

**Impact of Different Land Uses on Soil Quality and
Organic Carbon Stocks in Terrestrial Ecosystems of
North Eastern Zone of Kashmir**

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(2013-477-D)



**Division of Soil Science
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**Impact of Different Land Uses on Soil Quality and
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Sher-e-Kashmir

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This is to certify that the thesis entitled, “**Impact of Different Land Uses on Soil Quality and Organic Carbon Stocks in Terrestrial Ecosystems of North Eastern Zone of Kashmir**” submitted in partial fulfillment of the requirements for the award of the degree of Doctor of Philosophy in Agriculture (Soil Science), to the Faculty of Agriculture, Sher-e-Kashmir University of Agricultural Sciences and Technology of Kashmir, is a record of bonafide research work carried out by Mr. **Shakeel Ahmad Mir (Regd. No. 2013-477-D)** under my supervision and guidance. No Part of this thesis has been submitted for any other degree or diploma.

It is further certified that information received during the course of investigation has duly been acknowledged.

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ABSTRACT

Impact of different land uses on soil attributes were recorded from two depths (0 to 15cm and 15 to 30cm) from seven different land use types/ecosystems, namely wetlands, forests, grasslands, apple orchards, vegetable fields, as well as maize and paddy fields. The mean values of TOC in surface and sub-surface soil layers varied between 1.14 to 5.68 % and 0.92 to 5.03 % respectively. Average soil LOC in the surface soil layers was found to be relatively much higher under wetland use (11202 mgKg⁻¹) followed by forests (4407 mgKg⁻¹) and decreased in the order of grasslands (2820 mgKg⁻¹), apple orchards (2366 mgKg⁻¹), maize fields (2209 mgKg⁻¹), vegetable fields (2082 mgKg⁻¹), and paddy fields (2012 mgKg⁻¹) respectively. The greatest mean POC in surface layer was recorded in wetland soil and least in paddy soil. The mean MBC content in the surface soils varied from 276 to 952 mgKg⁻¹. The greatest CMI among the soils was recorded in forest land use followed by grassland and decreased in the order of apple, maize, paddy and vegetable soils. The total SOC stocks varied from 20.41 to 61.97 Mg ha⁻¹ in surface soils and from 17.27 to 60.21 Mg ha⁻¹ in subsurface soil. Largest mean organic carbon stock was recorded in mineral soils of wetlands followed by forest system, grassland, apple, vegetable, maize, and lowest in case of paddy soils. Varying quantities of major soil nutrients were recorded in this investigation and uncultivated soils exhibited higher values for N, P, K, S, Ca and Mg levels, indicating that these nutrients have been markedly influenced by

cultivation. Soil enzyme activities, which represent microbial activity, have been also affected by land use in the study area. Contrary to agro-ecosystem soils in general, uncultivated soils exhibited higher soil enzyme activity. In surface soil layer DHA activity varied from 10.4 to 22.2 $\mu\text{g TPF g}^{-1} \text{ soil h}^{-1}$, the Ac-P activity ranged from 283 to 519 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$, Alk-P activity varied from 285 to 500 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$, activity of Aryl-S was found to be in the range from 32 to 56 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$, whereas, mean FDA activity of the soils ranged from 10.08 to 17.47 $\mu\text{g g}^{-1} \text{ soil h}^{-1}$. Different indexing methods and scoring combination used in the investigation revealed the differences in soil quality among different land uses. Driven by adverse physical, chemical and biochemical changes, the overall SQ of paddy, maize, apple and vegetable fields has depleted. The SQ indices, in general, have revealed the following soil quality ranking among the land use types: forests > apple orchards > vegetables > maize > paddy soil. Based on soil carbon content, nutrient levels and enzyme activities, it is obvious that continuous disturbance has reduced soil quality in the crop areas. Evaluation of different indexing procedures reveals that regression based indices are most sensitive, followed by conceptual model based indices and PCA based (non-linear scoring) index. The soil quality indices used in the present investigation can be effectively used for different soils of the zone. However, further validation of the approaches used in indexing procedures is required.

Keywords: *Soil organic carbon fractions, carbon stocks, enzyme activities, soil quality index, carbon management index*

Signature of the student

Signature of the Major advisor

Dated.....

Dated.....

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ABBREVIATIONS & ACRONYMS

Ac-P	-	Acid phosphatase
Alk-P	-	Alkaline phosphatase
amsl	-	Above mean sea level
Av.K	-	Available K
Av.N	-	Available N
Av.P	-	Available P
BD	-	Bulk density
C.I	-	Confidence interval
CEC	-	Cation exchange capacity
CMI	-	Carbon management index
DHA	-	Dehydrogenase
Ex. Mg	-	Exchangeable Mg
Ex.Ca	-	Exchangeable Ca
FDA	-	Fluorescein diacetate
LOC	-	Labile organic carbon
LS	-	Linear score
LSF	-	Linear scoring function
LUC	-	Land use change
MBC	-	Microbial biomass carbon
MDS	-	Minimum data set
NLS	-	Non-linear score
NLSF	-	Non-linear scoring function
OC	-	Organic carbon
PCA	-	Principal component analysis
PNP	-	<i>p</i> -nitro phenol
POC	-	Particulate organic carbon
S.E	-	Standard error
SOC	-	Soil organic carbon
SOM	-	Soil organic matter
SQ	-	Soil quality
SQI	-	Soil quality index
TOC	-	Total Organic carbon
TPF	-	Triphenyl formazan
WBC	-	Walkley and Black organic carbon

Chapter – 1

INTRODUCTION

Soil is a dynamic, living, non-renewable resource that plays many key roles in terrestrial ecosystems (Doran and Parkin, 1994). Terrestrial ecosystems provide a number of vital services for people and society, such as biodiversity, food, fibre, water resources, carbon sequestration, and recreation. The future capability of ecosystems to provide these services is determined by changes in socio-economic characteristics, land use, biodiversity, atmospheric composition and climate (Metzger *et al.*, 2006). Land use, may be defined as the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it (Sharifabad *et al.*, 2014). It has been changing ever since humans first began to change their environment. However, the changes that have taken place over the last 50 years have been especially important and intense (Metzger *et al.*, 2006). Amongst the many aspects of global change, land use change has been stressed as a key human-induced effect on ecosystems (Turner *et al.*, 1997). Land use is one of the main drivers of many processes of environmental change, as it influences basic resources within the landscape, including the soil resources (González, 2014). It causes perturbation of the ecosystem and can influence the carbon stocks and fluxes. According to Lal (2003) changing natural ecosystems into agricultural ecosystems wastes about 30-50 tons of carbon ha⁻¹ in each year. Soils are among the largest terrestrial reservoirs of carbon and hold potential for expanded C sequestration. Thus, they provide a potential way to reduce atmospheric concentration of carbon dioxide (Lal, 2004a). At the same time, this process provides other important benefits in terms of increased soil and environmental quality. Small changes in the soil organic carbon stock could result in significant impacts on the atmospheric carbon concentration (Lal, 2004b; Stockmann *et al.*, 2013). The C balance of terrestrial ecosystems can be changed markedly by the impact of various human activities, including land-use change, which result in the release of trace gases that enhance the greenhouse effect

(Lorenz and Lal, 2005). About 120 Pg C has been released to the atmosphere due to change in land use throughout the world, most of which occurred after 1940 (Houghton, 1993). Different land uses have variable impacts on soil carbon pools and dynamics. Agricultural land, forests and pastures make up the most land area and these land uses have the potential to hold large quantities of Carbon (Lantz *et al.*, 1999). Lal (2004b) documented a reduction of about 60% in the levels of C in temperate environment and 75% or more in soils cultivated in the tropics after different periods of deforestation of natural forest to agricultural systems. According to Lal *et al.*, (1997) about 73.3% of all land use change is due to removal of forests and conversion of grasslands for arable land-use. Besides, the aforementioned land uses, wetlands are also major carbon sinks. While vegetation traps atmospheric CO₂ in wetlands and other ecosystems alike, the net-sink of wetlands is attributed to low decomposition rates in anaerobic conditions. Many riverine, estuarine and coastal wetlands also trap large quantities of sediment from natural and anthropogenic watershed sources, adding to the carbon accumulation. Estimates of carbon in global wetlands show a broad range from 202 to 535 Gt. However, degradation on a massive scale has already occurred in global wetland ecosystems (Mitra *et al.*, 2005). The condition of wetlands on the whole, in the state of J&K and in particular the study area is no different. Bombay Natural History Society (BNHS, 2014) reported that over the last 30 years, nearly 50% of the wetlands in the Kashmir valley have been encroached upon or severely damaged beyond restoration due to reckless developmental activities with no regard for nature conservation.

Accurate knowledge of the terrestrial component of the global carbon cycle has become a policy imperative, both for individual countries and globally. The policies and scientific research concerned with carbon depend on accurate and objective information about the state of, and changes in, various components of the terrestrial carbon cycle. Specifically, accurate estimation of soil organic carbon (SOC) stocks is necessary in order to meet the requirements of the Kyoto Protocol (United Nations, 1998) which commits all parties to determine its level

of SOC stocks in 1990, and to enable an estimate to be made of changes in SOC stocks in subsequent years (Krogh *et al.*, 2003). In addition to its environmental impact, soil carbon is an important attribute of soil quality and its productivity (Patil *et al.*, 2014). Land use changes, especially cultivation of deforested land, may rapidly diminish soil quality. As a result, severe degradation in soil quality may lead to a permanent degradation of land productivity (Islam and Weil, 2000a). Globally, decline in soil quality (SQ) has posed a tremendous challenge to increasing agricultural productivity, economic growth, and healthy environment. The underlying causes for SQ degradation are largely related to inappropriate land use and soil management, besides few other causes (Girmay *et al.*, 2008). Thus, land use, soil quality and SOC are the three intertwined factors that will continue to be essential in future as they influence the way the land is managed with respect to agricultural and environmental needs.

Soil quality may be defined as the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health (Doran and Parkin, 1994). Interest in evaluating the quality and health of our soil resources has been stimulated by increasing awareness that soil is a critically important component of the earth's biosphere, functioning not only in the production of food and fibre but also in the maintenance of local, regional, and global environmental quality (Glanz, 1995). It has been reported that soil land use influences biological soil quality more than soil type (Fromm *et al.*, 1993). Accurate, consistent assessment of soil quality requires a systematic method for measuring and interpreting soil properties that adequately serve as soil quality indicators (Granatstein and Bezdicek, 1992). A suitable soil quality indicator must be: (a) sensitive to as many degrading agents as possible; (b) show a consistent directional change in response to a given contaminant, and (c) be able to reflect different levels of degradation (Elliott, 1997). Mandal *et al.*, (2013) reported the superiority of forest land use system in terms of maintaining greater soil quality index (SQI) than other land-use systems.

Furthermore, the study indicates that soil quality in terms of physical properties, fertility status, and organic matter was highest in forest land use system followed by agroforestry, irrigated cropland and rainfed cropland. Marzaioli *et al.*, (2010) evaluated the impact of different land use types on soil and their results suggest that the permanent crop management had generally a strong negative impact on soil quality, while the moderate grazing activity and the crop management that leaves herb cover on the soil had a lower negative impact. Higher levels of chemical and physical soil quality indicators were observed in cultivated land use system due to addition of fertilizers and farmyard manure. SOC was the most dominant measured soil attribute as a SQI for 0-10 and 10-20 cm depths. Therefore, SOC could play an important role for monitoring soil quality (Shukla *et al.*, 2006). Armenise *et al.* (2013) in their study on soil quality found that physical indicators were less responsive to the management practices in comparison to the chemical parameters. Since, soil quality is affected by land use type as well as management practices, therefore, an inclusive study may involve its physical, chemical and biological properties that are able to readily change in response to variations in these conditions.

Soil is a complex system where in chemical, physical and biological factors are held in dynamic equilibrium. Studies of enzyme activities provide information on the biochemical processes occurring in soil. Evidences suggest that soil biological parameters may be potential and sensitive indicators of soil ecological stress and restoration (Kizilkaya and Bayrakli, 2005) as well as management induced changes in soil quality (Kennedy and Papendick, 1995). Enzymes are important in catalyzing several important reactions necessary for the life processes of micro-organisms in soils and the stabilization of soil structure, the decomposition of organic wastes, organic matter formation and nutrient cycling (Dick *et al.*, 1994). The enzyme levels in soil systems vary in amounts primarily due to the fact that each soil type has different amounts of organic matter content, composition and activity of its living organisms and intensity of the biological

processes (Stevenson, 1986). These enzymes are constantly being synthesized, accumulated, inactivated and/or decomposed in the soil, hence playing an important role in agriculture and particularly in nutrients cycling (Tabatabai, 1994). In practice, the biochemical reactions are brought about largely through the catalytic contribution of enzymes and variable substrates that serve as energy sources for micro-organisms (Kiss *et al.*, 1978). These enzymes may include amylase, arylsulphatases, β -glucosidase, cellulase, chitinase, dehydrogenase, phosphatase, protease and urease, having originated from mostly microorganisms and as well as plants and animals (Bandick and Dick 1999). The activities of hydrolytic enzymes are frequently measured to evaluate the effect of land use on biological processes in soils related to carbon (C), nitrogen (N), phosphate (P), and sulfur (S) cycling (Kandeler and Murer, 1993). Studies indicate that high enzyme activity signals mineral element limitation in the ecosystem (Ndakidemi, 2006). Enzyme activity generally increases with the rise of soil organic matter content. Higher enzyme activity indicates larger microbial communities and greater stability of enzymes adsorbed on humic materials (Marinari and Antisari, 2010). A better understanding of the role of these soil enzymes activity in the ecosystem will potentially provide a unique opportunity for an integrated biological assessment of soils due to their crucial role in several soil biological activities, their ease of measurement, and their rapid response to changes in soil management practices (Bandick and Dick, 1999).

Soil organic carbon includes plant, animal and microbial residues in all stages of decomposition (Post and Kwon, 2000). Soils represent the most important long-term organic carbon reservoir in terrestrial ecosystems, as they contain more C than plant biomass and the atmosphere (Schimel, 1995; Tarnocai *et al.*, 2009). The large SOC reservoir is not permanent but results from a dynamic equilibrium between organic and inorganic material entering and leaving the soil. Therefore, C stored in soils is affected by changes in vegetation and plant growth, removal of biomass by harvest, and mechanical soil disturbances, such as plowing. (Von

Lutzow and Kogel-Knabner, 2009; Janssens *et al.*, 2010). Precise determination of changes in organic carbon stocks is prerequisite to understand the role of soils in the cycling of carbon and to verify changes in stocks due to management. Changes in land use affect the cycling and storage of carbon in ecosystems (Guo and Giffard, 2002). The period of time needed for SOC stocks to change strongly enough to be detectable depends on the spatial variability of soil properties, the depth increment considered, and the rate of change. Historically, approximately 78 Pg C has been lost from the global soil pool due to land-use conversion for agriculture with approximately 26 Pg attributed to erosion and 52 Pg attributed to mineralization (Lal, 2004b). Conversion of native forest and pasture to cropland has been found to reduce SOC stocks by an average of 42% and 59%, respectively (Guo and Gifford 2002). Quasi-equilibrium value of SOC decreased from 1.78 to 0.68% in the first 30 cm depth, when the soils are used for agriculture instead of retaining them as forest. The highest threshold value of SOC was observed in forest system, followed by horticultural and lowest in agricultural system (Chandran *et al.*, 2009). Experimental results confirm the importance of management practices and pasture in determining first the steady-state SOC concentrations that are characteristic of given rotations and crop management systems, and second the rates of change of SOC concentrations as they approach steady-state concentrations in agricultural soils (Chan *et al.*, 2011). SOC stock as well as concentration was the highest (101.8 Mg ha^{-1} in 0-30cm, 227.97 Mg^{-1} in 0-100 cm) in temperate and lowest in sub-tropical climate (37 Mg^{-1} in 0-30 cm, $107.04 \text{ Mg ha}^{-1}$ in 0-100 cm). Pattern of SOC stock build up across the altitude was: temperate > lower alpine > upper alpine > sub-tropical (Singh *et al.*, 2011). Contribution of SOC concentration, bulk densities or fine earth fraction to the variance of SOC stocks varied with soil depth. While OC concentration was the most important determinant of OC stock variance in undisturbed forest and grassland sites with low stone contents, bulk densities or fine earth fraction became more important in topsoil layers of croplands and in stone-rich soils (Schrumpf *et al.*, 2011). Highest SOC stocks were found in temperate conditions

which they attributed to the optimum soil moisture and low soil temperature conditions of Himalayan Mountains of India. (Singh *et al.*, 2011)

The extent to which land use management influences soil organic matter (SOM) dynamics can be best evaluated by separating SOM into fractions (Jamala and Oke, 2013). The SOC is composed of diverse fractions varying in their degree of decomposition, recalcitrance, and turnover rate (Huang *et al.*, 2008). Microbial biomass typically comprises 1%–5% of the total organic matter content (Nsabimana *et al.*, 2004). Given its high turnover rate, soil microbial biomass can be used as a potential early and sensitive indicator of SOC changes (Cookson *et al.*, 2007). Soil labile organic carbon is the most active fraction of soil organic carbon with rapid turnover rates, and it changes substantially after disturbance and management (Coleman *et al.*, 1996). The particulate organic carbon (POC) consists of partly decomposed plant and animal residues with a rapid turnover; it is more responsive to management factors and it is believed to make a greater contribution to nutrient cycling (Janzen *et al.*, 1992). POC and MBC (microbial biomass carbon) are important C fractions that reflect key processes such as nutrient cycling and availability, soil aggregation, and soil C accrual (Wander, 2004). To estimate the effect of land use change (LUC) on the total SOC stocks, it is crucial to quantify and understand the sensitivity of the different functional SOC pools to disruptions such as LUC (Martin *et al.*, 1990). Results have shown that the POC fraction had the greatest sensitivity to changes in agricultural management practices, proving its ability as an early indicator of optimized practices to sequester C in soil. (Plaza-Bonilla *et al.*, 2014). Caravaca *et al.*, (2002) observed that there is a general decline of total organic carbon, extractable humic substances, water soluble carbon, water-soluble carbohydrates, Dehydrogenase, phosphatase and β -glucosidase activities following the different agricultural practices with respect to soil under grass cover. Dehydrogenase, urease and phosphatase activities were more sensitive and responded more frequently to land use than total organic carbon and water soluble carbon.

Land use change is a main driver of carbon storage in terrestrial ecosystems. Just like the other parts of the state of Jammu and Kashmir, the study area which spans over two districts viz., Bandipore and Ganderbal, has witnessed changes in land use, for instance, the conversion of forests/ pastures into cropland or a shift from agriculture to horticulture use. Besides, the loss of wetland area due to draining, filling and leveling. The impact of changes in land use and management on soil quality and cycling/storage of carbon cannot be overemphasized, particularly in the context of climate change and sustainability. With advances in climate change mitigation in recent years, emphasis has been put on SOC stocks particularly, as they vary with land use and change with forest loss. Thus, this below ground carbon may play a significant role in local and national carbon budgets and we may need more data on carbon stocks in the state in future. The present study is focused on assessment of carbon stocks and soil quality in agricultural and horticultural ecosystems, besides SOC stocks under forests, pastures and wetlands in the area. The study is based on the premise that any shift from the ecological equilibrium and land use change due to human activities has an impact on soil quality and SOC stocks across the ecosystems in this zone. The study entitled “Impact of Different Land Uses on Soil Quality and Organic Carbon Stocks in Terrestrial Ecosystems of North Eastern Zone of Kashmir” was conducted with the following overriding objectives:

- To Assess the Impact of Land Use on Soil Organic Carbon Stocks
- To Study Effect of Different Land Uses on Soil Quality
- To Develop Soil Quality Indices.

Chapter – 2

REVIEW OF LITERATURE

Overview

Agricultural use and management systems have been generally adopted without recognizing consequences on soil quality (SQ) in the study area. Moreover, land use change has impacted the overall quality of soils as well the SOC stocks with significant losses occurring through disturbance and cultivation. In this chapter perspective of soil quality and its assessment, soil enzymes as indicators of SQ, organic carbon stocks and various SOC fractions under different land uses and management systems, have been presented under various heads.

2.1 Impact of land use and management on soil quality

Solomon *et al.* (2002) reported that a change in organic matter content of the surface soil significantly influenced other key soil quality indicators. While studying the impacts of land use change on soil quality indicators, Ayoubi *et al.*, (2011) observed that clearing of the hardwood forest and tillage practices during 40 years led to a decrease in SOM by 71.5%. The reforestation of degraded land with olive and cupressus increased SOM by about 49% and 72%, respectively, compared to the cultivated control soil. The study showed that forest clearing followed by cultivation of the loessial hilly slopes resulted in the decline of the soil quality attributes, while reforestation improved them in the study area.

Kizilkaya and Dengiz (2010) investigated the effect of agricultural practices on properties of soil taken from four adjacent land use types. The study indicates that LUC and subsequent tillage practices resulted in significant decrease in organic matter, total porosity, total nitrogen and soil aggregate stability. The data shows that cultivation of natural forest soils resulted in alteration of physical and chemical properties of soils. In addition it was found that change of land use and land cover associated with change in soil organic matter levels can alter soil enzymes activities within soil profile.

Current anthropogenic influence on soil quality variation was double-edged, stakeholders must pay close attention to long-term soil quality. The results indicated that after 24 years of anthropogenic influence, soil organic matter, total nitrogen, cation exchange capacity and available phosphorus increased significantly and total phosphorus and available potassium decreased slightly. Among the total selected indicators, SOM was the main driving factor and pH was the limiting factor of soil quality based on path analysis (Qi *et al.*, 2011).

Su and Zhao, (2003) investigated the changes of soil physical, chemical and biological properties under different land use and management that lasted for 14 years. The results showed that among various land use systems marked differences exist in soil quality indicators, including soil particle distribution, composition, porosity, bulk density, water-holding capacity, organic matter, nutrient contents, pH, and enzyme activities. Most of these soil quality indicators were the highest in the orchard intercropped with crops and perennial grass (agroforestry systems), intermediate in the well-management irrigated farmland, and the lowest in the less-management dry farmland. Compared to the primary grassland soil, although some soil properties, including porosity, water-holding capacity, phosphorus content, and enzyme activities, were improved in the well-management systems, soil organic matter and nitrogen contents were significantly lower.

Dadhwal *et al.*, (2011) evaluated soil quality on the bases of seven identified soil attributes. With respect to the forest soil, bulk density increased by as much as 8.0% (1.6 to 8.0%) and organic carbon declined by as much as 39.5% (19.7 to 39.5%) in scrub and terraced agricultural lands. Substantial losses of total N (6.25 to 87.5%), and available K (21 to 69.9%) were also observed. However, extractable P exhibited the increasing trend (101%) in agricultural land. The study further revealed that cultivation on the hill slopes has resulted in a significant deterioration of soil quality as described by different physical and chemical soil attributes.

A study on assessment of soil quality in North Western Himalayas reveals that the forest had a higher fertility index and soil evaluation factor followed by grassland, horticulture, agriculture as compared to wasteland. Soil biological properties were highest under surface soil of forest (576 mg kg⁻¹, 31.24 mg kg⁻¹, 6.55 mg kg⁻¹, 29.6 µg PNP g⁻¹h⁻¹ and 35.65 µg TPF 24 h⁻¹ g⁻¹ dry soil, respectively for MBC, MBN, MBP, APHA and DHA) and least in sub-surface layers (45-60 cm) under wasteland.(Pal *et al.*, 2013).

The effects of land management practices on soil functions (water infiltration, storage and supply; nutrient storage, supply and cycling; and sustained biological activity) were evaluated. The index values indicated that the semi-direct management system resulted in the highest overall soil quality, i.e., the soil functions considered performed better, followed by the pre-germinated and conventional systems. It was found that with less intensive tillage, higher soil quality values were obtained than with conventional treatments. The evidence provided suggests that high yields are not compatible with high soil quality and, therefore, are not sustainable (Lima *et al.*, 2013).

Soil textural fractions varied with land use, while silt, clay, and bulk density differed with soil depths. Among soil chemical quality indicators SOC, available K⁺ and exchangeable bases varied significantly with land use. Human mismanagement land resources, excessive livestock grazing in the protected forest areas, and intensive agricultural production in adjacent farmland have caused deterioration in soil quality indicators (Moges *et al.*, 2013).

Nisar and Lone, (2013) observed that LUC in the form of deforestation, pasture land destruction and long-term cultivation caused decrease of soil organic matter, water holding capacity, available nitrogen and available potassium content. This has resulted in the degradation of soil quality in Sindh catchment of Kashmir Himalayas

Natural forest land systems have relatively good soil quality (SQ), whereas uncultivated marginal land systems have a seriously degraded soil. This study concludes that a PCA based SQI would be the best way to distinguish among land

use and soil management systems (LUSMS) since it appears more sensitive to disturbances and management practices and could thus help prevent further SQ degradation (Tsfahunegn, 2014).

Adeyolanu *et al.*, (2015) also observed higher soil quality under tree crops than the arable crops. Physical, chemical and overall soil quality indices ranged from moderate to high with the highest value occurring under perennials and the lowest under maize field. Both land use and soil type have significant effect on soil physical, chemical and overall quality. Arable crops especially maize will degrade soil quality quickly if not well managed.

Xia *et al.*, (2015) demonstrated the positive relationships between crop yields and soil quality indicators for both surface and subsurface soils. The SOM, TN, Av P, and Av K were regarded as key indicators in soil quality and crop yields for sustainability. The results of this study highlighted the importance of soil K content for soil quality improvement and crop yield sustainability, despite the fact that the need for K continues to be underestimated.

2.2 Soil quality indices of different land use systems

According to Saviozzi *et al.*, (2001) the indices most sensitive to management practices that may provide indications of the effects of soil cultivation, as well as of the differently undisturbed ecosystems were: organic carbon, water soluble carbohydrates, protease, β -glucosidase, urease and Hydrolysing Coefficient. Soil enzymes activities are well related with, and not more sensitive than OC. The results also indicate that long-term corn production at an intensive level caused a marked decline in all examined parameters. However, between undisturbed ecosystems, native grasslands showed higher value of soil quality parameters than forest.

Liebig *et al.*, (2001) adopted an indexing procedure adopted that was effective at discriminating between alternative (half fertilizer dose) and conventional treatments on continuous Maize and other sequences within and across agroecosystem functions. The alternative treatment had significantly lower levels of residual and early spring soil NO₃, significantly higher soil pH, and moderately

higher levels of soil organic C (5065 kg ha^{-1}) compared with the conventional treatment.

Over a period of about 30 years maize-based land use has had a negative effect on soil quality. Change from continuous maize to orchard around did not contribute significantly to an improvement in soil quality up to 5-7 years. Results show that the naturally re-grown secondary forest has the highest SQI followed by reforested areas. Orchards ranked third together with maize-maize. The maize-fallow use had the lowest SQI rank among all the land uses (Cho *et al.*, 2004).

Tiwari *et al.*, (2006) documented that Himalayan region rainfed terraces were found to have the highest nutrient reserves among the land uses. They also had the highest soil quality index (SQI=0.59) followed by forestlands (SQI= 0.45) and irrigated paddy lands (SQI=0.23). The SQI showed paddy lands to be in a 'degraded' condition and forest and rainfed terraces lands 'at risk'. Among the Cropping Patterns maize-millet had a higher SQI (SQI=0.62) than intensive vegetable - vegetable (SQI=0.54). Of the soil properties studied, SOC had the highest weight in determining soil quality. The change in pattern from maize-millet to vegetable - vegetable decreased SOC content from 16 to 14.5 g kg^{-1} within seven years.

Marzaioli *et al.*, (2010) established an SQI (without minimum data set selection) using physical, chemical and biological indicators such as aggregate stability, WHC, bulk density, particle size, pH, EC, CEC, SOC, N, nutrients, MBC, respiration and fungal mycelium. The authors observed a low soil quality in almost all permanent crops; an intermediate soil quality in shrublands, grazing lands, coniferous forest and middle-hill olive grove; and a high SQ in mixed forests. While as Ruiz *et al.* (2011) elaborated an index of biological soil quality (IBSQ) and concluded that well-managed crops and pastures may have better SQ than some forests.

Xie *et al.*, (2011) studied the effects of land use type on the soil quality by the method of soil quality index (SQI). It was observed that comparing with natural forest land, all the cultivated lands had somewhat decreased soil organic matter

content and higher soil bulk density, and the soil bulk density was significantly higher in paddy field and plantation crops. In cultivated lands, fertilization and reclamation made the soil available potassium and phosphorus contents maintained at a higher level, probably due to the input of mineral potassium and phosphorus and the decomposition of soil organic matter. The SQI of the few land use types was in the order of natural forest land (0.532) > maize field (0.516) > paddy field (0.416). The soils in higher altitude production demonstration areas (1614 +/-115 m amsl) had significant higher SQI, compared to the soils in lower altitude buffer areas (908 +/- 98 m amsl) and junction areas (926 +/- 131 m amsl).

While comparing soil fitness for agricultural use under different managements in the Mediterranean environment, it was observed that crop sequence significantly affected soil pH and had a measurable effect on plant available P, with lower pH and higher P availability for the legume-based annual rotation. Stubble incorporation enhanced the labile N pool in spring. The SQI indicated that soil quality was most affected by NPK rate. (Armenise *et al.*, 2013).

The index values computed by Lima *et al.*, (2013) indicated that with less intensive tillage, higher soil quality values were obtained than with conventional treatments. This study demonstrated that a few indicators, i.e., a MDS of 8 out of the 29 indicators or just 4 indicators as chosen by farmers, provided adequate management information on soil quality differences among the management systems. Therefore, the soil quality index approach is an appropriate way to develop a quantitative procedure to evaluate the effects of land management practices on soil functions. In this case, this meant that higher soil quality was associated with lower yield.

Armenise *et al.*, (2013) developed a SQI to evaluate the interaction between soil quality parameters. The fertilizer level appeared to drive SQI results. No effects due to cropping sequence and stubble management were found. The water extractable N was particularly sensitive to soil management. It was also observed that SQI was not significantly correlated with annual grain yield.

Kalu *et al.*, (2015) computed soil quality index on the basis of the soil management assessment framework for evaluation of soil quality under different land uses. It was observed that protected forest has the highest soil quality index (0.95) followed by community forest (0.91), pasture (0.88), paddy field (0.81), and maize-vegetable (0.79). Available phosphorus and soil organic carbon play major roles in making significant differences in the SQI among the different land use types. Less anthropogenic impact and vegetation in forest land result in better soil quality, whereas attempts to increase productivity in cultivated land degrade the soil quality. The proper application of fertilizer and giving priority to organic farming is recommended to improve soil quality.

2.3 Impact of land use and management systems on soil enzyme activities

All soils contain a group of enzymes that determine soil metabolic processes (McLaren, 1975). The activities of these enzymes in soils undergo complex biochemical processes consisting of integrated and ecologically-connected synthetic processes, and in the immobilization and enzyme stability (Khaziyev and Gulke, 1991). Together these findings indicate that soil enzyme activities have broad-scale spatial variability depending on the environmental conditions. The activities of hydrolytic enzymes are frequently measured to evaluate the effect of land use on biological processes in soils related to carbon (C), nitrogen (N), phosphate (P), and sulfur (S) cycling (Kandeler and Murer, 1993).

Enzymes are important in catalyzing several important reactions necessary for the life processes of micro-organisms in soils and the stabilization of soil structure, the decomposition of organic wastes, organic matter formation and nutrient cycling (Dick *et al.*, 1994). These enzymes are constantly being synthesized, accumulated, inactivated and/or decomposed in the soil, hence playing an important role in agriculture and particularly in nutrients cycling (Dick, 1997).

Most soil physico-chemical properties change slowly in response to the environmental stress, with significant changes commonly detected only after many years. By contrast, soil biological properties are sensitive indicators for soil

quality, which rapidly respond to minor environmental changes in the soil. Soil enzyme activity is a potential indicator of soil quality due to its high sensitivity to external interference and the ease of measurement (Bandick and Dick, 1999).

The activities of extracellular enzymes in soil vary significantly with seasons and geographical locations (Paz-Ferreiro, 2010) as well as soil depth (Wittmann *et al.*, 2004). A variety of quantitative measures, including soil physico-chemical properties indicative of the fundamental context of soil functions, have been extensively used to assess soil quality (Puglisi *et al.*, 2006). Enzyme activity generally increases with the rise of soil organic matter (OM) content. Higher enzyme activity indicates larger microbial communities and greater stability of enzymes adsorbed on humic materials (Marinari and Antisari, 2010).

2.3.1 Dehydrogenase

Dehydrogenase enzyme is often used as a measure of any disruption caused by pesticides, trace elements or management practices to the soil (Frank and Malkomes, 1993). Kucharski *et al.*, (1996) have explained that high dose or repeated use of fertilizers may restrict the microbial growth and consequently decrease the dehydrogenase activities in soil.

Many specific dehydrogenases transfer hydrogen to either nicotinamide adenine dinucleotide or nicotinamide adenine dinucleotide phosphate. Throughout mentioned co-enzymes hydrogen atoms are involved in the reductive processes of biosynthesis. Due to this fact, the overall DHA of a soil depends on the activities of various dehydrogenases, which are fundamental part of the enzyme system of all living microorganisms, like enzymes of the respiratory metabolism, the citrate cycle, and N metabolism (Subhani *et al.*, 2001). Thus, DHA serves as an indicator of the microbiological redox-systems and could be considered a good and adequate measure of microbial oxidative activities in soil.

Study performed by Włodarczyk *et al.*, (2002) indicated maximum DHA at pH 7.1; similar results were obtained by Ros *et al.*, (2003), where optimum for DHA was noted for pH 7.6-7.8. It was suggested by Chu *et al.* (2007) and Xie *et al.*, (2009) that long-term balanced fertilization greatly increased DHA level in

the soil environment, rather than nutrient-deficiency fertilization. Soil dehydrogenases (EC 1.1.) are the major representatives of the Oxidoreductase enzymes class (Gu *et al.*, 2009).

Samuel (2009) determined the soil enzyme activities (actual and potential dehydrogenase, catalase, acid and alkaline phosphatase) in the 0–10, 10–20, and 20–30 cm layers of a brown luvisol soil submitted to a complex fertilization experiment with different types of green manure. It was found that each activity decreased with increasing sampling depth. There were significant correlations of soil enzyme activities with chemical properties.

It was shown in many studies that DHA is significantly influenced by water content and dropped with the decrease of soil humidity. For example, Gu *et al.* (2009) observed higher DHA level (even by 90%) in flooded soil, rather than in non-flooded conditions. The higher DHA values in flooded conditions agreed also with results presented by Zhao *et al.* (2010) and Weaver *et al.*, (2012)

Zhao *et al.*, (2010) indicated, that soils with higher fertility are more capable of maintaining the original biological functions (i.e. have a higher functional stability). On the other hand, Moeskops *et al.*, (2010) compared the effect of organic and conventional farming practices on soil enzymatic activities. On the organic farms, soil fertility was maintained mainly with composted OM, in contrast to conventional farmers, who combined fresh manure and chemical fertilizers, and typically applied large amounts of pesticides. As a consequence, a strong negative impact of intensive fertilizer and also pesticide use on DHA was demonstrated

Dehydrogenases occur intracellular in all living microbial cells (Moeskops *et al.*, 2010; Zhao *et al.*, 2010; Yuan & Yue, 2012). Moreover, they are tightly linked with microbial oxido-reduction processes (Moeskops *et al.*, 2010). Dehydrogenases do not accumulate extracellular in the soil and they play a significant role in the biological oxidation of SOM by transferring hydrogen from organic substrates to inorganic acceptors (Zhang *et al.*, 2010).

A significant negative relationships between DHA and Eh was observed at surface layers of Mollic Gleysols, Eutric Fluvisols, Rendzina Leptosols and Haplic haezems, where determined correlation coefficients equaled as follows: $r=-0.91^*$, $r=-0.43^*$, $r=-0.47^{**}$ and $r=-0.48^{**}$ (Wolinska, 2010).

Several authors reported positive correlation between DHA and OM content (Zhao *et al.*, 2010; Yuan & Yue, 2012). Zhang *et al.*, (2010) also indicated that DHA, OM and CaCO_3 were correlated with each other in their spatial distribution.

Fernandez-Calvino *et al.* (2010) noted significantly positive correlations among soil DHA and pH in the range of 4.1 (pHKCl) and 4.9 (pHwater), suggesting that acidity suppressed potential enzyme activity.

Yuan & Yue (2012) stated the highest DHA level in autumn season and the lowest value of DHA in winter time. Depth of the soil profile is one of the most known and popular environmental factor reducing soil DHA level. It is well known that the highest microorganisms abundance is in the surface layer of the soil profile (till to the depth of 30 cm), at the deepest part of the soil the number of microbial cells is limited, and consequently also DHA level display diminishing trend.

Kalembasa and Symanowicz (2012) noted that the levels of enzymatic activities of acid phosphatase and soil dehydrogenases increased due to mineral fertilizers while the activity of alkaline phosphatase decreased. Furthermore, organic materials and ash significantly affected the activity of enzymes and the potential growth of corn.

Tan *et al.*, (2014) evaluated the spatial distribution of soil enzymatic properties in agricultural land on a county-wide (567 km²) scale. While studying various enzyme activities it was observed that phosphatase, and catalase activities were moderately spatially correlated, whereas dehydrogenase activity was weakly spatially correlated. Dehydrogenase activity ranged from 0.15 to 30.6 $\mu\text{gTPF}\cdot\text{g}^{-1}\text{h}^{-1}$, with a mean of 25.0 $\mu\text{gTPF}\cdot\text{g}^{-1}\text{h}^{-1}$. Coefficient of variation was 25% for phosphatase activity and 49% for dehydrogenase activity. OM content was found

significantly correlated with invertase, urease, phosphatase, and dehydrogenase activities.

Highest Dehydrogenase activity was recorded in deodar forest soil while the fodder crop and apple plantation showed significantly lower values. Contrary to inorganic fertilizer long term used agricultural farms, organically maintained agricultural farm exhibits higher DHA activity ($11.6 \mu\text{g TPF g}^{-1} \text{ soil h}^{-1}$). Agro-ecosystems in an average were found to have 36.54 % less DHA activity in comparison to forest ecosystems. In the study the organic farm soil exhibits 40.5 % higher dehydrogenase activity as compared to other soil (Singh *et al.*, 2014).

According to Wang *et al.*, (2015) biochemical variables- dehydrogenase and phosphatase activities, were positively correlated with soil moisture, water holding capacity, total N and SOC. Moreover, the principal component analysis indicates that dehydrogenase and phosphatase activities are the vital contributors to the soil of the undisturbed natural wetland.

Mangalasserya *et al.*, (2015) observed that zero tilled soils contained 9% more soil C, 30% higher microbial biomass C than tilled soil. The study found that microbial biomass was 30% greater in zero tilled soils indicating a more efficient functioning of the microbial community under zero tillage practice. Furthermore, microbial enzyme activities of dehydrogenase, cellulase, xylanase, β -glucosidase, phenol oxidase and peroxidase were higher in zero tilled soils of the temperate region.

2.3.2 Phosphatase

Phosphatase (EC 3.1.3.2) releases inorganic phosphate from organic moiety, and are known to play a key role in phosphorus cycle in soil ecosystems (Speir and Ross, 1978). A negative correlation between acid phosphatase and pH. has been documented by: Gehlen and Schroder (1990); Chhonkar and Tarafdar (1984); and Herbien and Neal (1990).

The phosphomonoesterases (acid and alkaline) differ in their substrate specificity and their pH optimum. They are excreted by plant roots and by

microorganisms. In the soil, microbial phosphatases dominate (Canarutto *et al.*, 1995).

In minimum tillage and reduced tillage higher phosphatase activity has been noted compared with other tillage systems (Piovaneli *et al.*, 1998), and these practices can induce positive effects on soil biochemical and chemical properties (Canarutto *et al.*, 1995). A positive correlation between phosphatases and C and N has been observed by Sarapatka and Krskova (1997): Bonmati *et al.*, (1991) and Chhonkar and Tarafdar (1984)

The activities of acid and alkaline phosphatases, β -glucosidase and urease were determined in soils under different land-use systems: primary forests, secondary forests, coffee plantations, and cultivated lands (upland fields and paddy fields after deforestation in hilly areas of Indonesia. Enzyme analyses showed that the activities of phosphatases, β -glucosidase, and urease were significantly higher in most cases in the primary forests or in the secondary forests than in the other two land-use systems, indicating that clearing the forests and converting them to other land-use systems significantly disturbed the growth of enzyme-producing organisms. There was no significant difference in the enzymatic activities between cultivated upland fields and paddy fields. The activity of alkaline phosphatase showed the most drastic decrease after the conversion to the other land uses (Salam *et al.*, 1998).

Phosphatase and arylsulfatase activities were measured in three wetland soils (bog, fen and swamp) over 12 months. The fen site (85-176 nmol g⁻¹ min⁻¹) showed the highest phosphatase activity of the three, whilst there was little difference between the arylsulphatase activities of the fen (4-14 nmol g⁻¹ min⁻¹) and the swamp (5-16 nmol g⁻¹ min⁻¹). For both enzymes, the lowest activity was observed in the bog site (20-61 nmol g⁻¹ min⁻¹ for phosphatase; 1-3 nmol g⁻¹ min⁻¹ for arylsulphatase). Hydrogen ion concentration was a dominant controlling factor for the phosphatase activities in all sites. Waterlogging and low temperature seem to restrict enzyme activities in the fen and the swamp sites, as both factors showed significant correlations with the enzyme activities. No temporal

relationships between the enzyme activities and the inorganic nutrient concentrations were detected. However, a negative relationship between phosphatase activity and phosphate content was discernable, when compared on a spatial basis (Kang and Freeman, 1999).

The green manure species influenced soil enzyme activity (urease, arylsulfatase and phosphatase) and C and N mineralization rates, both under the tree canopy and in-between rows. Cultivation of *Leucaena leucocephala* (N fixer) increased acid phosphatase and arylsulphatase activity and C and N mineralization both under the tree canopy and in-between rows (Balota and Chavez, 2010).

Phosphatase activity is sensitive to environmental perturbations such as organic amendments, tillage, waterlogging, compaction, fertilizer additions and thus it is often used as an environmental indicator of soil quality (Oprica *et al.*, 2011).

Reduction of 21% of soil moisture reduced acid phosphatase activity by 31–40% and no significant influences were found on alkaline phosphatase activity (Sardans and Penuelas 2005).

The soil acid phosphatase activity varied from 152 to 502 $\mu\text{g g}^{-1}$ soil. This parameter was significantly influenced by soil depth but was not affected by manure applications or by tillage practices. The soil alkaline phosphatase activity ranged from 24 to 71 $\mu\text{g g}^{-1}$ soil. Alkaline phosphatase activity was higher in tillage treatments compared to no-till plots. Furthermore, it was observed that soil alkaline phosphatase activity was always significantly lower at a depth of 15 to 30 cm compared to surface soil (Kheyrodin *et al.*, 2012).

Nine different land use system of similar soil type at depth 0-15cm were analyzed for soil enzymes (Dehydrogenase, Acid Phosphatase, Alkaline Phosphatase, Nitrate Reductase, Arylsulphatase, and Phytase) and genetic fingerprints (Randomly Amplified Polymorphic DNA) analysis. The land use systems investigated included Oak (*Quercus incana*), Deodar (*Cedrus deodara*), Pine (*Pinus roxburghii*) trees, Apple orchards and crop based systems in uplands and valleys. All the soil enzymes (including phosphatase) were significantly

higher in forest ecosystem followed by organic farm and conventional maize-wheat farm soil. (Singh *et al.*, 2014)

Enzyme activities of the soil are result of enzyme producing activity of soil microbes present in it. In rice crop, at zero day, alkaline phosphatase activity in rhizospheric soil samples taken from field conditions was observed to be 32.8 $\mu\text{g/g soil/hour}^{-1}$ at 33.5° C soil temperature.. At the time of harvesting, alkaline phosphatase activity in rhizospheric soil samples taken from field conditions was observed to be 25.0 $\mu\text{g/g soil/hour}$ at soil temperature of 25.5°C. (Kaur *et al.*, 2014).

In comparison with natural forests the cultivated soil exhibits lesser acid phosphatase activity: Fodder < Soybean–wheat < Upland rice < Apple plantation < Maize-wheat < Organic farm. Apple plantation soil shows significantly lower values of alkaline phosphatase (42.6 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) than acid phosphatase (295 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) values (Singh *et al.*, 2014). Reducing pH reduces soil alkaline phosphatase activity, arylsulphatase activity, nitrification potential, and respiration(Wang *et al.*, 2015).

2.3.3 Arylsulphatase

Arylsulphatases (EC 3.1.6.1) are typically widespread in soils (Tabatabai and Bremner, 1970; Ganeshamurthy *et al.*, 1995). They are secreted by bacteria into the external environment as a response to Sulphur limitation (McGill and Colle, 1981). Arylsulphatases are widespread in nature as well (Dodgson *et al.*, 1982). Occurrence of Arylsulphatases in different soil systems is often correlated with microbial biomass and rate of S immobilisation (Klose *et al.*, 1999; Vong *et al.*, 2003).

In a study of phosphatase and arylsulphatase activities in wetland soils (Kang and Freeman, 1999), pH was found to be positively correlated with phosphatase activity. Martinez and Tabatabai (2000) also observed a significant and positive correlation between alkaline phosphatase and arylsulphatase activity with pH, the correlation coefficient were 0.89 and 0.66, respectively, and a negative correlation of acid phosphatase and pH with a correlation coefficient 0.69.

A simplified assay was employed by Elsgaard *et al.*, (2002.) to determine the depth distribution of arylsulphatase activity of four soil types in Denmark. The arylsulphatase activity ranged from 0.8-30.5 $\mu\text{g p-nitrophenol g}^{-1}$ dry soil with the lowest levels in sandy soil. Overall, the highest activity was found in the plough layer (0-20cm), while the activity at soil depths of 30-50cm was much lower i.e., 8-26% of the highest activity. Thus, the potential for S cycling between organic and inorganic SO_4^{2-} was highest in the plough layer. There was a strong correlation between the arylsulphatase activity and organic C of the individual 10-cm soil segments.

Arylsulphatases are responsible for the hydrolysis of sulphate esters in the soil (Kertesz and Mirleau, 2004). Moscatelli *et al.*, (2005) recorded the greatest activity of arylsulphatase in autumn and lowest in winter, while for the rest of the year slight fluctuations around 22 mg pNP $\text{g}^{-1} \text{h}^{-1}$ for the FACE (free air CO_2 enrichment) plots and 18 mg pNP $\text{g}^{-1} \text{h}^{-1}$ for the control plots were recorded. Arylsulphatase resembles MBC trend as indicated by significant positive correlation between them. Moreover, CO_2 enrichment determined a positive effect in all the seasons, with an overall annual effect of +24 %..

The average arylsulphatase activity ranged from 11.57 $\mu\text{g PNP g}^{-1} \text{hr}^{-1}$ in apple plantation soil to 222.78 $\mu\text{g PNP g}^{-1} \text{hr}^{-1}$ in case of oak forest soil. Except oak and deodar soils, all the other ecosystem soil showed significantly reduced arylsulphatase activity followed in order: Upland rice > Maize-wheat farm > organic farm > Soybean-wheat > Fodder crops > Apple Plantation farm (Singh *et al.*, 2014).

Piuttia *et al.*, (2015) identified the dynamics of S in relation with functional microbial communities in soil amended with plant residues. Higher decomposition rates and S- SO_4^{2-} release from soils amended with low C/S ratio were recorded. Soil microbial biomass carbon increased more for residues with low C/S ratios. Besides this, Plant residues with low C/S increased the ratio bacteria: fungi possessing arylsulphatase activity and the C/S ratio of plant residues incorporated

in soil did not influence the arylsulfatase activity. Among the tested residues, mustard was the most labile and the most rapidly mineralized, leading to a significant 67% increase in the soil microbial biomass C compared with the control.

Reducing pH reduces soil arylsulphatase activity, nitrification potential, and respiration (Wang *et al.*, 2015). The impact of sulphur and soil compaction on soil microbial biomass C and the activities of urease and arylsulphatase was investigated under greenhouse conditions. The results showed that application of different levels of sulphur decreased microbial biomass carbon but urease and arylsulphatase activities decreased at the highest levels of sulfur. Microbial biomass carbon and arylsulphatase activity were not influenced by soil compaction (Shekoofeh *et al.*, 2015).

2.3.4 Fluorescein Diacetate

Fluorescein diacetate (FDA) is a non-polar esterified compound which can be hydrolysed by non-specific esterases (Guilbault and Kramer, 1964). It is generally assumed that FDA diffuses freely into intact cells (Rotman and Papermaster, 1966) where esterases hydrolyse it to the more polar fluorescein (Battin, 1997). Fluorescein diacetate [3',6'-diacetyl fluorescein] can be used to measure microbial activity in soils (Brunius, 1980; Lundgren, 1981; Schnurer and Rosswall, 1982).

Swisher and Carroll (1980) demonstrated that the amount of fluorescein produced by the hydrolysis of FDA was directly proportional to the microbial population growing on Douglas Fir foliage and a standardised method was developed. According to (Lundgren, 1981) the product of this enzymatic reaction is fluorescein, which can be visualized within cells by fluorescence microscopy. Schnurer and Rosswall (1982) documented that Fluorescein released in soil can also be measured by spectrophotometry.

Schnurer and Rosswall (1982) observed that FDA hydrolytic activities in soil samples from different layers of an agricultural soil were correlated with microbial respiration. Furthermore, spectrophotometric determination of the

hydrolysis of fluorescein diacetate (FDA) was shown to be a simple, sensitive, and rapid method for determining microbial activity in soil and litter and acetone was found to be suitable for terminating the reaction

Aseri and Tarafdar (2006) noted significant correlation between FDA hydrolysable enzyme activity and the other enzyme activities (acid phosphatase, alkaline phosphatase, phytase, dehydrogenase). A strong linear regression ($R^2 = 0.61$; $p < 0.01$) between the FDA hydrolysable enzyme activity and microbial biomass was observed. The results demonstrated that FDA hydrolysable enzyme activity is a potential biological indicator of arid soils and is a better indicator than dehydrogenase activity, already considered as a biological indicator.

Son *et al.*, (2006) investigated soil respiration and the hydrolysis of FDA as a measure of total microbial activity at three sites that had been changed from abandoned agricultural lands to natural vegetation: rice field conversion to forest (RF), crop field conversion to shrub (CS), and indigenous forest (IF). Mean annual FDA hydrolysis ($\mu\text{g FDA min}^{-1} \text{g}^{-1}$ dry soil) was higher at RF (4.56) followed by IF (4.61) and CS (3.65). At all three land-use change sites, soil respiration was only very weakly correlated with FDA hydrolysis.

Green *et al.*, (2006) recorded mean FDA hydrolytic activity, measured on three separate occasions for few soils. The activity ranged from 66.1 mg fluorescein kg^{-1} soil 3 h^{-1} to 226.3 mg fluorescein kg^{-1} soil 3 h^{-1} . The percent standard deviation for the eight soils tested with the method averaged 3.2% and ranged from 2.2 to 4.4%. Moreover according to these authors Fluorescein diacetate is hydrolyzed by a number of different enzymes, such as proteases, lipases, and esterases.

Freitas *et al.*, (2011) observed that Microbial carbon, CO_2 emission, and FDA activity were increased by 100 to 200% in the organic vineyards. Zhao (2012) observed that across the growing season, microbial biomass C and N contents, basal soil respiration, and fluorescein diacetate hydrolysis, dehydrogenase, urease, and invertase activities in the compost treated maize and combination of

compost and NPK treatment were 7- 203% higher than the balanced NPK treatment.

Application of diverse organic inputs significantly improved the microbial biomass carbon (MBC), organic carbon, dehydrogenase activity and FDA in paddy soil. MBC showed significant correlation with soil OC ($r=0.883$), DH ($r=0.922$) activity and FDA ($r=0.675$) hydrolase except phosphomonoesterase (PMEase) activity. Similarly, OC also displayed strong correlation with DH ($r=0.836$) activity and FDA ($r=0.842$) hydrolase but poorly with PMEase (0.269) activity (Nath *et al.*, 2012).

Degradation, season and interactions between degradation and season significantly influenced all soil microbial and enzyme response variables (FDA, DHA, and Cellulase). These variables were significantly higher under native vegetation than in restored land. Moreover, the microbial properties were significantly higher in restored land than in degraded land (both moderately and highly degraded land). FDA hydrolysis was significantly higher in rainy season. (Araujo *et al.*, 2013).

2.4 Organic carbon stocks as influenced by land use and management systems.

Jenny and Raychaudhuri (1960) conducted one of the first comprehensive studies on the distribution of OC in Indian soils. However, these authors did not provide estimates of the total soil C reserves. Nichols (1984) and Burke (1989) determined a strong relationship between clay and organic carbon concentration ($r^2=0.86$) in Southern Great Plains soils. That SOC content and clay concentration were related, was also determined by Burke *et al.*, (1989). Dadhwal and Nayak (1993) using ecosystem areas and representative global average C density, estimated organic C at 23.4–27.1 Pg in Indian soils.

Three scenarios of conservation tillage use, 27% (current usage), 57% (Scenario 2), and 76% (Scenario 3) of field cropland planted, were considered in the model based study. The SOC content for major field crops to 30-cm depth was 5304 to 8654 Tg, with 1710 to 2831 Tg C at 0- to 8-cm depth, and 1383 to 2240

Tg C at 8- to 15-cm depth. Maintaining current levels of conventional tillage until 2020 would result in 31 to 52 Tg SOC loss. Scenario 2 conventional tillage resulted in 18 to 30 Tg C SOC loss, and Scenario 3 yielded 9 to 16 Tg SOC loss, which were C savings of 21 to 36 Tg C over maintaining current levels of tillage. Conversion of conventional tillage to no-till resulted in 80 to 129 Tg C gain in soil for Scenario 2, and 286 to 468 Tg C for Scenario 3. No-till and conventional tillage had similar SOC contents below the 15-cm depth. Minimum tillage conserved current levels of SOC but did not consistently increase SOC above levels of conventional tillage (Kern and Johnson, 1993).

To sustain the quality and productivity of soils, the knowledge of OC in terms of its amount and quality in soils is essential (Bhattacharya *et al.*, 2000). The great capacity of wetland soils to store carbon (C) derives from the slow rate at which decomposition occurs under anaerobic conditions of poor drainage (Hobbie *et al.*, 2000). Drainage of wetlands during conversion to agriculture or forestry generally results in the loss of carbon, as soil organic matter previously stored under anaerobic conditions is aerated and exposed to atmospheric oxygen. In many cases, the organic carbon stores that had accumulated slowly over centuries to millennia can be lost in days (in the case of burning) or over decades (IPCC, 2001).

Guo and Gifford (2002) performed a meta-analysis of data from 74 publications indicating that soil C stocks decline after land-use changes from pasture to plantation (-10%), native forest to plantation forest (-13%), native forest to cropland (-42%), and pasture to cropland (-59%). They also found that soil C stocks increase after land-use changes from native forest to pasture (+8%), cropland to pasture (+19%), cropland to plantation forest (+18%), and cropland to secondary forest (+53%). The results varied, however, depending on factors such as annual precipitation, plant species and, not least, length of study periods.

Palmer *et al.*, (2002) found that in terms of dry matter production, the harvest index of modern production systems is between 65 and 75 per cent. So most of the carbon captured by the tree is transferred to the fruit and then exported from the

orchard as apples. Thus, there are reduced carbon inputs to the soil through roots, leaf drop, and prunings. Soil carbon inputs do not appear to be matching topsoil carbon losses in the alley in particular.

Chhabra *et al.*, (2003) estimated organic C pool at 6.8 Pg C in the top 1.0 m, using estimated SOC density and remote sensing based area under forest. Gupta and Rao (1994) reported SOC stock as 24.3 Pg for the soil ranging from surface to an average subsurface depth of 44–186 cm. These data however, were based on only 48 soil series. The present OC stock has been estimated as 63 Pg in the first 150 cm depth of soils. Soils with their geographical distribution through the country were used for computing SOC stock in different depths in various physiographic regions.

Wang *et al.*, (2004) measured SOC at soil depths of 0–10, 0–20, 0–30, 0–50, and 0–100 cm. It was observed that mean SOC densities in the top 1 m ranged from 4.65 kg/m² for bare ground to 17.32 kg/m² in grassland land cover. SOC in the top 1 m of soil was estimated at 82.5 ± 19.5 Pg C. The total SOC pool held in the top 10, 20, 30, 50, and 100 cm were 22%, 41%, 54%, 74%, and 100%, respectively.

The carbon sink capacity of the world's agricultural and degraded soils is 50 to 66% of the historic carbon loss of 42 to 78 Gt of carbon. The rate of soil organic carbon sequestration with adoption of recommended technologies depends on soil texture and structure, rainfall, temperature, farming system, and soil management. Strategies to increase the soil carbon pool include soil restoration and woodland regeneration, no-till farming, cover crops, nutrient management, manuring and sludge application, improved grazing, water conservation and harvesting, efficient irrigation, agroforestry practices, and growing energy crops on spare lands (Lal, 2004)

Puget and Lal (2005) investigated five land use and soil management treatments in Ohio through $\delta^{13}\text{C}$ techniques. Significant differences in soil properties were observed among land use treatments for 0 to 5-cm depth. The SOC concentration (g C kg⁻¹ of soil) in the 0 to 5-cm layer was 44.0 in forest, 24.0

in meadow, 26.1 in No-Till (NT) corn, 19.5 in NT corn-soybean, and 11.1 in PT (plow till) corn. The fraction of total C in corn residue converted to SOC was 11.9% for NT corn, 10.6% for NT corn-soybean, and 8.3% for PT corn. The proportion of SOC derived from corn residue was 96% for NT corn in the 0 to 5-cm layer, and it decreased gradually with depth and was 50% in PT corn. The mean SOC sequestration rate on conversion from PT to NT was 280 kg C ha⁻¹ y⁻¹. The SOC had a long turnover time when located deeper in the subsoil

Leifeld *et al.*, (2005) investigated Swiss soils and noted that Mean SOC density in the layer 0–20 cm ranged between 40.6±8.9 t ha⁻¹ for arable land and 50.7±122 t ha⁻¹ for favourable permanent grassland, and in the layer 0–100 cm from 62.9±15.2 t ha⁻¹ for unfavourable grassland to 117 . 4 ± 29.8 t ha⁻¹ for temporary grasslands. Land-use type, clay content, and altitude (serving as a climate proxy for grassland soils at higher altitudes) were identified as main SOC predictors in mineral soils. Clay content explained up to 44% of the variability in SOC concentrations in the fine earth of arable soils, but was not significantly related to SOC in grassland soils at higher altitudes. SOC concentration under permanent grassland increases linearly with altitude, but because soil depth and stone content limit carbon storage in alpine grassland soils, no relationship was found between altitude and SOC stock.

Wetlands only cover ~9 % of the land area, but store 35 % of the terrestrial organic C stocks (Mitra *et al.*, 2005) mostly due to their high plant productivity combined with slow decomposition rates underwater-saturated conditions (Reddy and DeLaune 2008).

Arrouays *et al.*, (2006) detected a very strong correlation between SOC and clay content. Plante *et al.*, (2006) examined soils in Ohio and Saskatchewan and found a statistically significant relationship ($r^2=0.48$, $p<0.01$) between clay and SOC. According to this study, as the amount of clay increased, the amount of carbon retained in soil also increased The organic C contents of most of the wetland soils in Saskatchewan are consistently higher by a factor of two to five times compared to adjacent well-drained soils. Dawson and Smith, (2007) noted

that the level of SOC in a particular soil is determined by many factors including climatic factors (temperature and moisture regime) and soil related factors (soil parent material, clay content, cation exchange capacity)).

Annual compost inputs at 5-10 t per year led to a 1.2 kg C per square metre difference in carbon in the top 30cm compared with that in a neighbouring apple orchard that did not use compost (Deurer *et al.*, 2008). This change in soil carbon stock also benefited orchard soil health (Deurer *et al.*, 2008; 2009).

Across bioregions, forested mineral soil wetlands accumulate significantly more SOC than upland forest soils from similar locations, indicating that with drying, these systems could lose large quantities of C (Fissore, 2009).

Results of a long-term (30 year) experiment in the Indian Himalayas under rainfed soybean (*Glycine max* L.)—wheat (*Triticum aestivum* L.) rotation, showed that addition of FYM with N or NPK fertilizers increased SOC and TSN contents. The overall gain in SOC in the 0- to 45-cm soil depth interval in the plots under NPK + FYM treatment over NPK was 17.18 Mg C ha⁻¹ in 30 year. The rate of conversion of input C to SOC was about 19% of each additional Mg C input per hectare. SOC content in large size aggregates was greater than in smaller size aggregates, and declined with decreased aggregate size (Bhattacharyya *et al.*, 2010).

Changes in soil properties and plant community biomass were evaluated by comparing sites with freshwater treatment versus reference sites following freshwater addition to wetlands of the Yellow River Delta for 7 years. The results indicated that SOC significantly increased in all wetland sites that were treated with freshwater compared to the reference sites. The treatment wetlands had greater total nitrogen (TN), lower pH and electrical conductivity and higher water content in the soil compared to the reference wetlands. The SOC mean concentration was 5.44 -0.57 g/kg for the freshwater treated wetlands and 3.45 - 0.24 g/kg for the reference wetlands for the entire 0-40 cm soil depth. (Wang *et al.*, 2011).

Singh *et al.*, (2011) studied carbon stocks in the western Himalayan Mountains of India, the investigation revealed that SOC stock as well as concentration was the highest (101.8 Mg ha⁻¹ in 0–30cm, 227.97 Mg ha⁻¹ in 0–100cm) in temperate and the lowest in sub-tropical climate (37 Mg ha⁻¹ in 0–30cm, 107.04 Mg ha⁻¹ in 0–100cm). Pattern of SOC stock build up across the altitude was: temperate>lower alpine>upper alpine>sub-tropical. SOC stocks in all land uses across the climatic conditions, except agriculture in lower alpine, was higher (0.7 to 41.6%) in the deeper than upper soil depth. SOC stocks in both the depths showed quadratic relations with soil temperature and soil moisture. Other factors like fine soil particles, land-use factor and altitude influenced positively whereas slope and pH, negatively to the SOC stocks. In all climatic conditions, other than temperate, SOC stocks were greater in natural ecosystems like forests and pastures (112.5 to 247.5 Mg ha⁻¹) than agriculture (63 to 120.4 Mg ha⁻¹). In temperate climate, SOC stock in agriculture (253.6 Mg ha⁻¹) on well formed terraces was a little higher than forest (231.3 Mg ha⁻¹) on natural slope.

The period of time needed for soil OC stocks to change strongly enough to be detectable depends on the spatial variability of soil properties, the depth increment considered, and the rate of change. Cropland sites, having small spatial variability, had lower minimum detectable differences (MDD) with 100 sampling points (105 ± 28 gC m⁻² for the upper 10 cm of the soil) than grassland and forest sites (206±64 and 246±64 gC m⁻² for 0–10 cm, respectively). Expected general trends in soil OC indicate that changes could be detectable after 2–15 year with 100 samples if changes occurred in the upper 10 cm of stone-poor soils (Schrumpf *et al.*, 2011).

Sakin, (2012) noted a highly significant relationship between carbon and nitrogen contents (R²=0.99; p<0.01).. Soil organic carbon (SOC) concentration was found to be directly related to soil texture at all depths.. The strongest relationship was detected between SOC and clay (R²=0.96) and silt (R²=0.95). The relationship between sand and SOC was determined as (R²=0.65). Carbon and N stocks were determined as 10.53 Tg C and 1.96 Tg N, respectively.

Yan *et al.*, (2012) studied the land use change from cereal fields to greenhouse vegetable production with intermediate and high management intensity. The conversion led to increases in SOC and TN and decreases in C:N ratios in the top soil. The accumulation rates of C and N in the surface soil (0–30 cm) were estimated to be 1.37 Mg C ha⁻¹ yr⁻¹ and 0.21 Mg N ha⁻¹ yr⁻¹ over an average period of 8 years after cereal fields to greenhouse vegetable production conversion. Moreover, the addition of readily available C (manure) and N (manure and inorganic N) leads to a temporary stabilization of C in relatively labile SOM fractions, but to a preferential stabilization of N in organo-mineral SOM fractions

Twongyirwe *et al.*, (2013) assessed paired sites of natural forest and major land uses converted between 1973 and 2010 in afro-montane region. Within forests was found a marked site to site variation in SOC from 54.6 to 82.6 Mg/ha. There was a tendency for higher SOC in converted land, associated with higher bulk density suggesting quality based land use selection with forest left on inferior soils. Cultivation, landscape position, slope and sampling depth were all significantly ($P < 0.05$) related to variation in SOC stocks following forest conversion but time since conversion had no detectable impact. Interestingly, there was no significant relationship between SOC in the top and sub-soils. Higher SOC stock is largely determined by higher bulk density.

Venkanna *et al.*, (2014) documented that like organic carbon concentration, forest system (87.29 Mg ha⁻¹) shows the highest organic C stocks, followed by fodder (60.03 Mg ha⁻¹), paddy (57.12 Mg ha⁻¹), maize (52.12 Mg ha⁻¹), chilli (49.50 Mg ha⁻¹), red gram (48.44 Mg ha⁻¹), intercrop (45.69 Mg ha⁻¹), cotton (44.98Mg ha⁻¹), permanent fallow (44.81 Mg ha⁻¹) and is lowest in case of castor (36.86 Mg ha⁻¹) system. The soil organic carbon (SOC) stock was highest in Alfisols (52.84 Mg ha⁻¹) followed by Inceptisols (51.26 Mg ha⁻¹) and Vertisols and associated soils (49.33 Mg ha⁻¹).

Dorji *et al.*, (2014) observed significant influence of land use and land cover (LULC) categories on SOC concentration, SOC density, SOC stocks and their

spatial distributions, although this influence decreased with increasing soil depth. The estimated mean SOC densities in the top 1m were highest in fir forest soils (41.4 kg m^{-2}) and lowest in paddy land (12.0 kg m^{-2}). Allowing for LULC relative areas, mixed conifer forest had the highest SOC stocks in the upper meter (12.4 Mt) with orchards the lowest (0.1Mt). The total SOC stocks for the whole study area for the 0–5, 5–15, 15–30, 30–60 and 60–100cm depths were 2.6, 5.0, 6.5, 7.5 and 5.4Mt, respectively. The relative SOC densities indicate that the conversion of even a fraction of forests to other LULC types could lead to substantial loss of SOC stocks. This loss of SOC stock is even greater when the decrease in aboveground biomass is also taken into consideration.

Hoffmann *et al.*, (2014) observed a large variability of estimated SOC stocks, and thus a low predictability of the stocks based on regional data sets in the mountainous study area.

Alvaro-Fuentes, (2014) observed that differences in SOC stocks were only found in the upper 5 cm soil layer in which 4, 11 and 20 years under NT (no-tillage) showed greater SOC stocks compared with 1 year under NT and the CT (conservation tillage) phase. Despite no significant differences were found in the total SOC stock (0–30 cm soil layer) there was a noteworthy difference of 5.7 Mg ha^{-1} between the phase with the longest NT duration and the phase under conventional tillage. However, more than 75% of the total SOC sequestered was gained during the first 11 years after NT adoption.

The results indicate that grazing led to an increase in soil compaction and soil surface salinity, which significantly lowered levels of total N, P, and TOC in the soil surface. Grazing decreased soil microbial biomass C and N concentration in the lower riparian floodplain wetland, whereas it significantly increased soil microbial biomass C and N concentration in the higher riparian floodplain wetland. Elevation differences in the riparian floodplain wetland increased spatial heterogeneity in the soil and thus resulted in different influence of grazing on wetland soils and ecosystem. Therefore, elevation differences and grazing

intensity were the main factors controlling soil characteristics in the riparian floodplain wetland of this region (Wang *et al.*, 2015).

Xiong *et al.*, (2014) have documented that SOC stocks significantly differed among LULC classes – sugarcane and wetland contained the highest SOC, followed by improved pasture, urban, mesic upland forest, rangeland, and pineland while crop, citrus and xeric upland forest remained the lowest. The surface 20 cm soils acted as a net sink for C with the median SOC significantly increasing from 2.69 to 3.40 kg m⁻² over the past decades. The SOC sequestration rate was LULC dependent and controlled by climate factors interacting with LULC. Remarkable site-specific SOC losses were involved in the conversions of wetland to other LULC types, and vice versa. At regional scale, restoration of wetlands contributed to the buildup of SOC stocks.

In temperate soils, low rate of OM decomposition leads almost invariably to the accumulation of organic material in soils that are poorly drained. Under prolonged rice (submerged) environment, there is a net addition of OM in soil, whereas the balance is generally negative under arable cropping systems. Soils under lowland rice system preferentially accumulate OM and are important for sequestering atmospheric C (Pal *et al.*, 2015).

Drewniak (2015) applied Community Land Model (CLM) to investigate the impacts of various management practices, including fertilizer use and differential rates of crop residue removal on the SOC storage of croplands in the continental United States over approximately a 170-year period. Results indicate that total US SOC stocks have already lost over 8 Pg C (10 %) due to land cultivation practices (e.g., fertilizer application, cultivar choice, and residue removal), compared to a land surface composed of native vegetation . After long periods of cultivation, individual sub-grids (the equivalent of a field plot) growing maize and soybean lost up to 65 % of the carbon stored compared to a grassland site. Crop residue management showed the greatest effect on soil carbon storage, with low and medium residue returns resulting in additional losses of 5 and 3.5 %, respectively,

in US carbon storage, while plots with high residue returns stored 2 % more carbon.

Shi *et al.*, (2015) observed that after farmland conversion to apple orchard, total SOC content and density and soil alkaline phosphatase activity significantly decreased. The LOC content, its proportion to TOC content, and carbon management index (CMI, changes in the total content and lability of SOC significantly increased in the 0–40 cm soil layer.. With increasing age of apple orchards, SOC stocks significantly increased after 10 years, being more than 10% larger in mature and over mature orchards than in adjacent farmlands. The LOC content and CMI value also had an increasing trend, while soil enzyme activities showed different response patterns. There were significant correlations between soil enzyme activities, SOC fractions, and CMI value ($P < 0.05$). Long-term apple cultivation was effective to restore SOC stocks, although it took over a decade to rebuild a new increasing trend after farmland conversion.

Rabbi *et al.*, (2015) observed the stocks of soil carbon (0–0.3 m, adjusted for variation in bulk density) varied widely under different land uses (2–241 Mg C ha⁻¹) across the 1482 sites surveyed. The SOC stock varied from 11 to 241 Mg C ha⁻¹ under pasture, whereas the range was 15–43 Mg C ha⁻¹ under irrigated cotton. Among the crop-pasture rotations, the crop dominant rotations had lower SOC stock (2–97 Mg C ha⁻¹) than pasture dominant rotations (6–166 Mg C ha⁻¹). The mixed annual-perennial pasture (11–230 Mg C ha⁻¹) and perennial pastures (8–241 Mg C ha⁻¹) generally had the widest range and maximum average SOC stock. The sites where residue was retained appeared to have higher soil carbon stock compared to other systems.

Zhang *et al.*, (2015) found that the average retention of maize-derived carbon plus manure-derived carbon during the early period of the trial (up to 11 years) was relatively high (10%) compared to the later period (22 to 27 years, 5.1–6.3%). About 11% of maize-derived carbon was converted to soil organic carbon, which was double the retention of manure-derived carbon (4.4–5.1%). This result

emphasized that organic amendments were necessary to a win-win strategy for both SOC sequestration and maize production.

Bajgai, *et al.*, (2015) observed that besides, scant crop residue input, tillage to control weeds in vegetable systems reduces soil organic carbon (SOC) levels. Organic fertilizers increased TOC whilst simulated tillage decreased it in laboratory. In field residue incorporation in soil increased TOC, but organic fertilizers behaved inconsistently. However, in a laboratory microcosm, both residue and organic fertilizers individually mitigated tillage-induced loss of TOC.

Hombegowda *et al.*, (2016) noted that conversion of forest to agriculture resulted in a large loss to the original SOC stock (50–61 %) in the top meter of soil depending on the climate zone. The most important variable regulating SOC stocks and its changes was tree basal area, possibly indicative of organic matter inputs. Furthermore, climatic variables such as temperature and precipitation, and soil variables such as clay fraction and soil pH were likewise all important regulators of SOC and SOC stock changes.

2.5 Influence of different land uses on soil organic carbon fraction

For characterization and functional purposes, SOM is generally subdivided into different fractions or compartments. The approaches for fractionation may broadly be categorized as chemical, physical and biological or biochemical. Additionally, some morphological characteristics are also used to distinguish the development of different humus forms in terrestrial ecosystems. Since SOM is a continuum of complex heterogeneous material, no single fractionation approach may be expected to adequately characterize the turnover rates of the whole soil. The biological or functional pools, that are mostly model-defined, may or may not be related to some chemically or physically defined fractions (Nieder and Benbi., 2008). The fractions of soil organic carbon included in the present study are: Microbial Biomass Carbon (MBC), Labile Organic Carbon (LOC) and Particulate Organic Carbon (POC), besides, the total organic carbon under the aforementioned land uses.

2.5.1 Microbial biomass carbon.

The microbial biomass accounts for only 1- 3 % of soil organic C but it is the eye of the needle through which all organic material that enters the soil must pass (Jenkinson, 1977). Microbial biomass C and N pools serve as a source (mineralization) or a sink (immobilization) of labile nutrients. Studies have shown that microbial biomass responded quickly to change in soil perturbation by tillage (Carter, 1986) and soil moisture (Skopp *et al.*, 1990).

Gupta and Germida (1988) and Anderson and Domsch (1989) investigated the effects of cultivation on the nature, distribution, and activity of microbial biomass content in different microaggregate size classes of a native and cultivated soil. Microaggregates (<0.25mm) of both soils contained lower organic-C, microbial biomass, enzymes and respiration compared to macroaggregates. The negative effects of cultivation were more pronounced on macroaggregates. These results suggest that microbial biomass, especially fungi, play an important role in the formation of macroaggregates and are the labile organic matter that serves as the primary source of C and nutrients released following cultivation.

Follett and Schimel, (1989) studied changes in microbial biomass dynamics and N cycling in soils formed under grassland vegetation in western Nebraska, and farmed under wheat-fallow since 1970. After 16 year of cultivation, soil microbial biomass levels in top 10 cm soil were decreased to 57, 52, and 36% for no-till, stubble mulch, and moldboard plow respectively, compared to grass. CO₂ respiration was proportional to microbial biomass, but N mineralization was not. It appeared that C availability for microbial growth declined with increased tillage intensity, which also decreased the soil's ability to immobilize and conserve mineral N.

Microbial biomass C (MBC) is recognized as a sensitive indicator of cultivation-induced changes in both SOC and biological properties of soil quality and soil health (Anderson and Domesch, 1989; Powelson *et al.*, 1987 ; Karlen *et al.*, 1997). The ratio of microbial biomass carbon (MBC) to total organic carbon (TOC) was measured by Anderson and Domesch, (1989) in soils from long-term agricultural experiments located in the temperate climatic zone of Central Europe.

This survey of MBC-to-TOC ratios indicated that a higher concentration of microbial carbon is characteristic of the crop rotation.. Mean % MBC in TOC amounted to 2.3 for permanent monocultures and to 2.9 for continuous crop rotations, respectively. The differences in C-to-C ratios between the two management systems is thought to be caused by the difference in cropping management rather than by soil texture, i.e. clay content. These MBC-to-TOC ratios rose to 4 or 3.7% in both plots under monoculture and plots under rotation, when the plots had received organic fertilizer the year prior to sampling. This rise in MBC over TOC is seen as a transient rather than an absolute phenomenon and is believed to be due to the easily available carbon fraction of the introduced organic material.

The microbial biomass itself may represent a labile pool of C and nutrient elements. In agricultural soils 200-1000 μg biomass C g^{-1} soil is often found. This cell mass fixes 100-600 kg N and 50-300 kg P per hectare in the upper 30 cm of soil. These amounts often exceed the annual application of nutrients supplied as fertilizer to soils in agricultural practice. The liberation or fixation of these nutrients depends on the life dynamics of the microorganisms. Growth of biomass and fixation of nutrients is promoted by rhizodeposits and plant debris and the liberation of nutrients is the consequence of microbial death (Martens, 1995).

The material added to soil are converted by microorganisms in order to generate energy and to produce new cellular metabolites to support their maintenance and growth. In the C-limited soil system available C in organic materials entering the soil is the driving force behind these processes but other essential nutrient elements (particularly N, P, K) are also involved. Under suitable environmental conditions the extent of the turnover will mainly be controlled by the size and activity of the microbial biomass (Martens, 1995). Microbial biomass C and N pools represent vital components of ecosystem cycling with a turnover time from days to years (Hu *et al.*, 1997),

Data indicated that young soils such as Inceptisols and Entisols had lower levels of microbial biomass carbon and labile organic carbon than older soils such

as Ultisols and Oxisols. However, these young soils did not differ from the older soils in the proportion of microbial biomass carbon to total organic carbon (Zou *et al.*, 2005).

While studying the impact of different land uses (forest, pasture, and agricultural lands) on soil microbial biomass carbon and nitrogen using the chloroform fumigation extraction (CFE) method. The average microbial biomass C were found as 1028.29 $\mu\text{g g}^{-1}$, 898.47 $\mu\text{g g}^{-1}$, and 485.10 $\mu\text{g g}^{-1}$, respectively, for forest, pasture, and agricultural soils. As with microbial biomass C, the average microbial biomass N was found as 129.99 $\mu\text{g g}^{-1}$, 100.90 $\mu\text{g g}^{-1}$, and 42.60 $\mu\text{g g}^{-1}$, respectively, for forest, pasture, and agricultural soils. Microbial biomass C and N were shown to be significantly correlated to the physico-chemical properties of the soil, such as organic C, total N, clay, and pH. The study shows that land use has a significant effect on microbial biomass C and N in soil by altering natural soil characteristics under the same ecological conditions (Kara and Bolat., 2008)

Zagal *et al.*, (2010) used maize as a reference plant material and natural ^{13}C for field assays of soil carbon dynamics. The labeling technique showed that a very low amount was incorporated to the soil (new C) during the time period of the experiment (4 years) and the ratio of remaining C / added C was very low suggesting that the high carbon content of the soil can therefore be considered as due to a large amount of passive carbon, or to ancient carbon inputs, that have saturated the sorption capacity.

Xu *et al.*, (2013) estimated the concentrations, stoichiometry and storage of soil microbial biomass carbon (C), nitrogen (N) and phosphorus (P) at biome and global scales. The results show that concentrations of C, N and P in soils and soil microbial biomass vary substantially across biomes; the fractions of soil elements C, N and P in soil microbial biomass are 1.2, 2.6 and 8.0%, respectively. The best estimates of C:N:P stoichiometry for soil elements and soil microbial biomass are 287:17:1 and 42:6:1, respectively, at global scale, and they vary in a wide range among biomes. The vertical distribution of soil microbial biomass follows the

distribution of roots up to 1 m depth. The global storage of soil microbial biomass C and N were estimated to be 16.7 Pg C and 2.6 Pg N in the 0–30 cm soil profiles, and 23.2 Pg C and 3.7 Pg N in the 0–100 cm soil profiles. The spatial patterns of soil microbial biomass C and N were consistent with those of soil organic C and total N, i.e. high density in northern high latitude, and low density in low latitudes and the Southern Hemisphere.

2.5.2 Particulate organic carbon.

Cambardella and Elliott (1991) determined that the POM C in the native sod represented 39% of the total soil organic C. Twenty years of bare-fallow, stubble-mulch, and no-till management reduced the TOC content in this fraction by 18, 19, and 25%, respectively. Analysis of the POM fraction for lignin and cellulose content indicated that this fraction was 47% lignin and had a lingo-cellulose index of 0.7. The stable C-isotope composition of the POM fraction suggests that wheat-derived POM turns over faster than grass-derived POM. The authors suggest that the POM fraction closely matches the characteristics of a SOM pool variously described as slow, decomposable, or stabilized organic matter.

Most studies dealing with POC in soils extract the POC fraction from the bulk soil on the basis of particle size separation, by extraction, density flotation or sieving (Oades and Waters, 1991; Norden *et al.*, 1992; Chow *et al.*, 2005). Due to its greater lability, POC was a more sensitive indicator of land use changes and management than SOC (Cambardella and Elliott, 1992, 1993).

Kapkiyai *et al.*, (1999) noted that addition of manure and retention of maize stover reduced the soil carbon loss by 49%.. Carbon balances suggest that particulate organic matter is more efficiently restocked by manure than maize stover. Particulate organic matter is a key fraction for understanding soil fertility changes.

According to Adejuyigbe *et al.*, (2000), the particulate organic matter is a more labile fraction of the soil organic matter which is the most readily formed and when it decomposes, it serves as an important substrate for mineralization

process in the soil. The level of this fraction therefore, could be an essential determinant of the fertility status of Savanna soils.

Chan (2001) studied changes in particulate organic carbon (POC) relative to total organic carbon (TOC) were measured in soils covering a wide range of different land use and management practices. POC made up 42–74% of TOC and tended to be greater under pasture and more conservative management than traditional cropping regimes. It was the form of organic carbon preferentially lost when soils under long-term pasture were brought under cultivation. It was also the dominant form of organic carbon accumulating under more conservative management practices (direct drilling, stubble retained and organic farming). Across all sites, changes in POC accounted for 81.2% (range 69–94%) of the changes in total organic carbon caused by differences in land use and management.

Yang *et al.*, (2005) observed that in the combined application of chemical fertilizers and farmyard manure or wheat straw, continuous waterlogging can markedly increase total soil organic carbon, but significantly reduce the labile paddy soil organic C fractions. In the alternative wetting and drying, incorporation of farmyard manure or wheat straw can increase the proportions of EOC, LFOC, POC, MBC, and MNC in TOC, but also improve soil water stable aggregation, compared to continuous waterlogging. The continuous waterlogging results in a lack of clear relationship between soil water stable aggregation and clay dispersion and soil organic C fractions

Fourty-four soils from under native vegetation and a range of management practices following clearing were analysed for 'labile' organic carbon using both the particulate organic carbon (POC) and the 333 mM KMnO₄ (MnoxC) methods. Although there was some correlation between the 2 methods, the POC method was more sensitive by about a factor of 2 to rapid loss in OC as a result of management or land-use change (Skjemstad *et al.*, 2006)

Bescansa *et al.*, (2006) stated that particulate organic carbon is an accurate soil quality indicator despite soil differences, and therefore advocated that it should be

considered when assessing the quality of different soil management practices. POC refers to the organic carbon which is combined with soil sand fractions (53~2000 μm) and comprises primarily semi-decomposed plant residues (Geng *et al.*, 2009). Soil aggregation is a process which is controlled by SOC and POC dynamics (Bouajila, 2010).

Figueiredo *et al.*, (2010) studied the dynamics of systems of tillage and crop rotation, alternating in time and space. Changes in organic C by land use occurred mainly in the fraction of particulate organic matter (> 53 μm). Proper management of grazing promoted increased levels of particulate organic matter by association with larger aggregates (2–8 μm), demonstrating the importance of the formation of this aggregate class for C protection in pasture

Sreekanth *et al.*, (2013) noted considerable differences in POC contents between soils of different forest categories. The values varied between 1.57 to 3.99% with an average of 2.6% for the bulk soil. The POC content ranged in the order: Shola Forest-SF(3.99%) > Deciduous Forest-DF(3.24%) > Riparian Forest –RF (1.61%) > thorny forest-TF (1.57%). The data showed extensive difference in the POC content between soils in different sub soil horizons in the study. The value ranged between 0.6 to 4.77% with an average of 2.6 %. As a general trend, the POC content increased with depth. However, POC concentration and stock has the same trend to SOC as this parameter was correlated with SOC. Since POC has been shown to be the labile fraction of SOC, it can be expressed as a percentage of SOC. The POC/SOC ratio represents the contribution of POC to SOC and for the bulk soil the maximum value was recorded under SF (86.92%) and the minimum (38.48%) in the TF. Across all sites, POC made up to 20 to 93 % of SOC.

There was a significantly positive relationship between the content of heavy fraction organic carbon, particulate organic carbon and total soil organic carbon. The ranges of soil light fraction organic carbon ratio and content were 0.008% - 0.15% and 0.10-0.40 g kg^{-1} , respectively, and the range of particulate organic carbon ratio was 8.83% - 30.58%, indicating that the non-protection component of

soil organic carbon was low and the carbon pool was relatively stable in this wetland (Dong, *et al.*, 2013).

Liu *et al.*, (2014) concluded that land- use strongly affected the amount of biomass input into the soil. Specifically, the occurrence of aggregates, aggregate SOC and aggregate easily oxidizable organic carbon (EOC) under grassland and forestland were normally higher than that under farmland. Furthermore, conventional cultivation destroyed aggregates. The dominant aggregate size fractions under farmland were < 0.5 mm, whereas the dominant aggregate size fractions under grassland and forestland were > 0.5 mm. In farmland, SOC and EOC tended to concentrate in < 1 mm macro-aggregates, whereas they tended to concentrate in > 1 mm macro-aggregates under grassland and forestland. EOC tended to concentrate in larger aggregates than SOC. The small fractions of aggregates formed large fractions by combining with fresh organic matter, and the fixed organic carbon in the smaller aggregates was more stable. So converting slope farmland to forestland and grassland could improve the storage and quality of SOC, and the tendency of SOC transfer. Wiesmeier *et al.*, (2014) observed that forest soils were characterized by distinctly lower proportions of intermediate and passive SOC and a high amount of active SOC in form of litter.

Kaori, *et al.*, (2014) evaluated the changes in SOC under different cropping periods followed by short fallow through changes in the composition and the amount of vegetation as input returned to soil. The results showed that SOC decreased significantly after 10-year cropping in response to a low amount of input. However, coarse organic matter carbon (COM-C) decreased after only 1-year cropping, and COM-C and POC decreased gradually during cropping for 2–5 years.

2.5.3 Labile organic carbon

Soil labile organic carbon is the most active fraction of soil organic carbon with rapid turnover rates, and it changes substantially after disturbance and management (Coleman *et al.*, 1996). Soil labile organic carbon has been represented operationally by specific fractions derived using chemical and

physical separation techniques. Chemical fractionation relies on the solubility of organic carbon in oxidizer, acid or base, producing soluble organic carbon (Tirol-Padre and Ladha., 2004). The labile fraction is more sensitive to short-term vegetation or land-use change (Sicardi *et al.*, 2004; Von-Lutzow *et al.*, 2007). The labile fraction of OC consists of micro-organisms, plant and soil fauna at different stages of degradation resulting in easily decomposable non-humic organic substances like carbohydrates, proteins, organic acids, amino acids, waxes, and other non-specific compounds (Poirier *et al.*, 2005).

Zou *et al.*, (2005) applied a modified fumigation–incubation technique to study five mineral soils and a forest Oa horizon to represent a wide range of organic carbon levels. The study noted that soil labile organic carbon varied from 0.8 mg/g in an Entisol to 17.3 mg/g in the Oa materials. Potential turnover time ranged from 24 days in an Alfisol to 102 days in an Ultisol. Soil labile organic carbon contributed from 4.8% in the Alfisol to 11.1% in the Ultisol to the total organic carbon.

Forest conversion from native broadleaf forest to plantation affects only the soil MBC but not the other labile SOC fractions (Wang and Wang 2007; Xu *et al.*, 2008 and Wang *et al.*, 2013). Results indicate that the response of labile SOC fractions to forest conversion depends on forest vegetation. Litter and root are the primary source of SOC. Therefore, the different quantities and qualities of organic matter input through litter fall and root activity (e.g., turnover and exudates) induced by the different tree species were responsible for the different responses of labile SOC to forest conversion (Vesterdal *et al.*, 2008).

Geng *et al.*, (2009) noted that terms used to characterize soil LOC in global soil science communities include, particulate organic carbon (POC), light fraction organic carbon (LFOC) readily oxidized carbon (ROC), soil microbial biomass carbon (SMBC) and dissolved organic carbon (DOC) etc.).

The soil organic carbon (SOC) and labile organic carbon (LOC) stores were investigated at arable land (AL), artificial grassland (AG), artificial woodland (AW), abandoned arable land (AAL) and desert steppe (DS) in a Loess Plateau.

The LOC stocks changed by a larger magnitude than the SOC stocks which suggests that it is a more sensitive index of carbon dynamics under a short-term LUC. The LOC stocks increased at 0–20 cm and 0–100 cm depths from DS to AW, which is opposite to that observed for SOC. The proportion of LOC to SOC ranged from 0.14 to 0.20 at the 0–20 cm depth for all the five land use types, indicating low SOC dynamics. The allocation proportion of LOC increased for four types of LUC conversion, and the change in magnitude was largest for DS to AW (40.91%). (Zhang *et al.*, 2012).

Cao *et al.*, (2013) elucidated the response of soil LOC under different grazing intensities. The results showed that the LOC including POC, light fraction organic carbon, and readily oxidizable carbon content at a depth of 0-15 cm decreased with the increasing grazing. The TOC and LOC content decreased with increasing depths of soil horizons. According to Wiesmeier *et al.*, (2014) the high amount of labile SOC in forest top-soils poses the risk of considerable SOC losses caused by mechanical or other disturbances.

Xia *et al.*, (2015) observed that microbial available carbon (MAC), microbial biomass carbon (MBC), easily oxidizable carbon (EOC), water-soluble organic carbon (WSOC), light fraction carbon (LFC) content increased along the elevation gradient across the soil depths, whereas LFC was higher in communities with low elevations compared to others across all soil depths. Positive correlations were found among SOC pools (MAC, MBC, EOC, WSOC, and SOC) ($P < 0.001$) across communities, except for LFC. LFC was positively correlated to other pools at low elevations and high elevations, respectively. Overall, LOC pools decreased with increasing soil depth across communities. The results suggest that LOC content principally depends on the amount of SOC, and LOC groups are good indicators for predicting minor changes of SOC in the C cycle.

Chapter – 3

MATERIALS AND METHODS

3.1 Description of the study area

3.1.1 Location

The study was carried out in north eastern zone of Kashmir in western Himalayan region. The area includes district Ganderbal and some parts of district Bandipore of Jammu and Kashmir State, and lies between the longitudes of 74°30' and 75°30'E and latitudes of 34°07' and 34°30'N. Ganderbal has an average elevation of 1619 m amsl. It is bordered by district Srinagar in the south, Bandipore to the north, Kargil in the north east, Anantnag to the south east and Baramulla in the south west. The sites in district Bandipora that formed the part of this study lie in the north of Ganderbal (Fig. 3.1). The area as a whole can be subdivided into three physio-graphic sub-divisions: plains, foothills and upper-hills.

3.1.2 Climate and Soil

The study area (Fig 3.1) falls under the temperate to Mediterranean type of climate and is characterized by mild summers and chilling winters. Due to altitudinal variation there is a wide variation in climatic conditions in different parts, ranging from a typical temperate climate in high altitude which experience snowfall and severe cold in the winter and warmer climate at low altitude. The winter commences from early November and lasts till end of March. Most of the precipitation received during this period is in the form of snow and the temperature, at times falls as low as -13°C . In December-January the minimum temperature is generally below freezing point. The period from March to June constitutes warm summers with temperature rising up to 33°C . However, it remains cold almost throughout the year in the uppermost reaches. The average precipitation is about 680 mm (CGWB, 2013).

The soils of the area are broadly divided into two types viz, Hapludalfs and Ochraqualfs. Hapludalfs are found on Karewa tops and uplands with a slope variation of 1-3%. The soils are very deep, well drained with moderate

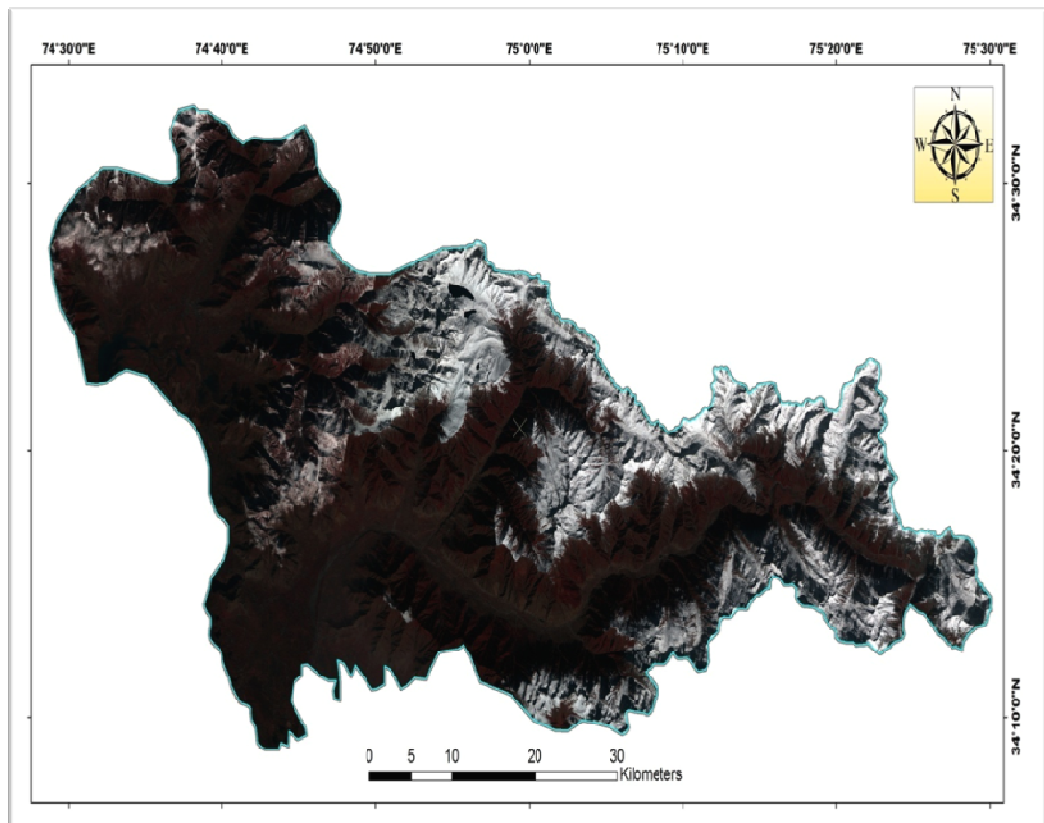


Fig. 3.1: Map of the study area.

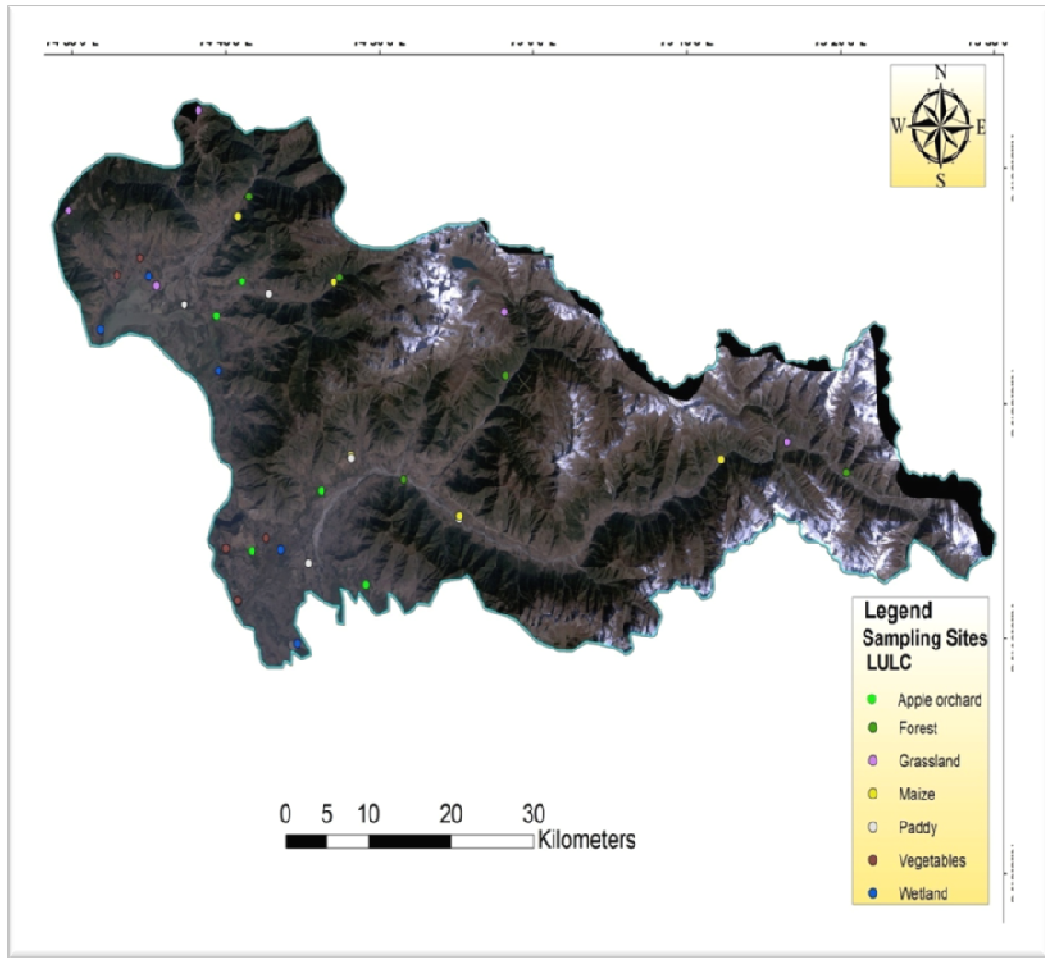


Fig. 3.2: Map showing study sites in different land use types of the study area.

permeability. These are medium to fine textured soils and the surface texture varies from clay loam to silty clay loam. The colour of the soils varies from Yellowish Brown to dark Brown. These soils are severely eroded resulting in the formation of gullies and ravines. Ochraqualfs are found in plain to mid upland topography. These soils are moderately fine textured with clay loam as the predominant surface texture. The extent of erosion on such soils is much less. These are dark brown to dark yellowish brown in colour with low permeability (CGWB, 2013). However, the study conducted on the soils of north-western Himalayan ecosystem by Sidhu and Surya (2014) also indicate the presence of Entisols and Inceptisols in the study area.

3.1.3 Land use and vegetation

Paddy is the main field crop of the zone, cultivated over 50% of the net sown area. It is cultivated in plains, terraces on foothills, as well on Karewas. Maize is the second most important crop, comprising about 25% of the net sown area. Maize is mostly cultivated at subsistence level in very small family-owned fields on hills, often adjacent to forests. Low yielding varieties of maize are cultivated for fodder and grain, largely for local consumption. Vegetables are cultivated in small fields or kitchen gardens throughout the study area, although commercial vegetable cultivation is practiced at few places near the town. Apple is the major fruit crop of the region which is cultivated on well drained plains, foot hills and even on hills up to the altitude, where local temperature and climate permits. Together, fruit and vegetable cultivation covers about 17% of the net sown area. Chemical fertilizers like Urea, DAP and MOP are widely used in apple orchards, paddy and vegetables. Organic manure in the form of cattle dung or farmyard manure is applied to all the crops. Sheep and goat dung as manure is mostly used in maize fields on hills, although small quantities of chemical fertilizers viz., urea and DAP, with no/ very little MOP are used at few places.

Temperate forests of the area are dominated by conifers including *Pinus wallichiana* (Blue Pine/Kail), *Cedrus deodara*(Himalayan Cedar/Deodar), *Abies pindrow*(Fir) and *Picea smithiana* (Spruce). Additionally associations of *Taxus*

baccata (Himalayan Yew) and *Juniperus recurva* (Juniper) can also be found (Wani *et al.*, 2013; 2014). The dominant species of the grasslands/pastures include *Cynodon dactylon L.*, *Bothriochloa pertusa L.*, *Medicago lupulina*, *Trifolium repens*, *Plantago lanceolata L.*, *Plantago major L.*, *Cyperus sps. etc.* The problem of grazing and degradation of pastures is more critical and intense here, due to the lifestyle of its rural people especially Gujjar and Bakerwal communities who manage herds of cattle. (Ahmad *et al.*, 2013).

There are many water bodies and wetlands in the area. Nallah Sindh, a major tributary to the Jhelum river flows through Ganderbal from north to south. Riverine wetlands in the area are located along side the Sindh. One of the Asia's biggest freshwater lakes; Wular, lies in Bandipore. The lake with its associated wetlands has been designated as a wetland site of international importance under the Ramsar Convention of 1990. The soils associated with the selected riverine and lacustrine wetland sites of the area, were chosen for the present study. Although thick organic layer, indicating the presence of Histic epipedon, was observed in areas immediate to the peripheries of water bodies, however, only the mineral soils located farther away from these organic layers of the wetlands were sampled and studied in the present investigation. Pertinently, these soils in the vicinity of wetlands are often encroached upon and used for cultivation, and therefore, observing their SOC stock and quality will be more useful as far as the impact of land use change is considered.

3.2 Design of survey and soil sampling

The ecosystems/land use types (Table 3.1) investigated in the study include: (A) Agricultural Ecosystem [(a) Paddy (b) Maize (c) Vegetables (d) Apple orchards]. (B) Grassland Ecosystem. (C) Forest Ecosystem (D) Wetland Ecosystem.

Table 3.1: **Ecosystems and land use types covered in the study**

Ecosystem	Land use type	Locations
Agriculture	Paddy	5
	Maize	5
Horticulture	Apple	5
	Vegetable	5
Forest	Forest	5
Grassland	Grassland	5
Wetland	Wetland	5

The geographic locations and altitudes (m amsl) of the study sites are presented in Table 3.2. Purposive sampling method was followed and samples were collected in autumn 2014. The samples were taken from two depths: 0-15 and 15-30 cm. Composite soil samples were collected from five locations in each land use type (Fig 3.2), with three replicates. From every replicate sample was composited from six subsamples randomly collected, pooled and sieved. The field moist soil samples were kept stored in refrigerator at temperature less than 4°C for preserving the enzyme activities till the analyses were over. Soil moisture was determined after drying at 105°C for 24 hours.

Table 3.2: Geographic location and altitude of the study sites.

Study site	Location	Altitude (m, amsl)
Paddy		
Baghurampora	34° 13.214' N 74° 45.358' E	1601
Gamroo	34° 24.203' N 74° 37.290' E	1608
Sumlar	34° 24.653' N 74° 42.797' E	1768
Margund	34° 15.078' N 74° 55.157' E	1830
Yarmuqam	34° 17.664' N 74° 48.159' E	1880
Maize		
Margund	34° 15.218' N 74° 55.177' E	1832
Athwot	34° 27.949' N 74° 40.785' E	1892
Yarmuqam	34° 17.787' N 74° 48.147' E	1927
Gagangir	34° 17.623' N 75° 12.227' E	2287
Kudara	34° 25.161' N 74° 47.013' E	2370

Contd. on Page 2

Apple		
Nadihal	34° 23.700' N 74° 39.365' E	1640
Badampora	34° 13.712' N 74° 41.696' E	1662
Khalmulla	34° 12.283' N 74° 49.095' E	1680
Watlar	34° 16.266' N 74° 46.213' E	1690
Arin	34° 25.164' N 74° 41.052' E	1716

Vegetable		
Aloosa	34° 25.290' N 74° 32.560' E	1585
Ahan	34° 13.710' N 74° 39.610' E	1598
Dab_Shalhara	34° 11.622' N 74° 40.759' E	1601
Barsoo	34° 14.281' N 74° 42.599' E	1619
Quilmuqam	34° 26.176' N 74° 34.422' E	1646

Forest		
Preng	34° 16.756' N 74° 51.562' E	1680
Athwotu	34° 28.794' N 74° 41.496' E	1881
Kudara	34° 25.331' N 74° 47.133' E	2430
Naranag	34° 21.128' N 74° 58.135' E	2507
Sonamarg	34° 17.055' N 75° 20.390' E	2750

Grassland		
Potshahi	34° 25.008' N 74° 35.476' E	1530
Sonamarg	34° 18.375' N 75° 16.538' E	2588
Nagmarg	34° 28.195' N 74° 29.751' E	2887
Trundkhul	34° 23.888' N 74° 58.103' E	3330
Tragbal	34° 32.425' N 74° 38.221' E	3539

Wetland		
Potshahi	34° 24.989' N 74° 35.246' E	1520
Tulmulla	34° 13.778' N 74° 43.585' E	1570
Kehnoosa	34° 23.151' N 74° 31.858' E	1577
Saderkoot	34° 21.397' N 74° 39.518' E	1580
Shalabugh	34° 09.671' N 74° 44.581' E	1586

3.3 Estimation of Physico-chemical properties of the soils

3.3.1 Bulk density

Bulk density of soil at each site was determined by the core method (Blake and Hartge, 1986)

3.3.2 Particle size analysis

Percent clay, sand and silt were determined by the hydrometer method (Bouyoucos, 1962)

3.3.3 Gravimetric water content

Soil moisture content was determined on oven-dry basis. Therefore, the actual soil moisture in the samples was considered (Hesse, 1971).

3.3.4 Porosity

Percent pore space (total porosity) was computed from bulk density (BD) and average particle density (PD) of 2.65 g cm^{-3} as $\{1 - \text{BD}/\text{PD}\} \times 100$ (Brady and Weil, 2002)

3.3.5 EC and pH

Electrical conductivity and reaction of the soil samples was determined in 1:2.5 soil :water ratio (w/v) with the help of combined electrode for EC and pH, as per the procedure given by Jackson(1973).

3.3.6 Available Nitrogen

Available N was determined by Alkaline Permanganate Method (Subbiah and Asija ,1956) involving distillation of the soil with alkaline KMnO_4 in presence of NaOH and determining the ammonia absorbed in a known volume of boric acid , through titration with standard sulphuric acid.

3.3.7 Available Phosphorus

Olsen's extractant 0.5 M NaHCO_3 (pH 8.5) was used for the extraction of available phosphorus. Phosphate in the extract was determined colorimetrically by the ammonium molybdate blue colour method using a spectrophotometer at 760 nm (Olsen *et al.* 1954).

3.3.8 Available Potassium

Available K was determined in the neutral normal ammonium acetate extract of soil with help of flame photometer (Jackson 1973).

3.3.9 Available Sulphur

Sulphur was determined with 0.15% CaCl_2 solution and the S in the extract was estimated turbidmetrically (Williams and Steinbergs 1959).

3.3.10 Exchangeable Calcium and Magnesium

Exchangeable calcium and magnesium was estimated in the ammonium acetate extract of soil by titration with EDTA (Jackson, 1973)

3.3.11 Cation exchange capacity

Cation exchange capacity (CEC) was estimated following method of Jackson (1974).

3.4 Estimation of soil organic carbon and its fractions

3.4.1 Organic Carbon (Walkley and Black organic carbon).

Soil organic carbon was determined by the method given by Walkley and Black (1934). In this method organic matter was first oxidized with chromic acid (potassium dichromate + concentrated sulphuric acid) and the unconsumed potassium dichromate was back-titrated against ferrous sulphate (redox titration).

3.4.2 Total soil organic carbon

Total organic carbon in the soil was determined by the wet oxidation method (Snyder and Trofymow 1984). For this purpose 1.0 gm of soil (passed through 1 mm sieve) was pretreated with 3.0 ml of 2N HCl to remove carbonates. Then soil was oxidized with $\text{K}_2\text{Cr}_2\text{O}_7$ in presence of conc. H_2SO_4 and H_3PO_4 in a ratio of 3:2 by heating on digestion block for 2 hrs. The CO_2 thus evolved was trapped in 2N NaOH and amount of CO_2 (trapped) was measured by back titration with 0.5N HCl using phenolphthalein indicator. Total organic carbon content was computed based on amount of CO_2 evolved.

3.4.3 Labile soil organic carbon

The amount of oxidizable carbon by 333 mM KMnO_4 (labile carbon) in soil was determined by following the procedure of Blair *et al.* (1995). For this purpose, 2.0

g of soil was taken in a centrifuge tube and oxidized, with 25 ml of 333 mM KMnO_4 by shaking on a mechanical shaker for 1 hr. The tube were centrifuged for 5 minute at 4000 r.p.m. and 1.0 ml of superannuated solution was diluted to 250 mL with double distilled water (DDW). The concentration of KMnO_4 was measured at 565 nm wavelength using spectrophotometer. The change in concentration of KMnO_4 was used to estimate the amount of carbon oxidized assuming that 1.0 mL of MnO_4^- was utilized in the oxidation of 0.7 mM (9.0 mg) of carbon.

3.4.4 **Particulate organic carbon**

Particulate organic carbon was determined by method given by Camberdella and Elliot (1992). For this purpose 10 gm of 2 mm sieved air dry soil sample was shaken with 0.5% sodium hexametaphosphate for 24 hr. Then soil suspension was passed through 0.053 mm sieve by spraying water from the top of sieve. The solid portion, retained on sieve was termed as particulate organic matter. The solid portion was transferred to small pre-weighed plastic boat by washing with spray of water. It contained both particulate organic matter and sand portion. The plastic boats were kept inside the forced air oven at 50 °C temperature for 72 hrs for drying and finally the weight of boat recorded. The solid material in the boat was grounded in a pestle and mortar to make a fine powder. The total organic carbon content in particulate organic matter was determined by wet digestion method as described in total organic carbon estimation (Snyder and Trofymow 1984).

3.4.5 **Microbial biomass carbon**

The biomass in the soil was estimated by the fumigation extraction method. Moist sample was taken in duplicate (to given approximately 10 g oven-dry weight) in 50 mL glass beaker. Another 10 g of the soil was weighted into an aluminium cane for moisture determination. One beaker with soil was kept inside vacuum desiccator and fumigated with fresh ethanol-free chloroform for 24 hours and the excess chloroform was removed by repeated back suction (Jenkinson and Powelson 1976). The fumigated and non-fumigated soil was extracted with 25 mL of 0.5M K_2SO_4 , carbon content in the extractant was determined by following the

oxidation with $K_2Cr_2O_7$ and finally titration with ferrous ammonium sulphate using ferroin as a indicator.

Microbial biomass carbon (MBC) was computed from the following relationship

$$\text{Microbial biomass carbon} = (\text{OC}_F - \text{OC}_{UF}) / K_{EC}$$

where, OC_F and OC_{UF} are organic carbon extract from the fumigated and unfumigated soil respectively, on oven dry basis and K_{EC} is the efficiency of extraction (0.25) as per Bremner and Kessel (1990).

3.5 Computation of soil organic carbon stocks

SOC stock of each site was measured by the product of SOC concentration , bulk density and soil depth. Carbon stocks related to various SOC fractions were similarly computed.

3.6 Carbon management index

Soil Organic Matter promotes modification in chemical, physical and biological properties of soil, and its presence is of great value to soil quality. The magnitude of OM influences is dependent on its quantity (total C) and quality (labile C) factors. The labile fraction is more easily affected by changes in soil management and, for this reason; it can be used as an indicator of management quality and agriculture sustainability. The Carbon Management Index (CMI) proposed by Blair *et al.* (1995) has the advantage of involving both quantity and quality factors of OM in its estimation. The carbon management index was computed from the following relationships:

$$i) \quad \text{Carbon pool index (CPI)} = \frac{\text{Sample total C (mg/g)}}{\text{Reference total C (mg/g)}}$$

ii) The loss of labile C is of greater importance than the loss of non-labile C.

To account for this, carbon lability index is calculated as follows:

$$a) \quad \text{Lability of Carbon} = \frac{\text{C in fraction oxidized by } KMnO_4}{\text{C remaining un-oxidized by } KMnO_4}$$

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

iii) Finally, the carbon management index (CMI) is calculated as follows;

$$CMI = C \text{ pool index (CPI)} \times \text{Lability index (LI)} \times 100$$

CMI was computed for all the land uses except for wetland, furthermore surface (0 to 15 cm) layer of grassland soil was used as reference for calculation

3.7 Estimation of soil enzyme activities

3.7.1 Dehydrogenase activity

Dehydrogenase activity was estimated by monitoring the rate of production of triphenyl formazan (TPF) from tri-phenyl tetrazolium chloride used as an electron acceptor. The method of Klein *et al.* (1971) was followed for the assay of dehydrogenase activity as outlined below.

One gram of air-dried soil was placed in 15 mL screw capped tube. To this 0.2 mL of 35% tri-phenyl tetrazolium chloride and 0.5 mL of 1% glucose were added and the tubes were incubated at 28 °C for a period of 24 hours. After incubation 10 mL of methanol was added and shaken exactly for one minute. It was allowed to stand in dark for six hours. The colour intensity developed was measured at 485 nm (blue filter). From the standard curve, drawn in the range of 0.004 to 0.4 mg TPF per 10 mL of methanol, the TPF produced in the sample were computed. Dehydrogenase activity was expressed as TPF formed per gram soil for one hour on oven dry weight basis.

3.7.2 Phosphatase activity

To one gram of soil in a 50 mL conical flask, 4 mL of modified universal buffer (pH 6.5 for acid phosphatase and pH 11 for alkaline phosphatase), 1 mL of *p*-nitro phenyl phosphate were added and incubated for one hour at 37 °C. After that 4 mL of 0.5 M NaOH and 1 mL of 0.5M CaCl₂ were added and the contents were filtered.

The amount of *p*-nitro phenol (PNP) present in the filtrate was measured colorimetrically at 410 nm. The phosphatase activity was calculated after deducting the PNP formed in the control (where no *p*-nitro phenyl phosphate was

added), from the PNP produced in the samples. The results were expressed as $\mu\text{g PNP g}^{-1} \text{ h}^{-1}$ (Tabatabai and Bremner 1969).

3.7.3 Fluorescein diacetate hydrolase

Fluorescein diacetate hydrolase was assayed according to Green *et al.* (2006). One gram of soil sample was incubated for 1 h at 37 °C with 9 mL of 50 mM phosphate buffer pH 7.0 and 1 mL of 5 mM fluorescein diacetate (FDA). The pH buffer used was 7.0 instead of 7.6 recommended by Alarcon-Gutierrez *et al.* (2008) to avoid the potential non-enzymatic interferences. The reaction was stopped by adding 2 mL of acetone, and immediately centrifuged. The amount of fluorescein released from FDA was measured in the supernatant at 490 nm.

3.7.4 Arylsulphatase activity

Arysulphatase activity was assayed as per method given by Tabatabai and Bremner (1970). For measuring arylsulphatase activity, 0.1 g of soil (<2 mm) of soil was placed in a 1.5 mL Eppendorf flask and 1 mL of acetate buffer, 0.25 mL of toluene and 0.25 mL of *p*-nitrophenyl sulphate (*p*-NPS) solution were added. The contents were mixed and incubated at 37 °C for 1 h. Then 0.1 mL of 0.5M CaCl_2 and 0.4 mL of 0.5M NaOH were added, swirled for a few seconds, and filtered (0.45 mm HA nitrocellulose, Millipore). Absorbance was measured at 405 nm against the reagent blank and *p*-nitrophenol content was computed by referring to a calibration curve.

3.8 Soil Quality Assessment

As a first step in soil quality assessment (Fig.3.3) critical parameters were identified, through different approaches, to arrange the minimum data set (MDS) . The nature of the land use, land use requirements and the management goals were considered while deriving the MDS. The possible soil functions namely, sustain plant growth, nutrient cycling, resist biochemical degradation, water transport and storage, filtering and buffering etc. that accomplish the recognized goals were identified. The next step is the detection of soil properties/ indicators that can explain the soil functions. The contribution of the individual soil properties to accomplish the desired soil function was distributed by assigning weights to

identified soil property. Indicator weights were derived by using statistical tools like principal component analysis, regression equations etc., as well as expert opinion and relevant literature. Steps followed in the development of SQI (Masto *et al.*, 2008) are shown in Fig 3.

3.8.1 **Linear scoring function**

After identifying the MDS indicators, every observation of each MDS indicators was transformed using a linear scoring method as suggested by Andrew *et al.* (2002). To assign the scores, indicators were arranged in order depending on whether a higher value was considered “good” or “bad” in terms of soil functions. In case of “more is better” indicators, each observation was divided by the highest observed value such that the highest score received a value of 1. For “less is better” indicators, the lowest observed value (in the numerator) was divided by each observation (in the denominator) such that lowest observed value received a score of 1. In this study, almost all of the indicators in the MDS were considered good from the view point of soil quality when they are in increasing order, and hence the “more is better” approach was followed, except in case of bulk density where ‘lesser is better’ approach was considered (Harris *et al.*, 1996).

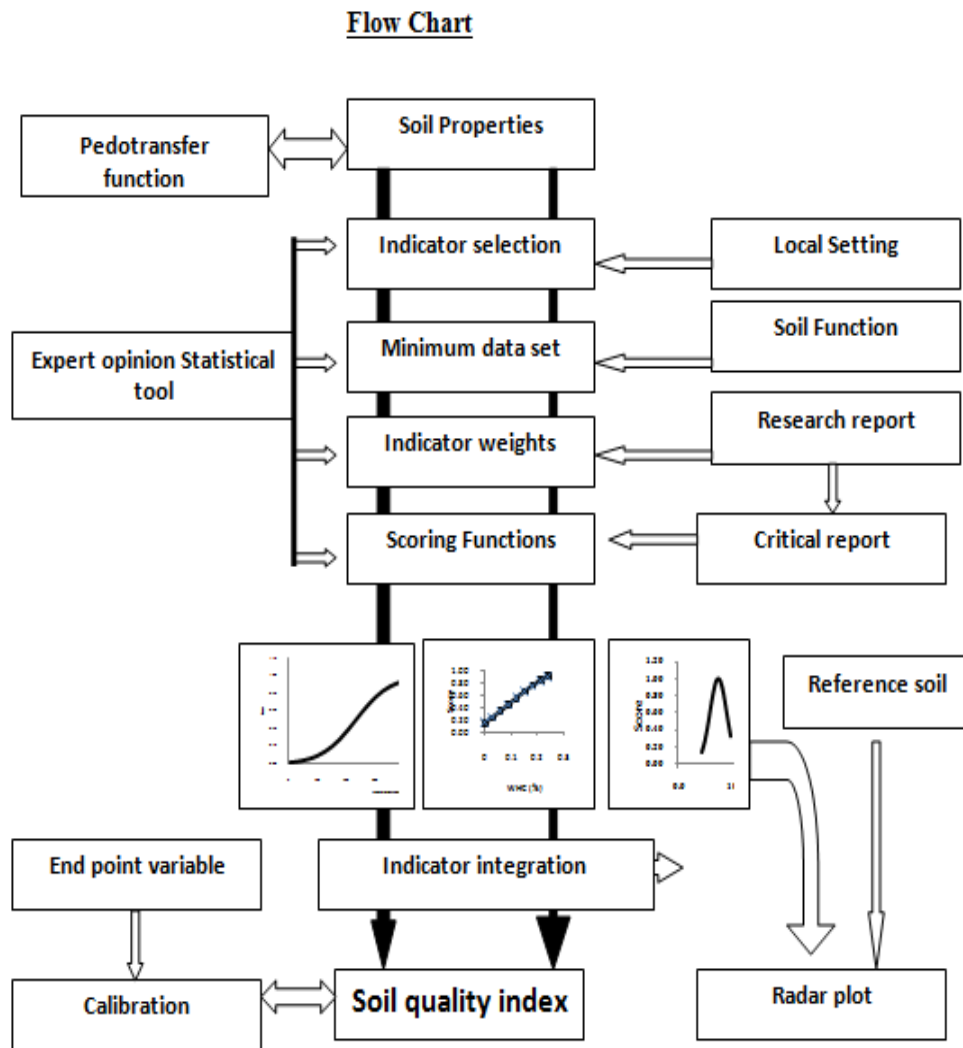


Fig 3.3: Outline of soil quality indexing procedure.

3.8.2 Non-Linear scoring function

The values of non-linear scores were computed by the equation (Masto *et al.*, 2008) as follows;

$$NLSF = 1/[1 + e^{-b(x-A)}]$$

Where, x is the soil property value, A the baseline or value of the soil property where the score equals 0.5 and b is the slope. Using the equation, standardized

scores for 'More is better' and 'Less is better' approach were determined for the indicators.

3.8.3 Indicator integration into indices

The following four soil quality indexing methods were developed.

1. Un-screened additive index
2. Principal component analysis based soil quality index
3. Regression equation based soil quality index
4. Conceptual framework for analyzing soil quality

3.8.3.1 Unscreened additive index

The additive index is the summation of scores of all the indicators studied, and the sum was divided by the total number of indicators used (Velmurgan, 2000), which is according to the following formula:

$$\text{Soil quality index (SQI)} = \frac{\sum_{i=1}^n S_i}{n}$$

Where, S denotes linear or non linear score of observed soil quality indicator and n is the number of indicators included in the index. Potential SQ indicators were selected based on their sensitivity to management practices, ability to describe major soil processes, ease and cost of sampling and laboratory analysis, and significance of increasing productivity and protecting environmental soil functions. Using Expert Opinion (EO) approach MDS variables were chosen from the potential indicators based on researcher knowledge and literature recommendations (Tesfahunegn, 2014; Doran and Parkin, 1994). POC was dropped from MDS and instead LOC was retained, because the former is obtained through physical fractionation method which does not always produce consistent results (Sollins *et al.*, 1999). Similarly WBC was dropped as it is essentially the same as TOC, differing only in the method of analysis. The variables pH and EC were also excluded from the data set because these two properties do not limit the

soil function as they are always found in normal range (Nisar and Lone 2013; Ramzan *et al.*,2014) in soils of the study area.

3.8.3.2 Principal component analysis based soil quality index

Principal component analysis (PCA) is the method for reducing correlated measurement variables to a smaller set of statistically independent linear combinations having certain unique properties with regard to characterizing individual differences. Principal component (PCs) for a data set are defined as a linear combinations of variables that account for maximum variance within the set by describing vectors of closets fit to n observations in p -dimensional space, subject to being orthogonal to one another (Dunteman, 1989). Standardized PCA of all (untransformed) data was performed using SPSS v 21.0. PCs that receive high eigen values were assumed to best represent the variation in the system. Therefore only the PCs with eigen value ≥ 1 (Kaiser, 1960) were examined. Additionally, PCs that explain $\geq 5\%$ of the variability in the soil data (Wander and Bollero, 1999) were included, when fewer than three PCs had eigenvalues ≥ 1 . Under a particular PC, each variable is given a factor loading that represents contribution of that variable to the composition of the PC. Only the variables with high factor loading were retained from each PC for soil quality indexing. High factor loading were defined as having absolute values within were defined as having within 10% of the highest factor loading (Wander and Bollero, 1999; Andrews *et al.* 2002). When more than one variable was retained under a single PC, multivariate correlations were employed to determine if the variables could be considered redundant and, therefore, eliminated from the SQI (Andrew *et al.* 2002). If the highly loaded factors were not correlated, then each was considered important, and thus, retained in the SQI. Among well correlated variables, the one with the highest factor loading (absolute values) was chosen for the SQI. Each PC explained a certain amount of variation (%) in the total data set; this percentage provided the weight for the variables chosen under a given PC. The final PCA based sol quality equation is as follows.

$$\text{Soil quality index (SQI)} = \sum_{i=1}^n W_i \times S_i$$

where, W is the PCA weighting factor and S is the indicator score. The equation was normalized to get a maximum SQI of one.

3.8.3.3 Regression equation based soil quality index

The results from the stepwise multiple regression equation between the soil quality indicators and total soil organic carbon was used to develop the soil quality index as per the following equation.

$$\text{Soil quality index (SQI)} = \sum_{i=1}^n S_i \times (\text{Beta})$$

Where, S denotes the linear or non linear scores obtained by the soil quality indicators, n is the number of indicators retained in the stepwise multiple regression equation and “Beta” is the standardized regression coefficients of the retained soil quality parameters. The equation was finally normalized to get a maximum SQI of one.

3.8.3.4 Conceptual framework for assessing soil quality

The approach used here to determine soil quality was primarily from Karlen and Stott (1994). In this study the critical soil functions were modified according to their importance in fulfilling the overall goals of maintaining soil quality under specific soil conditions and land-use purposes. Numerical weights (wt) to each soil function was assigned after taking into account specific research aims, local factors and conditions of cropping and land use, as well as the environmental concerns (Glover *et al.*, 2000; Wymore, 1993). The modified model is given under.

$$\text{Soil quality index (SQI)} = q_{\text{spg}} (\text{wt}) + q_{\text{cn}} (\text{wt}) + q_{\text{rbd}} (\text{wt}) + q_{\text{fb}} (\text{wt}) + q_{\text{wa}} (\text{wt})$$

where, q_{spg} is score (linear/non-linear) the for the soil’s ability to sustain plant growth, q_{cn} is the score (linear/non-linear) for soils’ ability to cycle nutrients, q_{rbd} is the score (linear/non-linear) for soil’ ability to resist biochemical degradation, q_{fb} is the score (linear/non-linear) for soils’ filtering and buffering ability and q_{wa}

is the score (linear/non-linear) for soil's ability to facilitate water absorption, wt is the numerical weight for each soil function. Numerical weights for each soil function were given according to the soil function's importance in fulfilling the overall goals of maintaining soil quality. Weights for all soil functions sum to 1.00. An ideal soil would fulfill all the functions considered important and, under the proposed framework will receive SQI of 1.00. As a soil fails to meet the ideal criteria, the SQI would fall, with zero being the lowest rating.

3.9 Evaluation of indexing procedures

In this study, soil quality indices were developed by the combination of two scoring functions and four indicator integration procedures. So, altogether, eight soil quality indices were tried. Indexing procedures were evaluated by the sensitivity of different procedures. The sensitivity was calculated using the following formula

$$\text{Sensitivity (Si)} = \text{SQI}_{(\text{max})} / \text{SQI}_{\text{min}}$$

Where, Si is the sensitivity and $\text{SQI}_{(\text{max})}$ are the maximum and $\text{SQI}_{(\text{min})}$ minimum SQI values.

3.10 Statistical procedures

Statistical analysis was carried out by using standard statistical procedures followed by Gomez and Gomez (1984). Data standardization was done before data reduction. Principal component analysis was used for computation of soil quality indices. The PCA, descriptive statistics and linear regressions (step wise; enter), were performed in SPSS version 21.0 (SPSS , 2014) . Non-linear scoring were analyzed in MS Excel TM spreadsheet using SOLVER Tab for cyclic iteration of 100, precision 0.000001, tolerance 5% and convergence 0.0001.

Chapter - 4

EXPERIMENTAL FINDINGS

The results obtained in the present study “Impact of Different Land Uses on Soil Quality and Organic Carbon Stocks in Terrestrial Ecosystems of North Eastern Zone of Kashmir” are grouped under the following headings.

- 4.1 Physico-chemical properties of soils
- 4.2 Soil organic carbon and its fractions.
- 4.3 Carbon management index
- 4.4 Soil organic carbon stocks
- 4.5 Soil enzyme activities under different land uses.
- 4.6 Soil quality assessment.
- 4.7 Sensitivity of indexing procedures.
- 4.8 Identification of limiting parameters.

4.1 Physico-chemical properties of soils.

4.1.1 Bulk density

The data on bulk density of soils are presented in Table 4.1. The BD of surface (0-15cm) layer of soils ranged from 0.73 to 1.38 Mg m⁻³. Highest mean value of BD (1.47 Mg m⁻³) was recorded in sub-surface (15-30cm) layer of apple orchards, whereas the lowest value (0.73Mg m⁻³) was recorded in surface layers of wetlands. The average bulk density of apple orchards as well as arable lands was generally higher than soils under native vegetation. Moreover, the bulk density of wetland soils was found to be markedly lower than that of average BD of other land uses. The confidence interval (CI) values obtained for different soils are given in Appendix (Table A.1).The upper and lower CI values were also highest for apple orchard soils and lowest in wetland soils in both the layers.

4.1.2 Porosity

Mean values of porosity (fraction, m^3/m^3) is given under table 4.1 As indicated by the data, porosity of the surface soil layers under different land uses, varied from 0.48 to 0.73 with the highest value related to wetland soil and lowest in apple orchards.

Table 4.1: Physical properties of different soils in the north eastern zone of Kashmir.

Land use type/ Location	Bulk density ($Mg\ m^{-3}$)		Porosity		Void Ratio	
	Surface	Sub-surface	Surface	Sub-surface	Surface	Sub-surface
Paddy						
Baghurampora	1.24	1.30	0.53	0.51	1.14	1.04
Gamroo	1.23	1.27	0.53	0.52	1.15	1.09
Sumlar	1.24	1.25	0.53	0.53	1.13	1.11
Margund	1.08	1.17	0.59	0.56	1.45	1.26
Yarmuqam	1.16	1.26	0.56	0.52	1.27	1.10
Mean	1.19	1.25	0.55	0.53	1.23	1.12
SE +/-	0.035	0.021	0.045	0.031	0.035	0.041
Maize						
Margund	1.30	1.38	0.51	0.48	1.03	0.92
Athwot	1.13	1.20	0.57	0.55	1.34	1.21
Yarmuqam	1.25	1.34	0.53	0.49	1.12	0.97
Gagangir	1.34	1.38	0.49	0.48	0.98	0.92
Kudara	1.22	1.25	0.54	0.53	1.17	1.11
Mean	1.26	1.30	0.53	0.51	1.13	1.03
SE +/-	0.043	0.028	0.044	0.027	0.046	0.032
Apple						
Nadihal	1.32	1.46	0.50	0.45	1.00	0.82
Badampura	1.42	1.47	0.46	0.44	0.87	0.80
Khalmula	1.35	1.48	0.49	0.44	0.96	0.79
Watlar	1.43	1.46	0.46	0.45	0.85	0.81
Arin	1.36	1.48	0.49	0.44	0.94	0.79
Mean	1.38	1.47	0.48	0.44	0.92	0.80

SE +/-	0.026	0.012	0.036	0.015	0.037	0.045
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Contd-Table 4.1

Vegetable						
Aloosa	0.98	1.10	0.63	0.58	1.69	1.41
Ahan	1.09	1.20	0.59	0.54	1.42	1.20
Dab shalahar	0.91	0.99	0.66	0.63	1.90	1.67
Barsoo	0.97	1.06	0.63	0.60	1.72	1.50
Quilmuqam	0.93	1.06	0.65	0.60	1.84	1.50
Mean	0.98	1.08	0.63	0.59	1.71	1.45
SE +/-	0.036	0.031	0.046	0.068	0.056	0.062
Forest						
Preng	0.90	0.99	0.66	0.63	1.94	1.67
Athwot	0.91	0.97	0.65	0.63	1.89	1.72
Kudara	0.81	0.88	0.69	0.67	2.25	1.99
Naranag	0.90	0.98	0.66	0.63	1.92	1.69
Sonamarg	0.94	1.03	0.65	0.61	1.82	1.56
Mean	0.90	0.98	0.66	0.63	1.97	1.73
SE +/-	0.049	0.046	0.059	0.038	0.049	0.029
	Bulk density		Porosity		Void Ratio	
Grassland						
Potushai	1.16	1.24	0.56	0.53	1.28	1.13
Sonamarg	1.06	1.17	0.60	0.56	1.49	1.26
Nagmarg	1.14	1.27	0.57	0.52	1.32	1.08
Trundkhul	1.20	1.26	0.54	0.52	1.20	1.10
Tragbal	1.20	1.26	0.54	0.52	1.20	1.10
Mean	1.16	1.24	0.56	0.53	1.30	1.13
SE +/-	0.029	0.025	0.029	0.046	0.039	0.048
Wetland						
Potushai	0.74	0.85	0.72	0.68	2.58	2.12
Tulmulla	0.60	0.73	0.77	0.72	3.37	2.62
Kehnoosa	0.77	0.80	0.71	0.70	2.44	2.31
Saderkoot	0.76	0.81	0.71	0.69	2.46	2.26
Shalabugh	0.75	0.79	0.72	0.70	2.52	2.34
Mean	0.73	0.80	0.73	0.70	2.67	2.33
SE +/-	0.039	0.018	0.049	0.036	0.051	0.039

SE-standard error of mean, Surface- 0-15cm layer, Sub-surface- 15-30cm layer

In general the porosity in the sub-surface layer in all the soils was lesser than the surface layer and varied in a manner opposite to soil BD. The 95% confidence interval values for porosity obtained under different land uses are given in Appendix (Table A.2). CI values were highest (0.69 and 0.76) under wetland soils and lowest (0.44 and 0.51) under apple orchards at 0-15cm depth. Similar trend in CI values was also observed in sub-surface soil layers.

4.1.3 Void ratio

Void ratios of soils are given in Table 4.1. The mean values ranged from 0.92 to 2.67 in the surface layers and from 0.80 to 2.33 in the sub-surface layers of soils. Highest mean values of void ratio, in both the layers were observed in wetland soils followed by forest soils. Apple orchard soils exhibited the lowest values among the soils. The void ratio in the wetland soils was nearly three times greater than the soils of apple orchards. Data on confidence intervals (lower and upper) are presented under Table A.3 in appendix. In surface soil 95% CI values were found to be highest (2.62 and 2.70) under wetlands and lowest (0.91 and 0.93) under apple orchards.

4.1.4 Clay content

The data on clay content of different soils are presented in Table 4.2. In general higher clay content was observed in wetland soils as compared to all other soils. The mean values of clay in surface and sub-surface layer of wetland soils were 42.75 and 43.73 % respectively. Soils under vegetable cultivation exhibited the lowest mean values of clay content that is 24.32 and 25.46 at surface and sub-surface layer, respectively. Only a slight variation in clay content was observed between 0-15cm and 15-30 cm depths at every location in the study area. The values of 95 % C.I are presented in Table A.4 & A.5 under appendix. CI (lower and upper) were estimated to be 42.27 to 43.23% and 23.77 to 30.55% in surface layers of wetland and vegetable soils, respectively. The 95% C.I varied in a manner, similar to mean clay content in soils.

4.1.5 Silt content

The mean values of silt content in the surface and sub-surface layers of soils are presented in Table 4.2. The data reveals that the silt content of the soils ranged from 32.77 to 46.79 % in surface and 32.74 to 47.12% in sub-surface layers. Highest mean silt content at 0-15cm depth was observed under forest land use and least under vegetable land use. Similarly, at 15-30 cm depth, highest % silt content was observed under maize and lowest under vegetable land use. The data on 95% C.I values of soil silt content are presented in Table A.4 & A.5 under appendix.

4.1.6 Sand content

The mean sand content in the soils under different land uses varied from 16.21 to 42.87% in surface soil layers and from 17.25 to 41.79% in the sub-surface layers (Table 4.2). However, at both the soil depths highest sand content was observed in soils under vegetables and lowest in wetland soils. The data on confidence interval presented in Table A.4 & A.5 (appendix) reveal that, in the surface layer the highest values for confidence interval (38.17 to 52.45%) were estimated for vegetable soils and lowest(16.04 to 16.37%) were recorded under wetland soils. The confidence interval values varied in a similar fashion in sub-surface soil layers.

Table 4.2: Percent sand, silt and clay in different soils of the north eastern zone of Kashmir.

Land use type /Location	-----Surface-----				-----Sub-surface-----			
	Sand	Silt	Clay	Textural Class	Sand	Silt	Clay	Textural Class
Paddy								
Baghurampora	19.50	47.60	32.90	<i>SiCL</i>	20.60	47.13	32.26	<i>SiCL</i>
Gamroo	31.50	39.16	29.33	<i>CL</i>	31.33	38.26	30.40	<i>CL</i>
Sumlar	31.10	40.13	28.76	<i>CL</i>	32.40	36.16	31.43	<i>CL</i>
Margund	22.73	45.00	32.26	<i>CL</i>	23.56	44.20	32.23	<i>CL</i>
Yarmuqam	43.20	22.03	34.76	<i>CL</i>	42.23	22.50	35.26	<i>CL</i>
Mean	29.61	38.78	31.60		30.02	37.65	32.32	
SE +/-	1.12	4.40	1.12		3.78	0.81	0.81	
Maize								
Margund	25.13	44.76	30.10	<i>CL</i>	20.30	47.73	31.96	<i>CL</i>
Athwot	20.23	50.33	29.43	<i>CL</i>	17.80	48.57	33.64	<i>CL</i>
Yarmuqam	20.00	48.02	32.00	<i>CL</i>	19.13	47.93	32.93	<i>CL</i>
Gagangir	26.33	43.90	29.76	<i>CL</i>	24.93	44.00	31.06	<i>CL</i>
Kudara	24.10	46.60	29.30	<i>CL</i>	24.36	47.36	28.27	<i>CL</i>
Mean	23.16	46.73	30.17		21.30	47.12	31.57	
SE +/-	1.29	1.15	0.54		1.42	0.80	0.93	
Apple								
Nadihal	43.33	22.66	34.00	<i>CL</i>	41.43	23.13	35.43	<i>CL</i>
Badampura	13.00	52.23	34.76	<i>SiCL</i>	12.50	51.83	35.66	<i>SiCL</i>
Khalmula	21.10	48.96	29.93	<i>CL</i>	21.33	48.27	30.40	<i>CL</i>
Watlar	16.33	50.33	33.33	<i>SiCL</i>	16.00	49.43	34.56	<i>SiCL</i>
Arin	41.70	23.00	35.30	<i>CL</i>	41.50	22.30	36.20	<i>CL</i>
Mean	27.09	39.44	33.46		26.55	38.99	34.45	
SE +/-	6.40	6.79	0.94		6.24	6.17	1.64	

Contd-Table 4.2	Sand	Silt	Clay	Textural Class	Sand	Silt	Clay	Textural Class
Vegetable								
Aloosa	34.53	34.23	31.23	<i>CL</i>	34.50	34.00	31.50	<i>CL</i>
Ahan	48.33	30.63	21.03	<i>L</i>	47.59	31.20	21.21	<i>L</i>
Dab shalahar	31.77	40.11	28.16	<i>CL</i>	32.22	36.35	31.42	<i>CL</i>
Barsoo	49.83	29.81	20.36	<i>L</i>	47.23	31.22	21.55	<i>L</i>
Quilmuqam	49.96	29.10	20.93	<i>L</i>	47.43	30.93	21.64	<i>L</i>
Mean	42.87	32.77	24.32		41.79	32.74	25.46	
SE +/-	3.11	2.06	1.11		0.85	1.99	2.79	
Forest								
Preng	14.23	56.86	28.90	<i>SiCL</i>	13.70	57.00	29.30	<i>SiCL</i>
Athwot	19.33	51.50	29.16	<i>SiCL</i>	19.33	50.50	30.16	<i>SiCL</i>
Kudara	23.43	47.53	29.03	<i>CL</i>	23.43	47.6	28.96	<i>CL</i>
Naranag	10.93	56.70	32.36	<i>SiCL</i>	10.66	57.30	32.03	<i>SiCL</i>
Sonamarg	40.86	21.36	37.76	<i>CL</i>	40.50	21.00	38.50	<i>CL</i>
Mean	21.76	46.79	31.44		21.52	46.68	31.79	
SE +/-	5.23	6.09	1.70		5.23	6.68	1.75	
Grassland								
Potushai	23.06	49.76	27.16	<i>SiL</i>	20.66	49.76	29.56	<i>SiL</i>
Sonamarg	40.20	21.83	37.96	<i>CL</i>	39.73	21.70	38.56	<i>CL</i>
Nagmarg	21.20	47.03	31.76	<i>CL</i>	22.33	46.06	31.60	<i>CL</i>
Trundkhul	19.06	48.00	32.93	<i>SCL</i>	18.20	47.73	33.73	<i>SiCL</i>
Tragbal	18.66	48.83	32.50	<i>SiCL</i>	17.66	48.16	34.16	<i>SiCL</i>
SE +/-	4.01	5.33	1.71		4.09	5.27	1.50	
Mean	24.43	43.09	32.46	<i>SiCL</i>	23.76	42.68	33.54	
Wetland								
Potushai	19.03	39.13	41.83	<i>C</i>	18.16	38.53	43.32	<i>C</i>
Tulmulla	13.50	45.16	41.33	<i>SiC</i>	15.53	40.96	43.51	<i>SiC</i>
Kehnoosa	14.70	41.23	44.06	<i>SiC</i>	18.03	39.23	42.73	<i>C</i>
Saderkoot	15.50	43.60	40.90	<i>SiC</i>	17.30	40.60	42.11	<i>SiC</i>
Shalabugh	18.33	36.03	45.63	<i>C</i>	17.23	35.70	47.06	<i>C</i>
Mean	16.21	41.03	42.75		17.25	39.00	43.73	
SE +/-	1.06	1.61	0.90		0.46	0.93	0.86	

CL-clay loam, SiCL-silty clay loam, L-loam, SiL-silt loam, SiC-silty clay, C-clay

4.1.7 Soil pH

The pH of soils at most of the sites was found to be in neutral range. The mean values of soil pH are presented in Table 4.3 and 4.4. In surface soil layers the lowest pH (6.42) was recorded in wetland soils and highest pH (7.30) was related to vegetable soils (Table 4.3). In sub-surface soil layers mean pH varied from 6.74 to 7.26 (Table 4.4). The mean pH of wetland soils was generally lower than other soils at both the depths in the study area. The confidence interval values related to different soils are given in Table A.6 under appendix. Lowest values for confidence interval (lower and upper) were found to be 6.31 to 6.54 in surface soil layer and 6.66 to 6.84 in sub-surface layer of wetland in the study area. Whereas, the highest values for confidence interval (lower and upper) were found to be 7.2 to 7.41 in surface layer and 7.10 to 7.42 in sub-surface layer of vegetable and grassland soils, respectively.

4.1.8 Electrical conductivity

The mean EC of the soils in this study varied from 0.11 to 0.60 dSm^{-1} in surface soils and from 0.09 to 0.65 dSm^{-1} in sub-surface soils (Table 4.3 and 4.4). Highest and lowest mean EC in surface and sub-surface soil layer was observed under wetland and grassland soil, respectively. The mean EC of forest and wetland soils increased with depth, whereas, in other land uses, it slightly declined with depth. The data on 95% C.I presented in Table A.7 under appendix, reveal that highest (lower and upper) value (0.48 to 0.70 dSm^{-1}) were observed under wetland and lowest (0.09 to 0.16) were recorded for surface layer of grassland soil. Similarly, for sub-surface layers the greatest (0.55 to 0.74) and lowest values (0.018 to 0.095) for CI (lower and upper) were estimated in wetland and forest soils, respectively.

4.1.9 Available N

Generally, the average available N declined with soil depth in all land uses. Mean available N content in surface layer (Table 4.3) was relatively much higher in wetland soil (202.57 mgKg^{-1}) followed by forest soil (178.70 mg Kg^{-1}) and decreased in the order of vegetable soil (145.32 mgKg^{-1}), grassland

soil(143.93mgKg⁻¹), apple (140.20 mgKg⁻¹), maize (103.49 mgKg⁻¹) and paddy soils (98.12 mgKg⁻¹) respectively. In 15-30cm soil layers, mean available N (Table 4.4) decreased in almost similar order, except that the value recorded under grassland soil (124.49 mg Kg⁻¹) was higher than that of vegetable soil (122.68 mgKg⁻¹). The data on 95% C.I are given in Table A.8 under appendix. In surface soil layer highest values for C.I (lower and upper) 197.22 and 208.54 mg Kg⁻¹ were recorded in wetland and lowest values 94.22 and 105.33 mg Kg⁻¹ were estimated in paddy soils. Wetland and paddy soils also exhibited the highest and least values for CIs in the sub-surface layers.

4.1.10 Available P

The average available P content of the soils varied from 11 to 19 mg Kg⁻¹ in the surface soil layers.(Table 4.3). In sub-surface layer it ranged from 8.01 to 13.26 mg Kg⁻¹ (Table 4.4). Decline in available P content with depth was observed under all the land use types. In the 0-15cm layer the least quantity of available P was observed in paddy soil and the maximum was recorded in wetland soil. In the 15-30 cm layer highest mean quantity of available P was observed in forest soil and least in vegetable soil. Among the surface soils of agricultural and horticultural ecosystems the values of mean available P were found to be highest in soils of apple orchards (12.89 mgKg⁻¹) followed by maize fields (11.83 mgKg⁻¹) and further decreased in the order of vegetable soil (11.72 mgKg⁻¹) and paddy soil (11.01 mgKg⁻¹), respectively. The data on 95% C.I are presented in Table A.9 under appendix.

4.1.11 Available K

Among the surface soils in agricultural and horticultural ecosystems the average value of available K (Table 4.3) was found to be highest under apple orchards (178 mgKg⁻¹) and least in paddy fields (149 mgKg⁻¹) and similar trend was recorded at lower depth (Table 4.4). Among all the land use types/ ecosystems, however, highest mean available K in both surface and sub-surface layers were recorded in wetland soils followed by that of forest soils. The data on 95% C.I are provided in Table A.10 under appendix. In surface soil layer highest

values for C.I (lower and upper) that is 255.67 and 270.34 mgKg⁻¹ were recorded in wetland and lowest values 138.56 and 154.56 mgKg⁻¹ were estimated in paddy soils.

4.1.12 Available S

The average available S content varied from 10.18 to 14.74 mgKg⁻¹ (Table 4.3) in surface soil layers, whereas in sub-surface layers it ranged from 8.48 to 14.72 mg Kg⁻¹ (Table 4.4). Highest available S content in both surface and sub-surface soil was observed in wetland soils and lowest in vegetable soils. The data on 95% C.I are presented in Table A.11 under appendix. The CI (lower and upper) related to surface layer of wetland soils were found to be 13.67 and 16.45 mgKg⁻¹ and for vegetable soils the lower and upper bounds were found to be 6.76 and 10.88 mgKg⁻¹, respectively.

4.1.13 Exchangeable Ca

Average exchangeable Ca (cmol(+)Kg⁻¹) in the surface soil layers (Table 4.3) was found to be highest (12.40) under wetland use followed by grassland (11.55) and decreased in the order of maize (10.85), forests (10.71), paddy fields (8.91), apple orchards (8.10) and vegetable fields (7.9), respectively. However, the highest and lowest values in sub-surface soils (Table 4.4) were recorded under wetland (12.29) and forest (6.94) land use, respectively. The data on 95% C.I are presented in Table A.12 under appendix. Highest values for C.I (lower and upper) in surface layer were recorded in wetland soil (10.54 and 14.05) and lowest values (5.26 and 9.77) were estimated in vegetable soils. In sub-surface layers the estimated values were 9.30 and 14.79 in wetland soil and 6.01 and 7.28 in vegetable soil.

4.1.14 Exchangeable Mg

The mean values of exchangeable Mg (cmol(+)Kg⁻¹) in the soils under different land uses varied from 1.57 to 3.03 in surface soils and from 2.02 to 3.19 in sub-surface soils (Table 4.3 and 4.4). In surface soil layer the highest exchangeable Mg was observed under wetland and lowest under vegetable land use, while the highest and lowest mean values in sub-surface layer were observed

in wetland and paddy soil, respectively. In the surface soil layer, highest values for 95% C.I (Table A.13 under appendix) were observed under wetland (2.43 and 3.25) and lowest(1.18 to

Table 4.3: Chemical properties of the surface (0-15cm) layer of different soils in the study area

Land use type/ Location	pH	EC dSm ⁻¹	Av. N -----mgKg ⁻¹ -----	Av. P	Av. K	Av. S	Ex. Ca -----Cmol(+)Kg ⁻¹ -----	Ex. Mg
Paddy								
Baghurampora	7.55	0.15	78.01	8.54	130.07	8.48	6.72	0.96
Gamroo	6.70	0.28	105.76	11.70	158.02	9.60	7.99	1.58
Sumlar	6.33	0.15	112.49	12.18	177.82	11.52	11.05	1.72
Margund	6.27	0.16	88.56	10.52	135.57	10.19	10.61	2.11
Yarmuqam	6.31	0.20	105.80	12.14	148.05	11.13	8.18	1.65
Mean	6.63	0.19	98.12	11.01	149.91	10.18	8.91	1.60
SE +/-	0.04	0.01	3.24	0.17	1.61	0.11	0.03	0.02
Maize								
Margund	7.60	0.08	82.29	10.15	143.12	10.77	10.79	1.90
Athwot	7.02	0.26	144.22	12.58	140.99	12.87	11.52	2.33
Yarmuqam	6.46	0.07	64.68	10.01	139.25	10.44	8.35	1.69
Gagangir	7.37	0.10	105.79	11.75	152.98	11.66	11.01	2.00
Kudara	6.75	0.19	120.45	14.68	142.18	12.05	12.57	2.13
Mean	7.04	0.14	103.49	11.83	143.70	11.56	10.85	2.01
SE +/-	0.08	0.01	4.56	0.18	1.12	0.16	0.02	0.02
Apple								
Nadihal	7.18	0.24	165.52	13.48	229.01	13.37	9.47	2.88
Badampura	7.05	0.12	104.66	11.44	142.43	14.87	8.23	1.86
Khalmula	6.56	0.33	145.43	12.65	202.53	11.98	5.12	1.97
Watlar	6.76	0.12	101.91	12.35	146.75	14.13	6.50	1.94
Arin	7.44	0.26	183.50	14.54	173.50	11.95	11.21	3.15
Mean	6.99	0.21	140.20	12.89	178.84	13.26	8.10	2.36
SE +/-	0.03	0.01	4.05	0.15	1.81	0.17	0.04	0.03
Vegetable								
Aloosa	7.00	0.26	129.25	9.61	201.83	10.47	8.86	1.41
Ahan	7.64	0.21	144.76	12.84	155.12	8.62	7.80	1.37

Contd. Table 4.3

	pH	EC	Av. N	Av. P	Av. K	Av. S	Ex. Ca	Ex. Mg
Dab shalahar	7.86	0.09	165.92	11.52	139.63	8.72	7.93	1.24
Barsoo	7.74	0.09	127.76	10.61	165.8	8.89	4.85	1.37
Quilmuqam	6.28	0.19	158.92	14.02	203.31	11.81	9.95	2.44
Mean	7.30	0.17	145.32	11.72	173.13	9.70	7.90	1.57
SE +/-	0.04	0.01	3.71	0.18	1.88	0.18	0.06	0.03
Forest								
Preng	7.12	0.22	127.15	13.80	200.09	12.05	9.42	3.05
Athwot	6.41	0.08	189.73	17.33	234.25	12.75	10.87	2.88
Kudara	6.66	0.06	210.24	19.14	240.07	13.76	14.82	2.67
Naranag	6.88	0.16	167.25	17.20	228.86	14.19	10.33	2.65
Sonamarg	6.46	0.07	199.13	18.94	237.57	14.59	8.10	3.21
Mean	6.70	0.12	178.70	17.28	228.17	13.47	10.71	2.89
SE +/-	0.06	0.02	3.63	0.21	2.19	0.16	0.01	0.02
Grassland								
Potushai	7.27	0.11	98.36	12.36	185.23	12.49	10.87	2.89
Sonamarg	6.75	0.14	186.73	16.58	252.66	14.31	12.32	1.44
Nagmarg	7.04	0.14	141.31	15.42	189.02	11.76	12.26	2.96
Trundkhul	7.22	0.11	164.91	14.55	255.54	13.29	10.69	2.86
Tragbal	7.48	0.08	128.34	11.9	184.80	12.34	11.62	2.31
Mean	7.15	0.11	143.93	14.16	213.45	12.84	11.55	2.49
SE +/-	0.05	0.02	2.05	0.20	2.23	0.13	0.03	0.02
Wetland								
Potushai	6.61	0.47	235.35	18.87	218.85	14.98	10.40	2.98
Tulmulla	6.10	0.76	284.11	17.91	372.75	16.63	13.91	2.83
Kehnoosa	6.77	0.54	184.17	17.85	220.38	12.46	9.71	3.18
Saderkoot	6.36	0.68	164.99	19.46	223.68	14.08	12.20	3.05
Shalabugh	6.30	0.52	144.24	21.52	273.36	15.57	15.80	3.10
Mean	6.42	0.60	202.57	19.12	261.81	14.74	12.40	3.03
SE +/-	0.050	0.04	2.65	0.56	1.69	0.18	0.04	0.03

Table 4.4: Chemical properties of the sub-surface(15-30cm) layer of different soils in the study area.

Land use type / Location	pH	EC dSm ⁻¹	Av. N -----mgKg ⁻¹ -----	Av. P	Av. K	Av. S	Ex. Ca -----Cmol(+)Kg ⁻¹ -----	Ex. Mg
Paddy								
Baghurampora	7.74	0.06	71.81	8.39	131.32	7.24	9.07	2.01
Gamroo	6.84	0.09	73.21	7.87	160.62	7.36	8.19	1.85
Margund	6.68	0.16	64.62	6.91	137.77	8.41	9.89	2.34
Sumlar	6.69	0.07	100.37	8.84	179.32	11.75	13.21	2.10
Yarmuqam	6.60	0.21	92.52	8.10	150.60	9.15	8.39	1.83
Mean	6.91	0.12	80.50	8.02	151.93	8.78	9.75	2.02
SE +/-	0.06	0.01	2.15	0.08	1.01	0.10	0.02	0.01
Maize								
Margund	7.54	0.08	79.61	6.85	145.90	8.18	10.40	2.20
Athwot	7.03	0.22	103.94	9.01	144.09	9.92	12.17	2.84
Yarmuqam	6.72	0.07	57.71	6.79	142.25	7.91	8.99	2.33
Gagangir	7.80	0.10	91.77	7.94	156.18	9.13	10.43	2.90
Kudara	6.88	0.17	108.08	9.95	144.96	9.42	11.82	2.71
Mean	7.19	0.13	88.22	8.10	146.68	8.91	10.76	2.59
SE +/-	0.08	0.01	1.24	0.07	1.02	0.09	0.03	0.01
Apple								
Nadihal	7.42	0.13	147.56	8.61	233.01	10.34	9.19	2.85
Badampura	7.11	0.15	89.54	7.64	144.43	12.05	7.74	2.21
Khalmula	7.06	0.24	117.23	8.39	205.53	9.84	4.97	2.18
Watlar	7.01	0.15	88.67	7.95	148.50	10.82	6.25	2.57
Arin	7.67	0.17	165.17	9.52	175.00	8.78	11.10	3.17
Mean	7.25	0.17	121.64	8.42	181.29	10.36	7.85	2.60
SE +/-	0.04	0.01	1.52	0.05	1.07	0.08	0.04	0.02
Vegetable								
Aloosa	7.15	0.34	102.26	6.67	165.33	7.66	8.07	2.61
Ahan	7.51	0.15	125.96	8.74	128.42	5.73	5.59	1.86
Dab shalahar	7.34	0.12	141.44	7.85	137.83	6.26	5.89	1.65
Barsoo	7.31	0.11	107.42	7.44	120.80	6.10	6.28	2.62
Quilmuqam	6.66	0.10	136.35	9.37	201.81	8.04	10.93	3.14
Mean	7.12	0.16	122.68	8.01	150.84	6.76	7.35	2.38
SE +/-	0.06	0.01	1.36	0.07	1.00	0.09	0.03	0.02

Contd.Table 4.4	pH	EC	Av. N	Av. P	Av. K	Av. S	Ex. Ca	Ex. Mg
Forest								
Preng	7.27	0.26	123.56	10.32	205.09	9.19	6.11	3.09
Athwot	6.60	0.10	180.18	13.09	238.45	10.11	6.78	2.92
Kudara	6.82	0.07	204.51	14.35	244.07	11.14	10.88	3.20
Naranag	7.42	0.19	165.82	13.96	233.66	11.31	6.13	2.01
Sonamarg	6.59	0.08	193.69	14.58	241.97	11.97	4.80	3.45
Mean	6.94	0.14	173.55	13.26	232.65	10.74	6.94	2.93
SE +/-	0.05	0.02	2.23	0.13	1.19	0.12	0.01	0.01
Grassland								
Potushai	7.28	0.10	70.09	9.81	193.23	9.86	10.29	2.01
Sonamarg	7.05	0.09	172.44	11.25	258.66	11.37	11.76	3.44
Nagmarg	7.17	0.07	119.15	10.84	195.02	9.33	11.94	2.91
Trundkhul	7.29	0.08	147.07	10.1	262.54	10.51	10.37	3.08
Tragbal	7.52	0.09	113.71	8.65	191.80	9.76	12.17	3.19
Mean	7.26	0.09	124.49	10.13	220.25	10.16	11.31	2.92
SE +/-	0.07	0.02	1.91	0.07	1.20	0.11	0.04	0.02
Wetland								
Potushai	6.90	0.52	199.03	10.66	223.85	12.25	10.14	3.66
Tulmulla	6.49	0.80	256.65	17.71	380.75	11.93	14.14	2.68
Kehnoosa	7.03	0.61	183.24	11.76	226.88	9.73	8.60	3.22
Saderkoot	6.66	0.72	176.15	11.91	228.68	11.21	12.46	3.16
Shalabugh	6.65	0.60	145.79	12.88	280.96	13.46	16.09	3.22
Mean	6.74	0.65	192.17	12.98	268.22	11.72	12.29	3.19
SE +/-	0.04	0.05	2.47	0.10	1.09	0.14	0.07	0.02

4.2 Soil organic carbon and its fractions.

4.2.1 Total organic carbon

TOC in surface soil layer was found to be invariably higher than the sub-surface layer under all the land uses. The mean values of TOC in surface (Table 4.5) and sub-surface (Table 4.6) soil layers varied between 1.14 to 5.68 and 0.88 to 5.03 %, respectively. The greatest mean TOC in surface layer was observed in wetland soil and least in paddy soil, whereas in sub-surface layer, highest and lowest values were recorded in wetland and maize soil, respectively. The average TOC in soils (0-30 cm depth) of native ecosystems was estimated as 3.06%, whereas, the value for all other land uses was found to be 1.16 %. Data on 95% C.I (Table A.14 in appendix) reveal that CI values followed the similar trend as mean, with the highest values viz., 5 to 6.34% estimated for wetland and lowest (1.02 to 1.26%) for paddy in surface layer .In subsurface layer, the values for 95% CI were found to be 4.46 to 5.58% in wetland soil and 0.79 to 1.04% in maize soil.

4.2.2 Walkley and Black organic carbon

In surface soil layers the average WBC varied from to 0.85 to 4.26 % (Table 4.5) whereas in sub-surface layers the WBC ranged from 0.68 to 3.54 % (Table 4.6).The values of WBC at both depths were found to be highest under wetland soil followed by forest soil at both surface and sub-surface level. In the surface layer of cultivated lands, the highest mean WBC was recorded in vegetable fields, followed by apple, maize and least in paddy fields. In sub-surface layers of arable lands the WBC varied in similar manner. The highest values (3.74 to 4.78 %) for 95% C.I of WBC in surface soil layer were observed under wetland and lowest(0.75 to 0.94%) were recorded under paddy. At sub-surface the highest and lowest values for CI were found to be 3.13 to 3.94% in wetland and 0.60 to 0.75% in paddy soil (Table A.15 under appendix). By and large the variations recorded in WBC content were similar to TOC, however, with differences in magnitude.

4.2.3 Labile organic carbon

Average LOC (mgKg^{-1}) in the surface soil layers (Table 4.5) was found to be relatively much higher under wetland use (11202) followed by forests (4407) and decreased in the order of grasslands (2820), apple orchards (2366), maize fields (2209), vegetable fields (2082), and paddy fields (2012) respectively. Similarly, in the sub-surface soil layers (Table 4.6) highest mean LOC was recorded under wetland use (9409) followed by forests (2873) and further decreased in the order of grasslands (1831), paddy fields (1324) apple orchards (1100), vegetable fields (1079) and maize fields (1006), respectively. The data on 95% C.I related to LOC (mgKg^{-1}) of soils at two depths are given in Table A.16 (appendix). The highest values (9576 to 12827) for C.I of surface layer were observed under wetland and lowest (1866 to 2156) were recorded under paddy. In the sub-surface the lowest and highest values for CI were found to be 824 to 1345 in maize and 6547 to 9763 in wetland soils.

4.2.4 Particulate organic carbon (POC)

In natural ecosystems as well as in disturbed ecosystems, average POC in surface soil layers was found to be considerably higher than the sub-surface layers. Moreover, on an average POC content was found to be higher than mean LOC at both the depths under all the land uses. The mean values of POC in surface (Table 4.5) and sub-surface (Table 4.6) soil layers varied between 2495 to 13420 (mgKg^{-1}) and 1043 to 11148 (mgKg^{-1}) respectively. The greatest mean POC in surface layer was recorded in wetland soil and least in paddy soil, whereas in sub-surface layer, highest and lowest values were recorded in wetland and vegetable soil, respectively. Mean POC (mgKg^{-1}) in the surface soil layers exhibited the 95% C.I (Table A.17) of 2085.06 to 2904.88, 2884.26 to 4504.73, 2606.01 to 3618.65, 2406.12 to 3179.47, 595.54 to 774.48, 3493.29 to 5303.50 and 11554.31 to 15286.14, for paddy, maize, apple, vegetable, forest, grassland and wetland soil, respectively. Whereas, POC in

Table 4.6: Organic Carbon and its fractions in sub-surface layer of different soils.

Land use type/ Location	TOC -----%-----	WBC	LOC	POC -----mgKg ⁻¹ -----	MBC
Paddy					
Baghurampora	0.77	0.64	1355	1429	95.3
Gamroo	0.89	0.62	1274	1294	133.6
Sumlar	1.18	0.82	1209	1424	245.1
Margund	0.76	0.55	1421	1519	173.4
Yarmuqam	1.01	0.75	1362	1415	214.5
Mean	0.92	0.68	1324	1416	172.4
SE +/-	0.06	0.34	36.55	125.22	10.22
Maize					
Margund	0.67	0.57	732	959	121.9
Athwot	1.23	0.91	1338	1465	264
Yarmuqam	0.51	0.48	624	799	132.7
Gagangir	0.97	0.80	1031	1144	157.6
Kudara	1.02	0.94	1305	1481	209.1
Mean	0.88	0.74	1006	1170	177.1
SE +/-	0.08	0.05	87.00	227.55	26.45
Apple					
Nadihal	1.25	0.92	1172	1539	340.8
Badampura	0.63	0.43	570	798	100.8
Khalmula	0.89	0.67	1350	1607	189.0
Watlar	0.69	0.49	891	1253	97.71
Arin	1.37	1.06	1517	2133	354.7
Mean	0.97	0.71	1100	1466	216.6
SE +/-	0.08	0.07	75.44	215.40	31.00
Vegetable					
Aloosa	1.18	0.96	1086	1105	285.7
Ahan	1.30	1.12	1123	1183	400.5

Contd –Table 4.6	TOC	WBC	LOC	POC	MBC
Dab shalahar	1.51	1.26	1372	1434	372.7
Barsoo	1.10	1.02	808	826	225.0
Quilmuqam	1.49	1.29	1008	666	304.8
Mean	1.32	1.13	1079	1043	317.7
SE +/-	0.05	0.05	87.54	143.22	27.33
Forest					
Preng	1.73	1.25	1961	2220	371.8
Athwot	2.13	1.62	2873	3297	584.2
Kudara	2.13	1.79	4386	5096	394.5
Naranag	1.84	1.30	2333	2888	561.6
Sonamarg	2.23	1.87	2812	3143	492.8
Mean	2.02	1.57	2873	3329	480.9
SE +/-	0.09	0.09	165.45	31.23	36.34
Grassland					
Potushai	0.88	0.63	1252	1612	156.1
Sonamarg	1.95	1.58	2305	2802	446.2
Nagmarg	1.52	1.11	1743	2175	567.8
Trundkhul	1.77	1.39	2071	2800	480.8
Tragbal	1.45	1.07	1783	2148	466.2
Mean	1.52	1.16	1831	2308	423.4
SE +/-	0.11	0.09	165.32	279.55	38.21
Wetland					
Potushai	5.16	3.43	8876	9729	777.6
Tulmulla	6.20	4.42	11253	13951	1011.2
Kehnoosa	4.78	3.45	10091	12180	751.2
Saderkoot	4.77	3.35	10245	12479	637.7
Shalabugh	4.23	3.05	6580	7402	592.3
Mean	5.03	3.54	9409	11148	754.0
SE +/-	0.26	0.18	145.56	674.30	47.55

4.3 Carbon management index

The data on Carbon management index (CMI) given in Table 4.7, reveals that greatest CMI among the soils of terrestrial ecosystems, was recorded in forest land use (162.5), followed by grassland (100) and decreased in the order of apple (87.5), maize (81.0), paddy (72.8) and vegetable soils (70.7), respectively. The carbon pool index and lability index decreased in the same order as CMI. For calculation of CMI the surface soil layer of grassland was used as reference.

Table 4.7: Carbon management index as affected by different land uses

Land use	SOC	LOC	CPI	UNOX-C	Lab C	LI	CMI
Paddy	11400	2012	0.64	9388	0.21	1.13	72.8
Maize	11800	2209	0.67	9591	0.23	1.22	81.0
Apple	12200	2366	0.69	9834	0.24	1.27	87.5
Vegetable	17100	2082	0.97	15018	0.14	0.73	70.7
Forest	23000	4407	1.30	18593	0.24	1.25	162.5
Grassland	17700	2820	1.00	14880	0.19	1.00	100.0

CPI- Carbon pool Index, UNOX-C –unoxidized C (mgKg^{-1}), Lab C–Lability of C, LI-Lability Index, SOC- mgkg^{-1} , LOC- mgKg^{-1}

4.4 Soil organic carbon stocks

Soil organic carbon stocks of different soils are presented in Table 4.8. The SOC stocks varied from 20.41 to 61.97 Mg ha^{-1} in surface soils and 17.22 to 60.21 Mg ha^{-1} in subsurface soils. Largest mean organic carbon stock was recorded in mineral soils of wetlands followed by forest system, grassland, apple, vegetable, maize, and lowest in case of paddy soils. In the sub-surface soil layers largest mean organic carbon stock was recorded in mineral soils of wetlands followed by forest, grassland, vegetable, apple, paddy and lowest in case of maize soils. The total SOC

Table 4.8: Soil organic carbon stocks (Mg ha⁻¹) in surface and sub-surface layers of different soils.

Land use type	-----Surface-----				-----Sub-surface-----			
	SOC	BD	Depth	SOC stock	SOC	BD	Depth	SOC stock
Paddy	1.14	1.19	0.15	20.41	1.05	1.25	0.15	17.27
Maize	1.18	1.26	0.15	22.29	1.04	1.30	0.15	17.22
Apple	1.22	1.38	0.15	25.23	0.98	1.47	0.15	21.42
Vegetable	1.71	0.98	0.15	25.17	1.53	1.08	0.15	21.46
Forest	2.34	0.90	0.15	31.45	2.32	0.98	0.15	29.55
Grassland	1.77	1.16	0.15	30.71	1.54	1.24	0.15	28.33
Wetland	5.68	0.73	0.15	61.97	5.03	0.80	0.15	60.21

SOC-soil organic carbon (%), BD- bulk density(Mg m⁻³), Depth-m

stock (Mg ha^{-1}) in 0-30cm depth (Table 4.13) was estimated as 37.68, 39.51, 46.65, 46.63, 61 and 59.04 for paddy, maize, apple, vegetable, forest, grassland and wetland soil, respectively.

4.4.1 Walkley and Black carbon stocks

Walkley and Black carbon (WBC) stocks of soils are presented in Table 4.9. The WBC stocks varied from 15.20 to 46.52 Mg ha^{-1} in surface soils and 12.73 to 42.38 Mg ha^{-1} in sub-surface soils. The highest and lowest values for WBC stocks, at both the depths were observed under wetland and paddy respectively. In WBC stocks similar trend as total SOC stocks were observed, although the estimated SOC stocks were found to be comparatively lower than the aforementioned total SOC stocks.

Table 4.9: Walkley and Black carbon stocks (Mg ha^{-1}) in surface and sub-surface layers of different soils

Land use	-----Surface-----		-----Sub-surface-----	
	WBC	WBC stock	WBC	WBC stock
Paddy	0.85	15.20	0.68	12.73
Maize	0.89	16.80	0.74	14.51
Apple	0.94	19.52	0.71	15.78
Vegetable	1.25	18.45	1.13	18.41
Forest	1.77	23.74	1.57	22.92
Grassland	1.33	23.02	1.16	21.60
Wetland	4.26	46.52	3.54	42.38

WBC- Walkley and Black carbon (%)

4.4.2 Labile organic carbon stocks

The LOC stocks (Table 4.10) varied from 3.6 to 12.22 Mg ha^{-1} in surface soils and 2.49 to 11.26 Mg ha^{-1} in sub-surface soil of different land uses. The LOC stocks under all the land uses were observed to be higher in surface layers than in the sub-surface layers. Largest mean value of LOC stock was recorded in wetlands followed by forest, grassland, apple, maize, paddy and lowest in case of

vegetable soils. The total LOC stocks (Mgha^{-1}) at 30cm depth were recorded as 6.09, 6.14, 7.32, 4.81,10.12, 8.28 and 23.48 for paddy, maize, apple, vegetable, forest, grassland and wetland soil, respectively (Table 4.13).

Table 4.10: Labile organic carbon stocks (Mgha^{-1}) in surface and sub-surface layers of different soils

Land use	----- Surface -----		----- Sub-surface -----	
	LOC	LOC stock	LOC	LOC stock
Paddy	2012	3.60	1324	2.49
Maize	2209	4.17	1006	1.97
Apple	2366	4.89	1100	2.43
Vegetable	2082	3.06	1079	1.75
Forest	4407	5.92	2873	4.20
Grassland	2807	4.87	1831	3.41
Wetland	11202	12.22	9409	11.26

LOC-Labile organic carbon (mgKg^{-1})

4.4.3 Particulate organic carbon stocks

The data on POC stocks in surface and sub-surface layers is presented in Table 4.11. The data reveal that the carbon stocks pertaining to particulate fraction varied from 4.11 (Mgha^{-1}) (vegetable soil) to 14.64 (Mgha^{-1}) (wetland soil) in surface layers and 1.81 to 13.24 (Mgha^{-1}) in sub-surface soils. In the sub-surface soil layers largest POC stock was recorded in wetlands (13.24 Mgha^{-1}) followed by forest (4.67 Mgha^{-1}), grassland (4.30 Mgha^{-1}), apple (3.24 Mgha^{-1}), paddy (2.66 Mgha^{-1}), maize soils, (2.29 Mgha^{-1}) and lowest in case of vegetable (1.81 Mgha^{-1}). In the surface soil layer the greatest POC stock was recorded in wetland (14.64 Mgha^{-1}), followed by forest land use (7.63 Mgha^{-1}) and decreased in the order of grassland (7.62 Mgha^{-1}), maize (6.98 Mgha^{-1}), apple (6.44 Mgha^{-1}), paddy (4.47 Mgha^{-1}) and vegetable soils (4.11 Mgha^{-1}), respectively. Total POC stocks at 30 cm depth are presented in Table 4.13.

Table 4.11: Particulate organic Carbon stocks (Mgha⁻¹) in surface and sub-surface layers of different soils.

Land use	----- Surface -----		----- Sub-surface -----	
	POC	POC stock	POC	POC stock
Paddy	2495	4.47	1416	2.66
Maize	3695	6.98	1170	2.29
Apple	3112	6.44	1466	3.24
Vegetable	2793	4.11	1116	1.81
Forest	5675	7.63	3329	4.87
Grassland	4398	7.62	2308	4.30
Wetland	13420	14.64	11148	13.24

POC-Particulate organic carbon (mgKg⁻¹).

Table 4.12: Microbial biomass carbon stocks (Mgha⁻¹) in surface and sub-surface layers of different soils.

Land use type	----- Surface -----		----- Sub-surface -----	
	MBC	MBC stock	MBC	MBC stock
Paddy	276.3	0.49	172.4	0.32
Maize	357.5	0.68	177.1	0.35
Apple	367.3	0.76	216.6	0.48
Vegetable	524.9	0.77	317.7	0.52
Forest	685	0.92	480.9	0.70
Grassland	592.3	1.03	423.4	0.79
Wetland	952.4	1.04	754.0	0.90

MBC-Microbial biomass carbon (mgKg⁻¹).

Table 4.13: Total Organic carbon stocks (Mgha⁻¹) and fractions in different soils

Land use type	TOC stock	WBC stock	LOC stock	POC stock	MBC stock
Paddy	37.68	27.93	6.09	7.13	0.81
Maize	39.51	31.31	6.14	9.27	1.03
Apple	46.65	35.3	7.32	9.68	1.24
Vegetable	46.63	36.86	4.81	5.92	1.29
Forest	61.00	46.66	10.12	12.5	1.62
Grassland	59.04	44.62	8.28	11.92	1.82
Wetland	122.18	88.9	23.48	27.88	1.94

4.4.4 Microbial biomass carbon stocks

Mean MBC stocks in surface layer were found to be much higher than that of the sub-surface layers under all land uses. The mean MBC content in the soils under different land uses varied from 0.49 to 1.04 Mgha⁻¹ in surface soil layers and from 0.32 to 0.90 Mgha⁻¹ in sub-surface soils (Table 4.12). Among the agricultural and horticultural ecosystems the average values of MBC stocks were found to be highest under vegetable land use (0.77 Mgha⁻¹) followed by apple orchards (0.76 Mgha⁻¹), maize (0.68 Mgha⁻¹) and lowest in paddy soils in 0-15 cm depth. In sub-surface layers the MBC stocks decreased in similar order across these land uses. The total MBC stocks (Mgha⁻¹) at 30cm depth were recorded as 0.81, 1.03, 1.24, 1.29, 1.62, 1.82 and 1.94 for paddy, maize, apple, vegetable, forest, grassland and wetland soil, respectively (Table 4.13).

4.5 Soil enzyme activities under different land uses

4.5.1 Dehydrogenase

The data (Table 4.14 and 4.15) reveal that the highest DHA activity was recorded in wetland soils. Under all the land uses the DHA activity declined with depth. Contrary to agro-ecosystem soils, forest and grassland soils exhibited higher DHA activity. The DHA activity varied from 10.40 to 22.28 µg TPF g⁻¹ soil h⁻¹ in surface soils and 5.90 to 17.16 µg TPF g⁻¹ soil h⁻¹ in sub-surface soil of the studied land uses. The average soil DHA activity in surface layer of native ecosystems was observed to 20.27 µg TPF g⁻¹ soil h⁻¹ whereas, the overall average activity was observed as 11.32 µg TPF g⁻¹ soil h⁻¹ in all other soils.

Mean DHA activity (µg TPF g⁻¹ soil h⁻¹) in the surface soil layer exhibited the 95% C.I (Table A.19 -appendix) of 10.39 to 10.80, 10.51 to 10.94, 13.22 to 13.88, 10.10 to 10.96, 20.51 to 21.52, 17.42 to 18.06 and 20.31 to 22.96 for paddy, maize, apple, vegetable, forest, grassland and wetland soil respectively. Whereas, the activity in the sub-surface soil layer recorded the C.I (Table A.19) of 6.14 to 6.58, 7.05 to 7.43, 7.82 to 8.32, 5.72 to 6.10, 12.02 to 14.81, 6.87 to 8.53 and

16.88 to 17.45 for paddy , maize , apple, vegetable, forest, grassland and wetland soil, respectively.

4.5.2 Acid phosphatase

The mean values of Ac-P activity ($\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) are presented in Table 4.14 and 4.15. The lowest activity (282.72) in surface layers was recorded in vegetable soils and highest activity (503.75) was related to wetland. The mean Ac-P activity of wetland soils was generally much higher than other soils in the study area. In the sub-surface layer soil layers, the lowest (130.29) and highest activity (436.43) were found to be associated with vegetable and wetland soils respectively.

The 95% C.I values related to the soils are presented in Table A.20 under appendix. The 95% CI values (lower and upper) of surface soils were also lowest (264.30 to 293.59) and highest (488.01 to 519.48) under vegetable soils and wetland soils, respectively.

4.5.3 Alkaline phosphatase.

Average values of alkaline phosphatase (Alk-P) activity in cultivated lands were observed to be lesser than the natural ecosystems systems in the study area. The data on mean Alk-P activity in surface soils are given in Table 4.14. In the surface soil layers highest Alk-P activity in $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$ was recorded in wetlands (500.29) followed by grassland (441.90), forest (425.25), paddy (316.67) maize soils (297.85), apple (289.74) and lowest in case of vegetable (285.05). In the sub-surface soil layer (Table 4.15) the greatest activity was also recorded in wetlands (426.33) followed by grassland (351.14), forest (338.70), paddy (227.96) maize soils (209.83), vegetable (177.60) and lowest in case of apple (166.84).

The data on confidence intervals pertaining to different land uses at two different depths are given in Table A.21. The highest values for 95% C.I of surface layer (498.65 to 512.40) were computed for wetland and lowest (278.39 to 291.70) for vegetable soil . In the sub-surface the highest and lowest values for CI (upper and lower) were found to be 418.08 to 434.58 in wetland and 176.02 to 179.18 in vegetable soils, respectively.

Table 4.14: Activities of enzymes in surface layer of different soils in the study area.

Land use type / Location	DHA	Ac-P	Alk-P	Aryl-S	FDA
Paddy					
Baghurampora	8.713	345.51	278.65	30.94	10.24
Gamroo	11.90	246.70	315.07	34.94	14.85
Sumlar	12.87	335.49	308.32	34.24	14.16
Margund	8.83	275.80	348.49	38.38	10.56
Yarmuqam	10.66	325.84	332.84	36.5	12.72
Mean	10.59	305.87	316.67	35.00	12.50
SE +/-	0.087	3.17	1.17	0.14	0.09
Maize					
Margund	8.26	214.94	296.24	32.95	8.87
Athwot	12.62	311.25	296.5	32.76	10.75
Yarmuqam	8.59	320.24	268.57	29.93	11.58
Gagangir	12.74	259.78	308.26	34.35	8.85
Kudara	11.4	307.41	319.72	35.41	10.34
Mean	10.72	282.72	297.85	33.08	10.07
SE +/-	0.090	3.93	1.08	0.12	0.09
Apple					
Nadihal	15.07	306.41	341.79	39.82	10.56
Badampura	11.49	285.69	264.69	31.40	13.96
Khalmula	17.48	280.81	287.27	34.08	14.31
Watlar	11.45	309.97	266.1	31.53	15.72
Arin	12.28	309.99	288.85	37.14	17.62
Mean	13.55	298.57	289.74	37.19	14.43
SE +/-	0.142	2.58	1.43	0.12	0.10
Vegetable					
Aloosa	10.45	229.83	252.55	29.18	16.77
Ahan	11.75	288.18	306.42	35.07	17.03
Dab shalahar	8.87	400.21	382.19	43.55	15.82

Contd-Table 4.14	DHA	Ac-P	Alk-P	Aryl-S	FDA
Barsoo	9.67	270.56	277.74	31.5	16.42
Quilmuqam	11.26	230.96	206.33	24.53	12.94
Mean	10.40	283.95	285.05	32.77	15.79
SE +/-	0.102	6.88	2.88	0.34	0.07
Forest					
Preng	16.12	451.23	380.86	41.28	16.81
Athwot	18.66	500.73	459.33	51.00	18.51
Kudara	18.43	505.45	466.74	51.91	16.58
Naranag	19.79	402.63	369.55	41.12	18.74
Sonamarg	21.07	491.18	450.79	50.4	16.69
Mean	20.81	470.24	425.45	47.14	17.47
SE +/-	0.132	9.37	1.86	0.25	0.05
Grassland					
Potushai	19.9	361.7	396.92	44.06	10.18
Sonamarg	16.58	542.41	506.25	56.53	16.77
Nagmarg	21.34	390.76	439.80	49.08	13.47
Trundkhul	17.91	525.22	502.51	56.24	15.56
Tragbal	12.97	352.73	364.02	40.38	11.76
Mean	17.74	434.56	441.9	49.25	13.54
SE +/-	0.139	3.70	2.83	0.32	0.14
Wetland					
Potushai	19.58	560.65	537.50	60.13	17.94
Tulmulla	25.19	622.46	589.33	68.04	17.48
Kehnoosa	23.89	403.13	455.31	51.46	16.33
Saderkoot	20.91	500.33	453.11	49.69	16.44
Shalabugh	21.85	432.2	466.22	51.74	14.55
Mean	22.28	503.75	500.29	56.21	16.54
SE +/-	0.142	6.80	2.98	0.35	0.05

DHA -Activity of dehydrogenase ($\mu\text{g TPF g}^{-1}$ soil h^{-1}), Ac-P -activity of acid phosphatase($\mu\text{g PNP g}^{-1}$ soil h^{-1}), Alk-P –Activity of alkaline phosphatase($\mu\text{g PNP g}^{-1}$ soil h^{-1}), Aryl-S –activity of arylsulfatase ($\mu\text{g PNP g}^{-1}$ soil h^{-1}) and FDA –activity of fluorescein diacetate hydrolase (FDA $\mu\text{g g}^{-1}$ soil h^{-1})

4.5.4 Arylsulphatase

In surface soil layers the average arylsulphatase (Aryl-S) activity varied from 32.77 to 56.21 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$ (Table 4.14) whereas in sub-surface layers it ranged from 10.63 to 20.62 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$ (Table 4.15). The mean values of Aryl-S activity in 0-15 cm depth were found to be highest under wetland followed by forest, and similar observation was recorded at 15 to 30 cm depth. The data on 95% CI are given in Table A.22 (appendix). The highest values for C.I (55.40 to 57.01) at surface layer were estimated for wetland and lowest values (30.82 to 32.32) were observed in vegetable soils, whereas the highest and lowest values of CI for sub-surface layers were found to be 20.36 to 20.90 and 11.31 to 11.96 for the same land uses.

4.5.5 Fluorescein diacetate hydrolase activity

The mean Fluorescein diacetate (FDA) hydrolase activity ($\mu\text{g g}^{-1} \text{ soil h}^{-1}$) of the soils ranged from 10.07 to 17.47 in surface layer (Table 4.14) and from 5.20 to 8.14 in sub-surface layer (Table 4.15) with highest and least activity related to forest and maize land use, respectively. The data on 95% C.I are presented in Table A.23 (appendix). Mean FDA activity in the surface soil layer exhibited the C.I of 12.30 to 12.72, 9.87 to 10.29, 14.19 to 14.68, 15.62 to 16.00, 17.34 to 17.60, 13.22 to 13.90 and 16.42 to 16.6 $\mu\text{g g}^{-1} \text{ soil h}^{-1}$ for paddy, maize, apple, vegetable, forest, grassland and wetland soil respectively. Whereas, the activity in the sub-surface soil layer recorded the 95% C.I of 6.50 to 6.82, 5.08 to 5.34, 6.45 to 9.69, 8.05 to 8.23, 7.06 to 7.39, 6.06 to 6.47 and 6.66 to 6.87 for paddy, maize, apple, vegetable, forest, grassland and wetland soil, respectively.

Table 4.15: Activities of enzymes in sub-surface layer of different soils in the study area.

Land use type / Location	DHA	Ac- P	Alk-P	Aryl-S	FDA
Paddy					
Baghurampora	4.91	128.89	189.11	11.24	4.89
Gamroo	8.26	134.33	274.9	11.80	9.01
Sumlar	8.70	147.66	225.77	13.33	7.10
Margund	4.28	126.67	218.99	9.42	5.68
Yarmuqam	5.63	143.30	231.07	12.4	6.57
Mean	6.35	136.17	227.96	11.63	6.65
SE +/-	0.06	2.22	1.38	0.14	0.06
Maize					
Margund	4.97	162.11	188.24	10.60	4.12
Athwot	8.36	125.52	185.63	10.76	5.02
Yarmuqam	5.42	137.89	177.48	9.99	7.66
Gangangir	8.51	104.37	274.37	12.61	4.02
Kudara	8.95	149.57	223.43	11.86	5.22
Mean	7.24	135.89	209.83	11.16	5.20
SE +/-	0.05	2.96	1.61	0.05	0.05
Apple					
Nadihal	11.73	174.25	232.93	12.55	8.16
Badampura	8.74	143.39	136.6	9.53	6.48
Khalmula	6.36	157.24	135.81	11.51	5.77
Watlar	6.37	183.06	125.98	9.89	7.26
Arin	7.14	130.00	202.88	10.61	7.65
Mean	8.07	157.58	166.84	10.81	7.06
SE +/-	0.22	2.86	2.21	0.03	0.22
Vegetable					
Aloosa	6.24	129.28	156.05	9.27	7.75
Ahan	7.13	117.38	171.67	11.61	8.77

Contd-Table 4.15	DHA	Ac- P	Alk-P	Aryl-S	FDA
Dab shalahar	4.11	121.88	191.85	14.25	8.31
Barsoo	5.02	122.33	176.90	10.44	7.87
Quilmuqam	7.02	160.59	191.54	7.58	7.99
Mean	5.90	130.29	177.60	10.63	8.14
SE +/-	0.03	3.83	1.08	0.12	0.03
Forest					
Preng	10.04	389.08	290.89	14.01	5.64
Athwot	13.32	402.01	402.51	16.67	7.06
Kudara	16.14	443.8	346.64	17.73	7.72
Naranag	11.25	356.07	284.91	13.78	7.86
Sonamarg	11.18	421.91	368.55	16.99	7.80
Mean	12.38	402.57	338.7	15.83	7.21
SE +/-	0.06	11.29	2.05	0.09	0.06
Grassland					
Potushai	7.24	159.22	237.62	15.51	4.6
Sonamarg	8.47	286.54	397.82	18.95	7.64
Nagmarg	7.29	205.77	373.79	16.31	6.40
Trundkhul	8.43	338.54	462.9	18.68	7.20
Tragbal	9.37	193.30	283.58	13.41	5.45
Mean	8.16	236.67	351.14	16.57	6.25
SE +/-	0.08	9.67	3.83	0.13	0.08
Wetland					
Potushai	15.32	501.86	412.61	21.62	7.07
Tulmulla	22.53	562.79	567.39	23.19	6.89
Kehnoosa	16.33	324.74	409.96	21.79	7.90
Saderkoot	17.13	403.46	395.47	17.53	6.42
Shalabugh	14.5	389.15	346.23	18.97	5.65
Mean	17.16	436.40	426.33	20.62	6.78
SE +/-	0.03	11.02	3.57	0.11	0.03

4.6 Soil Quality Assessment

4.6.1 Unscreened additive index

The unscreened additive index (*SQI-un*) computed through linear (*SQI-un.1*) and non-linear scoring (*SQI-un.2*) function are presented in Table 4.16. The maximum value for *SQI-un.1* was observed to be 0.997 for forest soil, followed by apple orchard soils(0.762) and further decreased in the order of vegetable soil (0.730), maize soil(0.711), and the lowest value was estimated for paddy soil (0.680). Similarly, the soil quality index values obtained through non-linear scoring were estimated as 0.645, 0.676, 0.720, 0.691 and 0.998 for paddy, maize, apple, vegetable and forest soil, respectively. The linear and non-linear scores of the indicators are the same as used for computation of other indices in this study.

Table 4.16: Unscreened additive index of different soils

Land use type	<i>SQI-un.1</i>	<i>SQI-un.2</i>
Paddy	0.680	0.645
Maize	0.711	0.676
Apple	0.762	0.720
Vegetable	0.730	0.691
Forest	0.997	0.998

SQI-un.1-unscreened SQI (linear scoring). *SQI-un.2*- unscreened SQI (non-linear scoring).

4.6.2 Principal component analysis based soil quality index

The 23 SQ indicators analyzed to evaluate various land uses were put through PCA. As a result five principal component (PC) groups were obtained that best explained variability in the data (Table 4.17). Communalities of the SQ indicators

show that individual indicators accounted for 65 to 96% of the variance. The first three PCs

Table 4.17: Principal component analysis results using 23 potential soil quality indicators to evaluate different land use types/ecosystems

Eigenvector	Principal Component (PC)					Communalities
	1	2	3	4	5	
BD	<u>-0.85</u>	0.34	-0.18	0.26	0.19	0.78
Water (w _m)	0.52	-0.37	-0.09	-0.17	-0.20	0.68
Sand	-0.41	-0.48	-0.39	0.40	-0.12	0.96
Silt	0.13	0.48	-0.50	<u>-0.61</u>	-0.17	0.88
Clay	0.69	0.12	<u>0.63</u>	0.40	0.16	0.79
pH	-0.38	0.01	0.43	-0.02	0.05	0.46
EC	0.59	-0.51	-0.18	-0.05	0.18	0.82
Av.N	<u>0.94</u>	-0.19	-0.05	0.06	0.10	0.96
Av.P	0.54	0.42	0.30	0.02	<u>0.71</u>	0.82
Av.K	0.56	<u>0.62</u>	0.18	0.19	-0.32	0.73
Av.S	0.63	<u>0.65</u>	-0.12	0.16	0.11	0.83
Ex.Ca	0.50	<u>0.68</u>	-0.01	0.27	0.04	0.75
Ex.Mg	0.48	<u>0.70</u>	0.01	0.14	0.06	0.74
DHA	<u>0.83</u>	0.08	0.53	-0.15	-0.10	0.87
Ac-P	<u>0.82</u>	0.28	0.17	-0.03	0.10	0.85
Alk-P	0.43	-0.06	-0.11	0.09	-0.07	0.88
Aryl-S	<u>0.84</u>	0.24	0.12	-0.02	0.21	0.88
FDA	0.53	-0.02	<u>0.62</u>	-0.32	0.15	0.79
TOC	<u>0.85</u>	-0.40	-0.17	0.07	-0.04	0.96
WBC	<u>0.83</u>	-0.42	-0.18	0.06	-0.04	0.96
POC	0.71	-0.34	0.10	-0.05	0.18	0.93
MBC	<u>0.81</u>	-0.34	-0.21	0.03	0.14	0.89
LOC	<u>0.92</u>	-0.16	-0.10	0.09	0.07	0.74
Eigen Value	<i>11.3</i>	<i>3.92</i>	<i>2.11</i>	<i>1.14</i>	<i>1.04</i>	-
Variance %	<i>48.49</i>	<i>16.36</i>	<i>8.81</i>	<i>4.78</i>	<i>4.34</i>	-
Cumulative Variance	48.49	64.86	73.68	78.46	82.8	-

Boldface eigenvector values correspond to the PCs highly weighted variables /the eigen vectors within 10% of the highest factor loadings examined for the index.

Bold-underlined factors correspond to the indicators retained in the SQ index

accounted for about 74% of the variance, indicating that these potential components would be most important to assess SQ .When combined, the first five PCs with eigen values >1 explained about 83% of the soil variability among the

land uses. As indicated in the Table 4.17, Soil BD, TOC, WBC, MBC, LOC, Av.N , DHA, Ac-P, and Aryl-S were the highly loaded factors attributed to PC1. Bulk density was not well correlated with other variables (Table 4.18) and therefore, it was retained. Av.N had a very high factor loading, and thus, was retained for the MDS . Significant correlation was observed among the variables like TOC, WBC, MBC, LOC and DHA, besides, the significant correlation of these with highly weighted Av.N, and therefore, these were not included in the MDS. The other variables retained for soil indexing include Ac-P, and Aryl-S, with factor loadings of 0.82 and 0.84 respectively. The highly loaded variables in PC2 were Av.K, Av.S, Ex.Ca and

Table 4.18: Inter-correlations between highly weighted variables under different PCs

PC1 variables	BD	TOC	WBC	MBC	LOC	Av.N	DHA	Ac-P	Aryl-S
BD	1								
TOC	0.66*	1							
WBC	0.65*	0.99**	1						
MBC	0.56	0.91**	0.93**	1					
LOC	0.49	0.87**	0.91**	0.76*	1				
Av.N	0.56	0.86**	0.84*	0.77*	0.81*	1			
DHA	0.55	0.73*	0.76*	0.72*	0.57	0.61	1		
Ac-P	0.45	0.65*	0.68*	0.58	0.66*	0.41	0.44	1	
Aryl-S	0.29	0.58	0.54	0.62	0.43	0.29	0.63*	0.53	1
PC2 variables	Av.K	Av.S	Ex.Ca	Ex.Mg					
Av.K	1								
Av.S	0.44	1							
Ex.Ca	0.49	0.61	1						
Ex.Mg	0.29	0.58	0.57	1					
PC3 variables	Clay	FDA							
Clay	1								
FDA	0.18	1							

**significant at P<0.01 * significant at P<0.05

Ex.Mg. All of these variables were not significantly correlated ($r < 0.6$) and therefore, retained in PC2 for integration into SQI. In PC3 the highest loaded factors were clay and FDA and these variables were found to be not correlated,

and therefore included in MDS for index development. The inclusion of clay in MDS is justifiable not only by virtue of its higher loading in PC3, but as well as for being frequently associated with higher SOC. Silt was the only highly loaded factor in PC4 and thus retained for indexing. The only highly loaded variable in PC5 was Av.P. Soil Av.P is often a limiting factor for both horticultural and field crop production in the study area, therefore, its inclusion in assessment of SQ seems rational.

The linear scores, non-linear scores, as well as weights used for PCA based soil quality indices (*SQI-pca*) are given in Table 4.19 and 4.20. Finally, the following equation was used for computation of the index.

$$SQI-pca = 0.586 (BD + Av.N + Ac-P + Aryl-S) + 0.198 (Av.K + Av.S + Ex.Ca + Ex. Mg) + 0.106 (clay + FDA) + 0.058(silt) + (0.052 Av.P)$$

The data on *SQI-pca* (Table 4.19 and 4.20) which includes indices computed through linear scoring (*SQI-pca.1*) and non-linear scoring (*SQI-pca.2*) reveal that the lowest value for *SQI-pca.1* was recorded for paddy soil (0.692) and increased in the order of maize soil (0.701), vegetable(0.718), apple (0.767) and forest soil (0.996). The *SQI-pca.2* varied in the similar order; however, the values estimated for the indices were generally lesser than those of *SQI-pca.1*. The *SQI-pca.2* decreased in the order forest (0.995) > apple (0.697) > vegetable (0.669) > maize (0.650) > paddy (0.641).

Table 4.19: Minimum data set of indicators, weights and linear scores used in PCA based SQI for different soils.

Principal component	<i>1</i>				<i>2</i>				<i>3</i>		<i>4</i>	<i>5</i>	SQI-pca.1
Weightage	----- <i>0.586</i> -----				----- <i>0.198</i> -----				---- <i>0.106</i> ----		<i>0.058</i>	<i>0.052</i>	
Indicator	BD	Av.N	Ac-P	Aryl-S	Av.K	Av.S	Ex.Ca	Ex.Mg	Clay	FDA	Silt	Av.P	
Land use	----- Linear Score -----												
Paddy	0.75	0.55	0.65	0.74	0.66	0.76	0.82	0.55	0.94	0.72	0.83	0.64	<i>0.692</i>
Maize	0.71	0.58	0.60	0.70	0.63	0.86	1.00	0.70	0.92	0.58	1.00	0.68	<i>0.701</i>
Apple	0.65	0.78	0.63	0.79	0.78	0.98	0.75	0.82	1.00	0.83	0.84	0.75	<i>0.767</i>
Vegetable	0.91	0.81	0.62	0.71	0.77	0.72	0.74	0.54	0.73	0.90	0.71	0.68	<i>0.718</i>
Forest	1.00	1.00	1.00	1.00	1.00	1.00	0.99	1.00	0.94	1.00	1.00	1.00	<i>0.996</i>

Table 4.20: Minimum data set of indicators, weights and non-linear scores used in PCA based SQI for different soils.

Principal component	<i>1</i>				<i>2</i>				<i>3</i>		<i>4</i>	<i>5</i>	SQI-pca.2
Weightage	----- <i>0.586</i> ----- -				----- <i>0.198</i> -----				---- <i>0.106</i> ----		<i>0.058</i>	<i>0.052</i>	
Indicator	BD	Av.N	Ac-P	Aryl-S	Av.K	Av.S	Ex.Ca	Ex.Mg	Clay	FDA	Silt	Av.P	
Land use	-----Non-Linear Score----- ---												
Paddy	0.45	0.56	0.64	0.74	0.65	0.74	0.78	0.55	0.93	0.72	0.83	0.64	<i>0.641</i>
Maize	0.41	0.58	0.60	0.71	0.63	0.84	1.00	0.64	0.90	0.56	1.00	0.68	<i>0.650</i>
Apple	0.34	0.75	0.63	0.78	0.76	0.98	0.70	0.75	1.00	0.83	0.84	0.68	<i>0.697</i>
Vegetable	0.69	0.78	0.60	0.69	0.71	0.68	0.68	0.51	0.73	0.88	0.70	0.75	<i>0.669</i>
Forest	1.00	1.00	1.00	1.00	1.00	1.00	0.98	1.00	0.94	1.00	1.00	1.00	<i>0.995</i>

4.6.3 Regression equation based soil quality index

Multiple linear regression (Table 4.21) was used to compute this index from soil physical, chemical and biological quality parameters and SOC (dependent variable). As result of stepwise multiple linear regression the parameters retained for indexing include BD, Clay, Av.N, Ex.Mg, LOC, MBC, and DHA. From the standardized regression coefficients of retained variables, the following SQI equation was developed.

$$SQI-reg = 0.091 \times S(Clays) + 0.140 \times S(BD) + 0.155 \times S(Av.N) + 0.114 \times S(Ex.Mg) + 0.64 \times S(LOC) + 0.217 \times S(MBC) + 0.150 \times S(DHA).$$

where, S is the score for the individual variables and the coefficients are the weighing factors obtained from the regression equation. The soil quality indices developed through linear (*SQI-reg.1*) and non-linear scoring function (*SQI-reg.2*) are presented in Table 4.22. Forest soil (1.0) recorded the highest value for *SQI-reg.1* followed by vegetable soil(0.62) and apple(0.62) and further decreased in the order of maize (0.57) and paddy(0.53). The *SQI-reg.2* for paddy, maize, apple, vegetable and forest soil were recorded as 0.48, 0.51, 0.55, 0.57 and 1.00, respectively.

Table 4.21: Stepwise regression of soil organic carbon with soil physical, chemical and biochemical properties

Parameter	B	Std. error	Beta	Significance
Constant	0.711	0.412		0.088
Clay	0.189	0.091	0.091	0.003
Bulk density	-0.914	0.217	-0.140	0.000
Available N	0.575	0.124	0.155	0.001
Exchangeable Mg	-0.220	0.045	-0.114	0.000
Labile organic carbon	0.031	0.008	0.642	0.000
Microbial biomass carbon	0.019	0.003	0.217	0.000
Dehydrogenase activity	-0.440	0.011	-0.150	0.000

R²=0.960, B-Unstandardized coefficients, Beta-Standardized coefficients.

Table 4.22: Regression based soil quality indices of different land use types.

Land use type	<i>SQI-reg.1</i>	<i>SQI-reg.2</i>	Indicators
Paddy	0.530	0.486	<i>Clay, Bulk density, Available N, Exchangeable Mg, Labile organic carbon, Microbial biomass carbon, Dehydrogenase activity</i>
Maize	0.574	0.514	
Apple	0.620	0.551	
Vegetable	0.622	0.572	
Forest	1.000	1.000	

SQI-reg.1: Regression based SQI developed through Linear scoring (LS)

SQI-reg.2: Regression based SQI developed through Non-Linear scoring (NLS)

4.6.4 Conceptual SQI

The details regarding conceptual framework model for assessment of soil quality are presented in Table 4.23 and Table 4.24. As depicted in the tables, the critical functions identified for the assessment of soil quality were; (1) Sustain plant growth (2) nutrient cycling (3) resist biochemical degradation (4) filtering and buffering and (5) facilitate water transport, with the assigned weights as 0.4, 0.2, 0.2, 0.1 and 0.1 respectively. Moreover, the linear scores and non-linear scores (Table 25 and 26) were used to compute indices through two different ways, as done with other indexing methods. The modified model is given under;

$$SQI-con = 0.4/3(qN+qP+qK) + 0.2(qDHA)+ 0.2 (qSOC) + 0.1(CEC) + 0.1(qBD)$$

Comparison of linear and non-linear scores of few soil quality indicators is given in Fig 4.1. The conceptual soil quality indices developed through linear scoring (*SQI-con.1*) and non-linear scoring (*SQI-con.2*) were found to be highest in forest soil (1.00) and least in paddy soils of the study area. The *SQI-con.1* for paddy, maize, apple, vegetable and forest soil was recorded as 0.589, 0.601, 0.731, 0.687 and 1.00,

Table 4.23: Conceptual framework model for assessment of soil quality using linear scoring function.

Function	Weight	Indicator	-----Land use type-----				
			Paddy	Maize	Apple	Vegetable	Forest
			-----Linear score-----				
1. Sustain plant growth	0.4	0.4/3 N	0.55	0.58	0.78	0.81	1.00
		0.4/3 P	0.64	0.68	0.75	0.68	1.00
		0.4/3 K	0.66	0.63	0.78	0.77	1.00
2. Nutrient cycling	0.2	DHA	0.51	0.52	0.85	0.50	1.00
3. Resist biochemical degradation	0.2	SOC	0.49	0.50	0.52	0.73	1.00
4. Filtering & buffering	0.1	CEC	0.68	0.75	0.85	0.65	1.00
5. Facilitate water transport	0.1	BD	0.75	0.71	0.65	0.91	1.00
<i>SQI-con.1:</i>			<i>0.589</i>	<i>0.601</i>	<i>0.731</i>	<i>0.687</i>	<i>1.000</i>

Table 4.24: Conceptual framework model for assessment of soil quality using non-linear scoring function.

Function	Weight	Indicator	-----Land use type-----				
			Paddy	Maize	Apple	Vegetable	Forest
			-----Non-linear score-----				
1. Sustain plant growth	0.4	0.4/3 N	0.56	0.58	0.75	0.78	1.00
		0.4/3 P	0.64	0.68	0.75	0.68	1.00
		0.4/3 K	0.65	0.71	0.76	0.71	1.00
2. Nutrient cycling	0.2	DHA	0.51	0.52	0.66	0.51	1.00
3. Resist biochemical degradation	0.2	SOC	0.51	0.53	0.66	0.80	1.00
4. Filtering & buffering	0.1	CEC	0.68	0.74	0.84	0.66	1.00
5. Facilitate water transport	0.1	BD	0.45	0.41	0.34	0.69	1.00
<i>SQL-con.2:</i>			<i>0.563</i>	<i>0.577</i>	<i>0.683</i>	<i>0.677</i>	<i>1.000</i>

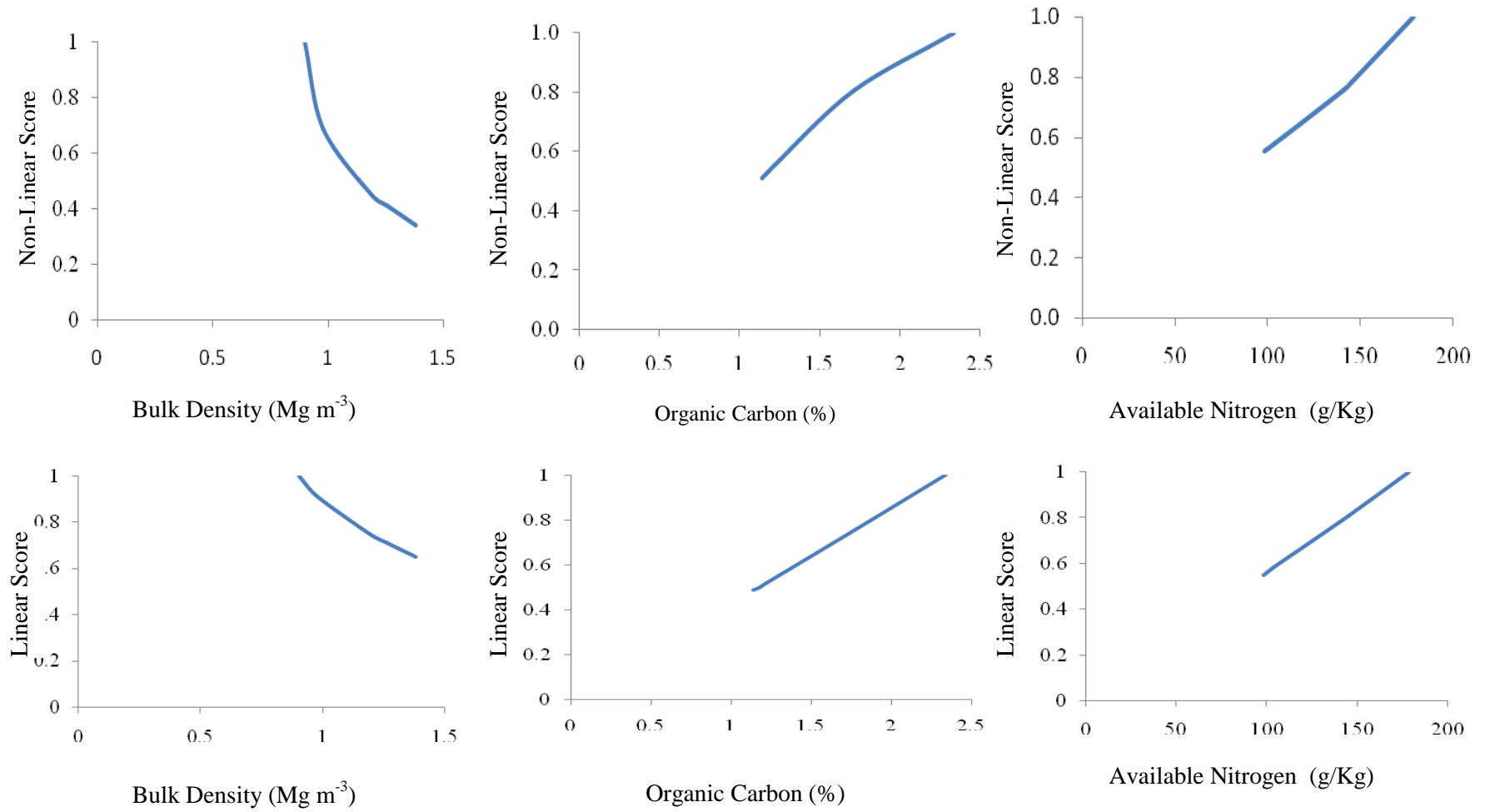


Fig.4.1: Comparison of non-linear and linear scores of few SQ indicators

Table 4.25: Overall soil quality indices for various land use types.

Land use type	Soil Quality Index							
	Unscreened (<i>SQI-un</i>)		Regression (<i>SQI-reg</i>)		PCA (<i>SQI-pca</i>)		Conceptual (<i>SQI-con</i>)	
	LSF	NLSF	LSF	NLSF	LSF	NLSF	LSF	NLSF
Paddy	0.680	0.645	0.530	0.486	0.692	0.640	0.589	0.563
Maize	0.711	0.676	0.574	0.512	0.701	0.651	0.601	0.577
Apple	0.762	0.720	0.620	0.551	0.767	0.697	0.731	0.683
Vegetable	0.730	0.691	0.622	0.572	0.718	0.669	0.687	0.677
Forest	0.997	0.998	1.000	1.000	0.996	0.995	1.000	1.000

LSF-Linear scoring function, NLSF-Non-linear scoring function,

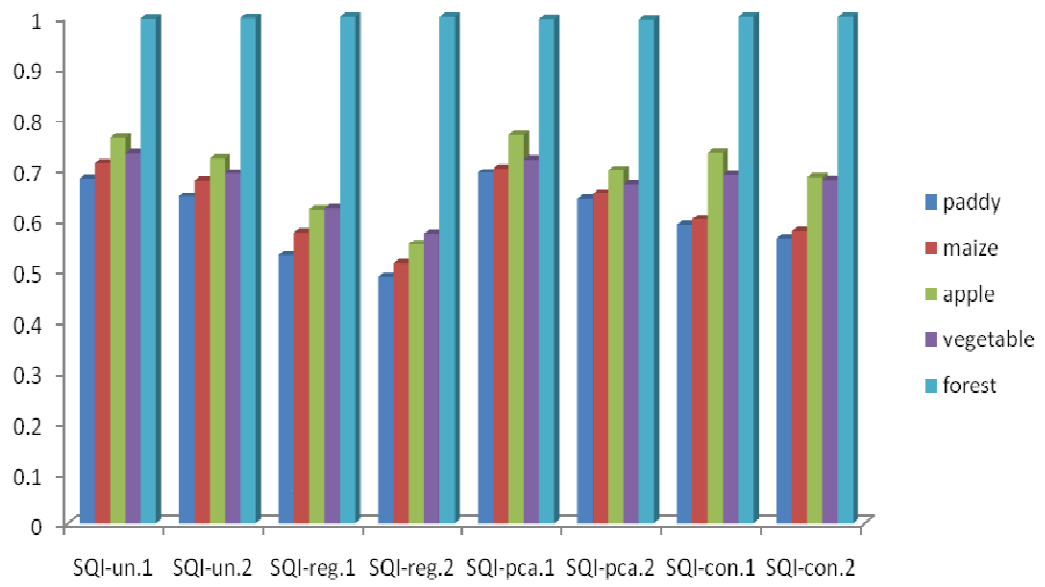


Fig.4.2: Soil quality indices of different land use types.

respectively. Similarly, *SQI-con.2* for paddy, maize, apple, vegetable and forest soil were estimated as: 0.563, 0.577, 0.683, 0.677 and 1.00, respectively

4.7 Sensitivity of indexing procedures

The evaluation of various soil quality indexing methods (Table 4.26) used in assessing SQI under different land use types was carried out by estimating their sensitivity. The sensitivity of indexing procedures (Table 4.26) decreased as follows: *SQI-reg.2* (2.05), *SQI-reg.1* (1.88), *SQI-con.2* (1.77), *SQI-con.1*(1.69), *SQI-pca.2* (1.55), *SQI-un.2*(1.54), *SQI-un.1*(1.46), and least in case of *SQI-pca.1*(1.43). Thus, the regression equation based index developed through non-linear scoring of indicators is the most sensitive method. Conceptual framework based indices (*SQI-con.1* and *SQI-con.2*) and PCA based (*SQI-pca2*) have also scored well in terms of sensitivity.

4.8 Identification of limiting parameters

Comparison of indicator scores (non-linear) between the reference(forest) soil and soils under other land use types is shown in Fig 4.3. In the radar diagram the areas enclosed by lines of different colors represent overall scores of different

Table 4.26: Evaluation of indexing and scoring methods applied for developing soil quality indices

Indexing procedure	<u>Linear Scoring Function</u>		<u>Non-linear scoring function</u>	
	Range	Sensitivity	Range	Sensitivity
Unscreened additive index	0.680 - 0.995	1.466	0.645 - 0.998	1.547
Principal component analysis	0.692- 0.996	1.439	0.640 - 0.995	1.554
Regression equation	0.530 -1.000	1.886	0.486 – 1.000	2.057
Conceptual framework	0.589 - 1.000	1.697	0.563 – 1.000	1.776

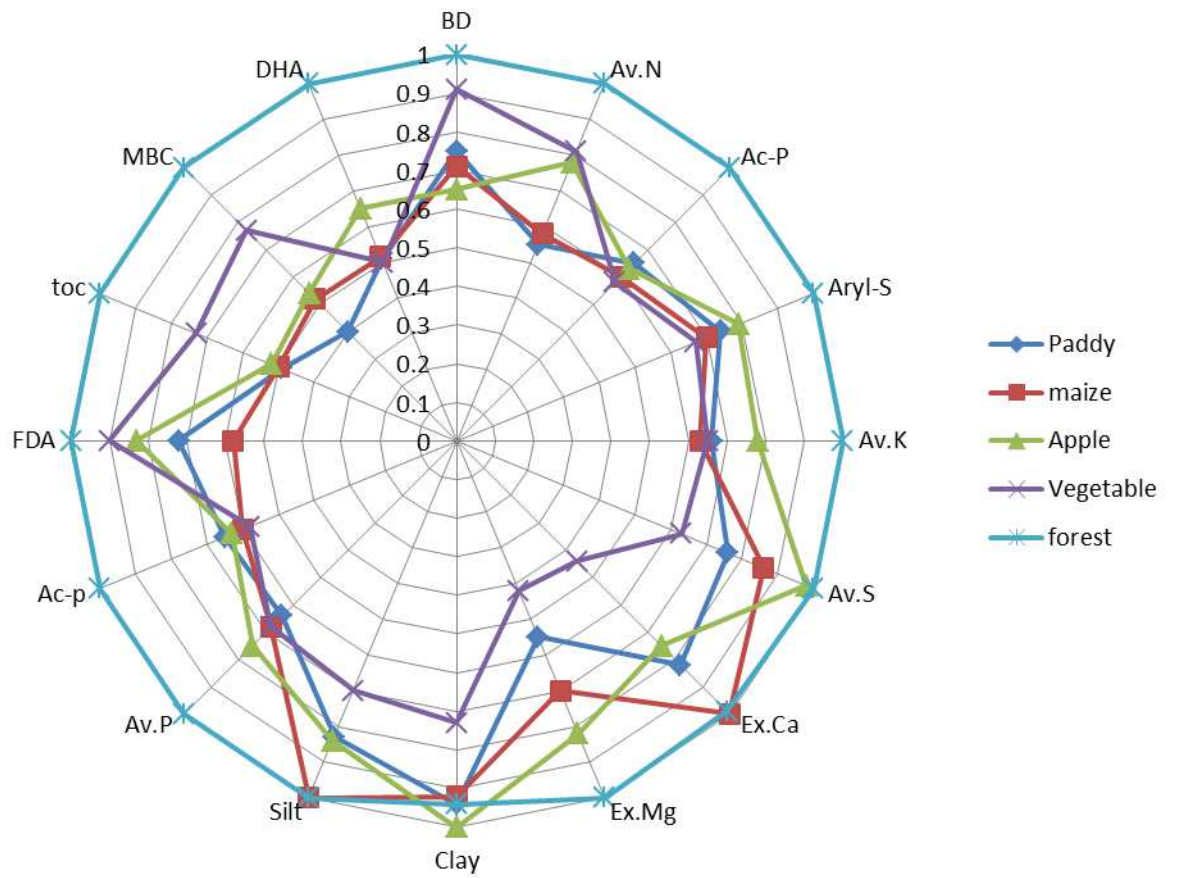


Fig. 4.3: Radar graph showing the average scores of key indicators as influenced by different land use types

indicators under different land use types/ecosystems, and the lines radiating from centre represent scores of different indicators. The figure indicates that the reference soil has almost no limiting parameter whereas other soils have many limiting parameters. Paddy soil has many shortcomings and maximum number of limiting parameters, which influence and degrade its overall quality by different magnitudes.

Chapter – 5

DISCUSSION

The present study was conducted to assess the impact of land use on soil organic carbon stocks as well as to study the impact of different land uses on soil quality and develop soil quality indices. To fulfill these objectives different soil physical, chemical and biochemical properties, as well as organic carbon fractions were determined and presented in the preceding part of this dissertation. In this chapter the results obtained during the investigation are elucidated and discussed in light of the relevant research conducted locally or internationally by different workers.

5.1 Physico-chemical properties of soils under different land uses/ecosystems.

5.1.1 Bulk density

The soil bulk density varied from 0.73 to 1.47 Mgha⁻¹ in surface and sub-surface layers, under different land use types. Apple orchard soils were observed to have highest bulk density in both the layers. Bhardwaj *et al.*, (2013) also recorded the maximum bulk density in the horticulture land use system and least in forest soils of temperate north western Himalayas. In general, higher soil bulk density was observed in cultivated land than native ecosystems. This may be due to continuous tillage. Similar results were obtained by Girma (1998), Celik (2005) and also by Islam and Weil (2000.b) who observed increase in soil bulk density from 1.22 to 1.38 Mgha⁻¹ as a result of tillage. Moreover, bulk density of grassland soil was observed to be higher than forest soil in this study. This can be attributed to the fact that surface soils in pastures are subject to overgrazing and compaction (Deuchars *et al.*, 1999; Carter *et al.*, 1998) and also due to lower OC content in these soils than those of the forest soils.

5.1.2 Soil texture

On the basis of texture, the studied soils were classified as clay loam, silty clay, silty clay loam, silty loam, loam and clay. Majority of the sites in the study area exhibited clay loam texture followed by silty clay loam. The average clay content of the cultivated surface soils including orchard soils varied from 24.32 to 33.46 % with an average of about 30 %. This is in harmony with the results of Ramzan *et al.*, (2014) who have also classified the soils as clay loam and silty clay loam at some study sites in the area. The texture of sub-surface soil layers did not differ from that of the surface layers in all land uses. Therefore, it can be concluded that land use has not had marked impact on inherent soil property like texture of surface soils in the studied ecosystems. Moreover, the wetland soils exhibited the highest average clay content of about 43% indicating some deposition of clay in the lower topographic positions of the watersheds (Cooper *et al.*, 1987). Thus, texture of the soils in the area reflects the nature of the parent material from which the soils were developed and the drainage pattern of the area. This is in accordance with the conclusions drawn by Ahukaemere *et al.*, (2012) in their study on soil quality in different agricultural land use types.

5.1.3 Soil pH

The Soil pH in the surface and subsurface soils varied from 6.42 to 7.30 and from 6.74 to 7.26 respectively. Therefore, only the surface mineral soils of wetlands in the area can be termed as moderately acid soils as per the classification of soil reaction suggested by Brady (1996) and rest of the soils under all the ecosystems fall in the neutral range. The mean pH of forest soils was however, observed to be lower than cultivated soils, this could be due to slower decomposition and higher organic matter in soil contributed by the coniferous vegetation. According to De Hann (1977) the reduction in pH of forest soils can be attributed to accumulation and subsequent slow decomposition of organic matter, which releases acids. The subsurface layers have higher pH than surface layers which may be due to differences in OM content (Hodges, 1996), use of inorganic fertilizers in arable soils and some leaching of bases from the surface layer. Average soil pH of maize fields was found to be higher than the adjacent

forest soils, as well the paddy soils elsewhere in the study area. Adams and Sidle (1987) have also recorded low pH in undisturbed natural forests as compared to disturbed ecosystems. The higher pH of maize soils in comparison to forest soil may be due to impact of tillage practices leading to loss of OM in maize soils. However, the lower pH of paddy soil, in comparison to maize soil, may be due to the fact that larger quantities of inorganic N fertilizer are used in paddy fields than in the maize fields of the study area. The inference is supported by the findings of Kizilkaya and Dengiz (2010).

5.1.4 **Electrical conductivity**

Electrical conductivity (EC) of surface soils ranged from 0.11 to 0.60 dSm⁻¹. The results of EC in the present investigation were found to be within the reported range of values for different land use types. All the soils in the study area including those of the wetlands have EC values well below 1dSm⁻¹ indicating that there is no salinity hazard. However, EC of paddy, maize, vegetable fields and apple orchard soils was observed to be slightly higher than forest and grassland soils. Kizilkaya and Dengiz (2010) explain that changing forest to cultivated land have increased EC values in their studied area, apparently due to fertilizer and management practices. On the other hand, higher EC was recorded in apple orchard soils compared to arable land. Sharma and Gupta (1989) have also recorded higher values of EC under tree-based land use system, as compared to arable land. Higher EC in mineral soils of wetlands may be attributed to deposition of fine sediments, nutrients and salts by runoff from elevated adjacent lands.

5.1.5 **Available nutrients**

The values of available N varied in different land use types of the study area, indicating the alteration of N dynamics in soils across a range of habitats (Garcia-Oliva, *et al.*, 2006; Monkiedje *et al.*, 2006). Quantity of available N in wetland and forest ecosystems was recorded to be higher than that of agriculture and horticulture ecosystems. This may be attributed to the effect of higher organic matter levels as well as enhanced enzymatic activities, mineralization and

mobilization of available nutrients including N. The inference is supported by findings of Marinari *et al.*,(2006). Yu and Ehrenfeld (2009) also reported larger pools of mineralizable N under both constant and fluctuating moisture regimes in some wetland areas. Furthermore, in wetlands nutrients can easily be transferred from upland mineral soils to downstream wetland areas through the movement of water. Higher available N in forest soil as compared to grassland, agriculture and horticulture land use has also been reported by Pal *et al.*, (2013). This may be due to higher water holding capacity that reduces nitrate leaching as well as higher humus content in surface soil layer of forests. Besides the reason that the availability of N depends to a large extent on the amount as well as properties of organic matter (De Hann, 1977).

Besides, available N, available P, K and S were also observed to be higher in forest and wetland soils. Pal *et al.*, (2013) reported higher P and K as well as higher fertility index of forest soils followed by grassland, horticulture and agriculture soil in north-western Himalaya. In the present study forest soil was found to have higher OC and available P. The OC has been reported to be positively correlated with available P in numerous studies. According to Tisdale *et al.*, (1997) nearly 50% soil phosphorus is present in organic form which explains presence of higher quantities of available P in forest, grassland and wetland soils.

The mean available K in wetland, forest and grassland soils were found to be higher than cultivated lands. However, the values of available K obtained in the present study are in accord with those recorded by some other investigators (Nisar and Lone, 2013) in similar areas of the Kashmir Himalaya. It is known that exchangeable K in soil largely depends on the composition of parent rock material, therefore, the differences in available K between native and cultivated ecosystems, as observed in the present study, can be assigned to its differential removal from soil as well as leaching and lessivage from cultivated lands. Moreover, the higher organic matter in soil has also apparently improved the retention of K in undisturbed soils. The conclusion is supported by findings of Boruah and Nath (1992) and other workers who have observed that OC, N, P and

K were positively correlated chiefly because all these attributes were intimately linked with soil humus. Furthermore, in this study, mean available K in maize soils was found to be lower than that of paddy soils, this may be due to the reason that higher quantities of fertilizer K are added to paddy soils and also because of the fact that maize crop removes more K per unit of dry matter produced.

In this study higher available S levels were observed in natural ecosystems, than disturbed ecosystem soils. Highest available S content in both surface and sub-surface soil was observed in wetland soils and lowest in vegetable soils. Higher SO_4^{2-} content in wetland soils has also been reported by Kang and Freeman(1999). As stated earlier the transfer of nutrients from upland mineral soils to topographically low lying wetlands, as well as contribution from organic matter may be the underlying reasons. Comparatively, the low available S in vegetable soil can be explained by its removal in larger amounts by cruciferous crops.

Exchangeable Ca and Mg in surface soils of native ecosystems were higher than agriculture and horticulture ecosystems. This is in accord with the findings of different investigators which indicate that continuous cultivation and use of acid forming inorganic fertilizers deplete exchangeable Ca and Mg (Aitken *et al.*, 1999; He *et al.*, 1999). Moreover, the value for mean exchangeable Ca in forest soil obtained in the present study is in accordance with the range (0.13 to 0.97%) reported by Siddiqui *et al.*, (2014) from coniferous forests of temperate Himalayan region. Higher exchangeable Ca and Mg in maize fields could be due to the reason that their removal from soil is comparatively lower due to subsistence level cultivation, and also because these are basically the recently (few decades) cultivated virgin lands where very little or no inorganic fertilizers are used. Exchangeable Ca was observed to decrease with depth in all soils except that of paddy, whereas, exchangeable Mg increased from surface to subsurface

layer in all land use types. Depletion of exchangeable Mg from surface soils is also supported by findings of Saikhe *et al.*, (1998).

5.2 Soil organic carbon and its fractions

5.2.1 Total soil organic carbon

Organic carbon of the soils was found in the order wetland > forest > grassland > vegetable > apple > maize > paddy. As a general trend, OC decreased with the increase in the soil depth. The higher capacity of wetland soils to store carbon is due to the slow rate at which decomposition occurs under conditions of poor drainage. Similar observation was made by Gorham (1991) and Hobbie *et al.*, (2000). Higher OC in surface layer of forest land-use followed by grassland and least in agriculture land-use is supported by findings of Pal *et al.*, (2013) and Johnson and Curtis (2001). Higher OC in surface soils of native ecosystems /natural forests have also been recorded by various other workers, for instance., Jamala and Oke (2013); Venkanna *et al.*, (2014); Hamkalo and Bedernicsek (2014) etc. Nisar and Lone (2013) also observed higher OC in forests and pastures of north-eastern Kashmir. This may be attributed to return of higher quantity and quality of plant litter, high altitude, low temperature and slow rate of mineralization. However, contrary to the findings of these workers, the average OC content of forest soils was found to be higher than that of grasslands in the present study. The plausible explanation for this could be that incessant overgrazing, as prevalent in the area, would reduce Primary Productivity from the ecosystem and ultimately the addition of organic matter to soil, notwithstanding, the fact that some organic matter is returned to soil through animal excreta.

Lowest OC, both in surface and subsurface layer was recorded in paddy soils. This is due to long-term cultivation under submerged conditions and application of mineral fertilizers, resulting in degradation of soil quality such as breakdown of stable aggregation and deterioration of SOM (Yang *et al.*, 2005).

5.2.2 Labile organic carbon

LOC in the soils followed the order wetland > forest > grassland > apple > maize > vegetable > paddy which is almost similar to the trend in SOC, except that

LOC of vegetable soil is lesser than that of maize and orchard soil. The higher quantity of LOC/active SOC as well as high OC density in wetland ecosystems have been reported by Wang *et al.*, (2012) and Fissore *et al.*, (2009). Similarity in trend of SOC and LOC could be due the reason that size of the LOC pool was mainly determined by the SOC pool (Rong *et al.*, 2011). Positive correlations among SOC and LOC pools have also been revealed in the findings of McLauchlan and Hobbie (2004).

Differences in LOC under different land use types indicates that vegetation may affect soil C pools by influencing the quality and quantity of detritus supplied, soil physical and chemical properties (Loya *et al.*, 2004). Surface soil layers invariably contained more LOC compared to that in sub-surface soil layers. Similar, finding was reported by Xia *et al.*, (2010). The degree of C and nitrogen limitation to C mineralization increased with soil depth (Fierer *et al.*, 2003). Therefore, the amount of LOC was much higher in the surface soil layer.

As indicated above LOC of forest and grassland soils was observed to be higher than that of arable lands. This is consistent with the results of Bayer (2002) who observed decrease of OC in soils under conventional land preparing/tillage, when compared to forest and also that the no-tillage system resulted in a biologically less oxidative environment, favorable to preservation of the organic matter labile fractions. This also explains the lower quantity of LOC as well as low LOC:TOC ratio in vegetable soils of the study area.

5.2.3 Particulate organic carbon

In the present investigation attempt was made to ascertain the impact of land use type on size of POC fraction in soil for the reason that it is often considered to be more sensitive to management or land-use-induced changes than the TOC. The results indicate that soils under paddy, maize, apple and vegetable cultivation have lower POC levels than that of wetland, grassland and forests. Higher POC or aggregate SOC in no tilled/conservative systems have also been reported by Liu *et al.*, (2014); Figueiredo *et al.*, (2010) and Chan (2001). Presence of large size of POC fraction in wetland is supported by Fissore *et al.*, (2009) who have reported

high quantities of active SOC from mineral wetland soils of China. The trend of higher quantity of POC fractions in pastures compared to croplands was consistent with many other studies (e.g., Davy and Koen, 2013; Chan *et al.*, 2011). Moreover, the smaller size of POC fraction in paddy land, as revealed by the present investigation, could be due to deterioration of soil aggregates by tillage which also increases the rate of organic carbon decomposition.

5.2.4 Microbial biomass carbon

MBC of soils under different land use types followed the same order as that of SOC. There are numerous evidences and it is well known that SOC strongly affects the amount and activity of soil microbial biomass (Diaz-Ravina *et al.*, 1988). MBC values obtained in the present investigation coincide well with the range (219 to 864 $\mu\text{g g}^{-1}$) reported by Sharma *et al.*, (2004) for different land uses, the range (166 to 1539 $\mu\text{g g}^{-1}$) for pasture land reported by Tracy and Frank (1998) and the mean value of 248 $\mu\text{g g}^{-1}$ for agricultural soil by Hu *et al.*, (1997). Higher MBC as well as SOC in forest soils than that of cropland soils, as observed in the present investigation, is supported by results reported by Xiangmin *et al.*, (2014) and Hu *et al.*, (1997). Higher MBC in forest and grassland soil as compared to agriculture soil is also supported by findings of Kara and Bolat (2008). The high values for soil microbial biomass found under native vegetation may be attributed to the relatively high quantity and quality of plant litter supporting a larger microbial biomass and activity due to the availability of above- and below ground carbon sources (Nsabimana *et al.*, 2004).

In the present study MBC represented 1.67 to 3.34 % of SOC in various ecosystems which is within the 0.3 to 7% range recorded by Wardle (1992). In forest ecosystems the MBC comprised 2.92 and 2.38 % of SOC in surface and sub-surface layer, whereas in surface agricultural soils it averaged around 3%. These results are consistent with 1.6–3.6% range for forest A-horizon and 0.3–4% for agricultural topsoil documented by Bundt *et al.*, (2001). Higher MBC and SOC contents in clayey mineral wetland soils as observed in this study could be attributed to the protection of microorganisms against predation as well as to the

protection of SOM against decomposition due to organo–mineral interactions in finer textured soils (Juma, 1993; Von Lutzow *et al.*, 2002)

5.3 Soil organic carbon stocks

SOC stocks as determined in this study were found to be markedly different, with agricultural/horticultural ecosystems on one hand and native ecosystems on the other. Of the land-use types investigated, wetlands, forests, as well as grassland exhibited the highest average SOC stocks in top 30 cm layer (61 to 122 Mg ha⁻¹) in contrast with the cereals viz., paddy, maize soil (37 to 39 Mg ha⁻¹), with apple orchard and vegetable soils representing an intermediate situation (46 Mg ha⁻¹). Singh *et al.*, (2011) estimated the mean SOC stock value as 101.8 Mg ha⁻¹ in 0–30 cm under various land uses in the temperate western Himalaya. Wetlands were found to store highest terrestrial organic C stocks in this study. This could be due to their high plant productivity combined with slow decomposition rates under water-saturated conditions (Reddy and DeLaune 2008). The higher clay content in wetlands could also have favored the greater storage of organic carbon. This has also been noted by Plante *et al.*, (2006) and Dawson and Smith, (2007). Average SOC stocks in wetlands was observed to be higher than upland forests in the study area. This finding is supported by the results of Fissore *et al.*, (2009).

Presence of higher carbon stocks in forests in comparison with paddy and maize, as revealed in this study, is also corroborated by the results of Venkanna *et al.*, (2014) and Dorji *et al.*, (2014). Puget *et al.*, (2005) also reported higher SOC concentration in forests, in contrast to pastures and agricultural lands. Furthermore, higher SOC stock in vegetable soil in comparison with cereals could be attributed to the fact that vegetables are cultivated in small home scale fields with heavy inputs of manure. Higher SOC stock in vegetable soil is supported by Yan *et al.*, (2012) who studied the land use change from cereal fields to vegetable production.

LOC stocks in different soils varies in the order wetland > forest > grassland > apple > maize > paddy > vegetable. The trend is almost the same as LOC content as

stated earlier, except that LOC stock in paddy soil is higher than that of vegetable soil. This is due to the fact that LOC:TOC ratio of vegetable soils is lesser and also the bulk density of paddy soil is higher in contrast to vegetable soil.

The soil POC and MBC stocks showed considerable variation ranging from 7 to 28 Mgha⁻¹ and 0.80 to 1.94 Mgha⁻¹, respectively, indicating the differences in carbon dynamics under different land use systems of the area under study. This is supported by large number of studies that have shown that both POC and MBC are sensitive to changes in land use, tillage, cover cropping and management (Cambardella and Elliott, 1992; Wander and Bidart, 2000; Grandy and Robertson, 2007). POC stocks have been found to be in the exactly in the same order as LOC stocks. However, the MBC stocks followed the trend; wetland > grassland > forest > vegetable > apple > maize > paddy. Thus, MBC stock in vegetable soils is greater than that of apple, maize and paddy. This can be attributed to proliferation of microbes in the vegetable soils, where high inputs of organic manures are applied. As most microorganisms are heterotrophic, their distribution and biological activity often depend on supply of organic matter (Moscatelli *et al.*, 2007; Yang *et al.*, 2010).

5.4 Carbon Management Index

Carbon management index (CMI) was measured on the basis of changes in the total C in the soil and its lability as determined by KMnO₄ oxidation. As stated in the previous part of this study, grassland soil was used as reference. The greatest CMI among the soils was recorded in forest land use (162.5), followed by grassland (100) and decreased in the order of apple (87.5), maize (81.0), paddy (72.8) and vegetable soils (70.7) respectively. The order exhibits the outcome of both the total SOC pool size and the lability (an estimate of turnover rate) that have been taken into account. Higher CMI in forest and grassland soil in contrast with agriculture/horticulture land indicates that cropping has resulted in a marked decline in both labile and non-labile C and hence the CMI has declined. Moreover, CMI is higher in the land use system where residue inputs and lability are higher. Indeed, these findings are supported by Blair *et al.*,(1995). Besides

this, another conclusion that may be drawn from the CMI measurement in this study is that due to low lability index, vegetable soil exhibited lowest value despite higher SOC pool size.

5.5 Enzyme activities of soils under different land use types

5.5.1 Dehydrogenase

Dehydrogenase (DHA) activity in soils displayed a trend, more or less similar to SOC. With wetland, forest and grassland soil exhibiting higher DHA activity than apple, vegetable, maize and paddy soil. This may be attributed to tillage and management practices, and most importantly the higher SOC, as several authors have reported positive correlation between DHA and OM content (Zhao *et al.*, 2010; Yuan & Yue, 2012 and Zhang *et al.*, 2010). Decline in DHA activity with soil depth was noted in this study. It is well known and can explained by the fact that the highest microorganisms abundance is in the surface layer of the soil profile (till to the depth of 30 cm), at the deepest part of the soil the number of microbial cells is limited, and consequently also DHA level display diminishing trend (Yuan & Yue, 2012).

Highest DHA activity in wetland soil as revealed in this investigation could be attributed to the reason that it is influenced by water content and drops with the decrease of soil humidity. The results of Gu *et al.* (2009) who observed higher DHA level in flooded soil rather than in non-flooded conditions support the inference. A significant negative relationships between DHA and redox potential as reported by Wolinska(2010) in various soils, also supports higher values for DHA activity in wetland soil.

Agriculture and horticulture ecosystems in an average were found to have less DHA activity in comparison to forest ecosystems. Similar, result was obtained by Singh *et al.*, (2014).These workers also recorded highest DHA activity in deodar forest soil while the fodder crop and apple plantation showed significantly lower values. These results are consistent with the findings of this study. Moreover, the average DHA activities of paddy, maize and vegetable soils as recorded in this study were not marked dissimilar. Therefore, it may be inferred

that these land use systems, with their associated management practices have had a similar impact/end -result as far as soil DHA activity is concerned.

5.5.2 Acid phosphatase

The lowest acid phosphatase activity ($282.72 \mu\text{g PNP g}^{-1}\text{soil h}^{-1}$) in surface layer of different soils was recorded in maize soils and highest ($503.75 \mu\text{gPNP g}^{-1}\text{soil h}^{-1}$) was related to wetland. Higher activity in wetland soil may be due to presence of adequate OM as well as ample moisture in wetland soils. The positive correlation of acid phosphatase activity with moisture is corroborated by findings of Sardans and Penuelas (2005) who recorded that reduction of 21% of soil moisture reduced acid phosphatase activity by 31 to 40%. In arable lands the activity varied from 282.72 to $305.87 \mu\text{g PNP g}^{-1}\text{soil h}^{-1}$. These values are consistent with the range (152 to $502 \mu\text{g g}^{-1}\text{soil}$) observed by Kheyrodin *et al.*, (2012) in agriculture soils. Forest soils exhibited higher acid phosphatase activity in contrast to grasslands, apple orchards and arable land. This result is supported partly by Singh *et al.*, (2014) as well as by Wang *et al.*, (2015) who observed higher acid phosphatase activity in forest ecosystems as compared with organic farms, apple orchards and conventional crop based systems. This may be attributed to sensitivity of phosphatase activity to perturbations such as tillage, compaction, fertilizer additions and organic amendments (Oprica *et al.*, 2011).

5.5.3 Alkaline phosphatase

Soil alkaline phosphatase activity was always lower at a depth of 15 to 30 cm compared to surface soil and the same observation has been recorded by Kheyrodin *et al.*, (2012). Average values of alkaline phosphatase in cultivated lands were observed to be lesser than the natural ecosystems systems in the study area. Higher alkaline phosphatase activity in forests was also reported by Singh *et al.*, (2014) and Salam *et al.*, (1998). In the present study, however, grassland soils were found to have higher alkaline phosphatase activity than forests. This could be due to higher and more suitable soil pH in grassland soils (Canarutto *et al.*, 1995). Moreover, apple plantation soil showed lower values of alkaline

phosphatase than acid phosphatase. This result is in accord with findings reported by Singh *et al.*, (2014).

5.5.4 Aryl sulphatase

In surface soil layers the average aryl sulphatase activity in native ecosystems varied from 47.14 to 56.21 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$ whereas in sub-surface layers it ranged from 15.83 to 20.62 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$. In agriculture and horticulture lands it varied from 32.77 to 37.19 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$ and 10.63 to 11.63 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$ in surface and sub-surface layers, respectively. A resemblance between OC content and arylsulphatase activity was observed with natural ecosystems on one hand and farmland ecosystems on another, irrespective of other soil physico-chemical properties. Strong correlation between the arylsulphatase activity and organic C of top soil as observed by Elsgaard *et al.*, (2002) supports this inference. Moreover, arylsulphatase resembles MBC trend as for as natural ecosystem soils are concerned which may be explained by the significant positive correlation between them as reported by Moscatelli *et al.*, (2005). Higher arylsulphatase in forests in comparison with apple plantation, maize and paddy, as revealed in the present study is consistent with results obtained by Singh *et al.*, (2014). Moreover, higher activity of this enzyme in the 0-15cm layer as compared with deeper layer as observed in this study is favored by the results obtained by Elsgaard *et al.*, (2002).

5.5.5 Fluorescein diacetate hydrolase

The mean Fluorescein diacetate (FDA) hydrolase activity of the soils ranged from 10.08 to 17.47 $\mu\text{g g}^{-1} \text{ soil h}^{-1}$ in surface layer and from 5.20 to 6.78 $\mu\text{g g}^{-1} \text{ soil h}^{-1}$ in sub-surface layer, with highest and least activity related to forest and maize land use, respectively. Decline in FDA with depth may be due difference in organic matter, since positive correlation between SOC and FDA has been reported by many workers (for example, Schnurer and Rosswall, 1982; Gasper *et al.*, 2001). Higher FDA in native forests in contrast to agriculture land has also been reported by Chaer *et al.*, (2009) and Araujo *et al.*, (2013). Higher FDA in

wetland and forest soils coincides with their higher SOC and higher enzyme activities in comparison with other soils. Contrary to this, vegetable and apple orchard soils have shown higher FDA activity than grassland soil which otherwise has higher SOC. It is speculated that it might be due to addition of diverse organic inputs in the soil. The same inference has been drawn by Nath *et al.*, (2012) in paddy soils.

5.6 Soil quality indices of different ecosystems/land use types

The mean value of all the soil physical, chemical and biological properties across the different ecosystems were evaluated for development of four types of indices viz, unscreened additive index, PCA based index, regression based index and Conceptual framework model based index. The indicators that formed the TDS and subsequently the MDS for indexing have been listed by various workers (For example Diack and Stott, 2001; Arshad and Martin, 2002; Kizilkaya and Dengiz (2010); Pal *et al.*, 2013; Cherubin *et al.*, 2016).

The unscreened additive indices developed through LSF and NLSF (*SQI-un.1* and *SQI-un.2*) revealed the following order for soil quality of different soils: forest(0.99, 0.99) >apple(0.76, 0.71) vegetable(0.73, 0.69) > maize(0.71, 0.67) >paddy(0.68, 0.64). The mean *un-SQI.1* and *un-SQI.2* for native ecosystem(forest) and cropland ecosystem were 0.99, 0.99 and 0.70 and 0.66, respectively. The maximum SQI for forest is due to better physical, chemical and biological properties as observed in this study. Indeed, higher soil quality of forests/uncropped lands in contrast with cropland has been reported in various studies (Su and Zhao, 2003; Ayoubi *et al.*, (2011; Dadhwal *et al.*, 2011; Tesfahunegn, 2014). Soil quality of apple orchard soils was observed to be higher than agriculture soils which is supported by the results obtained by Su and Zhao,(2003); Pal *et al.*, (2013); Adeyolanu *et al.*, (2015) and Lowest SQI value of agriculture soil among different land use types has also been observed by Tiwari *et al.*, (2006)

The PCA based SQI (*SQI-pca*) estimated through LSF and NLSF (*SQI-pca.1* and *SQI-pca.2*) were found to be maximum for forest soil (0.996, 0.995) followed

by apple orchard soils (0.767, 0.697), vegetable soil (0.708, 0.649), maize soil (0.705, 0.643) and least in paddy soil (0.692, 0.640). Thus, PCA based indexing reveals that forest and orchard soil quality is clearly better (as revealed in the *SQI-un*) and the quality of other anthropogenic ecosystem soils is lower and not distinctly different from one another. Least SQI value for paddy soil is supported by findings of Tiwari *et al.*, (2006).

Regression based soil quality indices computed through LSF and NLSF (*SQI-reg.1* and *SQI-reg.2*) were found in the order: forest soil (1.00, 1.00) followed by almost equal values for vegetable soil (0.62, 0.57) and apple orchard soil (0.62, 0.55) and further decreased in the order of maize (0.57, 0.51) and paddy (0.53, 0.48). As stated in the previous part of this study, multiple linear regression was used to develop this index from soil physical, chemical and biological quality attributes (independent variables) and SOC (dependent variable). Maintenance of SOC was hypothesized as the single management goal across all the land uses. Thus, the index has apparently varied as per the trend similar to SOC content in different land uses. Interestingly, in vegetable soil despite the higher SQ index, the carbon management index is lower than that of paddy and maize soil.

The attributes that are degrading the soil quality in paddy and maize soils in contrast to vegetable soil are BD, available N, MBC and DHA. Whereas, in case of apple orchard and vegetable soils lower scores for BD, Av.N, Ex.Mg, LOC, MBC and DHA are responsible for decline in the soil quality in comparison to that of forest soil. Furthermore, the large difference in index values of forest and maize soils reveals the impact of cultivation on forest soils, because in the study area, currently most maize soils, particularly in the forest fringes have been converted from natural forests in the past(few decades). Thus, a considerable degradation of quality has already taken place in maize soils at various study sites. This conclusion is favored by findings of Cho *et al.*, 2004; Xie *et al.*, (2011) and Kalu *et al.*, (2015).

The conceptual framework model based quality indices developed through linear scoring (*SQI-con.1*) for paddy, maize, vegetable, apple and forest soil were

recorded as 0.58, 0.60, 0.68, 0.73 and 1.00, respectively. The same model based quality indices developed through non-linear scoring (*SQI-con.2*) for paddy, maize, vegetable, apple, and forest soil were estimated as 0.56, 0.57, 0.67, 0.68, and 1.00, respectively. *SQI-con* displayed the same soil quality ranking among land uses as *SQI-pca* and *SQI-un*. This indicates that conceptual model based on potential indicators and assigning weights accordingly, was also effective and suitable for detecting the differences between soils of under various land use types/ecosystems. This is consistent with the conclusions drawn by Diack and Stott (2001). Thus, it may be concluded that soil quality of anthropogenic ecosystems in general, has declined in terms of key soil functions like ability to sustain plant growth, ability to cycle nutrients, ability to resist biochemical degradation, filtering and buffering ability and the ability to facilitate water absorption.

To interpret the overall results of different soil quality indices of soils, it may be concluded that the native vegetation (forest) soil had the highest SQI scores (0.99-1.00) suggesting they are functioning at 99 to 100% of their potential capacity. The apple orchard soils are functioning at 55 to 76% of their potential capacity, vegetable soils are functioning at 53 to 73% level, whereas maize and paddy soils are functioning at 57 to 71% and 48 to 68% of their potential capacity, respectively. Moreover, it could be concluded that the indexing procedures involving different MDS strategies were effective for detecting soil quality differences using a reduced number of indicators. Similar, results were reported by Andrews *et al.*, (2002) and Lima *et al.*, (2013), who both concluded that a reduced number of carefully chosen indicators could provide adequate information needed for soil quality assessment and decision making.

The identification of limiting parameters for different soils reveals that low Av.N, Av.P, MBC, and SOC are degrading the soil quality of paddy, whereas, in maize mostly low SOC, MBC and Av.P are the responsible factors. In case of apple orchard soil low SOC, Ex,Ca, Av.P and MBC are mostly affecting the soil

quality and in vegetable soil low Ex.Ca, Ex.Mg, Av.S, Ary-S and DHA are responsible for degrading the soil quality, in contrast to that of the forest.

Evaluation of different indexing procedures, undertaken by estimation of their sensitivity, reveals that regression based indices are most sensitive, followed by conceptual model based indices and PCA based (non-linear scoring) index. The higher sensitivity of regression equation and PCA based indexing procedures is vindicated by the results reported by Masto *et al.*, (2008). The indices computed through non linear scoring function have better sensitivity than their respective linear scoring methods. Thus, the indices developed can be effectively used to assess quality of soils under different land uses, although further validation may be required, particularly in case of regression equation based index as used in this study.

Chapter – 6

SUMMARY AND CONCLUSIONS

Land use change is a main driver of carbon storage in terrestrial ecosystems, particularly under the resource constrained and temperate mountain settings of Kashmir. Just like the other parts of the state of Jammu and Kashmir, the study area which spans over two districts viz., Bandipore and Ganderbal, has witnessed changes in land use, for instance, the conversion of forests/ pastures into cropland or a shift from agriculture to horticulture use. Besides, the loss of wetland area due to draining, filling, leveling and cultivation. The impact of changes in land use and management on soil quality and cycling/storage of carbon cannot be overemphasized, particularly in the context of climate change and sustainability. With advances in climate change mitigation in recent years, emphasis has been put on SOC stocks particularly, as they vary with management, land use and change with forest loss. Thus, this below ground carbon may play a significant role in local and national carbon budgets and we may need more data on carbon stocks in the state in future. The present study has focused on assessment of soil quality and changes in SOC stocks in croplands, besides SOC stocks under forests, pastures and wetlands in the area. The study is based on the premise that any shift from the ecological equilibrium and land use change due to human activities has an impact on soil quality and SOC stocks across the ecosystems in this zone. The study entitled “Impact of Different Land Uses on Soil Quality and Organic Carbon Stocks in Terrestrial Ecosystems of North Eastern Zone of Kashmir” was conducted, primarily with the following objectives:

- To assess the impact of land uses on soil organic carbon stocks
- To study effect of different land uses on soil quality
- To develop soil quality indices.

6.1 Physico-chemical properties of soils under different land use types.

Impact of different land uses on soil attributes were recorded from two depths viz., 0 to 15 and 15 to 30cm from seven different land use types/ecosystems, namely wetlands, forests, grasslands, apple orchards, vegetable fields, as well as maize and paddy fields.

Highest mean value of BD (1.47 Mg m^{-3}) was recorded in sub-surface layer of apple orchards, whereas, the lowest value (0.73 Mg m^{-3}) was recorded in surface layers of wetlands.. On the basis of texture the studied soils were classified as clay loam, silty clay, silty clay loam, silty loam, loam and clay. Majority of the sites in the study area exhibited clay loam texture followed by silty clay loam. The reaction of soils ranged from moderately acidic to neutral. The mean pH of forest soils was however, always observed to be lower than cultivated soils. All the soils in the study area including those of the wetlands have EC values well below 1 dSm^{-1} indicating that there is no salinity hazard.

Varying quantities of available N were observed in different soils. Quantity of available N in wetland and forest ecosystems was recorded to be higher than that of agriculture and horticulture ecosystems. Mean available N content in surface layer decreased in the order as follows: wetland soil (202 mgKg^{-1}) followed by forest soil (178 mg Kg^{-1}), vegetable soil (145 mgKg^{-1}), grassland soil (144 mgKg^{-1}), apple (140 mgKg^{-1}), maize (103 mgKg^{-1}) and paddy soils (98 mgKg^{-1}). Besides, Available N, available P, K and S were also observed to be higher in forest and wetland soils. Exchangeable Ca and Mg in surface soils of native ecosystems were higher than agriculture and horticulture ecosystems. Exchangeable Ca was observed to decrease with depth in all soils except that of paddy, whereas, exchangeable Mg increased from surface to subsurface layer in all land use types.

6.2 Soil organic carbon and its fractions.

TOC in surface layer were found to be invariably higher than the sub-surface layer under all land uses in the study area. The mean values of TOC in surface and sub-surface soil layers under different land uses varied between 1.14 to 5.68 % and 0.88 to 5.03 % respectively. SOC obtained through Walkley and Black

method (WBC) varied similarly as TOC. The greatest mean TOC in surface layer was observed in wetland soil and least in paddy soil, whereas in sub-surface layer, highest and lowest values were recorded in wetland and maize soil respectively. Average soil LOC in the surface soil layers was found to be relatively much higher under wetland use (11202mgKg^{-1}) followed by forests (4407 mgKg^{-1}) and decreased in the order of grasslands (2820 mgKg^{-1}), apple orchards (2366 mgKg^{-1}), maize fields (2209 mgKg^{-1}), vegetable fields (2082 mgKg^{-1}), and paddy fields (2012 mgKg^{-1}) respectively. On an average POC content was found to be higher than mean LOC at both the depths under all the land uses. The greatest mean POC in surface layer was recorded in wetland soil and least in paddy soil. The mean MBC content in the surface soils varied from 276 to 952 mgKg^{-1} . Among the agricultural and horticultural ecosystems the average values of MBC were found to be highest under vegetable land use followed by that of paddy fields in 0-15 cm depth and similar trend was recorded in 15 to 30 cm depth.

6.3 Carbon management Index

The greatest CMI among the soils was recorded in forest land use (162.5), followed by grassland (100) and decreased in the order of apple (87.5), maize (81.0), paddy (72.8) and vegetable soils (70.7) respectively. The grassland soil was used as reference in the calculation of CMI.

6.4 Soil organic carbon stocks

The total SOC stocks varied from 20.41 to 61.97 in surface soils and 17.27 to 60.21 (Mg ha^{-1}) in subsurface soil. Largest mean organic carbon stock was recorded in mineral soils of wetlands followed by forest system, grassland, apple, vegetable, maize, and lowest in case of paddy soils. The WBC stocks varied from 15.20 to 46.52 (Mg ha^{-1}) in surface soils and 12.73 to 42.38 in subsurface soil layers. The LOC stocks under all the land uses were observed to be higher in surface layers than in the sub-surface layers. Largest mean value of LOC stock was recorded in wetlands followed by forest, grassland, apple, maize, paddy and lowest in case of vegetable, soils. The carbon stocks pertaining to particulate fraction varied from 4.11 (vegetable soil) to 14.64 Mgha^{-1} (wetland soil) in

surface layers and 1.81 to 13.24 Mgha⁻¹ in sub-surface soil of the studied land uses. Among the agricultural and horticultural ecosystems the average values of MBC stocks were found to be highest under vegetable land use (0.77 Mgha⁻¹) followed by apple orchards (0.76 Mgha⁻¹), maize (0.68 Mgha⁻¹) and lowest in paddy soils.

6.5 Soil enzyme activities under different land use types.

Soil enzyme activities, which represent microbial activity, have been affected by land use in the study area. Contrary to agro-ecosystem soils, forest and grassland soils exhibited higher DHA activity. The DHA activity varied from 10.40 to 22.28 $\mu\text{g TPF g}^{-1} \text{ soil h}^{-1}$ in surface soils and 5.90--to 17.16 $\mu\text{g TPF g}^{-1} \text{ soil h}^{-1}$ in sub-surface of different soils. The lowest acid phosphatase (Ac-P) activity (282.72 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) in surface layers was recorded in maize soils and highest (503.75 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) was related to wetland. The mean Ac-P activity of wetland soils was generally much higher than other soils in the study area. Average values of alkaline phosphatase (Alk-P) activity in cultivated lands were also observed to be lesser than that of natural ecosystems. Vegetable soils exhibited the least Alk-P activity (285.05 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) and the maximum activity was recorded in wetland soils (500.29 $\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$). Regarding the activity of arylsulphatase (Aryl-S), the mean values in 0-15 cm layer were found to be highest under wetland followed by forest, similar observation was recorded at 15 to 30 cm depth. The least activity of Aryl-S was recorded in vegetable soils. The mean Fluorescein diacetate (FDA) hydrolase activity of the soils ranged from 10.08 to 17.47 $\mu\text{g g}^{-1} \text{ soil h}^{-1}$ in surface layer and from 5.20 to 6.78 $\mu\text{g g}^{-1} \text{ soil h}^{-1}$ in sub-surface layer, with highest and least activity related to forest and maize land use, respectively.

6.6 Soil Quality Assessment

The physical, chemical and biochemical attributes of soils were evaluated for development of four types of indices viz., unscreened additive index, PCA based index, regression equation based index and Conceptual framework model based

index. The unscreened additive indices *SQI-un.1* and *SQI-un.2* revealed the following order for soil quality of different soils: forest(0.99, 0.99) >apple(0.76, 0.71) > vegetable(0.73, 0.69) > maize(0.71, 0.67) >paddy(0.68, 0.64)> The mean *un-SQI.1* and *un-SQI.2* for native ecosystem(forest) and cropland ecosystem were 0.99, 0.99 and 0.70 and 0.66, respectively. The PCA based soil quality indices *SQI-pca.1* and *SQI-pca.2* were also found to be maximum for forest soil (0.99, 0.99) followed by apple orchard soils (0.76, 0.69), vegetable soil (0.70, 0.64), maize soil (0.70, 0.64) and least in paddy soil (0.69, 0.64).

Regression based soil quality indices (*SQI-reg.1* and *SQI-reg.2*) presented a slightly different ranking of soil quality among different land uses than other methods. *SQI-reg.1* and *SQI-reg.2* were found in the order: forest soil (1.00, 1.00) followed by almost equal values for vegetable soil (0.62, 0.57) and apple orchard soil (0.62, 0.55) and further decreased in the order of maize (0.57, 0.51) and paddy (0.53, 0.48).

The conceptual framework model based quality indices developed LSF (*SQI-con.1*) for paddy, maize, vegetable, apple and forest soil were recorded as 0.58, 0.60, 0.68, 0.73 and 1.00, respectively. The same model based quality indices developed through non-linear scoring (*SQI-con.2*) for paddy, maize, vegetable, apple, and forest soil were estimated as 0.56, 0.57, 0.67, 0.68, and 1.00, respectively. Thus the *SQI-con* displayed the same soil quality ranking among land uses as *SQI-pca* and *SQI-un*. This indicates that conceptual model based on potential indicators and assigning weights accordingly, was also effective and suitable for detecting the differences between soils under various land use types/ecosystems. To sum up all the indexing methods used for different soils, it may be concluded that the forest soil had the highest SQI scores, suggesting they are functioning at 99 to 100% of their potential capacity, whereas paddy soils with the lowest SQI scores are functioning at 48 to 68% of their potential capacity.

6.7 Sensitivity of indexing methods

Soil quality indexing methods were evaluated by estimating their sensitivity. The sensitivity of indexing procedures decreased as follows: *SQI-reg.2* (2.05),

SQI-reg.1(1.88), *SQI-con.2*(1.77), *SQI-con.1*(1.69), *SQI-pca.2*(1.55), *SQI-un.2*(1.54), *SQI-un.1*(1.46), and least in case of *SQI-pca.1*(1.43). Thus, the regression equation based index developed through non-linear scoring of indicators is the most sensitive method. Conceptual framework based indices (*SQI-con.1* and *SQI-con.2*) and PCA based(*SQI-pca2*) have also scored well in terms of sensitivity.

Conclusions

- Natural ecosystems were found to accumulate higher SOC than anthropogenic ecosystems. Wetlands have high SOC content whereas, cereal based crop cultivation has lead to decline in level of SOC in this zone.
- Land use has influenced the sizes of various fractions of SOC in agriculture and horticulture land uses. Negative impact of cropping on LOC level is particularly important, since it is a main component in the flux of CO₂ between the terrestrial ecosystems and atmosphere.
- Carbon management index of different soils was found in the order; Forest > grassland >apple orchard>maize>paddy> vegetable soil.
- Higher SOC stocks in native ecosystems in contrast to croplands imply that further intensification and extension of agriculture into native ecosystems will lead to higher loss of stored carbon from soils.
- Marked variations in soil chemical and physical attributes as well as enzyme activities were observed among different land use types. The undisturbed soils clearly exhibited better condition signifying that prolonged agricultural land use has altered the magnitude, diversity, and spatial variability of soil properties.
- Driven by adverse physical, chemical and biochemical changes, the overall SQ of paddy, maize, apple and vegetable fields has depleted. The SQ indices, in general, have revealed the following soil quality ranking among the land use types: forests > apple orchards > vegetables > maize >paddy soil.

- Different indexing methods and scoring combination used in the investigation revealed the differences in soil quality among different land uses. However, non-linear scoring exhibited higher sensitivity in the respective indices.
- Evaluation of different indexing procedures reveals that regression based indices are most sensitive, followed by conceptual model based indices and PCA based (non-linear scoring) index.
- The soil quality indices used in the present investigation can be effectively used for different soils of the zone. However, further validation of the approaches used in indexing procedures is required.

Overall, forest, grassland and wetland ecosystems appear to be most conducive for higher accumulation of SOC, thus helping in maintaining the soil quality. The results substantiate the hypothesis, on which this study was based, that any shift from the ecological equilibrium and land use change due to human activities has an impact on soil quality and SOC stocks across the ecosystems in this zone.

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APPENDIX

Table A.1: 95% Confidence interval and mean bulk density (Mg m^{-3}) of surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	1.19	0.035	1.11	1.27	1.25	0.021	1.20	1.30
Maize	1.26	0.043	1.13	1.37	1.30	0.028	1.21	1.39
Apple	1.38	0.026	1.31	1.43	1.47	0.012	1.44	1.50
Vegetable	0.98	0.036	0.89	1.06	1.08	0.031	1.01	1.15
Forest	0.89	0.049	0.78	1.00	0.97	0.046	0.86	1.08
Grassland	1.15	0.029	1.09	1.22	1.24	0.025	1.18	1.30
Wetland	0.73	0.039	0.63	0.82	0.80	0.018	0.74	0.85

+/- S.E(M) - standard error mean, C.I - confidence interval, LB - lower bound, UB - upper bound.

Table A.2: 95% Confidence interval and mean values of porosity in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	0.55	0.045	0.50	0.57	0.53	0.031	0.48	0.54
Maize	0.53	0.044	0.51	0.58	0.51	0.027	0.47	0.52
Apple	0.48	0.036	0.44	0.51	0.44	0.015	0.41	0.48
Vegetable	0.63	0.046	0.55	0.67	0.59	0.068	0.51	0.61
Forest	0.66	0.059	0.59	0.68	0.63	0.038	0.57	0.67
Grassland	0.56	0.029	0.53	0.60	0.53	0.046	0.51	0.57
Wetland	0.73	0.049	0.69	0.76	0.70	0.036	0.67	0.72

Table A.3: 95% Confidence interval and mean values of void ratios in surface and sub-surface layer of the soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	1.23	0.035	1.19	1.27	1.12	0.041	1.10	1.16
Maize	1.13	0.046	1.10	1.17	1.03	0.032	1.00	1.06
Apple	0.92	0.037	0.91	0.93	0.80	0.045	0.77	0.85
Vegetable	1.71	0.056	1.68	1.73	1.45	0.062	1.41	1.49
Forest	1.77	0.049	1.75	1.81	1.73	0.029	1.71	1.76
Grassland	1.30	0.039	1.28	1.33	1.13	0.048	1.10	1.15
Wetland	2.67	0.051	2.62	2.70	2.33	0.039	2.26	2.39

Table A.4: 95% Confidence interval and mean values of % clay, silt and sand of surface layer of different soils

Land use/ Ecosystem	Clay				Silt				Sand			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB			LB	UB
Paddy	31.6	1.12	28.48	34.73	38.78	4.40	26.38	51.18	29.60	4.12	18.15	41.05
Maize	30.17	0.54	28.67	31.67	46.70	1.15	43.52	49.93	23.16	1.29	19.57	26.74
Apple	33.46	0.94	30.84	36.08	39.44	6.79	20.56	58.31	27.09	6.4	9.23	44.97
Vegetable	24.32	1.11	23.09	30.30	32.77	2.06	27.25	38.71	42.87	3.11	38.17	52.45
Forest	31.44	1.70	26.70	36.18	46.79	6.09	28.49	65.09	21.76	5.23	7.23	36.28
Grassland	32.46	1.71	27.69	37.23	43.09	5.33	28.28	57.90	24.44	4.01	13.28	35.59
Wetland	42.75	0.90	42.27	43.23	41.03	1.61	40.69	41.36	16.21	1.06	16.04	16.37

Table A.5 : 95% Confidence interval and mean values of % clay, silt and sand of sub-surface layer of different soils

Land use/ Ecosystem	Clay				Silt				Sand			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB			LB	UB
Paddy	32.32	0.81	30.06	34.57	37.65	0.81	30.06	34.57	30.02	3.78	19.51	40.54
Maize	31.57	0.93	28.98	34.64	47.12	0.80	44.88	49.35	21.30	1.42	17.35	25.26
Apple	34.45	1.04	31.54	37.36	38.99	6.17	20.47	57.51	26.55	6.24	9.20	43.90
Vegetable	25.46	0.85	23.78	30.55	32.74	1.99	28.72	36.79	41.79	2.79	37.81	52.31
Forest	31.79	1.75	26.90	36.67	46.68	6.68	28.11	65.24	21.52	5.23	7.01	36.05
Grassland	33.52	1.50	29.35	37.70	42.68	5.27	28.02	57.34	23.72	4.09	12.36	35.07
Wetland	43.74	0.86	41.33	46.14	39.00	0.93	36.40	41.61	17.25	0.46	15.95	18.55

Table A.6: 95% Confidence interval and mean pH of surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	6.63	0.048	6.52	6.75	6.91	0.066	6.76	7.06
Maize	7.04	0.078	6.86	7.22	7.19	0.086	6.99	7.39
Apple	7.00	0.030	6.93	7.06	7.25	0.044	7.15	7.35
Vegetable	7.30	0.045	7.20	7.41	7.19	0.066	7.04	7.35
Forest	6.70	0.058	6.57	6.84	6.94	0.057	6.81	7.07
Grassland	7.15	0.054	7.03	7.28	7.26	0.071	7.10	7.42
Wetland	6.43	0.050	6.31	6.54	6.75	0.039	6.66	6.84

Table A.7: 95% Confidence interval and mean values of electrical conductivity (dSm^{-1}) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	0.19	0.012	0.171	0.226	0.12	0.015	0.101	0.187
Maize	0.14	0.005	0.136	0.161	0.13	0.004	0.127	0.149
Apple	0.21	0.008	0.192	0.225	0.17	0.009	0.163	0.225
Vegetable	0.17	0.006	0.164	0.195	0.16	0.017	0.155	0.247
Forest	0.12	0.012	0.118	0.173	0.14	0.012	0.018	0.095
Grassland	0.11	0.015	0.093	0.162	0.09	0.003	0.076	0.102
Wetland	0.60	0.048	0.489	0.700	0.65	0.051	0.55	0.74

Table A.8: 95% Confidence interval and mean values of available soil Nitrogen (mgKg^{-1}) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	98.12	3.246	94.22	105.33	80.50	2.15	77.56	83.23
Maize	103.49	4.566	94.34	110.34	88.22	1.24	85.87	92.34
Apple	140.20	4.053	132.00	154.36	121.64	1.52	118.34	126.45
Vegetable	145.32	3.711	139.45	153.23	122.68	1.365	116.55	125.76
Forest	178.70	3.635	170.31	187.07	173.55	2.23	166.45	178.56
Grassland	143.93	2.058	140.55	147.34	124.49	1.917	121.34	130.34
Wetland	202.57	2.650	197.22	208.54	192.17	2.47	186.74	198.32

Table A.9: 95% Confidence interval and mean values of available Phosphorus (mgKg^{-1}) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	11.01	0.177	10.84	11.91	8.02	0.082	7.83	8.21
Maize	11.83	0.184	10.64	12.12	8.11	0.071	7.94	8.27
Apple	12.89	0.155	12.66	13.92	8.42	0.047	8.31	8.53
Vegetable	11.72	0.183	11.53	11.91	8.01	0.072	7.85	8.18
Forest	17.28	0.216	17.02	18.15	13.26	0.130	12.96	13.56
Grassland	14.16	0.201	11.75	14.91	10.13	0.071	9.02	9.35
Wetland	19.12	0.562	15.75	16.49	12.98	0.101	12.74	13.22

Table A.10: 95% Confidence interval and mean values of available Potassium (mgKg^{-1}) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	149.91	1.614	138.56	154.56	151.93	0.746	148.62	154.67
Maize	143.70	1.129	140.33	146.42	146.68	0.819	141.96	149.55
Apple	178.84	1.811	164.81	186.46	181.29	1.076	175.95	187.68
Vegetable	173.13	1.881	158.32	187.56	150.84	1.00	138.61	153.96
Forest	228.17	2.198	210.66	246.76	232.65	1.198	127.65	241.57
Grassland	213.45	2.230	206.22	224.33	220.35	1.201	214.88	226.56
Wetland	261.81	1.691	255.67	270.34	268.22	1.090	256.54	274.22

Table A.11: 95% Confidence interval and mean values of available Sulphur (mgKg^{-1}) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	10.18	0.116	8.54	11.12	8.78	0.109	7.65	10.90
Maize	11.56	0.161	10.11	13.03.	8.91	0.092	8.10	9.21
Apple	13.26	0.171	11.01	15.24	10.36	0.086	9.54	11.01
Vegetable	9.70	0.189	6.76	10.88	6.76	0.087	6.26	8.66
Forest	13.47	0.163	11.01	14.43	10.74	0.120	8.76	11.31
Grassland	12.84	0.138	11.53	14.25	10.16	0.110	9.17	11.48
Wetland	14.74	0.183	13.67	16.45	11.72	0.141	9.79	12.44

Table A.12: 95% Confidence interval and mean values of exchangeable Calcium (cmol (+)Kg⁻¹) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	8.91	0.033	7.32	9.22	9.75	0.028	8.67	10.98
Maize	10.85	0.028	9.10	11.01	10.76	0.035	9.33	12.22
Apple	8.10	0.039	7.32	9.35	7.85	0.042	6.36	8.88
Vegetable	7.90	0.063	5.26	9.77	7.35	0.033	6.33	8.87
Forest	10.71	0.012	8.55	12.08	6.94	0.014	6.01	7.28
Grassland	11.55	0.038	10.19	13.26	11.31	0.046	9.55	14.01
Wetland	12.40	0.049	10.54	14.05	12.29	0.070	9.30	14.79

Table A.13: 95 % Confidence interval and mean values of exchangeable Magnesium (cmol (+)Kg⁻¹) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	1.60	0.023	1.55	1.65	2.02	0.013	2.00	2.16
Maize	2.01	0.021	2.00	2.41	2.59	0.010	2.51	2.65
Apple	2.36	0.031	2.12	2.76	2.60	0.021	2.50	2.65
Vegetable	1.57	0.027	1.18	2.49	2.38	0.023	1.91	2.76
Forest	2.89	0.020	2.00	3.10	2.93	0.015	2.55	3.22
Grassland	2.49	0.019	2.12	2.92	2.92	0.025	2.77	2.98
Wetland	3.03	0.027	2.43	3.25	3.19	0.026	2.53	3.32

Table A.14: 95% Confidence interval and mean values of % TOC in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	1.14	0.056	1.02	1.26	0.92	0.058	0.79	1.04
Maize	1.18	0.095	0.97	1.04	0.88	0.078	0.71	1.04
Apple	1.22	0.083	1.03	1.39	0.96	0.088	0.77	1.15
Vegetable	1.71	0.110	1.47	1.95	1.31	0.058	1.19	1.44
Forest	2.34	0.121	2.06	2.58	2.02	0.099	1.80	2.23
Grassland	1.77	0.080	1.59	1.94	1.52	0.111	1.27	1.75
Wetland	5.68	0.313	5.00	6.34	5.02	0.26	4.46	5.58

Table A.15: 95% Confidence interval and mean values of % WBC in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	0.85	0.044	0.75	0.94	0.68	0.034	0.60	0.75
Maize	0.89	0.072	0.73	1.04	0.74	0.057	0.67	0.86
Apple	0.94	0.066	0.80	1.08	0.71	0.071	0.56	0.86
Vegetable	1.25	0.082	1.07	1.42	1.13	0.053	1.01	1.24
Forest	1.76	0.091	1.53	1.99	1.57	0.095	1.36	1.77
Grassland	1.33	0.059	1.19	1.45	1.16	0.097	0.94	1.36
Wetland	4.26	0.220	3.78	4.74	3.54	0.187	3.13	3.94

Table A.16: 95% Confidence interval and mean values of LOC (mgKg⁻¹) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	2012	67.46	1866.90	2156.30	1324	36.55	1198.03	1487.04
Maize	2209	144.32	1899.75	2518.86	1006	87.00	824.45	1345.76
Apple	2366	125.60	2096.21	2636.14	1100	75.44	985.75	1301.45
Vegetable	2082	206.04	1640.12	2523.95	1079	87.54	815.97	1340.55
Forest	4407	393.95	3562.12	5252.10	2873	165.45	2534.66	3124.34
Grassland	2820	185.49	2422.13	3217.83	1831	165.32	1434.55	1756.90
Wetland	11202	757.86	9576.22	12827.60	9409	145.56	6547.43	9763.91

Table A.17: 95% Confidence interval and mean values of POC (mgKg⁻¹) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	2495	191.12	2085.06	2904.88	1416	125.22	1224.20	1646.36
Maize	3695	377.76	2884.26	4504.73	1170	227.55	724.32	1575.34
Apple	3112	236.07	2606.01	3618.65	1466	215.40	965.42	1734.25
Vegetable	2793	180.10	2406.12	3179.47	1043	143.22	786.54	1324.36
Forest	5675	41.71	595.54	774.48	3329	31.23	2986.35	3986.12
Grassland	4398	422.00	3493.29	5303.50	2308	279.55	1839.88	2677.41
Wetland	13420	870.01	11554.31	15286.14	11148	674.30	9254.32	12014.25

Table A.18: 95% Confidence interval and mean values of MBC (mgKg⁻¹) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	276.30	11.67	251.22	301.29	172.4	10.22	145.75	197.32
Maize	357.50	32.36	288.06	426.29	177.1	26.45	139.48	225.12
Apple	367.30	33.94	294.45	440.46	216.6	31.00	187.44	227.21
Vegetable	524.90	44.61	429.20	620.56	317.7	27.33	257.31	358.11
Forest	655.01	41.71	575.54	774.48	480.9	36.34	374.95	544.80
Grassland	592.30	34.36	518.61	666.01	423.4	38.21	327.54	547.53
Wetland	952.40	57.21	829.67	1075.10	754.0	47.55	621.47	843.70

Table A.19: 95% Confidence interval and mean values of Dehydrogenase activity ($\mu\text{g TPF g}^{-1} \text{ soil h}^{-1}$) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	10.59	0.087	10.39	10.80	6.36	0.095	6.14	6.58
Maize	10.72	0.090	10.51	10.94	7.24	0.082	7.05	7.43
Apple	13.55	0.142	13.22	13.88	8.07	0.108	7.82	8.32
Vegetable	10.40	0.102	10.10	10.96	5.90	0.079	5.72	6.10
Forest	20.81	0.132	20.51	21.52	12.38	0.109	12.02	14.81
Grassland	17.74	0.139	17.42	18.06	8.16	0.055	6.87	8.53
Wetland	22.28	0.142	20.31	22.96	17.16	0.120	16.88	17.45

Table A.20: 95% Confidence interval and mean values of Acid phosphatase activity ($\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	305.06	3.173	284.97	319.38	136.17	2.724	129.93	139.26
Maize	282.72	3.936	278.26	287.19	135.89	2.963	130.67	138.11
Apple	298.57	2.586	287.22	299.93	157.59	2.861	150.37	162.8
Vegetable	283.94	6.890	264.30	293.59	130.29	3.83	123.76	136.83
Forest	470.24	9.370	448.62	491.87	402.57	11.29	367.29	437.85
Grassland	434.58	3.701	426.01	443.11	236.67	9.67	226.05	261.30
Wetland	503.75	6.803	488.01	519.48	436.43	11.02	399.45	463.35

Table A.21: 95% Confidence interval and mean values of Alkaline phosphatase activity ($\mu\text{g PNP g}^{-1} \text{ soil h}^{-1}$) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	316.67	1.173	313.97	319.38	227.96	1.380	224.76	231.17
Maize	297.85	1.08	293.13	333.35	209.83	1.612	206.12	213.54
Apple	289.74	1.436	286.43	293.05	166.84	2.211	161.70	172.00
Vegetable	285.05	2.880	278.39	291.70	177.60	1.086	172.02	179.18
Forest	425.25	1.869	421.14	429.76	338.70	2.05	333.95	343.45
Grassland	441.90	2.83	435.35	448.35	351.14	3.830	342..30	359.98
Wetland	500.53	2.980	498.65	512.40	426.33	3.57	418.08	434.58

Table A.22: 95% Confidence interval and mean values of Aryl sulphatase activity ($\mu\text{g g}^{-1}\text{soil h}^{-1}$) in surface and sub-surface layer of different soils.

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
LB			UB	LB			UB	
Paddy	35.00	0.144	34.67	35.40	11.63	0.142	11.31	11.96
Maize	33.08	0.118	32.81	33.40	11.16	0.055	11.00	11.35
Apple	37.19	0.121	34.91	38.50	10.82	0.035	10.74	10.90
Vegetable	32.77	0.344	30.42	33.82	10.63	0.128	10.34	10.92
Forest	47.14	0.250	46.57	47.73	15.84	0.094	15.62	16.05
Grassland	49.26	0.326	48.51	50.01	16.57	0.136	16.26	16.89
Wetland	56.21	0.346	55.40	57.01	20.62	0.112	20.36	20.90

Table A.23: 95% Confidence interval and mean values of Fluorescein Diacetate (FDA) hydrolytic activity ($\mu\text{g g}^{-1}\text{soil h}^{-1}$) in surface and sub-surface layer of different soils..

Land use/ Ecosystem	-----Surface-----				-----Sub-surface-----			
	Mean	+/- S.E(M)	95% C.I		Mean	+/- S.E(M)	95% C.I	
			LB	UB			LB	UB
Paddy	12.51	0.090	12.30	12.72	6.65	0.063	6.50	6.82
Maize	10.08	0.090	9.87	10.29	5.20	0.056	5.08	5.34
Apple	14.43	0.107	14.19	14.68	7.06	0.22	6.45	9.69
Vegetable	15.80	0.076	15.62	16.00	8.14	0.037	8.05	8.23
Forest	17.47	0.053	17.34	17.60	7.21	0.069	7.06	7.39
Grassland	13.55	0.142	13.22	13.90	6.26	0.085	6.06	6.47
Wetland	16.55	0.055	16.42	16.68	6.78	0.036	6.66	6.87



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CERTIFICATE

Certified that corrections/amendments as suggested by the external Examiner Dr. Anil Sharma, Senior scientist (Soil Science), Central Institute of Temperate Horticulture, Srinagar, during Viva-Voce examination held on **16-03-2017** have been incorporated in the manuscript entitled “Impact of Different Land Uses on Soil Quality and Organic Carbon Stocks in Terrestrial Ecosystems of North Eastern Zone of Kashmir” submitted by **Shakeel Ahmad Mir (Regd. No. 2013-477-D)**.

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