri, C. Woomer, P., Nugroho, K. and Bernoux, M. (1997). Soil carbon dynamics in the humid tropical forest zone. Geoderma 79:187-225.

Van Noordwjik, M., Spek, L.Y. and De Wlligen, P. (1994). Proximal root diameter as predictor of total root system size for fractal branching models. I. Theory. Plant and soil 164: 107 – 118.

Warui (2000). Environmental and socio-economic impact charcoal production on the mount Marsabit area-pp. 12-30.

Woomer, P. L. and Palm, C. A. (1998). An approach to estimating system carbon stocks in tropical forests and associated land uses. Commonwealth Forestry Review 77: 181-190

Woomer, P.L, Martin, A., Albrecht, A., Resck, D.V.S. and Scharpenseel, H.W. (1994). The importance and management of soil organic matter in the tropics. In (Woomer, P.L and Swift, M.J. (eds) The Biological Management of Tropical Soil Fertility. Chichester. John Wiley & Sons.

Woomer, P.L. and Swift, M.J. (1993). Organic matter and sustainability of agricultural systems: Definitions and measurement. In (K. Mulongoy and R.Merckx (eds.) Soil Organic Matter Dynamics and Sustainability of Tropical Agriculture. Chichester. John Wiley & Sons.

Woomer, P.L., Palm, C.A., Qureshi, J., Okalebo, J.R., and Kottosame, J. (1997). Carbon sequestration and organic matter management by smallholder farmers in Africa. Advances in soil science.

Young A. (1997). Agroforestry for Soil Management, 2nd ed. CAB International, Wallingford, UK, 320 pp.

7.0 APPENDICES

APPENDIX I

Analysis of variance of different soil chemical properties at the three land use systems namely forest, cropped and pasturelands.

a.) pH

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	0.116359	0.058180	14.23	0.001
Residual	70	0.286217	0.004089		
Total	72	0.402576			

b.) Total carbon

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	15.3669	7.6835	43.15	0.001
Residual	70	12.4646	0.1781		
Total	72	27.8316			

c.) Total nitrogen

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	9.2902	4.6451	35.52	0.001
Residual	70	9.1552	0.1308		
Total	72	18.4454			

d.) CEC

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	0.77095	0.38547	13.95	0.001
Residual	70	1.93423	0.02763		
Total	72	2.70518			

e.) Magnesium

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	0.74243	0.37122	8.13	0.001
Residual	70	3.19784	0.04568		
Total	72	3.94027			

f.) Potassium

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	0.3950	0.1975	0.92	0.402
Residual	70	14.9777	0.2140		
Total	72	15.3727			

g.) Calcium

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	2.15854	1.07927	11.14	0.001
Residual	70	6.78294	0.09690		
Total	72	8.94148			

h.) Phosphorus

Source of variation	Degree of freedom	Sum of squares	mean squares	F value	P > F
Plot	2	22.3718	11.1859	13.46	0.001
Residual	71	58.9848	0.8308		
Total	73	81.3566			

APPENDIX II

Table: 2 Correlation coefficients (r^2) between measured and predicted for different soil properties for the validation data sets.

Soil property	n	R^2	Bias	RMSE
pH	57	0.95	0.0003	0.01
ExchCa Cmol/kg	57	0.76	0.0007	0.16
Exch K cmol/kg	56	0.35	0.01	0.25
Carbon g/kg	57	0.97	0.021	0.78
CEC cmol/kg	54	0.76	0.0008	0.09
Exch Mg cmol/kg	57	0.94	0.0009	0.18
Nitrogen g/kg	57	0.95	0.001	0.14

APPENDIX III

Nonlinear regression analysis for below ground carbon within the three land use systems.

Response variate: Carbon. Explanatory: Depth

a.) Land use system; Cropped

Summary of analysis

	Degree of freedom	Sum of squares	mean squares	F value	P > F
Regression	2	11.487	5.74335	151.83	0.001
Residual	28	1.059	0.03783		
Total	30	12.546	0.41819		

Percentage variance accounted for 91.0

Standard error of observations is estimated to be 0.194

Response variate: Carbon. Explanatory: Depth

b.) Land use system; Forest

Summary of analysis

	Degree of freedom	Sum of squares	mean squares	F value	P > F
Regression	2	44.637	22.3186	140.62	0.001
Residual	28	4.444	0.1587		
Total	30	49.081	1.6360		

Percentage variance accounted for 90.3

Standard error of observations is estimated to be 0.398

Response variate: Carbon. Explanatory: Depth

c.) Land use system; Pasture

Summary of analysis

	Degree of freedom	Sum of squares	mean squares	F value	P > F
Regression	2	11.51	5.7565	8.30	0.003
Residual	18	12.48	0.6932		
Total	20	23.99	1.996		

Percentage variance accounted for 42.2

Standard error of observations is estimated to be 0.833

APPENDIX IV

Above ground carbon estimates using the method diameter at breast height (DBH) data.

Species	D (cm)	Adjusted D (0.139 x D $^{2.32}$)	height (m)	Estimated Carbon (Mg/cm^3)
Strychnos heningsii	43	6160.9	5.5	856.3651
Strychnos heningsii	40	5209.3	5	724.0927
Rinorea covallaroides	56	11370.9	7	1580.555
Strychnos heningsii	40	5209.3	4.5	724.0927
Croton megalocarpus	108	52184.7	5	7253.673
Rinorea covallaroides	36	4079.6	4.9	567.0644
Strychnos heningsii	40	5209.3	4.5	724.0927
Dryptes geradii	17	715.6	1	99.4684
Strychnos heningsii	19	926.2	5	128.7418
Rinorea covallaroides	14	456.1	6	63.3979
Rinorea covallaroides	14	456.1	7	63.3979
Rinorea covallaroides	58	12335.4	>15	1714.621
Rinorea covallaroides	16	621.7	7	86.4163
Strychnos heningsii	6	63.9	3	8.8821
Rinorea covallaroides	34	3573	8	496.647
Dryptes geradii	10	208.9	5	29.0371
Strychnos heningsii	18	817	6	113.563
Rinorea covallaroides	10	208.9	8	29.0371
Croton megalocarpus	26	1917.5	10	266.5325
Dryptes geradii	22	1301.4	7	180.8946
Rinorea covallaroides	32	3104.2	7	431.4838
Strychnos heningsii	12	318.9	2.5	44.3271
Strychnos heningsii	10	208.9	3	29.0371
Strychnos heningsii	20	1043.3	6	145.0187
king	22	1301.4	5	180.8946
Croton megalocarpus	54	10450.9	12	1452.675
Rinorea covallaroides	24	1592.5	8	221.3575
Olea capensis	46	7204.4	9	1001.412
Croton megalocarpus	52	9574.8	12	1330.897
Strychnos heningsii	43	6160.9	4	856.3651
Rinorea covallaroides	29	2470.4	6	343.3856
Rinorea covallaroides	32	3104.2	8	431.4838
Rinorea covallaroides	6	63.9	2.3	8.8821
Dryptes geradii	10	208.9	2	29.0371
king	6	63.9	2.3	8.8821
Croton megalocarpus	32	3104.2	13	431.4838
Rinorea covallaroides	16	621.7	6.5	86.4163
Teclea nobilis	31	2883.8	9	400.8482
Croton megalocarpus	26	1917.5	10	266.5325

Species	D (cm)	Adjusted D (0.139 x D $^{2.32}$)	height (m)	Estimated Carbon (Mg/cm^3)
Strychnos heningsii	22	1301.4	8	180.8946
Rinorea covallaroides	52	9574.8	7	1330.897
Teclea nobilis	3	12.8	4	1.7792
Strychnos heningsii	17	715.6	7	99.4684
Croton megalocarpus	84	29129.2	14	4048.959
Rinorea covallaroides	7	91.3	4	12.6907
Rinorea covallaroides	12	318.9	6	44.3271
Rinorea covallaroides	46	7204.4	7	1001.412
Strychnos heningsii	54	10450.9	8	1452.675
Strychnos heningsii	15	535.2	4	74.3928
Dryptes geradii	12	318.9	2.5	44.3271
Canthium psyadrax	33	3333.9	10	463.4121
Rinorea covallaroides	21	1168.3	6	162.3937
Strychnos heningsii	6	63.9	3.5	8.8821
Croton megalocarpus	62	14399.5	12	2001.531
Croton megalocarpus	46	7204.4	13	1001.412
Strychnos heningsii	22	1301.4	7	180.8946
Strychnos heningsii	11	260.6	4	36.2234
Rinorea covallaroides	26	1917.5	11	266.5325
Canthium psyadrax	65	16067.9	10	2233.438
Olea capensis	42	5833.6	12	810.8704
Rinorea covallaroides	17	715.6	6	99.4684
Rinorea covallaroides	24	1592.5	8	221.3575
Strychnos heningsii	13	384	4	53.376
Rinorea covallaroides	21	1168.3	6	162.3937
Croton megalocarpus	46	7204.4	10	1001.412
Rinorea covallaroides	25	1750.7	7	243.3473
Strychnos heningsii	23	1442.8	7	200.5492
Rinorea covallaroides	42	5833.6	6	810.8704
Canthium psyadrax	18	817	9	113.563
Rinorea covallaroides	39	4912.1	8	682.7819
Teclea nobilis	32	3104.2	6	431.4838
Strychnos heningsii	24	1592.5	5	221.3575
Canthium psyadrax	24	1592.5	4	221.3575
Strychnos heningsii	14	456.1	5	63.3979
Rinorea covallaroides	28	2277.2	7	316.5308
Olea capensis	36	4079.6	12	567.0644
Rinorea covallaroides	32	3104.2	8	431.4838
Croton megalocarpus	39	4912.1	13	682.7819
Rinorea covallaroides	35	3821.5	8	531.1885
Rinorea covallaroides	16	621.7	7	86.4163
Rinorea covallaroides	14	456.1	8	63.3979
Rinorea covallaroides	34	3573	6	496.647

Species	D (cm)	Adjusted D (0.139 x $D^{2.32}$)	height (m)	Estimated Carbon (Mg/cm^3)
Croton megalocarpus	59	12834.4	10	1783.982
Croton megalocarpus	53	10007.4	11	1391.029
Teclea nobilis	29	2470.4	6	343.3856
Dryptes geradii	24	1592.5	6	221.3575
Dryptes geradii	12	318.9	7	44.3271
Dryptes geradii	11	260.6	6.5	36.2234
Dryptes geradii	16	621.7	4	86.4163
Croton megalocarpus	24	1592.5	11	221.3575
Dryptes geradii	6	63.9	5.2	8.8821
Dryptes geradii	17	715.6	6	99.4684
Dryptes geradii	21	1168.3	6.5	162.3937
Dryptes geradii	3	12.8	3	1.7792
Dryptes geradii	6	63.9	3.5	8.8821
Dryptes geradii	3	12.8	2	1.7792
Dryptes geradii	10	208.9	4.5	29.0371
Dryptes geradii	12	318.9	5	44.3271
Dryptes geradii	11	260.6	4	36.2234
Croton megalocarpus	23	1442.8	9	200.5492
Dryptes geradii	4	24.9	1.5	3.4611
Dryptes geradii	11	260.6	4	36.2234
Dryptes geradii	2	5	0.6	0.695
Dryptes geradii	30	2672.5	5.5	371.4775
ficus	100	43651.6	15	6067.572
Rinorea covallaroides	16	621.7	7	86.4163
Rinorea covallaroides	7	91.3	2.5	12.6907
Rinorea covallaroides	7	91.3	4	12.6907
Rinorea covallaroides	36	4079.6	7	567.0644
Rinorea covallaroides	5	41.8	4.5	5.8102
Rinorea covallaroides	14	456.1	6	63.3979
Croton megalocarpus	58	12335.4	10	1714.621
Toddalia asiataca	14	456.1	0.92	63.3979
Rinorea covallaroides	21	1168.3	4	162.3937
Rinorea covallaroides	7	91.3	4	12.6907
Rinorea covallaroides	8	124.5	4	17.3055
Rinorea covallaroides	12	318.9	5	44.3271
Rinorea covallaroides	6	63.9	4	8.8821
Rinorea covallaroides	7	91.3	7	12.6907
Rinorea covallaroides	6	63.9	4	8.8821
Osyris lanceolata	37	4347.4	10	604.2886
Rinorea covallaroides	11	260.6	4	36.2234
carol	3	12.8	1.5	1.7792
Rinorea covallaroides	2	5	1.5	0.695
clerodendrum	12	318.9	1	44.3271

Species	D (cm)	Adjusted D (0.139 x D $^{2.32}$)	height (m)	Estimated Carbon (Mg/cm^3)
Rinorea covallaroides	5	41.8	2.5	5.8102
Rinorea covallaroides	6	63.9	4	8.8821
Rinorea covallaroides	9.5	185.5	4	25.7845
Croton megalocarpus	23	1442.8	11	200.5492
Rinorea covallaroides	6	63.9	5	8.8821
Rinorea covallaroides	3	12.8	1.5	1.7792
Rinorea covallaroides	4	24.9	6	3.4611
Rinorea covallaroides	61	13866.4	12	1927.43
Rinorea covallaroides	25	1750.7	7	243.3473
Rinorea covallaroides	14	456.1	7	63.3979
Rinorea covallaroides	16	621.7	6.5	86.4163
Rinorea covallaroides	53	10007.4	14	1391.029
Rinorea covallaroides	32	3104.2	9	431.4838
Rinorea covallaroides	33	3333.9	8	463.4121
Rinorea covallaroides	12	318.9	8	44.3271
ficus	206	233438.2	>15	32447.91
Ritchiea albersii	31	2883.8	8	400.8482
Dovyalis abysinica	43	6160.9	9	856.3651
Rinorea covallaroides	7	91.3	6	12.6907
Rinorea covallaroides	9	163.6	6	22.7404
Rinorea covallaroides	9.5	185.5	6	25.7845
Rinorea covallaroides	12	318.9	7.5	44.3271
Rinorea covallaroides	26	1917.5	8	266.5325
Dovyalis abysinica	19	926.2	8	128.7418
Rinorea covallaroides	7	91.3	8	12.6907
Rinorea covallaroides	9.5	185.5	5	25.7845
Rinorea covallaroides	16	621.7	4	86.4163
Croton megalocarpus	61	13866.4	14	1927.43
Rinorea covallaroides	12	318.9	7	44.3271
Rinorea covallaroides	5	41.8	3	5.8102
Rinorea covallaroides	6	63.9	4	8.8821
Rinorea covallaroides	28	2277.2	8	316.5308
clausena	19	926.2	6	128.7418
clausena	26	1917.5	7	266.5325
Croton megalocarpus	29	2470.4	13	343.3856
Strychnos heningsii	43	6160.9	5.5	856.3651

APPENDIX V

Estimated surface litter data

L. Use	Plot number	Weight (g)
F	2	105.92
F	2	123.6
F	2	93.4
F	3	93.46
F	3	176.96
F	3	84.46
F	4	67.5
F	4	93.44
F	4	93.46
F	5	93.4
F	5	136.9
F	5	112.01
F	6	108.1
F	6	93.45
F	6	135.96
F	7	78.12
F	7	133.13
F	7	91.9
F	1	93.56
F	1	93.43
F	2	93.5
F	2	126.82
F	2	170.1
F	3	179.76
F	3	233.88
F	3	75.57
F	4	94.11
F	4	61.4
F	4	93.44
F	5	165.6
F	5	293.5
F	5	93.48
F	6	250.76
F	6	160.62
F	6	93.5
F	7	165.2
F	7	77.01
F	7	193.49
F	8	203.46
F	8	226.58
F	8	122.94
F	9	133.67
F	9	193.48

L. Use	Plot number	Weight (g)
F	9	118.9
C	1	63.32
C	1	93.45
C	2	36.28
C	2	42.03
C	2	22.21
C	3	195.56
C	3	44.02
C	4	21.32
C	4	18.07
C	5	79.91
C	5	64.85
C	6	31.93
C	6	63.23
C	7	48.63
C	7	91.83
C	8	53.26
C	8	92.96
R	1	64.97
R	1	58.9
R	1	120.87
R	2	46.76
R	2	40.24
R	2	26.5
R	3	54.48
R	3	44.34
R	3	112.5
R	4	54.85
R	4	103.08
R	4	26.01
R	5	45.81
R	5	24.1
R	5	30.67
R	6	34.62
R	6	33.2
R	6	29.73
R	7	55.2
R	7	35.91
R	7	39.55
R	8	49.63
R	8	50.2
R	8	48.4

APPENDIX VI

Estimated belowground cummulative carbon for the three land use systems.

L. use	Depth	Mean C	Cummulative carbon (Mg/cm³)
C	2	2.017	2.016726
C	5	3.269	5.286127
C	10	2.091	7.37663
C	15	1.332	8.70823
C	20	1.413	10.12082
C	25	1.229	11.34946
C	30	1.280	12.62975
C	35	1.040	13.66998
C	40	1.335	15.00538
C	45	0.933	15.93834
C	50	1.232	17.17066
C	55	1.225	18.39603
C	60	1.134	19.52963
C	65	1.205	20.73498
C	70	1.342	22.07737
C	75	1.024	23.10106
C	80	1.383	24.48448
C	85	1.331	25.81519
C	90	1.078	26.89355
C	95	1.003	27.89664
C	100	1.469	29.3659
C	105	1.176	30.54168
C	110	1.058	31.60011
C	115	1.202	32.80251
C	120	1.117	33.91946
C	125	1.122	35.04154
C	130	1.135	36.17611
C	135	1.442	37.61815
C	140	1.326	38.94399
C	145	1.326	40.27029
C	150	1.527	41.7977
R	2	2.772779	2.772779
R	5	1.913394	4.686173
R	10	5.306858	9.993031
R	20	1.036086	11.02912
R	30	3.469464	14.49858
R	40	1.106277	15.60486
R	50	1.251071	16.85593
R	60	1.606407	18.46234
R	65	1.265092	19.72743
R	70	0.857272	20.5847
R	75	1.166491	21.75119

L. use	Depth	Mean C	Cummulative carbon (Mg/cm³)
R	80	1.073573	22.82476
R	85	1.192526	24.01729
R	100	1.167658	25.18495
R	105	1.030263	26.21521
R	110	1.048856	27.26407
R	115	0.923234	28.1873
R	125	0.935545	29.12285
R	135	0.895129	30.01797
R	145	0.981578	30.99955
R	150	0.657047	31.6566
F	2	8.095	8.095
F	5	2.150	10.245
F	10	2.578	12.822
F	15	1.769	14.591
F	20	2.214	16.805
F	25	1.162	17.968
F	30	1.665	19.633
F	35	1.120	20.753
F	40	1.444	22.197
F	45	1.130	23.326
F	50	1.539	24.865
F	55	0.954	25.820
F	60	1.897	27.717
F	65	0.838	28.555
F	70	1.234	29.789
F	75	1.153	30.941
F	80	1.084	32.025
F	85	0.927	32.952
F	90	1.289	34.241
F	95	1.125	35.366
F	100	1.176	36.542
F	105	1.111	37.653
F	110	1.325	38.978
F	115	1.039	40.017
F	120	1.365	41.381
F	125	1.386	42.768
F	130	1.397	44.165
F	135	1.055	45.219
F	140	1.004	46.224
F	145	0.978	47.202
F	150	1.207	48.408