

CHAPTER 1

INTRODUCTION

There are ever increasing pressures for the development of sustainable agriculture. Over the next 25 years the population of the world will grow by about 40% to over 8,000 million people (Pinstrup-Anderson and Pandya-Lorch, 1994). At present, the number of people in the world who are undernourished is estimated to be between 500 million and 1 billion (World Resources Institute, 1992). Although world food production has dramatically increased since 1970, particularly in the developing countries, the per capita production has declined significantly in many African countries. The index of per capita production compiled by the Food and Agriculture Organization of the United Nation (FAO), using the 1979-1980 period as a baseline of 100, shows that 35 of 47 developing countries dropped below 100 in 1989, with the worst declines in Africa (World Resources Institute, 1992). These create a significant increase in the demand for food and consequently, feeding the rapidly growing world population has become a major agricultural development concern for the world.

High demand for food exerts pressure on the land resources in a number of ways, such as, 1) pressure to increase production on land currently in use through increased yields and cropping intensity in space and time, 2) pressure to expand the land in production, which inevitably means expansion to more marginal lands and 3) pressure on land through degradation of the resource and competition for non-agricultural uses (Lefroy and Craswell, 1996). Alexandratos (1988) projected that from 1983/1985 to the year 2000 agricultural production in the developing countries will increase by about 60%. Of this increase, 63% will come from increased yield, 15% from increased cropping intensity and 22% from net increase in arable land. In many countries in Asia, marginal land or cropping systems are being increasingly relied upon for agricultural production. As this reliance increases there will be a greater need to develop sustainable land management systems to preserve the natural resource base. Soil organic matter plays a major role in solving soil and environmental problem (Wallace, 1994b). Thus, the development of systems which preserve the resource base and avoid degradation of soil requires detailed understanding and monitoring of soil organic matter.

Soil organic matter is considered a key factor in maintaining soil quality and therefore it is crucial in determining long-term soil fertility (Wallace, 1994b). There is a great deal of evidence that the decline in crop yield with continued production in many temperate and tropical areas is correlated with declines in organic matter levels (Wood, 1985; Sanchez *et al.*, 1987). Therefore, the maintenance of soil organic matter must be considered as one of the goals of sustainable land management systems.

Soil organic matter provides a reservoir of plant nutrients and improves soil structure. In sandy soils, increased organic matter improves water retention and reduces the leaching of nutrients. In clay soils, added organic matter can improve soil porosity, reducing water runoff and erosion

losses. Organic matter can protect the soil against increases in acidity and alkalinity and can provide micronutrients not readily available in the most commonly used chemical fertilisers. Clearing and subsequent cropping affect the organic matter status of soils through changes in both the input of organic matter and its rate of turnover (Stevenson, 1986; Jenkinson, 1988).

As a consequence of decreasing crop yields resulting from soil organic matter decline, there is increasing interest in using plant residues and other organic materials to improve productivity in tropical agricultural systems. Maximising the return of crop residues is clearly important as it both reduces the removal of nutrients and returns organic matter. Many sources of organic materials, such as crop residues, green manures, compost, agro-industrial wastes or animal manure, have been widely used as soil amendments.

The breakdown of organic materials in soil is controlled by a large number of factors. The proportion of total carbon decomposed in the initial phase of breakdown is similar for a wide range of crop residues. Jenkinson (1981) found that ryegrass roots, ryegrass tops, green maize and mature wheat straw, all lost about two-thirds of their C after one year in the field. Allison *et al.* (1949) found that 34% of the C in green oats remained in the soil after one year. The interesting feature of these results is that similar amounts of C remain in the soil from fresh green residues and mature plant material once the initial phase of decomposition is over. The rates of decomposition of different plant materials, however, differ during the initial stages of decomposition (Jenkinson, 1981). Thus, Waksman and Tenney (1927) showed that immature plant material, with a large water-soluble fraction, decomposed more rapidly than did mature material. With certain exceptions, it is generally reported that most of the crop residue added to the soil is broken down in the first year.

One major factor that controls the decomposition rate is the quality of materials. The diversity in sources of plant derived organic matter inputs, as well as the complexity of these substrates, results in extreme heterogeneity in the microbial reactions involved in their decomposition (Alexander, 1977). Different sources of organic materials contain different types and/or amounts of organic compounds, and thus have different breakdown rates. Lignin levels and C:N ratios of plant residues are known to affect decomposition rate and nitrogen mineralisation (Peevy and Norman, 1948; Herman *et al.*, 1977). Although the C:N ratio is accepted as a good indicator for N mineralisation and immobilisation (Allison, 1973), polyphenol content also affects the processes (Vallis and Jones, 1973). Palm and Sanchez (1991) reported that mineralisation of leaves of some tropical legumes was not correlated with N or lignin contents, but negatively correlated with the polyphenol levels.

Apart from the quality of material, there are environmental factors that co-determine decomposition. Field studies suggest that decomposition rates in the tropics are generally higher than under temperate conditions (Jenny *et al.*, 1949; Jenkinson and Ayanaba, 1977). In addition, the management of organic matter application, such as the method of application and the rate, type and method of fertiliser application, can affect the breakdown rate. Often the incorporation of organic matter into the soil, especially with inorganic fertiliser, leads to more rapid breakdown. This is

presumably as a result of the more favourable conditions for microbial activity — including moisture content, temperature, nutrient availability and more intimate mixing of the organic matter with the soil.

In the past, organic matter management has been approached via the use of crop residue return and green manuring (Meelu and Morris, 1988). The use of green manure crops with high potential breakdown rates is appropriate in regions with cool spring temperatures, which slow the decay rate such that nutrients, particularly N, are released from the green manure at a rate which has some relationship with crop demands. In tropical systems, where mineralisation rates are potentially higher because of high soil temperatures at the beginning of the growing season (Jenkinson and Ayanaba, 1977), and potential leaching losses are greater because of higher intensity rainfall, a rapid release of nutrients from the residue is inappropriate. The rapid release of nutrients such as nitrogen, potassium and sulfur can lead to NO_3^- , K^+ and SO_4^{2-} , and the associated cations and anions, moving down the soil profile with the wetting front such that the establishing crop does not have ready access to the released nutrients. In soils of high hydraulic conductivity such nutrients may be leached below the rooting zone of the crop. This pattern of nutrient release is clearly the opposite of that required for sustainable agriculture.

If leaching of nutrients occurs, it is likely that there will be significant movement of carbon down the profile. When crop residues or green manure are added to tropical systems, where the turnover rate is high, there is an episodic introduction of carbon into the system and a rapid depletion of carbon as mineralisation progresses. This source of carbon and nutrients is readily decomposable by soil micro- and macro-organisms and represents a flush of carbon and nutrients into the system at a time when crop demand is low. Apart from significant loss of residue carbon as CO_2 , a part of the residue carbon, in the form of dissolved organic carbon compounds, can move down to the soil profile. This not only reduces the carbon content of the surface soil layers, but may also have significant effects on the N dynamics of the system. Where crop residue is added to soil there is a likelihood that denitrification may be increased following the addition of an energy source (Avalakki *et al.*, 1994). If soluble carbon compounds move down the profile with, or to, the nitrates and nitrites, the carbon compounds can provide a substrate for denitrifying bacteria, resulting in a loss of N from the soil to the atmosphere.

Lefroy *et al.* (1996) studied paired samples collected from adjacent areas of a Paleaquilt soil with different management at sites in Northeast Thailand. They found a large decline in total carbon and larger decline in labile carbon, down the profile to 40 cm. The carbon in the top 20 cm was substantially decreased by cropping. These changes in soil carbon, after periods of cropping, are associated with reduced fertility and low rice yields. The relatively large amounts of total and labile carbon in the 20 - 40 cm layer of the cultivated soil suggests there has been significant leaching of C and, presumably, other nutrients.

The fate of the carbon and nutrients released during organic matter decomposition is an important determinant of the short and long-term fertility of the system. For example, if the nitrogen mineralised from the leaf fall moves down the soil profile and is denitrified then it represents an important loss from the system. Little can be done to alter the fate of nutrients in leaf fall with the

possible exception of selecting for species with slower breakdown rates. When slow decomposition rate residue is incorporated into the soil, there can be a short-term reduction in available nutrients, particularly N, as they are immobilised in organic forms. Under certain conditions, and when plant demand is relatively low, the immobilisation of nutrients can reduce losses by leaching and, in the case of N, volatilisation. As a result, the application of slow decomposition residue should result in significant inputs of carbon and other nutrients which have a significant impact over the medium and longer term. Where the short term reduction in the availability of plant nutrients limits crop growth, it can be overcome by the addition of fertiliser. The integration of fertilisers with residues requires a balance between the stimulation of organic matter breakdown, which is the substrate for microbial activity and the means to sustain improvement in soil fertility.

Studies using residues as soil amendments have generally studied the yield response and nutrient uptake to various combinations of inputs with little or no measurement of nutrient cycling. To accurately assess the sustainability of the system, carbon and nutrient dynamics must be studied together. It has become apparent that residue management to optimise the efficiency of nutrient use by crops could be achieved on the basis of a detailed understanding of the processes regulating nutrient transfer and partitioning between residues, soil profile and plants, including losses by leaching. Unequivocal evidence of these processes can only be obtained using tracer isotopes. The dynamics of soil organic matter has generally been investigated by measuring the rate or degree of incorporation of applied isotopes into soil organic matter, rather than the chemical or physical form or the rate of breakdown or release. Stable and radioisotopes, when available, have been used in these studies for carbon (^{14}C and ^{13}C), nitrogen (^{15}N), phosphorus (^{32}P and ^{33}P) and sulfur (^{34}S and ^{35}S).

The primary concern of the work described in this thesis was to investigate aspects of the breakdown and nutrient releases of plant residues, the subsequent movement within the soil profile, including leaching losses, and the effects on soil properties. To achieve these goals a series of laboratory and glasshouse experiments were implemented and conducted at the Department of Agronomy and Soil Science, University of New England, Australia.

A laboratory experiment (Chapter 3) developed the UNE *in vitro* perfusion system as a technique for monitoring the decomposition and nutrient release from plant residues. The UNE perfusion system was then used to study the decomposition and nutrient release from residues with a wide range in quality and a pot experiment used the same residues to verify the results from the perfusion study (Chapter 4).

A glasshouse experiment, using drained pots, was conducted (Chapter 5) using ^{15}N and ^{35}S labelled plant residues of different quality. The impacts of these soil amendments on crop growth and soil organic matter dynamics were investigated. The natural abundance $\delta^{13}\text{C}$ technique was used to monitor changes in soil carbon.

The results of each of the experiments are discussed at the end of each chapter. The principal findings and their implications are considered further in the concluding discussion in Chapter 6.

CHAPTER 2

REVIEW OF LITERATURE

This study is aimed at understanding the effects of plant residue quality on the decomposition process, on the growth response of a subsequent crop and on the dynamics of soil organic matter (SOM). It is necessary to understand the process of organic matter formation, the role of SOM in agriculture, as well as the processes of organic matter decomposition in soil. The following extensively reviews these aspects of organic matter.

2.1 Definition of Soil Organic Matter

Schnitzer (1991) proposed that SOM refers to the sum total of all organic carbon-containing substances in soils. It consists of a whole series of products which range from undecayed plant and animal tissues through ephemeral products of decomposition to fairly stable brown to black material bearing no trace of the anatomical structure from which it was derived; it is the latter material that is normally defined as humus (Jenkinson, 1988). The organic matter content in soils can vary from less than 1% in coarse-textured sands (such as desert soils) to close to 100% in organic soils. A typical agricultural soil may contain 1 - 5% in the top 15 cm (Rasmussen and Collins, 1991; Schnitzer, 1991), which usually decreases rapidly with soil depth (Jenkinson, 1988). Rasmussen and Collins (1991) suggested that the percentage of organic matter equals the percentage of organic C x 1.724.

2.2 Constituents of Soil Organic Matter

As a result of its heterogeneous nature, several attempts have been made to subdivide SOM into groups which have similar morphological or chemical characteristics. According to Schnitzer (1991), SOM falls into two broad classes — non-humic and humic substances.

2.2.1 Non-humic Substances

Non-humic substances include those with still recognisable chemical characteristics such as carbohydrates, proteins, peptides, amino acids, nucleic acids, purines, pyrimidines, fatty acids, waxes, resins, pigments and other low-molecular weight organic substances. In general, these compounds are relatively easily degraded in soils and have short half lives (Schnitzer, 1991).

Carbohydrates in SOM account for 5 - 25% (Stevenson, 1986) and occur mainly as polysaccharides (Cheshire, 1979). Their origins are plant and animal residues, as well as extracellular gums produced by microorganisms and their cellular tissues (Schnitzer, 1991). Of all SOM, polysaccharides are most readily available to microorganisms as sources of food and energy (Schnitzer, 1991). The significance of carbohydrates in soils arises largely from the ability of complex polysaccharides to bind inorganic soil particles into stable aggregates. Other soil properties affected by polysaccharides include cation exchange capacity, biological activity and metal complexation. In certain

submerged soil, the production of microbial gums and mucilage may lead to an undesirable reduction in permeability, because of blocking of soil pores (Stevenson, 1986).

Nearly all of the nitrogen (N) found in soils is combined or closely associated with SOM. The main N-forms present in SOM are; protein-N, peptide-N, amino acids-N, amino sugar-N, and $\text{NH}_3\text{-N}$ (Schnitzer, 1991).

From 1.2 - 6.3% of organic matter in most soils occurs as fats waxes and resin, with somewhat higher values being reported for acid forest soils and peats (Stevenson, 1986). Fats, waxes and resin usually resist biodegradation due to their hydrophobic properties and, as a result, have a long residence times in soils (Jambu *et al.*, 1978 as cited by Schnitzer, 1991).

Between 20 and 80% of the total soil P in most surface soils may be in organic forms. Prominent organic forms exists mainly as complex organic compounds containing phosphate-ester linkages in which the phosphorus (P) is bonded to carbon (C) via oxygen, such as in nucleotides and nucleic acids (Stevenson, 1982).

Another group of non-humic substance is organic sulfur compounds. Over 90% of the total sulfur (S) in most non-calcareous soils is present in organic forms (Schnitzer, 1991). Freney (1986) has divided the organic S compounds into 2 broad categories — HI reducible organic S and C-bonded S. The HI reducible organic S accounts for 30 - 70% of the organic S in soils and is considered to include the most labile forms of organic S, largely ester sulfates, while C-bonded S accounts for 40 - 60% of organic S (Freney, 1986).

2.2.2 Humic Substances

These are the dark coloured, amorphous macromolecules synthesised within the soil as a result of biological and abiotic chemical and physical processes. They constitute the bulk of SOM (50 - 80%) and are known to range in molecular weight from a few hundred to several thousand (Schnitzer and Khan, 1972).

The content of humic substances in SOM has been estimated at about 65% (Schnitzer, 1991). Humic substances are widely considered to represent the most stable fraction of SOM, a property that has been attributed to their chemical structures, their entrapment within soil aggregates, and their interactions with metal cations and clay minerals (Theng *et al.*, 1989).

Based on their solubility in alkali and acid, Schnitzer (1991) stated that humic substances can be partitioned into the following three main fractions:

1. Humic acid — which is soluble in dilute alkali but is coagulated by acidification of the alkaline extract.
2. Fulvic acid — which is that humic fractions remaining in solution when the alkaline extract is acidified, i.e. soluble in both dilute alkali and dilute acid.
3. Humin — which is that humic fraction that cannot be extracted from the soil by dilute base or acid.

Stevenson (1986) stated that all soils would be expected to contain a broad spectrum of humic substances. However, distribution patterns will vary from soil to soil and with depth in the soil profile. The humus of forest soils (Alfisols, Spodosols and Ultisols) is characterised by a high content of fulvic acids; that of peat and grassland soils (Mollisols) contains high amounts of humic acids.

2.3 Formation of Soil Organic Matter

Although many different types of reactions can lead to the production of dark-coloured pigments (e.g. humic acids), the main pathway appears to be through condensation reactions involving polyphenols and quinones (Stevenson, 1982). Polyphenols derived from plants (e.g. lignin), or synthesised by microorganisms, are enzymatically converted to quinones, which undergo self-condensation or combine with amino compounds to form N-containing polymers (Stevenson, 1982). The role of microorganisms as sources of polyphenols has been emphasised by Kononova (1966), who gave a detailed account of research in which histological microscopic techniques and chemical methods were used to study the decomposition of plant residues. She suggested that the stages leading to the formation of humic substances were as follows:

Stage 1 : Fungi attack simple carbohydrates and parts of the protein and cellulose in the medullary rays, cambium and cortex of plant residues.

Stage 2 : Cellulose of the xylem is decomposed by aerobic myxobacteria. Polyphenols synthesised by the myxobacteria are oxidised to quinones by polyphenoloxidase enzymes, and the quinones subsequently react with N compounds to form brown humic substances.

Stage 3 : Lignin is decomposed and phenols released during decay are source materials for humus synthesis.

The C compounds from decomposing plant material are then incorporated into the humic substances. Sorensen (1963) fractionated the SOM after allowing ^{14}C -labelled barley straw to decompose in soil and studied the distribution of ^{14}C in fractions of the SOM. He found that the percentage of ^{14}C in each fraction was : 11.8 - 18.9% in fulvic acid, 19.5 - 26.4% in humic acid, 49.4 - 54.7% in humin. A summary on the formation of SOM prepared by Flaig (1984) is shown in Figure 2.1.

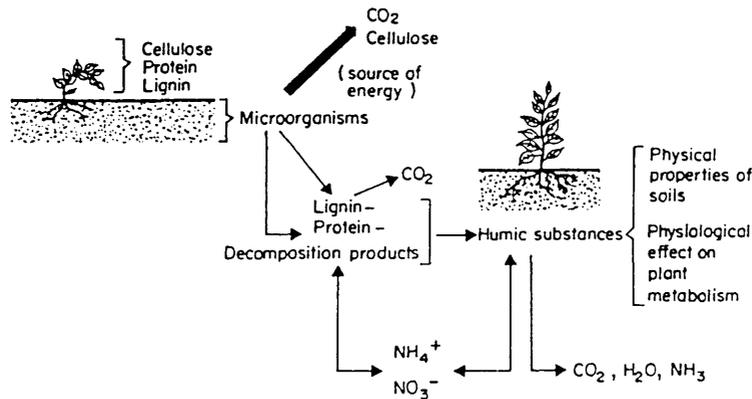


Figure 2.1 : Steps in the formation of SOM (Flaig, 1984)

2.4 Functions of Soil Organic Matter

The main functions of SOM are considered in terms of effects on soil physical, soil chemical and soil biological properties.

2.4.1 Soil Physical Properties

The effect of SOM on soil physical properties has been well established. Organic matter binds soil into aggregates, giving rise to soil structure and associated soil porosity, which are important properties in regard to root proliferation, gas exchange, and water retention and movement (Syers and Craswell, 1995). Humic substances and polysaccharides produced by microorganisms during the decomposition processes in soil promote and stabilise soil aggregation (Swift, 1991). The increase in fungal hyphae and bacterial biomass resulting from the increased microbial food supply also enhances aggregate stabilisation (Kladivko, 1994).

Crop residue left on the surface of the soil, and the subsequent humification of these materials, have numerous beneficial effects on the soil physical conditions. Sutrisno *et al.* (1995) found that using plant residue as mulches, in combination with terracing in slope areas in Indonesia, reduced the runoff and soil loss significantly, with an increase in water infiltration into soil. The effect of organic mulches in reducing erosion can be attributed to reducing the direct impact of raindrops, maintaining maximum soil infiltration and decreasing the quantity, velocity and transport capacity of runoff water (Gabriels and Michiels, 1991).

The other beneficial effects of the organic matter on soil physical properties include reductions in soil crusting (Gabriels and Michiels, 1991), bulk density (Black, 1973), soil temperatures (Horton *et al.*, 1994), and compaction (Gabriels and Michiels, 1991). Organic matter also tends to increase the volume of large transmission pores in the soil and, therefore, tends to increase saturated hydraulic conductivity (Lal, 1986).

2.4.2 Soil Chemical Properties

SOM arguably exerts its largest effect on chemical properties through direct and indirect effects on nutrient supply. The direct effect is that SOM is a major source of N, S, P and many micronutrients in soils (Allison, 1973; Stevenson, 1986; Jenkinson, 1988).

An indirect effect of SOM on nutrient supply is its positive role in enhancing cation and anion exchange capacity. This is particularly important in sandy soils where organic matter is the most important contributor to soil cation exchange capacity. Willett (1995) concluded that, for sandy soils in Northeast Thailand, SOM was an essential component for the provision of cation-exchange sites and buffering capacity, rather than just a source of nutrients which are produced by its decomposition.

The complexing ability of SOM for certain metals is well established (Jones and Jarvis, 1981). The ability of added organic matter to reduce exchangeable aluminium has been demonstrated by Hargrove and Thomas (1981); as has its ability to complex monomeric Al by Bell and Edwards (1987).

2.4.3 Soil Biological Properties

SOM stimulates the activity of fauna and microorganisms in soil, which contribute to nutrient release during the decomposition of plant and animal residues, and to synthesis of humified compounds (Syers and Craswell, 1995). The bulk of the active microbial and grazing faunal biomass operates in the active organic pool usually associated with fresh additions of plant residues (Lee and Pankhurst, 1992). Thus, any increase in the return of above or below ground residues within the soil should increase populations of soil fauna.

The fact that organic matter inputs can stimulate soil microbial biomass is shown by the work of Saffigna *et al.* (1989) in Queensland, Australia. The return of sorghum residues over a five year period increased biomass C by approximately 15% whereas total organic C increased by only 9%. The importance of fauna activity, stimulated by the addition of organic materials, in promoting aggregation and in enhancing water movement and aeration in soils has been reviewed by Hamblin (1985). Tian *et al.* (1993) also found that plant residues applied as mulch significantly increase earthworm or termite populations in soil. Soil organic matter creates an especially favourable habitat for soil arthropods and other invertebrates by increasing soil water content, modifying soil temperatures and providing a continuous food source (Burton and Burd, 1994).

Increasing the quantity of soil microbial substrates by the addition of crop residues may also benefit the soil flora. Dalal *et al.* (1991) studied the influence of residue management practices over a 20 years period on a Vertisol in subtropical Australia. They found that retention of residues of wheat or barley in combination with no-tillage and annual application of N fertiliser increased microbial biomass N

in the surface layer (0 - 25 mm), and residues combined with fertiliser had a similar effects in the 0 - 100 mm layer.

In addition, other workers (Hesse, 1984; Stevenson, 1986) reported that humic substances extracted from manures also increase the efficiency of N-fixing organisms like *Rhizobium* and *Azotobacter*.

2.5 Factors Affecting Levels of Organic Carbon in Soil

The level of organic matter in soils is influenced by climate, topography, parent material, vegetation, organisms, and time (Jenny, 1941; Stevenson, 1986). Jenny (1941) arranged the order of importance of these factors as climate > vegetation > topography = parent material > age. Organic matter in soil reaches a stable equilibrium when all factors, with the exception of time, change very little. The time needed for organic matter to reach a stable level in uncultivated soil can range from less than 100 to over 2000 years, depending upon climate conditions (Stevenson, 1986).

Jenny (1941) made some general statements about organic matter levels in virgin grassland soils that:

1. Grasslands soils have higher organic matter content than forest soils.
2. Organic matter content increases with increasing precipitation and decreases with increasing temperature
3. Fine-textured soils have higher organic matter content than coarse-textured soils.
4. Naturally moist and poorly drained soils have higher organic matter than well-drained soils.
5. Soils in lowlands have higher organic matter than soils on upland positions.

All of the above are affected when virgin land is cultivated. Rasmussen and Collins (1991) ascribe these changes to:

- 1) A much larger portion of biomass is removed for use as food, feed and fuel.
- 2) Crop selection, crop rotation, and residue utilisation influence the amount of biomass cycling in the ecosystem.
- 3) Accelerated wind and water erosion significantly modify the soil surface and alter organic matter accumulation, soil texture, and water holding capacity.
- 4) Soil aeration may be changed by drainage and tillage
- 5) Tillage to control weeds or prepare a seedbed increases soil disturbance and accelerates organic matter oxidation.

The following sections review the effect of four major factors resulting from cultivation on the content of SOM. The four factors are forest clearing and subsequent cropping, crop rotation, tillage and plant residue management.

2.5.1 Effects of Forest Clearing and Subsequent Cropping

Virgin grassland soils lose organic matter rapidly after they are first cultivated (Mann, 1986; Cerri *et al.*, 1991). Cultivation of a soil previously supporting native vegetation or pasture generally leads to reduced organic matter, organic C (Lefroy *et al.*, 1993), N (Ihori *et al.*, 1995), organic P (Tiessen *et al.*, 1992), deterioration of soil structure (Bauer and Black, 1981) and lower biological activity (Khan, 1970).

Dalal and Mayer (1986) showed in their study in the wheat belt in southern Queensland, Australia that soils which had been cropped for a period of 20 - 70 years suffered a decline in SOM, with particular reductions in total organic C, organic C in the light fractions, total N and mineralisable N. Other soil properties, including bulk density and extractable manganese, were also significantly affected by cultivation. This agrees with the observations of many workers, such as Ayanaba *et al.* (1976), Monreal and Janzen (1993), Arrouays *et al.* (1995), and King and Campbell (1994).

Cultivation results in a decline in organic matter due primarily to: 1) changes in soil temperature, moisture fluxes and aeration, 2) exposure of new soil surfaces resulting from aggregate disruption, and 3) reduced addition of organic materials, and frequently to increased soil erosion (Dalal and Mayer, 1986). The loss of SOM with cultivation is usually exponential, declining rapidly during the first 10 - 20 years, then more slowly, and finally approaching a new equilibrium in 50 - 60 years (Jenny, 1941; Arrouays *et al.*, 1995), however the time scale varies with climate and soil type.

Apart from this, site factors (e.g. climate and slope) also have additional effects to that of the forest clearing which enhance the depletion of SOM (Lugo *et al.*, 1986). Brown and Lugo (1990) studied the effect of forest clearing and succession on soil C content in different water regimes i.e. wet zone (more than 2,000 mm of rain per year); moist zone (1,250 mm per year), and dry zone (less than 750 mm per year). They reported that C contents to 50 cm depth of cultivated sites, expressed as a percentage of the forest soil, were 44% in the wet zone, 31% in the moist zone and 86% in the dry zone. They also found the decline in soil C (to a depth of 1 m) upon conversion to agriculture, which they suggested was due to reduction of organic input and a continuation of microbial activity and/or leaching.

Cultivation has also been found by van der Pol and Traore (1993) to reduce mineral nutrients such as N, P, K, Ca and Mg, in soil. Tiessen *et al.* (1992) studied the effects of shifting cultivation on SOM dynamics and reported that, as well as reducing organic C and N content, conversion from natural vegetation to cultivated land resulted in a significant decrease in soil mineral N content (NO_3^- and NH_4^+), but this could be recovered by increasing the length of fallow periods.

2.5.2 Effects of Cropping Systems

The effects of different cropping systems on SOM has been widely reported. Franzluebbers *et al.* (1994) studied the changes in soil organic C and N pools in wheat rotation systems and found that increasing cropping intensity (number of crops per unit of time) resulted in an increase in soil organic C, organic N, mineralisable C and N and biological properties. They suggested that this was due to an increase in the amount of plant residues added to the soil surface. Reduced cropping intensity or, in turn, increased fallow periods have marked effects on losses of C and N because of reduced vegetative

production and greater opportunity for both soil erosion and microbial oxidation of SOM (Biederbeck *et al.*, 1984). Similar findings were reported by Wood *et al.* (1990) and Robinson *et al.* (1996).

Inorganic fertiliser applied to the crop or cropping systems has been found to enhance the level of SOM. Ladd *et al.* (1994) reported that organic C and total N concentrations increased linearly with increasing N fertiliser rates applied to crop rotation systems. Liang *et al.* (1996) also reported that high fertiliser rates produced significantly greater levels of soil C and N than lower rates. The primary effect of inorganic fertiliser addition is to increase vegetative production and the amount of organic C available for recycling back into the soil system. This agrees with the observations made by many workers (Barber, 1979; Liang and Mackenzie, 1992; Rasmussen and Parton, 1994).

Generally, it can be concluded that rotations that return more organic matter to the soil over a run of years, tend towards a higher equilibrium SOM value than rotations that return less. Not surprisingly, organic matter declines most rapidly under bare fallow, when returns of plant material are very low or zero.

2.5.3 Effects of Organic Materials

The organic materials that have been widely used to improved SOM are crop residue, animal manure and green manure. There is now increasing interest in the use of urban and industrial wastes for this purpose.

a) Plant and Crop Residues

i) Type of Plant and Crop Residue

Residue application has been reported to raise SOM levels (Rasmussen *et al.*, 1980) and increase the formation and stability of aggregates (Rasmussen and Collins, 1991). However, in many cases, SOM content is only weakly related to the type of residue applied. Larson *et al.* (1972) found that alfalfa, cornstalks, oat straw, sawdust, and bromegrass produced similar increases in organic C in a Hapludoll in Iowa. The influence of residue type on organic N was similar but more variable.

Similar results were reported by Sowden (1968) who found little differences on SOM level between the incorporation of straw, alfalfa and deciduous leaves into soil over 20 years in Canada. They all prevented further decline in organic matter and N in a clay soil and increased levels in a sandy soil. This is in agreement with the findings of Almendras and Serohijos (1995) who found no difference between *Gliricidia sepium* and *Acacia auriculiformis* in raising soil organic C levels despite the different quality of these residues.

However, the type of plant residue has been found to affect the level of soluble C leached from decomposed residue. Harris and Safford (1996) found higher amounts of soluble C in leachate collected from maple leaves throughout the 6-month sampling period (July to December), compared with that found in leachate from pine and spruce, which decompose more slowly. They further suggested that the difference in levels of soluble C leached between species was due to their chemical composition. Maple leaf contains higher levels of water-soluble organics and less lignin, cellulose and

hemicellulose than pine and spruce leaves. The effect of chemical composition of plant residue on the decomposition rate is extensively reviewed in section 2.9.1.a.

ii) Residue Management

Generally, the rate of residue addition, its placement, the use of burning, and the amount of crop residue removed or returned, all affect the level of SOM.

Rate of Residue Addition : Several studies show that organic C and N in soil respond linearly to increasing the rate of residue addition. Horner *et al.* (1960) summarised some of the early research in USA, which found linear increases in soil C and N with rates of residue application from 0 to 3.5 t ha⁻¹. Barber (1979) reported the effect of residue addition rate on SOM which showed that doubling the amount of residue return in continuous corn production over 11 years resulted in a significantly increased level of SOM compared to the normal rate of return.

Residue Burning : Crop residue burning has long been used as a method of residue management and as the major method of straw disposal in many countries (Tanaka, 1978). Biederbeck *et al.* (1980) reported that the dry matter yield of wheat straw was reduced to 33% by burning, while a study by Rasmussen *et al.* (1980), which directly measured C loss, indicated that 35% remained after burning of wheat straw.

Annual burning of residues not only reduces SOM, but can also reduce microbial activity and, consequently, lead to the immobilisation of applied N (Boerner, 1982). Rasmussen *et al.* (1980) indicated that repeated burning decreased microbial SOM and microbial activity and Biederbeck *et al.* (1980) suggested an immediate decline of 95% in the fungal population and 70% in the bacterial population in the burnt treatments at one location.

Collins *et al.* (1992) found that burning of residue resulted in the decline of soil microbial biomass C and N. This is probably due to the smaller amount, compared to unburnt residue, of organic material returned to the soil and limited availability of mineralisable C to the microbial biomass. In addition, Ladd *et al.* (1994) found that after 3 years of residue management in wheat rotation systems, organic C and total N contents were not significantly different in plots with residue retained on the soil surface and incorporated into soil; but both were higher than in plots where residues were burned.

Burning also results in direct loss of nutrients. Burning of grasslands and wheat residues is estimated to result in a loss of between 50 to 70% of the residue N (Biederbeck *et al.*, 1980), while Lefroy *et al.* (1994) recorded losses of between 40 and 60% of S and 65% and 70% of N, when rice straw was burnt.

Residue Removal : Crop residues are used for fuel or animal feed in many areas of the world. This is potentially a major C and nutrient loss pathway. A 5 t crop of paddy (rice in the husk) removes about 2000 kg C, 150 kg N, 20 kg P, 150 kg K and 20 kg S in the paddy and straw (Ponnamperuma, 1984). Residue removal decreases C input into soil and thus lowers organic matter levels. Barber (1979) found that when the corn residue was removed for 10 years the organic matter level was 0.3% less than when

the residue was returned. The removal of surface crop residue was also found by Doran *et al.* (1984) to seriously reduce crop productivity.

b) Animal Manures

In general, manure contains about 500 g dry matter per kg and 150 g C and 11 g N per kg, with very high variation from animal species to species and year to year (Rasmussen and Collins, 1991). Thus, manuring can also affect the level of organic matter in soil. Allison (1973) mentioned that the increase in SOM from farmyard manure depended on the quality and amount of feed, and the age and the kind of animal.

In a long term experiment at the Rothamsted Agricultural Experiment Station, it was shown that when farmyard manure, containing 3 t C ha⁻¹, was applied annually, the C content of the soil more than trebled in 125 years, although equilibrium had still not been reached (Jenkinson, 1988). In addition, humic material derived from manure gives greater stability to soil (Jenkinson, 1988). Similar results were reported by Liang *et al.* (1996) who observed that in soil receiving 160 g C m⁻² of manure annually for 4 years, soil organic C increased significantly with greater stability in soil. In addition, Liang and Mackenzie (1992) concluded that crop production at an intensive level with manure application for 6 years, increased soil organic C significantly above the initial C level, but with no obvious effect on soil total N. Results from these workers were also supported by Rose (1991).

Collins *et al.* (1992) concluded that incorporation of barnyard manure produced higher microbial biomass than did the application of inorganic fertiliser. Animal manure supplies additional mineralisable C and N, that directly stimulates microbial activity and growth (McGill *et al.*, 1986). Rasmussen and Collins (1991) report that the effect of manure on organic C and N appears to be the same as that of legume and straw residues.

c) Green Manure

Green manure has been widely used as a sources of N for crops (Bin, 1983; Bouldin, 1988; Garrity and Flinn, 1988; Ladha *et al.*, 1988). In tropical areas the most common species used to improve soil fertility have been *Sesbania rostrata*, *Gliricidia sepium*, *Leucaena leucocephala* and *Azolla* (Li, 1984; Arunin *et al.*, 1988; Brewbaker and Glover, 1988; Rosegrant and Roumasset, 1988). However, green manures have generally been less effective than crop residues or animal manures in stabilising or increasing SOM levels (Pinck and Allison, 1951; Bin, 1983; Hesse, 1984). The primary function of green manure is to sequester biologically fixed N in sufficient quantity to meet the requirements of the following crop. As such, vegetation is normally incorporated into soil before it is mature, and as a result, does not have the same chemical composition (i.e. lignin) that is found in mature plants. Lignin content has a large influence on the stabilisation of C and N in recalcitrant soil fractions (Stevenson, 1986). In addition, ploughing in a green manure crop involves an extra mixing of the topsoil which leads to an accelerated decomposition of organic matter (Hesse, 1984).

d) Urban and Industrial Waste

Gabriels (1988) and Gabriels *et al.* (1989) incorporated industrial composts (paper mill sludge mixed with bark, and household waste) at a rate of 30 t ha⁻¹ in the upper 10 cm of a silt loam soil, and

reported that after 1 year SOM was increased by 1%. However, he suggested that care should be taken with successive additions of sludge, since this could result in an accumulation of heavy metals in the soil.

2.5.4 Effects of Tillage

In studies of the effect of tillage on SOM, the comparison is generally between conventional tillage and conservation tillage. Conservation tillage is described as non-inversion tillage that leaves a significant fraction of crop residue on, or only shallowly incorporated into the soil. This is done to control erosion, reduce energy use and conserve soil and soil water (Unger and MaCalla, 1980). Stubble mulch, eco-fallow, no-till, direct drilling and trashy-fallow are all forms of conservation tillage.

Many studies have shown that conservation tillage increases organic carbon and nitrogen in the top 5 - 15 cm of soil compared to conventional methods of tillage (Balesdent *et al.*, 1990; Buchanan and King, 1993; Reddy *et al.*, 1994). In general, the increase averages from 1 - 2% per year for both C and N (Rasmussen and Collins, 1991). Apart from C and N, conservation tillage has also been found to enhance the concentration of extractable P, extractable K and soil biological properties (White, 1990; Franzluebbers *et al.*, 1994).

Increased levels of C and N near the soil surface resulting from conservation tillage have been ascribed by Parr and Papendick (1978) to delayed residue decomposition, slower oxidation of soil C, reduced erosion, or any combination of these factors.

Effect of delayed tillage on SOM have also been reported. In the delayed tillage system, normal tillage is moved from the summer (after harvest), when soil temperatures are high, to the fall or spring, when the temperature is much lower. Unger (1982), as cited by Rasmussen and Collins (1991), reported that SOM in a wheat-fallow system in Texas after 36 years was 27% higher with delayed tillage compared to tillage immediately following wheat harvest. This is due to summer cultivation substantially stimulating oxidation rates, especially in those areas where summer rain is sufficient to moisten the soil following harvest.

From this section, it can be concluded that a decline in SOM almost inevitably accompanies the introduction of arable agriculture. Yet, despite this decline in organic matter, good yields are still obtained - provided the necessary fertilisers are applied each year (Sanchez *et al.*, 1987). This is not the case in many parts of the tropics, where production cannot be maintained under continuous cropping, and recourse must be made to shifting cultivation, with its attendant build-up of organic matter and soil nutrients during the fallow stage, if yields are to be maintained at acceptable levels.

2.6 Soil Organic Matter and Sustainable Agriculture

As world population and economic growth increases, there is an increased demand for food. To meet this increased demand with little increase in yield per unit area, an increase in the area of agricultural land is required. This forces the expansion of agricultural area into the forest or native grassland areas, and increase in cropping intensity. Alexandratos (1988) stated that by the year 2000, arable land expansion and cropping intensity increases are projected to account for around two-fifths of

the growth in production. As mentioned in the previous section, conversion of forest or native grassland to agricultural lands is one of the major reasons for land degradation, such as the decline in organic matter level. The decline in organic matter is the major reason for decreasing crop production in many areas, especially in the tropics, where the rate of decomposition is normally higher. As SOM levels decrease, erosion increases, compaction increases, water infiltration decreases, water holding capacity of soil decreases, soil fertility decreases and crop growth decreases (Wallace, 1994c). As organic matter and soil fertility decline, yields can be maintained only by the significant input of inorganic fertiliser. Sanchez *et al.* (1987) reported that in the medium-input systems developed in tropical America, yield was maintained at a reasonable level ($7.8 \text{ t grain ha}^{-1} \text{ yr}^{-1}$) over a 8 year period by regular addition of fertiliser and lime. Without fertiliser inputs crop yields were zero after only three crops.

In addition to deforestation, inappropriate land use and soil mismanagement cause soil degradation in many agricultural areas of the world. Soil degradation, caused by land misuse, ecologically incompatible farming systems and inappropriate soil management practices, can be a major cause of fertility and organic matter depletion from soil (Lal *et al.*, 1995). The World Resources Institute (1992) from the United Nations reported on the status of the soils of the world and contains some alarming conclusions;

1. Some 10 million ha of the best farm lands of the world have been so ruined by human activity since World War II that it is impossible to reclaim them.
2. Over 1.2 billion ha have been seriously damaged and can be restored only at great cost.
3. According to the report, this loss in soil capability could mean that there will be an enormous shortage of food in 20 to 30 years, and that poorer people will suffer.
4. Two-thirds of the seriously eroded land is in Asia and Africa. About 25% of the cropped land in Central America is moderately to severely damaged. In the USA and Canada, damaged land accounts for 4.4% of the total.
5. Food production has declined in 80 developing countries in the past decade, in spite of the green revolution. Soil degradation is the major factor.
6. In the rich prairie soils of North American, moderate damage from wind and water erosion continues.
7. Practices which are responsible for damaged soil include too much fertiliser that acidifies soil, failure to protect cultivated hillsides and fallow land from erosion, improper irrigation and drainage practices, which lead to salinity and waterlogging, deforestation and cultivation of drought-prone desert land.

Wallace (1994a) listed eight major mechanisms which may result in significant soil improvement. The aim of many of these mechanisms was to increase SOM, resulting in massive improvements in biological, physical and chemical properties. Wallace (1994c) suggested that truly sustainable agricultural systems will be achieved when SOM levels are restored to their original pre-cultivation amounts, or greater (Wallace, 1994c). In practice, this is difficult to achieve. It may be

possible in some agricultural systems, where the return of organic material from the system is far greater than in the natural system, such as sugarcane plantation and some other intensive agricultural systems. Even if significant increases in SOM are not achievable, many workers agree that agriculture will not be sustainable as long as SOM levels are on a continuous downward trend (Wallace, 1994a; Blair, *et al.*, 1995c). One key way to improve SOM is to maximise the return of organic materials (Gupta *et al.*, 1994; Wallace, 1994b; Blair *et al.*, 1995c). It has been estimated that an annual return of only 5 t ha⁻¹ of crop residues or manure, even with standard tillage, could keep soils in equilibrium with present levels of SOM (Follet *et al.*, 1987).

2.7 Sources of Organic Materials

2.7.1 Animal Waste

Animal manures are a good source of nutrients and organic matter. The excretion of dung and urine from domestic animals such as cattle, sheep, goats, pigs and poultry has the potential to provide enormous quantities of plant nutrients. For examples, Gaur (1980) reports that 15 tons N and 4 tons each of P and K are potentially available from 1000 tons of fresh cattle dung. For those who have access to the more exotic fauna, fresh elephant dung contains 1.3% N, 0.3% P and 0.1% K, with the C:N ratio of 43; fresh tiger dung has 2.8% N, 3.2% P and 0.03% K, with C:N ratio of 10; and fresh lion dung contains 3.6% N, 3.2% P and 0.04% K, with C:N ratio of 9 (Hesse, 1984).

Some countries, like China, take full advantage of such potential for recycling plant nutrients (Yuan, 1984). Apart from animal manure, slaughterhouse waste, such as bones, blood and skin, also have potential for providing plant nutrients.

However, farmers usually have access to relatively small amounts of farmyard manure. At best, farmyard manure can be used to improve soil fertility in small areas of the farm (Wonprasaid *et al.*, 1995). However, due to great diversity in the components, difficulties are encountered in making decisions for application (Yuan, 1984). In addition, proper storage, handling and application are necessary to reduce undesirable odour and nutrient losses, via runoff and leaching (Radke and Honeycutt, 1994).

2.7.2 Crop Residues

Crop residues include such materials as straw, stalks, husks and leaves. Some typical nutrient concentration figures are 0.5% N, 0.3% P and 1.2% K for cereal straw (Gaur, 1980); 0.7% N, 0.1% P and 1.4% K for maize stover; and 0.6% N, 0.1% P and 0.4% K for rice chaff (Yamamoto and Teramoto, 1978). The enormous quantity of straw production each year make it very promising for use as a source of SOM. Ranjhan (1986) estimates that more than 100 million tons of rice straw alone are produced each year in Southeast Asia and Ponnampereuma (1984) estimated that over 600 million tons are produced throughout the world. Crop residues can be recycled in three ways: composting, incorporation into soil or mulching (Ponnampereuma, 1984; Radke and Honeycutt, 1994).

Another major source of SOM are legume residues. Legume residues have considerably more potential for quality, due to its high N content, but much less for quantity. One major source of legume

material for incorporation into soil are perennial shrub or tree legumes, grown either adjacent to or within the crop area (Blair *et al.*, 1990). Tropical leguminous trees are found on every continent, and have been used to various degrees as fodder, fuel wood or organic matter resources, for many years in Africa, Asia, Australia and Americas (Van den Beldt and Huxley, 1982). Some species are now widely distributed and used outside their country of origin. Notable examples include *Leucaena* spp. and *Gliricidia sepium*, which originated from Central America and are now important components of agricultural systems in many countries of Asia and Africa (Brewbaker and Styles, 1982; Sumberg, 1985).

Recently NFTA (Nitrogen Fixing Tree Association) has popularised a valuable species of tree legume, namely *Flemingia macrophylla* (Budelman, 1989). The slow decomposition rate of its leaves, along with its dense growth and a moderate drought tolerance, make *Flemingia macrophylla* especially useful for mulching and soil protection (Budelman, 1989). *Flemingia macrophylla* is now increasingly used for soil conservation (Sutrisno *et al.*, 1995), as well as a SOM source (Budelman, 1988; Kachaka *et al.*, 1993; Almendras and Serohijos, 1995).

Appropriate plant species and management practices are necessary for successful utilisation of crop residues as a source of organic matter. The particular species and management practices need to be identified for different agroecosystems.

2.7.3 Green Manures

Legume green manure plants are grown to fix atmospheric N before being incorporated into soil. *Crotalaria* spp. contain, on average, about 4% N, *Sesbania* spp. 3% N and *Azolla* 4-5% N (Hesse, 1984). It is believed that this beneficial effect is gained by the current crop only and that there is little residual or cumulative benefit. Detailed species descriptions of many green manures are contained in Brewbaker and Glover (1988). Non-legume plants are sometimes used as green manure, such as water hyacinth, (Roger and Watanabe, 1984).

2.7.4 Urban and Industrial Waste

Human body wastes have greater potential than animal wastes for use as organic amendments. An average analysis of sewage has been given as 50 ppm N, 7 ppm P and 25 ppm K (Gaur, 1980), while another calculation indicates that about 2 kg N, 0.2 kg P and 0.8 kg K (Takahashi, 1978) are potentially available per year from one person.

Urban waste has considerably more potential in both quantity and nutrient content. In Germany, a computer analysis by Bassam and Thorman (1980) indicated that the N, P and K content of urban wastes exceeded the amount of chemical fertiliser used annually. Bastian (1977) reported that sludges produced at municipal waste water treatments plants in the USA amounted to about 4.5 million dry tons annually in 1977, and this figure was expected to reach 8 million by 1984 (Kurihara, 1984).

However, waste may contain large amounts of heavy metals, pest or diseases vectors, and/or toxic chemicals (Radke and Honeycutt, 1994). Thus, care should be taken with successive additions of sludge, since this could result in an accumulation of heavy metals, as well as pathogens in the soil (Kurihara, 1984).

2.8 Decay Processes of Organic Materials in Soil

Stevenson (1986) described the decomposition of plant materials in soil as constituting a basic biological process, in that C is circulated to the atmosphere as carbon dioxide (CO_2), N is made available as ammonium (NH_4^+) and nitrate (NO_3^-), and other associated elements (P, S and various micronutrients) appear in forms required by higher plants. In the process, part of the nutrients are assimilated by microorganisms and incorporated into microbial tissues (the soil biomass). The conversion of C, N, P and S to mineral forms is called mineralisation; the reverse process is called immobilisation (Figure 2.2).

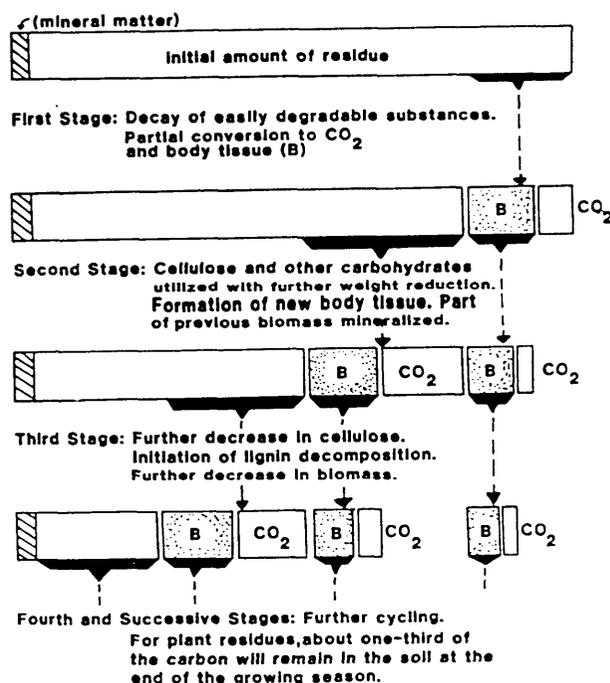


Figure 2.2 : Stages in the microbial decomposition of organic residues in soil. The letter B refers to the biomass. Each stage involves partial conversion of C to CO_2 and synthesis of microbial tissue (Stevenson, 1986)

The photosynthetic process is of primary importance in providing raw material for humus synthesis. By using solar energy, plus nutrients derived from the soil, higher plants produce lignin, cellulose, protein and other organic substances that make up their structures. Because of the complex nature of plant material, numerous species of microorganisms are involved in the decay process and several stages can be delineated in the decay (Stevenson, 1986).

Earthworms and other soil animals play an important role in reducing the size of plant material; further transformations are carried out by enzymes produced by microorganisms (Tian *et al.*, 1995a). The initial phase of microbial attack is characterised by rapid loss of readily decomposable organic substances and the production of by-products included NH_3 , hydrogen sulfide, CO_2 , organic acids and other incompletely oxidised substances. In subsequent phases, organic intermediates and newly formed biomass tissues are attacked by a wide variety of microorganisms, with production of new biomass and further C loss as CO_2 . The final stage of decomposition is characterised by gradual

decomposition of the more resistant plant parts, such as lignin and polyphenols, in which the actinomycetes and fungi play a major role (Stevenson, 1986; Jenkinson, 1988).

During decomposition by microorganisms, some of the C is released to the atmosphere as CO₂ and the remainder becomes part of the SOM. Part of the native humus is mineralised concurrently (Stevenson, 1986).

2.9 Factors Affecting Plant Residue Decomposition

Couteaux *et al.* (1995) stated that plant residue decomposition involves two simultaneous and fundamental sets of processes: 1) the concomitant mineralisation and humification of C compounds by microorganisms; and 2) the leaching downward in the soil of soluble compounds, whose C and N are progressively mineralised and immobilised. These processes are controlled by 3 main factors: climate, plant residue and its management, and abundance of decomposing organisms. Climate is the dominant factor in areas subjected to unfavourable weather conditions, whereas plant residue largely prevails as the regulator under favourable conditions (Couteaux *et al.*, 1995). Other complementary factors that have an effect on breakdown rate of residue are soil texture, plus effects of elevated O₃ and/or CO₂.

2.9.1 Effects of Plant Residue and Its Management

Effects of plant residue and its management on decomposition are usually considered in terms of quality of residue, fertiliser application, quantity of residue added and accessibility.

a) Effects of Plant Residues Quality

Plants contain 15 - 60% cellulose, 10 - 30% hemicellulose, 5 - 30% lignin, 2 - 15% protein, and soluble substances, such as sugars, amino acids, amino sugars and organic acids, which can constitute 10% of the dry weight (Paul and Clark, 1989). They also contain cutin (Gallardo and Merino, 1993), polyphenols (Tian *et al.*, 1995b) and silica (Goering and Van Soest, 1970). Plant residue decomposition and nutrient release are known to be affected by their chemical composition such as C:N ratio (Kimura *et al.*, 1990; Vanlauwe *et al.*, 1995), lignin content (Upadhyay *et al.*, 1989; Taylor *et al.*, 1989), polyphenol content (Palm and Sanchez, 1991; Oglesby and Fownes, 1992), nutrient content (Stewart *et al.*, 1966), plus cutin and litter toughness (Gallardo and Merino, 1993). Each of these factors are reviewed in detail below.

i) Effect of N Concentration and C:N Ratio

Several workers have searched for a general litter-quality predictor of mass loss. In many studies, litter N content or C:N ratio, have been shown to be of critical importance in litter decay (Iritani and Arnold, 1960; Fogel and Cromack, 1977; Yaacob and Blair, 1980; Cotrufo *et al.*, 1994). Taylor *et al.* (1989) pointed out that C:N ratio appeared to be a better predictor than lignin:N in two situations where substrates were low in lignin, and where a wide range of lignin contents were being considered. Generally, it has been found that plant residues with high N content show high decomposition rates and nutrient release (Janzen and Kucey, 1988; Douglas and Rickman, 1992). Frankenberger and Abdelmagid (1985) and Melillo *et al.* (1982) reported high correlation between N content, N release and

biomass loss. These are due to the high N content in residue reducing the competition of available N by microorganisms and consequently enhancing the decomposition by maintaining high microbial activity.

Parnas (1975) studied the relationship between N immobilised and C:N ratio, and concluded that N immobilisation occurs if the plant residue has a C:N ratio of more than 30. In addition, Poincelot (1975), as cited by Mote and Griffis (1980), undertook a composting study and concluded that the relationship between the dry weights of carbon and N were the major factors controlling the rate of decomposition. He further stated that a C:N ratio of 30 would seem to be the most desirable and that ratios between 26 and 35 have been observed to provide rapid and efficient composting. Allison and Klien (1962) found that the addition of plant material with a C:N ratio greater than 25 resulted in nutrient immobilisation and resulted in the production of ammonia.

ii) Effect of Lignin Content

The role of lignin as an inhibitor in the decomposition process has been elucidated in several studies (Meentemeyer, 1978; Berendse *et al.*, 1987; Fox *et al.*, 1990). Lignin is known to be a recalcitrant substance, highly resistant to microbial decomposition (Melillo *et al.*, 1982). There are relatively few microorganisms that can degrade lignin and they are exclusively aerobic (Jenkinson, 1988). Many workers found that increasing lignin concentration reduces the decomposition rate and nutrient release from plant residues, and also enhance nutrient immobilisation, especially of N (Aber and Melillo, 1982; Aber *et al.*, 1990; Tian *et al.*, 1992a). Fogel and Cromack (1977) have shown that lignin concentration of the substrate is an excellent index to use for prediction of the rate of disappearance and weight loss by forest litter samples. This is in agreement with Muller *et al.* (1988), who found that the lignin concentration of residue was a much better predictor of residue decomposition rate than N concentration.

iii) Effect of Polyphenol Content

The importance of polyphenolics in decomposition and mineralisation processes has been frequently debated (Swift *et al.*, 1979). Palm and Sanchez (1991) indicated that polyphenolic content may play a more important role in influencing mineralisation patterns for plant residue than %N or lignin to N ratio. The negative effect of polyphenols on decomposition and nutrient releases was also reported by Vallis and Jones (1973). They suggested that polyphenolic compounds bind to proteins and form complexes resistant to decomposition. Polyphenolics can also bind to organic N (e.g., amino acids and proteins) in the leaves, making the N unavailable, or can bind to the soluble forms of organic N released from the leaves, forming resistant complexes in the soil. Sivapalan *et al.* (1985) found lower net mineralisation from tea leaves that had high soluble N and high polyphenolic content in comparison with leaves with high soluble N content but low in polyphenolics. They suggested that polyphenolics made the soluble N unavailable.

iv) Effect of Leaf Toughness

Among features of litter that may affect its decomposition and that have been given little attention is physical leaf toughness. Gallardo and Merino (1993) developed a toughness index of

residues and proposed leaf toughness to be an index of substrate quality. Goering and Van Soest (1970) and Ma and Takahashi (1989) report that high silica content reduces the digestibility of plant.

Large differences in decomposition rates and nutrient release patterns have been observed among different residue materials, particularly in the early stages of decomposition (McClaugherty *et al.*, 1985). This is due to the diversity in sources of plant derived organic inputs, as well as the complexity of these substrates which results in extreme heterogeneity in the microbial reactions involved in their decomposition (Alexander, 1977; Tian *et al.*, 1995b). The percentage of each constituent contained in plants, varies depending on plant species, stage of growth, as well as plant part and seasons (Ross, 1989 as cited by Conteh, 1994). Thus the decomposition varies depending on plant species (Ajwa and Tabatabai, 1994), stage of growth (Waksman and Tenney, 1927) and plant part (Konboon, unpublished data-b (2nd); Lehmann *et al.*, 1995).

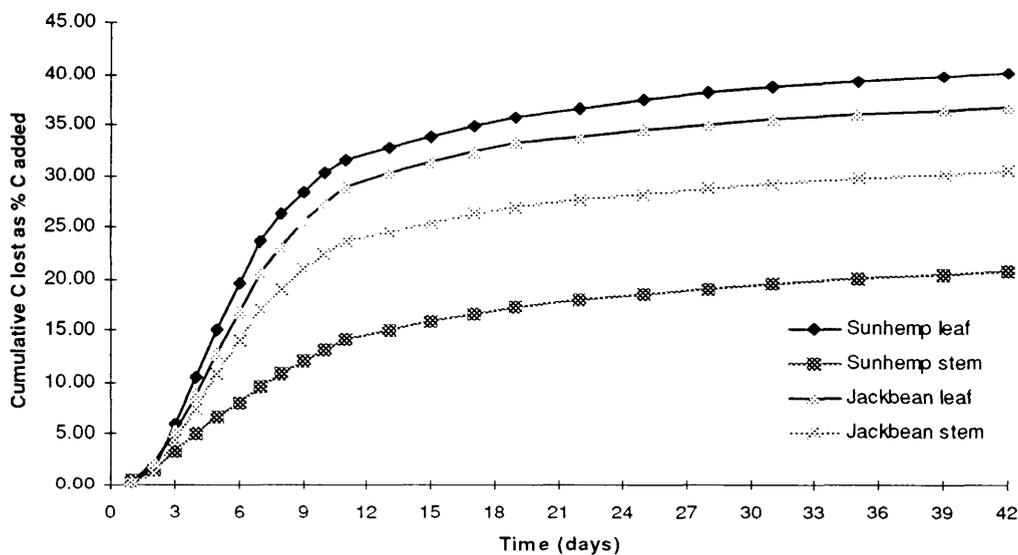


Figure 2.3 : Decomposition of different plant parts of two legumes (Konboon, unpublished data-b)

Attempts have been made to predict the rate and pattern of decomposition of organic substances from a range of organic materials based on its chemical components such as C:N ratio (Anderson, 1973; Parnas, 1975; Flanagan and Van Cleve, 1983), lignin content (Fogel and Cromack, 1977; Meentemeyer, 1978), lignin to N ratio (Moore, 1984; Melillo *et al.*, 1982), polyphenols content (Herman *et al.*, 1977; Tian *et al.*, 1992a), C: nutrient ratio (White and Ayoub, 1983; Wu *et al.*, 1993) and fibre (Latter and Howson, 1977). Herman *et al.* (1977) found that decomposition rates of plant residues could not be predicted from individual properties of the original material.

Recently, Tian *et al.* (1995b) developed a Plant Residue Quality Index (PRQI) for assessing the quality of plant residues using the C:N ratio, lignin content and polyphenol content. The PRQI was correlated with the decomposition rate of plant residues using litter bags and they concluded that the

PRQI can be used for selecting plant residues, and projecting their agronomic value. The PRQI is defined as;

$$\text{PRQI} = [1 / (a \text{ C:N ratio} + b \text{ Lignin} + c \text{ Polyphenols})] \times 100$$

Where *a*, *b* and *c* are coefficients of relative contribution of C:N ratio, lignin content (%) and polyphenol content (%) to plant residue quality.

b) Effects of Quantity Added

Evidence from laboratory and fields studies has generally demonstrated that the rates of decomposition of plant materials added to soils are proportional to the amounts initially added (Pinck and Allison, 1951; Jenkinson, 1965; 1977; Larson *et al.*, 1972). Pinck and Allison (1951) considered that the percentage decomposition of plant C in soils is nearly independent of input levels, provided that C additions do not exceed an amount equivalent to 1.5% of dry soil. Jenkinson (1977) observed a slightly higher percentage release of ¹⁴C₂O from a greater addition of ¹⁴C-labelled ryegrass in soils in laboratory incubations. In addition, Ladd *et al.* (1983) studied the effect of different quantities added on the decomposition of ¹⁴C and ¹⁵N-labelled legume (*Medicago littoralis*) material in soil, and found that the proportions of plant ¹⁴C and ¹⁵N retained in soil were not independent of the amounts of plant ¹⁴C and ¹⁵N added. They further concluded that this was especially apparent for residual organic ¹⁵N, and for soils sampled after the initially rapid decomposition period. The greater the amount of labelled plant material added, the smaller the proportions of residual organic ¹⁴C and ¹⁵N.

In contrast, Broadbent and Bartholomew (1948) studied the effect of rate of residue addition on the decomposition of ¹³C-labelled plant residue in soil and found that the rate of decomposition was inversely related to the quantity added, but no explanation was provided. This finding supported a study undertaken by Hallam and Bartholomew (1953).

c) Effects of Fertiliser Application

When organic material is added to soil the micro-organisms decomposing it can obtain the necessary inorganic nutrients (N, P, S, Ca etc.) for growth from two sources: those already present in the soil in plant-available forms and those in the added organic material itself. However, Jenkinson (1981) states that it is unusual for nutrient elements other than N to limit the decomposition of plant or animal material in soil, even though they can restrict microbial activity *in vitro*. Thus, although most elements considered as essential for plant growth can be utilised by soil microorganisms, it has been widely reported that the element required in the greatest abundance is N. Nitrogen is required by all decomposers because it is a constituent of extracellular and intracellular enzymes, nucleic acids and lipoprotein membranes, thus making it the most limiting nutrient for microbial activity. Therefore most studies on effects of inorganic nutrients on the activity of soil microbes have mainly been centred on N. However, the effect of added N on the decomposition of organic matter is variable (Jenkinson, 1981).

An important study in this area is that of Allison and Cover (1960), who added ammonium nitrate to raise the N level in shortleaf pine sawdust to be 0.15, 0.5, 1.0 and 2.0%, and found no clear positive effects of N on the decomposition rate. The results from this study showed that additions of ammonium nitrate increased the rate of decomposition of sawdust; but the reverse effect was observed

at the highest rate of N. They indicated that this negative effect was due in part to the lowering of the pH from its initial value and, in addition, increasing the salt concentration, which apparently affected the rate of microorganism growth. The report by Kowalenko *et al.* (1978) supports this observation.

A clear positive effect of N on decomposition rate of organic matter was reported by Azam *et al.* (1993), who studied the effect of N on decomposition of ¹⁴C-labelled ryegrass and reported that inorganic N, both native soil N and added N, enhanced the mineralisation of plant residues applied to soil. In addition, Azam *et al.* (1993) concluded from their study that the mineralisation of organic N from leguminous residues applied to soil is enhanced by the application of inorganic N, particularly ammonium. Supporting evidence is provided by the studies of Lueken *et al.* (1962). However, Fog (1988) observed that the negative effects of added N on decomposition occurred most often in experiments with a time scale of weeks to years, and where substrates had a significant lignin component.

Negative effects of N on decomposition have also been widely reported. Allison (1955) argued that the requirement of N during humus formation from plant residue was small and this requirement could almost always be met in agricultural soils since microorganisms are better able to compete for N than plants. Fog (1988) reported from his review that added N often has a negligible effect on decomposition. Little N may be required during decomposition, because C from plant residues is unavailable or slowly available to microorganisms and recycling of N and mineralisation of N from other sources may be adequate to meet the requirements. Positive decomposition responses to added N may only occur early in the decomposition process, when C availability and microbial activity are greatest (Tenney and Waksman, 1929; Knapp *et al.*, 1983). Moreover, decomposition may also be inhibited by the addition of N. In particular, the decomposition of lignin may be sensitive to high NH_4^+ -N levels, because addition of NH_4^+ -N inhibits or delays synthesis of ligninolytic enzymes by microorganisms (Keyser *et al.*, 1978; Swift *et al.*, 1979, Couteaux *et al.*, 1995). Moreover, the addition of N may reduce lignin decomposition due to competition by other less effective organisms (Fog, 1988).

Bremer *et al.* (1991) reported from a recent study that the addition of N to lentil and wheat straw reduced net N mineralisation, with a greater reduction with increasing N rates. They suggested that this was due to 1) more N immobilised because microbial biomass with a lower C:N ratio was synthesised, 2) increased losses of N by denitrification, and 3) reduced mineralisation or remineralisation, which may have been related to reduced microbial activity.

Apart from N, other elements have also been shown to have an effect on decomposition. Cheshire and Chapman (1996) studied the influence of N and P status of plant material and of added N and P on the mineralisation of C from ¹⁴C-labelled ryegrass in soil and concluded that P, whether intrinsic or added, can increase the rate of decomposition of organic residues but there is a strong interaction with N, which has a predominant influence. In addition, Enriquez *et al.* (1993) have demonstrated how the decomposition rate increases with both intrinsic N and P.

This suggests that all nutrients, such as, sulfur, phosphorus and potassium, can have a role in decomposition and nutrient release of plant residue, even if N is generally the most limiting.

d) Accessibility

The accessibility of plant residue to soil microbes is of primary importance in its rate of decomposition. The method of application of plant residue, such as residue particle size and placement, can provide a different degree of accessibility which, in turn, affects residue breakdown rate as well as the mineralisation-immobilisation processes.

Residue particle size appears to be an important factor in the decomposition and mineralisation processes, especially at the initial stage (Jensen, 1994). It has been suggested that plant residue with small particles decomposes faster than larger particles, because the increased surface area and greater dispersion in soil will increase the susceptibility to microbial attack, especially if residues are not readily penetrated by fungi and bacteria (Allison, 1973). This suggestion has been supported by many workers (Swift *et al.*, 1979; Amato *et al.*, 1984; Bremer *et al.*, 1991). Konboon (unpublished data-a) conducted a preliminary study on the effect of cut and uncut plant residues on decomposition and observed that the decomposition of cut residues occurred earlier, with a higher rate than uncut residues (Figure 2.4).

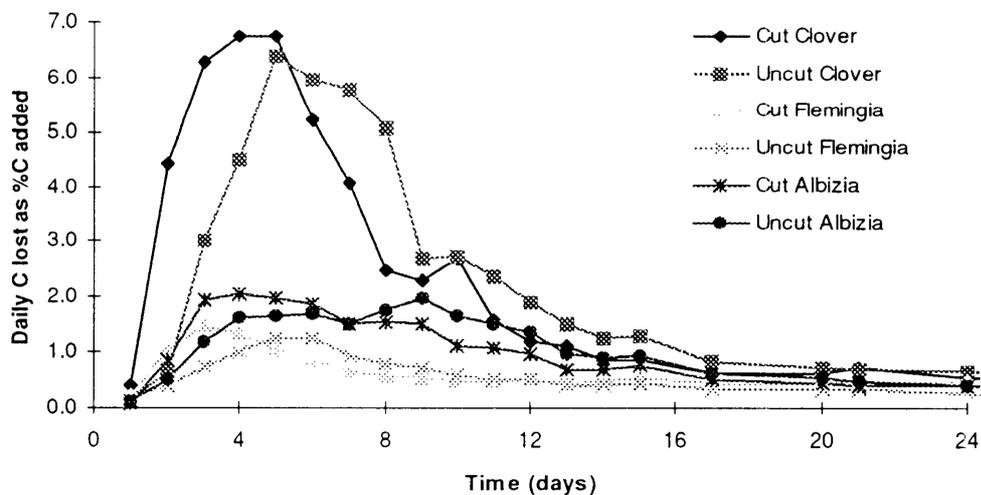


Figure 2.4 : Effect of cut and uncut plant residue on the decomposition rate (Konboon, unpublished data-a)

However, Jensen (1994) argued that the previous finding was true in the case of a residues with low C:N ratio, but not true in case of a residue with high C:N ratio. He further stated that during the initial decomposition of high C:N ratio residue the decomposition of larger sized residue may be N limited, resulting in a slower rate of decomposition compared to smaller residues.

The other residue management that affects accessibility is residue placement. Many studies indicate burying residue in soil increases the decomposition rate compared to placing residue on the soil surface (Harper and Lynch, 1981; Summerell and Burgess, 1989; Aulakh *et al.*, 1991; Douglas and Rickman, 1992; Buchanan and King, 1993; Schomberg *et al.*, 1994). Litter on the soil surface is more subject to unfavourable conditions for decomposition, particularly with respect to the moisture regime,

than is material buried in soil. The depth of incorporation also has an effect on decomposition. Kanal (1995) found that increasing the depth of residue incorporation from 5 to 20 cm resulted in a decrease in breakdown rate due to less biological activity. This is in contrast to the observation by Breland (1994) who found that increasing the incorporation depth up to 30 cm increased the decomposition rate due to more favourable moisture conditions in lower soil layers.

From this section, it can be concluded that the decomposition of plant residue is affected by 1) quality of plant residue, such as, C:N ratio, lignin content and polyphenol content and 2) management of plant residue, such as, fertiliser application and accessibility. These factors have to be determined in using plant residue to improve soil productivity and sustainability of agricultural systems in the longer term.

2.9.2 Environmental Factors

Environmental factors, especially temperature, soil moisture, soil oxygen and pH, control the rate of microbial growth and activity in soil. These factors, in turn, control the decomposition rate of plant residue in soil. The effect of environmental factors on the decomposition of SOM have been widely reported and reviewed as follows:

a) Temperature

Temperature affects the physiological reaction rates of organisms and the activity of microbial cells governed by the laws of thermodynamics. Therefore, changes in soil temperature have marked effects on microbial activity (Paul and Clark, 1989). Much work has been done to quantify the effect of temperature on the rate of microbial activity and decomposition of SOM. A classic example is the work by Jenkinson and Anayaba (1977). Using ^{14}C -labelled ryegrass, they showed that the pattern of decomposition in a tropical (mean temperature = 26.1 °C) and temperate (mean temperature = 8.9 °C) climate were similar; but the rates of decomposition were significantly different. Only 25% of the applied organic matter remained in the soil after 6 months in Nigeria, but under temperate English conditions it took 2 years to reach this stage (3rd). Jenkinson and Ladd (1981) also reported that decomposition rates observed under semi-arid conditions in South Australia were approximately double those reported for the United Kingdom. Jenny *et al.* (1949) compared the decomposition rates of organic matter in temperate and tropical region and found similar results.

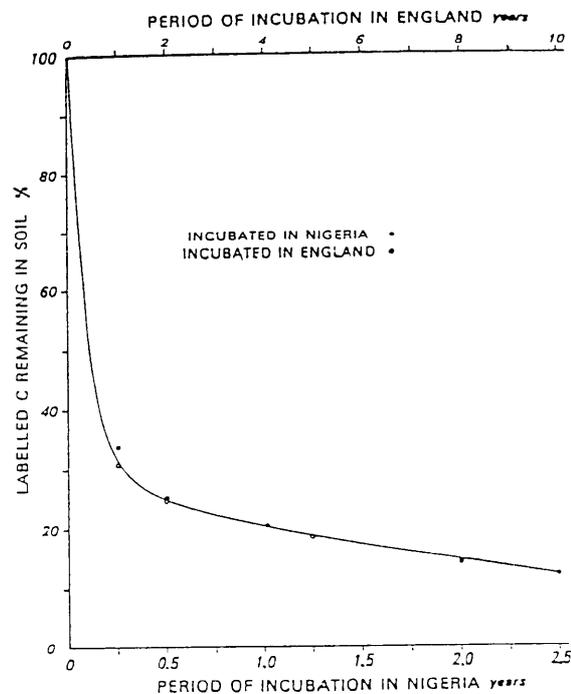


Figure 2.5 : Decomposition of uniformly ^{14}C labelled ryegrass in England and in Nigeria (Jenkinson and Anayaba, 1977)

Base on their C and N cycling study, Nicolardot *et al.* (1994) suggested that the decomposition of organic compounds in soil is markedly influenced by the temperature, the higher the temperature, the greater the extent of decomposition. This finding has been supported by many other workers, such as Howard and Howard (1993) and Vigil and Kissel (1995).

b) pH

The pH of soil affects both the nature and size of the microorganism population and the multiplicity of enzymes at the microbial level which subsequently affect the decomposition of SOM (Paul and Clark, 1989). Jenkinson (1977) studied the effect of pH on decomposition of ryegrass under field conditions and reported that 42% of the ryegrass-derived C still remained after one year in a soil pH 3.7, whilst 31% remained in soils of pH 4.4 and pH 6.9. However, by the end of five years this difference had largely disappeared, suggesting that only the early stages of decomposition were retarded by acidity.

Acid rain has also been reported to affect decomposition. Wilhelmi and Rothe (1990) studied the effect of acid rain (pH 1.5 to 6.5 range) on the C-mineralisation rates in organic soil layers and postulated that weakly acidic to neutral pH stimulated microbial CO_2 formation, whereas high acidity (pH < 2.4) inhibited it. However, Dursun *et al.* (1993) found that the concentration of sulphite in acid rain also had a combined effect with pH on decomposition.

c) Moisture

Growth and activity of soil microorganisms relies on soil moisture which, in turn, has a significant effect on plant residue decomposition and nutrient cycling (Standford and Epstein, 1974; Sommers *et al.*, 1981). Sommers *et al.* (1981) observed that soil dried to a water potential of -10 MPa evolved CO_2 at about half the rate observed if the soils were incubated at optimal water content, usually in the

range -20 to -50 kPa. The optimum conditions for microbially mediated decomposition of plant residue in soil is around field capacity (Standford and Epstein, 1974) or -0.03 MPa (Sommers *et al.*, 1981) Under laboratory conditions, Wilhelmi and Rothe (1990) found that low moisture limited the rate of C mineralisation and, as moisture level increased, C mineralisation increased until an optimum plateau was reached, which was 60% of soil water (w/w). Increased water beyond this optimum resulted in a decrease in C mineralisation. Andren *et al.* (1992) studied the decomposition of barley straw using litter-bags and reported that the decomposition rate was highly dependent on soil moisture. Similar results were obtained by Smith *et al.* (1993) and Reddy *et al.* (1994).

Water has another major impact on the decomposition of the plant material through the effect of wetting and drying. Many workers have reported that the decomposition of residue was enhanced in intermittently dried and rewetted soils (Amato and Ladd, 1980; Yaacob and Blair, 1980; Van Gestel *et al.*, 1993; Franzluebbers *et al.*, 1994). This is most likely related to the protection of organic matter in soil aggregates as wetting and drying leads to the breakdown of water-stable aggregates, with exposure of new surfaces for microbial attack (Stevenson, 1986).

d) Oxygen

Decomposition and mineralisation is less complete and slower under anaerobic conditions than under aerobic conditions (Jenkinson, 1981). When soil becomes so wet that the larger pores are water filled, decomposition of organic matter is limited by the rate at which oxygen can diffuse to the sites of microbial activity. The diffusion coefficient for oxygen in water is 10000 times smaller than that in air, so that even quite a modest oxygen demand cannot be met if the larger soil pores are water filled (Jenkinson, 1988). Parr *et al.* (1970) showed that although decomposition of organic matter, as measured by CO₂ evolution, was reduced when oxygen supply was cut-off, evolution of CO₂ was rapidly increased when aerobic conditions were restored, and that after about a week the total amount of CO₂ evolved from an incubation that had previously been anaerobic became the same as that from an incubation that had been aerobic throughout.

Aeration conditions in soil also affects nutrient transformations. In a study of the effect of soil aeration on nutrient transformations from decomposed residue in soil, Walters *et al.* (1992) observed that significant amounts of macronutrients were mineralised within a 10 day period under aerobic conditions, while decomposed residue enhanced anaerobiosis in poorly aerated soil (wet more than 10 days), resulting in chemical reduction and a several fold increase in DTPA extractable Mn, Fe and Cu after 20 days. After 30 days these micronutrients were re-oxidised resulting in decreased DTPA extractable concentrations.

Neue and Scharpenseel (1984) found that decomposition of plant residue under anaerobic conditions, or in submerged soil lead to the production of CO₂ and a range of compounds, such as methane (CH₄), NH₃, mercaptans, H₂, H₂S and partially humified residues. Schipper *et al.* (1994) reported that CO₂ evolution occurred 2-3 days prior to CH₄ production.

2.9.3 The Effect of Soil Texture

It has been suggested that soil texture, particularly silt plus clay content, plays a major role in determining the amount of organic matter in soil (Parton *et al.*, 1987). Organic C contents generally increase with the clay content of the soil, other factors being equal. Jenkinson (1988) stated that the amount of biomass, and biomass metabolites, formed during the first phase of the decomposition process, is greatly influenced by soil texture, clay soils retaining much more organic matter than sandy soils. Sorensen (1981) studied the decomposition of labelled cellulose in two soils of contrasting texture and found that soil containing 34% clay retained about twice as much of the cellulose-derived C than a soil with 4% clay. Moreover, more than twice as much of the cellulose C was converted to amino-acid C in the clay than in the sandy soil. The higher the level of elements retained in soil the greater they can be taken up by the plant. This was verified by Jordan *et al.* (1996) who studied the N availability to sorghum from organic residues in soil with different textures and found greater residue-N uptake by the plant and soil microbial biomass C and N in clayey soil (34% clay) compared with sandy soil (5% clay). Similar results were reported by Nowak and Nowak (1990).

It appears that the role of clay in stabilising organic matter in soils is more important in warmer soils where decomposition rates can be expected to be higher. In cold soils, cool temperatures may be the main factor slowing decomposition, and clay content may be less important (Anderson, 1995).

2.9.4 Soil Fauna

Soil fauna play an important role in enhancing and sustaining soil productivity, through their effects on SOM decomposition and availability of plant nutrients (Anderson, 1988). Witkamp and Ausmus (1976) report that exclusion of macro-fauna reduced decomposition rates and nutrient release from leaf litter.

Couteaux *et al.* (1995) stated that soil fauna contribute to litter breakdown by, 1) grinding plant residues and thus increasing the surface area of the detritus, 2) mixing SOM with the mineral horizon, and 3) channelling and improving the soil structure. Tian *et al.* (1995a) studied the effect of soil fauna on the decomposition of plant residues with contrasting chemical composition in the field, and observed increased breakdown rate following the addition of earthworms or millipedes. They further suggested that the role of soil fauna is relatively greater in the decomposition of materials with high C:N ratio, lignin and polyphenols, but have no obvious effect on the low C:N ratio residues as these residues are easily decomposed by microorganisms. Termites and ants are known to be efficient in digesting cellulose-containing substances, and in some cases also lignified substances (Lee and Wood, 1971). Millipedes break down plant litter and mix it with mineral soil, which they ingest (Kevan, 1968).

2.9.5 Other factors

Plant residue grown under elevated O₃ and/or CO₂ have been shown to affect structure, nutrient status and, as a result, its decomposition (Boerner and Rebbeck, 1995; Cotrufo *et al.*, 1994). The decomposition of plant residue in soil is also enhanced by living roots or, in turn, the presence of a crop (Cheng and Coleman, 1990; Dormaar, 1990), with the addition of lignite humic acid (Whiteley and Pettit,

1994). Allison and Cover (1960) and Condrón *et al.* (1993) showed that addition of lime enhanced decomposition, especially under acidic conditions.

2.10 Phytotoxicity of Plant Residues

Decomposed plant residue in soil has been found to have an adverse effect - known as allelopathy - on crop growth (Putnam, 1994). Substances produced during the decomposition of certain crop residues have long been considered as causes of poor growth and yield of many crops (Patrick *et al.*, 1963). Microorganisms can produce a large range of substances, potentially toxic to plant roots (Lynch, 1976 as cited by Cannell and Lynch, 1984) but they seldom accumulate in aerobic soil because they are metabolised rapidly by other microorganisms. However, the adverse effect of decomposed residue under aerobic condition on crops has been widely reported.

Bhowmik and Doll (1982) reported allelopathy effects from weed residues, such as giant foxtail and barnyard grass residues, that affected growth and reduced corn and soybean yields. Jessop and Stewart (1983) studied the effects of crop residues (rape, sorghum, field pea and wheat straw) on the emergence and early growth of wheat, and reported that wheat growth was severely depressed by crop residues, with the ranking of rape residue < pea residue < sorghum < wheat straw, with the depression varying from 50% to 70%, compared to the control. This is supported by the observation of Kimber (1973), who reported a marked depression in wheat germination under a wheat straw residue. This suggested major phytotoxic effects of residue during early growth (Jessop and Stewart, 1983).

Vaughan *et al.* (1983) found that the phenolic acid substances, such as Ferulic acid, *p*-Coumaric acid and *p*-HBA, released from the living and dead tissues of a variety of plant species caused adverse effects on the growth of crops.

2.11 Studying the Decomposition of Soil Organic Matter

Many methods have been used in studies of SOM changes and these have been reviewed extensively by Conteh (1994). The major methods reviewed here will be those significant to the study of SOM dynamics. Other methods will not be considered in any detail.

2.11.1 Measurement of Carbon Dioxide Evolution.

One aspect of understanding the turnover of SOM is the measure of the breakdown rate of different plant residues. In the past, plant residue decomposition rates in natural and agricultural systems have most often been determined using litter bags (Wieder and Lang, 1982). This traditional manner of following the decomposition of plant residues in soil has been to measure the loss of C (as CO₂) from soil and plant residues, incubated together, and to subtract from this the loss of C from soil incubated in the absence of residues. This approach, however, has limitations, including analytical errors in measuring long-time release of CO₂, particularly when the amount of plant residue added is kept small, relative to the amount of native organic matter. Also, the assumption is made that addition of plant residues to the soil does not alter the decomposition rate of the native organic matter (Stevenson, 1986).

In vitro perfusion methods have been used to study the decomposition of plant materials (Nyamai, 1992). These methods use a closed, continuous air-flow system, in which the plant material is allowed to decompose in a medium of constant ionic strength, with the evolved CO₂ absorbed in an alkali solution. The CO₂, produced by microbial respiration, is precipitated with BaCl₂ and subsequently estimated for CO₂, by titrating against standard HCl, with the assumption that 1 mL of 1 M HCl can release 22 mg CO₂. The reactions in this process are as follows:



The rate of CO₂ production is used to describe the rate of plant residue decomposition. This method allows the rapid screening of plant residues and ranking of potential materials. However, the perfusion apparatus of Nyamai (1992), which was made from glass, is difficult and expensive to construct and rather difficult to operate.

2.11.2 Total Organic Carbon Measurements

a) Oxidation by Dichromate

The status of the SOM resource base can be assessed by measurement of the total amount of organic C in the soil. The most commonly used techniques for measuring organic C in soil is based on the dichromate oxidation proposed by Schollenberger (1927) and modified by Walkley and Black (1934). The principal of this method is that all organic C is oxidised by the dichromate under acid conditions, and the amount of dichromate reduced gives an indication of the organic matter content. In spite of its wide usage, the dichromate procedure is subject to certain limitations. Firstly, the method is based on the assumption that C in SOM has an average valency of zero. Secondly, several substances interfere with the determination of oxidised C, notably chloride, iron (II) and higher oxides of Mn (Walkley, 1947). Thirdly, not all organic carbon is oxidised. In addition, dichromate procedures cannot be used to discriminate between C from carbonised materials and C from SOM.

b) Dumas-type Catalytic Combustion

Total C measurements have been improved by Dumas-type catalytic combustion of C to CO₂, with the resulting CO₂ being measured by a thermal conductivity detector or, more recently, by coupling the combustion instrument to a mass spectrometer in a continuous flow system. Although these methods provide an improvement on measurements of total C by oxidation, measurement of total C or total organic C is not sensitive to short-term changes in the amount, or more particularly the form of SOM, which results from changes in soil management (Lefroy *et al.*, 1995).

c) Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES)

Recently, Oweezkin *et al.* (1996) have successfully developed the method to determine the total dissolved organic C in soil solutions and waters derived from decomposing organic material, using ICP-AES. By this method, the soil solution or water sample are acidified and sparged by N gas (N₂) to remove inorganic C, prior to being introduced into the plasma for C determination. This method

provides a rapid and convenient method for the determination of dissolved organic C content of solutions, providing results with high accuracy. In addition to the measurement of C, the multi-element analysis capacity of the ICP-AES permits simultaneous analysis of all other elements of interest in the sample.

2.11.3 Isotopic Techniques

The dynamics of SOM have also been investigated by measuring the rate or degree of incorporation of applied isotopes into SOM, rather than the chemical or physical form, or the rate of breakdown or release. Stable and radioisotopes, when available, have been used in these studies for carbon (^{13}C and ^{14}C), nitrogen (^{15}N), phosphorus (^{32}P and ^{33}P) and sulfur (^{34}S and ^{35}S).

a) Using Natural Abundance of Isotopes

i) ^{13}C Natural Abundance

The $^{13}\text{C}:^{12}\text{C}$ ratio in the atmosphere varies with a number of physiographic parameters (temperature, altitude, latitude, etc.), and by certain biological processes (Lefroy *et al.*, 1995). The major biological process is photosynthesis, in which discrimination occurs during the carboxylation step against ^{13}C , in favour of the lower atomic mass ^{12}C . This discrimination varies with photosynthetic pathway, with greater discrimination against ^{13}C in C3 (Calvin cycle) plants than in C4 (Hatch-Slack cycle) plants, due to greater discrimination in the primary carboxylation step of C3 plants, catalysed by the enzyme ribulose biphosphate carboxylase (RuBP), resulting in lower $^{13}\text{C}:^{12}\text{C}$ ratios in C3 plants than in C4 plants. CAM (crassulacian acid metabolism) plants show variable discrimination; but more often similar to C4 plants.

The $^{13}\text{C}:^{12}\text{C}$ ratio in SOM, generally measured as $\delta^{13}\text{C}$, is comparable to that of the source plant material (Schwartz *et al.*, 1986), and thus every change in vegetation between C3 and C4 plants leads to a corresponding change in the $^{13}\text{C}:^{12}\text{C}$ value of SOM (Lefroy *et al.*, 1995). This principle has been used by Schwartz *et al.* (1986) to study changes in vegetation in the Congo, by Skjemstad *et al.* (1990) in studying the turnover of SOM under pasture, by Balesdent *et al.* (1987) and Lefroy *et al.* (1993) on changes in SOM as a result of cropping, by Bonde *et al.* (1992) in quantifying maize root derived soil organic C, and by Mary *et al.* (1992), in the biodegradation of root mucilage, roots and glucose in soils.

One advantage of natural abundance measurements, over the addition of stable or radioisotope, is that the addition of isotope invariably involves some perturbation of the system. If the perturbations are different from the normal perturbations which occur in the system, it can be difficult to separate the effects of the perturbations from true changes in SOM dynamics (Lefroy *et al.*, 1995).

ii) ^{15}N Nitrogen

Because of the intricate nature of soil N transformation, many facets of the soil N cycle are poorly understood. The introduction of ^{15}N techniques has provided a means for re-examining earlier concepts, and for developing sound principles from which efficient N management systems in soil can be devised. In a typical ^{15}N labelled experiment, a known amount of ^{15}N is applied to the soil, either directly or as labelled fertiliser, and measurements are made to assess the relative amounts of the soil

and fertiliser N utilised by plants, and the quantities of fertiliser N retained in mineral and organic forms. Any ^{15}N , not accounted for in the crop or in the soil, is assumed to have been lost by leaching, denitrification, or NH_3 volatilisation (Stevenson, 1986).

Recent work by Brooks *et al.* (1989) has led to the development of a diffusion procedure for automated isotope ratio analysis of inorganic N in soil extracts. With this procedure, the extract is treated with a mild alkali (MgO) and a reducing agent (Devarda's alloy) to convert $(\text{NH}_4^+ + \text{NO}_3^-)\text{-N}$ to $\text{NH}_3\text{-N}$. Conversion is carried out at room temperature in a disposable specimen container, and the $\text{NH}_3\text{-N}$ evolved is collected as $\text{NH}_4^+\text{-N}$ in an acidified disk, cut from a glass fibre filter with a paper punch. Although developed for diffusion of 2 M KCl extracts of soil, the method of Brooks *et al.* (1989) has application to other types of samples, including natural water samples. Thus, by using this method, it is possible to determine the ^{15}N in water or leachate samples.

^{15}N technique have been used in both laboratory and field experiments in the following types of studies;

- Nitrogen balance of the soil, including gains and losses from the soil-plant system (Bergersen *et al.*, 1992, He *et al.*, 1994).
- Immobilisation and mineralisation of N (Jedidi *et al.*, 1995).
- Uptake of soil and fertiliser N by plants (Jensen, 1994; Jordan *et al.*, 1996).
- Fate of residual fertiliser N in soil (Amato *et al.*, 1984; Ladd *et al.*, 1985).
- Fixation of NH_4^+ by clay and of NH_3 by organic matter, and the availability of the fixed N to plants and microorganisms (Amato and Ladd, 1980).
- Bacterial denitrification and chemodenitrification (Abubakar *et al.*, 1994).
- Relative use of NH_4^+ and NO_3^- by microorganisms and higher plants (Cortez and Cherqui, 1991; Diekmann *et al.*, 1993).
- Biological N_2 fixation (Senaratne and Ratnasinghe, 1995).
- Nitrate leaching (Abubakar *et al.*, 1994; Kamukondiwa and Bergstrom, 1994)
- As an indicator of SOM turnover (Ladd, 1981; Tiessen *et al.*, 1984; Muller and Sundman, 1988; Cadisch *et al.*, 1993; Green and Blackmer, 1995).

The $^{15}\text{N}:^{14}\text{N}$ natural abundance technique uses the difference in much the same way that the ^{13}C natural abundance technique relies on the difference in $^{13}\text{C}:^{12}\text{C}$ ratio between C3 and C4 species. This technique can be very powerful, as long as a good reference value for the non-fixed $^{15}\text{N}:^{14}\text{N}$ ratio is available. However, measurements of natural ^{15}N abundance requires high precision and are subject to variable and significant errors due to variations in the ^{15}N content of different organic and inorganic N components (Stevenson, 1986). Bremner and Hauck (1982) concluded that the many sources of error

in sample collection and processing, together with uncertainties in data interpretation, mitigate against use of variations in ^{15}N abundance as a mean of obtaining reliable information about N cycle processes in complex biological systems.

b) Using Radioisotopes

i) ^{14}C Carbon

The application of ^{14}C in studies of SOM seems to have been first brought to the attention in soil chemistry by Bingeman *et al.* (1953) and Hallam and Batholomew (1953). Since then ^{14}C has been used in many aspects of SOM research. The use of substrates labelled with ^{14}C made it possible to follow the decomposition of added residues with considerable accuracy, even in the presence of relatively large amounts of native organic matter. It has been possible also to identify plant C as it becomes incorporated into fractions of the soil humus (Stevenson, 1986).

The “bomb” ^{14}C technique has been pioneered by Rafter and Stout (1970) and Stout and O'Brien (1973). This technique uses variations in the elevated amounts of ^{14}C deposited in soils, as a result of atmospheric detonation of thermonuclear devices in the 40s, 50s and 60s (Goh, 1991). The annual input of C, the rate of decomposition, the turnover time and diffusivity, down the profile of C, have been determined using this technique (O'Brien and Stout, 1978; O'Brien, 1984). The main problem in using this method in soil studies, is the very low amount of ^{14}C present in the soil and the consequent difficulty and expense of analysis (Lefroy *et al.*, 1995).

ii) ^{35}S Sulfur and ^{32}P Phosphorus

The short half-lives of ^{32}P (14 days), ^{33}P (25 days) and ^{35}S (87 days) make them less appropriate than ^{14}C (half-life = 5730 years) for studies of SOM dynamics, particularly long-term studies in the field (Lefroy *et al.*, 1995). Despite these limitations, they have been successfully used in studies of the breakdown of residues, labelled with ^{33}P (Friesen and Blair, 1988) and ^{35}S (Lefroy *et al.*, 1994), and in the dynamics of ^{35}S in fractions of SOM (Eriksen *et al.*, 1995).

Lefroy *et al.* (1995) stated that the move from using ^{14}C to ^{13}C was likely to be concomitant with a move from using ^{35}S to ^{34}S , as long as the price of ^{34}S continued to decline and the availability of better and more manageable mass spectrometer systems improved. The interpretation of ^{34}S data is complicated, in much the same way as for ^{13}C , by the presence of different proportions of ^{34}S to ^{32}S in a range of biological and geological forms. There is no stable isotope to replace ^{32}P and ^{33}P .

2.11.4 Oxidation by Potassium Permanganate

Loginow *et al.* (1987) developed a method of fractionating soil organic matter, based on susceptibility to oxidation by permanganate. In its original form, the degree of oxidation with three different concentrations of KMnO_4 was used in conjunction with the total carbon content of the soil to obtain four fractions of soil C. The technique, however, cannot be used to obtain the organic fractions *in substantio*.

This technique was modified by Australian researchers (Lefroy *et al.*, 1993; Blair *et al.*, 1995b), by using a single strength of KMnO_4 to provide sufficient characterisation of the labile C to define the

state of soil systems. The total soil C, measured by combustion, and the amount of oxidising agent consumed by the KMnO_4 are used to calculate two fractions of organic C - one, which is oxidised by KMnO_4 and a second fraction, which is not oxidised by the KMnO_4 . They further demonstrated that the modified technique can be used to monitor small, relatively short term changes in SOM, characterising organic matter of various soils, and in evaluating changes occurring under the influence of organic and mineral fertilisation. In addition, the ability of the technique in measuring the amount of labile C in a soil, allowed a measurement of the relative sustainability of different agricultural systems, by comparison of a cropped and reference soil.

This method is based on the supposition that the oxidation action of KMnO_4 on soil organic C under neutral conditions is comparable to that of enzymes produced by soil microorganisms. The procedure involves reacting the SOM with an excess of KMnO_4 . As some classes of organic compounds are oxidised, the permanganate is reduced and the reduced permanganate, which is equivalent to the amount of C oxidised, is measured by spectrophotometry.

2.12 Fate of C, N and S in Agricultural Systems

2.12.1 Dynamics of Carbon in Agricultural System

Carbon is fixed in the plant and usually only a small fraction of the living plant is removed at harvest or consumed by herbivores. When the plant or parts of it die, decomposer organisms, i.e. microorganisms and soil fauna, attack the litter. Part of the litter is assimilated by the organisms, forming new biomass; or is respired (i.e. mineralised). The non-respired decomposition products are incorporated into the SOM, where they may remain for periods ranging from hours to several centuries before being mineralised. Most transfers involves respiration, i.e. CO_2 returns to the atmosphere which becomes available for photosynthesis. Atmospheric CO_2 is then eventually fixed by plants and the cycle is completed (Andren *et al.*, 1990).

2.12.2 Dynamics of Nitrogen in Agricultural System

The N cycle in soil (3rd) is an integral part of the overall cycle of N. The source of the soil N is the atmosphere, where the strongly bonded gaseous molecule (N_2) is the predominant gas (79.08% by volume). Gains in soil N occur through fixation of N_2 by microorganisms, such as blue-green algae, free-living bacteria and by symbiotic N_2 fixation by leguminous plants (Stevenson, 1986). The other source is from the return of ammonia (NH_3) and nitrate (NO_3^-) in rainwater. The amount of N added to the soil each year in atmospheric precipitation is normally too small to be of significance in crop production. Eriksson (1952), as cited by Stevenson (1986), estimated that accessions of NH_4^+ plus NO_3^- -N ranged from 0.78 to 22.0 kg ha^{-1} per year.

Nitrogen losses occur through crop removal, leaching and volatilisation. The conversion of N_2 to combined forms occurs through biological N_2 fixation. Organic forms of N, in turn, are converted to NH_3 and NO_3^- by mineralisation. The conversion to NH_3 is termed ammonification; the oxidation of this compound to NO_3^- is termed nitrification. The utilisation of NH_3 and NO_3^- by plants and soil organisms

constitutes assimilation and immobilisation, respectively. Combined N is ultimately returned to the atmosphere as N_2 , such as through biological denitrification, thereby completing the cycle (Stevenson, 1986).

2.12.3 Dynamics of Sulfur in Agricultural System

A pictorial representation of the S cycle is shown in 3rd. During weathering processes, the S of primary minerals is converted to sulfate (SO_4^{2-}) which in turn is utilised by plants and converted into organic forms, such as the cystine and methionine of proteins. When plant and animal residues are returned to the soil and subjected to decay by microorganisms, part of the organic S reappears as SO_4^{2-} , part is incorporated into microbial tissue, and hence into humus. Sulfur is added to soil in fertilisers, in pesticides of various types, in irrigation water and through absorption of S gases (e.g., SO_2) from the atmosphere. Losses of soil S result from runoff and leaching. Under certain circumstances the soil may serve as a source of hydrogen sulfide (H_2S) and other S gases to the atmosphere (Stevenson, 1986).

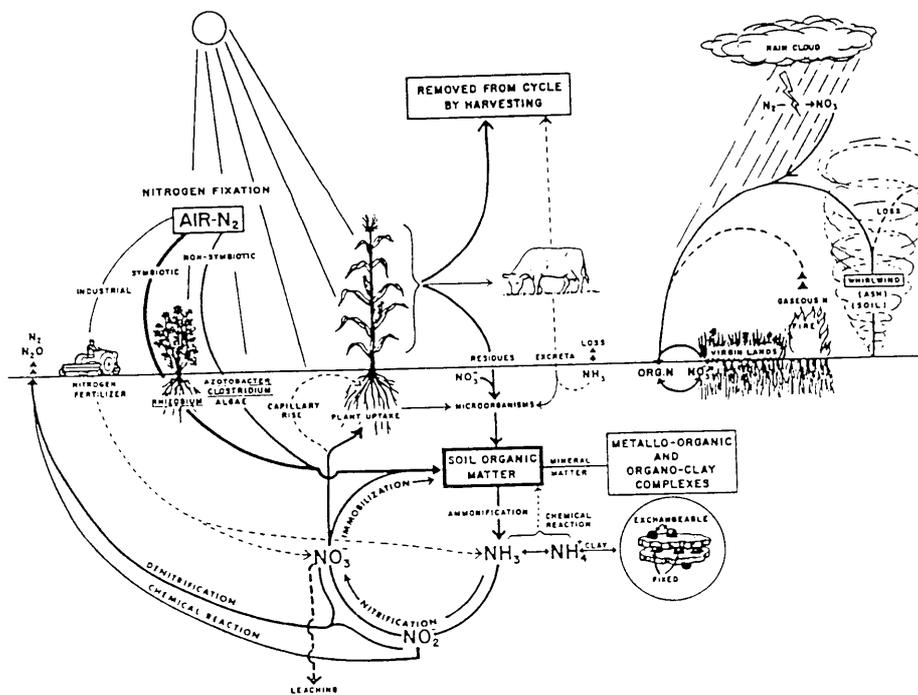


Figure 2.6 : The N cycle in soil (Stevenson, 1986)

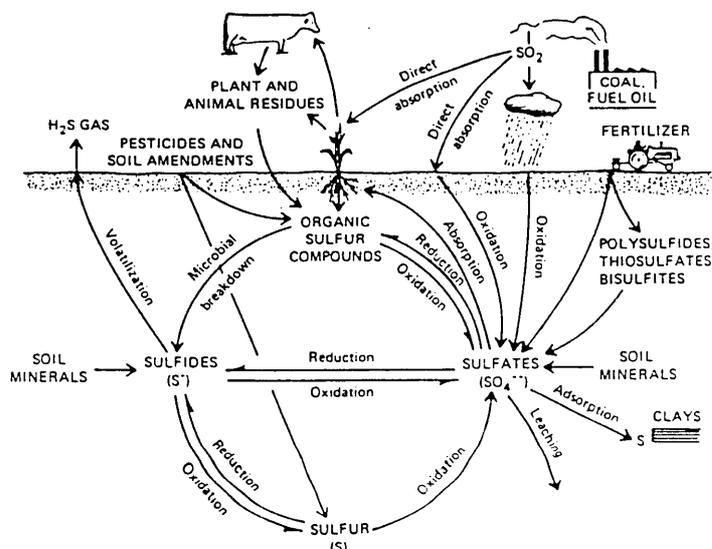


Figure 2.7 : The S cycle in soils (Stevenson, 1986)

2.13 Conclusion

Soil organic matter is defined as the sum total of all organic C-containing substances in soils. It consists of a whole series of products which range from undecayed plant and animal tissues through ephemeral products of decomposition to fairly stable brown to black material bearing no trace of the anatomical structure from which it was derived; it is the latter material that is normally defined as humus. Generally, SOM has been subdivided into 2 groups, which have similar morphological or chemical characteristics, namely non-humic and humic substances. The main functions of organic matter in soil are considered in terms of its effects on soil physical, soil chemical and soil biological properties which, in turn, affect crop productivity. Organic matter binds soil into aggregates, giving rise to soil structure and associated soil porosity, and stimulates the activity of soil fauna and microorganisms. Soil organic matter is also a major source of N, S, P and many micronutrients in soils.

The level of organic matter in soils is influenced by forest clearing and subsequent cropping, cropping systems and management, and organic material added. The decline in SOM is one of the major problems in many agricultural systems throughout the world. As SOM levels decrease, water-holding capacity of soil decreases, erosion increases, compaction increases, water infiltration decreases, soil fertility decreases and crop growth decreases. As organic matter and soil fertility decline, yields can be maintained only by significant input of inorganic fertiliser. To make the agricultural system more sustainable or truly sustainable, SOM levels need to be restored to an acceptable levels. One key way to improve SOM is to maximise the return of organic materials.

There is increasing interest in using organic material, such as animal waste, crop residues, green manures and urban and industrial wastes to improve soil organic matter and soil productivity. Generally, organic matter management has been approached via the use of crop residue return and green manuring. There are many factors that control the breakdown rate of these organic materials in

soil. One major factor is the "quality" of the added materials. The diversity in sources of plant derived organic matter inputs, as well as the complexity of these substrates, results in extreme heterogeneity in the microbial reactions involved in their decomposition. The different sources of organic materials contain different types and/or amounts of organic compounds, thus resulting in different breakdown rates. Lignin and polyphenol contents, C:N ratios, as well as nutrient contents are known to affect the breakdown rate and nutrient release of residues in soil. Apart from the quality of material there are other factors that co-determine the breakdown rate which include environmental factors (temperature, soil pH, soil aeration, soil moisture), biological factors (soil microorganisms and soil animals) and soil texture.

Attempts have been made to predict the rate and pattern of decomposition of organic substances from a range of organic matter quality parameters. Tian *et al.* (1995b) has developed a Plant Residue Quality Index (PRQI) using the C:N ratio, lignin and polyphenol concentration of plant residues, the PRQI obtained was then used for selecting plant residues. However, most often residue breakdown rates have been determined using litter bags (Wieder and Lang, 1982) and incubation studies (Jenkinson, 1966). Nyamai (1992) developed the *in vitro* perfusion technique and successfully used it to investigate the decomposition and nutrient release from foliage species. Apart from decomposition studies, the effect of using plant residues on SOM dynamics or changes in SOM has also been extensively studied. In these studies the techniques generally used included isotopic techniques, fractionation of SOM i.e. oxidation by potassium permanganate and total organic matter measurements. The study of the pattern of decomposition and nutrient release, as well as the subsequent fate of the released residues are important for the effective use and management of these materials to improve the level of SOM.

The experimental program undertaken concentrated on the decomposition of plant residues, including a measurement of carbon and nutrient dynamics in soil-plant systems, in order to evaluate role of plant residues in the development of sustainable cropping systems.