COMPARATIVE EFFECTS OF SUGARCANE MONOCULTURE ON SOIL ORGANIC MATTER STATUS AND SOIL BIOLOGICAL ACTIVITY

by

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B.Sc. Agric (Natal)

Submitted in partial fulfilment of

the requirements for the degree of

MASTER OF SCIENCE IN AGRICULTURE

in the

Discipline of Soil Science

School of Applied and Environmental Sciences

Faculty of Science and Agriculture

University of Natal

Pietermaritzburg

DECEMBER 2002

I hereby declare that the research reported in this study is my own work. Where use was made of the work of others it has been duly acknowledged in the text.

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ACKNOWLEDGMENTS

Without the support and assistance of many people none of this work would have been possible. Therefore it is with extreme gratitude that I acknowledge the contributions of the following people.

Prof. Dick Haynes, without whom none of this work would have been possible. Thank you for your foresight, supervision, flexibility with regards to trial objectives and of course the backbreaking digging!

Mr. Rianto van Antwerpen, my co-supervisor, thank you for your sponsorship, support and facilitating the taking of samples from the various farms.

Mr. Harvey Dicks, for the many hours spent over the statistical analyses. Thank you very much for your work and support. Your help was invaluable.

South African Sugar Association, thank you for the sponsorship of this project and allowing me to make use of this data.

Marlene Graham for introducing me to the life of the laboratory. Thank you for your experienced advise, support, and friendship.

Paula, Essak, Tad and the rest of the staff of the Soil Science Department, thank you for your technical assistance, without which it would have been impossible to run the trials necessary for this work.

Neil, my husband, thank you for your faith in me. This work would not be possible without your guidance, insight, encouragement, and of course your statistics.

My parents, thank you for teaching me to never give up, and to always do my best regardless of the circumstances.

Gran, thank you for all your concern and encouragement. I'm sorry you didn't see me finish.

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ABSTRACT

The effects of increasing periods under sugarcane monoculture (managed by preharvest burning) on soil organic matter content and related soil properties were investigated in the 0 to 10 cm layer of a sandy coastal Ochric Cambisol (Glenrosa soils) and a red Rhodic Ferralsol (Hutton soil) from the sugar belt of KwaZulu-Natal. The organic C content at both sites under undisturbed vegetation is about 48 g C kg^{-1}. This declined exponentially with increasing years under sugarcane. For the Glenrosa site, organic C reached a new equilibrium level of about 20 g kg^{-1} while at the Hutton soil the equivalent value was 41 g kg^{-1}. The higher organic matter content maintained at the Hutton site was attributed mainly to clay protection of organic matter since the clay content of the Hutton soil was 61 % compared to the 18 % for the Glenrosa soil. The loss of soil organic matter under sugarcane resulted in a concomitant decline in soil microbial biomass C, microbial quotient, basal respiration, aggregate stability, arylsulphatase and acid phosphatase activity. The activities of arylsulphatase and acid phosphatase like those for concentrations of microbial biomass and organic C, were higher for the Hutton than Glenrosa soils. At the Glenrosa site, the natural $\delta^{13}C$ abundance in soils was used to calculate the loss of forest-derived, native soil C and the concomitant input of sugarcane-derived C. Sugarcane-derived C increased over time until it accounted for about 61 % of organic C in the surface 10 cm in soils that had been under sugarcane for greater than 50 years.

The effects of agricultural land use (including burnt sugarcane) on organic matter content and related soil properties were compared with those under undisturbed native grassland in KwaZulu-Natal. Two separate farms situated on Oxisols were used and both contained fields with continuous long-term (>20 years) cropping histories. At site 1, soil organic C content in the surface 10 cm followed the order permanent kikuyu pasture > annual ryegrass pasture > native grassland > preharvest burnt sugarcane > maize under conventional tillage (CT). At site 2, organic C in the surface 20 cm decreased in the general order kikuyu pasture > native grassland > annual ryegrass pasture > maize under zero tillage (ZT) ≥ maize (CT). Organic C, microbial biomass C, microbial quotient, basal respiration and aggregate stability were substantially greater in the surface 5 cm under maize ZT than maize CT. In the undisturbed sites (eg native grassland and kikuyu pasture) the metabolic quotient increased with depth. By contrast under maize CT and sugarcane there was no significant stratification of organic C, yet there was a sharp decrease in metabolic quotient with depth. Aggregate stability was high under both native grassland and kikuyu pasture and it remained high to 40 cm depth under the deep-rooted kikuyu pasture. Although soil organic C was similar under maize CT and sugarcane, values for microbial biomass C, microbial quotient, basal respiration and aggregate stability were lower, and those for metabolic quotient and bulk density were higher, under sugarcane. This was attributed to the fallow nature of the soil in the interrows of sugarcane fields.

It was concluded that the loss of soil organic matter, microbial activity and aggregate stability is potentially problematic under maize CT and sugarcane and measures to improve organic matter status should be considered. For sugarcane, this could include green cane harvesting and the use of green manure crops in rotation.

CHAPTER ONE

GENERAL INTRODUCTION AND STUDY OBJECTIVES

In South Africa, about 40 000 ha of sugarcane is grown under a large variety of soils and climatic conditions (Meyer *et al.*, 1996). Grey sandy soils (Entisols) are the most prevalent soils under sugarcane cultivation, while red soils (Oxisols) are considered the second largest group of soils under cane (Meyer *et al.*, 1996).

Sugarcane (*Saccharum* spp) like many other crops, experiences reduced vigour and a plateau or even decline in yields after it has been under cultivation for many years (Moore *et al.*, 1991). This phenomenon, known as "yield decline", has been observed in many parts of the world including South Africa (Van Antwerpen and Meyer, 1996), Barbados (Anderson *et al.*, 1995), Fiji (Masilaca *et al.*, 1986), and Australia (Wood, 1985; Garside *et al.*, 1997).Such a yield decline occurs despite the introduction of new improved varieties which in trials have shown to reliably increase yields compared to the older varieties (Wood, 1985). Sugarcane yield is determined by three broad factors, namely climate, soil and management. It is difficult to associate sugarcane yield decline with any one factor (Anderson *et al.*, 1995). However, in a recent review, Garside (1997) defined sugar yield decline as "the loss of productive capacity of soils under monoculture" thus placing the emphasis on the soil factor.

Crop yield decline has been highly correlated with crop age and this has been attributed to successive ratoon damage and prolonged exposure to disease (Anderson *et al.*, 1995). Moore *et al.* (1991) suggested that unrecorded events such as untimely fertilizer applications, and reduced recoveries due to mill changes may play a silent role in yield decline. Inadequate fertilization may also play a role (Anderson *et al.*, 1995) since a balanced supply of nutrients is required to sustain sugarcane productivity (Yadav and Prasad, 1992). A build-up of root pathogens under sugarcane may not necessarily be a primary cause of yield decline, but does indicate an imbalance of other factors within the farming system (Garside, 1997; Hartemink and Kuniata, 1996). The overall decline in productivity may be partially due to increasing use of marginal land for cane cultivation, but the decline is still faster on older soils than the newer poorer soils (Wood, 1985). Soils are generally more degraded under old sugarcane compared to newly planted sugarcane (Garside *et al.*, 1997). Better yields are obtained on soils which have never had cane grown on them (Garside *et al.*, 1997; Wood, 1985), probably because they support a more advantageous biological environment (Wood, 1985) and have fewer detrimental soil organisms (Garside *et al.*, 1997). Soil crusting, erosion, compaction and waterlogging (depending on soil type) are other soil factors that limit cane yield production (Meyer *et al.*, 1996).

The most crucial factor associated with soil degradation under sugarcane is the loss of soil organic matter (Wood, 1985; Meyer *et al.*, 1996; Haynes and Hamilton, 1999). Cane yield decline is often associated with a decrease in soil organic matter content (Masilaca *et al.*, 1986). Older soils generally also have lower organic matter content, microbial biomass, pH and CEC, higher aluminium saturation and soil strength due to compaction compared to new sites (Fettell *et al.*, 1994; Garside *et al.*, 1997; Insam and Haselwandter, 1989). Thus it is thought that continuous cropping is responsible for soil degradation and hence cane yield decline (Wood, 1985).

A considerable decrease in organic matter content occurs in soils under sugarcane monoculture such that cane soils may have less than half the amount in uncultivated soils (Wood, 1985). The loss of organic matter has detrimental effects on soil biological, chemical and physical properties (Stevenson, 1994). An important objective of any sustainable agricultural system is to improve and maintain soil organic matter content (Gregorich *et al.*, 1994). Both the quantity and quality of organic matter are important. For example, the quality is determined by calculating the proportions of soil organic C and N in the mineralisable soil fractions, and these proportions indicate the quantity of readily-mineralisable C available to the microbial biomass, and the amounts of organic N that could become available at a later stage (Gregorich *et al.*, 1994).

The decrease in soil organic matter is concomitant with a decrease in soil biological activity, which is a cause for concern (Doran *et al.*, 1994; Doube and Schmidt, 1997) because biologically mediated processes are fundamental to their ecological functioning. Such functions include decomposition of organic residues, transformations of organic matter (Gregorich *et al.*, 1994), transformation and cycling of nutrients (Gregorich *et al.*, 1994), and formation and stabilization of soil aggregates (Oades and Waters, 1991; Dormaar and Foster, 1991).

Measurements of microbial biomass is relatively simple (Vance *et al.*, 1987), but do not indicate soil microbial activity which is an important soil biological parameter. Indices of microbial activity include basal respiration rate (CO_2-C evolved from a soil sample during incubation) and some enzyme activity assays eg acid phosphatase and arylsulphatase hydrolysis (Anderson and Domsch, 1993).

Sugarcane monoculture is a major land use on both the North and South coasts and into the midlands of KwaZulu-Natal. However, very little is known about the effects of sugarcane monoculture on soil organic matter content and quality, and soil biological activity (Haynes and

Hamilton, 1999). This information is relevant to sugarcane farmers, agricultural extension officers and environmental protection officers who are responsible for the management of land resources of the Province. However, comparisons of sugarcane soils to undisturbed veld and other major agricultural land uses in the area are necessary to make reasonable deductions.

This study was done in collaboration with staff of the Soil Science Department of the South African Sugar Association Experimental Station at Mount Edgecombe. The specific objectives were to:

a) Examine the effects of increasing periods under sugarcane on soil organic matter content, size and activity of microbial biomass, and aggregate stability in two contrasting regions of two different soil types within the sugarbelt of KwaZulu-Natal.

b) Compare the effects of long-term monoculture of sugarcane with that of maize (both conventionally and zero-tilled), kikuyu (perennial) and rye (annual) pastures, and undisturbed natural grassland on the amount and distribution of organic matter, microbial biomass size and activity, and aggregate stability within the soil profile of Oxisols in the midlands of KwaZulu-Natal.

There are five main sections to this thesis. After the introduction, the literature covering the important roles of soil organic matter, soil microbial biomass and activity, and aggregate stability in soil quality evaluation, and the effects of agricultural activities are reviewed and discussed in detail in Chapter 2. Chapters 3 and 4 relate and discuss the research outlined in the objectives a and b above. Conclusions and suggestions for further research are related in Chapter 5.

CHAPTER TWO

REVIEW OF LITERATURE

2.1 INTRODUCTION

Soil is not simply a substrate from which roots extract soluble nutrients, but a biological buffer, accommodating and transporting air, water, and nutrients, and interacting with the external environment (Doran *et al.*, 1994). The realisation that soil degradation is a serious problem under intensive agriculture has led to the increasing use of the concept of soil quality for assessments of agricultural sustainability (Doran and Parkin, 1994). Soil quality has been defined by Karlen *et al.* (1997), as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation." This definition is based on function which reflects the living and dynamic nature of soil (Doran *et al.*, 1996). Soil function requires the balance of three major components which are (i) sustainable biological productivity, (ii) environmental quality and (iii) plant and animal health.

Soil is recognised as a dynamic, living system that functions through a unique balance and interaction of its biological, chemical and physical components (Karlen *et al.*, 1997). Indicators of soil quality should measure more than soil function, be sensitive to soil changes over time due to disturbance and changes in land management, provide values which can be internationally compared, readily interpreted, and cost effective (Dalal, 1998). Generally, quality indicators can be separated into chemical (eg organic C, total N, pH and other soil test values), physical (eg bulk density and aggregate stability) and biological (eg microbial biomass C, enzyme analyses, and basal respiration) indicators (Doran and Parkin, 1994).

Organic matter is a fundamental aspect of soil quality (Doran and Parkin, 1994; Haynes, 1997) since it greatly influences soil chemical, physical and biological properties. It is composed of readily decomposable material such as plant litter and roots, dead and living organisms, and humus (Gregorich *et al.*, 1994). The non-humic substances generally have a rapid turnover and are used as substrates by soil microorganisms, whereas the humic substances are more resistant to decomposition due to their more resistant chemical structure (Gregorich *et al.*, 1994). Organic matter acts as a sink and source for plant nutrients, it maintains soil tilth, infiltration of water and air, water retention and reduces soil erosion. It is therefore useful in determining the quality of a soil (Gregorich *et al.*, 1994). The amount of mineralizable organic material in a soil can denote the quality of total organic matter, because it affects nutrient dynamics, the response of organic matter levels to changes in management, and carbon sequestration over prolonged periods of time (Gregorich *et al.*, 1994).

Total organic matter is quantified by measuring soil organic carbon and nitrogen. The size of many labile organic matter fractions can also be measured. Important pools include particulate organic matter, extractable and soluble fractions, potentially mineralisable C and N and microbial biomass C and N (Gregorich *et al.*, 1994; Heenan *et al.*, 1995).

Soil organic matter turnover is affected largely by climate, particularly soil moisture, and soil type, while sandy texture allows for a faster turnover rate by offering less protection to the organic matter than silt and clay (Cortez *et al.*, 2000). Soil organic matter content is determined by a balance between the magnitude of inputs of organic material and the rates of decomposition of organic matter. Organic carbon levels drop under intensive sugarcane cultivation (Hartemink, 1998; Van Antwerpen and Meyer, 1996), and cultivated soils may have less than half the amount of organic matter than uncultivated soils (Wood, 1985). This is because organic matter inputs are lower than in natural ecosystems and its rate of degradation is increased due to repeated tillage. However, sometimes organic matter levels may not be effected by tillage practices (Haynes and Knight, 1989). This may be due to changes in organic matter inputs and rates of decomposition balancing one another (Powlson and Jenkinson, 1981), or influenced by soil type (Meyer *et al.*, 1996). A redistribution of organic matter within the plough layer is caused by tillage, but it accumulates in the upper 5 cm of the soil when zero-tillage methods are used (Haynes and Knight, 1989). Soil management practice therefore affects both the quantity and distribution of organic matter in the soil profile.

There is increasing evidence that measurements of the size and activity of the soil biological community (eg microbial biomass C, basal respiratory rate, and enzyme activity) hold considerable promise as early indicators of soil degradation or improvement (Powlson and Jenkinson, 1981). In particular, such parameters are sensitive to changes in soil C availability, caused by alterations in soil management practice, and can change markedly before any changes in total organic matter content are detectable (Powlson *et al.*, 1987; Garside *et al.*, 1997).

When determining soil quality, researchers must examine the soil wholistically, not simply as independent functions. Data sets provide an overall picture of the soil quality, but the components of the data set are flexible to allow for a desired outcome in relation to the soil's function. Because soil organic matter plays a wide variety of roles in soil, it is universally accepted as an important indicator of soil quality (Gregorich *et al.*, 1994). The suitability of a particular soil to support plant growth depends on the biological activity, soil nutrition and soil structure. A well-structured data set will allow for accurate interpretation of the results in relation to the prevalent field conditions. In this review, the role of soil activity, soil enzyme activity and aggregate stability as indicators of soil quality in agricultural soils is reviewed and discussed. Particular emphasis is placed on the effects that agricultural land management have on these properties and the mechanisms.

2.2 ROLE OF SOIL ORGANIC MATTER IN SOIL QUALITY

The term "soil organic matter" refers to all the organic materials found in the soil including litter, light fraction, microbial biomass, water-soluble organics and stabilized organic matter (humus) (Stevenson, 1994). Soils vary greatly in organic matter content. Typically, a prairie grassland may contain 5 to 6 % organic matter in the top 15 cm of soil, while sandy soils may only contain 1 % organic matter (Stevenson, 1994).

It is difficult to measure organic matter content directly, but soil organic carbon and total nitrogen are indirect methods generally used (Gregorich *et al.*, 1994). The organic matter content is calculated from organic C measurements using a conversion factor, which is usually 1.727 (Stevenson, 1994). Total N measurements are multiplied by a conversion factor of 17.27 to depict the organic matter content of a soil (Stevenson, 1994).

2.2.1 Nature of Soil Organic Matter

Soil organic matter is composed of readily decomposable material such as plant litter and roots, dead and living organisms, and the resulting stable humus (Gregorich *et al.*, 1994). Organic C in soil represents materials of plant, animal and microbial origin (Kay, 1998). Plant residues are the major source of organic C in the soil (Paustian *et al.*, 1997). There are several stages of organic residue decay in soil (Stevenson, 1994). The first stage of decomposition is the decay of readily decomposable organic substances such as sugars and amino acids (Martin and Haider, 1986). The by-products of this first stage include CO_2, NH_3, H_2S, organic acids and other incompletely oxidised substances (Stevenson, 1994). The following stage involves the utilization of proteins and polysaccharides (such as cellulose and hemicellulose) which take longer to decompose (Martin and Haider, 1986). The final stage of decomposition involves the gradual decay of more resistant plant parts, including lignin, mainly by fungi and actinomycetes (Martin and Haider, 1986; Stevenson, 1994).

Initially, 40 to 60 % of the C is assimilated into the microbial biomass, and is later subject to biodegradation and transformation (Wagner, 1975), while only 5 to 10 % remains residual in the biomass (Jenkinson, 1971). Between 55 to 75 % of the C of most crop residues is lost as CO_2 within one year due to the respiratory activity of the soil biota (Martin and Haider, 1986). The remaining C is held within newly formed humus (Martin and Haider, 1986). Residual C becomes increasingly resistant to decomposition with time (Stevenson, 1994). Organic matter may be partitioned into two groups, namely the "active" and "inactive" pools (Stevenson, 1994). The "active" pool includes non-humic substances such as plant litter, light fraction, and other non-humic substances not bound to mineral constituents while the "inactive" pool includes humic substances (Stevenson, 1994).

2.2.1.1 Non-Humic Substances

Plant litter and particulate (light fraction) organic matter are non-humic substances (Stevenson, 1994) and are the precursors for the formation of other forms of organic matter. The non-humic fraction is readily decomposable and has a rapid turnover compared to humic substances (Gregorich *et al.*, 1994). Plant litter is deposited on the soil surface (leaves and trash after harvest), or within the profile via root turnover.

Light fraction is a transitory pool of organic matter (Gregorich and Janzen, 1996) and consists of mainly plant residues in various stages of decomposition in the soil (Stevenson, 1994). It can often account for 2 to 20 % of the soil's organic matter content (Janzen *et al.*, 1992) and even be as high as 40 % (Gregorich and Ellert, 1993). Variations are a result of different residue inputs and rates of decomposition (Stevenson, 1994). In agricultural ecosystems, the amount of total organic carbon held in the light fraction is highest under perennial forages and declines with time under arable crops or summer fallow (Janzen *et al.*, 1992).

Apart from plant litter and light fraction, another pool of non-humic substances includes readily metabolised soil organic compounds such as amino acids, carbohydrates, fats, waxes and organic acids (Tate, 1987). They may be of plant origin, or synthesised by microbial cells, and are quickly decomposed by soil microflora or stabilized into native soil organic matter (Tate, 1987). Thus, these compounds are generally transient within the soil and large concentrations do not usually accumulate (Tate, 1987).

The microbial biomass is another pool of non-humic substances (Stevenson, 1994), and is a dynamic, living component of soil organic matter (Jenkinson and Ladd, 1981). It represents the size of the total microbial population in the soil (Haynes, 1997). Bacterial- and fungal-feeding protozoa in soils are also important as they participate in organic matter turnover (Alexander, 1977b). Microbial biomass will be discussed in more detail in section 2.3.1.

Although the different labile fractions represent only a small portion of the total soil organic matter, they are very dynamic and account for much of the organic matter fluctuations over time (Cambardella and Elliott, 1992).

2.2.1.2 Humic Substances

In agricultural soils, most of the organic matter exists in the form of stable humus which is produced by biological degradation of plant and animal residues (Stevenson, 1994). Humus is the relatively biodegradation-resistant, mainly dark brown to black, fraction of the soil organic matter (Tate, 1987). The humus fraction consists of many organic compounds, but mainly humic acids and polysaccharides (Martin and Haider, 1986) and is of high molecular weight (Tate. 1987). Humic molecules are three-dimensional amorphous structures with many oxygen-containing functional groups (Tate, 1987). Humus molecules consist of a collection of organic compounds which bind to the free phenolic compounds (Tate, 1987). Thus, because of their complex structure and ability to be stabilized by bonding with clay (Edwards and Bremner, 1967), humic molecules are relatively stable (Gregorich *et al.*, 1994), resistant to microbial attack, and can last for between 20 and 200 years in a soil (Stevenson, 1994). Humic substances are thought to be formed by the condensation and polymerisation of phenols which are microbial metabolic products from litter decomposition (Tate, 1987).

The humified soil fraction provides negligible amounts of nutrients, unless the soils contain little or no easily metabolisable carbon substrates (eg in bare or fallow soils) in which case any nutrient supply is important (Tate, 1987). Humus binds to soil minerals to form soil aggregates, thus improving soil porosity and aeration. It improves water-retentive capacities of a soil, and acts as a sink for trace elements through complexation reactions. The association of humus with clay provides it with protection against microbial attack (Campbell and Lees, 1967), and the greater the amount of clay present in the soil, the greater the protection offered (Hassink *et al.*, 1993). Humus levels, however, may decline gradually due to microbial activity, and fall to a very low amount, particularly in soils which are continually cropped under poor management (Campbell and Lees, 1967). Increases in soil moisture will increase the decomposition of humus (Cortez *et al.*, 2000).

2.2.2 Functions of Organic Matter

Organic matter has a profound influence on many soil properties and is therefore a key attribute to soil quality (Doran and Parkin, 1994; Gregorich *et al.*, 1994). Many chemical, physical, and biological soil properties are dependent on the amount of soil organic matter (Moody *et al.*, 1999). The main properties and functions of humus are depicted in Table 2.1.

Table 2.1 General properties of humus and associated effects in soil (From Stevenson, 1994).

Property	Remarks	Effect on Soil
Colour	Dark due to presence of organic matter.	May facilitate warming.
Water retention	Organic matter can hold up to 20 times its own weight in water.	Helps prevent drying and shrinking, and improves moisture-retaining properties in sandy soils.
Association with clay minerals	Cements soil particles into structural units called aggregates.	Stabilises soil structure. Increases permeability.
Chelation	Forms stable complexes with polyvalent cations.	Increases micronutrient availability to higher plants.
Solubility in water	Organic matter is insoluble due to its association with clay.	Little organic matter is lost by leaching.
Buffer action	Induces buffering in slightly acid, neutral, and alkaline ranges.	Aids maintenance of uniform soil reactions.
Cation exchange	Total acidities of isolated fractions of humus range from 300 to 1400 cmoles/kg.	Increases cation exchange capacity (CEC) of the soil.
Mineralization	Decomposition of organic matter yields CO_2, NH_4^+, NO_3^-, PO_4^{3-} and SO_4^{2-}.	Source of nutrients for plant growth.
Combines with xenobiotics	Affects bioactivity, persistence and biodegradability of pesticides	Modifies application rate of pesticides for effective control.

Organic matter greatly affects nutrient availability to plants (Stevenson, 1994). It serves as a source of N, P and S after mineralization by soil microorganisms (Stevenson, 1994). Although all soil organic matter fractions are sources of nutrients such as nitrogen, sulphur and phosphate to some extent, the readily metabolisable fractions of the organic matter, are the primary contributors of plant nutrients (Tate, 1987). Such fractions include decomposing plant debris, roots and root exudates, and microbial biomass (Tate, 1987). Organic matter also influences nutrient supply from other sources. For example, it acts as a substrate for free-living N_2-fixing bacteria (Stevenson, 1994).

The large number of functional groups associated with humic molecules means that they have a high cation exchange capacity, are able to bind with polyvalent cations and xenobiotics, and form stable soil aggregates by association with clay minerals (Stevenson, 1994).

Soil organic matter greatly affects the structure of many soils (Stevenson, 1994). Aeration, water-holding capacity and permeability are all improved with high levels of soil organic matter (Stevenson, 1994). Soil organic compounds bind soil particles into stable structural units called aggregates (Tisdall and Oades, 1982). These aggregates maintain a loose, open, granular condition which aids gas exchange between the soil and the atmosphere, and water storage and transport (Stevenson, 1994). In a granular soil, particles are not easily carried by moving water, and granulation allows for water percolation down the profile, thus preventing soil erosion (Stevenson, 1994).

An important function of soil organic matter is to supply a metabolic energy source, such as carbon, to drive soil biological processes which then directly or indirectly effect other soil properties and processes (Roper and Gupta, 1995; Gregorich and Janzen, 1996). A major source of this carbon for use by heterotrophic organisms is plant residue input (Stevenson, 1994). Thus the particulate organic matter fraction hosts a large concentration of microorganisms hence great microbial activity because of ready substrate availability (Gregorich and Janzen, 1996).

Earthworms use surface litter as a carbon source, particularly under zero-tillage, and are thus involved in organic matter production and conservation (Haines and Uren, 1990). Most of the decomposition of plant residue is carried out by the microbial biomass, but the soil fauna enhance the process (Hendrix *et al.*, 1990). They do this by fragmenting plant residue thus increasing its surface area open to microbial attack (Hendrix *et al.*, 1990; Lavelle *et al.*, 1994). Comminution and mixing of soil during passage through the earthworms gut exposes physically protected organic matter to microbial attack, and intestinal mucilage promotes microbial activity (Lavelle, 1988). Macrofauna also distribute organic materials throughout the soil profile (Lee, 1985) creating greater contact between the substrate and attacking microflora.

The organic matter content of soil with an initially high level of organic matter will not necessarily increase with a further addition of inputs (Campbell *et al.*, 1991b). The difficulty in improving this soil characteristic was associated with a soil's limited capacity to hold organic matter (Campbell *et al.*, 1991b).

2.2.3 Organic C and Total N Content of Soils

Organic C and total N contents of a soil are a result of complex biochemical interactions between substate additions of C and N in fertilizers and plant and animal residues, and C and N losses through microbial decomposition and mineralisation, gaseous emissions to the atmosphere, leaching, and erosion (Gregorich *et al.*, 1994). Changes in inputs such as fertilizers and residues will be reflected in the total organic C and N content of the soil (Gregorich *et al.*, 1994). Moisture and temperature strongly influence organic matter decomposition in soil (Campbell *et al.*, 1981) such that the impact of management practices on organic C and total N will depend to some extent on the climate (Gregorich *et al.*, 1994). The effects of different land management practices will be reviewed and discussed in the following section.

The C:N ratio of soil organic matter may give an indication of a soil's ability to store and recycle energy and nutrients (Gregorich *et al.*, 1994). In agricultural soils, the C:N ratio generally remains within the range of 10 to 12 (Gregorich *et al.*, 1994; Stevenson, 1994). This ratio may change in response to land management changes such as fertilizer, tillage, and residue input changes (Gregorich *et al.*, 1994), depending primarily on soil type, and to some extent soil fertility (Dalal and Mayer, 1986). Prolonged sugarcane cultivation has been shown to increase the soil C:N ratio due to declines in N in excess of those of C (Bramley *et al.*, 1996; Van Antwerpen and Meyer, 1996). This may indicate increased humification of soil organic matter with increased sugarcane cultivation (Skjemstad *et al.*, 1999). However, cane soils under irrigation have been found to have a decrease in the C:N ratio compared to dryland (Van Antwerpen and Meyer, 1996). This reduction results from larger losses of C compared to N due to residue burning in irrigated fields (Van Antwerpen and Meyer, 1996). A greater return of plant residue return under dryland results in a higher C:N ratio where N losses are higher than C losses (Van Antwerpen and Meyer, 1996).

Changes in the C:N ratio are typically proportional to the rate of N loss or gain because the turnover of C may be delayed if there is not sufficient N for microbial decomposition (Rasmussen *et al.*, 1980). If inputs of each of these elements are significantly out of balance, then the efficiency of soil C sequestration is reduced (Paustian *et al.*, 1997). The C:N ratio of plant residues has an effect on the decomposition rates of the residues (Tate, 1987). Plant materials with a high C:N ratio do not provide sufficient nitrogen for metabolism of the decomposer populations under conditions of rapid microbial activity and thus decomposition is slowed down (Tate, 1987). If, however, N is added through fertilization, the decomposition rate resumes until carbon becomes limiting. In such a case the microbes are carbon-limited and require a carbon-source amendment to continue decomposition (Tate, 1987).

The C:N ratio of the mineralizable fraction of the organic matter indicates the composition of the active fraction of the soil organic matter (Gregorich *et al.*, 1994). For example, the addition of a cover crop with a high C:N ratio stimulates a fungal response, while an addition of a lower C:N ratio cover crop will increase bacterial activity (Lundquist *et al.*, 1999).

To make accurate assessments of the effects of land use and management practices on total organic C and N, the thickness and bulk densities of soil layers must be considered. However, because management practices may cause changes in bulk densities, comparisons of organic C and total N are made on a per area basis to a given soil depth (eg, Mg ha^{-1} to a depth of 30 cm) (Gregorich *et al.*, 1994).

2.2.4 Effects of Land Management Practices

A number of reviews cover the influence of land management practices on soil organic matter levels (Gregorich and Janzen, 1996; Haynes and Beare, 1996; Paustian *et al.*, 1997). The main features are reviewed below.

2.2.4.1 Arable Cropping

The most dramatic effect of agricultural land use on soil organic C is associated with initial cultivation of native soils (Paustian *et al.*, 1997). Initially, the organic C levels decline sharply within the first 10 years of cultivation, and eventually stabilise at a new equilibrium after 50 to 100 years (Paustian *et al.*, 1997). The decline is a result of both reduced inputs of organic material and an increased rate of organic matter decomposition induced by regular cultivation (Haynes and Tregurtha, 1998). Concomitant with the loss of organic C, there is a decline in both soil biological activity and physical properties (Haynes and Tregurtha, 1998). The level of the new equilibrium is determined by the ability of the soil to stabilize and protect organic C with clay, and the amount, quality and distribution of plant residue inputs (Tate, 1987).

The reasons for a decline in soil organic matter under arable systems are numerous. The rapid initial decline is an immediate response to the reduced carbonaceous residue input (Campbell *et al.*, 1991a). The input is reduced due to wide row spacings, harvesting of aerial plant parts, and burning of remaining trash. Another reason for a decline is the disruption of soil structure as cultivation disrupts soil aggregates exposing previously-protected organic matter to microbial attack (Tisdall and Oades, 1982; Gupta and Germida, 1988; Haynes and Beare, 1996). Cultivation also improves the abiotic conditions for microbial activity, namely soil moisture, aeration and temperature, thus there is an increase in microbial activity, further depleting the remaining organic matter (Paustian *et al.*, 1997).

The intensity with which the soil is cultivated can affect both the total amount of soil organic C and its distribution within the soil profile (Haynes and Knight, 1989). Some cultivated soils may have less than half the amount of organic matter than uncultivated soils (Wood, 1985). Residue inputs are generally more concentrated at the soil surface under zero-tillage, thus organic C (Franzleubbers *et al.*, 1995) and total N (Haynes and Knight, 1989) accumulate in the surface soil layers (Fig 2.1) (Powlson and Jenkinson, 1981; Paustian *et al.*, 1997). In contrast, under conventional tillage, soil organic matter is more evenly distributed throughout the soil profile due to inversion of soil layers by ploughing (Haynes and Knight, 1989; Woods, 1989; Haines and Uren, 1990). It must be noted however, that some workers have found that the difference in tillage practices may not be significant when considering the top 20 cm of soil (Powlson and Jenkinson, 1981; Haynes and Knight, 1989). Although different tillage practices may influence the depth distribution of microbial biomass and organic matter, they may not always cause an overall increase of these characteristics throughout the soil profile (Gupta *et al.*, 1994; Dalal, 1998). This may be due to organic input into the soils being the same for both conventional and zero-tillage methods (Powlson and Jenkinson, 1981), or determined by soil type (Meyer *et al.*, 1996).

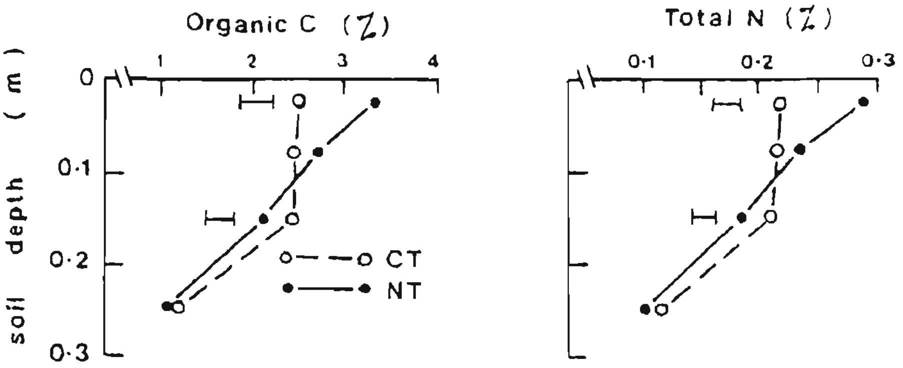

Fig 2.1 The effect of conservational tillage (CT) and zero-tillage (ZT) on organic carbon and total nitrogen concentrations down the soil profile of a stony silt loam (From Haynes and Knight, 1989).

Rotations that incorporate more organic matter into the soil allow for a higher organic matter equilibrium within the soil compared to rotations where the input is smaller (Blair *et al.*, 1998). The effects crop rotations have on soil organic matter content depend on the levels of plant residue input offered by the rotation combination (Paustian *et al.*, 1997). For example, organic matter returns are generally greater for cereal crops than root crops where below-ground matter is harvested (Haynes and Beare, 1996). The harvest of root crops also disturbs the soil structure thus encouraging the decomposition of native soil organic matter (Haynes and Beare, 1996).

Leaving land fallow is detrimental to organic matter, because the more extended that period of fallow, the less the plant residue input (McGill *et al.*, 1986; Janzen, 1987; Gregorich and Janzen, 1996; Moody *et al.*, 1999). In addition, fallow soils are often tilled to keep them weed-free, thus breaking up soil clods and promoting organic matter decompostion (Haynes and Beare, 1996).

Fertilizer additions increase crop production and therefore plant residue inputs, thus increasing soil organic matter contents of a soil (Biederbeck *et al.*, 1994; Paustian *et al.*, 1997). Fertilizer applications may thus prevent humus depletion (Campbell and Lees, 1967). However, fertilizer applications will not improve organic matter contents when the soils C-holding capacity is essentially saturated (Campbell *et al.*, 1991b). Nitrogen applications may be involved in C stabilization by improving the growth efficiencies of soil microbes therefore reducing the amount of C lost through respiration (Paustian *et al.*, 1992).

The loss of soil organic matter under sugarcane has been found to be substantial (Van Antwerpen and Meyer, 1996; Blair *et al.*, 1998; Hartemink, 1998; Skjemstad *et al.*, 1999). This may be due to the conventional cultivation every 6 to 7 years before replanting, the relatively wide row spacing leaving large interrows fallow for long periods of time, and the common practice of burning prior to harvest resulting in the loss of C and N which would otherwise be returned to the soil (Rasmussen *et al.*, 1980). In sugarcane fields, it has been found that organic carbon contents of a soil will decline in the surface layers of the soil, but increase in the lower layers of the soil profile after a prolonged period of time (Blair *et al.*, 1998; Skjemstad *et al.*, 1999). This may be due to the presence of charcoal from harvest burnings of the cane accumulating deep in the soil profile (Skjemstad *et al.*, 1999). This increase of charcoal accumulation may mask possible losses of labile forms of carbon which are necessary for soil structure and fertility (Skjemstad *et al.*, 1999). Perhaps a more likely explanation of C accumulation deeper within the profile would be the inversion of soil layers from tillage practices (Haynes and Hamilton, 1999).

Greencane harvesting releases active carbon thus the reduction of carbon in the surface layer will not be as large as when the crop is burnt (Blair *et al.*, 1998). Thus the large amounts of high quality residue offered by cane has a large effect on carbon dynamics within the soil (Blair *et al.*, 1998). A shift from burning to trash retention is a practice that can significantly increase soil organic matter levels under cane (Wood, 1985; Heenan *et al.*, 1995; Van Antwerpen and Meyer, 1996). In short, management practices that increase C inputs to soil and reduce decomposition rates associated with intense tillage will promote maintenance and buildup of soil C (Paustian *et al.*, 1997).

2.2.4.2 Pastoral Farming

Long-term pastoral management generally results in the accumulation of a high soil organic matter content (Haynes and Williams, 1993). The conversion of arable land to pasture results in an increase in organic matter content due to the higher plant dry matter production typical of pastures (Haynes and Beare, 1996) as a result of irrigation and fertilization. Again, it must be noted, that pastures will not necessarily increase a soil's organic matter content if the soil's capacity to hold organic matter is already reached (Campbell *et al.*, 1991b). Organic matter inputs under pasture come from senescing plant tops and roots, exudation of organic compounds from pasture roots, turnover of the large microbial biomass found in the root rhizosphere, and return of ingested material from animals in the form of dung (Haynes and Beare, 1996; Haynes and Williams, 1999). A large input of below-ground particulate organic matter due to grass root turnover and the absence of tillage are probably the two key components leading to organic C sequestration under permanent pasture (Paustian *et al.*, 1997).

The inclusion of perennial crops in rotations, particularly grasses, can sometimes improve the levels of soil organic matter due to the lack of tillage, improved aggregation by the roots of the crop, increased plant residue input, and reduced rate of organic matter decomposition (Paustian *et al.*, 1997; Haynes, 1999a; Haynes, 1999b). This system is known as ley-arable farming and includes 3 to 6 years of pasture alternated with 3 to 4 years of arable crops (Haynes and Beare, 1996). The equilibrium level of soil organic matter can be higher compared to continuous arable, but considerably lower than that in continuous pastures (Haynes and Beare, 1996). The soil organic matter content does not always increase in ley-arable farming, as the cultivation for arable crops will promote degradation of the organic matter accumulated during the pasture phase (Paustian *et al.*, 1997). For the soil C to increase over time, the contribution of pastures must be high enough to increase C inputs over the course of the rotation (Paustian *et al.*, 1997).

The effects of annual pastures, which are conventionally cultivated and resown each year (as is common practice in the South African dairy industry), on soil organic matter levels is largely unknown. However, in a 5 year study comparing annual grass with continuous grass, Haynes (1999a and 1999b) found that the use of annual grass did not result in an accumulation of organic matter, probably due to the destructive effects of annual tillage. When annual grass is sown by direct-drilling and tillage was therefore avoided, there was some accumulation of organic C in the surface soil layers compared to long-term arable fields, although the accumulation was not necessarily significant in the short-term (Haynes, 1999a). As already noted, direct-drilling has been found by many workers to aid the maintenance of soil organic carbon levels particularly in the surface soil layers in the long-term (Haines and Uren, 1990; Gupta *et al.*, 1994; Heenan *et al.*, 1995).

2.2.5 Potentially Mineralizable C and N

More than 75 % of the soil organic matter exists as slowly decomposable compounds, while the remainder consists of readily decomposable or mineralizable compounds (Gregorich *et al.*, 1994). The amounts of mineralizable organic material in a soil can denote the quality of total organic matter, because it affects nutrient (particularly N) cycling and availability within a single growing season, the response of organic matter levels to changes in management (Gregorich *et al.*, 1994). A distinction between the quality and quantity of soil organic matter is made possible by calculating the proportions of soil organic C and N in the mineralizable fractions. If the proportions of soil C and N remain unchanged through alterations in management techniques, regardless of changes in the absolute amounts of total or mineralizable C and N, the quality of soil organic matter is unchanged (Gregorich *et al.*, 1994).

Potentially mineralizable nitrogen is a measure of maximum N mineralised under defined conditions from soil organic N that may become available to plants (Dalal, 1998). It is measured as the release of inorganic N (NH_4^+ plus NO_3^-) after incubation for a set period of time (Gregorich *et al.*, 1994). Nitrogen mineralisation is measured to determine the ability of a soil to supply inorganic N mainly in the form of NO_3 which is the chief form of plant-available and mobile (leaching) N (Gregorich *et al.*, 1994). Moisture, N source, temperature and the soil matrix will determine the amounts of N mineralised (Paul, 1984). Similarly, Franzleubbers *et al.* (1995) found that the specific mineralisation activity of soil microbial biomass is influenced greatly by seasons.

Carbon mineralisation is the gross flux of CO_2 from the mineralizable fraction and it represents the total metabolic activity of heterotrophic soil organisms (Gregorich *et al.*, 1994). It can also be used to assess the decay of plant tissues, the persistence of organic wastes, the release of CO_2 from organic matter into the atmosphere, and the effect of pollutants on soil organism activities (Gregorich *et al.*, 1994). Carbon mineralisation is affected significantly by crop rotation, tillage, N fertilization, and stubble management (Pankhurst *et al.*, 1995).

Carbon and nitrogen mineralisation data is difficult to compare between studies due to a variety of techniques for their measurement (Gregorich *et al.*, 1994). Time series measurements indicate that the release of CO_2 and inorganic N from the mineralizable fraction is curvilinear during incubation of the samples (Gregorich *et al.*, 1994). The amounts of both mineralised C and N measured during any incubation time period depends on the conditions of incubation, thus soil moisture and temperature must be controlled (Gregorich *et al.*, 1994). The duration of incubation determines the amount collected (Gregorich *et al.*, 1994). Mineralisation of N is highest during the first week of incubation (Lundquist *et al.*, 1999). Treatment of the sample prior to incubation, for example drying and rewetting, will cause a flush of mineralisation and therefore a loss of C and

N prior to measurement (Gregorich *et al.*, 1994).

2.3 BIOLOGICAL INDICATORS OF SOIL QUALITY

Soil biota are the main driving force behind nutrient and energy transformations in soils (Banerjee *et al.*, 1999). A wide variety of both micro- and macroorganisms is found in soils. Microorganisms include algae, bacteria, fungi, and protozoa. Macroorganisms include some invertebrates for example, earthworms, micro- and macroarthropods, and termites. Soil biota affect nutrient availability to plants by transforming organically-held nitrogen into inorganic plant-available forms (Roper and Gupta, 1995). They also affect soil structure by being responsible for the decomposition and accumulation of organic matter as well as by burrowing and depositing their excreta in the soil (Roper and Gupta, 1995). The presence, diversity and activity of organisms are considerably affected by soil management (Roper and Gupta, 1995). Soil biological factors are considered by Garside *et al.* (1997) to be the most responsive of all soil factors to conditions imposed by sugarcane cropping.

Macroorganisms break up plant residues into smaller units which, in turn, promotes microbial decomposition (Hendrix *et al.*, 1990). The organic materials are also moved around within the soil profile by macroorganisms such as earthworms (Lee, 1985). Microorganisms metabolise cellulose, hemicellulose, other polysaccharides, hydrocarbons, and lignin, thus releasing carbon as an energy source for other microorganisms involved in nutrient transformations such as nitrogen and sulfur mineralisation (Roper and Gupta, 1995). The carbon made available by organic matter turnover is used as an energy source for microbial activity. Microorganism activities are limited by the availability of soil moisture and an energy source such as carbon

(Roper and Gupta, 1995).

Soils may differ in their ability to support a diverse range of micro- and macro-organisms due to differences in the physical and chemical properties of the soil (Alexander, 1977a). For example, many microorganisms are very specific as to which pH range is suitable for life (Alexander, 1977a). Nitrogenous fertilizer applications can cause a decrease in soil pH, which would result in the reduction of size and activity of many heterotrophic microorganisms, thus reducing the microbial biomass (Ladd *et al.*, 1994). However, if pH and other conditions remain unchanged as a result of fertilization, the size of the microbial biomass may increase due to the additional nutrients being made available (Biederbeck *et al.*, 1984). Soils with a higher clay percentage support more soil biota by offering protection from predation, parasitism (Roper and Marshall, 1978), and desiccation (Bushby and Marshall, 1977). Generally, the use of no-till practices instead of conventional tillage results in a greater number of microorganisms closer to the surface of the soil, due to the accumulation of plant residues as substrate on the soil surface (Haynes and Knight, 1989). Nevertheless, continuous herbicide applications can significantly reduce certain microorganism activities and populations (Roper and Gupta, 1995).

Because soil organisms and their activities strongly affect soil properties and processes, and their abundance is largely affected by soil management practice, they serve as important indicators of soil quality (Karlen *et al.*, 1997).

2.3.1 Soil Microbial Biomass

Microbial biomass is the small (1 to 4 %) living component of soil organic matter (Jenkinson and Ladd, 1981), excluding macrofauna and plant roots (Sparling, 1997). Microbial biomass is fundamental to organic matter cycling and carbon sequestration by the soil (Dalal, 1998) since it is both an agent for decomposition of plant residues (Haynes, 1997) and a sink for C, N, P, S and other nutrients (Dalal, 1998). Microbial biomass carbon and nitrogen measurements are useful in determining the response of the soil's total microbial biomass to changes in agricultural management (Roper and Gupta, 1995). The living and dynamic nature of the microbial biomass makes it more responsive to changes in soil management than the total organic matter content (Powlson *et al.*, 1987). The microbial component of the soil acts as a store of labile organic matter (Gregorich *et al.*, 1994; Haynes, 1997). Microbial biomass measurements are very useful in determining the effects of climate, cultural practices, land use and management on soil organic matter status although it is difficult to make comparisons between different climates, land uses and soil types (Sparling, 1997).

Microbial biomass is operative in plant production by performing a number of important functions. It serves as both a sink and a source for carbon, nitrogen, phosphorus, and sulfur. It is also involved in nutrient transformations, pesticide degradation, and the formation and stabilization of soil structure (Dalal, 1998). Bacteria and fungi, for example, produce polysaccharides and other organic materials which contribute to the binding of soil particles together to form macroaggregates thus contributing to soil structure (Tisdall and Oades, 1982). Fungi physically join soil particles creating stable aggregates with their hyphae, mucilage and extensive filamentous nature (Tisdall and Oades 1982; Gupta and Germida 1988).

2.3.2 Factors Affecting Microbial Biomass

The amount of microbial biomass a soil can sustain depends on many factors such as the protection of the microbial biomass offered by the clay fraction (Sørensen, 1983) and within aggregates (Breland and Eltun, 1999), climate (Insam and Haselwandter, 1989), organic matter content and soil management practice (Hassink and Whitmore, 1997).

2.3.2.1 Seasonal Changes

Temperature and moisture play a dominant role in determining the amount of microbial biomass in the soil (Wardle and Parkinson, 1990), and the cycling of C, S and P (He *et al.*, 1997). An increase in mean annual temperature has been found to decrease the amount of soil microbial biomass (Dalal and Mayer, 1987; Alvarez *et al.*, 1995). Microbial biomass is also limited by a higher ratio of precipitation to evaporation, thus the greater the soil moisture content, the smaller the microbial biomass (Insam *et al.*, 1989). However, at very low soil moisture contents, the microbial biomass activities are also limited (Roper and Gupta, 1995). Drying out of the soil results in a large decrease in total microbial biomass C and N contents (Cortez, 1989). Seasonal changes (moisture, temperature and substrate supply) result in relative changes in the microbial biomass and hence plant growth rate (Dalal, 1998). Higher microbial biomass values are measured at times of the year with the largest root biomass (Lynch and Panting, 1982) and greatest plant residue input (He *et al.*, 1997; Heenan *et al.*, 1995).

2.3.2.2 Clay Content

The soil matrix accommodates microbial biomass, holds substrates for microbial use, and provides a medium within which biosynthetic reactions can take place (Dalal, 1998). Fig 2.2 indicates that only part of the soil's organic matter content is protected at any one time (Hassink and Whitmore, 1997). The net amount of organic matter accumulated does not only depend on the protective capacity of the soil, but on the extent to which this capacity is already occupied by organic matter (Hassink and Whitmore, 1997).

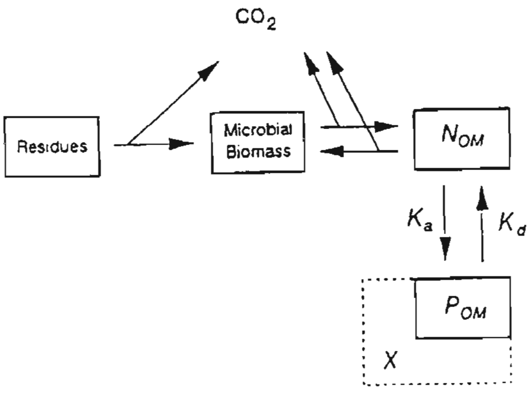

Fig 2.2 The turnover of organic matter through the microbial biomass. N_{OM} is the non-protected organic matter, P_{OM} is protected organic matter, X the capacity of the soil to protect organic matter, K_a the rate constant for protection/sorption, and K_d the rate constant for desorption. (From Hassink and Whitmore, 1997).

Clay allows for an increase in microbial biomass until the clay content reaches 50 % of the soils matrix, at which stage microbial biomass values tend to plateau (Dalal and Mayer 1987; Powlson and Jenkinson 1981). Clay provides more sites of refuge for protection for the microbial biomass and organic matter than sand (Hassink and Whitmore, 1997). Clay aids the survival of microorganisms in stressed conditions such as drying of the soil (Weigand *et al.*, 1995). It also generally stimulates microbial respiration (Weigand *et al.*, 1995). Thus clay soils generally contain more microbial biomass than sandy soils, and the microbial biomass turnover is higher in sandy soils (Hendrix *et al.*, 1998). This may be due to increased protozoan feeding due to the greater accessibility of the microflora within larger pores of sandy soils (Juma, 1993). Hendrix *et al.* (1998) also suggested that clay soils have a deeper distribution of microbial biomass in the soil profile than in sandy soils. This reflects a deeper incorporation of organic residues by earthworms in clay soils as well as more favourable conditions for microbial biomass accumulation.

2.3.2.3 Crop Rotations, Fertilizers and Soil Amendments

Plant biomass input may be increased by certain crop rotations (specifically those including pastures), thus increasing the size of the microbial biomass (Collins *et al.*, 1992). Crop rotations which include legumes provide an additional nitrogen source through symbiotic nitrogen fixation, thus affecting the size and activity of the soil microbial biomass (Biederbeck *et al.*, 1984). The amounts of N added depend on the type of legume, for example greater N mineralisation occurs under clover compared to lupins because a large proportion of N is removed from the soil in the harvested lupin grain (Heenan and Chan, 1992). Rotations that incorporate large amounts of organic matter into the soil generally maintain a higher soil microbial biomass (Blair *et al.*, 1998).

Nitrogen fertilizer, when applied to cereal crops, generally improves plant growth and plant biomass input, and as a result microbial biomass carbon and nitrogen are increased (Campbell *et al.*, 1991a). However, Ladd *et al.* (1994) found that an application of 80 kg N/ha resulted in a diminished microbial biomass due to soil acidification induced by nitrification. Applications of organic amendments such as farmyard manure generally increase soil microbial biomass unless the amendments are contaminated with heavy metals, or cause an unfavourable change in pH (Roper and Gupta, 1995).

2.3.2.4 Tillage Practices

Conventional tillage involves rotary cultivation or the use of a mouldboard plough for seedbed preparation, planting, weed control, and burial of plant residues (Haynes and Knight, 1989). Zero-tillage methods imply the use of herbicides for weed control, limited soil disturbance during seed planting and the lack of fertilizer incorporation into the soil (Haynes and Knight, 1989). Microbial biomass measurements provide an early indication of changes in soil quality due to altered tillage practices (Carter and Rennie, 1982). For example, a significant increase in microbial biomass C and N was found in the top 5 cm of the soil profile after only one year of no-till practice, but organic C and total N measurements remained similar over that time period for both conventional and no-till practices (Gupta *et al.*, 1994). Similar trends were found after a period of seven years (Haines and Uren, 1990). However, low plant yields and hence low plant residue input into the soil, may in some circumstances prevent a significant increase in microbial biomass, organic carbon and total nitrogen measurements even after 13 years of no-till practice (Fettell *et al.*, 1994).

Organic matter (ie organic C and total N) and microbial biomass levels decrease greatly with increasing soil depth when zero-tillage is practised compared to conventional tillage (Fig 2.3) (Powlson and Jenkinson, 1981). Tillage allows for a downward redistribution of organic matter and microbial biomass within the plough layer (Fig 2.3) (Carter and Rennie, 1982; Haynes and Knight, 1989; Haines and Uren, 1990; Franzleubbers *et al.*, 1995). Zero tillage may result in significantly greater microbial biomass measurements in the top five cm of soil compared to conventional tillage, but the difference in tillage practices may not be significant when considering the top 20 cm of soil (Powlson and Jenkinson, 1981). Indeed, although different tillage practices may influence the depth distribution of microbial biomass and organic matter, they may not cause an overall change in these characteristics when considered throughout the soil profile (Dalal, 1998).

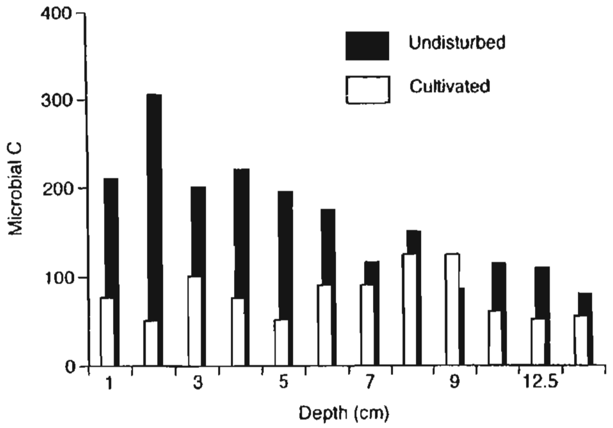

Fig 2.3 The effects of cultivation and zero-tillage on the distribution of microbial biomass carbon within a depth of 15 cm (From Sparling, 1997).

2.3.2.5 Burning of Crop Residues

The common practice of burning crop residues rather than returning them to the soil reduces the soil organic matter content as well as the microbial activity (Rasmussen *et al.*, 1980). It results in a decline of microbial biomass carbon and nitrogen due to the smaller return of organic matter, and the limited amount of mineralizable carbon for the microbial biomass (Collins *et al.*, 1992). Most of the C and N in the residues is lost via volatilization during burning (Biederbeck *et al.*, 1980). Burning indirectly reduces the soil microbial activity by reducing the soil organic matter, but it also has direct effects. For example, Biederbeck *et al.* (1980) found an immediate decline in both the fungal and bacterial populations following burning of cereal straw. Since burning generally reduces soil microbial biomass, in sugarcane production, the conversion of burning to green cane harvesting with retention of a trash blanket results in an increase in microbial biomass in the surface soil (Wood, 1991).

2.3.2.6 Pasture

Soils under pasture usually contain considerably larger amounts of microbial biomass than cultivated soils (Breland and Eltun, 1999). The major contributing factor is the large amounts of organic matter that are returned to the soil under pasture, especially that from root turnover (see section 2.2.4.2) (Campbell *et al.*, 1992). Carbon input is usually regarded as the limiting factor to the size of the soil microbial biomass (Anderson and Domsch, 1985) although sometimes inorganic phosphorus concentrations may be limiting due to P deficiencies (Srivastava, 1992). The microbial biomass is therefore characteristically high under pastures (eg 1200 μg C g^{-1}) and serves as a large labile pool of nutrients (Haynes and Williams, 1993).

2.3.3 Measurement Limitations

Microbial biomass measurements are laborious and cumbersome, they do not provide benchmark values, may be difficult to interpret, and do not clearly represent the productivity of the soil; these problems should be overcome if microbial biomass measurements are to be used as an indicator of soil quality (Dalal, 1998). Microbial biomass cannot independently provide a measure of microbial activity (Haynes, 1997). A measure of microbial biomass turnover such as CO_2 evolution or enzyme activity will indicate microbial activity (Anderson and Domsch, 1993). Differences induced by environmental conditions are reduced by using microbial indices such as the microbial quotient (the percentage of total organic C present as microbial biomass C) for soils of differing organic matter contents (Powlson and Jenkinson, 1981).

2.3.4 Microbial Quotient

Organic matter dynamics are partly reflected by the ratio of microbial biomass C to total organic C, ie the microbial quotient (Carter, 1991). Microbial biomass carbon has been found to be highly correlated to the organic carbon content of the soil (Anderson and Domsch, 1989). Climate was found by Insam *et al.* (1989) to have a low correlation with microbial biomass carbon and total organic carbon, but when compared with the microbial quotient, the correlation was high, demonstrating the importance of the use of the microbial quotient. The use of a ratio, as such, prevents the problems of working with absolute values, and changes in organic matter contents between soils (Sparling, 1997). The microbial quotient is used to monitor organic matter changes in agricultural soils (Carter and Rennie, 1982; Sparling, 1992). This ratio must be compared against a control such as native grassland on the same soil type (Carter, 1991). The control should have all conditions similar to those of the sample plot eg climate, mineralogy, and soil texture to allow for a real comparison (Sparling, 1992).

A pasture, when compared to an arable field, has a greater microbial quotient (Breland and Eltun, 1999; Sparling, 1992) due to greater carbon input via plant residues (Breland and Eltun, 1999).The microbial quotient does not always increase proportionally with an increase in total organic carbon content. For example, a greater increase in microbial biomass carbon compared to total organic carbon will increase the microbial quotient (Sparling, 1997). Monocropping can result in a decrease of the microbial quotient compared to multicropping (Anderson and Domsch, 1989). Generally, the more a soil is abused, and the faster the microbial biomass carbon pools will decline compared to the organic carbon content, the faster the microbial quotient will decrease (Sparling, 1997). This is evident in Fig 2.4 where the decline of both organic carbon and microbial biomass were erratic, probably due to spatial variability, but the microbial quotient showed a smoother decline with the sharpest decline being in the first 3 to 4 years (Sparling, 1992). In the early stages of ecosystem development the microbial quotient tends to be higher than in established soils (Insam and Haselwandter, 1989). Soil type will also affect the microbial quotient. Generally, sandy soils give lower microbial quotients, but different trends have been noticed elsewhere (Sparling, 1992).

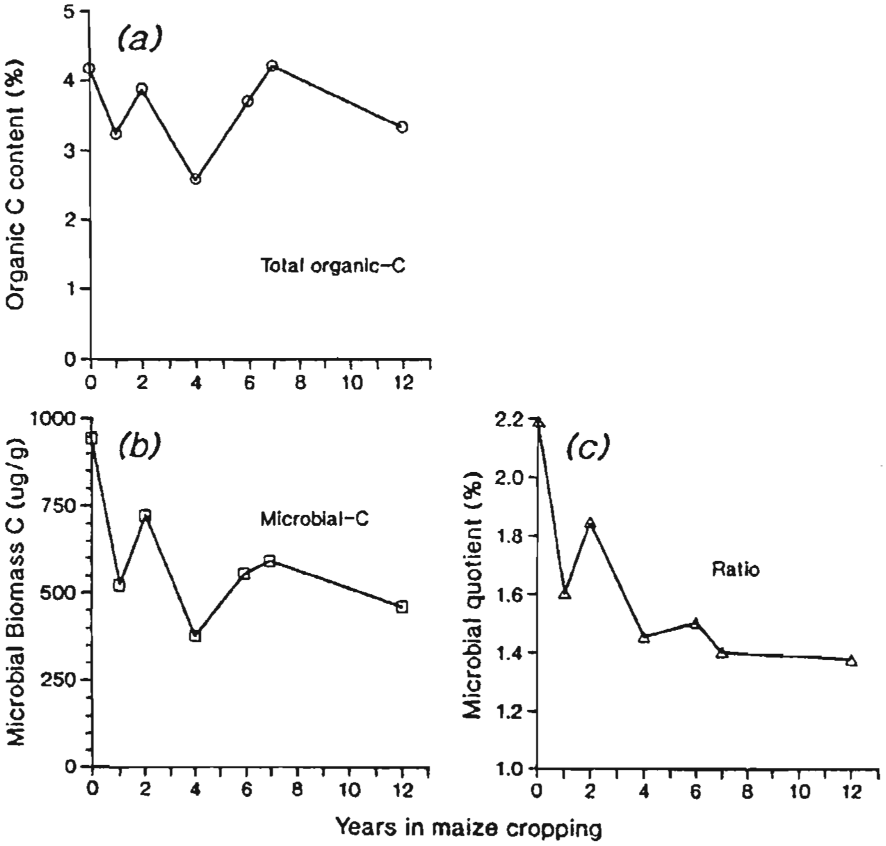

Fig 2.4 Changes in organic carbon (a) and microbial biomass carbon (b) contents over time under continuous maize cropping, and the resultant microbial quotient (c) (From Sparling, 1992).

2.3.5 Basal Respiration

CO_2 production, which represents the biological activity of a soil, is related to the amount of available organic matter, and not necessarily to the size of the microbial biomass (Sørensen, 1983). Soil respiration is highly variable depending largely on changes in substrate availability, moisture and temperature (Alvarez *et al.*, 1995). Thus respiration measurements are often made under controlled laboratory conditions (Anderson, 1982) and results are termed basal respiration. Where moisture and temperature are not limiting, the rate of CO_2 release may indicate organic matter quality and if the soil environment is conducive to decomposition (Sparling, 1997).

2.3.6 Metabolic or Respiratory Quotient

Organic matter dynamics may also be shown by the ratio of basal respiration (CO_2-C respired under incubation) to microbial biomass carbon, ie the respiratory quotient (qCO_2) (Sparling, 1997). The respiratory quotient may reflect the balance of input and output of organic matter to and from the soil, and the efficiency of the conversion of organic matter to microbial biomass C (Gregorich *et al.*, 1994). The respiratory quotient is an index of the effect of adverse environmental conditions, such as stress and disturbance, on the microbial biomass (Wardle and Ghani, 1995). With development of an ecosystem, the qCO_2 should decline, but stress and disturbance will induce a rise in the qCO_2 (Wardle and Ghani, 1995). Adverse conditions (stress and disturbance) reduce the efficiency of the microbial biomass, thus enhancing qCO_2 (Wardle and Ghani, 1995). The metabolic quotient must be used with care as interpretation becomes very difficult once quality and amounts of carbon substrates in the soil, and soil types, differ (Sparling, 1997).

Another complication is the unpredictable response of qCO_2 to disturbance, and the confounding effects of microbial responses to stress (Wardle and Ghani, 1995). For example, qCO_2 was found by Anderson and Domsch (1990) to be higher in rotational cropping compared to monoculture, while the opposite was found to be true by Lundquist *et al.* (1999). Lower qCO_2 values under pasture may be caused by a reduction in active biomass due to physical protection within aggregates, and larger organic matter supplies from roots, crop residues and manures (Breland

and Eltun, 1999). In a young developing ecosystem, there is likely to be a larger biomass and lower respiratory quotient than in developed ecosystems (Insam and Haselwandter, 1989). Alvarez *et al.*, (1995) suggested temperature increases cause an increase in the respiratory quotient, ie an increase in the microbial population's respiratory rate, relative to the amount of microbial biomass. The metabolic or respiratory quotient will generally be higher where a soil is more stressed (Insam and Haselwandter, 1989).

2.3.7 Soil Enzymes

2.3.7.1 General

Although enzymes are not classed as soil biota, they are the products of the microbial biomass. Soil enzymes are proteinaceous biological catalysts that function in reducing the energy required to activate biochemical reactions (Gregorich *et al.*, 1994), particularly those related to soil organic matter formation and degradation (Tate, 1987). They are involved in many soil functions such as decomposition of organic inputs, transformations of native organic matter, release of inorganic nutrients for plant use, N_2 fixation, nitrification, and denitrification (Dick, 1997). Each enzyme is specific to the type of chemical reaction in which it participates (Tabatabai, 1982). Soil enzymes are synthesized by living plants and soil organisms during metabolism, and found in living organisms (biotic enzymes), in dead material from plants and organisms, or complexed within soil organic and mineral colloids (abiotic enzymes) (Gregorich *et al.*, 1994). However, the enzymes from each of these sources cannot be measured individually (Tabatabai, 1982). Soil enzyme activities include both biotic and abiotic components (Burns, 1982). Extracellular activities are usually linked to various soil colloidal fractions, for example humic substances and clays (Tate, 1987). Intracellular enzymes are produced for cellular metabolism, and extracellular enzymes are exuded by microorganisms into soil solution to hydrolyse substrates that are too large for microbial uptake, detoxifying the surrounding environment, and creating a favourable environment for the survival of the microorganism (Dick, 1997). Enzyme activities are affected by changes in the environment and land management techniques, thus are useful indicators of soil quality (Gregorich *et al.*, 1994). However, the total enzymatic activity of a soil depends on the levels of both intra- and inter-cellular enzymes and separation of these two fractions is difficult (Gregorich *et al.*, 1994).

The decomposition of organic matter and amendments added to the soil, is catalysed by a series of enzymes with different functions acting in a cascade manner (Gregorich *et al.*, 1994). Enzymes are also involved in breaking down plant residue components such as cellulose by depolymerisation into the basic structure of humus (Gregorich *et al.*, 1994). Organic N is mineralised into NH_4^+ by a series of enzymes such as amidases, deaminases, proteases and ureases (Gregorich *et al.*, 1994). Arylsulphatase and acid and alkaline phosphomonoesterases partially control the dynamics of soil S and P respectively (Gregorich *et al.*, 1994).

Phosphatases are divided into acid and alkali phosphatases due to their activities within optimal acid and alkali pH ranges respectively. They are important in organic P mineralisation and plant nutrition. The inverse relationship between soil pH and acid phosphatase indicates that either the rate of synthesis and release of the enzyme from soil microorganisms, or the stability of the enzyme is related to soil pH (Tabatabai, 1982).

Arylsulphatase is partly involved in sulfur cycling in soils, by mineralising organic sulphur and making it available for plant growth (Tabatabai, 1982). The activity of this enzyme has been shown to decrease with increasing depth in the soil profile and decreasing organic matter content in soils of different physical and chemical properties (Tabatabai and Bremner, 1970).

Methods of measuring soil enzyme activity are generally quick, simple, accurate and reproducible (Tabatabai, 1982). Enzyme activity is determined by measuring the amounts of product produced by the enzymes, or the levels of substrate left after the enzymes have been activated for a specific period of time (Tabatabai, 1982). Conditions that must be standardised include: temperature; pH; substrate concentration; buffers; and time of incubation (Gregorich *et al.*, 1994). Generally, enzymatic activity can be related to soil organic matter content and thus microbial growth and activity, but the presence of extracellular enzymes means that this is not always possible (Tabatabai, 1982; Tate, 1987). In such cases, microbial biomass activity is not closely related to enzyme activities (Dick, 1997).

2.2.7.2 Factors Affecting Enzyme Activity

Enzyme activities fluctuate temporally due to changes in soil moisture (Ross *et al.*, 1984). Enzymes react rapidly to changes in cultivation, fertilization, and addition of organic amendments (Gregorich *et al.*, 1994). The degree of humification of the organic matter may determine enzyme activity, thus higher concentrations of easily decomposable substrates will promote enzyme synthesis (Tate, 1987). The activity of an enzyme depends on its tertiary structure, and any change in that structure inhibits or totally inactivates the enzyme. This can occur as a result of heavy metal pollution (Tate, 1987). The metal atom binds chemically with the sulphydral groups of the enzyme which are important for both structural integrity and active site formation (Tate, 1987). Salinity can decrease the activity of many enzymes (eg acid phosphatase and arylsulphatase), to various degrees depending on the enzyme (Frankenberger and Bingham, 1982).

Enzyme activities are influenced by pH, organic carbon, exchangeable potassium, calcium, and magnesium; organic carbon has the greatest effect (Baligar *et al.*, 1999). Other significant interactions with soil enzymes include moisture, total nitrogen and extractable phosphorus (Harrison, 1983). Harrison (1983) found that the influence each of the individual properties had on enzymatic activity varied according to soil depth, soil type, season, vegetative cover, and underlying rock type. These ecosystem properties accounted for 99 % of the variation in phosphatase activity. Enzyme-catalysed reactions are less sensitive to temperature changes than their uncatalysed counterparts because the catalysed reaction rate increases by a factor of less than 2, whereas the catalysed reaction rate doubles with every 10 °C rise in temperature (Tabatabai, 1982).

Because of the loss of organic matter and microbial activity, tillage will generally depress enzyme activities, for example, phosphatase was reduced to 49 %, and arylsulphatase to 65 % of their original activities (Gupta and Germida, 1988). This was also demonstrated by Haynes and Knight (1989) (Fig 2.5). Conservation tillage practices create higher levels of enzyme activity in the upper surface soil layer (generally <10 cm depth) (Doran, 1980).

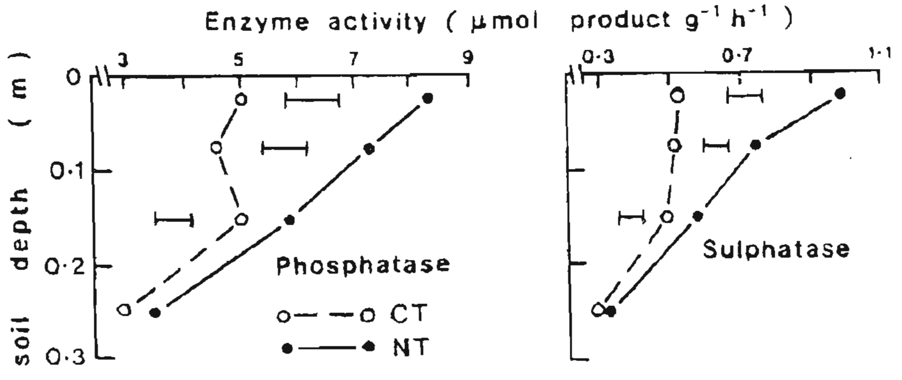

Fig 2.5 The effects of zero- (ZT) and conservational tillage (CT) on the activities of phosphatase and sulphatase down the soil profiles of a stony silt loam (Haynes and Knight, 1989).

As with many other biological indicators of soil quality, the levels of soil enzymes are higher in the surface soil layers than deeper down the profile (Duxbury and Tate, 1981; Tabatabai, 1982). The rhizosphere is a zone of increased microbial and enzyme activity, and lack of tillage and large C inputs contribute to consistently higher enzyme activities in grasslands compared to cultivated soils (Bandick and Dick, 1999). As shown in Table 2.2, enzyme (eg acid phosphatase) activities increase when above ground biomass and below ground root production increase (Duxbury and Tate, 1981). That is, grasses generally have a higher plant biomass and a larger input of plant residue compared with sugarcane, where generally, most of the plant residue is harvested or burnt in the field (Duxbury and Tate, 1981). When a soil is left fallow (Table 2.1), the organic matter inputs are insignificant, the microbial activity is low, and hence, so too is the enzyme activity (Duxbury and Tate, 1981).

Table 2.2 Variation in acid phosphatase activity in Pahokee Muck as a result of crop and position within the soil profile (Modified from Duxbury and Tate, 1981).

	Treatment			
Depth (cm)	Fallow	Sugarcane	St. Augustine-grass	Paragrass
0 - 5	1.24 ab (a)	2.34 a (b)	5.39 a (c)	4.31 a (cb)
30 - 35	1.58 a (a)	1.71 ab (ab)	2.83 b (b)	2.36 ab (ab)
60 -65	0.84 b (a)	0.49 bc (a)	0.93 c (ab)	1.59 b (b)
80 - 85	0.12 c (a)	0.14 c (ab)	0.30 c (ab)	0.34 c (b)

The values presented are means from a one-year period. Values with different letters are significantly different at the 5 percent level. Letters without brackets apply to the effect of soil depth for each individual crop. Letters with brackets apply to the effect of crop for each individual soil depth.

Soil enzyme activity is generally greater in freshly voided earthworm casts compared to uningested soil (Lee, 1985; Svensson *et al.*, 1986). Thus it is not surprising that the presence of earthworms in a silt loam, was found to increase respiration and cellulase and sulphatase activities (Ross and Cairns, 1982). The ryegrass, under which the earthworms were found also increased all biological activities in the soil. The combination of both earthworms and ryegrass maximised amylase, invertase, phosphatase and urease activities. Thus earthworms generally stimulate soil biochemical activities and nutrient cycling (Ross and Cairns, 1982).

From the above discussion it is evident that enzymes are affected by changes in environmental conditions and soil management systems, thus making them useful indicators of soil quality.

2.4 SOIL AGGREGATION AND STRUCTURAL ORGANISATION

Soil structure can be defined in terms of form and stability. Structural form is the arrangement of solid soil particles and the pores between them (Haynes and Beare, 1996). Structural stability is the ability of aggregates and pores to remain undamaged on exposure to stress eg drying and wetting (Haynes and Beare, 1996; Tisdall, 1996). Structure also implies the binding of particles of sand, silt and clay to form aggregates of different sizes by organic and inorganic materials, thus soils are not homogenous (Tisdall, 1996). Aggregate stability can change rapidly in response to changes in soil management, and thus serves as a very useful indicator of a soil's physical condition (Haynes and Beare, 1996).

The stability of aggregates and the pores between them is important as it affects the movement and storage of water, air and biological activity, and effects erosion and the growth of crops (Tisdall, 1996). The aggregates which are not water-stable slake into smaller subunits which disperse down the soil profile. This can result in limited water and air infiltration, restricted seedling emergence (Tisdall and Oades, 1982) and a dense plough layer (Haynes and Beare, 1996).

Agents involved in aggregate stabilization are humic substances (Haynes and Beare, 1996), microbially-produced polysaccharides (Tisdall and Oades, 1982), the linking and enmeshing effects of fine roots and fungal hyphae (Oades and Waters, 1991), earthworms (Lee and Foster, 1991), polyvalent cations (Martin, 1971) and wetting and drying cycles within the soil (Tisdall, 1996).

2.4.1 Models of Aggregation

Soils consist of microaggregates (<250 μm diameter), which are then bound together to form macroaggregates (>250 μm diameter). Larger aggregates tend to have larger pores and therefore more points of weakness than smaller aggregates. Thus, the smaller the aggregate, the greater the contact between particles, the stronger the bonds between particles, and the higher the stability of the aggregate (Dexter, 1988). Bonds within the microaggregates tend to be stronger than in macroaggregates (Edwards and Bremner, 1967). Edwards and Bremner (1967) proposed a model in which microaggregates less than 2 μm in diameter consist of clay particles joined to organic molecules (OM) by polyvalent cations (P). The microaggregates denoted as (Clay-P-OM) may combine together to form $(\text{Clay-P-OM})_x$ and $[(\text{Clay-P-OM})_x]_y$ through more polyvalent cation bridges. Particles such as (Clay-P-Clay) and (OM-P-OM) are also possible. Fragments of humified organic matter may bond to several clay particles (Edwards and Bremner, 1967). The most important mechanism of interaction between organic polymers and mineral surfaces involves bridges of polyvalent cations between the surface of clay particles, or hydroxy polymers and the ligand groups of organic polymers eg carboxyl groups (Tisdall and Oades, 1982).

Tisdall and Oades (1982) proposed a model suggesting that microaggregates are a result of stages of aggregation, provided that organic matter is the primary binding agent. Levels of aggregation may differ between soils. There are four stages of aggregation:

$$<0.2\ \mu\text{m} \rightarrow 0.2 - 2\ \mu\text{m} \rightarrow 2 - 20\ \mu\text{m} \rightarrow 20 - 250\ \mu\text{m} \rightarrow >2000\ \mu\text{m}$$

Oades and Waters (1991) modified the model for Alfisols and Mollisols where aggregates are primarily stabilized by organic materials. They suggest it is not possible to identify stages of aggregation in aggregates less than 20 μm in diameter, but the range of 20 to 250 μm could be subdivided into stages of 20 - 90 µm and 90 - 250 µm sized aggregates. These stages develop very slowly over the years and are probably not evident in young soils (Oades, 1993). However, in soils (eg Oxisols and Andosols) where aggregation is stabilized principally by inorganic materials, this model may not fit (Oades and Waters, 1991; Robert and Chenu, 1992).

2.4.2 Structural Organisation

The structural organisation of a soil is heterogenous, and can change quickly in both space and time. Thus only generalised models on micro- and macro-aggregate structure (Fig 2.6) and formation can be presented (Haynes and Beare, 1996).

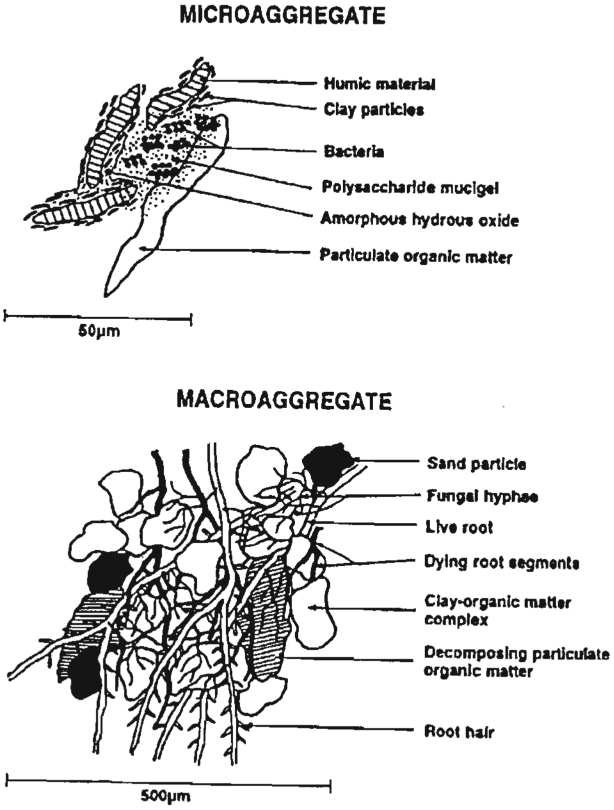

Fig 2.6 Drawing of the structures of both micro- and macro-aggregates (Haynes and Beare, 1996).

2.4.2.1 Microaggregates

a) Microaggregates less than 2 μm in diameter

Clay particles interact according to the balance of two forces: the attractive London van der Waal's forces and ion correlation forces; and repulsive forces as a result of ion hydration, and hydrogen bonds forming between water molecules found close to the clay-water interface (Quirk and Murray, 1991). The repulsive forces between the negatively charged particles of clay are weakened by polyvalent cations eg Al^{3+} and Ca^{2+}. This results in flocculation of the clay. Flocculation is also increased by high concentrations of electrolyte in soil solution, minimum disturbance of the soil and high organic matter (Rimmer and Greenland, 1976). Organic matter may however, prevent flocculation when aggregates are disrupted by heavy rainfall or tillage and it is adsorbed onto the surface of the clay (Quirk and Murray, 1991).

Small aggregates (<2 μm) are composed of fine clay (<0.2 μm) particles held together by organic matter and polyvalent cations. Although organic matter can fit between the lamellae of clay particles, it rarely does, and most enzymes and microorganisms are only adsorbed onto the outer surfaces of clay structures (Robert and Chenu, 1992). Organic matter associated with fine clay (<0.2 μm) was found to be mainly fulvic acids, and aliphatic humic acids which are rich in nutrients such as nitrogen and sulphur (Anderson *et al.*, 1981). However, organic matter in particles of 0.2 to 2 μm in diameter is derived from humified cell walls of microorganisms (Baldock *et al.*, 1992). Hassink *et al.* (1993) found that organic matter in particles of less than 2 μm, has no distinct shape and has a low C:N ratio. Thus, the organic matter held within the particles is highly degraded, protected by the clay, and originates from microorganisms.

b) Microaggregates of 2 - 20 μm in diameter

Microaggregates of this size are held together by both inorganic and organic bonds and sometimes floccules of particles less than 2 μm may form aggregates of 2 to 20 μm (Dexter, 1988). Other aggregates of this size may consists of fine silt and clay, and course sand (Tisdall and Oades, 1982), combined through microbial materials such as polysaccharides, hyphal fragments, and bacterial cells and colonies surrounded by inorganic materials (Oades and Waters, 1991). Bacterial cells or colonies are surrounded by a polysaccharide capsule which in turn is surrounded by clay particles. These clay particles align themselves either parallel to, or radially from the cell surface. As the polysaccharide is produced, the clay particles in the surrounding soil become compacted and reorientated, brought closer together by surface tension as the clay dries (Foster, 1988). These clay particles covering the bacteria to form aggregates also provide physical protection for the bacteria from decomposition, particularly when they are held in pores of less than 3 μm, and predacious protozoa and nematodes occupy larger pores. Because of the protection offered to bacteria, microaggregates of 2 to 20 μm contain 40 to 60 % of the soil's microbial biomass depending on the type and amount of clay (Jocteur Monrozier *et al.*, 1991).

The microorganisms protected by the clay have their osmotic barriers intact, even if they are dormant in the soil (Clark and Paul, 1970). The biomass dies as the soil dries out, but regrows once the soil is rewetted (McGill *et al.*, 1986). Once the bacterial cells have died, the microaggregate remains intact due to the polysaccharide, but can no longer be identified as a bacterial cluster (Foster, 1988). The growth of roots may exert pressure on the soil and cause compaction and reorientation of the soil, and the release of root exudates contributes to aggregation (Dormaar and Foster, 1991). The organic matter in these microaggregates has a low C:N ratio, and is composed of little, if any, plant material (Baldock *et al.*, 1992). The organic matter content is high, and contains a large portion of the microbial biomass (Jocteur Monrozier *et al.*, 1991). The aggregates of this size group are so strongly bound by persistant organic bonds, especially in pasture soils, that they are not normally disrupted by agricultural practices (Tisdall and Oades, 1982).

c) Microaggregates of 20 to 250 µm in diameter

Aggregates of 20 to 250 µm in diameter consist mainly of aggregates less than 20 µm cemented by plant and fungal debris, coated by inorganic materials, crystalline oxides and aluminosilicates (Tisdall and Oades, 1982). These microaggregates are stable to water in the field (Oades and Waters, 1991), and are part of the $[(\text{Clay-P-OM})_x]_y$ model (Edwards and Bremner ,1967). The organic matter in these aggregates is less decomposed than that in the smaller aggregates, has a higher C:N ratio, with a higher content of O-alkyl carbon (Baldock *et al.*, 1992). Aggregates within the range of 100 to 200 µm may contain a few remaining plant residues, but most have been decomposed by microorganisms (Oades and Waters, 1991). Microaggregates less than 100 µm contain a larger organic core of highly degraded plant residues, more organic matter, N, P and K compared to the larger aggregates (Oades and Waters, 1991). In a tilled soil, up to 56 % of aggregates may be microaggregates less than 100 µm and these contain twice the amount of organic matter of their larger (>100 µm) counterparts. In untilled soils, these aggregates (<100 µm) may contain up to four times the amount of organic matter compared to their tilled counterparts. The faecal pellets of microarthropods such as mites and collembola are microaggregates less than 100 µm, and are round and smooth. These pellets contain no mineral soil, but are densely colonised with fungal hyphae and bacteria (Lee and Foster, 1991).

2.4.2.2 Macroaggregates

The bonds between macroaggregates are not strong enough to hold the macroaggregates together against the swelling of clay, and release of trapped air and water breaks up the macroaggregates into smaller microaggregates (Emerson, 1991). However, some macroaggregates held together by iron aluminium oxides, for example in Oxisols, are very stable and can only be broken by ultrasound in laboratory conditions (Oades and Waters, 1991). Macroaggregates are otherwise mainly held together by fungal hyphae, fibrous roots and polysaccharides (Miller and Jastrow, 1990), which can occupy pores of 15 to 50 µm in diameter (Foster, 1988). A macroaggregate is stabilized by organic matter which stabilizes the larger pores (Quirk and Panabokke, 1962). The main stabilizers of macroaggregates under growing plants are roots, the hyphae of vesicular arbuscular mycorrhizal (VAM) fungi, and to some extent saprophytic fungi (Rasmussen *et al.*, 1980). Earthworm casts can be considered as macroaggregates and are also stabilized by organic matter and fungal hyphae (Lee and Foster, 1991).

Differences in water uptake by macroaggregates are not determined by pore-size distribution, but by the distribution of organic matter within the aggregate. Tilled aggregates have more points of weakness, allowing for collapse when wetted quickly (Quirk and Panabokke, 1962).

Depending on soil type, a large part of organic matter held in aggregates is chemically and/or physically protected from decomposition (Tisdall, 1996). A soil under pasture provides greater physical protection for the microbial biomass than one under cultivation (Breland and Eltun, 1999). Since they have a higher organic matter content and more stable macroaggregates, soils with more clay have finer pores thus providing more protection than soils with more sand (Hassink *et al.*, 1993). When a clay soil was passed through a fine sieve of 1 mm in diameter, there was increased N mineralization compared to a sandy soil, thus indicating a larger amount of organic matter was protected in the small pores between clay particles (Hassink *et al.*, 1993). Soil disturbed by tillage, soil fauna or rapid wetting often supports a flush of microbial activity, and thus C and N mineralisation. This is thought to be due to the exposure of organic matter once physically protected by macroaggregate structure (Gregorich *et al.*, 1989).

2.4.3 Indicators of Aggregate Stability

Both wet sieving and turbimetry are used to measure aggregate stability (Haynes, 1993). Wet sieving is used to determine the proportion of aggregates of a certain diameter remaining after exposure to water stress (Haynes, 1993). Turbimetry is the quantification of the amounts of clay and silt-sized particles produced (Haynes, 1993).

2.4.3.1 Soil Organic Carbon and Total N Content

Soil organic carbon and total N content of soils are closely correlated with the stability of soil aggregates, particularly in agricultural soils (Chaney and Swift, 1984; Haynes *et al.*, 1991). It has recently been found that both organic C content and total N content are more-or-less equally correlated to aggregate stability (Chaney and Swift, 1984; Haynes *et al.*, 1991). The close relationship between aggregate stability and organic carbon is depicted in Fig 2.7. Under sugarcane, the loss of soil organic matter content from burning results in a decrease in aggregate stability whilst the gain from trash retention causes an increase in aggregate stability (Van Antwerpen and Meyer, 1996).

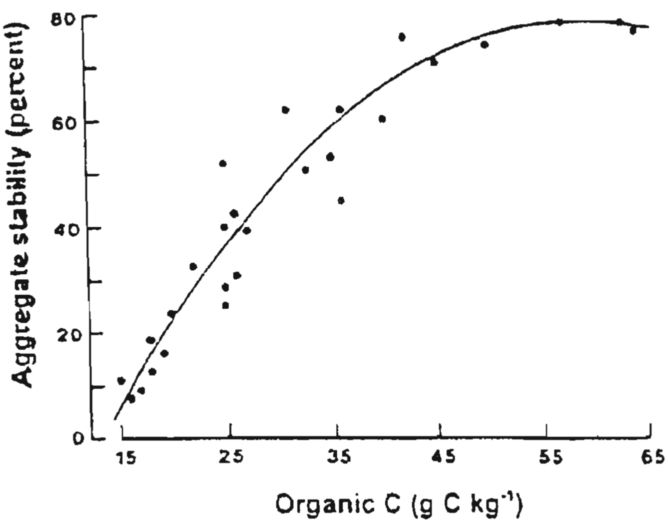

Fig 2.7 The effect organic carbon content has on soil aggregate stability (Haynes, 1997).

The greater the clay content of the soil, the greater the soil organic matter content and the more organic carbon that is required to reach a particular level of aggregate stability (Douglas and Goss, 1982; Haynes *et al.*, 1991). Each change in organic matter content usually results in a similar change in the stability index (Haynes and Beare, 1996).

2.4.3.2 Microbial Biomass

As stated previously, microbial biomass plays an important role in aggregate stability as it includes fungal hyphae which enmesh soil particles, and in addition both fungi and bacteria produce extracellular mucilage which acts as a binding agent. The microbial biomass is directly related to organic C content of soil in long-term trials, but is more indicative of short-term changes in soil organic matter dynamics due to its rapid turnover rate (Jenkinson and Ladd, 1981). Thus, microbial biomass is often used as an indicator of short-term changes in aggregate stability (Haynes and Beare, 1996).

2.4.3.3 Crop Root Density

Rhizodeposition is the supply of large amounts of organic material from root senescence and root exudates. The magnitude of rhizodeposition is closely correlated to total root mass (Shamoot *et al.*, 1968). Grasses usually tend to have a greater root mass than other crops, hence result in a larger soil microbial biomass and aggregate stability (Haynes and Francis, 1993). The buildup of microbial biomass increases the organic binding agent production. Thus the greater the root mass, the greater the influence on aggregate stability (Haynes and Francis, 1993).

2.4.3.4 Previous Cropping History

The cropping history will effect the stability of a soil, and has been found to be more closely correlated with short-term changes in aggregate stability compared to total nitrogen, organic carbon, or carbohydrate content (Haynes *et al.*, 1991). Short periods of arable cropping will decrease the aggregate stability, whilst periods of pasture will improve the aggregate stability even though total soil organic matter content remains unchanged (Haynes *et al.*, 1991).

2.5 CONCLUSIONS

Soil organic matter is fundamental to soil quality. It is composed of both humic and non-humic substances; the humic portion being more stable, while the non-humic substances have a faster turnover. The carbon made available from this turnover is used as a substrate by the heterotrophic soil microbial community. Total organic matter is represented by measurements of organic C and N, while labile organic matter is represented by fractions such as the potentially mineralizable C and N. The choosing of each soil property for a set of data is dependent on the desired outcome or field of interest of the research.

Soil biota are involved in the nutrient and energy transformations within the soil. The activity of biota is determined by carbon availability, and physical and chemical properties of the soil. The microbial biomass is a store of labile organic matter and is involved in organic matter cycling. It is the living component of organic matter and is dynamic and very responsive to changes in carbon availability. Thus, it serves as a more suitable indicator of carbon availability than total organic matter content. The microbial biomass is affected by environmental factors (eg temperature and moisture), as well as soil type, crop rotation, fertilizer regime, tillage practice, burning of crop residue, type of crop (eg pasture versus arable), and earthworm activity. Organic C and microbial biomass C content of soils are usually strongly related. The use of the microbial quotient (the ratio of microbial biomass C to total organic C) allows for comparison of the effect of different management practices at different sites having different initial organic C contents.

The size of the microbial biomass does not reflect its activity, thus separate measurements are necessary. Such activity measurements include basal respiration (CO_2 evolution from a sample of soil) and assays of overall enzymatic activity eg FDA hydrolysis rate. The respiratory quotient (ratio of basal respiration to microbial biomass C) indicates the efficiency with which the microbial biomass is using substrate C for microbial tissue production and maintenance. A high respiratory quotient suggests that the microbial community is under stress.

Aggregate stability indicates the ability of soil aggregates to withstand the degrading action of water. The arrangement of aggregates and therefore pores determines the ability of a soil to store and transport air and water. Agents involved in stabilizing aggregates include humic substances, microbially-produced polysaccharides, fine roots and fungal hyphae, earthworms, polyvalent cations, and wet-dry cycles in the soil. Management practices greatly affect aggregate stability in the short-term, thus aggregate stability is a good indicator of a soil's physical condition.

Soil organic matter content, microbial biomass size and activity, earthworm size and composition, and aggregate stability are sensitive to changes in soil management practices, and thus very important components of soil quality evaluations. However, until now, the use of these soil characteristics in evaluations of soil quality and degradation in South African agricultural soils has been very limited. These characteristics will serve as useful tools in assessing the effects of sugarcane monoculture on soil condition.

The assessment of soil quality and the extent of soil degradation under sugarcane in South Africa is important for two reasons. Firstly, the phenomenon of sugarcane yield decline is closely related to soil degradation and loss of soil biological activity in the Australian sugar industry. Secondly, because sugarcane production is a major land use in KwaZulu-Natal and sustainable land management is becoming more necessary, the comparison of the effects of sugarcane to those of other major agricultural crops and undisturbed veld on soil condition will be very important information.

CHAPTER THREE

EFFECT OF LONG-TERM SUGARCANE PRODUCTION ON SOIL ORGANIC MATTER STATUS AND RELATED PROPERTIES OF TWO CONTRASTING SOILS

3.1 INTRODUCTION

Soil degradation under sugarcane production has been a recent concern of many authors (Garside *et al.*, 1997; Haynes and Hamilton, 1999). In fact a plateau or decline in sugarcane yield per hectare observed in many countries throughout the world has been linked to soil degradation (Garside *et al.*, 1997). Several authors have suggested that the most serious factor associated with soil degradation under sugarcane is the loss of soil organic matter (Wood, 1985; Haynes and Hamilton, 1999).

A loss of soil organic matter can have detrimental effects on soil physical, chemical and biological properties. The decrease in soil biological activities is of particular concern (Doran and Parkin, 1994; Garside *et al.*, 1997) since biologically mediated processes are central to their ecological function. Important processes include organic residue degradation, transformations of organic matter, mineralisation of nutrients held in the organic form and formation and stabilization of aggregates.

C3 (eg sugarcane) and C4 (eg native vegetation possessing C4 metabolism) plants have different photosynthetic pathways enabling one to partition the soil organic matter as to its origin between the two types of plants. The naturally occurring isotope ^{13}C of the plant vegetation may serve as an *in situ* labelling of soil organic matter from the two contributing kinds of plant residue

(Balesdent *et al.*, 1988). This $\delta^{13}C$ methodology has been used to determine the long-term contributions of both maize and sugarcane to total soil organic matter content of soils in a number of places (Vitorello *et al.*, 1989; Gregorich *et al.*, 1995).

Sugarcane monoculture is a major land use in the KwaZulu-Natal Province of South Africa, but little is known of the effects of sugarcane on soil organic matter content and quality, soil biological activity, and soil physical conditions. This study covers the effects of increasing periods of sugarcane cultivation on the above soil properties in two contrasting regions within the sugar belt. In addition, $\delta^{13}C$ methodology is used to try quantify the contribution of sugarcane residues to the soil organic matter content.

3.2 MATERIALS AND METHODS

Fields with increasing years under sugarcane production were sampled from two separate localities in the South African province of KwaZulu-Natal. On the south coast region of the province, 20 fields were sampled from "Kinroy Estate" (30° 15' 36" S and 30° 30' 00" E) in autumn (March) to a depth of 10 cm (3 samples from each field were bulked). These samples provided a range of cropping histories ranging from undisturbed virgin veld to 69 years under sugarcane monoculture. The soil at the site was Glenrosa form (Glenrosa series) (Soil Classification Working Group, 1991) (Ochric Cambisol; FAO) with a clay content of about 18% and the clay fraction was predominantly kaolinite with some accessory vermiculite also present. The long-term mean monthly air temperature in that region is 19.9°C with a minimum of 9.0°C in July and maximum of 27.6°C in February. Mean annual rainfall is 965 mm and in this summer rainfall area 451 mm falls in the December to March period.

In the Midlands region of the province, 20 fields were sampled from "Seafield Estate" (29° 55' 48" S and 30° 24' 00" E) as described above, and these provided a range of histories from undisturbed natural grassland to 19 years under sugarcane monoculture. The soil at the site was a Hutton form (Farmingham series) (Rhodic Ferralsol, FAO) with a clay content of about 61%. Its mineralogy was dominated by kaolinite and halloysite, while there were also appreciable amounts of crystalline sesquioxides, gibbsite and interlayered chlorite. The long-term mean annual air temperature in the area is 18.3 °C with a minimum of 9.3 °C in June and a maximum of 25.8 °C in February. The mean annual rainfall is 821 mm and 440 mm falls between December and March.

As is common in the area, preharvest burning of the cane is practised at both sites. At the Glenrosa site, the cane is harvested on a 12 month cycle, with yields of about 60 to 80 Mg ha^{-1}, while the Hutton site cane is harvested every 18 months with a yield of about 100 to 120 Mg ha^{-1}. Fertilizers are applied after harvest and typical rates are approximately 110 kg N ha^{-1}, 30 kg P ha^{-1}, and 150 kg K ha^{-1}. Soils were tilled to about 20 to 25 cm at replanting after about one planted crop and seven ratoon crops. This comes to about every 8 years on the Glenrosa soil, and every 12 years on the Hutton soil.

Within 48 hours of collection, bulk field-moist samples were thoroughly mixed and divided into three sub-samples. One sub-sample was sieved (<2 mm), rewetted to field moisture, and stored in plastic bags at ± 4 °C before biological analysis. The second was sieved for collection of aggregates (2 - 4 mm diameter) which were air-dried for aggregate stability analysis. The third sub-sample was air-dried at room temperature, sieved (<2 mm) and ground (<0.5 mm) for analysis of organic C. The analysis of each sub-sample was carried out in triplicate.

Organic carbon was measured colorimetrically by the Walkley and Black dichromate oxidation method (Blakemore *et al.*, 1972). Microbial biomass carbon was calculated from the difference between C extracted by 0.5 M K_2SO_4 from ethanol-free chloroform fumigated and unfumigated soil samples when using the fumigation-extraction method. A K_c factor of 0.38 was used (Vance *et al*, 1987). The microbial quotient was calculated by expressing the microbial biomass C as a percentage of total soil organic carbon. Basal respiration was determined by the incubation of 30 g soil samples with 10 ml of 0.1 M NaOH in 1 l air-tight sealed glass jars at 25 °C for 10 days. The amount of CO_2-C evolved from the soil was collected in the NaOH and titrated against 0.2 M HCl. The metabolic or respiratory quotient is calculated as a ratio of basal respiration (μg CO_2-C) per mg of microbial biomass C per day.

Arylsulphatase activity was analysed as described by Tabatabai (1982). The amount of *p*-nitrophenol released from arylsulphatase activity was measured colorimetrically at a wavelength of 420 nm. Acid phosphatase was measured using the method of Tabatabai (1982), but the calibration graph was plotted using standards containing 0, 20, 40, 80 and 100 μg of *p*-nitrophenol. The amount of *p*-nitrophenol produced from acid phosphatase activity was measured spectrophotometrically at 400 nm.

Aggregate stability was determined through the use of the wet-sieving technique (Haynes, 1993). 30 g air-dried soil aggregates of 2 - 4 mm diameter were placed on a sieve of 2 mm aperture mesh which, when suspended at its highest oscillation level in water, allowed for the soil to be just covered by water. The oscillation rate was 25 cycles min^{-1}, and the amplitude of sieving action was 35 mm. After 15 minutes of oscillation in water, the remaining soil in the sieve was collected, oven-dried and weighed. The results were expressed as the percentage of stable aggregates remaining on the 2 mm sieve following sieving.

The $\delta^{13}C$ values of soils were determined on CO_2 evolved using a vacuum combustion system. The CO_2 was analysed in a Fision Sira 24 mass spectrometer. Each analysis was the mean of duplicate subsamples. The natural abundance of heavy isotopes was expressed relative to the international standard PDB (as $\delta^{13}C$ % (parts per thousand)). The proportion of C derived from sugarcane residues (X %) was calculated as:

$$X\ \% = \frac{\delta^{13}C \text{ cropped soil} - \delta^{13}C \text{ native soil}}{\delta^{13}C \text{ sugarcane} - \delta^{13}C \text{ native soil}} \times (100/1)$$

The $\delta^{13}C$ of sugarcane residues was determined from triplicate samples collected from across the experimental sites.

The Genstat Version 5.0 (1993) statistical programme (Lawes Agricultural Trust, Rothamstead) was used for all statistical analysis. In order for the results of two different localities to be statistically compared against each other on a common graph, a regression equation for each locality and a common correlation coefficient was obtained. The criteria for the selection of a suitable statistical model were the necessity to compare between years under sugarcane, between localities, and between years by locality from an unbalanced data set. Data for the various measured parameters were related to number of years under sugarcane monoculture using linear, quadratic, cubic, and exponential regressions to determine the best fitting predictive model. Regression equations and lines of best fit are presented.

3.3 RESULTS

Although the two sites had different natural vegetation, soils and climate, both had a relatively similar organic C content under undisturbed vegetation (Fig 3.1). Soil organic C content decreased with increasing years under sugarcane cultivation. For the Glenrosa site, a new equilibrium soil organic C content was reached after about 10 years of sugarcane monoculture. The new equilibrium level of organic C reached was markedly greater in the Hutton than in the Glenrosa soil.

Like that for organic C, there was an exponential decline in microbial biomass C with increasing years under sugarcane and the new equilibrium level reached was greater in the Hutton than Glenrosa soils (Fig 3.2). The decline for both sites was, however, considerably more pronounced than that of organic C content. As a result, the microbial quotient declined from about 2.2% to 1.5% (Fig. 3.3). Values for the microbial quotient were similar at both sites at the same cropping history.

Aggregate stability declined exponentially with increasing years under sugarcane and values were similar for the two different soils under both native vegetation or sugarcane (Fig 3.4).

The activity of the microbial biomass decreased with increasing years under sugarcane as indicated by decreases in the basal respiration, and arylsulphatase and acid phosphatase activities (Figs 3.5, 3.6 and 3.7 respectively). Basal respiration and the activities of acid phosphatase and arylsulphatase decreased sharply within the first 5 years of cultivation at both sites, thereafter tending towards an equilibrium. The Hutton site had significantly higher levels of both acid phosphatase and arylsulphatase compared to the Glenrosa site (Figs 3.6 and 3.7) but values for basal respiration were similar at the two sites (Fig 3.5).

Using values for basal respiration and microbial biomass C, the metabolic quotients were calculated. Values were unaffected by the number of years under sugarcane. Mean values were 21 (range 12 to 40) μg CO_2-C g^{-1} day^{-1} for the Glenrosa soil and 14 (range 4 - 28) μg CO_2-C g^{-1} day^{-1} for the Hutton Soil.

The relationships between the various measurements of organic matter and size and activity of the microbial biomass of both sites are presented in Table 3.1. Microbial biomass C was linearly related to organic C, and the regression equation and line of best fit are shown in Fig 3.8. Arylsulphatase and acid phosphatase activity were also linearly related to organic C (Table 3.1). Basal respiration, arylsulphatase and acid phosphatase activity were best related to microbial biomass C by quadratic relationships. Aggregate stability was linearly related to both organic C and microbial biomass C.

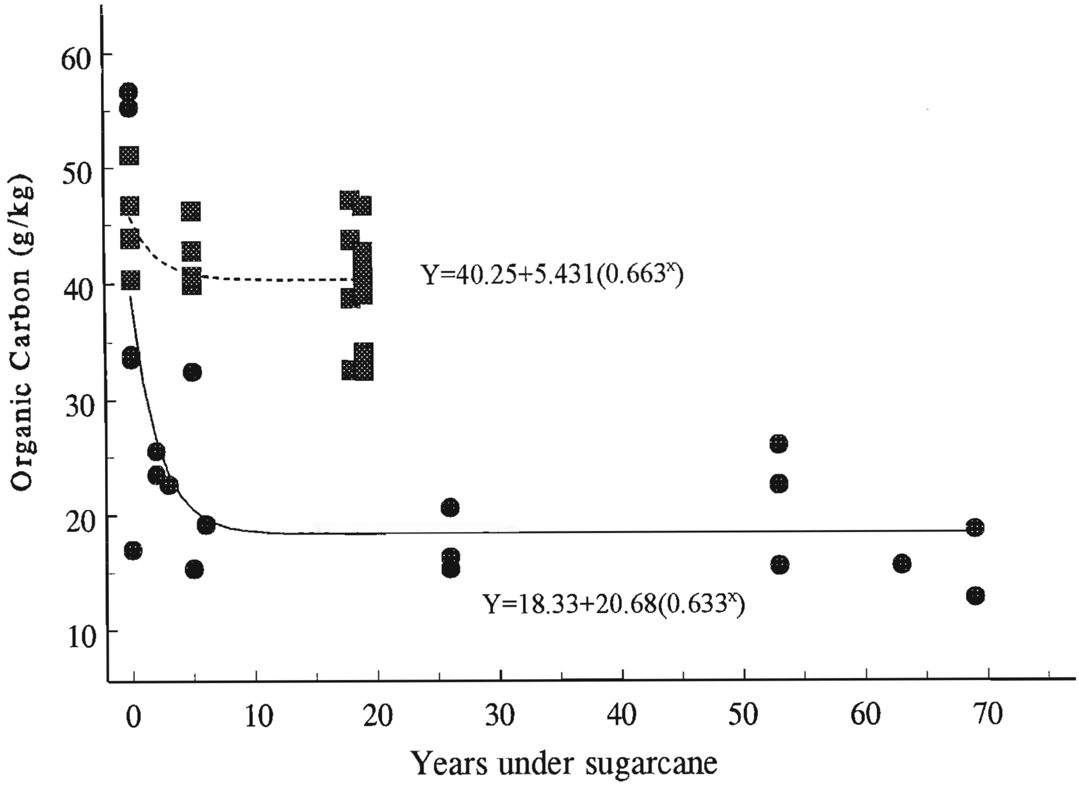

Fig 3.1 Effect of increasing time under sugarcane monoculture on soil organic C content at the Hutton (▨) and Glenrosa (●) sites (r=0.67; $P \leq 0.001$). Regression equations and lines of best fit shown.

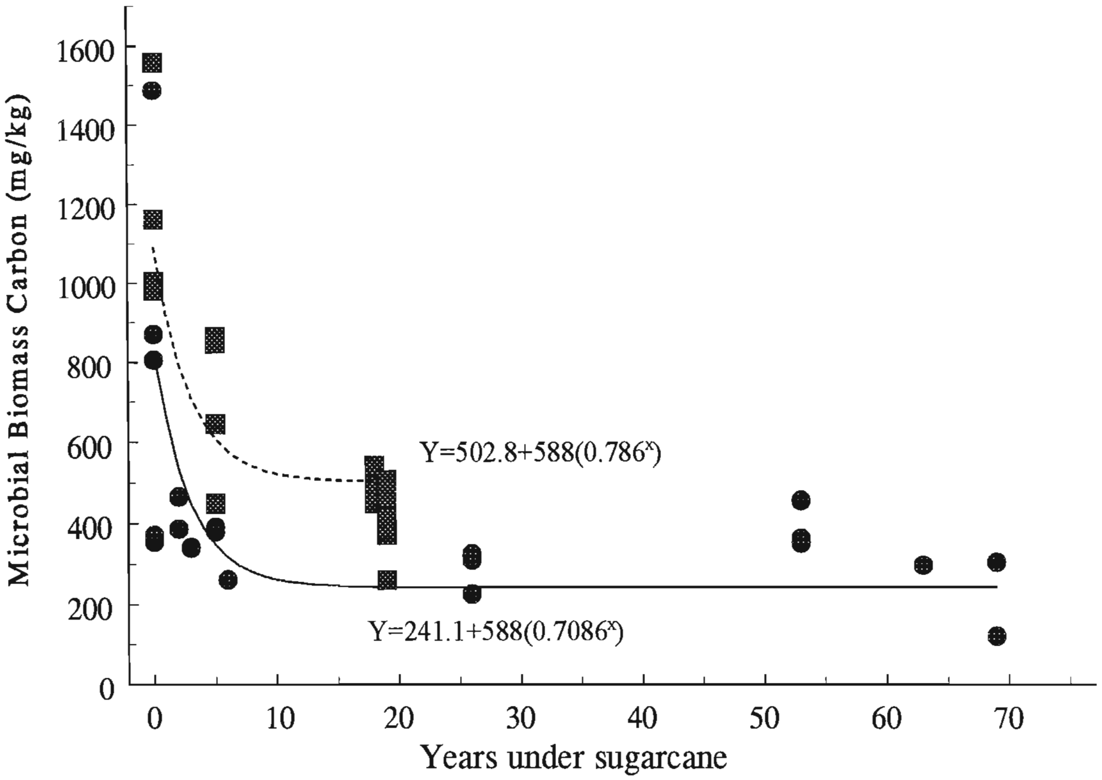

Fig 3.2 Effect of increasing time under sugarcane monoculture on soil microbial biomass C at the Hutton (▨) and Glenrosa (●) sites (r=0.60; $P \leq 0.001$). Regression equations and lines of best fit shown.

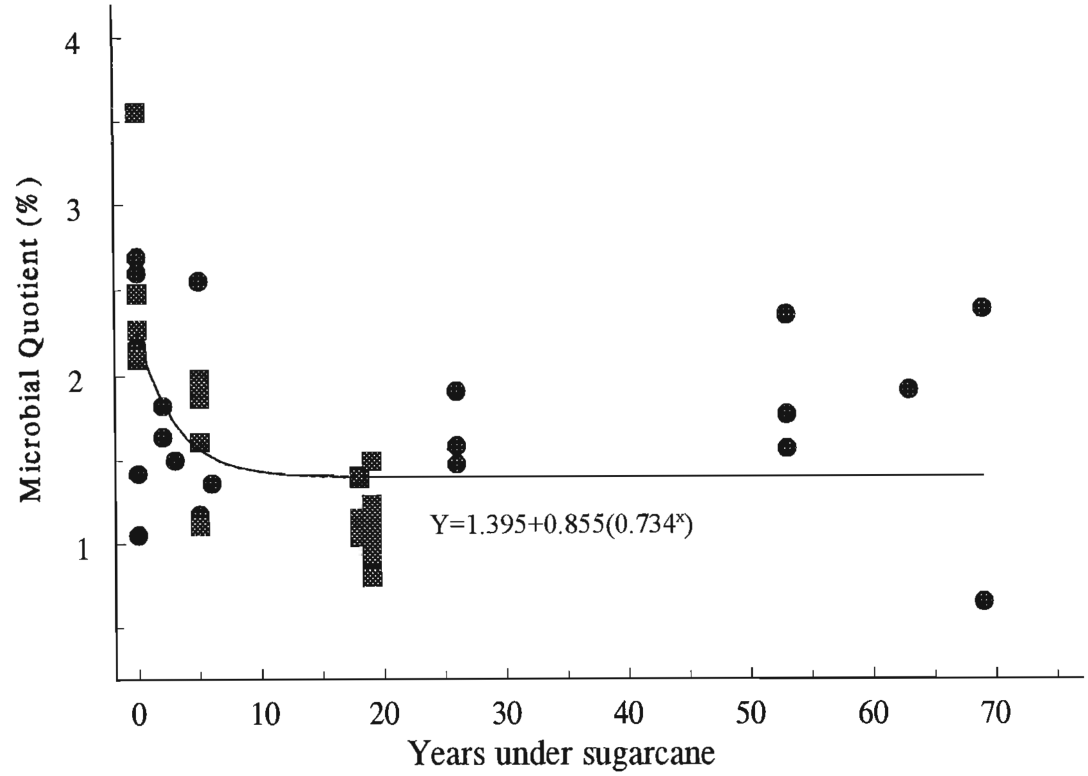

Fig 3.3 The effects of increasing time under sugarcane monoculture on microbial quotient at the Hutton (▩) and Glenrosa (●) sites (r=0.27; $P \leq 0.001$). Regression equation and line of best fit shown.

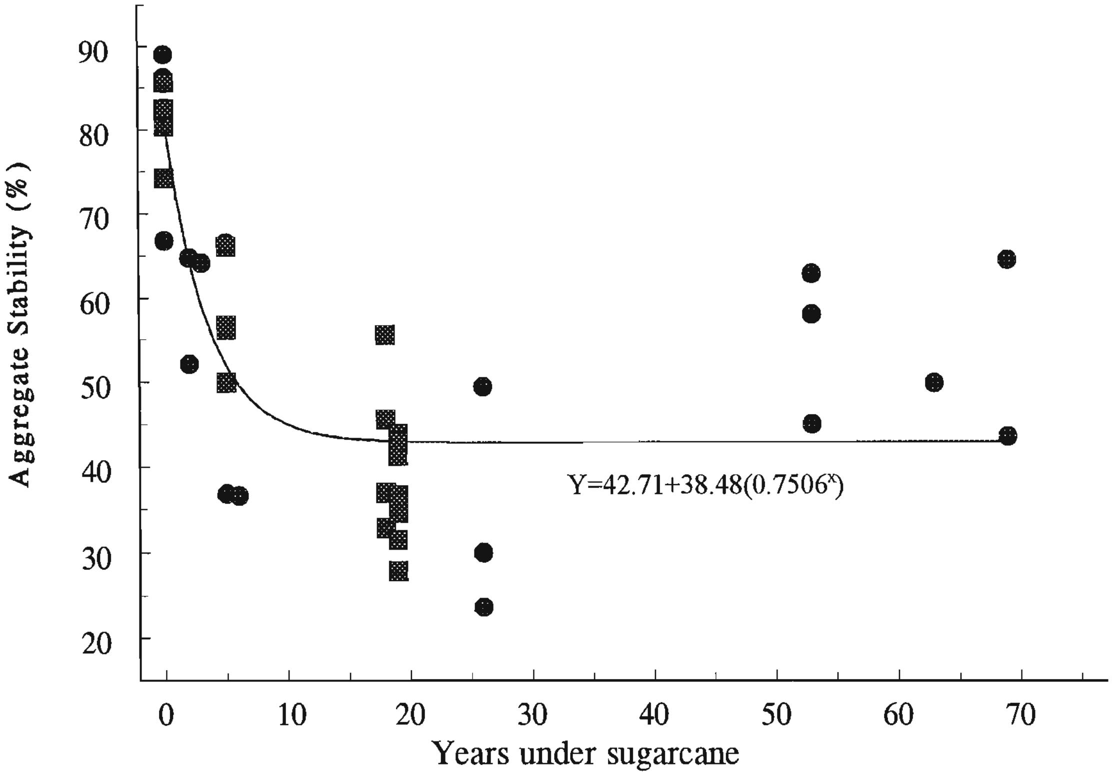

Fig 3.4 Effect of increasing time under sugarcane monoculture on soil aggregate stability at the Hutton (■) and Glenrosa (●) sites (r=0.69; $P \leq 0.001$). Regression equation and line of best fit shown.

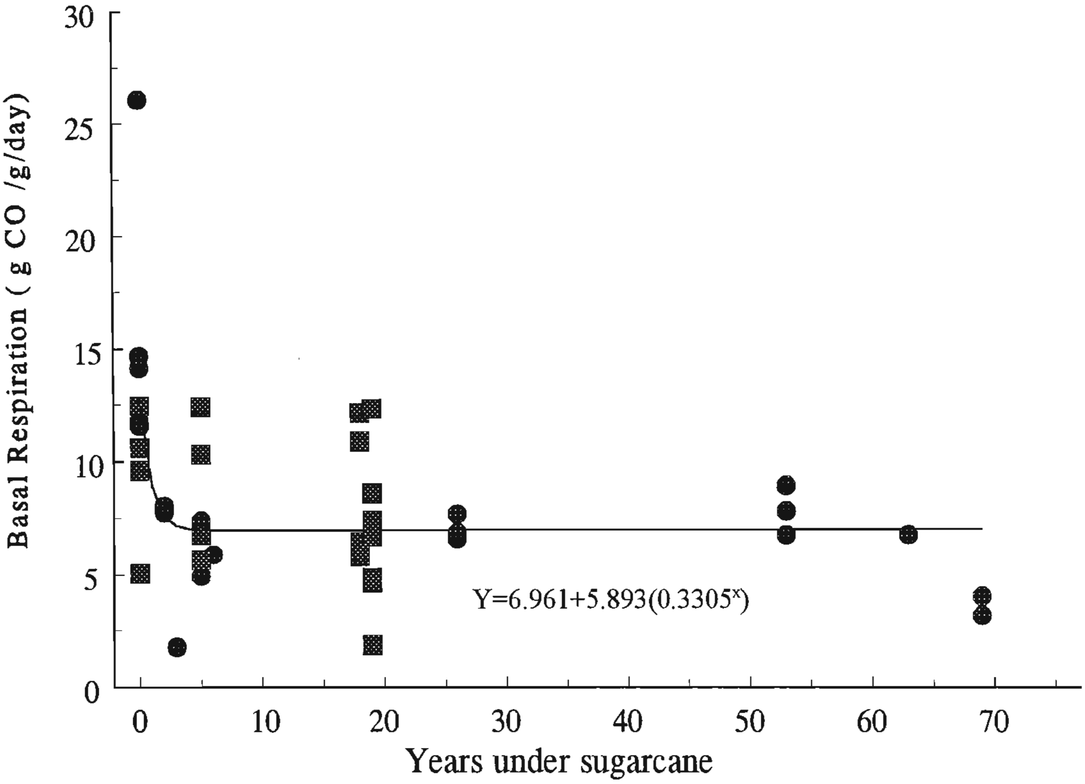

Fig 3.5 Effect of increasing years under sugarcane monoculture on basal respiration in the soil at the Hutton (▩) and Glenrosa (●) sites ($r=0.29$; $P \leq 0.001$). Regression equation and line of best fit shown.

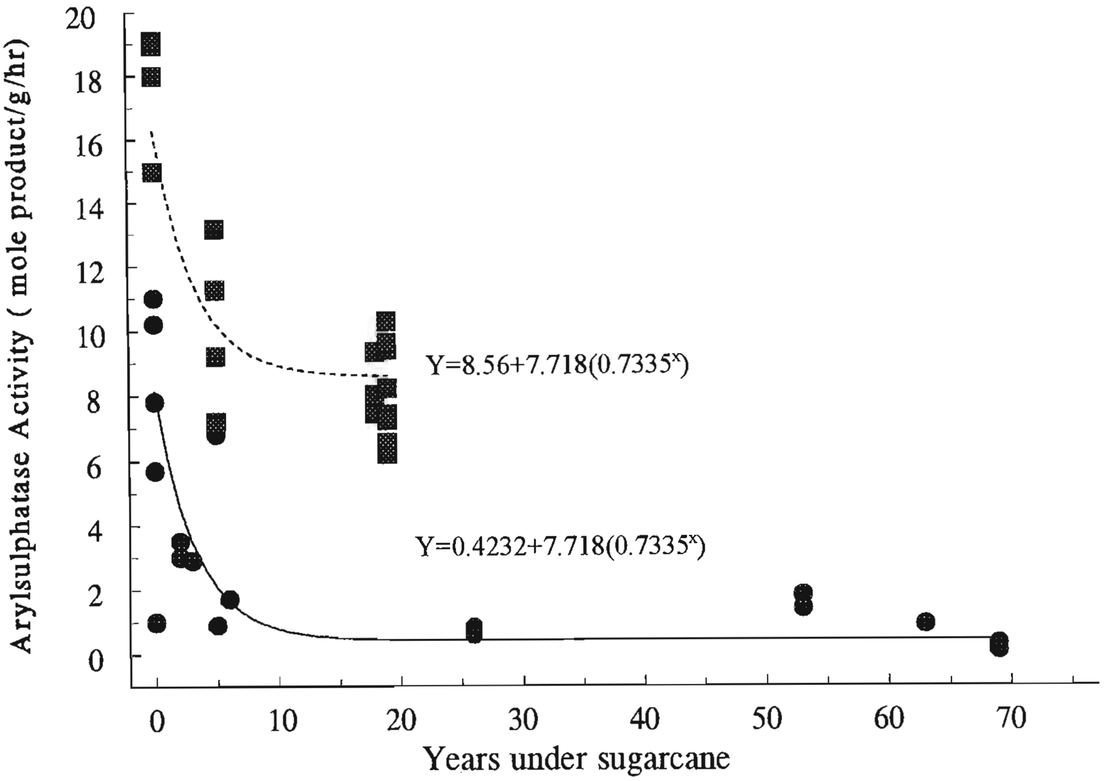

Fig 3.6 Effect of increasing years under sugarcane monoculture on arylsulphatase activity in soils at the at the Hutton (■) and Glenrosa (●) sites ($r=0.84$; $P \leq 0.001$). Regression equations and lines of best fit shown.

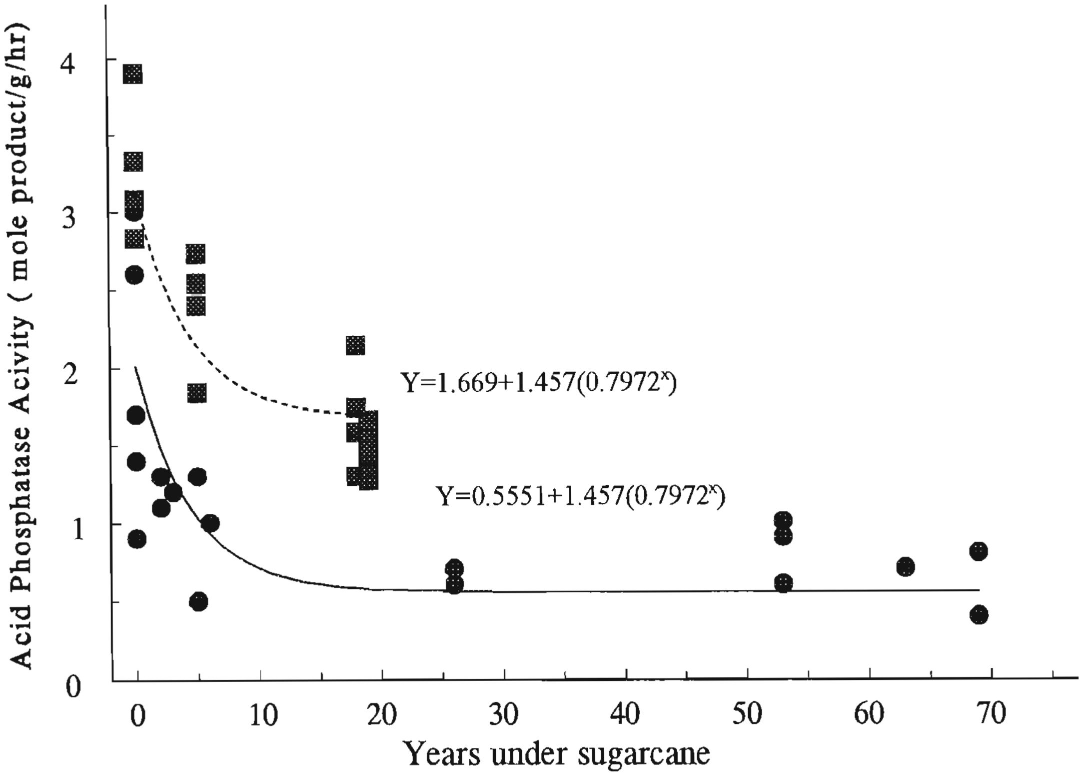

Fig 3.7 Effect of increasing time under sugarcane monoculture on acid phosphatase activity in the soils at the Hutton (■) and Glenrosa (●) sites ($r=0.77$; $P \leq 0.001$). Regression equations and lines of best fit shown.

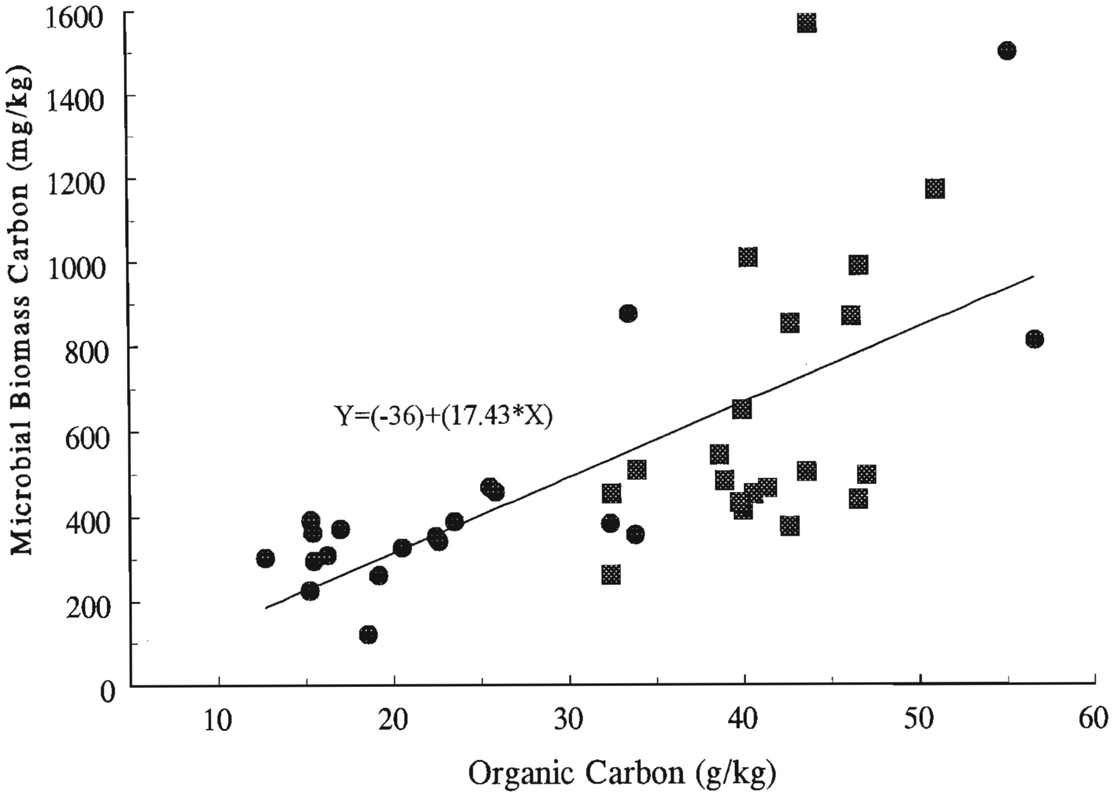

Fig 3.8 Correlation between organic C and microbial biomass C at the Hutton (▩) and Glenrosa (●) sites (r=0.44; $P \leq 0.001$). Regression equation and line of best fit shown.

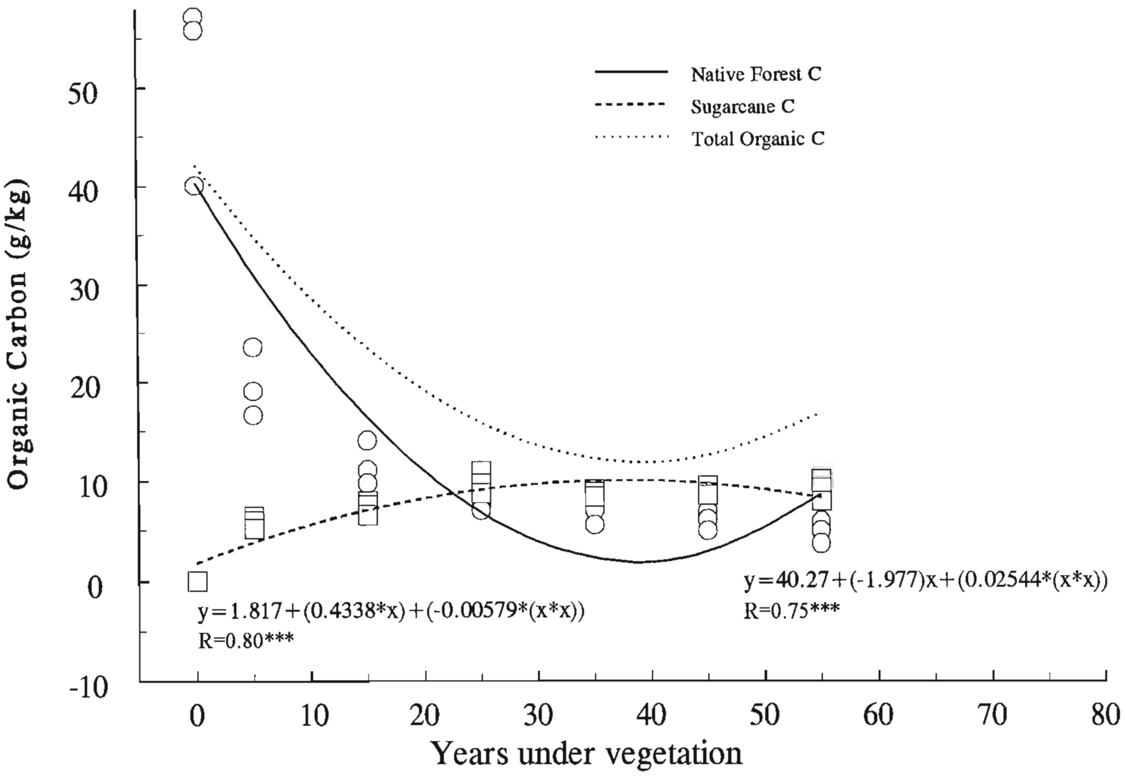

Fig 3.9 Effect of increasing time under sugarcane monoculture on forest-derived native C, sugarcane-derived C and the sum of both organic C sources in the Glenrosa soil. Regression equations, lines of best fit, correlation coefficients (R) and statistical significance shown. * $P \leq 0.05$, ** $P \leq 0.01$, *** $P \leq 0.001$.

Table 3.1. Correlation coefficients (r) and line of best fit between the various measurements of organic matter and the size and activity of the microbial biomass.

	Organic C	Microbial Biomass C
Microbial Biomass C	0.44 *** linear	-
Basal Respiration	0.53 ns quadratic	0.34 *** quadratic
Metabolic Quotient	0.10 ns linear	0.29 ** quadratic
Arylsulphatase	0.70 *** linear	0.84 * quadratic
Acid Phosphatase	0.69 *** linear	0.78 * quadratic
Aggregate Stability	0.72 *** linear	0.44 * linear

[a] Statistical significance shown as $^{*}P \leq 0.05$, $^{**}P \leq 0.01$, $^{***}P \leq 0.001$.

At the Glenrosa site $\delta^{13}C$ values for soil under native forest ranged from -25.38 to -25.36 % and values for forest plant material ranged from -27.6 to -25.9 %. Sugarcane plant parts had $\delta^{13}C$ values of -11.8 to -12.6 % and values for soil under sugarcane ranged from -22.4 to -17.1 %. At the Hutton site, the native grassy vegetation was dominated by C4 plants and as a consequence the contribution of sugarcane-derived C to soil organic C was not measured.

Sugarcane-derived soil organic C at the Glenrosa site increased over time until it accounted for about 61 % of the total C present in the surface 10 cm in soils that had been under sugarcane for more than 50 years (Fig 3.9). Over the same period, C derived from native vegetation declined exponentially until it accounted for only about 49 % of total soil organic C. After 20 to 30 years under sugarcane production, the amount of C in the soil derived from sugarcane approximately equalled that from native vegetation.

3.4 DISCUSSION

3.4.1 Organic Matter Content and Aggregate Stability

The loss of soil organic matter occurred in both Glenrosa and Hutton soils when the natural vegetation was converted to sugarcane monoculture. This is characteristic of soils under natural vegetation being converted to arable agriculture (Paustian *et al.*, 1997; Fenton *et al.*, 1999). As demonstrated in this study, soil organic C contents decrease rapidly in the first 10 to 20 years of cultivation (Fig 3.1). Such a decline under agricultural systems is attributable to i) a considerably lower return of carbonaceous residues to the soil (as a result of wide spacing of crops, removal of harvested products and removal or burning of crop residues) (Haynes and Beare, 1996), ii) tillage-induced aggregate disruption and hence exposure of physically-protected organic material to microbial action and iii) more favourable conditions for decomposition as created by tillage-induced aeration (Haynes and Tregurtha, 1998), irrigation (Campbell *et al.*, 1981) and fertilizer additions (Gregorich *et al.*, 1994).

The Glenrosa site also demonstrates stabilization of the organic matter content at a new equilibrium level after about 10 years (Fig 3.1). Generally, the new equilibrium level attained is determined by i) the ability of the soil to stabilize organic material (eg clay content and mineralogy), ii) the amount and quality of plant residue inputs and iii) the intensity and frequency of tillage (Haynes and Beare, 1996; Paustian *et al.*, 1997).

Carbonaceous residue inputs are minimal for sugarcane in the study area since the crop is preharvest burnt (to remove dead leaves) and the above-ground crop is then harvested and removed. Tillage is only performed at replanting of sugarcane, which occurs after one planted and seven ratoon crops. At the Glenrosa site tillage occurs every 8 years as the crop is harvested on a 12 month cycle, while at the Hutton site, an 18 month cycle leads to tillage every 12 years. Thus the more frequent disruption of aggregates is one possible cause for greater soil organic C losses at the Glenrosa site. The greater air temperature and mean annual rainfall at the Glenrosa site may also be important since this would interact with the greater frequency of tillage to promote organic matter decomposition.

However, considerably higher clay content of the Hutton soil (61%) compared to the Glenrosa soil (18%), is possibly the main reason for the higher organic C content maintained in the former soil (Hassink *et al.*, 1993). Many studies have demonstrated the positive correlation between clay and soil organic C content (Edwards and Bremner, 1967; Hassink *et al.*, 1993) since adsorption of organic molecules onto clay mineral surfaces provides a mechanism of stabilization of soil organic C against microbial attack (Hassink *et al.*, 1993). In fact, Johnston (1986) found that soil texture is one of the most important factors determining the equilibrium of soil organic matter status. Soil mineralogy may also play an important role as the Hutton soil contains substantial amounts of amorphous Al oxides which provide a substantial surface area and therefore a relatively large capacity to adsorb organic matter (Oades and Waters, 1991). In Oxisols aggregation is stabilized principally by inorganic materials (Oades and Waters, 1991; Robert and Chenu, 1992).

The ability of clay particles to adsorb organic matter also contributes to formation and stabilization of soil aggregates, thus protecting the organic matter from potential decomposition (Stevenson, 1994). The linear relationship between aggregate stability and organic C (Table 3.1) for both sites indicates the important role that soil organic matter serves in aggregate stability (Tisdall and Oades, 1982; Haynes and Beare, 1996). The lower clay content of the Glenrosa soil meant it had a lower potential to store organic matter under cultivation than the Hutton soil. However there is no significant difference in aggregate stability between the two sites (Fig 3.4). Thus, surprisingly, even though the organic matter content was substantially lower in the Glenrosa soil under sugarcane, aggregate stability was similar to that in the Hutton soil. In sandier soils, such as the Glenrosa, the enmeshment of organic material and sand particles by fine roots and associated micorrhizal fungal hyphae contributes greatly to the formation and stabilization of macroaggregates (Degens *et al.*, 1994; Degens *et al.*, 1996). Such a mechanism may explain the unexpectedly high aggregate stability in this soil.

3.4.2 Size and Activity of the Microbial Biomass

Increasing years under sugarcane cultivation lead to a pattern of decrease in microbial biomass C similar to that for organic C (Fig 3.2). The decline, however, was more pronounced particularly for the Hutton site. This is demonstrated by the microbial quotient (Fig 3.3) which decreased exponentially with increasing years under sugarcane. Such a trend is expected since when a virgin soil is put under cultivation, it is the labile readily metabolizable fractions of organic matter that are preferentially lost (Hart *et al.*, 1988). As a result, long term of cultivation leads to a proportionally smaller microbial community compared to undisturbed soil. Not only did the size of the microbial biomass decrease with increasing periods under sugarcane but, as expected, microbial activity, as measured by basal respiration, also decreased.

The higher microbial biomass content maintained in the Hutton soil compared to the Glenrosa is to be expected since microbial biomass C content is generally positively correlated to both clay and organic matter content (Table 3.1) of soils (Sparling, 1997; Dalal, 1998). The principle mechanism involved in physical protection of the microbial biomass is thought to be the ability of clay to retain soil organic matter and therefore more substrate C is available to support a larger microbial biomass.

It has been noted by several workers that the microbial quotient typically increases with increasing clay content (Sparling, 1997) up to about 50% clay content (Dalal, 1998). However in this study the two soils (whether under natural vegetation or sugarcane cultivation) did not show a significant difference in microbial quotients. Thus, soil type had no measurable effect on the proportions of soil organic C present in living form. This suggests that readily metabolizable C accounted for a similar proportion of total organic C in the two soils and that at least in this respect, organic matter quality was similar in the two soils under similar cropping histories.

Some have suggested that the metabolic quotient tends to be high in young and recently disturbed soils, and lower in native vegetation (Insam and Haselwandter, 1989). This is because soil disturbance reduces the efficiency of the soil microflora in converting substrate C into cellular C (Wardle and Ghani, 1995). However in this study there was no measurable change in metabolic quotient with increasing years under sugarcane cultivation. The lack of any effect of sugarcane production on the metabolic quotient confirms the conclusion of Wardle and Ghani (1995) that although cultivation represents a severe disturbance, metabolic quotient is not predictably enhanced by this perturbation.

Since basal respiration rates were similar in the two soils, values for the metabolic quotient were higher for the Glenrosa than Hutton soils. A higher metabolic quotient has been interpreted as a response of the soil microflora to adverse environmental conditions (Wardle and Ghani, 1995). The higher metabolic quotient at the Glenrosa site could be due to the greater loss of organic C in this soil. However, there is a very poor and non-significant correlation between organic C content and metabolic quotient (Table 3.1). Thus, a shortage of readily metabolisable C could not be the main stress to microbial activity in the Glenrosa soil. Another contributing factor could be the higher clay content in the Hutton soil. This may have helped maintain steady-state microenvironmental conditions for microbial activity, thus reducing environmental stress, independent of soil disturbance (Wardle and Ghani, 1995). This would favour a lower metabolic quotient in Hutton than Glenrosa soils with the same number of years under sugarcane.

It is also possible that there were differences in the composition of the microbial community in the two different soils. For example, bacterial communities are less efficient at converting substrate C into cellular C than fungi (Kazunori and Oba, 1994). The structure of the soil microbial communities at the two sites is a subject worthy of future study. Differences may exist due to the large differences in soil texture and climate between the sites.

Some workers have found that in arable soils, microbial biomass C is linearly related to organic C only up to about 25 g organic C kg^{-1} (Anderson and Domsch, 1989; Wiegand *et al.*, 1995). Above this level, microbial biomass was shown to increase more slowly. A relative enrichment of more recalcitrant soil organic matter (ie humic substances) in soil with a high organic matter has been suggested as the reason for subproportional increases in microbial biomass above the 25 g C kg^{-1} level (Wiegand *et al.*, 1995). The results of this study, however, indicate a linear relationship up to 56 and 51 g C kg^{-1} respectively for the Glenrosa and Hutton soils (Fig 3.8). As noted previously, the soils with the highest organic matter were from under undisturbed native vegetation and are also likely to have the highest amount and proportions of readily-metabolisable C.

The decline of arylsulphatase and acid phosphatase activities (Figs 3.6 and 3.7 respectively) with long-term sugarcane cultivation was to be expected due to the substantial decreases in soil organic matter content and in the size and activity of the microbial biomass (Gupta and Germida, 1988). This is confirmed by the strong correlations between arylsulphatase and acid phosphatase activities and both organic C and microbial biomass C (Table 3.1). Soil organic matter not only provides a substrate for the microbial biomass, but it also plays an important role in protecting soil enzymes by immobilizing them in a three-dimensional network of clay and humus complexes (Tabatabai, 1994). Thus, the Hutton site, with higher organic matter and clay contents, exhibited significantly higher arylsulphatase and acid phosphatase activities than the Glenrosa site.

Acid phosphatase and arylsulphatase are both hydrolases which are involved in mineralization of soil organic P and S with the release of orthophosphate and sulphate respectively (Tabatabai, 1982). Thus the declines in the activities of these enzymes with increasing years under sugarcane indicate the potential for reduced cycling of these nutrients within the soil system and a reduction in soil fertility.

3.4.3 $\delta^{13}C$ Data for the Glenrosa Soil

The rapid increase in the amount of crop-derived C in the soil during the first 10 to 20 years has been noted previously (Balesdent *et al*,. 1988; Collins *et al*., 1999), and as shown in Fig 3.9, it corresponds to the period when the bulk of the organic matter derived from native vegetation is lost. The amount of soil organic C derived from the crop after > 50 years of sugarcane (61 %) is high compared to results from most other studies where $\delta^{13}C$ techniques have been used (Balesdent *et al*., 1988; Vitorello *et al*., 1989; Collins *et al*., 1999). For example, on a Brazilian Oxisol that had been under sugarcane for 12 or 50 years, the amount of soil organic C derived from crop amounted to 11 and 36 % respectively (Vitorello *et al*., 1989). Collins *et al*. (1999) showed that under continuous maize in Ontario fro 25 years, about 30 % of the soil organic C in the plough layer was derived from the crop while in soil profiles that had been under maize for 8 to 35 years, equivalent values ranged from 22 to 40 %. Nevertheless, production of wheat in Missouri for 100 years resulted in replacement of 51 % of the soil organic C by wheat residue-derived C (Balesdent *et al*., 1988); the corresponding value after 27 years was about 27 %. The relatively high percentage of crop-derived C present in this soil under long-term sugarcane is presumably related to the massive loss of forest-derived C (89 %) that occurred.

It is clear from this data that a sugarcane crop even under preharvest burning contributes substantial amounts of organic matter to the soil. Such inputs would be considerably higher if green cane harvesting with retention of a trash blanket were employed as a farming practice (Wood, 1985; Heenan *et al*., 1995; Blair *et al*., 1998; Van Antwerpen and Meyer, 1996).

CONCLUSIONS

Long-term sugarcane production causes a marked decline in the soil organic matter content and related soil microbial and physical properties in the surface 10 cm. An equilibrium is reached after about 20 to 30 years, which is determined greatly by clay content of the soil. The sandy coastal soils that are commonly used in the cultivation of sugarcane in KwaZulu-Natal exhibit marked deterioration of the soil's condition. Such degradation is likely to lead to soil structural breakdown, soil erosion, compaction and surface runoff. The implementation of sustainable agricultural management techniques such as trash retention (as opposed to preharvest burning), zero-tillage, and the use of green crops in rotation should be considered as methods to arrest soil further degradation.

The extent of soil degradation under sugarcane production must be kept in perspective in relation to other common agricultural land uses within the province. How sugarcane production compares to maize production and cultivated pastures will be investigated in the following chapter.

CHAPTER FOUR

COMPARATIVE INFLUENCE OF AGRICULTURAL LAND MANAGEMENT PRACTICES ON ORGANIC MATTER CONTENT, MICROBIAL ACTIVITY AND AGGREGATE STABILITY IN THE PROFILES OF TWO OXISOLS

4.1 INTRODUCTION

The loss of soil organic matter is often considered as the most serious factor associated with agriculturally-induced soil degradation (Gregorich *et al.*, 1994; Paustian *et al.*, 1997). Such a loss can have detrimental effects on soil physical, chemical, and biological properties. To maintain and improve soil organic matter content is generally accepted as an important objective of any sustainable agricultural system (Gregorich *et al.*, 1994).

Although soil acidification and to a lesser extent salinization are considered as major aspects of soil degradation under commercial agriculture in South Africa, loss of soil organic matter is not generally accepted as an important issue. Thus, little information exists as to the effects of common agricultural practices on soil organic matter content, and related soil properties. However, there are large areas where commercial arable agriculture presents a high potential for organic matter degradation. In KwaZulu-Natal sugarcane (which is preharvest burnt) is the major crop in the coastal regions, while maize is more commonly cultivated further inland. In the maize industry, a trend away from conventional towards zero tillage aims at conserving soil water and soil organic matter. Dairy farming is also commonly practised and permanent kikuyu grass pastures are the basis of such agriculture. However, low production is experienced in the relatively cold winter months and farmers supplement with separate annual ryegrass pastures for winter production. These ryegrass pastures are conventionally cultivated each spring and resown with annual ryegrass.

The purpose of this study is to compare the effects of the above land uses (sugarcane, maize under conventional (CT) and zero tillage (ZT), and annual and permanent pasture) with that of undisturbed native grassland, with regards to soil organic matter and some related soil physical and biological properties.

For this purpose, two separate farms in the KwaZulu-Natal midlands, on the same soil type, containing fields of contrasting long-term cropping histories were chosen and studied. At one of the sites, sugarcane monoculture was growing alongside long-term maize, annual and perennial pasture, thus enabling a direct comparison between sugarcane and other land uses. Such a comparison is important in order to put the results of chapter 3 (which showed a substantial loss of organic matter) into perspective.

4.2 MATERIALS AND METHODS

Changes in soil organic matter status up to a depth of 40 cm in fields under different known cropping histories were compared at two sites in the midlands of the South African province of KwaZulu-Natal. The soils prevalent at both sites are classified as Hutton Form (Farmingham series) (Soil Classification Working Group, 1991) or as Rhodic Ferralsols (FAO). These soils have clay content of between 55 to 65 % and the mineralogy is mainly kaolinite and halloysite and there are appreciable amounts of crystalline sesquioxides, gibbsite and interlayered chlorite.

Baynesfield Estate (27° 22' S and 30° 45' E) was the first site and known cropping histories of fields were: >50 years permanent kikuyu pasture; >50 years annual ryegrass pasture; > 30 years continuous maize under conventional tillage (CT); > 30 years sugarcane under the practice of preharvest and crop-residue burning; and undisturbed native grassland. Site 2 was at Cedara Agricultural College / Research Station (29° 32' S and 30° 17' E) and cropping histories of fields were: >50 years permanent kikuyu pasture; >50 years annual ryegrass pasture; >25 years continuous maize under conventional tillage (CT); >25 years continuous maize under zero tillage (ZT); and undisturbed native grassland.

At Baynesfield, the mean annual rainfall is 844 mm and mean monthly temperatures range from a maximum of 21.1 °C in January to a minimum of 13.3 °C in June. At Cedara, the mean annual rainfall is 874 mm and mean monthly temperatures range from a maximum of 19.9 °C in January to a minimum of 11.3 °C in June.

Three plots (about 10m x 10m in area) were randomly chosen within each field. Five samples were taken from each plot using a bucket auger to a depth of 40 cm and sectioned into 0 - 5; 5 - 10; 10 - 20; 20 - 30; and 30 - 40 cm layers. Samples for each depth in each plot were bulked. There were three replicate samples of each depth of each field at each site.

Soil samples were analysed for organic C, microbial biomass C, basal respiration and aggregate stability as described in section 3.2. From the above measurements, the microbial and metabolic quotients were calculated (section 3.2). In order to calculate the organic matter content within the 40 cm soil profile, per unit area, results were converted to volumetric content. For this purpose, bulk density was measured in quadruplet in the 0 - 10, 10 - 20, 20 - 30, and 30 - 40 cm layers for each sample using the core method.

The standard error of the mean for the various measured parameters for each field and depth was calculated using values for the three replicate samples.

4.3 RESULTS

At site 1, the organic C content in the surface 10 cm followed the order kikuyu > annual ryegrass ≥ natural grassland ≥ sugarcane = maize (Fig 4.1a). Below 10 cm, the treatment effects are less clear.

Within the first 10 cm of soil, the microbial biomass content followed the order of kikuyu > natural grassland ≥ annual ryegrass ≥ maize ≥ sugarcane (Fig 4.1b). Below a depth of 10 cm, the microbial biomass C content of each treatment effect did not significantly differ from the others.

The basal respiration in the top 10 cm of the soil profiles followed the very clear order of kikuyu > natural grassland = annual ryegrass ≥ maize > sugarcane (Fig 4.1c). Below 20 cm, the basal respiration under maize was higher than that of sugarcane. The basal respiration of cane-planted soils did not change throughout the profile, while that of kikuyu, native grassland and annual ryegrass treatments decreased sharply in the top 15 cm.

Aggregate stability (Fig 4.1d) in the surface 20 cm showed the trend kikuyu = native grassland > annual ryegrass > maize > sugarcane. Below 20 cm, kikuyu pastures generally had greater aggregate stability than the other treatments.

The microbial quotient decreased with depth under kikuyu, annual ryegrass and native grassland (Fig 4.2a). However, no significant differences were noticed with increasing depth under sugarcane and maize. In the surface 5 cm the trend followed was kikuyu > native grassland ≥ annual ryegrass = maize ≥ sugarcane, while between 30 and 40 cm the trend was maize = sugarcane ≥ native grassland = annual ryegrass ≥ kikuyu.

The metabolic quotient (Fig 4.2b) for sugarcane was higher than the other treatments in the top 15 cm. Native grassland at a depth of between 30 and 40 cm had a metabolic quotient which was significantly higher than all other treatments other than kikuyu at the same depth.

Quantities of organic C and microbial biomass C calculated on a per-hectare basis to a depth of 40 cm at site 1 are shown in Table 4.1 along with bulk density values. The differences in organic C and microbial biomass C between land uses were generally greatest in the top 10 cm. Similar to the trends in Fig 4.1a, both organic C and microbial biomass C values for kikuyu pasture were higher than those for native grassland, and those under maize (CT) and sugarcane were generally lower than those under native grassland. Values for microbial biomass C were lower under sugarcane than under maize. The microbial biomass C values under annual ryegrass pasture were similar to those of native grassland. Bulk density values in the 0 - 10 cm layer were least for kikuyu pasture and native grassland and greatest under annual pasture and sugarcane.

At site 2, land management effects on organic C (Fig 4.3a) contents were evident to a depth of 40 cm unlike that of site 1, where differences were only really noticeable up to a depth of 10 cm. The treatment effects were, however, still most obvious in the top 5 cm where the trend was kikuyu pasture ≥ native grassland > maize (ZT) ≥ annual ryegrass pasture > maize (CT). Within the top 5 cm of the soil profile, the organic C content of maize (ZT) was higher than maize (CT), but this trend was reversed deeper than 10 cm.

The treatment effects on microbial biomass C (Fig 4.3b) and basal respiration (Fig 4.3c) followed the same trend as organic C, but kikuyu pasture values were much higher than those of native grassland at all depths. In the surface 5 cm, both values were considerably higher for maize (ZT) than maize (CT).

Aggregate stability (Fig 4.3d) followed the same trend as the other measurements but values for maize (ZT) remained higher than those of maize (CT) throughout the measured profile.

The microbial quotient values (Fig 4.4a) for treatment and depth showed a trend similar to that of microbial biomass C. In the surface 10 cm, the order for metabolic quotient was maize (CT) > maize (ZT) = ryegrass pasture >native grassland > kikuyu pasture (Fig 4.4b). The metabolic quotient for maize (CT) was notably higher than other treatments in the top 20 cm, while the values for kikuyu pasture were lower at all depths.

In the surface 10 cm, a clear order in bulk density values (Table 4.2) was followed, kikuyu pasture < native grassland < maize (CT) = annual ryegrass pasture < maize (ZT). Values for organic C and microbial biomass C in the top 40 cm were adjusted to a volume basis and both followed the order of kikuyu pasture = native grassland > ryegrass pasture > maize (ZT) = maize (CT). Both organic C and microbial biomass C in the surface 10 cm were notably higher for the treatment of maize (ZT) compared to maize (CT).

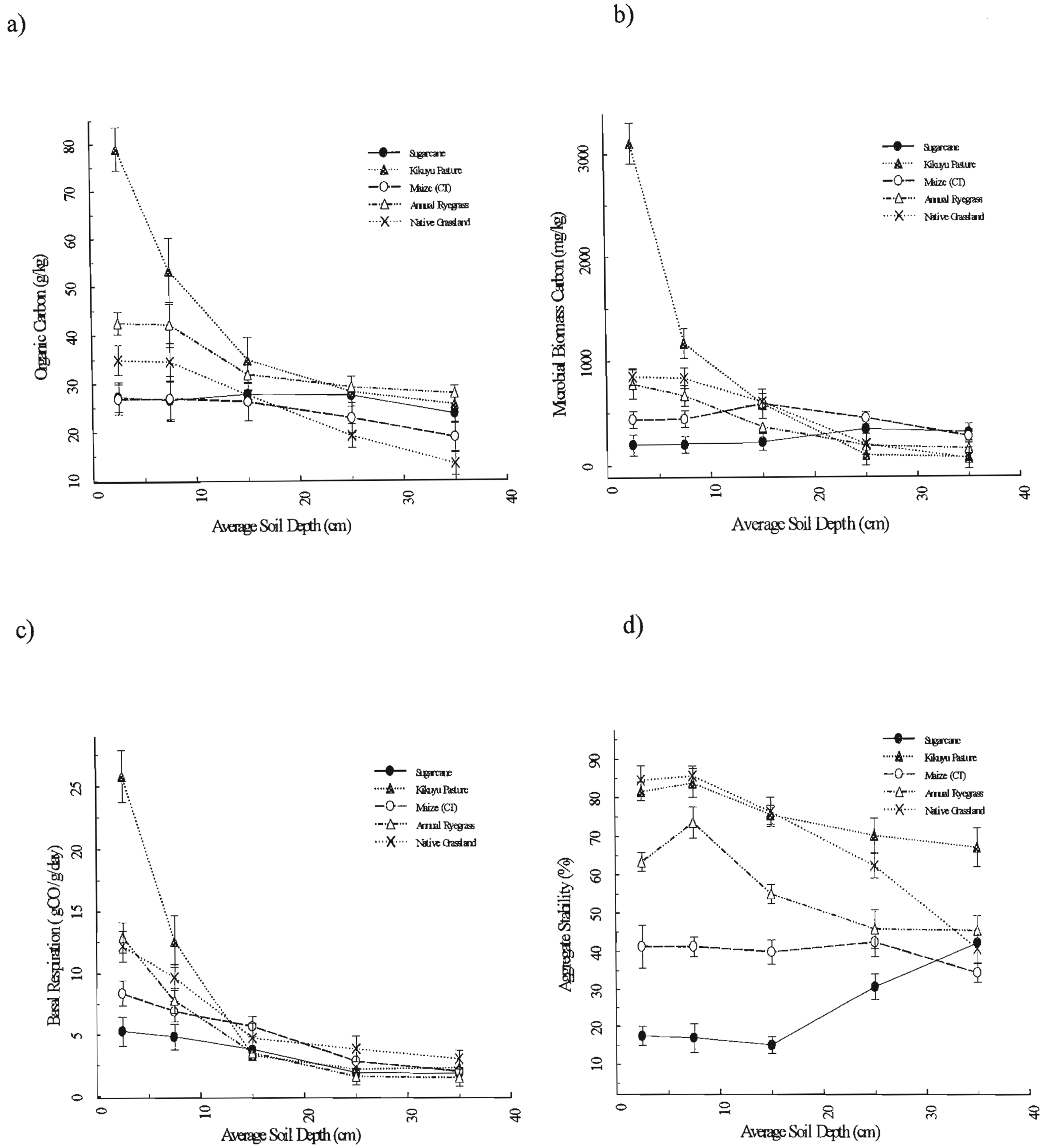

Fig 4.1 Effect of five long-term agricultural practices on a) organic C content, b) microbial biomass C content, c) basal respiration and d) aggregate stability in the soil profile up to a depth of 40 cm at Baynesfield estate (site 1).

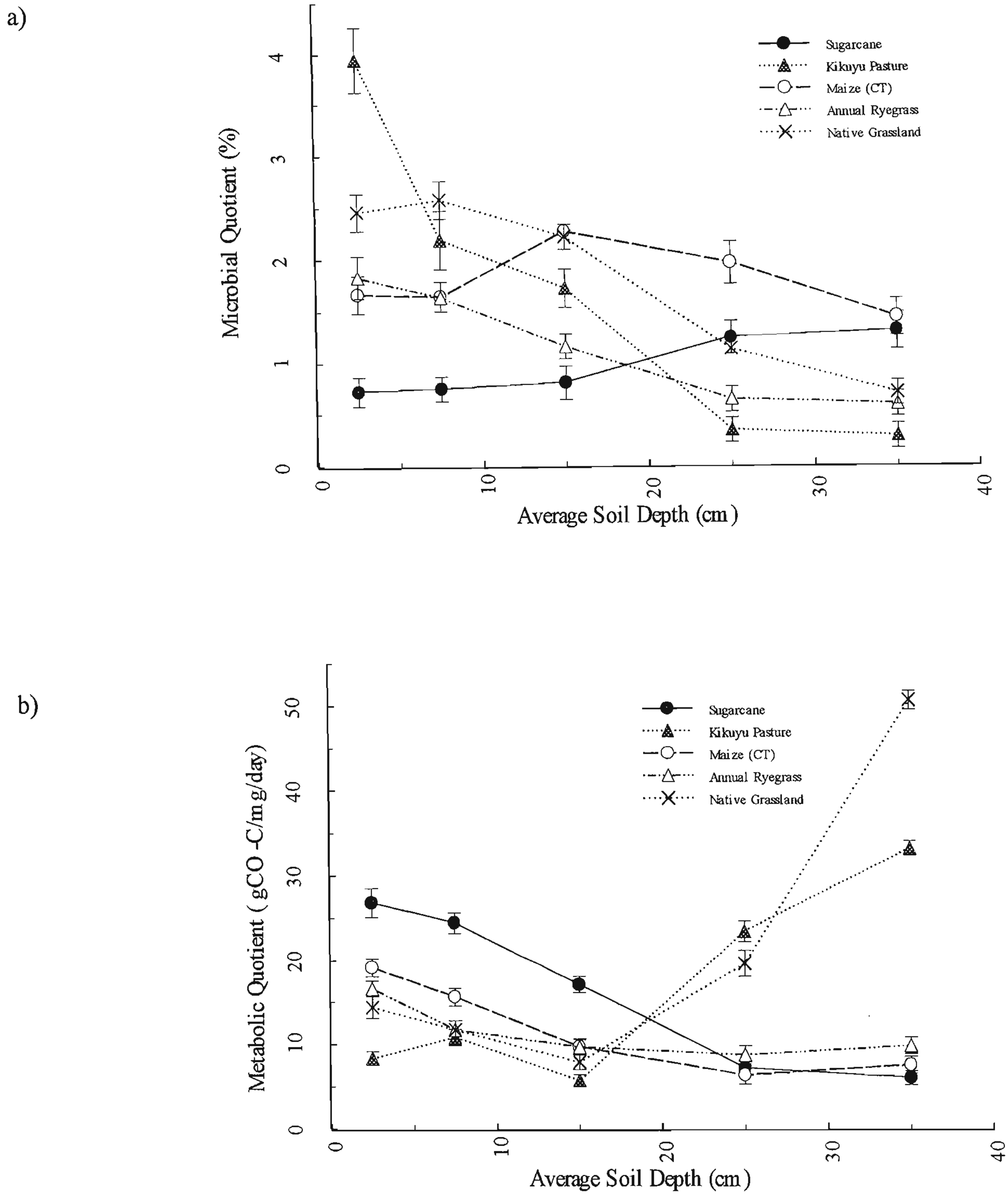

Fig 4.2 Effect of five long-term agricultural management practices on a) microbial quotient and b) metabolic quotient in the soil profile up to a depth of 40 cm at Baynesfield Estate (site 1).

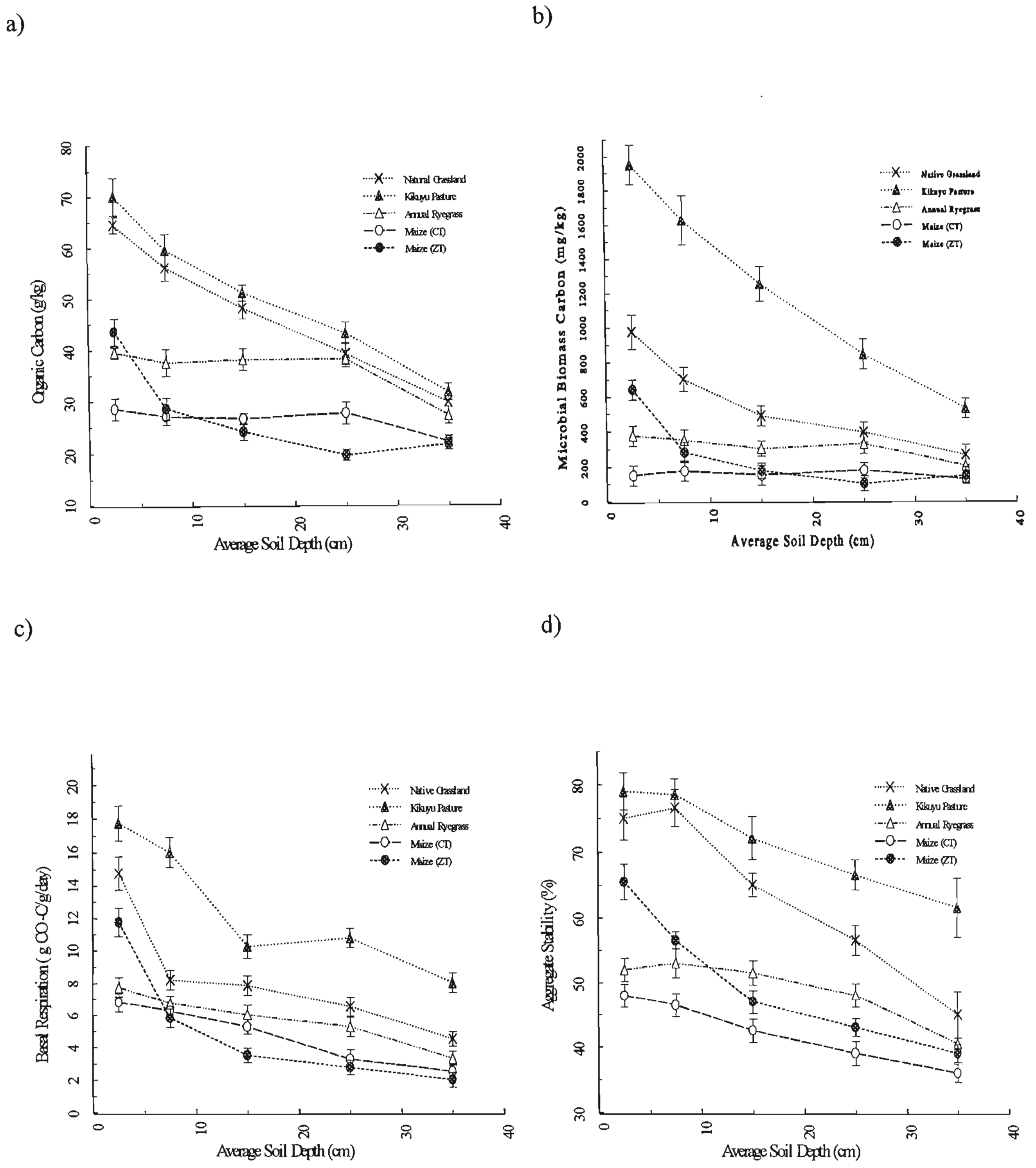

Fig 4.3 Effect of five long-term agricultural management practices on a) organic C content, b) microbial biomass C content, c) basal respiration and d) aggregate stability in the soil profile up to a depth of 40 cm at Cedara Research Farm (Site 2).

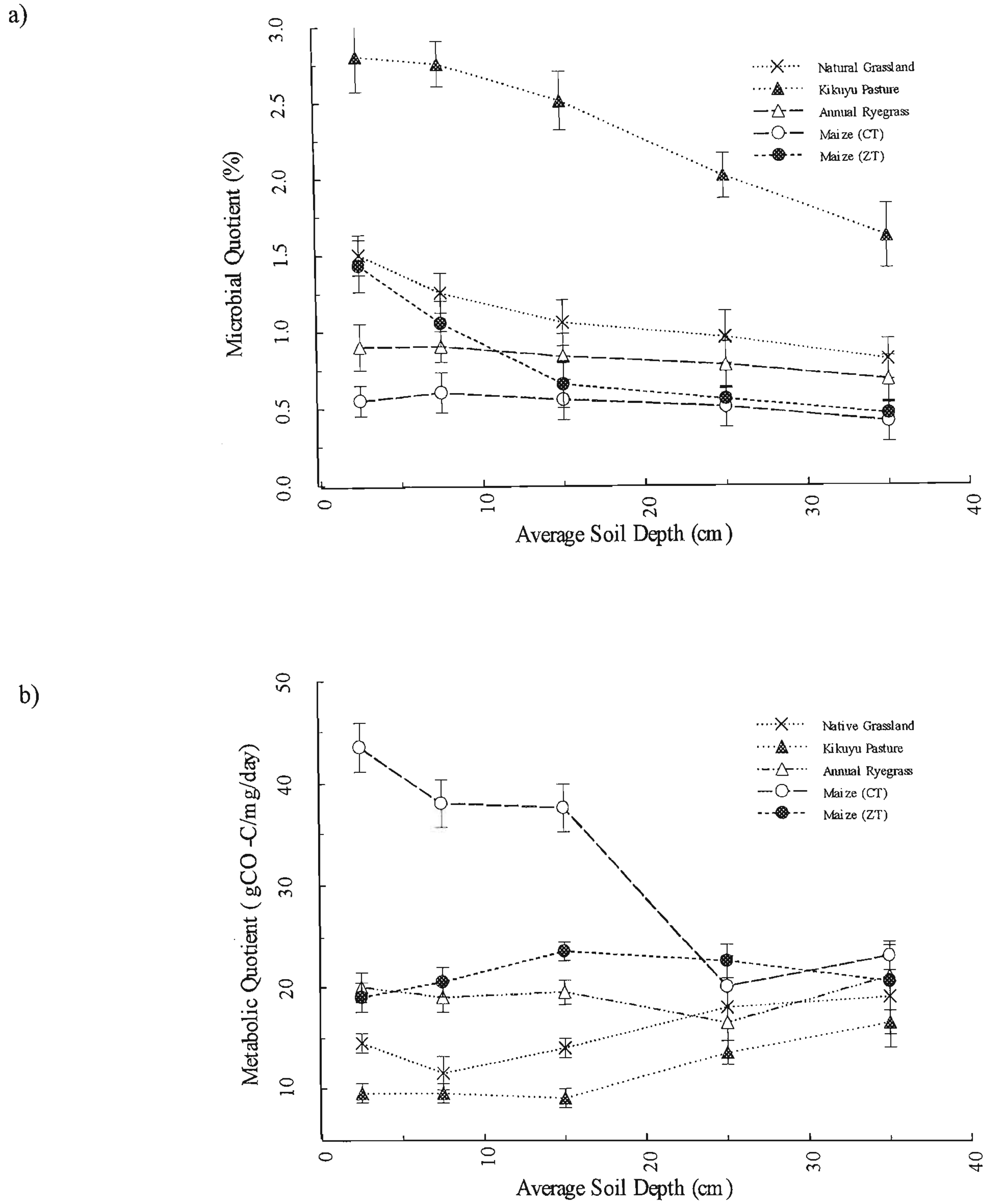

Fig 4.4 Effect of five long-term agricultural management practices on a) microbial quotient and b) metabolic quotient in the soil profile up to a depth of 40 cm at Cedara Research Farm (site 2).

Table 4.1 Effect of land management at Baynesfield Estate on bulk density, organic C and microbial biomass C in the soil profile on a volume basis (SE shown in parentheses).

Measurement and soil depth (cm)	Native Grassland	Kikuyu Pasture	Annual Ryegrass Pasture	Maize (CT)	Sugarcane
Bulk Density (mg/kg)					
0 - 10	1.17 (0.02)	1.05 (0.02)	1.32 (0.02)	1.25 (0.03)	1.37 (0.02)
10 - 20	1.30 (0.02	1.29 (0.02)	1.39 (0.02)	1.28 (0.02)	1.34 (0.02)
20 - 40	1.32 (0.02)	1.37 (0.03)	1.39 (0.03)	1.33 (0.03)	1.32 (0.03)
Organic C (mg/ha)					
0 - 10	45.7 (3.2)	69.3 (3.3)	59.7 (2.9)	33.6 (3.1)	36.8 (2.3)
10 - 20	39.3 (3.4)	44.9 (3.9)	44.2 (3.3)	33.4 (2.8)	37.1 (2.6)
20 - 40	67.1 (2.9)	73.1 (4.3)	63.8 (3.9)	56.9 (3.3)	67.0 (3.1)
Total (0 - 40)	152 (4.1)	187 (5.7)	168 (6.1)	124 (8.0)	141 (3.7)
Microbial Biomass C (kg/ha)					
0 - 10	985 (110)	2244 (146)	1210 (98)	550 (96)	269 (48)
10 - 20	784 (96)	755 (79)	626 (61)	653 (86)	295 (63)
20 - 40	504 (81)	349 (54)	439 (63)	936 (77)	847 (86)
Total (0 - 40)	2273 (164)	3348 (271)	2275 (169)	2139 (139)	1411 (154)

Table 4.2 Effect of land management at Cedara Research Station on bulk density, organic C and microbial biomass C in the soil profile on a volume basis (SE shown in parentheses).

Measurement and soil depth (cm)	Native Grassland	Kikuyu Pasture	Annual Ryegrass Pasture	Maize (CT)	Maize (ZT)
Bulk Density (mg/kg)					
0 - 10	1.11 (0.01)	1.06 (0.02)	1.28 (0.02)	1.25 (0.02)	1.40 (0.01)
10 - 20	1.30 (0.02)	1.26 (0.02)	1.31 (0.02)	1.33 (0.02)	1.32 (0.02)
20 - 40	1.32 (0.02)	1.35 (0.03)	1.38 (0.03)	1.33 (0.03)	1.34 (0.03)
Organic C (mg/ha)					
0 - 10	66.9 (2.9)	68.7 (2.1)	49.7 (2.4)	35.3 (2.6)	49.7 (2.8)
10 - 20	62.4 (2.3)	64.3 (2.8)	49.3 (2.6)	35.6 (3.0)	32.6 (2.1)
20 - 40	95.7 (3.1)	100.5 (3.1)	90.3 (3.3)	66.6 (3.4)	57.6 (2.9)
Total (0 - 40)	225 (6.3)	233 (5.4)	189 (8.1)	137 (7.3)	140 (4.6)
Microbial Biomass C (kg/ha)					
0 - 10	743 (81)	728 (39)	636 (28)	441 (56)	696 (51)
10 - 20	805 (73)	810 (69)	645 (33)	474 (62)	430 (39)
20 - 40	1308 (102)	1353 (110)	1244 (127)	882 (76)	771 (69)
Total (0 - 40)	2856 (173)	2891 (166)	2525 (169)	1797 (101)	1897 (99)

4.4 DISCUSSION

4.4.1 Soil Organic C

To determine the effects of various land uses on soil organic C and related microbial properties, it is necessary to compare values with those under undisturbed native vegetation (Gregorich *et al.*, 1994; Sparling, 1997). In other words, the values need to be compared with those in soil which has not been disturbed by agricultural or any other human activity. At each site, samples were therefore taken under local natural grassland on the same soil type.

At both sites a conversion to kikuyu pasture resulted in an increase in soil organic C content (Figs 4.1a and 4.3a) mainly due to an increase in organic matter input compared to that under native grassland. The above-ground production of kikuyu is between 10 and 16 Mg ha^{-1} yr^{-1} while that of native grassland is about 3 to 5 Mg ha^{-1} yr^{-1}. Thus, the above-ground input of organic matter in the form of dying shoots and animal dung will be greater than under native grassland. Presumably, kikuyu exhibits greater below-ground organic matter production in the form of turnover of the root biomass and associated rhizosphere microflora.

The soil organic C content decreased when native grassland was converted to long-term maize (CT) and sugarcane cultivation (Fig 41a). Tillage has been found to increase aeration and disrupt soil aggregates thereby increasing exposure of previously inaccessible organic matter to microbial attack (Haynes and Beare, 1996). However, although organic C contents of both maize (CT) and sugarcane are similar, their tillage programmes are very different. Conventional tillage of maize means the soils are ploughed every year. Sugarcane, on the other hand, is only ploughed after a series of one planted crop and seven ratoon crops, which is about every 16 years. Nonetheless, reduced organic matter inputs are also known to play a role in the loss of organic C that occurs under arable crops. The reduced inputs are a result of wider spacing of crops, harvesting, and removal of above-ground dry matter, as well as the burning of crop residues. Within a cane field, the interrow inputs are minimal, and compaction of the interrow is not conducive to root growth from the rows, thus a large proportion of the field is effectively left fallow. Indeed, preharvest burning of the cane, and the postharvest burning of residues (green cane tops) leads to minimal organic matter input except via root turnover in the row area. The distribution of returned organic matter differs for maize (CT). In a maize field, the above- and below-ground residues are spread randomly by tillage, and resowing each year takes place in different positions in the field. The crop residues are grazed and dung distribution is also random. Thus, although maize (CT) is tilled more frequently than sugarcane fields, the organic matter input is greater over the whole area, resulting in very similar measured organic C contents between the two land uses.

The accumulation of organic C in the surface 5 cm under maize (ZT) as noted at site 2 (Fig 4.3a) is common (Paustian *et al.*, 1997) and is a result of crop residue and weeds being left on the soil surface to decompose. However, under maize (CT), tillage results in a more uniform distribution of organic C up to a depth of 20 cm as noted at both sites. The total amount of organic C found within the profile to a depth of 40 cm is equal for both maize (ZT) and (CT) as has been found by other researchers (Powlson and Jenkinson, 1981; Haynes and Knight, 1989). This similarity is most likely due to similar organic matter inputs (such as stubble, roots and root exudates) and annual decomposition rates being similar. Thus zero-tillage results in a change in distribution of organic C, but not necessarily to a change in the total amount of organic C present.

Interestingly, in comparison with undisturbed grassland, the use of annual ryegrass pasture resulted in a slightly higher organic C content at site 1, but a decrease at site 2. The organic C content under an annual pasture is the result of a balance between large organic matter inputs from the pasture versus an increased decomposition rate induced by annual tillage. The dry matter production of grazed irrigated annual ryegrass pasture is 10 to 13 Mg ha^{-1} yr^{-1}. Depositions of animal dung, turnover of root biomass during the growing season, and the turning in of remaining pasture at the end of the season are all sources of organic matter. In all, annual ryegrass pasture will have a higher organic matter input than native grasslands. Tillage, however, breaks up soil aggregates, mixes and incorporates plant residues, and increases decomposition of both soil- and plant-derived C. From the results obtained from site 1, it appears that the effect of increased organic C inputs outweighed the greater decomposition rate and organic matter content increased. However at site 2, which has a higher initial organic matter content, the tillage effect outweighed the higher organic matter inputs and organic matter content equilibrated at a lower level than that under native grassland.

The production of sugarcane and maize resulted in lower organic C contents than annual ryegrass. Sugarcane and maize are spaced relatively widely and organic matter is removed during harvest of the cane and cobs. By contrast, annual ryegrass has greater planting densities, and the rapidly growing sward is utilized by grazing animals and much organic matter is returned to the field in the form of dung. It is therefore evident that organic matter inputs are greater under annual pasture than annual crops and as a result the equilibrium soil organic matter content attained is also greater.

4.4.2 Size and Activity of Microbial Biomass

The size (microbial biomass C) and activity (basal respiration rate) of the microbial biomass is influenced greatly by the supply of C, and thus trends for these parameters for all the treatments were found to be very similar to those of organic C. However, the effects of kikuyu pasture were far greater on microbial biomass C and basal respiration than on organic C. This indicates a large labile C input under permanent grazed pasture. This originates primarily from the turnover of the large ramified grass root system and the associated microbial biomass and from cycling of organic matter via dung deposition (Haynes and Knight, 1989). Thus the microbial quotient is highest under kikuyu pasture in the top 5 cm at site 1 and 40 cm at site 2. Although the organic C measured under both sugarcane and maize was very similar at site 1, microbial biomass C and basal respiration under maize was greater than under sugarcane. This most likely reflects the poorer labile C supply in the effectively fallow interrows of sugarcane fields (as discussed earlier).

Although the microbial quotient under sugarcane was lower, the metabolic quotient was higher than under maize. An increase in the metabolic quotient is a result of soil microflora in response to an increase in stress or disturbance (Wardle and Ghani, 1995). It seems likely that the main stress factor in the sugarcane interrow will be a low supply of labile C. Soil compaction could be an additional factor limiting microbial activity.

It is most likely that microbial activity is greater where the root density is greater, ie within the rows as opposed to the interrows of sugarcane fields. Root exudation of organic material, and root turnover are likely to be the major organic matter inputs in the row. Hartemink (1998) found a greater organic C content within the rows than interrow of a sugarcane plantation in Papua New

Guinea, and Spain *et al.* (1990) found a greater earthworm count within the rows of sugarcane plantations in Northern Queensland. Thus there is potential for further research into the spacial distribution of both the size and activity of the microbial biomass within a field (especially between rows and interrows).

Treatments resulting in a decrease of organic C content with depth (kikuyu pasture, annual ryegrass pasture and native grassland at site 1, and kikuyu pasture, native grassland, and maize (ZT) at site 2) also caused a decrease in the size (microbial biomass C) and activity (basal respiration) of soil microflora. While the microbial quotient also decreased with depth for these treatments, the metabolic quotient remained either unaffected, or increased. This is a result of a lesser input of fresh decomposable plant material with an increase in depth. Conversely, with increasing depth there is an increase in humified, and therefore less decomposable organic matter. A high microbial quotient indicates a large amount of readily available C substrate (Anderson and Domsch, 1989) and therefore a decrease in this measurement with depth was not surprising. Metabolic efficiency decreases when recalcitrant humus is used as a substrate in lieu of a readily decomposable C source, resulting in a high metabolic quotient. As the amount of recalcitrant organic substrate increases with depth, so does the metabolic quotient. Similar results have been reported by Lavahum *et al.* (1996).

While maize (CT) and sugarcane at site 1 did not show any real change in organic C and microbial biomass C contents, or microbial quotient with increasing depth, the metabolic quotient decreased significantly. This may be a result of great stress in the surface soil layers, in the form of both low C supply, and disturbance by tillage.

The metabolic quotient is often used as an indicator of ecosystem development (Insam and Haselwandter, 1989; Anderson and Domsch, 1990). The theory suggests that as an ecosystem matures, there is greater competition for available resources. Such competition allows for the survival of the most efficient users of substrate C. Thus a microbial biomass develops with a high metabolic efficiency and conservation of available substrate leads to a low metabolic quotient. Indeed, at both sites, native grassland showed a lower metabolic quotient in the surface 10 cm than the arable and annual pasture soils. Tillage would cause great disturbance and thus stress to the soil system leading to a less efficient use of available substrate. The higher metabolic quotient under maize (CT) compared to maize (ZT) underlines the negative effect of tillage. Although Wardle and Ghani (1995) suggest that the effects of tillage on metabolic quotient are not always consistent, other workers have found a positive correlation between the two (Breland and Eltun, 1999; Haynes, 1999b).

Surprisingly, metabolic quotient under permanent kikuyu pasture was lower than under native grassland. Although kikuyu pasture is not tilled, other perturbations such as treading and compaction caused by grazing dairy cattle occur. Such pastures receive dung and urine in localised patches, large fertilizer N applications (about 300 kg N ha^{-1} yr^{-1}), and maintenance applications of P, K and lime. Perhaps factors such as soil acidity, P deficiency and water stress under native grassland cause greater stress than that experienced under kikuyu pasture.

At site 2, the use of permanent kikuyu pasture instead of native grassland resulted in an increase in both the organic C and microbial biomass C contents throughout the soil profile while the metabolic quotient was decreased. On the other hand, the use of annual ryegrass decreased the organic C and microbial biomass C contents and caused an increase in the metabolic quotient.

Thus factors that decrease the size of the microbial biomass tend to increase the metabolic quotient and vice versa (Insam and Haselwandter, 1989; Breland and Eltun, 1999).

4.4.3 Aggregate Stability

Soil organic matter content is often closely related to the aggregate stability of a soil (Haynes and Beare, 1996). For example, at site 2, the trends of organic C content followed by each of the land uses is very similar to those of aggregate stability. The high aggregate stability under native grassland and kikuyu pasture is not necessarily only related to the high soil organic matter content and thus the binding actions of the humic material. The microbial biomass produces extracellular polysaccharides that act as glues whilst fine roots and associated mycorrhizal hyphae have enmeshing and stabilizing effects on soil aggregates (Degens, 1997). Thus the high root density and large microbial biomass under grasslands and pastures are important factors favouring the formation of stable aggregates. The greater aggregate stability in the 25 to 40 cm layers under kikuyu grass compared to other land uses, at both sites, is probably due to its extensive and deep root system.

The low aggregate stability of sugarcane compared to maize (CT) at site 1, despite their organic C contents being so similar, suggests that total organic C content is not always a good indicator of aggregate stability. The effectively fallow nature of sugarcane interrows leads to a smaller microbial biomass (section 4.4.2) which translates into a lower aggregate stability. This would enhance the soil's potential for structural breakdown in the surface layers of sugarcane compared to other land uses. Although the aggregate stability could have been greater within the rows due to a higher soil microbial biomass, the poor aggregate stability between the rows would still be of practical significance. On sloping land, for example, surface water runoff takes place between the rows, and a low aggregate stability will favour losses of soil by water erosion. The low aggregate stability will also predispose the soil to structural breakdown and compaction. The high bulk density measurements from the 0 - 10 cm layer under sugarcane (Table 4.1), are a reflection of compaction in the interrow area. Compaction is a common problem in the interrows of sugarcane plantations (Hartemink, 1998) and this leads to increased runoff of rain and irrigation water.

4.5 CONCLUSIONS

A few relatively simple measurements can be used to determine the organic matter content and the size and activity of the soil microbial biomass, so enabling a comparison of the effects of different agricultural land uses on soil quality.

The results from this trial showed that a permanent improved pasture increased the soil organic matter content compared to native grassland. On the other hand, conventional arable cultivation of maize and sugarcane created a significant decrease in the organic matter content. A comparison of zero-tillage versus conventional tillage practices showed a redistribution of organic matter within the soil profile, where most organic matter was found in the surface 5 cm layer under zero-tillage. The size and activity of the microbial biomass showed broadly similar trends.

Interestingly, microbial biomass C, basal respiration, microbial quotient and aggregate stability under sugarcane were lower than those under maize, even though the organic C contents were similar. The random samples taken from both within and between the rows under both crops led to this result. Because sugarcane remains in the same position for up to 16 years, and root turnover (as the only organic matter input under the practice of burning) is dominant within the row, the interrow of sugarcane plantations remains effectively fallow. Thus the supply of labile readily-metabolizable organic matter in the interrow presumably becomes depleted, and this results in a reduced microbial activity.

There is scope for future research on detailed comparisons of changes in organic matter content and size and activity of microbial biomass between rows and interrows of sugarcane fields.

CHAPTER FIVE

CONCLUSIONS

In an attempt to discover possible reasons for yield decline in long-term preharvest burnt, sugarcane monoculture, the quality of soils at two sites were determined using analyses of organic matter content and related microbial biomass and physical properties. The organic matter content, size and activity of the microbial biomass, and aggregate stability were greatly reduced within the first five years of cultivation. Generally, greater soil degradation took place at the Glenrosa site compared to the Hutton one. The lower clay content, and more frequent tillage coupled with greater temperatures and rainfall at the Glenrosa site led to a greater loss of soil organic matter and microbial biomass C. This was confirmed by the higher metabolic quotient which indicated greater stress and disturbance in the Glenrosa soil. Such soil degradation poses the threat in relation to soil structural breakdown, erosion, compaction and surface runoff. For sugarcane monoculture to be sustainable, it is vital for soil organic matter to be maintained and improved. The implementation of greencane harvesting (as opposed to preharvest burning) and trash retention, zero tillage and the use of green crops in rotation are considered as methods of arresting soil degradation.

When major land uses of KwaZulu-Natal were compared against native grassland, pastures such as annual ryegrass, and in particularly permanent kikuyu, greatly improved soil quality. Conventional cultivation of sugarcane and maize, on the other hand, caused a redistribution of organic matter and other related soil properties within the soil profile, while zero-tillage left most of the organic matter in the surface 5 cm of the soil. This high organic matter content near the surface under zero-tillage protects the soil from the degradative effects of rain and wind. The effects of tillage, organic matter inputs and spacing of crops, and not simply the type of crop, will determine the soil organic matter status under a particular crop. The low microbial activity and aggregate stability in the effectively fallow interrow area of burnt sugarcane fields may lead to losses of nutrients and soil via runoff and erosion. This is a particular problem on the rolling land used for sugarcane production in the region and results in silt accumulation in irrigation ditches, rivers and dams. Conversion from preharvest burning to green cane harvesting, where a blanket of trash is retained as a mulch over the soil surface of the interrow, would be a method of improving soil microbial activity and physical conditions and protecting the soil surface. Presumably, under a burning regime, soil microbial activity is concentrated in the plant rows of sugarcane fields and this deserves further study.

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