International Journal of Ecology and Environmental Sciences 35 (2-3): 129-161, 2009

Soil Ecology, Biodiversity and Carbon Management

S.R. GUPTA*, NEELAM AND RAVI KUMAR
Department of Botany, Kurukshetra University, Kurukshetra 136 119, Haryana, India
* Corresponding author; Email: *sgupta2002@yahoo.com*

ABSTRACT

This paper gives an overview of soil ecosystem services, soil organic matter dynamics, models of soil organic matter, biotic and abiotic control of litter decomposition, patterns of soil biodiversity, and potential of soil carbon sequestration to mitigate climate change. Healthy soil provides a wide range of ecosystem goods and services and the value of the ecosystem services provided by soil biodiversity amounts to US $1.5 trillion per year. Soil texture, soil mineralogy and soil organic matter play an important role in the functioning of an ecosystem. The global carbon pools and fluxes, and soil and vegetation carbon pools at biome and regional level have been discussed. The organic matter inputs and soil carbon pools vary in different biomes and ecosystems. Soil organic carbon simulation models are being increasingly used to describe soil carbon dynamics. Climate, soil fauna and litter quality regulate the rates of litter decomposition at local, regional and global scale. Litter quality has emerged as the most important direct regulator of the rates and patterns of litter decomposition at the global scale. There is much interest to analyze the effect of litter diversity, climate change and invasive species on litter decomposition rates. Modern techniques of molecular biology and high resolution microscopy elucidate the structural and functional aspects of soil biodiversity. Sustainable management and conservation of soil biota is important for conserving global biodiversity, as soil communities are species rich and affect ecosystem processes. Conservation agriculture, tree plantations on degraded lands, and agroforestry could enhance carbon storage in the soil-plant system. The integration of science, technology and traditional ecological knowledge can make substantial contribution to the science of soil ecology, ecosystem carbon management, and sustainability.

Key Words: Global Carbon Cycle, Carbon Pools, Litter Decomposition, Litter Quality, Decomposition Models, Soil Fauna, Microbial Diversity, Soil Biodiversity, Soil Carbon Management

INTRODUCTION

Soil is a vital component in the functioning of terrestrial ecosystems provides a habitat for diverse and interacting populations of soil organisms, accounts for decomposition processes, and a critical link in carbon sequestration to mitigate climate change. Soil organic matter contributes greatly to soil quality and plant health, and controls soil biological processes, and ecosystem properties. The management of soil organic matter, soil biodiversity, and the soil ecosystem services are central to the sustainability of both natural and managed ecosystems (Kennedy and Gewin 1997, UNESCO-SCOPE 2007, Palm et al. 2007). The exchange of carbon between the terrestrial ecosystems and the atmosphere is the key driver of the global carbon cycle (IPCC 2001, IPCC 2007, Houghton 2007). An understanding soil organic carbon dynamics is essential for restoring and maintaining soil health, sustained productivity from land, and to formulate soil carbon management strategies in present scenario of climate change (Pal et al. 2009).

Decomposition of organic matter plays a key role in global carbon cycle and nutrient cycling. During the last 80 years, much progress has been made in the field of ecology of litter decomposition at local, regional and global scale. There is now a fairly good understanding of the abiotic and biotic controls of the decomposition rates at local, regional and global scales (Singh and Gupta 1977, Swift et al. 1979, Heal et al. 1997, Zhang et al. 2008), the role of decomposition processes in soil nutrient cycling (Heal et al. 1997) and the role of the microbial enzymes in degradation of complex organic substrates (Deobald and Crawford 1997). Decomposition supports the diversity of microbial populations, and regulates the release of greenhouse gases to the atmosphere, and soil carbon sequestration at biosphere level (Berg and McClaugherty 2008).

The soils in terrestrial ecosystems support millions of genetically distinct prokaryote organisms (bacteria, archaea), and eukaryotic species across many taxonomic groups. Soil microbial communities occur at a broad range of scales from soil macroaggregates to plant rhizosphere, to field plot and to the ecosystem and global scale (Tiedje 1995). The functioning of terrestrial ecosystem depends on soil biodiversity as many of plant interactions take place below-ground. Soil biota play a key role in various ecosystem functions such as decomposition of organic matter, nutrient cycling, soil respiration, and formation and stabilization of soil structure (Swift et al. 1979, Brussaard et al. 1997, Coleman 2008). Climate change, the ecosystem degradation, land-use and land cover changes and soil pollution are posing a big threat to the diversity of organisms in soil.

The emissions of greenhouse gases from the combustion of fossil fuels and net emissions from land use change have contributed mainly to anthropogenic CO_2 fluxes (IPCC 2007, Canadell et al. 2007, Houghton 2007). The carbon balance of terrestrial ecosystems can be changed significantly by the direct impact of human activities by increasing the concentration of greenhouse gases in the atmosphere (IPCC 2001, IPCC 2007). A positive feedback of ecosystem carbon to climate change might occur at greater speed and with greater intensity as predicted in the carbon-cycle-climate models (Heimann and Reichstein 2008). Therefore, it is vital to manage carbon in ecological systems to formulate climate mitigation strategies for stabilizing atmospheric greenhouse gases (Trumper et al. 2009). Forest regeneration, tree plantations on degraded lands and agroforestry could enhance carbon storage in the soil-plant system. Agricultural soils can provide low-cost carbon sequestration through conservation tillage, crop diversification, organic farming, bioenergy crops, and crop residue return to soil (Smith et al. 2008a).

This paper gives an overview of soil ecology research with a focus on soil properties and ecosystem services, soil organic matter dynamics, models of soil organic matter, biotic and abiotic control of litter decomposition, patterns of belowground biodiversity, and soil carbon management in natural and managed ecosystems.

SOIL ECOSYSTEM SERVICES

Healthy soil provides a wide range of ecosystem goods and services that play a crucial role in sustaining biological diversity of planet earth. The soil functions and processes that benefit human society are referred to as ecosystem services. Achieving many of the Millennium Development Goals depends directly or indirectly on the ecosystem services of the soil (MA 2005). The soil functions such as decomposition of organic materials, soil nutrient cycling, and detoxification of soil contaminants, plant productivity and regulation of plant-soil water relationships that benefit humankind are some of the important soil ecosystem services (UNESCO and SCOPE 2007, Palm et al. 2007). Soil biodiversity is responsible for supplying the environment with a number of critically important ecosystem goods and services (Pimentel et al. 1997). The maintenance of fertile soil is one of the most vital ecological services the biota performs (Wall 2004, Coleman 2008).

Soils deliver provisioning, regulating, cultural and supporting ecosystem services, and are regulated by the physical, chemical and biological properties of the soil (Palm et al. 2007). There has been large increase in provisioning services of the soils due to large increase in food crops and livestock, the production of timber, and the production of fuel woods (MA 2005). The large increases in production from the land systems has caused degradation of soils and impacted the regulatory and supporting services of soils (MA 2005). The ability of soils to deliver the ecosystem services directly depends on soils regulatory services of filtering and detoxifying water, soil biodiversity, decomposition of organic materials, regulation of fluxes of greenhouse gases to and from the atmosphere, and plant-soil nutrient cycles (Palm et al. 2007). The relationship between provisioning ecosystem services, soil processes and soil properties are shown in Table 1. The processes mediated by the soil biota such as, waste recycling, soil formation, nitrogen fixation, bioremediation of chemicals, biotechnology, biocontrol of pests, and pollination by organisms having edaphic phase in their life cycle, provide the ecosystem services.

The economic value of biodiversity, including that of soil biodiversity has been estimated by Pimentel et al. (1997). The annual value of ecosystem services provided by soil biodiversity has been estimated to be US \$1.5 trillion (10^{12}) per year (Pimentel et al. 1997), which is approximately 2.5% of the combined global maximum Gross Domestic Product of US \$ 54 trillion

Table 1. Relationship between provisioning ecosystem services, soil processes and soil properties (from Palm et al. 2007).

Provisioning services	Soil processes	Soil property
Physical support for plants	Soil formation	Depth; State factors of soil formation, clay mineralogy
Provision of nutrients	Mineral weathering	Primary minerals
Provision of nutrients	Litter decomposition	Soil biota and soil Texture
Provision of nutrients	Soil organic matter mineralization	Soil organic matter quantity and quality ; Soil Texture
Provision of water	Infiltration	Macroporosity- aggregation, texture, soil organic matter, soil biota
Provision of water	Storage of water in soil	Aggregation, bulk density, depth ;texture, mineralogy, soil organic matter

per year (Constanza et al. 1997). Therefore, it is important to maintain the production systems as ecosystem service provider systems for food, timber, energy, biogeochemical regulation, and biodiversity and soil conservation (Porter et al. 2007). The ecosystem services can provide a framework for analyzing the differences in specific ecosystem services among soil types, and interconnections between the ecosystem services and key soil processes (Palm et al. 2007). Participatory approach could be practicable to integrate traditional knowledge of farmers with modern scientific advances so as to maintain ecosystem services, and soil health in agricultural systems (Kibblewhite et al. 2008).

Soil Properties

The soil consists of dynamic ecological systems characterized by diverse and interacting populations of soil biota and microorganisms, and its biophysical environment. The key soil properties are determined by the soil development processes, and the state factors of soil formation, i.e., climate, organisms, topography, parent material and time (Jenny 1941). Soils in different biomes of the world differ markedly in their colour, clay content, organic matter and depth. There are the 12 soil orders of soil taxonomy, originally based on the United States System of Soil Taxonomy, and now the reclassified FAO-UNESCO Digital Soil map (Palm et al. 2007). The extent of distribution of the twelve orders of soils along with some soil properties have been described in detail by Palm et al. (2007). The geographic distribution of the soil orders varies in tropical, temperate and boreal regions and among the major world biomes. Alfisols cover about 10.6% total world area in flooded grassland and savannas, temperate broad-leaved mixed forests; dry tropical/ subtropical forests; tropical/subtropical coniferous forests; Mediterranean biome. Aridisols characterized by low organic matter are widely distributed in desert biomes, many saline and alkali soils of non desert regions. Entisols, the most extensive and young soils, are found in desert biomes, tropical savanna and Mediterranean biomes. Oxisols, representing about 8% of total geographical area, cover large areas in tropical rainforest biomes, tropical/ subtropical broad-leaved moist forests, and tropical/ subtropical savanna.

Soil texture, soil mineralogy and soil organic matter play an important role in the functioning of an ecosystem. In addition to the importance of soil organic matter in the global carbon cycle, the soil organic matter determines a range of physical, chemical and biological properties of the soil. Clay mineralogy controls soil structure, porosity and stability through formation of micro-aggregates (Tisdall and Oades 1982). Physical fractionation techniques have also been used to separate soil organic matter pools into primary particles (sand, silt and clay), micro-aggregates (53-250 μm) and macro-aggregates (>250 μm). Primary soil particles (sand, silt and clay) are associated with organic matter to form micro-aggregates (<250 μm) and macro-aggregates (>250 μm) in the soil (Tisdall and Oades 1982).

Soil texture directly controls many other soil properties and has been considered an indicator of many soil processes (Parton et al. 1987). Soil minerals, both primary and secondary, are known to determine soil fertility and exert a stabilizing effect on soil organic matter. Soil mineralogy varies spatially as a function of climate and parent material and as a function of soil development over a period of time (Torn et al. 1997). Soil sink capacity for organic carbon depends on clay content and mineralogy, structural stability, moisture and temperature regimes and formation of soil aggregates (Lal 2004). There is growing interest in research using pedotransfer functions for estimating difficult-to-measure soil parameters (McBratney et al. 2003).

THE GLOBAL CARBON CYCLE

Carbon Pools and Fluxes

Terrestrial ecosystems contain carbon in the form of plants, animals, soils and microorganisms (bacteria and fungi). Plant biomass and soil organic matter constitute the major pool of carbon in terrestrial ecosystems. About 610 Pg C is stored in biotic pool in vegetation at any given time (Figure 1). The atmosphere contains approximately 750 Pg C (1 petagram = 1 gigaton= 10^{15} grams) of CO_2. The total amount of carbon in the world's soil organic matter is estimated to be 1500 to 1580 Pg C (Batjes 1996, Schlesinger 1991, Amundson 2001, Lal 2004, NASA 2008). Most of the carbon in soils enters in the form of litter from aboveground and belowground parts, which are broken down by microorganisms during the process of decomposition. The terrestrial plants remove 121.3 Pg C year^{-1} in gross primary production from the atmosphere. Approximately 60 Pg C year^{-1} is returned to the atmosphere from the land systems in autotrophic respiration. All organisms consuming plant material respire (i.e., terrestrial heterotrophic respiration) and return about 60 Pg C yr^{-1} to the atmosphere globally (Schlesinger 1991). The soil organic carbon (SOC) pool of 1580 Pg C is large compared to the annual fluxes of C of 121.3 Pg C to and from the terrestrial biosphere (Figure 1).

Soil Carbon Storage

Plant carbon and soil carbon in different biomes of the world indicate that tropical forests contain about 50% carbon stored in global vegetation (Sabine et al. 2004), Table 2. Forests are important as a major carbon pool as trees have more storage of carbon per unit area as compared to other types of vegetation (Houghton 2007).Tropical forests cover 7 to 10% of the global land and store 40 to 50% of carbon in terrestrial vegetation (Houghton 2005). A large amount of carbon remains stored in the frozen layers of soil (permafrost) in high latitudes including boreal forests and arctic tundra, and in tropical peat lands (Sabine et al. 2004).

The amount of organic matter in the soil is regulated by the vegetation type, net primary production, prevailing temperature and precipitation, current management practices, nature of parent material and

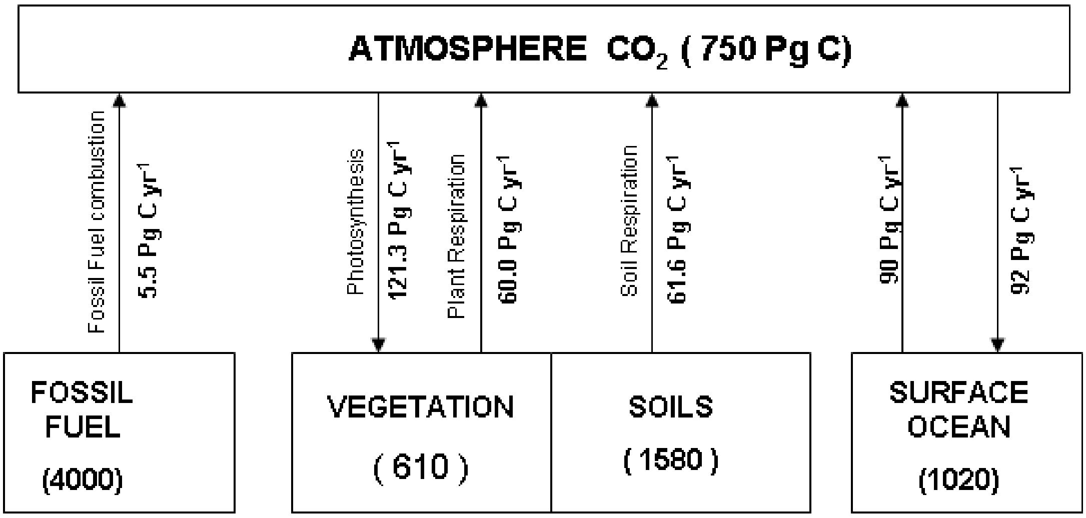


Figure1 . Schematic representation of carbon cycle showing major pools and fluxes of carbon; the values are in Pg C (1 petagram = 10^{15} grams). (based on data from Amundson 2001, IPCC 2001, NASA 2008).

Table 2. Net primary productivity (Pg C yr^{-1}), and carbon (Pg) in vegetation and soils of different biomes of the world and cropland ecosystems (from Sabine et al. 2004)

Biome /Ecosystem	NPP	Vegetation C	Soil C
Tropical forests	20.1	340	692
Temperate forests	7.4	139	262
Boreal forests	2.4	57	150
Arctic tundra	0.5	2	144
Mediterranean shrublands	1.3	17	124
Cropland	3.8	4	248
Tropical savannas and grasslands	13.7	79	345
Temperate grasslands	5.1	6	172
Deserts	3.2	10	208
TOTAL	**57.5**	**652**	**2344**

decomposition of organic matter. In various climatic regions, vegetation composition, climate and soil conditions have been found to effect soil carbon pool in the major biomes of the world (Sabine et al. 2004, Davidson and Janssens 2006). Tropical deforestation has contributed about 20% of total anthropogenic CO_2 emissions to the atmosphere (Houghton 2007). However, unaccounted selective logging in Amazonian rain forest ecosystems could give higher deforestation rates, and CO_2 emissions to the atmosphere (Asner et al. 2005).

Bhattacharyya et al. (2008) have analyzed soil carbon storage capacity of Indian soils by using spatial extent of agroclimatic zones, bioclimatic systems, and agroeco-subregions maps for prioritizing areas for carbon capture and storage. Soil organic carbon in major agro climatic zones of India are given in Table 3. The plateau and hills regions occupy about 45% of the total geographical area of the country are estimated to contain 38% of total organic carbon (Table 3). The western dry bioclimatic region, representing nine districts in Rajasthan, is characterized by poor vegetation growth and a very low organic carbon stock (1% of total). The Himalaya covers nearly 19% of the total geographical area of India, and contributes 33% of soil organic carbon reserves. The Indo-Gangetic plains contain about 9% of soil organic carbon stocks (Bhattacharyya et al. 2008). The soil inorganic carbon represents a large proportion of total soil carbon in Indian soils at different soil depths, the stock of inorganic carbon has been found to increase with soil depth (Table 4). There is also need to develop appropriate correction factor to the soil organic carbon estimates based on the Walkley and Black method to improve the accuracy of organic carbon stocks in Indian soils (Gopal Krishan et al. 2009).

Table 3. Soil organic carbon (Pg) at two soil depths in major agroclimatic zones of India (adapted from Bhattacharyya et al. 2008)

Agroclimatic zones	Soil Organic Carbon (0-30 cm)	Soil Organic Carbon (0-100 cm)
Himalaya	3.118	7.766
Indo-Gangetic Plains	0.893	2.308
Plateau and Hills	3.523	8.908
Coastal Plains	1.347	3.555
Gujarat	0.411	1.002
Western Dry	0.142	0.318
Island	0.121	0.183
Total	9.555	24.040

Table 4. Total carbon stocks (Pg C) in all agroclimatic zones of India at different soil depths (from: Bhattacharyya et al. 2008)

Soil Carbon Stock	Soil Depth (cm) 0-30	0-50	0-100	0-150
Soil organic C	9.550	15.074	24.040	29.920
Soil inorganic C	4.140	7.036	22.461	33.983
Total C	13.690	22.110	46.501	63.903

The Indo-Gangetic plains is among the most extensive fluvial plains of the world covering an area of 43.7 Mha in states of the northern, central and eastern parts of India, accounts for 13 % of the total geographical area of India, producing about 50% of the total food grains, and providing food to 40% of the population of the country (Pal et al. 2009). However, the soils of the Indo-Gangetic plain are poor in the organic carbon as compared to other parts of India, and tropical regions in general (Bhattacharyya et al. 2007, Pal et al. 2009).

The historical development of soils and the changes in the levels of organic carbon and agriculture have been analyzed by Pal et al. (2009). The nature and properties of the alluvium in the Indo-Gangetic plains vary in texture from sandy to clays, calcareous to non-calcareous and acidic to alkaline. The soil organic carbon and inorganic carbon stocks of the Indo-Gangetic plains (0-150 cm soil depth) are estimated to be 2.0 and 4.58 Pg C, respectively (Pal et al. 2009). In the upper 30cm soil layer, the soil organic carbon in the Indo-Gangetic plains accounts for 6.45% of total organic carbon stock in India and 0.09% of the world (Pal et al. 2009).

Regional Aspects of Global Carbon Cycle

Rapid climate changes can alter soils from sinks to sources of carbondioxide (Davidson and Janssens 2006). There is now increasing scientific and political interest in regional aspects of global carbon cycle (IPCC 2007, Houghton 2007). The studies on long-term monitoring plots across Amazonian region have shown that old growth forest trees are a sink of 0.62 to 0.023 Mg C ha^{-1} yr^{-1} (Phillips et al. 2008). The carbon balance of terrestrial ecosystems in China during the 1980s and 1990 period shows a net carbon sink ranging from 0.19 to 0.26 Pg C yr^{-1}, based on sample-based biomass and soil carbon inventories combined with remotely sensed vegetation greenness index, ecosystem models and atmospheric inversions of CO_2 concen-trations data (Piao et al. 2009). The regional differences in carbon sink are influenced by over harvest and degradation of forests, regional climate differences, and increase in the extent of forestry plantations (Piao et al. 2009). The terrestrial ecosystems in China have been found to absorb 28-37% of its cumulated fossil fuel carbon emission (Piao et al. 2009).

Smith et al. (2008b) have discussed about the various approaches for analyzing the sectoral carbon budget for better understanding of the global carbon cycle. These workers have emphasized the need for assessing the multi sectoral regional carbon budget for the proper management of timber, wood, food and fiber availability by using C isotope studies, eddy covariance, above and below ground field inventory for biomass, process modelling and experimental manipulations and remote sensing (Smith et al. 2008b).

The forest cover in India is 20.60% of the total geographical area, of which about 1.66% are very dense forest, 10.12% moderately dense and 8.82% open or degraded forests according to the state of forest report 2005 (FSI 2008). The major pools and fluxes in Indian forest based on growing stock volume approach including phytomass, soil, litter and fluxes of carbon due to litter fall and land-use changes have been analyzed by Chhabra and Dadhwal (2004), Ravindranath et al. (1997), and Ravindranath et al. (2008). During 1986, the forest carbon stock including vegetation and soil ranged from 8.58 to 9.57 Gt C (Chhabra and Dadhwal 2004). According to FAO report, the total carbon stock in Indian forests amounts to 10.01 Gt C, the forest soil account for 50% of total soil carbon (FAO 2006). On the basis of Comprehensive Mitigation Analysis Process (COMAP) model, Ravindranath et al. (2008) have shown the dominance of soil carbon in the total forest carbon stock in India. While projecting carbon stocks for the period 2006-2030, Ravindranath et al. (2008) have shown an increase of 11% in the forest carbon stocks for 2030 compared to the values in 2006.

SOIL ORGANIC MATTER

Organic matter is at the very foundation of soil ecology and is commonly divided into several pools depending upon resource quality, turnover time and functional pools (Woomer et al. 1994). The carbon fixed by the plants is the primary source of organic matter inputs into the soil both from aboveground and belowground parts of plants. Soil organic matter forms a highly heterogeneous mixture of organic materials in the soil along a continuum from freshly fallen litter to highly decomposed organic materials. There is inter-dependence between organic matter inputs into the soil, the activities of soil organisms, and litter decomposition (Swift and Woomer 1993). The distribution of soil organic matter into functional pools is an effective tool for ecosystem analysis for evaluating changes in climate and ecosystems management (Ardo and Olsson 2003).

Organic Matter Input to Soil

Plant and microbial residues represent the major sources of carbon input into the soil, which ultimately lead to the formation of soil organic matter. A large fraction of the terrestrial aboveground net primary production finds its way to the soil surface in the form of dead leaf, twig, and branch litter. The fine and coarse roots form the belowground litter or detritus also add an appreciable amount of organic matter into the soil (Raich and Nadelhoffer 1989). The total net primary productivity of tropical forests has been estimated to be 49.4 dry matter 10^9 Mg yr^{-1} (Whittaker and Likens 1975). In temperate grassland and forest ecosystems, the world net primary productivity is 24.5 and 5.4 dry matter 10^9 Mg yr^{-1}. The total net primary productivity of desert scrub, extreme desert and agriculture land amounts to 1.6, 0.07 and 9.1 dry matter10^9 Mg yr^{-1}, respectively (Whittaker and Likens 1975). About 50 to 97 % of the net primary production in major land ecosystems finds its way to the soil in the form of detritus. The total amount of detritus in land ecosystems has been reported to vary from 0.1 to 38 dry matter 10^9 Mg yr^{-1} (Schlesinger 1991). Distribution of carbon stocks in litter and fine roots in different biomes, and cultivated systems of the world are given in Table 5. The carbon stocks of litter are found to be greatest in boreal forests (24.0 Pg C).

Table 5. Distribution of carbon stocks (10^9 Mg C) in litter and fine roots in different biomes and crop systems of the world based on classification scheme of Whittaker (adapted from Schlesinger 1991, Jackson et al. 1997)

Biome	Total litter	Total fine root*
Tropical forest	3.6	7.00
Temperate forest	14.5	4.85
Boreal forest	24.0	3.60
Woodland and shrub land	2.4	2.20
Tropical savanna	1.5	7.45
Temperate grassland	1.8	6.85
Tundra and alpine	4.0	3.85
Desert scrub	0.20	2.45
Extreme desert, rock, and ice	0.02	-
Cultivated crops	0.7	1.05
Swamp and marsh	2.5	-
Total	55.22	39.1

*Total fine root biomass from Jackson et al. (1997) ;represented in terms of carbon assuming that dry matter has 50% carbon.

The mean annual litterfall from the vegetation shows a latitudinal gradient, the values increasing from boreal forests to the tropics (Vogt et al. 1986). On the basis of study from some 319 forest sites, Lonsdale (1988) with the help of the statistical model showed that the total litter fall decreases with increase in altitude, whereas leaf fall remained unaffected.

There are several studies on litterfall dynamics in forest ecosystems of India. The patterns of litter fall have been found to vary in relation to vegetation composition, climate and climatic seasonality in different forest ecosystems (Singh 1989, Kumar 2005). The litterfall in the forest of Western Ghats has been reported in the range of 8.5 to 15.4 Mg ha^{-1} yr^{-1} (Kumar and Deepu 1992).In the temperate Himalayan forests, litterfall ranges from 3.2 to 9.6 Mg ha^{-1} yr^{-1} (Singh and Singh 1987, Toky and Ramakrishnan 1981). In general, 70% material produced as litter in the temperate forests is leaf fall whereas, in tropical forests leaf fall accounts for 90% of the total litterfall (Kumar and Deepu 1992). In deciduous forests ecosystems, most of the litter fall is confined to a few months of the year.

The fine roots (1-2 mm diameter) constitute dynamic and active components of the root system, and their fast turnover plays a key role in soil carbon dynamics. Fine roots contribute a majority of belowground primary production with life expectancy ranging from a few weeks to years (Jackson et al. 1996, Jackson et al. 1997). Fine roots production forms a large proportion of total net production, and account for about 33% of the global annual net primary productivity (Gill and Jackson 2000).

Estimates of root litter input to soil have been found to be highly variable in different biomes as regulated by vegetation composition and climate. For estimating root biomass, there are a number of root excavation studies in natural and cultivated ecosystems (Jackson et al. 1996, Jackson et al. 1997). Tracer techniques provided a powerful tool to study functional rooting zones in some temperate grassland ecosystems (Singh and Coleman 1973). Rhizotrons, i.e., direct viewing roots through underground windows and tubes have been used for estimating input of root litter to the soil (Burke and Raynal 1994). The belowground litter input is reported in the range of 100 g m^{-2} yr^{-1} in northern hardwood forests to 1262 g m^{-2} yr^{-1} in a pacific silver fir forest (Vogt et al. 1996).

The carbon stocks in fine roots for different biomes of the world are given in Table 5.The total carbon stock in fine roots of the worlds ecosystems is 39.1 Pg C. A number of workers have estimated, biomass carbon, belowground production and turnover rates of roots in natural ecosystems of the world (see Jackson et al. 1996, 1997). Based on the database of 253 field studies, the standing root biomass has been reported to vary from 200 to 5000 g m^{-2} in different biomes of the world (Jackson et al. 1996). Root biomass in cropland, desert, tundra and the grassland systems was below 1500 g m^{-2}. In tundra, 94% of fine roots are located in the surface layer of soil (0-30cm). In forest biomes, tropical forests and savannas and deserts, 42 to 63% of fine roots are found in upper 30 cm of soil, whereas in boreal forests and temperate grasslands, 83% of roots are found in the surface layer of soil (Jackson et al. 1996, Jackson et al. 1997). Global warming is predicted to have profound effect on permafrost depth, rooting patterns and net carbon fluxes in tundra (Chapin et al. 1992). Deforestation in tropical rainforests will impact root biomass and soil carbon sequestration (Jackson et al. 1997).

The fine roots contributed 20 to 70% of total organic matter input in different forest ecosystems in India. In a dry deciduous forest, fine roots accounted for 40 to 48% of total organic matter in the soil-plant system (Singh and Singh 1991, Aggarwal 1997). In a tropical dry deciduous forest ecosystem, average fine root biomass at 0 to 15 cm and 45 to 60 cm soil depth was 2237 and 539 kg ha^{-1}, respectively (Aggarwal 1997). The maximum underground biomass in grassland ecosystems of India has been reported to range from 51 to 2368 g m^{-2} (Singh and Gupta 1992). In agriculture and agroforestry system, a majority of fine roots remain concentrated in the top 30 cm soil depth (Bhardwaj and Gupta 1993, Neelam 2006, Saini 2008). In the 6 to 7 year old agroforestry system, the fine root biomass varied from 2491 to 3832 kg ha^{-1} up to 60 cm soil depth (Saini 2008).

Woody litter, comprised of tree stems, stumps, branches, twigs, and roots (greater than 2mm), plays an important role in forest ecosystems (Harmon et al 1986). The amount of woody litter in forest ecosystems varies from 1.0 Mg ha^{-1}in dry tropical forests to 500 Mg ha^{-1} in old growth coniferous forests (Agee and Huff 1987). In most of ecosystems, woody litter has been reported to vary from 5 to 50 Mg ha^{-1} (see Berg and McClaugherty 2008). The woody litter in general has low nitrogen (0.30%) as compared to leaf litter (2.28%) and fine roots (2.0%) (Swift 1977, Fahey et al.1988).

Effect of Land Use on Soil Organic Carbon

The loss of soil organic matter due to conversion of natural ecosystems to permanent agriculture has been intensively studied in temperate ecosystems (Paul et al. 1997, Matson et al. 1997). Soil organic matter losses in temperate zone agriculture, are most rapid during the initial 24 years of cultivation, generally with loss of 50% original carbon (Paul et al. 1997). In tropical regions, the clearing of natural vegetation and intensive cultivation have caused loss of soil organic matter (Jenny 1980, Srivastava and Singh 1989).

The depletion of soil carbon, as high as 60-70%, has been reported in many Indian soils due to cultivation (Jenny and Raychaudhuri 1960). Based on Jenny and Raychaudhuri's (1960) data for cultivated and uncultivated soil C in India , the relative loss of SOC (% of original) caused by cultivation in the upper 20cm of Indian soils has been found to increase with decreasing mean annual temperature and to increase with increasing amounts of natural soil organic carbon (Amundson 2001, Davidson and Janssens 2006).

In tropical soils, losses of soil carbon are rapid even after 5 years of cultivation (Matson et al. 1997). In a seasonally dry tropical region, a marked decrease in soil organic matter and microbial biomass has been found due to the conversion of forest ecosystem into savanna and cropland (Srivastava and Singh 1989). In a dry sub-humid tropical region at Kurukshetra, forest system showed greater carbon pool in plant biomass and the soil as compared to the rice-wheat cropping system (Aggarwal 1997). After 30 years of cultivation, soil carbon in cropland was 50% of the forest soil. Due to introduction of *Populus deltoides* trees along with the cropping system, soil carbon content improved significantly (Bhardwaj and Gupta 1993, Saini 2008).

Soil Respiration and Carbon Balance

Soil respiration is the production of CO_2 by plant roots and organisms living in or on the soil (Raich and Schlesinger 1992, Singh and Gupta 1977). Soil respiration is the sum of root and microbial respiration with root respiration contributing 20-50% of the total CO_2 (Paul and Clark 1996). Soil respiration rates are regulated mainly by soil moisture content and temperature, and vegetation composition (Singh and Gupta 1977). Flux tower measurements, such as Eddy Covariance are being used to analyze CO_2 flux from community to ecosystems on a long-term basis. It has been estimated that fluxes of carbon from the soil to the atmosphere thorough organic matter decomposition and root respiration are about 10 folds greater than from fossil fuel and deforestation sources combined (Schimel et al. 2006). The studies on soil respiration are important to understand the ecosystem processes of carbon dynamics, energy flow and mineralization rates (Raich and Schlesinger 1992, Singh and Gupta 1977, Zhang et al. 2005).

Some studies on soil respiration are reported from a tropical grassland ecosystem (Gupta and Singh

1981a) and tropical forest, sub-tropical forest ecosystems (Rajvanshi and Gupta 1986), Rout and Gupta 1989) and tree plantations in a semi arid region (Saraswathi et al 2008). In the grassland at Kurukshetra, soil respiration rates were highest in rainy season, moderate in summer, and least during winter (Gupta and Singh 1981a). The soil CO_2 flux originates from root respiration plus microbial respiration derived from rhizo-deposition, microbial respiration from above ground and belowground litter. The contribution of root respiration was 42% to total carbon dioxide evolution from the soil respiration (Gupta and Singh 1981a). In the tropical *Dalbergia sissoo* dominated forest ecosystem at Kurukshetra in northern India, the soil respiration rates varied from 90 to 1120 $mgCO_2\ m^{-2}\ h^{-1}$(Rajvanshi and Gupta 1986). In the tree plantations of a semi arid region of Madurai, the variations in soil moisture caused seasonal variations in soil respiration rates (Saraswathi et al 2008).

Models of Soil Organic Matter

Soil organic carbon simulation models have been used to predict the effect of management practices and climate change on the fluxes and stocks of soil organic carbon. In the models, soil organic matter is commonly divided into several pools depending upon resource quality, turnover time and functional pools (Jenkinson et al. 1987, Parton et al. 1987). Different carbon pools existing in the soil have different rates of turnover, ranging from one year to few years to decades or more than 1000 years (stable fraction) as influenced by the biochemical composition of litter ((Jenkinson et al. 1987, Parton et al. 1987, Woomer et al. 1994). These models have been successfully used to simulate changes in total soil organic matter.

The two best-known models of soil carbon dynamics are the CENTURY (Parton et al. 1987 Jenkinson et al. 1987, Jenkinson 1990) and ROTH-C (Jenkinson et al. 1987, Jenkinson 1990). A simplified structure of the ROTH-C and CENTURY models of soil organic matter dynamics have been adapted from Davidson and Janssens (2006), Figure 2 .These models compartmentalize soil carbon into 5–7 conceptual pools, including 2–4 pools of decomposable plant material near the soil surface (litter layer) and three pools of carbon in the mineral soil, with mean residence time ranging from years to millennia. Roth-C only models soil processes, with plant residue carbon as the input.

CENTURY is an ecosystem model that recognizes three carbon pools in the mineral soil, i.e., fast, slow and passive pools (figure 2; Parton et al. 1987). The CENTURY model has been used extensively to simulate the long-term (10–100 yr), response of ecosystems to changes in climate, atmospheric CO_2 levels, and agricultural management practices (Parton and Rasmussen 1994). There is need to develop capability to replace the conceptual pool of soil organic carbon with measurable pools of different soil organic carbon fractions (Baldock 2007).

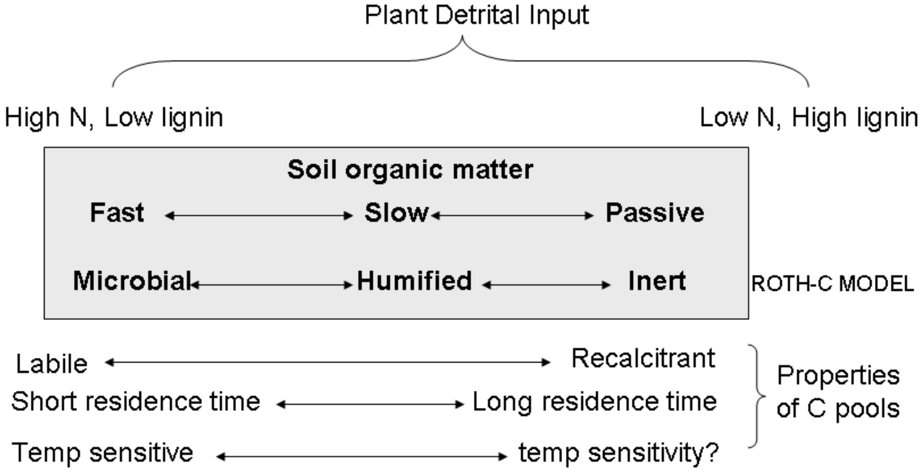


Figure2. The conceptual pools of soil organic matter in mineral soil along a continuum of decomposability and Mean Residence Time (MRT) of soil organic matter in CENTURY model and Rothamsted-Carbon model and the properties of soil carbon pools (adapted from Davidson and Janssens 2006).

The DAYCENT ecosystem model is the daily time-step version of the CENTURY model (Parton et al. 1994). The DAYCENT ecosystem model has been developed to link to atmospheric models and to better estimate trace gas fluxes from different ecosystems by incorporating the entire ecosystem processes represented in CENTURY (Parton et al. 1987). DAYCENT simulates exchanges of carbon, nutrients, and trace gases among the atmosphere, soil, and vegetation. The model is of intermediate complexity and requires site specific simple model input data including climate (daily maximum and minimum temperature and precipitation), soil texture and physical properties, vegetation cover, and land management. Decomposition of dead plant material and SOM are driven by the amount of material and C :N ratios of different pools, as well as water and temperature limitation. The effects of increased CO_2 concentration are also implemented in the DAYCENT model (Parton et al. 1994).

DECOMPOSITION OF LITTER

The pattern, process and population of decomposer organisms show variations at local, regional and global scale in relation to litter quality, biodiversity of soil organisms, soil conditions and climate (Heal et al. 1997). The decomposition process supports diversity in microbial populations by supplying a set of intermediate degradation products, which serve as energy and nutrient sources for different microbial population (Swift et al. 1979, Berg and McClaugherty 2008). Decomposition has a regulatory effect on the diversity and the stability of the ecological community and the soil food webs are based on the decomposition of organic matter.

Singh and Gupta (1977) compiled studies on plant litter decomposition and soil respiration in terrestrial ecosystems. Decomposition has been discussed from the perspective of carbon pools and fluxes in the ecosystems of world by Schlesinger (1977). The importance of soil fauna in litter decomposition, nutrient cycling, formation of soil organic matter and soil structure has been reviewed by several workers (Lavelle et al. 1994, Gupta and Malik 1996, Lavelle 1997). The general relationships between litter decomposition rates, resource quality, abiotic factors and decomposer organisms are well described in terrestrial ecosystems (Singh and Gupta 1977, Swift et al. 1979, Heal et al. 1997, Berg and McClaugherty 2008).

Decomposition Processes

Decomposition is a complex and multi step process of breaking down of complex organic matter by soil organisms to release free the nutrients for renewed uptake by the plants (Swift et al. 1979). During the process of litter decomposition, a large proportion of carbon is lost as respiration of decomposer organisms and nutrients are released during mineralization. Swift et al. (1979) gave the resource cascade model of decomposition and showed the participation of different substrates and soil biota in different phases of decomposition. The process of decomposition comprises of a series of modules coupled by inputs and output of carbon, nutrients and modifiers, and regulated by the decomposer organisms along with the physico-chemical environment (Heal et al. 1997).

The resource quality of litter, the activity of soil microorganisms and soil fauna and the environmental factors collectively determine the rates and completeness of decomposition in different ecosystems (Figure 3). Temperature and moisture are the two important abiotic factors regulating the rate of litter decomposition under natural conditions (see Singh and Gupta 1977, Swift et al. 1979, Berg and McClaugherty 2008). Generally, leaf litter with high nitrogen content is more colonized by both bacteria and fungi and rate of decomposition is high (Melillo et al. 1982). Lignin content exerted a major control over the rates of litter decomposition in forest ecosystems (Aber and Melillo 1982). The diversity and composition of functional group or difference in decomposition rates under identical environmental conditions have been attributed to leaf toughness, nitrogen, lignin, polyphenol concentration, and C :N and lignin : nitrogen ratios.

Rates of Decomposition

Litter and root decomposition have been quantified using the litter -bag technique as introduced by Bocock and Gilbert (1957) in different ecosystems of the world (see Singh and Gupta 1977, Gholz et al. 2000, Silver and Miya 2001, Zhang et al. 2008). Gholz et al. (2000) reported that k values of litter decomposition rates ranged from 0.032 to 3.734 g^{-1} yr^{-1} in different ecosystems from arctic tundra to tropical rain forests. For the root materials, Silver and Miya (2001) reported that k values ranged from 0.03 to 77.0 g^{-1} yr^{-1}. The variations in k values have been attributed to geographic locations, climatic conditions and litter quality. The difference in decomposition rates could

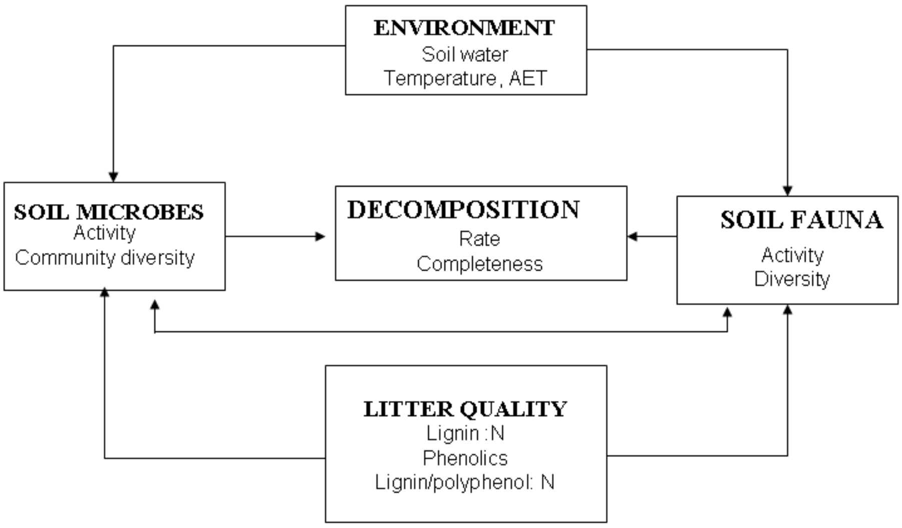


Figure 3. The decomposition of litter as regulated by the physico-chemical environment, litter quality and decomposer organisms (based on Swift et al. 1979)

also be due to variations in soil properties and the composition of microbial communities (Gholz et al. 2000, Zhang et al. 2008). The rates of decomposition have been found to vary with latitude towards both north and south; the k values were generally high in the tropical thorn forest, semi-desert and desert ecosystems in the equatorial regions (Zhang et al. 2008).

The decomposition rates of litter have been studied using the litter bag technique in various types of ecosystems in India (Gupta and Singh 1981b, Arun Lekha et al. 1989, Upadhyay and Singh 1989, Upadhyay et al. 1989, Gupta and Rout 1992). Decomposition of litter in some temperate and subtropical broad leaved forests in India is higher than in the coniferous forests (Upadhyay et al. 1989, Gupta and Rout 1992).

Litter decomposition rates have been studied in grassland, forest and agroforestry systems at Kurukshetra and compiled by Gupta and Malik (1999). The litter bag studies have provided useful information on decomposition rates in relation to climatic factors, litter quality and role of leaching and soil fauna in litter decomposition (Gupta and Malik 1999). The decomposition rates of different litter types in the grassland, forest and agriculture systems varied from 0.159 to 0.329% day^{-1} (Table 6). In an agroforestry system, the decomposition rates of *Populus* leaf litter, wheat straw and sugarcane straw have been reported to vary from 0.388 to 0.492% day^{-1} (Saini 2008).

Table 6. Decomposition rates for various plant materials in agroforestry, grassland and forest systems at Kurukshetra, India.

Ecosystem / plant species	Decomposition rate (percent day^{-1})
Agroforestry system[1]	
Populus deltoides	0.319
Leucaena leucocephala	0.329
Rice straw	0.274
Sorghum straw	0.288
Grassland[2]	
Chenopodium album	0.248
Desmostachya bipinnata	0.221-0.243
Dichanthium annulatum	0.338-0.345
Mixed grass	0.238-0.242
Sesbania bispinosa	0.315-0.356
Dry deciduous forest[3]	
Dalbergia sissoo	0.254-0.238
Acacia nilotica	0.176
Breynia rhamnoides	0.251
Butea monospherma	0.254
Capparis sepiaria	0.257
Carrisa spinarum	0.216
Cordia dichotoma	0.159
Diospyros cordifolia	0.255

[1] Arun Lekha and Gupta (1989); [2] Gupta and Singh (1981); Aggarwal (1997).

The decomposition rates of wheat and rice straw have been studied using the aerobic incubation method (Neelam 2006). The cumulative carbon mineralization rates for the straw and roots of rice and wheat are shown in Figures 4 and 5. During the total incubation period of 140 days, net decomposition rates (CO_2-C evolution from straw and root residue amended soil–control soil/ initial carbon in straw or roots rice and wheat) varied from 67.58% to 78.97%. The decomposition rates of rice straw were lower as compared to roots possibly due to the presence of suberin-lignin like fractions in rice roots. A large amount of surface residues could be respired as CO_2, whereas greater amounts of root-derived carbon may be conserved in the soil during the annual cycle of plant growth in cropping systems (Neelam 2006).

Models of Litter Decomposition

The models to describe the pattern of litter decomposition are given in Table 7. The single exponential model, as proposed by Jenny et al (1949) and Olson (1963), is widely used to describe the pattern of litter decomposition because of its relative simplicity and it provides a good fit for the early stages of litter decay. The single exponential model assumes that the absolute decomposition rate decreases linearly as the amount of the substrate remaining decreases. The exponential model also allows the calculation of half life (0.693/k) or time required to reach 95% loss (3/k).

Aber et al. (1990) suggested that the single exponential model works reasonably well for a variety of litters until only 20% of initial weight is remaining. The single exponential model showed a good fit for the decomposition of litter and roots in the tropical successional grassland (Gupta and Singh 1982). The decomposition of litter and changes in nutrient concentrations in the decomposing litter in a tropical dry deciduous forest ecosystem were best explained by single exponential model; the decomposition constant (day^{-1}) for the leaf litter of seven plant species varied from -0.00245 to -0.0076 (Aggarwal 1997). The pattern of litter decomposition of *Acacia nilotica* and *Dalbergia sissoo* leaf litter in a tropical dry deciduous forest ecosystem is shown in Figure 6.

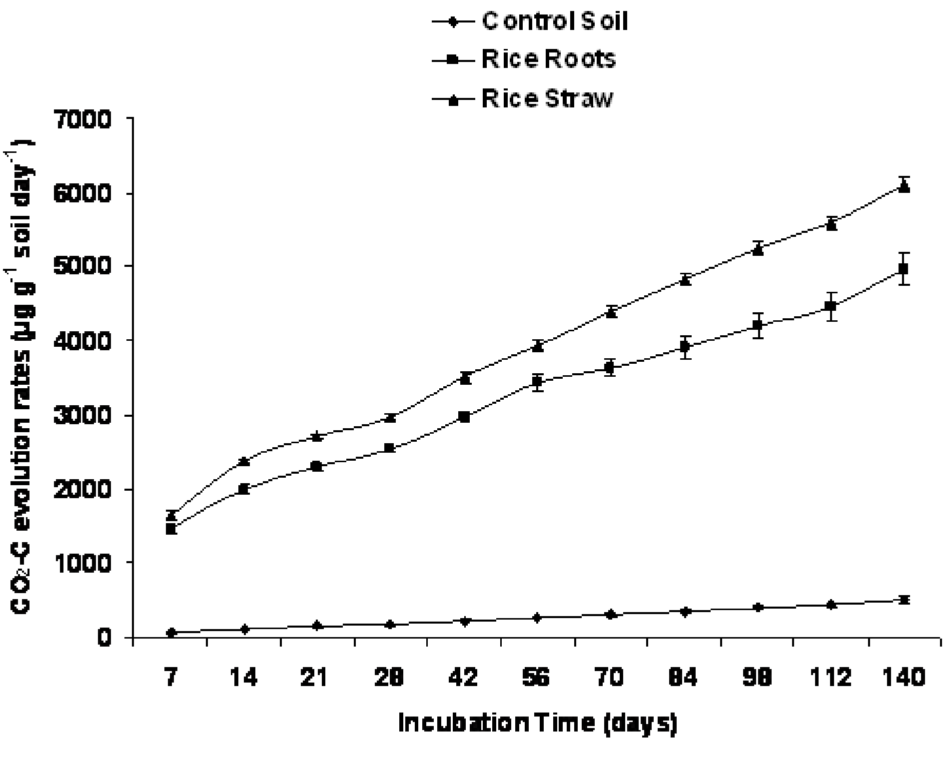


Figure 4 Variation in carbon mineralization rates in unamended and amended soil with rice roots and rice straw during incubation time of 140 days. (Neelam 2006).

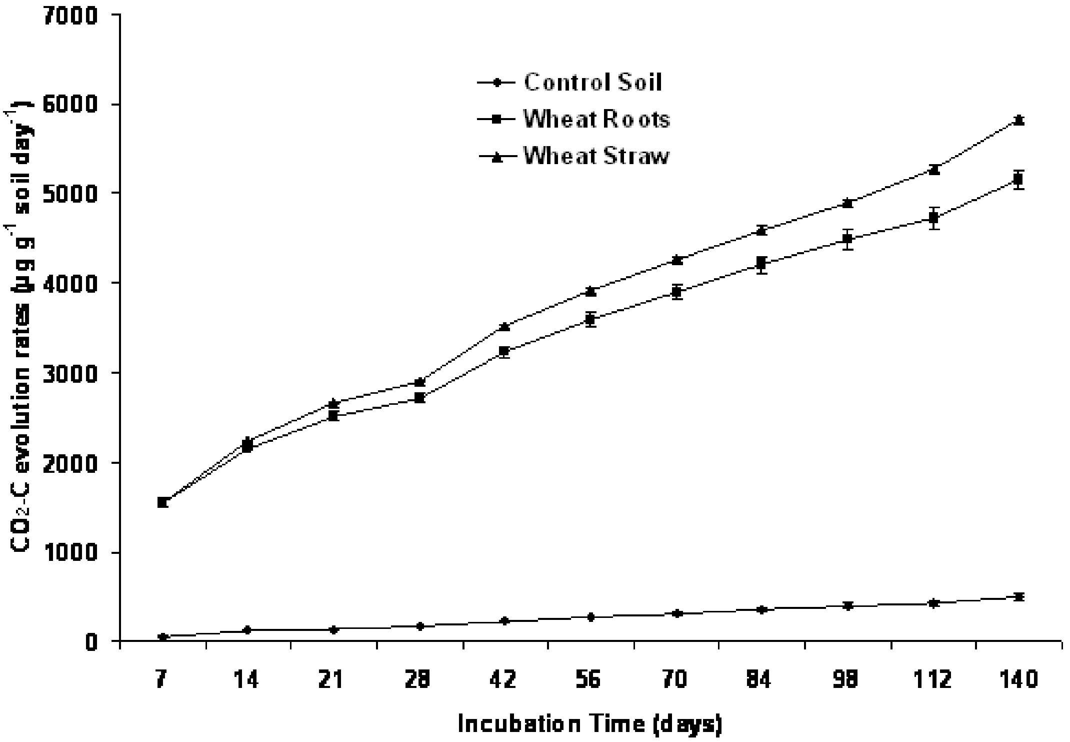


Figure 5. Variation in carbon mineralization rates in unamended and amended soil with wheat roots and wheat straw during incubation time of 140 days. (Neelam 2006).

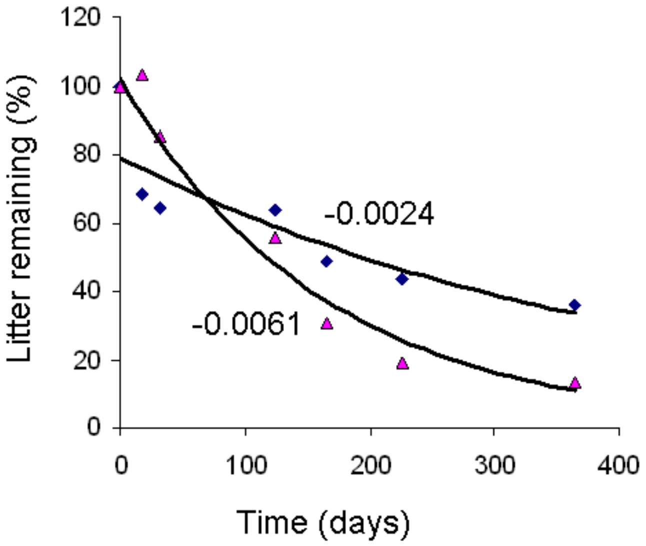


Figure 6. Pattern of weight loss of from decomposing leaf litter of *Acacia nilotica* (♦) and *Dalbergia sisssoo* (△) in a tropical dry deciduous forest (from Aggarwal 1997).

The double exponential model is based on the assumption that the litter has two main substrate quality components. There is a change in litter quality of plant residues with the progress of decomposition (Berg and Staaf 1980). Residue decomposition occurs in two distinct phases. The double exponential model (Weider and Lang 1982) is of the form:

$$M_t = Ae^{-k_1 t} + Be^{k_2 t}$$

where, M is dry weight/nutrient remaining, t is time, and k_1 and k_2 are rate constants for fast and slow decomposing fractions, A and B are the amount of each fraction initially.

The Rothamsted model of decomposition has defined two conceptual litter pools by fitting the model to long-term decomposition data (Jenkinson and Rayner 1977). During the early stages of decomposition, easily decomposable carbohydrates are lost from the litter, whereas lignin has the major control on decomposition during the later stages.

Table 7. Some models to describe the pattern of litter decomposition (from Berg and McClaugherty 2008)

Type of model	Formula	Reference
Single exponential	$M_t = M_0\, e^{-kt}$	Jenny et al. (1949), Olson (1963)
Double exponential	$M_t = Ae^{-k_1 t} + Be^{k_2 t}$	Lousier and Parkinson (1976)
Triple exponential	$M_t = Ae^{-k_1 t} + Be^{k_2 t} + Ce^{k_3 t}$	Couteaux et al. (1998)

Couteaux et al. (1998) used the triple exponential model to describe the decomposition of Scots pine needle litter and estimated the rates of three different components, i.e., labile, metastable and recalcitrant fractions. The triple exponential model is a development of the double exponential model in which the rate constants k_1,k_2 and k_3 are for rapid , slow and extremely slow decomposing fraction of the litter, where as A, B and C give the amount of each fraction, respectively (Table 7).

The Decomposition constant (k) for weight loss of leaf litter in sub-tropical forest ecosystems of Siwaliks in Morni hills in northern India are given in Table 8. The single exponential model showed a good fit for all the leaf litter (R^2= 0.59-0.83). There were significant differences in the pattern of litter decomposition due to litter quality, initial concentrations of lignin and nitrogen explained 86% and 77% of the variability in decomposition rates, respectively (Gupta and Rout 1992).

Table 8. Decomposition constant (k) for weight loss of leaf litter in sub-tropical forest ecosystems of Siwaliks in Morni hills in northern India (from Gupta and Rout 1992)

Plant species	Decomposition constant k (day^{-1})
Anogeissus latifolia	-0.00356
Carrissa spinarum	-0.00218
Grewia oppositifolia	-0.00922
Lannea coromandelica	-0.00760
Mixed-leaf	-0.00361
Rhus parviflora	-0.00186

Litter Quality Effects on Decomposition

Tenney and Waksman (1929) postulated that decomposition rates of the litter are regulated by the chemical composition of the substrate, sufficient supply of nitrogen to decomposer organisms, soil micro-organisms and environmental conditions, especially aeration, moisture supply, pH and temperature. During the last 80 years, much progress has been made in the field of ecology of litter decomposition, particularly the effects of abiotic factors and litter quality on decomposition rates. Mindermann (1968) showed that phenols, waxes and lignin are more resistant than cellulose, hemicelluloses and sugars. In general, the resistance varies in the order: sugars < starch < hemicelluloses, proteins and pectins < cellulose < lignins < suberins < cutins. Because different detritus contain these components in different proportions they decompose at different rates (Mindermann 1968). Nitrogen and lignin were recognized as major variables influencing the rates and pattern of decomposition (Fogel and Cromack 1977, Melilo et al. 1982). The physical, chemical and inhibitory components of litter regulate decomposition rates in different ecosystems (Swift et al. 1979, Cadisch and Giller 1997). The use of gas chromatography, mass spectrometry and stable isotopes have been useful for analyzing the litter quality and understanding organic matter transformations (see Heal et al. 1997, Gupta and Malik 1999).

In the central Himalayan forest ecosystems, initial lignin content was the best indicator of litter decomposition of the leaf litter varying in resource quality (Upadhyay et al. 1989). There is generally an inverse relationship between decomposition rate and lignin content of the litter (Upadhyay et al. 1989). For the climax humid tropical forests in north-east India, litter quality has been shown to influence the pattern of nutrient release from dominant tree species (Kheiwetam and Ramakrishnan 1993). Soil micro-organisms associated with decomposition litter have been studied in several grassland ecosystems under varying climatic conditions (Singh and Gupta 1992). In central Himalayan forest ecosystem, the rates of litter decomposition showed a positive relationship with the beta diversity of fungi (Singh and Singh 1989).

Zhang et al. (2008) have compiled a comprehensive global database of litter decomposition rate (k value) estimated by surface floor litter bags from 110 research sites and have analyzed the direct and indirect effects of latitude and altitude, climatic factors (mean annual temperature and mean annual precipitation) and the various litter quality factors on litter decomposition rates . At the large spatial scale, the *k* values decrease with latitude and lignin content of the litter whereas they increase with temperature, precipitation and nutrient concentrations (Zhang et al. 2008). This global scale analysis of decomposition rates has indicated that litter quality is the most important direct regulator of litter decomposition (Zhang et al. 2008).

Manzoni et al. (2008) have used a data set of about 2800 observations to show global nitrogen release patterns from decomposing litter. They have shown that the patterns of decomposition can be explained by fundamental stoichiometric relationships of decomposer activity, which acts through litter quality controls and the metabolic activity of decomposer organisms. The decomposer organisms across trophic levels lower their carbon-use efficiency to exploit residues with low initial nitrogen concentration (Manzoni et al. 2008).

Litter Diversity, Soil Fauna and Decomposition Rates

The effect of litter diversity on the composition and activities of soil communities and decomposition processes has been the subject of much interest in terrestrial ecosystems (Gartner and Cardon 2004, Hättenschwiler et al. 2005). The plant litter decomposition rates vary in response to diversity manipulation of plant litter (Hätten-schwiler and Gasser 2005). The effect of litter diversity on litter decomposition rates have been compiled for 30 studies (Gartner and Cardon 2004). About 50% of litter species showed synergistic effect on decomposition rates upon mixing that varied from 1 to 65% (Gartner and Cardon 2004). In about 30% of all the cases, there was no significant effect of litter mixtures on decomposition rates. The negative effects of litter mixture have been found to vary from 2 to 22% in 20 percent cases of the leaf litter mixtures (Gartner and Cardon 2004).

In temperate forest trees, the decomposition rates of the most recalcitrant species including *Fagus sylvatica*, *Quercus petraca*, and *Acer campestre*, increased significantly along the diversity gradient (Hättenschwiler and Gasser 2005). There was no diversity effect on rapidly decomposing species of *Carpinus betulus*, *Prurun avium*, and *Tilia platyphyllos* (Hättenschwiler and Gasser 2005). Various mechanisms that might explain litter mixture effects include nutrient transfers, litter types stimulating or inhibitory influence on specific litter components, microclimatic effects and the synergistic or antagonistic effects resulting from interactions among the trophic levels of decomposer organisms (Hättenschwiler and Gasser 2005).

Swift et al. (1979) hypothesized that the relative contribution of soil fauna (vs. microflora) to decomposition was dependent on the climatic region, being greatest at mid-latitudes and decreasing towards the poles. In a multi location study, decomposition of a common grass litter was monitored in animal-suppressed bags and untreated controls exposed at 30 sites distributed across broad climatic regions from 43°S to 68°N in six continents (Wall et al. 2008). The Global Litter Invertebrate Decomposition Experiment (GLIDE) has shown that soil animals significantly influence litter decomposition rates at the regional scale, and has shown the dominating influence of soil arthropods in litter decay over broad climatic regions, i.e., temperate and wet tropics (Wall et al. 2008). This global scale decomposition experiment has validated the conceptual model of Swift et al. (1979) that climate, litter quality and soil biota are the three primary drivers of litter decomposition

Global Change Effects on Decomposition

The decomposition rates have been analyzed also in the context of global climate change (Hobbie 1996, Arp et al. 1997), N deposition (Hobbie and Vitousek 2000), increase in atmospheric CO_2 concentration (Norby et al. 2001), and invasion of exotic species (Ashton et al. 2005). The role of microbial community composition and the community resource history are also important to understand the effect of global environmental change on decomposition (Strickland et al. 2009).

The effects of global warming on decomposition rates can vary by 20% (Hobbie 1996). The effect of nitrogen (Hobbie and Vitousek 2000) and elevated CO_2 (Norby et al. 2001) have been found to vary, generally from 0 to 60%.

Several workers have reported that invasive species exhibit high rates of decomposition than the native species possibly because of their high leaf nitrogen concentration (Vitousek et al. 1987, Kourtev et al. 2002, Ashton et al. 2005). Kourtev et al. (2002) indicated that invasive barberry (*Berberis thunbergii*) and stilt grass (*Microstegium vimivieum*) could alter the soil microbial community. The invasive species influenced the rates of litter decomposition rates through their litter quality effects, released nitrogen at a faster rate than the litter from native species in a mixed deciduous forest (Ashton et al. 2005). The decomposition rates of litter were also high on invaded sites (Ashton et al. 2005).

SOIL BIODIVERSITY

There is great variety of substrates and wide range of species rich habitat found in soils of land ecosystems (Wolters 2001). The population of soil fauna and microflora commonly found in the surface layer of fertile soils show wide variations in their numbers and biomass (Table 9). About 25% of all described living species strictly inhabit soil or litter in diverse types of land ecosystems (see Decaëns et al. 2006). Out of total number of 360, 000 of described soil organisms, 80 % are insects and 12% belong to Arachnida (Figure 7). The diverse groups of soil organisms are able to co-exist in the soil because of trophic niche partitioning, spatial and temporal segregation, density dependent regulation and high micro-habitat diversity (Giller 1996, Wolters 2001).

Soil biodiversity refers to the variety of life existing in the soil and play an important role in various ecosystem functions such as decomposition of organic matter, nutrient cycling and formation and stabilization of soil structure e.g. nitrogen fixing bacteria, mycorrhizae and other soil organisms for various biological controls (Brussaard et al. 1997). At regional, landscape and local ecosystem level soil structure, temperature, moisture regimes and land management practices strongly influence various soil biological processes, spatial and temporal distribution of species (Fox and MacDonald 2002). The linkages between aboveground and belowground diversity play a key role in ecosystem stability and functioning (Coleman and Whitman 2005, Allen et al. 2007, Bardgett et al. 2008).

Table 9. Numbers and biomass (live fresh weight) of microflora and fauna commonly found in the surface 15 cm of soil (from Brady and Weil 1999).

Organisms	Number (per m^2)	Biomass (g m^{-2})[a]
Microflora		
Bacteria	10^{13}-10^{14}	40-500
Actinomycetes	10^{12}-10^{13}	40-500
Fungi	10^{10}-10^{11}	100-1500
Algae	10^{9}-10^{10}	1-50
Fauna		
Protozoa	10^{9}-10^{10}	2-20
Nematodes	10^{6}-10^{7}	1-15
Mites	10^{3}-10^{6}	0.5-1.5
Collembola	10^{3}-10^{6}	0.5-1.5
Earthworms[b]	10-10^{3}	10-150
Other fauna	10^{2}-10^{4}	1-10

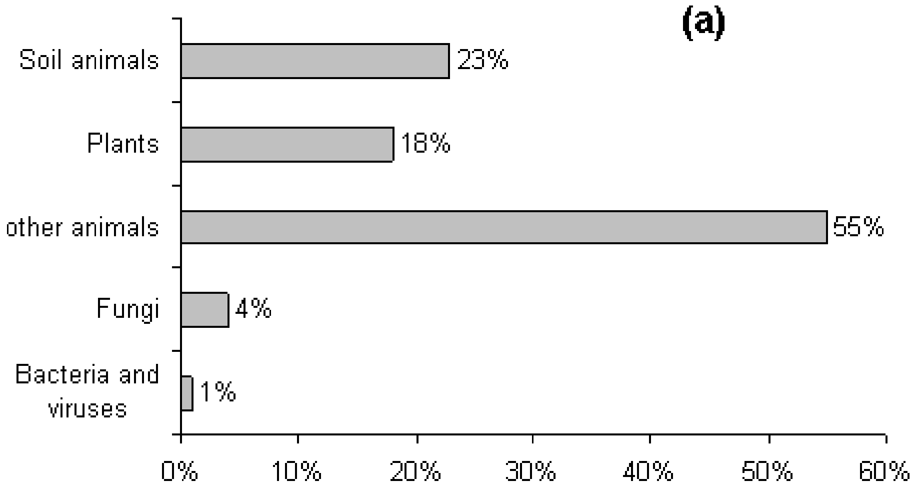
[a] Dry weights are about 20-25% of live fresh weight biomass.

[b] A greater soil depth is used for earthworms

Diversity of Soil Microorganisms

The soil micro-organisms including bacteria, fungi, and actinomycetes constitute the primary consumers of plant and animal residues and principal agents for the cycling of nitrogen and phosphorus. The diversity and abundances of microbial populations are related to litter diversity and resource quality of organic matter inputs. The plant residues with C:N ratios (>30:1) favour colonization by fungi, whereas litters with low C:N ratio favour bacteria , which in turn determine the diversity of consumers of bacteria and fungi (Hendrix et al. 1986, Moore and Hunt 1988). The physical and chemical nature of organic matter inputs into the soil have regulatory effect on the diversity of soil organisms as well as soil nutrient cycles (Moore et al. 2004).

Saprophytic soil fungi are active in decomposition of cellulose and lignin, whereas actinomycetes have important roles in decomposition of lignin and compost (Alexander 1977). Microbial diversity indices show ecological dynamics of a community and provide a promising tool to evaluate the patterns and processes natural and managed ecosystems. Soil microbial biomass, a labile pool of soil organic matter, comprises 1 to 3% of the total soil carbon and 3-5% of the total soil nitrogen (Jenkinson and Ladd 1981). Soil microbial biomass acts as a source and sink of plant nutrients (Smith and Paul 1990, Singh et al. 1989). Microbial



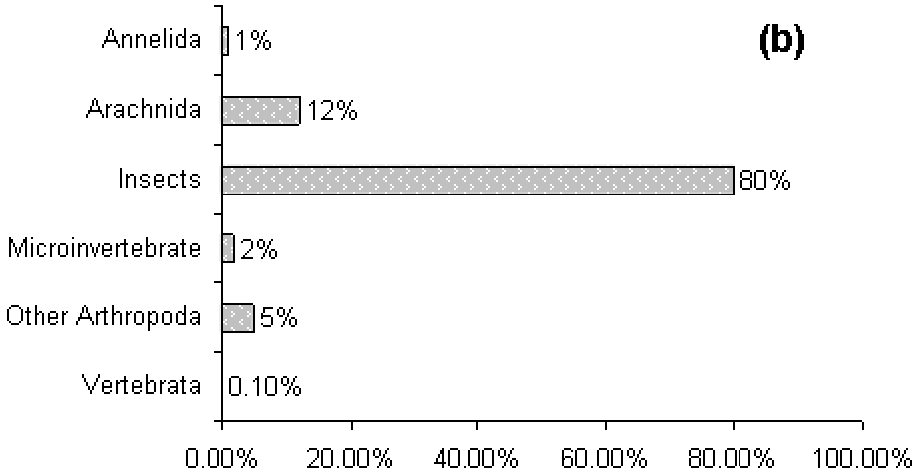


Figure 7 (a) Relative distribution of major groups of plants and animals in a total of 1.5 million species; (b) Relative importance of major taxa of soil animals (based on data from Decaens et al. 2006)

populations can provide an early indication of change in soil long before it can be measured by changes in soil organic matter (Powlson et al. 1987).

A significant stratification has been observed in soil microbial biomass carbon and nitrogen in the *Dalbergia sissoo* dominated forest. There was decrease in microbial biomass with increasing depth in the soil possibly due to the presence of distinct microbial communities for each soil layer. Differences in microbial biomass due to tree species and depth were related to soil organic matter in the tree plantations on moderate to highly sodic soils (Kaur et al. 2000, Kaur et al. 2002a). The size and dynamics of soil microbial biomass has been shown to vary with land use type (Kaur et al. 2000) and tree species (Kaur et al. 2000, Kaur et al. 2002a).

Soil microbial community composition and biomass have been studied using phospholipid fatty acid (PLFA) analysis of Great Plains grasslands spanning an 800 km transect from eastern Colorado to eastern Kansas (McCulley and Burke 2004). The microbial commu-nities differed among the different grassland community types; the relative abundance of fungi decreased while gram-negative anaerobic bacteria increased from short grass steppe to tall grass prairie (McCulley and Burke 2004).

Nitrification is a biological process carried out by nitrifying bacteria or nitrifiers in the soil. Nitrifiers are ubiquitous in the soil and play an important role in nitrogen turnover in ecosystems. Carney et al. (2004) have studied the effects of plant diversity and land use types on soil nitrifiers using polymerase chain reaction amplifications, cloning and sequencing of 16S rDNA. Several studies in India have reported that the populations of ammonia and nitrite oxidizing bacteria have been found to significantly relate to soil moisture, soil nutrient content, vegetation cover, soil characteristics and nitrogen mineralization rates (Jha et al. 1996, Ghosh and Dhyani 2005, Singh and Kashyap 2007). In a *Populus deltoides* agroforestry system in a semi arid region at Kurukshetra, the population size of ammonia oxidizing bacteria and nitrite oxidizing bacteria was found to be significantly related to seasonal variations in soil moisture content and soil nitrification rates (Saini 2008) (Table 10).

The enzymic index of carbon quality has been found to be highly correlated with decomposition rates (Sinsabaugh and Findlay 1995) and mass loss of litter has been predicted on the basis of extra cellular enzyme activity using a modeling approach (Sinsabaugh and Moorhead 1997). Using the MARCIE simulation model (Sinsabaugh and Moorhead 1997) predicted litter mass, microbial biomass and ligno-cellulose degrading enzymes for decomposing dogwood, maple and oak litter. Sinsabaugh et al. (2008) conducted a global-scale meta-analysis of the seven-most widely measured soil enzyme activities, using data from 40 ecosystems including grassland, shrub land, forest, tundra, cold desert, and herbaceous sere. The extra- cellular soil enzymes on a global scale provides a framework for comparing ecosystems as well as relating the soil microbial community function to global patterns of microbial biomass composition, nutrient dynamics and soil organic matter storage (Sinsabaugh et al. 2008).

Fierer and Jackson (2006) have analyzed bacterial community composition and diversity using the ribosomal DNA fingerprinting. Over a geographical gradient bacterial community diversity was affected by soil pH irrespective of site temperature, latitude of other soil variables; the diversity of bacteria being highest in neutral soils and lower in the case of acid soils (Fierer and Jackson 2006). At local scale, edaphic factors regulate microbial biogeography, whereas carbon and nutrient availability, and soil moisture influence the microbial community composition predominantly (Fierer and Jackson 2006).

Diversity of AM fungi

In terrestrial ecosystems, arbuscular mycorrhizal fungi characterize a delicate balance between plant, fungus and the soil (Mosse 1986) and are found in 80% of all species, including most agricultural crops (Trappe 1987, Smith and Read 1997). These are soil borne fungi belonging to six fungal genera (*Glomus, Sclerocystis, Acaulospora, Entrophospora, Gigaspora and Scutellospora)* of the single order Glomales within Zygomycetes. Recently, two new genera, *Archaeospora* and *Paraglomus*, have been added to the existing six genera of AM fungi (Redecker et al. 2000).

Mycorrhizal symbiosis is ecologically important in maintaining the productivity and diversity, stability of natural ecosystems (Jeffries et al. 2003, Kennedy 1998). Arbuscular mycorrhizal fungi have an important role in carbon cycling and soil carbon sequestration (Zhu and Miller 2003). In tropical system, mycorrhizal fungi could enhance plant productivity by increasing phosphorus uptake (Van der Heijden et al. 1998, 2006). From the microcosm study, Van der Heijden et al. (1998) reported that higher mycorrhizal fungal diversity is responsible for 105% higher plant diversity and 42% higher plant productivity. AM fungi and rhizobia enhance plant productivity by providing essential nutrients such as P and N to the plant, respectively. The role of arbuscular mycorrhizal fungi in regulating the productivity and diversity of plant communities

Table 10. Population of ammonia and nitrite oxidizing bacteria during January 2005 to April 2005 in *Populus deltoides* agroforestry system at Kurukshetra (from Saini 2008)

	January	February	March	April
Ammonia oxidizers (MPN × 10^4)	4.21-3.8	4.65-4.1	7.78-6.8	2.40-2.1
Nitrite oxidizers (MPN × 10^4)	1.36-1.3	1.82-1.73	5.81-4.83	1.42-1.3
Soil moisture (%)	18.63-17.76	20.28-19.4	22.7-21.67	9.40-9.52

has been studied in grassland ecosystems (Van der Heijden et al. 2008, Antoninka et al. 2009).

In the nutrient poor ecosystems, soil microbes have an important role in plant productivity by nutrient acquisition (Van der Heijden et al. 2008). Mycorrhizal fungi and nitrogen fixing bacteria have been found responsible for 5 to 80% of the total nitrogen and 75% of phosphorus acquired by the plants annually (Van der Heijden et al. 2008).

The occurrence of arbuscular mycorrhizal (AM) fungi in natural forest, agricultural systems, stressed systems and mangrove ecosystems in India have been studied by a number of workers (see Manoharachary et al. 2005). Various studies have indicated that the genus *Glomus* is widely distributed in different ecosystems because of its greater adaptability under a range of soil conditions (Manoharachary et al. 2005).

The diversity of AM fungi, spore density and relative spore density has been found to be affected by tillage in a rice-wheat system in northern India (Neelam 2006). In the zero-tillage system, the number of mycorrhizal fungi was higher compared to that of the conventional tillage and the furrow irrigated raised bed system. A total of 42 species of arbuscular mycorrhizal (AM) fungal species belonging to six genera (*Glomus, Acaulospora, Entrophospora, Gigaspora, Sclerocystis* and *Scutellospora*) were recorded in the wheat cropping system under different tillage practices (Neelam 2006; Figure 8). The relative density of four groups of mycorrhizal spores was: 64.45 to 73.08% (*Glomus* sp.), 14.14 to 20.38% (*Acaulospora* spp.); 2.70 to 13.68% (*Gigaspora* sp.) and 3.88 to 6.71% (other species). The AM fungal root colonization of wheat roots ranged from 78.98 to 93.96% in zero-tillage, furrow irrigated raised-bed tillage and the conventional tillage system at crop maturity in 7.5 to 15 cm soil depth (Figure 9).

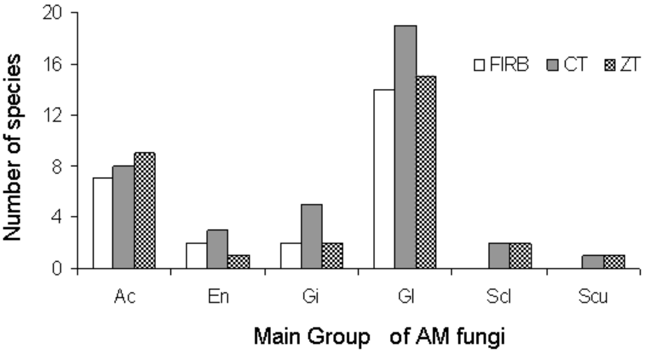


Figure 8. Diversity of arbuscular mycorrhizal (AM) fungi under conventional tillage (CT), zero-tillage (ZT) and furrow irrigated raised bed (FIRB) tillage practices in a wheat cropping system; Ac= *Acaulospora*, En= *Entrophospora*, Gi= *Gigaspora* , Gl= *Glomus*, Scl= *Sclerocystis*, Scu= *Scutellospora* (Neelam 2006).

Biodiversity of Soil Fauna

Soil fauna consist of a variety of organisms and the size relationship among various groups of soil fauna have been described by Swift et al. (1979). On the basis of size, three groups of soil fauna are: microfauna (protozoa and nematodes in water filled soil porosity), semi-micro fauna (collembola and acarids of litter and air filled pore space), and macrofauna (termites, earthworms and large arthropods) are described. Three major guilds of soil invertebrates i.e., micro food webs, litter transformers and ecosystem engineers, have been described on the basis of their interaction with soil micro organisms and the type of excretory products (Levelle 1997).

Earthworms convert plant residues into soil organic matter by increasing residue exposure to microbial activity and feeding on soil organic matter (Lee 1985). Termites are known to be efficient in cellulose and lignified subsystems as they produce a variety of enzymes due to the presence of associated microflora and protozoan in their guts to digest cellulose, lignin and other components (Lee and Wood 1971). In grassland and forest ecosystems of India, earthworms, termites and soil arthropods are widely distributed and abundant groups of soil fauna (Singh and Gupta 1992, Gupta and Malik 1996).

In sub humid grassland in Orissa, the oligochaetes formed 80% of total invertebrate biomass and processes about 18% of total energy input into soil (Senapati and Dash 1981). A high diversity of earthworm fauna has been reported in India due to varied climate and availability of diverse ecological niches (Julka and Paliwal 2005). A total of 413 species and subspecies of earthworms, belonging to 69 genera and 10 families are found in different biogeographical region of India. The eastern Himalayan Ago-climate zone exhibits high earthworm diversity accounting for 26% of all the species found in India (Julka and Paliwal 2005). The conversion of native forest by shifting agriculture has shown decline in species richness of earthworms in western Orissa (Senapati et al. 2005).

Termites play an important role in the decomposition of litter and roots in a grassland ecosystem at Kurukshetra (Gupta et al. 1981). In grassland and forest ecosystems, earthworms improved nutrient availability and termite activity resulted in the formation of nutrient rich microsites (Gupta et al. 1981, Singh and Singh 1989). The role of nutrient rich microsites due to the activity of earthworms and termites in vegetation regeneration and ecosystem stability needs to be analyzed in greater details.

Soil Food Web

The decomposer community forming the soil food webs is functionally complex as compared to the plant-based food chains. Soil food web models are based on grouping soil organisms into assemblages with similar trophic levels. These trophic groupings exhibit a high degree of taxonomic and functional diversity. In soil, bacteria and fungi are the primary decomposers and the mass of these components (microbial biomass) relates to the resource quality. One of the most extensive accounts of nutrient cycling through a soil food web is that of Hunt et al. (1987) in a short prairie ecosystem. Wardle (1995) reviewed and analyzed various studies on soil foodwebs under intensive and reduced or zero tillage conditions. In intensive agriculture systems, there is inhibitory effect of tillage on soil meso- and macro fauna, whereas in reduced tillage there is enhanced activity of mesofauna.

The linkages between plant litter quality, soil biota and decomposition processes have been analyzed by Lavelle (1997). Plant litter quality regulates the diversity of soil biota and the nature of soil biotic interactions, by operating through “micro foodwebs”, litter transformation system and ecosystem engineers for nutrient immobilization- mineralization (Lavelle 1997). Micro food webs mainly comprise micro-fauna (nematodes and protozoa) which are predators. The litter transformers show a mutualistic relationship with the micro flora based on internal rumen. Ecosystem engineers, such as earthworms and termites, have mutualistic relationship with the microflora in their gut and influence ecosystem processes by creating or modifying the physical environment for other species (Lavelle et al. 1997).

The detritus based food webs show the presence of distinct within habitat compartments, maintained by morphological differences between bacteria and fungi and differences in habitat niches to soil moisture and management practices (Hendrix et al. 1986). The high degree of omnivory leads to greater connectivity in soil food webs. In a microcosm experiment, Wardle (1995) showed that connectivity remained unaffected even through species richness increased. The bacterial feeding and fungal feeding in the detritus food webs are generalist feeders. Independent of species numbers, connectivity in fungi-based compartments is less than in bacteria-based system (Wardle 1995).

The concern over the functional consequences of declining biodiversity has brought about an increasing interest in linking below-ground food webs with aboveground food webs (Brussaard et al. 1997, Wardle 2002). Detritus food webs have been used for quantitative assessment of the nutrients and carbon between

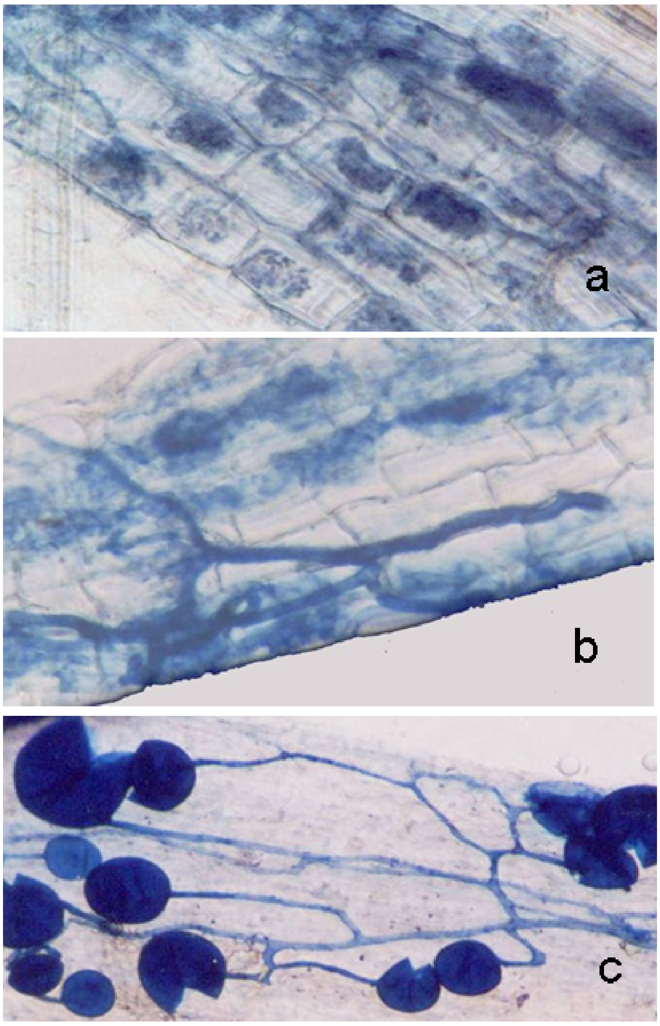


Figure 9. AM fungal infection in wheat roots under Zero-tillage. (a) An arbuscule with mycorrhizal hypha in a root cortical cell; (b) the penetration of infective AM hypha in cortical cells; (c) mycorrhizal hyphae with globose vesicles (Neelam 2006).

Table 11. Methods for Measuring the Soil Microbial Diversity

Method	Ecosystem Type / Soil	Selected References
Plate counts of colonies	Agriculture soils	Johnsen et al. (2001) Bakken (1997)
Most Probable Number technique	Agroforestry system	Saini (2008)
Nitrification rates	Agroforestry system	Saini (2008)
Decomposition rates	Global transect	Wall et al. (2008)
Soil respiration (CO_2 production)	Tropical grassland	Gupta and Singh (1981a)
Biomass C, or N,	Agroforestry system ,	Kaur et al. (2000)
Soil Enzymes	Forest Ecosystem	Sinsabaugh and Moorhead (1997)
Soil Enzymes	Global-scale meta-analysis	Sinsabaugh et al. (2008)
Phospholipids	Great Plains grasslands, USA	McCulley and Burke (2004)
DNA and RNA	Tropical forest, tree plantation, and pasture	Carney et al. (2004)
Ribosomal DNA –fingerprinting	North and South America	
	Soils of diverse ecosystems	Fierer and Jackson (2006)
Stable isotope probing	Methanol-utilizing microorganisms	Radajewski et al. (2000)
Minirhizotron method	Northern hardwood forest, USA	Hendrick and Pregitzer (1992)

various compartments, with a high degree of connections between predators. However, data are lacking for an assessment of the relative importance of temporal and spatial, trophic and non-trophic, evolutionary and ecological effects of soil fauna on microbes and subsequently, on ecosystem processes (Nieminen 2008).

Molecular Ecology and Soil Microbial Diversity

Some methods for analyzing soil microbial diversity are summarized in Table 11. A majority of soil microbes (99% of microbes) can not be grown in laboratory cultures. The cultivation-independent methods to study the indigenous microbial communities were adopted in the 1990s when the Polymerase Chain Reaction (PCR) and other DNA- based characterization methods became available (Narasimhan et al. 2003).

Information based on small subunits (16S rRNA of prokaryotes or 18S rRNA of Eucaryotes) reveal the phylogenetic relationship between the organisms from where the DNA or RNA arose. The new discipline of molecular microbial ecology is helping to provide new insights in the field of microbial diversity by using cultivation independent approaches (Narasimhan et al. 2003).

The molecular biological methods have been coupled with stable-isotope probing as cultivation-independent means of linking the identity of bacteria with their function in the environment. Stable isotope probing (SIP) was introduced to microbial ecology by Radajewski et al. (2000). This method has been used to characterize growing microorganisms in the environmental samples or determine those which have the genetic potential of metabolizing a labelled substrate. Radajewski et al. (2000) showed the application of this technique to investigate methanol-utilizing microorganisms in soil that involved two phylogenetically distinct groups of eubacteria; the α-proteobacterial and *Acidobacterium* lineages.

Visualization of natural belowground ecosystems has been difficult to analyze belowground biodiversity because of the physical inaccessibility of the rhizosphere. The minirhizotron method (Hendrick and Pregitzer 1992) offers a new opportunity to rhizosphere organisms in natural systems. The production, development, and mortality of fine roots in a northern hardwood forest have been monitored using minirhizotrons (Hendrick and Pregitzer 1992). With the automated minirhizotron camera (AMR), it is possible to capture images of roots, soil fungi, soil structure, and soil fauna, without excessively disturbing the ecosystem (Allen et al. 2007). New sensor technology can measure and monitor soil biodiversity and processes rapidly and continuously at different spatial and temporal scales. Experimental embedded networked sensors technology is leading to new understanding of spatio-temporal rhizosphere processes (Allen et al. 2007).

Scanning electron microscopy (SEM) or combined with staining of the native microbial community are being used to analyze soil microbial diversity (Sørensen et al. 2009). Environmental scanning electron microscopy (ESEM) could be useful for high resolution studies of microorganisms in undisturbed rhizosphere samples (Cabala and Teper 2007). Rapid development of novel florescence in situ hybridization (FISH) probes, staining technologies and confocal laser scanning microscopy (CLMS) has resulted in numerous root colonization studies(Eickhorst and Tippokotter 2008). The development of resin embedding and thin sectioning of FISH stained soil samples have been used for studying undisturbed soil and rhizosphere samples for analyzing soil microbial diversity (Eickhorst and Tippokotter 2008). There is need to integrate the microbial diversity with ecosystems processes, so that functional diversity can be studied at local, regional and landscape level.

Sequencing of soil genomes is a new area in microbial ecology, which will reveal the secrets of soil microbial communities in the ecological processes of a region. The genetic technology is helping to extract the genome of soil itself. Metagenomics can be useful to analyze the genetic diversity of the bacteria or rhizosphere soil sample (Rondon et al. 2000). The metagenomic alternative has demonstrated its utility to better understand the unidentified bacterial communities and the functioning of ecosystems (Rondon et al. 2000). Metagenomic DNA libraries have been constructed successfully from the rhizosphere metagenome of plants adapted to acid mine drainage and their screening adapted to detect novel microbial resistance genes (Mirete et al. 2007).

The present availability of a large number of whole genome sequences could be helpful in the understanding of microbial communities in the rhizosphere as well as genetic composition of their individual constituents (Sorensen et al. 2009). There is need to understand the functions of organisms in a community and developing suitable computational methods to manage the increasing amount of data obtained by metagenomics and metablomics (Sorensen et al. 2009).

SOIL CARBON MANAGEMENT

In recent years CO_2 has received much attention because its concentration in the atmosphere has risen to approximately 30% above natural background levels and will continue to rise in the near future (IPCC 2001, IPCC 2007). The global carbon sequestration potential in soil has been estimated at 0.4 to 1.2 Pg C yr^{-1} (Lal 2004; Figure 10).

There is a large potential for carbon management in agriculture, agrofortestry and plantation, forestry, and arid land ecosystems (Watson et al. 2000, Nair 2007, Trumper et al. 2009).

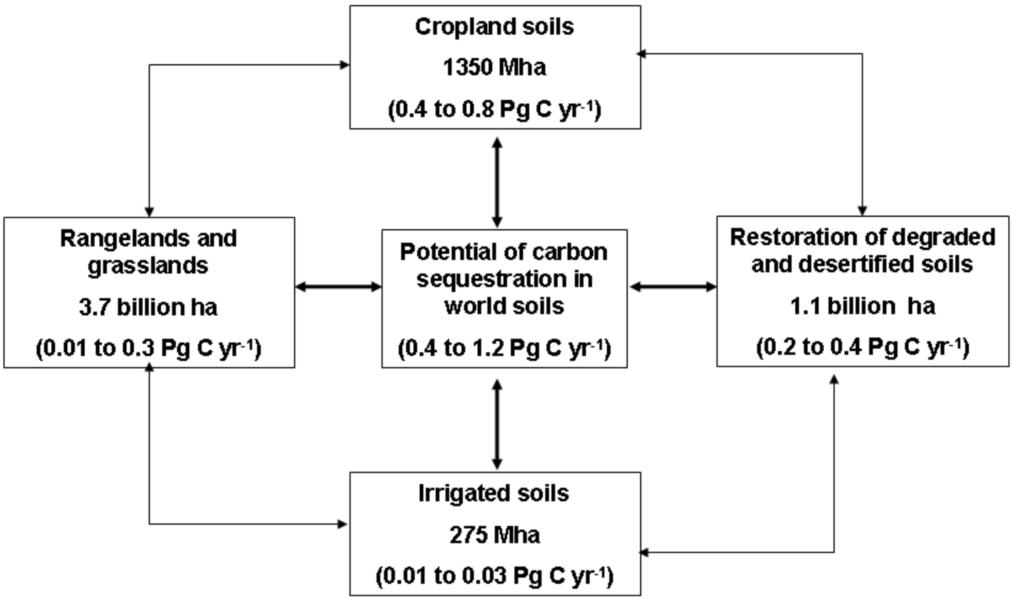


Figure 10. Soil Carbon sequestration potential in cropland, grazing land/rangeland, degraded/desertified lands, and irrigated soils (from Lal 2004)

Conservation Agriculture for Carbon Sequestration

Soil carbon levels are low in agricultural soils as compared to natural undisturbed ecosystems (Woomer et al. 1994). Foley et al. (2005) have reported that 40% of the land surface was covered by cropland and pasture. The large carbon sink potential in agricultural soils can be exploited as a mitigation option via agricultural management. The carbon sink in agricultural soils could be improved by introducing zero-tillage, improved efficiency of crop residue use, application of compost, crop rotation changes, bioenergy crops and organic farming (Porter et al. 2007).

The conservation tillage is a set of practices that leave crop residues on the soil surface to reduce erosion, and includes no-tillage, direct drilling, and minimum-tillage and/or ridge tillage to develop ecologically sustainable agricultural systems for the future. In recent years, the zero-tillage practices for establishment of wheat are being increasingly adopted in the Indo-Gangetic plains including regions of Haryana and Punjab. In the rice-wheat agriculture system, soil carbon increased in surface layer after seven years of zero-tillage (0.61%) as compared to conventional tillage (0.44%) (Neelam 2006).

Carbon Sequestration in Tree Plantations and Agroforestry

About 4% of the global forest area is represented by plantation (FAO 2006). There is large potential of carbon sequestration through tree plantations on marginal agricultural and degraded lands (Lal 2006). On the basis of synthesis of 600 observations, climate and economic modeling, Jackson et al. (2005) showed that plantations decreased stream flow by 227 mm per year globally (52%), with 13% of streams drying completely for at least 1 year. Based on the findings of Jackson et al. (2005), it may be stated that the evaluation of the benefits and trade-offs of tree plantations is crucial for the development and implementation of sustainable carbon sequestration policies in different regions of the world. It is also important to compare the value of other ecosystem services gained or lost with those of carbon sequestration (Jackson et al. 2005).

Agroforestry has a role in providing food and nutritional security and controlling land degradation and supporting environmental benefits across a range of landscapes and economies (Lal 2004, Nair 2007). Agroforestry has advantage of storing carbon through enhanced build up of soil organic matter and carbon storage in roots and deep soil layers. Agroforestry provides other multiple benefits including diversity conservation, water quality improvement, improvement of soil fertility and the ecological services (Lal 2004, Nair 2007, Pandey 2007, Schoeneberger 2008).

The average carbon storage in different types of agroforestry systems has been found to range from 10 to 50 Mg ha^{-1}. The smallholder agroforestry system could sequester 1.5 to 3.5 Mg C yr^{-1} ha^{-1} (Montagnini and Nair 2004). Trees in agroforestry systems play an important role in sequestration of carbon in the above and below-ground biomass. The total above-ground carbon storage varied from 29.41 to 44.43 Mg C ha^{-1} and total below-ground carbon storage increased from 6.28 to 9.51 Mg C ha^{-1} in the 5 to 7 year old *Populus deltoides* agroforestry system (Saini 2008). In tropical areas, agroforestry has a high potential for carbon sequestration than conventional agriculture (Nair et al. 2009), besides diverse co-benefits in terms of biodiversity conservation.

Carbon Sequestration in Arid and Degraded Lands

For restoration and maintenance of soil productivity, Gupta and Rao (1994) assessed the potential of wasteland for sequestering carbon by reforestation. About 2.8 million ha of land in the Indo-Gangetic plains are salt –affected. There is a large potential of sequestering carbon in soil and vegetation by protecting native vegetation, adopting reclamation agroforestry and silvopastoral agroforestry systems on salt affected soils (Gupta et al. 1990, Kaur et al 2002b). The tree-based land use system could sequester carbon in soil and vegetation and improve nutrient cycling within the system on highly sodic soils (Kaur et al 2002b). Carbon pools in soil, vegetation and soil microbial biomass in the silvopastoral agroforestry systems are shown in Figures 11 and 12. The total carbon storage in the tree + grass systems was 1.18 to 18.55 Mg C ha^{-1} and carbon input in net primary production varied between 0.98 to 6.50 Mg C ha^{-1} yr^{-1} . Carbon flux in net primary productivity increased significantly due to integration of *Prosopis* and *Dalbergia* with grasses. In silvipasture agroforestry systems, soil organic matter, biological productivity and carbon storage were greater than grass only systems (Kaur et al. 2002 b).

The soils of the arid and semi-arid regions of the Indo-Gangetic plains are poor in soil carbon (Bhattacharyya et al. 2008). There is large potential of carbon sequestration through improved land management. The

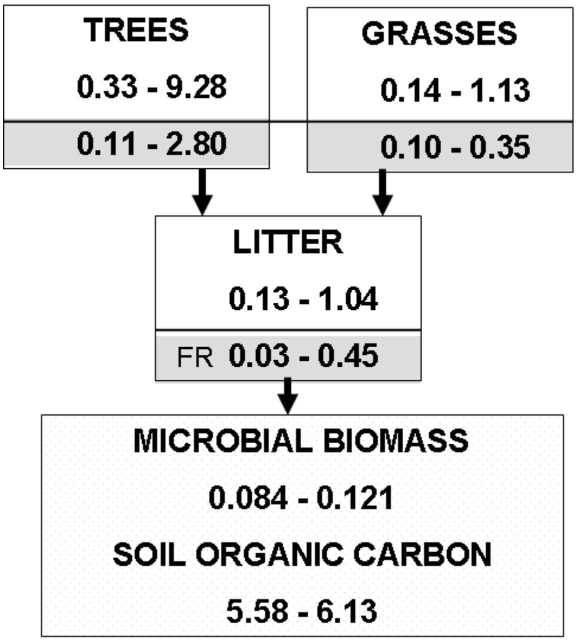


Figure 11. Carbon pools in soil, vegetation (the shaded portion of boxes represent the carbon pool in roots of trees and grasses, and fine roots (FR), litter and soil microbial biomass (Mg C ha^{-1}) in Trees + *Sporobolus marginatus* system in on a sodic soil at Bichian in northern India. (based on data from Kaur et al. 2002b).

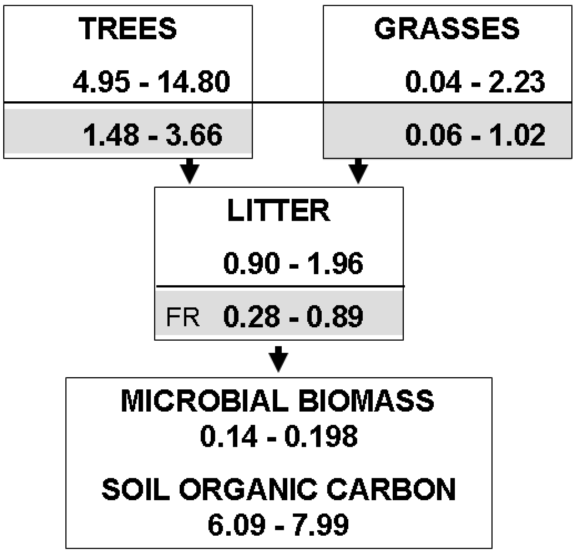


Figure 12. Carbon pools in soil, vegetation(the shaded portion of boxes represent the carbon pool in roots of trees and grasses, and fine roots, FR) and soil microbial biomass (Mg C /ha) in Trees + *Desmostachya bipinnata* system on a sodic soil at Bichian in northern India. (based on data from Kaur et al. 2002b).

thematic maps on soil organic carbon stocks can help to formulate carbon seques-tration programmes for different bioclimatic regions of the country (Bhattacharyya et al. 2008). On the basis carbon stocks of organic and inorganic carbon in different bioclimatic zones, Bhattacharyya et al. (2008) have reported that there is large potential of sequestration of atmospheric CO_2 in the form of soil inorganic carbon, i.e. pedogenic carbonate. New research initiatives are needed to create the historical soil climate-crop databank to make future projections on the sustain-ability of the rice-wheat cropping systems and agroforestry (Pal et al. 2009).

Some recent studies suggest that carbon uptake by desert is high and it can contribute significantly to the terrestrial carbon sink (Wohlfahrt et al. 2008) and there is need to quantify above- and belowground carbon pools over time in desert lands (Schlesinger et al. 2009).

Initiatives for Climate Mitigation

The United Nations Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol requires countries to take appropriate measures to reduce their overall greenhouse gas emissions to a level at least 5.2% below the 1990 level during the initial commitment period 2008-2012. Improved forest, cropland and rangeland management and agroforestry could result in the short term and long term sequestration of carbon in the plant-soil system. The first afforestation CDM pilot project in India , covering about 370 ha of the barren and rainfed lands in Sirsa, Haryana, has been approved during the year 2008 (UNFCCC 2008). This afforestation project is expected to sequester 11,591 tonnes of CO_2 equivalent of greenhouse gas every year.

A new project the Global Land Project (GLP) has evolved from Global Change and Terrestrial ecosystems (GCTE) and Land-Use and Land Cover Change(LUCC) with a mission to measure, model, and understand the coupled human environmental system (Ojima et al. 2007). The links between decision making, ecosystem services and global environmental change define various key pathways of coupled human-environmental activities at local, regional and global scales (GLP 2005). The main focus of the programme is to assess provision of ecosystem services as affected by the changes in the coupled socio-environmental system (GLP 2005, Ojima et al. 2007). The GLP project promotes greater integration of social and biophysical sciences so as to adapt to meet the challenges of global climate change. The carbon cycle can be better managed by assessing the carbon sequestration potential, the potential gain in carbon stocks in biomass and in soils within a given land area resulting from a change in land use, land cover and land management (Ojima et al. 2007).

Maintaining the stores and sink of carbon in natural ecosystems can play key role to reduce future emission of greenhouses gases (Lewis et al. 2009). Forests will help in adaptation to climate change by increasing resilience of people and ecosystems. To provide ecosystem services for strong carbon sinks will require formalizing and enforcing land rights along with payment of ecosystem services for forest dwellers living near forested areas (Lewis et al. 2009). Thus, protection of tropical forests would serve as carbon store in the long-term. Mitigation and adaptation options in the forest sector need to be fully understood and used in the context of promoting sustainable development.

CONCLUSIONS

The carbon balance of an ecosystem depends on aboveground and belowground processes. Belowground processes are still poorly understood, yet these provide a number of potentially important feedbacks in the global carbon-cycle-climate system. The ecological significance of decomposition and humus formation need to be analyzed from the perspectives of microbial ecology (Berg and McClaugherty 2008). Models should be developed for understanding and predicting key dynamics of soil C in relation to ecosystem processes and socioeconomic drivers under scenarios of global and regional change. In view of the global climate change, there is need to understand the role of soils in carbon storage in natural and cultivated systems.

Sustainable management and conservation of soil biota is important for conserving global biodiversity as soil communities are species rich and affect ecosystem processes (Decaëns et al. 2006). About 100 species of soil organisms are threatened to any degree and represent only one percent of total number of threatened species worldwide (IUCN 2004).There is need for the application of precautionary principle for conserving soil biodiversity and implementing conservation strategies for protecting soil biodiversity in different ecosystems (Decaëns et al.2006). Keeping in view the importance of soil organisms in maintaining ecosystem services and soil productivity, it is important to analyze various functional groups of below-ground biodiversity.

The extensive research in the field of soil ecology can be used for innovation and applications within the framework of general ecology by integrating empirical observations with general theories (Barot et al 2007). There is need to build bridges between soil ecology, general ecology and ecosystem management. Understanding the factors that affect the resistance and resilience of soils under changing environmental conditions is one of the frontiers in soil ecology research. The integration of science, technology and traditional ecological knowledge can make substantial contribution to the science of soil ecology, ecosystem carbon management, and sustainability.

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