मेघालय की पूर्वी खासी पहाड़ियों में विभिन्न भूमि उपयोग तंत्रों के अंतर्गत मृदा–जैविक पदार्थ की गुणवत्ता एवं संग्रह

Stocks and quality of soil organic matter under different land use systems in East Khasi hills of Meghalaya

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Stocks and quality of soil organic matter under different land use systems in East Khasi hills of Meghalaya

BY

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CERTIFICATE

This is to certify that the thesis entitled **"Stocks and quality of soil organic matter under different land use systems in East Khasi hills of Meghalaya"** submitted in partial fulfillment of the requirements for the award of the degree of **DOCTOR OF PHILOSOPHY** in **SOIL SCIENCE AND AGRICULTURAL CHEMISTRY,** to the **Post-Graduate School, Indian Agricultural Research Institute, New Delhi,** is a record of the *bona-fid*e research carried out by **Mr. RAMESH T.,** under my guidance and supervision. The subject of the thesis has been approved by the students's Advisory Committee.

No part of the thesis has been submitted for any other degree or diploma. All the assistance and help received during the course of this investigation have been duly acknowledged by him.

(K.M. MANJAIAH)
Chairman
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Place : New Delhi
Dated : 12th, December, 2012

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CHAPTER I

INTRODUCTION

Soil degradation has raised some serious debate, and it is an important issue in the modern era. The degree of soil degradation depends on soil's susceptibility to degradative processes, land use, the duration of degradative land use, and the management. Soil and water degradation are also related to overall environmental quality, of which water pollution and the "greenhouse effect" are two major concerns of global significance. The global concerns over increased atmospheric CO_2, which can potentially alter the earth's climate systems, have resulted in raising interest in studying soil organic matter (SOM) dynamics and carbon (SOC) sequestration capacity in various ecosystems (Schlesinger, 1997). Soils represent an important terrestrial stock of C and approximately two to three times as much as terrestrial vegetation and atmosphere, respectively, and the C in the SOM of agricultural land is composed of dominant terrestrial C stock.

The North Eastern parts of India, comprising the states of Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram, Nagaland, Sikkim, and Tripura, lies between $22^{o}05'$ and $29^{o}30'$ N latitudes and $87^{o}55'$ and $97^{o}24'$ E longitudes. The region is characterised by diverse agro-climatic and geographical situations. About 54.1 per cent of the total geographical area is under forests, 16.6 per cent under crops, and the rest either under non-agricultural uses or uncultivated land. The less area under agricultural crops is due to natural corollary of the physiographic features of the region, as major chunk of the land has more than 15 per cent slope, undulating topography, highly eroded and degraded soils, and inaccessible terrain. Continuous dilution of the forest cover in the region due to shifting cultivation, firewood, and timber collection is posing the most crucial problem of soil degradation resulting in poor soil health and environmental degradation in the hills.

Shifting cultivation commonly known as *slash and burn* agriculture, is believed to be originated during the Neolithic period around 7000 B.C. It was a remarkable innovation during primitive culture and regarded as the first step in transition from food gathering to food production. Yet this system of farming is still practiced in different parts of the world. It has been estimated that about 200 million people, 7% of mankind of the world is still practicing this type of cultivation in about 300 million hectare of various land i.e. 5% of cultivated soil throughout the world. In India, around two million tribal people cultivated approximately 11 million hectare of land under shifting cultivation. In North-East India, over a 100 of tribal ethnic minorities are practicing shifting cultivation and in certain parts of this region it is practised not only by the tribal minorities but also by the landless people and small landholding migrants. According to the report of National Commission on Agriculture (1976), 49,2000 tribal families of this region are involved in *Jhuming* and the total area affected by this practice is 26,94,000 ha. On an average, 3,869 km^2 area is put under shifting cultivation every year. Shifting cultivation in its more traditional and cultural integrated form is an ecological and economically viable system of agriculture as long as population densities are low and *jhum* cycles are long enough to maintain soil fertility. It became unsustainable today primarily due to the increase in population that led to increase in food demand. *Jhuming* cycle in the same land, which extended to 20–30 years in earlier days, has now been reduced to 3–6 years (Borthakur, 1992). Because of several causes, the land degradation in the region has reached a level of 36.64% of the total geographical area, which is almost double than the national average of 20.17%. Of various degradation types, water erosion, reduced infiltration, acidification, nutrient leaching, burning of vegetation, decline in vegetative cover, and biodiversity are important in context to the NE region. The total forest cover in the region is 1,41,652 km^2, which is about 54.1% of the geographic area as against the national average of 19.39% (State Forest Report, 2001). Since shifting cultivation is still practiced in the region, and every year dense forest is converted into *jhum* fields, there is drastic reduction in dense forest cover (canopy density > 40%) in most of the states.

Soil erosion under shifting cultivation is highly erratic from year to year depending on rainfall characteristics. Studies on steep slopes (44–53%) have indicated the soil loss to the tune of 40.9 t ha^{-1} and the corresponding nutrient losses per hectare are 702.9 kg of organic carbon, 63.5 kg of P and 5.9 kg of K (Prasad and Sharma, 1993). The soil loss from hill slopes (60–79%) under first year, second year, and abandoned *jhum* was estimated to be 147, 170, and 30 t ha^{-1} yr^{-1} (Singh and Singh, 1981). In general, tolerable soil loss (*T*) value is 11.2 Mg ha^{-1} yr^{-1} (5.0 t ac^{-1} yr^{-1}) while it is between 5.0 and 12.5 Mg ha^{-1} yr^{-1} (2.2 and 5.6 t ac^{-1} yr^{-1}) in North West Himalayas (Mandal et al., 2006). During first few years of clearing, carbon and nitrogen levels decrease rapidly. According to one estimate annual loss of top soil, N, P and K due to shifting cultivation is 88346, 10669, 0.372, and 6051 thousand tonnes in the region (Sharma, 1998). Singh et al. (1996) reported nutrient loss to the tune of 6.0 million tonnes of organic carbon, 9.7 tonnes of available P, and 5690 tonnes of K from the NEH region. The soil fertility decreases rapidly in the second year and is very poor in the third year. The cultivation during the third year and beyond is usually uneconomical. Although shifting cultivation is primitive as well as labour intensive and ecologically imbalance farming system yet it is very difficult to change traditional shifting cultivators even if we provide all the modern farm inputs. Firstly, it is very deep-rooted and secondly, it is a part of socio-cultural life of the tribal people which is linked to their religious rites and festivals. Thus, to replace *Jhuming* with alternate system of farming should be taken up on a priority basis in areas where the *Jhuming* cycle has come down to 3-5 years.

The soil organic matter (SOM) encompasses plant, animal and microbial bodies in all stages of decay and a diversity of heterogeneous organic substances intimately associated with inorganic component (Christensen, 1992). Soil organic matter is an important indicator of soil quality and productivity and sustaining SOM quality and quantity is critical to ensuring long-term soil fertility

(Tisdale et al., 1993). The type of land use system is an important factor controlling organic matter contents of soils since it affects the amount and quality of litter input, decomposition rates and the process of organic matter stabilization in soils. The changes in land use systems and management also influence the amount and rate of SOM losses (Guggenberger et al., 1995). Conversion of natural ecosystem to agricultural land and increasing intensity of tillage are major causes of decrease in SOM levels due to reduced inputs of organic matter, reduced physical protection of soil organic carbon (SOC) content, a lower fraction of non-soluble materials in more readily decomposed crop residues and soil erosion (Post and Kwon, 2000; Lal, 2003). The amount of SOC that exists in any given soil is determined by the balance between the rates of organic carbon input (vegetation, roots) and output (CO_2 from microbial decomposition). However, soil type, climate, management, mineral composition, topography, soil biota and the interactions between each of these are modifying factors that will affect the total amount of SOC in a profile as well as the distribution of SOC contents with depth. It is important to note that any changes made to the natural status of the soil systems (e.g. conversion to agriculture, deforestation, plantation) will result in different conditions under which SOC enters and exits the system. Therefore, perturbed systems may still be in the process of attaining a new equilibrium C content and any measurements of SOC have to take into account that the soil is in the process of re-establishing equilibrium, which could take more than 50 years (Baldock and Skjemstad, 1999).

It is now widely recognised that SOC plays an important role in soil biological (provision of substrate and nutrients for microbes), chemical (buffering and pH changes) and physical (stabilisation of soil structure) properties. In fact, these properties, along with SOC, N and P, are considered critical indicators for the health and quality of the soil. In particular, the suitability of soil for sustaining plant growth and biological activity is a function of physical (porosity, water holding capacity, structure and tilth) and chemical properties (nutrient supply capability, pH, salt content), many of which are a function of SOM content. Similarly, Elliott (1986) indicated that SOM was a key indicator of soil health but further suggested that particulate organic matter (POM) could be used as an indirect measure of soil health because of its short turnover time. In general, increases in SOM are seen as desirable by many farmers as higher levels are viewed as being directly related to better plant nutrition, ease of cultivation, penetration and seedbed preparation, greater aggregate stability, reduced bulk density, improved water holding capacity, enhanced porosity and earlier warming in spring (Carter and Stewart, 1996; Lal, 2002).

Understanding the mechanisms of carbon sequestration in SOM has attracted considerable attention in recent years, because of the important question of how organic C stocks will respond to a change in the global climate, which is expected to be the greatest in the developing countries (Vucetich et al., 2000). For predicting responses to changes in the environment, it is imperative to understand the processes and mechanisms that determine the decomposition and accumulation rates of SOM. The build-up of SOM is a highly complex process that is determined by both the quantity and quality of decomposing plant litter and the rate and completeness of its decomposition. Plant litter constitutes a wide range of C compounds that decompose at different rates (Janzen, 2005), and respond in different ways to ecological changes or changes in the abiotic environment (Vedrova et al., 2002). The concentration of easily decomposable, water-soluble substances generally enhances litter decomposition rates, whereas the concentrations of slowly decomposable tannins and other polyphenols retard the decomposition rates.

Sequestration of carbon in terrestrial ecosystems is a low-cost option that may be available in the near-term to mitigate increasing atmospheric CO_2 concentrations, while providing additional benefits. Storing carbon in terrestrial ecosystems can be achieved through maintenance of standing aboveground biomass, utilization of aboveground biomass in long-lived products, or protection of carbon (organic and inorganic) compounds present in soils. Soils are thought to have a finite carrying capacity for carbon based on parent material (texture), ambient temperature, annual precipitation, and net plant primary production. However, microbial transformation and turnover of soil organic matter can also influence the magnitude of soil carbon storage. The microbially mediated reactions fractionate organic material into various reactive carbon pools or fractions such as labile, slowly decomposable, and resistant with respect to their persistence in soil. The importance of carbon fractionation to soil carbon storage is evident by the intriguing possibility of manipulating the system into bypassing the labile pool with transformation directly into the slow and resistant pools with varying turnover rates from hours to hundreds of years. In this scenario, one would hypothesize that more of the carbon input would be sequestered in the soil for longer periods of time with less CO_2 emission to the atmosphere.

Maintenance and improvement of SOM quantity and quality are generally accepted as one of the most important criteria for sustainable soil management. Due to the high background SOM pool (Haynes, 1999), direct measurement of short-term total SOM losses resulting from land use change may not clearly show the effect of land use change. Hence, previous studies proposed some more sensitive indicators of SOM pools such as particulate organic carbon (POC) (Cambardella and Elliott, 1992), light fraction organic carbon (LFOC) (Six et al., 2002a), readily oxidized fractions by 333 *mM* $KMnO_4$ (Blair and Crocker, 2000) and microbial biomass carbon (MBC). These indicators often respond more rapidly to management-induced changes in the SOC pool than bulk SOM, and could serve as early indicators for the overall carbon stock change. On the premise that microbiological decomposition of organic matter in the soil is largely associated with an oxidation process of enzymatic character, the measurement of the labile carbon fractions by oxidation with 333 *mM* $KMnO_4$ developed by Blair et al. (1995) has successfully been used by numerous researchers in SOM studies (Blair and Crocker, 2000), however, the appropriate concentration of $KMnO_4$ has been widely debated (Vieira et al., 2007). The particulate fraction obtained through particle-size fractionation has similar characteristics and is also considered to contain labile C

(Cambardella and Elliott, 1994). Several authors (Ashagrie et al., 2005; Solomon et al., 2002) found differences in the quality and amount of SOM associated with mineral particles of different sizes, and also reported relatively greater losses of OC in the coarser than in the finer particle size separates as a result of changes in land use. All closely relate to the decomposition process, thus, the three labile organic carbon indicators seem to have a close association with one another and have an important impact on soil quality. However, there seems still no consensus as to which method or combination of methods is most suitable to evaluate the land use change impacts on soil quality (Skjemstad et al., 2004).

Monitoring programs, long-term experiments and modelling studies all show that land use significantly affects soil carbon stocks. Soil carbon losses occur when grasslands, managed forest lands or native ecosystems are converted to croplands. Vice versa soil carbon stocks are restored when croplands are either converted to grasslands, forest lands or natural ecosystems. The more carbon is present on the soil, the higher the potential for losing it. Therefore, the potential losses of unfavourable land use changes on organic matter rich soils are a major risk. The most effective strategy to prevent global soil carbon loss would be to halt land conversion to cropland, but this may conflict with growing global food demand unless per-area productivity of the cropland continues to grow. Given that land use change is often driven by demand and short term economic revenues, the most realistic option to improve soil carbon stocks is to protect the carbon stocks in the soils and to improve the way in which the land is managed to maximise carbon returns to the soil and minimise carbon losses.

Land-use change affects carbon stocks and also soil respiration (also called soil CO_2 efflux) rates (Trumbore et al., 1996). Soil respiration contributes 11 times more CO_2 emissions in atmosphere than the total CO_2 evolved from fossil fuel burning. To characterize the carbon exchange in ecosystems, an assessment of the magnitude and dynamics of soil CO_2 efflux is important, considering that soil respiration is a major CO_2 flux in the carbon cycle, second in magnitude to gross canopy photosynthesis (Raich and Schlesinger, 1992). The net flux of carbon between the soil and the atmosphere is determined by the rate at which SOC is converted to CO_2 by microorganisms and by autotrophic respiration. Soil respiration is influenced mainly by soil temperature and moisture but also by vegetation type and substrate availability (Vasconcelos et al., 2004). Understanding the impacts of agriculture, forest and horticulture land uses on soil carbon pools and fluxes will allow for a better understanding of the potential of soil to sequester and release carbon.

Numerous farming-system models integrating forestry, horticulture, agriculture, dairy, poultry, aquaculture, and pig, duck, and goat components have been introduced to complement soil conservation measures for long-term sustainable production (Prasad and Sharma, 1993) to replace the slash-and-burn cultivation in view of the socioeconomic conditions, topography, soil, and climatic conditions of the particular region for livelihood, income generation, and food security of the inhabitants (Bhatt et al., 2006). Short- and long-term studies to evaluate the interrelationships among soil physical, chemical, and biological properties under various land use systems will be essential for defining and/or refining management practices that expedite the restoration of productivity of specific farming systems. Assessment of soil quality, under different land use systems is of utmost importance for sustainable crop production and to maintain soil health. Because there was limited information available on SOM stocks and quality under various land use systems in the north-eastern Hill region, an attempt has been made to study the impact of more prominent land use systems on SOM stocks and quality at varying slopes and soil depths in relation to soil properties to assess the soil quality for sustainable crop production and soil productivity. Thus, the main objectives of the study are:

(i) To quantify soil organic matter stocks under different land use systems in East Khasi hills of Meghalaya.

(ii) To assess the quality of soil organic matter under different land use systems.

(iii) To study the carbon stability mechanisms under different land use systems.

CHAPTER II

BACKGROUND

2.1 Land use on soil fertility

Land use change is an important factor in global change phenomena. It is directly related to issues such as food security (Wilson, 1988; Tao et al., 2009), water and soil quality (Fu et al., 1999; Mueller-Warrant et al., 2012), and other important global life support issues. Land use change and changes in soil management often occur together, resulting in changes in soil quality, including soil microorganism activities (Saraswathy et al. 2007; Gol, 2009). As a consequence, one would expect close relationships between land use change and soil nutrient contents. In the last decades severe land-use changes occurred in tropical countries, due to increasing population and their demand for food resources (Lambin et al., 2000). Land degradation often resulted in nutrient depletion, SOM reduction, soil quality degradation, biodiversity decrease and soil erosion acceleration (Imoke et al., 2010; Ayoubi et al., 2011).

Braimoh and Vlek (2004) has shown that changes in land use, through altering the structure and functioning of ecosystems and influencing the biogeochemical cycle, could drive changes in soil properties and land productivity, resulting in further changes in soil quality and landscape over time and space. Forest land is rapidly converted into agriculture or pastureland. Land-use conversion may cause important changes in soil physical and chemical characteristics and can affect soil fertility, increase soil erosion or cause soil compaction (Neill et al., 1997). Increasing land conservation requires soil carbon and soil fertility management within a broader framework of sustainable development (Smith, 2008). Furthermore, it is important to establish sustainable land-use systems while conserving soil fertility in the long term.

Maintenance of soil fertility requires preservation of its organic matter, physical properties and nutrient levels. Due to the vulnerability of the soils in north-east India for several environmental factors and human activities, all the agricultural or land development related activities in the north-east region must consider soil fertility and its maintenance as a priority. One parameter to evaluate the sustainability of production systems is soil quality, which includes organic matter, acidity and/or alkalinity, availability and balance of nutrients, structure and aggregation, water infiltration and storage, as well as plant productivity. Modern studies establish correlations between the changes in the physical and biological properties, indicating that it is first necessary to know the soil characteristic of each type of soil in order to prevent the negative effects caused by human activity (Wilding, 1999).

Soil texture is one of the inherent soil physical properties less affected by management. It affects the infiltration and retention of water, soil aeration, absorption of nutrients, microbial activities, tillage and irrigation practices (Gupta, 2004). It is also an indicator of some other related soil features such as type of parent material, homogeneity and heterogeneity within the profile, migration of clay and intensity of weathering of soil material or age of soil (Brady and Weil, 2002). Bulk density provides information on the degree of compactness, aeration status, available water content and also the environment available to soil microorganisms. Bulk densities of soil horizons are inversely related to the amount of pore space and soil OM (Gupta, 2004). For instance, intensive cultivation increases bulk density resulting in reduction of total porosity. The study results of Yao et al. (2010) revealed that the bulk density of cultivated soils was higher than the bulk density of forest soils. Similarly, Araujo et al. (2004) and Yuksek et al. (2009) reported that soil bulk density under both cultivated, pasture and forest lands increased with increasing soil depth. Islam and Weil (2000) reported relatively low pH in forest land use compared to grassland and cultivated fields and stated that pre-weathered parent materials, amphoteric nature of Al and the intense leaching of basic cations are the likely contributing factors of low pH in forest soils.

In most tropical environments, the conversion of forest vegetation to agricultural land results in a decline of the soil OM content to a newer, lower equilibrium (Marzaioli et al., 2010). Most of the cultivated soils of north-east are poor in OM contents due to low amount of organic materials applied to the soil and complete removal of the biomass from the field, and also due to severe deforestation, steep relief condition, intensive cultivation and excessive erosion hazards (Saha et al., 2012). Biological degradation is frequently equated with the depletion of vegetation cover and OM in the soil, but also denotes the reduction of beneficial soil organisms that is important indicator of soil fertility (Laxminarayana, 2010). Zeidler et al. (2002) demonstrated a 20–50% reduction of SOM as a result of clearing tropical forests and their subsequent conversion into farmland. On the contrary, it is widely reported that organic matter content and soil aggregate stability are greatly enhanced in most of the soils converting from arable to forest plantation (Bouajila and Gallali, 2010). Geissen et al. (2009) reported that the concentration of organic carbon (OC) in soils under the forest, agriculture and fruit tree plantation decreased with depth.

The total N content of a soil is directly associated with its OC content and its amount is lower in continuously and intensively cultivated and highly weathered soils of the humid and sub humid tropics due to low OM content (Tisdale et al., 1995). Wakene (2001) reported that there was a 30% and 76% depletion of total N from agricultural fields cultivated for 40 years and abandoned land, respectively, compared to the virgin land. Average total N increased from cultivated to grazing and forest land soils, which again declined with increasing depth from surface to subsurface soils (Nega, 2006). Phosphorus (P) is known as the master key to agriculture because lack of available P in the soils limits the growth of both cultivated and uncultivated plants. Most of the soils in north-east India are acid soils known to have low P contents, not only due to the inherently low available P content, but also due to the high P fixation capacity of the soils (Bhat et al., 2006).

Wakene (2001) reported that the variation in the distribution of K depends on the mineral present, particles size distribution, degree of weathering, soil management practices, climatic conditions, degree of soil development, the intensity of cultivation and the parent material from which the soil is formed. The greater the proportion of clay mineral high in K, the greater will be the potential K availability in soils (Tisdale et al., 1995). Normally, losses of K by leaching appear to be more serious on soils with low activity clays than soils with high- activity clays. Soils in areas of moisture scarcity (such as in arid and semi arid regions) have less potential to be affected by leaching of cations than do soils of humid and sub-humid regions (Uzoho et al., 2007). Soils under continuous cultivation, application of acid forming inorganic fertilizers, high exchangeable and extractable Al and low pH are characterized by low contents of Ca and Mg mineral nutrients resulting in Ca and Mg deficiency due to excessive leaching (Uzoho et al., 2007).

Jiang et al. (2009) stated that availability of micronutrients is mainly affected by the factors like parent material, soil reaction, soil texture, soil OM, adsorptive surfaces and other physical, chemical and biological conditions in the rhizosphere. Nazif et al. (2006) stated that micronutrients have positive relation with the fine mineral fractions like clay and silt while negative relations with coarser sand particles. This is because their high retention of moisture induces the diffusion of these elements. Soil OM content also significantly affects the availability of micronutrients. According to Vijayakumar et al. (2011), the presence of OM may promote the availability of certain elements by supplying soluble complexing agents that interfere with their fixation. Brady and Weil (2002) indicated that the solubility, availability and plant uptake of micronutrient cations (Cu, Fe, Mn and Zn) are more under acidic conditions (pH of 5.0 to 6.5).

Soil microbial biomass carbon (SMBC) has been proposed as an index of soil stress and disturbance and its measurement is often essential for soil ecological studies (Hernandez et al., 1997). Microbial activity is fundamental in the processes that make energy and nutrients available for recycling in the ecosystem. The dynamics of microbial biomass and its role in plant nutrition under different ecosystem conditions is of greater significance (Onwonga et al., 2010). Arunachalam and Pandey (2003) observed higher and varied SMBC under forest land which may be related, directly to the quantity and quality of the litter, greater availability of nutrients due to the addition of higher plant litters and indirectly through the changes in soil physical and chemical properties. Witter and Kanal (1998) also observed a close relationship between organic C content and microbial biomass in soils under different land use systems

2.2 C stocks and fractions

Soil organic matter (SOM) is the central element of soil fertility, productivity and quality, as reduction in SOM is believed to create an array of negative effects on crop productivity. The importance of organic carbon to the physical, chemical, and biological aspects of soil quality is well recognized (Stevenson, 1986; Baretto et al., 2011). The SOM is an extremely important attribute of soil quality and soil health, since it influences soil physical, chemical, biological properties and processes. It is a source of energy and nutrients for soil biota which affects the nutrient supplying capacity of soil *via* mineralization. It also affects aggregate stability, trafficability, water retention and hydraulic properties (Haynes, 2005). In addition to being a direct source of plant nutrients, SOM also indirectly influences the nutrient availability in soil. Besides, it is extremely important in maintaining overall quality of environment as soil contains significant part of global carbon stock (Lal et al., 1998; Verma et al., 2010).

Several studies (Cambardella and Elliott, 1992; Chan, 1997; Bhattacharyya et al., 2011) suggest that certain fractions of soil organic matter are more important in maintaining soil quality and are, therefore, more sensitive indicators of the impact of management practices. Martin et al. (2010) and Singh et al. (2011) reported decrease in soil organic carbon under agriculture land use in comparison to forestry and horticulture land uses. However, it is likely that both quantity and quality of soil organic carbon sequestered under different land uses are different, and these, in turn, can have different but important effects on soil quality, such as soil structural stability and chemical fertility. This knowledge is important for the selection of suitable land use systems, either singly or as mixer, to moderate the impact of changing land use pattern on global climate change.

Although soil organic matter includes a continuum of materials ranging from highly decomposable to very recalcitrant, it is divided into two major pools, labile and stabilized fractions for convenience (Haynes, 2005). The labile carbon pool is the fraction of SOC, which has rapid turnover rates and sensitive to alteration in land use management practices. The pool of SOC fuels the soil food web and therefore greatly influences the nutrient cycling for maintaining soil quality and its productivity (Majumder et al., 2008a). Cropping systems and management practices that ensure greater amount of crop residue to be returned to the soil are expected to cause a net build up of SOC stock (Kaur et al., 2008; Majumder et al., 2008b).

The soil organic C pool and the C lability directly influence soil physical, chemical and biological attributes as well as the self-organization capacity of soils (Blair and Crocker, 2000). Blair et al. (1995) proposed labile C as oxidizable in 333 mM $KMnO_4$ solution. Therefore, the integration of both soil organic C pool and C lability into the C management index (CMI), originally proposed by Blair et al. (1995), can provide a useful parameter to assess the capacity of management systems into promote soil quality (Blair et al., 1995, 2006a; Diekow et al., 2005a). In spite of the existence of a large number of studies that can provide a lot of relevant data pertaining to soil management, however, few are the studies that integrate the total soil organic C pool and the C lability into the CMI as a way to assess the capacity of management systems into promote soil quality.

Human beings are accelerating the rate of increase in atmospheric CO_2 concentration through fossil fuels burning, land use, land-use changes and forestry activities, resulting in global warming and climate change during the recent times (Upadhyay et al., 2005). Over the last 20 years, majority of the emission is attributed to burning of fossil fuel, while 10-30% is attributed to land use change and deforestation (IPCC, 2001). The average atmospheric CO_2 concentration has increased from pre-industrial concentration of 280 ppm to 379 ppm in 2005, and is currently increasing at a rate of about 1.5 ppm per year (IPCC, 2007). The International Panel on Climate Change (IPCC) in its fourth assessment report has strongly recommended to limit the increase in global temperature below 2°C as compared with pre-industrial level (i.e. measured from 1750) to avoid serious ecological and economic threats. A rise in global mean temperature by 0.74°C has already been recorded, and hence climate scientists are focusing on an urgent action to curb global warming (IPCC, 2007; Kerr, 2007).

According to the article 4 of the United Nation Framework Convention on Climate Change (UNFCCC), climate change can be prevented and minimised by “limiting anthropogenic emissions of greenhouse, and protecting and enhancing greenhouse gas sinks and reservoirs”. As a mitigation measure to global climate change due to greenhouse gas emission, it is required to cut the rate of emission either through reducing tropical deforestation or to enhance the natural carbon sequestration potential of different land use systems through forest plantations, raising orchards and scientific management of agricultural lands. The degraded lands by *Jhuming* in north-east India have a large potential to sequester carbon in the soil; storage in vegetation is preferable due to their longer residence time and less risk of rapid release to the atmosphere (Lal, 2001). The protection of existing forests, regeneration of degraded forests, raising of forest and horticulture plantations and conservation agriculture in India have been contributing to enhanced carbon stock (Ravindranath et al., 2008). The potential of a land use for increased carbon sequestration capability can be assessed either through the amount of carbon stored or estimating the annual carbon sequestration rate (Iverson et al., 1993).

In last decade, the greenhouse effect has been of great concern, and has led to several studies on the quality, kind, distribution and behaviour of SOC. Global warming and its effect on soils in terms of SOC management have led to several quantitative estimates for global C content in the soils. The current global stock of soil organic carbon is estimated to be 1,500-1,550 Pg (Adams et al., 1990; Eswaran et al., 1993). The first estimate of the organic carbon stock in Indian soils was 24.3 pg (1 Pg = 10^{15} g) based on 48 soil samples (Gupta and Rao, 1994). Forest soils are one of the major carbon sinks on earth, because of their higher organic matter content. About 40% of the total SOC stock of the global soils resides in forest ecosystem. According to Buringh (1984), global agricultural soil organic C stock is about 142 Pg C, which represents 8-10% of the total soil carbon. The SOC pool can be depleted by 15–40% in a 2-year period to 1-m depth when tropical forest is converted to agricultural land use (Ingram and Fernandes, 2001) or as much as 50–75% (Lal, 2004; Post and Kwon, 2000). Such depletion of the SOC pool creates the potential to accumulate (sequester) C in soils upon adoption of a restorative land use and less harmful agricultural practices.

Soils can act as sinks or as a source for carbon in the atmosphere depending on the changes happening to soil organic matter. Equilibrium between the rate of decomposition and rate of supply of organic matter is disturbed when land use is changed (Lal, 1999). Soil organic matter can also increase or decrease depending on numerous factors, including climate, vegetation type, nutrient availability, disturbance, and land use and management practices (Leifeld and Kogel-Knabner, 2005). Physical soil properties, such as soil structure, particle size, and composition, have profound impact on soil carbon. Soil particle size has an influence on the rate of decomposition of soil organic carbon. The release of nutrients from litter decomposition is a fundamental process in the internal biogeochemical cycle of an ecosystem, and decomposers recycle a large amount of carbon that was bounded in the plant or tree to the atmosphere (Larrey-Larrouy et al., 2004).

About 53% of the total area of north-east India is under forest cover which is higher than that of national average of about 20%. Forest Survey of India (1999) estimated that about 0.179 M ha of area every year is to be affected by Jhuming in north-east India. Jhuming with reduced fallow cycle of 2-3 years (10-15 years in the past) along with deforestation and forest fires resulted in loss of biodiversity and severe soil erosion, leading to depletion of soil productivity in the north-east India. On average, 36.64% of the total geographical area is degraded as a result of shifting cultivation in north-eastern India, which is almost two-fold more than the national average of 20.17% (Bhatt et al., 2006). Almost 88.3 Mt of soil is estimated to be lost annually as a result of shifting cultivation in the North-Eastern Hill (NEH) Region (Prasad and Sharma, 1993), with the rate of loss ranging from 30 to 170 t ha^{-1} yr^{-1} (Singh and Singh, 1980), compared with the national average of 16.35 t ha^{-1} yr^{-1} (Narayana and Ram Babu, 1983). Such high rates of soil erosion result in considerable leaching of nutrients from the topsoil, ultimately leading to poor productivity of crops. Prasad et al. (1981) estimated that 18.1 Mt of soil eroded annually from the land subjected to shifting cultivation would lead to losses of 603 t of organic carbon, 97 t of available P_2O_5, and 5690 t of available K_2O.

2.3 Humic acids characterization, elemental composition and functional groups

Humic substances (HS) belong to the most important class of naturally occurring complex agents, comprising a complex mixture of physically and chemically similar substances that show a large number of oxygen-containing functional groups, particularly -COOH and -OH groups. They are generally divided into three classes of materials on the basis of their alkaline and acid solubility (fulvic acids which are soluble under all pH conditions, humic acids soluble at pH >2.0 and humin insoluble at all pHs). Humin and humic acids generally represent the major fraction of humic substances and appear to display similar analytical characteristics and chemical structure (Schulten and Schnitzer, 1997). HS are formed by secondary synthesis reactions during the decay process and by transformation of biomolecules originating from dead organisms and microbial activity. These compounds are major sinks of refractory organic carbon and their resistance to microbial degradation is attributable to self-associations of the molecules through associations with difficult-to-degrade substances, such as long-chain hydrocarbons in fatty acid and esters, to associations with the soil's mineral colloids and to entrapment in soil aggregates. HS play an important role in soil conservation, for water holding capacity, and for the complexation of metals in terrestrial and aquatic systems (Hayes and Graham, 2000; Hayes and Malcolm, 2001).

The properties of HS in the environment involve their structural characteristics and composition, which are controlled by the process of humification of organic matter (Rosa et al., 2002) and are dependent on soil type (Baldock et al., 1997), vegetation (Quideau et al., 2001), and climatic conditions (Dai et al., 2002). Humification is defined as the transformation of macromorphologically identifiable matter into amorphous compounds, as a rule involving the changes that occur in vegetal residues or soil organic matter during the humification process. It has been related to the preferential oxidation of plant polysaccharides, the selective preservation of more recalcitrant organic compounds such as lignin and phenolic structures, and to the incorporation of organic compounds of microbial origin (Zech et al., 1997). The best method for measuring the degree of humification is still being debated because there is no well-defined model of HS structure (Piccolo, 2001). Several techniques have been used to characterize the progress of humification, including measurement of the E4/E6 ratio, defined as the ratio of optical absorbance at 465 and 665 nm in aqueous solution, which indicates the structural condensation (Stevenson, 1994); elemental composition; functional groups analysis and C/H, C/O, and C/N ratios, defined as the atomic ratio between elements (carbon, hydrogen, oxygen, and nitrogen; Stevenson, 1994; Rosa et al., 2001).

Allard (2006) reported higher elemental composition in the humic acids extracted from forest land use compared to maize cultivated lands. The C/N ratio of humic acids was higher under forest land use than cultivated field indicating a high degree of humification and/or a greater microbial contribution under cultivated fields. Nierop et al. (2001) and Jolivet et al. (2001) also observed similar trend for whole organic matter from forest soils, soils under grasslands and under maize. Conversion of grasslands and pastured grasslands into fallows and arable lands leads to a transformation of humic acids in the direction of increasing the degree of their chemical maturity (Ch'ng et al., 2011). Cunha et al. (2009) in their study have shown that different land uses (tillage or its absence, and different amounts and quality of organic refuse input) have a significant statistical influence on humus and total carbon and nitrogen with decreased values under grassland, and especially arable land, in contrast to forest, while N enrichment of humus is significantly higher under grassland than under forest. They also concluded that human influences altered the humus composition significantly by increasing the stability of humus under grasslands and arable lands in contrast to forest soils.

Soil organic matter is composed of different compartments which differ from each other in biochemical composition, biological stability and carbon turnover rates (Paustian et al., 1992). The humified SOM (Humic acid, fulvic acid and humin) represents the most microbially recalcitrant and thus stable reservoir of organic carbon (OC) in soil and an important component for the control of soil erosion by water and wind (Piccolo, 2001). The general constancy of C/N values for the different humic fractions suggests a resistance to microbial decomposition attributable to their recalcitrant nature deriving from either physical protection in microaggregates or chemical protection by their hydrophobic composition (Spaccini et al., 2000).

2.4 Aggregate cementing agents and thermal stability

Soil aggregation is an important mechanism for stabilization of soil organic matter (SOM), especially in organic rich soils (Lutzow et al., 2006). Soil aggregation affects various soil physical, chemical, and biological processes, such as soil aeration, soil water infiltration, and soil microbial activities. If soil organic matter (SOM) is within soil aggregates, aggregate formation physically protects SOM from biodegradation. Therefore, aggregate formation promotes long term carbon sequestration and soil structural stability (Six et al., 2000b).

Aggregate formation and stabilization are affected by various factors, including clay content, mineralogy of the clay fraction, and types and amount of SOM (Denef and Six, 2005). The main agents of aggregate formation and stabilization are organic materials, including persistent cementing agents, such as humic matter involved in stabilizing microaggregates, and transient bonding agents (e.g. polysaccharides derived from plants and microorganisms) as well as temporary binding agents (e.g. fungal hyphae, fine roots, bacterial cells) related to formation and stabilization of macroaggregates. Denef et al. (2001) and De Gryze et al. (2005) showed that fungi significantly affected the macroaggregate formation. Besides physical effects of enmeshment of macroaggregates by hyphae, with extracellular polysaccharides produced by hyphae, microaggregates are attached and bound into stable macroaggregates (Neufeldt et al., 1999). In addition, the hydrophobicity of microbial extracellular polysaccharides contributes to the stabilization of macroaggregates by decreasing their wettability (Liu et al., 2005). Besides fungi, bacteria also exude extracellular polysaccharides to bond soil particles and increase inter-particle cohesion (Degens, 1997).

Aggregates of different sizes can be formed in soils because of the organic matter where, the primary particles and clay microstructure are bound together with bacterial and fungal debris into extremely stable microaggregates which may be bound together with fungal and plant debris giving a larger microaggregates. These microaggregates are bound into macroaggregates, due to the effect of transient binding agents (polysaccharides derived from plants and microorganisms) and temporary binding agents (fungal hyphae, fine roots, bacterial cells) (Tisdall and Oades, 1982; Oades, 1993). In many soils, intensive cultivation has been reported to degrade the soil structure which is a measure of soil aggregates stability. The lower stability is usually associated with decrease in SOM content and significantly affects the plant development. Eynard et al. (2004) have registered decreased SOM and aggregate stability under cultivation of forested land. They also found a significant correlation between SOM and aggregate stability due to the binding action of humic substances and other microbial by-products in their study on comparing cultivated soils and forest soils for aggregate stability.

Soil carbohydrates, which represent from 5 to 25% of SOM (Stevenson, 1994), constitute a significant part of the labile pool of SOM and are most affected by land use changes (Spaccini et al., 2001). Due to the temporary biological stability of carbohydrates (Insam, 1996), their long-lasting role in improving soil physical properties may not be assumed in all soil conditions (Degens and Sparling, 1996) and large emphasis had been given to the action of polymeric carbohydrates in stabilizing soil structure (Tisdall and Oades, 1982). The effect of cultivation on the nutrient and microbial characteristics of soils are observed in the C and N-enriched small macroaggregate fractions (2.00–0.25mm) (Cambardella and Elliott, 1992). Dormaar (1983) and Gajic et al. (2006) reported that SOM and polysaccharides were associated with the >0.25mm water-stable aggregates (WSA). Christensen (1992) observed that whereas the C/N, C/P and N/P ratios of water-stable macroaggregates were smaller than those of microaggregates, the microaggregates contained less SOM associated with silt plus clay than the macroaggregates.

Considerable SOC storage and stabilization is associated with organo-mineral complexation phenomena controlled by soil mineralogy and pedogenesis (Six et al., 2002; Sollins et al., 1996). In one mechanism, C protection takes place by chelation of organic acids with Fe^{3+} and Al^{3+} to form metastable intermediate organo-metal complexes that slow SOC decomposition. Prevalence of this type of association is a good index for slow SOC turnover (Tate, 2002) because Al tends to promote sorption of Al-OM complexes on other surfaces and enhances polyvalent cation bonding with other OM and mineral surfaces (Rasmussen et al., 2005). The concept of physical protection emphasizes the importance of aggregation in the processes of SOC stabilization and turnover (Christensen, 2001). The location of organic matter within the soil structural units has been demonstrated to control SOM dynamics (Angers and Chenu, 1997, Feller and Beare, 1998). Recent studies indicate that the macroaggregate structure exerts some physical protection on soil organic matter (Beare et al., 1994), whereas soil organic matter is mostly protected in free microaggregates (Six et al., 1998) and in microaggregates within the macroaggregates (Bossuyt et al., 2002; Tang et al., 2010).

Biochemical stabilization of SOM in the soil matrix is a strong function of the inherent chemical and structural stability of the organic biomolecules (Krull et al., 2003). Constituents of SOC such as alkyl-C, O-alkyl-C, aromatic-C and carbonyl-C exist in varying proportions depending on the stage of SOC decomposition. Traditionally, it is assumed that, as decomposition progresses, humic materials containing high proportions of aromatic C from lignin structures accumulate in SOM, while carbohydrates and hemicellulose are preferentially utilized in decomposition processes dominated by bacteria (Sollins et al., 1996). Recently, it has been observed that the proportion of alkyl-C and carbonyl-C increases while the content of O-alkyl-C progressively decreases with increasing decomposition. As the density of the soil fractions increases, because of the preferential decomposition of the carbohydrates in the SOM, the ratio of alkyl/O-alkyl C (A:O-A) increases. Thus, either carbon content or carbonyl-C may be used as an indicator for increasing degree of natural OM decomposition resulting from oxidative-degradation processes (Baldock et al., 1997). Moreover, accumulation of alkyl-C signifies the selective preservation of recalcitrant litter materials or microbial synthesis of new alkyl compounds (Quideau et al., 2001).

Soil aggregates are dynamic in nature, the size distribution of aggregates being affected by the change in land use and management practices in tropical conditions (Spaccini et al., 2001; Ashagrie et al., 2005). The ability of the soil to function as a component of an ecosystem may be degraded, aggraded or sustained as use-dependent properties change in response to land-use and management (Fesha et al., 2002). Deuchare et al. (1999) have reported that when forest land is converted to pasture land, soils are subject to compaction and subsequent decreased porosity. On the other hand, with increasing forest age, infiltration increases and soil erosion decreases when pasture land is converted in to forest land (Carter et al., 1998). Several studies have found decreased soil aggregates stability for soils under annual crops (Angers et al., 1999); continuous tillage and arable crop production (Kavdir et al., 2005) due to the destruction of macroaggregates (Elliott, 1986), as a result, soil becomes more susceptible to erosion since macroaggregates are disturbed (Six et al., 2000). Similarly, conversion of forests into croplands known to deteriorate soil properties, especially reduce soil organic carbon (SOC) and changes in distribution and stability of soil aggregates (Singh and Singh, 1981).

Previous studies have reported some factors affecting aggregate stability as follows: clay minerals (Denef and Six, 2005), clay content (Lloyd et al., 2006), soil organic matter (Bipfubusa et al., 2008), pH (Tayel et al., 2010), cation exchange capacity (Dimoyiannis, 2011), exchangeable cations

(Bronick and Lal, 2005), and Fe_2O_3 (Barthes et al., 2008). Lloyd et al. (2006) observed that clay disperse readily with increasing soil pH. Aggregation is formed by flocculation, cementation and the rearrangement of particles (Duiker et al., 2003). In general, high content of base minerals increased the stability of soils, due to the applied chemical bonding of the aggregates (Dimoyiannis, 2011).

Glomalin contributes to the stabilization of aggregates by sloughing off hyphae onto the surrounding organic matter, binding to clays (probably via cation bridging by iron), and providing a hydrophobic coating (Wright and Upadhyaya, 1999). This is demonstrated in a number of experiments, where total and, especially, immunoreactive concentration of glomalin are positively correlated with present water-stable soil aggregates in both agricultural and native soils (Wright and Upadhyaya, 1998; Wright et al., 1999).

Different sized aggregates are bonded by different organic binding agents suggesting that different sizes contain different amounts of C, N, P and other elements. Although Badlock et al. (1987) found that total carbohydrate content significantly increased as aggregate sizes decreased for aggregates <0.5 mm in diameter, many other studies have shown that larger aggregates contain higher amounts of organic carbon and polysaccharides, for example, Christensen (1992) found high organic matter in 0.25-1 mm fractions. Dormaar (1984) observed a reduction in carbohydrate content with decreasing aggregate size when carbohydrate content was expressed per unit weight of the total soil materials collected in aggregate size fractions. The average carbohydrate contents of >250 µm diameter fraction were much higher than those of the 100 µm diameter fractions for three different soils that contained relatively low organic matter. For more intensively cultivated soils, he also found that the total carbohydrate content decreased as aggregate size decreases. It would appear that the association between carbohydrate materials of different water-stable aggregate size fractions may differ in different soils.

2.5 Soil CO_2 efflux

Soil carbon has been extensively studied because of the huge soil carbon pool in terrestrial ecosystems (Houghton et al., 2001), the large quantity of soil carbon fluxes (Raich and Schlesinger 1992; Raich et al., 2002), and its sensitivity to environmental conditions. It is still uncertain whether or not soil carbon will exert a positive feedback to global warming (Kirschbaum 1995; Trumbore et al., 1996; Cox et al., 2000; Giardina and Ryan, 2000; Kirschbaum, 2000; Lou et al., 2004). The global change and its effects on our future environment require a better understanding and quantification of the processes of greenhouse gas emission (Ohashi et al., 1999). CO_2 efflux from soil to atmosphere is a major component of greenhouse gas emission and is a crucial pathway of the C cycle. Soil respiration consists of organic matter oxidation, root respiration and rhizosphere respiration (i.e., microbial consumption of root exudates and contents of sloughed cells) (Hanson et al., 2001). Soil respired CO_2 represents the ultimate oxidative fate of soil C, and the C lost from terrestrial ecosystems occurs mainly through soil respiration (Amundson, 2001). These C losses are generally many times greater than losses from leaching and erosion (Lal, 2003), although erosion can be a major contributor in some systems (Jacinthe et al., 2004).

On a global scale, soil respiration was estimated to produce 80.4 Pg C per year (petagram carbon per year) with a range of 79.3–81.8 Pg C per year (Raich et al., 2002), accounting for 60-90 percent of total respiration of global terrestrial ecosystems (Schimel et al., 2001), which is more than 11 times the current rate of fossil fuel combustion (Butler et al., 2012). Accordingly, small changes in the magnitude of soil respiration could have a large effect on the concentration of CO_2 in the atmosphere. It is highly sensitive to temperature and global changes may have a great influence on the magnitude of soil CO_2 efflux. The potential increase in CO_2 release from the soil caused by future elevated temperature may have a positive feedback effect on atmospheric CO_2 and global change (Kirschbaum, 1995). In the context of increasing CO_2 concentration in the atmosphere and the related potential change in climate, knowledge of soil CO_2 emission is of great importance to estimate future atmospheric CO_2 concentration and global change (Liang et al., 2004). Therefore, it is important to obtain accurate estimates of soil CO_2 efflux and to understand controls on the underlying process.

Soil surface CO_2 efflux, or soil respiration, is a major component of the biosphere's carbon cycle because it constitutes about three-quarters of total ecosystem respiration (Law et al., 2001). In recent years, soil CO_2 efflux has been the subject of intense studies because of its potential and controversial role in amplifying global warming (Cox et al., 2000; Giardina and Ryan, 2000; Kirschbaum, 2000; Lou et al., 2004). Soil carbon modelers generally view soil CO_2 efflux as a function of soil temperature or a combination of soil temperature and moisture (Raich and Schlesinger, 1992; Davidson et al., 1998; Xu and Qi, 2001a). Land use change is one of several anthropogenic activities causing a global increase in the atmospheric concentration of CO_2 and other GHGs (IPCC, 2001; Houghton and Hacker, 1999). Emission of CO_2 due to land use change and deforestation together with that from the soil is estimated to be between 55±30 Gt (Giga ton = 1015 g) (IPCC, 1995) and 78±17 Gt (Lal, 1999). Land use and soil management practices significantly influence soil organic carbon (SOC) dynamics and C flux from the soil (Batjes, 1996), although the mechanisms and processes of C sequestration in soil are not completely understood (Lal et al., 1995).

The majority of soil CO_2 evolves from the metabolic activities of soil organisms (Mielnick and Dugas, 1999). Both heterotrophic and autotrophic organisms contribute to soil CO_2 efflux through respiration. An additional, non-metabolic source of soil CO_2 is chemical oxidation of soil minerals. Kelting et al. (1998) described the movement of soil carbon between three compartments: plant root tissue, the rhizosphere, and root-free soil. The rhizosphere is the soil matrix in close proximity to plant roots and supports a large microbial community that utilizes root-derived organic matter as its primary energy substrate. Root-free soil supports a smaller microbial community, deriving nourishment via secondary products diffused from soil organic matter.

Photosynthetically fixed carbon accounts for the majority of soil organic matter through litterfall, root exudates, and root mortality (van Veen et al., 1989). Faunal mortality also contributes to the organic matter pool. Heterotrophic organisms (microbes and other soil fauna) release CO_2 as they consume soil organic matter. Plant root respiration makes a significant contribution to soil CO_2 efflux when carbon fixed aboveground is assimilated by the plant into root tissue and other compounds belowground. The contribution of plant roots to overall soil CO_2 efflux varies depending on a myriad of factors including biome, season, stage of succession, and microclimate. Estimates for root respiration contribution in grasslands range from 17% to 60% (Kucera and Kirham, 1971). In temperate forests, plant roots produce an estimated 40-50% of total soil CO_2 (Ohashi et al., 1999). In assessing soil CO_2 efflux for a two-yearold pine plantation, Pangle (2000) attributed 30% of total soil CO_2 efflux on an annual basis to root respiration.

On a global basis there are wide ranging differences in both soil CO_2 efflux rates among differing biomes and vegetation types (Kirschbaum, 2000; Raich and Schlesinger, 1992; Raich and Tufekcioglu, 2000). Raich and Schlesinger (1992) summarized many studies and reported that tropical lowland forests, temperate forests and tropical grasslands had the highest annual soil respiration rates estimated at 1092, 662, and 629 g C m^{-2} yr^{-1}, respectively. Cultivated lands, temperate grasslands, and boreal forests were intermediate with rates of 544, 442, and 322 g C m^{-2} yr^{-1} each. Desert scrub vegetation, swamps and marshes, and tundra exhibited the lowest CO_2 efflux rates at 224, 200, and 60 g C m^{-2} yr^{-1} (Raich and Schlesinger, 1992).

Raich and Tufekcioglu (2000) further examined the effect of vegetation type on soil respiration rates by analyzing the results from a number of soil respiration studies where measurements were taken from forests, grasslands, and cropped fields using the same methods, same sampling period, and on sites with similar parent materials and topography. In their comparison they reported that soil respiration rates in grasslands were on average about 25% higher than nearby cropped fields. Furthermore, they reported that soil respiration rates in grasslands were on average about 20% higher than in forests with similar growing conditions (Raich and Tufekcioglu, 2000). They hypothesized that the higher rates in grasslands could be attributed to a high allocation of productivity to belowground biomass. Using the same criteria stated above they also reported that rates in broadleaf forests were approximately 10% higher than conifer forests (Raich and Tufekcioglu, 2000). This observation did not agree with the observations of Raich and Potter (1995) who found no discernable differences in soil respiration rates between deciduous and coniferous forests located in moist biomes. The influence of root biomass on soil CO_2 efflux rates is considered to be large and many studies have estimated that root respiration represents between 10% to 90% of observed soil CO_2 efflux depending on ecosystem type (Raich and Tufekcioglu, 2000).

Raich and Tufekcioglu (2000) and Rustad et al. (2001) in a summary of factors controlling soil respiration rates and carbon dioxide efflux state that the following factors are influential: 1) temperature, 2) moisture, 3) vegetation and substrate quality, 4) ecosystem net primary productivity (NPP), 5) plant rooting density and belowground biomass allocations, 6) soil physical and chemical properties, 7) population and community dynamics for above/belowground flora and fauna, and 8) land-use and/or disturbance regimes. Of these factors temperature and precipitation are considered the most influential environmental factors affecting soil CO_2 efflux rates because they interact to influence the productivity of terrestrial ecosystems and the decomposition rate of detritus/soil organic carbon (Singh and Gupta, 1977; Raich and Schlesinger, 1992). For example, the highest average soil CO_2 efflux rates are most often observed in tropical evergreen forest ecosystems where NPP is high and decomposition is not limited by either temperature or moisture constraints. In contrast, decomposition of detritus/soil organic carbon in boreal and tundra ecosystems is often constrained by low temperatures and soil aeration limitations (Schlesinger, 1997).

Temperature and precipitation are considered the most influential environmental factors affecting soil CO_2 efflux rates because they interact to influence the productivity of terrestrial ecosystems and the decomposition rate of detritus/soil organic carbon (Schlesinger, 1997; Singh and Gupta, 1977; Raich and Schlesinger, 1992). Reinforcing this notion is the fact that the highest average soil CO_2 efflux rates are usually observed in tropical evergreen forest ecosystems while the lowest rates are usually observed in tundra ecosystems. In tropical forests, primary productivity is high and decomposition rates are seldom limited by temperature or moisture constraints. In contrast, primary productivity in tundra ecosystems is low and decomposition rates are often constrained by low temperatures and excessive soil moisture.

The response of soil respiration to temperature and moisture content varies with extremities in either factor. According to Schlentner and Van Cleve (1985), temperature increases have little effect on respiration at gravimetric water contents below 75%; however, respiration is quite responsive to temperature changes at higher gravimetric water contents (100-250%). Similarly, changes in moisture content have little effect on respiration at temperatures below 5^oC while respiration is quite responsive to moisture changes at temperatures between 10^oC and 20^oC. Wildung et al. (1975) examined the effects of soil temperature and soil moisture content on soil CO_2 efflux rates in arid grassland soils. While soil temperature alone was not significantly correlated with efflux changes, they noted that soil temperature and efflux were positively correlated when volumetric soil moisture was greater than 10%. The investigators concluded that soil CO_2 efflux was controlled by the interaction of soil temperature and soil moisture content.

While the temperature dependence of soil CO_2 efflux is clear, there has been difficulty in describing the relationship mathematically. Lloyd and Taylor (1994) evaluated the adequacy of various mathematical relationships using 15 data sets collected by various researchers in terrestrial ecosystems around the world. Simple linear, exponential (Q_{10}), and Arrhenius relationships were evaluated for their ability to predict soil respiration rates in the absence of soil moisture limitations. The investigators rejected a simple linear relationship outright simply by graphing respiration as a function of temperature and observing marked curvilinearity.

According to Lloyd and Taylor (1994), the Q_{10} relationship was first described by van't Hoff in 1898 and applies to many chemical reactions with temperature dependence. The Q_{10} relationship is exponential and is defined as the ratio of the rate of respiration at temperature T to the rate at temperature T + 10^{o}C. The investigators found that the bestfit curve for the 15 data sets (n=149) gave a Q_{10} of 2.4 and explained 70% of the variation in respiration as a function of temperature. However, they also found that the exponential form underestimated respiration rates at low temperatures and overestimated at high temperatures. Based on their results, they concluded that the relationship between respiration and temperature is not a simple exponential over the normal range of physiological temperatures.

The investigators also evaluated an Arrhenius type dependence of respiration rate upon temperature. The Arrhenius equation is an exponential function that accounts for activation energy of a chemical process. However, the equation assumes constant activation energy across a range of temperatures and while more predictive than the simple exponential relationship (R^2=0.74), it too is inadequate, similarly underestimating respiration rates at low temperatures and overestimating respiration rates at high temperatures. The investigators found that the relationship between respiration and temperature was best described by an Arrhenius type equation where the effective activation energy for respiration varies inversely with temperature. The equation produced the best fit for the data set studied with R2=0.79 and provided an accurate, unbiased estimate of soil respiration rates across a wide range of ecosystem types and soil temperatures. Q_{10} values cited across a range of studies vary from 1.3 to 3.3 (Raich and Schlesinger, 1992). Similarly, Howard and Howard (1979) reported Q_{10} values that ranged from 1.96 to 2.83 over a range of soil types and temperatures.

Davidson and Ackerman (1993) reported soil C loss of approximately 20–40% after forest clearing and conversion to agriculture in the humid tropics within the first 1or 2 years following soil disturbance. Losses of soil organic C due to land clearing may result from several processes including decreased inputs and changes in composition of plant litter and increased rates of soil organic matter decomposition and soil erosion (Feller and Beare, 1997). In addition, tillage increases the rate of soil organic matter decomposition by burying surface residues, disrupting soil aggregates, aerating the soil, and exposing new surfaces to microbial attack (Brown et al., 1994). Therefore, the land use changes will affect the amount of soil organic matter loss or gain. Rapid initial losses of soil organic C following forest clearing and conversion to agriculture are primarily losses of the biologically-labile or active soil organic C pool (Brown et al., 1994). Changes in the active organic C pool can be monitored by measurement of rates of soil CO_2 efflux or soil respiration, although other methods including biological, chemical, physical, and isotopic procedures have also been proposed to distinguish active from more stable organic C pools (Townsend et al., 1995).

Compared with extensive studies in management impacts on soil carbon pools, the studies on the soil respiration affected by management actions are few. Nakane et al. (1986) found soil respiration decreased after harvesting due to the cessation of root respiration. Toland and Zhak (1994) reported that soil respiration in intact and clear-cut plots did not differ significantly between two plots because the increase in microbial respiration in clear-cut plots offset the decrease in root respiration after clear-cut. Striegl and Wickland (1998) concluded that clear-cutting in a mature jack pine woodland reduced soil respiration due to the disruption of soil surface and the death of tree roots. Studying the impact of land use change and management on soil carbon will help to address such an open question, “can we and how can we sequester more carbon in soils by management activities?”

CHAPTER III

MATERIALS AND METHODS

The materials and methods related to the study on "**Stocks and quality of soil organic matter under different land use systems in East Khasi hills of Meghalaya**" comprising of the laboratory analysis and incubation experiments are presented in this chapter.

3.1 Land use systems

North-east India is one of the resource rich regions of the country and following wide range of land use practices. The majority of the mountainous population of the north-east India depends upon agricultural and forest based natural resources for their livelihood. The soil and climatic condition of the region is suitable for growing different types of agricultural crops from cereals to fruits in both tropical and temperate climatic environment occurring on different altitudes. Significant changes in land use/land cover changes have been reported both on temporal as well as spatial scale, mainly due to the economic development and population growth. The traditional farming system in north-east is a mixture of crop, forestry and animal husbandry and more recently horticulture. The adoptions of different land uses which have its distinct nature of management are also known to have significant impact on the soil fertility status, carbon dynamics thus having a potential impact on climate change. Of the several land use practices followed, the most commonly followed land use systems are chosen and the details of the selected land use systems for the study are given in table 1. For this study three main land uses namely, agriculture, horticulture and forest were identified. In each main land uses, a total of five cropping systems including one reference site (without tree or natural fallow) were selected with varying ages.

3.2 Location of the study site

The study was carried out with different multipurpose tree species (MPTs) planted under agri-silvipastural system in 1983 (26 years old), horticulture tree species planted in 1994 (15 years old) and agricultural crops continuously cultivated for the past 10 years at Indian Council of Agricultural Research (ICAR) Complex for North-East Hill (NEH) Region, Umiam located at 25^{o}41′21″ North (latitude) and 91^{o}55′25″ East (longitude) with an altitude of 980 m above mean sea level. For each land use systems, a control plot (natural fallow) was also selected from the nearby area for comparison. The station is situated in the central part of Meghalaya in the East *Khasi* Hills of North-East India. The climate of the study area is seasonal with distinct warm-wet and cold-dry seasons in a year. The mean annual rainfall is about 2208.5 mm, of which more than 90% is received during May to October; August and September being the wettest months of the year. Mean minimum and maximum temperature varies from 6.8 ^{o}C in February to 29.7 ^{o}C in April. Relative humidity ranges between 40% during winter and 88% during rainy season. The location and annual rainfall, mean minimum and maximum temperature of the region are given in Figure 3.1 and 3.2, respectively.

3.3 Collection and preparation of soil samples

The experiments were laid out in a completely randomized block design within each land use with five treatments each under agroforestry (four MPTs and one control), horticulture (four horticulture crops and one control) and agricultural (four agricultural crops and one control) land use systems. Under agroforestry land use, each treatment had standard plot size of 130 m^2 accommodating twelve plants per plot whereas under horticulture and agricultural land uses, each treatment had the plot size of 250 m^2. The soils of the study area are classified as *Typic Paleudalf* (USDA System). Soil samples were collected during October-November in the year 2009 from 0-15, 15-30, 30-45, 45-60 and 60-75 cm soil depth at different blocks under all the treatments including control (natural fallow). For all the treatments except control, soil samples were collected from three different locations each at a distance of 1 m from the tree species under agroforestry land use, 0.5 m from the tree species under horticulture land use as suggested by Dhyani and Tripathi (1999). Soil samples from plots under agricultural land use and controls in all the land uses were collected randomly from three locations. The soil samples were brought to the laboratory, air-dried at room temperature, ground to pass through 2-mm sieve and used for the analysis of various parameters. For the determination of bulk density, undisturbed soils were collected using core samplers. About 250 g of moist soil samples were kept in deep freezer for moisture, microbial biomass carbon determination and incubation experiments.

Table 3.1 Land use systems used for the present study

Main land use	**Agriculture**	**Horticulture**	**Agroforestry**
Age (years)	10	15	26
Sub-land use	-Maize	-Pear	-Champak
	(*Zea mays*)	(*Pyrus communis*)	(*Michelia oblonga*)
	-Potato	-Peach	-Tree bean
	(*Solanum tuberosum*)	(*Prunus persica*)	(*Parkia roxburghii*)
	-Rice	-Khasi mandarin	-Alder
	(*Oryza sativa*)	(*Citrus reticulata*)	(*Alnus nepalensis*)
	-Turmeric	-Guava	-Khasi pine
	(*Curcuma longa*)	(*Psidium guajava*)	(*Pinus kesiya*)
	-Control (No crop)	-Control (No tree)	-Control (No tree)

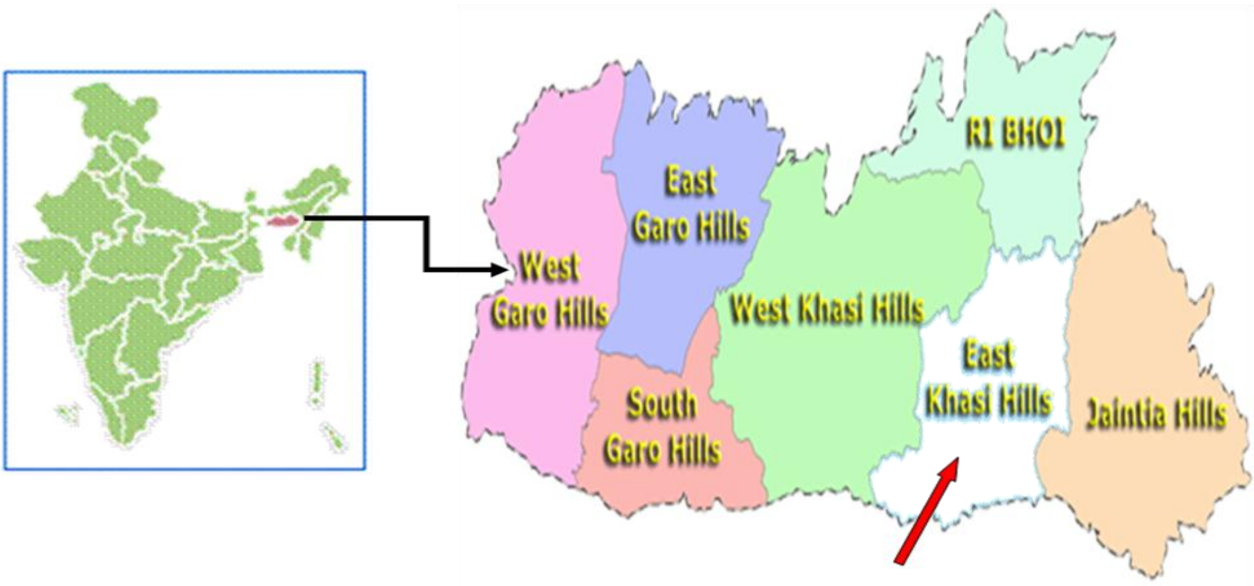


Figure 3.1 Location of the study site

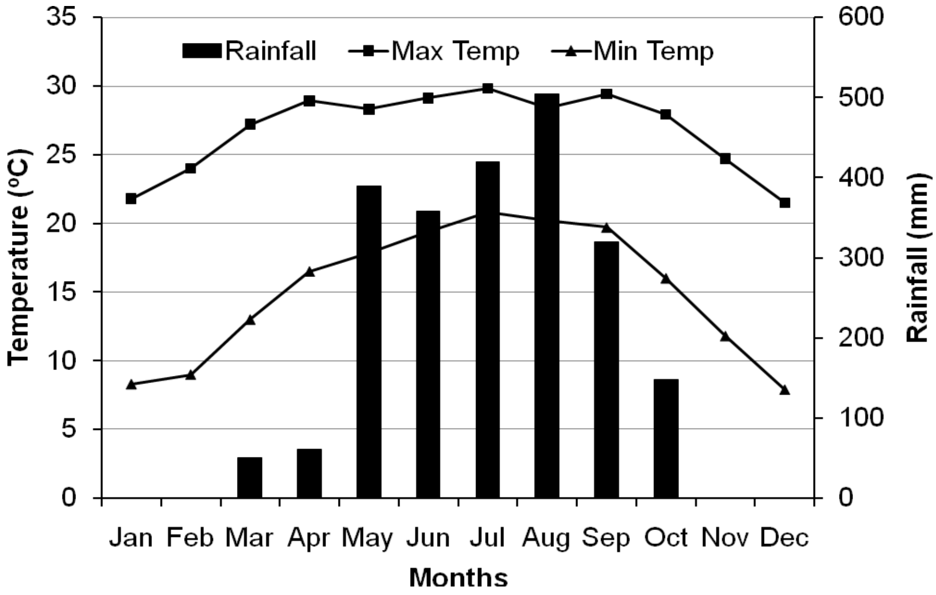


Figure 3.2 Mean minimum and maximum temperature of the region selected for the study during 2009.

3.4. General characteristics of the soils

3.4.1 Soil reaction (pH)

The pH of the soil samples was determined in 1: 2.5 (soil: water) suspension using combined electrode (glass and calomel) in a digital pH meter (Jackson, 1973).

3.4.2 Electrical conductivity (EC)

The electrical conductivity was measured in the supernatant liquid of the soil water suspension (1: 2.5) with the help of Conductivity Bridge and expressed in dS m^{-1} at 25 °C.

3.4.3 Mechanical analysis and moisture

Particle size distribution of the soil was determined by using Bouyoucos hydrometer (Bouyoucos, 1962) and textural class was deduced with the help of International triangular chart. Soil moisture content was determined gravimetrically by oven drying 10 g of fresh soil (Allen, 1989).

3.4.5 Mean-weight diameter (MWD)

The mean-weight diameter (MWD) was determined on air-dried samples, pre-sieved through a 4 mm mesh according to the procedure outlined by Kemper and Chepil (1965). Essentially this involved weighing 20 g of the < 4 mm aggregates into the topmost sieve in a nest of five sieves of diameters 2, 1, 0.5, 0.25 and 0.1 mm and pre-soaking in distilled water for 10 min. Thereafter, the whole sieve set and its contents were oscillated in water for 20 times along a 4 cm stroke at the rate of 1 oscillation per second. Care was taken to ensure that the soil on the topmost sieve was under the water level throughout the period of oscillation. After the oscillation the resistant aggregates on each sieve were collected quantitatively, oven-dried at 105 °C for 24 hours and their masses recorded. The resistant aggregates were thus categorized into the following size ranges: 4-2 mm, 2-l mm, 1-0.5 mm, 0.5-0.25 mm and < 0.25 mm. The last class was obtained as the difference between the original mass of the soil that was sieved (i.e., 20 g) and the sum of the first four classes. The respective masses were used to compute the mean-weight diameter as,

$$\text{MWD} = \sum_{j=1}^{k} Wj \, \bar{\text{X}}\text{j}$$

where, MWD = mean-weight diameter, (mm); $\bar{\text{X}}\text{j}$ = arithmetic mean diameter of the j-1 and *j* sieve openings (mm); *Wj* = proportion of the total sample weight (uncorrected for sand and gravel) occurring in the fraction (dimensionless); and k = total number of size fractions (in this case 5). Three replicate determinations were made on each soil sample.

3.4.6 Available nutrients

Available nitrogen content was determined by alkaline permanganate oxidation method using Kjeldahl distillation unit (Subbaiah and Asija, 1956). Phosphorous content was measured by using colorimetrically with a molybdenum blue reagent after 0.03 mol L^{-1} NH_4F- 0.025 mol L^{-1} HCl extraction (Bray and Kurtz, 1945). Available-K was determined by using flame photometrically after 1 *N* NH_4OAc extraction. For sulphur, soil samples were extracted with 0.15% $CaCl_2$ (Hanway and Heidel, 1952) and sulphur content in the extracts was determined by turbidimetric method (Williams and Steinbergs, 1969). For calcium and magnesium, soil samples were extracted with 1 *N* NH_4OAc and extractant was analysed for Ca and Mg by following EDTA titration method (Cheng and Bray, 1951). Available micronutrients like Fe, Mn and Cu content were measured by extraction with DTPA-$CaCl_2$-TEA solution and then analyzing the extract by atomic absorption spectrometer (Lindsey and Norvell, 1978).

3.4.7 Organic carbon

Organic carbon was determined in soil samples passed through 100 mesh sieve by wet digestion method of Walkley and Black (1934) as described by Jackson (1973).

3.4.8 Total organic carbon

Total organic carbon in soil was determined by wet oxidation method (Synder and Trofymow, 1984). For this purpose, 1.0 g of soil (passed through 1mm sieve) was pre-treated with 3.0 ml of 2 *N* HCl to remove carbonates, then soil was oxidized with $K_2Cr_2O_7$ in presence of 25 ml of conc. H_2SO_4 and H_3PO_4 in a ratio of 3:2, by heating on digestion block for 2 hrs. Thus evolved CO_2 was trapped in 2 *N* NaOH and amount of CO_2 (entrapped) was measured by back titration with 0.5 *N* HCl using phenolphthalein indicator. Total organic carbon content was computed based on the amount of evolved CO_2.

3.4.9 Carbon stock calculations

The carbon stocks were estimated to a depth of 75 cm. Total organic carbon and bulk density were used to calculate the carbon stocks. For total organic carbon the method described by Snyder and Trofymow (1984) as described in the following section was followed. Soil bulk density values were determined by the core method, using oven-dried soil mass and field volume of sample (Allen et al., 1974). Calcium carbonate estimation in the soils was carried out using HCl titrations and inert carbon in the soils was considered. The inert carbon in the soil was computed using below equation (Falloon et al., 1998):

$$C_I = 0.049\ C_T^{1.139}$$

where, $C_{I\ is}$ the inert carbon content (t ha^{-1}), and C_T is total organic carbon in soil.

C stocks (Mg ha^{-1}) = C_{TOT} * BD * D where, C_{TOT} is total organic carbon (g 100^{-1});

BD= Bulk density (Mg m^{-3})

D= Depth (cm)

3.4.10 Labile carbon

The amount of oxidizable carbon by 333 *mM* $KMnO_4$ (labile carbon) in soil was determined by following the procedure of Blair et al. (1995). For this purpose, 2.0 g of soil was taken in centrifuge tube and oxidized, with 25 ml of 333 *mM* $KMnO_4$ by shaking in a mechanical shaker for 1 hour. The tubes were centrifuged for 5 minutes at 4000 rpm and 1.0 ml of supernatant solution was diluted to 250 ml with double distilled water. The concentration of $KMnO_4$ was measured at 565 nm wavelength using spectrophotometer. The change in concentration of $KMnO_4$ is used to estimate the amount of carbon oxidized assuming that 1.0 *mM* of MnO_4 was consumed (Mn VII – Mn IV) in the oxidation of 0.75 *mM* (9.0 mg) of carbon.

3.4.11 Carbon Management Index (CMI)

Blair et al. (1995) proposed Carbon Management Index (CMI), a multiplicative function of Carbon Pool Index (CPI) and Lability Index (LI) as an indicator of the rate of change of soil organic matter in response to land management changes, relative to a more stable reference soil.

Non-labile carbon (C_{NL}) was calculated from the difference between total carbon (C_T), and labile carbon (C_L). The relative amounts of these two fractions and the total carbon in a cropped and reference soil have been used by Blair et al. (1995) to calculate a CMI. This index compares the changes that occur in total and labile carbon as a result of agricultural practice, with an emphasis on the changes in C_L, as opposed to C_{NL} in SOM. The CMI is calculated as follows:

a) Change in total C pool size:

The loss of C from a soil with a large carbon pool is of less consequence than the loss of the same amount of C from a soil already depleted of C or which started with a smaller total C pool. Similarly, the more a soil has been depleted of carbon the more difficult it is to rehabilitate. To account for this a CPI is calculated as follows:

$$\text{CPI} = \frac{\text{Sample total carbon (mg g}^{-1}\text{)}}{\text{Reference total carbon (mg g}^{-1}\text{)}} \qquad \frac{C_T \text{ sample}}{C_T \text{ reference}}$$

b) The loss of C_L is of greater consequence than the loss of C_{NL}. To account for this, since it is the turnover of labile carbon which releases nutrients and the labile carbon component of SOM appears to be of particular importance in affecting soil physical factors (Whitbread et al., 1998) a LI of C is calculated as follows:

$$\text{Lability of C} = \frac{\text{C in fraction oxidized by KMnO}_4 \text{ (mg labile C g}^{-1}\text{ soil)}}{\text{C remaining unoxidized by KMnO}_4 \text{ (mg labile C g}^{-1}\text{ soil)}}$$

$$= \frac{C_L}{C_{NL}}$$

$$\text{Lability Index (LI)} = \frac{\text{Lability of C in sample soil}}{\text{Lability of C in reference soil}}$$

c) The Carbon Management Index (CMI) is then calculated as follows:

$$\text{Carbon Management Index (CMI)} = \text{C Pool Index * Lability Index *100}$$

$$= \text{CPI *LI * 100}$$

3.4.12 Microbial biomass carbon

Microbial biomass carbon (MBC) was determined by the chloroform fumigation incubation (CFI) technique as per the procedure of Jenkinson and Powlson (1976). Shade-dried, 0.5-mm-sieved soil (10 g) was placed in a 50 mL beaker in which 1.0 mL of distilled water was added and kept in an air-tight desiccator. Fumigation was carried out with ethanol-free chloroform by applying vacuum until the chloroform clearly evaporated. The tap of the desiccator was closed, and the desiccator was kept in the dark for 5 days. After incubation, the soils were transferred into 125 ml extraction bottles, and the samples were shaken with 0.5 *M* potassium sulphate (K_2SO_4) for 30 minutes and filtered through Whatman No. 42 filter paper. The MBC was calculated from the net amount of total C (fumigated C and nonfumigated C) using a factor of 2.64 (Vance et al., 1987). Microbial quotient was calculated from the ratio of microbial biomass carbon (C*mic*) to the total organic carbon (C*tot*).

3.4.13 Particulate organic carbon

Particulate soil organic matter is a labile intermediate in the soil organic matter continuum from fresh organic materials to humified matter. The isolation of SOM particulate from the mineral-associated fraction was performed by physical fractionation (Cambardella and Elliott, 1992). The soil samples were air-dried, crushed with a wood roll and sieved (< 2 mm) and stored in plastic pots. Soil subsamples of 20 g were placed in snap-cap flasks and dispersed with 60 mL of sodium hexametaphosphate [$(NaPO_3)_6$] solution at 8.17 mmol L^{-1} (5 g L^{-1}) and horizontal shaking (60 cycles min^{-1}) for 15 hrs. Thereafter the suspension was passed through a 53 µm mesh and washed with distilled water to separate organic material from sand. The material retained in the sieve was considered the particulate fraction and the material that passed through the sieve was considered the mineral-associated fraction, which was collected in a plastic bucket. The particulate associated fractions were dried at 90 ºC initially and then at 50 ºC until dry. After drying and weighing, the particulate fraction samples were ground with a pestle and a mortar for C analysis.

3.4.14 Extraction and characterization of soil organic matter

The soil samples were first equilibrated to a pH of 1.0 to 2.0 with 1 *M* HCl at room temperature, and adjusted solution volume with 0.1 *M* HCl to provide a final concentration that had ratio of 10 mL liquid $1g^{-1}$ dry sample. The suspension was shaken for 1 hour and the supernatant was separated from the residue by decantation after allowing solution to settle (or by low speed centrifugation). Neutralized the soil residue with 1*M* NaOH to pH=7.0 and then added 0.1 *N* NaOH under an atmosphere of N_2 to give a final extractant to soil ratio of 10:1. Extraction was done under N_2 atmosphere with intermittent shaking for a minimum of 4 hours and allowed the alkaline suspension to settle overnight and collected the supernatant by means of centrifugation. The supernatant was acidified with 6 *M* HCl with constant stirring to pH=1.0 and then allowed the suspension to stand for 12-16 hours. The humic acid (precipitate) and fulvic acid (supernatant - FA Extract 2) fractions were separated by centrifugation.

The extracted humic acids were purified by removing the impurities in it. This was done first by re-dissolving the humic acid fraction by adding a minimum volume of 0.1 *M* KOH under N_2 atmosphere. The solution was treated with solid KCl to attain 0.3 *M* (K^+) and then centrifuged at high speed to remove suspended solids. The supernatant was acidified with 6 *M* HCl with constant stirring to pH=1.0 so as to re-precipitate the humic acids. The supernatant was separated from the humic acid precipitate by centrifugation. The humic acid precipitate was suspended in 0.1 *M* HCl/0.3 *M* HF solution in a plastic container and was shaken overnight at room temperature. Centrifugation and HCl/HF treatment was repeated, if necessary, until the ash content was below 1 per cent. The precipitate was transferred to a Visking dialysis tube by slurring with water and dialyzed against distilled water until the dialysis water gave a negative Cl^- test with the $AgNO_3$. The humic acid was freeze-dried and used for estimation of Carbon and Nitrogen content, functional groups and E4/E6 ratio (Schnitzer, 1982; Stevenson, 1994).

3.4.14.1 Total acidity, carboxylic and phenolic OH

Total acidity and carboxylic OH groups were determined by the procedure suggested by Schnitzer and Gupta (1965). The phenolic OH groups (meq g^{-1} of humic substances) were calculated from the difference between total acidity and carboxylic OH groups as described below.

Calcuim acetate ($Ca(OAc)_2$) method for carboxylic OH (COOH groups)

To 50 to 100 mg samples of humic substances in a 125-ml ground-glass stoppered Erlenmeyer flask 10 ml of 1 *N* $Ca(OAc)_2$ solution and 40 ml of CO_2-free distilled water were added. A blank was set up simultaneously, consisting of 10 ml of 1 *N* $Ca(OAc)_2$ and 40 ml of CO_2-free distilled water only. After shaking for 24 hours at room temperature, the suspension was filtered, the residue washed with CO_2-free distilled water and the filtrate plus washings titrated potentiometrically (glass-calomel electrodes) with standard 0.1 *N* NaOH solution to pH 9.8. The following calculation was used:

$$COOH\ groups\ (meq/g\ of\ humic\ substances) = \frac{(\mathrm{T.V\ for\ sample - T.V\ for\ blank)\ x\ N\ of\ base\ x\ 1000}}{\mathrm{Weight\ of\ sample\ (mg)}}$$

Barium hydroxide ($Ba(OH)_2$) method for total acidity

To 50 to 100 mg of organic matter in a 125 ml ground-glass stoppered Erlenmeyer flask, 20 ml of 0.25 *N* $Ba(OH)_2$ was added. Simultaneously, a blank was set up consisting of 20 ml of 0.25 *N* $Ba(OH)_2$ only. The air in the flask was displaced by N_2 and the system shaken for 24 hours at room temperature. Following this, the suspension was filtered, the residue washed thoroughly with CO_2-free distilled water and the filtrate plus washings titrated potentiometrically (glass-calomel) with standard 0.5 *N* HCl solution to pH 8.4. The calculation was as follows:

$$Total\ acidity\ (meq/g\ of\ humic\ substances) = \frac{(\mathrm{T.V\ for\ blank - T.V\ for\ sample)\ x}\ N\ \mathrm{of\ acid\ x\ 1000}}{\mathrm{Weight\ of\ sample\ (mg)}}$$

3.4.14.2 Elemental composition and E4/E6 ratio

The carbon and nitrogen content of the humic acid was determined by the procedure outlined by Chen et al. (1977). The E4/E6 ratios were determined from the absorbances at 465 and 665 nm (E4/E6) after dissolving 2 mg HA or FA in 10 ml 0.05 *N* $NaHCO_3$ (0.01 to 0.05% (wt/vol) of HA or FA) (Chen et al., 1977). Soil pH measurements were done with pH meter; pH's were adjusted with dilute NaOH and HC1 solutions. With HA's, E4/E6 measurements were possible only at pH 7 or higher because these materials were insoluble in distilled water at lower pH levels. This problem, however, was not encountered with the FA and the fractions derived from it, and it was possible to determine E4/E6 ratios over the pH range 1-12. Optical densities of all solutions were measured at 465 and 665 nm on spectrophotometer.

3.4.14.3 Fourier transform infrared spectroscopy (FTIR)

The FTIR spectra of HA samples were recorded on a Win-IRrez (Bio-Rad, Hercules, CA, USA) using the potassium bromide (KBr) disc technique. The samples (2 mg) were mixed with potassium bromide (about 100 mg) in a clean glass pestle and mortar and compressed to obtain a pellet. The base line was corrected and scanning was performed from 4000–400 cm^{-1}.

3.5. LABORATORY EXPERIMENTS

The soil samples were first air dried, sieved and stored under room condition. The soils were characterized for physico-chemical properties as detailed above. The protocols used for separation and characterization of soil aggregates to realize the set objectives are detailed below.

3.5.1. Characterization of macro (>250 µm) and micro (<250 µm) aggregates and thermal stability

About 500 g of air-dried soil samples were passed through a 250 µm sieve to separate them into macro (>250 µm) and micro aggregates (<250 µm). Macro (>250 µm) and micro aggregates (<250 µm) separated from the soils were used for studying the carbon and nitrogen stabilization in them (Puget et al., 2000; Six et al., 1998). The aggregates were characterized for initial carbon and nitrogen content (Jackson, 1973). Following parameters were also analysed in the aggregates.

3.5.1.1 Polysaccharides

Total and dilute acid extractable polysaccharides were performed on soil/ aggregate samples from the field. The total and dilute acid-extractable polysaccharides were analyzed with the technique modified from Whistler and Wolfrom (1962) by Lowe (1994). For analysis of the total polysaccharides, a 0.5 g subsample of a soil fraction was put in an Erlenmeyer flask, and 4.0 mL of 12 *M* H_2SO_4 was added to the flask. The flask was covered with a large glass sphere and left to stand for 2 hrs. The H_2SO_4 in the flask was diluted to 0.5 *M by* adding 92 mL distilled water, and the flask was autoclaved for 1 hour (103 kPa, 121 ^{o}C). After cooling, the polysaccharides in the flask were distilled through 9 cm Whatman # 2 filter paper, and distilled water was added to the filtrate to a volume of 100 mL for analysis. One mL of the filtrate was pipetted into a cuvette, and 1 mL phenol solution and 5 mL of concentrated H_2SO_4 were added. After standing for 10 min, the cuvette was placed in a water bath at 25 to 30 ^{o}C for 25 min. Absorbance at 490 nm of the aliquot was read on a spectrophotometer. A standard curve was prepared by dilution of stock glucose solution to determine polysaccharide content in the sample. The analysis of dilute acid extractable polysaccharide was the same as that of total polysaccharides except that 100 mL 0.5 *M* H_2SO_4 (not 12 *M* H_2SO_4) was added directly into the flask with the soil/ aggregate sample.

3.5.1.2 Glomalin content

1g of soil/ aggregate samples was extracted with 50 *mM* sodium citrate, pH 8.0, at 121 ^oC for 1 hr (Wright and Upadhyaya, 1999). Extraction was repeated until the supernatant was straw-coloured (up to three more times). All supernatants from each 1 hr extraction were combined. A modified Bradford protein assay (Wright et al., 1996) was used to measure glomalin content in the extracts. Samples were diluted in PBS (phosphate buffered saline) and reacted with Bio-Rad® (Hercules, CA) Bradford protein dye reagent. Absorbance was read at A_{595} after 5 mins. Glomalin concentration was determined by comparison with a bovine serum albumin (BSA) standard curve.

3.5.1.3 Thermal stability of macro (>250 μm) and micro (<250 μm) sized soil aggregates

Incubation experiment was carried out to study the temperature response of aggregates (macro and micro) at 15, 30 and 60 days. The samples were incubated in wide mouth bottles at 25 ^oC, 30 ^oC and 35 ^oC. The treatments were replicated thrice. Aggregates were analyzed for the carbon and nitrogen content (Jackson, 1973) after the specific time period at different temperature gradients for the calculation of the percent carbon remained in the aggregates. During the experiment, at most care was taken to maintain the moisture content near to field capacity. By using these data, the temperature sensitivity of the carbon and nitrogen losses were calculated by using Arrhenius equation **(k= A exp (-E/ RT)** and Q_{10} values (Knorr et al., 2005).

3.5.2 CO_2 efflux studies from soils as affected by temperature and land uses

A 150 days incubation experiment was carried out in the laboratory. The stored moist soil samples equivalent to 100 g (oven dry basis) were placed into Schott jars after 20 mL of deionized water was carefully mixed into them. Small cups filled with 25 mL of 1 mol L^{-1} NaOH were placed inside the jars to trap the evolved CO_2. The jars were fastened airtight and incubated at three temperatures: 25, 30 and 35 ^oC. The selection of the temperatures were based on the range, mean annual minimum and maximum of temperatures experienced at the experimental sites of the region (Fig. 2). For each temperature, a series of blanks without soil were included, using cups only filled with 25 mL of 1 mol L^{-1} NaOH in the Schott jars to trap the CO_2 from the atmosphere enclosed in the jars. Moisture of the soil samples was periodically adjusted to a value of 60% of the soil water holding capacity (WHC). The constant soil moisture was maintained by weighing each sample once a week and adjusting the water content to a target value. Three replications were set up for each treatment of the experiment. The CO_2 that evolved from the soil was measured at 3, 6, 9, 12, 15, 20, 30, 45, 60, 90, 120 and 150 days after incubation by way of titrating the NaOH solution against 0.1 mol L^{-1} HCl after the addition of $BaC1_2$. NaOH solution, incubated as described above, was also titrated but without incorporating any soil (blank). The CO_2 efflux data thus generated for different temperature gradients was used to calculate the temperature dependency of CO_2 efflux using Arrhenius activation energy (k= A exp (-E/ RT) by plotting 1 / ln k against T and Q_{10} value (Knorr et al., 2005). The Q_{10} value is calculated using the following equation (Metcalf and Eddy, 1991).

$$Q_{10} = [k_2/k_1]^{10/\Delta T}$$

where, k_1 and k_2 are the rate constants calculated at temperature T_1 and T_2 and ΔT is the change in temperature from T_1 to T_2.

3.5.2.1 Specific respiratory activity

Specific respiratory activity of soil microbial biomass carbon was estimated by dividing the CO_2-C produced by the value of soil microbial biomass following the procedure of Campbell et al. (1991).

3.6 Statistical analysis

Data obtained from this research programme was statistically analysed following standard statistical methods (Gomez and Gomez, 1984). Analysis of variance method to elucidate the effect of different land use systems and depth on the various soil physical, chemical and biological properties, soil organic pools and their stabilization was carried out using computer based statistical programme SPSS (version 16.0). Duncan's Multiple Range Test (DMRT) coefficients have been used to segregate significance of difference among the mean values obtained for various parameters studied during the current investigation. Unless otherwise stated, the level of significance referred to in the results is $p<0.05$. Further correlation study was carried out to access the relationship between different parameters studied in this investigation.

CHAPTER IV

Effect of Land Use Systems on Soil Properties in Subtropical Region of Meghalaya, India

4.1 Abstract

Land conversion from fallow to agroforestry (AF), horticulture fruit trees plantation and agriculture crop cultivation in general resulted in higher soil organic carbon, available nitrogen (N), phosphorus (P) and potassium (K) content. Amongst the land use systems, AF land use recorded maximum values for all these parameters compared to horticulture and agriculture land uses. The decline in soil organic carbon, available N and K content in the soils of cultivated land was to the extent of 30.4, 17.8 and 17.2%, respectively as compared to the AF lands and 1.8, 13.9 and 22.1%, respectively as compared to horticulture tree species. Soil bulk density, moisture content and mean weight diameter (MWD) and other soil chemical properties studied were significantly affected by land uses ($P \leq 0.05$ and/or $P \leq 0.01$). Soil moisture and MWD values were highest under AF plantation (28.3 g $100g^{-1}$ and 2.08 mm, respectively) except soil bulk density which was at lowest level (1.22 Mg m^{-3}). In contrast, soil pH, available P and exchangeable Mg in AF land use; moisture content and available P in horticulture land use were not significantly ($P > 0.05$) different. Highest average mean values of exchangeable Ca (2.31 meq $100g^{-1}$), exchangeable Mg (1.01 meq $100g^{-1}$) and available P (20.3 kg ha^{-1}) and sulphur (3.83 kg ha^{-1}) were observed under the horticulture land compared to the other two land uses. With respect to all the attributes studied except soil pH, the values were highest in AF land use and lowest in agriculture land use. Furthermore, considering the soil depths, higher mean values of organic carbon (1.91 g $100g^{-1}$), available N (446.0 kg ha^{-1}), P (35.14 kg ha^{-1}), K (325.7 kg ha^{-1}), S (4.05 kg ha^{-1}), exchangeable Ca (2.65 meq $100g^{-1}$), and Mg (1.07 meq $100g^{-1}$) were recorded in the surface (0-15 cm) soil layer than in the subsurface (15-75 cm) soil layers. In dissimilarity with the above observation, soil bulk density was highest in the subsurface soil layers (1.42 g cm^{-3}) compared to the surface soil layer (1.22 g cm^{-3}). Soil microbial biomass concentration was maximum under AF tree plantation (425.1 mg kg^{-1}) and lowest in agriculture land use (272.9 mg kg^{-1}). Correlations showed that soil pH was positively and significantly correlated with basic cations such as K, Ca and Mg whereas; it had negative and significant correlation with micronutrient cations (Fe and Mn). Soil organic carbon showed positive and significant correlation with available N, Fe and Mn, MWD and soil MBC whereas, as expected, it showed negative and significant relation with soil Cu. The correlation between SOC with available P and soil P with Fe and Mn was not significant.

Key words: Land use systems, MWD, soil fertility, microbial biomass, Meghalaya.

4.2 Introduction

Soil is an important component of terrestrial ecosystems because it preserves nutrient reserves, supports many biological processes, related to nutrient cycles, and filters, keeps and transforms pollutants reducing their toxic effect. To preserve this resource and its functions, it is necessary first of all to know the conditions and the processes occurring in it, for example, through the determination of soil quality (Karlen et al., 1997). Soil quality is a combination of soil physical, chemical and biological properties that are able to readily change in response to variations in soil conditions (Brejda et al., 2000). Islam and Weil (2000) and Sanchez et al. (2002) have reported that soil quality may be affected by land use type and agricultural management practices because these may cause alterations in soil physical and chemical properties and in soil biotic community determining, in turn, a reduction in land productivity. Fromm et al. (1993) also described the dominance of soil land use on biological soil quality over soil type.

Maintenance of soil fertility requires preservation of its organic matter, physical properties and nutrient levels. Due to the vulnerability of the soils in north-east India for several environmental factors and human activities, all the agricultural or land development related activities in the north-east region must consider soil fertility and its maintenance as a priority. Several studies establish correlations between the changes in the physical and biological properties, indicating that it is first necessary to know the soil characteristic of each type of soil in order to prevent the negative effects caused by human activity (Wilding, 1999). However, the effects of conversion of different land use types in the north-east Indian situation of shifting cultivation and the impacts are incomplete.

The physical properties of soils determine their adaptability to cultivation and the level of biological activity that can be supported by the soil. Soil physical properties also largely determine the soil's water and air supplying capacity to plants. Many soil physical properties change with changes in land use system and its management such as intensity of cultivation, the instrument used and the nature of the land under cultivation, rendering the soil less permeable and more susceptible to runoff and erosion losses (Sanchez, 1976).

Soil chemical properties are the most important among the factors that determine the nutrient supplying power of the soil to the plants and microbes. The chemical reactions that occur in the soil affect processes leading to soil development and soil fertility build up. Minerals inherited from the soil parent materials overtime release chemical elements that undergo various changes and transformations within the soil.

Various farming systems models have been developed in Meghalaya, north-east India, integrating agroforestry, horticulture, livestocks, aquaculture, etc. to complement soil conservation measures for long-term sustainable production and also to replace the slash and burn cultivation in view of the socio-economic conditions, topography, soil and climatic conditions for livelihood, income generation and food security of the inhabitants (Bhat et al., 2006). Agroforestry is an ideal scientific approach for eco-restoration of degraded lands and sustainable management. Several studies have described the beneficial effects of agroforestry systems, compared to horticulture and agriculture land uses, in long-term soil productivity and sustainability (Kirby and Potvin, 2007; Nair et al., 2009) but the magnitude of the beneficial effects may vary with a number of site specific factors and attributes of associated tree species. Increased nutrient inputs and recycling, reduction in nutrient losses, and improved soil physical properties are all characteristics of agroforestry systems as compared to sole cropping systems under hilly ecosystems (Nair, 1993). Udawatta et al. (2008b) reported improved soil aggregate stability, nutrients availability and microbial activity under agroforestry systems in comparison to other land use systems. Although directly fulfilling the multifaceted needs of farmer's *viz.*, fuel wood, timber, feed, etc. agroforestry systems indirectly provides many ecological benefits especially storing carbon through C sequestration both in tree biomass and soil. Therefore, it is crucial to evaluate the soil fertility status under different land use systems particularly in the hilly ecosystems of Meghalaya where the level of land degradation is very high. With this background, the present study was carried out to evaluate the impact of different land use systems on soil fertility in terms of physical, physico-chemical, chemical and biological proprties and to select the best land uses which can restore the soil fertility for long-term sustainability and provide ecological benefits through increased carbon sequestration.

4.3 Materials and methods

4.3.1 Location of the study site

The study was carried out with different multipurpose tree species (MPTs) planted under agri-silvipastural system in 1983, horticulture tree species planted in 1994 and agricultural crops continuously cultivated for the past 10 years at Indian Council of Agricultural Research (ICAR) Complex for North-East Hill (NEH) Region, Umiam. The station is situated in the central part of Meghalaya in the East *Khasi* Hills of North-East India.

4.3.2 Collection and preparation of soil samples

The experiments were laid out in a completely randomized block design within each land use with five treatments each under agroforestry (four MPTs and one control), horticulture (four horticulture crops and one control) and agricultural (for agricultural crops and one control) land use systems. Under agroforestry land use, each treatment had standard plot size of 130 m^2 accommodating twelve plants per plot whereas under horticulture and agricultural land uses, each treatment had the plot size of 250 m^2. The soils of the study area are classified as *Typic Paleudalf* (USDA System). Soil samples were collected during October-November in the year 2009 from 0-15, 15-30, 30-45, 45-60 and 60-75 cm soil depth at different blocks under all the treatments including controls (natural fallow). For all the treatments except control, soil samples were collected from three different locations each at a distance of 1-m from the tree species under agroforestry land use, 0.5-m from the tree species under horticulture land use as suggested by Dhyani and Tripathi (2000). Soil samples from plots under agricultural land use and controls in all the land uses were collected randomly from three locations. The soil samples were brought to the laboratory, air-dried at room temperature, ground to pass through 2-mm sieve and used for the analysis. For the determination bulk density, undisturbed soils were collected using core samplers. About 250g of moist soil samples were kept in deep freezer for moisture, microbial biomass carbon determination and incubation experiments.

4.3.3 Methods of analysis

The soils were characterized for different physico-chemical properties and their detailed protocols are followed are listed in general Material and Methods chapter: Soil pH, electrical conductivity, particle size distribution and soil texture, soil moisture, soil bulk density, mean weight diameter, organic carbon, available soil nutrients (N, P, K, S, Ca, Mg, Fe, Cu, Mn) and soil microbial biomass carbon.

4.3.4 Statistical analysis

Data obtained from this research programme was statistically analysed following standard statistical methods (Gomez and Gomez, 1984). Analysis of variance method to elucidate the effect of different land use systems and depth on the various soil physical and chemical properties was carried out using computer based statistical programme SPSS (version 16.0). Duncan's Multiple Range Test (DMRT) coefficients have been used to segregate significance of difference among the mean values obtained for various parameters studied during the current investigation.

4.4 Results

4.4.1 Physical properties

The effects of different land use systems on soil bulk density (BD) and moisture content are given in table 4.1 and 4.2. Adoption of various agroforestry (AF) tree species, fruit trees and agriculture crops significantly influenced the soil BD and moisture content. Significant changes due to soil depths on both soil BD and moisture content were also observed in all the three land use systems but interaction of land use systems and depth did not show any considerable changes on these attributes. Soil BD and moisture content ranged from 1.14-1.30, 1.36-1.10 and 1.31-1.39 Mg m^{-3} and 24.6-32.2, 23.0-29.6 and 25.8-28.8 g $100g^{-1}$ in AF, agriculture and horticulture land use systems, respectively. In AF land use Khasi pine showed highest soil moisture content (32.2 g $100g^{-1}$) and lowest soil BD (1.14 Mg m^{-3}); in agriculture land use, rice recorded highest moisture content (29.6 g $100g^{-1}$) and maize showed lowest BD (1.36 Mg m^{-3}); in horticulture land use, peach recorded highest soil moisture (28.8 g $100g^{-1}$) and guava showed lowest BD (1.31 Mg m^{-3}) compared to their respective control plots (24.6 and 1.30; 23.0 and 1.40; 25.8 and 1.39, respectively). Irrespective of the land uses, soil moisture content was highest in surface soil than the subsurface soil layers (30.2, 28.5 and 30.0 g $100g^{-1}$ in AF, agriculture and horticulture land use, respectively). On the other hand, soil BD was lowest in surface layer (0-15 cm) (1.11, 1.30 and 1.26 Mg m^{-3} in AF, agriculture and horticulture land use, respectively) and highest in subsurface (15-75 cm) soil layers (1.32, 1.49 and 1.44 Mg m^{-3} in AF, agriculture and horticulture land use, respectively). Agriculture crops and fruit trees did not show considerable variation on mean weight diameter (MWD) while AF tree species showed significant influence on MWD (Fig. 4.3). Khasi pine (2.36 mm), turmeric (1.69 mm) and Khasi mandarin (2.04 mm) showed highest MWD compared to their respective controls in AF (1.37 mm), agriculture (1.07 mm) and horticulture (1.22 mm) land uses. In general, 31, 27 and 12 % content and 65, 46 and 55 % increase in MWD and 14, 4 and 6 % decrease in soil BD was noticed in AF, agriculture and horticulture land uses, respectively. Overall, highest soil moisture content (Fig. 4.2) and MWD (Fig. 4.4) and lowest soil BD (Fig. 4.1) was recorded in AF tree plantation compared to agriculture and horticulture land uses. Table 4.15 reveals that soil BD was positively correlated with soil moisture (r=0.057) and negatively and significantly correlated with MWD (r=0.970**). On the other hand, MWD showed positive and significant correlation with soil moisture (r= 0.606*).

4.4.2 Chemical properties

It was found that there was a significant difference between the land use systems and soil depths on soil organic carbon (Table 4.4). With respect to the soil pH, only agriculture and horticulture land uses showed considerable variations and AF land use did not show any effects on soil pH. In all the three land uses, soil depths had noteworthy effect on soil pH and organic carbon content but interaction of land uses and soil depths did not show any significant effect on these attributes (Table 4.14). In AF land use, the mean soil pH was lower under tree species compared to control and varied from 4.54 in the upper (0-15 cm) soil layer to 4.29 in the lower (60-75 cm) soil layer (table 3). Among the tree species, Khasi pine (4.29) had the lowest soil pH followed by Alder (4.33) and highest soil pH was recorded in control (4.59). In contrast to AF, agriculture and horticulture land uses had lower soil pH in control plots compared to fruit trees and agricultural crops (Table 4.3). In agricultural land use, the mean soil pH varied from 4.68-4.98 with highest value in turmeric (4.98) followed by potato (4.93) and lowest was in control (4.68). Soil pH varied from 4.66-6.02 in horticulture land use with highest value in peach (6.02) followed by pear (5.37), Khasi mandarin (5.27) and lowest in control (4.66). Among the three main land uses, horticultural land use recorded highest pH followed by agriculture and the lowest soil pH was recorded in agroforestry land use (Fig. 4.5). Overall, adoption of agricultural crops and horticulture fruit trees increased the soil pH by 3 and 16%, respectively whereas adoption of AF tree species decreased the soil pH by 5%. Soil pH showed a positive and significant relation (Table 4.15) with the basic cations such as K (r=0.659**), Ca (r= 0.656*) and Mg (r= 0.654**). In contrast, it was negatively and significantly related with micronutrients like Fe (r= -0.538**) and Mn (r= -0.564**). With regard to soil P, soil pH showed the positive relation but did not show any significant effect. Soil organic carbon (SOC) content ranged from 1.41-1.89, 0.98-1.33 and 0.95-1.37 g $100g^{-1}$ in AF, agriculture and horticulture land uses, respectively (Table 4.4). In all the three land uses SOC decreased with soil depths (Table 4.4) with highest accumulation in the top (0-15 cm) soil layer (2.41, 1.58 and 1.76 g $100g^{-1}$ in AF, agriculture and horticulture land uses, respectively) compared to the deeper (15-75 cm) soil layers

(0.95, 0.66 and 0.5 g 100g^{-1} in AF, agriculture and horticulture land uses, respectively). In AF land use, highest SOC was recorded in Alder plot (1.89 g 100g^{-1}) followed by Champak (1.67 g 100g^{-1}) and the lowest was found in control plot (1.41 g 100g^{-1}). Turmeric (1.33 g 100g^{-1}) followed by potato (1.19 g 100g^{-1}) showed highest SOC and control plot (0.98 g 100g^{-1}) showed lowest SOC in case of agriculture land use. Among the horticulture fruit trees, peach (1.37) recorded highest SOC followed by guava (1.25 g 100g^{-1}) and the lowest was recorded in control plot (1.02 g 100g^{-1}). On the whole, agroforestry land use increased the SOC by 17%, agriculture by 18.4% and horticulture land use by 14.7% compared to their respective control plots. Among the three major land uses, agroforestry land use was found to have highest SOC followed by horticulture and the lowest was in agriculture land use (Fig.4.6). As indicated in the table 4.15, soil OC was positively and significantly correlated with nitrogen (r=0.587**), MWD (r= 0.872**), MBC (r=0.803**), Fe (r= 0.312*) and Mn (0.691*). Conversely, it was negatively and significantly related with soil Cu (r=-0.517**).

Depth-wise distribution of soil available nitrogen, phosphorous and potassium under different land use systems are presented in tables 4.5, 4.6 and 4.77, respectively. In agroforestry land use, tree species, soil depths and their interaction had significant effect on the soil available N content but in horticulture and agriculture land use only land use and soil depths had the considerable influence on available N content. Interaction of both land use and soil depth did not show any effect. Like available N, available K also significantly affected by AF tree species, fruit trees and agricultural crops and also by soil depths in all the three land use systems. Interaction of land use and soil depths, like available N, did not show any noteworthy effect on soil available K content. In contrast to soil available N and K, different AF tree species, fruit trees and agricultural crops did not influence the soil available P content significantly but only soil depths had the noteworthy effects. Interaction of land uses and soil depths also had no considerable effect on available K in all the three land uses. The surface and subsurface available N, P and K concentration differed significantly across the land use systems (Tables 4.5, 4.6 and 4.7). In all the three land use systems, surface soils (0-15 cm) had highest concentration of available N, P and K than the subsurface soil layers. On the whole adoption of various AF tree species increased the soil available N, P and K content by 22.0, 54.0 and 53.3%, fruit trees by 11.0, 52.3 and 14.7% and agricultural crops by 6.4, 71.0 and 94.0%. In general, soil available N, P and K content was highest under AF tree species followed by horticulture fruit trees and the lowest was observed in agriculture land use (Fig. 4.7). With respect to the soil depths, adoption of various land use systems increased the surface soil (0-15 cm) SOC content by 71.5, 56.8 and 78.2 % in AF, agriculture and horticulture land uses, respectively. Surface (0-15 cm) soil available N, P and K contents also increased by 40.7, 33.2 and 22.7% in AF land use; 153, 173 and 118% in agriculture land use; 23.3, 34.1 and 34.0% in horticulture land use, respectively. Overall, highest increase in available N and P content was observed in AF land use followed by agriculture land use while, horticulture land use showed highest increase in soil available K content followed by agriculture land use and the lowest was in AF land use compared to control plots in their respective land uses.

From the table 4.8, 4.9, 4.10 and 4.14, it can be inferred that different land use types had the significant influence on soil exchangeable calcium (Ca) and available sulphur (S) content. On the other hand, only horticulture and agriculture land uses significantly influenced soil exchangeable magnesium (Mg) content. Soil depths in all the three land use types showed noteworthy effects on all the secondary nutrients but interaction of all the three land use types and soil depths did not show any considerable variations on soil exchangeable Ca and Mg. However, interaction of AF tree species and soil depths and also horticulture fruit trees and soil depths had significant influence on S content. Soil available S content ranged from 1.54-1.99, 2.33-2.94 and 2.69-5.76 kg ha^{-1} in AF, agriculture and horticulture land uses, respectively. Among the AF tree species, Alder (1.99 kg ha^{-1}); fruit trees, Khasi mandarin (5.76 kg ha^{-1}); agriculture crops, potato (2.94 kg ha^{-1}) showed highest available S content. All the land uses had the lowest exchangeable Ca and Mg content in the control plots but with respect to available S, Alder, Maize and Guava had the lowest concentration than other associated crops and tree species in their respective land use systems. Irrespective of the land uses, surface soils had highest exchangeable Ca and Mg and available S content than the subsurface soil layers. About 70, 47 and 48%; 80, 49 and 46%; 72, 37 and 90% increase in exchangeable Ca and Mg, and available S content was observed under AF tree species, fruit trees and agriculture crops in surface soil layer (0-15 cm) compared to subsurface soil layers (15-75 cm) in the respective land uses. In relative to control plots, AF land use increased the exchangeable Ca by 9.5-28 % (average 17.8%); agricultural land use by 14.4-73.3 % (avg. 42.4%); horticultural tree species by 2.7-34.5% (avg. 34.5%). On an average, AF and agricultural land uses showed only slight increase of 1.7 and 6.3% in exchangeable Mg content compared to control plots, however rice in agricultural land use showed 28.4% increase compared to control. Unlike exchangeable Ca and Mg, available S content decreased in all the AF tree species (4.4%) although agricultural and horticultural land uses recorded increased available S content (7.5 and 53.0%, respectively) compared to control plots. Overall, unlike available N, P and K, exchangeable Ca and Mg, and available S content was highest under horticulture land uses followed by agriculture land use and the lowest content was observed under AF land use (Fig. 4.8).

DTPA extractable iron (Fe), manganese (Mn) and copper (Cu) under each land use systems and soil depths are given in tables 4.11, 4.12 and 4.13, respectively. Soils under AF tree plantation were highest in Fe and Mn, and lowest in Cu compared to soils under agriculture and horticulture plots (Fig. 4.9 and 4.10). All the three land uses and soil depths significantly influenced the soil Fe, Mn and Cu contents. However, interaction of land uses and soil depths had the influence only on soil Fe and Mn under horticulture land use; only on soil Cu in agriculture land use; only on soil Mn in AF land use (Table 4.14). On an average, AF tree plantation, fruit trees and agriculture crops increased soil Fe, Mn and Cu contents in relative to their control plots, however, the per cent increase was highest in agriculture land use compared to other two land uses. Adoption of various AF, fruit tree species and agriculture crops increased the soil Fe and Mn content while Cu content decreased in AF land use than other two land uses. Regardless of the land uses, surface soil layer (0-15 cm) recorded highest amount of Fe, Mn and Cu than subsurface soil layers (15-75 cm) (Table 4.11, 4.12 and 4.13, respectively). Overall, 35.6, 34.1 and 35.0% increase under AF land use; 40.6, 36.9 and 43.1% increase under agriculture land use; 30.2, 32.1 and 32.6% increase under horticulture land use in Fe, Mn and Cu contents, respectively was observed in the surface layer (0-15 cm) compared to subsurface soil layers (15-75 cm). Available Fe and Mn had the positive and significant relation with each other (r= 0.733**); however, Fe and Cu (r= -0.517**), and Mn and Cu (r= -0.636**) showed significant and negative correlation (Table 4.15). Available Fe (r= 0.312*) and Mn (r= 0.691**) also showed positive and significant correlation with soil OC but soil Cu showed negative and significant correlation with soil OC (r= -0.517**)

4.4.3 Soil microbial biomass carbon

Soil microbial biomass carbon (MBC) differed significantly only among the AF tree species and no considerable variation was observed in the agriculture and horticulture land uses (Fig. 4.11 and 4.12). It ranged from 333 – 548 mg kg^{-1} in AF land use, 217 – 309 mg kg^{-1} in agriculture land use and 282 – 401 mg kg^{-1} in horticulture land use. Among the three land uses, AF (425.1 mg kg^{-1}) had relatively higher MBC followed by horticulture (340.6 mg kg^{-1}) land use and the lowest was recorded in agriculture (272.9 mg kg^{-1}) land use (Fig. 12). Within each land uses, Alder (548 mg kg^{-1}), turmeric (309 mg kg^{-1}) and Khasi mandarin (401 mg kg^{-1}) showed highest MBC in AF, agriculture and horticulture land uses, respectively (Fig. 4.11). Nearly 34, 31 and 26 per cent increase in soil MBC was observed in AF, agriculture and horticulture land uses, respectively compared to their control plots. Overall, 56% increase in soil MBC under AF land use was observed compared to agriculture land use (Fig. 4.12). In general adoption of various AF tree species, agriculture crops and horticulture fruit trees did not show any significant changes on soil MBC (Fig. 4.12). As expected, MBC had positive and significant association with soil moisture (r= 0.566**), OC (r= 0.803**), available N (r= 0.794**) and MWD (r= 0.82**) while showing the negative association with soil BD (r= -0.804**) (table 4.15).

4.5 Discussion

4.5.1 Soil Physical properties

Land use types can have a significant impact on soil texture characteristics, which, in turn, can reflect the soil fertility status of a given area (Yao et al., 2010). Despite the fact that texture is an inherent soil property, management practices contributed no significant changes in soil texture of the selected study area. A noteworthy effect of all the land uses on soil bulk density, mean weight diameter and moisture content was observed in this study. AF tree species, agriculture crops and horticulture fruit trees had the average bulk density of 1.30 Mg m^{-3} which is about 5% lower than control plots (1.37 Mg m^{-3}). The ranges of bulk density values observed in this study are within the ranges expected in most mineral soils as indicated by Hillel (1980). On an average, 14% decrease in soil BD was observed in AF land use compared to agriculture land use (Fig. 4.1). Loss of organic matter by cultivation or relatively less annual accumulation of OM in agriculture land use resulted in a higher BD. The highly significant correlation (-0.745**) between SOC and BD observed in the present study confirmed the dependence of BD on SOC (Kizilkaya and Dengiz, 2010; Abbasi and Rasool, 2005). Conversion of fallow land to forest and horticulture plantation often results in soil particles aggregation thereby decreasing the soil bulk density (Feller et al., 2001). Compaction resulting from intensive cultivation and low plant residues and organic inputs in agriculture land use might have caused the relatively higher bulk density values in all soil depths than that of in the AF and horticulture land uses. Similar results were also reported by Islam and Weil (2000). Irrespective of land uses, control plots showed highest BD than soils under trees and crop cover. It may be attributed to the poor vegetation cover and soil compaction due to rain drop impact (Isirimah et al., 2003). The MWD increased considerably under these land uses as compared to control plots (Figure 3). Soils under AF land use showed the highest MWD followed by horticulture and agriculture land uses. Overall, in forest land uses 65%, in agriculture land use 46% and horticulture land use 55 % increase in MWD was seen as compared to control plots. This could possibly due to higher root density, root activity and root biomass under AF and horticulture tree species than under agriculture crops (Tripathi et al., 2009; Saha et al., 2007). A lower bulk density coupled with higher MWD under AF and horticulture land uses could be attributed to their impressive root system and root biomass resulting from accumulation of higher organic carbon, proliferation of rhizosphere and micro floral and faunal activities and root exudation below ground (Arunachalam and Pandey, 2003). Significantly higher moisture content under tree covers (21.8% over control) was attributed primarily to the continuous vegetative covers with litter fall of trees and also to their subsequent decomposition and decayed root biomass. This was in agreement with Wakene (2001) who reported 27.35% increase in available water content under forest land use compared to agriculture land use. Lal (1996b) has also reported similar improvement in water content under various land use systems. Overall, such improvement in physical properties by these AF and horticulture trees could be ascribed to effect of living biomass of trees, as there was a constant addition of organic matter to the soil through decaying the large volume of dead roots as well as leaf litter thus forming stable aggregates (Myer et al., 1994).

4.5.2 Soil Chemical properties

The soil pH ranged from 4.29 to 4.43, 4.78 to 4.98 and 4.97 to 6.02 under AF, agriculture and horticulture land uses, respectively falling under moderately to strongly acidic range (Table 4.3). On the whole, 9-16% reduction in soil pH was noticed in AF land use compared to agriculture and horticulture land uses. In AF land use, the lowest soil pH in Khasi pine may be related with the chemical composition of Khasi pine needle and its slow decomposition rate (Tripathi et al., 2009). As alder and tree bean are nitrogen fixing trees, the nitrification process that increases soil NO_3

concentrations is one of major determinant factors of lower soil pH in these tree species (Neill et al., 2006). In general, under agroforestry land use, this acidification phenomenon may be related to cation uptake by plant with subsequent release of H^+ ions, high litter fall, accumulation and decay of organic matter thus releasing organic acids, increased CO_2 levels through root respiration (Yao et al., 2010). This was supported by significant and negative correlation of SOC with soil pH (-0.368**), which might be due to the formation of organic and inorganic acids as a result of decomposition and oxidation of organic matter. Islam and Weil (2000) explained that the pre-weathered parent materials, the amphoteric nature of aluminium and the intense leaching of basic cations during rainfall are all the most likely contributing factors to decrease pH in forest soils. Similarly, a persistent solvent action of acids on the mineral constituents of the soil is responsible for the removal of base forming cations through dissolution and leaching resulting in decrease in soil pH (Brady and Weil, 1996). Considering the five soil depths, the higher mean values of pH (4.54, 5.11 and 5.59 in AF, agriculture and horticulture land uses, respectively) were observed within the surface soils. In general, pH values decreased with increasing soil depth (Table 4.3) which might be attributed to the corresponding decrease in SOM and basic cations such as Ca and Mg ions down the soil layers. Present study shows that significant correlation of soil pH with available nutrients indicating the importance of soil pH in making the nutrients available to the plants and ability of the soils to retain nutrients. Therefore, any change in soil pH will have an intensive effect on plant growth and soil quality (Abbasi and Rasool, 2005).

The amount of organic carbon in soils represents a balance between primary productivity, as influenced by environmental conditions and biologically-mediated decomposition processes. The soils under different land uses showed a significant difference with respect to SOC (Table 4.4 and Fig. 4.6). Within five depths, OC decreased with soil depths (Table 4) depicting that it was not only the land uses, which significantly affected the SOC, but soil depths also had s significant effect on SOC. Soil organic content, on average, of forest land uses increased 17.0 per cent as compared to other land uses. In general, it followed the trend of AF>Horticulture>Agriculture. The accumulation of SOC under tree species depends on the quantity as well as quality of chemical composition (lignin/nitrogen ratio, carbon/nitrogen ratio, cellulose, hemi-cellulose etc.) of tree roots and litter and varies widely as a function of climate and soil type (Parton et al., 1987; Saha et al., 2007). The superiority of Alder, Champak, Tree bean and Khasi pine in improving soil properties was mainly due to higher fine root biomass and greater leaf fall of these tree species and quality litter production leading to improvement in soil carbon status (Geissen et al., 2009). On the other hand, in general, agriculture utilization of soils has been found to decrease their OC content through disruption of the equilibrium between the competing processes of humus formation and mineralization (Saviozzi et al., 2001; Kizilkaya and Dengiz, 2010). Addition of vegetative material (leaves, litters, plant roots) is the main source of OM in soil (Abbasi et al., 2002). The lack of organic substrate due to the absence of continuous vegetation cover contributed low OC in the agriculture land use as observed in this study. Low content of SOC in agriculture land use might also be due to the continuous ploughing and cultivation which, in turn, reduces the level of SOC and enhances surface run-off and mineralization (Dalal et al., 2003). Ellert and Gregorich (1996) reported 34% less OC in agriculture land use than adjacent forest soils, while according to Smith et al. (2000), agriculture soils are considered to have lost about 25-35% of their OC due to cultivation. In addition, most of the soil organic matter produced in cultivated lands was removed with harvest while crop residues left over the soil and were placed under the soil with plough (Kizilkaya and Dengiz, 2010).

Soil available N content, as expected, was significantly more in AF land use than agriculture and horticulture land uses and followed the same trend of soil organic carbon since organic nitrogen constitute the bulk of total nitrogen in soils (Table 4.5). In contrast, available P and K were highest in horticulture land use than the other two land uses (Table 4.6 and 4.7). In all the land uses, available N, P and K contents were high in surface soils than subsurface soils. Overall, 18% increase in available N content and 19 and 17% decrease in available P and K content, respectively were observed in AF land use over agriculture and horticulture land uses. The impact of N-fixing trees on soil mineral N content was clearly evident in AF land use as Alder recorded highest available N content among the AF tree species. It indicates that Alder being the N-fixing tree species, contributed significant increase in available N in soil in comparison to other land uses. The highest available N in AF land use may be ascribed to the high amount of SOC addition through more litter fall and root biomass in AF land use which is clearly evident from the strong positive and significant correlation of available N with SOC (Table 4.15). On the other hand, the lower levels of available N in agriculture land use may have resulted from a combination of low C inputs because of less biomass return on harvested land and greater C losses due to aggregate disruptions, increased aeration by tillage, crop residue burning, microbial disturbance and livestock grazing (Islam, 2000).

The concentration of available phosphorus (AP) in surface layers under different land use systems ranged from 30.48 to 39.15 kg ha^{-1} (Table 4.6), which was high according to Havlin et al. (1999), who rated available P values as, low (<10), medium (10.1-25), and high (>25). However, the AP content decreased with depth in all the land uses because P is a less mobile plant nutrient in the soil profile (Table 4.6). The highest concentration of AP was recorded in the horticulture land use, whereas the lowest was found under the AF land use. In all the three land uses, AF, fruit tree plantations and crop cultivation increased the soil AP in comparison to control plots. Abbasi and Rasool (2005) obtained very high and positive correlation (r=0.79**) obtained between OC and AP which indicates organic matter highly contributes to AP of soils. But, in our study we did not get significant correlation between SOC and AP. However, the highest concentration of AP under the these land uses, in relative to control plots, is attributed to high accumulation of organic matter due to litter fall, root biomass and addition of crop residues (Saha et al., 2007). The high organic matter content and its mineralization ensure release of phosphate ions, though most of the phosphate ion released in this way will be in topsoil. The differences in soil AP storages may be resulted from changes in biological and geochemical processes at different depths after human disturbances (Gong et al., 2005). In contrast to N, P is very insoluble in soil systems. Maintaining proper soil pH is critical to provide adequate P for optimum plant growth. When soil pH drops below 5.0-5.5, aluminium becomes soluble and binds P, making it less available for plant use. In our study, except horticulture land use, all other land use systems had soil pH below 5.0. The lowest soil pH in AF land use (4.4) could have resulted in the P fixation by Fe and Al as insoluble Fe and Al phosphates (Szott and Melendez, 2001). On the other hand, in horticulture land use, the increased AP might be due to a conversion of Fe^{3+} phosphate to Fe^{2+} phosphate and hydrolysis of Al/Fe phosphate due to slightly increased pH in horticulture land use compared to other land uses. Soil P did not show significant correlation with the studied soil properties. The much emphasized P and Fe, P and Ca were not noticeable in our study although soil P showed negative relationship with available Fe and Mn (Table 4.14). In general, the leaf litter of forest tree species with high concentration of nutrients may therefore enhance soil available N and P content by adding high quality of litter to the soil systems. The poor quality of Khasi pine litter in terms of nutrient concentration was compensated by greater amount of litter input (Tripathi et al., 2009). In contrast to both available N and P, available K had different trend; Horticulture>AF>Agriculture. The lowest content of available K in agriculture land use indicates the extractive effect of cultivation, agriculture practices and crops on the soil. Gong et al. (2005) also indicated that harvest could take high K values from the soil and hence results in lowering K levels in agriculture fields.

Exchangeable Ca is the most dominant cation followed by Mg in nearly all soil systems except those with a very low pH. In soils with a pH above about 4.8 is usually present in amount as adequate for crop growth. In our study, both AF and agriculture land use were with pH below 4.8 thus showing less exchangeable Ca and Mg compared to horticulture land use (Fig. 4.8). This is clearly evident from the strong positive and significant correlation of soil PH with exchangeable Ca (0.656**) and Mg (0.654**) (Table 4.15). Exchangeable Ca and Mg were significantly related to available S, Fe, Mn and Cu. The negative effect of Ca and Mg on Fe and Mn was observed in this study. Adeoye (1984) also observed negative relation between exchangeable Ca and available Mn in soils under different land use systems. The control of SOC over cations like Ca and Mg has already been widely reported (Asadu et al., 1997). However, in our study, the more unexpected was the no significant correlation of SOC with Ca and Mg (Table 4.14). Many research results indicated that weathering, intensive cultivation, leaching and generally harvesting all parts of the crop for different purposes affect the distribution of exchangeable cations like K, Ca and Mg in the soil systems and enhances their depletion (Saikhe et al., 1998b).

Significant variation was observed among the different land use systems in DTPA extractable Fe and Mn (Table 4.11 and 4.12, correspondingly). In spite of variation among and within the land use systems in available Fe and Mn concentrations, Fe and Mn were at the toxic levels as reported by Lindsay and Norvel (1969). Moreover, the trend in available Fe and Mn concentration under different land use systems and soil depths were similar. This implies that these two elements have the same chemical behaviour as described by Krauskof (1972). Available Fe and Mn were highest under AF and horticulture land use and lowest in agriculture land use. The management practices like addition of liming and fertilization might have caused increase in soil pH which in turn reduced the available Fe and Mn concentration in agriculture land use (Geissen et al., 2009). Soil Fe showed positive and significant correlation with SOC and negative and significant correlation with soil pH, K, Ca and Mg suggesting the dependency of soil Fe on these properties in the soil selected for the study (Vijayakumar et al., 2011). The high available Fe and Mn in the surface horizons may be due to complex formation with organic compounds that protects their leaching. As a result their concentrations decreased with depth in line with previous report by Havlin et al. (1999). The r-value between soil Fe and pH and soil Mn and pH was -0.538** and -0.564**, respectively. The soil pH was negatively and significantly correlated with soil Fe and Mn. It can be observed that Fe and Mn decreases with the increase in soil pH. These results were supported by Chinchmalatpure et al. (2000) and Patiram et al. (2000) who reported negative and significant correlation between soil pH and Fe and Mn. The correlation coefficient between soil Fe, Mn and SOC was 0.312* and 0.691**, respectively revealing the positive and significant correlation of SOC with Fe and Mn. These results confirm that soil rich in SOC increases the availability of Fe and Mn. These results are in agreement with Chinchmalatpure et al. (2000) but in contradictory with Ibrahim et al. (2011) who reported negative and significant correlation between SOC and available Mn. They also reported that soil Fe and Mn had a negative correlation with soil Ca. In our study also, we observed negative and significant correlation between soil Fe and Mn and soil Ca (Table 4.15) concluding that liming decreases the availability of soil Fe and Mn by increasing soil pH. The concentration of Cu was also affected by land uses and soil depths (Table 4.13) but interaction of soil depths and land uses did not show any significant effect (Table 4.14). In contrast to Fe and Mn, available Cu content was highest in horticulture and agriculture land use systems than AF land use. This is attributed to the strong and negative association of soil Cu with SOC which is clearly evident from the highest SOC under AF land use (Table 4) and significant and negative correlation between these two parameters (Table 4.14). This is in corroboration with the findings of Solomon et al. (2002). The r –value between soil Cu and pH was 0.828**. It shows that, unlike Fe and Mn, there was positive and significant correlation between soil pH and Cu. These results were supported by Saddiq et al. (2008) and Samndi et al. (2007) who reported Cu was positively correlated with soil pH. On the other hand, soil Cu showed negative and significant correlation with SOC (-0.517**). This is in conformity with the findings of Ibrahim et al. (2011) and Samndi et al. (2007). Overall, the changes in soil micronutrient concentration may be attributed to the differences in SOC content and pH among the three land uses.

4.5.3 Soil microbial biomass

Soil microbial biomass, a living part of soil organic matter, is an agent of transformation for added and native organic matter and acts as labile reservoir for plant available nitrogen, phosphorous and sulphur (Jenkinson and Ladd, 1981). The activity of the microbial biomass is commonly used to characterize the microbial status of soil and to determine the effect of cultivation and field management practices on soil microorganisms (Perott et al., 1992). In our study, results showed that soil microbial biomass differed significantly only among the AF tree species and no considerable variation was observed in the agriculture and horticulture land uses (Fig. 4.11 and 4.12). Among the three land uses, AF had the highest MBC followed by horticulture land use and the lowest was recorded in agriculture land use (Fig. 4.12). In all the three land uses, control plots had lowest MBC and nearly 34, 31 and 26 per cent increase in soil MBC was observed in AF, agriculture and horticulture land uses, respectively compared to their control plots. Overall, 56% increase in soil MBC under AF land use was observed compared to agriculture land use (Fig. 4.12). The chief contributory factor for the higher soil MBC in the AF land use than the other land uses seems to be the greater availability of nutrients due to the addition of higher plant litters and root biomass (Arunachalam and Pandey, 2003). This was clearly evident from the strong positive correlations between MBC and SOC (r = 0.803**, P < 0.01). Witter and Kanal (1998) also observed a close relationship between organic C content and microbial biomass in different soils but our results were in contrast with the findings of Pietri and Brookes (2008) who reported no correlation between SOC and MBC due to the changes of land uses. The characteristics of vegetation differences directly through quality of litter and indirectly through changes in soil chemical and physical properties may have the influence on MBC in soil (McLean and Hunta, 2002). The differences in soil MBC within and among the land uses could also be due to the variation in soil moisture, stages of plant growth, soil temperature and substrate availability. Similar observations have also been reported by several other workers (Chang and Juma, 1996; Campbell et al., 1991). The reduced organic inputs through crop residues along with tillage practices could be the possible reason for lowest MBC in agriculture land use than AF and horticulture land uses (Ross, 1987).

4.5.4 Relation between the land use systems, soil depths and soil properties

The soil properties considered in this study were influenced by both land use systems and soil depth (Table 4.14). Significant effects of soil depths in the differences among the selected soil properties except soil moisture content only in AF land use were observed. On the other hand, land use influenced all the soil properties considered, except pH only in AF land use, moisture content only in horticulture land use, available P in both AF and horticulture land sues and exchangeable Mg only in AF land use. This indicates that land use practices largely determine changes in soil properties. Particularly, inappropriate land use aggravates the rate of degradation of these properties. Land uses affects basic processes such as erosion, soil structure and aggregate stability, nutrient cycling, leaching, carbon sequestration, and other similar physical and biochemical processes, although processes in the soil depend on many factors such as land use systems, soil types, topography, and climatic conditions (Maddonni et al., 1999; Singh et al., 1995; Saikhe et al., 1998). However, the absence of significant influences on some of the soil properties shows the existence of some properties, which were not responsive to land use changes. Geissen et al. (2009) and Marzaioli et al. (2010) also showed that soil pH, bulk density, moisture content and available P did not change significantly with different land use systems. In general, soil functions depend on both stable and dynamic soil properties. The difference between land use change dependent and inherent soil properties vary at temporal scales ranging from seconds to centuries and at spatial scales from millimeters to hundreds of kilometers. Over a sufficient time period, practices that affect dynamic soil properties and stable soil properties will change. Because land attributes are based on climatic factors, internal soil properties (temperature- regime, moisture regime, effective depth, pH, EC), and external soil properties (soil slope, occurrence of flooding, erosion and soil accessibility) (Lal, 1996).

4.6 Conclusions

Soil physical properties such as bulk density, moisture content and MWD showed notable variations among land use systems, irrespective of soil depths. Overall, conversion of fallow land to AF, agriculture and horticulture land uses increased the soil moisture content and MWD while reducing the soil BD. The soil pH of the selected study area ranges from moderately to strongly acidic. Conversion of fallow land (control plots) to agriculture and horticulture land uses, in fact, increased the soil pH; however, AF tree plantations in the fallow lands decreased the soil pH to a level of about 6%. With regard to agriculture crops and horticulture fruit trees adoptions, soil pH decreased by about 9.3 and 16.3%, respectively due to the adoption of agroforestry plantation in the study site. The soil pH was significantly and positively correlated with the basic cations such as K, Ca and Mg while, it showed negative and significant correlation with soil micronutrients suggesting the important role of soil pH in making the nutrient available to the crop plants. Land conversion from fallow to agriculture cultivation, agroforestry and horticulture fruit trees plantation increased the soil organic carbon, available nutrients like nitrogen, phosphorus, potassium, sulphur, exchangeable Ca and Mg, and DTPA extractable micronutrients (Fe, Mn and Cu). However, the increase was highest under land conversion from fallow to agroforestry plantations excepting available phosphorus, exchangeable Ca and Mg and Cu and these properties were higher in the land conversion from fallow to horticulture plantation. In most cases, agroforestry tree plantation proved to be the best land use with respect to improvement in soil physical and chemical properties in relative to agriculture and horticulture land uses. Soil microbial biomass was highest under agroforestry plantation than the other two land uses. Amongst the 13 trees and crop species studied, *A. nepalensis* (Alder) recorded highest soil MBC, in overall as well as among the agroforestry tree species, followed by turmeric amongst the agriculture crops and Khasi mandarin amongst the horticulture fruit tree species.

Soil moisture content, organic carbon, available N, P, K, exchangeable Ca, and Mg and DTPA extractable micronutrients contents decreased with increasing depth while soil BD was high under subsurface soil layers. As expected soil organic carbon had significant correlation with most of the soil nutrients, MBC, MWD and bulk density but, the most unexpected was the insignificant correlation of SOC with available phosphorus.

By and large, the study revealed that most of the soil properties are influenced by land use systems and soil depths. Cultivation of agriculture crops, in relation to AF tree plantation and horticulture fruit trees, has negative effects on physico-chemical and biological properties of the soils. However, conversion from fallow land to agriculture land improved all the physico-chemical and biological properties but the magnitude of the benefits obtained from the conversion of fallow land to other land uses was maximum under agroforestry tree plantation due to increased accumulation of soil organic matter by highest leaf litter and root biomass addition to the soil, despite decreasing the soil pH. Therefore, it could be recommended that inclusion of agroforestry tree species in the land use practices that increases organic carbon and other properties in the ecosystems is a viable option to restore the soil fertility status for long-term productivity and also signifies the ecological benefits (mitigating global warming) through increased organic carbon sequestration in the highly degraded soils of North-east India, Meghalaya in particular.

Table 4.1 Effect of different land use systems on soil bulk density (Mg m^{-3})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	1.11	1.14	1.19	1.26	1.32	1.20b
Tree bean	1.15	1.18	1.25	1.28	1.34	1.24b
Alder	1.07	1.09	1.14	1.22	1.3	1.16bc
Khasi Pine	1.05	1.07	1.13	1.19	1.25	1.14c
Control (No tree)	1.18	1.25	1.31	1.36	1.39	1.30a
Mean	1.11d	1.15d	1.20c	1.26b	1.32a	
			Agriculture			
Maize	1.27	1.30	1.35	1.41	1.45	1.36c
Potato	1.31	1.35	1.37	1.42	1.49	1.39bc
Rice	1.33	1.27	1.40	1.47	1.51	1.40ab
Turmeric	1.25	1.31	1.35	1.46	1.49	1.37bc
Control (No crop)	1.33	1.37	1.45	1.48	1.50	1.42a
Mean	1.30d	1.32d	1.38c	1.45b	1.49a	
			Horticulture			
Pear	1.28	1.32	1.36	1.38	1.42	1.35abc
Peach	1.23	1.27	1.32	1.36	1.40	1.32bc
K Mandarin	1.29	1.31	1.35	1.39	1.45	1.36ab
Guava	1.19	1.26	1.31	1.37	1.44	1.31c
Control (No tree)	1.30	1.33	1.38	1.43	1.48	1.39a
Mean	1.26d	1.30d	1.35c	1.39b	1.44a	

Table 4.2 Effect of different land use systems on soil moisture content (g 100g^{-1})

Land use system	**Depth (cm)**					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	31	27.7	27.5	25.1	24	27.06b
Tree bean	26.1	28.9	27.4	25.5	26.3	26.84b
Alder	31.8	32.4	31.8	30.9	30.2	31.42a
Khasi Pine	33.8	34.2	31.6	30.4	31.2	32.24a
Control (No tree)	28.4	23.4	24.2	23.7	23.1	24.56b
Mean	30.22a	29.32a	28.5a	27.12a	26.96a	
			Agriculture			
Maize	29.9	27.6	26.8	24.5	22.7	26.3b
Potato	28.9	26.7	23.8	23.7	24.3	25.5b
Rice	32.2	30.3	30.6	28.3	26.7	29.6a
Turmeric	26.7	24.9	25.6	24.4	24.6	25.2b
Control (No crop)	24.8	23.7	23.2	22.5	20.7	23.0c
Mean	28.5a	26.6b	26.0bc	24.7cd	23.8d	
			Horticulture			
Pear	31.6	30.1	26.5	25.5	25.4	27.8a
Peach	32.7	30.4	29.4	26.1	25.3	28.8a
K Mandarin	28.7	26	26.5	26.2	26.3	26.7a
Guava	29.9	28.6	28	26.4	25.6	27.7a
Control (No tree)	27.1	27.5	25.9	25.3	23.4	25.8a
Mean	30.0a	28.5ab	27.3ab	25.9bc	25.2c	

Table 4.3 Effect of different land use systems on soil pH (1:2)

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	4.6	4.48	4.4	4.38	4.3	4.43ab
Tree bean	4.59	4.43	4.23	4.31	4.27	4.37ab
Alder	4.38	4.36	4.33	4.3	4.26	4.33b
Khasi Pine	4.36	4.33	4.31	4.27	4.2	4.29ab
Control (No tree)	4.76	4.69	4.58	4.46	4.44	4.59a
Mean	4.54a	4.46ab	4.37bc	4.34bc	4.29c	
			Agriculture			
Maize	5.12	4.98	4.81	4.72	4.69	4.86ab
Potato	5.24	5.10	4.91	4.77	4.65	4.93a
Rice	5.09	4.91	4.76	4.62	4.51	4.78bc
Turmeric	5.29	5.06	4.97	4.82	4.74	4.98a
Control (No crop)	4.81	4.73	4.71	4.62	4.53	4.68c
Mean	5.11a	4.96b	4.83bc	4.71cd	4.62d	
			Horticulture			
Pear	5.71	5.65	5.64	4.93	4.93	5.37b
Peach	6.24	6.13	6.04	5.81	5.89	6.02a
K Mandarin	5.75	5.50	5.11	5.03	4.96	5.27b
Guava	5.43	5.14	4.90	4.79	4.59	4.97c
Control (No tree)	4.82	4.78	4.59	4.58	4.51	4.66d
Mean	5.59a	5.44ab	5.26bc	5.03cd	4.98d	

Table 4.4 Effect of different land use systems on soil organic carbon (g $100g^{-1}$)

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	2.45	2.01	1.56	1.34	0.98	1.67b
Tree bean	2.32	1.92	1.49	1.11	0.78	1.52bc
Alder	3.12	2.18	1.65	1.39	1.09	1.89a
Khasi Pine	2.27	1.71	1.47	1.19	1.02	1.53bc
Control (No tree)	1.91	1.81	1.41	1.08	0.86	1.41c
Mean	2.41a	1.93b	1.52c	1.22d	0.95e	
			Agriculture			
Maize	1.62	1.49	1.20	0.94	0.52	1.15ab
Potato	1.72	1.40	1.11	0.84	0.86	1.19a
Rice	1.50	1.22	1.07	0.62	0.43	0.97c
Turmeric	1.71	1.67	1.41	0.98	0.89	1.33a
Control (No crop)	1.37	1.21	0.95	0.74	0.61	0.98bc
Mean	1.58a	1.40b	1.15c	0.82d	0.66d	
			Horticulture			
Pear	1.71	1.13	0.89	0.67	0.33	0.95c
Peach	1.81	1.63	1.41	1.12	0.86	1.37a
K Mandarin	1.89	1.43	0.96	0.80	0.49	1.11b
Guava	1.78	1.51	1.32	0.96	0.69	1.25a
Control (No tree)	1.61	1.38	0.99	0.65	0.48	1.02bc
Mean	1.76a	1.42b	1.12c	0.84d	0.57e	

Table 4.5 Effect of different land use systems on soil available nitrogen (kg ha^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	522.3	403.4	354	322.5	293.3	379.1bc
Tree bean	496.3	369.7	360.7	349.5	300	375.2c
Alder	584.3	475.3	389.9	351.7	322.5	424.7a
Khasi Pine	464.8	425.9	389.9	369.7	333.7	396.8b
Control (No tree)	403.4	325.2	319.9	300.5	269	323.6c
Mean	494.2a	399.9b	362.9c	338.8d	303.7e	
			Agriculture			
Maize	394.4	331.5	311.3	273.1	252.8	312.6bc
Potato	414.6	360.7	295.5	266.3	248.3	317.1bc
Rice	421.4	363.0	345.0	311.3	295.5	347.2a
Turmeric	398.9	363.0	322.5	297.8	270.8	330.6ab
Control (No crop)	387.7	302.3	288.8	279.8	277.5	307.2c
Mean	403.4a	344.1b	312.6c	285.6d	269.0d	
			Horticulture			
Pear	437.1	407.9	369.7	327.0	306.8	369.7b
Peach	455.1	430.4	392.2	360.7	320.3	391.7a
K Mandarin	428.1	401.2	365.2	349.5	329.2	374.6ab
Guava	446.1	425.9	398.9	365.2	333.7	394.0a
Control (No tree)	434.9	363.0	331.5	306.8	288.8	345.0c
Mean	440.3a	405.7b	371.5c	341.8d	315.8e	

Table 4.6 Effect of different land use systems on soil available phosphorous (kg ha^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	32.1	25.2	17.4	7.1	6.4	17.64a
Tree bean	30.1	18.7	10.1	8.7	5.1	14.54a
Alder	47.2	24.7	11.4	10.4	12.9	21.32a
Khasi Pine	23.7	16.5	15	8.9	6.5	14.12a
Control (No tree)	19.3	14.8	9.47	6.9	4.4	10.97a
Mean	30.48a	19.98ab	12.67bc	8.40bc	7.06c	
			Agriculture			
Maize	49.1	19.8	14.7	11.5	10.3	21.1ab
Potato	65.6	56.8	20.7	8.2	7.9	31.8a
Rice	23.5	20.7	10.1	11.3	8.9	14.9b
Turmeric	36.9	14.9	12.8	9.3	8.0	16.4b
Control (No crop)	20.7	11.7	11.2	10.5	7.5	12.3b
Mean	39.15a	24.77b	13.90bc	10.14c	8.54c	
			Horticulture			
Pear	43.2	19.8	15.2	12.6	9.2	20.0a
Peach	39.1	30.4	14.5	21.2	7.4	22.5a
K Mandarin	45.3	22.6	22.5	8.4	8.4	21.5a
Guava	24.6	30.1	35.4	17.2	8.1	23.1a
Control (No tree)	26.9	18.7	11.8	8.7	5.4	14.3a
Mean	35.8a	24.3ab	19.9bc	13.6bc	7.7c	

Table 4.7 Effect of different land use systems on soil available potassium (kg ha^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	313.6	307.6	274.8	253.9	241.9	278.4b
Tree bean	286.7	277.8	263	274.8	247.9	270.0b
Alder	361.2	332.1	316.6	277.8	271.8	311.9a
Khasi Pine	276	254.1	198.4	210.8	180.3	223.9c
Control (No tree)	248.4	233.6	170.2	125.7	105.9	176.8d
Mean	297.2a	281.0a	244.6b	228.6bc	209.6c	
			Agriculture			
Maize	292.7	230.0	230.0	227.0	203.1	236.5b
Potato	365.1	362.6	312.1	295.7	261.9	319.5a
Rice	322.2	287.7	245.6	213.8	184.9	250.8b
Turmeric	306.1	273.1	247.6	206.4	186.3	243.9b
Control (No crop)	203.1	146.1	110.2	111.6	105.3	135.3c
Mean	297.8a	259.9b	229.1c	210.9cd	188.3d	
			Horticulture			
Pear	323.6	290.3	270.4	255.4	224.2	272.8b
Peach	495.2	439.4	322.8	290.7	249.0	359.4a
K Mandarin	419.6	379.4	352.7	275.0	240.3	333.4a
Guava	343.5	322.3	273.7	255.0	229.8	284.8b
Control (No tree)	328.5	304.8	280.0	238.1	211.2	272.5b
Mean	382.1a	347.2b	300.0c	262.8d	230.9e	

Table 4.8 Effect of different land use systems on soil exchangeable calcium (meq 100g^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	2.05	1.52	1.42	1.30	1.32	1.52bc
Tree bean	2.58	1.65	1.62	1.42	1.43	1.74a
Alder	2.72	1.67	1.33	1.30	1.27	1.66ab
Khasi Pine	2.43	1.70	1.17	1.00	1.17	1.49bc
Control (No tree)	1.82	1.47	1.25	1.19	1.07	1.36c
Mean	2.32a	1.60b	1.36c	1.25c	1.24c	
			Agriculture			
Maize	2.68	2.37	1.93	1.62	1.38	2.00b
Potato	2.47	1.73	1.35	1.22	1.17	1.59cd
Rice	3.04	2.78	2.35	2.03	1.83	2.41a
Turmeric	2.35	1.91	1.77	1.67	1.79	1.90bc
Control (No crop)	1.97	1.55	1.21	1.14	1.10	1.39d
Mean	2.50a	2.07b	1.72c	1.54c	1.45c	
			Horticulture			
Pear	3.33	2.77	2.52	2.21	1.89	2.54a
Peach	3.64	3.07	2.77	2.31	2.05	2.77a
K Mandarin	3.39	2.84	2.61	2.13	1.90	2.57a
Guava	2.87	1.83	1.68	1.52	1.41	1.86b
Control (No tree)	2.42	2.25	1.77	1.48	1.15	1.81b
Mean	3.12a	2.55b	2.27c	1.93d	1.68e	

Table 4.9 Effect of different land use systems on soil exchangeable magnesium (meq 100g^{-1})

Land use system	Depth (cm)					
	0-15	15-30	30-45	45-60	60-75	Mean
			Agroforestry			
Champak	0.90	0.67	0.62	0.41	0.44	0.61a
Tree bean	1.02	0.62	0.52	0.44	0.43	0.60a
Alder	1.01	0.67	0.53	0.42	0.42	0.61a
Khasi Pine	0.86	0.63	0.46	0.40	0.35	0.54a
Control (No tree)	0.79	0.61	0.53	0.53	0.43	0.58a
Mean	0.91a	0.64b	0.53c	0.44d	0.41d	
			Agriculture			
Maize	1.06	0.87	0.67	0.57	0.53	0.74ab
Potato	0.81	0.64	0.55	0.44	0.37	0.56b
Rice	1.14	1.00	0.82	0.72	0.61	0.86a
Turmeric	0.90	0.75	0.69	0.59	0.54	0.69ab
Control (No crop)	0.90	0.79	0.66	0.56	0.44	0.67ab
Mean	0.96a	0.81ab	0.68bc	0.58c	0.50c	
			Horticulture			
Pear	1.28	1.09	1.12	0.94	0.77	1.04b
Peach	1.58	1.32	1.19	1.07	0.94	1.22a
K Mandarin	1.50	1.19	1.11	0.92	0.72	1.09b
Guava	1.17	1.00	0.82	0.64	0.63	0.85c
Control (No tree)	1.20	0.96	0.78	0.70	0.55	0.84c
Mean	1.35a	1.11b	1.01b	0.85c	0.72d	

Table 4.10 Effect of different land use systems on soil available sulphur (kg ha^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	3.66	1.94	1.64	1.33	1.20	1.95a
Tree bean	1.95	1.45	1.61	1.46	1.24	1.54b
Alder	3.37	2.20	1.84	1.44	1.10	1.99a
Khasi Pine	2.48	2.14	1.88	1.42	1.38	1.86a
Control (No tree)	2.48	2.21	2.12	1.52	1.27	1.92a
Mean	2.79a	1.99b	1.82b	1.43c	1.24c	
			Agriculture			
Maize	3.35	2.27	1.88	2.12	2.04	2.33b
Potato	3.69	3.39	3.10	2.25	2.27	2.94a
Rice	3.12	2.59	2.55	1.86	1.82	2.39b
Turmeric	2.90	2.61	2.52	2.29	2.14	2.49b
Control (No crop)	2.89	2.52	2.33	2.03	2.01	2.36b
Mean	3.19a	2.68b	2.48b	2.11c	2.05c	
			Horticulture			
Pear	6.97	2.86	2.50	2.08	1.95	3.27c
Peach	8.31	5.78	4.72	3.72	2.82	5.07b
K Mandarin	6.95	6.42	5.63	4.95	4.84	5.76a
Guava	3.33	2.45	2.43	1.98	1.64	2.36d
Control (No tree)	5.25	2.72	2.25	1.69	1.52	2.69d
Mean	6.16a	4.05b	3.51b	2.88c	2.55c	

Table 4.11 Effect of different land use systems on soil DTPA extractable iron (mg kg^{-1})

Land use system	Depth (cm)					
	0-15	15-30	30-45	45-60	60-75	Mean
			Agroforestry			
Champak	136.6	134.2	113.1	102.5	101.5	117.6b
Tree bean	195.9	140.1	129.6	116.9	95.9	135.7a
Alder	148.3	124.4	119.8	105.5	92.1	118.0b
Khasi Pine	138.1	128.8	121.8	107.5	104.1	120.1b
Control (No tree)	148.0	125.5	104.0	103.9	91.2	114.5b
Mean	153.4a	130.6b	117.7c	107.3cd	97.0d	
			Agriculture			
Maize	137.5	127.4	113.9	91.8	77.9	109.7c
Potato	151.6	151.5	118.1	111.5	92.2	125.0b
Rice	167.2	131.0	124.5	99.7	90.4	122.5bc
Turmeric	160.7	148.4	143.4	136.7	109.1	139.7a
Control (No crop)	108.3	104.1	75.6	76.9	85.9	90.1d
Mean	145.1a	132.5a	115.1b	103.3bc	91.1c	
			Horticulture			
Pear	61.7	59.2	55.1	34.3	25.4	47.1b
Peach	52.2	51.5	53.2	49.4	34.3	48.1ab
K Mandarin	78.5	59.8	48.9	47.3	35.4	54.0a
Guava	46.8	40.9	41.5	36.8	35.5	40.3c
Control (No tree)	42.8	42.6	39.4	38.6	37.4	40.2c
Mean	56.41a	50.80ab	47.63b	41.28c	33.60d	

Table 4.12 Effect of different land use systems on soil DTPA extractable manganese (mg kg^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	79.4	61.4	44.1	33.7	22.0	48.1c
Tree bean	75.4	67.7	65.2	65.4	49.3	64.6b
Alder	74.0	71.8	77.8	69.0	59.1	70.3a
Khasi Pine	58.1	46.5	44.1	37.6	34.0	44.1cd
Control (No tree)	51.1	46.5	42.3	37.4	33.6	42.2d
Mean	67.60a	58.77b	54.71b	48.60c	39.63d	
			Agriculture			
Maize	43.3	42.7	36.8	35.7	30.3	37.7b
Potato	37.1	28.2	26.0	22.4	21.6	27.1c
Rice	45.4	38.0	34.3	33.6	30.3	36.3b
Turmeric	57.2	55.6	52.8	37.5	35.1	47.7a
Control (No crop)	40.2	25.7	22.5	22.3	21.0	26.3c
Mean	44.65a	38.06b	34.47bc	30.31cd	27.65d	
			Horticulture			
Pear	26.7	26.1	25.5	25.1	24.1	25.5ab
Peach	30.1	27.5	20.8	19.5	21.2	23.8b
K Mandarin	41.6	26.7	23.1	24.9	21.6	27.6a
Guava	27.1	26.4	23.8	22.4	22.6	24.5b
Control (No tree)	29.3	24.1	20.8	21.8	20.8	23.4b
Mean	30.98a	26.14b	22.83c	22.74c	22.07c	

Table 4.13 Effect of different land use systems on soil DTPA extractable copper (mg kg^{-1})

Land use system	**Depth (cm)**					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	2.16	1.54	1.73	1.56	1.46	1.69b
Tree bean	2.63	2.11	2.04	1.89	1.64	2.06ab
Alder	2.89	2.18	2.11	1.94	1.79	2.18a
Khasi Pine	2.33	2.08	2.01	1.80	1.76	2.00ab
Control (No tree)	2.22	2.01	1.79	1.44	1.35	1.76b
Mean	2.46a	2.00b	1.94bc	1.74bc	1.61c	
			Agriculture			
Maize	5.58	5.06	4.16	3.88	2.72	4.28b
Potato	4.73	4.17	4.02	2.92	2.71	3.71c
Rice	6.49	5.78	4.97	3.42	2.81	4.69a
Turmeric	4.05	3.72	3.19	3.01	2.86	3.37c
Control (No crop)	4.19	3.43	2.78	2.39	2.03	2.96d
Mean	5.01a	4.43b	3.82c	3.12d	2.63e	
			Horticulture			
Pear	5.69	5.03	4.74	2.98	2.64	4.22b
Peach	7.87	7.27	6.33	5.74	3.73	6.19a
K Mandarin	7.42	6.52	5.84	5.79	4.58	6.03a
Guava	4.96	4.76	4.58	3.49	3.14	4.19b
Control (No tree)	5.23	4.82	4.36	4.06	3.56	4.40b
Mean	6.23a	5.68ab	5.17b	4.42c	3.53d	

Table 4.14 Effect of land use systems, depth and their interaction on various soil properties

Dependent variables	Agroforestry			Agriculture			Horticulture		
	LUS	Depth (D)	LUS x D	LUS	Depth (D)	LUS x D	LUS	Depth (D)	LUS x D
pH (1:2)	NS	**	NS	**	**	NS	**	**	NS
Bulk density (Mg m^{-3})	**	**	NS	**	**	NS	**	**	NS
Moisture content (g $100g^{-1}$)	**	NS	NS	**	**	NS	NS	**	NS
Organic carbon (g $100g^{-1}$)	**	**	NS	**	**	NS	**	**	NS
Available N (kg ha^{-1})	**	**	**	**	**	NS	**	**	NS
Available P (kg ha^{-1})	NS	**	NS	*	**	NS	NS	**	NS
Available K (kg ha^{-1})	**	**	NS	**	**	NS	**	**	NS
Exchangeable Ca (meq $100g^{-1}$)	**	**	NS	**	**	NS	**	**	NS
Exchangeable Mg(meq $100g^{-1}$)	NS	**	NS	*	**	NS	**	**	NS
Available S (kg ha^{-1})	**	**	**	**	**	NS	**	**	**
DTPA extractable Fe (ppm)	*	**	NS	**	**	NS	**	**	**
DTPA extractable Mn (ppm)	**	**	**	**	**	NS	*	**	*
DTPA extractable Cu (ppm)	*	**	NS	**	**	**	**	**	NS

Table 4.15 Pearson's correlation matrix for various soil physical, chemical and biological properties

Variables	pH	BD	Moisture	OC	N	P	K	Ca	Mg	S	Fe	Mn	Cu	MWD	MBC
pH	1	0.328*	0.392**	-0.368*	-0.033	0.119	0.659**	0.656**	0.654**	0.707**	-0.538**	-0.564**	0.828**	-0.576*	-0.267
BD		1	0.057	-0.745**	-0.533**	0.023	-0.008	.099	0.121	0.221	-0.247	-0.603**	0.514**	-0.970**	-0.804**
Moisture			1	-0.281	0.054	0.072	0.218	0.161	0.127	0.193	-0.259	-0.372*	0.391**	0.606*	0.566**
OC				1	0.587**	-0.036	0.107	-0.047	-0.099	-0.164	0.312*	0.691**	-0.517**	0.872**	0.803**
N					1	0.104	0.374*	0.209	0.150	0.160	-0.008	0.337*	-0.114	0.682**	0.794**
P						1	0.194	0.142	0.067	0.189	-0.029	-0.070	0.109	-0.152	0.125
K							1	0.546**	0.666**	0.635**	-0.317*	-0.216	0.637**	0.110	0.467
Ca								1	0.655**	0.621**	-0.419**	-0.310*	0.677**	-0.386	-0.045
Mg									1	0.711**	-0.566**	-0.362*	0.728**	-0.463	-0.125
S										1	-0.676**	-0.527**	0.656**	-0.369	-0.031
Fe											1	0.733**	-0.517**	0.807**	0.539*
Mn												1	-0.636**	0.455	0.097
Cu													1	-0.654	-0.354
MWD														1	0.827**
MBC															1

** Correlation is significant at the 0.01 level (2-tailed); *. Correlation is significant at the 0.05 level (2-tailed); BD-bulk density; OC-organic carbon; MWD-mean weight diameter; MBC-microbial biomass carbon

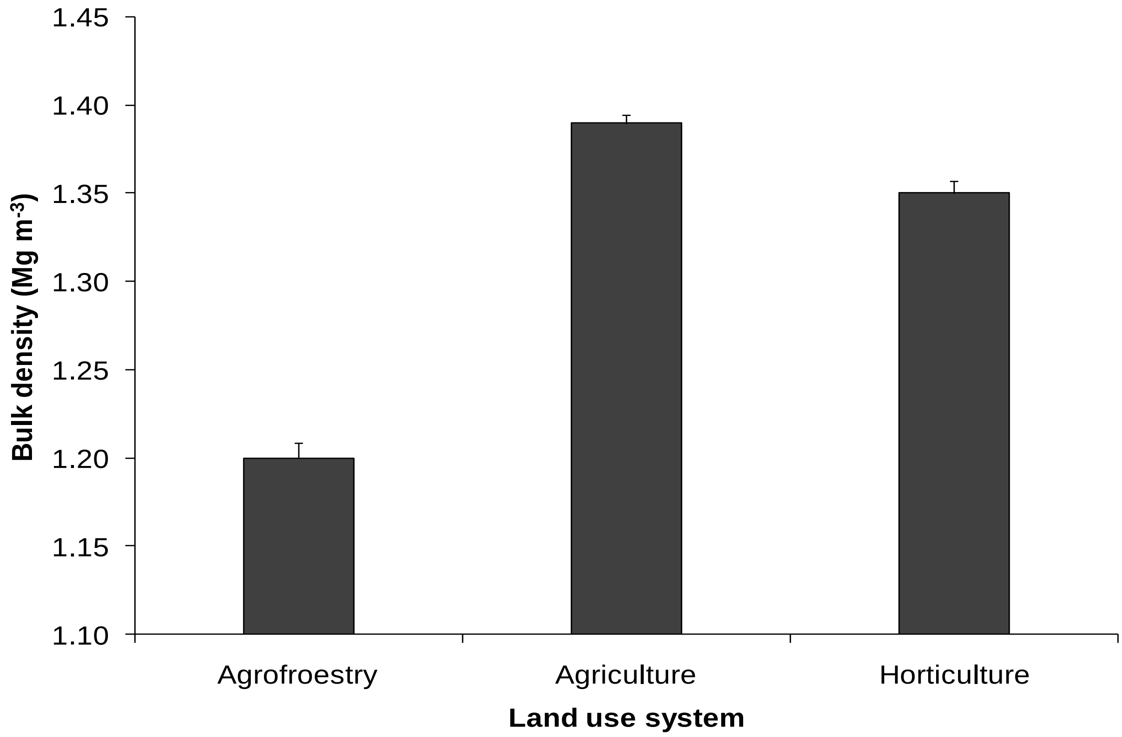


Fig. 4.1 Effect of land use systems on soil bulk density

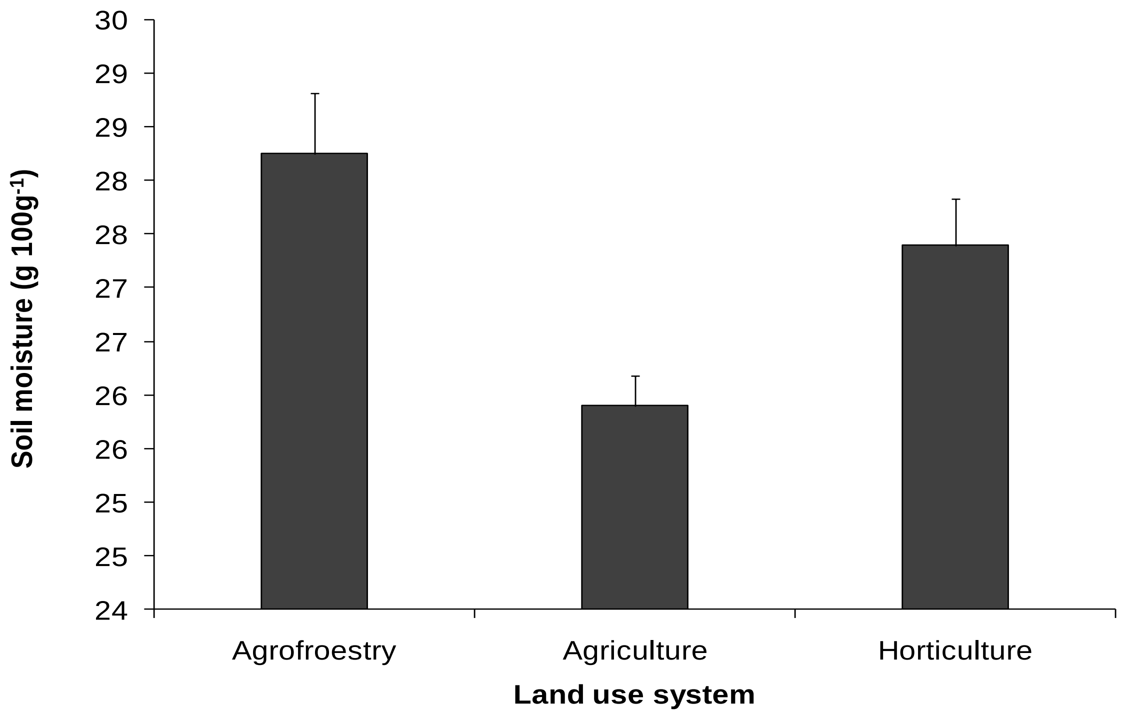


Fig. 4.2 Effect of land use systems on soil moisture content

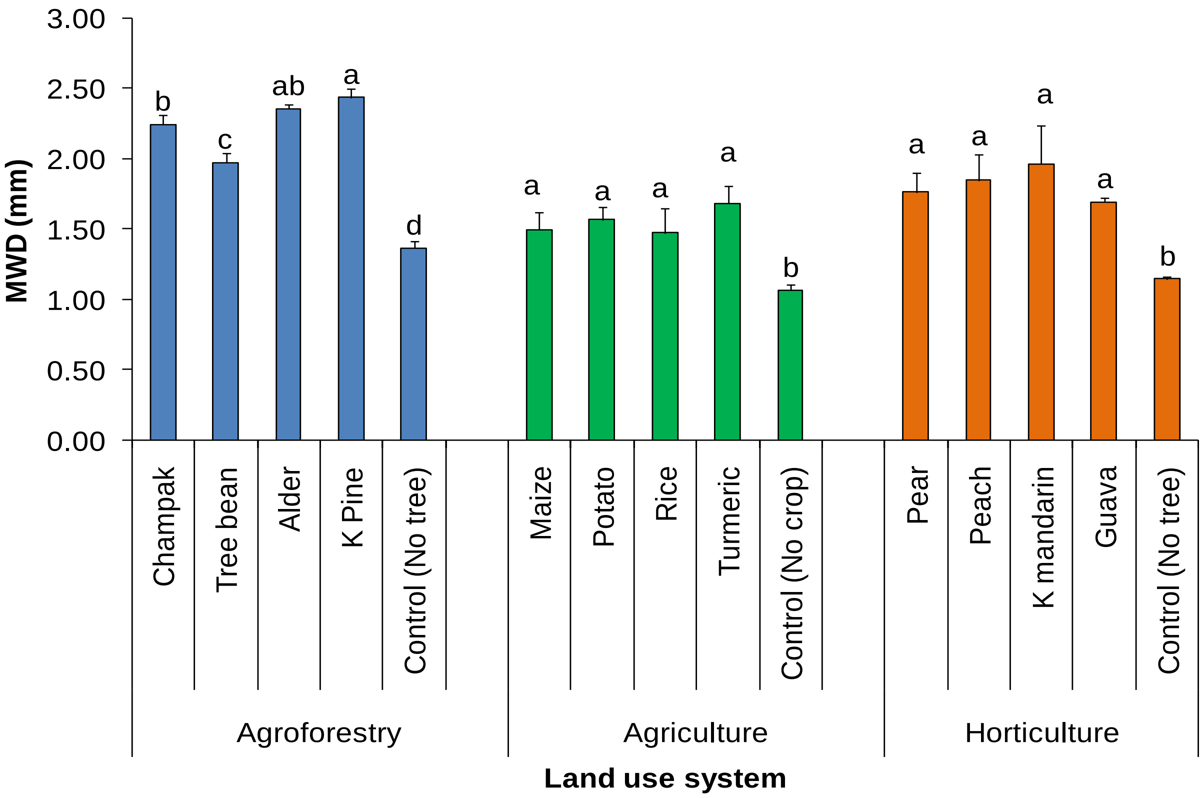


Fig. 4.3 Effect of different land use systems on mean weight diameter (MWD)

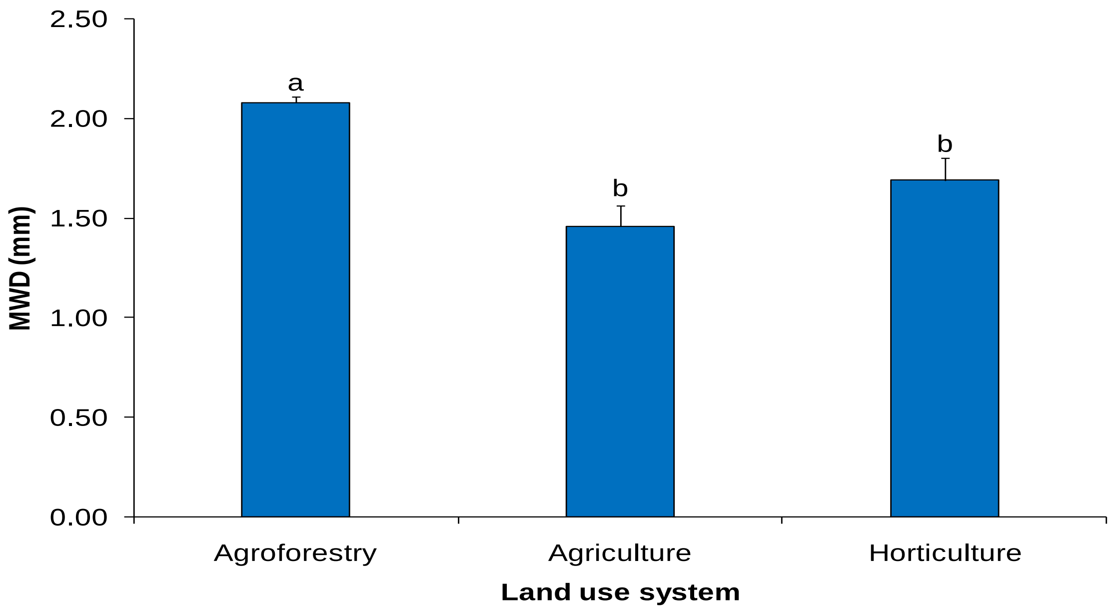


Fig. 4.4 Effect of main land use systems on mean weight diameter (MWD)

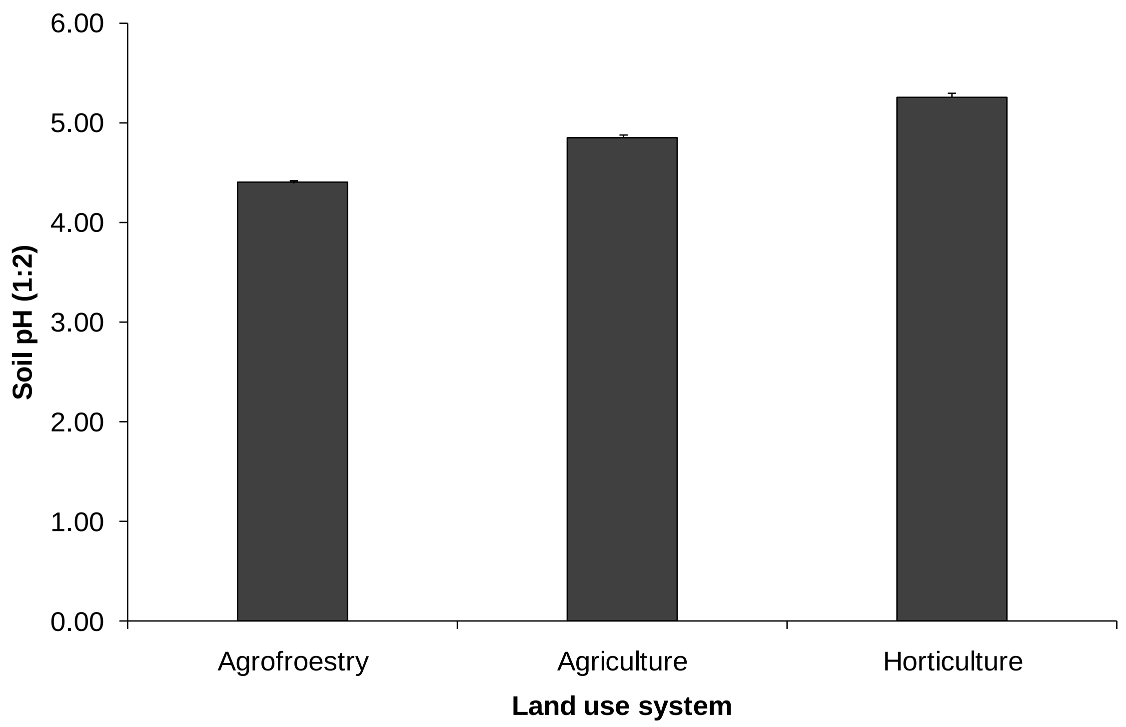


Fig. 4.5 Effect of main land use systems on soil pH

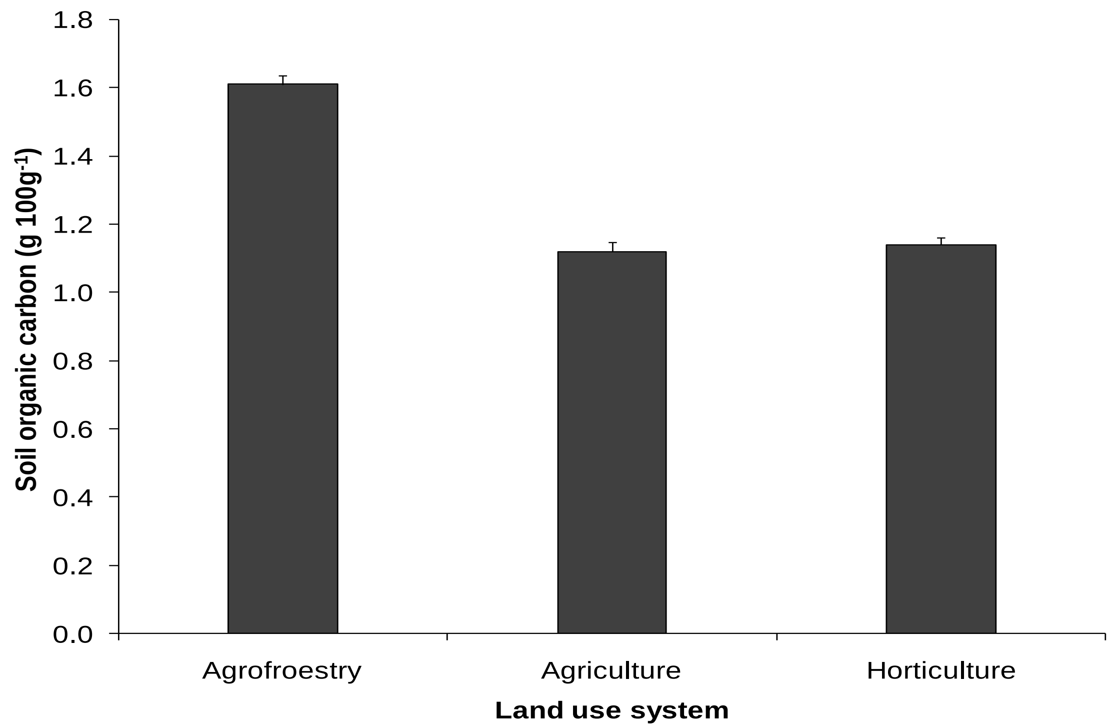


Fig. 4.6 Effect of main land use systems on soil organic carbon content

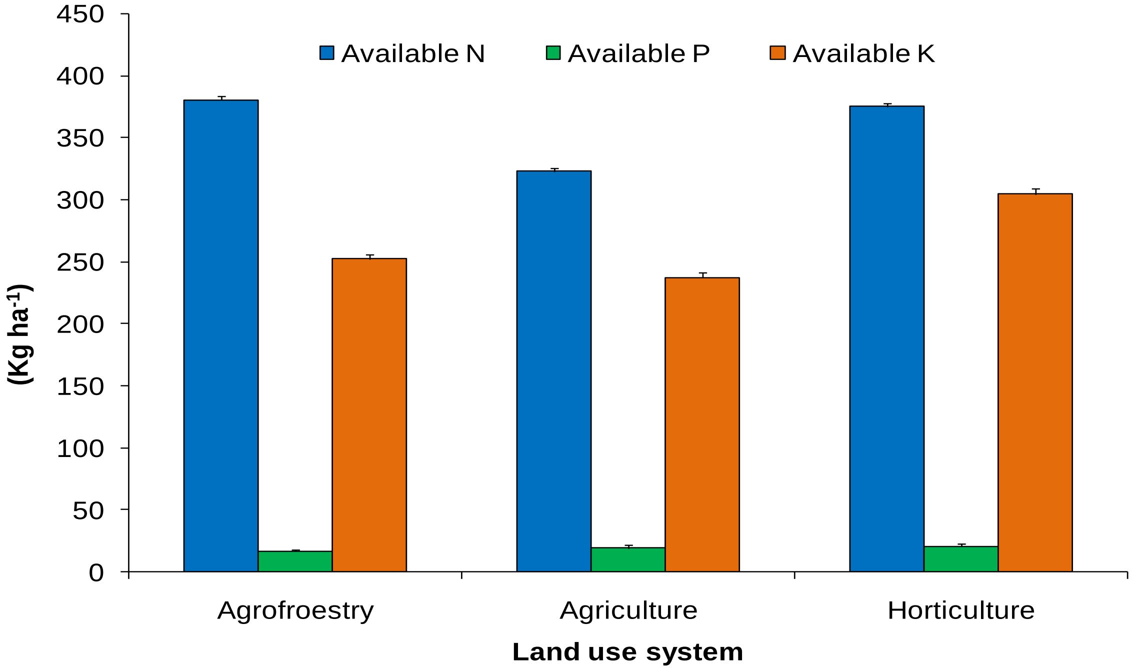


Fig. 4.7 Effect of main land use systems on soil available nitrogen, phosphorous and potassium

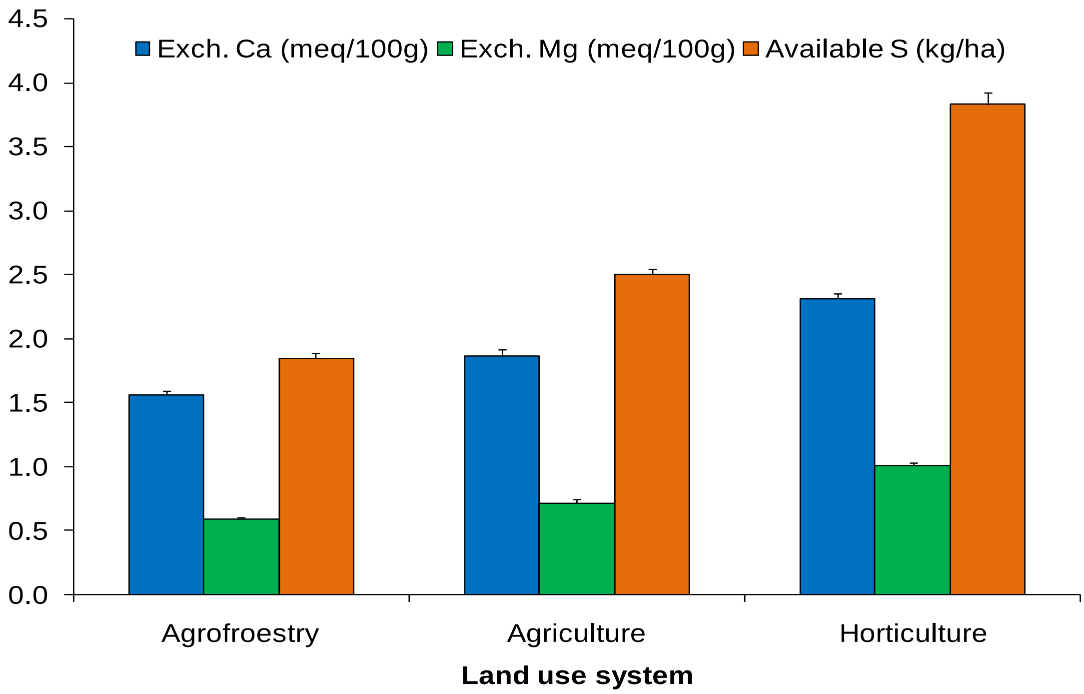


Fig. 4.8 Effect of main land use systems on available sulphur, exchangeable calcium and magnesium

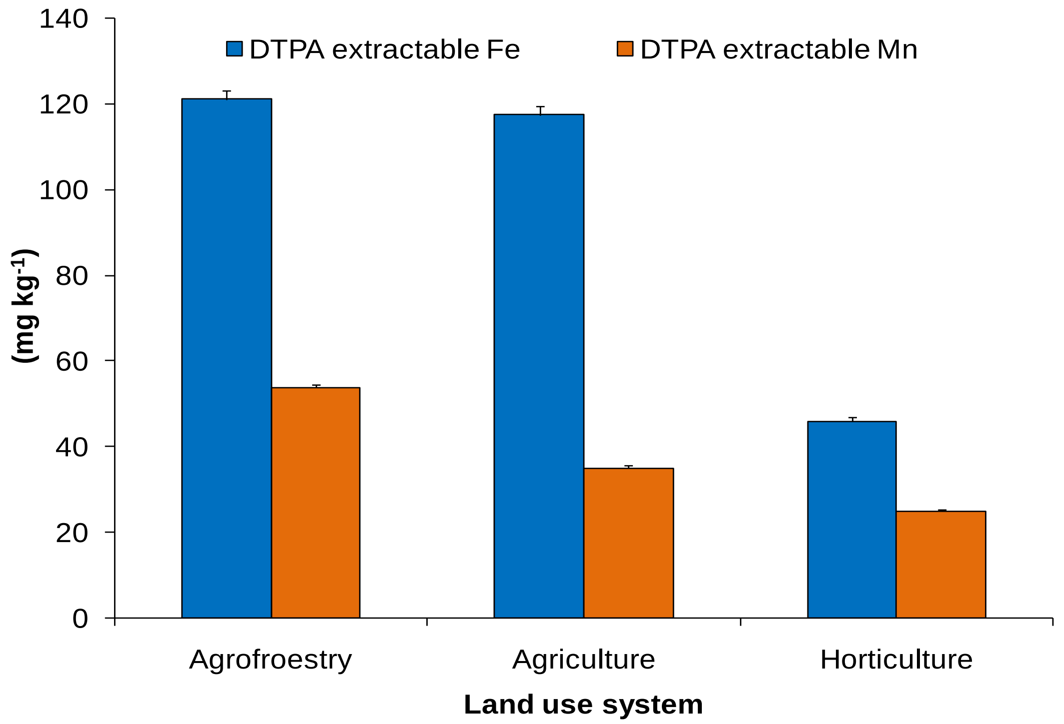


Fig. 4.9 Effect of main land use systems on DTPA extractable iron (Fe) and manganese (Mn)

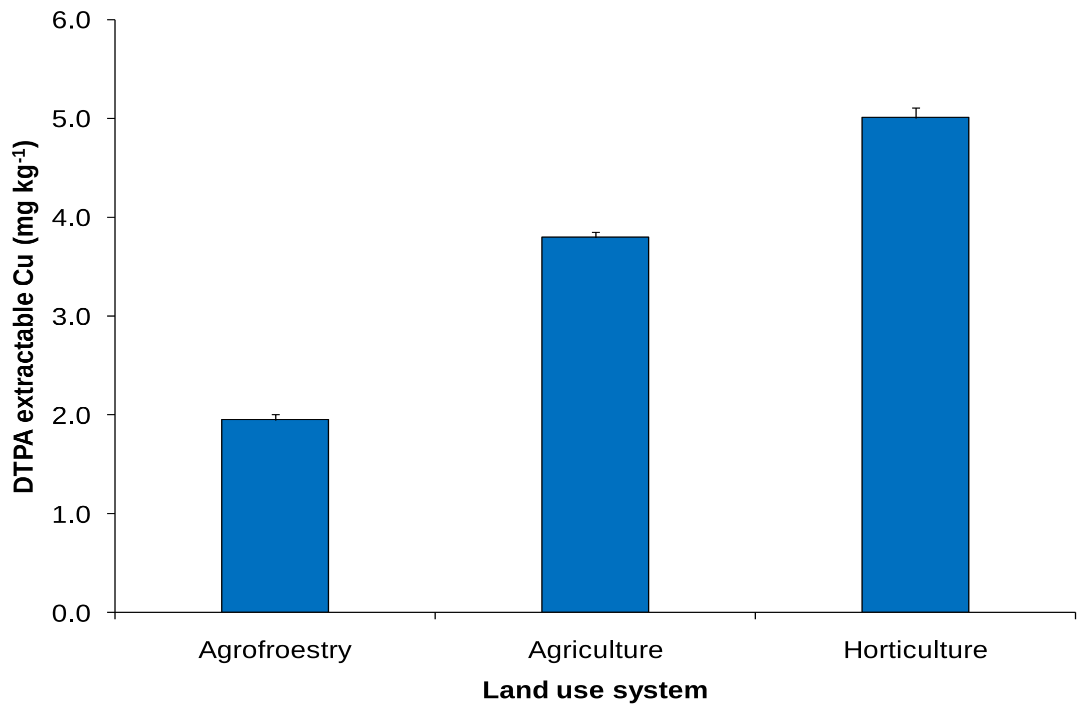


Fig. 4.10 Effect of main land use systems on soil DTPA extractable copper (Cu)

CHAPTER V

Assessment of the soil organic carbon stocks and fractions under different land use systems in East Khasi hills of Meghalaya

5.1 Abstract

The data showed that total organic carbon (TOC), microbial biomass carbon (MBC), labile carbon (C_L), non-labile carbon (C_{NL}) and particulate organic carbon (POC) followed the order: agroforestry>horticulture plantation>agriculture crops. All the carbon fractions were higher in the adopted land use practices compared to the fallow lands. In the soils of agroforestry land use, TOC, POC, C_L, C_{NL} and MBC increased by 27.8, 107, 131.7, 18.0 and 55.7%, respectively compared to the agriculture lands. On the other hand, conversion of fallow lands to agriculture, horticulture and agroforestry land uses significantly increased all these fractions by 27, 68.8, 51, 27 and 31 % in agriculture land use, 25, 43.5, 17, 38 and 26 % in horticulture land use and 26.4, 54.8, 27, 46 and 34 % in agroforestry land uses. All these fractions were highest in surface soil layers and decreased with soil profile. SOC stocks under different agricultural land uses and management practices significantly decreased downward the soil profile. Amongst the three land uses, SOC stocks were highest through the 0-75-cm soil under agroforestry (54.3 Mg ha^{-1}) followed by horticulture plantation (53.7Mg ha^{-1}) and lowest in soils under agriculture crops (49.3Mg ha^{-1}). Conversion of fallow lands to AF, agriculture and horticulture land uses increased the SOC stocks in the surface (0-15 cm) soil by 35, 34.5 and 36%, respectively in relative to subsurface soil layers (15-75cm). The carbon management index (CMI) was ranged from 91-130 with an average value of 110.4 in agroforestry land use; 122-182 with an average value of 142.6 in agriculture land use and 97-137 with an average value 104.2 in horticulture land use. The highest CMI in our study followed the order of agriculture > horticulture > agroforestry demonstrating that land conversion from fallow to agriculture crops cultivation have more sensitivity to the changes in SOC and other fractions than other land uses. The labile soil carbon fractions were significantly positively (P<0.05) correlated with TOC indicating that the changes in TOC content of soils is mainly influenced by the labile C pools. The correlation between the TOC and MBC (0.493**) was higher than that between POC and MBC (0.487**). MBC was highly correlated with $KMnO_4$ oxidizable C (0.544**).

Key words: Land use, carbon management index, labile carbon, particulate organic carbon, carbon stock, microbial biomass carbon, total carbon.

5.2 Introduction

Soil organic matter (SOM) is the central element of soil fertility, productivity and quality, as reduction in SOM is believed to create an array of negative effects on crop productivity. It becomes highly essential to maintain and improve its level in the soil is a pre-requisite to ensuring soil quality, future productivity, and sustainability (Katyal et al., 2001; Stevenson, 1986; Haynes, 2005; Lal et al., 1998). Several studies (Cambardella and Elliott, 1992; Chan, 1997) suggest that certain fractions of soil organic matter are more important in maintaining soil quality and are, therefore, more sensitive indicators of the impact of management practices. Chan et al. (1998) reported decrease in soil organic carbon under agriculture land use in comparison to forestry and horticulture land uses.

Changes in SOC due to management practices are difficult to quantify as these changes occur slowly, are relatively small compared to the vast SOC pool size, and vary both spatially and temporally (Russell et al., 2004). Most conventional methods used in soil organic carbon determination have been developed to maximize oxidation and recovery of C (Walkley and Black, 1934; Heanes, 1984). However, total organic carbon measurements might not be sensitive indicators of changes in soil quality due to land use changes. Adoption of procedures that can extract the more labile fraction preferentially might be a more useful approach for the characterization of soil organic carbon resulting from different management practices. Blair et al. (1995) used potassium permanganate oxidizible organic carbon as a measure of soil organic carbon lability and combined labile and non-labile carbon fractions to derive a carbon management index (CMI). Particulate organic carbon (POC) (Cambardella and Elliott, 1992) and soil microbial biomass carbon (SMBC) (Jenkinson and Powlson, 1976) are the other sensitive indicators for direct measurements of changes in SOM pools on short-term scale. In spite of the existence of a large number of studies that can provide a lot of relevant data pertaining to soil management, however, few are the studies that integrate the total soil organic C pool and the C lability into the CMI as a way to assess the capacity of management systems into promote soil quality.

Among the components of global change, land use change has the greatest impact on terrestrial ecosystems, profoundly altering land cover, biota and biogeochemical cycles (Walker and Desanker, 2004). Historically, the most important change of land use in north-east India was the shifting agriculture locally called *Jhum* cultivation. About 53% of the total area of north-east India is under forest cover which is higher than that of national average of about 20%. Forest Survey of India (1999) estimated that about 0.179 M ha of area every year is to be affected by Jhuming in north-east India. Jhuming with reduced fallow cycle of 2-3 years (10-15 years in the past) along with deforestation and forest fires resulted in loss of biodiversity and severe soil erosion, leading to depletion of soil productivity in the north-east India. On average, 36.64% of the total geographical area is degraded as a result of shifting cultivation in north-eastern India, which is almost two-fold more than the national average of 20.17% (Bhatt et al., 2006).

Tree plantations and agricultural crops, fertility maintenance including use of inorganic fertilizers and organic manures, tillage methods, and other cropping system components improve soil productivity through ecological and physicochemical changes that depend upon the quantity and quality of litter reaching soil surface and rate of litter decomposition and nutrient release. To sustain the quality and productivity of soils, knowledge of SOC in terms of its amount and quality is essential. The first comprehensive study of organic carbon (OC) status in Indian soils was conducted (Jenny and Raychaudhuri, 1960) by collecting 500 soil samples from different cultivated fields and forests with variable rainfall and temperature patterns. However, information on the changes in soil carbon stocks and fractions due to land use changes are limited in general (Sehgal et al., 1992; Velayutham et al., 2000; Chhabra et al., 2003; Tripathi et al, 2010; Rizvi et al., 2011) and particularly lacking in north-east India. Hence, there is a critical need for the development or identification of best land use and management practices that enhance soil organic C (SOC) sequestration. Increased sequestration of C in forest and agricultural soils has the potential to mitigate the increase in atmospheric greenhouse gases (Wei et al., 2008; Sampson and Scholes, 2006). Under the backdrop of aforesaid facts, the present study was carried out with the major objective to analyze the effect of land use change involving agroforestry, horticulture fruit trees and agriculture cultivation in East Khasi hills of Meghalaya on the potential of soil carbon sequestration in terms of the concentration of soil organic carbon and their fractions.

5.3 Materials and methods

5.3.1 Location of the study site and collection and preparation of soil samples

The study was carried out with different multipurpose tree species (MPTs) planted under agri-silvipastural system in 1983, horticulture tree species planted in 1994 and agricultural crops continuously cultivated for the past 10 years at research farm of Indian Council of Agricultural Research (ICAR) Complex for North-East Hill (NEH) Region, Umiam. The station is situated in the central part of Meghalaya in the East *Khasi* Hills of North-East India.

Soil samples were collected (Dhyani and Tripathi, 2000) during October-November in the year 2009 from 0-15, 15-30, 30-45, 45-60 and 60-75 cm soil depth at different blocks under all the treatments including controls (natural fallow). The soil samples were brought to the laboratory, air-dried at room temperature, ground to pass through 2-mm sieve and used for further analysis.

5.3.4 Methods of soil analysis

Organic carbon was determined in soil samples passed through 100 mesh sieve by wet digestion method of Walkley and Black (1934) as described by Jackson (1973).

The carbon stocks were estimated to a depth of 75-cm. Total organic carbon and bulk density were used to calculate the carbon stocks. For total organic carbon the method described by Snyder and Trofymow (1984) was followed. Soil bulk density values in the fields were measured by the core method (Allen et al., 1974).

Carbon Management Index (CMI) was computed by using the method as proposed by Blair et al. (1995). The amount of oxidizable carbon by 333 mM $KMnO_4$ (labile carbon) in soil was determined by following the procedure of Blair et al. (1995). Microbial biomass C (MBC) was determined by the chloroform fumigation incubation (CFI) technique as per the procedure of Jenkinson and Powlson (1976). The isolation of SOM particulate from the mineral-associated fraction was performed by physical fractionation (Cambardella and Elliott, 1992). Data obtained from this research programme was statistically analysed following standard statistical methods (Gomez and Gomez, 1984).

5.4 Results

5.4.1 Total organic carbon (TOC)

Results on soil TOC are shown in the table 5.2. Soil TOC tended to decrease with soil depths and was significantly affected by land uses and soil depths. However, interaction of land uses and soil depths did not show significant effects on soil TOC (Table 5.2). At the surface (0-15 cm) soil layer, TOC ranged from 2.14-4.18 g $100g^{-1}$ in agroforestry (AF), 1.51-3.20 g $100g^{-1}$in agriculture and from 1.82-3.60g $100g^{-1}$ in horticulture land use. In specific, TOC, on an average, was increased 46.7 % in the surface (0-15 cm) soil layer in relative to subsurface (15-75 cm) soil layers and the increase was greater in AF land use (49.1%) than horticulture (46.1%) and agriculture (44.8%) land uses. Amongst the land use systems, AF land use had significantly higher mean TOC (3.08%) followed by horticulture (2.69%) and the lowest was under agriculture land use (2.41%). In AF land use, Alder (3.60 %) showed the highest TOC content followed by Champak (3.20 %) and the lowest was found in control plots (No tree) (2.54%). Similarly, Potato (2.7%) followed by Turmeric (2.63) in agriculture land use and Peach (3.03%) followed by Guava (2.95%) in horticulture land uses recorded highest TOC content compared to other tree and crops species in their respective land uses. In all the land use systems, control plots contained low TOC contents. Between the 14 trees and crop species, Alder (41.8%) showed highest increase in TOC content followed by Potato (36%) and the lowest increase was found under Peach field (15%) compared to control plots. On an average, agriculture land use showed highest increase of 27% in TOC content followed by AF land use (26.4) and the lowest was horticulture (25.0%) land use than control plots. The soil TOC contents of the selected land uses followed the order of AF>Horticulture>Agriculture (Fig. 1). TOC content in soil was strongly influenced by the concentration of POC (0.564**), labile C (0.433*), non-labile C (0.973**) and MBC (0.493**) (Table 5.10,). Among the other soil properties, TOC content was positively and significantly related with available N (0.486**) while showing the negative and significant relation with soil bulk density (-0.491**).

5.4.2 Particulate organic carbon (POC) and POC/TOC ratio

The average soil POC ranged from 5315 to 1426 mg kg^{-1} between various land use systems with AF land use yielding the highest (4576 mg kg^{-1})(Fig. 2). The difference in soil POC due to the influence of land uses and soil depths were found to be highly significant in all the land uses.

However, interaction of land uses and soil depths did not show any considerable changes in soil POC content. In the surface soils, on an average, AF land use recorded highest POC content followed by horticulture and agriculture land use (Table 5.3). The per cent decrease in POC content at 15-30 cm depth was more under AF and agriculture land uses than horticulture land use compared to the surface soil (0-15cm). At 30-45 cm depth, only AF land use showed highest per cent decrease than the other two land uses. At 45-75 cm depths, all the three land uses showed similar degree of per cent decrease in POC content over the surface soil. Irrespective of the soil depths, adoption of various AF tree species increased the POC content by 54.9%; agriculture crops by 68.8% and horticulture tree species by 43.5% in relative to their control plots. The highest increase in POC concentration was found in Tree bean (67.1%), Potato (110.3%) and Khasi mandarin (59.5%) in AF, horticulture and agriculture land uses, respectively over control plots. Overall, soil POC content followed the order: Tree bean > Alder > Khasi Pine > Champak > Control in AF land use, Potato > Turmeric > Rice > Maize > Control in agriculture land use and Guava > Peach > Khasi mandarin > Pear > Control in horticulture land use. The POC to TOC ratio varied from 0.129-0.184, 0.057-0.119 and 0.131-0.148 in AF, agriculture and horticulture land uses, respectively (Table 5.4). Irrespective of the soil depths, the POC/TOC ratio was highest in the surface soil layer (0.105-0.194) and decreased down the soil layers. The average POC/TOC ratios in AF, agriculture and horticulture land uses were 0.153, 0.09 and 0.141 which were slightly lower than the control plots in their respective land uses. However, alder in AF land use, Potato and Peach in agriculture land use and Khasi mandarin in horticulture land use were relatively higher in the POC/TOC ration compared with control plots. In general, AF tree plantation and horticulture fruit trees increased the POC/TOC ratio by 62.6 and 49.7%, respectively in relative to the agriculture crop species (Fig. 3). Like TOC, POC also showed the similar trend in relation with other soil carbon fractions and pools except that POC had strong and positive relation with lability of C (0.321*) whereas, TOC had strong and negative correlation with lability of C (-0.384**) (Table 5.10).

5.4.3 Labile carbon, non-labile carbon and labile C to TOC ratio

The effects of different land uses on $KMnO_4$ oxidizable labile C and non-labile C content in the soil are presented in table 5.5. Adoption of various AF tree species significantly increased the labile C content in soil over the control plot while horticulture fruit trees plantation and agriculture cropping did not show significant changes on labile C (Table 5.5). The average labile C content was increased by 27, 51 and 17% by converting the fallow lands to AF, agriculture and horticulture land uses, respectively. On average, the labile C content was 132 and 63% higher in AF and horticulture land use compared to agriculture land use (Fig. 4). The changes in non-labile C were similar in pattern of those of the labile C at all the three land uses (table 5.5). Similar to that of labile C, non-labile C also was significantly influenced by the adoption of various agroforestry tree species. However, horticulture plantation and agriculture crops species did not show any considerable changes on non-labile C (Table 5.5). AF and horticulture practices showed 18 and 6.0% increase in non-labile C over agriculture land use practices (Fig. 5.5). Conversion of fallow lands to AF and horticulture land use decreased the labile C to TOC ratio by 14.4 and 20.0%, respectively while fallow lands to agriculture practices increased the ratio by 8.7% (Table 5.5). However, in comparison to agriculture land use, AF land use increased the labile C to TOC ratio by 74.1% and horticulture land use increased the ratio by 67.7% (Fig. 3). Labile C had highly significant and positive correlation with TOC (0.433**), POC (0.820**), MBC (0.544**) and lability of C (0.581**) while showing the significant and negative correlation with soil pH (-0.295*) and soil bulk density (-0.602**) (table 5.10). Non-labile carbon also showed the similar trend of labile C but it showed highest significant and positive correlation with SOC stocks (0.960**).

5.4.4 Lability of carbon and Lability Index (LI)

The effects of different land use systems on lability of C and lability index (LI) are given in the tables 5.5 and 5.7. Both the attributes did not vary significantly among the land use systems with the respective values ranging from 0.22 to 0.30 for C lability and 0.80-1.02 for LI in AF land use, from 0.11 to 0.17 for C lability and 1.06-1.68 for LI in agriculture land use and from 0.21 to 0.31 for C lability and 0.75-1.87 for LI in horticulture land use. The lability of C exhibited an increase of 9.0% by converting the fallow lands to agriculture land use whereas, fallow lands to AF and horticulture land use conversion decreased the lability of C by 20.0 and 27.0%, respectively. However, in relative to agriculture land use, AF and horticulture land use practices increased the lability of C by 97.7 and 89.8%, respectively. In AF land use, relatively high C lability was found under Champak field despite Khasi pine showed highest decrease over control plots. Similarly, in horticulture land use, Khasi mandarin decreased the lability of C by 43.2% followed by Pear (32.2%) over control plots. On the other hand, Potato in agriculture crops increased the lability of C by 42% followed by Maize while Rice and Turmeric decreased the lability of C compared to control. In contrary to the lability of C, LI was highest under agriculture land use than horticulture and AF land uses (Fig. 5.8). Adoption of AF and horticulture land uses decreased the LI by 52.9 and 20.8%, respectively compared to agriculture land use. Highest LI was noticed in Champak, Maize and Khasi mandarin in AF, agriculture and horticulture land uses, respectively (Table 5.7).

5.4.5 Carbon Pool Index (CPI) and Carbon Management Index (CMI)

CPI and CMI as influenced by the AF, agriculture and horticulture land use systems are given in table 5.7. Table 5.9 indicates that agriculture crops and horticulture fruit tress had no influence on CPI; however AF tree species significantly changed the CPI values. Similarly, the effects of AF and horticulture tree species and agriculture crops on CMI were found to be non-significant. CPI values were higher in AF land use (1.27) than agriculture land use while CMI followed the opposite trend being highest in agriculture land use (142.6) than AF and horticulture land uses (Fig. 5.8 and 5.9). Among the crop and tree species, Alder, Maize and Turmeric, and Guava recorded the highest CPI values while, Alder, Potato and Khasi mandarin were highest in CMI values in AF, agriculture and horticulture land uses, respectively. On the whole, AF land use increased the CPI values by 9.5% over agriculture land use; in contrary to this agriculture land use increased the CMI values by 29.2 and 36.9% over AF and horticulture land uses, respectively. Correlation studies between the CMI and various soil carbon fractions reveals that CMI had positive correlation with labile C, lability of C and LI while TOC, POC, MBC and CPI had the negative correlation with CMI (Table 5.10). Among the studied properties, only LI (0.884**) showed significant positive relation with CMI whereas, CPI (-0.109*) showed the negative and significant relation with CMI.

5.4.6 Microbial biomass carbon (MBC) and Microbial quotient (MQ)

Soil microbial biomass differed significantly only among the AF tree species and no considerable variation was observed in the agriculture and horticulture land uses (Table 5.6 and Fig. 5.6). It ranged from 333-548 mg kg^{-1} in AF land use, 217-309 mg kg^{-1} in agriculture land use and 282-401 mg kg^{-1} in horticulture land use. Among the three land uses, AF (425.1 mg kg^{-1}) had relatively the highest MBC followed by horticulture (340.6 mg kg^{-1}) land use and the lowest was recorded in agriculture (272.9 mg kg^{-1}) land use. Within each land uses, Alder (548 mg kg^{-1}), turmeric (309 mg kg^{-1}) and Khasi mandarin (401 mg kg^{-1}) showed highest MBC in AF, agriculture and horticulture land uses, respectively (Table 5.6). In all the three land uses, control plots had lowest MBC (333, 217 and 282.4 mg kg^{-1} in AF, agriculture and horticulture land uses, respectively). Nearly 34, 31 and 26 per cent increase in soil MBC was observed in AF, agriculture and horticulture land uses, respectively compared to their control plots. Overall, 56% increase in soil MBC under AF land use was observed compared to agriculture land use (Fig. 5.6). In general adoption of various AF tree species, agriculture crops and horticulture fruit trees did not show any significant changes on soil MBC. Microbial quotient (MQ) was calculated as a fraction of TOC. From the table 5.6, it is clearly evident that land use systems did not change the MQ significantly. MQ values varied from 1.73-1.97% and the values were more or less similar in all the three land uses with relatively higher values in horticulture land use than those of agriculture and AF land uses. Land use change from fallow to AF, horticulture and agriculture practices increased the MQ with highest increase in horticulture land use (13%) (Fig. 5.7). In AF land use, the MQ values followed the order of Champak>Control>Tree bean>Alder>Khasi pine, in agriculture land use, Control>Turmeric>Rice>Maize>Potato and Pear>Guava>K. Mandarin>Peach>Control in horticulture land use.

5.4.7 Carbon stocks

Soil carbon stocks at various depths varied significantly and tended to be higher under AF land use than those under agriculture and horticulture land uses (Table 5.8). It ranged from 42.2 (60-75 cm) to 68.6 (0-15 cm) Mg ha^{-1}in AF land use; 33.6 (60-75cm) to 62.1 (0-15 cm) Mg ha^{-1}in agriculture land use and 39.1 (60-75 cm) to 68.0 (0-15 cm) Mg ha^{-1} in horticulture land use. Conversion of fallow lands to AF, agriculture and horticulture land uses increased the SOC stocks in the surface (0-15 cm) soil by 35, 34.5 and 36%, respectively in relative to subsurface soil layers (15-75cm). However, the per cent increase was greater under agriculture land use (80%) followed by horticulture (74%) and the lowest was under AF (60%) land use in the surface soil (0-15cm) in relative to the subsurface soil (60-75 cm). The statistical analysis of SOC stocks did not show significant differences among the AF tree species and among the horticulture tree species. However, following agriculture farming systems significantly influenced the SOC stocks (Table 5.9). The average (0-75 cm) SOC stocks in AF land use ranged from 47.8 Mg ha^{-1}(Control) to 60.2 Mg ha^{-1}(Alder) with an average carbon stocks of 54.3 Mg ha^{-1}, while it ranged from 40.7 Mg ha^{-1}(Control) to 55.4 Mg ha^{-1} (Potato) with an average of 49.3Mg ha^{-1} and 46.1Mg ha^{-1} (Control) to 59.2 Mg ha^{-1}(Peach) with an average of 53.7 Mg $ha^{-1}$for the agriculture and horticulture land use systems, respectively (Table 5.8). In all the three land uses, control plots showed the lowest SOC stocks. Indeed, land conversion from fallow to AF, agriculture and horticulture land uses increased the SOC stocks by 17, 26.5 and 20.5%, respectively. However, amongst the three land uses, adoption of AF tree species stored highest SOC stocks followed by horticulture tree species and the lowest was found under agriculture land use (Fig. 5.10). TOC, POC Labile C, non-labile C, MBC and CPI showed positive correlation with soil C stocks; however, significant correlation of SOC stocks was observed only with TOC (0.929**), POC (0.368*), non-labile (0.960**) and MBC (0.297*). SOC stocks were negatively correlated with lability of C, LI and CMI (Table 5.10) although significant negative correlation observed only between SOC stocks lability of C (-0.530**). Among the other soil properties, SOC stocks had significant positive relation with available N (0.320*), potassium (0.336*). SOC stocks were negatively and significantly correlated with lability of C (-0.230**) whereas, it had negative relation with Li and CMI (Table 5.10).

5.5 Discussion

Soils are thought to have a finite carrying capacity for carbon based on parent material (texture), ambient temperature, annual precipitation, and net plant primary production. However, microbial transformation and turnover of soil organic matter can also influence the magnitude of soil C storage. The microbially mediated reactions fractionate organic materials into various reactive C pools such as labile, slowly decomposable, and resistant with respect to their persistence in soil. Currently, interest in the effects of SOC fractions on soil quality indicators of physical, chemical and biological properties under different land-use types is increasing (Campbell et al. 1999; Chan et al. 2002). The importance of C fractionation to soil carbon storage is evident by the intriguing possibility of manipulating the system into bypassing the labile pool with transformation directly into the slow and resistant pools.

5.5.1 Total organic carbon (TOC)

The amount of organic C in the soil results from the net balance between the rate of organic material inputs and rate of mineralization in SOC (Post and Kwon, 2000). A good farming practice can decrease CO_2 evolution from soil into the atmosphere and enhance soil fertility and thus productivity (Lal, 2004). This is more important in tropical and subtropical region where soils are inherently low in organic carbon content and production system is fragile. Studies have shown that such an increase in soil organic carbon levels is directly linked to the amount and quality of organic residues return to the soils. Amongst the three land uses studied, greater increase of TOC content was observed in soils under agroforestry plantation and lowest TOC was in soils under agriculture crops. The highest TOC concentration in agroforestry land use is attributed to the chemical stabilization of organic carbon in those soils (Percival et al., 2000). On the other hand, lack of vegetation cover, no organic input and high erosion due to rain's direct impact on the surface soil resulted in low SOC could be the reasons for the lowest TOC content in control plots (Saha et al., 2011). However, agriculture crops have larger potential to store organic carbon in soil that can be achieved by better management practices. The management practices to increase the SOC should be directed towards increasing residue inputs and decreasing decomposition rates which can be achieved by crop rotation, the use of cover crops and vegetative fallows (Lal and Kimble, 2000; Oelbermann et al., 2004). Irrespective of the land uses, surface soils were high in TOC content than the subsurface soil layers; however agroforestry and horticulture land uses contained high TOC in the subsurface soil layers than the subsurface soil layers of agriculture land use. The deeper root biomass C of trees and their decreased mineralization in both land uses might have increased the TOC accumulation in the subsurface soil layers. Kaiser et al. (2002) observed that forest subsoil has about 45% of total SOC of the profile. In fact, the confinement of SOC under the trees subsoil is essential for long term storage carbon due to reduced biological decomposition.

5.5.2 Labile carbon, non-labile carbon and lability index (LI)

Relative sizes of labile C in various ecosystems and their responses to disturbance could have important implications in understanding SOC stability. Also, identification of such fraction may serve as an indicator or even as a verification tool for SOC changes in terms of accounting for C stocks in the Kyoto Protocol. The labile C constitutes a small portion in TOC and is characterized by rapid mineralization due to labile nature of its constituents and lack of protection by soil colloids (Turchenek and Oades, 1979). This indicates that labile C responds to a greater extent to the change of land use (Jinbo et al., 2006). For this study, it was hypothesized that the conversions of fallow lands into the agroforestry plantation, fruit trees plantation and agriculture cultivation can alter the size of soil labile C. As hypothesized, levels of labile C in the soil significantly increased after the conversion (Table 5.5). These results are in accordance with the findings of others (Chen et al., 2004a; Xu et al., 2008). In addition to the differences in microclimatic conditions, explanations given for the differences between the three land uses have included differences in the ground vegetation cover, quantity and quality of organic matter inputs to soils. Management practices such as tillage operation for soil preparation and less crop residue addition with high mineralization rates resulted in lower labile carbon in agriculture land use compared to other land use systems. Lack of vegetation covers and no organic matter inputs in the control plots caused high erosion that not only resulted in a direct depletion of SOC in the agriculture land use, but also led to a decrease of physical protection of organic C by soil mineral particles and soil aggregates (Yang et al., 2009). These factors lowered labile C in the control plots compared with those in agroforestry, horticulture tree and agriculture crop species. Non-labile C also followed the similar trend of labile C being highest in the agroforestry land use and lowest under agriculture land use. Furthermore, the declined nutrients supplied from crop residues, lower soil humic matter content and poor aggregate stability in agriculture land use could enhance soil and water losses, and have the carbon which has been incorporated into soil aggregates at molecular or aggregate level to release and decompose. Thus, the balance between labile and non-labile C was disturbed, and the non-labile C was constantly transferred and decomposed, leading to the decrease of both C fractions. This may be the main reason for decrease of labile and non-labile C under agriculture land use (Xinyu et al., 2006). Six et al. (2002) reported that land-use change not only influenced the quantity of SOM and resulted in the change of labile C and total organic carbon contents, but also influenced the stability and quality of SOM and led to the transfer of labile and non-labile C in soil. Amongst the three land uses, AF land use had highest labile C to TOC ratio; however, all the three land uses were highest in the labile to TOC ratio in relative to the control plots. This result suggests that the highest quality of soil organic matter under these land uses over control plots (Zhou et al., 2006).

5.5.3 Particulate organic carbon (POC) and POC/TOC ratio

Since SOC is heterogeneous in nature, it is likely that the amounts of labile fractions of SOC may change due to changes in the amount and nature of C inputs, even though the total amount of SOC remains essentially unaffected. Cambardella and Elliott (1992) suggested that the soil organic matter C fraction >53 µm could be used as the labile fraction in grasslands and cultivated soils since it was more sensitive to change in management than the whole soil organic matter C. However, Zach et al. (2006) found that the POC >53 µm fraction was no more sensitive as an indicator to change in management than the whole soil in both pasture and cultivated soils. This was also confirmed by Dou et al. (2008) for cultivated soils, where they found that no labile SOC pool, including the POC >53 µm fractions, was more sensitive than the whole SOC for soils. It represents the un-decomposed plant residues and partly decomposed plant materials at an early stage of decomposition (Leifeld and Kogel-Knabner, 2005).

Agroforestry systems yielded highest POC than horticulture and agriculture land uses. The highest accumulation of SOC in agroforestry land use might have increased the aggregation which can effectively protect POC from the decomposition and enhanced the POC accumulation in agroforestry land use compared with other land uses (Blanco-Caqui and Lal, 2004; Mao et al., 2011). The increased POC accumulation under these land uses also suggests that soils under these land uses build active C pools (Saha et al., 2011). The lower POC concentration in agriculture land use than agroforestry land use may be related to the lower organic input addition and more losses of SOC due to cultivation practices such as tillage causing high mineralization and losses through erosion. Yang et al. (2009) also reported that cultivation affected POC content harmfully and significantly decreased the POC content after forest conversion. The magnitudes of POC revealed a clear decline with soil depths (Table 5.3). Because they are strongly related to root C inputs (Gao et al., 2000), and other organic residues are often accumulated at the soil surface. The increased POC content in the subsurface layers of all the land uses compared to the subsurface layers of control plots was mainly related to the left over biomass in agriculture land use, high root biomass under AF and horticulture land uses and also due to increased microbial biomass debris under these land uses. It is also suggested that the greater biochemical recalcitrance of root litter might have also increased the POC contents in soils depending upon the root biomass produced (Purakayastha et al., 2009). Our results are also consistent with the previous findings of Chen et al. (2004a) and Sarkhot et al. (2008). Table 5.9 confirms the findings of Yang et al. (2009) who had reported that POC content in soil significantly influenced by the land use systems and soil depths.

In the three land use systems, POC accounted for 19.7 (agriculture) to 32.0% (horticulture) of the SOC (POC/TOC ratio) at 0–75 cm depth, this was slightly higher than those obtained by Cambardella and Elliot (1992), which ranged between 18% and 25%. In general, the POC usually represents a smaller proportion (10–20%) of the SOC in warm and wet tropical and subtropical regions (Bayer et al., 2002). However, Franzluebbers and Arshad (1997) found a higher proportion of about 50% under dry or cold climate. The smaller proportions POC to SOC in agriculture land use observed in our study (Table 5.4) may probably be related to the higher biological decomposition of recent organic material inputs due to high aeration and losses by erosion, leading to less accumulation of POC (Chen et al., 2004b).

5.5.4 Microbial Biomass Carbon (MBC) and microbial quotient (MQ)

Soil microbial biomass, a living part of soil organic matter, is an agent of transformation for added and native organic matter and acts a labile reservoir for plant available nitrogen, phosphorous and sulphur (Jenkinson and Ladd, 1981). The activity of the microbial biomass is commonly used to characterize the microbial status of soil and to determine the effect of cultivation and field management practices on soil microorganisms (Perott et al., 1992). In our study, the results showed that soil microbial biomass differed significantly only among the AF tree species and no considerable variation was observed in the agriculture and horticulture land uses (Table 5.6 and Fig. 5.7). Soil MBC decreased following the land use change from fallow to agroforestry, horticulture plantations and agriculture crops cultivation. Among the three land uses, AF had the highest MBC followed by horticulture land use and the lowest was recorded in agriculture land use (Fig. 5.7). In all the three land uses, control plots had lowest MBC and nearly 34, 31 and 26 per cent increase in soil MBC was observed in AF, agriculture and horticulture land uses, respectively compared to their control plots. Overall, 56% increase in soil MBC under AF land use was observed compared to agriculture land use (Fig. 5.7). The chief contributory factor for the higher soil MBC in the AF land use than the other land uses seems to be the greater availability of nutrients due to the addition of higher plant litters and root biomass (Arunachalam and Pandey, 2003). The characteristics of vegetation differences directly through quality of litter and indirectly through changes in soil chemical and physical properties may have the influence on MBC in soil (McLean and Hunta, 2002). The differences in soil MBC within and among the land uses could also be due to the variation in soil moisture, stages of plant growth, soil temperature and substrate availability. Similar observations have been reported by several other workers (Chang and Juma, 1996; Campbell et al., 1999). The reduced organic inputs through crop residues along with tillage practices could be the possible reason for lowest MBC in agriculture land use than AF and horticulture land uses. According to Ross (1987), crop residues can have a larger effect on soil microbial biomass and activity, which, in turn, affect the ability of soil to supply nutrients to plants through soil organic matter turnover. In conclusion, although agriculture land use have lowest in soil MBC, land conversion from fallow to AF and fruit tree plantation and also cultivated land may profoundly modified the microclimate with different vegetation canopy and litter input thus favouring the enhancement of overall soil fertility status.

The ratio of MBC to SOC (microbial quotient) indicates the proportion of organic C that may be readily metabolized. It usually falls within the range of 1–4% (Sparling, 1992). However, because of differences in soil and management practices, variations in sampling date and analytical methods, wider ranges of MBC/SOC ratio from 0.27% to 7.0% have been reported (Insam et al., 1989; Omay et al., 1997). The present study showed that the ratio of MBC to SOC ranged from 1.61–2.15% (Table 5.6), which lies well within the range reported. In comparison with agriculture soils, agroforestry and horticulture land uses showed higher proportions of MBC to SOC at the 0–15 cm depth (1.67-2.15%) (Fig. 5.8). The result can be explained on the basis of the fact that more diversified organic substrate production and input under these different land use systems support more interdependent food web which allows the maintenance of higher MBC per unit soil organic C (Anderson and Domsch, 1989). The increased organic inputs to agroforestry land use increased the microbial activity and caused higher conversion to microbial biomass leading to better stability of organic carbon in agroforestry land use system.

The $KMnO_4$ oxidizable C (labile C), POC and MBC have been proposed to be used as indicators to evaluate the effect of different soil management practices because these fractions may precede future changes of SOC (He et al., 2008; Xu et al., 2008). In this study, significant differences in $KMnO_4$ oxidizable C, POC and MBC resulting from the land use change from fallow to agroforestry, horticulture plantation and agriculture cultivation were observed. The labile C, POC and MBC concentration values for the surface soil ranged from 3509-8129 mg kg^{-1} (a 2.31-fold difference), 2211 to 4576 mg kg^{-1} (a 2.07-fold difference), and 273 to 425 mg kg^{-1} (a 1.56-fold difference) among the three land use systems, while SOC content ranged from 1.12 to 1.61 g $100g^{-1}$ (a 1.43-fold difference) (Chen et al., 2005b). Also, significant correlations among labile fraction organic C and SOC were found. This demonstrates that it is desirable to understand the impacts of labile C pools on soil quality under different land uses or management practices because of their highest sensitivity to land use changes as evident above and also for maintaining soil physical and chemical properties and fertility (Blair and Crocker, 2000). The variations in all the C fractions among the tree and crop species may be related to the quantity of C and nutrient inputs added to the soils by litter fall, root biomass and crop residues which is directly dependent on the trees and crop productivity, soil management practices and site-specific edaphic and climatic conditions (Nair, 1993). The quality of the organic C inputs also a key factor for the allocation of the C in to different pools (Oelbermann et al., 2004).

5.5.5 Carbon stocks

The most sensible approach to study SOC, would be on an unit area basis for a specified depth interval which requires information on the spatial distribution of soil types, SOC and bulk density of soils. It would thus provide a better understanding of the terrestrial reservoir of SOC far beyond the general objectives of C sequestration in soils and the detrimental effects of global warming. The SOC stocks present in the soil represent a dynamic balance between the input of dead plant materials and the loss from the decomposition. Carbon can be stored in diverse forms with a wide range of mean residence time (Torn et al., 1994). SOC stocks values under different agricultural land uses and management practices significantly decreased downward the soil profile. Degryze et al. (2004) and Liu et al. (2003) also reported decreased SOC stocks down the soil layers under various land use systems. The effects of agricultural land uses and their interaction with soil depths did not show significant changes on SOC stocks; however, agriculture land use significantly influenced the SOC stocks. SOC stocks values were higher in the upper soil layers than those in the lower soil layers under all the land uses, which may be related to higher litter and root biomass inputs from tree crops, and crop residues and root biomass of agriculture crops. However, in relative to the subsurface soil layers of agriculture land use, the subsurface soil layers in both agroforestry and horticulture land uses contained highest SOC stocks. This may be attributed to the deeper root systems and more root biomass with relatively less mineralization compared to agriculture crops (Liu et al., 2003; Xinyu et al., 2006). Amongst the three land uses, SOC stocks were high through the 0-75-cm soil under agroforestry followed by horticulture plantation and lowest in soils under agriculture crops, which may be related to the highest addition of litter and root biomass. The results on SOC stocks from our study indicates that agroforestry and horticulture fruit trees are beneficial for the prevention of water and wind erosions and because it increased SOC contents and stocks values within the 75cm profiles and therefore it is an ideal option for land uses and management practices to improve income, sustain soil quality, and sequestrate carbon.

5.5.6 Correlation coefficients between soil organic carbon fractions

Correlation coefficients between the organic carbon fractions in all the land uses are dissimilar and shown in table 5.10. This indicates the differential accumulation pattern of soil organic carbon under these land use systems. Computing the relationship with total organic carbon, as an independent variable, TOC showed high correlation with POC, labile C and MBC. On the contrary, bulk density negatively correlated with all the C fractions in the soil. A negative correlation between SOC content and bulk density has been reported earlier (Aragon et al., 2000). A strong and positive correlation between these labile C pools and TOC indicated that the changes in TOC content of soils is mainly influenced by the labile C pools. Among the three labile C pools, POC had higher significant correlation (0.579**) with TOC than the other two labile C pools indicating that POC in these land uses largely depends upon the quantity of TOC in the soil. This is in agreement with the findings of Luan et al. (2010) who reported positive and significant correlation between TOC and POC. The correlation between the TOC and MBC (0.493**) was higher than that between POC and MBC (0.487**). However, an opposite trend was also reported by Camberdella and Elliott (1994) and Gregorich et al. (1994). Such relationship also indicates that MBC is largely dependent on the other labile C pools. In the present study, MBC was highly correlated with $KMnO_4$ oxidizable C (0.544**). This is also supported by the findings of Rudrappa et al. (2006). However, the concentration of labile C is less dependent on TOC than POC and MBC as indicated in the table 5.10. In contrast to our findings, Zhang et al. (2010) reported no significant correlation between labile C and MBC. On the whole, these findings in our study clearly signify the differential pattern of accumulation of various organic C pools in different land use systems.

5.5.7 Carbon management Index (CMI)

CMI is an index for SOM change induced by soil management practices, and a systematic and sensitive monitoring method for SOC change. It could reflect the degree of degradation and regeneration of soil quality (Loginow et al., 1987). CMI combined index and lability of soil carbon pool under anthropogenic influence, and could reflect environmental effects on quantity changes of total organic carbon and labile C. Thus, CMI could comprehensively and dynamically reflect the environmental effects on SOC quality (Xu et al., 2006). The results further proved that soil carbon pool and CMI were influenced by agroforestry, horticulture and agriculture land use systems, and CMI could comprehensively and dynamically reflect the effect of these land use change and management practices on soil carbon pool and fertility. Several studies (Vieira et al., 2007; Rudrappa et al., 2006; Wei et al., 2008;) used undisturbed land which is more stable in soil organic carbon as the reference for calculating the CMI and indicated that cultivation of undisturbed lands decreased the SOC content and other C fractions and thus CMI. In the present study, we used three different control plots for each land uses as reference land which are barren and had very low organic carbon and other C fractions. These barren/fallow lands conversions into agroforestry plantation, fruit trees plantation and agriculture cropping systems increased total, labile, non-labile, microbial biomass and particulate organic carbon content due to higher litter and root biomass input, crop residue incorporation and thus increased the CMI in all the land uses. The highest CMI in our study followed the order of Agriculture>Horticulture>Agroforestry indicating land conversion from fallow to agriculture crops cultivation have more sensitivity to the changes in soil organic carbon and other fractions than other land uses. This also indicates that due to its high sensitivity if agriculture lands are not properly managed more losses of SOC and other C fractions particularly labile C fractions will occur which may have direct impact on global climate change. The variations in CMI among the tree and crop species may be attributed to the differences in root and shoot biomass, decomposition characteristics, soil properties, vegetation types and microclimate of the soil. CMI in all the three lands uses in the present study was not correlated significantly with total carbon, labile and non-labile carbon and microbial biomass carbon. This was in contrast to the findings of Wei et al. (2008) who had reported positive and significant correlation of CMI with the above C fractions. Among the soil properties, only soil available P had significant and positive correlation with CMI which is in accordance with the findings of Shen et al. (2000).

5.5.8 Conclusions

In this study, total, organic and labile carbon concentrations were significantly higher in agroforestry tree plantations than in the horticulture tree plantation and agriculture farming systems. In the agroforestry land use system Alder and Champak; in horticulture tree plantation Guava and Peach, and in agriculture land use Potato and Turmeric had significantly higher levels of all the carbon fractions than other tree and crop species in their respective land uses. Consistent with other studies, active (labile) soil carbon components reacted more sensitively to changes in land management than total or organic carbon fractions. However, all the carbon fractions were highest under agriculture, horticulture and agroforestry land uses compared with the control or fallow lands. Hence, total and labile fractions were combined to derive the carbon management index (CMI), a more complete indicator of soil carbon dynamics.

Although total soil organic carbon varied significantly across the land use types, relatively highest significant changes in particulate organic carbon, labile carbon and microbial biomass carbon were observed in all the land uses, with agroforestry yielding the most and croplands containing the least. These organic matter fractions are closely associated with other soil properties including soil pH, bulk density and available nitrogen.

Results showed that the SOC stocks in different land use systems were markedly affected in the 0–75cm soil layers with highest stocks in the surface layer and decreased sharply with increasing soil depths. SOC contents were higher in soils under agroforestry and horticulture fruit trees plantation but lower in soils under agriculture crops. There were no significant effects of the land use systems on SOC stocks; however soil depths significantly influenced the SOC stocks.

Table 5.1 Initial soil properties of different land uses selected for the study

Land use system	Soil properties						
	pH	BD (Mg m^{-3})	MWD (mm)	Moisture (g $100g^{-1}$)	N (kg ha^{-1})	P	K
				Agroforestry			
Champak	4.60	1.11	31.0	31.0	522.3	32.1	313.6
Tree bean	4.59	1.15	26.1	26.1	496.3	30.1	286.7
Alder	4.38	1.07	31.8	31.8	584.3	47.2	361.2
Khasi Pine	4.36	1.05	33.8	33.8	464.8	23.7	276.0
Control (No tree)	4.76	1.18	28.4	28.4	403.4	19.3	248.4
				Agriculture			
Maize	5.12	1.27	29.9	29.9	394.4	49.1	292.7
Potato	5.24	1.31	28.9	28.9	414.6	65.6	365.1
Rice	5.09	1.33	32.2	32.2	421.4	23.5	322.2
Turmeric	5.29	1.25	26.7	26.7	398.9	36.9	306.1
Control (No crop)	4.81	1.33	24.8	24.8	387.7	20.7	203.1
				Horticulture			
Pear	5.71	1.28	31.6	31.6	437.1	43.2	323.6
Peach	6.24	1.23	32.7	32.7	455.1	39.1	495.2
K Mandarin	5.75	1.29	28.7	28.7	428.1	45.3	419.6
Guava	5.43	1.19	29.9	29.9	446.1	24.6	343.5
Control (No tree)	4.82	1.30	27.1	27.1	434.9	26.9	328.5

Table 5.2 Effect of different land use systems on depth-wise soil total organic carbon (g $100g^{-1}$)

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	4.17	3.76	3.08	2.74	2.24	3.20ab
Tree bean	3.99	3.55	3.07	2.54	2.07	3.04b
Alder	5.52	3.68	3.52	2.85	2.44	3.60a
Khasi Pine	4.08	3.32	2.97	2.41	2.21	3.00bc
Control (No tree)	3.12	3.01	2.68	2.12	1.76	2.54c
Mean	4.18a	3.46b	3.06b	2.53c	2.14c	
			Agriculture			
Maize	3.20	2.71	2.57	2.18	1.45	2.42a
Potato	3.51	3.21	2.80	2.34	1.64	2.70a
Rice	3.08	2.52	2.27	2.05	1.60	2.30ab
Turmeric	3.62	3.17	2.47	2.22	1.69	2.63a
Control (No crop)	2.59	2.40	2.14	1.58	1.19	1.98b
Mean	3.20a	2.80ab	2.45bc	2.07cd	1.51d	
			Horticulture			
Pear	3.56	2.86	2.41	2.12	1.87	2.57ab
Peach	3.84	3.43	3.25	2.63	2.01	3.03a
K Mandarin	3.69	3.09	2.72	2.00	1.81	2.66ab
Guava	4.07	3.39	2.92	2.44	1.92	2.95a
Control (No tree)	2.83	2.65	2.27	1.95	1.49	2.24b
Mean	3.60a	3.08ab	2.72bc	2.23cd	1.82d	

Table 5.3 Effect of different land use systems on depth-wise soil particulate organic carbon (mg kg^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	7920	5440	3653	2720	2000	4347a
Tree bean	7640	6040	5320	4680	2893	5315a
Alder	8720	6400	4160	4240	2560	5216a
Khasi Pine	7480	5960	4693	3587	2400	4824a
Control (No tree)	5160	4000	3080	2074	1587	3180b
Mean	7384a	5568b	4181c	3460c	2288d	
			Agriculture			
Maize	3120	2440	2080	1560	907	2021bc
Potato	4000	3360	2960	2640	2033	2999a
Rice	3006	2716	2490	1480	999	2138bc
Turmeric	4800	2560	2160	1760	1067	2469ab
Control (No crop)	2160	1850	1440	1120	560	1426c
Mean	3417a	2585b	2226bc	1712cd	1113d	
			Horticulture			
Pear	5000	4127	3313	2467	1910	3363bc
Peach	6200	5800	4080	2320	2011	4082ab
K Mandarin	5840	5520	3120	2760	1573	3763ab
Guava	6240	5360	4000	3640	2320	4312a
Control (No tree)	3960	3280	3100	2100	1080	2704c
Mean	5448a	4817a	3523b	2657c	1779d	

Table 5.4 Effect of different land use systems on particulate organic carbon to total organic carbon ratio

Land use system	Depth (cm)					
	0-15	15-30	30-45	45-60	60-75	Mean
			Agroforestry			
Champak	0.190	0.160	0.105	0.100	0.088	0.129a
Tree bean	0.193	0.181	0.186	0.205	0.157	0.184a
Alder	0.159	0.188	0.118	0.154	0.104	0.145a
Khasi Pine	0.183	0.176	0.162	0.148	0.106	0.155a
Control (No tree)	0.245	0.175	0.132	0.126	0.104	0.156a
Mean	0.194a	0.176a	0.141ab	0.146ab	0.112b	
			Agriculture			
Maize	0.093	0.096	0.085	0.080	0.079	0.087ab
Potato	0.116	0.103	0.104	0.119	0.149	0.118a
Rice	0.058	0.067	0.066	0.058	0.036	0.057b
Turmeric	0.138	0.087	0.089	0.084	0.063	0.092ab
Control (No crop)	0.120	0.121	0.125	0.100	0.126	0.119a
Mean	0.105a	0.095a	0.094a	0.088a	0.091a	
			Horticulture			
Pear	0.150	0.145	0.139	0.114	0.106	0.131a
Peach	0.185	0.204	0.148	0.094	0.112	0.148a
K Mandarin	0.184	0.189	0.114	0.146	0.088	0.144a
Guava	0.152	0.162	0.135	0.149	0.110	0.142a
Control (No tree)	0.194	0.188	0.144	0.113	0.076	0.143a
Mean	0.173a	0.177a	0.136ab	0.123b	0.098b	

Table 5.5 Effect of different land use systems on labile carbon, non-labile carbon, lability of carbon and labile C to TOC ratio

Land use system	Labile C (mg kg^{-1})	Non-labile C (mg kg^{-1})	Lability of C	LC/TOC ratio
		Agroforestry		
Champak	8761ab	32906b	0.27a	0.210a
Tree bean	7858bc	32042b	0.25a	0.197a
Alder	10156a	45044a	0.23a	0.184a
Khasi Pine	7191bc	33609b	0.22a	0.181a
Control (No tree)	6680c	24520b	0.30a	0.226a
		Agriculture		
Maize	4670a	27330a	0.17a	0.146a
Potato	3925ab	31175a	0.13a	0.112a
Rice	3134ab	27666a	0.12a	0.105a
Turmeric	3331ab	32869a	0.11a	0.097a
Control (No crop)	2487b	23413a	0.12a	0.106a
		Horticulture		
Pear	5870a	29730a	0.21a	0.172a
Peach	5572a	32828a	0.22a	0.170a
K Mandarin	6114a	30786a	0.30a	0.202a
Guava	6048a	34652a	0.18a	0.149a
Control (No tree)	5043a	23223a	0.31a	0.217a

Table 5.6 Effect of different land use systems on soil microbial biomass carbon and microbial quotient

Land use system	MBC (mg kg^{-1})	Microbial quotient (%)
	Agroforestry	
Champak	358.0ab	1.89a
Tree bean	314.0ab	1.76a
Alder	360.0a	1.76a
Khasi Pine	302.0b	1.67a
Control (No tree)	270.0b	1.76a
	Agriculture	
Maize	148.0a	1.62a
Potato	216.0a	1.72a
Rice	190.0a	1.84a
Turmeric	224.0a	1.85a
Control (No crop)	182.0a	1.61a
	Horticulture	
Pear	276.0a	2.15a
Peach	242.0a	1.88a
K Mandarin	232.0a	2.11a
Guava	212.0a	1.91a
Control (No tree)	200.0a	1.78a

Table 5.7 Effect of different land use systems on lability index (LI), carbon pool index (CPI) and carbon management index (CMI)

Land use system	LI	CPI	CPM
		Agroforestry	
Champak	1.02a	1.20b	119.4a
Tree bean	0.90a	1.15b	101.5a
Alder	0.85a	1.59a	129.6a
Khasi Pine	0.80a	1.16b	91.1a
		Agriculture	
Maize	1.68a	1.11a	181.7a
Potato	1.36a	1.21a	149.0a
Rice	1.36a	1.09a	117.9a
Turmeric	1.06a	1.21a	122.0a
		Horticulture	
Pear	1.08a	1.15a	97.2a
Peach	0.75a	1.13a	86.2a
K Mandarin	1.87a	1.16a	136.6a
Guava	0.82a	1.26a	96.9a

Table 5.8 Effect of different land use systems on soil carbon stocks (Mg ha^{-1})

Land use system	Depth (cm)					
	0-15	**15-30**	**30-45**	**45-60**	**60-75**	**Mean**
			Agroforestry			
Champak	67.0	60.9	52.5	50.1	43.3	54.7ab
Tree bean	66.4	61.1	54.8	47.7	41.3	54.3ab
Alder	86.5	58.8	59.4	50.7	45.7	60.2a
Khasi Pine	68.9	58.0	54.6	46.4	44.2	54.4ab
Control (No tree)	54.3	54.8	51.6	41.6	36.7	47.8b
Mean	68.6a	58.7b	54.6bc	47.3cd	42.2d	
			Agriculture			
Maize	61.1	52.7	52.2	46.0	31.3	48.7ab
Potato	68.6	64.6	57.5	49.8	36.3	55.4a
Rice	61.3	51.8	49.2	45.5	36.0	48.8ab
Turmeric	67.7	62.0	49.8	48.6	37.5	53.1a
Control (No crop)	51.7	45.5	44.8	34.6	26.8	40.7b
Mean	62.1a	55.3ab	50.7bc	44.9c	33.6d	
			Horticulture			
Pear	68.4	56.4	49.2	44.1	40.1	51.6ab
Peach	70.9	65.7	64.5	53.5	41.7	59.2a
K Mandarin	71.9	61.2	55.5	42.0	39.5	54.0ab
Guava	73.3	64.3	57.4	50.2	41.3	57.3ab
Control (No tree)	55.5	53.1	47.0	41.9	32.9	46.1b
Mean	68.0a	60.1ab	54.7bc	46.3cd	39.1d	

Table 5.9 Effect of land use systems, depth and their interaction on C stocks, CMI and various soil C fractions

Dependent variables	Agroforestry			Agriculture			Horticulture		
	LUS	Depth (D)	LUS x D	LUS	Depth (D)	LUS x D	LUS	Depth (D)	LUS x D
TOC	**	**	NS	**	**	NS	*	**	NS
POC	**	**	NS	**	**	NS	**	**	NS
POC/TOC ratio									
Stocks	NS	**	NS	**	**	NS	NS	**	NS
Labile C	**	-	-	NS	-	-	NS	-	-
Labile C/TOC ratio		-	-		-	-		-	-
Non-labile C	**	-	-	NS	-	-	NS	-	-
MBC	*	-	-	NS	-	-	NS	-	-
Microbial quotient	NS	-	-	NS	-	-	NS	-	-
Lability of C	NS	-	-	NS	-	-	NS	-	-
Lability Index	NS	-	-	NS	-	-	NS	-	-
CPI	*	-	-	NS	-	-	NS	-	-
CMI	NS	-	-	NS	-	-	NS	-	-

Table 5.10 Correlation coefficients between various C fractions and soil properties

	PH	BD	MOIST	N	P	K	TOC	POC	STOCKS	LABILE	NONLAB	MBC	LABILITY	LI	CPI	CMI
pH	1	0.328*	0.392**	-0.033	0.119	0.659**	-0.034	-0.160	0.163	-0.295*	0.039	-0.096	-0.110	0.051	-0.047	-0.140
BD		1	0.057	-0.533**	0.023	-0.008	-0.491**	-0.660**	-0.159	-0.692**	-0.355*	-0.602**	-0.196	0.147	-0.246	0.093
Moisture			1	0.054	0.072	0.218	-0.101	-0.249	-0.042	-0.206	-0.056	-0.036	-0.081	-0.122	0.086	-0.274
N				1	0.104	0.374*	0.486**	0.579**	0.320*	0.590**	0.375*	0.529**	0.210	-0.099	0.267	-0.026
P					1	0.194	0.039	0.021	0.073	0.040	0.032	-0.113	-0.041	0.429**	-0.185	0.435**
K						1	0.261	0.240	0.336*	0.145	0.245	0.229	0.082	-0.082	0.145	-0.113
TOC							1	0.564**	0.929**	0.433**	0.973**	0.493**	-0.384**	-0.086	0.304	-0.026
POC								1	0.368*	0.820**	0.400**	0.487**	0.321*	-0.125	0.102	-0.088
Stocks									1	0.182	0.960**	0.297*	-0.530**	-0.047	0.234	-0.029
Labile C										1	0.212	0.544**	0.581**	-0.034	0.248	0.101
Non-labile C											1	0.394**	-0.565**	-0.088	0.274	-0.059
MBC												1	0.083	-0.257	0.304	-0.189
Lability of C													1	-0.031	0.114	0.077
LI														1	-0.656**	0.884**
CPI															1	-0.41*
CMI																1

** Correlation is significant at the 0.01 level (2-tailed); *. Correlation is significant at the 0.05 level (2-tailed); TOC-Total organic carbon; POC-Particulate organic carbon; MBC-microbial biomass carbon; LI-Lability index; CPI-Carbon pool index; CMI-Carbon management index.

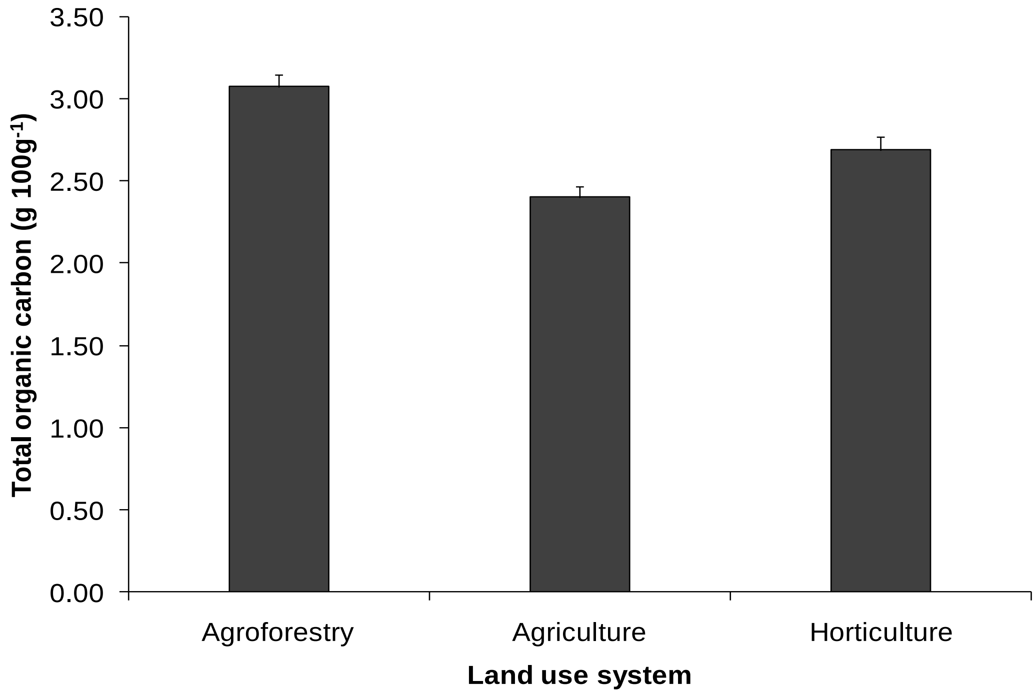


Fig. 5.1 Effect of different main land use systems on soil total organic carbon (g 100^{-1})

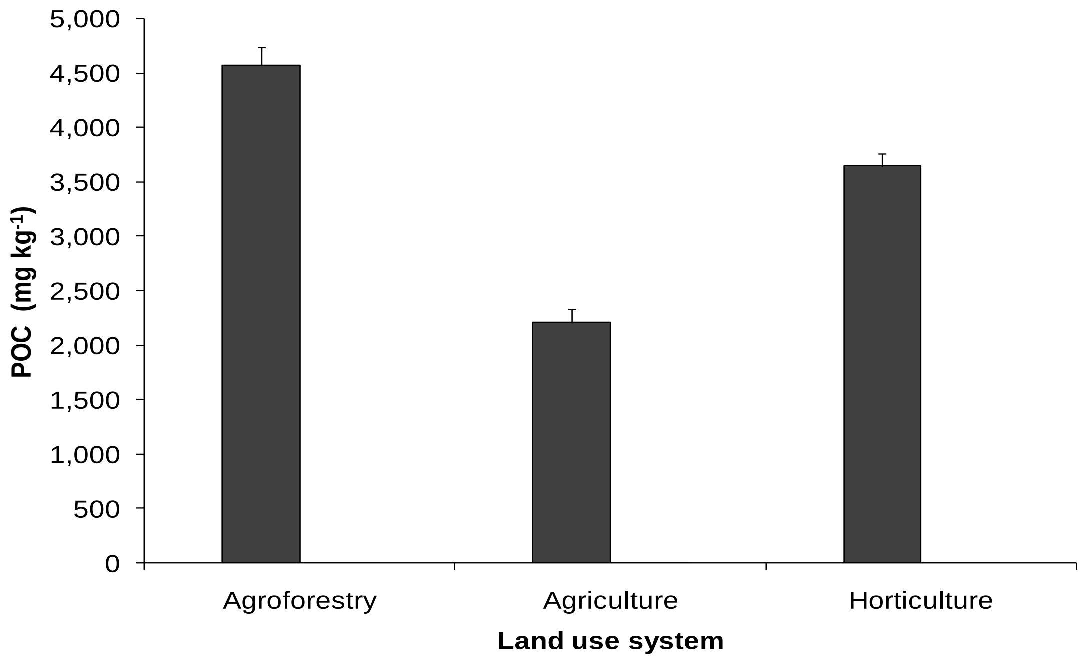


Fig. 5.2 Effect of different main land use systems on soil particulate organic carbon (mg kg^{-1})

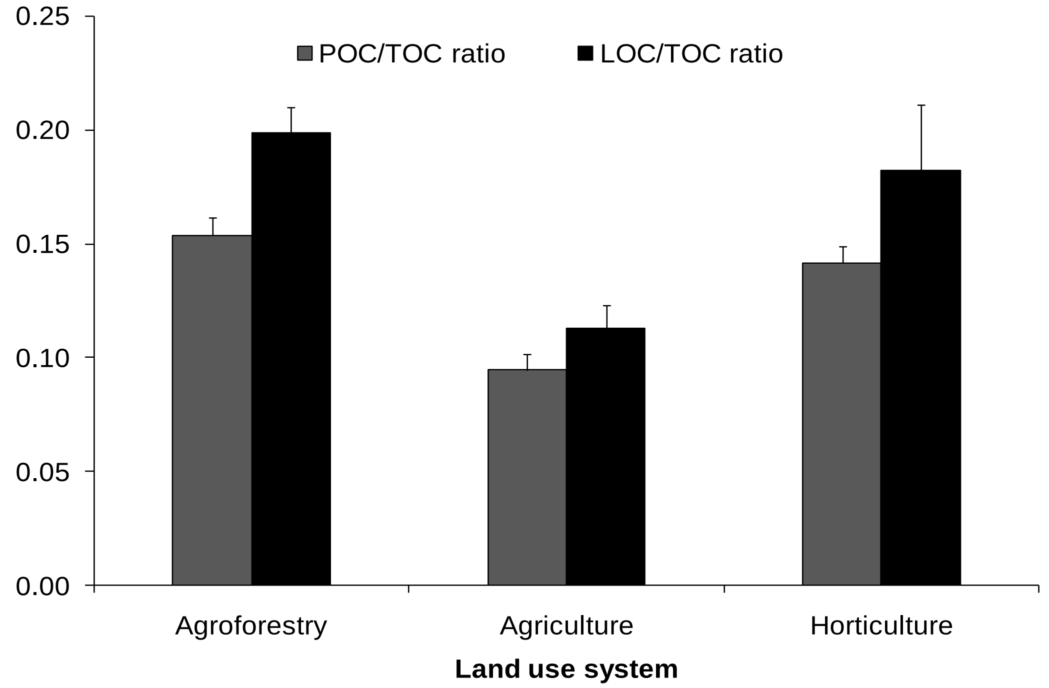


Fig. 5.3 Effect of different main land use systems on soil POC/TOC ratio and LOC/POC ratio

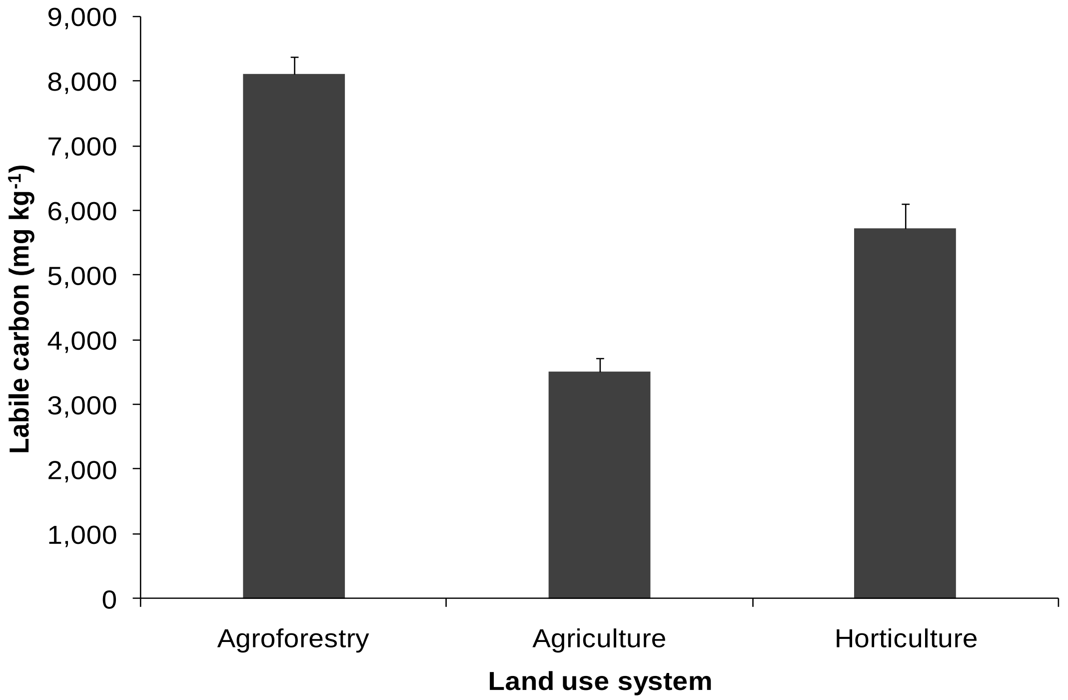


Fig. 5.4 Effect of different main land use systems on soil labile carbon (mg kg^{-1})

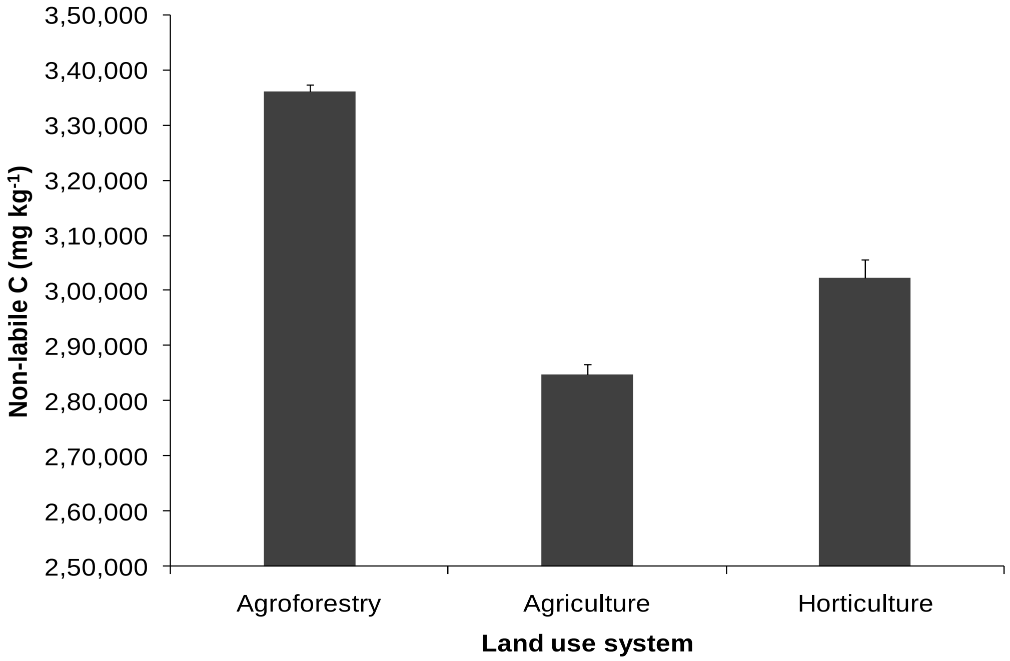


Fig. 5.5 Effect of different main land use systems on non-labile carbon (mg kg^{-1}) in soil

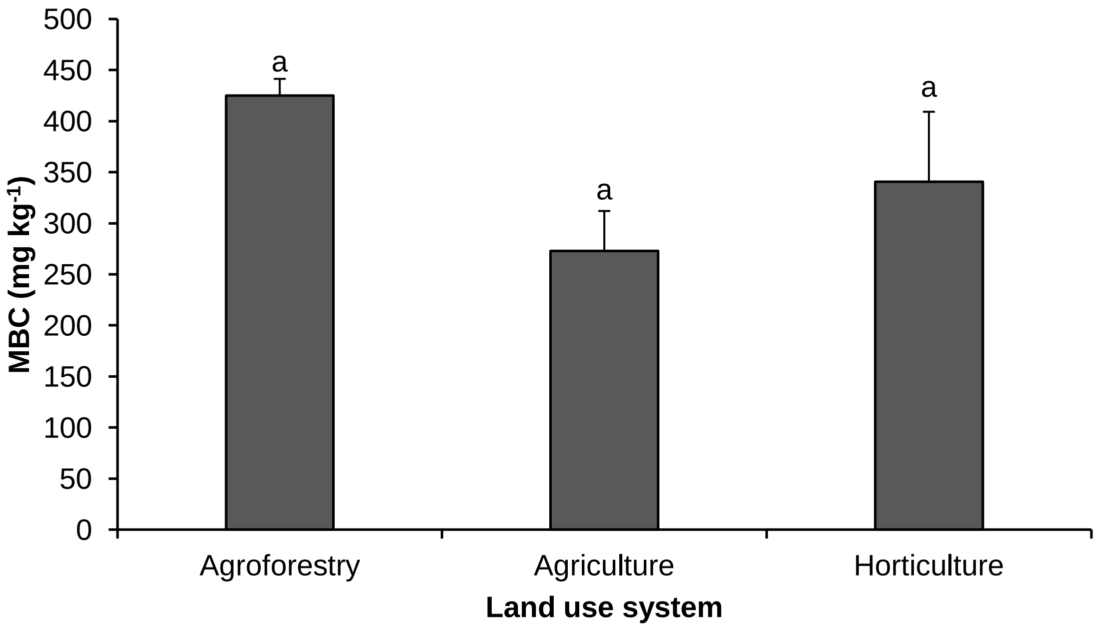


Fig. 5.6 Effect of different main land uses on microbial biomass carbon (mg kg^{-1})

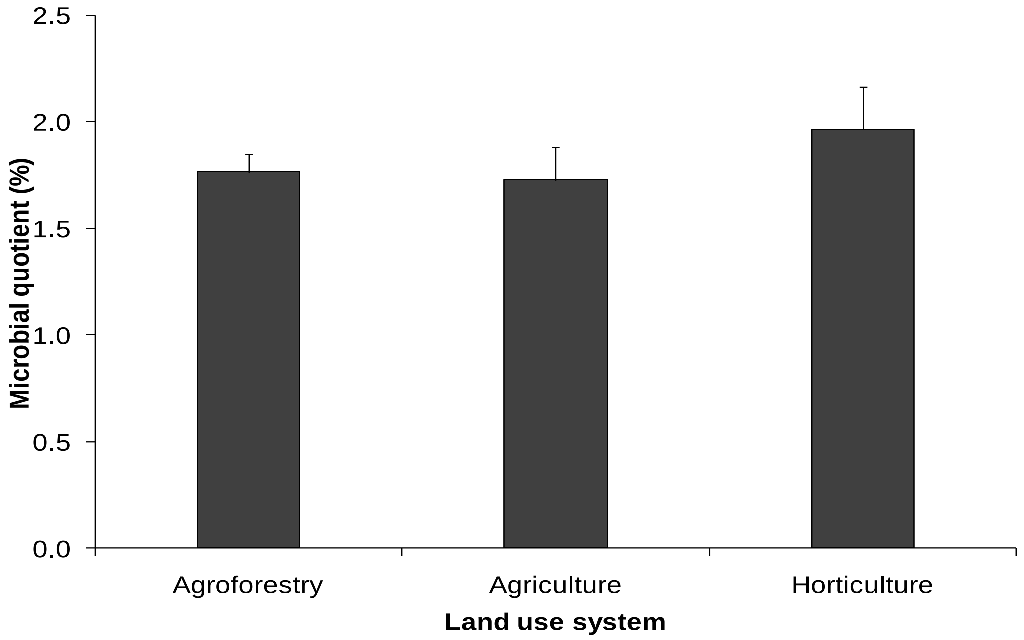


Fig. 5.7 Effect of different main land uses on microbial quotient (%)

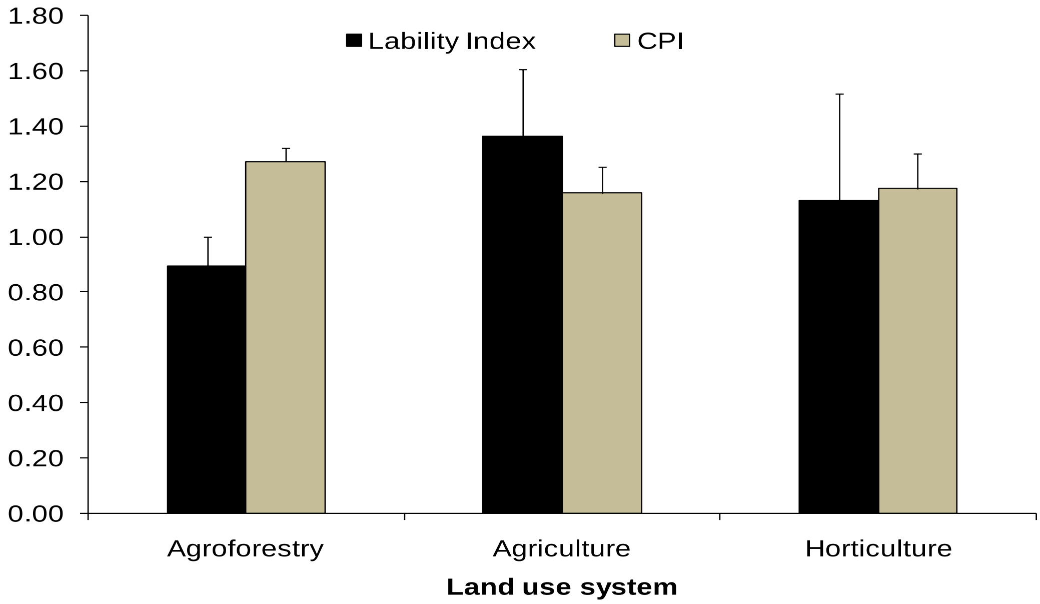


Fig. 5.8 Effect of different main land use systems on lability index (LI) and carbon pool index (CPI)

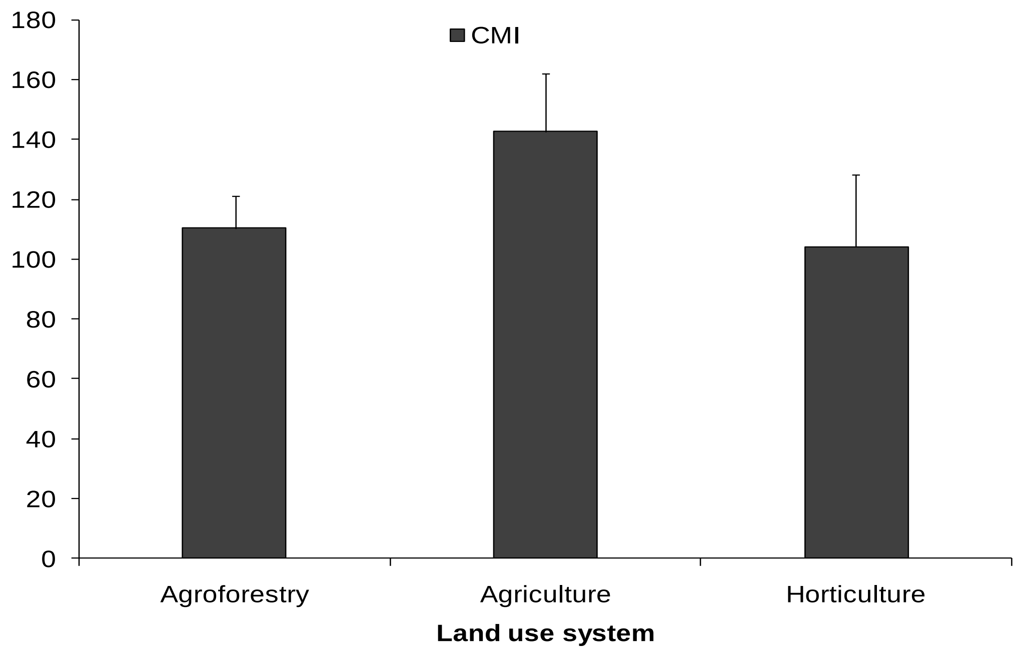


Fig. 5.9 Effect of different main land use systems on carbon management index (CMI)

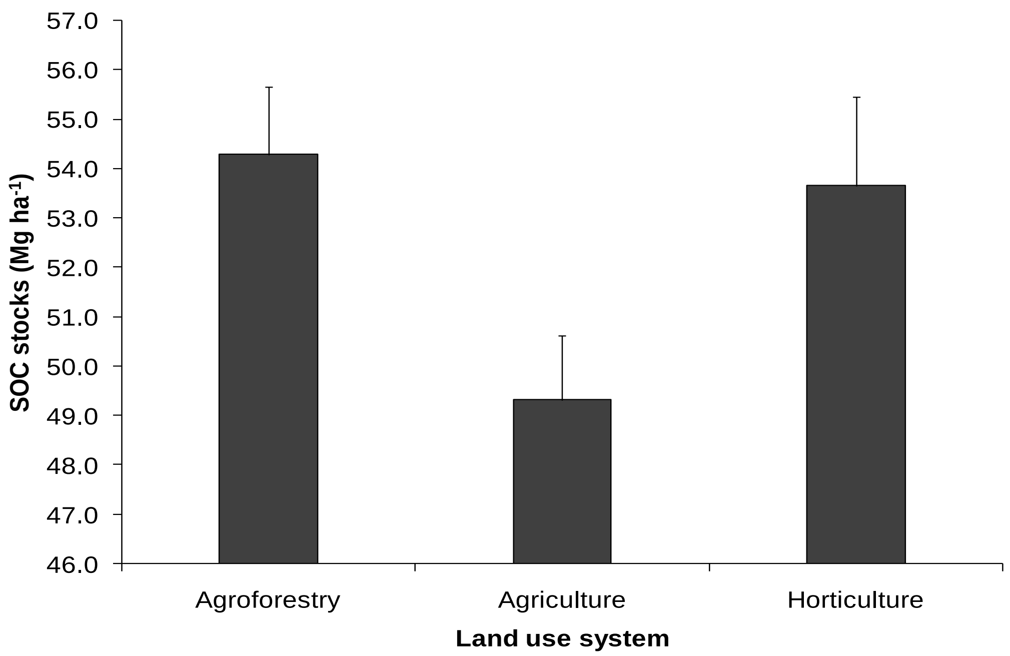


Fig. 5.10 Effect of different main land uses on soil organic carbon stocks (Mg ha^{-1})

CHAPTER VI

Assessment of quality of soil organic matter under different land use systems in East Khasi hills of Meghalaya

6.1 Abstract

Humic substances (HS) represent an important part of soil organic matter (SOM) which plays many roles in this ecosystem. Very little is known about the quality of organic matter in the soils of north-eastern India with varying vegetative cover or land use systems. Bulk soil samples were collected from three major land use systems namely agroforestry, agriculture and horticulture. In each land use, four different tree or crop species (Champak, Tree bean, Alder and Khasi pine in agroforestry land use; Maize, Potato, Rice and Turmeric in agriculture land use; Pear, Peach, Khasi mandarin and Guava in horticulture land use) were selected for this study and compared with control plots (without any tree or crop species). The humic acids (HAs) were characterised by Fourier Transform Infrared Spectroscopy (FTIR), elemental composition and total acidity and functional groups. Humic acids from agroforestry land use showed higher C and N contents (66.7 and 5.53 g kg^{-1}) compared to other two land use systems. Generally, C and N contents of HAs were decreasing in the following order of agroforestry>horticulture> agriculture. The relatively low E4/E6 values (4.86) of HAs in the agroforestry land use system indicate prominence of aromatic components, suggesting that the HAs were of high molecular weight. The relatively high E4/E6 ratios (5.63) of HAs under agriculture land use indicated the dominance of aliphatic components, suggesting that HAs were of low in molecular weight. The total acidity, carboxylic (-COOH) and phenolic (-OH) groups of HAs were found to be significantly higher under agroforestry land use followed by horticulture and agriculture land use. Conversely, adoption of various agroforestry tree species, horticulture fruit trees and agriculture crop species increased the C and N contents, total acidity, carboxylic OH and phenolic OH groups of HAs compared to control plots.

Key words: Land use, FTIR spectra, elemental composition, functional groups, E4/E6 ratio.

6.2 Introduction

The study of SOM is important for its role as a source and reservoir of atmospheric CO_2 and plant nutrients and a main factor linking the physical, chemical and biological properties of soils (Vergnoux et al., 2011). Soil organic matter is composed of different compartments which differ from each other in biochemical composition, biological stability and carbon turnover rates (Marinari et al., 2010). The humified SOM (HA, FA and HU), called as humic substances, represents the most microbially recalcitrant and thus stable reservoir of organic carbon (OC) in soil and an important component for the control of soil degradation (Jastrow et al., 2007; Lugato and Berti, 2008).

Humic substances (HS) belong to the most important class of naturally occurring complex agents, comprising a complex mixture of physically and chemically similar substances that show a large number of oxygen-containing functional groups, particularly -COOH and -OH groups. They are generally divided into three classes of materials on the basis of their alkaline and acid solubility (fulvic acids which are soluble under all pH conditions, humic acids soluble at pH >2.0 and humin insoluble at all pHs). Humin and humic acids generally represent the major fraction of humic substances and appear to display similar analytical characteristics and chemical structure (Schulten and Schnitzer, 1997). HS are formed by secondary synthesis reactions during the decay process and by transformation of biomolecules originating from dead organisms and microbial activity. These compounds are major sinks of refractory organic carbon and their resistance to microbial degradation is attributable to self-associations of the molecules through associations with difficult-to-degrade substances, such as long-chain hydrocarbons in fatty acid and esters, to associations with the soil's mineral colloids and to entrapment in soil aggregates. HS play an important role in soil conservation, for water holding capacity, and for the complexation of metals in terrestrial and aquatic systems (Hayes and Graham, 2000; Hayes and Malcolm, 2001).

The North Eastern region is characterised by diverse agro-climatic and geographical situations. About 54.1 per cent of the total geographical area is under forests, 16.6 per cent under crops, and the rest either under non-agricultural uses or uncultivated land. The low area under agricultural crops is due to natural corollary of the physiographic features of the region, as major chunk of the land has more than 15 per cent slope, undulating topography, highly eroded and degraded soils, and inaccessible terrain. Continuous dilution of the forest cover in the region due to shifting cultivation, firewood, and timber collection is posing the most crucial problem resulting in poor soil health and environmental degradation in the hills. Intensive forestry or agricultural uses without appropriate soil and water conservation measures in northeast Indian soils have often led to severe changes in soil organic matter (SOM) turnover (Patram and Sharma, 2006). This depends on climatic constraints and the impact of local management practices ranging from extensive commercial farming to low-input traditional agriculture. Most soils in northeast India are acidic and weathered with a productivity closely linked to SOM levels. Litter properties and some SOM fractions may act as buffers to seasonal variations and perturbations in nutrient cycles. In addition, the litterfall may protect the soil surface hence promoting high SOM content. Clearance of natural vegetation by shifting cultivation for arable agriculture modifies the amount and quality of SOM, whereas soils that temperature/moisture regimes and biological processes affect litter decomposition and SOM dynamics.

The properties of HS in the environment involve their structural characteristics and composition, which are controlled by the process of humification of organic matter (Rosa et al., 2002) and are dependent on soil type (Baldock et al., 1997), vegetation (Quideau et al., 2001), and climatic conditions (Dai et al., 2002). Humification is defined as the transformation of macro-morphologically identifiable matter into amorphous compounds, as a rule involving the changes that occur in vegetal residues or soil organic matter during the humification process. It has been related to the preferential oxidation of plant polysaccharides, the selective preservation of more recalcitrant organic compounds such as lignin and phenolic structures, and to the incorporation of organic compounds of microbial origin (Zech et al., 1997). The best method for measuring the degree of humification is still being debated because there is no well-defined model of HS structure (Piccolo, 2001). However, it is usually evaluated through indirect measurements that reflect the structural changes which occur during the humification process. Several techniques have been used to characterize the progress of humification, including measurement of the E4/E6 ratio, defined as the ratio of optical absorbance at 465 and 665 nm in aqueous solution, which indicates the structural condensation (Stevenson, 1994); elemental composition; functional groups analysis and C/H, C/O, and C/N ratios, defined as the atomic ratio between elements (carbon, hydrogen, oxygen, and nitrogen; Stevenson, 1994; Rosa et al., 2001).

The effect of land use on soil organic matter and the properties of humic substances have been in a focus of attention of researchers worldwide. Agreeing with early investigators of organic matter in soil, modern authors have also found that its status in soil results from various physical and chemical processes taking place in soil and the character of local plant communities and land uses (Tate, 2001). The dynamics of organic matter in soil, character and intensity of pedogenesis, and effects of agricultural practices on those processes in virgin soils, arable lands, forest soils or degraded soils are being studied by looking closely at the group and fractional composition of humus and characteristics of humus matter (Donisa et al., 2003). The important influence of land use has been acknowledged in a large number of studies worldwide focusing on the effects of various agricultural practices, primarily different soil cultivation, on the composition of humus in soil (Doane et al., 2003). The dominant effect of annual rainfall and vegetation on the composition of HAs has been also reported by Arshad and Schnitzer (1989). Other characteristics of soil humic substances like functional group analysis, spectral characteristics in visible and infrared range (Zhang et al., 1998) and molecular weights (Mukhopadhyay and Banerjee, 1985) are too influenced by cultivation and vegetation type. Nevertheless, there is very little or no information available on the chemical composition of the SOM under different land use systems in north-east India. The present study thus was undertaken to study the effects of different land uses viz. agroforestry, horticulture orchards and agriculture on soil organic matter quality, which could be of immense important in sequestering the soil organic carbon in these areas.

6.3 Materials and methods

6.3.1 Location of the study site

The study was carried out with different multipurpose tree species (MPTs) planted under agri-silvipastural system in 1983, horticulture tree species planted in 1994 and agricultural crops continuously cultivated for the past 10 years at research farm of Indian Council of Agricultural Research (ICAR) Complex for North-East Hill (NEH) Region, Umiam.

Soil samples were collected brought to the laboratory, air-dried at room temperature, ground to pass through 2-mm sieve and used for the analysis of various parameters.

6.3.2 Extraction and characterization of soil organic matter

The soil samples were first equilibrated to a pH value between 1.0 to 2.0 with 1 *M* HCl at room temperature, and adjusted solution volume with 0.1 *M* HCl to provide a final concentration that had ratio of 10 ml liquid/1 g dry sample. The suspension was shaken for 1 hour and the supernatant was separated from the residue by decantation after allowing solution to settle (or by low speed centrifugation). Neutralized the soil residue with 1 *M* NaOH to pH=7.0 and then added 0.1 *N* NaOH under an atmosphere of N_2 to give a final extractant to soil ratio of 10:1. Extracted the suspension under N_2 with intermittent shaking for a minimum of 4 hours and allowed the alkaline suspension to settle overnight and collected the supernatant by means of centrifugation. The supernatant was acidified with 6 *M* HCl with constant stirring to pH=1.0 and then allowed the suspension to stand for 12-16 hours. The humic acid (precipitate) and fulvic acid (supernatant - FA Extract 2) fractions were separated by centrifugation.

The extracted humic acids were purified for removing the impurities in it. This was done by first re-dissolving the humic acid fraction by adding a minimum volume of 0.1 *M* KOH under N_2. The solution was treated with solid KCl to attain 0.3 *M* (K^+) and then centrifuged at high speed to remove suspended solids. The supernatant was acidified with 6 *M* HCl with constant stirring to pH=1.0 so as to re-precipitate the humic acids. The supernatant was separated from the humic acid precipitate by centrifugation. The humic acid precipitate was suspended in 0.1 *M* HCl/0.3 *M* HF solution in a plastic container and was shaken overnight at room temperature. Centrifugation and HCl/HF treatment was repeated, if necessary, until the ash content was below 1 percent. The precipitate was transferred to a Visking dialysis tube by slurring with water and dialyzed against distilled water until the dialysis water gave a negative Cl^- test with the $AgNO_3$. The humic acid was freeze-dried and used for estimation of Carbon and Nitrogen content, functional groups and E4/E6 ratio (Schnitzer, 1982; Stevenson, 1994). Total acidity and carboxylic OH groups were determined by the procedure suggested by Schnitzer and Gupta (1965). The phenolic OH groups (meq/g of humic substances) were calculated from the difference between total acidity and carboxylic OH groups as described below.

The carbon and nitrogen content of the humic acid was determined by the procedure outlined by Chen et al. (1977). The E4/E6 ratios were determined from the absorbances at 465 and 665 nm (E4/E6) after dissolving 2 mg HA or FA in 10 ml 0.05 *N* $NaHCO_3$ (0.01 to 0.05% (wt/vol) of HA or FA) (Chen et al., 1977). Soil pH measurements were done with pH meter; pH's were adjusted with dilute NaOH and HC1 solutions. With HA's, E4/E6 measurements were possible only at pH 7 or higher because these materials were insoluble in distilled water at lower pH levels. This problem, however, was not encountered with the FA and the fractions derived from it, and it was possible to determine E4/E6 ratios over the pH range 1-12. Optical densities of all solutions were measured at

465 and 665 nm on spectrophotometer. The FTIR spectra of HA samples were recorded on a Win-IRrez (Bio-Rad, Hercules, CA, USA) using the potassium bromide (KBr) disc technique. The samples (2 mg) were mixed with potassium bromide (about 100 mg) in a clean glass pestle and mortar and compressed to obtain a pellet. The base line was corrected and scanning was performed from 4000–400 cm^{-1}.

6.3.3 Statistical analysis

Data obtained from this research programme was statistically analysed following standard statistical methods (Gomez and Gomez, 1984).

6.4 Results

6.4.1 FTIR spectra of humic acids

The FTIR spectra of the humic acids extracted from the soils under different land use systems are illustrated in the figures 6.1, 6.2 and 6.3 for agroforestry, agriculture and horticulture land uses, respectively. More or less all the spectra of humic acids showed similarity with respect to absorption peaks irrespective of the land uses. The identification of the absorption bands was done based on data published by Barancikova et al. (1997), Silverstein et al. (1991) and Stevenson (1994). In this study, the most important bands at 3400 cm^{-1} could not be detected which is associated to OH stretch of OH groups and the bands at 2933 cm^{-1} associated to aliphatic C-H stretching in none of the humic acids extracted from each land use systems. On the other hand, all the humic acids extracted from each land uses showed many bands which are characteristics to the original humic acid substances. In the region below 1,800 cm^{-1}, some important differences were observed. In agroforestry land use systems, the FTIR spectra of humic acids extracted from soils under Champak (Fig. 6.1 A-1), Alder (Fig. 6.1 C-1) and control (Fig. 6.1 E-1) plots showed clear bands compared to other agroforestry tree species. Similarly, FTIR spectra of humic acid extracted from soils under potato (Fig. 6.2 B-2) and turmeric (Fig. 6.2 D-2) in agriculture land use; all the plots except FTIR spectra of Khasi mandarin (Fig. 6.3 C-3) showed the characteristics bands of humic acids. In general, the FTIR spectra showed some characteristics and strong absorption bands in the regions from 1700-1725, 1457-1640, 1215-1225, 1014-1030, 719-754 and 517-533 cm^{-1}wavenumber.

6.4.2 Total acidity, functional groups and E4/E6 ratio of humic acids

Table 6.2 shows the total acidity, carboxylic OH, phenolic OH and E4/E6 values as influenced by the various agroforestry trees, horticulture fruit trees and agriculture crop species. It is evident from the statistical analysis that all the crops and tree species had significant effects on carboxylic OH and phenolic OH groups whereas, only agriculture crops showed significant influence on the concentration of total acidity. In general, the total acidity, carboxylic OH and phenolic OH groups followed the order of agroforestry>horticulture>agriculture (Fig. 6.4) whereas, the E4/E6 ratio showed the opposite trend and followed the order: agriculture>horticulture>agroforestry (Fig. 6.6). Total acidity ranged from 9.01 (control) to 11.0 (Khasi pine) with an average of 9.77 meq g^{-1} of humic acids in agroforestry land use; 5.2 (Potato) to 8.40 (Turmeric) with an average of 6.6 meq g^{-1} of humic acids in agriculture land use and from 7.4 (Khasi mandarin) to 9.40 (Pear) with an average of 8.18 meq g^{-1} of humic acids under horticulture land use. With respect to carboxylic OH groups, Khasi pine, Turmeric and Pear recorded the maximum values of 6.8, 4.8 and 6.2 meq g^{-1} of humic acids, respectively while Alder (5.2), Potato (2.0) and Guava (3.2) recorded the lowest values corresponding to agroforestry, agriculture and horticulture land uses. The phenolic OH groups were ranged from 3.2-4.2, 2.8-3.6 and 3.0-4.6 meq g^{-1} of humic acids under agroforestry, agriculture and horticulture land use systems, respectively. Overall, agroforestry tree species showed higher total acidity, carboxylic OH and phenolic OH concentration in humic acids by 48, 74 and 19%, respectively compared to agriculture crop species.

The E4/E6 ratios of the HAs from bulk soil samples from various land use systems ranged from 4.41-5.22 under agroforestry land use, 4.89-6.37 under agriculture land use and 4.43-5.52 under horticulture land use (Table 6.2), with significant variations among the various tree and crop species. For HAs from agroforestry land use, E4/E6 ratios were highest for Champak (5.22); Rice (6.37) for agriculture land use and Pear (5.52) for horticulture land use. In contrast to the C/N ratio, the E4/E6 ratios in HAs from the control plots, on an average basis, were lower compared to the E4/E6 ratios in humic acids from soils under tree and crop species. The mean E4/E6 ratio of humic acids from agriculture land use was 14% higher than the mean E4/E6 ratio of humic acids from agroforestry land use (Fig. 6.6).

6.4.3 Elemental composition and C/N ratio of humic acids

The effects of different land use systems on the elemental composition and C/N ratio of humic acids extracted from soils under various tree and crop species are given in the table 6.2. It is clear from the table that conversion of fallow lands to agroforestry, agriculture and horticulture land uses increased the concentration of both carbon and nitrogen in the humic acids regardless of the tree and crop species. None of the selected land use systems for this study showed significant changes in the humic acid carbon content; however, all the tree and crop species significantly influenced the nitrogen content of humic acids. The humic acid C content ranged from 61.6 (Tree bean) to 64.0 (Khasi pine) with an average of 62.73 g $100g^{-1}$ in agroforestry land use; 55.5 (Rice) to 59.7 (Turmeric) with an average of 57.16 g $100g^{-1}$ under agriculture land use and 58.3 to 63.0 (Guava) with an average of 60.68 g $100g^{-1}$ under horticulture land use (Table 6.2 and Fig. 6.5). Similar to carbon, nitrogen content was also highest under agroforestry land use (5.53 g $100g^{-1}$) followed by horticulture land use (4.92 g $100g^{-1}$) and the lowest was under agriculture (4.13 g $100g^{-1}$) land use (Fig. 6.5). Amongst the tree and crop species, Alder (6.32 g $100g^{-1}$) under agroforestry land use, maize (4.51 g $100g^{-1}$) under agriculture land use and peach (5.48 g $100g^{-1}$) under horticulture land use showed the maximum nitrogen content in humic acids (table 6.2). In all the three land use systems, control plots recorded the lowest content of humic acid carbon and nitrogen. On the whole, conversion of fallow lands to agroforestry, agriculture and horticulture land uses did not increase the C content of humic acids considerably however, it increased the humic acids N content by 17, 10 and 15%, respectively. On an average, adoption of agroforestry tree species increased the C and N contents in humic acids by 10 and 34%, respectively compared to agriculture land use (Fig. 6.5). In contrary to both C and N content, the C/N ratio of humic acid was maximum under agriculture land use (13.86) followed by horticulture (12.39) and the lowest ratio was found in humic acids under agroforestry land use (11.44). In agroforestry land use, it ranged from 9.86 (Alder) to 12.71 (Control); from 12.71(Maize) to 14.8 (Control) in agriculture land use and from 11.2 (Peach) to 13.68 (Control) in horticulture land use. It is clear from the results that control plots under all the land uses recorded the maximum C/N ratios of humic acids.

6.5 Discussion

Humic substances (HS) from environmental compartments can strongly differ in their chemical and physical characteristics, as a result of the diversified humification conditions. In soils, the structure and composition of humic substances seem to be influenced, among other parameters, by parent material, soil pH, vegetation, soil management system and cultivation. Also the soil type, including its mineralogy, which in turn, is related to soil age and climate, can affect the quality of humic substances.

6.5.1 FT-IR spectra of humic acids

The FT-IR spectra showed features typically attributed to humic acids and demonstrated the uniformity of the extracted materials (Figures 6.1, 6.2 and 6.3). Irrespective of their origin all humic acids present more or less similar infrared spectra. In fact, those polar chemical functionalities absorbing infrared radiation such as phenolic and alcoholic hydroxyls, aromatic and aliphatic carboxyls and carbonyls, aliphatic C-H and amides, might be present in all the studied humic substances, regardless of their origin. Such interpretation is in line with the concepts attributed to humus mixtures, according to which they might consist of associations of molecules, with a “universal average formula unit”, held together via supramolecular interactions (Cozzolino et al., 2001). The relative contents of these groups, on the other hand, can differ from one sample to another but in our study we could not calculate the concentration of these functional groups from the FTIR spectra; instead we calculated the same by potentiometric titration and are discussed in the chapter 6.5.2.

In this study, the absorption peaks at 2924 and 2852 cm^{-1} could not be observed which are attributed, respectively, to the asymmetrical and symmetrical stretching of methylene ($-CH_2-$) groups this being the characteristics of aliphatic and non-strained cyclic hydrocarbons (Silverstein et al., 1991). This may indicate the high aromatic characteristics of the humic acids extracted from all the tree and crop species in this study. The different characteristics observed in the area below 1,800 cm^{-1} of the spectra of FTIR were concentrated in the 1,720 to 1,705 cm^{-1} region, attributed to the stretching of C=O of carboxylic acids, aldehydes and ketones, whose relative intensities are strong for the humic acids extracted from soils under Champak, Alder and control plot in agroforestry land use, potato in agriculture land use and pear, peach, guava and control plot under horticulture land use. The bands in this range of the infrared spectrum have several assignments including aromatic C=C stretching, amide group ($RCONH_2$) C=O stretching and amide -NH bending. An absorption band between 1,661 and 1,628 cm^{-1} attributed to the stretching C = O of groups amide (band I of the amide), and also C=O of quinones and H tied to conjugated ketones which are mostly seen Alder and control plots under agroforestry land use, potato and turmeric under agriculture land use and pear, peach, guava and control plots under horticulture land use; a fourth peak between 1,614 and 1,612 cm^{-1}, generally attributed to the stretching of aromatic C=C and symmetrical stretching of the anion COO^-, that are only present in the humic acids of the soils under Alder and control plots, potato, and pear and control plots in agroforestry, agriculture and horticulture land uses, respectively. Carbon-carbon stretching usually appears at about 1650 cm^{-1} for doubled bonds, but is shifted to about 1600 cm^{-1} by conjugation. Substituted or unsubstituted amides, on the other hand, show the C=O band in the 1640–1690 cm^{-1} region and the -NH bending at 1600–1640 cm^{-1} (Marinari et al., 2010). All these bands are produced by aromatic C=C, C=O in amide (I), ketone and quinone groups of the humic acids (Amir et al., 2004). Absorption bands in 1540-1550 cm^{-1}, attributed to the N-H deformation of the amide (band II of the amide) and stretching of C=N just observed in the spectra of the humic acids of the Alder, potato, pear, peach and control plots; a band in the region from 1,354 to 1,327 cm^{-1}, attributed to the symmetric deformation of CH_3, just observed in the spectra of the humic acids from potato field. The strong absorption in 1015-1037 cm^{-1} in Alder, potato, turmeric, guava and control plots represents C-O of alcohol and polysaccharide (Adekunle et al., 2007); absorption in the region from 762 to 669 cm^{-1}, attributed to the angular deformation out of the plan of connections C-H of aromatic rings, observed in the spectra of the Khasi pine, potato, pear, peach and control plots. In general, the data obtained through the FTIR spectroscopy, suggest a mixture of aromatic/aliphatic characteristics, a great amount of carboxylic groups and more numbers of n-containing groups for the humic acids of the studied land use systems. In humic acids, all these types of vibrations are expected to exist and their corresponding frequency signals might be overlapped in this region, the shifts to the right or left might therefore be results of the relative influence of each group (Cunha et al., 2009; Palanivel et al., 2012).

6.5.2 Total acidity, functional groups and E4/E6 ratio of humic acids

The tendency of HA to condition soil depends partly on the presence of oxygen containing functional groups such as carboxylic COOH and phenolic OH (Gondar et al., 2005). The carboxylic COOH and phenolic OH values (Table 6.2) were found to be 3.2-6.8 and 2.8-4.6 meq g^{-1} humic acids, respectively. These values were within the ranges reported for carboxylic COOH and phenolic OH (Tan, 2003; Ch'ng et al., 2011). In this study, carboxylic OH groups contributed 62, 52 and 56% of total acidity in agroforestry, agriculture and horticulture land use systems, respectively. These results are in agreement with the findings of Plaza et al. (2006) who have reported in their study that about

51-64% of total acidity of soil humic acids was contributed by carboxylic OH groups. The cation exchange capacity of humic acids is represented by the total acidity (summation of carboxylic COOH and Phenolic OH values) of the humic acids. The total acidity value (Table 6.2) of humic acids was found to be in the range of 5.2-11 meq g^{-1} humic acids, a value that was found to be within the range reported by other workers (Tan, 2003; Ch'ng et al., 2011). The high total acidity in agroforestry land uses was a consequent of high content of carboxylic OH groups in the humic acids as evident from the table 6.2.

The ratio of optical or absorbances of dilute aqueous humic acid (HA) solution at 465 and 665 nm is widely used by soil scientists for the characterization of humic materials. This ratio, usually referred to as E4/E6, reportedly is independent of concentrations of humic materials but varies for humic materials extracted from different soil types (Kononova, 1966; Schnitzer and Khan, 1972). Kononova (1966) believed that the magnitude of the E4/E6 ratio is related to the degree of condensation of the aromatic C network, with a low ratio indicative of a relatively high degree of condensation of aromatic humic constituents. Conversely, a high E4/E6 ratio reflects a low degree of aromatic condensation and infers the presence of relatively large proportions of aliphatic structures. Other workers have suggested (Schnitzer and Khan, 1972) that light absorption of aqueous HA solution in the visible region of the electromagnetic spectrum increases with: (i) the ratio of Carbon in aromatic *nuclei* to C in aliphatic side chains; (ii) the total C content; and (iii) the molecular weight. All of these factors are likely to affect the magnitude of E4/E6 ratios.

The data in the table 6.2 shows that humic acids under agroforestry land use showed lower E4/E6 ratios than agriculture and horticulture land uses thus confirms that humic acids under agroforestry tree species are associated with a relatively large molecular size or high molecular weight (i.e. a large molecule) with high degree of aromatic condensation. This molecule has high carbon content, but relatively low in oxygen (not measured), COOH groups, and total acidity. A high E4/E6 ratio, by contrast, in agriculture land use indicates a smaller molecule which contains less carbon but more oxygen, COOH groups, and total acidity. As reported by Hanmin et al. (2012), the humic acids with high E4/E6 ratios in agriculture land use implies lower degree of aromatic condensation and the presence of a relatively large proportion of aliphatic structure (Rivero et al., 2004; Campitelli et al., 2006; Palanivel et al., 2012) and this observation is consistent with a previous study (Amir et al., 2004; Wei et al., 2007; Palanivel et al., 2012). It is frequently suggested that the magnitude of the E4/E6 ratio of humic substances is related to the relative concentration of condensed aromatic rings in these materials. A low E4/E6 ratio supposedly indicates a relatively high concentration of condensed ring structures and the presence of relatively large proportion of aliphatic structures and for a high ratio the reverse is the case.

6.5.3 Elemental composition and C/N ratio of humic acids

A review of the factors controlling organic matter dynamics in tropical soils has been reported by Zech et al. (1997). They claimed that changes in organic matter during the process of decomposition are affected by the dominant site-specific characteristics like temperature, soil moisture content, pH and nutrient levels. Our results also indicated an influence of land use on the supposed stable humified pool of soil organic matter. The C and N contents and the C/N ratios of humic acids (Table 6.2) were within the ranges reported for other soils (Stevenson, 1994; Spaccini et al., 2006; Pospisilova et al., 2008) and varied substantially from agroforestry to agriculture soils except humic acid C content due to large variations in the degree of humification among the tree and crop species selected for this study. With cultivation, both C and N contents of humic acids in agriculture lands decreased due to oxidative degradation of C and N compounds upon cultivation. The general constancy of C/N values for the different humic fractions suggests a resistance to microbial decomposition attributable to their recalcitrant nature deriving from either physical protection in microaggregates (Dutarte et al., 1993) or chemical protection by their hydrophobic composition (Spaccini et al., 2000). The data indicated that higher C and N values were recorded for agroforestry land use than for the humic acids extracted from agriculture land use. The C/N atomic ratios of humic acids extracted from soils under agriculture land use were higher than rest of the land uses indicating a higher degree of humification (Guggenberger et al., 1995) and/or a greater microbial contribution for agroforestry land use. A similar trend has also been observed for whole organic matter from forest soils (Nierop et al., 2001; Jolivet et al., 2001). The decomposition of the organic material in soil tends to lead to the formation of phenolic structures deriving from lignin. These structures are hardly decomposed with respect to sugars and proteins (Rosa et al., 2005). Therefore, the C/H, C/N, and C/O atomic ratios have been used as important indicators of the aromaticity and level of organic material decomposition although we could not estimate the C/H and C/O ratios of humic acids in our study. According to the literature (Stevenson, 1994), higher value of C/N atomic ratios are associated with higher degrees of humification due to decreased acid, carbohydrate, and amino acid/protein content.

6.6 Conclusions

The spectra of all humic acids from each tree and crop species were similar because there was no significant difference in the quantities of C in humic acids regardless of the land use systems with different trees and crops of varying ages and were mainly dominated by aromatic components thus indicating the high maturity of HAs. In general, the FTIR spectra showed some distinct characteristics and strong absorption bands in the regions from 1700-1725, 1457-1640, 1215-1225, 1014-1030, 719-754 and 517-533 cm^{-1}wavenumber.

The total acidity which is a measure of the cation exchange capacity of the humic acids was relatively higher in agroforestry land use followed by horticulture and agriculture land use. The carboxylic (-COOH) and phenolic (-OH) groups of humic acids of all the land use systems were found to be consistent with the range reported by other researchers and followed the same trend of total acidity. The relatively higher E4/E6 values of HAs under agriculture land use systems indicated the prominence of aliphatic components and the HAs were of low molecular weight. Conversely, the lower E4/E6 ratios of the humic acids under agroforestry land use systems indicated the dominance of aromatic components and high molecular weight humic acids.

The average carbon and nitrogen contents were relatively higher under agroforestry land use and decreased upon cultivation in the agriculture land use. On the other hand, the C/N ratios of humic acids were higher under agriculture land use. Compared to trees and crop species, all the control plots showed lower values of humic acids C, N, total acidity, carboxylic and phenolic OH groups; higher values of E4/E6 and C/N ratios. On the whole, it can be concluded that adoption various agroforestry, horticulture fruit tree and agriculture crops species in the fallow lands increased the degree of humification thus resulted in the higher aromatic condensation of humic acids and increased stabilization or quality of soil organic matter. Amongst the three land use systems, agroforestry land use showed higher aromatic condensation and increased stabilization or quality of soil organic matter whereas, agriculture land use showed lower aromatic condensation and thus lower stabilization or quality of soil organic matter.

Table 6.1 Initial soil properties of different land uses selected for the study

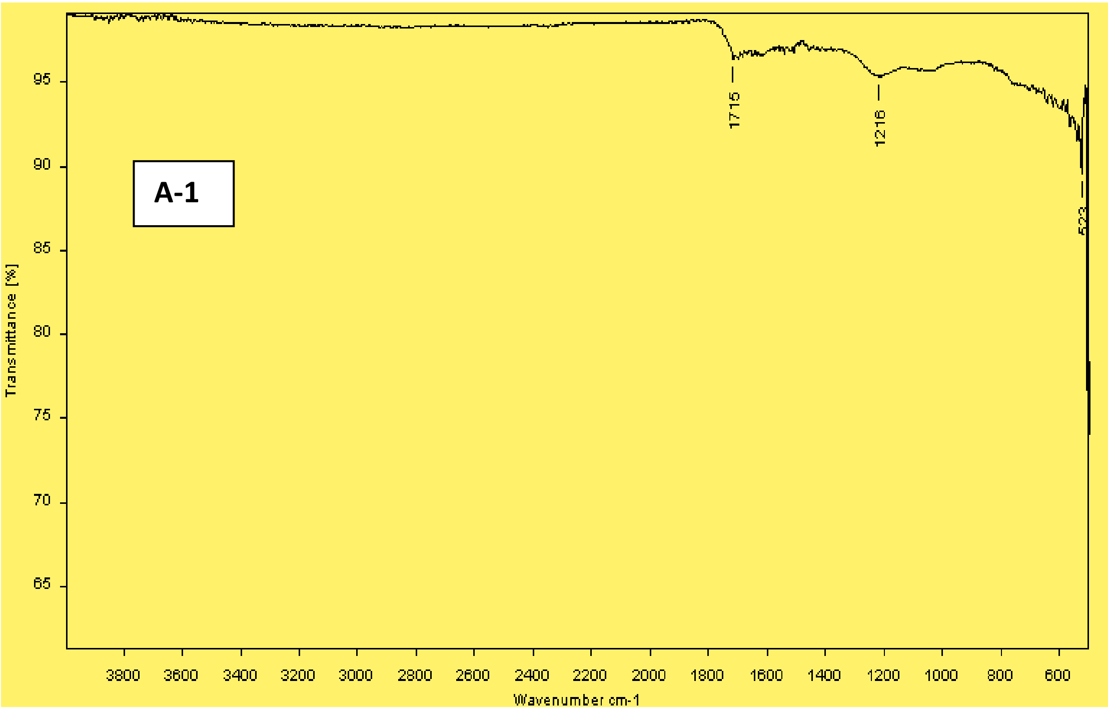
Land use system	Soil properties						
	pH	BD (Mg m^{-3})	MWD (mm)	Moisture (g $100g^{-1}$)	N	P	K
					(kg ha^{-1})		
			Agroforestry				
Champak	4.60	1.11	31.0	31.0	522.3	32.1	313.6
Tree bean	4.59	1.15	26.1	26.1	496.3	30.1	286.7
Alder	4.38	1.07	31.8	31.8	584.3	47.2	361.2
Khasi Pine	4.36	1.05	33.8	33.8	464.8	23.7	276.0
Control (No tree)	4.76	1.18	28.4	28.4	403.4	19.3	248.4
			Agriculture				
Maize	5.12	1.27	29.9	29.9	394.4	49.1	292.7
Potato	5.24	1.31	28.9	28.9	414.6	65.6	365.1
Rice	5.09	1.33	32.2	32.2	421.4	23.5	322.2
Turmeric	5.29	1.25	26.7	26.7	398.9	36.9	306.1
Control (No crop)	4.81	1.33	24.8	24.8	387.7	20.7	203.1
			Horticulture				
Pear	5.71	1.28	31.6	31.6	437.1	43.2	323.6
Peach	6.24	1.23	32.7	32.7	455.1	39.1	495.2
K Mandarin	5.75	1.29	28.7	28.7	428.1	45.3	419.6
Guava	5.43	1.19	29.9	29.9	446.1	24.6	343.5
Control (No tree)	4.82	1.30	27.1	27.1	434.9	26.9	328.5

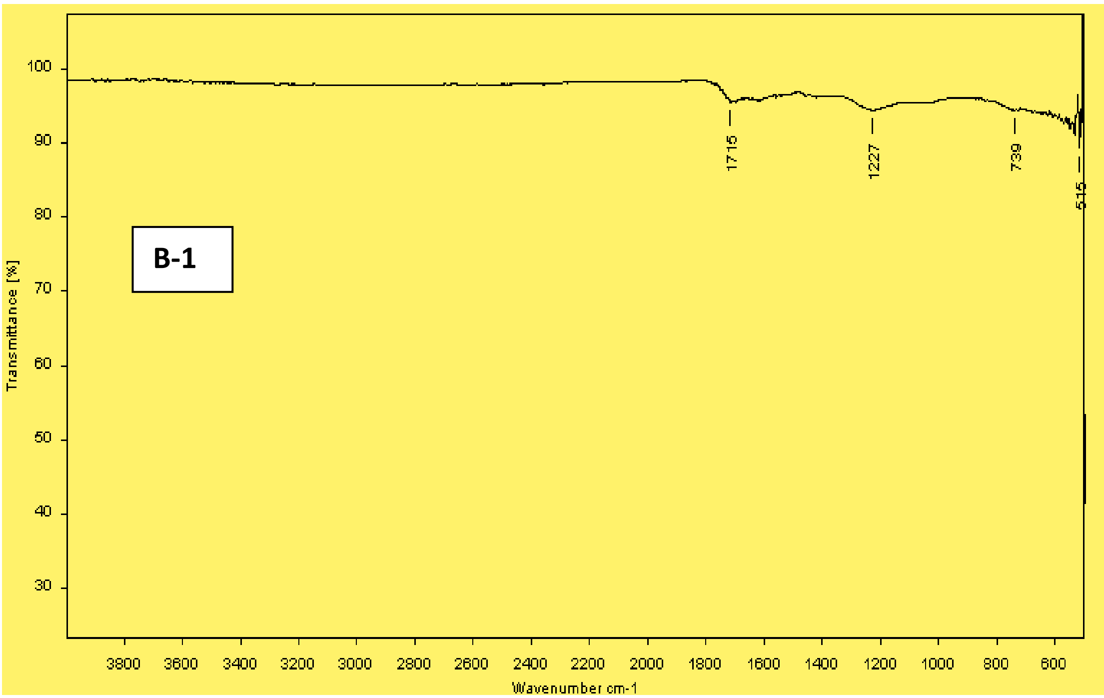
Table 6.2 Properties of humic acids extracted from different land uses

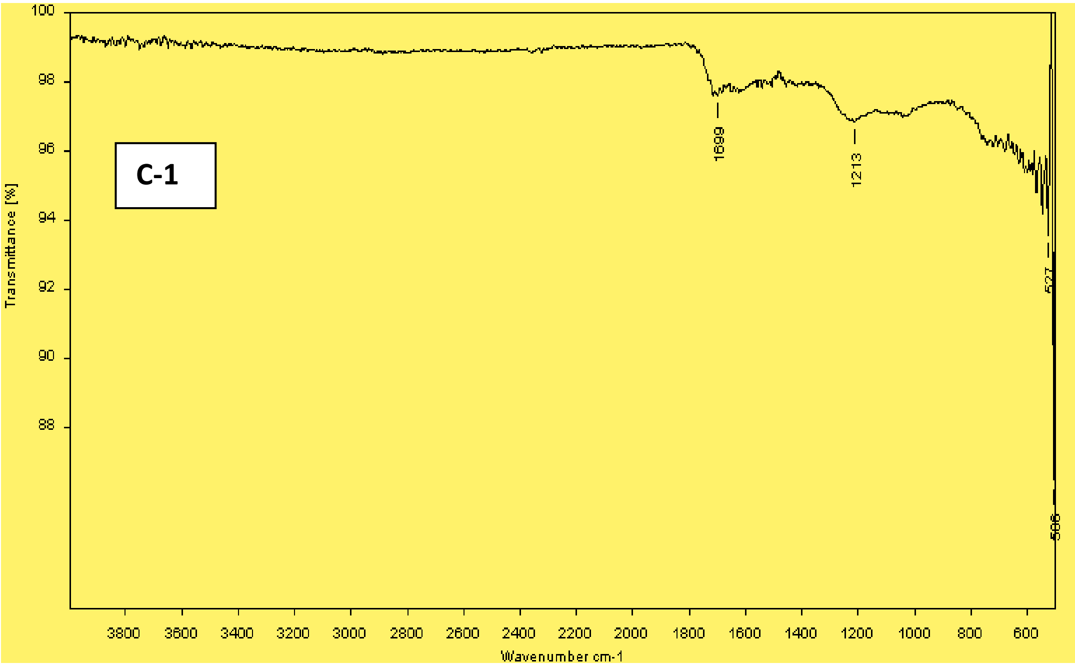
Land use system	Total acidity	Carboxylic OH	Phenolic OH	C	N	C/N ratio	E4/E6 ratio
	(meq g^{-1} of HA*)			(g $100g^{-1}$ of HA*)			
				Agroforestry			
Champak	10.0a	6.30a	3.70bc	63.9a	5.94a	10.76ab	5.22a
Tree bean	9.50a	6.20ab	3.30cd	61.6a	5.43b	11.34ab	5.13a
Alder	9.30a	5.20b	4.10ab	62.3a	6.32a	9.86b	4.76b
Khasi Pine	11.0a	6.80a	4.20a	64.0a	5.10bc	12.55a	4.80b
Control (No tree)	9.01a	5.80ab	3.20d	61.9a	4.87c	12.71a	4.41c
				Agriculture			
Maize	6.80b	4.00b	2.80c	57.3a	4.51a	12.71a	5.02c
Potato	5.20c	2.00d	3.20b	56.0a	4.11bc	13.63a	6.07ab
Rice	6.60bc	3.40bc	3.20b	55.5a	3.93cd	14.12a	6.37a
Turmeric	8.40a	4.80a	3.60a	59.7a	4.27b	13.98a	4.89c
Control (No crop)	6.00bc	3.20c	2.80c	56.8a	3.82d	14.87a	5.81b
				Horticulture			
Pear	9.40a	6.20a	3.20b	60.8a	4.94ab	12.31a	5.52a
Peach	8.00ab	4.80b	3.00b	61.4a	5.48a	11.20a	5.31a
K Mandarin	7.40b	3.80c	3.60b	58.3a	4.76b	12.25a	5.26a
Guava	7.80b	3.20c	4.60a	63.0a	5.04ab	12.50a	4.88ab
Control (No tree)	8.30ab	5.10b	3.20b	59.9a	4.38c	13.68a	4.43b

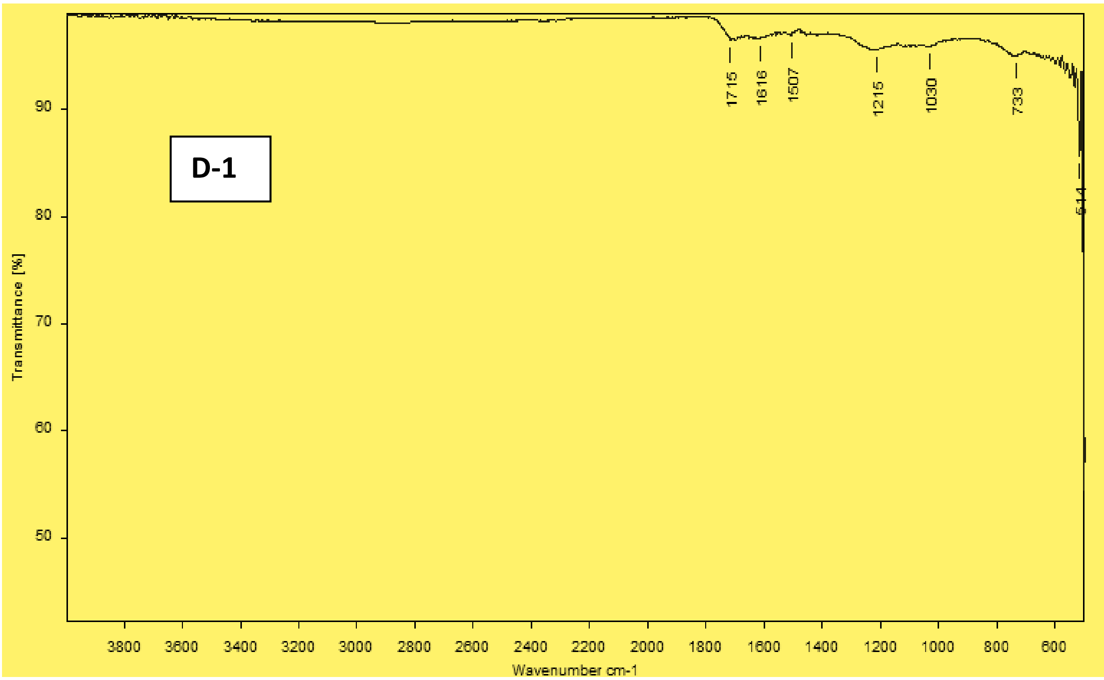
* Humic acids

Fig. 6.1 FTIR spectra of humic acids extracted from the soils under champak (A-1), tree bean (B-1), alder (C-1), Khasi pine (D-1) and control site (E-1) in agroforestry land use systems






C-1
Transmittance [%]
100
98
96
94
92
90
88
1699
1213
527
506
3800 3600 3400 3200 3000 2800 2600 2400 2200 2000 1800 1600 1400 1200 1000 800 600
Wavenumber cm-1


D-1
Transmittance [%]
90
80
70
60
50
1715
1616
1507
1215
1030
733
614
3800 3600 3400 3200 3000 2800 2600 2400 2200 2000 1800 1600 1400 1200 1000 800 600
Wavenumber cm-1

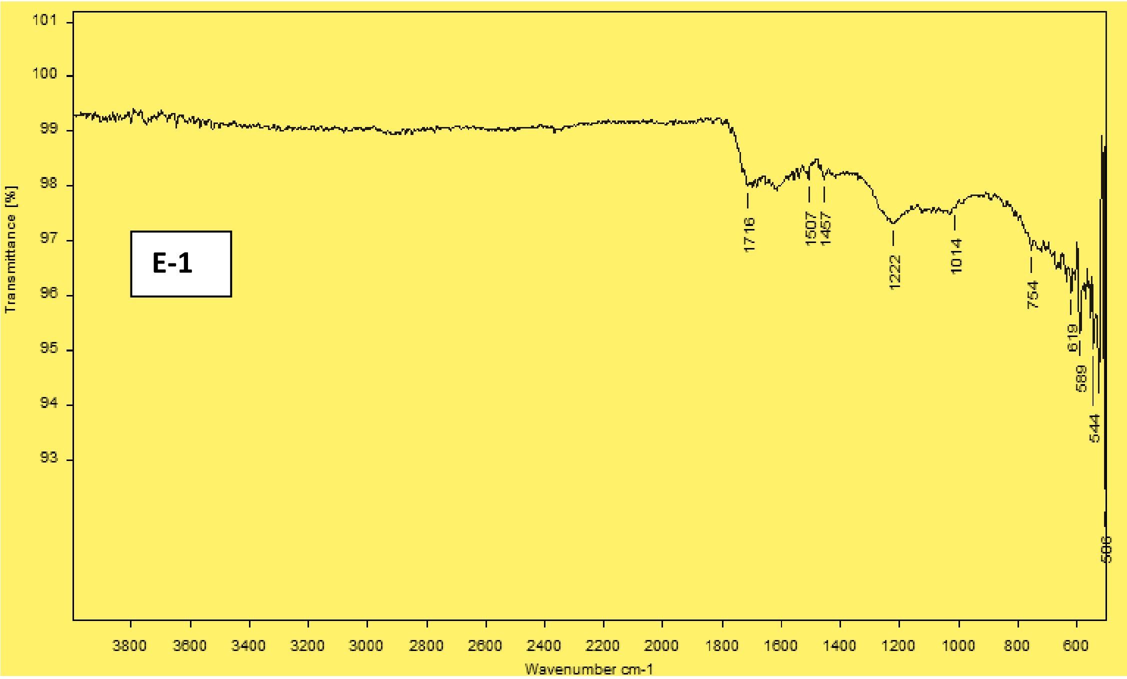
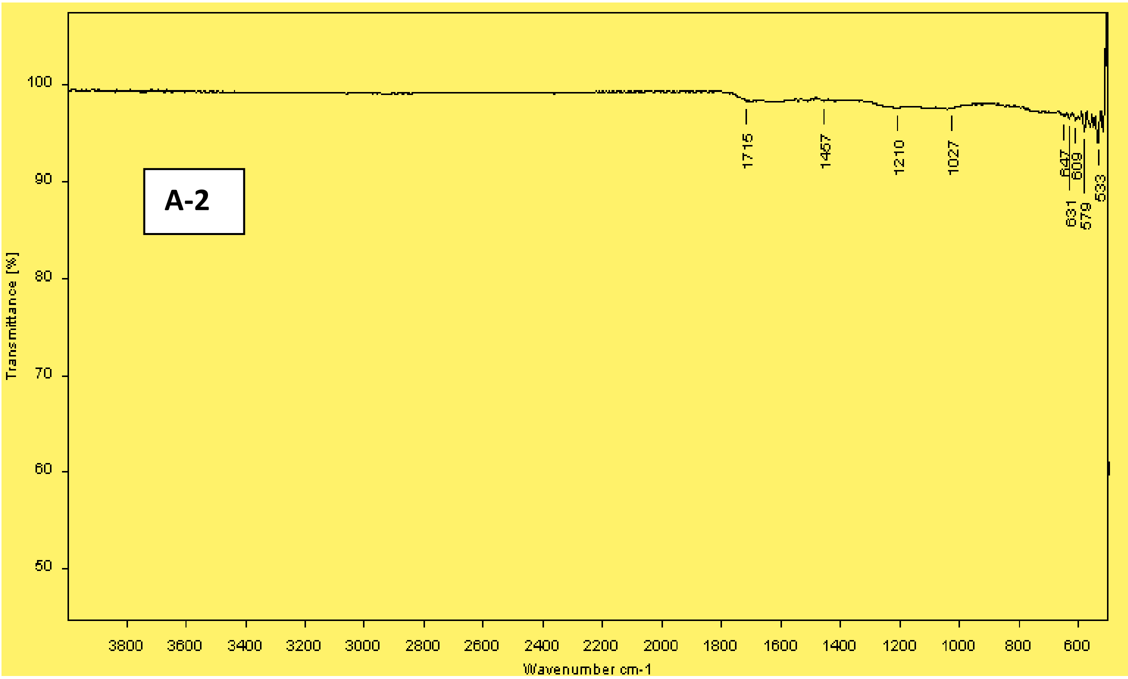
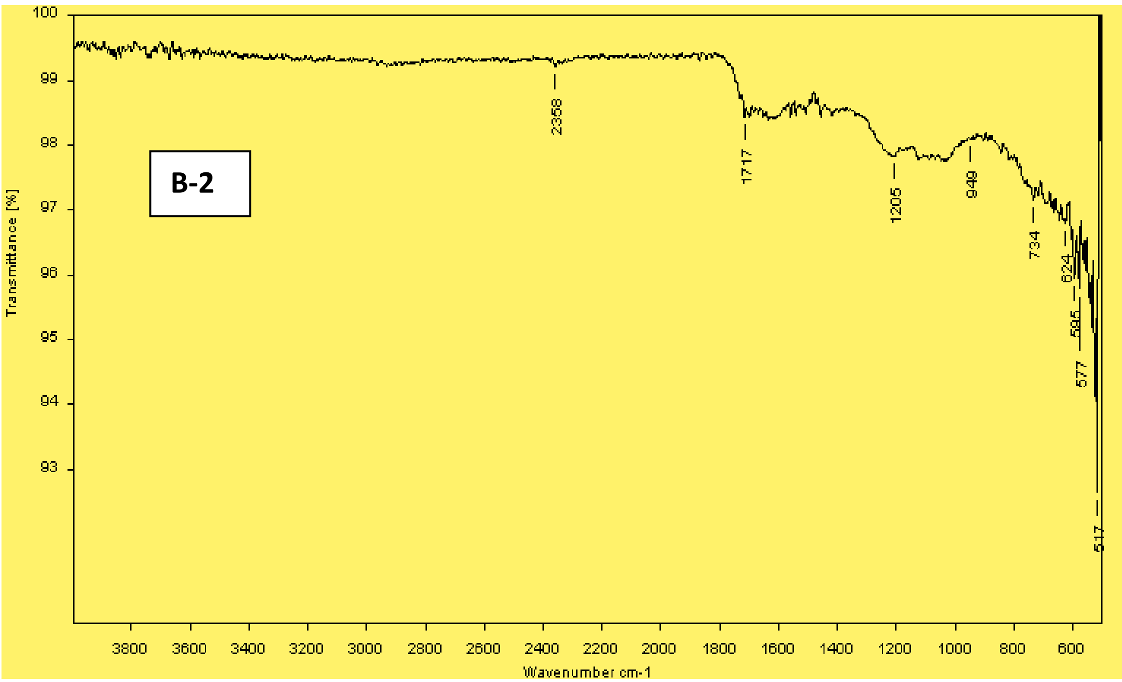
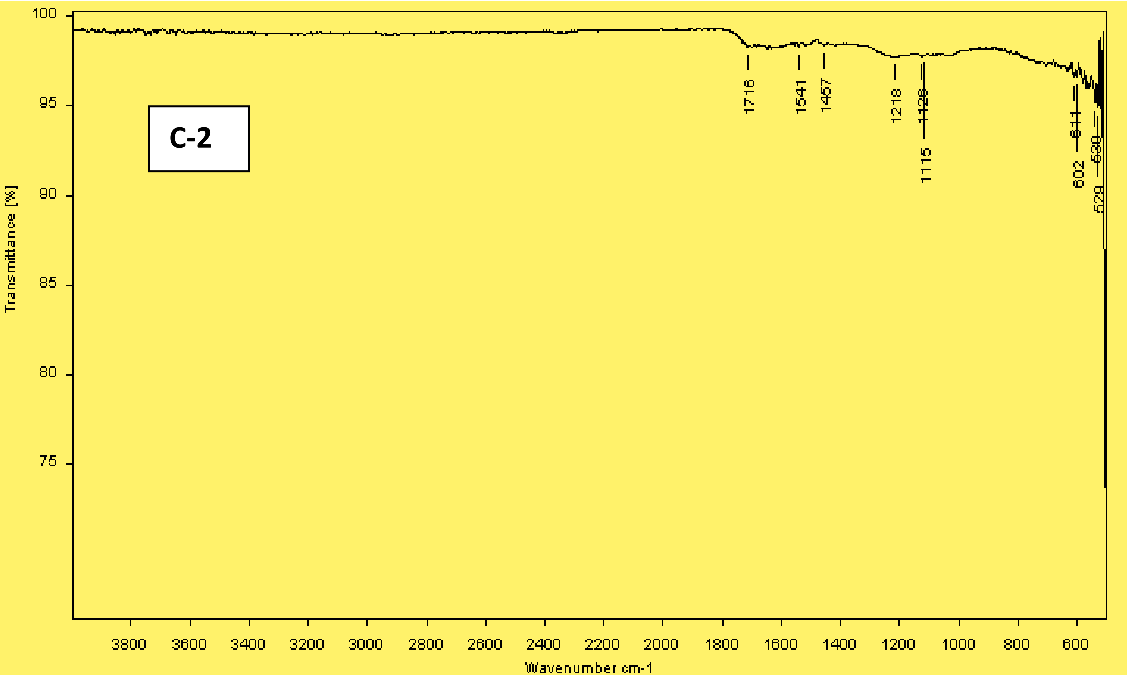
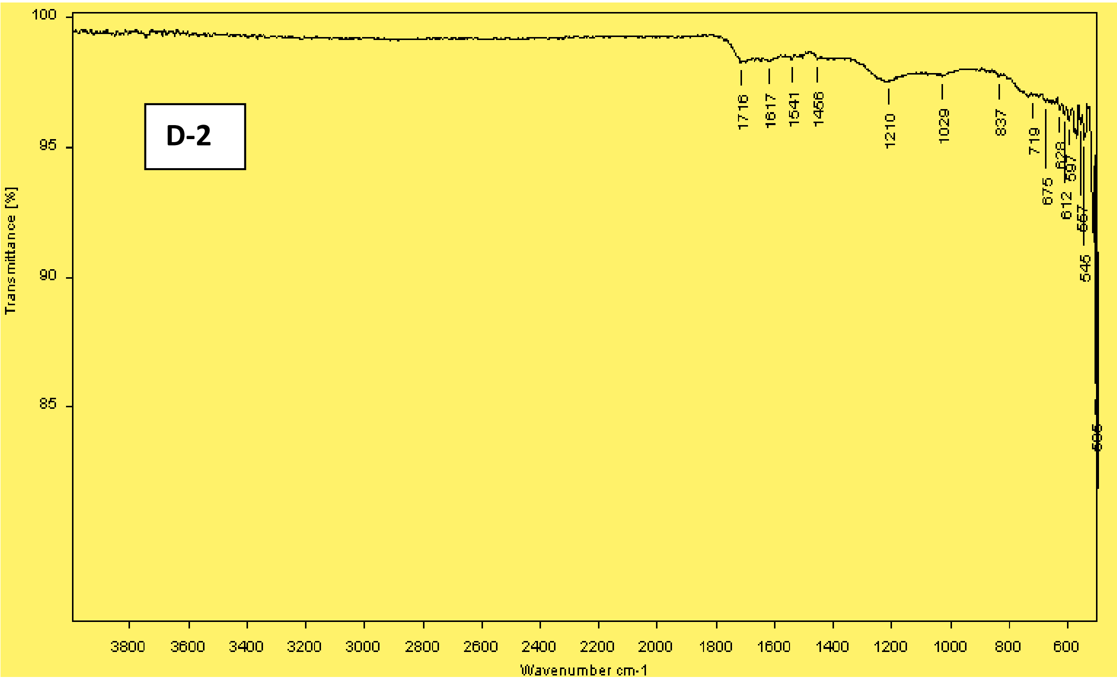

E-1
Transmittance [%]
Wavenumber cm-1
1716
1507
1457
1222
1014
754
619
589
544
506

Fig. 6.2 FTIR spectra of humic acids extracted from the soils under maize (A-2), potato (B-2), rice (C-2), turmeric (D-2) and control site (E-2) in agriculture land use systems






C-2
Transmittance [%]
Wavenumber cm-1
1716
1541
1457
1218
1128
1115
611
602
539
529


D-2
Transmittance [%]
Wavenumber cm-1
1716
1617
1541
1456
1210
1029
837
719
675
628
612
597
567
545
505

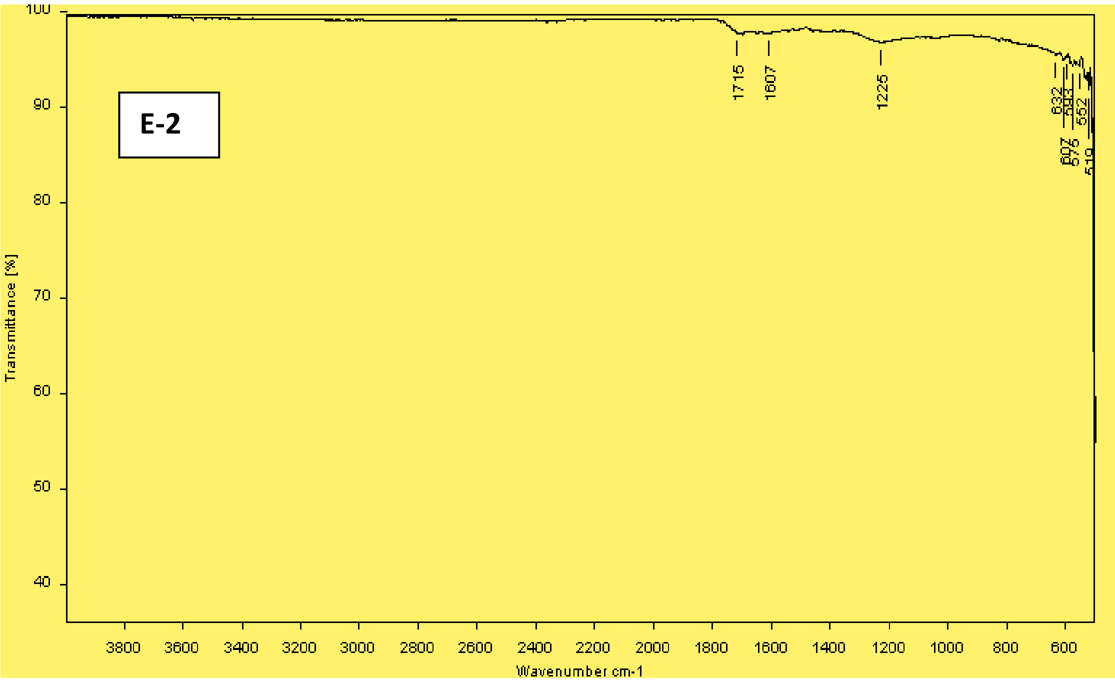
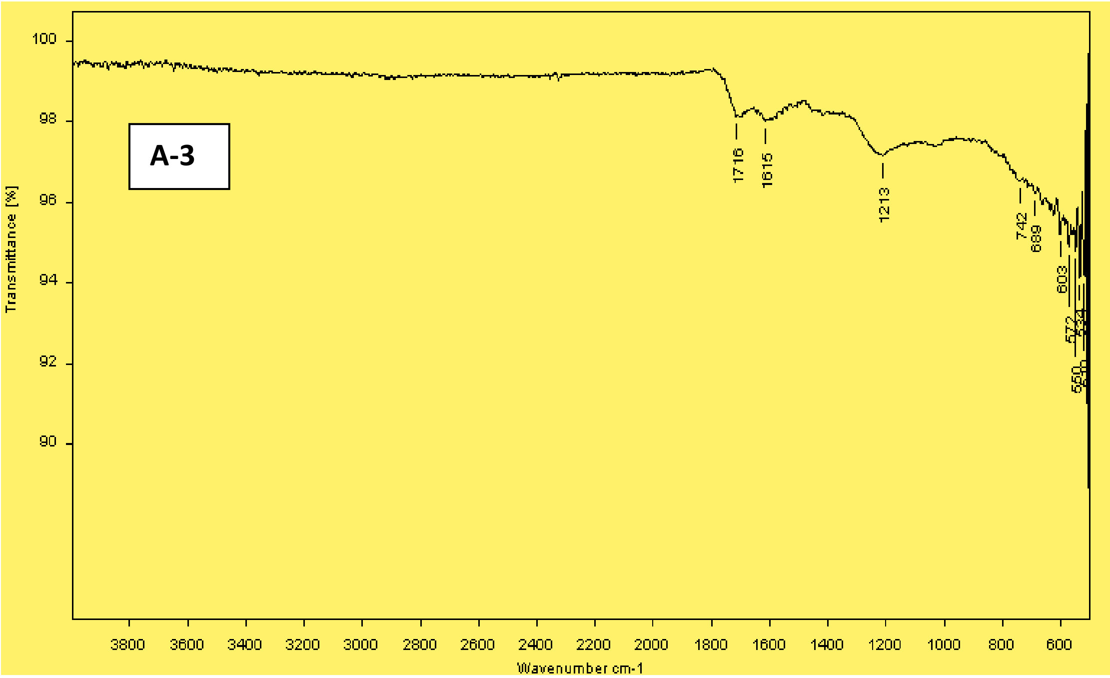
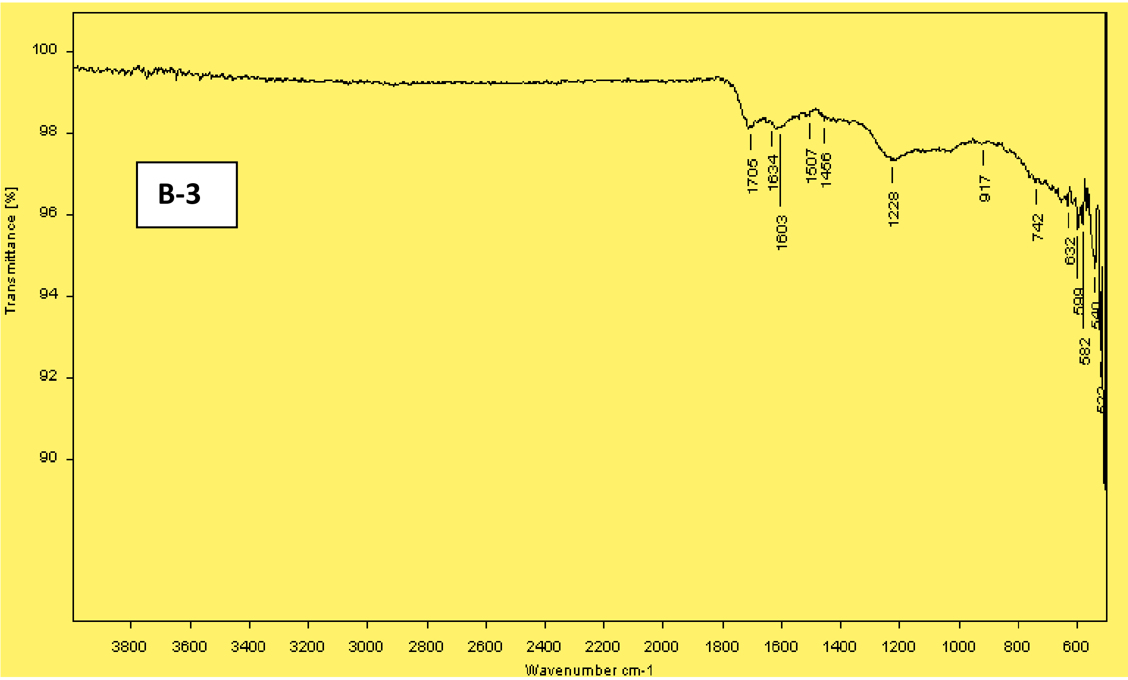
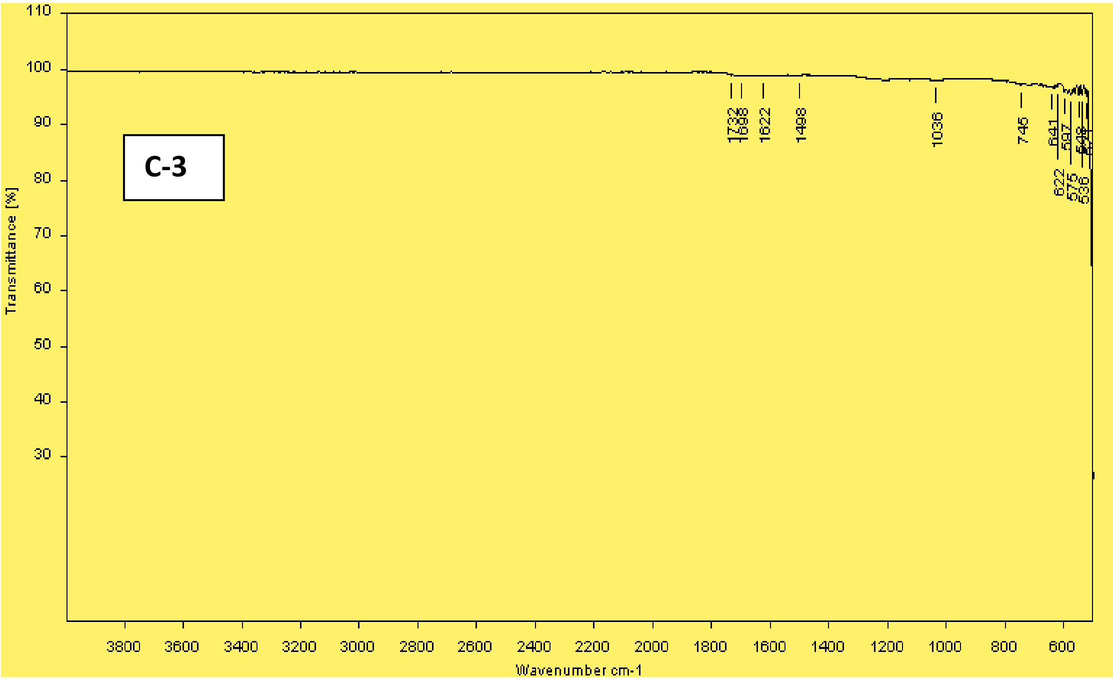
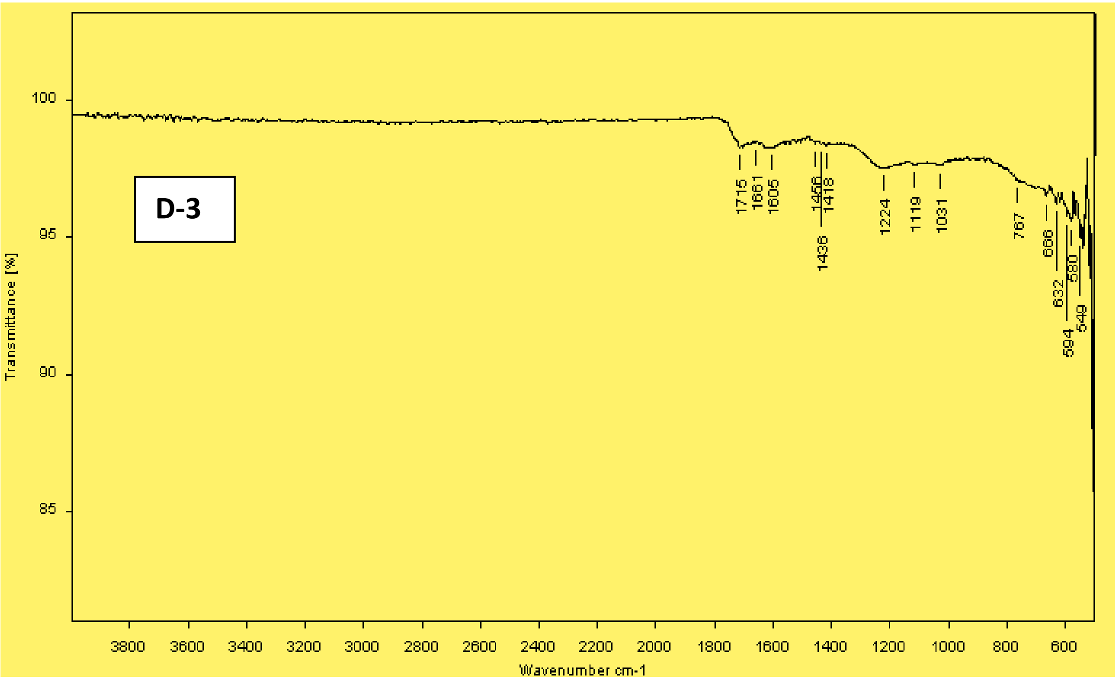

E-2
Transmittance [%]
Wavenumber cm-1
1715
1607
1225
632
607
593
575
552
519

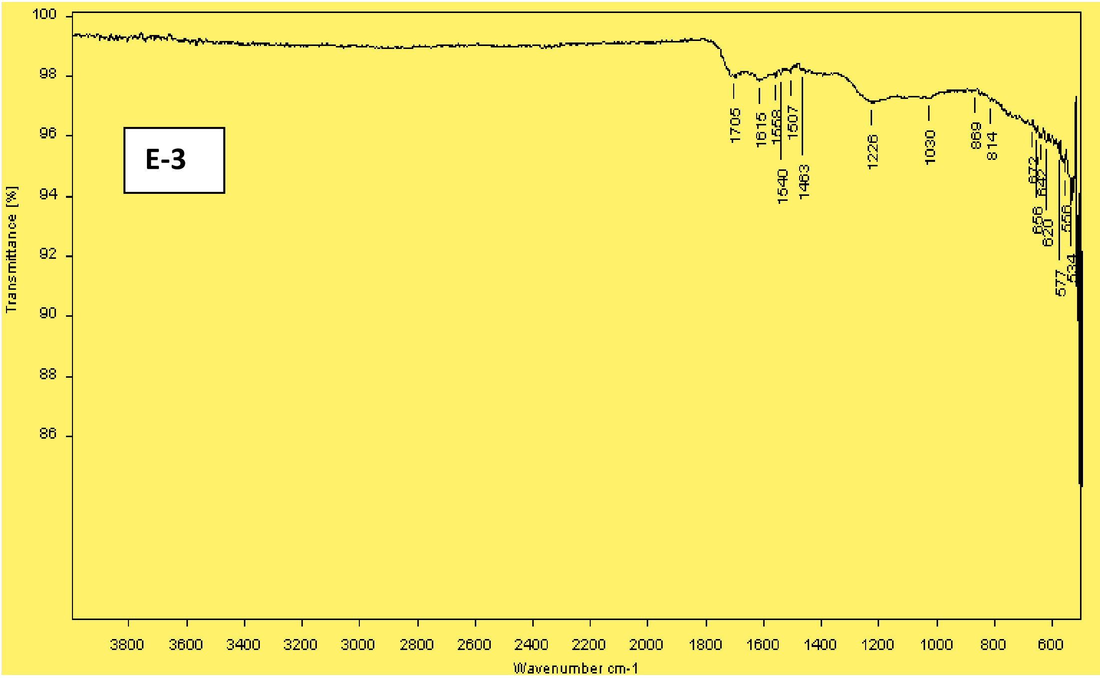
Fig. 6.3 FTIR spectra of humic acids extracted from the soils under peach (A-3), pear (B-3), Khasi mandarin (C-3), guava (D-3) and control site (E-3) in horticulture land use systems






C-3
Transmittance [%]
Wavenumber cm-1
1732
1698
1622
1498
1036
745
641
597
622
575
536


D-3
Transmittance [%]
Wavenumber cm-1
1715
1661
1605
1456
1418
1436
1224
1119
1031
767
666
632
580
594
549



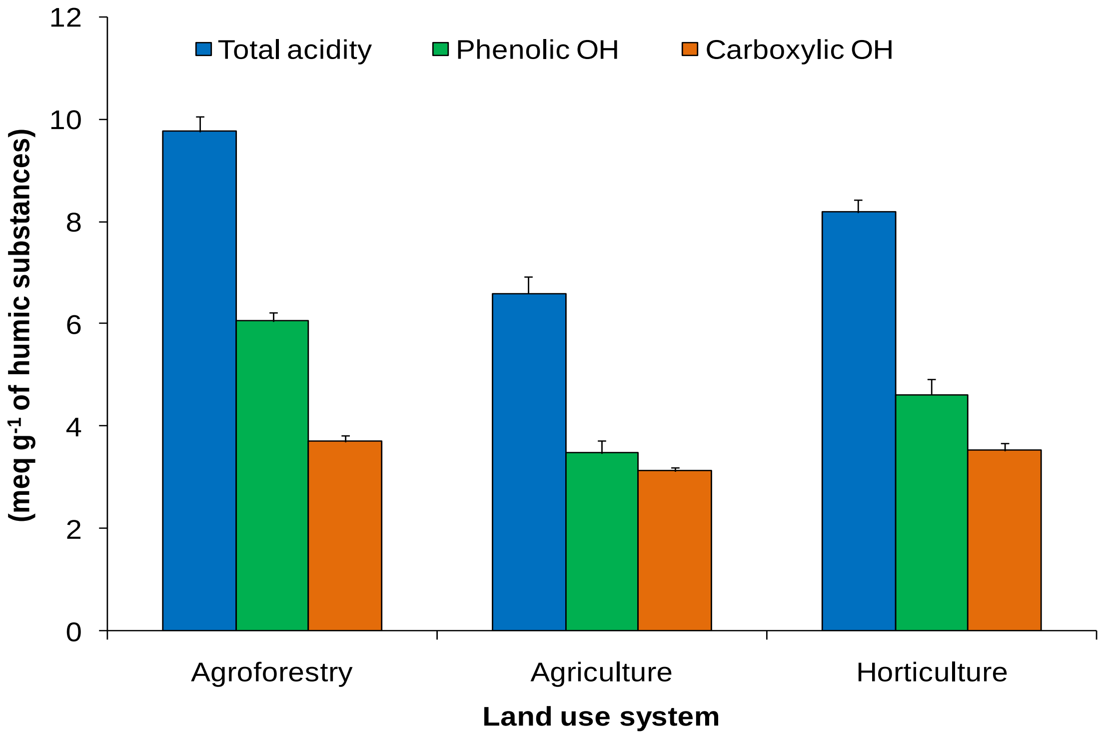


Fig. 6.4 Effect of different land uses on total acidity, phenolic OH and carboxylic OH groups (g $100g^{-1}$) of humic acids

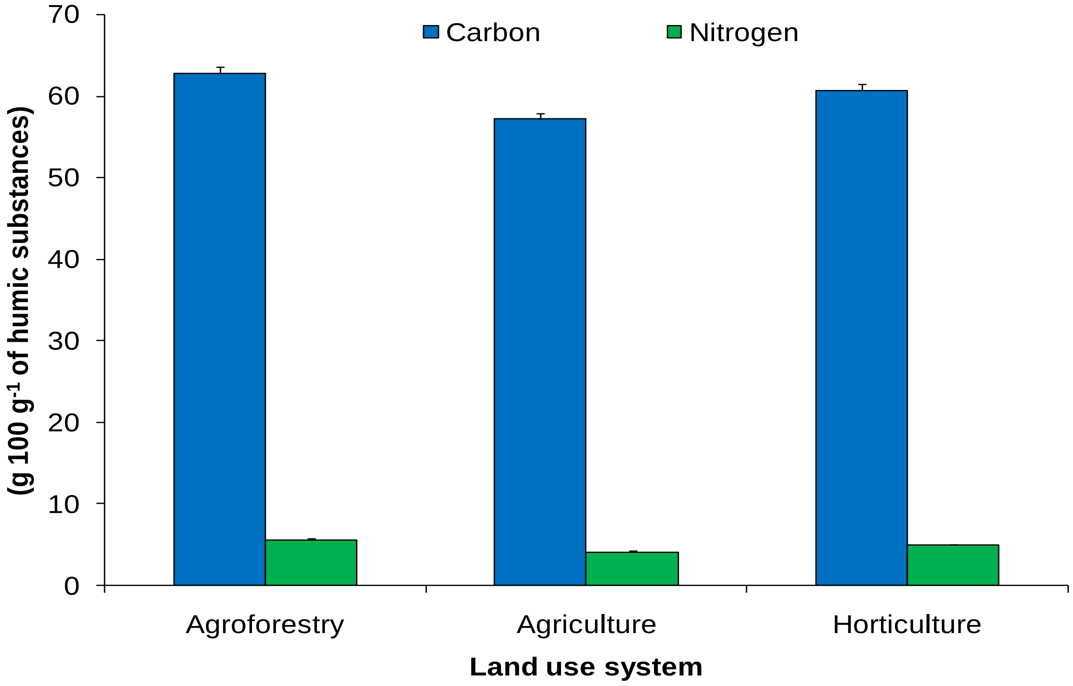


Fig. 6.5 Effect of different land uses on carbon and nitrogen (g 100g^{-1}) of humic acids

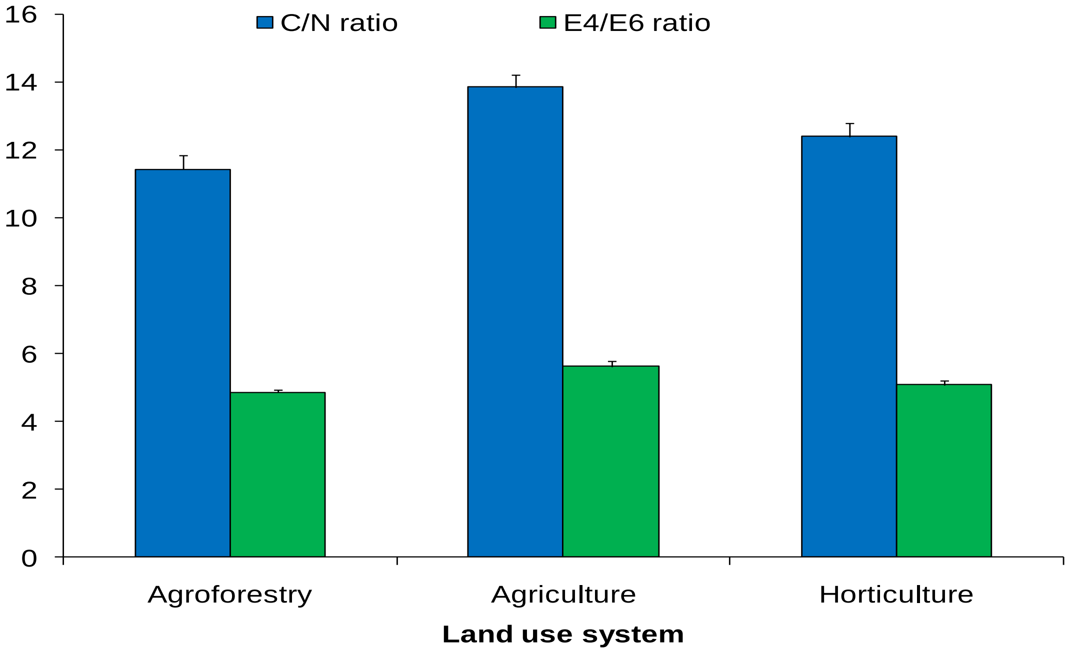


Fig. 6.6 Effect of different land uses on C/N ratio and E_4/E_6 ratio of humic acids

CHAPTER VII

RESEARCH PAPER- IV

Assessment of soil carbon stability under different land use systems in East Khasi hills of Meghalaya

7.1 Abstract

This study was conducted to evaluate the carbon, nitrogen and other binding agents mainly polysaccharides and total glomalin on aggregate stability of soils collected from agroforestry, agriculture and horticulture land uses in east Khasi hills of Meghalaya, north-east India. Surface soil samples were collected from each land uses, separated in to macro (>250 µm) and microaggregates (<250 µm) and used for the analysis of various binding agents such as carbon, nitrogen, polysaccharides and total glomalin content in bulk soils as well as in macro and microaggregates. The overall pattern indicated that the mean weight diameter (MWD) of the studied land uses followed the order of agroforestry>horticulture>agriculture indicating the highest aggregate stability under agroforestry land use. In general, the average distribution of organic carbon, nitrogen, total polysaccharides (TPS), dilute acid extractable polysaccharides (DAEP) and total glomalin were highest under agroforestry land uses compared to horticulture and agriculture land uses. Amongst the aggregates, macroaggregates recorded highest values of all these attributes compared to microaggregates. In the present study, all these attributes followed the order of macroaggregates>bulk soils>microaggregates. Agriculture land use decreased the concentration of total carbon, nitrogen, TPS, DAEP and total glomalin by 31, 15, 6, 15 and 17 % in bulk soil; 29, 11, 10, 13 and 19 % in macroaggregates; 25, 12, 10, 12 and 29% in microaggregates compared to agroforestry land use.

The rate of carbon losses in macro and microaggregates, as calculated by first order rate constant, were higher under agriculture land use (0.0025 and 0.0018) followed by horticulture land use (0.0016 and 0.0019) and the lowest was under agroforestry land use (0.0017 and 0.0014). Overall, the per cent loss of carbon and nitrogen, in terms of mineralization, was highest in macroaggregates (13, 17 and 14 for carbon; 20, 23 and 24 for nitrogen) than microaggregates (11, 15 and 13 for carbon; 18, 21 and 19 for nitrogen) in agroforestry, agriculture and horticulture land use systems, respectively. The temperature sensitivity of aggregate carbon and nitrogen mineralization was examined by using the Arrhenius activation energy and Q_{10} values. In both carbon and nitrogen mineralization, the average activation energy was highest for microaggregates (88 and 62.8 kJ mol^{-1}, respectively) than macroaggregates (73.4 and 54.9 kJ mol^{-1}, respectively) indicating the high stability nature of the microaggregates. Highest average Q_{10} values for macroaggregates C and N decomposition (0.871 and 0.897, respectively) indicate the faster reaction rates of macroaggregates than microaggregates (0.845 and 0.878, respectively). It was found that macroaggregates stored more carbon, nitrogen, TPS, DAEP and total glomalin than microaggregates. On the other hand, microaggregates lost less carbon and

nitrogen upon incubation up to a period of 60 days indicating the highest stability of the microaggregates compared to macroaggregates irrespective of the land uses.

Key words: Land use, aggregate stability, macroaggregates, microaggregates, carbon and nitrogen mineralization, total polysaccharides, DAEP, total glomalin.

7.2 Introduction

Soil aggregation is an important mechanism for stabilization of soil organic matter (SOM) (Lutzow et al., 2006; Six et al., 2000). Soil aggregation is caused by a variety of aggregate stabilizing compounds, which work simultaneously at different spatial scales. Organic matter (OM) is one of the most important constituents of soils due to its capacity in affecting plant growth indirectly and directly. Indirectly, OM improves the physical conditions of soils by enhancing aggregation, aeration and water retention, thereby creating a suitable environment for root growth (Senesi and Loffredo, 1999). The relationship between OM and soil aggregation or structure formation was described by Tisdall and Oades (1982) in a conceptual model, affected by three types of aggregation agents.

In soils, where the OM is the main binding agent, aggregates of different sizes can be formed. Primary particles and clay microstructure are bound together with bacterial and fungal debris into extremely stable microaggregates which may be bound together with fungal and plant debris giving a larger microaggregates. The humic matter, considered as a persistent cementing agent, is involved in stabilizing microaggregates. These microaggregates are bound into macroaggregates, due to the effect of transient binding agents (polysaccharides derived from plants and microorganisms) and temporary binding agents (fungal hyphae, fine roots, bacterial cells) (Tisdall and Oades, 1982; Oades, 1993). This aggregate hierarchy theory, which states an uneven distribution of organic compounds among different aggregate-size fractions, has been used by many authors to explain correlations between a reduction of aggregation and a loss of SOM (Six et al., 2004).

Soil carbohydrates, which represent from 5 to 25% of SOM (Stevenson, 1994), constitute a significant part of the labile pool of SOM and are most affected by land use changes (Spaccini et al., 2001). Due to the temporary biological stability of carbohydrates (Insam, 1996), their long-lasting role in improving soil physical properties may not be assumed in all soil conditions (Degens and Sparling, 1996) and large emphasis had been given to the action of polymeric carbohydrates in stabilizing soil structure (Tisdall and Oades, 1982). The effect of cultivation on the nutrient and microbial characteristics of soils are observed in the C and N-enriched small macroaggregate fractions (2.00–0.25 mm) (Cambardella and Elliott, 1993). Dormaar (1983) reported that SOM and polysaccharides were associated with the >0.25 mm water-stable aggregates (WSA). Christensen (1992) observed that whereas the C/N, C/P and N/P ratios of water-stable macroaggregates were smaller than those of microaggregates, the microaggregates contained less SOM associated with silt plus clay than the macroaggregates.

Organic C storage in soil is increased by adsorption onto mineral surfaces (Tisdall and Oades, 1982) and by physical protection within the soil matrix, where it is inaccessible to microbes (Puget et al., 2000). The concept of physical protection emphasizes the importance of aggregation in the processes of SOC stabilization and turnover (Christensen, 2001). The location of organic matter within the soil structural units has been demonstrated to control SOM dynamics (Angers and Chenu, 1997, Feller and Beare, 1998). Studies indicate that the macroaggregate structure exerts some physical protection on soil organic matter (Beare et al., 1994), whereas soil organic matter is mostly protected in free microaggregates (Six et al. 1998) and in microaggregates within the macroaggregates (Bossuyt et al., 2002).

Even though it is generally known that land use strongly affects soil aggregation (John et al., 2005; Ashagrie et al., 2005), the temporal dimension of this impact remains unknown. The ability of the soil to function as a component of an ecosystem may be degraded, aggraded or sustained as use-dependent properties change in response to land-use and management (Fesha et al., 2002). For instance, some studies have shown that when forest land is converted to pasture land, soils are subject to compaction and subsequent decreased porosity (Deuchare et al., 1999). Conversely, when pasture is converted to forest land, infiltration increases and soil erosion decreases with increasing forest age (Carter et al., 1998). Soil aggregates stability has also been shown to decrease for soils under annual crops (Angers et al., 1999); continuous tillage and arable crop production (Kavdir et al., 2005). Similarly, conversion of forests into croplands known to deteriorate soil properties, especially reduce soil organic carbon (SOC) and changes in distribution and stability of soil aggregates (Singh and Singh, 1980). Loss of SOC with cultivation is connected to the destruction of macroaggregates (Elliott, 1986), as a result, soil becomes more susceptible to erosion since macroaggregates are disturbed (Six et al., 2000). Soil aggregates are thus dynamic soil properties that tend to respond rapidly to environmental changes (Coote et al., 1988); and different land use types would exercise their effects on soil properties such as aggregates formation and stabilization in various ways and magnitudes. Their effects may be bigger or smaller in terms of degradation (Taboada-Castro et al., 2006), hence the conduct of the study under different land use types is at most important.

The low structural stability of the tropical and subtropical soils generally and soils of north-eastern India in particular, is one of the most serious soil physical constraints to increased and sustained high level crop production in these regions. This, coupled with high rainfall and climatic erosivity, is the major cause of the high rate of water erosion experienced in these regions. The breakdown of these aggregates and their subsequent dispersion have therefore been reported to be the first step to crusting and surface sealing which would in turn reduce water infiltration into the soil and soil water storage and availability; and consequently accelerate surface runoff and soil erosion (Lado et al., 2004). The problem of high erosion damage has been described for some areas in north-eastern India (Prasad et al., 1981; Prasad and Sharma, 1993) associated the high erosion intensity in the region to combination of very high erosive rainfall and highly erodible soil.

Knowledge of the impact of different land use types on the stability of their various soil aggregates, especially at macro and microaggregate level, has therefore become imperative to ensure better management of these soils. As aggregate stabilizing compounds we choose to study carbohydrates and glomalin. Carbohydrates in soils are derived from bacteria, fungi or root mucilage (Jolivet et al., 2006). Glomalin is a thermostable protein produced by arbuscular mycorrhizal fungi (Wright and Upadhyaya, 1998). As the glomalin fraction gained by high temperature extraction is not completely pure, it is addressed as glomalin-related soil protein and has been shown to strongly contribute to soil aggregation (Rosier et al., 2006). Thus, considering the high rainfall erosivity and high soil erodibility or the dynamic nature of surface soil properties in the region, there is a need to conduct an in-depth study on the aggregate stability and stabilizing agents of these soils, with a view to evaluating land use effects on soil systems, thus developing stable and viable soil management strategies for optimizing their structural stability. The information obtained from the study would also be important for appropriate and rational land use planning in accordance with the potential of each land use type. This has been shown to be a key to successful land development for sustainable agricultural production (Lal, 1986). However, little or no information is available on aggregate stability and associated stabilizing agents of soils as affected by the different land use types considered in this study in north-eastern India. Thus, this study was conducted with the objectives of: (1) quantifying effects of some land use types on the quantity of aggregate stabilizing agents of the various soils, and (2) determining the magnitude of rate of decomposition of carbon and nitrogen in aggregates from various land uses with respect to temperature increase by using Arrhenius activation energy and Q_{10} values.

7.3 Materials and methods

7.3.1 Location of the study site

The study was carried out with different multipurpose tree species (MPTs) planted under agri-silvipastural system in 1983, horticulture tree species planted in 1994 and agricultural crops continuously cultivated for the past 10 years at research farm of Indian Council of Agricultural Research (ICAR) Complex for North-East Hill (NEH) Region, Umiam. The station is situated in the central part of Meghalaya in the East *Khasi* Hills of North-East India.

Soil samples were collected during October-November in the year 2009. The air-dried soil samples were characterized with respect to following properties. The mean-weight diameter (MWD) was determined on air-dried samples, pre-sieved through a 4 mm mesh according to the procedure outlined by Kemper and Chepil (1965). The total and dilute acid-extractable polysaccharides were analyzed with the technique modified from Whistler and Wolfrom (1962) by Lowe (1994). Total glomalin content in soils and aggregates was estimated by following the method of Wright and Upadhyaya (1999).

Incubation experiment was carried out to study the temperature responses of aggregates (macro and micro). Aggregates were analyzed for the carbon and nitrogen content (Jackson, 1973) after the specific time period at different temperature gradients for the calculation of the percent carbon remained in the aggregates. By using these data, the temperature sensitivity of the carbon and nitrogen losses were calculated by using Arrhenius equation **(k= A exp (-E/ RT)** and Q_{10} values (Knorr et al., 2005).

Data obtained from this research programme was statistically analysed following standard statistical methods (Gomez and Gomez, 1984).

7.4 Results

7.4.1 Land use, duration and temperature effects on aggregate carbon

Land uses, duration and temperature significantly affected carbon decomposition in both macroaggregates and microaggregates; however interaction effect was not significant (Table 7.24). The interactive effects of land use and duration, land use and temperature, and duration and temperature on soil macroaggregate carbon decomposition are given in the table 7.2, 7.4 and 7.6, respectively. From the table 7.2, it clearly evident that increase in the incubation period up to 60 days increased the carbon decomposition irrespective of the land use systems. However, the mean per cent loss of aggregate carbon with respect to initial macroaggregate carbon was highest under agriculture land use system (24%) than agroforestry (18%) and horticulture (20%) land use systems (Table 7.2). The maximum loss was observed at 60 days of incubation and was highest in agroforestry (4.60 g kg^{-1}) followed by agriculture (4.37 g kg^{-1}) and the lowest was under horticulture (4.26 g kg^{-1}) land uses. Increase in temperature from 25 to 35 ^{o}C increased the aggregate carbon loss significantly in all the land uses although the increase was highest at 35 ^{o}C (Table 7.4). On an average, the macroaggregate carbon loss increased from 1.86-3.79 g kg^{-1}; 1.57-3.92 g kg^{-1} and 1.46-3.76 g kg^{-1} under agroforestry, agriculture and horticulture land uses, respectively by increasing the temperature from 25 to 35 ^{o}C. In comparison to the initial macroaggregate carbon, increase in temperature from 25 to 35 ^{o}C, increased the carbon loss by 15.0, 22.0 and 17.0% with the maximum increase in Khasi pine, maize and guava under agroforestry, agriculture and horticulture systems, respectively. However, with respect to the tree and crops species, control plots lost slightly more macroaggregate carbon in all the land uses (Fig. 7.8).

The microaggregate carbon content was highest under agroforestry land use (range: 20.3-27.2 g kg^{-1} with an average of 22.4 g kg^{-1}) followed by horticulture land use (range: 17.5-21.4 g kg^{-1} with an average of 19.47 g kg^{-1}) and was lowest under agriculture land use systems (range: 14.7-17.7 g kg^{-1} with an average of 16.67 g kg^{-1}). Similar to macroaggregates, microaggregates also followed the same trend in carbon decomposition due to the increased incubation period as well as temperature in all the land use systems (Table 7.3, 7.5 and 7.7). During the incubation period of up to 60 days, the rate of carbon loss was more under control plots (3.73, 3.24 and 3.18 g kg^{-1}in agroforestry, agriculture and horticulture land uses, respectively) compared to the average loss from tree and crop species (Table 7.3). Like macroaggregate, the per cent carbon losses in microaggregate, in comparison to initial microaggregate carbon, was maximum under agriculture (19%) land uses than agroforestry (15%) and horticulture (15%) land uses (Fig. 7.9 and table 7.5). The mean macro and microaggregate carbon content, at the end of the incubation period was highest under agroforestry land use (23.0 and 20.4 g kg^{-1}, respectively) followed by horticulture land use (19.29 and 17.6 g kg^{-1}, respectively) and the lowest was under agriculture (15.56 and 14.7 g kg^{-1}, respectively) land uses, respectively (Fig. 7.1, 7.2 and 7.3). Amongst the tree and crop species, the maximum mean macro and micro-aggregate C was recorded in Alder (27.3 and 25.3 g kg^{-1}); rice (17.0 g kg^{-1}) and turmeric (16.4 g kg^{-1}); Khasi mandarin (20.8 and 19.8 g kg^{-1}), respectively in agroforestry, agriculture and horticulture land use systems (Fig. 7.2 and 7.3). Similarly, in terms of C content in the macro and microaggregates, lowest values were recorded at 60 days of incubation among the incubation periods and at 35 ^{o}C amongst the three temperatures used in the experiment in agriculture land use than other land uses (Fig. 7.4, 7.5, 7.6 and 7.7). The per cent carbon mineralization in macro and microaggregates was highest under agriculture land use (17 and 15, respectively) followed by horticulture (14 and 13, respectively) and the lowest was under agroforestry land use (13 and 11, respectively). On the whole, the per cent carbon mineralization was higher in macroaggregates than microaggregates irrespective of the land uses due to increased temperature over a period of 60 days (Fig. 7.8 and 7.9).

7.4.2 Land use, duration and temperature effects on aggregate nitrogen

Tables 7.12, 7.13, 7.14, 7.15, 7.16, 7.17 and 7.18, shows the effects of land uses, temperature, duration and their interaction on initial macro- and microaggregate nitrogen content and their subsequent mineralization or losses over the period of 60 days at three different temperatures *viz*., 25, 30 and 35 ^{o}C. The macro and microaggregate nitrogen almost followed the similar trend of aggregates carbon decomposition during the incubation period irrespective of the tree and crop species. However, the magnitude of loss of aggregate nitrogen was slightly higher than the aggregate carbon loss (Fig. 7.17 and 7.18). Parallel to aggregate carbon, land uses, duration and temperature significantly affected nitrogen losses in both macroaggregates and microaggregates; however interaction of any of these factors did not change the nitrogen contents of the aggregates significantly (Table 7.24). The initial macro and microaggregate nitrogen content was maximum under agroforestry land use (2.43 and 2.07 g kg^{-1}, respectively) followed by horticulture (2.24 and 1.97 g kg^{-1}, respectively) and the lowest was under agriculture (2.13 and 1.82 g kg^{-1}, respectively) land uses. Amongst the tree and crop species, Alder (2.77 and 2.35 g kg^{-1}), Rice (2.22 and 1.96 g kg^{-1}) and Guava (2.37 and 2.06 g kg^{-1}) recorded highest values of nitrogen in both macro and microaggregates, respectively in each land use systems. However, with respect to control plots in each land uses, adoption of tree and crop species increased the macroaggregate nitrogen content by 16, 4.0 and 6 %; microaggregate nitrogen content by 15, 9.0 and 8.0% in agroforestry, agriculture and horticulture land uses, respectively. In general, the magnitude of increase in nitrogen content due to the adoption of agroforestry tree species was slightly higher with respect to macroaggregates whereas, agriculture crops and horticulture tree species increased the microaggregates nitrogen more than the macroaggregate nitrogen compared to control plots. Upon the incubation up to the period of 60 days, macroaggregate nitrogen was lost by 0.37-0.59, 0.41-0.58 and 0.50-0.60 g kg^{-1}; microaggregate nitrogen lost by 0.39-0.55, 0.40-0.52 and 0.35-0.61 g kg^{-1} in agroforestry, agriculture and horticulture land uses, respectively (Table 7.12 and 7.13). However, at 60 days of incubation, the nitrogen losses were comparatively more in macroaggregates (0.48, 0.50 and 0.54 g kg^{-1}) than microaggregates (0.47, 0.44 and 0.44 g kg^{-1}) in agroforestry, agriculture and horticulture land uses, respectively. With respect to the temperature, nitrogen losses in both the aggregates were more at 35 ^oC irrespective of the land uses and duration (Table 7.14, 7.15, 7.16 and 7.17). However, the temperature increase from 25 to 35 ^oC caused 0.44, 0.44 and 0.54 g kg^{-1} losses of nitrogen in macroaggregate, and 0.41, 0.40 and 0.39 g kg^{-1} of nitrogen losses in microaggregate under agroforestry, agriculture and horticulture land uses, respectively. On the whole, due to the interaction of land uses, temperature and duration, the per cent nitrogen losses was comparatively minimum in microaggregates (17.6, 21.4 and 19.3, respectively) than macroaggregates (19.7, 23.5 and 24.1, respectively) in agroforestry, agriculture and horticulture land uses. On the other hand, the magnitude of nitrogen losses in both aggregates due to the interaction of these factors was relatively higher than aggregate carbon losses (Fig. 7.17 and 7.18).

7.4.3 Rate of reaction, activation energy and Q_{10} values of aggregates carbon decomposition

The results of the changes in the reaction rates at different temperatures for the decomposition of carbon in macroaggregates under various land use systems are given in table 7.10. In all the land use systems, the rate of reaction over the 5 ^oC increase in temperature range was more in control plots, on an average, compared to the tree and crop species. Temperature increase from 25 to 30 ^oC increased the rate constant by 2.2, 1.8 and 1.9 times whereas, temperature increase from 30 to 35 ^oC increased the rate constant by 15, 1.7 and 1.6 times in agroforestry, agriculture and horticulture systems, respectively. In general the rate of reaction was highest in agriculture land use (range: 0.0008-0.0025) followed by horticulture land use (range: 0.0006-0.0019) and the lowest was under agroforestry land use (range: 0.0005-0.0017). The mean activation energy of microaggregate carbon loss followed the order of horticulture (85.5 kJ mol^{-1}) > agriculture (74.6 kJ mol^{-1})>agroforestry (60.2 kJ mol^{-1}) (Table 7.10). On contrary, Q_{10} values followed the order of agroforestry (0.895)> agriculture (0.866) >horticulture (0.852) land use systems (Table 7.10) with maximum values in the soils under Champak (0.955), turmeric (0.894) and guava (0.874) in each land uses. With respect to control plots, the AE was relatively higher than tree and crop species but the trend was reverse in case of Q_{10} values showing highest values in tree and crop species and lowest values under control plots in each land uses.

Table 7.11 shows the effects of land uses on rate of reaction, activation energy and Q_{10} values of microaggregate carbon decomposition. The mean rate constant of the carbon mineralization in microaggregate followed the similar trend of macroaggregate carbon decomposition and was maximum under agriculture (0.0018) followed by horticulture (0.0016) and the lowest was under agroforestry (0.0014) land uses. Increase in temperature from 25 to 30 ^{o}C increased the rate constant by 1.5, 2.1 and 1.5 times; from 30 to 35 ^{o}C increased the rate constant by 1.40, 1.3 and 1.5 times in agroforestry, agriculture and horticulture land uses, respectively. In comparison to the macroaggregates, the rate of reaction was slightly slower in microaggregates as evident from the tables 7.10 and 7.11. In contrast to macroaggregates, the activation energy in microaggregate carbon decomposition was relatively higher and maximum was under agroforestry (91.9 kJ mol^{-1}) land use followed by horticulture (87.2 kJ mol^{-1}) and the lowest values were recorded in agriculture (84.9 kJ mol^{-1}) land use. Opposite to activation energy, Q_{10} values were highest in macroaggregate carbon loss in relative to microaggregate carbon decomposition and followed the order of: horticulture (0.847)>agriculture (0.845) >agroforestry (0.842) (Table 7.11).

7.4.4 Rate of reaction, activation energy and Q_{10} values of aggregates nitrogen decomposition

The study showed that both macro and microaggregates under various tree and crop species had a comparatively lower rate constants values than that of control plots, at all the study temperatures (Tables 7.20 and 7.21). The mean rate constants of microaggregate nitrogen decomposition ranged between 0.008 to 0.0018 in agroforestry land use; 0.0012 to 0.0022 in agriculture land use and 0.0011 to 0.0022 in horticulture land use at 25 and 35 ^{o}C, respectively. Regarding macroaggregate nitrogen decomposition, it ranged from 0.0012 to 0.0026 in agroforestry, 0.0013 to 0.0029 in agriculture and 0.0012 to 0.0028 in horticulture land uses at 25 and 35 ^{o}C, respectively (Table 7.21).

The maximum activation energy (AE) value of 25.8 kJ mol^{-1} for agroforestry systems followed by 57.68 kJ mol^{-1} for horticulture systems and the lowest AE of 48.66 kJ mol^{-1} for agriculture systems were obtained under macroaggregate nitrogen decomposition. On the other hand, for Q_{10} values of macroaggregate nitrogen decomposition, agriculture land use showed the maximum value of 0.909 followed by 0.889 for agroforestry and 0.890 for horticulture land uses (Table 7.20). In microaggregate nitrogen decomposition, the AE were 58.03, 60.85 and 68.19 kJ mol^{-1}; Q_{10} values were 0.842, 0.845 and 0.847 for agroforestry, agriculture and horticulture systems, respectively (Table 7.21). The AE values were relatively higher under control plots than under tree and crop species in both macro and microaggregates nitrogen decomposition whereas, the Q_{10} values were lower under control plots than tree and crop species. Amongst the two aggregates, AE and Q_{10} values were considerably higher in microaggregate nitrogen decomposition compared to macroaggregate nitrogen decomposition (Table 7.20 and 7.21).

7.4.5 Effect of land uses on polysaccharides in soil and aggregates

Land use changes from fallow to various land uses significantly affected total polysaccharides (TP) in whole soil, as well as in the macro and microaggregates (Table 7.22). However, none of the land use systems affected the dilute acid extractable polysaccharides (DAEP) content in soil, macro and microaggregates (table 7.22). Conversion of land use from fallow to agroforestry, agriculture and horticulture land uses increased the TP by 31, 16 and 11% in soil, 25, 11 and 8.0% in macroaggregates and 24, 30 and 15% in microaggregates, respectively. Considering DAEP, adoption of agroforestry trees, horticulture fruit trees and agriculture crop species increased the DAEP content by 16, 54 and 27% in bulk soil, 14, 19 and 31% in macroaggregates and 24, 43 and 60% in microaggregates. However, in comparison to agriculture land use, agroforestry land use recorded 7.0, 11.0 and 12.0% increase in TP content; 17.0, 15.0 and 14.0% increase in DAEP content of soil, macroaggregates and microaggregates, respectively (Table 7.22). In general, the TP and DEAP contents were in the order of: macroaggregates>soil>microaggregates in both agroforestry and agriculture land use systems whereas, horticulture land use showed the order of macroaggregates>microaggregates>soil. Amongst the three land uses, the TP and DAEP were higher in agroforestry (1.47 and 1.16; 1.78 and 1.41; 1.36 and 1.09 g $100g^{-1}$) and the lowest values under agriculture land use system (1.18 and 0.99; 1.48 and 1.23; 1.22 and 0.96 g $100g^{-1}$) in soil, macroaggregate and microaggregates, respectively.

7.4.6 Effect of land uses on total glomalin in soils and aggregates

From the table 7.23, it is clearly seen that none of the land use systems affected the total glomalin content in soil, macro and microaggregates significantly. The mean total glomalin contents of agroforestry land use were 0.24, 0.37 and 0.28 g $100g^{-1}$; agriculture land use were 0.20, 0.30 and 0.20 g $100g^{-1}$; horticulture land use were 0.2, 0.36 and 0.26 g $100g^{-1}$ in soil, macro and microaggregates, respectively. In contrast to TP and DAEP, total glomalin followed the order of macroaggregates>microaggregate>soil. Land conversion from fallow (control plots) to agroforestry, agriculture and horticulture systems, increased the total glomalin content by 32, 28 and 38%; 29, 35 and 40%; 41, 35 and 64% in soil, macro and microaggregates, respectively. However, agriculture land use recorded 21, 23 and 40% lower total glomalin content compared to agroforestry land use and 7.0, 20 and 30% less total glomalin content than horticulture land use in soil, macro and microaggregates, respectively.

7.4.7 Correlation between MWD, aggregate carbon and nitrogen, and soil properties

Table 7.25 indicates that MWD had positive and significant correlation with macro and microaggregate carbon and nitrogen and also with soil organic carbon and nitrogen; however, it showed negative and significant relationship with bulk density (-0.692**). Both micro and macro aggregate carbon and nitrogen were positive and significantly correlated with soil organic carbon, nitrogen and MBC whereas, they exhibited a positive and negative correlation with soil BD (Table 7.25). Total polysaccharides and DAEP had positive and significant correlation with both macro and microaggregate carbon and nitrogen whereas, only soil glomalin content showed positive and significant correlation with them. Macroaggregate glomalin was significant and positively related with MWD (0.31*), macroaggregate nitrogen content (0.31*), soil TP (0.59**), macroaggregate TP (0.36*) and soil DAEP (0.51**) whereas, microaggregate glomalin showed positive and significant relationship only with soil TP (0.48**) and soil DAEP (0.34*) (Table 7.25)

7.5 Discussion

7.5.1 Aggregates carbon and nitrogen contents

Agroforestry land use had significantly highest quantity of aggregate carbon and nitrogen compared to agriculture land use. These results could be related to the larger amount of living and decaying plant roots, polysaccharides and glomalin under agroforestry land use that would have been rapidly destroyed by cultivation under agriculture land use. These results confirm earlier observations that aggregates are dynamic in nature, the size distribution of aggregates being affected by the change in land use and management practices in tropical conditions (Spaccini et al., 2001; Ashagrie et al., 2005). Higher organic C in aggregates than in bulk soil indicates that organic agents have bound mineral particles to form aggregates. An increase of organic C with increasing aggregate sizes confirmed the aggregate hierarchy (Oades, 1984). Compared with the control plots, aggregate stability was improved under tree and crop species. The lower amount of aggregate carbon and nitrogen in the control plots is due to the relatively less carbon inputs that are added to these soils either through external addition or by way of crop residue additions and variations in soil properties. Studies on the mass distribution of organic carbon among aggregate classes (Cambardella and Elliott, 1993; Six et al., 2000) suggest that an increase in soil aggregation through adoption of best management practices usually leads to carbon sequestration. Increased SOC improves soil aggregate stability, which tends to decrease with increasing aggregate size (Puget et al., 2000). Another important factor in these kaolinite rich soils is that inorganic binding agents, such as alumina-silicates and crystalline iron oxides, form more stable aggregates (Tisdall and Oades, 1982), which covers the effects of organic matter on aggregate stability, as reported by Zhang and Horn (2001). This suggested that physical and biological processes should both be involved in the interaction of soil aggregation and SOC sequestration in an ecosystem.

Polysaccharides derived from microbial activity acted as transient cementing agents between micro-aggregates and hence were important factors in macro-aggregation (Tisdall and Oades, 1982; Conant et al., 2004). Compared to agroforestry systems, the declines of SOC in agriculture land use among the aggregates were the largest for the macroaggregates and smallest for the microaggregates because the smaller aggregates provide a greater degree of physical protection of SOC than macro-aggregates (Conant et al., 2004). The studies of Balesdent et al. (2000) showed that when the macroaggregates break down, they release microaggregates and microbially processed organic matter particles. The present study established that when microaggregates are formed from disruption of macroaggregates the intermediary organic carbon holding these microaggregates also get lost in the disintegration process. The lower total organic carbon content in the microaggregates than the macroaggregates in all the studied land use systems are due to these reasons and these is in confirmation with the earlier studies. However, microaggregates are still stable enough and not as sensitive to disruptive forces as the macroaggregates, and therefore become resistant to microbial attack.

The N content of the agroforestry soils was significantly higher than those of agriculture soils regardless of aggregate sizes. The nitrogen concentrations associated with macroaggregate in the agroforestry land use, were 15% higher than the corresponding values in the agriculture land use. On the other hand, microaggregates of all the land uses had higher concentration of nitrogen compared to microaggregates of control plots in each land uses. This could be attributed partly to the redistribution and/or transfer of SOC from the large aggregates to smaller ones either in the process of biodegradation or by mechanical disruption of the large macroaggregates (Ashagrie et al., 2005). These distribution patterns of soil N were consistent with soil C. When SOC in aggregates changed, the soil N storage changed correspondingly. Sainju et al. (2003) reported the retention of SOC depended on soil total N while Gao et al. (2000) showed the trend of SOC fluctuation corresponded with soil N. Cultivation practices caused a lower accumulation of C and N in the aggregates of the studied soils. Exposure of more surface area for microbial attack, oxidation, burning effect of temperature and preferential removal of the smaller aggregates by erosion could be increased following the cultivation. The macro and microaggregate carbon and nitrogen were significantly correlated with SOC (Table 7.25). Several studies have shown that SOC was the major cementation material and had an important effect on the stability and composition of water-stable aggregates (Chaney and Swift, 1984; Elliott, 1986).

7.5.2 Thermal stability of aggregates carbon and nitrogen

The relationship of aggregate stability to soil organic carbon was possibly related to the binding agents (hyphae and roots) and fungal exudates such as polysaccharides and glomalin (Wright and Upadhyaya, 1996; Wright et al., 1999) which improve the stability of aggregates. These organic compounds may impart some degree of water repellency, thereby improving soil stability (Eynard et al., 2004). The analysis of first order rate kinetics in different land use systems showed that aggregates in the agroforestry system showed a better protection of the organic carbon compared to agriculture system in both the aggregates (Tables 7.10 and 7.20). However, the rate of reaction was faster in macroaggregates than microaggregates irrespective of the land use systems. The rate constants in both carbon and nitrogen mineralization of aggregates showed lower values at 25 ^{o}C than at 35 ^{o}C in all the land use systems. The study also found that there was a gradual increase in the rate of the reaction with temperature in both aggregates of various land use systems. The higher rate constants stand for the relatively lower stability of carbon in the macroaggregates of soils in agriculture land use which in turn is related to the higher soil disturbance. This is because of the changes in soil carbon availability between and within the aggregates among the land use systems. Aggregates physically protect soil organic matter by forming physical barriers between microbes and enzymes and their substrates and controlling food web interactions and consequently microbial turnover (Elliott and Coleman, 1988).

When SOC decrease from forest soils to agriculture lands, aggregate stability decreases as well. The reason for this is the destruction of macroaggregates by tillage or other management practices causing exposure of the inner core of particulate organic carbon facilitating rapid decomposition by microorganisms of this important organic carbon reserve in soil (Six et al., 2004). Differences in stability within the different land use systems soils may be mainly due to the differences in organic inputs through litter and root biomass, crop residues and the intensity of soil physical disturbance. These aspects and the susceptibility of the aggregates to be disrupted by the action of the dry and wet cycles, can drastically affect the dynamics of soil aggregate formation and stabilization. Hence, the enhanced protection of soil organic matter by aggregates in soils under agroforestry systems results in an accumulation of more carbon than under agriculture land use. The higher activation energies in the agroforestry system than agriculture land use further confirms the above studies and shows that lower soil disturbance or increased accumulation aggregate cementing agents could provide more protection to organic carbon stored in aggregates.

Hernantez and Hernandez (2002) found more C and N mineralization in macroaggregates than in microaggregates for cultivated soils. Similarly, Elliott (1986) and Gupta and Germida (1988) also found more C and N mineralization in macroaggregates than microaggregates, in cultivated as well as in not cultivated soils. They also stated that changes in soil structure, due to different management, could alter the interaction between microorganisms and the soil organic substrate by affecting substrate availability within and between the aggregates. This fact could indicate the presence of a higher proportion of easily mineralisable C in less stable aggregates i.e. macroaggregates inducing an increment in microbial activity thus mineralization than the well protected microaggregates. This is supported by the differences in soil microbial biomass carbon and its mineralization activity in the different size fractions in our study (Tables 7.1 and Figure 7.8, 7.9, 7.17, 7.18). According to these results the nature and the quantity of microorganisms present in the soil may be different in the two aggregate size fractions. A heterogeneous distribution of the microbial biomass in the different aggregate size fractions has also been reported by other authors (Singh and Singh, 1980; Mendes et al., 1999).These differences were probably due to variation in tillage practices, amount, type, and placement of plant biomass, soil type, and environmental conditions between the locations (Sainju et al., 2003). Sainju et al. (2009) reported in their study that organic matter that decomposes quickly can produce rapid but transient stable aggregates while organic matter that decomposes slowly can produce longer lasting stable aggregates. Faz Cano et al. (2002) reported that influences of soil order, parent material, vegetation, and climatic characteristics on the abundance of C functional groups of SOM and C and N mineralization of aggregates.

Soil organic matter can be physically stabilized, or protected from decomposition, through micro-aggregation, or intimate association with silt and clay particles, and can be biochemically stabilized through the formation of recalcitrant soil organic matter compounds. A portion of soil carbon is labile and decomposes rapidly, but most is resistant: it is retained in the soil much longer due to physical, chemical, or biochemical protection from decomposition. Besnard et al. (1996) reported that macroaggregates break down and release microaggregates and microbially processed soil organic matter particles. Microaggregates provide more protection to the carbon than that of macroaggregates. The microaggregates were found to provide a lower reaction rate and higher activation energy irrespective of the land uses (Table 7.11 and 7.21). The higher protection of the carbon enabled them to show a lower reaction rate at all temperatures compared to the macroaggregates in the same systems. The stronger bonding responsible for the high activation energies of carbon in smaller aggregates and clay bound fractions were also explained by Wershaw, (1999). In the formation of clay bound fractions, strong non-covalent bonds are formed with a variety of different types of compounds and, in particular, with carbohydrates and proteins which in turn demands large amounts of energy for their removal.

The study also indicates that among the tree and crops species, the plots receiving highest amount of organic matter could conserve a good amount of their microaggregates' carbon over a wide range of temperature without substantial loss. The higher rate in these plots compared to other plots including controls may be attributed to the higher amount of easily available carbon sources in these aggregates. The results are similar to the ones obtained by Holland et al., (2000) who proposed that soils with a larger initial proportion of labile organic matter exhibited a significantly greater response to increases in temperature. The study also revealed higher activation energy for the microaggregates derived from plots having highest amount of organic carbon. The higher activation energy in these microaggregates is an indication of the higher protection capacity of the microaggregates from these soils. Physical protection by aggregates prevents organic molecules from coming into contact with organisms, enzymes, or conditions that enable decomposition (e.g., aggregates limit O_2 diffusion). Otherwise readily decomposable organic materials that are associated with minerals, either directly or through cation-bridging, are chemically protected and less susceptible to decomposition. Increases in temperature may increase desorption from soil minerals, but chemical protection effectively increases energy of activation and leads to decreased decomposition rates. Condensation and complexation reactions also lead to the formation of organic macromolecules that resist decomposition because organisms are unable to make efficient use of them or lack the enzymes to attack them; these compounds are biochemically protected. This resistance may be overcome by co-metabolism, but biochemical protection renders a significant portion of soil carbon immune to description by standard decomposition kinetics (Jastrow and Miller, 1997).

Aggregate stability is generally strongly correlated with soil organic matter content (Tisdall and Oades, 1982; Chaney and Swift, 1984) but is sometimes more strongly correlated with labile pools of organic matter such as microbial biomass C or extractable carbohydrates (Haynes and Beare, 1996) as observed in our study (Table 7.25) suggesting the importance of increasing organic C to improve soil aggregation to large extent.

7.5.4 Q_{10} values of aggregate carbon and nitrogen mineralization

Incubation experiments have been criticized as underestimating temperature sensitivity because the quality of soil organic matter pools changes during incubation (Kirschbaum, 1995). Accordingly, Reichstein et al. (2000) suggested calculating Q_{10} values from rate constants derived from longer incubation periods rather than from instantaneously derived activities at different temperature intervals. There is not much evidence that respiration was affected by changes in soil organic matter quality during incubation: The respiration rate over a period of 60 days was fairly constant in a preliminary experiment to determine the temperature sensitivity of C and N mineralization. The sensitivity of soil microbial processes to temperature changes and the implications of these responses for the global soil carbon dynamics are of foremost importance with respect to the climate change. The relationships between decomposition rates and temperature or the temperature sensitivity of the C and N mineralization in aggregates were investigated in the present study using Q_{10} values. In agroforestry land use system there was an increase in Q_{10} values amongst the three land use systems; and values were higher under tree and crop species in relative to control plots (Tables 7.10 and 7.20) Carbon and nitrogen mineralization involves the sequential or parallel exploitation of the chemical fractions of a decomposing material (Andren and Paustian, 1987). The higher amounts carbon and nitrogen in soils under agroforestry tree species, becoming available at higher temperatures, could sustain and maintain a higher reaction rates over a wider range of temperatures compared to agriculture land use. It is postulated that a temperature-dependent stabilization mechanism was operating in the soils rich in organic matter. Because the observed temperature responses were based on the biodegradability of the carbon (composed of the sum of undecomposed straw, humified material and microbial biomass) and greater stabilization of these pools at higher temperatures that could have resulted in lower thermal optima for C and nitrogen mineralization (Dallal et al., 2003). Also reactions with large activation energies have large Q_{10} values (Fierer et al., 2006) so the protection of some organic molecules could, in theory, be more sensitive to warming than soil respiration. The smaller Q_{10} values for the agriculture land use and control plots were explained satisfactorily by Fierer et al. (2005, 2006). They found that reactions with low Ea have low Q_{10} values; that is, they are relatively less sensitive to temperature. The higher Q_{10} values under agroforestry tree species was due to the higher adapted microbes in these soils. The studies of Dallal et al., (2003) also showed that microorganisms able to perform better under suitable organic sources can increase the reaction rates over a range of temperature increase.

The Q_{10} values of the microaggregate C and N decomposition was higher than the macroaggregates in all the land use systems. This shows that at higher temperatures the microaggregates decomposes and exposes the stored carbon to decomposition. Hulscher and Cornelissen, (1996) reported that warming can accelerate desorption of organic molecules; thus making them more amenable to decomposition. Similar to macroaggregates, the Q_{10} values in agroforestry land uses were higher than agriculture land use; and also the values were lower in control plots than the under tree and crop species (Tables 7.11 and 7.21). The lower reaction rates of microaggregates can be attributed to the increased physico–chemical protection offered by these treatments. It is proposed that soil physico-chemical reactions, which stabilize soil carbon and nitrogen, and protect it from microbial respiration, may be accelerated by warming. Many reactions are involved in stabilization, some of which are likely to have positive activation energies, notably chemical adsorption (Schulten and Leinweber, 1999). Plante and McGill (2002) reported that soil disturbances arising from wet-dry cycles or tillage increase the rate of soil aggregate breakdown and turnover and studies of Besnard et al. (1996) showed that microaggregates had a higher protection capacity for organic carbon and thus nitrogen. The varied Q_{10} values for C and N mineralization in both aggregates may also be related to the substrate quality which implies higher activation energy for lower substrate quality.

According to Bosatta and Agren (1999), the carbon quality of the soil organic matter can be defined as the total number of enzymatic steps required to mineralize carbon to the end product CO_2. Hence more enzymatic steps lead to higher activation energy for the C mineralization and greater temperature sensitivity. Soil mineral particles can stabilize organic C through adsorption to surfaces, potentially resulting in different temperature dependencies (Mikan et al., 2002). The nitrogen quality of soil organic matter was defined following the same concept. The relationship between the substrate quality and the Q_{10} were similar for C and N mineralization. This suggests similar temperature sensitivity for both processes under equal substrate quality conditions. However, the Q_{10} values were higher for N mineralization than C mineralization as indicated in our study, pointing to differences in the use of soil organic matter pools (Koch et al., 2007).

7.5.4 Polysaccharides and total glomalin contents in soil and aggregates

Polysaccharides have proved to be a useful tool for measuring the effect of land uses and management practices. The total polysaccharides (TPS) content of the soil and aggregates was significantly affected by the different tree and crop species (Table 7.24). However, agriculture crops had the significant changes only on microaggregate TPS. On the other hand, none of the tree and crop species showed considerable variation in soil, macro and microaggregates dilute acid extractable polysaccharides (DAEP) concentration. The polysaccharides (both total and DAEP) concentrations reported in this study increased very rapidly after the conversion of fallow land to agroforestry, agriculture and horticulture land uses. On the other hand, although the land conversion increased the total glomalin content in all the three land uses in bulk soils, macro and microaggregates, the magnitude of increase was highest in polysaccharides than total glomalin content. This fast increase of the carbohydrate concentrations in comparison to the total glomalin clearly reveals the more labile nature of polysaccharides. Soils under agriculture land use had lower polysaccharides content than the agroforestry and horticulture land use in the bulk soil samples, macro and microaggregates (Table 7.23). Agriculture land use caused 7, 11 and 12% decrease in total polysaccharides and 17, 15 and 14% decrease in DAEP concentration compared to soils under agroforestry. The results of polysaccharides distribution within the aggregates (Table 7.22) shows that soil polysaccharides concentration decreased with decreasing aggregates sizes irrespective of the land uses. These lowest contents of polysaccharides in microaggregates were also found in reported studies (Angers and Giroux, 1996; Liu et al., 2005). The variation in the contents of polysaccharides in aggregates of different size classes can be due to an accumulation of decomposition products from plant residues in some size classes of aggregates (Guggenberger et al., 1995) and to the greater presence of fungal hyphae and small roots in macroaggregates (Tisdall and Oades, 1982). Similarly, Jolivet et al. (2006) reported that conversion of the forest land to cropland decreased the polysaccharides concentration by 34-65 %. The relatively fast reaction of the concentrations of polysaccharides towards land use change shows that polysaccharides can be used as a quick responding biomarker to track short-term changes in carbon dynamics following land use change. In relative to controls, all these values were high under tree and crop species.

Wattel-Koekkoek et al. (2001) and Liu et al. (2005) stated that the variation may also be attributed to the sorption of polysaccharides by clay minerals mainly in the presence of divalent cations in the soil. However, Spaccini et al. (2004) obtained a wider range of variation for TPS (3.5–32.6 g kg^{-1}) while, Dalal and Henry (1988) demonstrated lower variation in carbohydrate contents (3.1–6.6 g kg^{-1}). Besides the different types of sources of polysaccharides in soil, the difference nature and quantity of clay minerals associated with organic compounds in different soils can be one of the main factors that explain the difference between the results of other works and those of the present study. Dontsova and Bigham (2005) observed a twofold increase in the adsorption of a polysaccharide on the kaolinite surfaces compared to smectite surfaces. This result is important to the present study since most of the clay fractions in northeast soils are kaolinite in nature. These studies also showed that the sorption of the polysaccharides by clay minerals was related to other properties, such as surface area of the minerals and cation exchange capacity. Therefore, the variation of these properties for different soils can explain the different relations between polysaccharides and aggregation properties in different studies.

In the present study, it was found that the total glomalin concentration in the bulk soil, macro and micro aggregates increased by 21, 23 and 40% in agroforestry land use than agriculture land use. This goes in line with Preger et al. (2007), who observed a 33% increase in the surface soils glomalin content under forest land use compared to cultivated lands. The ratios of total glomalin to TOC increased in agriculture land use compared to agroforestry and horticulture land uses in bulk soil, macro and microaggregates (Table 7.23). This is in accordance with Preger et al. (2007), who reported that the ratio of glomalin to TOC increased from 0.14 in forest land to approximately 0.27 over the course of cropland use. These findings demonstrate that young SOM, which is continuously produced by soil biota, proportionally increases following C-depletion of soils with cultivation.

Results also indicated a significant and positive correlation between polysaccharides concentration and aggregate stability (MWD). This is in agreement with the findings of Jouguet et al. (2008) but in contrast with the findings of Spaccini et al. (2004) who found poor correlation between polysaccharides and MWD and reported that in tropical conditions polysaccharides cannot be always considered as persistent structural stabilizers because of their rapid degradation by microbial activities. Similarly we observed a strong positive and significant correlation between total glomalin and soil aggregates carbon content. This is in conformity with the findings of Wright et al. (2007) who also observed significant correlation between glomalin and different aggregate size fractions under various farming systems. Harner et al. (2004) also described a sigmoidal correlation between the glomalin and the aggregate stability of soils.

7.6 Conclusions

In our study, land use change from fallow to agriculture, horticulture and agroforestry has significantly increased all the above aggregate stabilizing agents. The magnitude of increase was highest under agroforestry land use compared to the agriculture and horticulture land uses stating the importance of agroforestry tree species in restoring the soil aggregate stability thus soil quality. Macroaggregates are enriched with all the stabilizing agents compared to microaggregates irrespective of the land use systems. In general, the order of these stabilizing agents in this study was macroaggregates>bulk soils>microaggregates. Of the tree and crop species, Alder (Alnus nepalensis) recorded highest values of these attributes than the other tree and crop species. Amongst the stabilizing agents, SOC had strongest correlation with mean weight diameter signifying that SOC was the major cementation material and had an important effect on the stability and composition of water-stable aggregates.

It was revealed that the rate of reaction was faster under agriculture land use amongst the three land uses studied and in macroaggregates compared to microaggregates suggesting the higher stability of the microaggregates over macroaggregates. It was also confirmed by the higher Q_{10} values of macroaggregates than microaggregates. Increase in temperature increased the rate of mineralization in both aggregates and was highest at 35 ^{o}C than at lower temperatures. Owing to its higher stability, activation energies of carbon and nitrogen decomposition in microaggregates were higher than activation energies of macroaggregate carbon and nitrogen decomposition in all the land uses including control plots. Overall, the per cent mineralization of carbon and nitrogen was highest under agriculture land use system irrespective of the aggregates. However, the magnitude of mineralization of nitrogen was relatively higher than carbon mineralization and higher in macroaggregates than microaggregates in all the land use systems.

Table 7.1 Initial soil properties of different land uses selected for the study

Land use system	Soil properties						
	pH	BD (Mg m^{-3})	MWD (mm)	Moisture (g 100g^{-1})	N (kg ha^{-1})	P (kg ha^{-1})	K (kg ha^{-1})
				Agroforestry			
Champak	4.60	1.11	31.0	31.0	522.3	32.1	313.6
Tree bean	4.59	1.15	26.1	26.1	496.3	30.1	286.7
Alder	4.38	1.07	31.8	31.8	584.3	47.2	361.2
Khasi Pine	4.36	1.05	33.8	33.8	464.8	23.7	276.0
Control (No tree)	4.76	1.18	28.4	28.4	403.4	19.3	248.4
				Agriculture			
Maize	5.12	1.27	29.9	29.9	394.4	49.1	292.7
Potato	5.24	1.31	28.9	28.9	414.6	65.6	365.1
Rice	5.09	1.33	32.2	32.2	421.4	23.5	322.2
Turmeric	5.29	1.25	26.7	26.7	398.9	36.9	306.1
Control (No crop)	4.81	1.33	24.8	24.8	387.7	20.7	203.1
				Horticulture			
Pear	5.71	1.28	31.6	31.6	437.1	43.2	323.6
Peach	6.24	1.23	32.7	32.7	455.1	39.1	495.2
K Mandarin	5.75	1.29	28.7	28.7	428.1	45.3	419.6
Guava	5.43	1.19	29.9	29.9	446.1	24.6	343.5
Control (No tree)	4.82	1.30	27.1	27.1	434.9	26.9	328.5

Table 7.2 Interactive effect of land use and duration on soil macroaggregate carbon stability (g kg^{-1})

Land use system	Duration (days)			Mean
	15	30	60	
Agroforestry				
Champak	25.06	24.10	22.70	23.74b
Tree bean	22.66	21.72	19.52	21.40c
Alder	28.73	27.63	25.41	27.26a
Khasi pine	23.31	22.31	19.87	21.83c
Control (No tree)	21.79	21.16	18.93	20.63c
Mean	24.31a	23.38a	21.22b	
Agriculture				
Maize	16.33	15.48	13.32	15.05b
Potato	16.92	16.30	14.41	15.88ab
Rice	18.33	17.46	15.13	16.97a
Turmeric	16.49	15.66	13.88	15.34b
Control (No crop)	15.92	15.07	12.82	14.60b
Mean	16.80a	15.99a	13.91b	
Horticulture				
Pear	19.93	19.52	17.43	18.96b
Peach	22.22	21.51	18.71	20.81a
Khasi mandarin	19.55	17.95	16.47	17.99b
Guava	21.70	21.40	19.34	20.81a
Control (No tree)	19.28	18.47	15.84	17.86b
Mean	20.54a	19.77a	17.56b	

Table 7.3 Interactive effect of land use and duration on soil microaggregate carbon stability (g kg^{-1})

Land use system	Duration (days)			Mean
	15	30	60	
		Agroforestry		
Champak	20.18	19.72	18.82	19.57b
Tree bean	19.18	18.75	17.00	18.31b
Alder	26.32	25.57	24.07	25.32a
Khasi pine	20.92	19.95	18.47	19.78b
Control (No tree)	20.33	18.87	17.47	18.89b
Mean	21.39a	20.57a	19.16b	
		Agriculture		
Maize	15.55	14.30	13.00	14.28c
Potato	16.68	15.47	14.37	15.51b
Rice	15.88	14.70	13.57	14.72c
Turmeric	17.67	16.62	14.83	16.37a
Control (No crop)	13.77	12.83	11.60	12.73d
Mean	15.91a	14.78a	13.47b	
		Horticulture		
Pear	18.45	16.81	15.88	17.05bc
Peach	19.08	18.07	16.80	17.98b
Khasi mandarin	16.75	16.07	15.27	16.03c
Guava	20.67	20.15	18.71	19.84a
Control (No tree)	17.93	17.12	15.62	16.89bc
Mean	18.58a	17.64a	16.45b	

Table 7.4 Interactive effect of land use and temperature on soil macroaggregate carbon stability ($g\ kg^{-1}$)

Land use system	Temperature (^oC)			Mean
	25	30	35	
		Agroforestry		
Champak	24.51	23.72	22.99	23.74b
Tree bean	22.25	21.47	20.47	21.40c
Alder	28.28	27.19	26.31	27.26a
Khasi pine	23.02	21.71	20.76	21.83c
Control (No tree)	21.73	20.53	19.61	20.62c
Mean	23.96a	22.92ab	22.03b	
		Agriculture		
Maize	16.20	15.13	13.81	15.05b
Potato	17.00	15.74	14.89	15.88ab
Rice	18.06	16.90	15.75	16.90a
Turmeric	16.37	15.26	14.19	15.27b
Control (No crop)	15.91	14.47	13.18	14.52b
Mean	16.71a	15.50b	14.36b	
		Horticulture		
Pear	19.90	18.93	18.05	18.96b
Peach	21.87	20.74	19.82	20.81a
Khasi mandarin	19.33	17.85	16.78	17.99b
Guava	21.75	20.83	19.86	20.81a
Control (No tree)	18.97	17.82	15.79	17.53b
Mean	20.36a	19.23ab	18.06b	

Table 7.5 Interactive effect of land use and temperature on soil microaggregate carbon stability ($g\ kg^{-1}$)

Land use system	Temperature (^oC)			Mean
	25	30	35	
	Agroforestry			
Champak	20.55	19.33	18.83	19.57b
Tree bean	19.13	18.33	17.47	18.31b
Alder	26.47	25.30	24.19	25.32a
Khasi pine	20.88	19.78	18.67	19.78b
Control (No tree)	19.83	18.92	17.92	18.89b
Mean	21.37a	20.33ab	19.41b	
	Agriculture			
Maize	15.22	14.33	13.30	14.28c
Potato	16.45	15.43	14.63	15.51b
Rice	15.63	14.72	13.80	14.72c
Turmeric	17.00	16.65	15.46	16.37a
Control (No crop)	13.67	12.83	11.70	12.73d
Mean	15.59a	14.79b	13.78c	
	Horticulture			
Pear	18.12	16.92	16.11	17.05bc
Peach	18.93	17.95	17.07	17.98b
Khasi mandarin	16.63	16.02	15.43	16.03c
Guava	20.62	20.05	18.86	19.84a
Control (No tree)	17.78	17.00	15.88	16.89bc
Mean	18.42a	17.59ab	16.67b	

Table 7.6 Interactive effect of duration and temperature on soil macroaggregate carbon stability ($g\ kg^{-1}$)

Duration (days)	Temperature (^{o}C)			Mean
	25	**30**	**35**	
		Agroforestry		
15	25.15	24.17	23.61	24.31a
30	24.15	23.45	22.55	23.38ab
60	22.57	21.16	19.93	21.22b
Mean	23.96a	22.93ab	22.03b	
		Agriculture		
15	17.56	16.70	16.14	16.80a
30	17.02	15.98	14.98	15.99ab
60	15.56	13.83	12.34	13.91b
Mean	16.71a	15.50ab	14.49b	
		Horticulture		
15	21.18	20.52	19.91	20.54a
30	20.74	19.57	18.99	19.77ab
60	19.18	17.61	15.88	17.56b
Mean	20.3a	19. 24ab	18.26b	

Table 7.7 Interactive effect of duration and temperature on soil microaggregate carbon stability ($g\ kg^{-1}$)

Duration (days)	Temperature (^oC) 25	30	35	Mean
		Agroforestry		
15	21.88	21.51	20.77	21.39a
30	21.44	20.40	19.87	20.57a
60	20.80	19.09	17.60	19.16b
Mean	21.37a	20.33ab	19.41b	
		Agriculture		
15	16.32	16.05	15.36	15.91a
30	15.54	14.70	14.11	14.78b
60	14.92	13.63	11.87	13.47c
Mean	15.59a	14.79b	13.78c	
		Horticulture		
15	19.06	18.53	17.14	18.24a
30	18.34	17.69	16.90	17.64ab
60	17.85	16.54	14.97	16.45b
Mean	18.42a	17.59ab	16.34b	

Table 7.8 Interactive effect of land use, duration and temperature on soil macroaggregate carbon stability (g kg^{-1})

Land use system	Initial	15 days			30 days			60 days		
		25 °C	30 °C	35 °C	25 °C	30 °C	35 °C	25 °C	30 °C	35 °C
Agroforestry										
Champak	26.6	25.8	25.0	24.3	24.8	24.1	23.4	22.9	22.0	21.3
Tree bean	24.1	23.4	22.6	21.9	22.5	21.7	21.0	20.9	20.0	18.5
Alder	30.8	29.8	28.4	27.9	28.3	27.8	26.8	26.7	25.3	24.2
Khasi Pine	24.7	24.2	23.1	22.7	23.2	22.6	21.2	21.7	19.5	18.4
Control (No tree)	22.9	22.5	21.7	21.2	22.0	21.0	20.4	20.7	18.9	17.2
Agriculture										
Maize	17.9	17.2	16.4	15.4	16.4	15.9	14.2	15.0	13.1	11.8
Potato	18.3	17.8	16.7	16.3	17.4	16.3	15.2	15.9	14.2	13.2
Rice	19.7	18.8	18.3	17.9	18.5	17.4	16.5	16.9	15.0	13.5
Turmeric	18.2	17.2	16.4	15.9	16.7	15.5	14.8	15.3	13.9	12.5
Control (No crop)	17.3	16.8	15.7	15.2	16.2	14.8	14.2	14.8	12.9	10.8
Horticulture										
Pear	21.4	20.6	19.8	19.3	20.3	19.3	18.9	18.7	17.7	15.9
Peach	23.3	22.8	22.1	21.7	22.2	21.6	20.7	20.6	18.5	17.0
K Mandarin	20.8	20.1	19.8	18.8	19.7	17.5	16.7	18.3	16.3	14.9
Guava	23.3	22.5	21.5	21.1	22.0	21.4	20.8	20.8	19.6	17.7
Control (No tree)	20.3	19.9	19.4	18.6	19.5	18.1	17.8	17.6	16.0	14.0

Table 7.9 Interactive effect of land use, duration and temperature on soil microaggregate carbon stability (g kg^{-1})

Land use system	Initial	15 days			30 days			60 days		
		25	30	35	25	30	35	25	30	35
					Agroforestry					
Champak	21.5	21.1	20.1	19.4	20.6	19.4	19.2	20.1	18.5	17.9
Tree bean	20.3	19.5	19.6	18.5	19.4	18.5	18.4	18.5	17.0	15.6
Alder	27.2	26.8	26.4	25.8	26.6	25.4	24.7	26.1	24.1	22.1
Khasi Pine	21.8	21.4	21.1	20.3	21.0	19.9	19.0	20.3	18.4	16.7
Control (No tree)	21.2	20.7	20.4	19.9	19.7	18.9	18.1	19.1	17.5	15.8
					Agriculture					
Maize	16.6	16.2	15.5	15.0	15.2	14.3	13.4	14.3	13.2	11.5
Potato	17.6	17.3	16.8	16.0	16.4	15.2	14.8	15.7	14.3	13.1
Rice	16.8	16.5	16.1	15.1	15.5	14.7	13.9	14.9	13.4	12.4
Turmeric	17.7	17.4	18.2	17.4	17.1	16.5	16.3	16.5	15.3	12.7
Control (No crop)	14.7	14.3	13.7	13.3	13.5	12.8	12.2	13.2	12.0	9.6
					Horticulture					
Pear	19.6	19.2	18.2	18.1	17.9	16.7	15.8	17.3	15.9	14.4
Peach	20.1	19.7	19.1	18.6	18.9	18.1	17.3	18.3	16.7	15.4
Khasi Mandarin	17.5	17.1	16.7	16.5	16.6	16.2	15.5	16.3	15.2	14.4
Guava	21.4	21.1	20.8	20.2	20.7	20.4	19.4	20.1	19.1	17.0
Control (No tree)	18.8	18.4	18.0	17.5	17.7	17.2	16.5	17.3	15.9	13.7

Table 7.10 Effect of different land use systems and temperature on rate constant, activation energy and Q_{10} values of macroaggregate carbon loss

Land use system	Rate constant (k)			AE	Q_{10}
	Temperature (^{o}C)			(kJ mol^{-1})	
	25	30	35		
			Agroforestry		
Champak	0.0005	0.0011	0.0013	24.25	0.955
Tree bean	0.0006	0.0013	0.0018	54.27	0.905
Alder	0.0003	0.0009	0.0015	37.00	0.932
Khasi Pine	0.0005	0.0013	0.0019	74.97	0.868
Control (No tree)	0.0008	0.0014	0.0022	110.51	0.815
			Agriculture		
Maize	0.0011	0.0016	0.0026	66.24	0.879
Potato	0.0009	0.0016	0.0021	88.88	0.847
Rice	0.0009	0.0017	0.0022	76.37	0.863
Turmeric	0.0005	0.0012	0.0025	57.64	0.894
Control (No crop)	0.0008	0.0014	0.0031	83.96	0.849
			Horticulture		
Pear	0.0009	0.0015	0.0022	88.62	0.847
Peach	0.0007	0.0013	0.0019	88.88	0.847
K Mandarin	0.0005	0.0010	0.0014	89.08	0.847
Guava	0.0004	0.0008	0.0017	72.75	0.874
Control (No tree)	0.0006	0.0012	0.0023	88.08	0.845

Table 7.11 Effect of different land use systems and temperature on rate constant, activation energy and Q_{10} values of microaggregate carbon loss

Land use system	**Rate constant (k)**			**AE**	**Q_{10}**
	Temperature (^{o}C)			**(kJ mol^{-1})**	
	25	**30**	**35**		
		Agroforestry			
Champak	0.0008	0.0009	0.0011	73.13	0.874
Tree bean	0.0007	0.0009	0.0014	83.97	0.852
Alder	0.0008	0.0010	0.0013	122.99	0.802
Khasi Pine	0.0006	0.0012	0.0016	102.06	0.824
Control (No tree)	0.0004	0.0010	0.0017	77.19	0.858
		Agriculture			
Maize	0.0008	0.0017	0.0019	65.56	0.874
Potato	0.0006	0.0011	0.0016	64.74	0.879
Rice	0.0007	0.0015	0.0019	68.32	0.873
Turmeric	0.0008	0.0014	0.0017	122.81	0.788
Control (No crop)	0.0007	0.0015	0.0021	103.21	0.810
		Horticulture			
Pear	0.0005	0.0008	0.0016	68.22	0.873
Peach	0.0005	0.0011	0.0016	76.25	0.863
K Mandarin	0.0005	0.0014	0.0016	78.67	0.865
Guava	0.0005	0.0007	0.0013	110.32	0.815
Control (No tree)	0.0006	0.0013	0.0019	102.50	0.819

Table 7.12 Interactive effect of land use and duration on soil macroaggregate nitrogen stability ($g\ kg^{-1}$)

Land use system	Duration (days)			Mean
	15	30	60	
	Agroforestry			
Champak	2.287	2.158	1.980	2.142b
Tree bean	2.200	2.140	2.019	2.120b
Alder	2.491	2.350	2.183	2.341a
Khasi pine	2.128	2.030	1.920	2.026b
Control (No tree)	1.946	1.857	1.684	1.829c
Mean	2.210a	2.107a	1.957b	
	Agriculture			
Maize	1.801	1.690	1.470	1.654b
Potato	2.001	1.917	1.793	1.904a
Rice	1.994	1.869	1.730	1.864a
Turmeric	1.974	1.834	1.706	1.838a
Control (No crop)	1.794	1.716	1.479	1.663b
Mean	1.913a	1.805b	1.636c	
	Horticulture			
Pear	1.838	1.740	1.591	1.723b
Peach	2.060	1.917	1.803	1.927a
Khasi mandarin	1.988	1.860	1.750	1.866a
Guava	2.121	1.990	1.850	1.987a
Control (No tree)	1.818	1.658	1.527	1.668b
Mean	1.965a	1.833b	1.704c	

Table 7.13 Interactive effect of land use and duration on soil microaggregate nitrogen stability ($g\ kg^{-1}$)

Land use system	Duration (days)			Mean
	15	30	60	
	Agroforestry			
Champak	2.038	1.947	1.867	1.951ab
Tree bean	1.760	1.613	1.556	1.643c
Alder	2.090	2.010	1.939	2.013a
Khasi pine	1.917	1.770	1.729	1.805b
Control (No tree)	1.700	1.566	1.481	1.582c
Mean	1.901a	1.781b	1.714b	
	Agriculture			
Maize	1.550	1.398	1.338	1.429c
Potato	1.680	1.531	1.463	1.558b
Rice	1.837	1.690	1.576	1.701a
Turmeric	1.793	1.597	1.503	1.631ab
Control (No crop)	1.514	1.459	1.304	1.426c
Mean	1.675a	1.535b	1.437c	
	Horticulture			
Pear	1.733	1.643	1.549	1.642b
Peach	1.907	1.727	1.638	1.757a
Khasi mandarin	1.923	1.789	1.638	1.783a
Guava	1.917	1.743	1.699	1.786a
Control (No tree)	1.740	1.617	1.441	1.599b
Mean	1.844a	1.704b	1.593c	

Table 7.14 Interactive effect of land use and temperature on soil macroaggregate nitrogen stability (g kg^{-1})

Land use system	Temperature (^{o}C)			Mean
	25	30	35	
	Agroforestry			
Champak	2.262	2.136	2.027	2.142b
Tree bean	2.200	2.137	2.022	2.120b
Alder	2.454	2.320	2.250	2.341a
Khasi pine	2.114	2.010	1.953	2.026b
Control (No tree)	1.932	1.823	1.731	1.829c
Mean	2.192a	2.085a	1.997b	
	Agriculture			
Maize	1.744	1.657	1.560	1.654b
Potato	1.984	1.907	1.820	1.904a
Rice	1.960	1.867	1.767	1.865a
Turmeric	1.930	1.839	1.746	1.838a
Control (No crop)	1.761	1.672	1.556	1.663b
Mean	1.876a	1.788b	1.690c	
	Horticulture			
Pear	1.841	1.727	1.601	1.723b
Peach	2.043	1.930	1.807	1.927a
Khasi mandarin	1.998	1.873	1.727	1.866a
Guava	2.118	1.990	1.853	1.987a
Control (No tree)	1.811	1.658	1.533	1.667b
Mean	1.962a	1.836b	1.704c	

Table 7.15 Interactive effect of land use and temperature on soil microaggregate nitrogen stability (g kg^{-1})

Land use system	Temperature (°C)			Mean
	25	30	35	
	Agroforestry			
Champak	2.059	1.914	1.778	1.917ab
Tree bean	1.771	1.640	1.520	1.644c
Alder	2.180	1.990	1.870	2.013a
Khasi pine	1.950	1.790	1.680	1.807b
Control (No tree)	1.730	1.560	1.460	1.583c
Mean	1.938a	1.779b	1.662c	
	Agriculture			
Maize	1.570	1.410	1.310	1.430c
Potato	1.670	1.560	1.440	1.557b
Rice	1.820	1.690	1.590	1.700a
Turmeric	1.770	1.620	1.500	1.630ab
Control (No crop)	1.570	1.420	1.280	1.423c
Mean	1.680a	1.540b	1.424c	
	Horticulture			
Pear	1.760	1.640	1.530	1.643b
Peach	1.900	1.740	1.630	1.757a
Khasi mandarin	1.910	1.790	1.660	1.787a
Guava	1.910	1.770	1.670	1.783a
Control (No tree)	1.730	1.610	1.460	1.600b
Mean	1.842a	1.710b	1.590c	

Table 7.16 Interactive effect of duration and temperature on soil macroaggregate nitrogen stability (g kg^{-1})

Duration (days)	Temperature (^{o}C)			Mean
	25	30	35	
		Agroforestry		
15	2.295	2.188	2.148	2.210a
30	2.194	2.095	2.031	2.107a
60	2.089	1.972	1.811	1.957b
Mean	2.193a	2.085ab	1.997b	
		Agriculture		
15	1.993	1.904	1.842	1.913a
30	1.873	1.811	1.732	1.805b
60	1.762	1.650	1.495	1.636c
Mean	1.876a	1.788a	1.690b	
		Horticulture		
15	2.079	1.961	1.855	1.965a
30	1.957	1.837	1.705	1.833b
60	1.851	1.709	1.553	1.704c
Mean	1.962a	1.836b	1.704c	

Table 7.17 Interactive effect of duration and temperature on soil microaggregate nitrogen stability (g kg^{-1})

Duration (days)	Temperature (^{o}C)			Mean
	25	30	35	
		Agroforestry		
15	2.029	1.884	1.789	1.901a
30	1.889	1.762	1.633	1.761b
60	1.890	1.690	1.560	1.713b
Mean	1.936a	1.779b	1.661c	
		Agriculture		
15	1.800	1.657	1.568	1.675a
30	1.651	1.535	1.418	1.535b
60	1.590	1.440	1.280	1.437c
Mean	1.680a	1.544b	1.422c	
		Horticulture		
15	1.946	1.836	1.750	1.844a
30	1.818	1.710	1.583	1.704b
60	1.760	1.580	1.440	1.593c
Mean	1.841a	1.709b	1.591c	

Table 7.18 Interactive effect of land use, duration and temperature on soil macroaggregate nitrogen stability (g kg^{-1})

Land use system	Initial	15 days			30 days			60 days		
		25 °C	30 °C	35 °C	25 °C	30 °C	35 °C	25 °C	30 °C	35 °C
Agroforestry										
Champak	2.40	2.36	2.28	2.21	2.29	2.15	2.03	2.13	1.97	1.84
Tree bean	2.29	2.27	2.20	2.13	2.19	2.13	2.09	2.15	2.07	1.83
Alder	2.67	2.62	2.44	2.42	2.44	2.32	2.30	2.31	2.21	2.04
Khasi Pine	2.27	2.21	2.08	2.08	2.11	2.02	1.94	2.01	1.91	1.82
Control (No tree)	2.05	2.01	1.93	1.88	1.93	1.84	1.78	1.84	1.68	1.52
Agriculture										
Maize	1.90	1.87	1.79	1.73	1.76	1.69	1.60	1.60	1.48	1.34
Potato	2.10	2.07	1.99	1.93	1.97	1.91	1.85	1.90	1.80	1.66
Rice	2.12	2.08	1.98	1.91	1.92	1.88	1.80	1.87	1.73	1.58
Turmeric	2.08	2.05	1.96	1.87	1.85	1.81	1.75	1.72	1.70	1.55
Control (No crop)	1.96	1.89	1.77	1.71	1.77	1.72	1.64	1.61	1.51	1.31
Horticulture										
Pear	2.02	1.95	1.83	1.72	1.83	1.74	1.63	1.73	1.59	1.44
Peach	2.23	2.14	2.05	1.97	2.03	1.91	1.79	1.94	1.81	1.64
K Mandarin	2.15	2.09	1.98	1.88	1.99	1.86	1.71	1.90	1.76	1.57
Guava	2.27	2.24	2.12	2.00	2.11	1.99	1.86	2.00	1.85	1.69
Control (No tree)	2.03	1.97	1.80	1.67	1.80	1.65	1.51	1.65	1.51	1.41

Table 7.19 Interactive effect of land use, duration and temperature on soil microaggregate nitrogen stability (g kg^{-1})

Land use system	Initial	15 days			30 days			60 days		
		25 ^{o}C	30 ^{o}C	35 ^{o}C	25 ^{o}C	30 ^{o}C	35 ^{o}C	25 ^{o}C	30 ^{o}C	35 ^{o}C
					Agroforestry					
Champak	2.18	2.15	2.03	1.93	2.01	1.84	1.69	2.01	1.87	1.71
Tree bean	1.93	1.87	1.76	1.65	1.72	1.62	1.49	1.71	1.54	1.41
Alder	2.35	2.29	2.04	1.94	2.13	2.01	1.89	2.11	1.92	1.79
Khasi Pine	2.07	2.02	1.90	1.83	1.90	1.77	1.64	1.92	1.69	1.58
Control (No tree)	1.85	1.81	1.69	1.60	1.68	1.57	1.44	1.69	1.42	1.33
					Agriculture					
Maize	1.74	1.69	1.55	1.41	1.53	1.37	1.29	1.48	1.31	1.22
Potato	1.80	1.78	1.68	1.58	1.64	1.54	1.41	1.60	1.48	1.31
Rice	1.96	1.94	1.82	1.75	1.80	1.69	1.58	1.72	1.57	1.43
Turmeric	1.92	1.90	1.78	1.70	1.72	1.60	1.47	1.68	1.49	1.34
Control (No crop)	1.71	1.69	1.46	1.39	1.57	1.48	1.33	1.46	1.34	1.11
					Horticulture					
Pear	1.87	1.84	1.74	1.62	1.76	1.63	1.54	1.67	1.54	1.44
Peach	2.04	2.01	1.89	1.82	1.84	1.74	1.60	1.84	1.59	1.48
Khasi Mandarin	2.03	2.02	1.91	1.84	1.90	1.80	1.67	1.80	1.65	1.47
Guava	2.06	2.01	1.91	1.83	1.87	1.75	1.61	1.86	1.66	1.58
Control (No tree)	1.86	1.85	1.73	1.64	1.72	1.63	1.50	1.61	1.48	1.23

Table 7.20 Effect of different land use systems and temperature on rate constant, activation energy and Q_{10} values of macroaggregate nitrogen loss

Land use system	Rate constant (k) Temperature (°C) 25	30	35	AE (kJ mol^{-1})	Q_{10}
		Agroforestry			
Champak	0.0010	0.0017	0.0027	57.04	0.893
Tree bean	0.0018	0.0024	0.0033	83.61	0.855
Alder	0.0012	0.0016	0.0022	41.59	0.920
Khasi Pine	0.0010	0.0014	0.0021	38.91	0.927
Control (No tree)	0.0012	0.0018	0.0029	73.66	0.865
		Agriculture			
Maize	0.0017	0.0028	0.0032	49.87	0.902
Potato	0.0011	0.0018	0.0028	63.07	0.886
Rice	0.0011	0.0016	0.0024	52.85	0.900
Turmeric	0.0016	0.0022	0.0030	25.57	0.948
Control (No crop)	0.0011	0.0015	0.0032	52.78	0.895
		Horticulture			
Pear	0.0008	0.0014	0.0024	56.26	0.892
Peach	0.0017	0.0026	0.0038	56.61	0.893
K Mandarin	0.0009	0.0015	0.0023	68.16	0.873
Guava	0.0012	0.0016	0.0024	64.64	0.879
Control (No tree)	0.0012	0.0016	0.0031	39.02	0.921

Table 7.21 Effect of different land use systems and temperature on rate constant, activation energy and Q_{10} values of microaggregate nitrogen loss

Land use system	Rate constant (k) Temperature (°C) 25	30	35	AE (kJ mol^{-1})	Q_{10}
	Agroforestry				
Champak	0.0009	0.0014	0.0019	75.78	0.856
Tree bean	0.0005	0.0007	0.0015	46.21	0.904
Alder	0.0011	0.0013	0.0019	46.21	0.910
Khasi Pine	0.0009	0.0011	0.0015	56.55	0.893
Control (No tree)	0.0008	0.0014	0.0021	67.26	0.869
	Agriculture				
Maize	0.0013	0.0018	0.0025	48.39	0.901
Potato	0.0007	0.0011	0.0016	71.27	0.863
Rice	0.0010	0.0014	0.0020	59.48	0.885
Turmeric	0.0015	0.0016	0.0021	47.94	0.902
Control (No crop)	0.0014	0.0018	0.0028	81.25	0.843
	Horticulture				
Pear	0.0011	0.0016	0.0023	83.80	0.846
Peach	0.0010	0.0015	0.0021	61.36	0.874
K Mandarin	0.0009	0.0014	0.0022	71.59	0.866
Guava	0.0009	0.0014	0.0021	52.81	0.897
Control (No tree)	0.0015	0.0021	0.0025	72.22	0.859

Table 7.22 Effect of various tree and crop species on total polysaccharides and dilute acid extractable polysaccharides (DAEP) (g 100g^{-1}) content in bulk soil, macro and microaggregates

Land use system	Total polysaccharides (g 100g^{-1})			DAEP (g 100g^{-1})		
	Soil	Aggregates		Soil	Aggregates	
		>250 µm	<250 µm		>250 µm	<250 µm
Agroforestry						
Champak	1.49ab	1.76a	1.41a	1.26ab	1.34a	1.12a
Tree bean	1.53ab	1.97a	1.36ab	1.23ab	1.42a	1.07a
Alder	1.74a	1.95a	1.53a	1.32a	1.50a	1.24a
Khasi pine	1.41bc	1.74ab	1.38ab	0.98b	1.53a	1.09a
Control (No tree)	1.17c	1.49b	1.14b	1.03ab	1.27a	0.91a
Mean	1.47	1.78	1.36	1.16	1.41	1.09
Agriculture						
Maize	1.11ab	1.23b	1.44a	1.15a	1.44a	1.31a
Potato	1.15ab	1.45b	1.23ab	1.22a	1.01b	0.94ab
Rice	1.35a	1.47b	1.42a	0.80ab	1.29ab	1.04ab
Turmeric	1.18ab	1.85a	1.03b	1.10ab	1.34ab	0.78ab
Control (No crop)	1.08ab	1.39b	0.98b	0.69b	1.07b	0.71b
Mean	1.18	1.48	1.22	0.99	1.23	0.96
Horticulture						
Pear	1.44ab	1.58a	1.17c	0.94b	1.32a	1.04ab
Peach	1.46a	1.72a	1.24b	0.84b	1.54a	1.13ab
Khasi mandarin	1.32ab	1.58a	1.26b	1.15ab	1.36a	1.06ab
Guava	1.44ab	1.67a	1.59a	1.27a	1.69a	1.51a
Control (No tree)	1.22b	1.48a	1.14c	0.83b	1.13a	0.74b
Mean	1.38	1.60	1.28	1.01	1.41	1.09

Table 7.23 Effect of various tree and crop species on total glomalin content (g $100g^{-1}$) in bulk soil, macro and microaggregates

Land use system	Soil	Aggregates		Ratio of TG to OC		
		>250 µm	<250 µm	Soil	>250 µm	<250 µm
			Agroforestry			
Champak	0.26ab	0.35b	0.28a	0.10	0.13	0.13
Tree bean	0.22ab	0.39b	0.31a	0.09	0.16	0.15
Alder	0.31a	0.48a	0.26a	0.10	0.16	0.09
Khasi pine	0.22ab	0.34b	0.35a	0.10	0.14	0.16
Control (No tree)	0.19b	0.30b	0.22a	0.09	0.13	0.10
Mean	0.24	0.37	0.28	0.10	0.14	0.13
			Agriculture			
Maize	0.19ab	0.31b	0.21ab	0.11	0.17	0.13
Potato	0.22ab	0.37a	0.18ab	0.12	0.20	0.10
Rice	0.18ab	0.29b	0.20ab	0.11	0.15	0.12
Turmeric	0.25a	0.31b	0.24a	0.14	0.17	0.14
Control (No crop)	0.16b	0.23c	0.15b	0.11	0.13	0.10
Mean	0.20	0.30	0.20	0.12	0.16	0.12
			Horticulture			
Pear	0.21a	0.39a	0.31a	0.11	0.18	0.16
Peach	0.22a	0.37a	0.24b	0.12	0.16	0.12
Khasi mandarin	0.22a	0.30a	0.32a	0.12	0.14	0.18
Guava	0.25a	0.48a	0.25b	0.13	0.20	0.12
Control (No tree)	0.16b	0.28a	0.17c	0.10	0.14	0.09
Mean	0.21	0.36	0.26	0.12	0.17	0.13

Table 7.24 Effects of land use, duration, temperature and their interaction on aggregate carbon and nitrogen contents

Factors	Agroforestry		Agriculture		Horticulture	
	Aggregates carbon		Aggregates carbon		Aggregates carbon	
	>250 µm	<250 µm	>250 µm	<250 µm	>250 µm	<250 µm
Land use (LUS)	**	**	NS	**	**	**
Duration (D)	*	**	NS	**	*	**
Temperature (T)	*	**	*	**	*	*
LUS x D	NS	NS	NS	NS	NS	NS
LUS x T	NS	NS	NS	NS	NS	NS
D x T	NS	NS	NS	NS	NS	NS
LUS x D x T	NS	NS	NS	NS	NS	NS
			Aggregate nitrogen			
Land use (LUS)	**	**	**	**	**	**
Duration (D)	**	**	**	**	**	**
Temperature (T)	**	**	**	**	**	**
LUS x D	NS	NS	NS	NS	NS	NS
LUS x T	NS	NS	NS	NS	NS	NS
D x T	NS	NS	NS	NS	NS	NS
LUS x D x T	NS	NS	NS	NS	NS	NS

Table 7.25 Pearson's correlation matrix between MWD, macro and microaggregate carbon, and soil properties

Variables	MWD	Soil OC	MAC-C	MIC-C	Soil N	MAC-N	MIC-N	MBC	Soil TP	MAC-TP	MIC-TP	Soil DP	MAC-DP	MIC DP	Soil G	MAC G	MIC G
MWD	1	0.55**	0.56**	0.43**	0.54**	0.50**	0.56**	0.57**	0.43**	0.45**	0.32*	0.28	0.27	0.15	0.43**	0.31*	0.16
Soil OC		1	0.91**	0.67**	0.48**	0.51**	0.45**	0.46**	0.48**	0.41**	0.45**	0.37*	0.35*	0.14	0.35*	0.23	0.19
MAC-C			1	0.70**	0.49**	0.58**	0.54**	0.46**	0.32*	0.37*	0.44**	0.32*	0.36*	0.15	0.39**	0.17	0.08
MIC-C				1	0.46**	0.53**	0.57**	0.47**	0.33*	0.35*	0.41**	0.38**	0.33*	0.25	0.43**	0.28	0.16
Soil N					1	0.64**	0.69**	0.55**	0.16	0.31*	0.23	0.24	0.21	0.09	0.38**	0.05	0.04
MAC-N						1	0.89**	0.33*	0.31*	0.58**	0.37*	0.31*	0.14	0.12	0.50**	0.31*	0.10
MIC-N							1	0.40**	0.27	0.43**	0.37*	0.18	0.16	0.22	0.46**	0.23	0.16
MBC								1	0.35*	0.22	0.31*	0.43**	0.08	0.04	0.35*	0.24	0.14
Soil TP									1	0.44**	0.49**	0.44**	0.24	0.16	0.55**	0.59**	0.48**
MAC TP										1	0.49**	0.42**	0.07	0.23	0.46**	0.36*	0.21
MIC TP											1	0.43**	0.38*	0.47**	0.32*	0.21	0.19
Soil DP												1	0.35*	0.22	0.50**	0.51**	0.34*
MAC DP													1	0.47**	0.21	0.13	-0.10
MIC DP														1	0.05	0.03	-0.07
Soil G															1	0.53**	0.38*
MAC G																1	0.36*
MIC G																	1

** Correlation is significant at the 0.01 level (2-tailed); *. Correlation is significant at the 0.05 level (2-tailed); BD-bulk density; OC-organic carbon; MWD-mean weight diameter; MBC-microbial biomass carbon; MAC: macroaggregate; MIC: microaggregate; C: carbon; N: nitrogen; TP: total polysaccharides; DP: dilute acid extractable polysaccharides; G: glomalin

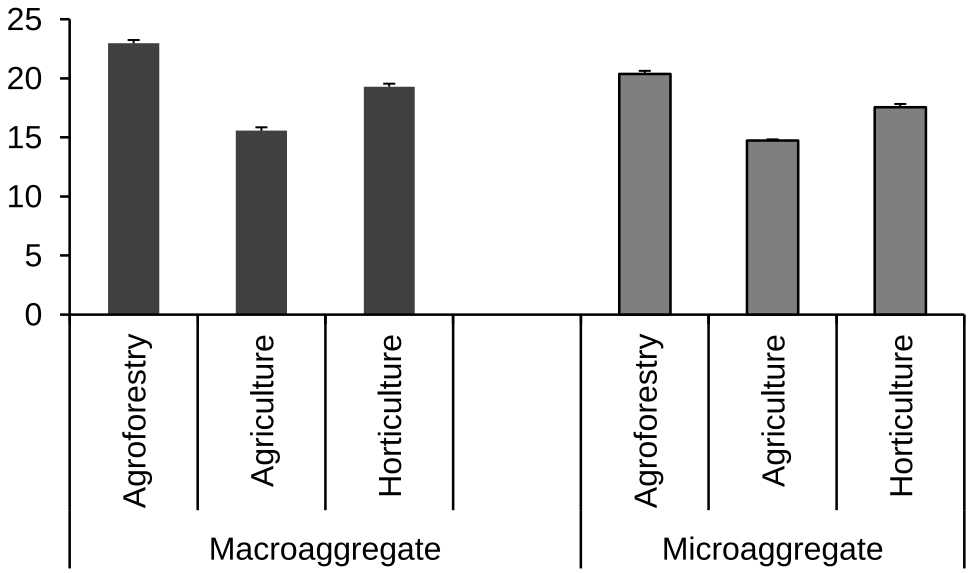


Fig. 7.1 Mean carbon content of macro- and microaggregates (g kg^{-1}) under various land use systems

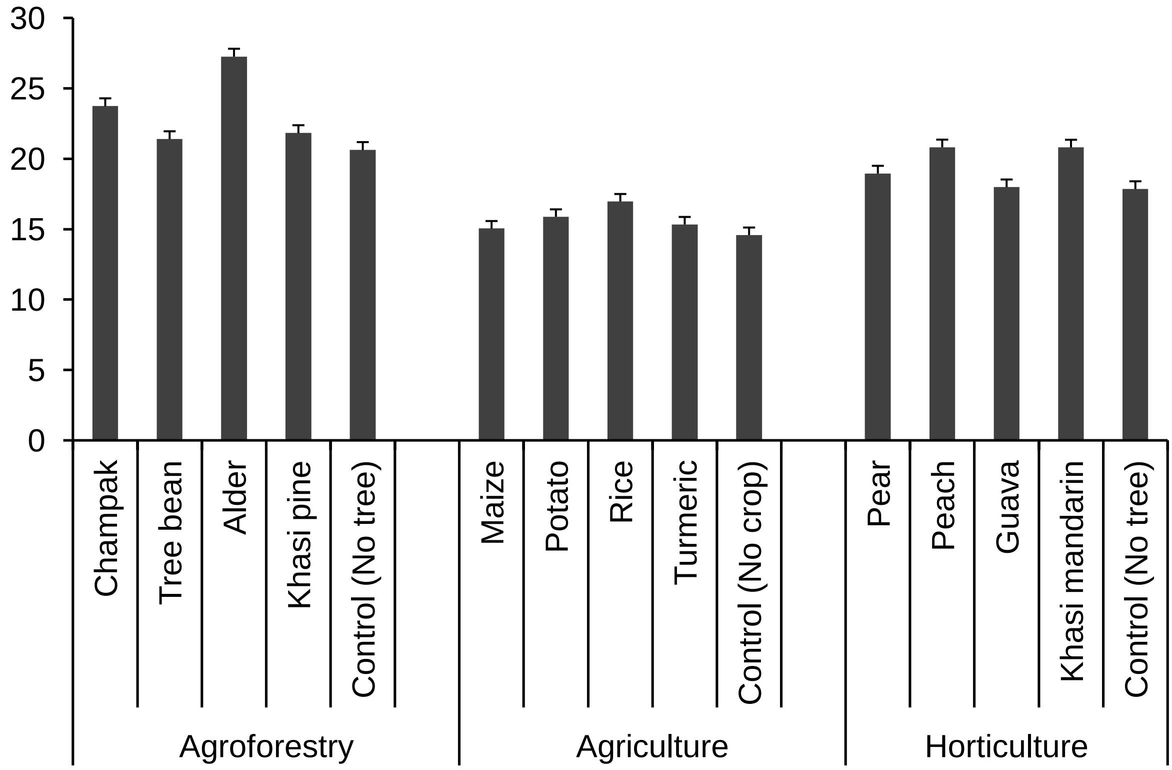


Fig. 7.2 Effect of various tree and crop species on carbon content of macroaggregates (g kg^{-1})

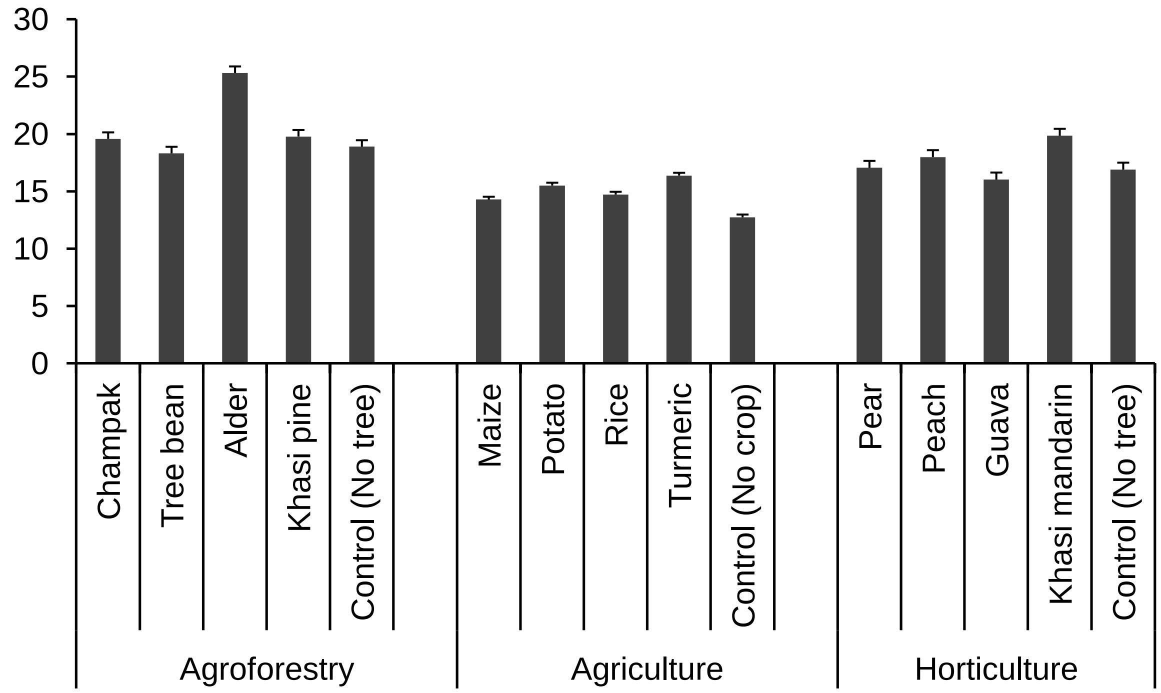


Fig. 7.3 Effect of various tree and crop species on carbon content of microaggregates (g kg^{-1})

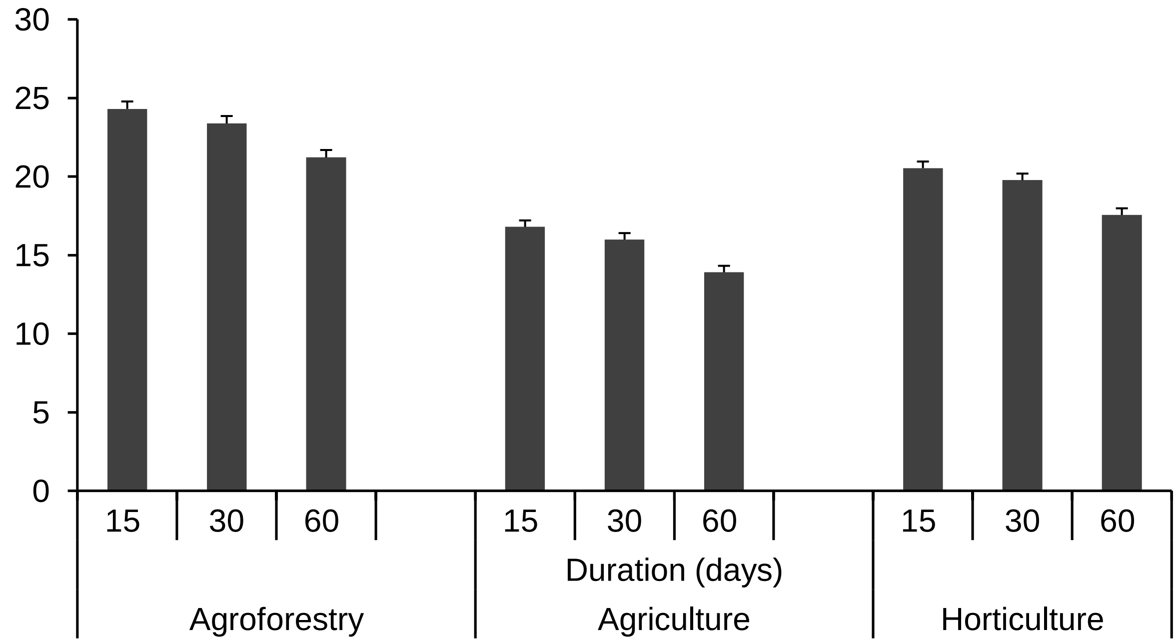


Fig. 7.4 Effect of duration on average carbon content of macroaggregates (g kg^{-1}) under various land use systems

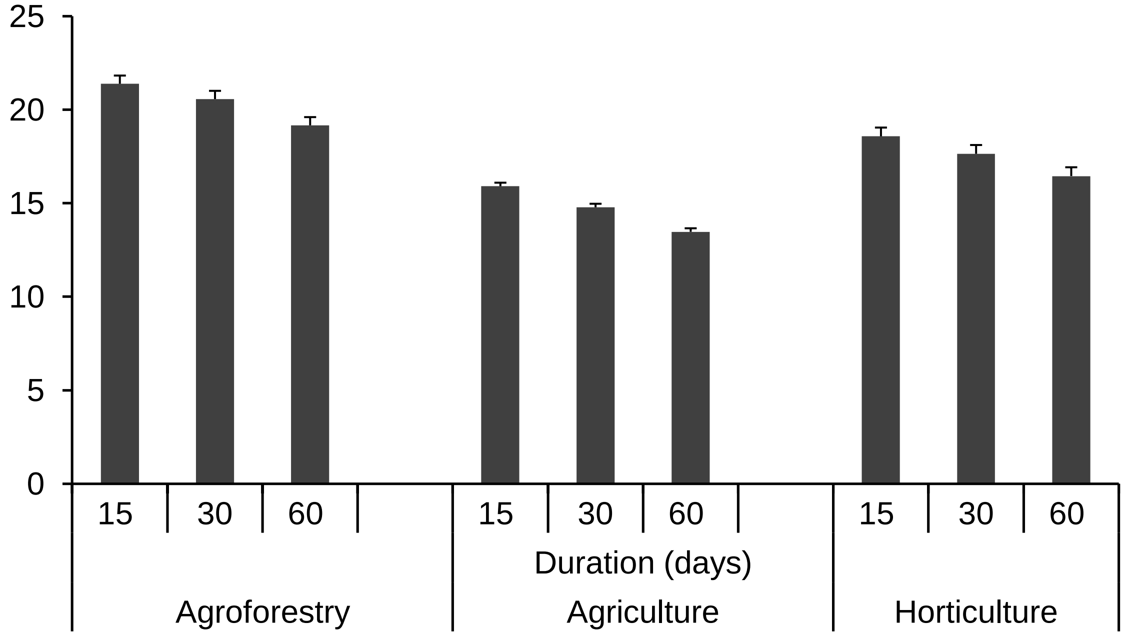


Fig. 7.5 Effect of duration on average carbon content of microaggregates (g kg^{-1}) under various land use systems

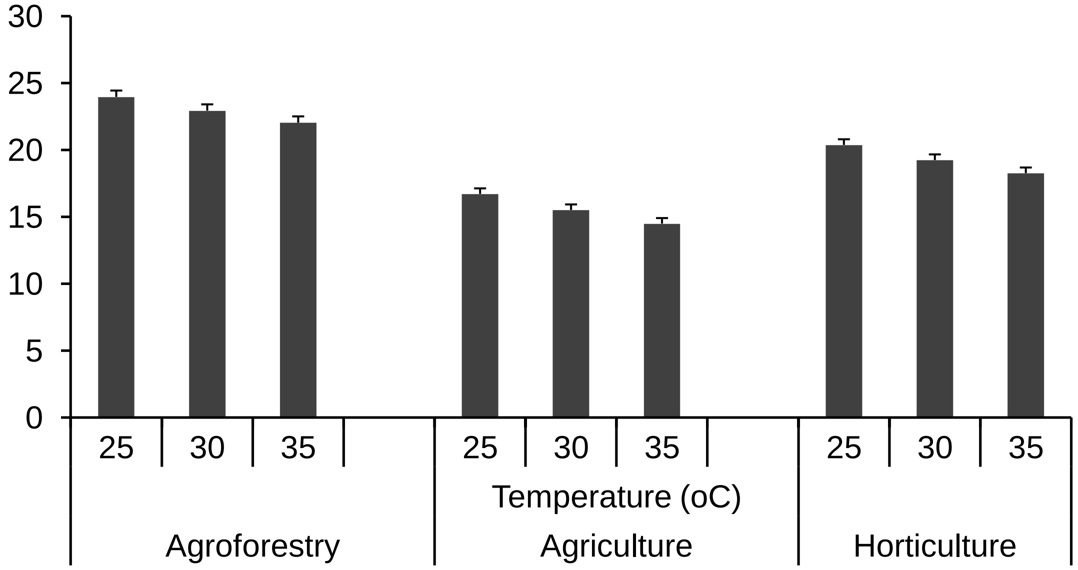


Fig. 7.6 Effect of temperature on average carbon content of macroaggregates (g kg^{-1}) under various land use systems

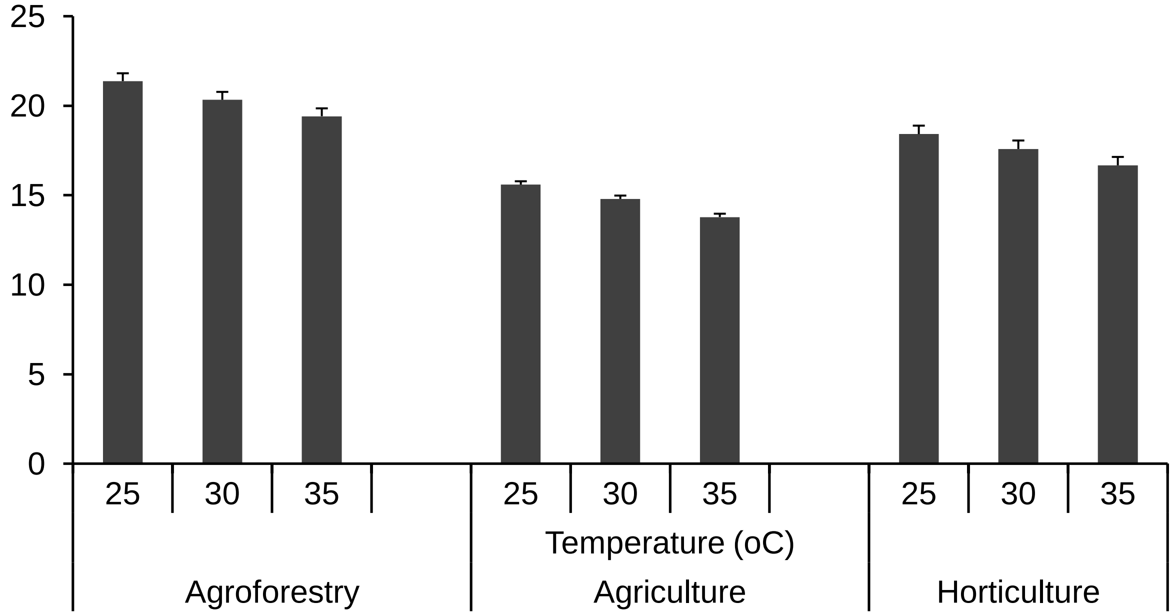


Fig. 7.7 Effect of temperature on average carbon content of microaggregates (g kg^{-1}) under various land use systems

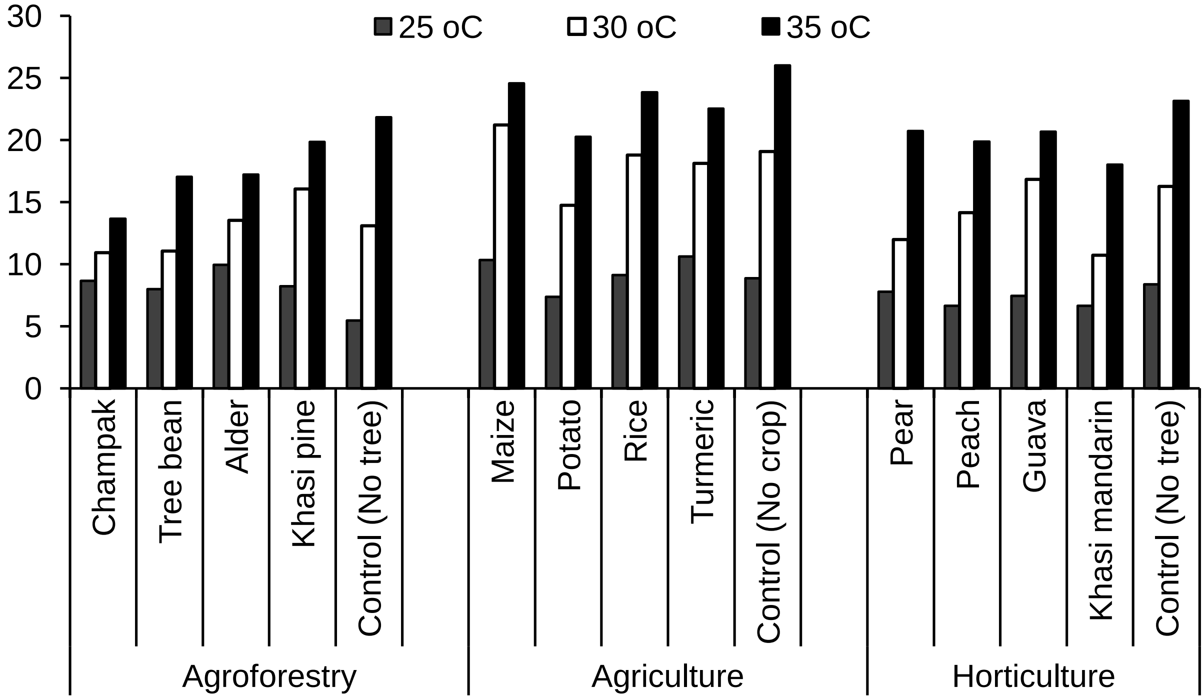


Fig.7.8 Per cent carbon mineralized from macroaggregates from different tree and crop species under various temperatures

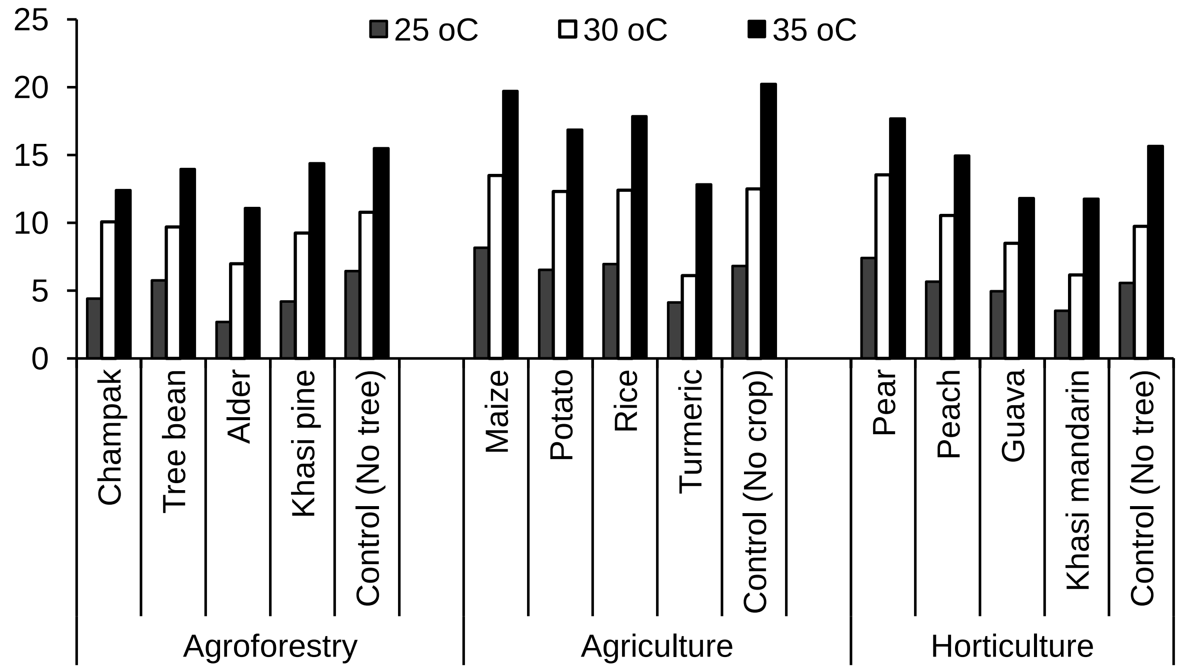


Fig.7.9 Per cent carbon mineralized from microaggregates of different tree and crop species under various temperatures

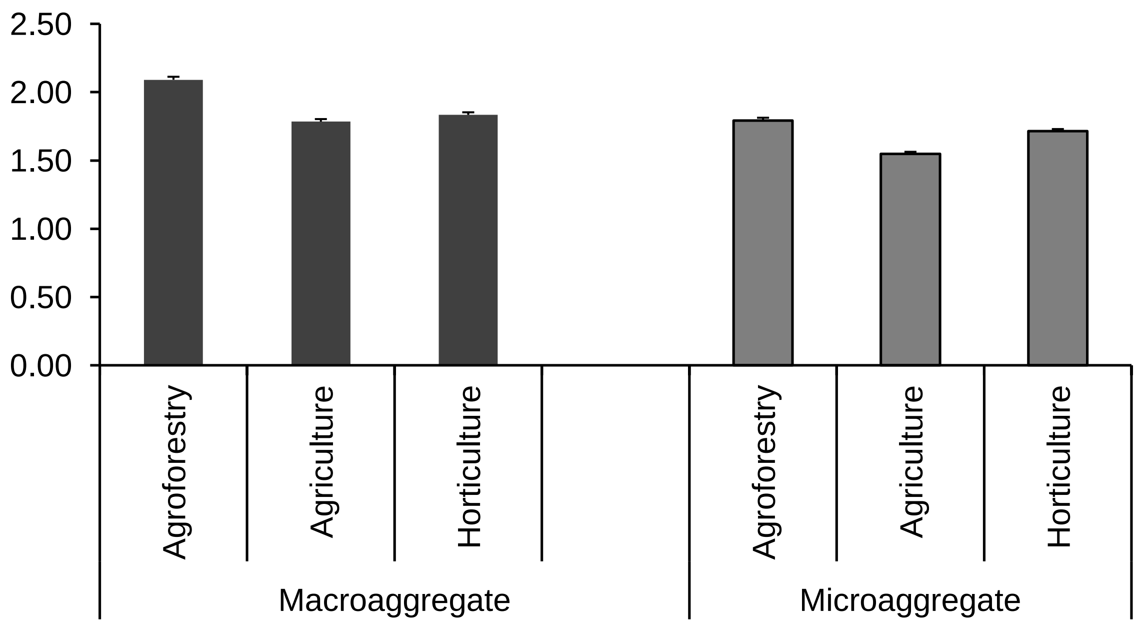


Fig. 7.10 Mean nitrogen content of macro- and microaggregates (g kg^{-1}) under various land use systems

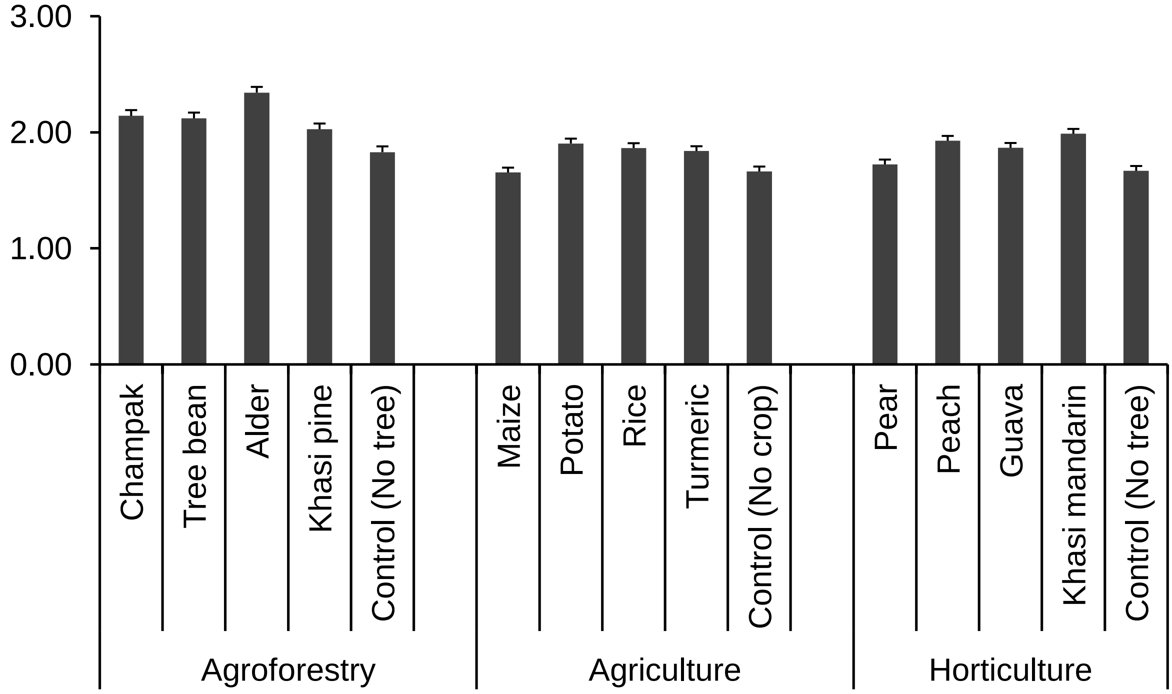


Fig. 7.11 Effect of various tree and crop species on nitrogen content of macroaggregates (g kg^{-1})

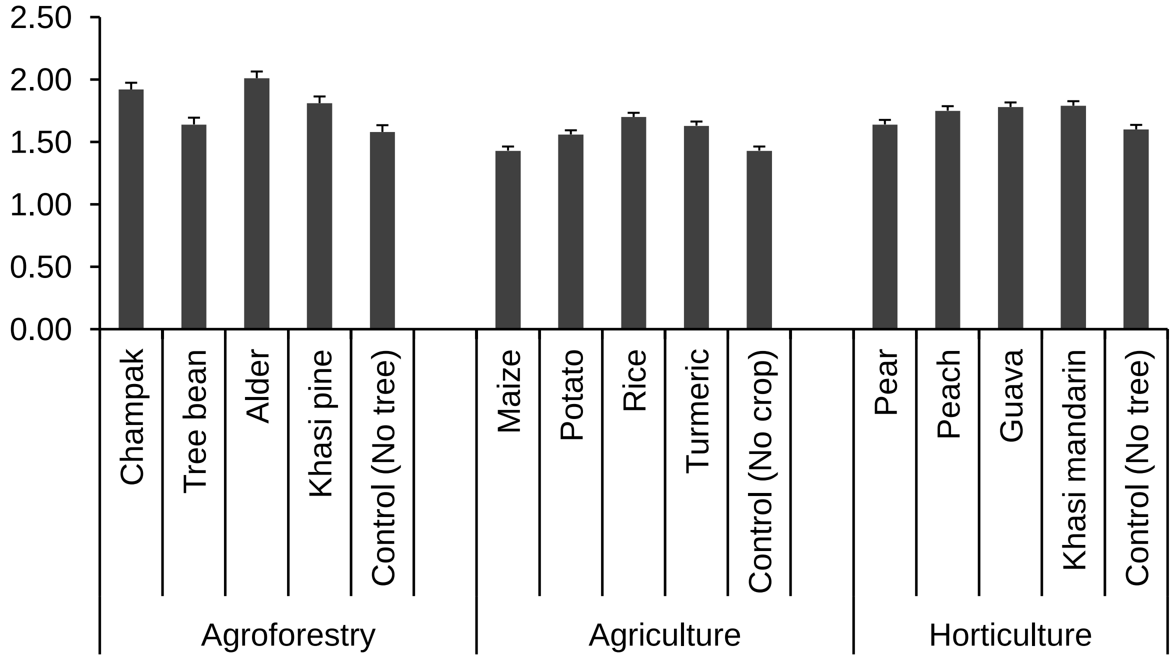


Fig. 7.12 Effect of various tree and crop species on nitrogen content of microaggregates (g kg^{-1})

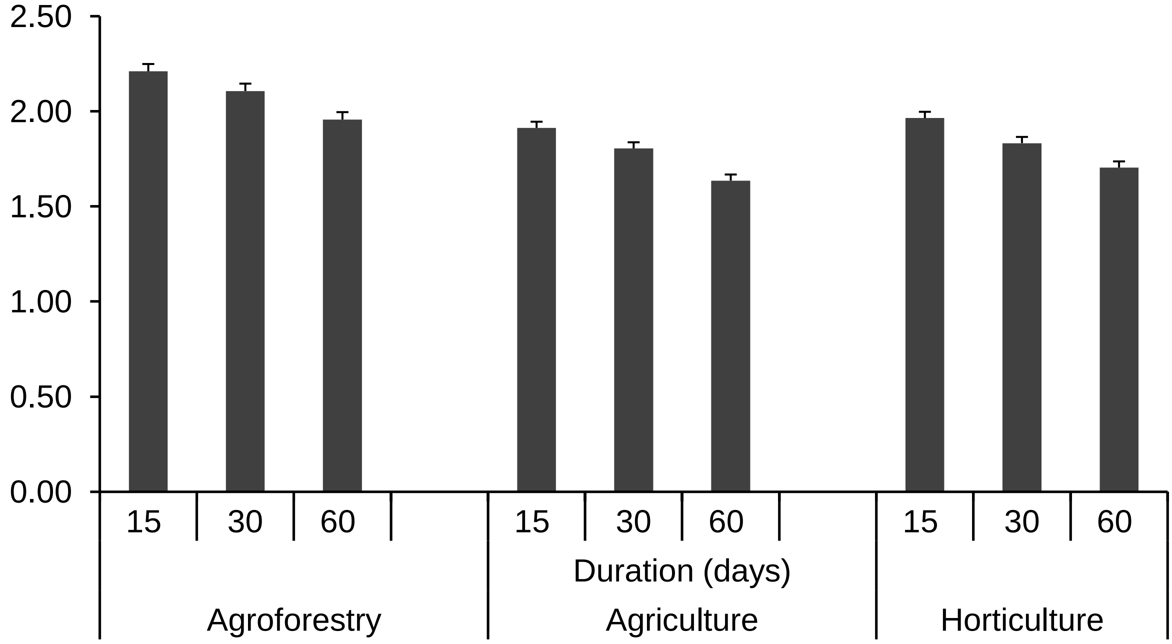


Fig. 7.13 Effect of duration on average nitrogen content of macroaggregates (g kg^{-1}) under various land use systems

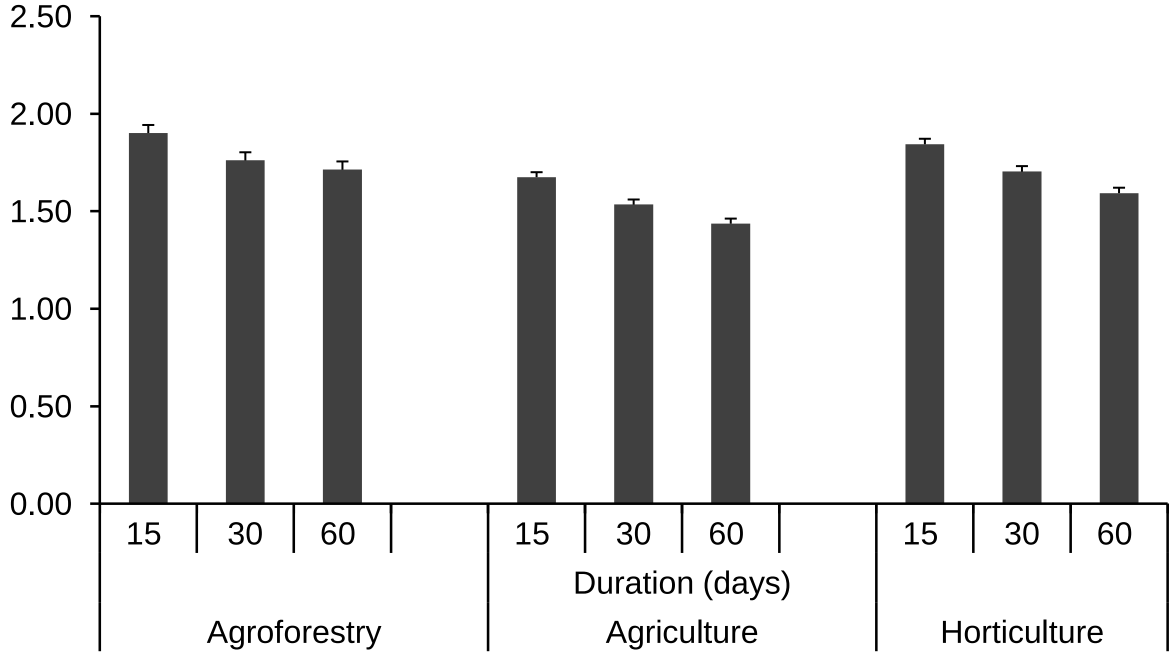


Fig. 7.14 Effect of duration on average nitrogen content of microaggregates (g kg^{-1}) under various land use systems

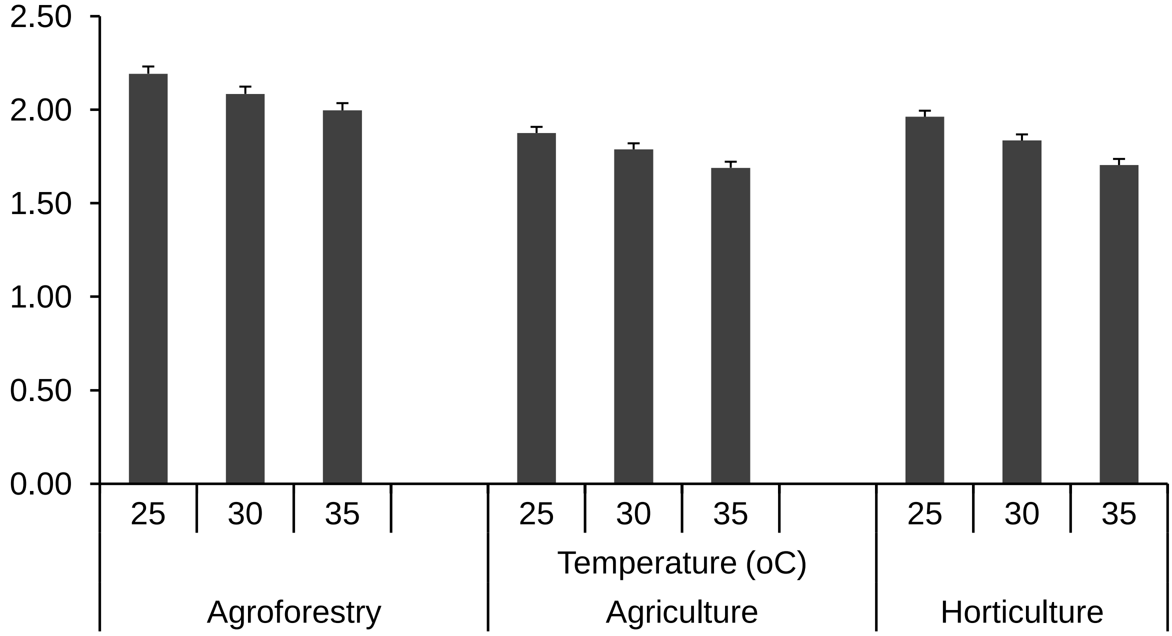


Fig. 7.15 Effect of temperature on average nitrogen content of macroaggregates (g kg^{-1}) under various land use systems

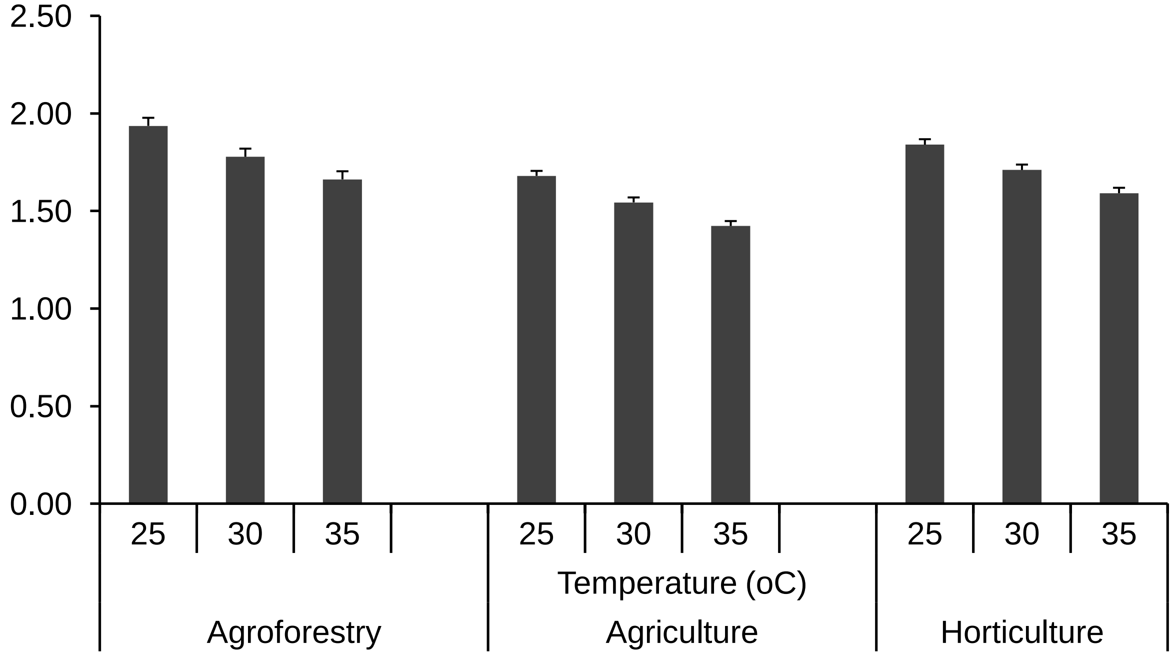


Fig. 7.16 Effect of temperature on average nitrogen content of microaggregates (g kg^{-1}) under various land use systems

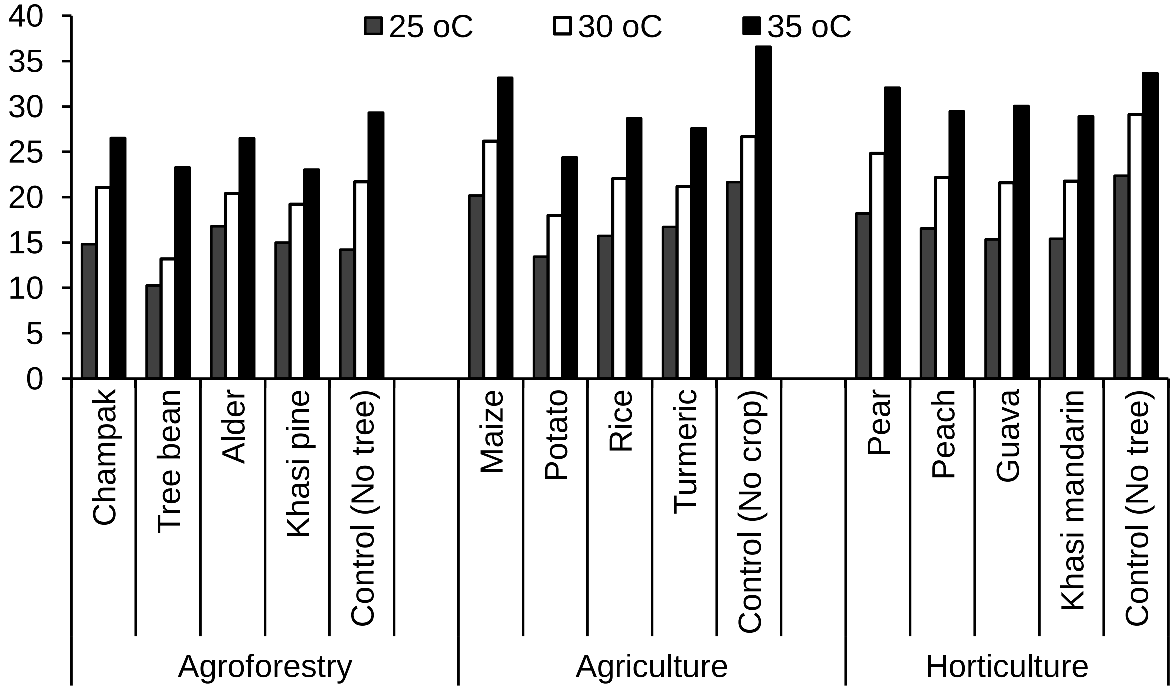


Fig.7.17 Per cent nitrogen mineralized from macroaggregates from different tree and crop species under various temperatures

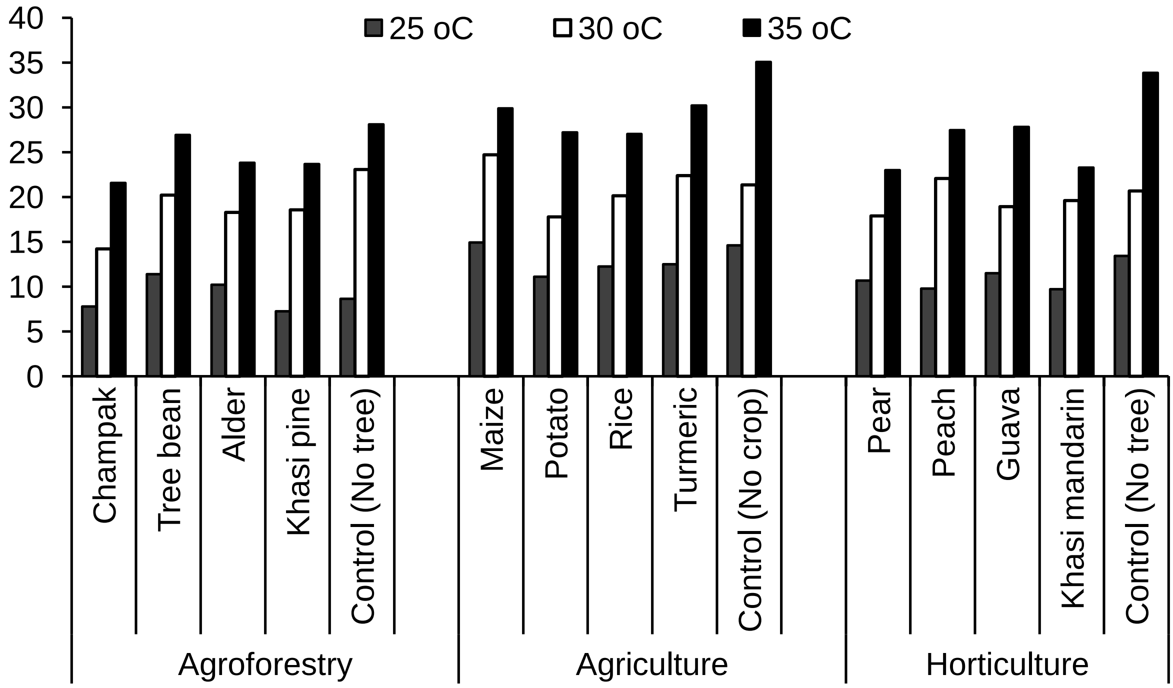


Fig.7.18 Per cent nitrogen mineralized from microaggregates of different tree and crop species under various temperatures

CHAPTER VIII

RESEARCH PAPER- V

Assessment of carbon stability mechanisms under different land use systems in East Khasi hills of Meghalaya – by CO_2 efflux study

8.1 Abstract

This study investigated the effects of land conversion from fallow to agriculture, horticulture and agriculture land use systems on soil CO_2 efflux in the East Khasi hills of Meghalaya wherein the *Jhuming* is a common farming practice. Results of the 150 days incubation experiment showed that soil CO_2 efflux ranged from 1.63 to 2.01 g 100g^{-1}, 1.62 to 1.94, 1.64 to 1.97 g 100g^{-1} soil in agroforestry, agriculture and horticulture land uses, respectively. Highest increase of 14.6% under agroforestry land use, 5.9% under agriculture land use and 14.3% under horticulture land use was observed in cumulative CO_2 efflux over control plots. The corresponding per day emission of CO_2 efflux ranged from 10.89 (Control) to 13.4 mg 100g^{-1} (Alder), 10.79 (Potato) to 12.95 mg 100g^{-1} (Rice) and 10.94 (Pear) to 13.14 mg 100g^{-1} (Guava) with an average of 12.16, 11.74 and 12.19 mg 100g^{-1} in agroforestry, agriculture and horticulture land uses. Soil CO_2 efflux significantly increased by the increase in temperature from 25 to 35 ^{o}C and the maximum per cent increase was under agriculture land use (74%) followed by horticulture (68%) and agroforestry land uses (61%). Among the crop and tree species, the highest increase was recorded in Alder (64%) under agroforestry, Maize (89%) under agriculture and Khasi mandarin (86%) under horticulture land uses at 150 days of incubation. The temperature dependency or sensitivity of soil CO_2 efflux to temperature was studied by using Arrhenius activation energy (AE) and Q_{10} values. AE was highest under agriculture (51.5 kJ mol^{-1}) and lowest under agroforestry land use (51.5 kJ mol^{-1}); on the contrary Q_{10} values were higher in agroforestry (0.88) and lowest under agriculture land uses (0.86) indicating the highest sensitivity of CO_2 efflux to temperature due to land use change. Microbial metabolic quotient was greater in agriculture land use (range: 43.1 to 54.9 with an average of 47.2 µg CO_2 mg MBC^{-1}) followed by horticulture land use (range: 32.2 to 47.6 with an average of 40 µg CO_2 mg MBC^{-1}) and lowest in agroforestry land use (range: 24.9 to 33.4 with an average of 29.5 µg CO_2 mg MBC^{-1}). The land uses, temperature and their interaction significantly influenced the soil CO_2 efflux. Soil CO_2 efflux and microbial metabolic quotient correlated significantly with most of the soil properties studied in this investigation.

Key words: Land use change, temperature, soil CO_2 efflux, activation energy, Q_{10} values, microbial metabolic quotient.

8.2 Introduction

The wide-reaching concern with global change and its effects on our future environment requires a better understanding and quantification of the processes of greenhouse gas emission (Ohashi et al., 1999). Soils are the largest carbon pool in terrestrial ecosystems, containing more than 1500 Pg C (Raich and Schlesinger, 1992; Eswaran et al., 1993). CO_2 efflux from soil to atmosphere is a major component of greenhouse gas emission and is a crucial pathway of the C cycle. Soil respiration consists of organic matter oxidation, root respiration and rhizosphere respiration (i.e., microbial consumption of root exudates and contents of sloughed cells) (Hanson et al., 2001). Soil respired CO_2 represents the ultimate oxidative fate of soil C, and the C lost from terrestrial ecosystems occurs mainly through soil respiration (Amundson, 2001). These C losses are generally many times greater than losses from leaching and erosion (Lal, 2003), although erosion can be a major contributor in some systems (Jacinthe et al., 2004).

On a global scale, soil respiration was estimated to produce 80.4 Pg C per year (petagram carbon per year) with a range of 79.3–81.8 Pg C per year (Raich et al., 2002), accounting for 60-90 percent of total respiration of global terrestrial ecosystems (Schimel et al., 2001), which is more than 11 times the current rate of fossil fuel combustion (Marland et al., 2001; Andre butler et al., 2011). Accordingly, small changes in the magnitude of soil respiration could have a large effect on the concentration of CO_2 in the atmosphere. It is highly sensitive to temperature and global changes may have a great influence on the magnitude of soil CO_2 efflux. The potential increase in CO_2 release from the soil caused by future elevated temperature may have a positive feedback effect on atmospheric CO_2 and global change (Kirschbaum, 1995). In the context of increasing CO_2 concentration in the atmosphere and the related potential change in climate, knowledge of soil CO_2 emission is of great importance to estimate future atmospheric CO_2 concentration and global change (Liang et al., 2004). Therefore, it is important to obtain accurate estimates of soil CO_2 efflux and to understand controls on the underlying process.

Soil organic matter (SOM) has a very complex and heterogeneous composition and is generally mixed or associated with the mineral soil constituents to form soil aggregates (Del Galdo et al., 2003). Soil organic matter dynamics are mainly influenced by its recalcitrance and accessibility, and interactions between SOM and soil components (Sollins et al., 1996). Recalcitrance is the inherent molecular level resistance of a molecule to microbial and enzymatic breakdown. Accessibility is the extent to which the location of substrates controls access by microbes, fungi or enzymes. Interactions of organic substrates with organic or inorganic molecules may alter the degradation rate of the substrates (Swanston et al., 2002). Soil organic matter is a major factor in ecosystem functioning and determines whether soils act as sinks or sources of carbon in the global carbon cycle. Carbon input, magnitude of soil organic carbon pools and finally carbon mineralization depend on many factors

Soil CO_2 efflux is mainly regulated by the oxidation of soil organic matter during litter decomposition by heterotrophic microorganisms and the respiration by plant roots. Thus, the population dynamics of soil microorganisms (e.g. bacteria and fungi) and the soil abiotic factors (i.e., temperature, moisture, organic matter content) are the major factors playing important role in the emission of CO_2 by soil (Muhr et al., 2008). In addition, abiotic factors affect the gaseous diffusion and metabolic activity of soil microorganism and therefore, control the dynamics of soil microorganisms and their metabolic activities within sites (Davidson et al., 1998). Soil temperature and soil moisture are among the most important factors controlling the CO_2 efflux (Raich and Schlesinger, 1992; Davidson et al., 1998). The direct relationship between CO_2 efflux and temperature is well documented (Raich and Schlesinger, 1992; Fang and Moncrieff, 2001).

Temperature is a primary control on CO_2 production in most soils (Kirschbaum, 2000; Raich and Schlesinger, 1992), but not all studies concur (Giardina and Ryan, 2000). As global temperatures rise, any changes in soil CO_2 emissions will in part be determined by the temperature dependence of soil CO_2 production. Because root and microbial sources of CO_2 show increased activity as a function of temperature (Boone et al., 1998) and since new studies suggest that global temperature increases are amplified in the ground, it is critical that the temperature dependence of soil CO_2 production be examined. To describe the temperature dependency of CO_2 efflux, most of the studies use different principles like linear regression analysis (Witkamp, 1966), Q_{10} (Maljanen et al., 2006), power relationship (Kucera and Kirkham, 1971) and Arrhenius equation (Howard and Howard, 1979). In our study, we have used the Q_{10} and Arrhenius form to describe the temperature dependency of soil CO_2 efflux.

Land use change is one of several anthropogenic activities causing a global increase in the atmospheric concentration of CO_2 and other GHGs (IPCC, 1995; Houghton and Hacker, 1999). To characterize the carbon exchange in ecosystems, an assessment of the magnitude and dynamics of soil CO_2 efflux is important, considering that soil respiration is a major CO_2 flux in the carbon cycle, second in magnitude to gross canopy photosynthesis (Raich and Schlesinger, 1992). The net flux of carbon between the soil and the atmosphere is determined by the rate at which soil organic C is converted to CO_2 by microorganisms and by autotrophic respiration. Emission of CO_2 due to land use change and deforestation together with that from the soil is estimated to be between 55±30 Gt (Giga ton = 1015 g) (IPCC, 1995) and 78±17 Gt (Lal, 1999). Land use and soil management practices significantly influence soil organic carbon (SOC) dynamics and C flux from the soil (Paustian et al., 1995; Batjes, 1996; 1998), although the mechanisms and processes of C sequestration in soil are not completely understood (Lal et al., 1995; Bajracharya et al., 1998).

Conversion of natural ecosystems to agricultural systems has resulted in substantial soil carbon loss as a consequence of deforestation, tillage and other physical soil disturbances, removal of crop residues, diminished nitrogen fertility, and changes in soil temperature and moisture as a consequence of cultural practices such as clean cultivation (Lal, 2002). Davidson and Ackerman (1993) reported soil C loss of approximately 20–40% after forest clearing and conversion to agriculture in the humid tropics within the first 1or 2 years following soil disturbance. Losses of soil organic C due to land clearing may result from several processes including decreased inputs and changes in composition of plant litter and increased rates of soil organic matter decomposition and soil erosion (Feller and Beare, 1997). In addition, tillage increases the rate of soil organic matter decomposition by burying surface residues, disrupting soil aggregates, aerating the soil, and exposing new surfaces to microbial attack (Brown et al., 1994). Therefore, the land use changes will affect the amount of soil organic matter loss or gain. Rapid initial losses of soil organic C following forest clearing and conversion to agriculture are primarily losses of the biologically-labile or active soil organic C pool (Brown et al., 1994). Changes in the active organic C pool can be monitored by measurement of rates of soil CO_2 efflux or soil respiration, although other methods including biological, chemical, physical, and isotopic procedures have also been proposed to distinguish active from more stable organic C pools (Townsend et al., 1995).

Many studies of soil respiration have been reported in many ecosystem; temperate forest (Xu and Qi, 2001), boreal forest (Soe and Buchmann, 2005), rain forest (Schwendenmann et al., 2003), semi-arid steppe (Rey et al., 2011), sub alpine forest (Scott-Denton et al., 2003), tropical bare soil (La scala et al., 2000), tropical forests (Sundarapandian and Swamy, 2006), cropland (Rochette et al., 1991), grassland (Wei-Yu Shi et al., 2011) and plantation (Epron et al., 2004). However, studies on comparative account of soil respiration under agroforestry, horticulture plantation and agriculture land use systems are very limited particularly in northeast India. Therefore, in this investigation we hypothesized that conversion of fallow lands to agroforestry plantation, horticulture plantation and agriculture crop cultivation increases the accumulation of C and thus increases the C respired from more recalcitrant C sources in soils from these land uses relative to the fallow land soils. At the same time, we also proposed that the magnitude of soil respiration increases under fallow lands and agriculture crop cultivation because long-term cultivation decreases non-recalcitrant C in the soil.

Land use, soil, and climatic factors prevailing in a given ecosystem regulate the fluxes of GHGs. These driving factors vary by ecological zone, and their effect on gaseous flux is further influenced by land use changes and management practices. Ongoing land use changes in developing countries are likely to have a profound impact on fluxes of GHGs (Houghton and Hackler, 1999). There are major gaps in our knowledge and database on fluxes of GHGs for the north-eastern region of India, an area of rapid land use changes. Slash and burn cultivation, locally branded as *Jhum* cultivation, with reduced fallow cycle of 2-3 years (10-15 years in the past) is the predominant form of agriculture in the north-eastern region of India resulted in loss of biodiversity and severe soil erosion causing decline in soil productivity. On an average, shifting cultivation along with raised beds along the slope, unabated deforestation and high rainfall caused 36.64% of land degradation with a soil loss of 30.2-170.2 t ha^{-1} yr^{-1} which is almost two-fold more than the national average of 20.17% (Bhatt et al., 2006). The *Jhuming* practices also causes decline in soil organic matter, which is the main soil quality indicator and is responsible for a number of soil properties in the plant-animal ecosystems. Woomer et al. (1994) reported that changes in soil organic carbon (SOC) due to land management practices can have significant impact on soil physico-chemical properties and potentially affect the global C cycle. This study aims to initiate work in this direction by measuring efflux of CO_2 from soils of different land use systems and the effect of temperature and soil biotic and abiotic variables on fluxes of CO_2 from the soils in East Khasi hills of Meghalaya. In addition, effects of other relevant factors such as temperature, soil BD, MWD, moisture, OC and MBC may assist us in the interpretation of data to derive a conclusion that larger variations in these factors will significantly influence the CO_2 fluxes under various land use systems.

8.3 Materials and methods

8.3.1 Location of the study site

The study was carried out with different multipurpose tree species (MPTs) planted under agri-silvipastural system in 1983, horticulture tree species planted in 1994 and agricultural crops continuously cultivated for the past 10 years at research farm of Indian Council of Agricultural Research (ICAR) Complex for North-East Hill (NEH) Region, Umiam. The station is situated in the central part of Meghalaya in the East *Khasi* Hills of North-East India.

Soil samples were collected during October-November in the year 2009 under all the treatments including controls (natural fallow). About 250 g of moist soil samples were kept in deep freezer for moisture and incubation experiments.

A 150 days incubation experiment was carried out in the laboratory at three temperatures: 25, 30 and 35 ^{o}C. The CO_2 that evolved from the soil at different time intervals was measured. And the CO_2 efflux data thus generated for different temperature gradients was used to calculate the temperature dependency of CO_2 efflux using Arrhenius activation energy ($k= A \exp (-E/ RT)$ and Q_{10} value (Knorr et al., 2005; Metcalf and Eddy, 1991).

The **s**pecific respiratory activity of soil microbial biomass carbon was estimated by dividing the CO_2-C produced by the value of soil microbial biomass (Campbell et al., 1991). The data obtained from this research programme was statistically analysed following standard statistical methods (Gomez and Gomez, 1984).

8.4 Results

8.4.1 Land uses and CO_2 efflux

The influence of crop and tree species on CO_2 efflux was shown by the comparison of the soil CO_2 efflux progression between the crop and tree species, and no tree or no crop (control) in each land use systems. Figure 8.1 represents the effects of various multipurpose tree species on the cumulative CO_2 efflux released from the soils with respect to CO_2 efflux from control plot (no tree) during the 150 days of incubation period. It is clear from the figure that adoption of various agroforestry tree species increased the mean cumulative CO_2 efflux on an average by 14.6 % with the lowest increases in Khasi pine (8.8%) to the highest increase in alder field (23.2%) as compared to control. It was also found that up to 20 days of incubation, no significant differences in CO_2 efflux was observed among the tree species; thereafter agroforestry tree species significantly changed the soil CO_2 emission. Increase in duration significantly increased the CO_2 efflux from soils under all the multipurpose tree species (Fig. 8.7A). The per day emission of CO_2 efflux ranged from 10.9 to 13.4 mg $100g^{-1}$ being highest from alder field and lowest from control plot. At the end of the incubation period (150 days), cumulative CO_2 efflux under agroforestry land use followed the order: Alder>Champak>Tree bean>Khasi pine>Control (Fig. 8.5).

Similar to agroforestry land use, land conversion from fallow to agriculture crop cultivation significantly increased the cumulative CO_2 efflux from the soils. The CO_2 efflux ranged from 1.618 (control) to 1.943 g $100g^{-1}$soil (Rice) (Fig 8.2). However, only, an average, 5.9% increase in cumulative CO_2 efflux was noticed in soils under agriculture crops compared to control plot. Amongst the agriculture crops, rice soils resulted in highest cumulative CO_2 efflux than other crops which is 15.5% higher compared to control plot (Fig. 8.5). Incubation of agriculture soils up to 90 days of duration did not show considerable variation but thereafter significant increase was observed in cumulative CO_2 efflux up to the period of 150 days (8.7B). The daily CO_2 efflux from the soils under agriculture land use ranged from 10.79 (Potato) to 12.95 (Rice) mg $100g^{-1}$ with an average of 11.75 mg $100g^{-1}$ soil which is relatively lower than agroforestry land use system (12.16 mg $100g^{-1}$ soil). In general, agriculture crops significantly increased the soil CO_2 efflux and followed the order: Rice>Turmeric> Maize >Control> Potato (Fig. 8.5).

Figure 8.3 illustrates the effect of various horticulture tree species on the soil CO_2 efflux. The soil CO_2 efflux from peach field was significantly higher (1.945 g $100g^{-1}$) while Khasi mandarin field samples showed lower values (1.746 g $100g^{-1}$) as compared to control plot (1.642 g $100g^{-1}$). Similar to agroforestry and agriculture land uses, increase in the incubation period significantly increased the CO_2 efflux from soils under all the horticulture tree species including control although we did not find substantial differences in soil CO_2 efflux up to 45 days of incubation (8.7C). The maximum increase in soil CO_2 efflux at 150 days of incubation was observed under Guava field (20.1%) followed by Peach (18.5%) and lowest was under Khasi mandarin (6.5%) in relative to control plot (Fig. 8.5). Overall, in relation to control plot, 14.3% increase in soil CO_2 efflux was recorded under horticulture tree species which is almost similar to that of agroforestry land use but relatively lower than agriculture land use. Soil CO_2 efflux under horticulture land use followed the decreasing order of guava, peach, pear, Khasi mandarin and control (Fig. 8.5).

On the whole, adoption of various agroforestry, agriculture crops and horticulture tree species increased the soil CO_2 efflux by 14.6, 5.9 and 14.3%, respectively over the control plots. In relative to the initial days of incubation period, highest increase of about 11% in soil CO_2 efflux was observed under agriculture land use followed by horticulture land use (8.95%) and lowest increase was noticed in agroforestry land use (8.26%) at 150 days. During he initial period of incubation (up to 3-9 days), soil CO_2 efflux in all the three land uses did not show any considerable changes and thereafter increased the efflux significantly up to the end of the incubation period (Fig 8.4). As compared to agriculture land use, the cumulative CO_2 efflux was highest under agroforestry and horticulture land uses (Fig. 8.6).

8.4.2 Temperature and CO_2 efflux

Effect of temperature (25, 30 and 35 ^oC) on soil CO_2 efflux under agroforestry, agriculture and horticulture land use systems during the incubation period of 150 days are presented in figure 8.8, 8.9 and 8.10, respectively. Figure 8.8 clearly illustrates that soil CO_2 efflux under agroforestry land use increased significantly as the temperature increases from 25 to 35 ^oC. However, at the initial stage of incubation period there was no considerable variation and the difference became wider as the incubation period advanced up to 150 days. At 150 days, the cumulative CO_2 efflux from soil ranged from 1.21 (Control) to 1.49 g $100g^{-1}$ (Alder) with an average value of 1.37 g $100g^{-1}$ soil at 25 ^oC. At 35 ^oC, it varied from 2.05 (Control) to 2.45 g $100g^{-1}$ (Alder) with an average value of 2.21 g $100g^{-1}$ soil (Fig. 8.12A). Increase in temperature from 25 to 35 ^oC increased the average soil CO_2 efflux by 61% in agroforestry land use; however, the rate of increase was highest under control plot (about 69%) and lowest under Khasi pine (about 53%).

Cumulative soil CO_2 efflux rates ranged from 1.10 (Potato) to 1.38 g $100g^{-1}$ (Turmeric) with an average of 1.28 g $100g^{-1}$ soil at 25 ^oC and increased from 2.05 to 2.23 g $100g^{-1}$ (Rice and Turmeric) with an average of 2.18 g 100^{-1} soil under agriculture land use system. As compared to agroforestry land use, the differences in soil CO_2 efflux due to temperature increase from 25 to 35 ^oC in agriculture land use was clearly visible from the initial stage of incubation and increased significantly up to 150 days of incubation (Fig. 8.9). At 150 days of incubation, with increase in temperature from 25 to 35 ^oC, nearly 91.6% increase in soil CO2 flux was observed in control plot, 89.2% in maize and the lowest increase under rice field (35%) (Fig. 8.12B).

Horticulture tree species behaved more or less similar to the agroforestry systems with respect to soil CO_2 efflux due to increase in temperature from 25 to 35 ^oC (Fig. 8.10). However, the average per cent increase in CO_2 efflux due to the increase of temperature from 25 to 35 ^oC was about 68% (range: 50.1 g 100^{-1} under Peach to 86.6 g 100^{-1} under Khasi mandarin) which is higher than the agroforestry land use (61%) but relatively lower than the agriculture land use (91.6%) (Fig. 8.12C). The cumulative CO_2 efflux at 25 ^oC ranged from 1.12 (Control) to 1.49 g 100^{-1} (Peach) with an average of 1.31 g 100^{-1} and ranged from 2.08 (Control) to 2.30 g 100^{-1} (Pear) with an average of 2.24 g 100^{-1} at 35 ^oC. In general, the cumulative CO_2 efflux was higher at 35 ^oC over other two temperatures in all the three land use systems although agroforestry and horticulture land uses showed highest and agriculture land use recorded lowest cumulative CO_2 efflux at 150 days of incubation (Fig. 8.11).

8.4.3 Rate constant, activation energy and Q_{10} value

First order rate constant (k) and activation energy of carbon-dioxide evolution from soils under different land use systems soils at 25, 30 and 35 ^oC are given in table 8.2. The rate of the reaction was higher at 25 ^oC and tended to increase as the temperature increases to 35 ^oC in all the land use systems. The average rate constant was highest under agriculture land use (0.019) followed by horticulture and the lowest was found under agroforestry land use (0.012). At 35 ^oC, Khasi pine=Parkia, Rice and Peach=Guava showed the maximum rate constant compared to associated tree and crop species. However, control plots under all the three land uses showed greater rate constant values compared to tree and crop species. The temperature sensitivity was calculated by using Arrhenius activation energy and Q_{10} values and is given in the table 8.2. The activation energies of the different land uses varied between 39.0 (Champak=Alder) to 59.1 kJ mol^{-1} (Khasi pine) with an average of 46.5 kJ mol^{-1} in agroforestry land use; 31 (Rice) to 63.7 kJ mol^{-1} (Control) with an average of 51.5 kJ mol^{-1} in agriculture land use and from 44.9 (Peach) to 58.3 kJ mol^{-1} (Khasi mandarin) with an average of 51.7 kJ mol^{-1} in horticulture land use. In general, the activation energies showed dissimilar trends in all the land use systems and were found to be higher in agriculture and horticulture land uses. Specifically, trees like Khasi pine, Pear and Khasi mandarin increased the activation energy over control and in general, other tree and crop species increased the activation energy than control. In contrast to activation energy, Q_{10} values were higher under agroforestry land use and ranged from 0.85 (Khasi pine) to 0.90 (Alder = Champak) with an average of 0.879 (Table 8.2). The Q_{10} values were more or less similar in agriculture and horticulture land uses with highest value in Rice (0.905) and Peach = Guava (0.872) under each land use systems, respectively. However, control plots in all the land uses were lower in Q_{10} values compared to associated tree and crop species.

8.4.4 Specific respiratory activity (SRA) or microbial metabolic quotient

Specific microbial respiratory activity (also called microbial metabolic quotient) as influenced by different land use systems and temperature is given in the table 8.3 and depicted in the figures 8.12 and 8.13. SRA was significantly influenced by agroforestry land use; however no significant changes in SRA were noticed under agriculture and horticulture land use systems. The SRA in agroforestry land use ranged from 26.14 to 38.09 µg CO_2 mg MBC^{-1} with an average of 30.57 µg CO_2 mg MBC^{-1}. The highest SRA was found under the Tree bean and Khasi pine in agroforestry land use. Rice and Guava also had highest SRA under agriculture and horticulture land uses, correspondingly. Nevertheless, control plots in all the land uses showed higher SRA than the allied tree and crop species. Increase in temperature from 25 to 35 ^oC increased the SRA significantly and found to be higher at 35 ^oC than 25 and 30 ^oC irrespective of the land uses. The highest increase of 71.3% was observed under agriculture land use compared to agroforestry (69%) and horticulture land use (70%) due to increase in temperature from 25 to 35 ^oC. However, interaction of both land uses and temperature did not show any significant changes in microbial metabolic quotient. In general, the metabolic quotient in our study was 54.5 and 18.5% higher in agriculture land use in relative to agroforestry and horticulture land uses, respectively and followed the order Agriculture>Horticulture>Agroforestry (Fig. 8.13).

8.4.5 Relation of soil CO_2 efflux and SRA with soil biotic and abiotic variables

Soil CO_2 efflux was statistically correlated with soil properties *viz*. soil BD, MWD, moisture, soil organic carbon (SOC) and microbial biomass carbon (MBC) after 150 days of incubation (Table 8.5). Soil CO_2 efflux was positively related with SOC and MBC but was not significant. However, in comparison to MBC (r= 0.094), soil CO_2 efflux had more dependency on SOC (r= 0.279). Soil pH also exhibited considerable influence on soil CO_2 efflux but to a lesser extent than SOC and SMBC. Conversely, soil BD (r= -0.335*) had negative and significant relationship with soil CO_2 efflux. MWD and available N had positive correlation with soil CO_2 efflux however, significant correlation was found with only available N (r=0.363*). Specific respiratory activity, also called metabolic coefficient, was also statistically correlated with soil biotic and abiotic variables (Table 8.5). SRA had strong positive and significant correlation with soil bulk density (0.456**) and mean weight diameter (0.496**); however, MWD influenced the SRA to a higher extent in relative to soil BD. In contrast, SRA was negative and significantly related with soil organic carbon (-0.433**), microbial biomass carbon (-0.912**) and available nitrogen (-0.348*).

8.5 Discussion

8.5.1 Land use and soil CO_2 efflux

Land use change can strongly influence soil microclimate, plant carbon allocation pattern, substrate availability and input, and hence the soil CO_2 efflux which is an useful indicator to determine energy flow patterns, specifically the mineralization of nutrients and the rate of organic material decomposition (Li et al., 2011). Our study revealed that land use change significantly influenced the rate of soil CO_2 efflux although, within the land uses, we observed significant changes only in agriculture and horticulture land uses (Fig. 8.5 and table 8.5). Our results are in conformity with the findings of Raich and Tufekcioglu (2000); in contrast, Schwendenmann et al. (2003) did not find significant effect of land use change on soil CO_2 efflux. Amongst the land use systems, highest rate of soil CO_2 efflux was observed under agroforestry land use followed by horticulture and agriculture land use. Among the agroforestry system higher soil CO_2 efflux was observed from soils from Alder and Champak fields. It has also been reported that the changes in land use might have modified the organic carbon transformation due to the changes in substrate quality (Feigl et al., 1995), altered microbial community size (Cleveland et al., 2003) and/or changes in soil porosity and water retention (Martinez and Zinck, 2004). The low values of CO_2 efflux rates recorded for soils under agriculture land use overall may reflect their low total C content due to the lower carbon input or net primary productivity, lower substrate quality, loss of soil carbon by frequent soil erosion events and poor structure (Schwendenmann et al., 2003).

Compared to control plots, the total CO_2 efflux from the soils under tree and crop species was higher because of the heterotrophic respiration and the C input by root exudates. A possible assumption for the higher increase of CO_2 in the plots under tree and crop species instead of plots without tree and crop species (control) could be a higher C input by trees and crop plants. Besides, it was assumed that the labile C of the exudates simplifies the mineralisation of native soil organic matter (Kuzyakov, 2002) and thus stimulated the microbial activity (Kuzyakov et al., 2000). On the one hand the decomposition depended on temperature and substrate, but on the other hand a constant exudate addition by the roots of trees and crop plants may have decreased the mineralisation of SOM (Kuzyakov et al., 2007). The mean daily soil CO_2 efflux ranged from 10.94 to 13.40 mg $100g^{-1}$ soil. In general, the magnitude and variability of soil CO_2 efflux that found under different land use systems may be attributed to the variation in soil biotic and abiotic factors controlling the spatial variability in soil respiration *viz*. the heterogeneity of vegetation coverage, root biomass, depth of the top soil layer, litter quantity and quality, soil organic matter, soil temperature and moisture content, etc. (Kucera and Kirkham, 1971; Jassal et al., 2005; Tang et al., 2005; Tufekcioglu et al., 2009).

The effects of incubation period on soil CO_2 efflux from the selected land use systems indicated that increase in the incubation period significantly increased the cumulative soil CO_2 efflux irrespective of the land uses (Fig. 8.4). However, the per cent increase in soil CO_2 efflux due to the increase in incubation duration was higher at the initial stage and decreased as the incubation progressed up to the period of 150 days. Two reasons may cause the decline in soil CO_2 efflux in the later stage: (1) roots are removed from the soil samples which may deactivate the microbes at the later stage of incubation; (2) soil organic matter may be eventually exhausted by microbial decomposition which therefore causes decline in respiration rate, although the organic matter content was reported to be stable after some incubation time (Winkler et al., 1996; Grisi et al., 1996).

8.5.2 Temperature and soil CO_2 efflux

The effects of temperature on soil CO_2 fluxes are well known and have been demonstrated in numerous studies (Iqbal et al., 2008; Iqbal et al., 2009b). Factors such as temperature, moisture availability, and substrate properties that simultaneously influence the production and consumption of organic matter are more important in controlling the overall rate of soil CO_2 efflux than vegetation type in most cases. However, our study showed significantly varied CO_2 efflux under different land uses; agroforestry had 14.6% higher rates of soil CO_2 efflux than agriculture land use demonstrating that vegetation type does in some cases significantly affect rates of soil CO_2 efflux (Raich and Tufekcioglu, 2000). In this study, a significant increase in soil CO_2 efflux with increase in temperature gradients was observed under all the tree and crop species including control plots. Overall, due to increase in temperature from 25 to 35 ^{o}C, highest increase (74%) was noticed in agriculture land use than agroforestry and horticulture land uses (Fig. 8.11 and 8.12). This could be possibly explained by the quality of litter and root biomass, crop residues and soil properties which are relatively inferior in agriculture land use in comparison to other land uses (Saha et al., 2007). The highest rate of soil CO_2 efflux under agriculture land use may also be attributed to the increased mineralization of the relatively less protected organic carbon in the aggregates and easily degradable organic carbon of agriculture land use compared to agroforestry and horticulture land uses (Lal, 2002). Anon et al. (2001) also reported that increased temperature coupled with highest addition of organic inputs under different land use systems supported more microbial growth and utilization of available substrates by altering the respiratory quotient of microbes, which is clearly evident in our study (Table 8.3), and thus increase in CO_2 efflux.

The temperature sensitivity of soil organic matter decomposition is a highly debated topic due to its important implications for the global carbon cycle and feedbacks to the climate system (Davidson and Janssens, 2006). In our study, the temperature dependency or sensitivity of CO_2 efflux to temperature was determined by using Q_{10} and Arrhenius equation. The first order rate kinetics (k) and activation energy of CO_2 efflux under different land use systems are given in the table 8.2. The first order rate constants showed a gradual increase when the temperature increased from 25 to 35 ^oC in all the land uses. In relative to agroforestry land use, agriculture land use showed highest rate constant; on the other hand, control plots in all the three land uses had highest rate constant. This indicates that greater aggregate stability and protection provided for the organic carbon in the agroforestry system than agriculture land use. Tillage-induced soil disturbance in agriculture land use may increase soil organic carbon mineralization by bringing crop residue closer to microbes, more soil-residue contact, and within the plough layer where soil moisture conditions favour higher mineralization rates (Gregorich et al., 1998). It also physically disrupts/breaks aggregates and exposes hitherto encapsulated (protected) carbon to microbial processes (Kay, 1998). A noteworthy effect of tree species on activation energy (AE) of CO_2 efflux was observed and the value ranged from 38.9 to 63.7 kJ mol^{-1} being highest in agriculture land use (51. 7 kJ mol^{-1}) and lowest in agroforestry land use (46.5 kJ mol^{-1}) specifying the quality of substrate for decomposition. These results are in consistent with the findings of Davidson and Janssens (2006) who reported highest sensitivity of low quality organic carbon to temperature change.

The Q_{10} coefficient, which is the relative increase of soil CO_2 efflux rate for a 10 ^oC change in temperature, serves as an index for the sensitivity of soil CO_2 efflux rate to temperature. Based on the three temperature used in the present incubation experiment *viz.*, 25, 30 and 35 ^oC, the calculated Q_{10} values of agroforestry, agriculture and horticulture land uses were 0.879, 0.855 and 0.857, respectively. This demonstrates that the soil CO_2 efflux under agroforestry land use had the highest sensitivity to temperature than the agriculture and horticulture land uses. It also suggested that conversion of lands from fallow to agroforestry, agriculture and horticulture land uses increases the sensitivity of soil CO_2 efflux to temperature as evident from the present investigation. This result is in accordance with the findings of Zhou et al. (2006). The Q_{10} value could be affected by a range of factors including calculation model, temperature range, soil water availability, substrate quality and microbial population (Davidson et al. 2006). In the present study, the variation in Q_{10} values under different tree and crop species may be attributed to the differences in litter and root biomass, crop residue addition, SOC quality or microbial activities among the tree and crop species. Besides this, control plots in all the land uses were low in Q_{10} values compared to the plots under tree and crops species. The possible reasons for the decreased values in control plots as evident from several studies (Balesdent et al., 1988; McCulley et al., 2007) may be due to the increased organic matter decomposition by the enhanced activities of soil microorganisms as a result of land management practices or intensification thus increasing the temperature sensitivity of soil CO_2 efflux under tree and crop species compared to control plots.

8.5.3 Specific respiratory activity or microbial metabolic quotient

The specific respiration rate is a relative measurement of how efficiently the soil microbial biomass is utilizing C resources, and the degree of substrate limitation for the soil microbial biomass (Wardle and Ghani, 1995). The microbial metabolic quotient did not change significantly with land uses and decreased by 35% in agroforestry land use in comparison to the agriculture land use (Table 8.3). As per the reported literature, the average rate of soil CO_2 efflux increased with agroforestry land use due to the increased carbon inputs to the soil but respiratory activity as a percent of microbial biomass carbon (metabolic coefficient) decreased in the agroforestry land use. This suggests that agroforestry was most efficient land use in preserving C in soil. On the other hand, the highest microbial metabolic quotient in agriculture indicates the ecosystem stress in agriculture land use due to the cultivation disturbance. This result in our study is in accordance with the findings of Goyal et al. (1992), Lupwayi et al. (1998) and Yao et al. (2000). An enhanced respiration rate has been attributed to a low efficiency of substrate utilization for growth when microorganisms are under environmental stress (Merrington et al., 2002). Therefore, the specific respiration rate might be one of the most influential soil microbial indicators responsive to land use change. An increased specific respiration rate also indicates the energy shifting from growth to maintenance in an ecosystem (Aoyama and Nagumo, 1997).

8.5.4 Relationships of soil CO_2 efflux and SRA with soil variables

Soil CO_2 effluxes were statistically correlated with soil properties *viz*., soil BD, MWD, moisture, OC and MBC after 150 days of incubation (Table 8.5). Soil CO_2 efflux was positively related with MWD, SOC, available N and SMBC but it showed significant correlation only with available N (0.363*). However, in comparison to MBC, soil CO_2 efflux had more dependency on SOC. This result is in agreement with the findings of Lou et al. (2004) but in contrast with Li et al. (2011). Soil moisture exhibited no considerable influence on soil CO_2 efflux. Conversely, soil BD (r= -0.335*) had negative and significant relationships with soil CO_2 efflux as shown in the table 4, implying that dense soils with high soil BD reduces the capacity of soil fluid and gas transport, which in turn reduces the accessibility of soil microorganisms to soil air and water, creating a less favourable environment for soil microorganisms mediated organic matter decomposition thus reducing the soil CO_2 efflux. A similar observation was also reported by Pengthomkeerati et al. (2005). A strong significant positive relation of SRA with soil bulk density and MWD, and significant negative correlation with SOC, MBC and available N were observed in this study. This is in conformity with the findings of Ananyeva et al. (2008) who had reported a close relationship between SRA and organic carbon and available N.

8.6 Conclusions

In the present study, the cumulative soil CO_2 efflux rates for the three land uses differed significantly whereas, remarkable variation within crops and tree species was observed in agriculture and horticulture land uses. The results revealed that conversion of fallow lands to agroforestry, horticulture and agriculture land uses resulted in the accumulation of organic carbon and thus increasing the loss of soil organic C as CO_2 emission. However, more importantly, in relative to the agroforestry and horticulture land uses, the rate of CO_2 efflux was higher in soils under agriculture land use. This investigation has also shown that in relative to the three land uses studied, soils from the control plots resulted in significantly more soil CO_2 per unit mass of microbial biomass signifying the importance of tree and crop species in sequestering the soil carbon by reducing soil CO_2 efflux.

Conversion from fallow to agriculture, horticulture and agroforestry increased the Q_{10} values while decreasing the Arrhenius activation energy (AE) suggesting the increased sensitivity of soil CO_2 efflux to land use change. On the other hand, the increased Q_{10} values and decreased AE in the agroforestry land use in relative to agriculture and horticulture land use implies the importance agroforestry systems in protecting the soil organic carbon by increasing the aggregate stability and thus soil carbon storage.

Microbial metabolic quotient was significantly lower in agroforestry land uses than agriculture and horticulture land uses to make it the most efficient land use to preserve organic carbon in soil and also suggests a higher accumulation of resistant organic C pool in this land use system. The highest microbial metabolic quotient in agriculture indicates the ecosystem stress in agriculture land use due to the cultivation disturbance. Therefore, the specific respiration rate might be one of the most influential soil microbial indicators responsive to land use change.

Soil CO_2 efflux appears to be mainly regulated by the abiotic and biotic variables. Because biotic variables like microbial biomass carbon are significantly affected by altering the pattern of temperature. The significant impact of soil variables on CO_2 efflux and specific respiratory activity indicates the strong implications of this study on one of important global change phenomena of increasing atmospheric CO_2 concentration due to the change in land use and management practices.

Table 8.1 Initial soil properties of different land uses selected for the study

Land use system	Soil properties						
	pH	BD (Mg m^{-3})	MWD (mm)	Moisture (g $100g^{-1}$)	N (kg ha^{-1})	P	K
				Agroforestry			
Champak	4.60	1.11	31.0	31.0	522.3	32.1	313.6
Tree bean	4.59	1.15	26.1	26.1	496.3	30.1	286.7
Alder	4.38	1.07	31.8	31.8	584.3	47.2	361.2
Khasi Pine	4.36	1.05	33.8	33.8	464.8	23.7	276.0
Control (No tree)	4.76	1.18	28.4	28.4	403.4	19.3	248.4
				Agriculture			
Maize	5.12	1.27	29.9	29.9	394.4	49.1	292.7
Potato	5.24	1.31	28.9	28.9	414.6	65.6	365.1
Rice	5.09	1.33	32.2	32.2	421.4	23.5	322.2
Turmeric	5.29	1.25	26.7	26.7	398.9	36.9	306.1
Control (No crop)	4.81	1.33	24.8	24.8	387.7	20.7	203.1
				Horticulture			
Pear	5.71	1.28	31.6	31.6	437.1	43.2	323.6
Peach	6.24	1.23	32.7	32.7	455.1	39.1	495.2
K Mandarin	5.75	1.29	28.7	28.7	428.1	45.3	419.6
Guava	5.43	1.19	29.9	29.9	446.1	24.6	343.5
Control (No tree)	4.82	1.30	27.1	27.1	434.9	26.9	328.5

Table 8.2 Effect of temperature on rate constant, activation energy and Q_{10} value of CO_2 efflux from different multipurpose tree species in agroforestry land use

Land use systems	Rate constant Temperature (°C) 25	30	35	Activation energy (kJ mol^{-1})	Q_{10}
Agroforestry					
Champak	0.006	0.008	0.010	38.98c	0.9000a
Tree bean	0.007	0.011	0.013	47.33b	0.8752a
Alder	0.006	0.008	0.010	38.98c	0.9002a
Khasi pine	0.006	0.010	0.013	59.07a	0.8489a
Control (No tree)	0.008	0.013	0.015	48.08b	0.8698a
Agriculture					
Maize	0.009	0.015	0.019	57.1ab	0.841a
Potato	0.008	0.013	0.016	53.0b	0.856a
Rice	0.014	0.018	0.021	31.0c	0.905a
Turmeric	0.009	0.014	0.018	52.9b	0.853a
Control (No crop)	0.010	0.019	0.023	63.7a	0.819a
Horticulture					
Pear	0.008	0.013	0.017	57.6a	0.844a
Peach	0.010	0.015	0.018	44.9b	0.872a
Khasi mandarin	0.007	0.012	0.015	58.3a	0.846a
Guava	0.010	0.016	0.018	45.0b	0.872a
Control (No tree)	0.010	0.016	0.020	53.0ab	0.849a

Values indicated by same letter were not significantly different ($P \leq 0.05$) by Duncan's test method.

Table 8.3 Effect of land use and temperature and their interaction on specific microbial respiratory activity (μg CO_2 mg MBC^{-1}) at 150 days of incubation from soils under various land use systems

Land use systems	**SRA (μg CO_2 mg MBC^{-1})**			**Mean**
	Temperature (^oC)			
	25	**30**	**35**	
		Agroforestry		
Champak	21.03	26.77	30.63	26.14bc
Tree bean	23.04	33.68	39.38	32.03ab
Alder	18.25	25.68	29.93	24.62c
Khasi pine	21.87	33.89	40.06	31.94ab
Control (No tree)	26.67	40.30	47.30	38.09a
Mean	22.17c	32.07b	37.46a	**30.57**
		Agriculture		
Maize	30.14	47.89	57.00	45.01a
Potato	29.44	45.39	54.50	43.11a
Rice	43.36	51.37	58.57	51.10a
Turmeric	31.48	43.25	52.63	42.46a
Control (No crop)	36.09	58.12	69.42	54.54a
Mean	34.11b	49.20a	58.42a	**47.24**
		Horticulture		
Pear	24.00	36.87	44.22	35.03a
Peach	31.57	44.36	47.46	41.13a
Khasi mandarin	21.80	34.18	40.54	32.17a
Guava	33.12	48.23	51.51	44.28a
Control (No tree)	32.87	50.40	59.40	47.56a
Mean	28.67b	42.81a	48.63a	**40.03**

Table 8.4 Effect of land use, temperature, duration and their interaction on CO_2 efflux from soils under various land use systems

Factors	Agroforestry	Agriculture	Horticulture
LUS	**	**	**
Temperature	**	**	**
Duration	**	**	**
LUS x Temperature	**	**	**
LUS x Duration	**	**	**
Temperature x Duration	**	**	**
LUS x Temperature x Duration	NS	**	**

** Significant at 0.01 level; NS-not significant

Table 8.5 Pearson's correlation between various soil biotic and abiotic factors and soil CO_2 efflux and between SRA and soil biotic and abiotic variables at 150 days of incubation

Soil variables	r value	Significance level	r value	Significance level	No. of samples
	Soil CO_2 efflux		**SRA**		
Soil pH	0.046	0.763	0.073	0.635	45
Bulk density	-0.335*	0.024	0.456**	0.01	45
MWD	0.254	0.092	0.496**	0.01	45
Moisture	-0.128	0.403	0.01	0.946	45
OC	0.279	0.063	-0.433**	0.003	45
MBC	0.094	0.537	-0.912**	0.000	45
Available N	0.363*	0.014	-0.348*	0.019	45
Available P	-0.066	0.669	0.153	0.317	45
Available K	0.145	0.343	-0.179	0.239	45

* Correlation is significant at the 0.05 level

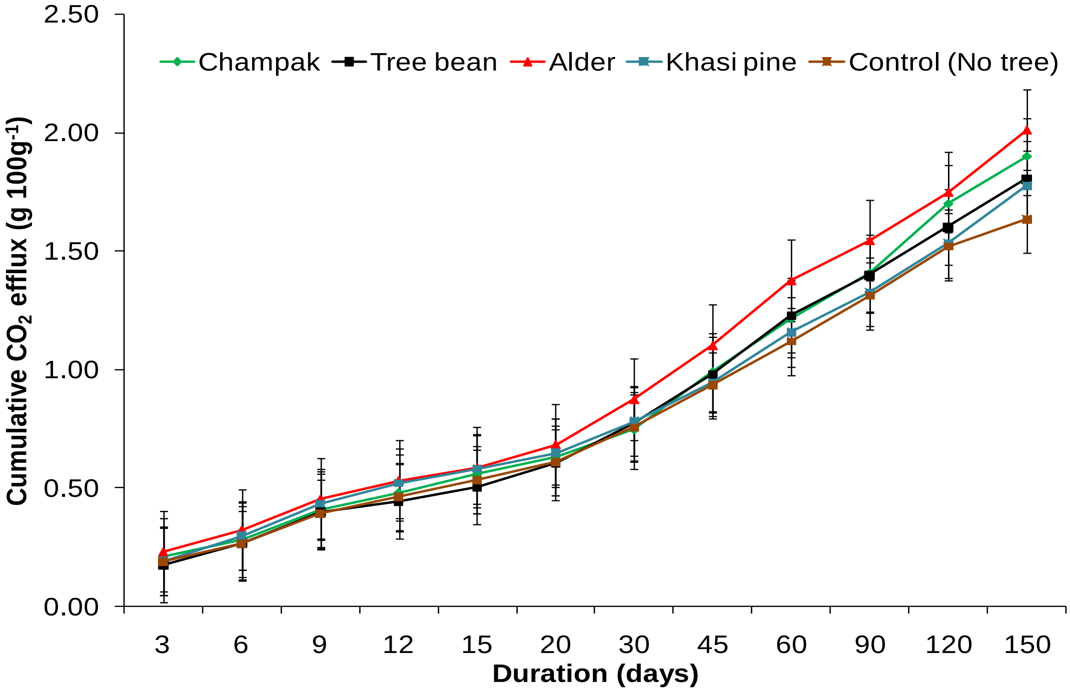


Fig. 8.1 Cumulative CO_2 efflux (g $100g^{-1}$ soil) in soils from various multipurpose tree species under agroforestry land use system

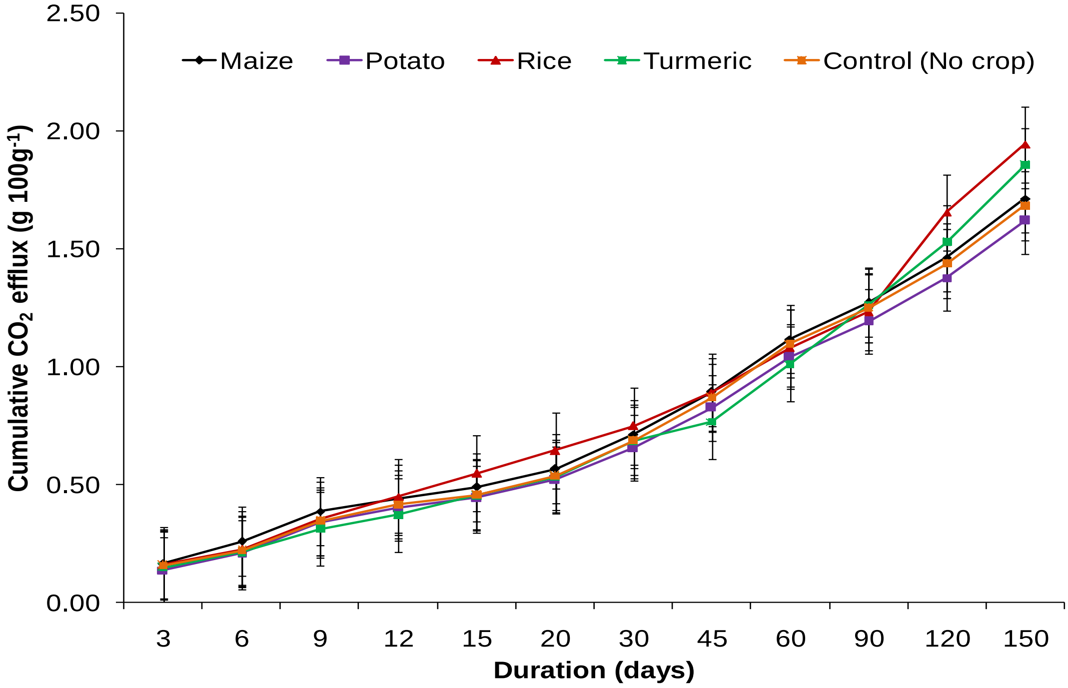


Fig. 8.2 Cumulative CO_2 efflux (g $100g^{-1}$ soil) in soils from various agriculture crops under agriculture land use system

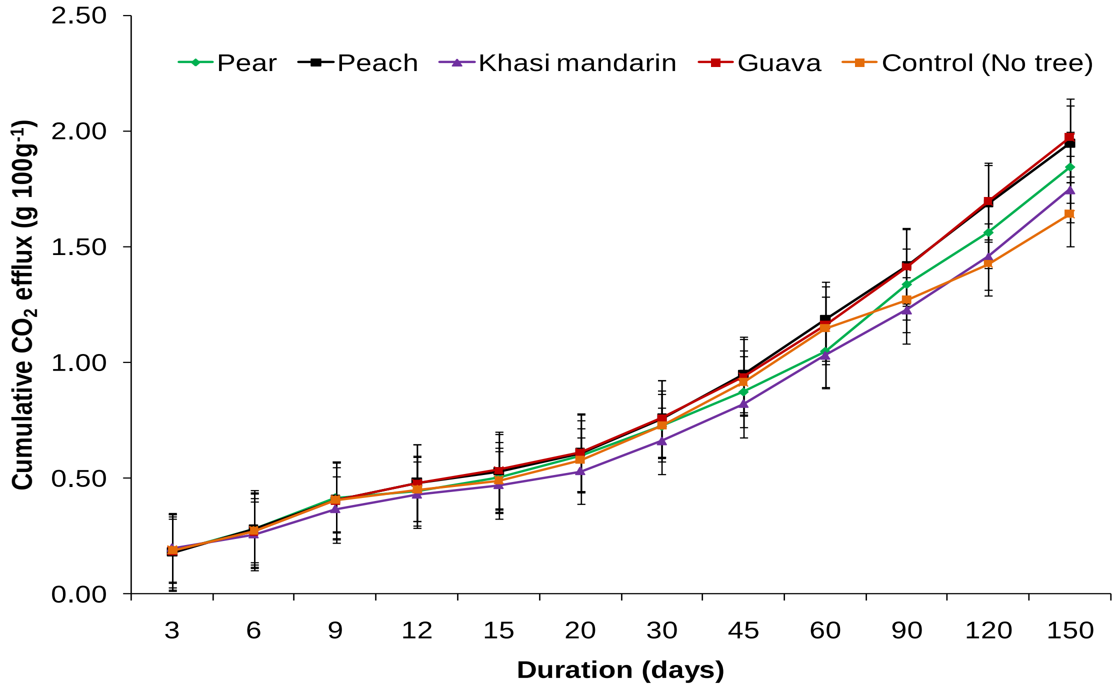


Fig. 8.3 Cumulative CO_2 efflux (g 100g^{-1} soil) in soils from various horticulture tree species under horticulture land use system

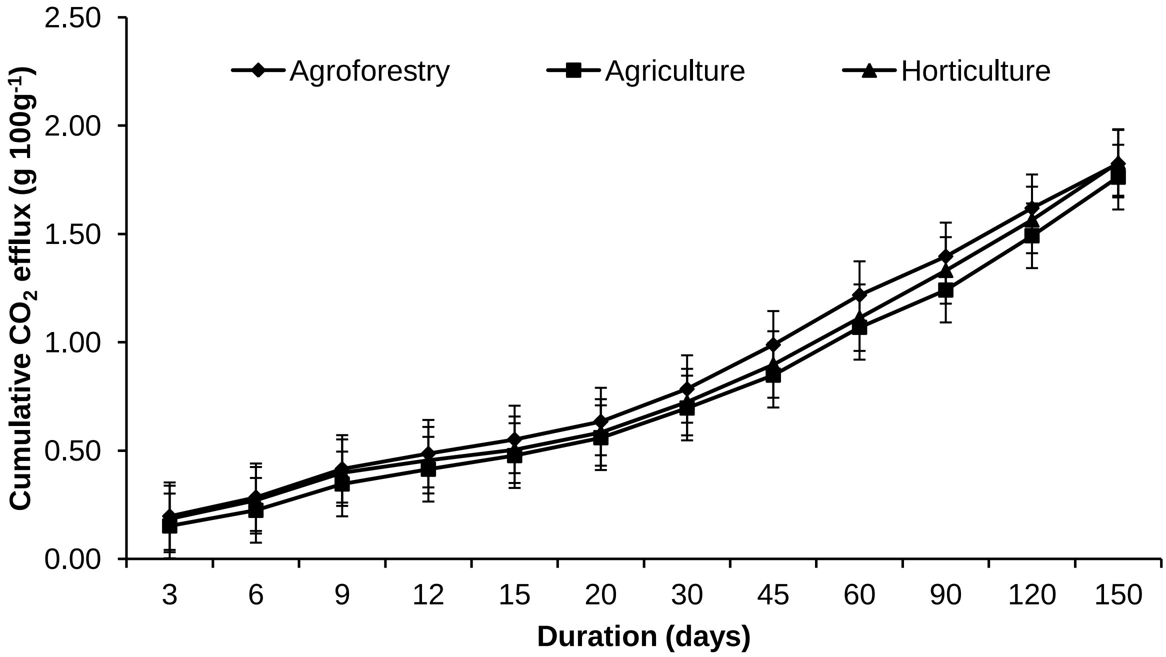


Fig.8.4 Effect of duration and land use systems on mean cumulative CO_2 efflux (g 100g^{-1}soil) from soils

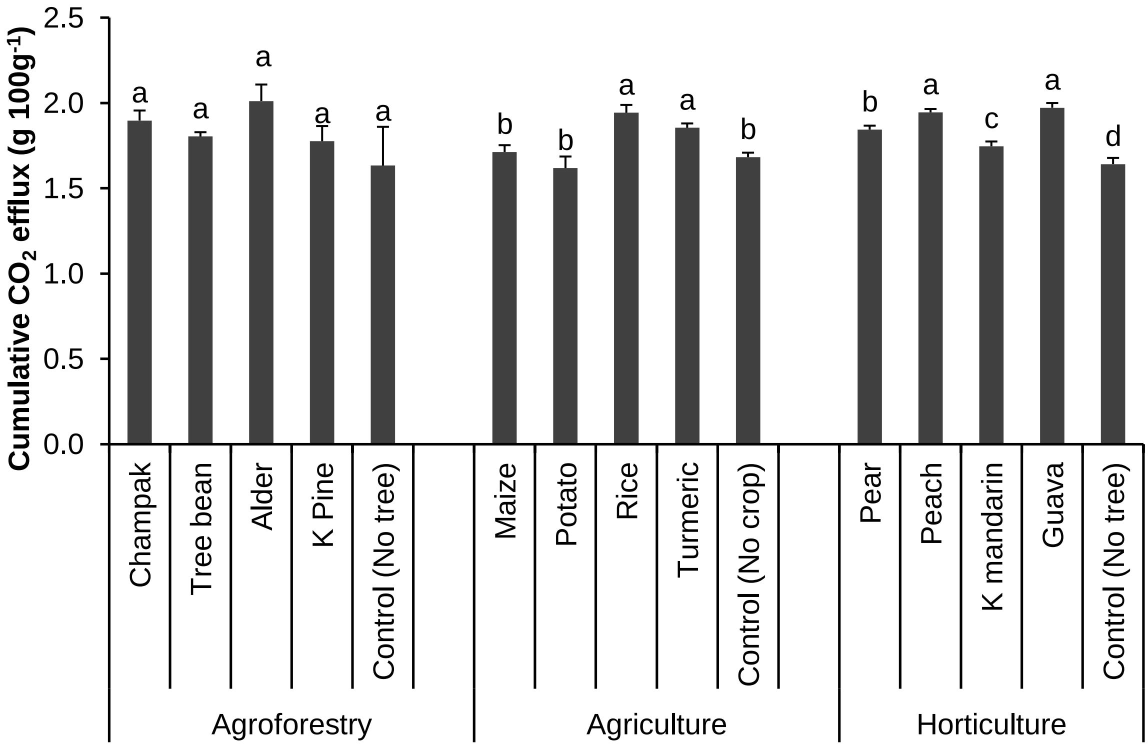


Fig. 8.5 Effect of various tree and crop species on cumulative CO_2 efflux (g $100g^{-1}$ soil) at 150 days of incubation

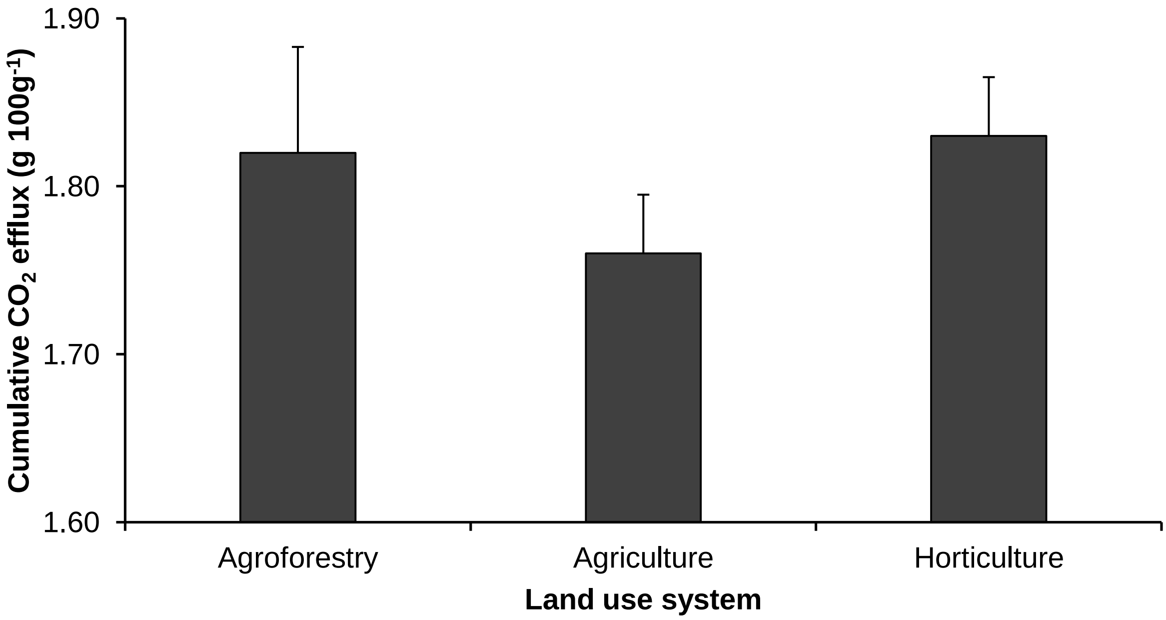


Fig. 8.6 Mean cumulative CO_2 efflux (g $100g^{-1}$ soil) in soils from various main land use systems at 150 days of incubation

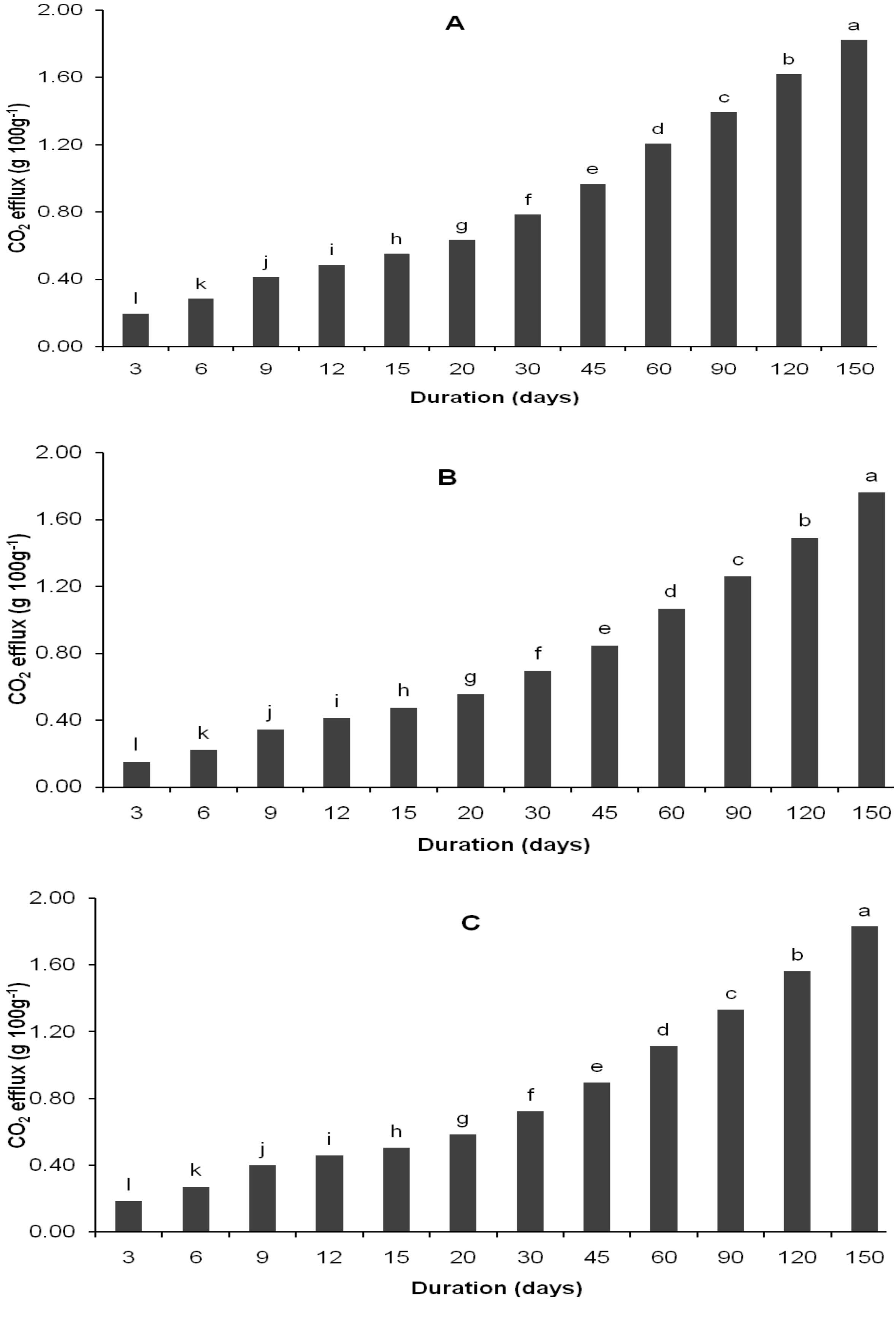


Fig. 8.7 Effect of duration on CO_2 efflux from soils under agroforestry (A), agriculture (B) and horticulture (C) land use systems

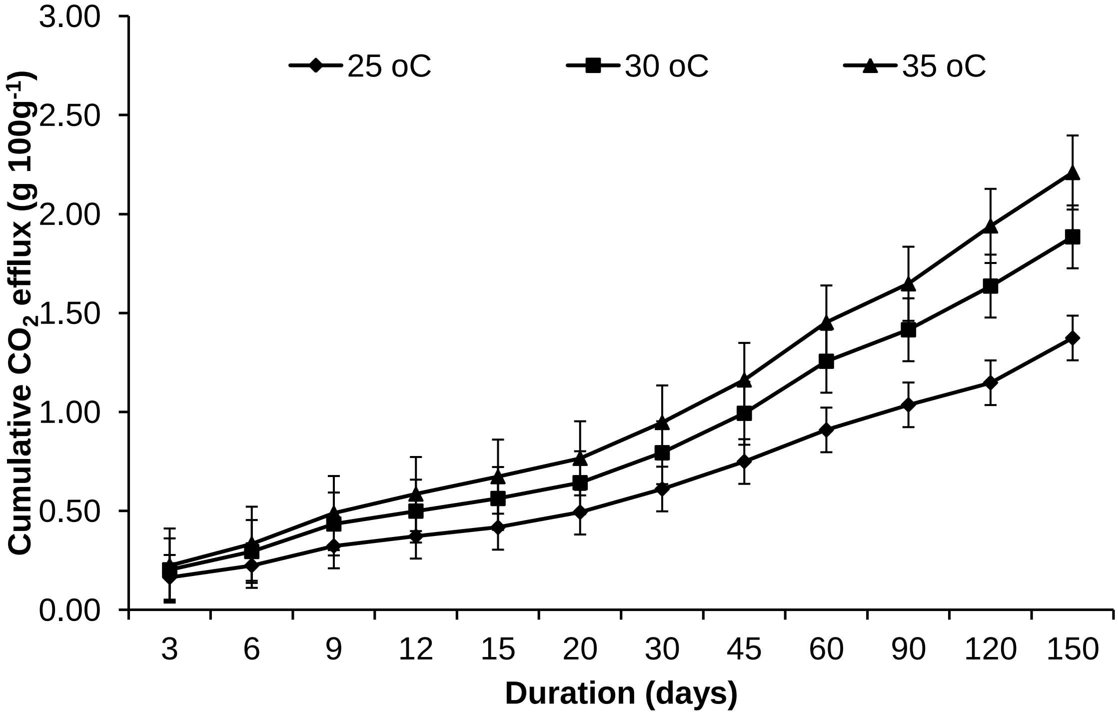


Fig. 8.8 Effect of temperature and duration on cumulative CO_2 (g 100g^{-1}) efflux under agroforestry land use system

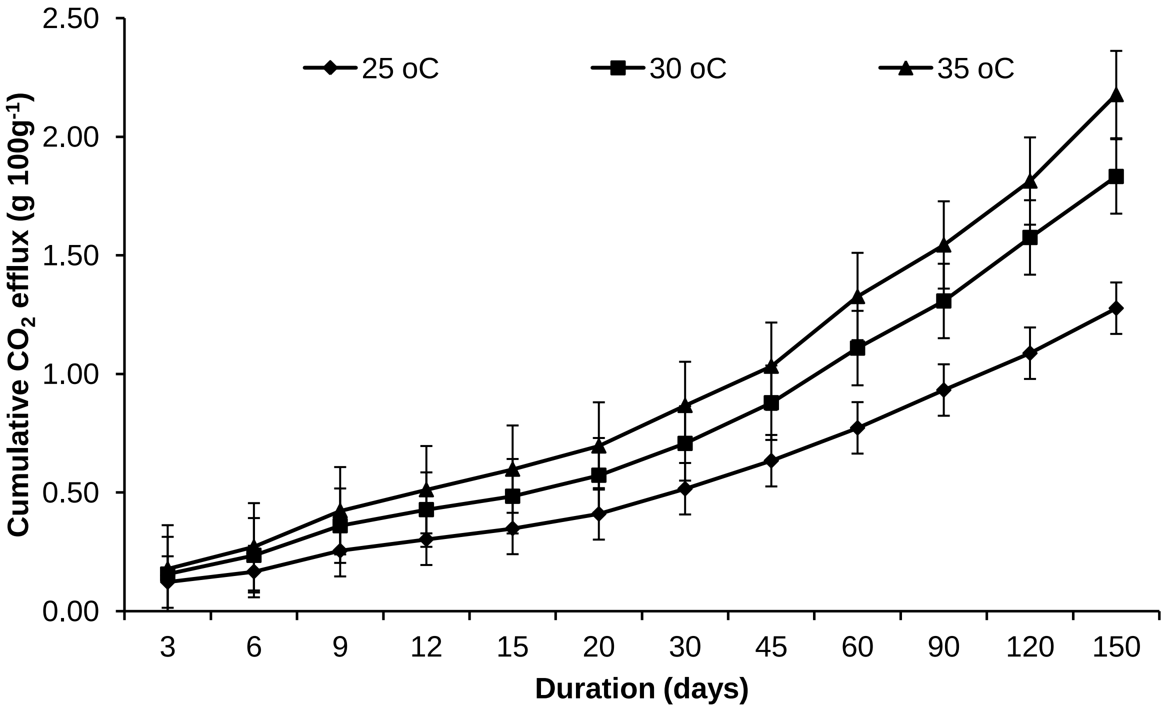


Fig. 8.9 Effect of temperature and duration on cumulative CO_2 (g 100g^{-1}) efflux under agriculture land use system

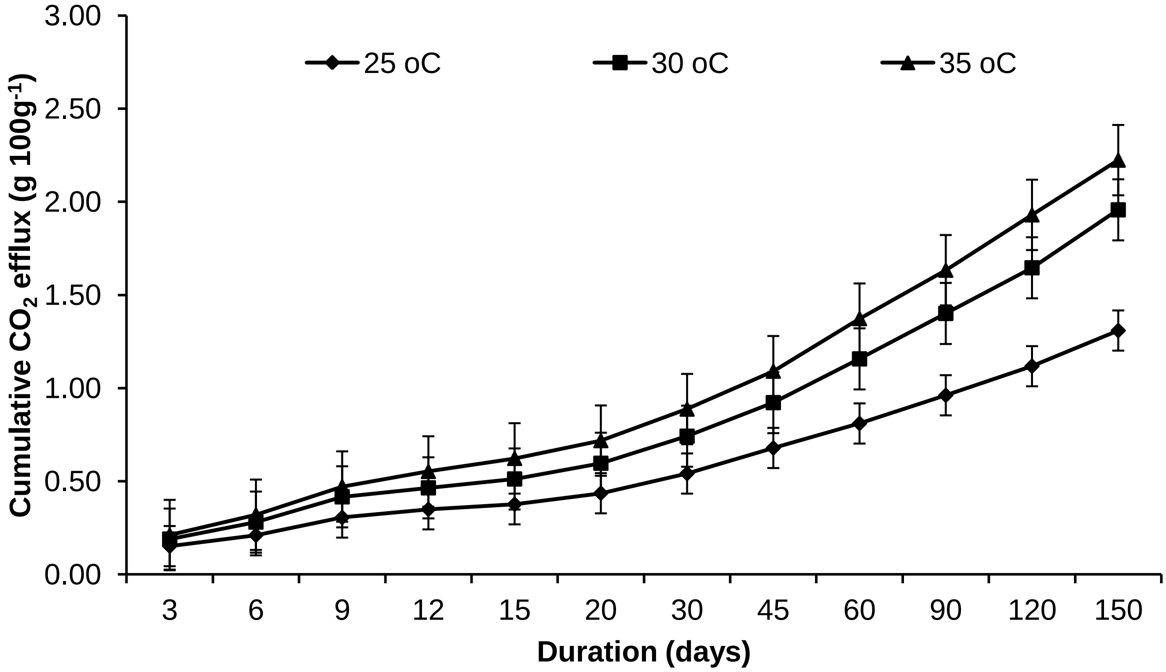


Fig. 8.10 Effect of temperature and duration on cumulative CO_2 (g 100g^{-1}) efflux under horticulture land use system

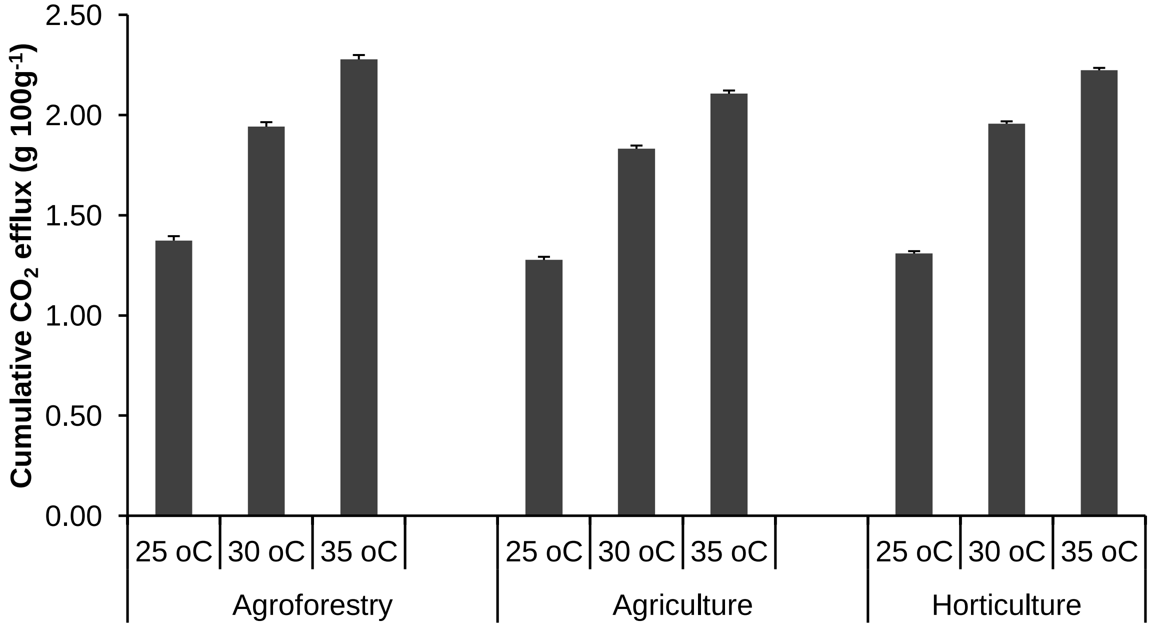


Fig. 8.11 Effect of temperature on mean cumulative CO_2 (g 100g^{-1}) efflux under various land use system at 150 days of incubation

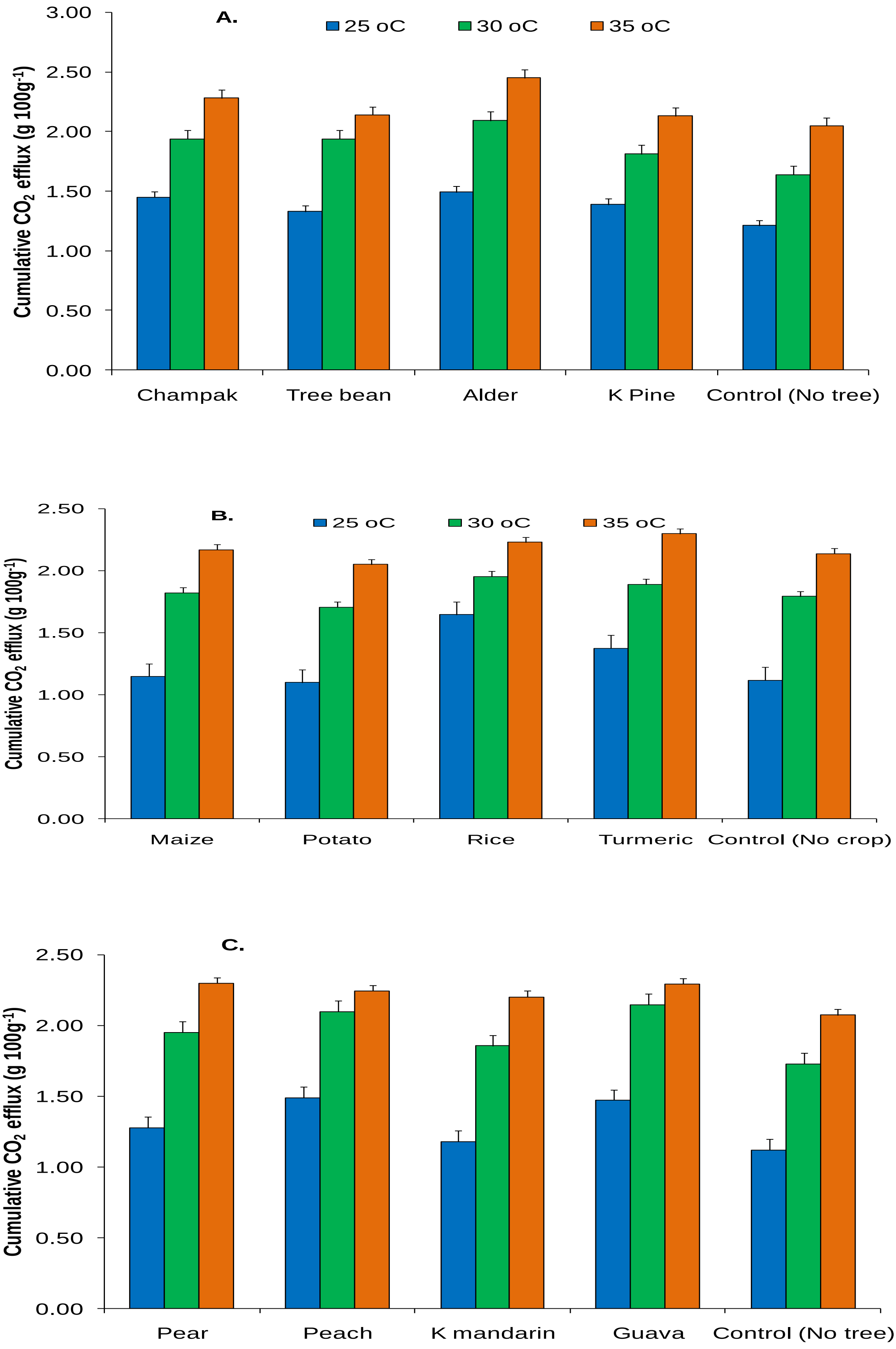


Fig. 8.12 Effect of temperature on cumulative CO_2 efflux from agroforestry (A), agriculture (B) and horticulture (C) land uses

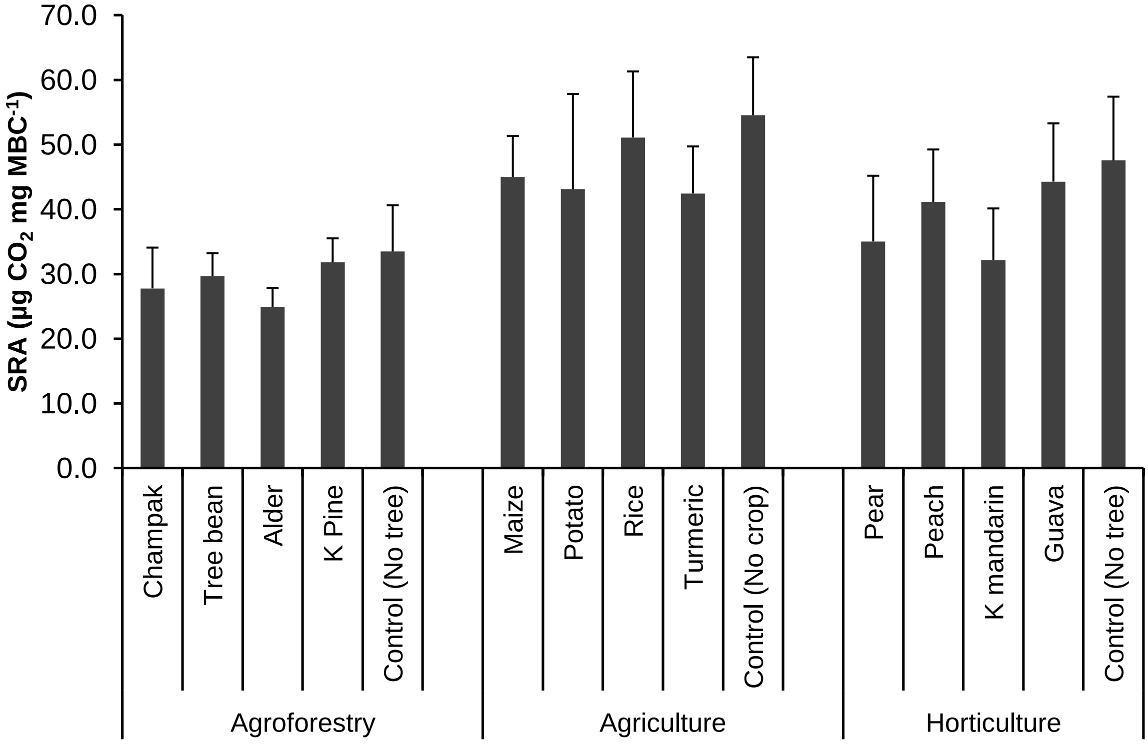


Fig. 8.13 Effect of various tree and crop species on specific microbial respiratory activity (µg CO_2 mg^{-1} MBC) at 150 days of incubation

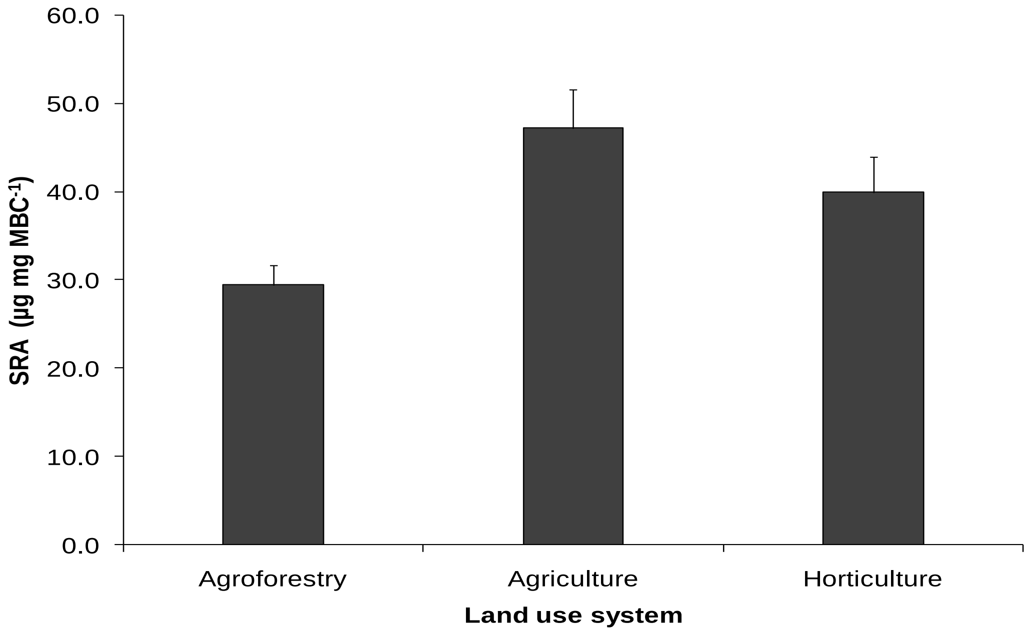


Fig. 8.14 Mean specific microbial respiratory activity (µg CO_2 mg^{-1} MBC) under various land use systems at 150 days of incubation

CHAPTER IX

DISCUSSION

Soil organic matter is an important component of the agroecosystem because of its role in the dynamics of greenhouse gases (Kirschbaum, 2000). It affects soil physical, chemical, and biological properties and hence is regarded as a key attribute of soil fertility (Verma et al., 2010). In terrestrial ecosystems, land use change or management practices can disrupt SOM dynamic equilibrium and produce a marked effect on soil organic C stocks and fractions (Lal, 2004). In order to study the dynamics of SOC under different land uses, three main land uses *viz*., agroforestry, agriculture and horticulture were identified in the East Khasi hills of Meghalaya, India. Under each land use systems, four different crops or tree species (Champak, Tree bean, Alder and Khasi pine under agroforestry land use; Maize, Potato, Rice and Turmeric under agriculture land use; Pear, Peach, Khasi mandarin and Guava under horticulture land use) were selected which are commonly practiced in north-eastern India. For the comparison, control plots (natural fallow) were also selected for each land use in the study site. Different parameters such as SOC stocks, carbon fractions (total organic carbon, particulate organic carbon, labile C, non-labile C and microbial biomass carbon), stability of the soil aggregates for carbon and nitrogen, aggregate cementing agents like carbon, nitrogen, polysaccharides and glomalin, CO_2 efflux from soils were evaluated under the selected land use systems. In addition to this, the possible changes in soil fertility status under different land use systems were also looked into.

In agreement with the several previous reports (Saha et al., 2007; Tripathi et al., 2009), agroforestry land use systems in the current study improved the soil physical properties such as mean weight diameter (MWD), moisture content and bulk density (BD). Under agroforestry land use, MWD increased by 43% and moisture content by 9% while soil BD decreased by 14%. On the other hand, land conversion from fallow to different land uses such agroforestry, agriculture and horticulture increased the MWD and soil moisture content while decreasing the soil BD. The overall improvement in soil physical properties under different land use systems may be attributed to the effect of living biomass of trees, as there was a constant addition of organic matter to the soil through decaying the large volume of dead roots as well as leaf litter thus forming stable aggregates (Saha et al., 2007; Saha et al., 2012). Abbasi and Rasool (2005) and Kizilkaya and Dengiz (2010) in their study explained that the loss of organic matter by cultivation or relatively less annual accumulation of OM in agriculture land use resulted in a lower MWD and moisture content and relatively higher soil BD compared to the other two land use systems.

Despite a significant decline in soil pH (9-16%), the nutrient availability in the soil was substantially increased by the adoption of agroforestry tree species compared to control plots as well as horticulture and agriculture land uses. However, agriculture crops and horticulture tree species increased the soil pH by 5 and 16%, respectively as compared to control plots. This result finds some support from the earlier observation of Islam and Weil (2000); Tripathi et al. (2009); and Uria et al.

(2011) who have described that the reduction of soil pH under agroforestry or forest land use systems could be due to the pre-weathered parent materials, the amphoteric nature of aluminium and the intense leaching of basic cations during rainfall, decomposition of organic materials and the nitrogen fixing ability of the agroforestry tree species like *Alnus nepalensis* and *Parkia roxburghii*. Furthermore, compared to control plots, all the tree and crop species increased the SOC, primary nutrients (N, P and K), secondary nutrients (Ca, Mg and S) and also the micronutrients (Fe, Mn and Cu) in the current study. In general the soil OC and nutrient status of the studied land use systems followed the order of agroforestry >horticulture> agriculture. The changes in the SOC and nutrient content of the land uses may be ascribed to the accumulation of SOC under tree species depends on the quantity as well as quality of chemical composition (lignin/nitrogen ratio, carbon/nitrogen ratio, cellulose, hemi-cellulose etc.) of tree roots and litter and varies widely as a function of climate and soil type (Parton et al., 1987; Saha et al., 2007). On the other hand, in general, agriculture utilization of soils has been found to decrease their OC and available nutrient contents through disruption of the equilibrium between the competing processes of humus formation and mineralization, absence of continuous vegetation cover and also due to the continuous ploughing and cultivation which, in turn, reduces the level of SOC and enhances surface run-off and mineralization (Dalal et al., 2003; Saviozzi et al., 2001). According to Smith et al. (2000), agriculture soils are considered to have lost about 25-35% of their OC due to cultivation. Guo and Gifford (2002) and Murty et al. (2002) report that the magnitude of the SOC loss may vary from 25 to 75%, depending on the antecedent C pool, land use, management and climate. SOC and nitrogen losses are caused by a number of factors, including lower organic matter input to soil, increases in mineralization, soil erosion and leaching (Guo and Gifford, 2002). In addition, most of the soil organic matter produced in cultivated lands was removed with harvest while crop residues left over the soil and were placed under the soil with plough (Kizilkaya and Dengiz, 2010). In agreement with Berthrong et al. (2009), the decline in basic cations like K, Ca and Mg under agriculture land use in this study may be explained by weathering, intensive cultivation, leaching and generally harvesting all parts of the crop for different purposes that affects the distribution of exchangeable these cations in the soil systems and enhance their depletion.

Like soil physical and chemical properties, soil microbial biomass carbon (MBC) was also significantly affected by the land use changes. Soil MBC increased by nearly 34, 31 and 26 per cent under AF, agriculture and horticulture land uses, respectively compared to their control plots. Conversely, 56% increase in soil MBC under AF land use was observed compared to agriculture land use. The chief contributory factor for the higher soil MBC in the AF land use than the other land uses seems to be the greater availability of nutrients due to the addition of higher plant litters and root biomass (Arunachalam and Pandey, 2003). This was clearly evident from the strong positive correlations between MBC and SOC (r = 0.803**, P < 0.01). Zhang et al. (2011) and Wanga and Wanga (2011) also observed a close relationship between organic C content and microbial biomass in different soils but our results were in contrast with the findings of Pietri and Brookes (2008) who reported no correlation between SOC and MBC due to the changes of land uses. The reduced organic inputs through crop residues along with tillage practices could be the possible reason for lowest MBC in agriculture land use than AF and horticulture land uses. According to Xiu-Mei et al. (2008) and Logah et al. (2010), crop residues can have a larger effect on soil microbial biomass and activity, which, in turn, affect the ability of soil to supply nutrients to plants through soil organic matter turnover. The ratio of MBC to SOC (microbial quotient) indicates the proportion of organic C that may be readily metabolized. It usually falls within the range of 1–4% (Sparling, 1992). Our study showed that the ratio of MBC to SOC ranged from 1.61–2.15%, which lies well within the range reported. In comparison with agriculture soils, agroforestry and horticulture land uses showed higher proportions of MBC to SOC (1.67-2.15%) and this can be explained on the basis of the fact that more diversified organic substrate production and input under these different land use systems support more interdependent food web which allows the maintenance of higher MBC per unit soil organic C (Anderson and Domsch, 1989). According to Anderson and Domsch (2010), the increased organic inputs to agroforestry land use in this study increased the microbial activity and caused higher conversion to microbial biomass leading to better stability of organic carbon in agroforestry land use system.

According to Lal (2004), a good farming practice can decrease soil CO_2 evolution in to the atmosphere and enhance soil fertility and thus productivity. This is most essential in tropical and subtropical regions of our country where the soils are inherently low in organic carbon content and the production system is fragile. Amongst the three land uses studied, greater increase of soil TOC content was observed under agroforestry plantation and lowest under agriculture crops. This result is conformity with Percival et al. (2000) who have recorded the highest TOC in soils under agroforestry land use due to the chemical stabilization of organic carbon in those soils. On the other hand, lack of vegetation cover, no organic input and high erosion due to rain’s direct impact on the surface soil resulted in low SOC could be the reasons for the lowest TOC content in control plots (Saha et al., 2011).

Identification and quantification of labile C fractions in various ecosystems and their responses to land use changes have important implications in understanding SOC stability and also serve as an indicator or even as a verification tool for SOC changes in terms of accounting for C stocks in the Kyoto Protocol. The labile C constitutes a small portion in TOC. Due to its rapid mineralization (because of lack of protection by soil colloids) (Turchenek and Oades, 1979), labile C responds to a greater extent to the change of land use (Jinbo et al., 2006). In conformity with the previous findings of Chen et al. (2004a) and Xu et al. (2008), the levels of labile C in the soil in our study significantly increased after the tree and crop species compared to control. In addition to the differences in microclimatic conditions, explanations given for the differences between the three land uses have included differences in the ground vegetation cover, quantity and quality of organic matter inputs to soils. Non-labile C also followed the similar trend of labile C. The balance between labile and non-labile C was disturbed due to the land use change and the non-labile C was constantly transferred and decomposed, leading to the decrease of both C fractions under agriculture land use (Xinyu et al., 2006). Similar to labile and non-labile C, POC and $KMnO_4$ oxidizable C were also highest under agroforestry systems than horticulture and agriculture land uses. The highest accumulation of SOC in agroforestry land use might have increased the aggregation which can effectively protect POC from the decomposition and enhanced the POC accumulation in agroforestry land use compared with other land uses (Blanco-Caqui and Lal, 2004; Mao et al., 2011). The increased POC and $KMnO_4$ oxidizable C accumulation under these land uses also suggests that soils under these land uses build active C pools (Saha et al., 2011). Yang et al. (2009) also reported that cultivation significantly decreased both of these fractions after forest conversion. This demonstrates that it is desirable to understand the impacts of labile C pools on soil quality under different land uses or management practices because of their highest sensitivity to land use changes as evident above and also for maintaining soil physical and chemical properties and fertility (Blair, 2000; Nahrawi et al., 2012).

SOC stocks estimation provide a better understanding of the terrestrial reservoir of SOC far beyond the general objectives of C sequestration in soils and the detrimental effects of global warming. The SOC stocks present in the soil represent a dynamic balance between the input of dead plant materials and the loss from the decomposition. SOC stocks values under different agricultural land uses and management practices significantly decreased downward the soil profile. Martin et al. (2010) and Mosquera et al. (2012) also reported decreased SOC stocks down the soil layers under various land use systems. The greater SOC stocks under agroforestry and horticulture fruit trees in our study indicates that these two land uses are beneficial for the prevention of water and wind erosions and because it increased SOC contents and stocks values within the 75cm profiles and therefore it is an ideal option for land uses and management practices to improve income, sustain soil quality, and sequestrate carbon.

Carbon management index (CMI) is an index for SOM changes induced by soil management practices, and a systematic and sensitive monitoring method for SOC change. It could reflect the degree of degradation and regeneration of soil quality (Loginow et al., 1987). CMI combined index and lability of soil carbon pool under anthropogenic influence, and could reflect environmental effects on quantity changes of total organic carbon and labile C. Thus, CMI could comprehensively and dynamically reflect the environmental effects on SOC quality (Xu et al., 2006; Gong et al., 2009). The results of our investigation further proved that soil carbon pool and CMI were significantly influenced by agroforestry, horticulture and agriculture land use systems, and CMI could comprehensively and dynamically reflect the effect of these land use change and management practices on soil carbon pool and fertility. The highest CMI in our study followed the order of Agriculture >Horticulture >Agroforestry indicating land conversion from fallow to agriculture crops cultivation have more sensitivity to the changes in soil organic carbon and other fractions than other land uses. This also indicates that due to its high sensitivity if agriculture lands are not properly managed more losses of SOC and other C fractions particularly labile C fractions will occur which may have direct impact on global climate change.

The carboxylic OH and phenolic OH values in soil humic acids were within the ranges reported by Tan (2003) and Ch'ng et al. (2011) and ranged from 3.2-6.8 and 2.8-4.6 meq g^{-1} humic acids, respectively. In this study, carboxylic OH groups contributed 62, 52 and 56% of total acidity in AF, agriculture and horticulture land use systems, respectively. These results are in agreement with the findings of Plaza et al. (2006) who have reported in their study that about 51-64% of total acidity of soil humic acids was contributed by carboxylic OH groups. The total acidity (representative CEC) of humic acids was found to be in the range of 5.2-11 meq g^{-1} humic acids, a value that was found to be within the range reported by other workers (Tan, 2003; Ch'ng et al., 2011). Agroforestry land use showed lower E4/E6 ratios than agriculture and horticulture land uses thus confirms that humic acids under agroforestry tree species are associated with a relatively large molecular size or high molecular weight (i.e. a large molecule) with high degree of aromatic condensation. This molecule has high carbon content, but relatively low in oxygen (not measured), COOH groups, and total acidity. A high E4/E6 ratio, by contrast, in agriculture land use indicates a smaller molecule which contains less carbon but more oxygen, COOH groups, and total acidity.

The C and N contents and the C/N ratios of humic acids, important indicators of the aromaticity and level of organic material decomposition, were within the ranges reported for other soils (Stevenson, 1994; Spaccini et al., 2006; Pospisilova et al., 2008) and varied substantially due to large variations in the degree of humification among the tree and crop species selected for this study. C and N contents of humic acids in agriculture lands decreased due to oxidative degradation of C and N compounds upon cultivation. The C/N atomic ratios of humic acids extracted from soils under agriculture land use were higher than that of humic acids extracted from soils under AF and horticulture land uses indicating a higher degree of humification (Guggenberger et al., 1995) and/or a greater microbial contribution for agroforestry land use. A similar trend has been observed for whole organic matter from forest soils (Nierop et al., 2001; Allard, 2006).

Soil aggregates, which have significant influence on soil physical and chemical properties, are the most basic units of soil structure and an important component of the soil (Tisdall and Oades, 1982). Aggregates are formed by the various organic cementing agents like polysaccharides, glomalin and fungal hyphae (Tisdall and Oades, 1982). In our study, the various aggregate cementing agents *viz*., organic carbon, nitrogen, total and dilute acid extractable polysaccharides, and glomalin were determined in bulk soil, macro- and microaggregates. In conformity with several previous studies (Spaccini et al., 2001; Ashagrie et al., 2007), all these fractions were significantly higher under AF land use and were greater in macroaggregates than microaggregates. On the contrary, agriculture land use showed decline in all these parameters in both the aggregate sizes. These results could be related to the larger amount of living and decaying plant roots and litter addition under agroforestry land use that would have been rapidly destroyed by cultivation under agriculture land use (Six et al., 2004). An increase of organic C with increasing aggregate sizes in our study also confirmed the aggregate hierarchy, as proposed by Oades (1984). Compared with the control plots, aggregate stability was improved under tree and crop species. The lower amount of aggregate carbon and nitrogen in the control plots was due to the relatively less carbon inputs that are added to these soils either through external addition or by way of crop residue additions and variations in soil properties. Studies on the mass distribution of organic carbon among aggregate classes (Cambardella and Elliott, 1993; Six et al., 2000) suggest that an increase in soil aggregation through adoption of best management practices usually leads to carbon sequestration (Puget et al., 2000). The greater degree of physical protection of SOC by smaller aggregates resulted in lower C and N losses than macro-aggregates (Conant et al., 2004).

Incubation experiment conducted to the study the stability of aggregates under different temperature revealed that both C and N were lost more from macroaggregates than microaggregates. It was also observed that the per cent losses of C and N from the aggregates were greater under agriculture land use and lower under horticulture land use. This is clearly evident from the lower and higher rate of reactions under AF land use and macroaggregates, respectively. This result demonstrates that highest stability of aggregates under AF land use and lowest stability under agriculture land use. Under AF land use, aggregates physically protect soil organic matter by forming physical barriers between microbes and enzymes and their substrates and controlling food web interactions and consequently microbial turnover (Elliott and Coleman, 1988). On the other hand, lower organic matter quality and the destruction of macroaggregates by tillage or other management practices causing exposure of the inner core of particulate organic carbon facilitating rapid decomposition by microorganisms of this important organic carbon reserve in soil under agriculture land use (Six et al., 2004).

The sensitivity of soil microbial processes to temperature changes and the implications of these responses for the global soil carbon dynamics are of foremost importance with respect to the climate change. The relationships between decomposition rates and temperature or the temperature sensitivity of the C and N mineralization in aggregates were investigated in the present study using Q_{10} values and activation energy (AE). In AF land use system there was an increase in Q_{10} values and AE amongst the three land use systems; and values were higher under tree and crop species in relative to control plots. The higher amounts of carbon and nitrogen in soils under agroforestry tree species, becoming available at higher temperatures, could sustain and maintain a higher reaction rates over a wider range of temperatures compared to agriculture land use. The smaller Q_{10} values and AE for the agriculture land use and control plots were explained by Fierer et al. (2005, 2006). Dallal et al., (2003) also showed that microorganisms able to perform better under suitable organic sources can increase the reaction rates over a range of temperature increase.

Higher Q_{10} and AE in microaggregate C and N decomposition shows that at higher temperatures the microaggregates decomposes and exposes the stored carbon to decomposition. According to Hulscher and Cornelissen, (1996) and Conant et al. (2011) it can be attributed to the increased physico-chemical protection offered by microaggregates. It is also proposed that soil physico-chemical reactions, which stabilize soil carbon and nitrogen, and protect it from microbial respiration, may be accelerated by warming. Many reactions are involved in stabilization, some of which are likely to have positive activation energies, notably chemical adsorption (Schulten and Leinweber, 2000). The relationship between the substrate quality and the Q_{10} were similar for C and N mineralization. This suggests similar temperature sensitivity for both processes under equal substrate quality conditions. However, the Q_{10} values were higher for N mineralization than C mineralization as indicated in our study, pointing to differences in the use of soil organic matter pools (Koch et al., 2007).

Land use change can strongly influence soil microclimate, plant carbon allocation pattern, substrate availability and input, and hence the soil CO_2 efflux which is an useful indicator to determine energy flow patterns, specifically the mineralization of nutrients and the rate of organic material decomposition (Li et al., 2011). In our incubation experiment, highest rate of soil CO_2 efflux was observed under AF land use followed by horticulture land use and the lowest rate was found under agriculture land use. The highest organic carbon with better substrate quality and more favourable soil environment for microbial growth compared under agroforestry land use could have contributed for the highest rate of CO_2 efflux from the soil. However, the per cent rate of CO_2 efflux was highest under agriculture land use than AF and horticulture land uses. According to various studies, the changes in land use might have modified the organic carbon transformation due to the changes in substrate quality (Feigl et al., 1995), altered microbial community size (Cleveland et al., 2003) and/or changes in soil porosity and water retention (Martinez and Zinck, 2004). In general, the magnitude and variability of soil CO_2 efflux that found under different land use systems may be attributed to the variation in soil biotic and abiotic factors controlling the spatial variability in soil respiration *viz*. the heterogeneity of vegetation coverage, root biomass, depth of the top soil layer, litter quantity and quality, soil organic matter, soil temperature and moisture content, etc. (Jassal et al., 2005; Tang et al., 2005).

Temperature effects along with soil moisture, substrate quality on soil CO_2 fluxes are well known and have been demonstrated in numerous studies (Iqbal et al., 2008; Iqbal et al., 2009b). We found significant increase in soil CO_2 efflux with increase in temperature gradients irrespective of the land uses including control plots. Overall, the highest increase of 74% due to increase in temperature from 25 to 35 ^{o}C under agriculture land use and this could be possibly explained by the quality of litter and root biomass, crop residues and soil properties which are relatively inferior in agriculture land use in comparison to other land uses (Saha et al., 2007). It may also be attributed to the increased mineralization of the relatively less protected organic carbon in the aggregates and easily degradable organic carbon of agriculture land use than AF and horticulture land uses (Lal, 2002). Q_{10} and Arrhenius equation were used to determine the temperature dependency or sensitivity of CO_2 efflux in this investigation. The reduced rate constant and increased AE and Q_{10} values under AF land use indicates that greater aggregate stability and protection provided for the organic carbon in the agroforestry system than agriculture land use. These results are in consistent with the findings of Davidson and Janssens (2006) who reported highest sensitivity of low quality organic carbon to temperature change. It also suggested that conversion of lands from fallow to agroforestry, agriculture and horticulture land uses increases the sensitivity of soil CO_2 efflux to temperature as evident from the present investigation (Cunyu et al., 2005).

The microbial metabolic quotient did not change significantly due to the change in land use and decreased approximately by 35% in agroforestry land use in comparison to the agriculture land use. As per the reported literature, the average rate of soil CO_2 efflux increased with agroforestry land use due to the increased carbon inputs to the soil but respiratory activity as a percent of microbial biomass carbon (metabolic coefficient) decreased in the agroforestry land use. This suggests that agroforestry was most efficient land use in preserving C in soil. This finding indicates that when the microbial biomass becomes more efficient in the use of the ecosystem resources, less CO_2 (per unit of C*mic*) is lost through respiration and a higher amount of the C is incorporated into microbial biomass, resulting on qCO_2 decrease (Silva et al., 2007; Ferreira et al., 2011). On the other hand, the highest microbial metabolic quotient in agriculture indicates the ecosystem stress in agriculture land use due to the cultivation disturbance (Mao and Zeng, 2010). Therefore, the specific respiration rate might be one of the most influential soil microbial indicators responsive to land use change.

CHAPTER X

SUMMARY AND CONCLUSIONS

Experimental observations across the various ecosystems world over have established that there have been noteworthy influences of land use/land cover change on the soil fertility status, soil organic matter (SOM) dynamics, aggregate stability and the soil CO_2 efflux in the tropical countries particularly in India. However, the impacts of conversion of fallow lands to various land uses like agroforestry, agriculture and horticulture land use systems; also the comparison of these three land use systems for their influence on the above said attributes are scarce in India; nil in Northeast India in particular.

Climate change is one of the primary concerns of the human activity. The IPCC Third Assessment Report (TAR) concludes that there is strong evidence that human activities have affected the world climate through their inappropriate land use/land cover change, unscientific soil and water conservation measures and deforestation. The atmospheric CO_2 concentration has increased from 280 ppmv in 1970 to 367 ppmv in 1999 and is currently increasing at the rate of 1.5 ppmv/year. CO_2 enrichment in the atmosphere along with other greenhouse gasses has led to an increase in the average global surface temperature of 0.6 ^{o}C since the last 19th century with the current warming rate of 0.17 ^{o}C/decade (IPCC, 2001). The observed rate of global mean temperature is in excess of the critical rate of 0.1 ^{o}C/decade beyond which the ecosystems cannot adjust. These changes may decrease the SOC pools and structural stability, increase soil's susceptibility to water run-off and erosion, and disrupt the nutrient cycles including carbon in soils, and cause adverse impacts on biomass productivity, biodiversity and the environment.

One of the possible ways of mitigating the global warming by reducing the atmospheric CO_2 emission from is the soil is sequestering organic carbon in the terrestrial ecosystems particularly in soil which is the largest carbon reservoir next to ocean. It is therefore important to understand the dynamics of SOC as well as its role in terrestrial ecosystem carbon balance and the global carbon cycles. Hence, this present investigation was under taken in north-east India, Meghalaya in particular, to examine the impacts of land conversion from fallow to agroforestry, agriculture and horticulture land use systems on soil fertility status, carbon stocks and fractions, quality of organic matter, aggregate stability and soil CO_2 efflux. Also, these three main land use systems were compared for their fertility status, potential to sequest as well as to stabilize SOC to identify the suitable land use systems in the highly degraded soils of north-east India.

For this study, three major land use namely agroforestry, agriculture and horticulture were identified in East Khasi hills of Meghalaya. Under each land uses, four major tree or crops species viz. Champak, Tree bean, Alder and Khasi pine under agroforestry; Maize, Potato, Rice and Turmeric under agriculture and, Pear, Peach, Khasi mandarin and Guava under horticulture land uses were selected along with the control plots (Natural fallow) nearby in each land use systems. Depth-wise soil samples were collected (0-15, 15-30, 30-45, 45-60 and 60-75 cm) during the month of October-November in the year 2009 and used for various analysis and incubation experiments. The summary and conclusions of the results obtained under various objectives of the study are presented here.

Physical properties

The soils of the study site were silty clay loam in texture. Soil physical properties such as bulk density, moisture content and mean weight diameter (MWD) showed notable variations amongst the land use systems, particularity in all the soil depths. The highest bulk density in surface soil layer was recorded under agriculture land use, whereas the lowest was in agroforestry (AF) land use. On the other hand, soil moisture content and MWD was highest in AF land use followed by horticulture land use and the lowest was found in agriculture land use. Overall, conversion of fallow land to AF, agriculture and horticulture land uses increased the soil moisture content by 20, 16 and 8%, respectively and MWD by 65, 46 and 58%, respectively while reducing the soil BD by 9, 11 and 4%, respectively. However, the present study reveals that adoption of various AF tree species, on the whole, increased the soil moisture content (average: 6.2%) and MWD (average: 28.3%) to the maximum while reducing soil BD (average: 13.1%) to a minimum compared to agriculture and horticulture land uses.

Chemical properties

The soil pH of the selected study area ranges from moderately to strongly acidic. Conversion of fallow land (control plots) to agriculture and horticulture land uses, in fact, increased the soil pH; however, AF tree plantations in the fallow lands decreased the soil pH to a level of about 6%. With regard to agriculture crops and horticulture fruit trees, adoption of agroforestry plantation decreased the soil pH by about 9.3 and 16.3%, respectively in the study site. The soil pH was significantly and positively correlated with the basic cations such as K, Ca and Mg while, it showed negative and significant correlation with soil micronutrients suggesting the importance of soil pH in nutrient availability to crops. Land conversion from fallow to agriculture, agroforestry and horticulture fruit trees plantation increased the soil organic carbon, available nutrients like nitrogen, phosphorus, potassium, sulphur, exchangeable Ca and Mg, and DTPA extractable micronutrients (Fe, Mn and Cu). However, the increase was prominent under agroforestry plantations excepting available phosphorus, exchangeable Ca and Mg, and Cu. These later properties were higher under the land conversion from fallow to horticulture plantation. In most cases, agroforestry tree plantation proved to be the best land use with respect to improvement in soil physical and chemical properties. Soil microbial biomass carbon was highest under agroforestry plantation (average: 424 mg kg^{-1}) as compared to other two land uses. Amongst the 13 trees and crop species studied, *A. nepalensis* (Alder) recorded highest soil MBC of 548 mg kg^{-1}, in overall as well as among the agroforestry tree species, followed by turmeric (309 mg kg^{-1}) amongst the agriculture crops and Khasi mandarin (401 mg kg^{-1}) amongst the horticulture fruit tree species.

Irrespective of the land use systems, soil moisture content, organic carbon, available N, P, K, exchangeable Ca and Mg, and DTPA extractable micronutrients contents decreased with increasing depth while soil BD increased with increasing soil depth. Soil organic carbon showed significant correlation with most of the soil nutrients, MBC, MWD and bulk density but, except with available phosphorus.

Cultivation of agriculture crops, compared to AF tree plantation and horticulture fruit trees, resulted in less build-up of soil physico-chemical and biological properties. However, conversion from fallow land to agriculture use improved soil fertility status like other two land uses but the magnitude of the benefit was highest under agroforestry tree plantation.

Carbon stocks and fractions

Total, particulate, labile and non-labile carbon contents were 27, 107, 131 and 18% higher in agroforestry tree plantations than in horticulture tree plantation and agriculture farming systems. In the agroforestry land use system Alder and Champak; in horticulture tree plantation Guava and Peach, and in agriculture land use Potato and Turmeric had significantly higher levels of these carbon fractions than the associated tree and crop species in their respective land uses. Carbon management index (CMI), a more complete indicator of soil carbon dynamics was 29 and 37% higher in agriculture land use compared to AF and horticulture land uses, respectively.

Although TOC varied significantly across the land use types, the influence of land use on particulate organic carbon, labile carbon and microbial biomass carbon were relatively highest compared to TOC. These organic matter fractions were also closely associated with other soil properties including soil pH, bulk density and available nitrogen. Thus, these labile carbon fractions such as particulate organic carbon, labile carbon and microbial biomass carbon may provide an early indication of the loss in SOM fractions most associated with soil fertility.

SOC stocks in different land use systems were markedly affected in the 0-75cm soil layers with highest stocks in the surface layer (average: 67.4 Mg ha^{-1}) and decreased sharply with increasing soil depth. We did not observe any significant effects of the land use systems on SOC stocks; however we could find the significant influence of soil depths on SOC stocks. The SOC stocks calculated for different land uses (up to 75 cm depth) as follows: : AF (54.3 Mg ha^{-1}) > Horticulture (53.6 Mg ha^{-1}) > Agriculture (49.3 Mg ha^{-1}) land uses. Soils under agriculture crops also have large potential to accumulate carbon and returning to the SOC pools if only a portion of these plant residues are removed from the lands for food or livestock fees.

Humic acids characteristics

In general, the FTIR spectra of humic acids showed some distinct characteristics and strong absorption bands in the regions from 1700-1725, 1457-1640, 1215-1225, 1014-1030, 719-754 and 517-533 cm^{-1} wavenumber. There was no significant difference in the quantities of C in humic acids regardless of the land use systems with different trees and crops of varying ages and were mainly dominated by aromatic components thus indicating the high maturity of HAs.

In general, agroforestry land use increased the total acidity, a measure of the cation exchange capacity of the humic acids, by 48 and 10% compared to horticulture and agriculture land use. The carboxylic (-COOH) and phenolic (-OH) groups of humic acids of all the land use systems followed the same trend of total acidity. Agroforestry land use increased the carboxylic OH by 48% and phenolic OH by 19% compared to agriculture land use. The relatively high E4/E6 values of HAs under agriculture land use system (5.63) indicated the prominence of aliphatic components and the HAs were of low molecular weight. Conversely, the lower E4/E6 ratios of the humic acids under agroforestry land use system (4.86) indicated the dominance of aromatic components and high molecular weight humic acids.

The average carbon and nitrogen contents of humic acids were relatively higher under agroforestry land use (62.7 and 5.5%, respectively) and decreased upon cultivation in the agriculture land use. On the other hand, the C/N ratios of humic acids were higher under agriculture land use (13.9) than agroforestry (11.4) and horticulture (12.4) land uses. Compared to trees and crop species, control plots in all the land uses showed lower values of humic acids C, N, total acidity, carboxylic and phenolic OH groups; higher values of E4/E6 and C/N ratios. On the whole, adoption of agroforestry, horticulture fruit tree and agriculture crops species in the fallow lands increased the degree of humification thus resulted in the higher aromatic condensation of humic acids and increased stabilization or quality of soil organic matter. Amongst the land use systems, agroforestry land use showed higher aromatic condensation and increased stabilization or quality of soil organic matter whereas, agriculture land use showed lower aromatic condensation and thus lower stabilization or quality of soil organic matter.

Aggregate stability and cementing agents

Land use change from fallow to agriculture, horticulture and agroforestry significantly increased all the aggregate stabilizing agents such as organic carbon, nitrogen, polysaccharides and glomalin. The effect was much greater under agroforestry land use compared to rest of the land uses. Macroaggregates were enriched with all these stabilizing agents compared to microaggregates irrespective of the land use systems. Amongst the tree and crop species, Alder (Alnus nepalensis) recorded highest values of these attributes than the other tree and crop species. Of the stabilizing agents, SOC had strongest correlation with mean weight diameter signifying that SOC was the major cementation material and had an important effect on the stability and composition of water-stable aggregates.

Incubation experiments conducted to determine the stability of carbon and nitrogen in both macro- and microaggregates with varying temperature (25, 30 and 35 ^{o}C) revealed that the rate of reaction was faster under agriculture land use and slower under agroforestry land use marking the importance of agroforestry land use in stabilizing the carbon; macroaggregates registered faster decomposition rate compared to microaggregates suggesting the higher stability of the microaggregates over macroaggregates. It was also confirmed by the higher Q_{10} values of macroaggregates than microaggregates. The increased aggregation promoted by the agroforestry land use systems may be responsible for the well protected aggregate carbon and nitrogen with lower decomposition rate under agroforestry land use. Increase in temperature increased the rate of mineralization in both aggregates and was highest at 35 ^{o}C than at lower temperatures. Owing to its higher stability, activation energies of carbon and nitrogen decomposition in microaggregates were higher than activation energies of macroaggregate carbon and nitrogen decomposition in all the land uses including control plots. Overall, the per cent mineralization of carbon and nitrogen was highest under agriculture land use compared to other land uses irrespective of the aggregates. However, the magnitude of mineralization of nitrogen was relatively higher than carbon mineralization and higher in macroaggregates than microaggregates in all the land use systems.

Soil CO_2 efflux and Specific respiratory activity or metabolic quotient

Conversion of fallow lands to agroforestry, horticulture and agriculture land uses resulted in the accumulation of organic carbon and thus significantly increased the loss of soil organic C as CO_2 emission. However, more importantly, in relative to the agroforestry and horticulture land uses, the rate of CO_2 efflux was higher in soils under agriculture land use. This study has also shown that in relative to the three land uses studied, soils from the control plots resulted in significantly more soil CO_2 per unit mass of microbial biomass signifying the importance of tree and crop species in sequestering the soil carbon by reducing soil CO_2 efflux. Furthermore, the changes in soil physical and chemical properties due to varied organic carbon inputs also significantly influenced the soil CO_2 efflux by altering the gas diffusion in the soil.

Temperature is a major factor that has a critical effect on annual carbon balance and hence, the temperature dependency or sensitivity of soil CO_2 efflux to temperature was studied in this investigation. The soil CO_2 effluxes were found to increase with increase in temperature irrespective of the land uses. We observed highest soil CO_2 efflux of 2.20 g 100g^{-1}(on average) at 35 ^{o}C which is 17 and 66% higher than the CO_2 efflux at 25 and 30 ^{o}C, correspondingly. The results also revealed that land conversion from fallow to agriculture, horticulture and agroforestry systems increased the Q_{10} values and Arrhenius activation energy (AE) suggesting the increased sensitivity of soil CO_2 efflux to land use change. On the other hand, the increased Q_{10} values and AE in the agroforestry land use in relative to agriculture and horticulture land use implies the importance of agroforestry systems in protecting the soil organic carbon from the decomposition by increasing the aggregate stability and thus soil carbon storage.

Agroforestry land uses had significantly lower microbial metabolic quotient or specific respiration activity (30.6 µg CO_2 mg MBC^{-1}) than agriculture (47.2 µg CO_2 mg MBC^{-1}) and horticulture land uses (40.0 µg CO_2 mg MBC^{-1}) signifying the dominance of this land use to preserve organic carbon in soil. It also suggests a higher accumulation of resistant organic C pool in this land use system. The highest microbial metabolic quotient in agriculture indicates the ecosystem stress in agriculture land use due to the cultivation disturbance. The significant impact of soil variables on CO_2

efflux and specific respiratory activity indicates the strong implications of this study on one of important global change phenomena of increasing atmospheric CO_2 concentration due to the change in land use and management practices.

The salient findings of the present investigation are summarized as follows:

- From the overview of all the results of the present investigation, it can be concluded that land conversion from fallow lands to agroforestry, agriculture and horticulture land uses resulted in improvements in all the physical, chemical and biological properties; carbon stocks and fractions and quality of organic carbon.
- Compared to TOC, the labile C fractions such as $KMnO_4$ oxidizable carbon, microbial biomass carbon and particulate organic carbon can be used as the sensitive indicators to measure the impact of land use change on C dynamics compared to total organic carbon.
- The stabilization of soil organic carbon and thus soil aggregates were increased under all these land use systems compared to fallow lands.
- In contrast, the soil CO_2 efflux increased under these land uses because of their highest organic carbon contents due to the increased addition of organic inputs; however, the specific microbial activity i.e. the metabolic quotient was considerably declined under these land uses compared to control plots thus indicating the overall capacity of these land use systems in sequestering carbon or releasing soil CO_2 to atmosphere.
- Amongst the three land uses studied, agroforestry tree species improved the overall soil quality, aggregate stability and sequestered more organic carbon by emitting less soil CO_2 to the atmosphere.
- Therefore, it can be concluded from our study that planting agroforestry tree or horticulture fruit trees or agriculture crops cultivation with better management practices in the fallow lands could improve the soil fertility, reduces the soil losses and increase the carbon sequestration in one hand; on the other hand, it could be recommended that inclusion of agroforestry tree species, compared to agriculture crops and horticulture fruit trees, in the land use practices that increases organic carbon and other soil properties in the ecosystems is a viable option to restore the soil fertility status for long-term productivity and also signifies the ecological benefits (mitigating global warming) through increased organic carbon sequestration in the highly degraded soils of North-east India, Meghalaya in particular.

Stocks and quality of soil organic matter under different land use systems in East Khasi hills of Meghalaya

ABSTRACT

Soils of north-eastern India are being seriously degraded and destructed due to land use change, shifting cultivation with unscientific soil managements and are usually accompanied by decreasing in concentrations of soil organic carbon (SOC) and nutrients thus soil fertility in these regions. The purpose of this investigation was to assess the impacts of land use and land use change on organic matter dynamic and carbon sequestration. We hypothesized that the conversion of fallow lands to agroforestry (AF), horticulture and agriculture land use systems favour conservation of SOC besides improving soil fertility. Also, we hypothesized that AF land use system is the superior one to sequester more carbon by emitting less soil CO_2 to the atmosphere. To test these hypotheses, an investigation was carried out in the East Khasi hills of Meghalaya, which is located at $25^{o}41'21''$ North (latitude) and $91^{o}55'25''$ East (longitude) by selecting three major land use systems viz. AF (26 years), agriculture (10 years) and horticulture (15 years) with four major crops or trees under each land uses (Champak, Tree bean, Alder and Khasi pine under AF land use; Maize, Potato, Rice and Turmeric under agriculture; Pear, Peach, Khasi mandarin and Guava under horticulture land use). A fallow land or control plot (without tree or crops) nearby each major land use was selected for comparisons. Soil samples were collected from each land use systems from five depths (0-15, 15-30, 30-45, 45-60 and 60-75 cm) and used for the analysis of various soil fertility parameters, carbon stocks and fractions, and humic acid characterization. Incubation experiments were also carried out to study the thermal stability of aggregates and soil CO_2 efflux with varying temperatures.

The texture of experimental soils is silty clay loam. Soil pH varied from moderately to strongly acidic. SOC, available nitrogen, phosphorus and potassium content increased exponentially following land conversion from fallow to AF, horticulture fruit trees plantation and agriculture crop cultivation. Amongst the three major land use systems, AF land use recorded maximum value of these attributes followed by horticulture land use and the lowest was under agriculture land use. Agriculture land use systems lost SOC, available N and K content by 30.4, 17.8 and 17.2 %, respectively as compared to the AF land use. Soil bulk density, moisture content and mean weight diameter and all of the soils chemical properties studied were significantly affected by land uses and was highest under AF plantation (28.3 g $100g^{-1}$ and 2.08 mm, respectively) except soil bulk density which was at lowest level (1.22 g cm^{-3}). However, the highest average mean values of exchangeable Ca (2.31 meq $100g^{-1}$), exchangeable Mg (1.01 meq $100g^{-1}$) and available P (20.3 kg ha^{-1}) and sulphur (3.83 kg ha^{-1}) were observed under the horticulture land use compared to the other two land uses.

The results showed that the contents of total organic carbon (TOC), microbial biomass carbon (MBC), labile carbon, non-labile carbon and particulate organic carbon (POC) followed the order: AF>horticulture>agriculture. However, all the carbon fractions were highest in the adopted land use

practices like AF, horticulture and agriculture systems compared to the fallow lands. TOC, POC, labile C, non-labile C, MBC and C stocks in the soils of AF land use increased by 28, 107, 132, 18.0, 56 and 10 %, respectively as compared to the agriculture lands. On the other hand, conversion of fallow lands to agriculture, horticulture and AF land uses significantly increased all these fractions by 27, 69, 51, 27, 31 and 35% in agriculture land use, 25, 44, 17, 38, 26 and 34 % in horticulture land use and 26, 55, 27, 46, 34 and 36 % in AF land uses.

The FTIR spectra of humic acids showed some distinct characteristics and strong absorption bands in the regions from 1700-1725, 1457-1640, 1215-1225, 1014-1030, 719-754 and 517-533 cm^{-1} wavenumber. Humic acids from agroforestry land uses recorded maximum elemental composition (C and N), total acidity, carboxylic and phenolic OH groups; conversely, the lower E4/E6 ratios of the humic acids under AF land use system (4.86) indicated the dominance of aromatic components and high molecular weight humic acids.

AF land use significantly increased all the aggregate stabilizing agents such as organic carbon, nitrogen, polysaccharides and glomalin compared to other land uses. All these attributes were higher in macroaggregates than microaggregates irrespective of the land uses. The first order rate kinetics was higher under agriculture land use; also in macroaggregate than microaggregates marking the importance of AF and microaggregates in stabilizing and protecting SOC. It was also confirmed by the higher Q_{10} values, activation energy (AE) of macroaggregates than microaggregates; also in AF land use than agriculture and horticulture land uses.

Incubation experiment on soil CO_2 efflux study showed that CO_2 efflux from soil increased with increase in temperature and was greater in soils from AF land use than the other land uses including fallow lands. However, the metabolic quotient ($q$$CO_2$) was maximum under agriculture land use. The rate of CO_2 efflux was faster under agriculture land use as indicated by the first order rate constant (0.19) compared to AF and horticulture land uses. In support of this finding, the AE and Q_{10} values were found higher under AF land use compared to agriculture land use. Furthermore, the changes in soil physical and chemical properties due to varied organic carbon inputs also significantly influenced the soil CO_2 efflux by altering the gas diffusion in the soil.

The present investigation thus concludes that planting agroforestry tree or horticulture fruit trees or agriculture crops cultivation with better management practices in the fallow lands could improve the soil fertility, reduces the soil losses and increase the carbon sequestration in one hand; on the other hand, it could be recommended that inclusion of agroforestry tree species, compared to agriculture crops and horticulture fruit trees, in the land use practices that increases organic carbon and improves other soil properties in the ecosystems is a viable option to restore the soil fertility status for long-term productivity and also signifies the ecological benefits (mitigating global warming) through increased organic carbon sequestration in the highly degraded soils of North-east India, Meghalaya in particular.

मेघालय की पूर्वी खासी पहाड़ियों में विभिन्न भूमि उपयोग तंत्रों के अंतर्गत मृदा–जैविक पदार्थ की गुणवत्ता एवं संग्रह

सार

उत्तर–पूर्वी भारत की मृदाओं का भूमि–उपयोग परिवर्तन एवं अवैज्ञानिक मृदा–प्रवंधनों सहित स्थानांतरी खेती के कारण गंभीर रूप से निम्नीकरण एवं विनास हो रहा है तथा प्रायः इसके साथ– साथ मृदा–जैविक कार्बन □एस ओ सी□एवं पोषक तत्त्वों की सान्द्रताएं भी कम हो रही हैं। इस प्रकार से इन क्षेत्रों में मृदा–उर्वरता कम होती जा रही है। इस अन्वेक्षण का उद्देश्य□मृदा जैविक पदार्थ गतिकी एवं कार्बन–प्रच्छादन पर भूमि–उपयोग परिवर्तन के प्रभावों का मूल्यांकन करना था। हमने परिकल्पना की कि परती भूमि को कृषि– वानिकी □ए एफ□□ कृषि एवं कृषि भूमि उपयोग तंत्रों में परिवर्तित करना□मृदा–उर्वरता में सुधार करने के अतिरिक्त एस ओ सी के संरक्षण सहायक है। हमने यह भी परिकल्पना की कि ए फ भूमि–उपयोग तंत्र□ वातावरण में कम मृदा–कार्बन डाइऑसाइड उत्सर्जन के कारण अधिक कार्बन के प्रच्छादन हेतु श्रेष्ठ है। इन परिकल्पनाओं के परीक्षणार्थ□मेघालय की पूर्वी खासी पहाड़ियों□जो **25 41’ 21’’’** उत्तर (अक्षांश) एवं **91 55’ 25’’** पूर्व □अक्षांश□पर स्थित हैं□में तीन मुख्य भूमि–उपयोग तंत्रों यथा□एफ □**26** वर्ष□□कृषि □**10** वर्ष □एवं उद्यानिकी □**15** वर्ष□का वरण कर तथा प्रत्येक भूमि–उपयोग के अन्तर्गत चार मुख्य फसलों अथवा वृक्षों ए एफ भूमि उपयोग के अन्तर्गत चंपक□ट्री बीन□आल्डर एवं खासी पाइन ः कृषि के अन्तर्गत मक्का□आलू□धान एवं हल्दी ः उद्यानिकी संबंधी भूमि–उपयोग के अन्तर्गत नाशपती□आड़ू□खासी मेंडेरिन एवं अमरूद□के साथ एक अध्ययन किया गया। प्रत्येक मुख्य भूमि–उपयोग के समीप एक परती भूमि या कंट्रोल □बिना किसी फसल या वृक्ष के□का तुलना हेतु वरण किया गया। प्रत्येक भूमि–उपयोग तंत्र से पाच□गहराइयों □**0–15**□**15–30**□**30–45**□**45–60** एवं **60–75** सेमी□से मृदा–नमूने एकत्र किए गए तथा उनका उपयोग कई मृदा–उर्वरता प्राचलों के विश्लेषण कार्बन संग्रहों एवं प्रभाजों तथा हयूम्कि अम्ल के अभिलक्षण के लिए किया गया। परिवर्तनशील तापमानों पर समुच्चयों के ताप–संबंधी स्थयित्व एवं मृदा कार्बनडाइऑक्साइड बहिर्वाह के अध्ययन हेतु उदभवन प्रयोग भी किए गए।

प्रायोगिक मृदाओं की बनावट सिल्टी मृत्तिका दोमट थी। मृदा का पी एच मान मध्यम स्तर से लेकर प्रबल अम्लीय था। परती भूमि से ए एफ में बदलने□उद्यानिक फलदायी वृक्षों के रोपण तथा कृषि फसलों की खेती करने के बाद एस ओ सी□उपलब्ध नाइट्रोजन□फॉस्फोरस एवं पोटेशियम अंश में चरघातंकीय रूप से बढ़ोत्तरी हुई। तीन मुख्य मृदा–उपयोग तंत्रों में के मध्य□ए एफ भुमि– उपयोग उद्यान संवंधी फलदायी वृक्षों का स्थान रहा तथा इन गुणों का न्यूनतम मान कृषि भूमि उपयोग तंत्र में था। ए एफ भूमि ने एस ओ सी□उपलब्ध नाइट्रोजन एवं पोटेशियम

अम्श □मशः **30.4□174** एवं **17.2%** खो दिया। मृदा स्थूल घनत्व□जमी अंश एवं औसत भार व्यास तथा अध्ययन किए गए मृदा के सभी रासायनिक गुण□मृदा-उपयोगों से प्रभावित हुए तथा ए एफ वृक्षारोपण के अन्तर्गत मृदा स्थूल घनत्व को छोदकर जो कि न्यूनतम स्तर □1.**22** ग्रा सेमी **-3**□पर था□मृदा के अन्य सभी रासायनिक गुण अधिकतम □□मशः **28.3** ग्रा प्रति **100** एवं **2.08** मिमी□थे। वैसे अन्य दोनों भूमि-उपयोगों की तुलना में उद्यान-संबंधी भूमि-उपयोग के अन्तर्गत विनिमय योग्य कैल्शियम □2.**31** एम ई क्यू प्रति **100** ग्रा□□विनिमय योग्य मैग्नीशियम □1.**01** एम ई क्यू प्रति **100** ग्रा□□उपलब्ध फॉस्फोरस □3.**83** कि ग्रा प्रति है□एवं सल्फर □3.**83** कि ग्रा प्रति हे□के औसत माध्य मान अधिकतम पाए गए।

परिणामों ने दर्शाया कि कुल जैविक कार्बन □टी ओ सी□सूक्ष्मजीव संबंधी जैवमात्रा कार्बन □एम बी सी□□अस्रि कार्बन अपरिवर्ती कार्बन एवं विविक्त जैविक कार्बन □पी ओ सी□के अंशों ने निम्नलिलिखित □म का अनुसरण किया ः ए एफ>उद्यानिकी कृषि। वैसे परती भूमि की तुलना में अपनाई गई भुमि-उपयोग ि□याओं यथा□ए एफ□उद्यानिकी एवं कृषि- तंत्रों में□कार्बन के सभी प्रभाज अधिकतम पाए गए। कृषि-भूमियों की तुलना में□ए एफ भूमि उपयोग के अन्तर्गत मृदाओं के टी ओ सी□पी ओ सी□अस्थिर एवं कार्बन संग्रहों में □मशः **28□107□132□18□56** एवं **10%** की बढ़ोत्तरी देखी गई। दूसरी ओर, परती भूमि के कृषि, उद्यानिकी एवं एफ भूमि उपयोग में बदलने से इन सभी प्रबाजों में महत्वपूर्ण बढ़ोतरी हुई यथा□कृषि भूमि उपयोग में □मश ः **27□69□51□27□31** एवं **35%** की उद्यान-संबंधी भूमि उपयोग से **25□44□17□38□26** एवं **34%** की तथा ए एफ भूमि-उपयोग में **26□55□27□46□34** एवं **36%** की बढ़ोत्तरी पायी गई।

ह्यूमिक अम्लों के□एफ टी आई आर स्पेक्ट्रा ने **1700-1725□1457-1640□1215-1225□1014-1030□719-754** एवं **517- 533** प्रति सेमी तरंग संख्या वाले क्षेत्रों में शक्तिशाली अधिशोषण पट्टिकाएं तथा कुछ सुव्यक्त गुण दर्शाए।

कृषि वानिकी भूमि उपयोग से ह्यूमिक अम्लों में तत्त्व संबंधी संघटन □कार्बन एवं नाइट्रोजन□कुल अम्लीयता□कार्बोज़ायलिक एवं फीनालिक ओ एच समूह अधिकतम रेकार्ड किए गए ; इसके विपरीत □ह्यूमिक अमलों के कम ई**4** क्रई**6** अनुपातों ने ऐरोमैटिक घटकों एवं उच्च आण्विक भार वाले ह्यूमिक अमलों की प्रभाविता को दर्शाया।

अन्य भूमि उपयोगों ककी तुलना में□ए एफ भूमि-उपयोग द्वारा□जैविक कार्बन□नाइट्रोजन□पॉलीसैकेराइड्स एवं ग्लोमेलिन□में महत्वपूर्ण रूप से अधिक बढ़ोत्तरी देखी गई। भूमि उपयोगों से अप्रभावित□सूक्ष्म समुच्चयों की तुलना में□

स्थूल समुच्चयों में ये सभी गुण अधिक थे। कृषि भूमि –उपयोग के अन्तर्गत तथा सूक्ष्म-समुच्चयों में प्रथम □म दर गतिकी अधिका थी। जिससे एस ओ सी के स्थायीकरण एवं संरक्षण में ए एफ एवं सूक्षम समुच्चयों के महत्व का पता चलता है। सूक्ष्म समुच्चयों की तुलना में स्थूल समुच्चयों की स□्रण ऊर्जा □ए ई□के उच्चतर क्यू **10** मानों द्वारा भी इसकी पुष्ठि हुई ः कृषि एवं उद्यान संबंधी भूमि –उपयोगों की तुलना में ए एफ में भी ऐसा ही था।

मृदा–कार्बन डाइऑसाइड–बहिर्वाह पर उद्भवन-प्रयोग ने दर्शाया कि तापमान में बढ़ोत्तरी के साथ मृदा से कार्वनडाइऑसाइड का बहिर्वाह बढ़ गया तथा यह परती भूमि सहित अन्य भूमि–उपयोगों की तुलना में□ए एफ भूमि–उपयोग की मृदाओं से यह अधिक □**2.21** ग्रा प्रति **100** ग्रा□था। वैसे कृषि भूमि-उपयोग के अन्तर्गत उपापचयी भागफल □क्यू सी ओ **2**□ अधिकतम था। ए एफ एवं उद्यान संवंधी भूमि–उपयोग की अपेक्षा कृषि भूमि–उपयोग में कार्व न डाइ–ऑक्साइड की बहिर्वाह–दर अधिक तेज थी जैसा कि प्रथम □म–दर स्थिरांक □**0.19**□ने दर्शाया। इन परिणामों के समर्थन में□कृषि भुमि–उपयोग की तुलना में□ए एफ भूमि –उपयोग के अन्तर्गत ए ई एवं क्य **10** मान अधिक पाए गए। इसके अलावा□परिवर्तित जैविक कार्बन निवेशों के कारण मृदा में गैस–विररण में परिवर्तन द्वारा मृदा का कार्बन डाइऑसाइड बहिर्वाह महत्वपूर्ण रूप से प्रभावित होता है। इस प्रकार□प्रस्तुत अध्ययन से यह निष्कर्ष निकलता है कि परती भूमि पर कृषि वानिकी से संबंधित वृक्षारोपण अथवा कृषि –फसलों की खेती के साथ–साथ बेहतर प्रबंधन ि□्रयाओं से मृदा उर्वरता में सुधार□मृदा क्षति में कमी तथा कार्बन के विविक्तकरण में बढ़ोत्तरी होती है वहीं दूसरी आर इस बाद की संस्तुति की ज सकती है कि भमि उपयोग ि□्रयाओ में कृषि फसलों एवं उद्यान संबंधी फलदायी वृक्षों के कृषि वानिकी से संबंधित□ वृक्ष –प्रजातियों का समावेश किया जाना चाहिए जिससे परिस्थिति –तंत्रों में जैविक कार्बन एवं मृदा के अन्य गुणों में बढ़ोात्त्री होती है तथा उत्तर–पूर्व भारत□विशेष रूप जैविक कार्बन–विविक्तकरण में बढोतरी के माध्यम से यह पारिस्थिकीय – लाभों □वैश्विक तापन का शमन करने□को दर्शाता है तथा दीर्घावधि उत्पादिता हेतु□मृदा–उर्वरता स्तर की पुनर्प्राप्ति का यही एक सक्षम विकल्प है ।

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