CHANGES IN SOIL AGGREGATION, MICROBIAL POPULATION, CARBON AND NUTRIENTS UNDER DIFFERENT LAND USE SYSTEMS OF MIZORAM

A THESIS SUBMITTED IN PARTIAL FULFILMENT OF THE
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CHANGES IN SOIL AGGREGATION, MICROBIAL POPULATION, CARBON AND NUTRIENTS UNDER DIFFERENT LAND USE SYSTEMS OF MIZORAM

BY

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Submitted
In partial fulfilment of the requirement of the degree of
Doctor of Philosophy in Forestry of Mizoram University, Tanhril, Aizawl

DECLARATION

I, **Mr. Etsoshan Y Ovung**, hereby declare that the subject matter of this thesis entitled, "*Changes in soil aggregation, microbial population, carbon and nutrients under different land use systems of Mizoram*" is the record of work done by me and that the contents of the thesis did not form basis for the award of any previous degree to me or anybody else, and that the thesis has not been submitted by me for any research degree in any other University/Institute.

This thesis is being submitted to the Mizoram University for the degree of **Doctor of Philosophy** in the Department of Forestry.

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CERTIFICATE

This is to certify that the thesis entitled "**Changes in soil aggregation, microbial population, carbon and nutrients under different land use systems of Mizoram**" submitted to the Mizoram University, Aizawl for the award of the degree of Doctor of Philosophy in Forestry is the original work carried out by Mr. Etsoshan Y Ovung (Regd. No.MZU//Ph.D./997 of 31.05.2017) under my supervision. I further certify that the thesis is the result of his original investigation and neither the thesis as a whole nor any part of it was submitted earlier to any University or Institute for the award of any degree. The candidate has fulfilled all the requirements laid down in the Ph.D. regulations of Mizoram University.

His passion oriented hard work for the completion of the research is to be duly appreciated.

Date:
Place: Mizoram University

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Place: MZU Campus **Etsoshan Y Ovung**

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Table of contents

LIST OF TABLES

LIST OF FIGURES

LIST OF NOTATIONS AND ABBREVATIONS

NOTATIONS		**ABBREVATIONS**	
%	Percent	ANOVA	Analysis of variance
e.g.	Exempli gratia	AP	*Acacia pennata* plantation
et al	Et alia	BD	Bulk density
etc	Et cetera	Ca	Calcium
cm^2	Centimetre square	CEC	Cation exchange capacity
ha	Hectare	CFU’s	Colony forming units
$p<0.01$	Significant level at 1 percent	CJ	Current *Jhum*
$p<0.05$	Significant level at 5 percent	GMD	Geometric mean diameter
ºC	Degree Celsius	HG	Homegarden
Mha	Million hectares	JF	*Jhum* fallow
g	Gram	K	Potassium
mg	Milligram	LSD	Least significant difference
m^2	Metre square	m amsl	Meters above mean sea level
kg	Kilogram	MBC	Microbial biomass carbon
sq. km	Square kilometere	Mg	Magnesium
$mg\ kg^{-1}$	Milligram per kilogram	Mn	Manganese
R	Correlation coefficient	MWD	Mean weight diameter
cm	Centimetre	Na	Sodium
mm	Millimetre	NAF	Natural forest
m	metre	P	Phosphorus
$g\ cm^{-3}$	Gram per centimetre cube	SE	Standard error
$Mg\ ha^{-1}$	Mega grams per hectare	TN	Total Nitrogen
$\mu g\ g^{-1}$	Microgram per gram	TOC	Total organic carbon
$cmol\ kg^{-1}$	Centi mole per kilogram	USDA	United states department of agriculture

CHAPTER 1

INTRODUCTION

1.1. Concept of land use change

Land is classified as a physical terrestrial entity structured through temporal and spatial scales and comprises of topography and natural resources including soil, micro-organisms, minerals and water. The term "land use" and "land cover" are not same and differs in their nature, while land use refers to the physical existence of land and its management and modifications, land cover reflects the biophysical state and takes into account the form of vegetation, water and earth matter including human structures (Patel *et al.*, 2019; Meyer and Turner, 1994; Moser, 1996). Thereby, the transformation and modification of the Earth's terrestrial surface (Land) as a result of anthropogenic activities gradually came to be commonly known as land use change. Subsequently, the use of this land resource, referred to as land use, varies with the purpose to which it is being utilized, for example, cultivation, shelter, mining, recreational activities etc. Land use patterns can therefore be indicated as the interaction between anthropogenic activities and the ecosystems, and their exploitation in temporal and spatial scales. The term land use is concerned with the intention, for which the land is utilized and includes the human actions and impacts concerned with land. Thus, the necessities of human and environmental characteristics influences and shapes land use. Land use change has varied impact on the environment and thus, the identification of issues related to land use change and finding alternatives to tackle it has attracted major attention of ecologists and environmentalists. More specifically, the impact of land use change on environment, commodities and services are of major concern that includes the impact on biodiversity, soil health and atmosphere (Sala *et al.*, 2000; Trimble and Crosson, 2000; Tripathi *et al*., 2008). Therefore, the knowledge on land use change is a crucial attribute for instituting policies and schemes for the benefit of the people while promoting conservation and sustainable use of natural resources (Rawat and Kumar, 2015).

1.2. Global land use change scenario

In view of the rapid change in the global environment, efforts are being put regularly by researchers around the globe to find out key inputs for sustainable use and conservation of natural resources. Initially, addressing the problem and understanding the trend of changes are the fundamental keys for tackling the issue of land use change. In the current scenario, maximum of the changes in land use can be attributed to the alarming rate of growth in human population resulting in increased anthropogenic activities. Land use change is primarily related to the increase in population of an area coupled with anthropogenic interventions (Achmad *et al.,* 2015). According to a recent report from the United Nations, it is estimated that 55 % of the total world population are residing or either dwelling in the urban areas and proposes that by the year 2050 it will rise to 68% (United Nations, 2018). This reflects an addition of about 2.5 billion urban dwellers by the year 2050 and 90% of this increment would take place in Africa and Asia suggesting that the issue would be more pronounced in developing countries.

In general, there is a dearth in the availability of recent data or trends related to global change in forest cover. Nevertheless, the latest available data reflects the loss of gross tree cover at 30 million ha in 2017 from an average of 23 million ha in the past 5 years (Hansen *et al.,* 2018), while, net loss in natural forest was 6.5 million hectares in the year 2015 (FAO, 2015). In addition, since 2001 the 2nd greatest loss in gross tropical tree cover was reported in 2017 from the tropical regions (Weisse and Goldman, 2018).

The driving factors for loss of forest cover is attributable to clearing of forests for cultivation, logging and other activities aimed at gaining products and commodities such as palm oil, coal and gold resulting to about a quarter of the losses in forest cover (Curtis *et al.,* 2018). These activities coupled with shifting cultivation are more restricted in the tropical regions and is anticipated to cause permanent forest loss in this regions. Moreover, the availability of studies and findings on global land use change are limited and generation of such information is vital in the current scenario of undesirable climate change. Several studies have focused on addressing the issue of global land use change and they overlap partially but presents consensus on some of the most important trends, magnitude and direction of changes in land use by taking into account indicator variables such as cropland extend and fallow lands, changes in forest cover, degraded land and urbanization (Ramankutty *et al.,* 2002a; Ramankutty *et al.,* 2002b; Lambin *et al.,* 2003; Leff *et al.,* 2004; Lepers *et al.,* 2005). These studies are aimed at addressing trends in land use change and its impact on the fertility of soil and its appropriate management. Even though land modification for livelihood and other necessary purposes has been prevalent for hundreds and thousands of years, the magnitude and pace of land use change is much greater now than it was in the past decades. Land use change is one of the most destructive and driving factor responsible for changes in the local, regional and global environmental and ecological processes. Therefore, the analytical research and study on land use change plays an impeccable task and contribute critically in the global changing environment through data generation which is fundamental for devising crucial inputs for sustainable management of environment and ecology for future generations (Zhao *et al.,* 2004; Erle and Pontius, 2007).

1.3. Implications of land use change in India

The rapid pace of exploitation has brought forth exponential changes in the ecological processes and environment at the local, regional and global level owing to the alarming rate of growth in human population. Today, India ranks second among the countries with highest population and it is going through a substantial course of urbanization with one in every three Indian residing in an urban area. Recent census shows that for the first point in time after independence, the population growth is higher in urban areas in comparison to rural areas (India Census, 2011). It was estimated that the population of the urban areas in India was approximately 380 million as per the 2011 census and that it is expected to hit the 600 million mark by 2030 and may rise almost over 900 million by 2050 (Ahluwalia *et al.,* 2014; Hoelscher and Aijaz, 2016). Thus, this rapid course of population growth and urbanization has placed a substantial pressure on the natural ecosystem services and ecological processes with negative environmental repercussions especially soil health that may lead to degradation. The biggest repercussion of this issue is the irreparable damage and loss of natural ecosystems while changing the local climatic conditions and thus affecting the health of the environment (Scolozzi and Geneletti, 2012; Ng *et al.,* 2011; Sharma *et al.,* 2013). In a developing country such as India, researchers and policy makers are more obligated to find out new and pioneering ideas to tackle this issue and adapt to the rapid changing environment (Alankar, 2015; Hoelscher and Aijaz, 2016).

In terms of area, India is the 7th largest country in the world having a population of approximately 1.3 billion in 2015 (FAO, 2017a; UN-Pop, 2017). The country is characterized by massive diversity in case of flora and fauna, culture and traditions, climatic conditions, topography, land use systems and socioeconomic privileges (FAO, 2017b). Over the course of about 140 years, the country has witnessed significant changes in land use comprising of forest cover loss as a result of deforestation, expansion of croplands and urbanization (Roy *et al.,* 2015; Tian *et al.,* 2014). India is one of the largest producers of agricultural supplies in the world with over half of its territory occupied by croplands (FAO, 2017a; Teluguntla *et al.,* 2015). Over the past decades, India has seen noteworthy growth in the agricultural production sector (Chand and Parappurathu, 2012; Pingali, 2012). However, there are still considerable gaps in the yield of various crops all around the country (Brahmanand *et al.,* 2013; Sharma, 2016) and this gaps can be attributable to aspects including the dominant and continuous practice of subsistence farming, deprivation or no access to chemical inputs, limited equipments and poor management (George, 2014; Bhattacharyya *et al.,* 2015; ICAR, 2015). The country is expected to reach a population of about 1.6 billion by the year 2050 (UN-Pop, 2017) and along with the varying diet preferences, there is a need to enhance the food productivity and supply (Alexandratos and Bruinsma, 2012).

When a developing country such as India evolves and grows economically, there is an increase in land area being diverted to non-agricultural purposes. The country has prioritized on urbanization as a policy for development and this gave rise to increase in land area occupied by non-agricultural activities greater than average increase over the decade. During the year 2010-2011, about 21 % of the geographical area was under forest cover where 8 % was occupied by non-agricultural activities, 7.5 % was under fallow and 5 % remained barren and unoccupied (GOI, 2015). In India, the typical land holding is 1.1 ha and this has been creating worry for livelihood and food security for millions of farmers practicing subsistence farming. Thus, the sole approach to enhance the farmer’s livelihood and productivity is the efficient and sustainable use of the available marginal land. During the year 1951 to 2009, the area occupied by non-agricultural activities increased from 3.3 % to 8.6 %, while during the same phase, non-arable and barren areas shrank from 13 % to 5.5 %. In case of arable land, there was a diminution from 66.7% in 1951 to 59.7% in 2009 (Rathee, 2014). Presently, the country affords a food supply to about 18 % of the world's total population and yet it occupies 2.4 % of the total land area in the world (Teluguntla *et al.*, 2015; Bhattacharyya *et al.*, 2015). Studies have shown that the country has great possibility to elevate the production of agricultural goods through development and adoption of improved management techniques, sustainable land use systems and use of new crop varieties (Mauser *et al.,* 2015). To achieve this there is a need to invest in further research and development in the agricultural field and likewise the probable detrimental implications on the environment as a result of the expansion and growth of the agricultural sector should not be compromised (Tilman and Clark, 2014; Ramankutty *et al.,* 2018; Rockström *et al.,* 2017; Springmann *et al.,* 2018; Srivastava *et al.,* 2016; Tilman *et al.,* 2017).

India falls under one of the richest countries in the world in terms of biodiversity and the remaining attached forest cover (22 % of the total area) reflects a potential stock for carbon and it is imperative to conserve it as a means for climate change mitigation (Nadagoudar, 2016; Swaminathan and Bhavani, 2013). Land use change has increased unproductive or less productive areas in the country and therefore it is of the greatest priority to tackle this issue. The dynamics of land use change in India has witnessed hasty changes in the past decades where the correlation between the country's massive population and land use dynamics share an intricate association which is driven by an elevated level of heterogeneity in the ecology and environment, extremity in socioeconomic variations, language and culture. Changes in land use are usually a result of fulfilling the demands of the ever increasing population and thus adding to the chances of contracting irreversible and long term damage to soil health and the environment. It can be rightly put forth that land use change is the most prominent indicator of transformation of the terrestrial Earth's surface due to anthropogenic activities in any nation.

1.4. Land use change in North-east India and its impact on soil health

North eastern region of India is one of the most diverse regions in the country in terms of flora and fauna and falls among one of the world’s biodiversity hotspots. The North eastern region of India extends to about 2,61000 km^2 with diverse physiographic features and biodiversity, well supported by favourable climatic conditions specifically elevated rainfall supporting higher vegetation both below and above ground is a distinct place in India (Chatterjee *et al.,* 2006). These distinct characteristics of the region provide favourable conditions for lush vegetation growth. Over the last three decades, the forest cover has changed drastically as a result of people relocating to different places, increased clearing of forest, varied farming practices, expansion of agriculture and industries induced by population growth and elevated economic activities.

Land use change in the North eastern region of India differs and portrays a unique trend in comparison to the rest of the regions in the country. Majority of the area in the region is characterized by steep slopes and undulating terrain except in the state of Assam where the plain area extends to about 84.4% of its geographical area, whereas the rest of the states amount to about 35% in terms of plain area and the remaining undulating terrains. Approximately 15.7% of its geographical area is occupied by cultivation, open and dense forest cover in the region ranges from 40.27% in Assam to 72.99% in Arunachal Pradesh (NRCS, 2011). While area under shifting cultivation (*Jhum*) covers about 2.88% (0.754 Mha), grasslands cover 6.06% of the geographical area, wastelands occupy about 6.22% and water bodies occupy 4.53% of the total geographical area (Choudhury *et al.,* 2014).

Recently, there has been a rapid change in land use in the North eastern region as a result of extensive deforestation. There has been massive modification of forest areas into short term agricultural practice largely attributable to shifting cultivation (*Jhum*) coupled with conversion of native forests and grasslands into horticulture and plantation areas (Choudhury *et al.,* 2013) which led to enormous loss of vegetation in the region and increased loss in soil fertility. The incineration of the accumulated phytomass at 8.5 million tonnes each year during shifting cultivation (Das *et al.,* 2011) and extensive cultivation on steep slopes under heavy and erratic rainfall conditions has accelerated soil erosion greater than 88 Mg ha^{-1} year^{-1} with an estimated annual loss of 6.0 million tonnes of soil organic carbon and other available nutrients (Ghosh *et al.,* 2009).

Approximately about 1.3 Mha of land is exposed to severe soil erosion out of the 4.0 Mha net sown areas of the region. Deforestation due to shifting cultivation is still the key factor for the loss of forest cover in the region and consequent imbalance in the ecological process degrades the buffering capacity of the environment and the soil in particular. Since most of the agricultural practices in the region are rainfed and therefore, there is sporadic drought and water shortage during the height of crop growing season which is another issue affecting optimum crop productivity and sustainable farming. Ecosystems in the North eastern region are highly vulnerable and more than 70% of the population is still dependent on subsistence, marginal, low input agriculture, low yield and rainfed agricultural practices. In addition, majority of the farmers have marginal land holdings with about 40% living in poverty (Barah, 2007). All these challenges coupled with meagre socio-economic status, increasing population and traditional form of agricultural and cultivation has induced a trend of land use change in the region which is distinct from the rest of the country. In the past years, the rural mass has been exposed to multiple challenges which placed sustainable agricultural yield and livelihood at a susceptible stage. Land use change has a colossal impact on the soil fertility and subsequently the sustainable agricultural production practices. As the problem persists, accepting the adverse impact of land use change patterns and its interactions between anthropogenic practices and environment on soil fertility and crop productivity are the most essential challenges that is needed to be considered at present for achieving the goals of sustainability through land management.

1.5. Implications of and use change in Mizoram

Majority of the rural population in Mizoram is still dependent on agriculture and allied sector, animal husbandry and other small scale or cottage industries for their subsistence and economic returns. Agriculture and allied sector is the most dominant activity on which the rural masses depend on and they shape the rural economy (Grogan *et al.,* 2012). However, these activities are controlled by the existing physiographic characteristics and therefore there have been complications in relation to land use activities in the past decades. Shifting cultivation (*Jhum*) is the most dominant form of cultivation in the state and given the extreme topographic conditions, this form of cultivation could be the only probable option of cultivation. Each year almost 1.5% of the total area is affected by shifting cultivation or *Jhum* in Mizoram (Maithani, 2005b).

The traditional form of *Jhum* or shifting cultivation has been the sole and major source of economy for the farming communities of Mizoram since time immemorial. This form of agriculture has been practiced sustainably in the previous years, however, due to exponential increase in population in recent years the practice has become unsustainable as a result of substantial decrease in the length of fallow periods (Grogan *et al.,* 2012). This has led to increased changes in land use patterns leading to loss of soil fertility as a result of erosion and loss of natural forest which are strongly linked to food security for the poor farming community of the region. Land use pattern of the state is shown in Table 1.1.

Table 1.1 Land use pattern of Mizoram

Land use	Area (in 000' ha)	Percentage
Geographical Area	21081	
Reporting area for land utilization	2039	100
Forests	1585	77.75
Not available for land cultivation	75	3.69
Permanent pastures and other grazing lands	11	0.54
Land under misc. tree crops and groves	41	2.03
Culturable wasteland	8	0.37
Fallow land other than current fallows	127	6.24
Current fallows	47	2.28
Net area sown	145	7.10

Source: Land Use Statistics, Ministry of Agriculture, GOI, (2014-15)

The drawback with shifting cultivation is that once the process of cultivation is carried out and the yield has been harvested, the land area is left barren. Under such steep topography with high incidence of erratic rainfall each year, deterioration of soil health is accelerated which lengthens the process of soil reclamation and period of recovery, but this basic problem is seldom considered. Moreover, the productivity of the land during the next cultivation is decreased on this particular patch of land (Tripathi *et al.,* 2017) leading to low output per unit area. This kind of land use system is not only destructive but unproductive and demands great labour involvement and as a result, the state has suffered massive loss in soil fertility, natural forest, wildlife, exhaustion of underground water leading to drying of rivers and springs. Therefore, to battle this issue and with an aim to uplift the farmers of the state that comprises about 70 % of the total population, in 1984, the Government of Mizoram came up with a policy known as the New Land Use Policy (NLUP) which was implemented in fractions and small scale during the year 1985 to 1992. This policy had a few setbacks with the implementation being called off and on several occasions as this was a new concept which has never been attempted in the state in its history. However, driven by the hardships faced by the people of the state, the policy developed into a more meaningful attempt during 1993 to 1998 with a large scale implementation than the preceding tenure and involved about 35000 families over a 2 year interval (Zothansanga and Bobby, 2019). Later in the year 2011, the policy came into force again and showed better and promising plans and framework as compared to the previous periods considering the advices from Government of India. Post implementation of NLUP, the state of Mizoram has seen notable changes in the status of shifting cultivation. During the year 2010 to 2011, shifting cultivation areas extend to about 28,562 ha and involved 68, 433 families. However, the area and number of *Jhum* practicing families decreased to 19, 851 ha and 48,417 during 2015 to 2016 which accounted to an increase of about 30.50% and 29.25% respectively (Zothansanga and Bobby, 2019).

Apart from the mainstream agricultural practice of *Jhum*, other land use practices are initiated including horticultural plantations such as Rubber, Areca nut, Citrus fruits, Anthurium flower and oil palm plantations. However, these necessitate significant support and investment form the government making it practically achievable only for a small group of the population or farmers. Pig rearing and poultry are the basic practices of animal husbandry in the state. Cattle rearing and subsequent cultivation of agricultural crops seems to be a viable form of land use system but it is limited on gentle slopes and terraces (Grogan *et al.*, 2012). Wetland rice cultivation is also practiced annually on moderate slopes and valleys yet such feasible areas are exceptionally less. Thus, there is a huge necessity for the government and policy makers to look into the options which would bring out efficient use of land while promoting ecological processes and conservation of environment.

1.6. Impact of land use change on soil properties

The hastened rate of growth in human population have been the major driving force responsible for the type of land use and its expansion which in return affect the soil health through conversion and modification of natural ecosystems to man-made land uses. This increase in conversion and modification of natural ecosystems and its expansion could lead to an irreversible damage to soil properties in the long run. Changes in land use have been well attributed to alterations in soil properties since time and several studies have been carried out till date. Soil properties such as soil organic matter (SOM), soil biota, water absorption capacity, resistance to erosion, sufficient oxygen and nutrients are considered as major soil health indicators (Jing *et al.,* 2011).

The conversion and modification of land use in the tropical regions have imparted a considerable alteration in the soil properties by affecting the organic matter accumulation in the soil. Land uses differ in their soil organic carbon distribution as a result of the variations in their vegetation cover, root distribution pattern, quality and quantity of litter fall. Don and Schumacher (2011) observed that conversion of croplands to fallow or grasslands in the tropical regions led to increase in the soil organic carbon (SOC) stocks within the 20 and 40 cm soil depth by 32.2 % and 25 % respectively. In the recent decades, the global SOC stocks at 15 cm soil depth increased by 0.19% C y^{-1} in areas which did not experience any land use change (Chen *et al.,* 2015) which clearly indicates that soil properties are significantly affected by land uses (Rolando *et al.,* 2018). Soil organic carbon is one of the most important soil quality indicators and represents one of the chief constituents of the global carbon cycle and its concentration varies among the land uses (Ali *et al.*, 2017). Alterations in the vegetative cover induces changes in C cycling, soil organic matter (SOM) content and emission of CO_2 depending on the amount of litter input and decomposition kinetics (Murty *et al.,* 2002). Higher C/N ratio in plant litter inhibits the rate of decomposition and as a result there is greater accumulation of litter on the surface of the soil (Tejada *et al.,* 2009). On the contrary, plant litter having lower C/N ratio (<25) have faster decomposition rates, thereby, augmenting the release and availability of nutrients in the soil. In acidic soils which occur in most tropical regions, there is less microbial activity as a result of decreased population, which in turn, greatly reduces mineralization rate of SOM as soil microorganisms play a significant role in the decomposition process of SOM. The availability of nutrients in the soil is dependent on the characteristics of soil microorganism such as microbial biomass, enzymatic activities and metabolic quotient and their effects on the soil ecological processes (Jiang *et al.*, 2009). These characteristics are primarily responsible for alterations in the productivity of various land uses and therefore, it can be regarded as one of the most important soil quality indicators (Dick *et al.*, 1997). The various soil ecological processes are carried out and maintained by soil microbes as a result of their enzymatic actions largely by decomposing soil organic matter, serving as catalysts in various biochemical reactions concerned with energy shifts and nutrient cycling (Sinsabaugh *et al.*, 1994). Thus, microbial population and its activities are regarded as significant indicators in identifying changes in soil properties as a result of undesirable soil management practices and signify disturbances to ecosystem through early response.

Land uses do not necessarily have a negative impact on the soil properties. Appropriate soil management practices such as conservation or minimal tillage, organic manure addition, mulching, mixed cropping or crop rotation are known to augment the soil properties and thus achieving sustainability (Ovung *et al.*, 2021). While the practice of poor management practices such as excessive use of acidic fertilizers, continuous cropping without fallowing and poor or no soil conservation measures lead to soil health deterioration (Willy *et al.*, 2019). Under such circumstances, it is of great importance to address the issue and understand which particular soil property is affected by the type of land use management practice or the prevalent land use system as a whole. Thus, there is a need to direct more research and development (R&D) pertaining to the impact of land use systems, management practices and environmental factors on soil properties.

1.7. Land use change and soil aggregate stability

Soil aggregates are significantly affected by land uses and their management practices such as tillage, use of fertilizers and organic amendment addition. Soil aggregate formation and distribution are dependent on various factors such as soil organic matter content, decomposition, presence of soil biota, amount of exudates, root density, structure and biomass. The fundamental unit of soil structural stability is soil aggregate and it is profoundly affected by various management practices (Chen *et al.*, 2017; Chivenge *et al.*, 2011). Land uses such as agricultural lands and other cultivated areas are known to impact the soil physical, chemical and biological properties as a result of management practices such as tillage, use of chemical fertilizers and agro-chemicals. Tillage practices negatively impacted the soil physical properties such as soil bulk density, soil aggregation and texture which in turns affects the proper functioning of soil processes such as infiltration, soil moisture content and water holding capacity (Gomez *et al.*, 1992). Soil physical properties such as soil aggregates play a significant role in maintaining soil structural stability and improve other soil physical properties such as bulk density and infiltration (Singh *et al.,* 2017; Zhong *et al.,* 2019). While, conversion of degraded land to plantations is known to reverse soil deterioration process through the addition of SOM which enhances the formation of soil aggregates and thereby improving the structural stability of soil (Singh *et al.,* 2002; Ge *et al.,*2018). Soil organic matter (SOM) is one of the most important factors which enhance the formation of soil aggregates and its continuous loss degrades soil aggregate stability (Liu *et al.,* 2014). Several studies have reported that changes in land use have a direct impact on the soil aggregate stability (Tripathi *el al*., 2008; Tripathi *et al.*, 2012; Singh *et al.*, 2015). Soil aggregate stability may be considered as one the most important physical property which can be used as an indicator to analyse the changes in the soil quality since soil aggregates influences various soil processes such as nutrient availability, soil erosion, infiltration, soil aeration and water holding capacity (Six *et al.*, 2004). Thus, in order to maintain soil fertility, enhancing soil productivity, inhibiting soil erosion and maintaining soil aggregate stability is of vital importance (Erktan *et al.*, 2016).

The concentration of soil organic carbon (SOC) in the soil is directly correlated with soil aggregate stability since SOC acts as a catalyst in the formation and stabilization of soil aggregates due to the presence of humic substances which act as a binding agents (Tripathi *et al*., 2008; Mishra *et al.*,2014). Thus, higher SOC content in the soil favours soil aggregate formation and improves structural stability of soil and vice-versa. Soil aggregate stability signifies the behavior of soil aggregate fractions with the changing land uses which allows the proper understanding of the pattern of land use impact on soil structural stability and helps in assessing the magnitude of soil degradation (Gupta *et al.,* 2009).

Changes in soil aggregate stability and SOC varies with land uses, which in turn, affects the C sequestration capacity. Land use that experiences minimum disturbances holds more SOC as a result of better soil aggregation which conserves and protects the SOC present in the soil. On the contrary, land uses that experiences high disturbances in the soil due to conversion of forest to cropland or other management practices results in rapid loss of SOC due to increased rate of decomposition induced by the breakdown of aggregates (Murty *et al.,* 2002). Healthier soil aggregate stability and increased SOC could be reported from natural ecosystems such as forests and grasslands as a result of the availability of carbon inputs from belowground as well as above ground biomass. Even though there are numerous studies which have assessed the impact of land use change on soil aggregate stability (Tripathi *et al*., 2008; Tripathi *et al.,* 2012; Chen *et al.,* 2017; Liu *et al.,* 2014), there is very less information available on the impact of land use change on soil aggregate stability in the hilly ecosystems of North-east India where shifting cultivation is more pronounced. Thus, it is of great importance to assess the impact of land use change such as conversion of forests to shifting cultivation and other land use practices on soil aggregate stability given the hilly nature of the region.

1.8. Rationale of the study

Ecosystems in Mizoram are characterized by steep slopes, undulating terrain and high occurrence of rainfall each year amounting to more than 2500 mm which varies annually (Saha *et al.,* 2015). About 70% of the total geographical area of the state (~21,081 km^2) is sloped at angles higher than 33° (Anonymous, 2009). Soils in the state are typically young and highly acidic with higher loose sediment concentration making it susceptible to erosion and landslides. They can be grouped under inceptisols, entisols and ultisols with generally low potassium and phosphorus concentrations (Misra and Saithantuaanga, 2000). The most dominant agricultural practice in the state is shifting cultivation also known as *Jhum* or slash and burn cultivation on which majority of the rural population is dependent on it for their subsistence and economy. Substantial increase in the practice of shifting cultivation has affected the soil fertility as a result of run-off and erosion leading to uncontrolled loss of soil nutrients in this region (Wapongnungsang *et al.,* 2018).

According to Aayog NITI (2018), shifting cultivation extends to about ~2617.56 sq. km which sums up to about 12.4 % of the total geographical area of the state. Shifting cultivation over the past decades negatively affected the environment as a result of the increasing practice which gradually led to reduction of fallow period to as low as 3-5 years. This shortened fallow period induced greater surface run-off and soil erosion and thus leading to poor soil fertility, low crop productivity affected food security (Grogan *et al.,* 2012; Wapongnungsang *et al.,* 2018). Thus, chief causes of land and soil degradation in the state are a combined impact of cultivation on steep and fragile soils with minimum soil conservation measures or vegetation cover, decreased fallow period, poor recycling of plant and animal residues, erratic and erosive rainfall patterns and forest deterioration. Shifting cultivation adds about 44 - 50% of the total soil loss and deforestation in Mizoram (Singh, 1996).

Therefore, analysis of aspects that contributes to soil health enhancement in specific landscape are decisive and can only be achieved through the adoption of sustainable practices and ideal land use system which will provide both economic prosperity and ecological sustainability. As soil productivity and sustainability depends on dynamic equilibrium among its physical, chemical and biological properties, an assessment of soil properties in relation to the existing land use acts as potential tool to provide information on the nutrient storage in semi-natural and cultivated ecosystems (Lal, 1998). These properties are continuously influenced by land use systems and practices with profound influence on soil properties and thus, help in monitoring and restoration of soil health (Deekor *et al.,* 2012). Without the proper maintenance of soil fertility, it is impossible to achieve increment of agricultural production in feeding the increasing human population. Soil fertility maintenance is the key to optimize, sustained and sufficient crop production as well as in overcoming undesirable climate change in the long-run. Henceforth, it is paramount to assess the influence of land use systems on soil health for evolving sustainable land use practices and policy frameworks. The present study was therefore, initiated with a motive to quantify and assess the status of soil physico-chemical and biological characteristics and the impact of different prominent land use systems on soil fertility in Mizoram.

1.9. Objectives

Keeping in view the priorities discussed above, the present study was instituted to achieve the following major objectives:

1. To assess soil physico-chemical properties under different land use systems.
2. To determine water stable soil aggregates and their associated chemical characteristics under different land use systems.
3. To estimate soil microbial population under different land use systems, and to relate soil microbial population with aggregates sizes (micro- and macro-aggregates).

CHAPTER 2

REVIEW OF LITERATURE

2.1. Studies on land use change effect on various ecosystems

The number of studies conducted over the world emphasized urbanization as one of the important and dominating factors of global phenomena affecting the natural environment as well as human surroundings and their livelihood options through various social, economical and ecological processes (Mandelas *et al.*, 2007; Tripathi *et al*., 2008; Tripathi *et al.*, 2012; Liu *et al.*, 2014; Chen *et al.*, 2017). Urban growth is a key driving factor which influences environments at all levels including local, regional and global. During the past three hundred years, this impact gave rise to undesirable land use change which has shifted from negligible to hazardous magnitude where uncontrolled population growth of humans signifies the fundamental driving factor of land use change (Vitousek *et al.,* 1997). The alarming increase in population and economic growth coupled with land use change arises at the price of alterations in the natural ecosystems (Ifatimehin and Ufuah, 2006; Ifatimehin and Musa, 2008). Tan *et al.* (2005) reflected that the development and extension of 145 metropolitan cities in China during the period 1990 to 2000 mostly transpired on arable lands.

The interaction of anthropogenic activities with the environment has been accepted as the key driving factor in altering the biosphere specifically the landscapes. According to Fasal (2000), anthropogenic activities are the causes of most part of the changes in shape and processes of the biosphere rather than natural causes. In addition, Fasal (2000) also reflected that land conversion and modification have been the major driving factor for anthropogenic transformation of the natural ecosystems. In majority of the countries, the ever increasing human population deteriorates the natural forest cover as a result of the changing land use patterns (Nicolson, 1987) while inducing social and economic changes in its surroundings. Nanda (2005) expressed that the proliferation of urbanization has influenced exponential changes in terms of demographic characteristics and land use patterns which has also led to devious and severe ecological and environmental issues (Zhau *et al.,* 2006). According to Kalnay and Cai (2003), urbanization can have a direct influence on the transformation of local climatic conditions at urban and adjoining areas.

In the present scenario, rapid changes in the environment, climate and loss of forest cover and vegetation has paramount repercussions on affecting various ecological processes and ecosystem services (Tripathi *et al.*, 2017). Adesina (2005) focussed on these implications and concluded important implications of land use change on ecosystems as: a) fragmentation and loss of wildlife habitat; b) destruction of genetic pool; c) exhaustion of medicinal plants and food; d) occurrence of drought and desertification; e) increased emission of green house gases; f) deterioration of soil fertility. Weng (2001) asserted that diverse studies from multi-disciplinary fields showed that changes in land use has emerged as key factors for various applications including forestry, geology, hydrology, ecology and environment, agriculture etc and these applications relates to loss of crop lands, soil health deterioration, urbanization and industrialization, changes in water quality etc.

In general, land use change plays a very crucial role between land and the atmosphere and their interactions affect local, regional and global climatic conditions. There are varied evidences of how land use and land cover change affects the climatic conditions at all levels. For instance, the conversion of natural vegetation to agriculture has resulted in drastic changes in the air temperature and moisture at near-surface (Fall *et al.,* 2010; Karl *et al.,* 2012). These implications could be experienced in the Midwest and Great Plains in the US where the regularity of incidence of severe dew point temperatures increased as a result of conversion to agricultural lands (Mahmood *et al.,* 2008). Riebsame *et al.* (1994) stressed that land use and land cover change can be related to the changing land use patterns which is fuelled by various anthropogenic activities and these affects the environment and ecological processes, water and air conditions which ultimately forms a combined effect on the climatic conditions and the biosphere. Globally, the changes in forest areas, farmlands, water, air and soil health are instigated by the demands and needs to supply food, fiber and shelter to more than 6 billion people with expansion in the urban areas which comes along with massive intensification in consumption of water, energy sources, fertilizers and substantial loss of biodiversity (Foley *et al.*, 2005). There are several means in which urbanization impacts the environment and ecology of an area and its surroundings which includes modifications in the climatic conditions, fragmentation and loss of indigenous habitats and increase in the concentration of man-made pollutants in air, water and soil where all these implications on the ecosystem and ecological processes are directly attributable to the changing pattern of land use altering the landscapes (Mandelas *et al.*, 2007; Guru and Anubhooti, 2015). Johnson *et al.* (2000) and Bonan *et al.* (2008) also reflected on the impact of increasing temperature which affected the proper functioning of processes such as fluxes in hydrological cycles, rate of decomposition and faster physiological growth which results in reduced productivity and accelerated maturity. According to UNEP (2004), the depreciating pace of forest area is placed at second highest globally which ranges from about 4 to 6% per year and this decrease in forest area could be correlated with the undesirable climate change which have escalated over the past years as a result of anthropogenic interventions.

2.2. Ecological studies on land use change

The increasing human population coupled with the extensive use of land for various activities have led to modification of majority of the terrestrial ecosystems into contemporary man made-biomes or anthromes which has caused an emergence of various ecological processes and trends (Ellis, 2011). Challenges relating to land use change have got the attention of researchers from multiple fields varying from enthusiasts attempting to address the issues or understanding the causes, implications and consequences and researchers trying to find possible alternatives to tackle the severe issues (Brown *et al.*, 2000; Theobald, 2001).

Long *et al.* (2007) asserted that anthropogenic interventions driven by social and economic aspects instigate conversions and modifications in both urban and rural areas in spite of the limitations from physiographic conditions. Meyer (1995) also reflected that there are other instances in which anthropogenic activities have affected land cover such as the destruction and deterioration of the tropospheric ozone, agricultural crops, forest cover and water bodies by acid rain occurring as a result of excessive burning of fossil fuels from various sources.

A study by Tekle and Hedlund (2000) reported an enhancement in the extend of shelter and settlements and open areas at the cost of forests and shrub lands during the year 1958 to 1986 in the district of Kalu, Southern Wello in Ethiopia. In addition, a study by Woien (1995) attributed the increasing human population density to expansion of homestead in the central highlands during the year 1957 to 1986. Another instance of impact of land reforms and settlement on land use change can be seen in the study by Mark and Kudakwashe (2010) in the Midlands Province of Zimbabwe wherein the increase in settlements led to increase in the expanse of croplands, deforestation for various cultivation purposes and other activities including clearing of forests for fuelwood and poles for building shelters could be reported.

Begum *et al.* (2010) reported from that the urban settlements and other built-ups grew twofold near the water bodies around Davangere city, Karnataka in India at the expense of scrub and cultivation area during the year 1970 to 2005. Another study by Prakasam (2010) on land use/land cover for about 40 years in Kodaikanal taluk, Tamil Nadu, India revealed increase in settlements, built-up areas and crop lands whereas extend of forest cover and water bodies decreased. Javed and Khan (2012) also reported significant decrease in forest area and water body but increase in barren areas, settlements and wastelands as a result of mining operations.

2.3. Studies on land use change and soil properties

Globally, land use change and management practices are the most dominating factors which influence major properties and ecological processes of soils (Lal, 2001; Pacheco *et al.,* 2018; Valle Junior *et al.,* 2014). Unsustainable land use and management practices which lead to soil health deterioration can be regarded as a severe challenge for food security and environmental sustainability worldwide (Oldeman, 1998; Lal, 2009). However, this issue is more pronounced in the developing regions experiencing pressure due to increase in human population (Lal, 2006; Pricope *et al.,* 2013; Tully *et al.,* 2015) and where conversion of natural forests and grasslands to other land uses such as croplands and uncontrolled grazing have degraded the soil properties (Sun *et al.,* 2003; Giertz *et al.,* 2005; Jangid *et al.,* 2011; Gregory *et al.,* 2015; Valera *et al.,* 2016). Impact of land use change on soil properties is more pronounced in the hilly regions as it can greatly degrade soil quality with increased soil loss through runoff and erosion. A number of studies (Ney *et al.,* 2019; Safar *et al.,*2019; Celik, 2005; Khormali *et al.,* 2009; Azizsoltani *et al.,* 2019) have shown that clearing of forest and changes in land use can greatly degrade important soil properties such as soil carbon sequestration potential, soil porosity, soil microbial biomass, infiltration and nutrient content. In addition, it could also lead to increased soil bulk density, higher runoff and eventually leading to heavy soil losses (Saravanan *et al.*, 2019) especially in hilly landscapes.

Several studies have now regarded land uses and management practices such as agricultural lands, intensive cultivation without any conservation measures, deforestation and removal of plant and crop residues leads to maximum loss of soil organic carbon (Gelaw *et al.,* 2014; Abegaz *et al.,* 2016; Assefa *et al.,* 2017; Negasa *et al.,* 2017) and exposure of soil and land to hazards such as erosion by water (Haregeweyn *et al.,* 2017; Fenta *et al.,* 2019). According to Elliot and Efetha (1999), continuous tillage and grazing were observed to lower the soil organic matter content, soil aggregate stability and infiltration, thus making the soil vulnerable to loss of nutrients through surface runoff (Pardini *et al.,* 2003).

Studies have been carried out in relation to the impact of land use changes on soil properties in the North-east region of India (Ovung *et al.*, 2021, Wapongnungsang *et al.*, 2017; Lalnunzira and Tripathi, 2018; Hauchhum and Tripathi, 2017; Singha and Tripathi, 2017). Shepherd *et al.* (2000) asserted that land use change in tropical regions cause substantial alterations in the soil physical, chemical and biological properties. Increased levels of soil organic carbon (SOC), total nitrogen (TN) and cation exchange capacity (CEC) of soils were reported from undisturbed forests in comparison to disturbed (Offiong *et al.,* 2009). Shifting cultivation is a major issue in most tropical regions especially in North-east India where this practice is the most prominent form of agriculture which causes maximum loss of vegetation cover while inducing land use change. Decreasing fallow period has led to soil degradation, while increase in the fallow period following shifting cultivation have been observed to enhance the SOM build up in the soil and thus increased nutrient availability (Wapongnungsang *et al.*, 2017).

One of the key indicators of soil fertility is soil organic matter (SOM) which is suitably used for soil health and fertility assessment in tropical areas (Paniagua *et al.*, 1999). According to GoI (2009), conversion of natural vegetation to croplands led to significant decrease in SOC and TN. A report by IPCC (2000) stated that almost 14 % of the terrestrial carbon is sequestered in the soils of tropical regions; however, to meet the demands and needs of food, timber and other needs due to rapid growth in population carbon pools are diminishing as a result of the conversion of natural vegetation to other land use practices.

2.4. Studies on land use change, SOC and soil aggregate stability

Soil organic carbon (SOC) plays a very important role in the global carbon cycle of terrestrial ecosystems while regulating soil fertility in various ecosystems (Balesdent *et al.,* 2018; Brahma *et al.,* 2018; Fontaine *et al.,* 2007). Having a great storage in nature, minute alterations in SOC can have a significant impact on the CO_2 concentrations in the atmosphere and may result in climate change at the global level (Ajami *et al.,* 2016; Li *et al.,* 2017). Majority of the soil biogeochemical cycles of nutrients and soil processes such as transfer of nutrients and substances for microbial functions are collectively linked to SOC concentration and therefore, SOC can be a considered as a valid indicator for evaluating changes in soil quality (Li *et al.,* 2016; Stevenson *et al.,* 2016). In addition, higher SOC concentrations in the soil can substantially enhance soil health (Hati *et al.,* 2007; Lal, 2004).

Land use and management practices with minimal soil disturbances adds to the increase in soil organic carbon concentration, on the contrary, higher disturbances on soil leads to decrease in soil organic carbon and subsequent soil degradation (Luo, 2010). Conversion of land uses from natural forest and grasslands to croplands cause loss of C up to 50% (Bhatia *et al.,* 2011; Bruun *et al.,* 2015; Guo and Gifford, 2002). While, increase in vegetation cover on barren and abandoned cultivated lands augments C storage (Post and Kwon, 2002). On the contrary, Roose and Barthes (2001) reflected that cultivation over time leads to reduction of C concentration as a result of decreasing C input each year and enhanced mineralization rates due to disturbances of soil surface. Many studies have shown that ability of increasing SOC content in soil to create favorable conditions for improved soil productivity and help reduce emission of greenhouse gases (Hati *et al.,* 2007; Lal, 2004). A study by Ajami *et al.* (2016) showed that cultivated soils have much decreased C storages than uncultivated or natural soils and increase in SOC concentration not only augments the soil nutrient availability and soil structural stability but it also helps greatly in mitigating CO_2 emissions through C sequestration in the soil (Chang *et al.,* 2016; Han *et al.,* 2017; Kabiri *et al.,* 2016; Zhang *et al.,* 2018).

Soil organic carbon (SOC) is one of the most important indicators of soil fertility on which various processes in the soil are dependent. However, due to rapid change in land use such as conversion of natural forests to secondary vegetation and agricultural land uses, there has been drastic loss in SOC content (Yan *et al.,* 2012). Conversion of forest to agricultural lands has led to a global decline in SOC to about 24% (Murty *et al.,* 2002). Several studies have also shown that conversion of forest to not only cultivated land but also to various plantations leads to severe decrease in SOC content (Guillaume *et al.,* 2015; van Straaten *et al.,* 2015; de Blécourt *et al.,* 2013), while, a loss of SOC up to 25 to 30% have been reported from conversion of natural forest to cultivated lands (Don *et al.,* 2011). Loss of forest cover in the tropical regions at the cost of various economically valuable tree plantations has led to significant changes in soil carbon concentration. A study by van Straaten *et al.* (2015) showed that loss of carbon up to one-half in the soil is attributed to the conversion of natural forest to tree plantations such as rubber, oil palm and cacao.

Land use change of forest to cultivated lands usually lead to decrease in the soil aggregate stability and the associated carbon concentration (Golchin and Asgari, 2008). Islam and Weil (2000) indicated that native forest and reforested soils generally have higher stability of aggregates in comparison to soils under cultivation which signifies that higher SOM content in the forest soil favors soil aggregation.

Stable soil structure represents higher productivity of the soil and is able to regulate health of environment through the protection and conservation of SOC with its stable aggregation (Amezketa, 1999). Soil aggregates play a significant role in the structural stability of soil and thereby improving soil physical qualities such as aeration, water holding ability, soil porosity and infiltration (Singh *et al.,* 2017; Zhong *et al.,* 2019).

According to Tisdall and Oades (1982), soil aggregates, depending on the size are usually classified as micro-aggregates (<0.250 mm); meso-aggregates (0.250-1.00 mm) and macro-aggregtaes (>1.00). Good soil aggregation is vital for maintenance of soil quality since disruption of macro-aggregates to meso- and micro-aggregates leads to poor soil structure and thus they are more vulnerable to be affected by erosion and surface runoff (Amézketa, 1999). Stockmann *et al.* (2013) reflected that soil associated C distribution among various aggregate size fractions determines the SOM dynamics in different soil ecosystems.

Soil aggregate stability is generally influenced by land use type and more importantly the management practices carried out. Govaerts *et al.* (2007) reflected that soils with no tillage and retention of residues enhance soil aggregate stability in comparison to tilled soils. Study on zero tillage land management with straw retention treatment on the soil also showed positive response to the soil's structural stability (Zhang *et al.*, 2009) which reflects that retention of organic matter on the soil conserves soil aggregates from disruption. Studies by Limon-Ortega *et al.* (2002) and Chan (2001) also reported that retaining plant/crop residues on the surface of the soil results in enhancement of SOC and augments soil aggregate stability with the growth and return of soil biota especially earthworms.

The availability of SOM in the soil greatly affects the formation of soil aggregates and thus improves soil structure (Onweremadu *et al.*, 2007; Durigan *et al.*, 2017). Several significant studies have recommended that SOM can aid in the development of soil aggregates thereby increasing the mechanical stability of soil aggregates by maintaining the coherence of inter-particle bonds through the binding of mineral particles (Yu *et al.*, 2015; Peng *et al.*, 2015; An *et al.*, 2010, Somasundaram *et al.*, 2016). SOM helps in the formation of soil aggregates particularly macro-aggregates which is the key determinant of the structural stability of soil (Elliott, 1986) and the macro-aggregate proportions decreases with increasing soil depth (Chen *et al.*, 2017). Six *et al.* (2004) reported that land uses such as agroforestry systems promotes soil aggregate stability as a result of the presence of plant residues on the surface soil.

2.5. Studies on soil aggregate stability and microorganisms

It is well known that land use change and management activities affect the C turnover in the soil, however, the primary causal processes remain largely unidentified. As such, there is a gap in the information related to the correlation between management activities and soil ecological processes and the understanding of the role of microorganism and their associated functions (Wardle *et al.,* 2004; Xiao *et al.,* 2007; De Deyn *et al.,* 2008; Jin*et al.,* 2010). In various ecosystems, soil microbes play a vital in almost all ecological processes such as nutrient cycling and decomposition (Singh *et al.,* 2010; Trivedi *et al.,* 2013). The production of enzymes, chemical excretes and other aggregate binding agents and decomposition of SOM associated with aggregates by the microbial community in the soil has been related to the increase in stability of aggregates (Six *et al.,* 2006; Wilson *et al.,* 2009; Tiemann and Grandy, 2015).

It has been hypothesized that the distribution of SOM resources within different aggregate size fractions (Macro-, meso- and micro-aggregates) acts as niches for the allocation of diverse functional microorganism groups which in turn regulates ecosystem processes such as decomposition and mineralization (Davinic *et al.,* 2012; Tiemann and Grandy, 2015; Trivedi *et al.,* 2015). Changes in land use management practices impact the soil structure by disrupting soil aggregates (Six *et al.,* 2006; Tiemann *et al.,* 2015) which in turn alters the soil physico-chemcial properties and subsequently affects the functional microbial group distribution and their functions within various aggregate sizes.

CHAPTER 3

MATERIALS AND METHODS

3.1. Study area

Mizoram is one of the eight sister states of North-east India (Figure 3.1) located in the southernmost part of the region, covering an area of 21,081 km^2. Mizoram is a hilly state blessed with a huge coverage of green foliage and diverse range of biodiversity. The tropic of cancer runs through the state of Mizoram in North Lungsai. The State lies between 21°56'N to 24°31'N lat and 92°16'E to 93°26'E long and shares inter-state borders with Assam and Manipur in the north and Tripura in the west. The state also shares an international boundary with Myanmar and Bangladesh.

The physiography of the state is characterized by undulating, rugged, steep hills and interspersed range of valleys. The climate pattern is moist tropical to moist sub-tropical. The region is influenced by monsoons, raining heavily from May to September with little rain in the winter season, which makes this period cold and dry. The state receives an annual rainfall of about 2100 mm-3000 mm and the annual temperature ranges from 11°C to 24°C and 18°C to 29°C in winter and summer months, respectively. Majority of the area is categorized under land use capability class-II to class-IV, which requires proper land management (Mizoram SAPCC, 2012-17). The most common soil order found in the state falls under inceptisols which is generally composed of sandy loam and clay loamy soil having high organic carbon richness (Colney and Nautiyal, 2013). The soils of the area are significantly altered by changes in land use as result of shifting cultivation due to decreased fallow periods. Cultivation on steep slopes with minimum soil conservation measures leads to surface runoff and erosion losses of soil due to erratic and erosive rainfall patterns.

The most common type of forest is tropical wet-evergreen forest and other forest type includes semi-evergreen forest and tropical-moist deciduous forest with occurrence of various species of bamboo forest in pockets. Bamboos spring out profusely in the middle and lower stories of the evergreen forest. *Melocanna baccifera* is the most dominant bamboo forest seen in the state (MIRSAC, 2007). The most important tree species occurring in the tropical wet-evergreen forest includes; *Dipterocarpus turbinatus, Artocarpus chaplasha, Terminalia myriocarpa, Amoora wallichii, Michelia champaca* and *Mesua ferrea*. The forest cover in the state extends to 18,005.51 km^2 which amounts to 85.41% of the total geographical area of the state. The state has experienced a decrease in forest cover from 86.27% (ISFR, 2017) to 85.41% (ISFR, 2019) which amounts to about 180.49 km^2 loss in comparison to the previous ISFR (2017) report. The forest cover in terms of canopy density class as per ISFR (2019) is shown in Table 3.1.

Table 3.1 Forest cover of Mizoram (in km^2).

Forest cover (as per canopy density class)	Area	Geographical area (%)
Very dense forest	157.05	0.74
Moderately dense forest	5800.75	27.52
Open forest	12,047.71	57.15
Total	**18005.51**	**85.41**
Scrub	0.90	0.00

Source: ISFR (2019)

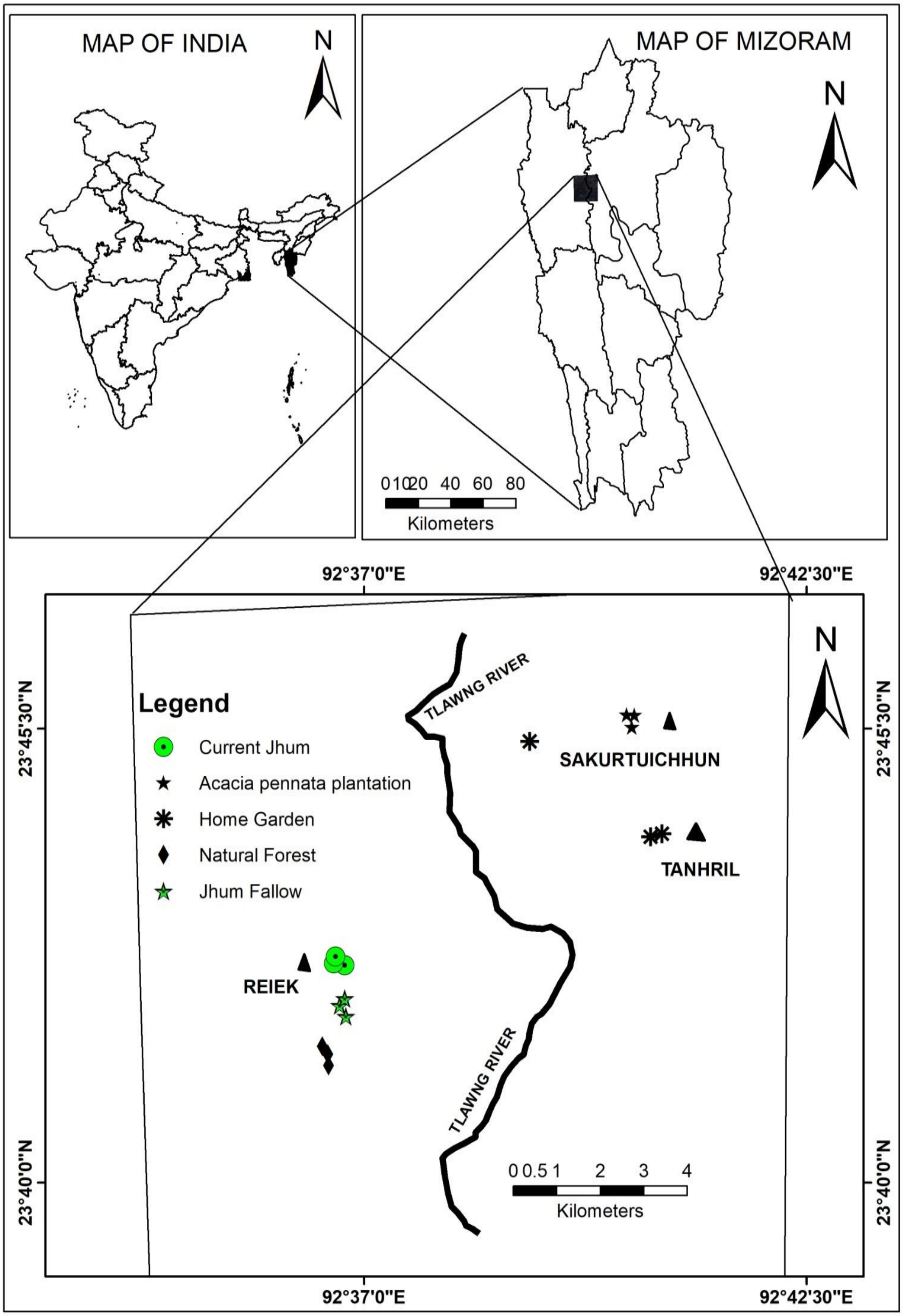


Figure 3.1 Map of study area showing different land use systems and study sites.

3.2. General lithology and statigraphy of the study site

The study sites fall under the age of miocene to upper oligocene and surma group, represented by bhuban formation and categorized as upper bhuban unit which is characterized by the occurrence of arenaceous with sandstones, shales and siltstones having a thickness of more than 1100 m with shallow marine and near to lagoonal depositions as represented in Figure 3.2 and Table 3.2.

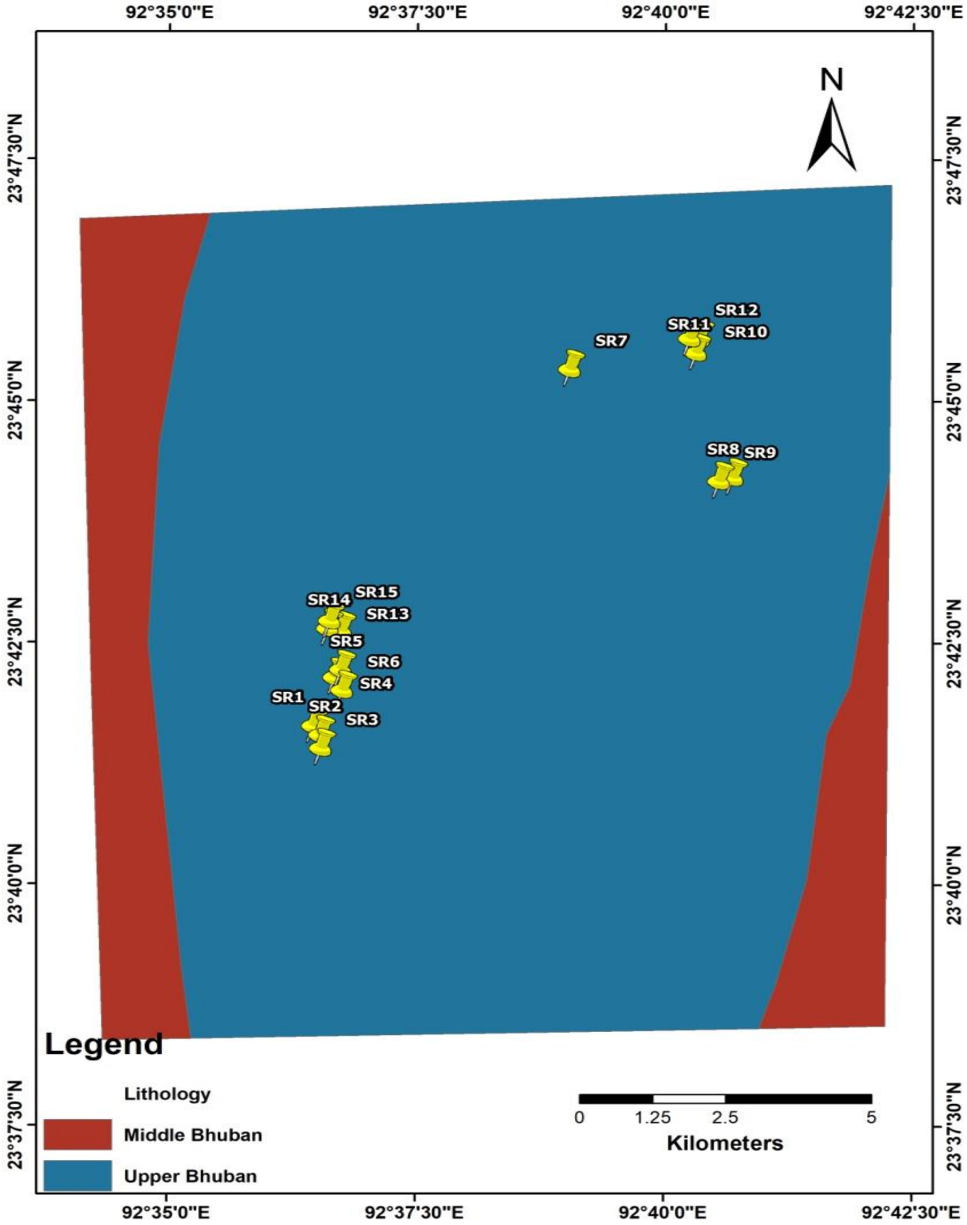
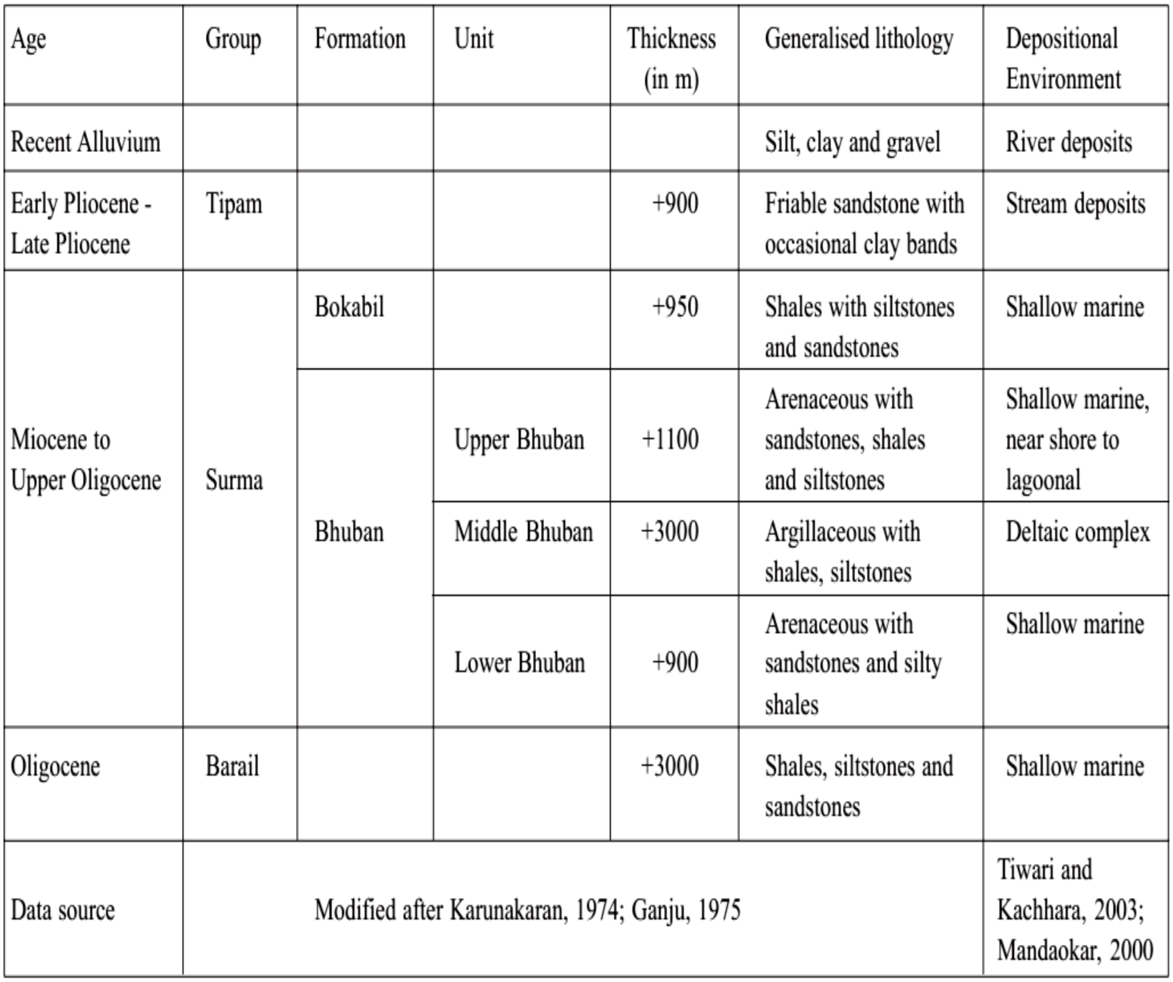


Figure 3.2 Lithological map showing study sites (Source: Geological Survey of India).

Table 3.2 General stratigraphy of Mizoram.



Age	Group	Formation	Unit	Thickness (in m)	Generalised lithology	Depositional Environment
Recent Alluvium					Silt, clay and gravel	River deposits
Early Pliocene - Late Pliocene	Tipam			+900	Friable sandstone with occasional clay bands	Stream deposits
Miocene to Upper Oligocene	Surma	Bokabil		+950	Shales with siltstones and sandstones	Shallow marine
		Bhuban	Upper Bhuban	+1100	Arenaceous with sandstones, shales and siltstones	Shallow marine, near shore to lagoonal
			Middle Bhuban	+3000	Argillaceous with shales, siltstones	Deltaic complex
			Lower Bhuban	+900	Arenaceous with sandstones and silty shales	Shallow marine
Oligocene	Barail			+3000	Shales, siltstones and sandstones	Shallow marine
Data source	Modified after Karunakaran, 1974; Ganju, 1975					Tiwari and Kachhara, 2003; Mandaokar, 2000

3.3. Criteria behind selection of land use systems for the study

Though, Mizoram has been one of the fastest developing states in the North-east region of India, majority of the rural population is still dependent shifting cultivation, locally called *Jhum*, for their primary source of livelihood since time immemorial. This practice is carried out annually to date, and each year hundreds of hectares of forest fallows are converted into current *Jhum* cultivation (cropping). The time taken for full recovery (10-15 years) until next cropping of these fallow lands varies as the altitude, topography and climate changes. Some other factors like past management practices, slope and aspect may also add to the variation in the recovery (time taken) of the current fallow. Therefore, there is an irregularity in the time of recovery of the fallow lands, especially when the fallow vegetation composition is slow growing and produces low biomass or the impact of inappropriate past land use management practices on soil health or other poor or no soil and water conservation measures. This poses immense pressure on the Government to identify potential land use systems as an alternative to *Jhum* or to implement innovative techniques within *Jhum* cultivation so as to promote sustainable land use management systems.

Considering the continuing land degradation, soil erosion and landslides, and the above views mentioned related to the study area, five prominent land use systems were selected for investigating their soil physical and bio-chemical properties so as to drawn any relevant inference that will aid in understanding how vegetative composition, land use and management practices and other extraneous factors affects the soil fertility and stability. The four important factors were thus formulated to represent the criteria's accounted for the selection of the different land use systems for this study; a) popularity of the land use; b) area coverage of the land use; c) relevance of the land use to livelihood; d) ecological significance of the land use.

3.4. Research approach

Five different land use systems were selected as experimental sites from Aizawl and Mamit district namely *Acacia pennata* plantation (AP), Home garden, (HG) Current *Jhum* (CJ), 12-15 years *Jhum* fallow (JF) and Natural forest (NAF) which served as control, against which the differences were compared. The ages of the lands sustaining the particular land use system were identified through interviews and interactions from the associated local farmers and land owners. The five different land use systems varied in their vegetative composition as well as management practices and hence, the assumption that the differences in soil properties would be attributed to the type of its land use, management practices and vegetation cover.

3.5. Description of study sites

The study sites falls within the two adjoining districts of Aizawl and Mamit which share a similar type of climate, vegetation and soil. A total of 15 study sites (5 land use systems x 3 site replicates) were located and identified within the two districts. Mamit is located at about 28 km away from Aizawl. Considering all the three forest class according to canopy density (very dense forest, moderately dense forest and open forest), Aizawl has a forest cover of about 3078.91 km^2 while Mamit covers 2716.87 km^2 (ISFR, 2019). The soils of the study sites are generally rich in organic carbon but poor in phosphorus content. The surface soils of the hilly areas are dark, leached and poor in primary nutrients but rich in iron and aluminium. The soil pH is highly acidic and varies from 4.3 to 5.5. A general description of the various land use systems along with their location, geo-coordinates and vegetative composition is shown in Table 3.3.

Table 3.3 Study sites along with their location, GPS coordinates and vegetative composition.

Land use systems	**GPS Coordinates**	**Location**	**Vegetative composition**
Natural Forest (NAF)	N 23° 41’ 32.8” E 092°36’33.4” 1209 m amsl	Reiek	*Schima wallichii, Toona ciliata, Castanopis tribuloides, Melocana baccifera, Erythrina indica, Derris robusta, Firmiana colorata, Sterculia villosa, Macaranga peltata, Mallotus paniculatus, Trema orientalis, Albizia chinesis*
Jhum fallow (JF)	N 23° 42’ 09.1” E 092°36’41.9” 1030 m amsl	Reiek	*Schima wallichii, Trema orientalis, Rhus semialata,, Thysanolaena maxima*
Home garden (HG)	N 23° 45’ 30.3” E 092°39’42.6” 964 m amsl	Tanhril	*Saccharum officinarum, Zea mays, Capsicum annum, Solanum melongena, Abelmoschus esculentus, Ananas comosus, Carica papaya, Manihot esculentum, Phaseolus vulgaris, Ipomea batatus, Zingiber officinalis, Clerodendrum colebrookianum, Mangifera indica, Areca catechu, Psidium guajava, Schima wallichi, Parkia roxburghii Musa sp. and Livestock*
Acacia pennata plantation (AP)	N 23° 45’ 37.6” E 092°40’22.4” 785 m amsl	Sakurtuichhun	*Acacia pennata*

Current *Jhum* (CJ)	N 23° 42’ 43.8” E 092°36’41.9” 889 m amsl	Reiek	*Solanum lycopersicum*, *Abelmoschus esculentus*, *Capsicum annum*, *Colocasia esculenta*, *Musa paradisiaca*, *Zea mays*, *Solanum melongena*, *Zingiber officinalis*, *Curcuma longa*

3.6. Methodology

3.6.1. Experimental design

Initially, for each land use system, three replicated sites (SR1, SR2 & SR3) measuring at least 20 x 25 m with similar topography, soil and vegetation were identified and established to signify true representatives of each land use type (Figure 3.3).

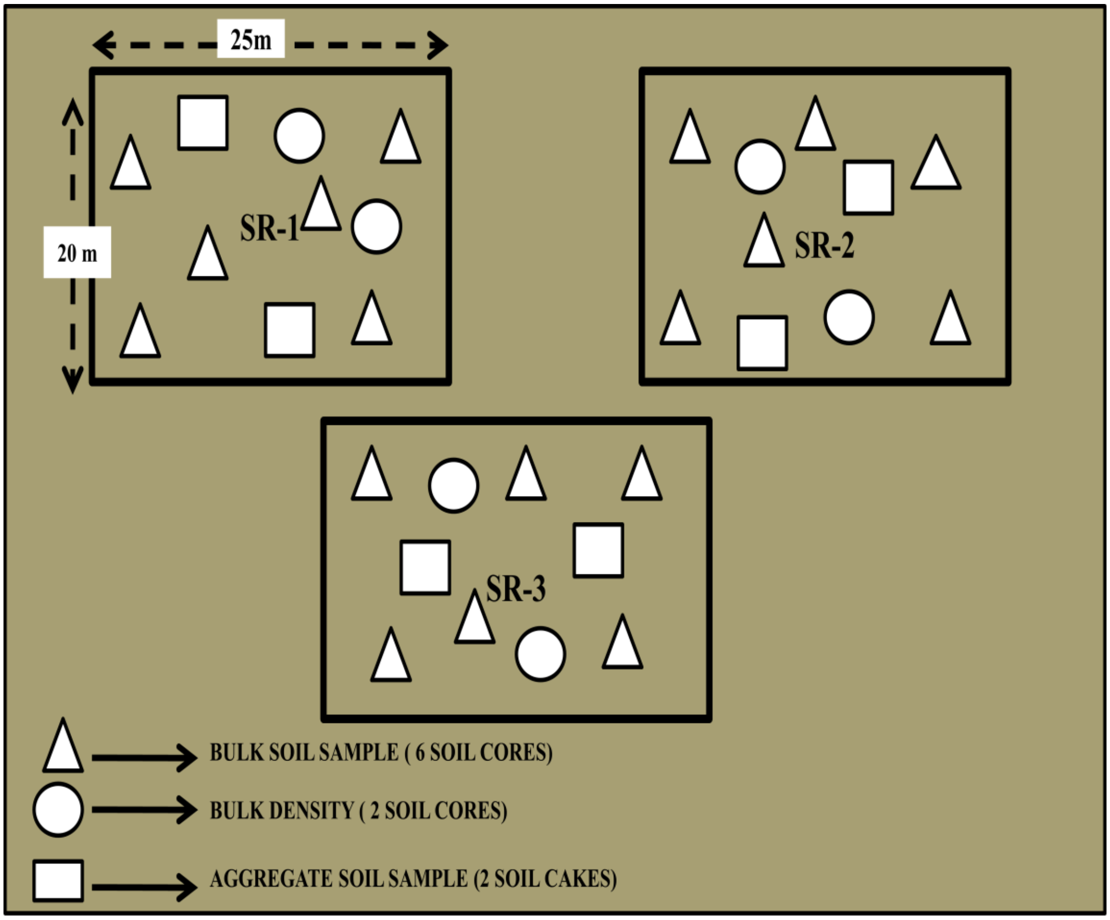


Figure 3.3 Representation of random sampling across different site replicates (SR1-site replicate 1, SR2-site replicate 2 & SR3-site replicate 3) for each land use systems.

3.6.2. Bulk soil sampling

All soil samples were collected twice consecutively during the year 2017 and 2018 from all sites in the month of May and June. Soil samples for physical and bio-chemical analysis were collected randomly from each site replicate by inserting a 10 cm scaled soil corer having an inner diameter of 5.2 cm. Soil samples were then drawn at three different depths i.e., 0-10 cm, 10-20 cm and 20-30 cm depth. Within each site replicate, 2 composited soil samples were collected for each depth where a single composite sample was composed of 3 random soil cores. Thus, a total of 90 samples (2 composite samples x 3 depths x 3 site replicates x 5 land use systems) were collected for the study. In case of bulk density, soil samples were collected using the same soil corer and a total of 90 samples (2 soil cores x 3 depths x 3 site replicates x 5 land use types) were collected for the analysis. Samples were then enclosed in a properly labelled zip locks and transported to the laboratory for further processing. The soil samples were then passed through a 2 mm mesh and further divided into two sub-samples, one was air dried and stored while the other was stored in a zip lock in -20° C for analysis.

3.6.3. Sampling for soil aggregate stability analysis

Soil samples for aggregate stability analysis were collected using a large knife with a smooth surface and a specially modified machete for easy slicing and to avoid any compression and disturbance of the soil and to minimize the risk of compaction. A soil cake measuring about 1 square feet in size was excavated using the specially prepared tools up to 30 cm soil depth. The soil cakes were then carefully broken by hand along their natural plane of weakness and the part of the soil not contacted by the tools were labelled and stored in hard plastic containers so as to avoid disturbance and mixing of samples. A total of 90 samples (2 aggregate samples x 3 depths x 3 site replicates x 5 land use systems) were collected for the analysis.

3.7. Laboratory analysis of soil properties

3.7.1. Analysis of soil physical properties

Soil bulk density (BD) was estimated from undisturbed soil using a core sampler of known volume as described by Robertson *et al.* (1974), after oven drying the samples at 105 °C till constant mass the dry weight was taken and calculated.

Soil moisture content (SMC %) was estimated by gravimetric method (Anderson and Ingram, 1993) by oven drying the known weight of field moist soil at 105°C for 24 hours. Soil colour was determined using a Munsell soil colour chart (Munsell, 1905). Soil texture was established following the Buoyoucous hydrometer method (Piper, 2005) and soil textural class was assigned using the USDA soil textural classification.

3.7.2. Soil aggregate stability analysis

Water stable soil aggregates were estimated following the method of wet sieving (Yoder, 1936; Elliott, 1986; John and Kim, 2002). Soil samples were air-dried for about a week and aggregates of 8 mm size were separated for the process of wet sieving. Eight different sieve sizes i.e., 4.00, 3.35, 2.80, 2.00, 1.40, 1.00, 0.50 and 0.212 mm were used to retain nine different size classes (>4.00 mm, 3.35 mm–4.00 mm, 2.80 mm–3.35 mm, 2.00 mm–2.80 mm, 1.40 mm–2.00 mm, 1.00 mm–1.40 mm, 0.50 mm–1.00 mm, 0.212 mm – 0.50 mm and <0.212 mm) using a mechanical wet sieve shaker developed by S.D. HARDSON CO. Kolkata. The 8 mm sieved aggregate was placed on the upper most sieve and the nest of sieves were then sited in a water column just covering the top sieve and allowed to oscillate for 30 min. The soils from each sieve were then washed down with water into beakers using a wash bottle. The labelled beakers containing the soil samples were then dried in the oven at 40 °C. After drying, the soil from each sieve were weighed and grouped under three aggregate size fractions (>2.00 mm, 0.25–2 mm and < 0.25 mm) namely macro-aggregate, meso-aggregate and micro-aggregate (Zhong *et al.,* 2019; Deb *et al.,* 2019). The different aggregate size fractions were then analyzed for associated chemical properties including TOC and TN. The resulting weights of the soil from each sieve size were represented as percentages relative of the total dry weight of sample. The following stability indices including water stable aggregates (WSA %), mean weight diameter (MWD, mm) and geometric mean diameter (GMD, mm) were computed as follows (Tripathi *et al*., 2012):

WSA % = Dry weight (g) of aggregates in each sieve size

/total dry weight (g) of aggregates x 100

$$\text{MWD} = \Sigma\, x_i y_i$$

where, y_i is the proportion of each size class concerning the total sample and x_i represents the mean diameter of the mesh size (mm).

$$GMD = \exp (\Sigma (w_i \ln x_i) \Sigma w_i)$$

where, w_i is the weight of the aggregate size class I (g) and x_i is the mean diameter of i^{th} size class.

3.7.3. Analysis of soil chemical properties

Soil pH was measured on the basis of the potentiometric principle (Peech, 1965) in a 1:25 soil/water solution using a pH meter. Soil organic matter (SOM %) was estimated following the loss on ignition (LOI) method, 25 grams of air dried soil samples were transferred in a crucible and dried in the oven at 40°C for about 6 hours to remove any excess moisture. The moisture-free samples were then heated in a muffle furnace at 660°C and the loss in weight was expressed as a percentage of SOM. Air dried soil samples passed through 2mm mesh sieve were used for the estimation of Soil Organic Carbon (SOC) and Total Nitrogen (TN) using CHNS/O Elemental Analyzer with autosampler and TCD detector –Euro Vector (Model: Euro EA3000, LECO). Air dried soil samples were extracted using Mehlich-I solution (0.05 M HCl + 0.025 M H_2SO_4) and analyzed for exchangeable nutrients (P_{avail}, Ca, Mg, Na, K and Mn) using Inductive coupled plasma Spectrometer (ICP-spectrometer, Model-6000 series, Thermo scientific). For the assessment of total nutrients like P_{total}, Ca, Mg, K, Na and Mn) air dry soil samples were digested using Aqua-regia (HNO_3 + 3 HCL) in a microwave oven which was then, diluted and filtered using Whatman filter paper. The aliquot samples were then analyzed using the Inductive coupled plasma Spectrometer (ICP-spectrometer, Model-6000 series, Thermo scientific).

3.7.4. Estimation of soil microbial biomass carbon (MBC)

Fresh soil samples were used for the analysis of soil microbial biomass carbon (MBC) following the chloroform-fumigation extraction method as outlined by Brookes and Joergensen (2006). For the estimation of MBC, 50 g of freshly collected soil samples were divided into two equal parts of 25 g each and placed in a 50 ml beaker. One part of of the samples was placed in the air tight dessicator containing chloroform vapours and the other part was kept as control without chloroform for 24 hours. Both the samples were then extracted using 0.5 M K_2SO_4 (1:4 ratio of soil to extractant) and shaken for 30 minutes. The samples were then filtered through Whatman no. 42 filter papers. Further, 10 ml of the supernatant aliquots were used for the estimation of C by wet oxidation method similar to the SOC estimation method described by Walkley and Black (1947). Thereafter, the difference between the fumigated and non-fumigated samples was taken out and MBC was calculated (Vance *et al.,* 1987; Wu *et al.,* 1990; Dilly and Munch, 1998).

3.7.5. Soil microbial population count

Soil microbial population was estimated using 1 g of freshly collected soil samples through serial dilution method outlined by Martin (1950). Soil sample of 1 gram of soil was added to the initial test tube containing 10 ml (dilution factor 10^{-1}) of distilled water. After mixing well, 1 ml of the solution from the first test tube was transferred to the subsequent test tube which contained 9 ml of distilled water which was labeled as dilution factor 10^{-2}. The same process was carried out to acquire dilution factors of 10^{-3}, 10^{-4}, 10^{-5}, 10^{-6} and 10^{-7}. Dilution plate technique experiment was carried out for obtaining colony forming units for the count of microbial populationas method given by Waksman (1922). The agar based media used for the population count was different for each of the different targets (fungi, actinomycetes and bacteria). Potato dextrose agar (PTA) was mixed with antibiotic ~ 0.08% of penicillin and chloramphenicol along with rose Bengal for the estimation of Fungi population. For the estimation of actinomycetes populaituon count, starch casein agar (SCA) mixed with nystatin (~0.08%) as antibiotic was used as the medium. Nutrient agar added with 0.08% of nystatin and actidione was used as media for assessment of bacterial population. Dilution of 10^{-3} to 10^{-5} were used for isolation of fungi, 10^{-4} to 10^{-6} for actinomycetes and 10^{-5} to 10^{-7} were used for bacterial isolation. After media preparation, 1 ml of each dilution was transferred into the petri-plates containing the solid media. Following, the media plates were incubated at 28 ± 1^0C for fungi and 25 ± 1^0C for bacteria and actinomycetes. Actinomycetes and bacteria population count was carried out after 24 hours of incubation has been completed while for fungi, it was incubated for 72 hours prior to population count. All the glassware's and media which were used were sterilized in an autoclave for 20 minutes at 120^0C.

3.7.6. Computations

Cation exchange capacity (CEC) for various land use systems and soil depths were established by summation of exchangeable cations (K^+, Ca^{+2}, Mg^{+2} & Na^+) to provide a measure of cation exchange capacity (CEC) as described by Robertson *et al.* (1999).

Soil carbon stock (Mg ha^{-1}) for each depth and land use system was computed following the formula given by of Blanco-Canqui and Lal (2008), while, the soil carbon stock for all the layers was summed to obtain the soil carbon stock for 0–30 cm depth in all the land use systems.

SOC stock (Mg ha^{-1}) = 10^4 (m^2/ha) x Soil depth (m) x BD (Mg/m^3) x SOC %/100
where, BD is bulk density and SOC (%) is soil organic carbon concentration.

3.8. Statistical analysis

All the data obtained are presented as mean (average) and standard error (SE). The resulting data's were subjected to two-way analysis of variance (ANOVA). Significant differences among soil variables, aggregate size fractions and microbial population were determined and least significant difference (LSD) were calculated to determine significant differences between means at $p \leq 0.05$. All statistical analysis was carried outusing open source OPSTAT (free Online Agriculture Data Analysis Tool developed by O.P. Sheoran, Computer Programmer at CCS HAU, Hisar, India). Again, Pearson correlation analysis between various soil properties was carried out using OPSTAT.

CHAPTER 4

RESULTS

4.1. Soil physico-chemcial properties

4.1.1. Changes in soil physical properties across different land use systems and soil depths

Soil colour ranged from brown to dark yellowish brown in the surface layer (0-10 cm), while the deeper depths (10-20 and 20-30) were dominated by light yellowish brown to brownish yellow (Table 4.2) across the different land use systems. Soil physical properties such as SMC, BD, sand and silt content were found to be significantly ($p<0.05$) affected by different land use systems (L), soil depths (D) and an interaction (LxD) of both factors. However, the effect of land use systems (L) on clay content was observed to be non-significant. The soil BD values in the surface layer was in the order: NAF (0.92 g cm^{-3}) < JF (1.13 g cm^{-3}) < CJ (1.2 g cm^{-3}) < AP (1.22 g cm^{-3}) < HG (1.25 g cm^{-3}) **(**Table 4.1**)**. Soil moisture content (SMC) was highest in JF with value amounting to 25.0 % in the surface layer (0-10 cm). While the least SMC content was recorded in AP with 12.4 % at the surface layer (0-10 cm). No noteworthy trend of SMC content with respect to soil depths could be observed. Soil textural class varied from sandy clay loam to loam in the surface layer (0-10 cm) and silty clay to clay loam in the deeper soil layers. Soil textural class ranged from sandy clay loam to clay loam in NAF, JF and CJ while in case of HG and AP, sandy clay loam and loamy texture were observed. The clay concentration in soil was highest in JF followed by CJ, whereas, the silt content was recorded higher in the sub-surface depths of HG by 50 % (Table 4.1). With respect to soil depths, clay was found to increase with increasing depth while sand concentration decreased with increasing depth. There was no notable trend observed in the concentration of silt with respect to depth. However, silt content was found to increase with increasing depth in HG and JF.

Table 4.1 Effects of different land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Homegarden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*) and soil depths (0-10, 10-20 & 20-30 cm) on soil physical properties.

	Soil Physical properties					
Land uses	**SMC (%)**	**BD (g/cm^3)**	**Clay (%)**	**Silt (%)**	**Sand (%)**	**Textural Class**
NAF						
(0-10)	23.9 ±0.2	0.92 ±0.03	24.1 ±1.3	21.4 ±1.8	54.5 ±1.8	Sandy clay loam
(10-20)	22.6 ±0.8	1.05 ±0.05	29.8 ±1.8	21.1 ±2.0	50.1 ±2.4	Sandy clay loam
(20-30)	22.2 ±1.4	1.15 ±0.05	31.5 ±0.7	21.3 ±1.2	47.2 ±0.7	Clay loam
JF						
(0-10)	25.0 ±0.4	1.13 ±0.03	29.8 ±2.4	22.0 ±2.3	48.2 ±0.7	Sandy clay loam
(10-20)	21.7 ±0.4	1.22 ±0.02	32.3 ±1.4	24.7 ±1.3	43.1 ±2.4	Clay loam
(20-30)	23.5 ±0.3	1.25 ±0.02	34.9 ±1.0	24.7 ±2.4	40.4 ±1.2	Clay loam
HG						
(0-10)	23.1 ±1.4	1.25 ±0.03	25.3 ±1.2	32.0 ±4.0	42.7 ±3.0	Loam
(10-20)	22.2 ±1.0	1.32 ±0.03	28.7 ±1.2	54.5 ±2.9	16.8 ±1.2	Silty clay loam
(20-30)	21.8 ±0.6	1.37 ±0.04	31.9 ±1.8	56.7 ±0.7	11.4 ±1.8	Silty clay loam
AP						
(0-10)	12.4 ±0.2	1.22 ±0.03	25.3 ±2.4	31.3 ±3.7	43.3 ±2.2	Loam
(10-20)	15.1 ±0.3	1.27 ±0.02	29.7 ±2.5	34.6 ±1.3	35.7 ±0.7	Clay loam
(20-30)	17.8 ±0.3	1.32 ±0.03	31.5 ±2.4	30.7 ±0.7	37.9 ±2.9	Clay loam
CJ						
(0-10)	14.6 ±0.3	1.17 ±0.02	28.5 ±0.7	26.0 ±2.0	45.5 ±2.4	Sandy clay loam
(10-20)	19.6 ±1.0	1.24 ±0.03	32.1 ±1.8	27.3 ±2.7	40.6 ±1.2	Clay loam
(20-30)	21.3 ±0.5	1.30 ±0.06	35.9 ±1.2	22.7 ±1.8	41.5 ±1.8	Clay loam
LSD$_{0.05}$						
L=	**1.232**	**0.065**	**NS**	**4.624**	**4.173**	
D=	**0.954**	**0.05**	**3.31**	**3.582**	**3.232**	
L x D=	**2.134**	**0.113**	**NS**	**8.009**	**7.227**	

Note: L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths, LSD$_{0.05}$: $p<0.05$, NS=Non-significant.

Table 4.2 Soil colour at different soil depths (0-10, 10-20 & 20-30 cm) across different land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Home garden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*).

	Munsell colour notation			
Land uses	**Hue**	**Value**	**Chroma**	**Soil colour**
NAF				
(0-10)	10 YR	4	3	Brown
(10-20)	10 YR	6	8	Brownish yellow
(20-30)	10 YR	7	8	Yellow
JF				
(0-10)	10 YR	4	4	Dark yellowish brown
(10-20)	10 YR	6	6	Brownish yellow
(20-30)	10 YR	7	6	Yellow
HG				
(0-10)	10 YR	4	6	Dark yellowish brown
(10-20)	10 YR	6	4	Light yellowish brown
(20-30)	10 YR	6	6	Brownish yellow
AP				
(0-10)	10 YR	5	3	Brown
(10-20)	10 YR	5	6	Yellowish brown
(20-30)	10 YR	6	6	Brownish yellow
CJ				
(0-10)	7.5 YR	5	3	Brown
(10-20)	10 YR	6	8	Light yellowish brown
(20-30)	10 YR	6	6	Light yellowish brown

4.1.2. Soil chemical properties

4.1.2.1. Changes in soil total organic carbon (TOC), total nitrogen (TN) and carbon to nitrogen ratio (C/N) across different land use systems and soil depths

ANOVA indicated significant ($p<0.05$) difference in TOC and TN concentrations in soil due to different land use systems (L) and soil depths (D) as well as their interaction (LxD). The TOC and TN values decreased significantly with increasing soil depth ($p<0.05$), however, an interesting trend of TN content in the soil of AP could be observed wherein, the highest TN content was recorded higher in the deeper soil depth (20-30 cm) than in the sub-surface soil layer (10-20 cm) with values 0.20% and 0.18% respectively (Fig. 4.2). The highest values of TOC and TN with values corresponding to 6.2 % and 0.50 % were observed in surface layer (0-10 cm) of NAF (Fig. 4.1). Among the various land use systems, the least contents of TOC and TN with values 1.10 % and 0.1 % respectively were reported at 20-30 cm soil layer of HG. The C/N of soil was also estimated to be significantly affected by land use systems and soil depths ($p<0.05$). C/N decreased with increasing depth in all the land use systems except in AP where the highest C/N was reported from the sub-surface depth (10-20 cm) of the soil. The highest C/N was recorded from JF in all the soil depths (Fig. 4.3), while the lowest was observed in the deeper soil depth (20-30 cm) of CJ with observed mean value of 8.05. At the surface layer (0-10 cm), C/N was observed to be in ascending order of; AP (11.41) < HG (11.56) < CJ (12.40) < NAF (13.12) < 10YJF (14.63).

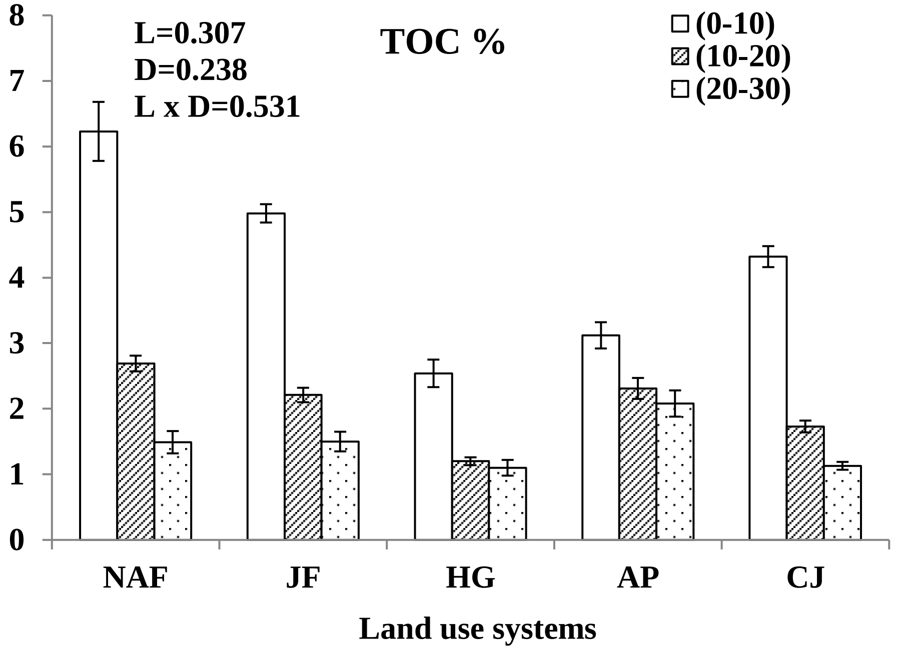


Figure 4.1 Effects of land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Homegarden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*) and soil depths (0-10, 10-20 & 20-30 cm) on total organic carbon (TOC %) content (L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

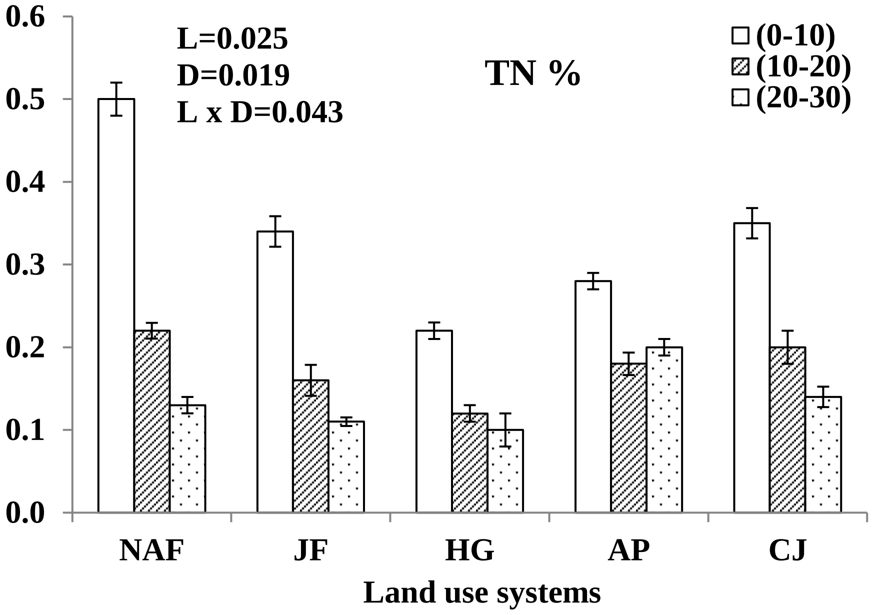


Figure 4.2 Effects of land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Homegarden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*) and soil depths (0-10, 10-20 & 20-30 cm) on total nitrogen (TN %) content (L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths, LSD$_{0.05}$: $p<0.05$, NS=Non-significant).

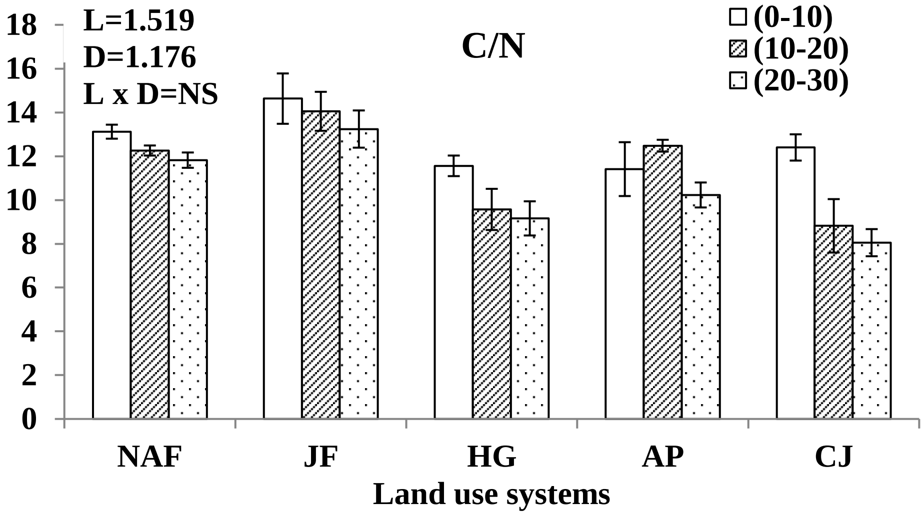


Figure 4.3 Effects of land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Homegarden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*) and soil depths (0-10, 10-20 & 20-30 cm) on Carbon:Nitrogen (C/N) ratio (L= Land use systems,

D= Soil depths, L x D= Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

4.1.2.2. Dynamics of C and N stocks across different land use systems and soil depths

Soil C stock ranged from 47.42 to 86.27 Mg ha^{-1} in the surface layer among the various land use systems. The highest soil C stock with respect to soil depth was estimated in the surface layer (0-10 cm) of NAF (86.27 Mg ha^{-1}), whereas the lowest was observed in HG (47.42 Mg ha^{-1}). Land use systems (L) and soil depths (D) significantly affected the concentration of C ($p<0.05$), wherein C stock declined with increasing soil depth (Fig. 4.4). Similarly in case of N stock the highest sequestration was recorded at the surface soil layer (0-10 cm) in all the land use systems**.** The highest N stock was recorded in the surface layer (0-10 cm) of NAF (6.95 Mg ha^{-1}) and the lowest in HG (4.09 Mg ha^{-1}) (Fig. 4.5)**.** The N concentration in the soil layer were significantly ($p<0.05$) affected by land use systems (L), soil depths (D) along with their interaction (LxD). A general trend of decreasing N stock with increasing soil depth was observed (Fig. 4.5) in almost all the land use systems except in AP where the N stock in the deeper soil layer (20-30 cm) was higher than the sub-surface soil depth (10-20 cm). A very peculiar trend of higher N sequestration could be observed in the 10-20 and 20-30 cm soil depths of AP where the highest N stock was observed among all the land use sysytems in those soil depths (Fig. 4.5). In general soil C and N stock were conserved highest in the surface layer (0-10) of NAF among all the land use systems.

In addition, total stock (0-30 cm) of C and N were significantly affected ($p<0.05$) by various land use systems (L). The trend of total C storage in 0-30 cm soil layer among the various land use systems was observed to be in the descending order of; NAF (153.36) > JF (152.66) > AP (140.38) > CJ (130.54) > HG (86.98) (Fig. 4.6). In case of N stock, CJ reported the highest followed by NAF (12.57), AP (12.39), JF (10.87) and lowest in HG (8.43 Mg ha^{-1}) (Fig. 4.7).

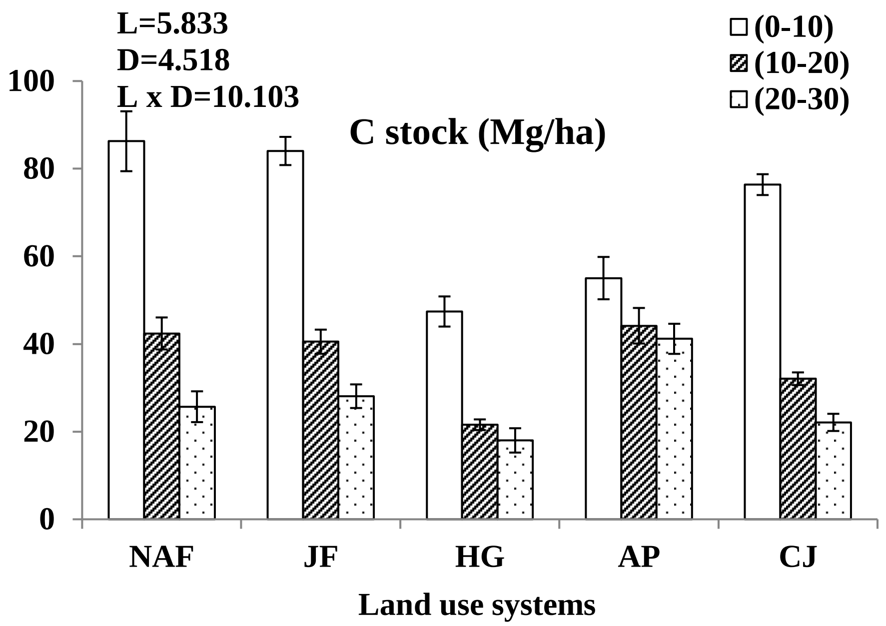


Figure 4.4 Effects of land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Homegarden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*) and soil depths (0-10, 10-20 & 20-30 cm) on C stock (L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths, LSD$_{0.05}$: p<0.05, NS=Non-significant).

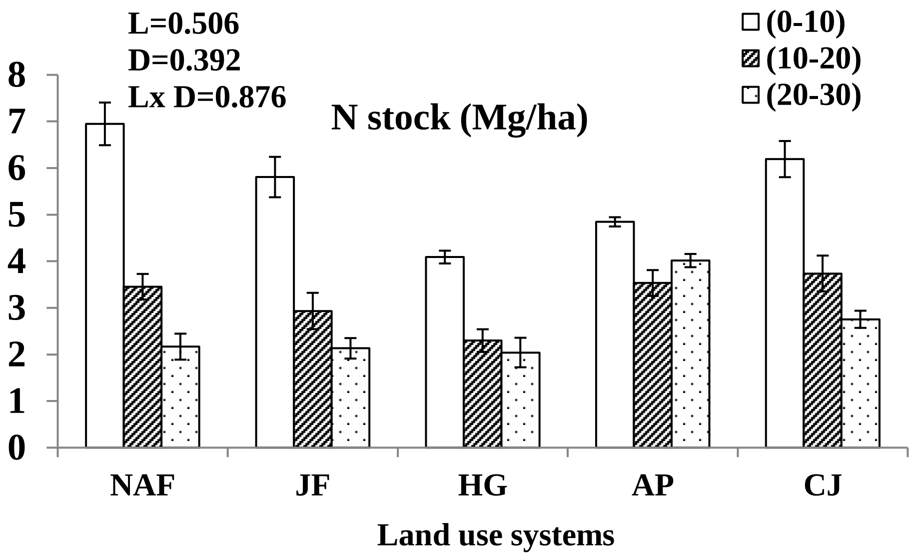


Figure 4.5 Effects of land use systems (NAF–Natural forest, JF–*Jhum* fallow, HG–Homegarden, AP–*Acacia pennata* plantation & CJ–Current *Jhum*) and soil depths (0-10, 10-20 & 20-30 cm) on N stock (L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths, LSD$_{0.05}$: p<0.05, NS=Non-significant).

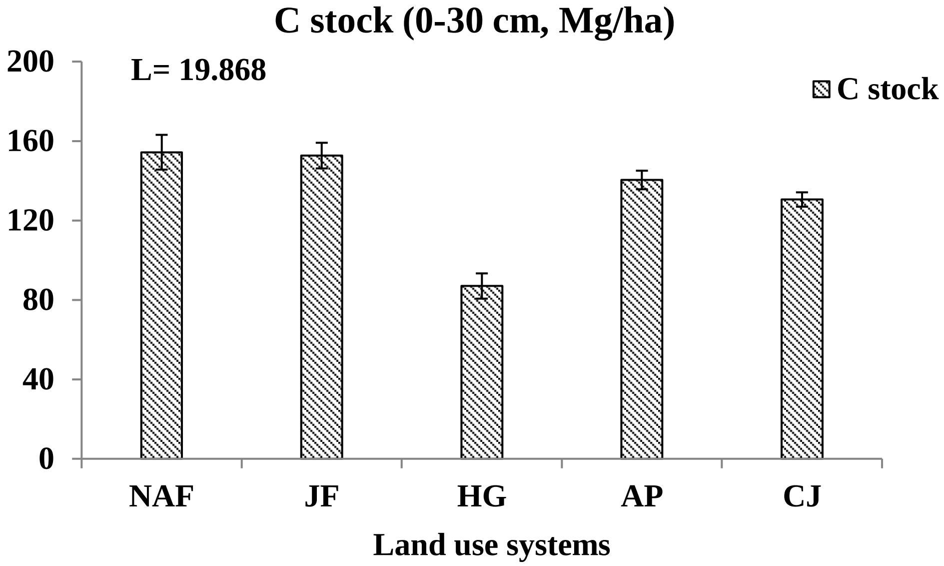


Figure 4.6 Effect of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum* –CJ) on C stock up to 30cm. (L= Land use systems, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

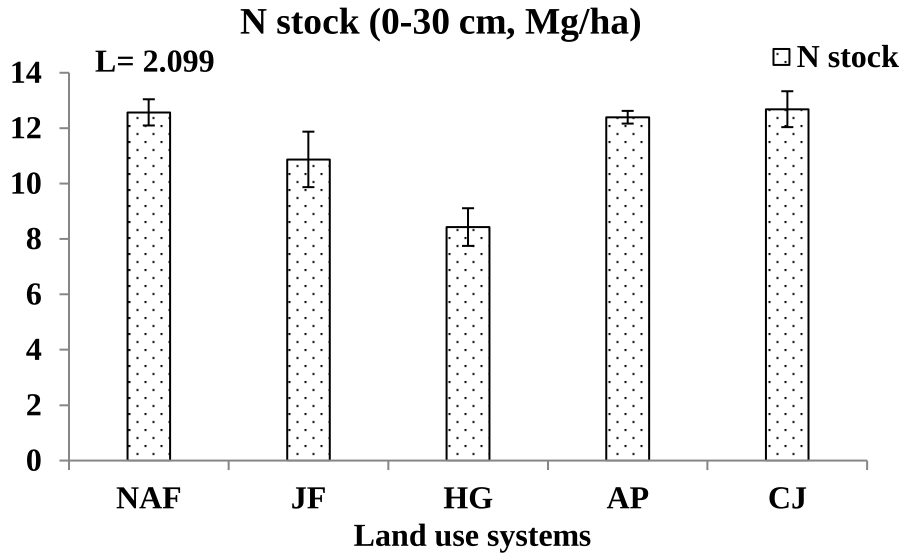


Figure 4.7 Effect of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) on N stock up to 30cm. (L= Land use systems, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

4.1.2.3. Magnitude (%) of changes in soil C and N stocks (Mg ha^{-1}) at 0-30 cm soil depth considering native forest (NAF) as a control site

Maximum decrease in soil C and N stock upto 0-30 cm depth was observed in HG where the decrease in the amount of C and N stocks were 67.37 Mg ha^{-1} and 4.14 Mg ha^{-1}, respectively. Minimum decrease in the amount of C stock was reported from JF (1.69 Mg ha^{-1}) followed by AP and CJ with values 13.98 Mg ha^{-1} and 23.81 Mg ha^{-1} respectively (Fig. 4.8). In case of decrease in N stock realtive to NAF, there was no loss recorded in CJ and an increase in about 0.11 Mg ha^{-1} could be reported, while JF and AP recorded a loss of 1.80 Mg ha^{-1} and 0.18 Mg ha^{-1}, respectively (Fig. 4.9).

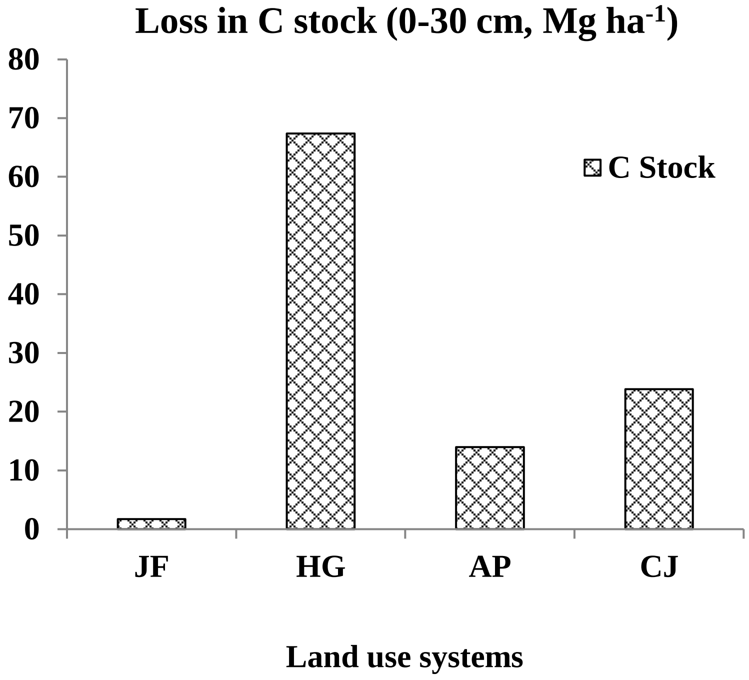


Figure 4.8 Decrease in soil C stock at 0-30 cm across different land use systems compared from native forest (NAF).

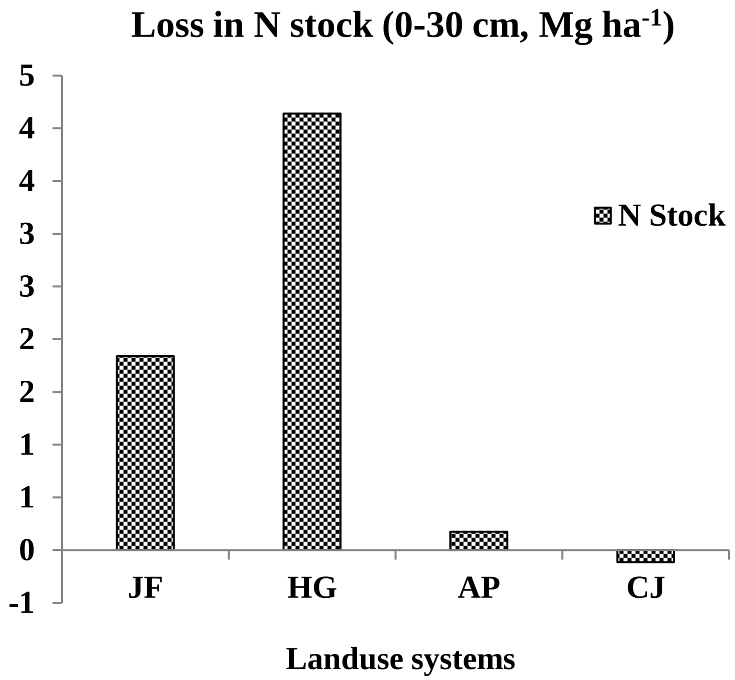


Figure 4.9 Decrease in soil N stock at 0-30 cm across different land use systems compared from native forest (NAF).

4.1.2.4. Soil organic matter (SOM %) across different land use systems and soil depths

Soil organic matter (SOM) was significantly ($p<0.05$) affected by different land use systems (L), soil depths (D) and an interaction of both factors (LxD). The highest SOM was observed to be conserved in NAF followed by JF and lowest was in HG. Highest SOM was reported from NAF at the 0-10 cm soil layer with a content of 13.79 % which followed a decreasing trend with increasing soil depths in all the land use systems (Fig. 4.10). The lowest SOM concentration at 20-30 cm soil depth was reported from HG with a value of 3.65 %.

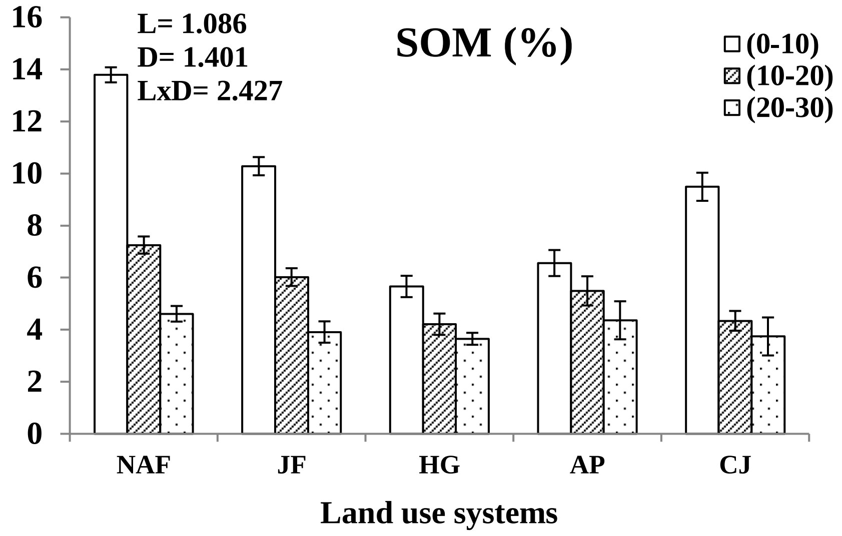


Figure 4.10 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on SOM content (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

4.1.2.5. Changes in soil pH, available phosphorus (P_{avail}), total phosphorus (P_{total}), exchangeable and total nutrients (Mg, K, Ca, Na and Mn) across different land use systems and soil depths

Generally, soil pH was acidic in all the land use systems and values decreased with increasing depth. It was observed to be highest in CJ (5.4) and lowest in AP (4.7) at the surface soil layer (0-10 cm). Soil pH was significantly varied among the different land use systems (L) and soil depths (D) ($p<0.05$). P_{avail} concentrations in soil was also found to be significantly affected by different land use systems (L), soil depths (D) as well as an interaction (LxD) of both factors ($p<0.05$). The highest P_{avail} concentrations at the surface layer (0-10 cm) was reported highest from HG followed by JF and the lowest in AP with values 30.83, 20.35 and 7.26 mg kg^{-1}, respectively (Table 4.3). In case of Mg, K and Mn, the highest concentration was recorded from JF in all the soil depths among all the land use systems. In the surface layer (0-10 cm), highest values of Mg, K and Mn in JF were recorded to be 751.32, 812.52 and 353.8 mg kg^{-1}, respectively. While the least concentration of Mg and K at the surface layer (0-10 cm) was reported from AP (207.42 mg kg^{-1} and 169.21 mg kg^{-1}, respectively). Mn and Ca were lowest in HG and CJ at the surface layer (0-10 cm) with values 102.1 and 119.65 mg kg^{-1}, respectively. Ca concentration was reported to be substantially higher in HG in comparison to other land use systems in all the soil depths. Highest Ca concentration was recorded at 720.1, 596.3 and 453.1 mg kg^{-1} at 0-10, 10-20 and 20-30 cm soil depths respectively in HG. In case of Na, the highest content was reported from HG with a value of 27.11 mg kg^{-1} and least in AP with 16.84 mg kg^{-1} (Table 4.3) in the surface layer (0-10 cm). These cations were significantly affected by different land use systems (L), soil depths (D) as well as an interaction (LxD) of both these factors ($p<0.05$). As a general trend, Mg, K, Ca, and Mn reflected depreciating values with increasing depth in all the land use systems.

Almost all the total concentration of nutrients were significantly ($p<0.05$) affected by land use systems (L), soil depths (D) and their interaction (LxD) except in case of Na where soil depths ($p>0.05$) did not show any significant affect on Na concentration in the soil. NAF and CJ showed a decreasing trend of concentration with increasing soil depth for all the total nutrients (P, Na, Mg, K, Ca and Mn) estimated. JF also showed a decreasing concentration with increasing depth in almost all the nutrients except for Na content. In all the land use systems studied, total P, Ca and Mn followed a decreasing trend of concentration with increasing soil depths. Total P content were reported highest in soil of NAF while Na concentration was observed greatest in the sub-surface depths of HG. Total concentrations of Mg, K, Ca and Mn were observed greatest in HG in comparison to the other land use systems. Total P content was recorded highest in the surface layer (0-10 cm) of NAF with a concentration of 1480.6 mg kg^{-1} in comparison to the rest of the land uses which recorded just about 1/3 of the value in all soil depths (Table 4.4). Highest concentration of Na (80.33 mg kg^{-1}) and K (4759.36 mg kg^{-1}) were recorded from the deeper depth (20-30 cm) of the soil layer in HG. Greatest concentrations of total Mg (4611.46 mg kg^{-1}) was observed at 10-20 cm soil depth of HG, while, highest Ca and Mn were observed at 3138.5 mg kg^{-1} and 863.35 mg kg^{-1} respectively at the surface layer (0-10 cm) of HG (Table 4.4). In general, soils of HG recorded highest concentration of nutrients except for P content which was substantially higher in NAF.

Table 4.3 Effects of different land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum* –CJ) and soil depths (0-10, 10-20 & 20-30cm) on soil pH, P_{avail} and exchangeable cations.

Land uses	**Soil pH & exchangeable nutrients (mg kg^{-1}, dry soil basis)**						
NAF	**pH**	**P_{avail}**	**Na**	**Mg**	**K**	**Ca**	**Mn**
(0-10)	4.8 ±0.10	14.21 ±0.22	20.49 ±2.62	366.89 ±1.77	198.27±8.99	281.87±22.8	193.11±4.1
(10-20)	4.7 ±0.03	10.38 ±0.25	22.54 ±1.28	225.33 ±1.71	132.91±2.09	143.19±6.18	122.6±1.4
(20-30)	4.5 ±0.13	4.61 ±0.24	11.89 ±1.06	215.88 ±2.00	117.79±1.57	124.57±2.06	103.01±1.8
JF							
(0-10)	4.8 ±0.12	20.35 ±2.02	21.27 ±1.27	751.32±21.4	812.52 ±22.2	217.94±21.3	353.8±24
(10-20)	4.6 ±0.03	5.95 ±0.38	25.85 ±1.47	315.22±10.5	214.39 ±25.3	117.31±3.32	173.3±14
(20-30)	4.5 ±0.06	3.66 ±0.21	18.73 ±2.22	247.6±18.32	108.53 ±13.7	84.83±5.31	96.65±5.2
HG							
(0-10)	5.1 ±0.07	30.83 ±2.27	27.11 ±2.29	441.24±17.3	224.79±14.2	720.1±30.1	102.1±9.2
(10-20)	4.5 ±0.07	12.57 ±1.23	17.39 ±1.01	202.32±24.8	87.47±7.52	596.30±20.0	33.94±1.3
(20-30)	4.3 ±0.09	9.03 ±0.67	18.66 ±1.70	173.81 ±1.5	60.65 ±8.63	453.10±15.1	16.48±2.1
AP							
(0-10)	4.9 ±0.07	7.26 ±0.12	16.84 ±1.69	207.42±10.3	169.21±15.5	610.1±21.54	131.9±13.6
(10-20)	4.5 ±0.07	4.13 ±0.15	13.20 ±1.33	86.59 ±2.46	101.35±10.2	193.19±20.5	69.92 ±7.28
(20-30)	4.2 ±0.11	3.51 ±0.21	15.94 ±1.43	57.46 ±1.99	78.32 ±15.13	132.69±12.8	52.35 ±3.15
CJ							
(0-10)	5.4 ±0.22	9.65 ±0.35	18.29 ±0.96	276.20 ±2.25	369.63±13.4	119.65±20.7	146.1±16.1
(10-20)	4.7 ±0.30	5.40 ±0.14	17.76 ±1.55	146.92 ±2.45	160.02 ±9.70	148.57 ±0.77	55.15±11.5
(20-30)	4.3 ±0.06	4.48 ±0.18	14.13 ±1.60	127.69 ±1.78	131.98 ±2.78	136.02 ±8.81	70.54 ±2.02
$LSD_{0.05}$							
L=	**0.215**	**1.487**	**2.739**	**19.113**	**22.045**	**36.642**	**17.029**
D=	**0.167**	**1.152**	**2.122**	**14.805**	**17.076**	**28.383**	**13.191**
L x D=	**0.373**	**2.575**	**4.745**	**33.105**	**38.183**	**63.465**	**29.496**

Note: L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths $LSD_{0.05}$: $p<0.05$, NS=Non-significant.

Table 4.4 Effects of different land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on total soil nutrient contents.

Landuses	**Total soil nutrients content (mg kg^{-1}, dry soil basis)**					
	P	**Na**	**Mg**	**K**	**Ca**	**Mn**
NAF						
(0-10)	1480.6±27.4	78.54±3.45	2328.29±42.45	1933.43±25.0	1748.39±43.4	191.62±4.17
(10-20)	459.21±23.44	66.90±3.01	2120.59±33.23	1835.53±33.96	537.75±25.45	102.68±1.63
(20-30)	328.28±36.58	41.76±3.77	1700.76±40.21	1353.20±46.88	444.59±21.60	75.50±3.59
JF						
(0-10)	368.59±5.57	39.75±3.45	2593.85±54.16	2087.07±10.54	855.6±43.5	290.54±37.82
(10-20)	270.31±4.79	66.15±5.00	2500.27±51.84	1960.10±43.21	554.09±32.5	150.08±2.62
(20-30)	240.42±4.64	58.83±5.39	2381.52±49.68	1672.05±63.63	404.66±23.9	158.54±3.53
HG						
(0-10)	527.41±23.48	59.57±6.69	4611.46±102.1	4561.35±199.5	3138.5±103.7	863.35±79.58
(10-20)	391.20±23.36	86.60±12.1	3811.03±333.1	4491.84±232.5	833.33±92.13	772.34±28.26
(20-30)	383.52±10.29	80.33±9.63	4429.76±204.9	4759.36±157.3	658.10±25.93	723.58±19.28
AP						
(0-10)	340.73±5.34	45.38±5.40	1981.00±46.45	1123.12±24.81	246.42±8.55	190.13±3.55
(10-20)	223.87±29.46	37.27±2.67	2171±32.40	1234.42±32.36	126.20±4.17	97.25±4.27
(20-30)	165.28±8.81	40.48±3.57	2092.04±153.9	1384.36±21.11	100.81±4.01	76.09±2.24
CJ						
(0-10)	384.87±3.09	52.76±5.09	3609.36±46.17	2369.03±43.77	712.76±32.66	303.38±5.38
(10-20)	382.68±3.39	40.67±3.45	3515.74±39.73	2310.02±55.07	655.21±20.64	293.64±4.06
(20-30)	299.02±3.04	38.00±3.07	3465.38±65.38	2244.50±180.8	133.98±57.01	235.96±24.39
$LSD_{0.05}$						
L=	**23.461**	**7.516**	**153.891**	**137.481**	**56.338**	**32.94**
D=	**30.289**	**NS**	**198.672**	**177.488**	**72.732**	**42.526**
L x D=	**52.461**	**16.806**	**344.111**	**307.418**	**125.975**	**73.657**

Note: L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths

$LSD_{0.05}$: $p<0.05$, NS=Non-significant.

4.1.2.6. Soil cation exchange capacity (CEC) across different land use systems and soil depths

The CEC across various land use systems ranged from 3.902 to 9.53 c mol kg^{-1} at the surface layer (0-10 cm). Maximum cation exchange capacity (CEC) was recorded in JF (9.53 c mol kg^{-1}) in the surface layer, whereas, in the subsurface layers, CEC were observed highest in HG with values 4.94 and 3.91 cmol kg^{-1}, respectively for 10-20 and 20-30 cm soil depths (Fig. 4.11). The lowest values of CEC at the surface layer (0-10 cm) was reported from CJ (3.902 c mol kg^{-1}), while the lowest vaues of CEC with values 2.005 cmol kg^{-1} and 1.412 cmol kg^{-1} were recorded at 10-20 and 20-30 cm soil depths respectively in AP. CEC was also observed to change in all the land use systems with a decrease in level with increasing soil depths. Land use systems (L), soil depths (D) and an interaction of these factors was observed to have a significant effect ($p<0.05$) on the soil CEC levels.

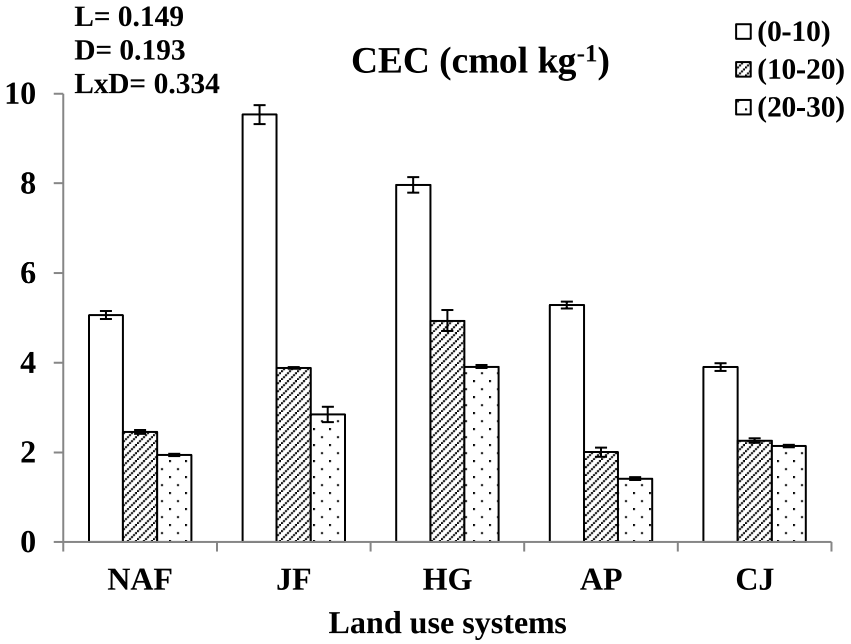


Figure 4.11 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on soil CEC levels. (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, LSD$_{0.05}$: $p<0.05$, NS=Non-significant).

4.2. Soil aggregate stability and aggregate sizes associated TOC and TN

4.2.1. Changes in soil aggregate sizes (macro-, meso- and micro-aggregates) in the soil layer across different land use systems

Water stable soil aggregates varied widely among the different land uses and soil depths. Highest macro-aggregate concentration was observed in the surface layer (0-10 cm) in all the land use systems and dereased with increasing soil depths. Water stable soil macro-aggregates at the surface soil layer (0-10 cm) ranged from 59.57 to 83.36 % with the maximum contetnt in JF (83.36 %). Macro-aggregates declined drastically at 10-20 and 20-30 cm soil depths of HG with values 29.68 and 14.68 %, respectively. Whereas, the highest levels of macro-aggregates at the sub-surface soil (10-20 cm) were recorded in NAF at 58.36 % (Fig. 4.12).

Proportion of water stable meso-aggregates was significantly affected by land use systems (L), soil depths (D) and an interaction (LxD) of both factors ($p<0.05$). Meso-aggregates did not show any common trend of distribution in the soil layer across the various land use systems, however, an increase in meso-aggergate with increasing soil depths was observed in JF and AP. Highest levels of water stable soil meso-aggregates were recorded at values 46.51 and 45.08 %, respectively, at 10-20 and 20-30 cm soil depths in HG (Fig. 4.13). Similarly, the highest proportion of soil micro-aggregates was obsereved at 20-30 and 10-20 cm soil depths with values 20.35 and 13.54 %, respectively in HG (Fig. 4.14). It was observed that soil micro-aggregates increased with increasing soil depths except in CJ where the sub-surface layer (10-20 cm) had lesser micro-aggregate proportion than the surface soil layer (0-10 cm). Water stable soil micro-aggregates was found to be significantly affected by land use systems (L), soil depths (D) and an interaction (LxD) of both factors ($p<0.05$). In general, water stable soil macro-, meso- and micro-aggregates were significantly affected by land use systems (L), soil depths (D) and an interaction (LxD) of both factors ($p<0.05$).

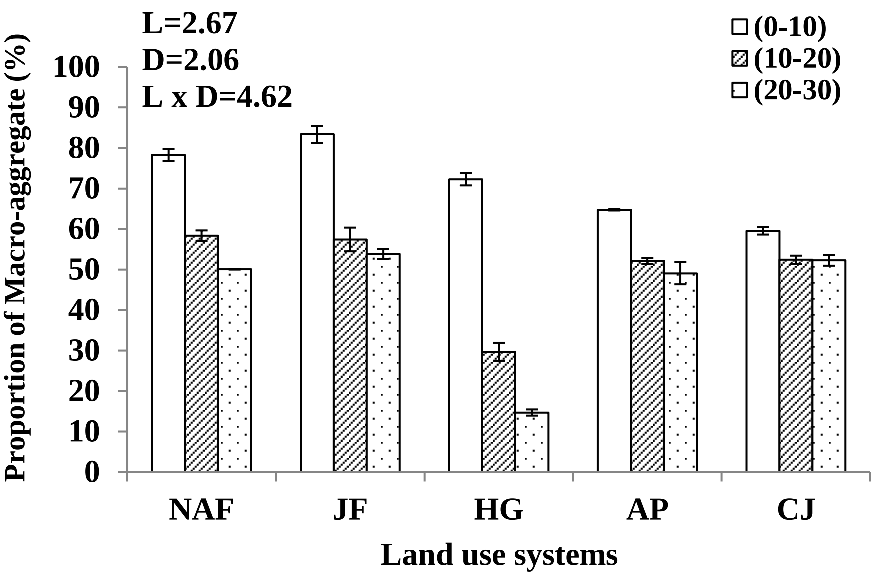


Figure 4.12 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on water stable soil macro-aggregates. (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

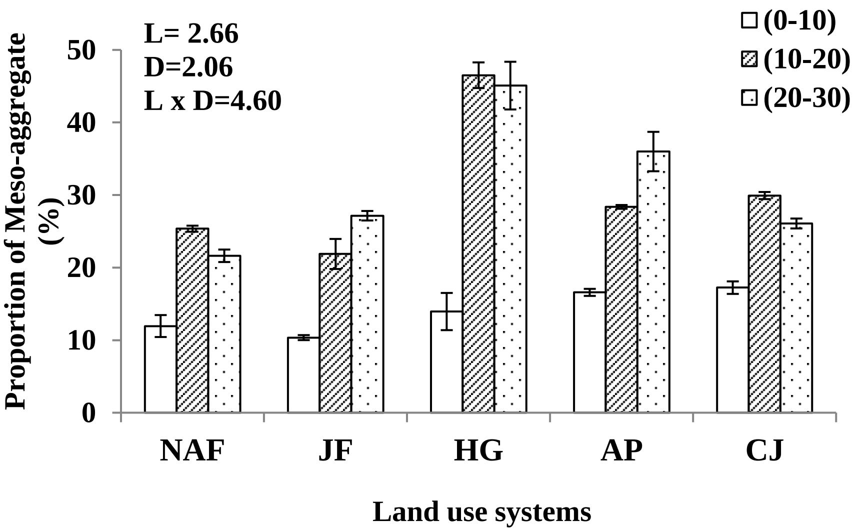


Figure 4.13 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on water stable soil meso-aggregates. (L= Land

use systems, D= Soil depths, LxD=Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant.

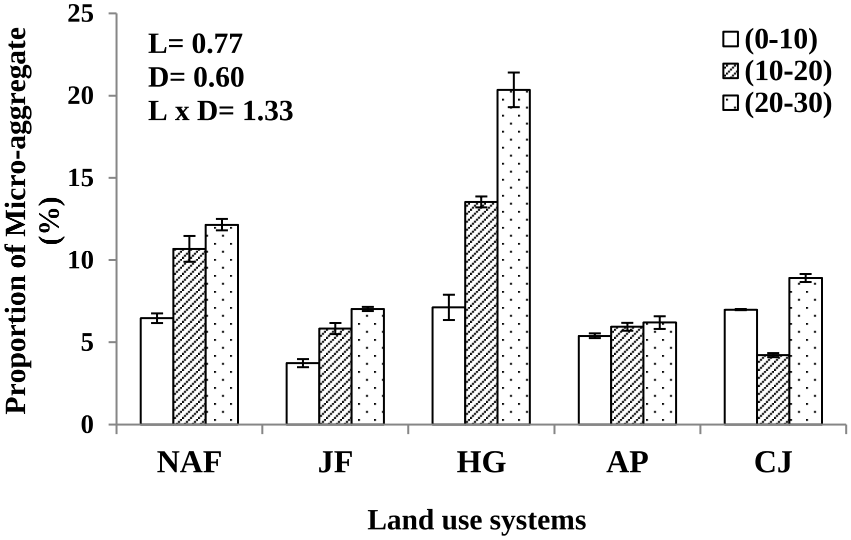


Figure 4.14 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on water stable soil micro-aggregates. (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

4.2.2. Changes in mean weight diameter (MWD) and geometric mean diameter (GMD) across different land use systems and soil depths

Mean weight diameter (MWD) and geometric mean diameter (GMD) significantly ($p<0.05$) varied among the diffrent land use systems (L) and soil depths (D). Land use systems and soil depths interaction (LxD) was also found to be significant ($p<0.05$). Both MWD and GMD decreased with increasing soil depth in all the land use systems. Values of MWD ranged from 4.10 to 5.15 (Fig. 4.15), while GMD ranged from 1.63 to 1.90 (Fig. 4.16) at the surface layer (0-10 cm). In the surface layer (0-10 cm), highest MWD (5.15) and GMD (1.90) were recorded in JF, while the lowest values of MWD (4.10) and GMD (1.63) were recorded in CJ. MWD and GMD at the sub-surface depths (10-20 and 20-30 cm) were obsereved to be lowest in HG in comparison to other land use systems. The MWD values in HG at 10-20 and 20-30 cm soil depths were observed at 2.16 and 1.74 respectively. Similarly, among all land uses, GMD values were observed to be lowest in HG at both 10-20 and 20-30 cm soil layer with values 1.17 and 1.04 respectively.

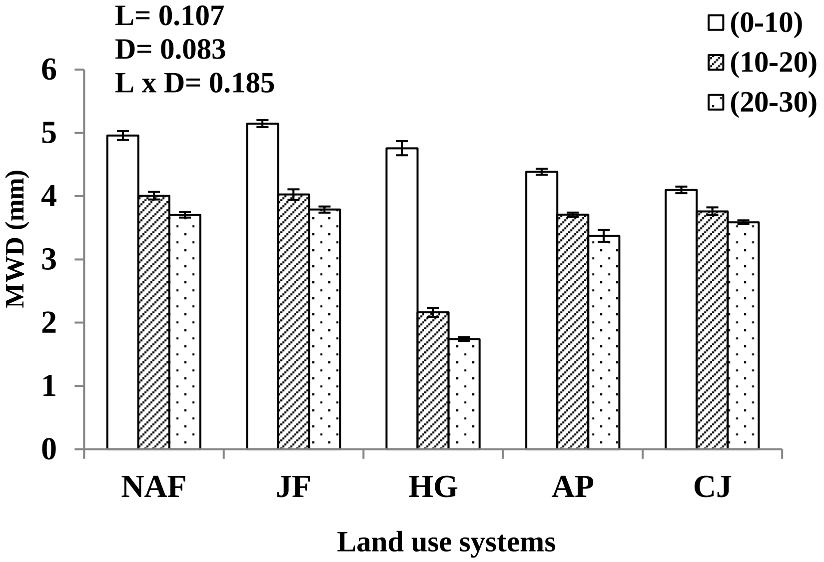


Figure 4.15 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on mean weight diameter (MWD). (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

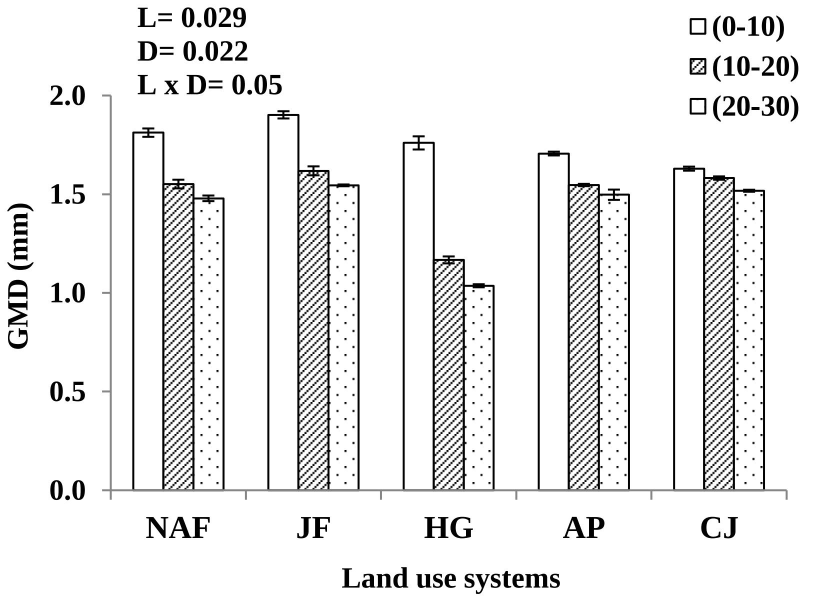


Figure 4.16 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on geometric mean diameter (GMD). (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, $LSD_{0.05}$: $p<0.05$, NS=Non-significant).

4.2.3. Associated total organic carbon (TOC %) and total nitrogen (TN %) in different soil aggregate sizes (Macro-, meso- and micro-aggregates) across different land use systems and soil depths.

Associated TOC and TN in different soil aggregate size fractions (macro-, meso- and micro-aggregate) were observed to be significantly affected by different land use systems (L), soil depths (D) and their interaction (LxD). Macro-, meso- and micro-aggregate associated TOC and TN decreased with increasing soil depths in all the land use systems. In addition, associated TOC and TN were found to be higher in macro-aggregates in comparison to meso- and micro-aggregate. Highest aggregate associated TOC and TN were recorded in NAF at the surface layer (0-10 cm) with values 5.65 and 0.42 %, respectively (Table 4.5). Comparatively, the least concentration of aggregate associated TOC and TN were recorded in HG for all aggregate fractions (macro-, meso- and micro-aggregates). Following NAF, CJ also showed relatively higher TOC and TN content in macro-, meso- and micro-aggregate fractions in comparison to JF, HG and AP land uses. At the surface layer (0-10 cm), aggregate associated TOC ranged from 1.79 to 5.65 % in macro-aggregate, 1.61 to 4.4 % in meso-aggregate and 1.48 to 3.5 % in micro-aggregates across the different land use systems. Similarly, aggregate associated TN concentration at the surface soil layer (0-10 cm) ranged from 0.21 to 0.42 % in macro-aggregate, 0.19 to 0.35 % in meso-aggregate and 0.16 to 0.28 % in micro-aggregates in across the various land use systems (Table 4.5).

Table 4.5 Effects of different land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum* –CJ) and soil depths (0-10, 10-20 & 20-30 cm) on associated TOC and TN in different aggregate size fractions (Macro-, meso- and micro-aggregates).

Land uses	**Assocaited TOC (%)**			**Associated TN (%)**		
NAF	**Macro-aggregate**	**Meso-aggregtae**	**Micro-aggregate**	**Macro-aggregate**	**Meso-aggregate**	**Micro-aggregate**
(0-10)	5.65±0.11	4.4±0.04	3.58±0.06	0.42±0.03	0.35±0.013	0.28±0.01
(10-20)	3.405±0.13	3.03±0.12	2.39±0.13	0.26±0.03	0.23±0.01	0.22±0.01
(20-30)	2.295±0.04	2.2±0.11	1.84±0.08	0.2±0.01	0.19±0.02	0.17±0.02
JF						
(0-10)	2.395±0.04	2.07±0.2	1.9±0.05	0.23±0.01	0.19±0.11	0.17±0.01
(10-20)	1.455±0.10	1.35±0.05	1.23±0.03	0.15±0.03	0.14±0.02	0.125±0.01
(20-30)	0.895±0.02	0.86±0.04	0.79±0.03	0.12±0.02	0.11±0.011	0.09±0.01
HG						
(0-10)	1.795±0.02	1.61±0.05	1.48±0.02	0.21±0.03	0.21±0.017	0.16±0.01
(10-20)	1.36±0.02	1.26±0.03	1.08±0.01	0.18 9±0.02	0.16±0.01	0.14±0.01
(20-30)	1.32±0.03	1.01±0.05	0.96±0.03	0.17±0.02	0.15±0.013	0.15±0.01
AP						
(0-10)	2.485±0.02	2.38±0.02	2.07±0.07	0.27±0.04	0.25±0.01	0.23±0.01
(10-20)	1.965±0.03	1.85±0.06	1.73±0.05	0.22±0.01	0.20±0.01	0.18±0.01
(20-30)	1.475±0.06	1.46±0.05	1.43±0.02	0.185±0.02	0.17±0.01	0.15±0.01
CJ						
(0-10)	2.975±0.27	2.41±0.03	2.25±0.15	0.34±0.01	0.31±0.02	0.28±0.02
(10-20)	1.465±0.04	1.29±0.05	1.23±0.05	0.20±0.03	0.19±0.01	0.18±0.01
(20-30)	1.215±0.05	1.11±0.05	1.07±0.12	0.18±0.03	0.169±0.01	0.13±0.01
$LSD_{0.05}$						
L=	**0.185**	**0.108**	**0.192**	**0.017**	**0.015**	**0.013**
D=	**0.239**	**0.14**	**0.247**	**0.022**	**0.02**	**0.017**
LxD=	**0.414**	**0.242**	**0.428**	**0.038**	**0.034**	**0.03**

Note: L= Land use systems, D= Soil depths, L x D= Land use systems x Soil depths $LSD_{0.05}$: $p<0.05$, NS=Non-significant.

4.3. Soil microbial properties

4.3.1. Soil microbial biomass carbon (MBC) across different land use systems and soil depths

Soil microbial biomass carbon (MBC) followed a trend of decreasing levels with increasing soil depth. The highest MBC was reported from NAF in all the soil depths in comparison to other land use systems. MBC at the surface soil layer (0-10 cm) ranged from 391.43 to 634.53 mg kg^{-1} (Fig. 4.17). MBC content was also observed to be significantly affected by various lands use systems (L), soil depths (D) and their interaction (LxD) ($p<0.05$). The highest MBC content in soil was reported from NAF with 634.53 mg kg^{-1} at the surface soil layer (0-10 cm) and followed a descending order of JF>CJ>AP>HG. Lowest values of MBC at 10-20 and 20-30 cm soil depths were observed in HG and CJ with values 229.31 and 79.04 mg kg^{-1} respectively.

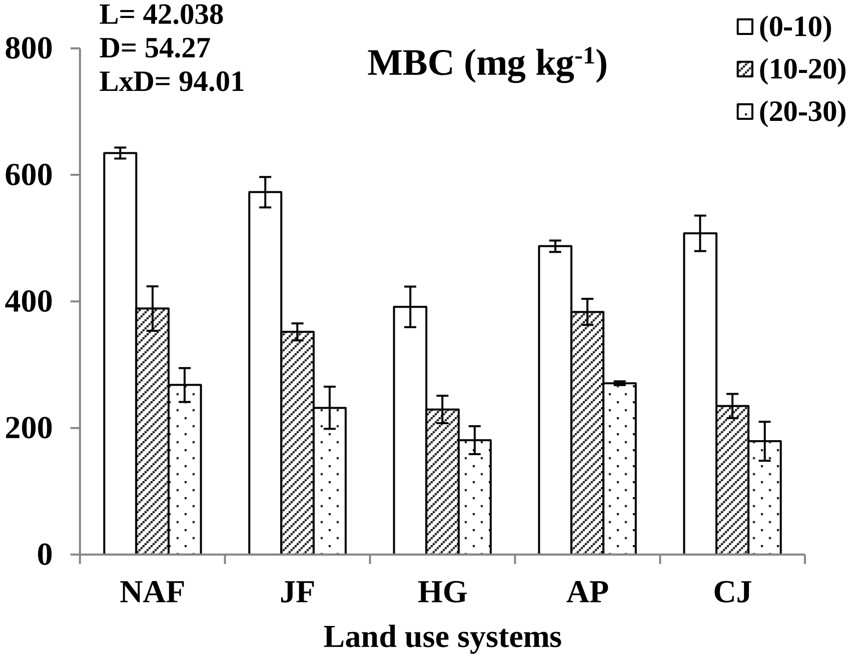


Figure 4.17 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on soil microbial biomass carbon (MBC). (L= Land use systems, D= Soil depths, LxD=Land use systems x Soil depths, LSD$_{0.05}$: $p<0.05$, NS=Non-significant).

4.3.2. Soil microbial population (fungi, bacteria and actionmycetes) across different land use systems and soil depths

Fungi population was found to exist highest in JF at 0-10 cm with 22.5 CFU’s g^{-1} (Fig. 4.18) soil in comparison to rest of the land use systems and soil depths. It was also observed that the population followed a decreasing trend of incidence with increasing soil depths in all the land use systems. The least population of fungi at the surface layer (0-10 cm) was reported from HG (18.3 CFU’s g^{-1} soil). Similarly, the least population of fungi at the deeper soil depth (20-30 cm) was observed from HG and CJ with a common value of 8.0 CFU’S g^{-1} soil. In case of bacteria, the population at the surface soil (0-10 cm) across the land use systems ranged from 140.3 to 186.0 CFU’S g^{-1} soil (Fig. 4.19), with the highest incidence in NF and lowest in HG. It was also observed that the bacterial population decreased with increasing soil depths in all the land use systems. Similarly, actinomycetes population was higher in NAF at the 0-10 cm soil layer (54.7 CFU’S g^{-1} soil) followed by JF (51.2 CFU’S g^{-1} soil) and the least was observed in HG (28.6 CFU’S g^{-1} soil). While at the sub-surface soil layer (10-20 cm), the highest incidence of actinomycetes was recorded in AP with 46.7 CFU’S g^{-1} soil (Fig. 4.20). Similar to the trend of fungi and bacteria population in the soil layers, actinomycetes also followed a decreasing trend of incidence with increasing soil depths in all the land use systems. As a general trend, population of fungi, bacteria and actinomycetes were observed to occur lowest in HG in comparison to the rest of the land use systems in all the soil depths.

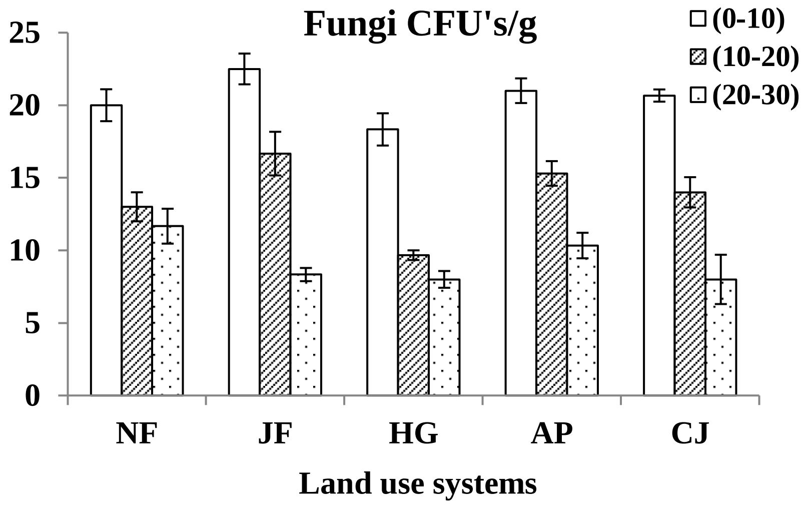


Figure 4.18 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on soil fungi population.

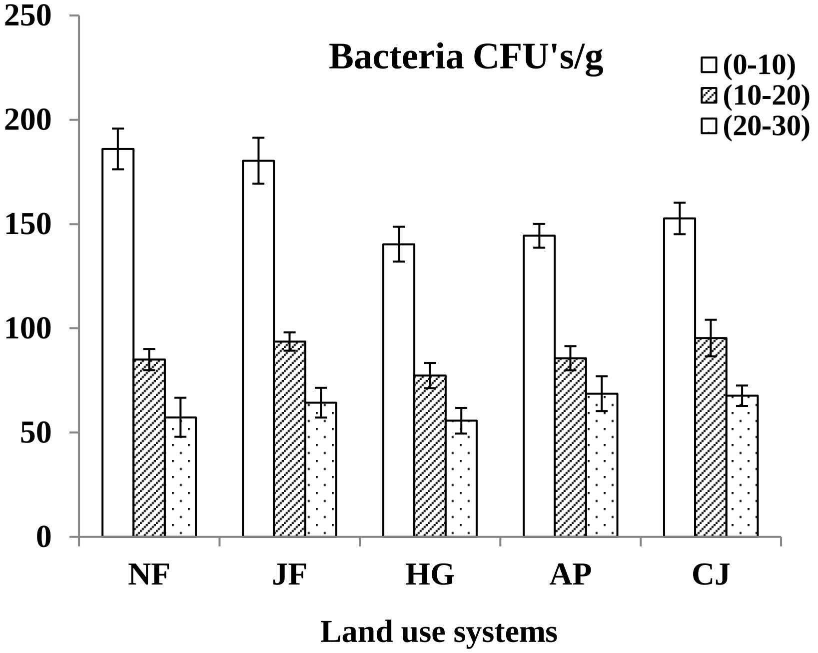


Figure 4.19 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on soil bacterial population.

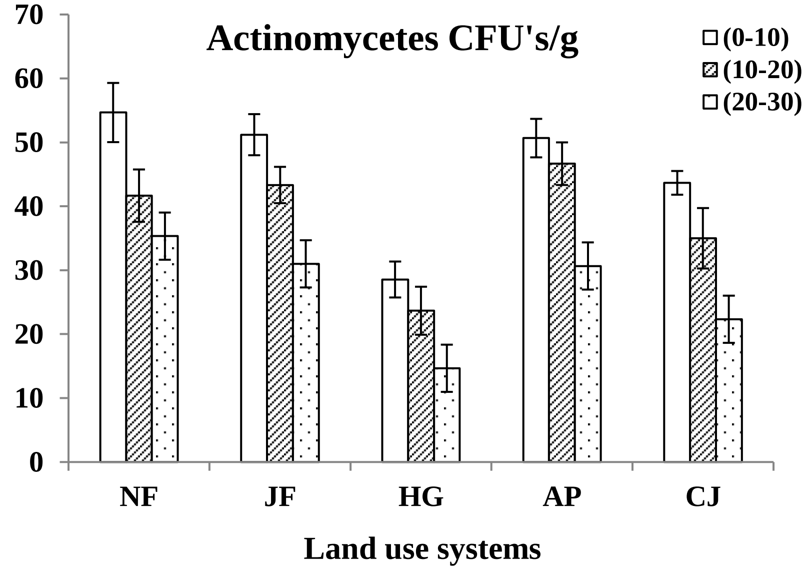


Figure 4.20 Effects of land use systems (Natural forest–NAF, *Jhum* fallow–JF, Home garden–HG, *Acacia pennata* plantation–AP & Current *Jhum*–CJ) and soil depths (0-10, 10-20 & 20-30 cm) on soil actinomycetes population.

4.4. Correlation between soil physical and bio-chemical parameters at various soil depths (0-10 cm, 10-20 cm, 20-30 cm and 0-30 cm)

Pearsons correlation test revealed significant correlations between various soil parameters at different soil depths (0-10, 10-20 and 20-30 cm) as shown in Table 4.6, 4.7 and 4.8. At the surface soil layer (0-10 cm), SOC showed strong positive correlation ($p<0.01$) with TN and strong negative correlation with silt content. BD was negatively correlated with SOC ($p<0.05$) and also negatively correlated with TN ($p<0.01$). Macro-aggregate was observed to have strong positive correlation with MWD as well as GMD, while meso-aggregates had strong negative correlation with MWD and GMD ($p<0.01$). Strong negative correlations ($p<0.01$) between macro- and meso-aggregate could also be reported (Table 4.6). GMD was observed to be positively correlated with MWD ($p<0.01$).

At the sub-surface soil layer (10-20 cm), macro-aggregate was found to have strong negative correlation with silt content ($p<0.01$) while meso-aggregate was found to be negatively correlated with sand and positively correlated with silt ($p<0.05$). MWD was observed to be negatively correlated with silt content ($p<0.01$) while sand concentration was reported to share a positive correlation with MWD. GMD was also observed to be negatively correlated with silt content ($p<0.05$).

Macro-aggregate was observed to be have strong negative correlation with meso-aggregate concentration ($p<0.01$) while it was found to share a strong positive correlation with MWD and GMD ($p<0.01$). Strong negative correlation of meso-aggregate with MWD and GMD could also be observed (Table 4.7). MWD was also observed to be positively correlated with GMD.

Fungi population was found to be negatively correlated with meso-aggregates but positively correlated with GMD. In addition, actinomycetes was also found to have a strong positive correlation with C/N ($p<0.01$). In the deeper soil depth (20-30 cm), sand was observed to have a negative correlation with silt and meso-aggregate ($p<0.05$) while also sharing a strong negative correlation with bacteria ($p<0.01$). Strong negative correlations ($p<0.01$) of silt with macro-aggregate, MWD and GMD were also observed (Table 4.8). Bacteria was positively correlated with silt ($p<0.05$). Micro-aggregate was negatively correlated with macro-aggregate and GMD ($p<0.05$). Srong positive correlations of MWD and GMD with macro-aggregates were also observed. Meso-aggregate was negatively correlated with MWD but positively correlated with bacteria. Micro-aggregate had negative correlation with GMD. Similar to the other depths, GMD and MWD were observed to have strong positive correlation with each other.

A correlation analysis was carried out for the total soil depth of 0-30 cm by taking average values of the various soil parameters from the three soil depths where significant correlations were observed between different soil parameters. Significant positive correlations of SOC with OM ($p<0.01$) and positive correlation with macro-aggregate and MWD ($p<0.05$) were observed. BD was negatively correlated with OM ($p<0.01$) and negatively correlated with SOC ($p<0.05$). Meso-aggregate was found to be negatively correlated with SOC and GMD and shared strong negative correlations with macro-aggregate and MWD (Table 4.9). Strong positive correlations of macro-aggregate with MWD and GMD were observed ($p<0.01$). In addition, macro-aggregate was positively correlated with fungi ($p<0.05$). Fungi also showed strong positive correlations with MWD and GMD ($p<0.01$). MWD was observed to be positively correlated with GMD ($p<0.01$).

Table 4.6 Correlation coefficients (R) between sand, silt, clay, bulk density (BD), moisture content (MC), Total organic carbon (TOC), total nitrogen (TN), carbon/nitrogen (C/N), macro-aggregates (Macro-), meso-aggregates (Meso-), micro-aggregates (Micro-), mean weight diameter (MWD), geometric mean diameter (GMD), Fungi, Bacteria (Bact) and Actinomycetes (Acti) at 0-10 cm soil layer.

(0-10 cm)	Sand	Silt	Clay	BD	MC	SOC	TN	C/N	Macro-	Meso-	Micro-	MWD	GMD	Fungi	Bact	Acti
Sand	1															
Silt	-0.393^{NS}	1														
Clay	-0.854^{NS}	-0.142^{NS}	1													
BD	-0.604^{NS}	0.826^{NS}	0.183^{NS}	1												
MC	0.583^{NS}	-0.282^{NS}	-0.468^{NS}	-0.486^{NS}	1											
SOC	0.456^{NS}	**-0.962****	0.053^{NS}	**-0.938***	0.307^{NS}	1										
TN	0.428^{NS}	-0.849^{NS}	0.020^{NS}	**-0.972****	0.301^{NS}	**0.960****	1									
C/N	0.383^{NS}	-0.877^{NS}	0.084^{NS}	-0.511^{NS}	0.197^{NS}	0.726^{NS}	0.505^{NS}	1								
Macro-	0.834^{NS}	-0.570^{NS}	-0.576^{NS}	-0.480^{NS}	0.627^{NS}	0.483^{NS}	0.316^{NS}	0.729^{NS}	1							
Meso-	-0.785^{NS}	0.636^{NS}	0.486^{NS}	0.512^{NS}	-0.641^{NS}	-0.538^{NS}	-0.361^{NS}	-0.783^{NS}	**-0.993****	1						
Micro-	-0.390^{NS}	0.371^{NS}	0.210^{NS}	0.034^{NS}	0.294^{NS}	-0.230^{NS}	-0.026^{NS}	-0.675^{NS}	-0.559^{NS}	0.536^{NS}	1					
MWD	0.855^{NS}	-0.496^{NS}	-0.640^{NS}	-0.442^{NS}	0.662^{NS}	0.418^{NS}	0.262^{NS}	0.659^{NS}	**0.995****	**-0.981****	-0.518^{NS}	1				
GMD	0.810^{NS}	-0.521^{NS}	-0.578^{NS}	-0.382^{NS}	0.497^{NS}	0.415^{NS}	0.228^{NS}	0.738^{NS}	**0.985****	**-0.970****	-0.682^{NS}	0.978**	1			
Fungi	-0.585^{NS}	-0.182^{NS}	0.733^{NS}	-0.249^{NS}	-0.277^{NS}	0.290^{NS}	0.432^{NS}	-0.190^{NS}	-0.672^{NS}	0.604^{NS}	0.555^{NS}	-0.717^{NS}	-0.741^{NS}	1		
Bact	0.510^{NS}	**-0.889***	-0.046^{NS}	-0.760^{NS}	0.658^{NS}	0.823^{NS}	0.693^{NS}	0.828^{NS}	0.757^{NS}	-0.821^{NS}	-0.213^{NS}	0.713^{NS}	0.671^{NS}	-0.053^{NS}	1	
Acti	-0.325^{NS}	-0.409^{NS}	0.581^{NS}	-0.144^{NS}	0.411^{NS}	0.275^{NS}	0.169^{NS}	0.394^{NS}	0.107^{NS}	-0.210^{NS}	0.299^{NS}	0.070^{NS}	0.013^{NS}	0.314^{NS}	0.586^{NS}	1

***Correlation is significant at 0.05 level;** ** **Correlation is significant at 0.01 level**

Table 4.7 Correlation coefficients (R) between Sand, silt, clay, bulk density (BD), moisture content (MC), Total organic carbon (TOC), Total nitrogen (TN), Carbon/Nitrogen (C/N), Macro-aggregates (Macro-), Meso-aggregates (Meso-), Micro-aggregates (Micro-), Mean weight diameter (MWD), Geometric mean diameter (GMD), Fungi, Bacteria (Bact) and Actinomycetes (Acti) at 10-20 cm soil layer.

(10-20 cm)	Sand	Silt	Clay	BD	MC	SOC	TN	C/N	Macro-	Meso-	Micro-	MWD	GMD	Fungi	Bact	Acti
Sand	1	**-0.933***	0.028^{NS}	-0.857^{NS}	0.060^{NS}	**0.955***	0.769^{NS}	0.695^{NS}	**0.936***	**-0.917***	-0.349^{NS}	**0.908***	0.820^{NS}	0.651^{NS}	-0.058^{NS}	0.614^{NS}
Silt		1	-0.387^{NS}	0.781^{NS}	-0.034^{NS}	-0.826^{NS}	-0.850^{NS}	-0.476^{NS}	**-0.975****	**0.937***	0.571^{NS}	**-0.966****	-0.921*	-0.704^{NS}	-0.245^{NS}	-0.340^{NS}
Clay			1	0.027^{NS}	-0.061^{NS}	-0.151^{NS}	0.390^{NS}	-0.456^{NS}	0.309^{NS}	-0.253^{NS}	-0.691^{NS}	0.356^{NS}	0.458^{NS}	0.286^{NS}	0.829^{NS}	-0.629^{NS}
BD				1	-0.374^{NS}	-0.795^{NS}	-0.788^{NS}	-0.367^{NS}	-0.690^{NS}	0.606^{NS}	-0.051^{NS}	-0.643^{NS}	-0.499^{NS}	-0.180^{NS}	0.314^{NS}	-0.330^{NS}
MC					1	-0.150^{NS}	-0.136^{NS}	-0.062^{NS}	-0.148^{NS}	0.153^{NS}	0.555^{NS}	-0.197^{NS}	-0.290^{NS}	-0.411^{NS}	-0.055^{NS}	-0.147^{NS}
SOC						1	0.751^{NS}	0.727^{NS}	0.875^{NS}	-0.852^{NS}	-0.295^{NS}	0.849^{NS}	0.754^{NS}	0.615^{NS}	-0.257^{NS}	0.713^{NS}
TN							1	0.120^{NS}	0.807^{NS}	-0.688^{NS}	-0.426^{NS}	0.802^{NS}	0.740^{NS}	0.389^{NS}	-0.015^{NS}	0.075^{NS}
C/N								1	0.590^{NS}	-0.693^{NS}	-0.177^{NS}	0.563^{NS}	0.510^{NS}	0.677^{NS}	-0.147^{NS}	**0.969****
Macro-									1	**-0.984****	-0.653^{NS}	**0.997****	**0.968****	0.819^{NS}	0.232^{NS}	0.472^{NS}
Meso-										1	0.673^{NS}	**-0.982****	**-0.965****	-0.891*	-0.281^{NS}	-0.564^{NS}
Micro-											1	-0.705^{NS}	-0.822^{NS}	-0.843^{NS}	-0.743^{NS}	-0.035^{NS}
MWD												1	**0.983****	0.843^{NS}	0.282^{NS}	0.440^{NS}
GMD													1	**0.900***	0.425^{NS}	0.372^{NS}
Fungi														1	0.495^{NS}	0.545^{NS}
Bact															1	-0.356^{NS}
Acti																1

* **Correlation is significant at 0.05 level;** ** **Correlation is significant at 0.01 level**

Table 4.8 Correlation coefficients (R) between Sand, silt, clay, bulk density (BD), moisture content (MC), Total organic carbon (TOC), Total nitrogen (TN), Carbon/Nitrogen (C/N), Macro-aggregates (Macro-), Meso-aggregates (Meso-), Micro-aggregates (Micro-), Mean weight diameter (MWD), Geometric mean diameter (GMD), Fungi, Bacteria (Bact) and Actinomycetes (Acti) at 20-30 cm soil layer.

(20-30 cm)	Sand	Silt	Clay	BD	MC	SOC	TN	C/N	Macro-	Meso-	Micro-	MWD	GMD	Fungi	Bact	Acti
Sand	1															
Silt	**-0.950***	1														
Clay	0.357^{NS}	-0.630^{NS}	1													
BD	-0.841^{NS}	0.730^{NS}	-0.093^{NS}	1												
MC	0.024^{NS}	-0.066^{NS}	0.137^{NS}	-0.391^{NS}	1											
SOC	0.485^{NS}	-0.316^{NS}	-0.263^{NS}	-0.146^{NS}	-0.663^{NS}	1										
TN	0.370^{NS}	-0.333^{NS}	0.077^{NS}	0.073^{NS}	**-0.914***	0.810^{NS}	1									
C/N	0.544^{NS}	-0.363^{NS}	-0.267^{NS}	-0.628^{NS}	0.425^{NS}	0.389^{NS}	-0.179^{NS}	1								
Macro-	**0.923***	**-0.976****	0.625^{NS}	-0.610^{NS}	-0.038^{NS}	0.443^{NS}	0.434^{NS}	0.386^{NS}	1							
Meso-	**-0.898***	0.938*	-0.574^{NS}	0.866^{NS}	-0.343^{NS}	-0.053^{NS}	-0.035^{NS}	-0.402^{NS}	-0.848^{NS}	1						
Micro-	-0.758^{NS}	0.811^{NS}	-0.543^{NS}	0.300^{NS}	0.282^{NS}	-0.636^{NS}	-0.619^{NS}	-0.325^{NS}	**-0.918***	0.570^{NS}	1					
MWD	**0.953***	**-0.988****	0.586^{NS}	-0.700^{NS}	0.058^{NS}	0.401^{NS}	0.350^{NS}	0.458^{NS}	**0.992****	**-0.898***	-0.870^{NS}	1				
GMD	**0.911***	**-0.963****	0.618^{NS}	-0.573^{NS}	-0.084^{NS}	0.479^{NS}	0.475^{NS}	0.372^{NS}	**0.998****	-0.817^{NS}	**-0.938***	**0.984****	1			
Fungi	0.642^{NS}	-0.427^{NS}	-0.320^{NS}	-0.816^{NS}	-0.050^{NS}	0.374^{NS}	0.217^{NS}	0.412^{NS}	0.310^{NS}	-0.517^{NS}	-0.077^{NS}	0.383^{NS}	0.289^{NS}	1		
Bact	**-0.969****	**0.901***	-0.286^{NS}	**0.947***	-0.207^{NS}	-0.322^{NS}	-0.171^{NS}	-0.585^{NS}	-0.827^{NS}	**0.936***	0.582^{NS}	-0.885*	-0.802^{NS}	-0.732^{NS}	1	
Acti	0.846^{NS}	-0.864^{NS}	0.483^{NS}	-0.756^{NS}	0.454^{NS}	0.180^{NS}	-0.055^{NS}	0.696^{NS}	0.853^{NS}	-0.877^{NS}	-0.688^{NS}	0.899*	0.831^{NS}	0.298^{NS}	-0.848^{NS}	1

* **Correlation is significant at 0.05 level;** ** **Correlation is significant at 0.01 level**

Table 4.9 Correlation coefficients (R) between soil bulk density (BD), Total organic carbon (TOC), Total nitrogen (TN), Carbon/Nitrogen (C/N), Soil organic matter (SOM), Macro-aggregates (Macro-), Meso-aggregates (Meso-), Micro-aggregates (Micro-), Mean weight diameter (MWD), Geometric mean diameter (GMD), Fungi, Bacteria (Bact) and Actinomycetes (Actino-) upto 0-30 cm soil layer.

(0-30 cm)	BD	SOC	TN	C/N	SOM	Macro-	Meso-	Micro-	MWD	GMD	Fungi	Bact	Actino
BD	1	**-0.917***	-0.846^{NS}	-0.525^{NS}	**-0.987****	-0.695^{NS}	0.780^{NS}	0.084^{NS}	-0.705^{NS}	-0.538^{NS}	-0.442^{NS}	-0.472^{NS}	-0.281^{NS}
SOC		1	0.872^{NS}	0.696^{NS}	**0.963****	**0.911***	**-0.933***	-0.456^{NS}	**0.916***	0.815^{NS}	0.763^{NS}	0.544^{NS}	0.455^{NS}
TN			1	0.263^{NS}	0.852^{NS}	0.715^{NS}	-0.771^{NS}	-0.382^{NS}	0.741^{NS}	0.657^{NS}	0.572^{NS}	0.151^{NS}	0.114^{NS}
C/N				1	0.617^{NS}	0.771^{NS}	-0.707^{NS}	-0.420^{NS}	0.744^{NS}	0.687^{NS}	0.742^{NS}	0.874^{NS}	0.801^{NS}
SOM					1	0.800^{NS}	-0.868^{NS}	-0.228^{NS}	0.807^{NS}	0.661^{NS}	0.565^{NS}	0.503^{NS}	0.333^{NS}
Macro-						1	**-0.980****	-0.745^{NS}	**0.999****	**0.975****	**0.912***	0.473^{NS}	0.449^{NS}
Meso-							1	0.652^{NS}	**-0.984****	**-0.933***	-0.816^{NS}	-0.396^{NS}	-0.317^{NS}
Micro-								1	-0.747^{NS}	-0.874^{NS}	-0.863^{NS}	-0.062^{NS}	-0.241^{NS}
MWD									1	**0.975****	**0.906***	0.443^{NS}	0.420^{NS}
GMD										1	**0.944***	0.347^{NS}	0.387^{NS}
Fungi											1	0.515^{NS}	0.625^{NS}
Bact												1	**0.938***
Actino													1

* **Correlation is significant at 0.05 level;** ** **Correlation is significant at 0.01 level**

CHAPTER 5

DISCUSSION

5.1. Effects of land use systems and soil depths on soil physical properties

Land use systems and soil depths profoundly affected soil physical properties as result of variations in the plant species, their ages and ecological characteristics. Increase in soil BD with increasing soil depth in different land use systems can be associated with decreasing SOM as well as the particle size distribution and the overlying weight (Anteneh *et al.,* 2013). The lower BD in the surface soil layer is widely reported by the higher buildup and accumulation of SOM (Biswas *et al.,* 2012; Lalnunzira and Tripathi, 2018). In NAF and JF profused root growth and increased root density in soils may have significantly decreased soil BD compared to cultivated soils (AP, HG and CJ). Whereas, the conversion of native forests to cultivated lands or plantations leads to loss in SOM and thus the higher BD in cultivated lands (AP, HG and CJ) than in native forests and fallow lands. Our findings are in agreement with other studies which indicated higher BD in cultivated soils than in native forests and restored/rehabilitated lands (Yitbarek *et al.,* 2013; Takele *et al.,* 2014; Abad *et al.,* 2014). In addition, Lelisa and Abebaw (2016) also reported higher BD at 0-10, 10-20 and 20-30 cm soil depths in grasslands than in restored lands. Our values of BD are in concurrence within the value range of 1.1 to 1.4 g cm^{3} as given by Gupta (2004) for mineral soils, while BD of soil texture falling in the class of clay and silt loam usually range from 1.0 to 1.6 g cm^{-3} according to Brady (1990).

Highest SMC in JF followed by NAF in comparison to other land uses maybe attributable to the presence of continuous vegetation cover in these land uses since lack of vegetation during rains lead to decreased infiltration rates due to losses through runoff which influences the conservation of moisture in the soil (Sadeghi *et al.,*2007). SOM is considered as one of the most important dependent factors for SMC in the soil owing to its hydrophilic character and its ability to improve soil structure (Haynes and Naidu, 1998; Kimble *et al.,* 2007) which favours SMC conservation. However, no prominent trend of SMC in the soil layer could be observed even though it was significantly affected by varying soil depths which maybe the result of the varying vegetation cover and management practices across the various land use systems. In addition, increased SMC in land uses with increased vegetation cover as in the case of NF and JF has also been reported by Duma (2000).

In case of particle size distribution, clay was observed highest in the sub-surface depth (20-30 cm) of CJ which is in accordance with the findings in the study by Tufa *et al.* (2019) from cultivated lands. Increase in clay content with increasing soil depth in all the land use systems maybe attributable to the profuse growth and development of root channels enhancing the downward movement of clay particles and its subsequent accumulation at the sub-surface layer through movement of clay particles (Ketema and Yimer, 2014). Our findings on increasing clay content with increasing soil depth are in accordance with the findings from Awdenegest *et al.* (2012). Chemada *et al.* (2017) also asserted that due to continuous and longer period of cultivation, clay content increased from surface layer to sub-surface and deeper layers in cultivated lands. Soil textural class in NAF, JF, AP and CJ were well within the class of sandy clay loam, clay loam and loam indicating similar attributes which may be related to the fact that particles size distribution takes time to be altered (Brady, 1990; Osman, 2012). Higher content of sand content was also observed in the surface layer than the sub-surface layers. which coincides with the study of Gebrelibanos and Assen (2013) where higher sand concentration was observed in the surface layer of cultivated lands which is chiefly due to the fact that sand concentration usually remains higher in the surface soil layer (Yimer *et al.*, 2008). Variation in soil particle size distribution indicates the influence of land use types on soil characteristics induced by variations in management practices operated on the land (Abbasi *et al.*, 2007). The dominance of brown and dark yellowish brown soil colour in the surface layer (0-10 cm) of all the land use systems may be well related to the presence of moderate amount of organic matter and considerable presence of iron oxides. The sub-surface layers of the soil in almost all the land use systems exhibited yellow to light yellowish and brownish yellow colour which may be well attributed to the high concentration of iron and aluminium oxides which is a common characteristic of acidic soils in Northeast India.

5.2. Changes in soil pH, P_{avail}, P_{total}, exchangeable and total nutrients, and CEC as influenced by different land use systems and soil depths

Soil pH was highly acidic at the sub-surface soil depths (10-20 and 20-30 cm) in all the land use systems. Higher soil pH in the surface layer of CJ is well attributed to the release of potash as a result of the traditional slash and burn technique in the CJ land use system. Burning enhances the release of nutrients in the soil and thus increasing the soil pH (Moraes *et al.,* 1996). The higher values of pH in the cultivated lands (HG, AP and CJ) may also be a result of the conversion of natural forest into cultivation which leads to increase in pH at the surface as well as the sub-surface soil layers (Lumbanraja *et al.,* 1998). The present values of pH are in accordance with other findings from the study area which indicated strong acidic nature of reaction in these soils (Gorgan *et al.,* 2012; Tripathi *et al.,* 2017; Lungmuana *et al.,* 2017). P_{avail} was observed to be highest in HG at the surface layer and maybe well related to the addition of organic manure in the form of animal waste by the land owners. Whereas, the higher P_{avail} in JF and NAF may be attributed to net P mineralization as a result of the continuous addition and occurrence of litter in these land uses. Sarkar *et al.* (2010) reported that P_{avail} of soil increased with the addition and presence of litter in on the soil surface. While the decrease in soil P_{avail} with increasing soil depths may also be a result of decreasing SOM content that contributes significantly to the P pool in the soils of the study area. In addition, SOM influences the P_{avail} through anion replacement of H_2PO from adsorption sites and the formation of organophosphate complexes which are readily taken up by plants as reported in different studies (Abebe and Endalkachew, 2012; Nega and Heluf, 2013; Yihenew and Getachew, 2013). Our values of P_{avail} falls within the range of low to medium range reported for various land use systems and soil depths (Cottenie, 1980). In addition, studies by Murphy (1968), Tekalign *et al.* (2002) and Abebe and Endalkachew (2012) reported that P_{avail} in soils including inceptisols and vertisols are typically low and correlates it to various processes such as erosion, fixation and abundant crop harvests especially in tropical regions.

Present findings on P_{total} content in bulk soils across different land use system ranged from 340.73 to 1480.6 mg kg^{-1} in the surface soil (0-10 cm) which is in parallel with the findings of Zhang *et al.* (2020) from the sub-tropical regions of Australia. The highest P_{total} concentration in native forest and lowest in the cultivated land indicated that the conversion of natural forest to other land uses significantly decreases the P_{total} content in the soil, which reflected a consistent reduction in P_{total} content in soils under cropping in relation to native forest because of the disruption in nutrient cycling (Tripathi and Singh, 1994, Tripathi et al. 2008). Chacon and Dezzeo (2004) stressed that conversion of native forests to cultivated lands leads to decrease in P concentration in the soil while increasing fractions of non-available P forms. The decrease in P_{total} with increasing soil depths may be well related to the accelerated soil erosion induced by land use change which significantly reduces the SOM by a magnitude of more than half or more (Zheng *et al.,* 2005) in the soil layer, which is the major source of organic substrate for nutrient concentration and availability such as P (Groppo *et al.,* 2015; Pimentel *et al.,* 1995).

As per ANOVA, soil exchangeable cations (Na, Mg, K, Ca and Mn) were significantly affected by different land use systems and soil depths. The interaction of both factors was also significant. Similar findings have been reported from different parts of the world (Selassie and Ayanna, 2013; Aytenew and Kibret, 2016; Ufot *et al.,* 2016). The higher exchangeable Mg, K and Mn contents in JF maybe related to the rapid emergence of herbaceous vegetation and closed canopy that protected the soil from the direct impact of rainfall and minimized the loss of soil nutrients through runoff and erosion. In addition, Ramakrishnan and Kushwaha (2001) indicated that longer fallow periods (>20 years) contained more soil available nutrients leading to better crop productivity in comparison to younger fallow lands. The period of fallow played a vital role in promotion of soil nutrients, and therefore, longer period of fallow favours nutrient conservation. Wapongnungsang (2017) also indicated that a fallow period of more than 10 years lead to higher conservation of nutrients in the soil than shorted fallow during cultivation throughout the cropping season. In comparison to the other land use systems, the lower values of exchangeable Na, Mg, K, Ca and Mn in cultivated soils of AP and CJ may also be attributed to the nutrient uptake by plants for their growth and development. Our lower values of exchangeable Na, K, Ca and Mg in cultivated lands (AP and CJ) with higher concentrations in the surface layer (0-10 cm) are in conformity with the study carried out by Yimer *et al.* (2008) in Ethiopia where the values of the elements were observed to be lower in cultivated lands than in grasslands or native forests.

Incase of total nutrients concentration, the elevated levels of total Mg, K, Ca and Mn concentrations in HG in comparison to the other land uses may be attributable to the application of animal and household waste like dung manure and ash since these amendments are important source of Ca, P, K and other nutrients (Voundi *et al.,* 1998). The greatest concentration of Mg, K, Ca and Mn in the surface layer (0-10 cm) than the sub-surface layers (10-20 and 20-30 cm) may be related to the higher availability of plant and animal residues in the surface layer than beneath. Furthermore, it can also be attributed to the vegetation that pumps the bases from sub-surface layer to surface layers (Yimer *et al.,* 2008). Kiflu and Beyene (2013) reported higher nutrient content in the surface soil layer as a result of organic residue accumulation and related biological activity which is in accord with our findings.

Highest CEC in the surface layer of JF may be attributed to the highest clay concentration as well as soil organic carbon content in the surface layer of JF soil in comparison to other land use systems, while the highest CEC at the 10-20 and 20-30 cm soil layer of HG may be related to the higher silt concentration in those layers which readily trap soil exchangeable cations coupled with the addition of continuous organic amendments. Silt tends to trap cations more readily as a result of its negatively charged sites which enable them to adsorb and hold cations. On the contrary, lower values of CEC in NAF could also be due to the higher sand content which may have led to increased leaching. Dai yunan *et al.* (2018) found strong negative correlation of CEC with sand content. The higher values of CEC in JF with higher SOC content is in accordance with other studies which showed a positive correlation of CEC with SOC in uncultivated lands (Tegene, 2000; Eshetu *et al.,* 2004). Our findings indicated that changes in CEC is susceptible to management practices such as use of fertilizers, organic amendments, soil particle size distribution and also depends on the SOC content in the soil. Our values of CEC ranges from 3.902 to 9.53 c mol kg^{-1} across the various land use systems and soil depths and falls under the classification of weak fertility as given by Wu Q (2011), where soils with CEC of <10 c mol kg^{-1}, 10-20 c mol kg^{-1} and

>20 c mol kg^{-1} were grouped as weak fertility, moderate fertility and high fertility soils, respectively.

5.3. Changes in TOC, TN, C/N, SOM, and stocks of C and N as influenced by different land use systems and soil depths

Significant differences in TOC and TN could be observed across the different land use systems and soil depths. A decreasing trend of TOC and TN could be observed with increasing soil depths in all of the land uses. The higher TOC and TN in the surface layer (0-10 cm) may be well attributed to the higher organic matter content (Poorter, 2016). Several other studies have also reported similar reports of decreasing TOC with increasing soil depths (Eunice *et al.,* 2020; Moges, 2013). In addition, positive correlation of organic matter with SOC was observed in this study. Maximum TOC and TN in NAF and JF may be a result of elevated organic matter inputs from both above and below ground biomass in these soils (Materechera, 2010; Murovhi, 2012). Higher rate and amount of organic matter in the soils of NAF and JF can be expected owing to its vegetation and profuse litter. Several studies from sub-tropical regions of Northeast India have indicated the role of root and leaf litter accumulation in maintaining soil carbon and nitrogen (Ovung *et al.,* 2021; Wapongnungsang *et al.,* 2017, Lalnunzira and Tripathi, 2018). On the contrary, lesser concentration of SOC and TN in the cultivated land uses may be attributed to the effect of land use management approaches, since these approaches play a vital role in nutrient conservation (Ovung *et al.,* 2021). The consistent removal of the surface vegetation through weeding and clearing of fallen leaf litter leads to decreased microbial activity. These management approaches generally removes soil cover and decreases organic matter inputs leaving the soil bare and vulnerable to erosion by soil and water (Foley *et al.,* 2005; Giller *et al.,* 1997; Mills and Fey, 2004). Another important factor contributing to loss of TN in cultivated soils such as HG can be related to the uptake of N by crops since N is one of the most rapidly absorbed nutrients (Salcedo, 2008). Several studies have reported rapid loss of soil carbon, nitrogen and alteration of microbial community as a result of conversion of native forests to cultivated lands including plantations (Murty *et al.,* 2002; Saggar *et al.,* 2001, Xiangmin *et al.,* 2014). However, highest concentrations of TN in the sub-surface layer (20-30 cm) could be observed in AP in comparison to the other land uses (JF, HG, CJ and NAF). This may be due to the rhizospheric nitrogen fixation in the soil at the sub-surface layer, attributable to the leguminous nature of the planted species (*Acacia pennata*) in AP. These N fixing species exhibits a competitive characteristic and are able to enhance soil fertility particularly N through the transfer of N to the soil systems (Barea *et al.,* 1992; 1996; Rode, 1995; Geesing *et al.,* 2000).

C:N ratio followed a decreasing trend with increasing soil depths in all the land use systems and indicated decreased C input in deeper depths through leaf litter. A recent study from Ethiopia by Tufa *et al.* (2019) also reported similar findings of decreasing C:N values with increasing soil depths in various ecosystems. The lowest C:N ratio in HG can be related to the management practices in HG as well as the increased microbial activities leading to evolution of CO_2 and its higher subsequent loss in the surface layer than the sub-surface layers. The higher values of C: N in the soils of JF and NAF signifies the optimal incidence of biological processes. Present finding on C:N is in accordance with the reports of Gebrelibanos and Assen (2013) where highest C:N was recorded in forest lands in comparison to cultivated lands. In addition, higher C:N in native forests than adjacent plantation, grazing land and cultivated lands has also been reported by Selassie and Ayanna (2013). Furthermore, they also suggested that the optimal range of C:N for adequate supply of nitrogen for supporting microbial activities is from 10:1 to 12:1. Our present values of C: N in all the land uses except in JF were also well within the given optimal range.

Higher SOM accumulation in NAF and JF compared to the other land uses indicates that the natural ecosystems with diverse species composition along with profuse root systems tightly bounds and efficiently recycles soil organic matter within the system (Singh *et al.,* 2015; Lalnunzira and Tripathi, 2018). However, in the other ecosystems SOM content is lost because of forest degradation and management practices operating within the ecosystems (Tripathi *et al.,* 1999; Singha and Tripathi, 2017). The lower values of SOM in HG can be well related to continuous cultivation without fallow leading to removal of crop residues and increased organic matter oxidation rates which reduces organic matter sources as reported by Nega and Heluf (2009). The pattern of SOM accumulation was considerably higher in the surface layer in all the land use systems which decreased with increasing soil depths. The higher

SOM build up in the surface soil layer and the decreasing trend with increasing soil depths can be linked to the incessant accumulation of undecomposed and partially decomposed plant and animal residues in the surface soil layer (Singh *et al.,* 2015; Singha and Tripathi, 2017; Lalnunzira and Tripathi, 2018). Generally, clearing of forest areas and its subsequent conversion to other land uses like agricultural croplands concedes a considerable loss in SOM (Weldeamlak and Stroosnijder, 2003; Genxu *et al.,* 2004). Ashagrie *et al.* (2007) also indicated that cultivation of land for agricultural produce leads to rapid decomposition of SOM as a result of alterations in factors such as temperature, soil water content and aeration. SOM plays a major role in the productivity of soil and is dependent on land management practices and the intensity and quality of organic matter inputs from the particular land use system (Gaiser and Stahr, 2013).

Soil C and N stock were significantly higher in NAF in comparison to other land uses which indicated that highly diverse species composition along with profuse root systems were tightly coupled with efficient recycling of soil soil C and N in the system (Singha and Tripathi, 2017; Lalnunzira and Tripathi, 2018, Manpoong *et al.,* 2020). Highest stocks of C and N in the soil were reported from the surface layer (0-10 cm) in all the land uses. Significant differences in C and N stock was observed across different land uses and soil depths indicating the role of vegetation and management practices in C and N sequestration in the soil. Maximum C and N stocks are quantified in the 0-10 cm soil layer as a result of the higher accumulation of organic matter (Lalnunzira and Tripathi, 2018). Conversion of native forests to agricultural land uses is the major cause for loss of organic matter content leading to decrease in C and N stocks in the soil (Maia *et al.,* 2013) and thus, the lower soil C and N stock in the cultivated lands such as HG. Higher C and N stocks in NAF in comparison to other land uses can be correlated to the minimum disturbance in the ecosystem which is important for conservation and maintenance of C stock as well as microbial carbon (Maia *et al.,* 2019).

Total soil C and N stock (0-30 cm) were observed to be conserved highest in NAF and JF which may be well linked to the higher return of organic matter in the soil due to the presence of continuous vegetation cover and its root distribution. On the contrary, least soil C and N stock in HG is attributable to the lower organic matter input and the rapid and constant utilization of N and other nutrients by crops coupled with prolonged cultivation which enhances soil mineralization (Lal, 2018). Magnitude of loss in C stock in comparison to NAF was observed to be least in JF among the other land uses (HG, AP and CJ) which correlates to the longer recovery period in this land use leading to larger accumulation of litter inputs which gradually sequesters maximum soil C.

5.4. Impacts of land use systems and soil depths on soil aggregate stability

Soil aggregate stability indicated by MWD and GMD varied significantly across the land uses and soil depths as a result of variations in the vegetation composition and land management practices (Manpoong, 2019). Number of studies have emphasized the impact of land use change on soil aggregates fractions and its stability as a result of differences in land use type and management practices (Somasundaram *et al.,* 2017; Tisdall and Oades, 2012; Dalal *et al.,* 2011) and nutrient amendments (Sarker *et al.,* 2018; Tripathi *et al.,* 2008, Tripathi *et al.,* 2012) in various ecosystems. Among all the land uses, JF and NAF showed significantly higher aggregate stability.

Dominance of soil macro-aggregates in the surface layer (0-10 cm) leading to higher aggregate stability in the surface as denoted by higher MWD and GMD in the surface layer is in accord with other findings (Kumri *et al.,* 2020; Tripathi *et al.,* 2012; Singh *et al.,* 2017) from a dry tropical forest of India. Increased water stable macro-aggregates could be observed in the surface layer (0-10 cm) in all the land uses which decreased with increasing soil depths. This could be linked to the higher concentration of SOM in the surface layer than the lower soil depths which acts as cementing agents that facilitates in structuring soil macro-aggregates (Mohanty *et al.,* 2012) that protects the soil from physical disturbances and restoring soil properties (Al-Kaisi *et al.,* 2014).

Positive correlations of TOC with soil macro-aggregates and MWD could be observed in this study which indicates that TOC is an important factor responsible for higher aggregate stability. Higher TOC in JF and NAF in comparison to other land uses might have facilitated formation of more water stable macro-aggregates leading to higher stability in these soils. Increased accumulation of organic carbon improves soil aggregate stability as a result of the adherence and binding of mineral particles which ensures protection to the soil structure from slaking and favours water stable aggregate formation (Demenois *et al.,* 2018). In addition, increase in litter and organic matter accumulation enhances aggregate formation (Nath *et al.,* 2018b).

Lower soil aggregate stability in HG, AP and CJ can be attributed to the mechanical and physical disturbances of soil as a result of the cultivation and management patterns associated with the specific land uses which alters the process of macro-aggregate formation and decreases aggregate stability (Chen *et al.*, 2017). Likewise, conversion of native forest to other land uses alters the plant community structure, litter production, organic matter accumulation and decomposition kinetics that together alter soil fauna composition and micro-climate which profoundly affected the pattern of soil aggregate formation (Chen *et al.*, 2017). Thus, the lower fractions of soil macro-aggregates and aggregate stability in CJ, AP and HG can be linked to the management interventions and cultivation patterns resulting due to conversion of NAF to other land uses.

MWD and GMD were observed to be considerably lower in the sub-surface depths (10-20 and 20-30 cm) of HG in comparison to other land uses at those depths. This can be well related to the presence of very high silt particles representing a silty clay loam texture at those depths. A strong negative correlation of silt with MWD, GMD and soil macro-aggregates has also been observed at the sub-surface depths (10-20 and 20-30 cm) in this study. Generally, silty textured soils are more prone to slaking than soils with different textures. Our observation on lower soil aggregate stability in layers dominated by silt particles (silty-clay loam texture) in HG corroborates with the study by Carrizo *et al.* (2015) carried out in the humid sub-tropical regions of Argentina. They presented a lesser stability of soil as a result of the presence of greater fine silt content. A study by Chenu *et al.* (2000) also reported that the disruption of continuously cultivated loam soils is attributable to the slaking forces. Furthermore, soils exhibiting these attributes generally have lesser soil structural stability as well as buffering capacity as asserted by Taboada *et al.* (2008), with very less or inexistence of micro-cracking due to the persistence of a matrix that exhibits no flexibility (expansion and contraction) process and skeletal structure.

Our reports on increasing proportions of meso- and micro-aggregates with increasing depth are in accordance with the findings from other studies (Kumri *et al.*, 2020; Singh *et al.*, 2017). This can be linked to the increasing compaction of soil which increases with increasing soil depths, since soil compactness inhibits the penetration and expansion of roots in the sub-surface soil layers, it decreases the activity of microbes including fungi (Hoorman *et al.,* 2009) which leads to decrease in production of polysaccharides and other organic compounds which act as gluing agents in the formation of macro-aggregates. Macro-aggregates formation is augmented by the accumulation and decomposition of plant litter and animal residues which are the major source of C for microbial activities which in turn provides the binding agents (Golchin *et al.,* 1994; Puget *et al.,* 1995; Jastrow *et al.,* 1996; Six *et al.,* 1999). This can be well linked to the decreasing macro-aggregates with increasing soil depths since organic matter as well as soil microbial biomass decreases with increasing soil depths and thus the higher soil macro-aggregates in the surface layer than the lower depths. Increased proportions of macro-aggregates in the surface soil layer as observed in this study is also reported by similar other studies (Li *et al.,* 2007; Saha *et al.,* 2010).

Higher aggregate stability in JF in comparison to other land uses can be related to the increased concentration of TOC (Kalhoro *et al.,* 2017) and clay particles (Sainju, 2006). Similar positive correlation of TOC with MWD is observed in this study which corroborates with the findings of Kumri *et al.,* (2020). Furthermore, Ge *et al.* (2018) indicated that addition of fresh organic matter enhanced soil aggregation. Franzluebbers *et al.* (2000) reported that increase in the content of clay particles led to increase in soil macro-aggregates in comparison to micro-aggregates which enhanced aggregate stability. Thus, soil under long-term fallow (JF) of >12 years with minimal or no disturbance led to improved soil structure and aggregate stability indicating that the inherent parent material and management practices associated with land uses alters soil aggregate stability.

5.5. Changes in soil aggregate size fractions and aggregate size associated C and N across different land use systems and soil depths

Aggregate size fractions and associated C and N varied siginfiantly in different land uses and soil depths. Aggregate associated C and N were observed to be highest in macro-aggregate in comparison to meso- and micro-aggregates in all the land uses. Several other studes have also reported higher TOC and TN concentrations in macro-aggregates relative to meso- and mciro-aggregates (Six *et al.,* 2000; Fernandez *et al.,* 2010; Sarker *et al.,* 2018b). In addition, according to Tisdall and Oades (1982) hierarchy model of aggregates, the fundamental unit for formation of aggregates of the greater order (e.g., mega- and macro-aggregates) is composed of micro-aggregates adhered together by binding agents derived from organic remains which adds to the organic carbon content in macro-aggregates. Meso- and macro-aggregates are dominated by silt plus clay particles and therefore, SOM may be more stable in these aggregate classes due to the higher surface area of the particles. This leads to decrease in turnover rates of SOM and probably assists in prolonging the mean residence time (MRT) as indicated by O'Brien and Jastrow (2013). Higher concentration of TOC and TN in macro-aggregates and meso-aggregates in NAF and JF in comparison to other land uses can be related to the higher accumulation and retention of soil organic matter in the inter- micro-aggregate spaces within meso- and macro-aggregates. A study by Ayoubi *et al.* (2012) revealed higher associated organic carbon in aggregate size fractions of native forest in relation to cultivated land and degraded forest, linking it to increased availability of organic matter inputs coupled with lesser anthropogenic interventions which enhanced the concentration of associated organic carbon in macro-, meso- and micro-aggregates as well as establishment of macro-aggregates leading to greater aggregate stability. Likewise, in case of cultivated lands such as HG, the least concentration of TOC and TN in the aggregate size fractions can be attributable to the lesser inputs of organic matter and management practices causing mechanical disturbances (Liu *et al.,* 2014).

Management practices causing physical disturbances to the soil profile are known to disrupt and break soil macro-aggregates and induce rate of organic matter mineralization leading to loss of organic carbon from the inter-aggregate spaces (Six *et al.,* 1998). Other studies have also shown the influence of various management practices on SOC and TN content in various aggregate size fractions (Six *et al.,* 2000; Yang *et al.,* 2007; Devine *et al.,* 2014; Somasundaram *et al.,* 2017). Decrease in macro-, meso- and micro-aggregate associated TOC and TN in the other land uses (JF, HG, AP and CJ) in comparison to NAF indicated impact of land use change (conversion of NAF to other land use) on these attributes. Further, it also indicated that land use change leads to redistribution of TOC and TN in the aggregate size fractions among the various land uses.

5.6. Effects of land use systems and soil depths on soil microbial population, microbial biomass carbon and their inter-relations with soil aggregate size fractions

Soil microbial populations (i.e. Fungi, Bacteria and Actinomycetes) differed across the various land uses and soil depths with higher population being recorded at the surface layer (0-10 cm) in all the land uses observed. The decrease in population of fungi, bacteria and actinomycetes with increasing depths may be attributed to the decreasing organic matter which acts as a source of carbon for the microbes for their metabolism (Bhattarai *et al.,* 2015). In addition, optimal environmental conditions like soil aeration and temperature in the surface soil favours the growth of the population and vice versa in the deeper soil depths (Bhattarai *et al.,* 2015).

Higher population of bacteria and fungi could be observed in NAF and JF, respectively, indicating that the persistence of vegetation in these land uses led to increased concentration of carbon sources in the soil through higher organic matter accumulation and thus sustaining a higher microbial population (Ndour, 2008). The lower pH in the soils of NAF and JF may also be responsible for the higher population in these land uses indicated by positive correlation of soil pH with soil microbial population (Ibekwe *et al.,* 2012). However, it has been indicated that the degree of soil aggregation and stability can be closely related to microbial diversity, population and community structure than other properties such as pH and types of organic compounds (Treonis *et al.,* 2010; Yin *et al.,* 2010; Wakelin *et al.,* 2008) which coincides with our findings of higher soil aggregation in JF and NAF and thus the higher microbial population in those land uses. In addition, the lower or negligible disturbance of soil in these systems may protect the microbial habitats by means of conserving soil moisture and maintaining soil temperature swings (Derpsch *et al.,* 1991). The build up of C sources enhances soil aggregation representing a major source of nutrients and energy which fuel the development and functions of microbes. On the contrary, agricultural management practices can exert significant impact on microbial population and activities (Wakelin *et al.,* 2008; Rasche *et al.,* 2006; Ferreira *et al.,* 2000) as in the case of HG land use system. Present findings of highest microbial counts in NAF followed by JF and least in HG, are in consistence with other findings which indicated significant effect of land uses and soil depths on soil microbial population count with highest count at the surface soil layer from native forests followed by grasslands and least in cultivated lands (Wani *et al.,* 2018; Bello *et al.,* 2013; Asadu *et al.,* 2015; Silva *et al.,* 2013).

Higher number of fungi in JF and NAF signifies that the dominance of trees and other vegetation might have enhanced the population of ectomycorrhizal fungi which is present in most species of trees. In addition, the presence of tree and vegetation cover reduced the impact of falling rain on the soil and created favorable micro-climatic conditions for increased growth and multiplications of fungi as indicated by Asadu *et al.* (2015). The least count of fungi in HG in the present study may be attributed to the management practices which caused disturbances to the soil structure since fungi are easily affected by alterations in soil as well as environmental factors (Sui *et al.,* 2012). Our current findings are in accordance with the observations of Bello *et al*., (2013), Asadu *et al.* (2015) and Wani *et al.* (2018) wherein, highest counts of fungi was reported from natural forest soils in both the surface and sub-surface layers. Similarly, our findings on bacteria and actinomycetes corroborates with the studies carried out by Okonkwo (2010) and Kumar *et al.* (2017) where highest count of bacteria and actinomycetes were reported from natural forest in comparison to cultivated lands at the surface and sub-surface layers. It is suggested that incessant cultivation leads to disturbances in soil processes and structure that depletes organic matter in long run and thereby reducing the population of microbial communities.

Significant differences in land use systems and soil depths could also be observed in case of soil microbial biomass carbon (MBC). Increased levels of MBC was observed in NAF and JF in comparison to the other land uses (HG, AP and CJ) which may be well linked to the persistence of vegetation which continuously adds organic matter to the soil enhancing organic carbon that provides the substrate to the microbes in the soil which supports higher microbial activities. The present study also signifies that increased TOC in NAF and JF in the soil lead to increased microbial biomass. A number of studies have also shown a relationship between diverse and rich plant community and soil microbial biomass carbon (Broughton *et al.,* 2000; Spechn *et al.,* 2000; Bardget *et al.,* 1999). Increase in soil carbon enhances the growth and multiplication of soil microbes and results in higher microbial biomass accumulation (Tripathi *et al.,* 2008; Chen *et al.,* 2005).

The lowest value of MBC in the cultivated land (HG) is attributable to the absence of surface vegetation with reduced inputs of organic matter that led to decreased microbial activity (Degens *et al.,* 2000). It has been reported that maximum depletion of organic matter in the soil due to periodic harvest and loss of above-ground crop biomass supported carbon limitation for a limited time frame and results in decreased levels of MBC in the soil (Kushwaha *et al.,* 2001; Singh and Ghoshal, 2014). This can also be related to the decrease in MBC with increasing soil depths since soil organic matter is the fundamental unit for sustenance of soil microbes. Impact of MBC as a result of vegetation cover has also been well established and recognized by several studies (Jin *et al.,* 2010; Xu *et al.,* 2008).

The impact of soil microbial community on soil aggregate stability is well established (Kushwaha *et al.,* 2001; Tripathi *et al.,* 2012). Microbial activities enhance the decomposition of organic matter and production of enzymes and other binding agents such as polysaccharides and glomalin which leads to bind mciro-aggregates to form soil macro-aggregates (Six *et al.,* 2006; Wilson *et al.,* 2009; Tiemann and Grandy, 2015). The results of the present sudy showed strong correlation of fungi with soil macro-aggregates and MWD indicating that the fungal community influences the formation of macro-aggregates and enhances soil aggregate stability in these ecosystems and vice versa. This may be well attributed to hyphal entanglement which is the most prominent characteristic feature of fungi contributing to the formation of soil aggregates especially macro-aggregates (Molope and Page, 1986). However, studies have argued that function of fungal polysaccharide associated binding is highly underestimated (Aspiras *et al.,* 1971; Burns and Davies, 1986; Beare *et al.,* 1997). The findings of the present study are in accordance with several other works which indicated that fungi have a significant role in the formation of soil macro-aggregates (Low and Stuart, 1974; Tisdall and Oades, 1982; Lynch and Bragg, 1985; Beare *et al.,* 1997). In addition, positive correlation of bacteria with soil meso-aggregates has also been observed in this study. This suggests that soil fungal population has more influence in stabilization of soil aggregates relative to bacteria. Studies by Monreal and Kodama (1997) and

Neumann *et al.* (2013) have also shown that higher fractions of bacteria are usually linked with soil micro-aggregates than soil macro-aggregates which corroborates with the present findings. Present findings also suggest that different size classes of aggregates hosts specific habitats of microbes while allowing colonization of various specific microbial communities as supported by other several studies (Davinic *et al.*, 2012; Trivedi *et al.*, 2015). In general, our findings illustrate that management activities affect soil physico-chemcial and biological diversity within aggregates of various size fractions and the subsequent composition of various microbial populations and their associated functions.

CHAPTER 6

SUMMARY AND CONCLUSION

Land use change induced by anthropogenic interventions has a significant effect on soil characteristics and subsequently affects the sustainability of different ecosystems. Clearing of native forests for cultivation has led to negative effects on soil properties as indicated by the poorer soil fertility in cultivated land in comparison to natural forest. Important soil fertility indicators such as TOC, TN, OM, microbial population and MBC were significantly affected by various land use systems with lower levels in Homegarden of the study. On the contrary, *Jhum* fallow of more than 12 years exhibited improved soil properties as a result of less incidence of disturbance on the ecosystem. Present findings corresponds to the fact that soil under cultivation (Homegarden) is exposed to several disturbances owing to the various management activities incorporated coupled with the periodic growth and harvesting of crops with less organic matter addition that enhance depletion of organic matter in the soil.

Soil carbon and nitrogen stocks were also considerably lesser in Homegarden relative to the rest of the land uses. These stocks are significantly affected by the vegetation cover and magnitude of disturbances exerted on the particular ecosystem. Natural forest and *Jhum* fallow (>12 years) exhibited higher carbon and nitrogen sequestering capacity owing to the fact that these land use systems are free of any management practices leading to fluency of natural functions and processes. In addition, the nutrient distribution pattern is also greatly affected by various management practices and not only the vegetation. Lower values of TOC and TN but higher concentration of P_{avail}, Na and Ca in HG indicated the role of management practices. Soil texture and inherent parent material contributes to the variation in the chemical composition of the soil apart from the vegetation harboured.

Higher values of N % in the lower depths of AP may be well attributed to the leguminous feature of the plant and signifies the impression that it could be a potential landuse system for practicing intercropping/mixed cropping or for the purpose of restoration and reclamation of degraded lands. The considerable higher stocks of N and

C stock in the sub surface layer of AP in relation to other land uses supports the above argument relative to AP as a potential land use for sustainable farming practice with good scope for soil C and N sequestration.

MWD and GMD showed a strong negative correlation with silt content and decreased significantly with increased silt content. TOC and macro-aggregates were also found to have a strong positive correlation with MWD and GMD. These findings suggest that soil aggregate stability is significantly affected by the vegetation cover influencing the organic matter inputs in the soil and the inherent soil particle size distribution. The results also indicated that higher soils exhibiting higher silt content are more vulnerable to erosion. Higher associated TOC and TN content in macro-aggregates of land uses with higher TOC and TN in bulk soils implies that TOC and TN content in soil contributes profoundly to the structural stability of the soil.

Soil microbial populations were significantly higher in JF and NAF relative to the other land uses (HG, AP and CJ) indicating the impact of conversion of native forests to other land uses on the soil microbial composition and community. In addition, the disruption of soil macro-aggregates to lower order aggregate sizes owing to conversion of native forests and management practices profoundly influences the soil microbial population structure, their distinct habitats and colonization patterns in different aggregate sizes. Positive correlations of fungal and actinomycetes population with MWD and GMD indicates the role of fungi in aggregate stability with increased stability in JF and NF as evident from the higher fungal population count in these land uses. Positive correlation of actinomycetes with soil macro-aggregates represents the function of hyphae in improving the soil aggregate stability. On the contrary, negative correlations of fungi and bacteria with soil micro-aggregates signify that soils with lower stability impacts the microbial community of soil and vice-versa

Overall, the results showed a distinct change in soil properties with land use change ultimately leading to negative feedbacks between soil properties (eg. TOC, TN) and land use system as in the case of conversion of Natural forest to settled Homegardens. Comparing the observed land uses with natural forest, *Acacia pennata* plantation (AP) land use system can be a potential land use system for developing sustainable land use practices owing to its ability to recover and sequester higher carbon and nitrogen even in its sub surface soil layers as indicated by the results. The findings from the present study is expected to contribute in the development of land use management policies or practices and for further research ventures aimed at sustainable use of land and rejuvenating soil quality of degraded lands in Mizoram.

Our findings signify that conversion of native forests leading to cropping and different management practices in small scale agro-ecosystems instigates degradation of soil quality and represses sustainability. The results indicated that TOC and SOM strongly influences a wide range of soil properties especially soil aggregation which is one of the most important soil characteristic responsible for conservation of soil nutrients and soil fertility in general. Owing to the tremendous importance of TOC and SOM and its influence on soil health, its management aimed at repletion and retention is a key factor in developing management policies and selection of land use systems. It is therefore of great importance to adopt suitable cultivation practices and systems which will enhance soil properties. The results and inferences from the current study will help in enhancing our understanding on the impact of conversion of native forests to other land uses including cropping, and management practices on soil physico-chemical and biological characteristics. In order to achieve sustainability, appropriate management practices aimed at conserving and enhancing soil properties should be the key factor for adoption of any agro ecosystems in the current scenario of land use change.

REFERENCES

Aayog, NITI. (2018). Shifting Cultivation: Towards a Transformational Approach. (Accessed on 23 September 2019).

Abad, J.R.S., Hassan, K. and Alamdarlou, E.H. (2014). Assessment the effects of land use changes on soil physicochemical properties in Jafarabad of Golestan province, Iran. Bulletin of Environment, *Pharmacology and Life Sciences* 3: 296-300.

Abbasi, M.K., Zafar, M. and Khan, S.R. (2007). Influence of different land-cover types on the changes of selected soil properties in the mountain region of Rawalakot Azad Jammu and Kashmir. *Nutrient Cycling in Agroecosystems*, 78(1): 97110.

Abebe, N. and Endalkachew, K. (2012). The contribution of coffee agro-ecotype to soil fertility in Southwestern Ethiopia. *African Journal of Agricultural Research*, 7(1), 74-81.

Abegaz, A., Winowiecki, L.A., Vågen, T.G., Langan, S. and Smith, J.U. (2016). Spatial and temporal dynamics of soil organic carbon in landscapes of the upper Blue Nile Basin of the Ethiopian Highlands. *Agriculture Ecosystem Environ*ment 218, 190–208.

Achmad, A., Hasyim, S., Dahlan. B. and Aulia. D.N. (2015). Modeling of Urban Growth in Tsunami-prone City Using Logistic Regression: Analysis of Banda Aceh, Indonesia. *Applied Geography* 62:237–246.

Adesina, F.A. (2005). Geoinformation and Natural Resources exploitation in Africa; United Nations Economic and Social Council Paper delivered in Fourth meeting of the committee on Development Information, Addis Ababa on 23 – 28 April.

Ahluwalia, I., Kanbur, R. and Mohanty, P. (2014). Urbanisation in India: challenges, opportunities and the way forward. SAGE Publications, New Delhi.

Ajami, M., Heidari, A., Khormali, F., Gorji, M. and Ayoubi, S. (2016). Environmental factors controlling soil organic carbon storage in loess soils of a sub-humid region, northern Iran. *Geoderma*, 281, 1–10.

Alankar. (2015). India’s megacities and climate change: explorations from Delhi and Mumbai (STEPS working paper no. 79). STEPS Centre, Sussex.

Alexandratos, N. and Bruinsma, J. (2012). World agriculture towards 2030/2050: The 2012 revision (ESA Working Papers). ESA Working Paper No. 12-03, Agricultural Development Economics Division, Food and Agriculture Organization of the United Nations. Rome.

Ali, S., Begum, F., Hayat, R. and Bohannan, B.J.M. (2017). Variation in soil organic carbon stock in different land uses and altitudes in Bagrotvaley, Northern Karakoram. Acta Agriculture Scandinavica Section B- *Soil & Plant Science*, 67, 551-561.

Al-Kaisi, M.M., Douelle, A. and Kwaw-Mensah, D. (2014). Soil micro-aggregate and macroaggregate decay over time and soil carbon change as influenced by different tillage systems. *Journal of Soil & Water Conservation*, 69(6).

Amezketa, E. (1999). Soil aggregate stability: a review. *Journal of Sustainable Agriculture*, 14 (2–3).

An, S.S. Mentler, A., Mayer, H. and Blum, W.E.H. (2010). Soil aggregation, aggregate stability, organic carbon and nitrogen in different soil aggregate fractions under forest and shrub vegetation on the Loess Plateau, China. *Catena*, 81, 226–233.

Anderson, J.M. and Ingram, I.S.I. (1993). Tropical Soil Biology and Fertility - A Handbook of Methods. 2ndedn. CAB International, Wallingford, UK.

Anonymous. (2009). New land use policy: executive summary. Government of Mizoram, Aizawl. (Accessed on 21 August 2019).

Anteneh, A., Birru, Y., Yihenew, G. and Tadele, A. (2013). The role of biochar on acid soil reclamation and yield of Teff (Eragrostistef [Zucc] Trotter) in Northwestern Ethiopia. *Journal of Agricultural Science*. 6.

Asadu, C.L.A., Nwafor, I.A. and Chibuike, G.U. (2015). Contributions of Microorganisms to Soil Fertility in Adjacent Forest, Fallow and Cultivated Land Use Types in Nsukka, Nigeria. *International Journal of Agriculture and Forestry*. 5(3): 199-204.

Ashagrie, Y., Zech, W., Guggenberger, G. and Mamo, T. (2007). Soil aggregation and total and particulate organic matter following conversion of native forests to continuous cultivation in Ethiopia. *Soil Tillage Research*. 94, 101e108.

Aspiras, R.B., Allen, O.N., Harris, R.F. and Chester, G. (1971). Aggregate stabilization by filamentous microorganisms. *Science* 112, 282–284.

Assefa, D., Rewald, B., Sandén, H., Rosinger, C., Abiyu, A., Yitaferu, B. and Godbold, D.L. (2017). Deforestation and land use strongly effect soil organic carbon and nitrogen stock in Northwest Ethiopia. *Catena* 153, 89–99.

Ayoubi, S., Karchegani, P.M., Mosaddeghi, M.R. and Honarjoo, N. (2012). Soil aggregation and organic carbon as by topography and land use change in western Iran. *Soil and Tillage Research* 121:18–26.

Aytenew, M. and Kibret, K. (2016). Assessment of soil fertility status at dawja watershed in Enebse Sar Midir district, Northwestern Ethiopia. *International Journal of Plant & Soil Science* 11(2): 1-13.

Azizsoltani, E., Honarjoo, N. andAyoubi, S. (2019). How soil pore distribution could help in soil quality studies as an appropriate indicator? *Eurasian journal of Soil Science,* 52:654–660.

Balesdent, J., Basile-Doelsch, I., Chadoeuf, J., Cornu, S., Derrien, D., Fekiacova, Z. and Hatté, C. (2018). Atmosphere–soil carbon transfer as a function of soil depth. *Nature,* 559, 599–602.

Barah, B.C. (2007). Strategies for agricultural development in the North-East India: challenges and emerging opportunities. *Indian Journal of Agricultural Economics*, 62(902-2016-67379).

Bardgett, R. D. and Shine, A. (1999). Linkages between plant litter diversity, soil microbial biomass and ecosystem function in temperate grasslands. *Soil Biology & Biochemistry*, 31, 317-321.

Barea, J.M, Requena, N. and Jimenez, I. (1996). A revegetation strategy based on the management of arbuscular mycorrhizae, Rhizobium and rhizobacteria for the reclamation of desertified Mediterranean shurbland ecosystems. In Mycorrhization of Forest Plants under Arid and Semi-arid Conditions and Desertification Control in the Mediterranean. pp. 75–86. CIHEAM-IAMZ, Zaragoza.

Barea, J.M., Azcón, R. and Azcón-Aguilar, C. (1992). The use of 15N to assess the role of VA mycorrhiza in plant N nutrition and its application to evaluate the role of mycorrhiza in restoring Mediterranean ecosystems. In Mycorrhizas in the Ecosystem. Eds. D J Read, D H Lewis, A H Fitter and I J Alexander. pp. 190–197. CAB International, Cambridge.

Beare, M.H., Hu, S., Coleman, D.C. and Hendrix, P.F., (1997). Influences of mycelial fungi on soil aggregation and organic matter storage in conventional and no-tillage soils. *Applied Soil Ecol*ogy. 5, 211–219.

Begum, N., Narayana, J. and SL, A.K. (2010). Land Use/Land Cover Changes in the Catchment of Water Bodies in and Around Davangere City, Karnataka. *International Journal of ecology and environmental sciences*, *36*(4), 277-280.

Bello, H.S, Isa, T, Isa, M.A. and Akinmuisere, K. (2013). Effects of land use on the nature and population of microorganisms in the semi-arid region of north-eastern Nigeria. *International journal of environment*. 2(1): 224-230.

Bhatiam A., Aggarwal, P.K., Jain, N. and Pathak, H. (2011). Greenhouse gas emission from rice-wheat growing areas in India: spatial analysis and up scaling, Greenhouse Gases: *Science and Technology*, vol. 2, no. 2, pp. 115–125.

Bhattacharyya, R., Ghosh, B., Mishra, P., Mandal, B., Rao, C. and Sarkar, D. (2015). Soil degradation in India: Challenges and potential solutions. *Sustainability*, 7(4), 3528–3570.

Bhattarai, A., Bhattarai, B. AND Pandey, S. (2015). Variation of soil microbial population in different soil horizons. *J Microbiol Exp*, *2*(2), 00044.

Biswas, A., Alamgir, M., Haque, S.M.S. and Osman, K.T. (2012). Study on soils under shifting cultivation and other land use categories in Chittagong Hill Tracts, Bangladesh. *Journal of Forest Research*, 23(2): 261-265.

Blanco-Canqui, H. and Lal, R. (2008). No-tillage and soil-profile carbon sequestration: an on farm assessment. *Soil Science Society of America Journal*, 72, 693–701.

Bonan, G. (2008). Forest and climate change: forcing, feedbacks and the climate benefits of forests. *Science* 320(5882):1444–1449

Brady, N.C. (1990). The Nature and Properties of Soils 10th Edition. Prentice-Hall Limited, India. 621p.

Brahma, B., Pathak, K., Lal, R., Kurmi, B., Das, M., Nath, P. C. and Nath, A.J. (2018). Ecosystems carbon sequestration through restoration of degraded lands in northeast India. *Land Degradation & Development*, 29, 15–25.

Brahmanand, P.S., Kumar, A., Ghosh, S., Chowdhury, S.R., Singandhupe, R. B. and Singh, R. (2013). Challenges to food security in India. *Current Science*, 841–846.

Brown, D.G., Pijanowski, B.C. and Duh, J.D. (2000). Modeling the relationships between land use and land cover on private lands in the Upper Midwest, USA. *Journal of Environmental Management, 59*(4), 247-263.

Bruun, T.B., Elberling, B., de Neergaard, A. and Magid, J. (2015). Organic carbon dynamics in different soil types after conversion of forest to agriculture. *Land Degradation & Development*, *26*(3), 272-283.

Burns, R.G. and Davies, J.A. (1986). The microbiology of soil structure. In: Lopez-Real, J.M., Hodges, R.D. (Eds.), The Role of Microorganisms in a Sustainable Agriculture. A.B. Academic Publishers, Berkhamstead, UK, pp. 9–27.

Carrizo, M.E., Alesso, C.A., Cosentino, D.and Imhoff, S. (2015). Aggregation agents and structural stability in soils with different texture and organic carbon contents. *Scientia Agricola*, *72*(1), 75-82.

Chacon, N. and Dezzeo, N. (2004). Phosphorus fractions and sorption processes in soil samples taken in a forest-savanna sequence of the Gran Sabana in Southern Venezuela. *Biology Fertility Soils* 40, 14–19.

Chand, R. and Parappurathu, S. (2012). Temporal and spatial variations in agricultural growth and its determinants. *Economic and Political Weekly*, 47(26–27), 55–64.

Chang, J., Ciais, P., Viovy, N., Vuichard, N., Herrero, M., Havlík, P. and Soussana, J. (2016). Effect of climate change, CO2 trends, nitrogen addition, and land-cover and management intensity changes on the carbon balance of European grasslands. *Global Change Biology*, 22(1), 338–350.

Chatterjee, S., Saikia, A., Dutta, P., Ghosh, D. and Worah, S. (2006). Review of Biodiversity in Northeast India. Background Paper No. 13, WWF-India, Delhi.

Chen, C., Liu, W., Jiang, X. and Wu, J. (2017). Effects of rubber-based agroforestry systems on soil aggregation and associated soil organic carbon: implications for land use. *Geoderma* 299, 13–24.

Chen, J., Li, S., Liang, C., Xu, Q., Li, Y., Qin, H. and Fuhrmann, J.J. (2017). Response of microbial community structure and function to short-term biochar amendment in an intensively managed bamboo (Phyllostachys praecox) plantation soil: effect of particle size and addition rate. *Science of the Total Environment*, 574, 24-33.

Chen, L., Smith, P. and Yong, Y. (2015). How has soil carbon stock changed over recent decade? *Global change biology*. 21, 3197–3199.

Chen, T.H., Chiu, C. Y. and Tian, G.L. (2005). Seasonal dynamics of soil microbial biomass in coastal sand dune forest. *Pedobiologia*, 49, 645-653.

Chenu, C., Le Bissonnais, Y. and Arrouays, D. (2000). Organic matter influence on clay wettability and soil aggregate stability. *Soil Science Society of America Journal* 64: 1479-1486.

Chivenge, P., Vanlauwe, B., Gentile, R. and Six, J. (2011). Organic resource quality influences short-term aggregate dynamics and soil organic carbon and nitrogen accumulation. *Soil Biology & Biochemistry*, 43, 657-666.

Choudhury, B.U., Mohapatra, K.P., Das, A., Das, P.T., Nongkhlaw, L., Fiyaz, A.R., Ngachan, S.V., Rajkhowa, D.J. and Munda, G.C. (2013). Spatial variability in distribution of organic carbon stocks in the soils of North East India. *Current Science*, 104(5): 604-614.

Choudhury, B.U., Pratibha, T., Das, S.V., Ngachan., Mokidul Islam., Anup Das, B.C., Verma, K.P., Mohapatra, L., Nongkhlaw., Syed, B., Islam. and Munda, G.C. (2014). Landuse land cover change detection, soil health assessment and socioeconomy in Northeast India: a remote sensing and GIS approach. *Research Bulletin, NAIP publication* 7 (2014): 53.

Cohen, B. (2004). Urban growth in developing countries: a review of current trends and a caution regarding existing forecasts. *World development*, *32*(1), 23-51.

Colney, L. and Nautiyal, B.P. (2013). Characterization and evaluation of soils of Aizawl district, Mizoram, India using remote sensing and GIS techniques. *Journal of Geomatics*, 7, 83-91.

Cottenie, A. (1980). Soil and plant testing as a basis of fertilizer recommendations. FAO soil bulletin 38/2. Food and Agriculture Organization of the United Nations, Rome, Italy.

Curtis, P., Slay, C., Harris, N., Tyukavina, A. and Hansen. M.C. (2018). Classifying Drivers of Global Forest Loss. *Science,* September 14.

Dalal, R.C. and Bridge, B.J. (1996). Aggregation and organic matter storage in sub-humid and semi-arid soils. pp. 263-307. In Carter, M.R. & B. A. Stewart (eds.). Structure and organic matter storage in agricultural soils. *Adv. Soil Sci.* Lewis Publishers, CRC Press, Inc., Boca Raton, FL.

Das, A., Munda, G.C., Ghosh, P.K., Ramkrushna, G.I., Patel, D.P., Choudhury, B.U. and Jayanta, L. (2011). Managing degraded land and soil organic carbon for climate resilient agriculture in North East India. *Soil carbon sequestration for climate change mitigation and food security,* 241-265.

Davinic, M., Fultz, L.M., Acosta-Martinez, V., Calderón, F.J., Cox, S.B., Dowd, S.E., Allen, V.G., Zak, J.C. and Moore-Kucera, J. (2012). Pyrosequencing and mid-infrared spectroscopy reveal distinct aggregate stratification of soil bacterial communities and organic matter composition. *Soil Biology Biochemistry* 46: 63-72.

Davy, M.C. and Koen, T.B. (2013). Variations in soil organic carbon for two soil types and six land uses in the Murray Catchment, New South Wales, Australia. *Soil Research*, 51, 631–644.

de Blécourt, M., Brumme, R., Xu, J., Corre, M. D. and Veldkamp, E. (2013). Soil carbon stocks decrease following conversion of secondary forests to rubber (Hevea brasiliensis) plantations. *PLoS One*, 8, e69357.

De Deyn, G.B., Shiel, R.S., Ostle, N.J., McNamara, N.P., Oakley, S., Young, I., Freeman, C., Fenner, N., Quirk, H., and Bardgett, R.D. (2011). Additional carbon sequestration benefits of grassland diversity restoration. J *Applied Ecol*ogy 48: 600-608.

De Fries, R.S., Foley, J.A. and Asner, G.P. (2004). Land-use choices: balancing human needs and ecosystem function. *Frontiers in Ecology and the Environment* 2 (5): 249–257.

Deb, S., Kumar, D. and Chakraborty, S. (2019). Comparative carbon stability in surface soils and subsoils under submerged rice and upland non-rice crop ecologies: a physical fractionation study. *Catena*, 175, 400–410.

Deekor, T.N, Iwara, A.I. and Ogundele, F.O. (2012). Changes in soil properties under different land use covers in parts of Odukpani, Cross River State, Nigeria. *Journal of Environment and Ecology*, *3*(1), 86-99.

Degens, B.P., Schipper, L.A. and Sparling, G.P. (2000). Decrease in organic C reserves in soils can reduce the catabolic diversity of the soil microbial communities. *Soil Biology & Biochemistry,* 32, 189-196.

Demenois, J., Carriconde, F., Bonaventure, P., Maeght, J.L., Stokes, A. and Rey, F. (2018). Impact of plant root functional traits and associated mycorrhizas on the aggregate stability of a tropical Ferralsol. *Geoderma* 312, 6–16.

Derpsch, R., Roth, C.H. and Sidiras, N. (1991). Controle da Erosão no Paraná, Brasil: Sistemas de Cobertura do Solo, Plantio Direto e Preparo Conservacionista do Solo. Deutsche Gesellschaft für Technissche Zusammenarbeit (GTZ), Eschborn, Germany/IAPAR, Londrina, Brazil, p. 272.

Devine, S., Markewitz, D., Hendrix, P. and Coleman, D. (2014). Soil aggregates and associated organic matter under conventional tillage, no-tillage, and forest succession after three decades. *PLos One* 9 (1), e84988.

Dick, R.P. (1997). Soil enzyme activities as integrative indicators of soil health, in: Pankhurst, C.E., Doube, B.M., & Gupta V.V.S.R. (Eds.), *Biological Indicators of Soil Health*, CA, International, Wallingford, 121-156.

Don, A. and Schumacher, F.A. (2011). Impact of tropical land-use change on soil organic carbon stocks: meta analysis. *Global Change Biology*. 17, 1658–1670.

Don, A., Schumacher, J. and Freibauer, A. (2011). Impact of tropical land-use change on soil organic carbon stocks- a meta-analysis. *Global Change Biology*, 17, 1658-1670.

Duguma, L.A., Hager, H. and Sieghardt, M. (2010). Effects of land use types on soil chemical properties in smallholder farmers of central highland Ethiopia. Ekológia. Bratislava. *Ekológia (Bratislava)*, *29*(1), 1-14.

Duma, M.F. (2000). A Comparative Study of Soil Degradation between Rangelands under Communal Grazing and Controlled Grazing in Alice, Eastern Cape, M.Sc. Thesis, Rhodes University,

Durigan, M.R., Cherubin, M.R., de Camargo, P.B., Ferreira, J.N., Berenguer, E., Gardner, T.A., Barlow, J., Dias, C.T.S., Signor, D. and Junior, R.C.O. (2017). Soil Organic Matter Responses to Anthropogenic Forest Disturbance and Land Use Change in the Eastern Brazilian Amazon. *Sustainability*, 9, 379.

Dwivedi, R.S., Sreenivas, K. and Ramana, K.V. (2005). Land-use/land-cover change analysis in part of Ethiopia using Landsat Thematic Mapper data. *International Journal of Remote Sensing, 26(7):1285–1287.*

Elliott, E.T. (1986). Aggregate structure and carbon, nitrogen and phosphorus in native and cultivated soils. *Soil Science Society of America Journal*, 50, 627– 633.

Elliott, J.A. and Efetha, A.A. (1999). Influence of tillage and cropping system on soil organic matter, structure and infiltration in a rolling landscape. *Canadian Journal of Soil Science*, 79, 457–463.

Ellis, E.C. (2011). Anthropogenic transformation of the terrestrial biosphere. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *369*(1938), 1010-1035.

Ellis, E.C. and Ramankutty, N. (2008). Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment*, 6 (8): 439–447.

Erktan, A., Cécillon, L., Graf, F., Roumet, C., Legout, C. and Rey, F. (2016). Increase in soil aggregate stability along a Mediterranean successional gradient in severely eroded gully bed ecosystems: combined effects of soil, root traits and plant community characteristics. *Plant and Soil*, 398, 121-137.

Eshetu, Z., Giesler, R. and Hogberg, P. (2004). Historical land use pattern affects the chemistry of forest soils in the Ethiopian highlands. *Geoderma* 118, 149–165.

Fall, S., Diffenbaugh, N.S. Niyogi, D., Pielke Sr, R.A and Rochon, G. (2010). Temperature and equivalent temperature over the United States (1979–2005). *International Journal of Climatology*, 30, 2045-2054.

Fan, F., Weng, Q. and Wang, Y. (2007). Land use land cover change in Guangzhou, China, from 1998 to 2003, based on Landsat TM/ETM+ imagery. *Sensors*, 7:1323–1342.

FAO. (2015). Forest Resources Assessment.http://www.fao.org/3/a-i4808e.pdf.

FAO. (2017a). FAOSTAT: Food and agricultural data. Retrieved September 25, 2017, from http://www.fao.org/faostat/en/#home

FAO. (2017b). India at a glance. Retrieved May 14, 2019, from http://www.fao.org/india/fao-in-india/india-at-a-glance/en/.

FAO. (2006). Guidelines for Soil Description 4th Edn. Food and Agriculture Organisation of the United States, Rome.

Fasal, S. (2000). Urban expansion and loss of agricultural land – A GIS based study of Saharanpur City, India; *Environment and Urbanization*, Vol 12. (2). 133 – 149

Fenta, A.A., Yasuda, H., Shimizu, K., Haregeweyn, N.and Negussie, A. (2016). Dynamics ofsoil erosion as influenced by watershed management practices: a case study of the Agula watershed in the semi-arid highlands of northern Ethiopia. *Environment Management*, 58, 889–905.

Fernandez, R., Quiroga, A., Zorati, C. and Noellemeyer, E. (2010). Carbon contents and respiration rates of aggregate size fractions under no-till and conventional tillage. *Soil Tillage Research*, 109, 103–109.

Ferreira, M.C., Andrade, D.S. and Chueire, L.M.O. (2000). Tillage method and crop rotation effects on the population sizes and diversity of bradyrhizobia nodulating soybean. *Soil Biol Biochem*, 32: 627-37.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R. and Helkowski, J.H. (2005). Global consequences of land use. *Science*, 309(5734), 570-574.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N. and Snyder, P.K. (2005). Global consequences of land use. *Science* 309,570e574.

Fontaine, S., Barot, S., Barré, P., Bdioui, N., Mary, B. and Rumpel, C. (2007). Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature*, 450, 277–280.

Franzluebbers, A.J., Wright, S.F. and Stuedemann, J.A.(2000). Soil aggregation and glomalin under pastures in southern Piedmont USA. Soil Science. *Society of American Journal*, 64, 1018–1026.

Gaiser, T. and Stahr, K. (2013). Soil organic carbon, soil formation and soil fertility. In *Ecosystem Services and Carbon Sequestration in the Biosphere*, (pp. 407-418). Springer, Dordrecht.

Ge, Z., Fang, S., Chen Han, Y.H., Zhu, R. and Peng, S. (2018). Soil aggregation and organic carbon dynamics under poplar plantations. *Forests*, 9 (9), 508.

Gebrelibanos, T. and Assen, M. (2013). Effects of land-use/cover changes on soil properties in a dry land watershed of Hirmi and its adjacent agro ecosystem: Northern Ethiopia. *International Journal of Geosciences Research* 1(1): 45-57.

Geesing, D., Felker, P. and Bingham, R.L. (2000). Influence of mesquite (Prosopis glandulosa) on soil nitrogen and carbon development: implications for global carbon sequestration. *Journal of Arid Environment*, 46(2), 157–180.

Gelaw, A.M., Singh, B. and Lal, R. (2014). Soil organic carbon and total nitrogen stocks under different land uses in a semi-arid watershed in Tigray, Northern Ethiopia, *Agriculture Ecosystems and Environment*, 188, 256–263.

Genxu, W., Haiyan, M., Ju, Q. and Juan, C. (2004). Impact of land use changes on soil carbon, nitrogen and phosphorus and water pollution in an arid region of northwest China. *Soil Use and Management*. 20: 32-39.

George, T. (2014). Why crop yields in developing countries have not kept pace with advances in agronomy. *Global Food Security*, 3(1), 49–58.

Ghosh, P.K., Saha, R., Gupta, J.J., Ramesh, T., Das, Anup, Lama, T.D., Munda, G.C., Bordoloi, J.S., Verma, M.R. and Ngachan, S.V. (2009). Long term effect of pastures on soil quality of North-East India. *Australian Journal of Soil Res*earch, 47:372-379.

Giertz, S., Junge, B. and Diekkrüger, B. (2005). Assessing the effects of land use change on soilphysical properties and hydrological processes in the sub-humid tropical environment of West Africa. *Physic and Chemistry of the Earth*, Parts A/B/C 30, 485–496.

Giller, K.E., Beare, M.H., Lavelle, P., Izac, A.M.N. and Swift, M.J. (1997). Agricultural intensification, soil biodiversity and agroecosystem function. *Applied Soil Ecology*, 6, 3e16.

GoI. (2015). Directorate of Economics and Statistics. Ministry of Agriculture and Farmers Welfare, Government of India.

Gol, C. (2009). The effects of land use change on soil properties and organic carbon at Dagdami river catchment in Turkey. *Journal of Environmental Biology*, 30(5), 825-830.

Golchin, A. and Asgari, H. (2008). Land use effects on soil quality indicators in north-eastern Iran. *Soil Research*, *46*(1), 27-36.

Golchin, A., Oades, J.M., Skjemstad, J.O. and Clarke, P. (1994). Soil structure and carbon cycling. *Australian Journal of Soil Research*, 32:1043–1068.

Gómez, J.A., Giráldez, J.V., Pastor, M. and Fereres, E. (1999). Effects of tillage method on soil physical properties, infiltration and yield in an olive orchard. *Soil and tillage research*, *52*(3-4), 167-175.

Gregory, A.S., Ritz, K., McGrath, S.P., Quinton, J.N., Goulding, K.W.T., Jones, R.J.A. and Whitmore, A.P. (2015). A review of the impacts of degradation threats on soil propertiesin the UK. *Soil Use and Management*. 31, 1–15.

Grogan, P., Lalnunmawia, F. and Tripathi, S.K. (2012). Shifting cultivation in steeply sloped regions: a review of management options and research priorities for Mizoram state, Northeast India. *Agroforestry Systems*, *84*(2), 163-177.

Groppo, J.D., Lins, S.R.M., Camargo, P.B., Assad, E.D., Pinto, H.S., Martins, S.C., Salgado, P.R., Evangelista, B., Vasconcellos, E., Sano, E.E., Pavao, E., Luna, R. and Martinelli, L.A. (2015). Changes in soil carbon, nitrogen and phosphorus due to land-use changes in Brazil. *Biogeosciences*, 12, 2533–2571

Guillaume, T., Maranguita, D., Murtilaksono, K. and Kuzyakov, Y. (2016). Sensitivity and resistance of soil fertility indicators to land-use changes: new concept and examples from conversion of Indonesian rainforest to plantations. *Ecological Indicators*, 67, 49-57.

Guo, L.B. and Gifford, R.M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global change biology*, *8*(4), 345-360.

Gupta, N., Kukal, S.S., Bawa, S.S. and Dhaliwal, G.S. (2009). Soil organic carbon and aggregation under poplar based agroforestry system in relation to tree age and soil type. *Agroforestry Systems*, 76, 27-35.

Guru, B., Anubhooti. (2015). Urban growth effects for land use changes leading to natural hazards in parts of Mumbai metro city, India. Disaster Adv 8(10):1–14.

Han, D., Wiesmeier, M., Conant, R. T., Kühnel, A., Sun, Z., Kögel-Knabner, I. and Ouyang, Z. (2018). Large soil organic carbon increase due to improved agronomic management in the North China Plain from 1980s to 2010s. *Global change biology*, *24*(3), 987-1000.

Hansen, M.C. (2018). Global Forest Watch, University of Maryland, Google, USGS, and NASA.

Haregeweyn, N., Tsunekawa, A., Poesen, J., Tsubo, M., Meshesha, D.T., Fenta, A.A., Nyssen, J. and Adgo, E. (2017). Comprehensive assessment of soil erosion risk for better land use planning in river basins: case study of the Upper Blue Nile River. *Science of the Total Environment*, 574, 95–108.

Hati, K.M., Swarup, A., Dwivedi, A.K., Misra, A.K. and Bandyopadhyay, K.K. (2007). Changes in soil physical properties and organic carbon status at the topsoil horizon of a vertisol of central India after 28 years of continuous cropping, fertilization and manuring. *Agriculture, Ecosystems and Environment*, 119, 127–134.

Hauchhum, R. and Tripathi, S.K. (2017). Rhizosphere effects of Melocanna baccifera on soil microbial properties under different fallow phases following shifting cultivation. *International Journal of Plant and Soil Science*, 17, 1-9.

Haynes, R.J. and Naidu, R. (1998). Influence of Lime, Fertilizer and Manure Applications on soil organic matter content and soil physical conditions: a review. *Nutrient Recycling in Agroecosystem*, 51: 123-137.

Hoelscher, K. and Aijaz, R. (2016). Challenges and opportunities in an urbanising India. *International Area Studies Review*, *19*(1), 3-11.

Hoorman, J.J., Moraesa, J.C. and Randall, R. (2009). The biology of soil compactness, Fact Sheet, Agriculture and Natural Resources. The Ohio State University, Columbus, Ohio,pp. 1–7.

Ibekwe, A.M., Poss, J.A., Grattan, S.R., Grieve, C.M. and Suarez, D. (2012). Bacterial diversity in cucumber (Cucumis sativus) rhizosphere in response to salinity, soil pH, and boron. *Soil Biology and Biochemistry*. 42. 567- 575.

ICAR. (2015). Vision 2050. New Delhi. Retrieved from http://www.icar.org.in/files/Vision-2050-ICAR.pdf.

Ifatimehin, O.O. and Musa, S.D. (2008). Application of Geoninformatic Technology in Evaluating Urban Agriculture and Urban Poverty in Lokoja, Nigeria. *Journal of Geography and the Environment*, *1*(1), 21-31.

Ifatimehin, O.O. and Ufuah, M.E. (2006). An analysis of urban expansion and loss of vegetation cover in Lokoja, using GIS techniques. *Zaria Geography*, *17*(1), 28-36.

IPCC. (2000). Land Use, Land Use Change and Forestry. A Special Report of IPCC. Cambridge University Press, Cambridge. pp. 377.

ISFR. (2017). India State of Forest Report. http://fsi.nic.in/isfr2017/mizoram-isfr-2017.pdf.

ISFR. (2019). India State of Forest Report. http://fsi.nic.in/isfr2019/mizoram-isfr-2019.pdf.

Islam, K.R. and Weil, R.R. (2000). Land use effects on soil quality in a tropical forest ecosystem of Bangladesh. *Agriculture, Ecosystems and Environment*, 79, 9-16.

Jangid, K., Williams, M.A., Franzluebbers, A.J., Schmidt, T.M., Coleman, D.C. and Whitman, W.B. (2011). Land-use history has a stronger impact on soil microbial community composition than aboveground vegetation and soil properties. *Soil Biology and Biochemistry*, 43, 2184–2193.

Jastrow, J.D. (1996). Soil aggregate formation and the accrual of particulate and mineral-associated organic matter. *Soil Biology and Biochemistry*, 28:665–676.

Javed, A. and Khan, I. (2012). Landuse/Land cover change due to mining activities in Singrauli Industrial belt, Madhya Pradesh using remote sensing and GIS. *Journal of Environmental Research and Development*, *6*(3A), 834-843.

Jiang, J.P., Xiong, Y.C., Jiang, H.M., Ye, D.Y., Song, Y.J. and Li, F.M. (2009). Soil microbial activity during secondary vegetation succession in semiarid abandoned lands of Loess Plateau. *Pedosphere*, 19, 735-747

Jin, H., Sun, O. J. and Liu, J. (2010). Changes in soil microbial biomass and community structure with addition of contrasting types of plant litter in a semiarid grassland ecosystem. *Journal of Plant Ecology*, 3, 209-217.

Jing, W.J., Yue, X., Hui-Chun, Y., Chong, Y.S. and Yuan, F.H. (2011).Efects of land use and soil management particles on soil fertility quality in north china cities urban fringe. *African Journal of Agricultural Research*, 6(11):2059–2065.

John, R.N. and Kim, S.P. (2002). Aggregate stability and size distribution. In: Dane, J.H., Topp, G.C. (Eds.), Methods of Soil Analysis. Part 4. Physical Methods. Soil Sci. Am. Book Series No. 5. *Soil Science of America*, Madison, pp. 425–442.

Johnson, D.W., Susfalk, R.B., Ghloz, H.L. and Hanson, P.J. (2000). Stimulated effects of temperature and precipitation change in several forest ecosystems. *Journal of Hydrology*, 235(3–4):183–204.

Kabiri, V., Raiesi, F. and Ghazavi, M.A. (2016). Tillage effects on soil microbial biomass, SOM mineralization and enzyme activity in a semi-arid Calcixerepts. *Agriculture, Ecosystems and Environment*, 232, 73–84.

Kalhoro, S.A., Xuexuan, X., Chen, W., Hua, R., Raza, S. and Ding, K. (2017). Effects of different land-use systems on soil aggregates: a case study of the loess plateau (Northern China). *Sustainability*, 9, 1349.

Kalnay, E. and Cai, M. (2003). Impact of urbanization and land use change on climate. *Nature*, Vol. 423: 528-31.

Karl, T.R., Gleason, B.E., Menne, M.J., McMahon, J.R., Heim Jr, R.R., Brewer, M.J. and Groisman, P.Y. (2012). US temperature and drought: Recent anomalies and trends. *Eos, Transactions American Geophysical Union*, *93*(47), 473-474.

Ketema, H. and Yimer, F. (2014). Soil property variation under agroforestry based conservationtillage and maize based conventional tillage in Southern Ethiopia. *Soil Tillage Research*.141, 25–31.

Khormali, F., Ajamai, M., Ayoubi, S., Srinivasarao, C. and Wani, S.P. (2009). Role of deforestation and hillslope position on soil quality attributes of loess-derived soils in Golestan province. Iran, *Agriculture Ecosystem and Environment*, 134(3-4):178–189.

Kiflu, A. and Beyene, S. (2013). Effects of different land use systems on selected soil properties in South Ethiopia. *Journal of Soil Science and Environment Management*, 4(5): 100-107.

Kimble, J.M., Rice, C.W., Reed, D., Mooney, S., Follett, R.F. and Lal, R. (2007). Soil Carbon Management: Economic, Environmental and Societal benefits. *Taylor & Francis Group*, LLC.

Kuemmerle, T., Müller, D., Griffiths, P. and Rusu, M. (2009). Land use change in Southern Romania after the collapse of socialism. *Regional Environmental Change*, *9*(1), 1.

Kurmi, B., Nath, A.J., Lal, R. and Das, A.K. (2020). Water stable aggregates and the associated active and recalcitrant carbon in soil under rubber plantation. *Science of The Total Environment*, *703*, 135498.

Kushwaha, C.P., Tripathi, S.K. and Singh, K.P. (2001). Soil organic matter and water-stable aggregates under different tillage and residue conditions in a tropical dryland agroecosystem. *Applied Soil Ecology*, *16*(3), 229-241.

Lal, R. (1998). Soil quality and agricultural sustainability. CRC press.

Lal, R. (2001). Soil degradation by erosion. *Land Degradation*. Dev. 12, 519–539.

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *Science*, 304, 1623–1627.

Lal, R. (2006). Enhancing crop yields in the developing countries through restoration ofthe soil organic carbon pool in agricultural lands. *Land Degradation*. Dev. 17, 197–209.

Lal, R. (2018). Digging deeper: A holistic perspective of factors affecting soil organic carbon sequestration in agroecosystems. *Global Change Biology*, *24*(8), 3285-3301.

Lal, R. (2018). Saving global land resources by enhancing eco-efficiency of agroecosystems. *Journal of Soil Water Conservation*, 73, 100–106.

Lal, R. (2009). Soils and world food security. *Soil Tillage Research.* 102, 1-4.

Lalnunzira, C. and Tripathi, S.K. (2018). Leaf and root production, decomposition and fluxes of carbon and nitrogen during stand development in tropical moist forests of northeast India. *Soil Research*, (In press).

Lalnunzira, C., Brearley, F.Q. and Tripathi, S.K. (2019). Changes in fine and coarse root biomass, production and element return during recovery of wet tropical mountain forest following stone mining in Mizoram. *Journal of Mountain Science*, (In Press).

Lambin, E.F., Geist. H.J. and Lepers, E. (2003). Dynamics of land-use and land-cover change in tropical regions. *Annual Review of Environment & Resources*. 28: 205-241.

Lambin, E.F., Turner, B.L. and Geist, H.J. (2001). The causes of land-use and land-cover change: moving beyond the myths. *Global Environmental Change*, vol. 11, no. 4, pp. 261–269.

Lelisa, A. and Abebaw, A. (2016). Study on selected soil physicochemical properties of rehabilitated degraded bare land: The case of Jigessa rehabilitatıon site, Borana zone, Ethiopia. *Global Journal of Advanced Research*, 3(5): 345-354.

Lepers, E., Lambin, E.F., Janetos, A.C., DeFries, R., Achard, F., Ramankutty, N. and Scholes, R.J. (2005). A synthesis of rapid land-cover change information for the 1981–2000 period. *BioScience, 55*(2), 19-26.

Li, L., Zhang, X., Zhang, P., Zhang, J. and Pan, G. (2007). Variation in organic carbon and nitrogen in aggregate size fractions of a paddy soil under fertilization practices from Tai Lake Region, China. *Journal of Science and Food Agriculture*, 87:1052–1058.

Li, Q.Q., Zhang, H., Jiang, X.Y., Luo, Y.L., Wang, C.Q., Yue, T.X. and Gao, X.S. (2017). Spatially distributed modeling of soil organic carbon across China with improved accuracy. *Journal of Advances in Modeling Earth Systems*, 9, 1167–1185.

Li, S., Zhang, S.R., Pu, Y.L., Li, T., Xu, X.X., Jia, Y.X. and Gong, G.S. (2016). Dynamics of soil labile organic carbon fractions and C-cycle enzyme activities under straw mulch in Chengdu Plain. *Soil & Tillage Research*, 155, 289–297.

Liu, M.Y., Chang, Q.R., Qi, Y.B., Liu, J. and Chen, T. (2014) Aggregation and soil organic carbon fractions under different land uses on the tableland of the Loess Plateau of China. *Catena*, 115:19–28.

Liu, M.Y., Chang, Q.R., Qi, Y.B., Liu, J. and Chen, T. (2014). Aggregation and soil organic carbon fractions under different land uses on the tableland of the Loess Plateau of China. *Catena*, 115, 19-28.

Long, H., Li, Y., Liu, Y., Woods, M. and Zou, J. (2012). Accelerated restructuring in rural China fueled by ‘increasing vs. decreasing balance’ land-use policy for dealing with hollowed villages. *Land use policy*, *29*(1), 11-22.

Low, A.J. and Stuart, P.R. (1974). Microstructural differences between arable and old grassland soils as shown in the scanning electron microscope. *Journal of Soil Science*, 25, 135–142.

Lumbanraja, J., Syam, T., Nishide, H., Mahi, A.K., Utomo, M., Kimura, S. and Kimura, M. (1998). Deterioration of soil fertility by land use changes in South Sumatra, Indonesia: from 1970 to 1990. *Hydrological Processes*, 12, 2003–2013.

Lungmuana, Singh, S.B., Vanthawmliana, Saha, S., Dutta, S.K., Rambuatsaiha, Singh, A.R. and Boopathi, T. (2017). Impact of secondary forest fallow period on soil microbial biomass carbon and enzyme activity dynamics under shifting cultivation in north eastern hill region, India. *Catena*, 156:10-17.

Luo, Z., Wang, E. and Sun, O.J. (2010). Soil carbon change and its responses to agricultural practices in Australian agro-ecosystems: a review and synthesis. *Geoderma*, *155*(3-4), 211-223.

Lynch, J.M. and Bragg, E. (1985). Microorganisms and soil aggregate stability. *Advanced Soil Science*. 2, 134–170.

Mahmood, R., Hubbard, K.G., Leeper, R.D. and Foster, S.A. (2008). Increase in near-surface atmospheric moisture content due to land use changes: evidence from the observed dewpoint temperature data. *Monthly Weather Review*, *136*(4), 1554-1561.

Maia Andrade, E., Valbrun, W., Almeida, A.M. M.D., Rosa, G. and Silva, A.G.R.D. (2020). Land-Use Effect on Soil Carbon and Nitrogen Stock in a Seasonally Dry Tropical Forest. *Agronomy*, *10*(2), 158.

Maia, S.M.F., Carvalho, J.L.N., Cerri, C.E.P., Lal, R., Bernoux, M., Galdos, M.V. and Cerri, C.C. (2013). Contrasting approaches for estimating soil carbon changes in Amazon and Cerrado biomes. *Soil Tillage Research*, 133, 75–84.

Maia, S.M.F., Otutumi, A.T., de Sá Mendonça, E., Neves, J.C.L. and de Oliveira, T.S. (2019). Combined effect of intercropping and minimum tillage on soil carbon sequestration and organic matter pools in the semiarid region of Brazil. *Soil Research*, 57, 266–275.

Maithani, B.P. (2005b). Shifting Cultivation in North-East India Policy Issues and Options. Published and printed by Krishna for Mittal Publications, Mohan Garden, New Delhi, India.

Mandelas, A.E., Hatzichristos, T. and Prastacos, P. (2007). A fuzzy cellular automata based shell for modeling urban growth- a pilot application in Mesogia area. In: *10th AGILE international conference on geographic information science Aalborg University, Denmark.*

Manpoong, C., De Mandal, S., Bangaruswamy, D.K., Perumal, R.C., Benny, J., Beena, P.S. and Tripathi, S.K. (2020). Linking rhizosphere soil biochemical and microbial community characteristics across different land use systems in mountainous region in Northeast India. *Meta Gene*, *23*, 100625.

Manpoong. (2019). Influence of rhizosphere in different land use systems of Mizoram. Mizoram. Ph.D. Thesis.

Mark, M. and Kudakwashe, M. (2010). Rate of land use/land cover changes in Shurugwi District, Zimbabwe: Drivers for Change. *Journal of Sustainable Development in Africa*, *12*(3), 107-121.

Materechera, S. (2010). Soil physical and biological properties as influenced by growth of vetiver grass (Vetiveria zizanioides L.) in a semi-arid environment of South Africa. In: *19th World Congress of Soil Science, Soil Solutions for a Changing World* (pp.1-6).

Mauser, W., Klepper, G., Zabel, F., Delzeit, R., Hank, T., Putzenlechner, B. and Calzadilla, A. (2015). Global biomass production potentials exceed expected future demand without the need for cropland expansion. *Nature Communications*, 6(8745).

Meyer, W.B. (1995). Past and present land use and land cover in the USA. *Consequences*, *1*(1), 25-33.

Mills, A.J. and Fey, M.V. (2004). Declining soil quality in South Africa: effects of land use on soil organic matter and surface crusting. *South African Journal of Plant and Soil*, 21, 388e398.

MIRSAC. (2007). Natural resources mapping of Mizoram using remote sensing and GIS, Aizawl district (A Project Report), Mizoram remote sensing application centre, Science Technology & Environment, Aizawl. pp, 2-37.

Mishra, B.P. (2010). A study on the micro-environment, litter accumulation on forest floor and available nutrients in the soils of broad-leaved, mixed pine and pine forests at two distinct altitudes in Meghalaya, northeast India. *Current Science*, 90(12), 1829-1833.

Misra, U.K. and Saithantuaanga, H. (2000). Characterization of acid soils of Mizoram. *Journal of the Indian Society of Soil Science*, *48*(3), 437-446.

Moges, A., Dagnachew, M. and Yimer, F. (2013). Land use effects on soil quality indicators: a case study of Abo-Wonsho Southern Ethiopia. *Applied and Environmental Soil Science*, *2013*.

Moges, A., Dagnachew, M. and Yimer, F. (2013). Land use effects on soil quality indicators: A case study of Abo-Wonsho Southern Ethiopia. *Applied and Environmental Soil Science*. (2013). Hindawi Publishing corporation. 9p.

Mohanty, M., Sinha, K.K., Hati, M., Painuli, D.K. and Chaudhary, R.S. (2012). Stability of soil aggregatesunder different vegetation covers in a vertisol of Central India. *Journal of Agricultural Physics*, 12 (2), 133–142.

Molope, M.B. and Page, E.R. (1986). The contributions of fungi, bacteria and physical processes in the development of aggregate stability of a cultivated soil. In: Lopez-Real, J.M., Hodges, R.D. (Eds.), *The Role of Microorganisms in a Sustainable Agriculture*. A.B. Academic Publishers, Berkhamstead, UK.

Monreal, C.M. and Kodama, H. (1997). Influence of aggregate architecture and minerals on living habitats and soil organic matter. *Can J Soil Sci*, 77: 367-377.

Moraes, J.F.L., Volkoff, B., Cerri, C.C. and Bernoux, M. (1996). Soil properties under Amazon forest and changes due to pasture installation in Rondoˆnia, Brazil. *Geoderma*, 70 (1), 63–81.

Moser, S.C. (1996). A partial instructional module on global and regional land use/cover change: assessing the data and searching for general relationships. *GeoJournal*, 39(3), 241-283.

Munsell, A.H. (1905). *A Color Notation*. Boston: G. H. Ellis Co.

Murovhi, N.R., Materechera, S.A. and Mulugeta, S.D. (2012). Seasonal changes in litter fall and its quality from three sub-tropical fruit tree species at Nelspruit, South Africa. *Agroforestry Systems*, *86*(1), 61-71.

Murphy, H.F. (1968). A report on the fertility status and other data on some soils of Ethiopia, Experiment Station Bulletin No. 44, College of Agriculture Haile Sellassie I University, Dire Dawa, Ethiopia.

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

Nadagoudar, S.V. (2016). A critical analysis of law relating to biodiversity conservation and forest ecosystem management in India. In S. Nautiyal, R. Schaldach, K. V. Raju, H. Kächele, B. Pritchard, & K. S. Rao (Eds.), *Climate change challenge (3C) and social-economic- ecological interface-building* (pp. 121–130). Cham: Springer.

Nanda, T. (2005). Urban Sprawl and occupational change in Raipur City; Unpublished M.Sc. thesis, Dept of Geo, University of Delhi, Delhi.

Nath, A.J., Brahma, B., Sileshi, G.W., Das, A.K., (2018b). Impact of land use changes on thestorage of soil organic carbon in active and recalcitrant pools in a humid tropical region of India. *Science of the Total Environment*. 624, 908–917.

Ndour, N.Y.B., Achouak, W. and Christen, R. (2008). Characteristics of microbial habitats in a tropical soil subject to different fallow management. *Appl Environ Microbiol*, 38: 51-61.

Nega, E. and Heluf, G. (2009). Influence of land use changes and soil depth on cation exchange capacity and contents of exchangeable bases in the soils of Senbete watershed, Western Ethiopia. *Ethiopian Journal of Natural Resources,* 11(2), 195-206.

Nega, E. and Heluf, G. (2013). Effect of land use changes and soil depth on soil organic matter, total nitrogen and available phosphorus contents of soils in Senbat watershed, western Ethiopia. ARPN *Journal of Agricultural and Biological Science*, 8(3), 206-212.

Negasa, T., Ketema, H., Legesse, A., Sisay, M. and Temesgen, H. (2017). Variation in soil properties under different land use types managed by smallholder farmers along the toposequence in southern Ethiopia. *Geoderma*, 290, 40–50.

Neumann, D., Heuer, A., Hemkemeyer, M., Martens, R. and Tebbe, C.C. (2013) Response of microbial communities to long-term fertilization depends on their microhabitat. FEMS *Microbiolology Ecology,* 86: 71-84.

Ney, M., Graf, A., Bogena, H., Diekkrüger, B., Drüe, C., Esser, O., Heinemann, G., Klosterhalfen, A., Pick, K., Pütz, T., Schmidt, M., Valler, V. andVereecken, H. (2019). CO2 fluxes before and after partial deforestation of a Central European spruce forest. *Agriculture and Forest Meteorology,* 274: 61–74.

Ng, C., Xie, J. and Yu, J. (2011). Measuring the spatio-temporal variation of habitat isolation due to rapid urbanization: a case study of Shenzen River cross-boundary catchment, China. *Landscape and Urban Planning*, 103:44–54.

Nicolson, L.D. (1987). The Greening of the cities; Routledge and Kegan Paul, London.

Nkana, J.V., Demeyer, A. and Verloo, M.G. (1998). Chemical effects of wood ash on plant growth in tropical acid soils. *Bioresource Technology*, *63*(3), 251-260.

O'Brien, S.L. and Jastrow, J.D. (2013). Physical and chemical protection in hierarchical soil aggregates regulates soil carbon and nitrogen recovery in restored perennial grasslands. *Soil Biology and Biochemistry*, 61, 1–13.

Offiong, R.A., Atu, J.E., Njar, G.N. and Iwara, A.I. (2009). Effects of land use change on soil physicochemical properties in a South-Southern Nigeria. *African Journal of Environmental Pollution and Health*, 7, 47-51.

Okonkwo, C.I. (2010). Effect of Burning and Cultivation on Soil Properties and Microbial Population of Four Different Land Use Systems in Abakaliki. *Research Journal of Agriculture and Biological Sciences*.6 (6): 1007-1014.

Oldeman, L.R. (1998). Soil Degradation: A Threat to Food Security. International Soil Reference and Information Center, Wageningen.

Onweremadu, E.U., Onyia, V.N. and Anikwe, M.A.N. (2007). Carbon and nitrogen distribution in water-stable aggregates under two tillage techniques in Fluvisols of Owerri area, southeastern Nigeria. *Soil Tillage Research*. 97, 195–206.

Osman, K.T. (2012). Soils: Principles, Properties and Management. Springer, New York. 271p.

Ovung, E.Y., Kumar, K.S., Manpoong, C., Tripathi, S.K., Khanduri, V.P. and Singh, S.K. (2020). Influence of land-use pattern on soil quality in a steeply sloped tropical mountainous region India. *Archives of Agronomy and Soil Science*, DOI: 10.1080/03650340.2020.1858478.

Pacheco, F.A.L., Sanches Fernandes, L.F., Valle Junior, R.F., Pissarra, T.C.T. and Valera, C.A. (2018). Land degradation: multiple environmental consequences and routes to neutrality. *Current Opinion in Environmental Science & Health*, 5, 79–86.

Paniagua, A., Kammerbauer, J., Avedillo, M. and Adrews, A.M. (1999). Relationship of soil characteristics to vegetation successions on a sequence of degraded and rehabilitated soils in Honduras. *Agriculture, Ecosystems & Environment*, 72, 215-225.

Pardini, G., Gispert, M. and Dunjó, G. (2003). Runoff erosion and nutrient depletion in five Mediterranean soils of NE Spain under different land use. *Science of the Total Environment*. 309,213–224.

Patel, S.K., Verma, P. and Singh, G.S. (2019). Agricultural growth and land use land cover change in peri-urban India. *Environmental monitoring and assessment*, *191*(9), 1-17.

Peech, M. (1965). Hydrogen-ion activity. In: Black, C.A. (Ed.), Methods of Soil Analysis, Part 2, Chemical and Microbiological Properties Vol 9. *American Society for Microbiology*, Agron, Madison, Wisconsin, pp. 914–925.

Peixoto, R.S., Coutinho, H.L.C. and Madari, B. (2006). Soil aggregation and bacterial community structure as affected by tillage and cover cropping in the Brazilian Cerrados. *Soil Till Res*, 90: 16-28.

Peng, X., Yan, X., Zhou, H., Zhang, Y.Z. and Sun, H. (2015). Assessing the contributions of sesquioxides and soil organic matter to aggregation in an Ultisol under long-term fertilization. *Soil Tillage Research*, 146, 89–98.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R. and Blair, R. (1995). Environmental and economic costs of soil erosion and conservation. *Science,* 267, 1117–1123.

Pingali, P.L. (2012). Green Revolution: Impacts, limits, and the path ahead. Proceedings of the National Academy of Sciences of the United States of America, 109(31), 12302–12308.

Piper, C.S. (2005). Soil and Plant Analysis. New Delhi: Srishti Book Distributors, 368 pp.

Poorter, L.B., Ongers, F. and Aide, T.M. (2016). Biomass resilience of Neotropical secondary forests. *Nature*, 530, 211–214.

Post, W.M. and Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential, *Global Change Biology*, vol. 6, no. 3, pp. 317–327.

Prakasam, C. (2010). Land use and land cover change detection through remote sensing approach: A case study of Kodaikanal taluk, Tamil nadu. *International journal of Geomatics and Geosciences*, *1*(2), 150.

Pricope, N.G., Husak, G., Lopez-Carr, D., Funk, C., Michaelsen, J. (2013). The climate population nexus in the East African Horn: emerging degradation trends in rangeland and pastoral livelihood zones. *Global Environmental Change*, 23, 1525–1541.

Puget, P., Angers, D.A. and Chenu, C. (1995). Total and young organic matter distributions in aggregates of silty cultivated soils. *European Journal of Soil Science*, 46:49–59.

Qi, M.M., Li, J., Lin, Z., Li, Z.J., Li, Y.T., Yang, X.D. and Zhao, B.Q. (2016). Temperature effects on soil organic carbon, soil labile organic carbon fractions, and soil enzyme activities under long-term fertilization regimes. *Journal of Integrative Agriculture*, 15, 918–928.

Ramakrishnan, P.S. and Kushwaha, S.P.S. (2001). Secondary forests of the Himalaya with emphasis on the north-eastern hill region of India. *Journal of Tropical Forest Science*, 727-747.

Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M. and Rieseberg, L.H. (2018). Trends in global agricultural land use: Implications for environmental health and food security. *Annual Review of Plant Biology*, 69(1), 789–815.

Rasche, F., Hödl, V. and Poll, C. (2006). Rhizosphere bacteria affected by transgenic potatoes with antibacterial activities compared to the effects of soil, wild-type potatoes, vegetation stage and pathogen exposure. FEMS *Microbiol Ecol*, 56: 219-35.

Rathee, G. (2014). Trends of land-use change in India. In *Urbanization in Asia* (pp. 215-238). Springer, New Delhi.

Rawat, J.S. andKumar, M. (2015). Monitoring landuse/cover change using remote sensing and GIS techniques: a case study of Hawalbagh block, district Almora, Uttarakhand, India. *Egyptian Journal of Remote Sensing & Space Sciences*, 18, 77-84.

Riebsame, W.E., Meyer, W.B. and Turner, B.L. (1994). Modeling land use and cover as part of global environmental change. *Climatic change*, 28(1-2), 45-64.

Robertson, G.P., Sollins, P. and Ellis, B.G. (1999). Exchangeable ions, pH, and cation exchange capacity. *Standard soil methods for long-term ecological research*, Oxford University Press, New York, 106-114.

Robertson, W.K., Pope, P.E. and Tomlinson, R.T. (1974). Sampling tool for taking undegraded soil cores. *Soil Science Society of America Journal*, Proc. 38, 855–857.

Rockström, J., Williams, J., Daily, G., Noble, A., Matthews, N. and Gordon, L. (2017). Sustainable intensification of agriculture for human prosperity and global sustainability. *Ambio*, 46(1), 4–17.

Rode, M.W. (1995). Aboveground nutrient cycling and forest development on poor sandy soil. *Plant and Soil*. 168/169, 337–343.

Rolando, J.L., Dubeux, J., Ramirez, D. and Moreno, M.R. (2018). Land use effects on soilfertility and nutrient cycling in the Peruvian High-Andean Puna grasslands. *Soil Science Society of America Journal*, 82 (2), 463–474.

Roose, E. and Barthes, B. (2001). Organic matter management for soil conservation and productivity restoration in Africa: a contribution from francophone research. *Nutrient Cycling in Agroecosystems*, vol. 61, pp. 159–170.

Roy, P., Roy, A., Joshi, P., Kale, M., Srivastava, V. and Srivastava, S. (2015). Development of Decadal (1985–1995–2005) Land use and land cover database for India. *Remote Sensing*, 7(3), 2401–2430.

Sadeghi, S.H.R., Vangah, B.G. and Safaeelan, N.A. (2007). Comparison between effects of open grazing and manual harvesting of cultivated summer rangelands of Northern Iran on infiltration, runoff and sediment yield. *Land Degradation & Development*, 18: 608-620.

Safar, N.V.H., Magnago, L.F.S., Rolim, S.G. and Schaefer, C.E.G.R. (2019). Atlantic Forest topsoil nutrients can be resistant to disturbance and forest clearing. *Biotropica*, *51*(3), 342-354.

Saggar, S., Yeats, G.W. and Shepherd, T.G. (2001). Cultivation effect on soil biological properties, micorfauna and organic matter dynamics in Eutric Glaysol and Gleyic Luvison soils in New Zealand. *Soil and Tillage Research*, 58: 55-68.

Saha, S., Chakraborty, D., Choudhury, B.U. (2015). Spatial variability in temporal trends of precipitation and its impact on the agricultural scenario of Mizoram. *Current Science*, 2278-2282.

Saha, S.K., Nair, P.K., Nair, R., Kumar, V.D. and Mohan, B. (2010). Carbon storage in relation to soil size-fractions under tree-based land-use systems. *Plant and Soil*, 328:433–446.

Sainju, U.M. (2006). Carbon and nitrogen pools in soil aggregates separated by dry and wet sieving methods. *Soil Science.* 171, 937–949.

Salcedo, I.H. (2008). Fertilidade do Solo e Produção de Biomassa no Semi-Árido, 1st ed.; UFPE: Recife, Brazil, p. 291

Saravanan, S., Jennifer Jacinth, J., Singh, L., Saranya, T. andSivaranjani, S. (2019). Impact of land-use change on soil erosion in the Coonoor Watershed, Nilgiris Mountain Range, Tamil Nadu, India. In: ElAskary H, Lee S, Heggy E, Pradhan B (eds) Advances in remote sensing and geo informatics applications. *Advances in Science, Technology & Innovation (IEREK Interdisciplinary Series for Sustainable Development)*. Springer, Cham

Sarkar, U.K., Saha, B.K., Goswami, C. and Chowdhury, M.A.H. (2010). Leaf litter amendment in forest soil and their effect on the yield quality of red amaranth. *Journal of Bangladesh Agriculture Univeristy*, 8(2): 221226.

Sarker, J.R., Singh, B.P., Cowie, A.L., Fang, Y., Collins, D., Badgery, W. and Dalal, R.C. (2018). Agricultural management practices impacted carbon and nutrient concentrations in soil aggregates, with minimal influence on aggregate stability and total carbon and nutrient stocks in contrasting soils. *Soil and Tillage Research*, *178*, 209-223.

Sarker, J.R., Singh, B.P., Cowie, A.L., Fang, Y., Collins, D., Dougherty, W.J. and Singh, B.K. (2018b). Carbon and nutrient mineralisation dynamics in aggregate-size classes from different tillage systems after input of canola and wheat residues. *Soil Biol. Biochem*. 116, 22–38.

Scolozzi, R. and Geneletti, D. (2012). A multi-scale qualitative approach to assess the impact of urbanization on natural habitats and their connectivity. *Environmental Impact Assessment*, Rev 36:9–22.

Selassie, Y.G. and Ayanna, G. (2013). Effects of different land use systems on selected physico-chemical properties of soils in Northwestern Ethiopia. *Journal of Agricultural Science*, 5(4): 112-120.

Sharma, B. P. (2016). Present position of agriculture in India. *International Journal of Science and Research*, (IJSR), 5(4), 240–243.

Sharma, R., Ghosh, A. and Joshi, P. (2013).Spatio-temporal footprints of urbanization in Surat, the Diamond City of India (1990–2009). *Environmental Monitoring and Assessment*, 185:3313–3325

Silva, A.P., Babujia, L., Matsumoto, M.F., Guimarães, M.F. and Hungria, M. (2013). Bacterial diversity under different tillage and crop rotation systems in an oxisol of Southern Brazil. *Embrapa Soja-Artigo em periódico indexado (ALICE)*.

Singh, A.N., Raghubanshi, A.S. and Singh, J.S. (2002). Plantations as a tool for mine spoil restoration.*Current Science*, 82 (12), 1436–1441.

Singh, D. (1996). *The last frontier: people and forests in Mizoram*. Tata Energy Research Institute.

Singh, M.K. and Ghoshal, N. (2014). Variation in soil microbial biomass in the dry tropics: impact of land-use change. *Soil Research*, *52*(3), 299-306.

Singh, M.K., Singh, S. and Ghoshal, N. (2017). Impact of land use change on soil aggregate dynamics in the dry tropics. *Restoration Ecology*. 25 (6), 962–971.

Singh, P., Heikkinen, J., Ketoja, E., Nuutinen, V., Palojärvi, A., Sheehy, J., Esala, M., Mitra, S., Alakukku, L. and Regina, K. (2015). Tillage and crop residue management methods had minor effects on the stock and stabilization of top soil carbon in a 30-year field experiment. *Science of the Total Environment*, 518, 337- 344.

Singha, D. and Tripathi, S.K. (2017). Variations in fine root growth during age chronosequence of moist tropical forest following shifting cultivation in Mizoram, Northeast India. *Tropical Ecology*, 58(4), 769-779.

Sinsabaugh, R.S. (1994). Enzymic analysis of microbial pattern and process. *Biology and Fertility of Soils*, *17*(1), 69-74.

Six, J., Bossuyt, H., Degryze, S. and Denef, K. (2004). A history of research on the link between (micro) aggregates, soil biota, and soil organic matter dynamics. *Soil & Tillage Research*, 79, 7-31.

Six, J., Elliott, E.T. and Paustian, K. (2000). Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. Soil Biology and Biochemistry. 32, 2099–2103.

Six, J., Elliott, E.T. and Paustian, K. (1999). Aggregate and soil organic matter dynamics under conventional and no-tillage systems. *Soil Science Society of America Journal*, 63:350–358.

Six, J., Elliott, E.T., Paustian, K. and Doran, J.W. (1998). Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Science Society of America Journal*, 62:1367–1377

Six, J., Frey, S.D., Thiet, R.K. and Batten, K.M. (2006). Bacterial and fungal contributions to carbon sequestration in agroecosystems. *Soil Science Society of America Journal*, 70, 555-569.

Six, J., Frey, S.D., Thiet, R.K. and Batten, K.M. (2006). Bacterial and fungal contributions to carbon sequestration in agroecosystems. *Soil Science Society of America Journal*, *70*(2), 555-569.

Six, J., Paustian, K., Elliott, E.T. and Combrink, C. (2000). Soil structure and organic matter. I. Distribution of aggregate-size classes and aggregate-associated carbon. *Soil Science Society of America Journal*, 64:681–689.

Somasundaram, J., Reeves, S., Wang, W., Heenan, M. and Dalal, R. (2017). Impact of 47 years of no tillage and stubble retention on soil aggregation and carbon distribution in a Vertisol. *Land Degradation & Development*, 28, 1589-1602.

Somasundaram, J., Reeves, S., Wang, W., Heenan, M. and Dalal, R. (2017). Impact of 47 years of no tillage and stubble retention on soil aggregation and carbon distribution in a vertisol. *Land Degradation & Development*, *28*(5), 1589-1602.

Spechn, E. M., Joshi, J., Schmid, B., Alphei, J. and Korner, C. (2000). Plant diversity effects on soil heterotrophic activity in experimental grassland ecosystems. *Plant and Soil*, 224, 217-230.

Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B.L. and Lassaletta, L.(2018). Options for keeping the food system within environmental limits. *Nature*, 562(7728), 519–525.

Srivastava, P., Singh, R., Tripathi, S. and Singh Raghubanshi, A. (2016). An urgent need for sustainable thinking in agriculture, An Indian scenario. *Ecological Indicators*, 67, 611–622.

Stevenson, B.A., Sarmah, A.K., Smernik, R., Hunter, D.W.F. and Fraser, S. (2016). Soil carbon characterization and nutrient ratios across land uses on two contrasting soils: Their relationships to microbial biomass and function. *Soil Biology & Biochemistry*, 97, 50–62.

Stockmann, U., Adams, M.A., Crawford, J.W., Field, D.J., Henakaarchchi, N., Jenkins, M. and Wheeler, I. (2013). The knowns, known unknowns and unknowns of sequestration of soil organic carbon. *Agriculture, Ecosystems & Environment*, *164*, 80-99.

Sun, B., Zhou, S. and Zhao, Q. (2003). Evaluation of spatial and temporal changes of soil quality based on geostatistical analysis in the hill region of subtropical China. *Geoderma*, 115, 85–99.

Swaminathan, M.S. and Bhavani, R.V. (2013). Food production & availability, Essential prerequisites for sustainable food security. *The Indian Journal of Medical Research*, 138(3), 383–391.

Taboada, M.A., Barbosa, O.A. and Cosentino, D.J. (2008). Null creation of air-filled structural pores by soil cracking and shrinkage in silty loamy soils. *Soil Science*, *173*(2), 130-142.

Takele, L., Chimdi, A. and Abebaw, A. (2014). Dynamics of Soil fertility as ınfluenced by different land use systems and soil depth in West Showa Zone, Gindeberet District, Ethiopia. *Agriculture, Forestry and Fisheries*, 3(6): 489-494.

Tan, M., Li, X., Xie, H. and Lu, C. (2005). Urban land expansion and arable land loss in China—a case study of Beijing–Tianjin–Hebei region. *Land use policy*, *22*(3), 187-196.

Tegene, B. (2000). Characteristics and classification of soils of Gora Daget Forest, South Wollo highlands, Ethiopia. SINET: *Ethiopian Journal of Science*, 23 (1), 35–51.

Tejada, M., Hernandez, M.T. and Garcia, C. (2009). Soil restoration using composted plant residues: effects on soil properties. *Soil & Tillage Research*, 102, 109-117.

Tekalign, M., Richter, C. and Heiligtag, B. (2002). Phosphorus availability studies on ten Ethiopian Vertisols. *Journal of Agriculture and Rural Development in the Tropics and Subtropics*, 103(2), 177-183.

Tekle, K. and Hedlund, L. (2000). Land cover changes between 1958 and 1986 in Kalu District, southern Wello, Ethiopia. *Mountain research and development, 20*(1), 42-51.

Teluguntla, P.G., Thenkabail, P.S., Xiong, J.N., Gumma, M.K., Giri, C. and Milesi, C. (2015). Global Cropland Area Database (GCAD) derived from remote sensing in support of food security in the twenty-first century: Current achievements and future possibilities. In P.S. Thenkabail (Ed.), *Remote sensing handbook*. Volume II, Land resources monitoring, modeling, and mapping with remote sensing (Chap. 7, p. 849), Chapter 7. Boca Raton: CRC Press.

Theobald, D.M. (2004). Placing exurban land-use change in a human modification framework. *Frontiers in Ecology and the Environment*, *2*(3), 139-144.

Tian, H., Banger, K., Bo, T. and Dadhwal, V.K. (2014). History of land use in India during 1880-2010: Large-scale land transformations reconstructed from satellite data and historical archives. *Global and Planetary Change*, 121, 78–88.

Tiemann, L.K., and Grandy, A.S. (2015). Mechanisms of soil carbon accrual and storage in bioenergy cropping systems. *GCB Bioenergy*, 7: 161-174.

Tiemann, L.K., Grandy, A.S., Atkinson, E.E., Marin-Spiotta, E. and McDaniel, M.D. (2015).Crop rotational diversity enhances belowground communities and functions in an agroecosystem. *Ecology Letters*, 8, 761e771.

Tilman, D. and Clark, M. (2014). Global diets link environmental sustainability and human health. *Nature*, 515(7528), 518–522.

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S. and Packer, C. (2017). Future threats to biodiversity and pathways to their prevention. *Nature*, 546(7656), 73–81.

Tisdall, J.M. and Oades, J.M. (2012). Landmark Papers: No. 1. Organic matter and water-stable aggregates in soils. *European Journal of Soil Science*, 63, 8-21.

Tisdall, J.M. and Oades, J.M. (1982). Organic matter and water-stable aggregates in soils. *Journal of Soil Science*, 62:141–163.

Tisdall, J.M. and Oades, J.M., (1982). Organic matter and water-stable aggregates in soils. *Journal of Soil Science*. 33, 141–163.

Treonis, A.M., Austin, E.E. and Buyer, J.S. (2010). Effects of organic amendment and tillage on soil microorganisms and microfauna. *Appl Soil Ecol*, 46: 103-10.

Tripathi, S.K. and Singh, K.P. (1994). Productivity and nutrient cycling in recently harvested and mature bamboo savannas in the dry tropics. *Journal of applied ecology*, 109-124.

Tripathi, S.K., Kushwaha, C.P. and Basu, S.K. (2012). Application of fractal theory in assessing soil aggregates in Indian tropical ecosystems. *Journal of Forest Research*, 23, 355-364.

Tripathi, S.K., Kushwaha, C.P. and Singh, K.P. (2008). Tropical forest and savanna ecosystems show differential impact of N and P additions on soil organic matter and aggregates structure. *Global Change Biology*, 14, 2572-2581.

Tripathi, S.K., Singh, K.P. and Singh, P.K. (1999). Temporal changes in spatial pattern of fine-root mass and nutrient concentrations in Indian bamboo savanna. *Applied Vegetation Science*, *2*(2), 229-238.

Tripathi, S.K., Vanlalfakawma, D.C. and Lalnunmawia, F. (2017). Shifting cultivation on steep slopes of Mizoram, India: Impact of policy reforms. In: Cairns M. (ed.), *Shifting Cultivation Policies: Balancing Environmental and Social Sustainability*: CABI Publishing, London. (In press).

Trivedi, P., Anderson, I.C. and Singh, B.K. (2013). Microbial modulators of soil carbon storage: integrating genomic and metabolic knowledge for global prediction. *Trends in Microbiology*, 21: 641-651.

Trivedi, P., Rochester, I.J., Trivedi, C., Van Nostrand, J.D., Zhou, J., Karunaratne, S., Anderson, I.C. and Singh, B.K. (2015). Soil aggregate size mediates the impacts of cropping regimes on soil carbon and microbial communities. *Soil Biology and Biochemistry*, 91: 169-181.

Tufa, M., Melese, A. and Tena, W. (2019). Effects of land use types on selected soil physical and chemical properties: The case of Kuyu District, Ethiopia. *Eurasian Journal of Soil Science*, *8*(2), 94-109.

Tully, K., Sullivan, C., Weil, R. and Sanchez, P. (2015). The state of soil degradation in Sub-Saharan Africa: baselines, trajectories, and solutions. *Sustainability*, 7, 6523–6552.

Ufot, U.O., Iren, O.B. and Chikere Njoku, C.U. (2016). Effects of land use on soil physical and chemical properties in Akokwa area of Imo State, Nigeria. *International Journal of Life Sciences Scientific Research*, 2(3): 273-278.

UNEP, (2004). Annual report. United Nations Organization.

United Nations, (2018). World Urbanization Prospects 2018 Highlights. Department of Economic and Social Afairs Population Division, United Nations, NY.

UN-Pop, (2017). World Population Prospects: The 2017 Revision, Online Demographic Profiles. Retrieved November 26, 2018.

UNWUP, (2010). URL: http://data.worldbank.org/indicator/SP.URB.TOT.

Valera, C.A., Valle Junior, R.F., Varandas, S.G.P., Sanches Fernandes, L.F. and Pacheco, F.A.L. (2016). The role of environmental land use conflicts in soil fertility: a study onthe Uberaba River basin, Brazil. *Science of the Total Environment*, 562, 463–473.

Valle Junior, R.F., Varandas, S.G.P., Sanches Fernandes, L.F. and Pacheco, F.A.L. (2014).Environmental land use conflicts: a threat to soil conservation. *Land Use Policy*, 41,172–185.

van Straaten, O., Corre, M.D., Wolf, K., Tchienkoua, M., Cuellar, E., Matthews, R.B. and Veldkamp, E. (2015). Conversion of lowland tropical forests to tree cash crop plantations loses up to one-half of stored soil organic carbon. *Proceedings of the National Academy of Sciences*, 112 (32), 9956-9960.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. and Melillo, J.M. (1997). Human domination of Earth's ecosystems. *Science*, *277*(5325), 494-499.

Wakelin, S.A., Macdonald, L.M. and Rogersa, S.L. (2008). Habitat selective factors influencing the structural composition and functional capacity of microbial communities in agricultural soils. *Soil Biol Biochem*, 40: 803-13.

Wani, F.S., Akhter, F., Mir, S., Baba, Z.A., Maqbool, S., Zargar, M.Y., and Nabi, S.U. (2018). Assessment of Soil Microbial Status under Different Land Use Systems in North Western Zone of Kashmir. *International Journal of Current Microbiology and Applied Sci*ence, *7*(8), 266-279.

Wapongnungsang, (2018). Impact of fallow periods and treatments on the pattern of recovery of soil fertility and plant productivity in shifting cultivation sites in aizawl district, Mizoram. Ph.D Thesis.

Wapongnungsang., Hauchhum, R. and Tripathi, S.K. (2017). Litter decomposition Vis-a-Vis carbon and nitrogen dynamics of Tephrosia candida components in different fallow periods following shifting cultivation in Mizoram. *Indian Journal of Ecology*, 44(4), 791-796.

Wapongnungsang., Lalnunzira, C. and Tripathi, S.K. (2019). Dynamics of litter carbon and nitrogen in forest fallows following shifting cultivation in Mizoram, Northeast India. *International Journal of Plant and Environment*, 4, 55-63.

Wapongnungsang., Manpoong, C. and Tripathi, S.K. (2018). Changes in soil fertility and rice productivity in three consecutive years cropping under different fallow phases following Shifting cultivation. *International Journal of Physical and Social Science*, 25 (6):1-10.

Wardle, D.A., Bardgett, R.D., Klironomos, J.N., Setälä, H., Van Der Putten, W.H. and Wall, D.H. (2004). Ecological linkages between aboveground and belowground biota. *Science,* 304: 1629-1633.

Weisse, M. and Goldman. E.D. (2018). 2017 Was the Second-Worst Year on Record for Tropical Tree Cover Loss. World Resources Institute. Insights (blog), June 26.

Willy, D.K., Muyanga, M. and Jayne, T. (2019). Can economic and environmental benefits associated with agricultural intensification be sustained at high population densities? A farm level empirical analysis. *Land use policy*, *81*, 100-110.

Wilson, G.W., Rice, C.W., Rillig, M.C., Springer, A. and Hartnett, D.C. (2009). Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular mycorrhizal fungi: results from long-term field experiments. *Ecology Letters*, 12: 452-461.

Wøien, H. (1995). Deforestation, information and citations. *GeoJournal*, *37*(4), 501-511.

Woldeamlak, B. and Stroosnijder, L. (2003). Effects of agroecological land use succession on soil properties in the Chemoga watershed, Blue Nile basin, Ethiopia. *Geoderma*. 111: 85-98.

Wu, Q. (2011). Environmental edaphology [M]. Beijing: Chinese Agricultural Press: 82-83. (In Chinese).

Xiangmin, F., Qingli, W., Wangming, Z., Wei, Z., Yawei, W., Lijun, N. and Limin, D. (2014). Land use effects on soil organic carbon, microbial biomass and microbial activity in Changbai Mountains of Northeast China. *Chinese Geographical Science*, 24(3):297306.

Xiao, C., Janssens, I.A., Liu, P., Zhou, Z. and Sun, O.J. (2007) Irrigation and enhanced soil carbon input effects on below-ground carbon cycling in semiarid temperate grasslands. *New Phytololgist*, 174: 835-846.

Xu, Z.H., Ward, S., Chen, C.R., Blumfield, T., Prasolova, N. and Liu, J. (2008). Soil carbon and nutrient pools, microbial properties and gross nitrogen transformations in adjacent natural forest and hoop pine plantations of subtropical Australia. *Journal of Soils and Sediments*, 8, 99-105.

Yan, Y., Tian, J., Fan, M., Zhang F., Li, A., Christie, P., Chen, H., Lee, J., Kuzyakov, Y. and Six, J. (2012). Soil organic carbon and total nitrogen in intensively managed arable soils. *Agriculture, Ecosystems & Environment*, 150, 102-110.

Yang, Z., Singh, B.R., Hansen, S., Hu, Z. and Riley, H. (2007). Aggregate associated sulfur fractions in long-term (> 80 years) fertilized soils. *Soil Science Society of America Journal*, 71, 163–170.

Yihenew, G. and Getachew, A. (2013). Effects of different land use systems on selected physicochemical properties of soils in Northwestern Ethiopia. *Journal of Agricultural Science*, 5(4), 112-120.

Yimer, F., Ledin, S. and Abdulakdir, A. (2008). Concentrations of exchangeable bases and cation exchange capacity in soils of cropland, grazing and forest in the Bale Mountains, Ethiopia. *Forest Ecology and Management*, 256, 1298–1302.

Yimer, F., Ledin, S. and Abdelkadir, A. (2008). Concentrations of exchangeable bases and cation exchange capacity in soils of cropland, grazing and forest in the Bale Mountains, Ethiopia, *Forest Ecology and Management*, vol. 256, no. 6, pp.1298–1302.

Yin, C., Jones, K.L. and Peterson, D.E. (2010). Members of soil bacterial communities sensitive to tillage and crop rotation. *Soil Biol Biochem*, 42: 2111-8.

Yitbarek, T., Gebrekidan, H., Kibret, K. andBeyene, S.(2013). Impacts of Land Use on Selected Physicochemical Properties of Soils of Abobo Area, Western Ethiopia. *Agriculture, Forestry and Fisheries*, 2(5): 177-183.

Yoder, R.E. (1936). A direct method of aggregate analysis of soils and a study of the physical nature of erosion losses. *Journal of American Society of Agronomy*. 28, 337–351.

Yu, M., Zhang, L., Xu, X., Feger, K.H., Wang, Y., Liu, W. and Schwärzel, K. (2015). Impact of land-use changes on soil hydraulic properties of Calcaric Regosols on the Loess Plateau, NW China. *Journal of Plant Nutrition and Soil Science*, *178*(3), 486-498.

Yunan, D., Xianliang, Q.and Xiaochen, W. (2018). Study on Cation Exchange Capacity of Agricultural Soils. IOP Conf. Ser.: Mater. *Sci. Eng*. 392 042039.

Zhang, Q., Wu, J.J., Lei, Y., Yang, F., Zhang, D.D., Zhang, K.R. and Cheng, X.L. (2018). Agricultural land use change impacts soil CO2 emission and its 13C-isotopic signature in central China. *Soil and Tillage Research*, 177, 105–112.

Zhang, Y., Bhattacharyya, R., Dalal, R C., Wang, P., Menzies, N.W. and Kopittke, P.M. (2020). Impact of land use change and soil type on total phosphorus and its fractions in soil aggregates. *Land Degradation & Development*, *31*(7), 828-841.

Zhao, G.X., Lin, G. and Warner, T. (2004). Using Thematic Mapper data for change detection and sustainable use of cultivated land: a case study in the Yellow River delta, China. *International Journal of Remote Sensing*, 25(13):2509–2522.

Zhao, S., Da, L., Tang, Z., Fang, H., Song, K. and Fang, J. (2006). Ecological consequences of rapid urban expansion: Shanghai, China. *Frontiers in Ecology and the Environment*, *4*(7), 341-346.

Zheng, F., He, X., Gao, X., Zhang, C. and Tang, K. (2005). Effects of erosion patterns on nutrient loss following deforestation on the Loess Plateau of China. *Agriculture, Ecosystem and Environment*. 108, 85–97.

Zhong, Z., Han, X., Xu, Y., Zhang, W., Fu, S., Liu, W., Ren, C., Yang, G. and Ren, G. (2019). Effects of land use change on organic carbon dynamics associated with soil aggregate fractions on the Loess Plateau. China. *Land Degradation and Development*, 30 (1070), 1082.

Zothansanga, D. and Bobby, B. (2019). The New Land Use Policy: A Panacea for Shifting Cultivation in Mizoram. *Senhri Journal of Multidisciplinary Studies*, *4*, 21-29.

PHOTO-PLATES


Natural forest (NAF)


Current *Jhum* (CJ)


>12 year old *Jhum* fallow (JF)


***Acacia pennata* Plantation (AP)**


Home garden (HG)

Photo plate 1: Different land use systems (study sites).

Photo plate 2: Collection of soil samples (bulk soil and aggregate samples).

Photo plate 3: Laboratory analysis (wet sieving for analysis of aggregate stability).

Photo plate 4: Laboratory analysis.

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ABSTRACT

CHANGES IN SOIL AGGREGATION, MICROBIAL POPULATION, CARBON AND NUTRIENTS UNDER DIFFERENT LAND USE SYSTEMS OF MIZORAM

A THESIS SUBMITTED IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

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Background

The term land use is concerned with the intention for which the land is utilized and includes the human actions and impacts concerned with land. The transformation and modification of the Earth's terrestrial surface (Land) as a result of anthropogenic activities gradually came to be commonly known as land use change. Subsequently, the use of this land resource, referred to as land use, varies with the purpose to which it is being utilized, for example, cultivation, shelter, mining, recreational activities etc. Land use patterns can therefore be indicated as the interaction between anthropogenic activities and the ecosystems, and their exploitation in temporal and spatial scales. Land use change has varied impact on the environment and thus, the identification of issues related to land use change and finding alternatives to tackle it has attracted major attention of ecologists and environmentalists. More specifically, the impact of land use change on environment, commodities and services are of major concern that includes the impact on biodiversity, soil health and atmosphere.

North eastern region of India is one of the most diverse regions in the country in terms of flora and fauna and falls among one of the world's biodiversity hotspots. The North eastern region of India extends to about 2,61000 km^2 with diverse physiographic features and biodiversity, well supported by favourable climatic conditions specifically elevated rainfall supporting higher vegetation both below and above ground is a distinct place in India. However, over the last three decades, the forest cover has changed drastically as a result of people relocating to different places, increased clearing of forest, varied farming practices, expansion of agriculture and industries induced by population growth and elevated economic activities. These led to massive modification of forest areas into short term agricultural practice largely attributable to shifting cultivation (*Jhum*) coupled with conversion of native forests and grasslands into horticulture and plantation areas which led to enormous loss of vegetation in the region and increased loss in soil fertility. The incineration of the accumulated phytomass at 8.5 million tonnes each year during shifting cultivation and extensive cultivation on steep slopes under heavy and erratic rainfall conditions has accelerated soil erosion greater than 88 Mg ha-1 year-1 with an estimated annual loss of 6.0 million tonnes of soil organic carbon and other available nutrients. This indicates that land use change has a colossal impact on the soil fertility and subsequently the sustainable agricultural production practices. As the problem persists, accepting the adverse impact of land use change patterns and its interactions between anthropogenic practices and environment on soil fertility and crop productivity are the most essential challenges that is needed to be considered at present for achieving the goals of sustainability through land management.

Similar to other north eastern states, ecosystems in Mizoram are characterized by steep slopes, undulating terrain and high occurrence of rainfall each year amounting to more than 2500 mm which varies annually. About 70% of the total geographical area of the state (~21,081 km2) is sloped at angles higher than 33°. Substantial increase in the practice of shifting cultivation under such environmental features has affected the soil fertility as a result of run-off and erosion leading to uncontrolled loss of soil nutrients. The chief causes of land and soil degradation in the state are a combined impact of cultivation on steep and fragile soils with minimum soil conservation measures or vegetation cover, decreased fallow period, poor recycling of plant and animal residues, erratic and erosive rainfall patterns and forest deterioration.

Therefore, analysis of aspects that contributes to soil health enhancement in specific landscape are decisive and can only be achieved through the adoption of sustainable practices and ideal land use system which will provide both economic prosperity and ecological sustainability. As soil productivity and sustainability depends on dynamic equilibrium among its physical, chemical and biological properties, an assessment of soil properties in relation to the existing land use acts as potential tool to provide information on the nutrient storage in semi-natural and cultivated ecosystems. These properties are continuously influenced by land use systems and practices with profound influence on soil properties and thus, help in monitoring and restoration of soil health. Without the proper maintenance of soil fertility, it is impossible to achieve increment of agricultural production in feeding the ever increasing human population.

Soil fertility maintenance is the key to optimize, sustained and sufficient crop production as well as in overcoming undesirable climate change in the long-run. Henceforth, it is paramount to assess the influence of land use types on soil health for evolving sustainable land use practices and policy frameworks. The present study was therefore, initiated with a motive to quantify the status of soil physico-chemical and biological characteristics under various prominent land use systems and assess its impact on soil fertility and stability in Mizoram.

Objectives

Keeping in view the priorities discussed above, the present study was instituted to achieve the following major objectives:

1. To assess soil physico-chemical properties under different land use systems.
2. To determine water stable soil aggregates and their associated chemical characteristics under different land use systems.
3. To estimate soil microbial population under different land use systems, and to relate soil microbial population with aggregates sizes (micro- and macro-aggregates).

Methods

Study sites

Five different land use systems were selected as experimental sites from Aizawl and Mamit district namely *Acacia pennata* plantation (AP), Home garden, (HG) Current Jhum (CJ), *Jhum* fallow (JF) >12 years and Natural forest (NAF) which served as control, against which the differences were compared. The ages of the lands sustaining the particular land use system were identified through interviews and interactions from the associated local farmers and land owners. The five different land use systems varied in their vegetative composition as well as management practices and hence, the assumption that the differences in soil properties would be attributed to the type of its land use, management practices and vegetation cover.

Experimental design

Initially, for each land use system, three replicated sites measuring at least 20 x 25 m with similar topography, soil and vegetation were identified and established to signify true representatives of each land use type. All soil samples were collected twice consecutively during the year 2017 and 2018 from all sites in the month of May and June.

Sample collection

Soil samples for physical and biochemical analysis were collected randomly from each site replicate by inserting a 10 cm scaled soil corer having an inner diameter of 5.2 cm. Soil samples were then drawn at three different depths i.e., 0-10 cm, 10-20 cm and 20-30 cm depth. Within each site replicate, 2 composited soil samples were collected for each depth where a single composite sample was composed of 3 random soil cores. Thus, a total of 90 samples (2 composite samples x 3 depths x 3 site replicates x 5 land use systems) were collected for the study. In case of bulk density, soil samples were collected using the same soil corer and a total of 90 samples (2 soil cores x 3 depths x 3 site replicates x 5 land use types) were collected for the analysis.

Soil samples for aggregate stability analysis were collected using a large knife with a smooth surface and a specially modified machete for easy slicing and to avoid any compression and disturbance of the soil and to minimize the risk of compaction. A soil cake measuring about 1 square feet in size was excavated using the specially prepared tools up to 30 cm soil depth. The soil cakes were then carefully broken by hand along their natural plane of weakness and the part of the soil not contacted by the tools were labelled and stored in hard plastic containers so as to avoid disturbance and mixing of samples. A total of 90 samples (2 aggregate samples x 3 depths x 3 site replicates x 5 land use systems) were collected for the analysis.

Laboratory analysis

Soil bulk density (BD) was estimated from a core sampler of known volume, after oven drying the samples at 105 °C till constant mass the dry weight was taken and calculated. Soil moisture content (SMC %) was estimated by gravimetric method. Soil colour was determined using a Munsell soil colour chart. Soil texture was established following the Buoyoucous hydrometer method and soil textural class was assigned using the USDA soil textural classification.

Water stable soil aggregates were estimated following the method of wet sieving. Eight different sieve sizes i.e., 4.00, 3.35, 2.80, 2.00, 1.40, 1.00, 0.50 and 0.212 mm were used to retain nine different size classes (>4.00 mm, 3.35 mm–4.00 mm, 2.80 mm–3.35 mm, 2.00 mm–2.80 mm, 1.40 mm–2.00 mm, 1.00 mm–1.40 mm, 0.50 mm–1.00 mm, 0.212 mm – 0.50 mm and <0.212 mm) using a mechanical wet sieve shaker developed by S.D. HARDSON CO. The soil from each sieve were weighed and grouped under three aggregate size fractions (>2.00 mm, 0.25–2 mm and < 0.25 mm) namely macro-aggregate, meso-aggregate and micro-aggregate. Soil pH was measured on the basis of the potentiometric principle in a 1:25 soil/water solution using a pH meter. Soil organic matter (SOM %) was estimated following the loss on ignition (LOI) method**.**

Soil total organic carbon (TOC) and Total Nitrogen (TN) was analyzed using CHNS/O Elemental Analyzer with autosampler and TCD detector –Euro Vector (Model: Euro EA3000, LECO). Air dried soil samples were extracted using Mehlich-I solution (0.05 M HCl + 0.025 M H_2SO_4) and analyzed for exchangeable nutrients (Pavail, Ca, Mg, Na, K and Mn) using Inductive coupled plasma Spectrometer (ICP-spectrometer, Model-6000 series, Thermo scientific). For the assessment of total nutrients like Ptotal, Ca, Mg, K, Na and Mn) air dry soil samples were digested using Aqua-regia (HNO_3 + 3 HCL) in a microwave oven which was then, diluted and filtered using Whatman filter paper. The aliquot samples were then analyzed using the Inductive coupled plasma Spectrometer (ICP-spectrometer, Model-6000 series, Thermo scientific).

Results

The studied soil parameters varied significantly ($p<0.05$) across the different land use systems and soil depths. Soil colour across the various land use systems ranged from brown to dark yellowish brown in the surface layer (0-10cm), while the deeper depths (10-20 & 20-30) were dominated by light yellowish brown to brownish yellow. Soil physical properties such as SMC, BD, sand and silt content were found to be significantly ($p<0.05$) affected by different land use systems, soil depths as well as an interaction of both factors. The soil BD values in the surface layer were in the order of: NAF < JF < CJ < AP < HG. Soil moisture content (SMC) was highest in JF in the surface layer (0-10 cm), while the least was recorded in AP. Soil textural class varied from sandy clay loam to loam in the surface layer (0-10 cm) and silty clay to clay loam in the deeper soil layers. Soil textural class ranged from sandy clay loam to clay loam in NAF, JF and CJ while in case of HG and AP, sandy clay loam and loamy soil were observed.

TOC and TN concentrations in soil was observed to be significantly ($p<0.05$) affected by different land use systems (L), soil depths (D) as well as their interactions (LxD). The TOC and TN values decreased significantly with increasing soil depth ($p<0.05$), however, an interesting trend of TN content in the soil of AP could be observed wherein, the highest TN content was recorded higher in the deeper soil depth (20-30 cm) than in the sub-surface soil layer (10-20 cm) with values 0.20% and 0.18% respectively. The highest values of TOC and TN with values were observed in surface layer (0-10 cm) of NAF. Among the various land use systems, the least contents of TOC and TN were reported from the 20-30 cm soil layer of HG. The C/N of soil was also estimated to be significantly ($p<0.05$) affected by land use systems and soil depths ($p<0.05$). C/N decreased with increasing depth in all the land use systems except in AP, where the highest C/N was reported from the sub-surface depth (10-20 cm) of the soil. The highest C/N was recorded from JF in all the soil depths, while the lowest was observed in deeper depths of CJ.

Land use systems (L) and soil depths (D) significantly ($p<0.05$) affected the concentration of C stock, wherein C stock declined with increasing soil depth. Similarly in case of N stock the highest sequestration was recorded in the surface soil layer (0-10 cm) in all the various land use systems. The highest soil C stock with respect to soil depth was estimated in the surface layer (0-10 cm) of NAF, whereas the lowest was observed at HG. Likewise, the highest N stock was recorded in NAF and the lowest in HG. Soil total stock (0-30 cm) of C and N were significantly affected ($p<0.05$) by various land use systems (L). The trend of total C storage in 0-30 cm soil layer among the various land use systems was in the order; NAF > JF > AP > CJ > HG. In case of N stock, CJ reported the highest N content followed by NAF, AP, JF and lowest in HG. Maximum decrease in soil C and N stock upto 0-30 cm soil depth relative to NAF was observed in HG.

Soil organic matter (SOM) was observed to be significantly ($p<0.05$) affected by different land use systems (L), soil depths (D) and an interaction of both factors (LxD).

The highest SOM was observed to be conserved in NAF followed by JF and the lowest was in HG. SOM content in soil followed a decreasing trend with increasing soil depths in all the land use systems.

Generally, soil pH was acidic in all the land use systems and values decreased with increasing depth. It was highest in CJ (5.4) and the lowest in AP (4.7) at the surface soil layer. P_{avail} concentrations in soil was also found to be significantly affected by different land use systems (L), soil depths (D) as well as an interaction (LxD) of both the factors ($p<0.05$). The highest P_{avail} concentrations at the surface layer (0-10 cm) were reported from HG followed by JF and the lowest in AP. In case of Mg, K and Mn, the highest concentration was recorded from JF in all the soil depths among all the land use systems. Mn and Ca were lowest in HG and CJ at the surface layer (0-10 cm). Ca concentration was reported to be substantially higher in HG in comparison to other land uses in all soil depths. These cations were significantly affected by different land use systems (L), soil depths (D) as well as an interaction (LxD) of both these factors ($p<0.05$). As a general trend, Mg, K, Ca, and Mn reflected depreciating values with increasing depth in all the land use systems.

Almost all the total concentration of nutrients were significantly ($p<0.05$) affected by land use systems (L), soil depths (D) along with their interactions (LxD). NAF and CJ showed a decreasing trend of concentration with increasing soil depth for all the nutrients estimated (P, Na, Mg, K, Ca and Mn). JF also showed a decreasing concentration with increasing depth in almost all the nutrients except for Na content. In all the land use systems studied, P, Ca and Mn followed a decreasing trend of concentration with increasing soil depths. P content was reported highest in soil of NAF while, concentration of Na, Mg, K, Ca and Mn were observed greatest in HG in comparison to the other land use systems. In general, soils of HG land use system had highest concentration of total nutrients except total P content, which was substantially higher in NAF. CEC was also observed to change in all the land use systems with a decrease in level with increasing soil depths. Land use systems (L), soil depths (D) and an interaction of both these factors were observed to have a significant effect ($p<0.05$) on the soil CEC levels.

Soil aggregate size fractions (Macro-, meso- and micro-aggregates) were observed to be significantly ($p<0.05$) affected by different land use systems, soil depths as well as an interaction of both the factors. Macro-aggregates were observed to decrease with increasing soil depths in all the land use systems, while meso- and micro-aggregates increased with increasing soil depths in almost all the land use systems except in CJ and HG. Mean weight diameter (MWD) and geometric mean diameter (GMD) significantly ($p<0.05$) varied among different land use systems (L) and soil depths (D). Interaction effect of land use and soil depths (LxD) was also found to be significant ($p<0.05$). Both MWD and GMD decreased with increasing soil depth in all the land use systems. Highest MWD and GMD were observed in surface layer (0-10 cm) of JF and NAF. In the subsurface layers, the lowest values of GMD and MWD were observed in HG.

Associated TOC and TN in different soil aggregate fractions (macro-, meso- and micro-aggregate) were observed to be significantly affected by different land use systems (L), soil depths (D) and their interactions (LxD). Macro-, meso- and micro-aggregate associated TOC and TN decreased with increasing soil depths in all the land use systems. In addition, associated TOC and TN were found to be higher in macro-aggregates in comparison to meso- and micro-aggregate. Highest aggregate associated TOC and TN were recorded in NAF. Comparatively, the least concentration of aggregate associated TOC and TN were recorded in HG for all aggregate fractions (macro-, meso- and micro-aggregates).

Soil microbial biomass carbon (MBC) followed a trend decreasing with increasing soil depth. The highest MBC was reported from NAF in all the soil depths in comparison to other land use systems. The highest MBC content in the surface layer (0-10 cm) followed a descending trend of; NAF> JF>CJ>AP>HG. As a general trend, population of fungi, bacteria and actinomycetes were observed to occur lowest in HG in comparison to the rest of the land use systems in all the soil depths. Fungi population was found to exist highest in JF at 0-10 cm with 22.5 CFU’s g^{-1} soil in comparison to rest of the land use systems and soil depths. In case of bacteria, the population at the surface soil (0-10 cm) across the land use systems ranged from 140.3 to 186.0 CFU’S g^{-1} soil, with the highest incidence in NF and lowest in HG. Similarly, actinomycetes population was higher in NAF at the 0-10 cm soil layer (54.7 CFU’S g^{-1} soil) followed by JF (51.2 CFU'S g^{-1} soil) and the least was observed in HG (28.6 CFU'S g^{-1} soil). While at the sub-surface soil layer (10-20 cm), the highest incidence of actinomycetes was recorded in AP with 46.7 CFU'S g^{-1} soil.

Conclusion

Overall, the results showed a distinct change in soil properties with land use change ultimately leading to negative feedbacks between soil properties (eg. TOC, TN) and land use system as in the case of conversion of Natural forest to settled Homegardens. Comparing the observed land uses with natural forest, *Acacia pennata* plantation (AP) land use system can be a potential land use system for developing sustainable land use practices owing to its ability to recover and sequester higher carbon and nitrogen even in its sub surface soil layers as indicated by the results. The findings from the present study is expected to contribute in the development of land use management policies or practices and for further research ventures aimed at sustainable use of land and rejuvenating soil quality of degraded lands in Mizoram.

Our findings signify that conversion of native forests leading to cropping and different management practices in small scale agro-ecosystems instigates degradation of soil quality and represses sustainability. The results indicated that TOC and SOM strongly influences a wide range of soil properties especially soil aggregation which is one of the most important soil characteristic responsible for conservation of soil nutrients and soil fertility in general. Owing to the tremendous importance of TOC and SOM and its influence on soil health, its management aimed at repletion and retention is a key factor in developing management policies and selection of land use systems. It is therefore of great importance to adopt suitable cultivation practices and systems which will enhance soil properties. The results and inferences from the current study will help in enhancing our understanding on the impact of conversion of native forests to other land uses including cropping, and management practices on soil physico-chemical and biological characteristics. In order to achieve sustainability, appropriate management practices aimed at conserving and enhancing soil properties should be the key factor for adoption of any agro ecosystems in the current scenario of land use change.