# PROCESSES CONTROLLING CARBON AND NITROGEN DYNAMICS ACROSS VEGETATION TYPES AND LAND USES IN SELECTED SOUTH AFRICAN SITES

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**DECLARATION**

I declare that this research report is my own, unaided work. It is being submitted for the Degree of Master of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination in any other University.

M. Peters

24 day of March 1997

## ABSTRACT

An understanding of the biogeochemistry of carbon and nitrogen in ecosystems is necessary for the sustainability of system function. Transformations, including different land uses, disrupt the natural input:output of soil organic matter and often result in changes in the cycling of carbon and nitrogen. Consequently it is imperative to know how different land uses are likely to alter the pool sizes, flux rates and turnover of carbon and nitrogen in the soil.

The savanna and grassland biomes of South Africa include large areas which have been transformed by man and are the main sites of primary and secondary production. Sites in these biomes along a vegetation and soil type gradient have been investigated. Soil samples from a conserved area, a cultivated area and a livestock area have been sampled. A range of soil properties including the potential rate of nitrogen mineralization, total soil carbon and nitrogen, microbial carbon and nitrogen, soil texture, bulk density, pH and standing dead herbaceous biomass have been quantified. These along with values reported in the literature have been used to validate the CENTURY model, which simulates the turnover of ecosystem attributes on the basis of soil organic matter inputs and outputs.

Results show that the soil organic matter pool sizes for the sites and land uses were positively correlated with the percentage fines (silt-plus-clay) and site aridity. Sites which were moist and had a percentage of fines greater than 45% tended to have 3 times more C and N. Land use, especially cultivation, reduced the amount of SOM at sites by 50% mainly because of the effects on the light fraction mass. The potential rate of N mineralization was not significantly different between sites but the cultivated land use led to the immobilization of N. Possible reasons for this included the negative impact that cultivation has on soil macroaggregates, the lower (< 1.0 g/kg) input of light fraction, and the low (< 10%) percentage fines at these sites. Simulations of the SOM fractions using the CENTURY model for six functional types indicate that similar trends emerged but the model greatly overestimated absolute amounts of SOM.

In conclusion, the absolute quantities of soil carbon and nitrogen are influenced by climate, soil texture, and land use; but the proportion of soil organic matter fractions do not appear to differ per biome or per land use indicating similar turnover times.

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## 1.1 Introduction

Soil nitrogen is often a limiting resource to plants which require it for metabolism and growth. The principal source for this nutrient is the decomposition of plant and animal litter, while additional sources include the input of atmospheric nitrogen by biological and industrial fixation, artificial inputs via fertiliser additions, and the nitrogen obtained from the store of soil organic matter present in the soil.

Considerable understanding already exists for many of the processes listed above and the need is now to integrate them into a model. Such an undertaking will allow for the development of a simulation model which can be used to determine the likely changes in the nitrogen cycle following changes in the environment. Such changes are postulated to occur as a result of increased levels of $CO_2$ and land use changes (Vitousek 1994). It will therefore be necessary to ensure that the cycling of nitrogen be maintained for any disruption to this exchange will inevitably lead to situations of unsustainable use. This is because available nitrogen is closely associated with the level of organic matter in the soil (Parton *et al* 1987). Soil organic matter stores, in turn, are often decreased by land use changes like the conversion of forest to grassland, grassland to pasture and the effects of urbanisation and over-population (Ayanaba *et al* 1976; Campbell and Zentner 1993)).

In light of the above, however, it is perhaps ironic that the exact relationship between the inputs and outputs of N to the soil in South Africa are not well understood, and this project sets out to partially address this need. The project aims to quantify the amount of soil carbon and nitrogen and the potential rate of nitrogen mineralization in selected sites of South Africa. The process of mineralisation describes the conversion of organic nitrogen into ammonium ($NH_4^+$- N). Changes in the environment determine this rate which varies according to several biological and environmental variables (Paul and Clark 1989). Laboratory analysis in conjunction with past and current literature will be used to test the CENTURY model developed by Parton *et al* (1987). Such a model quantifies the turnover of various soil organic matter fractions and includes N mineralization from the organic pools in each. This is a suitable framework from which to estimate plant yield and appropriate land use in the various biomes of South Africa.

## 1.2 Literature Review

### 1.2.1 The cycling of nitrogen

The nitrogen cycle is a good example of a biogeochemical cycle. Such a cycle describes the movement of elements and organic compounds that are essential for life and is generally designated as nutrient cycling. Each cycle is composed of pools or reservoirs which vary in size. These pools can be either large and slow moving having a non-biological component known as the reservoir pool, or it can be a smaller and more active exchange pool that is moving back and forth rapidly between organisms and their immediate environment (Odum 1971).

An essential feature of a biogeochemical cycle is that the fluxes between the pools are regulated primarily by the activity of biological organisms. Such organisms include the soil macro-, meso-, and micro-fauna which contribute to the turnover and size of the various pools, especially those of nitrogen. Such organisms typically function within a narrow abiotic range, which implies that when the environment deviates from this optimum their functioning is impeded (Wardle 1992). Consequently, in the case of nitrogen cycling, the flow of the nutrient will be altered and this often leads to losses from the system.

Nitrogen has a global atmospheric reservoir pool size of approximately $3.9 \times 10^9$ Tg, while the soil in contrast only has a pool size of approximately 105 000 Tg (Paul and Clark 1989). Despite this large atmospheric reservoir only a small percentage of plants have symbiotic associations which aid the incorporation of N into plant tissues by means of Biological Nitrogen Fixation (BNF) (Charley and Richards 1986). Other kinds of biological nitrogen fixers, e.g. free living bacteria, similarly do not play a significant role in fixing nitrogen, i.e. converting atmospheric N to forms of N that are available for use by plants, e.g. $NH_4^+$ or $NO_3^-$. This emphasises the important role that the soil N pool has for maintaining the source of N for biological activity.

Charley and Richards (1986) emphasised that the soil-litter subsystem acts as a control valve through which pass virtually all the nutrients. The rate at which energy and nutrients flow through this gate governs the productivity of the whole ecosystem. Several processes are involved in this cycling and some include mineral acquisition or inputs, redistribution as well as transformation and relocation, and mineral or nutrient losses. In the case of N cycling in semi-arid savannas it is the return of N to the soil that is considered to be the rate limiting step (Scholes and Walker 1993). This return occurs via several processes and determines how productive the vegetation is.

Nitrogen is commonly returned to the soil in the form of litter although additional sources include the inputs from wet and dry deposition, BNF, and fertilizer inputs. Litter is comprised of several nutrients which vary according to the time of year that it is formed, the supply of the nutrients and supply of resources such as carbon dioxide, water and light. In cases where nutrients may be limiting, and resources like $CO_2$ in abundance, then the quality of the litter may become less suitable for decomposition, i.e. become more resistant to decomposition (Swift *et al* 1979; Chapin 1980; Chapin 1991).

Decomposition is the process by which litter is oxidized or broken down and its rate determines how quickly nutrients such as nitrogen are made available to plants and animals for growth (Horner *et al* 1988). A common index used to quantify how susceptible the litter is to microbial attack, and therefore decomposition, is the carbon (C) or lignin (L) to nitrogen (N) ratio, i.e. C:N or L:N (Swift *et al* 1979; Taylor *et al* 1989). When this ratio is greater than 25:1 then the litter will be more recalcitrant to decomposition because of the higher quantities of condensed tannins, structural carbon, and lignin remaining in the tissue (Horner *et al* 1988; Paul and Clark, 1989). When the ratio is less than 25:1 then the litter generally decomposes quicker since the microbial population can gain access to the nitrogen in sufficient quantities.

### 1.2.2 SOM dynamics

Soil organic matter (SOM) defines all carbon containing substances in the soil. It owes it's origin to the decomposition of above- and below-ground plant and animal substrates, and refers to the remaining litter and residues in the soil.

SOM is composed of various pools which are divided according to whether the organic material is labile or recalcitrant to microbial attack and decomposition. Several classifications exist, ranging from those which subdivide the SOM into various chemical and physical fractions (Mckean 1993) to those which use a biological classification based on turnover times (Parton *et al* 1987). The latter classification is seen to be more useful as it relies on understanding the input and output of organic material to the soil. It can thus be used to develop predictions of soil fertility and sustainability (Woomer 1993)

Soil organic matter turnover is a function of the decomposition rate. This regulates the rate of release of nutrients to the soil. The decomposition rate in turn is a function of the activity and size of the microbial population, which is affected by both abiotic and biotic factors in the ecosystem. As no two ecosystems posses the same driving variables and vary according to their geology, climate, and topography, the decomposition rate and SOM turnover for each fraction in the litter will therefore be different.

Decomposition includes the physical and chemical breakdown of organic matter. The physical processes include those of comminution, i.e. the breakdown of organic material by organisms themselves, and weathering, i.e. by the action of wind, water or abiotic damage. Chemical breakdown or the catabolism of organic material includes the oxidation of complex organic molecules to inorganic ions or gases. Typically this conversion results in the release of $CO_{2\,(g)}$ as a result of respiration losses.

Both of the above decomposition processes involve micro-organism and microbial activity. Any factor which constrains this activity and therefore the decomposition rate will in turn inhibit the release of nutrients. This can often lead to reduced ecosystem production and fertility (Odum 1971; Charley and Richards 1986; Scholes 1993).

### 1.2.3 Microbial activity

According to Sparling (1985) the soil microbial biomass, defined as the living microbial component of the soil, is the primary agent of the soil ecosystem responsible for litter decomposition, nutrient cycling and energy flow (Charley and Richards 1986). This it achieves by influencing both the transformation of organic matter and the storage of carbon and mineral nutrients. The microbial biomass thus serves as a source and sink in most terrestrial ecosystems (Vitousek and Matson 1984; Ladd *et al* 1985), and is increasingly becoming recognised as being closely linked to the mineralization potential of soils and the decay rate of leaf litter (Wardle 1992 ).

Mineralization refers to the conversion of an element from an organic to an inorganic form. Gross mineralization is the total amount converted, and net mineralization is the gross minus the growth demand or immobilization by the decomposer organisms (Paul and Clark 1989). This decomposition process catabolizes organic matter and commonly is brought on by a succession of organisms in the soil.

Immobilization refers to the locking away of nutrients into the microbial component in the soil. Although it reduces the availability of the nutrient during the growing season, it may still be beneficial for temperate and tropical regions during the autumn months when a flush of mineralization following organic matter disturbance releases the nitrogen. Immobilization of mineral nitrogen into the microbial biomass reduces the risk of leaching and allows a steady re-mineralization of the nitrogen from the microbial biomass for growth the following season (Wood 1989).

A commonly held view (Wood 1989; Paul and Clark 1989; Prasad *et al* 1993; Wild 1993) is that micro-organisms immobilize nutrients in sufficient quantities to maintain a particular balance in their tissues. For example bacteria require a C:N ratio of approximately 4:1 and are considered to be 'heavy users' of N, while fungi such as *Penicillium chrysogenum* have a C:N ratio of approximately 13:1 (Jenkinson and Ladd 1981). Substrates with a C:N ratio of approximately 25 to 30, or 16 in the case of savannas (Scholes and Walker 1993), normally have sufficient N to balance C assimilation, and no net release or immobilization of N will occur. Substrates with ratios greater than this will usually result in net immobilization, i.e. locking away of nitrogen free to move otherwise, and substrates with C:N ratios lower than this generally result in net mineralisation of N (Berg and Staaf 1981). This illustrates the well known Mineralization-

Immobilization Turnover or MIT model regulating the accumulation of nutrients such as $NH_4^+$ in the soil (Wood 1989).

Consequently any factors that increase the C:N ratio or alter microbial activity will in turn alter the availability of plant available N.

### 1.2.4 Factors influencing the cycling of N

Microbial activity is largely dependent on the way that the chemical composition of the litter and the environmental effects of temperature and moisture vary over space and time. In periods of high rainfall the mineralisation rate is increased because of the greater microbial populations in the soil. However mineralisation like nitrification is oxygen dependent and will be reduced in the case of flooding.

Controls on the mineralisation process include those of temperature, moisture, oxygen concentration (tension), substrate quality and availability, soil depth, soil pH and kinds of land use.

a) Temperature effects - Changes in soil temperature have marked effects on microbial activity as, like other organisms, they are governed by the laws of thermodynamics (Paul and Clark 1989). Temperature also influences the physicochemical characteristics of the environment including the soil volume, pressure, oxidation-reduction potentials, diffusion, Brownian movement, viscosity, surface tension, and water structure.

Microbiological activity generally increases exponentially from 5°C to 25°C while growth below 5°C is almost non-existent (Paul and Clark 1989). Beyond the maximum activity at 25 - 35°C their activity declines linearly as the temperature increases to 60°C (Wood 1989). Very few soils maintain a uniform temperature in their upper layers and variations can be either seasonal or diurnal.

Soil temperature depends upon the atmospheric temperature and inputs and losses from radiation. It is also influenced by the presence or absence of vegetation (Hofstede 1995), the soil water content and the depth within the soil. Temperatures in soils are often associated with dry conditions and the resulting effects on soil organisms and processes can be complex. In temperate grassland soils, microbial activity is limited by low temperatures during the winter, whereas the major limitations to soil organisms in tropical savanna soils during the same period is drought (Hofstede 1995). In tropical rain forest neither temperature nor soil moisture limit microbial activity throughout the year, leading to relatively rapid rates of organic matter turnover in these soils (Marrs *et al* 1988; Prasad *et al* 1994).

b) Moisture availability - along with changes in temperature will occur changes in soil moisture content. Soil water affects not only the moisture available for use by organisms but also influences the soil aeration status, the nature and amount of soluble materials, and the pH of the soil solution (Paul and Clark 1989). Water is also necessary if plants are to be able to make use of the available nutrients such as $NH_4^+$ and $NO_3^-$. This is because such nutrients can only be transported to the root surface as water is drawn there by bulk flow. This occurs when transpirational water losses set up a diffusion gradient between the soil and the root surface. As water moves to even out this water potential gradient so nutrients enter into the roots and root hairs.

Water is also important as a requirement of adequate soil moisture for microbial activity. According to Paul and Clark (1989) the water potential at which microbial activity is optimal occurs near -0.01 MPa, and decreases as the soil becomes either waterlogged near zero water potential or more arid at large, negative water potentials (*cf.* -8 MPa). Different groups of organisms do not all show the same responses and some may still be active at more negative water potentials (Kladivko and Keeney 1987; Atlas and Bartha 1993; Scholes 1993).

c) Substrate quality - The influence of substrate quality on microbial activity has been covered above. However it is worthwhile reiterating the point that the C:N and /or L:N ratios of the plant material greatly influences the dynamics of microbial activity. These ratios are typically a function of nutrient availability (Melillo *et al* 1982; Scholes 1993; Bloemhof and Berendse 1995).

d) Oxygen availability - Oxygen availability will strongly influence those micro-organisms that are aerobic decomposers. The highest levels of decomposer activity are found in aerobic environments (Swift *et al* 1979). Further more when soils become anaerobic nitrogen is lost from the system. This is produced by the anaerobic bacteria as a by product of the reduction of $NO_3^-$ to $N_2$ and other oxides of nitrogen, i.e. NO and $N_2O$.

e) Substrate availability - Micro-organisms require organic substrates for energy acquisition, cell maintenance and growth. These substrates enter the soil in the form of above ground litter, including shoot material and animal waste, or below ground litter, i.e. as root material and soil organic matter. Wood (1989) has shown that a bacterial population in a United Kingdom forest soil would require 1.3 kg organic matter $m^{-2}$ $yr.^{-1}$ for maintenance alone. Similarly, for a Canadian grassland soil the estimated maintenance requirement for a bacterial population of 55 g $m^{-2}$ was 19 270 g $m^{-2}$ $yr.^{-1}$ compared to an estimated input of substrate of 500 g $m^{-2}$ $yr.^{-1}$. Wood (1989) concluded that these calculations supported the view that growth of the soil bacterial population is severely limited by substrate supply. Similar conclusions exist for fungal populations. Consequently it can be inferred that system fertility declines with a decline in the microbial component (Wardle 1992; Prasad *et al* 1994; Srivastava and Lal 1994)

There is also considerable spatial variability in substrate supply in the soil. Litter and animal excreta are not uniformly distributed on the soil surface, and roots provide localized zones of high nutrient concentration. This will explain the wide spatial variation in the quantity of nitrogen found in a semi-arid landscape (McKean 1993; Scholes 1993).

f) Soil texture - This soil property varies according to the climate, topography, and parent material from which the soil particles are derived (Gerrard 1981; Scholes 1993). Soil consists of mineral particles of various sizes, shapes, and chemical characteristics. The formation of clay-organic matter complexes and the stabilization of clay, sand, and silt particles into aggregates are the dominant structural features of most soils (Tisdall and Oades 1982; Wood 1989). In terms of micro-organism function and the cycling of N in the soil, it is the clay fraction of soils that plays the most important role. This is because most clays, and micro-organisms and SOM have a net negative charge and therefore adhere cations. Soil aggregation is one of the most important factors controlling microbial activity and SOM turnover. Aggregates form when microflora and roots produce filaments and polysaccharides that combine with clays to form organic matter-mineral complexes (Tisdall and Oades 1982; Wood 1989).

The importance of soil texture on microbial activity is most clearly seen by how easily the organisms gain access to organic matter encased in soil aggregates. The pore sizes of aggregates may allow for the protection of organic matter if they prevent micro-organism access. Most organisms exist on the outside of aggregates and in the small pore spaces between them; relatively few reside within the aggregate. Pore neck space determine the accessibility to pores by organisms according to their body sizes. Occupancy is also affected by the water content of the [illegible] (Wood 1989).

g) pH - Acid soils, i.e. those with a pH below 7, commonly occur where soils are strongly leached. The intensity of acidity is also affected by the parent rock, the climate and the vegetation. Processes which act as a source of proton ions include the dissolution of carbon dioxide in the soil solution, nitrification of $NH_4^+$ and sulphur oxidation as a result of fertilizer ($(NH_4)_2SO_4$) addition, combustion of fossil fuels and acid rain (Wood 1989). As soils become more acidic, basic cations are displaced from exchange sites and leached down the soil profile. Exchangeable $H^+$ ions take their place on the clay minerals and organic matter. Clays with appreciable amounts of exchangeable $H^+$ release aluminium, magnesium and silica. After prolonged weathering the clay minerals are destroyed and mainly gibbsite, silicates and various iron oxide minerals remain (Trudgill 1977).

h) Salinity - Soluble salts (mainly sodium sulphate and chlorides of sodium and calcium) accumulate in the surface soils under hot dry conditions when the ground water comes within a few meters of the soils surface. Situations where this can occur include desert regions, following forest clearing and under the application of poor irrigation methods. If the salts are mainly sodium, then as they are washed out, sodium hydrogen carbonate is formed which causes the soil pH to increase to > 9. At such high pH nitrogen may be lost from the system as $NH_3$ is volatilized. Soil micro-organisms are inhibited by the osmotic effect cased by salt accumulation and this is equivalent to a condition of drought stress. Subsequently the mineralization of organic matter will be altered and litter could accumulate. The nitrogen would then be susceptible to losses following combustion of the litter (pyrodentrification), physical mineralization and volatilization.

The influence of each of the above factors on the availability of nitrogen can be understood in isolation but it is the interaction of these factors that determines how easily mineralisation is changed. An example would include the interaction between temperature and water on microbial activity.

Fluctuations in soil temperature and moisture may have a more profound effect on soil microbial processes and mineralization than constant or extreme conditions. For example, air drying soil kills some of the microbial population and renders soil organic matter more decomposable. The release of soluble organic carbon, nitrogen and phosphorous following air drying, leads to a subsequent increase in respiration and nitrification when the dry soil is re-wetted (Wood 1989). This is known as the Birch effect and illustrates interaction between temperature and moisture on microbial activity.

### 1.2.5 Land transformations

Soil micro-organisms are sensitive to extreme temperature and require moisture to survive. Thus at temperatures greater than about 50°C and below 5°C soil organism activity will decline (Runge 1983; Wood 1990) and the flux of mineral-N entering the soil will decline. Not only will land use changes result in deleterious conditions for the survival of micro-organisms, but the supply of substrate will diminish, the decomposition process decline over time and litter will accumulate on the surface. Such changes will consequently result in lower levels of mineral-N in the soil due to pyrodenitrification and leachate losses. Pyrodenitrification is a process where the combustion of organic materials leads to the loss of litter nutrients (Cook 1994). Reductions in surface cover also expose the system to losses in the form of runoff as the infiltration rate declines, but increase the temperature of soil leading to increased rates of mineralization.

The continued cycling of N in the soil relies on there being an almost constant supply of organic matter to the soil. In the event of there being any change in this balance between the inputs of N as organic matter and the outputs of N as plant available and gaseous nitrogen, the system will develop into one with lower soil fertility and sustainibility over time. Such changes are frequently manifested by land use changes and transformations (Vitousek 1994).

Land cover change is defined as alterations of the physical or biotic nature of a site, i.e. the conversion of forest to grassland, while land use change involves alteration of the way humans use land, as in the conversion of low input agricultural land to high input uses or visa versa (Meyer and Turner 1992). Here they are both used synonmously as they both represent changes in the landscape.

Vitousek (1994) list three well documented global changes as those including (1) increasing concentrations of $CO_2$ in the atmosphere, (2) alteration in the biogeochemistry of the global N cycle, and (3) ongoing land use and land cover changes. These he hypothesized were bound to climatic consequences which similarly would have direct effects on biota in all earth's terrestrial ecosystems.

Consequences of land use change invokes changes at many scales and incorporates changes in the physical, climatic, and biological sense. Apparent effects of land use change include increases in the concentration of atmospheric gases, i.e. $CO_2$, $N_2O$, $NO_x$, $N_2$ and CO; alterations of the climate both locally and regionally (Lean and Warrilow 1989; Shukla *et al* 1990; Dickenson 1991), and increases in albedo and local temperature

In addition to fixing N in the form of fertilizers, estimated to yield > 80 Tg year$^{-1}$, human activity also mobilizes N from long term storage pools through biomass burning, land clearing and conversion, and the drainage of wetlands (Vitousek 1994). Such transformation of an often limiting nutrient can be expected to affect biological systems at all levels of organisation and at spatial scales from local to global. At the atmospheric scale, anthropogenic fixation has occurred along with increases in the atmospheric concentrations of the stable greenhouse gas nitrous oxide (Prinn *et al* 1990). Such alterations were proposed by Vitousek (1994) to result in global sources and sinks that were 30% out of balance. He also suggested that the increases in fixation, mobilization or application of N would have positive effects on net primary production and biomass. However such additions of N will also aid the decline in many microbial species, i.e. ectomycorrhizal fungi in areas of Europe (Arnolds 1991). Additional changes as a result of alterations in the cycling of N include the effects on ecological processes like N mineralization (Couteaux *et al* 1995).

Stevenson (1965) noted that N accumulation in soils bore a close relationship to that of organic matter. Undisturbed grassland soils accumulate organic matter over many years with contents of, for example, 3.6% in a long term (30 year old) lowland pasture (Tyson *et al* 1990). Ryden (1984) notes that older pastures may contain large amounts of organic N, i.e. 5 - 15 t $ha^{-1}$ in the top 10 cm. Ayanaba *et al* (1976) similarly report that the observed decline in the soil reserves of C, N, and S under secondary lowland rain forest with cropping is usually less rapid when crop residues (maize) were returned to the soil as a mulch than when they were removed. The decomposition of litter is therefore critical to nutrient cycles.

Many authors similarly report a close association of N mineralization and the total soil organic N content of soils (Pastor *et al* 1987, Ayanaba *et al* 1976) with various factors altering the natural fluxes of nutrients in the soil (Hofstede 1995, Du Preez and Du Toit 1995).

Contrasting viewpoints are expressed in terms of the impact that grazing has on the availability of N in the soil. Shariff *et al* (1994) report that under heavy grazing, 77% of the annual aboveground growth is removed. Annual litter and root decomposition rates average 16% and over 59 $ug$ N $g^{-1}$ is immobilized into the soil. In contrast a moderately grazed site had annual litter and root decomposition rates exceeding 55% and a N mineralization rate of 60 - 269 $ug$ N $g^{-1}$ for the growing season. These authors concluded that the standard grazing rate of "take half leave half" may have a significant impact on N conservation and the supply of mineral N for plant growth (Woomer 1993; Shariff *et al* 1994).

Opposing this conservative view is the results of an investigation into the effects that grazing and burning had on soil and plant nutrient concentrations in Columbian paramo grasslands. Hofstede (1995) report that after long-term heavy grazing, soils had a higher bulk density and a lower moisture content. These study sites did however have greater decomposition rates. He concluded that the effects of burning and grazing on Paramo soils was principally restricted to changes in soil physical characteristics, and that differences in chemical characteristics of the soil do not cause differences in vegetation structure between grazed, burned, and undisturbed sites. These trends were hypothesized to exist because fires triggered a fast recycling of nutrients (Rundel 1981; Vogl 1974), grazers compressed the soil due to trampling (McNaughton *et al* 1988), greater temperatures at the soil surface meant faster rates of decomposition which all yielded a greater mineralization of nutrients (Chapin 1980; Milchunas *et al* 1988).

Higher nutrient availability would generally result in higher nutrient concentrations in plant tissues (Bryant *et al* 1983; Chapin 1980; Ruess 1984) assuming that nutrients made available are taken up by the plants. This in turn would lead to a positive feedback where the more favourable C:N ratios in plant materials would render the material more susceptible to decomposition.

Global climatic changes are also expected to result in similar changes in the way that nitrogen and other nutrients flux in the environment (Parton *et al* 1987). The increase in global $CO_2$ from 205 parts per million in 1850 to 340 ppm today is expected to continue to 600 ppm by the year 2050 (Meyers 1985). Although exact estimates are difficult to make, increasing $CO_2$ is likely to result in more extreme climatic events, a warming of the atmosphere and greater C:N ratios in the vegetation. Such changes will ellicit changes in the mineralization and nitrification rate owing to their influence on microbial activity and population size. However increasing temperatures results in faster turnover times.

Vitousek (1992) and many others, recognise land use change to be the single most important of the many interacting components of global change, that effects ecological systems. However it is also realised that no single *bona fide* quantitative measurement for this change is in use today due to the summation of many local changes in many local areas prevents such a measure (Vitousek 1994).

### 1.2.6 Brief overview of N mineralization in the biomes of South Africa

Recent (*cf.* 1990's) estimates for rates of Nmin exist for several sites in South Africa (Prinsloo et al 1990; Ellery 1992; Carter G.A. 1993; McKean 1993; Scholes and Walker 1993; Du Preez and Du Toit 1995).

Du Preez and Du Toit (1995) quantified Nmin in the grasslands of South Africa by investigating 5 agro-ecosystems along an east-west gradient. At each site a cultivated soil and its virgin counterpart were sampled to a depth of 200 mm and total nitrogen and mineralizable N were determined as indices of N fertility. They concluded that cultivation, irrespective of the period, caused a significant decrease in the N fertility of all five agro-ecosystems. This was attributed to the intensive cropping practises which severely depleted N fertility despite N fertilization. The quantity of mineralizable N for the virgin soils increased almost linearly from 11.8 mg $kg^{-1}$ in the dry west to 70.3 mg $kg^{-1}$ in the moist east according to the fine silt-plus-clay content, and exponentially with increasing aridity. Similarly the quantity of mineralizable N increased from 6.8 mg $kg^{-1}$ to 15.7 mg $kg^{-1}$ for the cultivated soils across the west-east gradient. The pattern for loss of mineralizable N was very rapid during the first few years of cultivation and followed the loss of total N. This is because the latter serves as a reservoir for the former (Du Preez and Du Toit 1995).

Other estimates for the rate of Nmin in the grasslands of South Africa include the study by Ellery (1992). He showed that Nmin ranged from 1.6 mg N $kg^{-1}$ $d^{-1}$ in the sweetveld grasslands of the west to 2.7 mg N $kg^{-1}$ $d^{-1}$ for an intermediate and 2.2 mg N $kg^{-1}$ $d^{-1}$ for a sour grassveld.

In a similar study of the savanna biome, Carter, G.A. (1993) showed that Nmin ranged from 2.2 mg $kg^{-1}$ $d^{-1}$ at Nylsvley to 5.5 mg $kg^{-1}$ $d^{-1}$ in the east. A site which was more arid, Messina in the far north, had an Nmin rate of 2.4 mg $kg^{-1}$ $d^{-1}$ while a site at the southern extremity of the savanna biome (Rooipoort game farm) had a Nmin of 2.9 mg $kg^{-1}$ $d^{-1}$.

McKean (1993) also reported Nmin rates of the same order of magnitude for the Nylsvley study site although it was apparent that Nmin was greater under the canopy of trees than in the open.

Table 1.1 The rate of nitrogen mineralization in biomes of South Africa. Conversions are calculated using a bulk density of 1.6 g/cm$^3$ to a depth of 0.2 m.

| **Biome and sites** | **Nmin rate** **mg N kg$^{-1}$ d$^{-1}$ ~ g N m$^{-2}$ yr$^{-1}$** | **Incubation** | **Source** |
|---|---|---|---|
| Savanna biome | | | |
| Rooipoort | 2.93 ~ 342 | Anaerobic | Carter (1993) |
| Messina | 2.36 ~ 275 | Anaerobic | Carter (1993) |
| Nylsvley | 2.22 ~ 259 | Anaerobic | Carter (1993) |
| Nylsvley | 2.18 ~ 255 | Anaerobic | McKean (1993) |
| Nylsvley | 0.1 to 1.3 ~ 12 - 775 | Anaerobic | Scholes & Walker (1993) |
| Mkuze | 5.54 ~ 647 | Anaerobic | Carter (1993) |
| Timbavati | 4.35 ~ 508 | Anaerobic | Carter (1993) |
| | | | |
| Grassland biome | | | |
| Uncultivated | 40.15 to 81.4 (g m$^{-2}$ yr$^{-1}$) | Leachate | Prinsloo *et al* (1990) |
| Reverted | 27.74 to 91.6 (g m$^{-2}$ yr$^{-1}$) | Leachate | Prinsloo *et al* (1990) |
| Dry to moist | 0.076 to 0.456 ~ 9 - 53 | Leachate | Du Preez and Du Toit (1995) |
| Sweet to sour | 1.63 to 2.18 ~ 190 - 255 | Anaerobic | Ellery (1992) |
| | | | |
| Fynbos biome | | | |
| Coastal | 0.260 to 0.438 ~ 30 - 51 | Aerobic | Stock *et al* (1988) |

Results such as those given above illustrate that Nmin does vary between natural sites in South Africa and this variation is typically a function of soil texture (Du Preez and Du Toit 1995) and aridity (Carter, G.A. 1993; Du Preez and Du Toit 1995). However the rates tend to fluctuate within an order of magnitude. Stock *et al* (1988) estimated Nmin in the coastal fynbos along a succession with values for Nmin ranging from 0.260, 0.310, and 0.438 mg N kg$^{-1}$ d$^{-1}$ for sites which were 1 year, 6 year and 20 years old.

### 1.2.7 N availability indices

According to Bundy and Meisinger (1994) N availability indices are calculated using chemical or biological tests to measure or predict the amounts of available N released from a specific soil under a specific set of test conditions. Predictive estimates of N availability in soils are necessary owing to the pressures of economic and environmental incentives to use available N more efficiently in crop production and minimise the losses of N from cropland to the environment.

### 1.2.8 A brief review of some of the methods to determine available N.

Chemical methods refer to the simple and rapid approach to estimating N needs and provide a convenient assessment of relative differences among various experimental treatments. New methods include determining the quantity of $NH_4^+$- N released from organic matter on heating soil samples with 2 M KCl; measurement of $NH_4^+$- N released by steam distillation of soil samples with pH 11.2 phosphate-borate buffer, and UV absorbance of $NaHCO_3$ soil extracts at 200 *n*m (Bundy and Meisinger 1994).

Undisturbed soil core incubation refers to isolating a core of soil *in situ* in a stainless steel cylinder and measuring the change in $NH_4^+$- N and $NO_3^-$- N concentration over the incubation period. This method is seen as being a more realistic method for assessing N availability than the disturbed methods (see later). However discrepancies exist as standardised conditions can not be maintained owing to fluctuating soil water content, i.e. increasing water content in the cylinders because of condensation (Scholes and Walker 1993). More importantly is that plants would not get access to roots and changes in the bulk density of the soil would follow compaction. These would lead to unrealistic estimates for the rate of Nmin.

The above aerobic technique is also seen as being impractical for field scale predictions as a large number of soil cores would be needed to accurately assess Nmin. It is also a very laborious technique. The method would however be appropriate for investigations into tillage studies, pastures, forests and fallow soils where the variation in the degree of soil disturbance is an integral component of the systems under evaluation (Bundy and Meisinger 1994).

There are also a number of laboratory techniques which have been developed to quantify the rate of Nmin. These include those which are biological as well as those which are chemical in nature. The biological methods usually involve incubation of soil under conditions that promote Nmin from organic sources and the measurements of inorganic N produced. Chemical techniques follow those described above, and there selection depends largely on the objectives of the study (Bundy and Meisinger 1994). The biological methods are based on the assumption that the same biological processes that cause release of plant available N are also responsible for production of inorganic N in the laboratory procedures. An important caveat of using the biological N availability indices is that they must be viewed as relative indications of soil N availability.

Two laboratory techniques commonly used to measure N mineralization include the aerobic or waterlogged (anaerobic) incubation. The waterlogged technique was first proposed by Waring and Bremner (1964) and was recommended by Keeney (1982) due their being several advantages to the technique. It is simple, easily adapted to a laboratory routine, has a short incubation period of only 7 days, requires little or no sample pre-treatment, eliminates concerns related to optimum water content and water loss during incubation, and makes use of minimal apparatus and reagents (Keeney 1982). The long-term aerobic laboratory incubation technique first came to the fore in 1972 when Stanford and Smith used it to estimate the Nmin rates for various soils. This method involves measurement of inorganic N produced during aerobic incubation of soil or soil amended with sand or vermiculite under near optimum conditions of temperature, moisture and oxygen supply for up to 30 weeks. Inorganic N is usually removed by periodic leaching of soil samples incubated in a combination filtration-incubation container. Although these authors used a lengthy incubation period, shorter time periods (< 8 weeks) prove critical for understanding the N availability status of soils and adequately describes the Nmin in the active fraction of soil organic N. The N that is released later in the incubations explains the release from the more stable soil organic matter pool and is frequently used to model soil Nmin and characterise various components of the labile N pool in soils. Aerobic methods do however have a coefficient of variation between 20 and 60% (Bundy and Meisinger 1994).

Summaries of early research (Harmsen and Van Schreven 1955; Allison 1965; Bremner 1965) reached the general conclusion that (1) chemical extraction methods were likely to be unsuccessful because they could not simulate the action of soil micro-organisms (but see Keeney and Bremner 1966), (2) biological incubations under standardised conditions were most successful because they used the same microbial agents under field conditions, and (3) methods based on soil $NO_3^-$- N levels were of very limited value because the soil $NO_3^-$- N pool was too transient to be useful.

Stanford (1982) and Keeney (1982) similarly concluded that long-term biological mineralizations were most suitable but were not practical, short-term mineralizations were acceptable but were affected by sample handling and pre-treatment (Keeney and Bremner 1966) with $NH_4^+$- N production after 7 days of anaerobic incubation being the recommended procedure by Keeney (1982).

In 1984 Meisinger suggested that N availability assessments should involve use of both residual $NO_3^-$- N and Nmin tests, expanded use of local soil conditions through soil taxonomic classification, and an integration of this information for each specific site by use of computer models and local weather data.

### 1.2.9 The utility of a simulation model

Modelling provides a convenient tool to use in assessing the susceptibility of processes to change. A simple way of investigating the importance that any particular change is likely to have can be described by means of a simulation or sensitivity analysis. This is the procedure where individual parameters in the model are systematically altered and the output from the model compared with previous outputs. Depending on what these results indicate, the change can be classified into sensitivity classes, i.e. those parameters which produce the greatest deviation in output will be most sensitive to change whilst those which produce similar results are more robust.

Currently several models have been developed to model the soil organic matter dynamics of soil (Parton *et al* 1987). Jenny (1941) used a single-state variable model to assess the decline of soil organic C and N in cultivated soils. Campbell *et al* (1978) divided soil organic matter into two different compartments including the stable organic matter and labile organic matter with quick (53 years) and slow (1429 years) turnover rates respectively. Paul and Van Veen (1978) and Van Veen and Paul (1981) further divided the plant residues into recalcitrant and decomposable fractions and introduced the concept of physically protected soil organic matter. The latter class was assumed to have a much lower decomposition rate than non-physically protected soil matter. Parton *et al* (1987) developed a model having multiple SOM compartments based on turnover times. Decomposition rates varied as a function of monthly soil temperature and precipitation, and included both C and N flows.

Parton *et al* (1987) developed a model to simulate the steady-state organic matter levels for 24 grassland locations in the Great Plains, United States of America. The CENTURY model simulates both the labile (rapid turnover) and stabilised (slow turnover) fractions, and thus simulates the nutrient-supply capacity of the soil organic matter. The choice of focusing on soil organic matter was given that organic matter is central to the cycling of plant nutrients, influences water relations and erosion potential, and is a key factor in soil structure (Tisdale and Oades 1982). The model comprises three submodels namely the SOIL and DECOMPOSITION (SOM) submodel, the PLANT submodel and the NITROGEN submodel. The soil and decomposition or SOM submodel investigates the turnover of carbon in the soil. The plant submodel simulates plant production as the monthly dynamics of C and N in the live and dead aboveground plant material, live roots, and structural and metabolic surface and soil residue pools. The nitrogen model (Figure 1) has the same structure as the carbon model, and they assumed that most N was bonded to C.

Several models have also been developed to estimate the contribution of nitrogen mineralization to the nitrogen supply of plants. Matus and Rodriguez (1994) based their estimates on the decomposition of the active SOM pool. This pool is comprised of stabilised and labile soil organic N where the stabilised N is built up from accumulated inputs of fresh organic N during plant rotation, and the labile N is a fraction of total N added, which mineralizes faster than the stabilised N. Bloemhof and Berendse (1995) described a dynamic simulation model of carbon losses due to microbial respiration, mineralization and immobilization of N from above-ground and dead plant materials. They showed that the model was adequate in predicting the quantity of nitrogen and carbon in all litter types. Similarly Paustian *et al* (1992), using a CENTURY model, showed that the treatment differences in SOM could be explained by the rate of organic matter input, its lignin content, and the C:N ratio. However, there appeared to be additional positive effects of N supply on SOM accumulation that were not fully explained by the model. These included the quality of organic amendments through the controls on N mineralization and immobilization.

Models allow one to postulate what impact future changes in the environment are likely to hold. Processes such as N mineralizatiom and nitrification can be adequately modelled using only a small set of driving variables. These variables also often change between and within sites which can be used to describe regional trends. Thus on the basis of such trends one is able to predict how the system function is likely to change.

Parton and Rasmussen (1994) further stress the increasing need to develop models to assess the long-term effects of management practices on soil and environment quality, and to test these models across a wide range of environments.

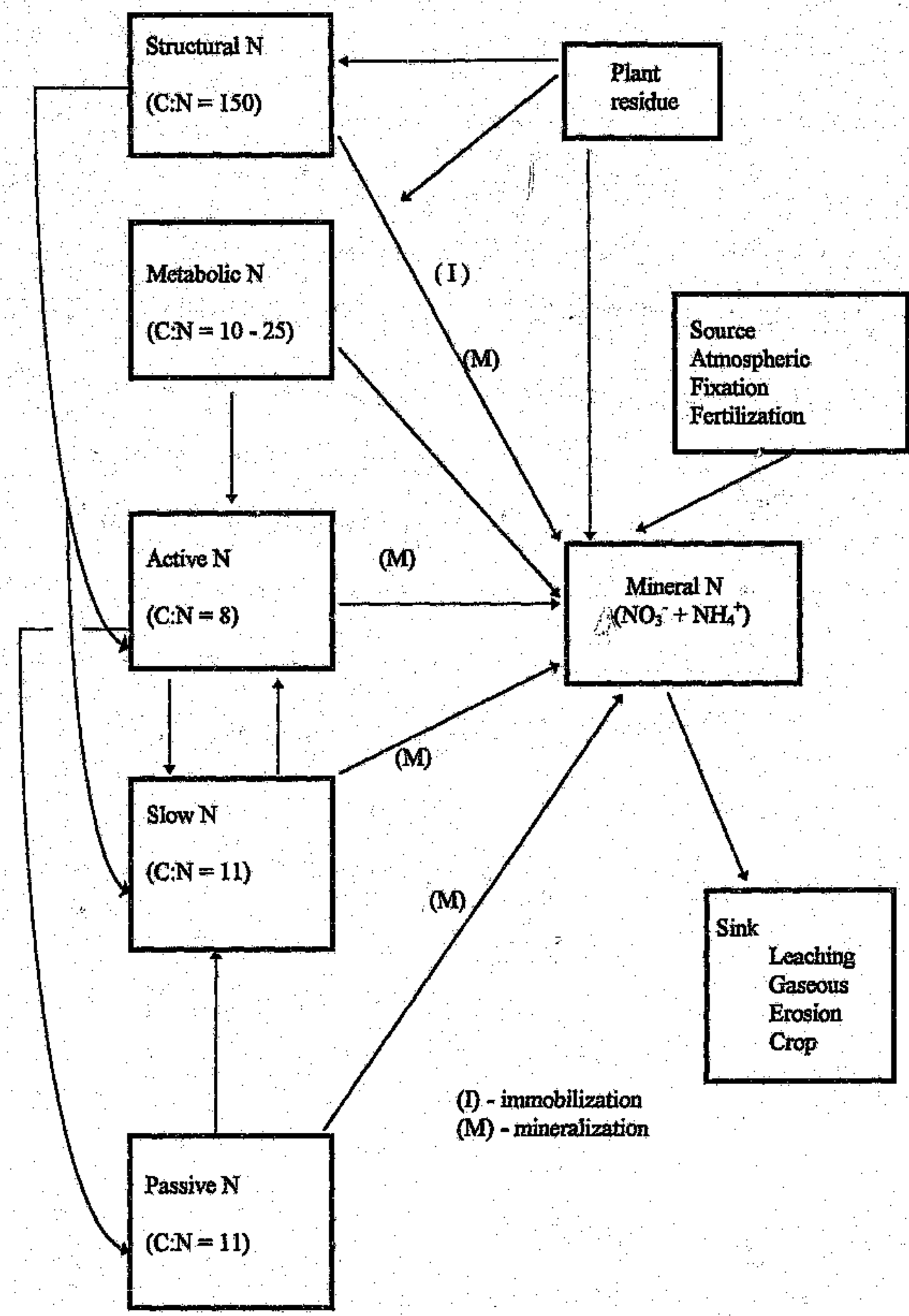


Figure 1.1 A flow diagram for the N submodel of the CENTURY model (after Parton *et al* 1987).

## 1.3 MOTIVATION

A current focus in South African plant and soil science revolves around understanding the biogeochemistry of managed and natural ecosystems (Du Preez and Du Toit 1995; Scholes, M.C. 1996 unpublished manuscript). This involves understanding the impacts of land use, climate and atmospheric change so that the availability and supply of nutrients can be ascertained. Biomes such as the grassland and savanna biomes include areas with different land uses in which the amount of soil organic matter could be expected to be reduced in comparison with natural areas. Such changes have commonly been reported to have slower rates of Nmin owing to the correlation between Nmin and organic matter content.

Three objectives have been set for a research plan into investigating the biogeochemistry of South African ecosystems (Scholes, M.C. 1996 unpublished manuscript). The first aims to better quantify the relative importance of and environmental controls on the processes driving the cycling of carbon and nitrogen in semi-natural ecosystems in South Africa as a basis for predicting primary and secondary production. The second seeks to better understand the linkage between carbon and nitrogen biogeochemistry across a range of soil and vegetation types, while the third looks at incorporating this knowledge in predictive simulation models to extrapolate the processes through time and space. Such research will be able to answer questions regarding how N fluxes in the environment with an important caveat being that productivity can be maintained. This is especially important in light of the threats that anthropogenic influences such as land use change, land transformation, and pollution, etc. have on the ecosystem.

## 1.4 AIM

To quantify the relative importance of processes driving carbon and nitrogen dynamics and how flux rates of these processes vary across vegetation types and land use.

## 1.5 OBJECTIVES

1) - To quantify the amounts of soil organic matter (SOM) for each site and land use type.
2) - To quantify the rate of potential N mineralization in samples from different sites and land uses.
3) - To quantify the proportion of N immobilized in the microbial pool for different sites and land uses.
4) - To test whether Total N, Total C, microbial N and C, potential N mineralization and light fraction N and C in areas of South Africa can be predicted by the CENTURY model.
5) - To improve understanding on how the biological and physiochemical factors of the soil system interact to determine the turnover of the carbon and nitrogen pools in each site.

## 1.6 KEY QUESTIONS:

1) What are the sizes of the soil organic matter pool fractions in each of the sampled land use areas and sites?
2) Does the rate of potential N mineralization differ within and between land use types and between sites?
3) What quantity of plant available N is immobilized into the microbial biomass?
4) How well does the CENTURY model predict the mineralization of soil organic nitrogen in each land use type and site?
5) Do N mineralization, biological immobilization and the physiochemical factors including texture, pH, and moisture content of the soil system adequately explain the turnover of soil carbon and nitrogen in these sites and land uses?

The analysis and interpretation of the samples will be conducted in two stages. In the first stage sample site characteristics and sample analysis for soil organic matter fractions and N mineralization will be quantified (laboratory stage). The second stage involves fitting the collected data to the CENTURY model, and following this, relevant data obtained for predictions regarding soil N turnover (modelling stage).

# Chapter 2

## Soil Organic Matter fractionation and quantification

### 2.1 Introduction

The cycling of carbon and nitrogen in an ecosystem is postulated to be governed by the turnover of soil organic matter. According to Anderson and Ingram (1989) soil organic matter can be delimited into two main fractions. The light fraction comprises the soil which consists of microbial biomass and partially humified/cellular organic matter with a short turnover time of 1 to 5 years. The heavy fraction comprises humified soil comprising physically protected and/or chemical forms of organic matter that are resistant to decomposition with turnover times from 20 to 40 years (slow soil) or up to 200 to 1500 years (passive soil). For practical purposes the heavy fraction is defined as the organic C and N pool in soil samples after the removal of the light fraction (Anderson and Ingram 1989).

Parton *et al* (1987) in formulating the CENTURY model, subdivided the continuum of SOM in the soil into three main pools based on the turnover times of each. These include the active pool, the slow pool and the passive pool. The active pool includes mainly the microbial biomass and has a turnover time of 1 - 3 years. The slow pool, of which the mass of light fraction is a primary constituent, has a turnover time of decades ranging 25 - 75 years. This pool also has an intermediate fraction that is recalcitrant but not normally resistant to decomposition, forming particles smaller than 250 *u*m in diameter. This pool is nevertheless included in the slow pool because it turns over more rapidly than the passive pool. The passive pool, which includes humified material adsorbed onto clay particles, has a turnover time of 100's to 1000's of years. The definitions for the various SOM fractions as outlined by Parton *et al* (1987) are used here.

Any sufficiently large disruption to the turnover of these SOM fractions will inevitably lead to situations of unsustainable use. This is because the SOM fractions determine the nutrient capacity of the soil (Parton *et al* 1987). Soil organic matter in turn is often reduced by land use changes like the conversion of forest to grassland, grassland to pasture and the effects of urbanization and over-population (Vitousek 1994; Woomer 1995).

This chapter aims to quantify the amount of soil carbon and nitrogen in the soil organic matter fractions of selected sites in South Africa.. These quantifications will provide a better understanding of the factors influencing the carbon and nitrogen dynamics of different land uses along an environmental gradient in South Africa.

## 2.2 Material and Methods

### 2.2.1 Sample sites

Study sites were chosen on the basis of the criteria given below and these provide a broad range in geomorphology, climate and vegetation. This follows a similar selection procedure as that used by Ellery (1992), Carter, G.A (1993), and Du Preez and Du Toit (1995) who have also quantified soil nutrient dynamics as part of larger projects in South Africa.

a) Soil type
b) Geographic division of climate
c) Kind of vegetation
d) Land use type
e) Rainfall
f) Geomorphology

Within each biome, study sites along a rainfall and broad soil type gradient extending from the dry west to the moist east were sampled (Table 2.1). These sites are thus useful for investigating which environmental processes were important in regulating the amount of carbon and nitrogen in the soil. At each study site, samples were collected in each of a maximum of three land use types namely a conserved or relatively undisturbed area (Con), a cultivated area (Cult) and an area used for livestock production (Lvstk). Each land use type was not necessarily on the same soil texture or mineralogy owing to the suitability of the site for that particular land use. Consequently statistical differences for the relevant nutrient amounts and rates per land use are confounded by differing soil textures, and differences per soil texture are confounded with different climates. However this is unavoidable. The experimental design does allow one to postulate which processes are important in determining nutrient levels and generating hypotheses based on the trends in the data (Carter, M.R. 1993). This was the aim of this chapter.

Table 2.1 The broad climatic, vegetation and soil type characteristics for each of the 11 study sites investigated.

| Biome | Site | Vegetation type[1] | Soil form[2] |
|---|---|---|---|
| Savanna - | | | |
| | Towoomba Hutton | Mixed bushveld savanna | Hutton sands |
| | Towoomba clay[4] | Mixed bushveld savanna | Arcadia (smectitic clays) |
| | Nylsvley | Mixed bushveld savanna | Hutton and Felsite sands, Arcadia (smectitic clays) |
| | Messina | Mopani savanna | Red-yellow apedal, freely drained, with high base status |
| | Klaserie[4]<br>Klaserie Communal[3] | Lowveld savanna | Regic sands |
| | Mkuze | Lowveld savanna | Arcadia (smectitic clays) |
| Grassland - | | | |
| | Bloemhof | Dry *Cymbopogon-Themeda* grassland | Avalon / Clovelly |
| | Bethlehem | *Cymbopogon-Themeda* grassland | Avalon / Clovelly |
| | Bethal | *Themeda* grassveld | Avalon |
| | Vryheid | Northern Tall grassveld | Avalon |

[1] After Low and Rebelo (1996)
[2] After Macvicar *et al* (1977) and Land Type Series maps, Soil and Irrigation Research Institute, Pretoria (1979)
[3] Communal grazing lands of the Okkenhoutboom village near Acornhoek, Mpumalanga.
[4] No conserved sample for the Klaserie site, No livestock area for the Towoomba Clay.

### 2.2.2 Sample collection

A composite soil sample (500 g) of 10 or more (Bundy and Meisinger 1994) bulked random samples to a depth of 15 cm were taken from each of three replicates in each of the land use types in each site using a soil corer (52 mm diameter). Samples were placed into plastic bags, loosely knotted, and stored cool in cooler boxes for as short a time period as possible before analysis. This was done so that the samples would not dry out. This was necessary so that the microbial population was altered as little as possible.

However storage of samples at 4°C in cooler boxes in the field proved difficult and the subsequent effects of keeping samples moist under favourable oxygen and temperature conditions could have initiated growth and therefore an overestimation of the microbial biomass. Nevertheless, the same sampling methodology was applied for all the samples and so the relative differences between samples would still be valid.
Replicates were from areas representative of the land uses in the area. All replicates are assumed to come from a homogeneous soil type within a land use.

### 2.2.3 Site measurements

Soil variables per land use at each site that were recorded included soil texture, bulk density and soil pH.

*Soil texture* - The percentages of sand, silt and clay in a combined soil sample of three replicates of the land use at a site were analysed by the hydrometer method (Anderson and Ingram 1993) at the Institute for Soil Climate and Water (Pretoria).

*pH* - A Crison micropH 2001 pH meter (Laboratory and Scientific Company (Pty) Ltd) was used to measure the pH of 20 g soil mixed in 50 ml distilled water for 10 minutes and allowed to stand for 30 minutes (Okalebo *et al* 1993).

*Bulk density* - Two methods were used to estimate the bulk density at the various sites and land uses depending on whether the soil was sandy or stony. For sandy soils a core (105 mm by 70 mm diameter) was used to extract a known volume of soil. This was dried at 60°C in a convection oven until constant weight and the bulk density calculated following equation 1 (Okalebo *et al* 1993). In cases where the soil was stony, an infill method was used to calculate soil bulk density (Anderson and Ingram 1993). Soil was excavated from a hole of 10 cm x 10 cm x 10 cm, lined with a plastic bag and filled with water. This was used to calculate the volume of the hole (V). The stones and soil which were removed from the hole were oven dried at 60°C until constant weight (W) and used to calculate the bulk density using equation (1).

$$\text{Bulk density } (g/cm^3) = W / V \qquad (1)$$

*Soil moisture content* - The gravimetric change in weight of a sieved (2 mm sieve) soil sample after drying in a convection oven set at 60°C provides an estimate for the soil moisture content (Anderson and Ingram 1993). This was used to express quantities of nutrients on a soil dry mass basis.

*Aridity index* - An aridity index was used to differentiate sites along a climatic gradient and was calculated using the ratio of mean annual precipitation over the last 20 years to mean annual potential evaporation (South African Weather Bureau).

### 2.2.4 Sample analyses

Samples were analysed as soon as possible after they had been collected. In cases where this was not possible, samples were stored at 4°C in a refrigerated store room. Analysis of the soil samples, which comprise replicates for the land uses at each site, included determining several chemical and physical attributes. Owing to time constraints, only two replicates were collected for the Towoomba cultivated site, otherwise three replicates were collected for each land use.

***Soil organic matter fractionation***

The size of the soil organic matter fractions for the microbial biomass N and C pools (Vance *et al* 1987), the mass of the light fraction (Anderson and Ingram 1989), amounts of light fraction N and light fraction C, total N and C (Anderson and Ingram 1989), and by difference, the size of the passive N and C pools, were estimated for all of the soil samples.

The size of the three soil organic matter fractions were determined for each sample as follows.

*- The active or microbial biomass pool*

Quantification of the microbial pool follows the method proposed by Vance *et al* (1987). This simple method for determining the size of the microbial population is the chloroform-fumigation extraction technique where composite field samples were returned to the laboratory and sieved using a 2mm sieve. Aliquots (2 ), each of about 50 g, were weighed into polytops. Distilled water (2.5 ml) was added to the soil to bring it to 60% field capacity, and these were incubated for 5 days in the dark at room temperature (± 25°C). This was used to prime the microbial population in the soil samples. After the priming period had elapsed, one half of the subsamples was fumigated by placing the polytops of moist soil in a large glass vacuum dessicator with a beaker of ± 50 ml of alcohol free chloroform. The dessicator was evacuated until the chloroform boiled, and placed into the dark for 5 days at 25°C (Anderson and Ingram 1989). The other half of the subsamples were not fumigated. Samples were extracted with 0.5 M $K_2SO_4$ at a ratio of 5:1. Samples were centrifuged at 5000 revolutions per minute for 5 minutes, and the supernatant analysed for ammonium N by the colorimetric method (Anderson and Ingram 1993).

Although the technique does not directly measure microbial activity, as no differentiation between quiescent and active organisms is made (Lake 1995), it can be inferred that a greater microbial biomass equates with greater nutrient turnover (Wardle 1992).

Microbial biomass N was estimated by calculating the difference between a $T_0$ assay and an assay which was incubated in 100% chloroform for five days (Vance *et al* 1987). The chloroform causes the microbial membranes to lyse which should result in an increase in the amount of nutrients in the soil. Nitrogen released by the fumigation process was measured by the colorimetric determination of ammonium (Anderson and Ingram 1993).

The amount of carbon in the microbial biomass was measured in a similar method described above, but the extracted carbon was determined using the complete 'wet' oxidation method by acidified potassium dichromate (Anderson and Ingram 1993). This method requires heating the sample at 150°C for 30 minutes, following which barium chloride is added and the supernatant solution is analysed by colorimetry at 600 *n*m.

*- The slow or light fraction pool*

The light fraction is defined as that material which, when dispersed in water, floats, and which passes a 2 mm, but not a 0.25 mm, sieve. A simple method which was used for collecting the light fraction included dispersing 50 g of sieved soil, through a 2 mm sieve, and then decanting the floating residue into a 0.25 mm sieve (Anderson and Ingram 1989). Material was dried and weighed over a three day period in a convection oven set at 60°C. Following combustion in a furnace at 550°C for 6 hours the percentage of fine sand (less than 2 mm but greater than 0.25 mm) was calculated, and used to correct the light fraction mass used in the chemical determinations of light fraction N and C (see below).

Two corrections were required to accurately estimate light fraction C and N in soil samples from each land use and biome. The first correction was used to compensate for the fine sand that remained in the light fraction after sieving. 59% of the sieved soil sample from the land uses of each biome was light fraction. The second correction included using biome and land use specific correction factors (i.e. % N and %C) to estimate the amount of N and C in the light fraction (Table 2.2)

Table 2.2 Correction factors that were used to estimate the amount of N and C in the light fraction.

| Land use | % N | | % C | |
|---|---|---|---|---|
| | **Savanna** | **Grassland** | **Savanna** | **Grassland** |
| Conserved | 11.65 | 23.96 | 22.57 | 39.82 |
| Cultivated | 17.18 | 18.37 | 48.73 | 54.11 |
| Livestock | 29.48 | | 43.00 | |

Corrections were derived by measuring the percentage N and C (see methods for total N and C) in subsamples that possessed a light fraction mass greater than 0.4 g. Following this the un-ashed light fraction + fine sand component was multiplied by 59% to get the mass of light fraction and then an estimate of the amount of N in the light fraction was obtained by multiplying by the correction factors (Table 2.2)

Similar methodology applies for obtaining an estimate of light fraction C.

*- The intermediate pool*

To calculate an estimate for the intermediate N and C pool sizes, it was necessary to determine the total N and C for the soil samples. Total N was determined by Kjeldahl digestion followed by steam distillation (Institute for Soil Climate and Water). Total C was quantified with a 'wet' complete oxidation procedure (Anderson and Ingram 1993) in which a 1 ± 0.001 g soil sample was digested at 135°C for 1 hour before the addition of 50 ml of 0.4% barium chloride. Samples were left to stand overnight before they were read at 600 *nm* using a spectrophotometer.

Intermediate N and C was calculated as the difference between the total N and C and the N and C that is in the microbial biomass and light fraction pools, respectively. Thus, subtracting the total pool size from the sum of the microbial and light fraction pools, gives the pool size that is complexed onto soil particles, i.e. clay.

### 2.2.5 Data analysis

***Parametric statistical methods***

Histograms of all the variable means were used to determine if the data was normally distributed. An examination of the residual plots following an analysis of variance test or after regression analysis, was used to determine if transformations were necessary. A log transformation was applied to the data if a fan-shaped pattern, indicating heteroscedasity of the variance, was evident when plotting the predicted and the residual. Transformed variables included, light fraction mass, microbial N, microbial C, Light fraction N, and Light fraction C. Nmin and other variables were not transformed, and transformations were not conducted on the regression analysis because an examination of the residual plots indicated the existence of no clear patterns.

The means for each land use at each site (n = 11) of the SOM variables including microbial biomass N and C, light fraction mass, light fraction N and C have been compared using a one way analysis of variance for biomes, sites and land uses (SYSTAT). If the ANOVA was shown to be significant, differences were detected using a Student

Newman Keuls multiple comparison test testing at the 95% level (SAS 1986). No interaction term could be investigated in any of the ANOVA's as the design was unbalanced and had missing cells. This was because certain land uses could not be sampled at two of the sites investigated. These were the livestock on clay at Towoomba, and the conserved on regic sands at Klaserie (see Table 2.1).

To investigate the impact that land use had on the quantity of the SOM variables at each biomes, site and land use, an analysis of covariance was used with the variables aridity index and percentage fines in the soil of each site and land use as covariates. A Tukey post hoc multiple comparison test was used to detect differences if the analysis was significant at the 5% level.

Similar methods as described above were used for comparing Nmin in the biomes, sites and land uses. Linear regression analysis was used to determine whether any trends existed in the distribution of Nmin in the various sites and land uses under investigation. Similarly Nmin was regressed with total N, light fraction mass, light fraction N, aridity index, and percentage fines to determine if any association existed with these variables. Relations per land use have also been compared and the slopes of the regression lines statistically compared (Sokal and Rolf 1994).

Chi-square tests of difference, used to indicate significant associations between the proportion of N and C in the various soil and SOM fractions of each biome and land use, have been calculated using EPISTAT (Gustafson 1986) at the 95% level.

Pearson correlation analysis and bonferroni adjusted probabilities were used to compare the correlations among all the variables.

***- Multivariate analyses***

The CANOCO version 3.12 (ter Braak 1988 and 1990) canonical ordination program was used to describe the location of sites according to the influence of 'species' and environmental variables. A Detrended Correspondence Analysis, used to aid selection of a linear or unimodal model, showed that the gradient length was less than 1.5 standard deviations. Therefore a linear indirect, i.e. RDA, method was used to describe which variables were most important in explaining the variation in the location of sites and land uses. Gauch (1982) states that the assumptions of PCA, i.e. that they have normal distributions and be uncorrelated, can rarely, if ever, be valid when using field data sets. However deviations from these requirements are tolerable when the technique is used for descriptive purposes (Gauch 1982). Furthermore the test of significance does not depend on parametric distribution assumptions, therefore there is no need to concern oneself with transforming data to conform to a normal distribution. Therefore the untransformed data were entered into the program and no transformations were performed.

A multivariate direct gradient analysis, i.e. Redundancy Analysis or RDA, was used to explain the species responses by ordination axes that are constrained to be linear combinations of supplied environmental variables (ter Braak 1988). The ordination diagram obtained from a direct gradient analysis has therefore a known environmental basis.

The full species and environmental data sets were run with the RDA option, and all defaults were chosen. The final output from this initial analysis was used to define the data sets to be tested. Variables with an inflation factor greater than 20 were discarded as these do not have a unique contribution to the ordination. Similarly strongly correlated variables make the plot unstable (Gauch 1982). Species variables that were therefore used included microbial biomass C and N, light fraction N and the potential rate of Nmin. Environmental variables included the percentage fines, aridity index, pH, bulk density, and light fraction mass. The latter variable was included, even though it was strongly correlated with the percentage fines, to note its influence.

Interpretation - In a RDA triplot the species variables are displayed as arrows whereas samples are displayed as points. With the appropriate scaling in use (ter Braak 1990), the length of an arrow indicates the importance of the variable, and it's direction indicates how well the environment is correlated with the various species composition axes. The cosine of the angle between the arrows indicates the correlation between the variables, the location of the site scores relative to the arrows indicates the environmental characteristics of the sites, and the location of the species scores to the arrows indicates the environmental preferences of each species (see ter Braak 1988 and 1990 for further explanation of the interpretation of the PCA biplot). The eigen analysis gives a measure of the importance of the axis (ter Braak 1990). The closer the values are to 1 the greater their importance.

## 2.3 Results

### 2.3.1 Biome and site comparisons

Table 2.3 and 2.5 show the results obtained for the determinations of the various fractions of soil organic matter (SOM) found in the soil of the 11 sites and Figure 2.1 shows how the sites differed in the mass of light fraction.

Table 2.3 The size of the soil organic matter C fractions, as a mean for the three land uses, in each of the sites of the savanna and grassland biome.

| | | Soil organic matter fractions | | | |
|---|---|---|---|---|---|
| **Biome** | **Site** | **Microbial C (mg C/g)** | **Lfr C (mg C/g)** | **Intermediate C (mg C/g)** | **Total C (mg C/g)** |
| Savanna | | | | | |
| | Towoomba1 | 0.17 | $0.80^{abc}$ | 6.58 | $7.55^{cde}$ |
| | Towoomba2 | 0.23 | $2.24^{ab}$ | 17.03 | $19.50^{b}$ |
| | Nylsvley | 0.08 | $0.65^{abc}$ | 5.25 | $5.98^{cde}$ |
| | Messina | 0.13 | $0.14^{c}$ | 4.72 | $4.99^{de}$ |
| | Klaserie | 0.10 | $1.77^{ab}$ | 5.06 | $6.93^{cde}$ |
| | Klaserie C | 0.11 | $0.34^{bc}$ | 5.32 | $5.77^{cde}$ |
| | Mkuze | 0.21 | $2.60^{ab}$ | 18.03 | $20.84^{b}$ |
| Grassland | | | | | |
| | Bloemhof | 0.10 | $1.17^{abc}$ | 2.70 | $3.97^{e}$ |
| | Bethlehem | 0.21 | $0.86^{bc}$ | 8.69 | $10.07^{cd}$ |
| | Bethal | 0.17 | $1.14^{abc}$ | 10.09 | $11.40^{cb}$ |
| | Vryheid | 0.10 | $2.64^{a}$ | 36.34 | $39.08^{a}$ |

Note that significant differences ($P < 0.05$) exist between sites which have different letters.

No significant differences were found for the amount of C in the microbial biomass for either of the biomes (Savanna = 147.23 mg C/kg versus Grassland = 145.38 mg C/kg). Site differences (Table 2.3) were significant ($F_{(10,81)} = 2.389$, $P = 0.015$). The Towoomba2 clay site had 2.8 times the amount of C in the microbial biomass of the Messina site. The variance of the microbial biomass C pool at each site was large and showed that sites were not generally unique in the level of microbial biomass C. The size of the microbial biomass C pool was significantly correlated with the percentage fines and negatively correlated with the bulk density of the soil (Table 2.4). No significant correlation of the amount of C in the microbial biomass was found with the aridity index or the amount of total C.

Table 2.4 Pearson correlation coefficients for the variable means investigated in this study (n = 31, significant correlations indicated by italized bold). Bartlett Chi-square statistic = 346.565, D.F. = 105, P < 0.05.

| | AI | BD | % fines | LFrC | LFrN | Lfrmass | McrbC | McrbN |
|---|---|---|---|---|---|---|---|---|
| Aridity index | 1.0 | | | | | | | |
| Bulk density | -0.387 | 1.0 | | | | | | |
| % fines | 0.425 | ***-0.784*** | 1.0 | | | | | |
| Light fraction C | ***0.374*** | ***-0.467*** | 0.481 | 1.0 | | | | |
| Light fraction N | ***0.375*** | ***-0.652*** | 0.600 | ***0.925*** | 1.0 | | | |
| Light fraction mass | 0.260 | ***-0.391*** | ***0.588*** | ***0.695*** | ***0.743*** | 1.0 | | |
| Microbial C | 0.222 | ***-0.365*** | ***0.508*** | 0.239 | 0.195 | 0.293 | 1.0 | |
| Microbial N | 0.314 | -0.122 | 0.231 | 0.334 | 0.152 | 0.200 | 0.199 | 1.0 |
| N mineralization | 0.333 | -0.080 | 0.019 | 0.249 | 0.126 | 0.112 | 0.274 | ***0.401*** |
| pH | ***-0.376*** | -0.083 | 0.220 | 0.049 | 0.113 | 0.333 | 0.362 | 0.136 |
| Total C | ***0.600*** | ***-0.607*** | 0.753 | ***0.419*** | ***0.482*** | 0.460 | 0.112 | ***0.365*** |
| Total N | ***0.700*** | ***-0.680*** | ***0.765*** | ***0.535*** | ***0.562*** | ***0.525*** | 0.312 | ***0.442*** |
| Soil C:N | ***0.503*** | ***-0.505*** | ***0.768*** | 0.373 | 0.464 | 0.408 | 0.258 | 0.336 |
| Light fraction C:N | -0.01 | 0.017 | -0.169 | 0.341 | 0.050 | -0.007 | 0.078 | ***0.436*** |

| | Nmin | pH | Total C | Total N | s C:N | Lfr C:N |
|---|---|---|---|---|---|---|
| Nmineralization | 1.0 | | | | | |
| pH | -0.128 | 1.0 | | | | |
| Total C | 0.276 | -0.137 | 1.0 | | | |
| Total N | ***0.375*** | -0.148 | ***0.914*** | 1.0 | | |
| Soil C:N | 0.165 | 0.111 | ***0.846*** | ***0.701*** | 1.0 | |
| Light fraction C:N | 0.306 | -0.138 | -0.012 | 0.128 | -0.130 | 1.0 |

The amount of C in the light fraction was not significantly different between biomes (Savanna = 19.45 mg C/g v Grassland = 25.12 mg C/g, $F_{(1,90)} = 0.943$, $P = 0.334$), but significant site differences existed (Table 2.3). Light fraction C was significantly positively correlated with the aridity index, and the total N and C contents of the soil (Table 2.4).

The Vryheid site had a significantly greater amount of total C (Table 2.3). The sites of Towoomba2, Mkuze and Vryheid had the greatest total C which was on average 3 times greater than the total C content of the other sites, which tended to have lower percentage fines in the soil. The Bloemhof site had the lowest amount of total C, but this was not significantly different to the remaining sites. Total C was significantly correlated with the aridity index, percentage fines in the soil, and C in the light fraction (Table 2.4).

Table 2.5 The size of the soil organic matter N fractions, as a mean for the three land uses, in each of the sites of the savanna and grassland biome.

| Biome | Site | Soil organic matter fractions | | | |
|---|---|---|---|---|---|
| | | **Microbial N (ug N/g)** | **Lfr N (ug N/g)** | **Intermediate N (ug N/g)** | **Total N (ug N/g)** |
| Savanna | | | | | |
| | Towoomba1 | 8.72 | $314.54^{bc}$ | 416.35 | $739.61^{cbd}$ |
| | Towoomba2 | 9.74 | $1187.73^{ab}$ | -257.75 | $939.72^{b}$ |
| | Nylsvley | 7.34 | $289.98^{bc}$ | 231.13 | $528.45^{ccd}$ |
| | Messina | 7.22 | $66.38^{c}$ | 363.92 | $437.52^{cd}$ |
| | Klaserie | 7.69 | $732.45^{ab}$ | -70.26 | $669.88^{cbd}$ |
| | Klaserie C | 6.95 | $169.93^{bc}$ | 347.93 | $524.81^{ccd}$ |
| | Mkuze | 8.97 | $1490.76^{ab}$ | 45.52 | $1545.25^{a}$ |
| Grassland | | | | | |
| | Bloemhof | 6.01 | $455.21^{bc}$ | -102.30 | $358.92^{c}$ |
| | Bethlehem | 7.03 | $347.96^{bc}$ | 439.34 | $794.33^{cb}$ |
| | Bethal | 9.17 | $616.98^{ab}$ | 331.04 | $957.19^{b}$ |
| | Vryheid | 9.45 | $1259.41^{a}$ | 676.50 | $1945.36^{a}$ |

Note that significant differences ($P < 0.05$) exist between sites which have different letters.

The amount of N in the microbial biomass was not significantly different between the two biomes (Savanna = 8.09 mg N/kg versus Grassland = 7.92 mg N /kg, $F_{(1,88)} = 0.886$, $P = 0.349$). Similarly the differences between the sites were not significant ($F_{(10,81)} = 1.508$, $P = 0.152$). No relationship existed between microbial biomass N and the aridity index nor the percentage fines in the soil (Table 2.4). Microbial biomass N was however positively correlated with total soil N and C, the potential rate of N mineralization, and the C:N ratio of the light fraction.

The amount of N in the light fraction of the savanna biome (13 536.44 ug N/g) versus that in the grassland biome (15 850.69 ug N/g) was not significantly different ($F_{(1,90)} = 0.973$,

$P = 0.327$). However, sites had significantly different amounts of N in their light fraction ($F_{(10,81)} = 6.235$, $P = 0.000$). Light fraction N was also significantly correlated with the mass of light fraction, due to the way it was calculated (see methods section), and the total N and C in the soil (Table 2.4).

Errors in the use of mean values for calculating the light fraction mass for land use are proposed to be the reason for the negative intermediate fraction values.

The light fraction mass was not significantly different ($F_{(1,90)} = 0.160$, $P = 0.690$) between biomes with the savanna biome having a light fraction mass of 2.34 g/kg soil and the grassland having a slightly greater mass of 2.49 g/kg soil. There were however significant differences between sites ($F_{(10,81)} = 4.346$, $P = 0.000$) (Figure 2.1). Sites which had a high percentage of fines in the soil similarly had a large mass of light fraction (Table 2.4). The same relationship was not however found using the aridity index. Light fraction mass was also positively correlated with the total N in the soil.

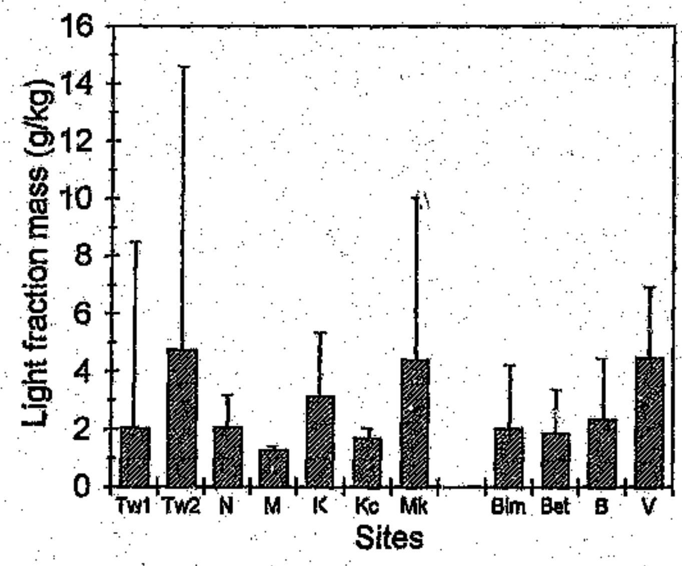


Figure 2.1 The mean (+ SD) mass of the light fraction at each site (n = 3) of the savanna and grassland biome. Similar trends exist for the difference in light fraction N and C between the 11 sites.

### 2.3.2 Land use comparisons

The mean quantity of N in the microbial biomass of the land uses were significantly different (con = 8.08$^{a}$, cult = 6.82$^{ab}$, lvstk = 9.09$^{ac}$, $F_{(2,89)}$ = 5.214, P = 0.007) but significant differences were not found for the amount of C in the microbial biomass of each land use type (con = 161.91 *ug* C/kg, cult = 127.60 *ug* C/kg, lvstk = 143.03 *ug* C/kg). None of the sites of the land uses were significantly different in the amount of N in the microbial biomass (Table 2.6). The Mkuze site had the greatest microbial biomass, which was only slightly larger than that of the other sites. Sites were not significantly different in the size of the microbial biomass C pools of each land use, but the Mkuze conserved site had the greatest C content.

Table 2.6 The size of the soil organic matter fractions for the land uses of the 11 sites in the savanna and grassland biome.

| | | Soil organic matter fractions | | | |
|---|---|---|---|---|---|
| **Biome and site** | **Land use** | **Microbial C ($\mu$g C/g)** | **Microbial N ($\mu$g N/g)** | **Lfr C (mg C/g)** | **Lfr N (mg N/g)** |
| *Savanna* | | | | | |
| Towoomba1 | Con | 162.71 | 10.48 | $10.70^{d}$ | $5.52^{e}$ |
| Towoomba2 | Con | 215.33 | 11.11 | $142.26^{a}$ | $73.40^{a}$ |
| Nylsvley | Con | 143.86 | 9.11 | $20.59^{bcd}$ | $10.62^{abc}$ |
| Messina | Con | 131.35 | 7.77 | $4.82^{d}$ | $2.49^{e}$ |
| KlaserieC | Con | 167.89 | 8.93 | $13.42^{cd}$ | $6.93^{bc}$ |
| Mkuze | Con | 308.39 | 6.81 | $61.08^{abc}$ | $31.51^{ab}$ |
| *Grassland* | | | | | |
| Bloemhof | Con | 117.65 | 4.98 | $9.90^{d}$ | $5.96^{e}$ |
| Bethlehem | Con | 248.80 | 4.25 | $18.05^{cd}$ | $10.86^{abc}$ |
| Bethal | Con | 46.78 | 8.52 | $85.03^{ab}$ | $51.16^{a}$ |
| Vryheid | Con | 76.30 | 8.84 | $74.53^{ab}$ | $44.85^{a}$ |
| *Savanna* | | | | | |
| Towoomba1 | Cul | 183.94 | 6.90 | $4.80^{ab}$ | $3.29^{ab}$ |
| Towoomba2 | Cul | 253.40 | 7.68 | $13.33^{ab}$ | $9.14^{ab}$ |
| Nylsvley | Cul | 40.21 | 6.31 | $8.89^{ab}$ | $6.09^{ab}$ |
| Messina | Cul | 132.10 | 6.95 | $3.00^{ab}$ | $2.06^{ab}$ |
| Klaserie | Cul | 96.58 | 6.70 | $22.38^{ab}$ | $15.34^{ab}$ |
| Klaserie (C) | Cul | 48.99 | 6.19 | $9.16^{ab}$ | $6.28^{ab}$ |
| Mkuze | Cul | 171.86 | 4.85 | $149.34^{a}$ | $102.39^{a}$ |
| *Grassland* | | | | | |
| Bloemhof | Cul | 75.58 | 6.21 | $10.03^{ab}$ | $6.88^{ab}$ |
| Bethlehem | Cul | 157.38 | 6.91 | $2.76^{b}$ | $1.89^{b}$ |
| Bethal | Cul | 99.95 | 8.72 | $4.93^{ab}$ | $3.38^{ab}$ |
| Vryheid | Cul | 185.57 | 7.62 | $52.92^{a}$ | $36.28^{a}$ |
| *Savanna* | | | | | |
| Towoomba1 | Lvs | 151.07 | 8.79 | $68.02^{ab}$ | $23.98^{ab}$ |
| Nylsvley | Lvs | 56.35 | 6.62 | $38.32^{abc}$ | $13.51^{abc}$ |
| Messina | Lvs | 119.70 | 6.95 | $6.73^{c}$ | $2.37^{c}$ |
| Klaserie | Lvs | 100.41 | 8.69 | $100.86^{ab}$ | $35.56^{ab}$ |
| Klaserie (C) | Lvs | 104.60 | 5.72 | $12.78^{bc}$ | $4.51^{bc}$ |
| Mkuze | Lvs | 159.21 | 15.24 | $60.96^{ab}$ | $21.49^{ab}$ |
| *Grassland* | | | | | |
| Bloemhof | Lvs | 115.55 | 6.84 | $101.96^{ab}$ | $34.61^{ab}$ |
| Bethlehem | Lvs | 227.85 | 9.93 | $69.28^{abc}$ | $23.52^{abc}$ |
| Bethal | Lvs | 158.95 | 10.25 | $28.78^{abc}$ | $9.77^{abc}$ |
| Vryheid | Lvs | 236.64 | 11.89 | $147.72^{a}$ | $50.15^{a}$ |

Note that significant differences ($P < 0.05$) exist between sites which have different letters.

The mass of light fraction was significantly different between land uses with the cultivated land use having a smaller mass (1.64 g/kg) as opposed to the conserved and cultivated land uses (3.00 g/kg and 2.87 g/kg) respectively (Figure 2.2). The cultivated land uses usually had half as much light fraction mass. There was also a significant difference between the amount of N in the light fraction for the three land uses ($F_{(2,89)} = 4.741$, $P = 0.011$) with the cultivated land use having 50% less N in the light fraction than either the conserved (12 823.31 ug N/g) or the livestock land uses (13 458.60 ug N/g) (Table 2.6). Similarly there was a significant land difference in the amount of C in the light fraction ($F_{(2,89)} = 12.290$, $P = 0.000$). The cultivated land use had the lowest amount of C in the light fraction (10.33 mg C/g) versus that in the conserved (25.00 mg C/g) and livestock (40.46 mg C/g) land uses.

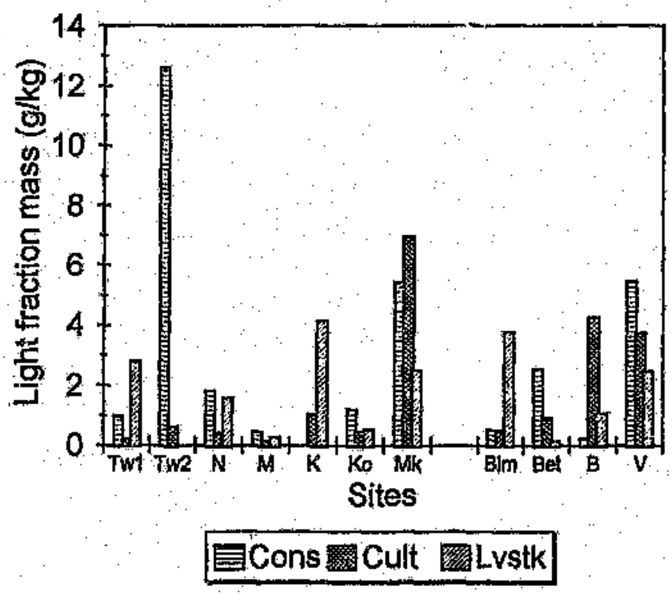


Figure 2.2 The mean mass of light fraction (g/kg) in each land use of the 11 sites investigated in the savanna and grassland biome.

The total soil N was significantly greater in the conserved (950.60 mg N /kg) and livestock (857.04 mg N/kg) land uses ($F_{(2,89)} = 8.677$, $P = 0.000$) than it was in the cultivated land use (535.80 mg N/kg). The differences between the sites of a particular land use shows that sites tended to differ in the amount of soil C and N ($F_{(10,81)} = 15.578$, $P = 0.000$ and $F_{(10,81)} = 16.375$, $P = 0.000$, respectively) with sites having a close correlation between the amount of N and C in the soil (Table 2.4). Sites which had a high percentage of fines, generally had greater amounts of total N and C than sites which had a high percentage of sands (Appendix 1). This seems to imply that soil C and N varied in accordance with the soil texture, or more specifically the percentage of fines (silt + clay). This was evident from the close correlation between the percentage fines and the level of soil C and N. The C:N ratio at each site appeared to be fairly constant fluctuating around 10 or 11 (data not presented). Discrepancies exist where the soil was very clayey, eg. Towoomba 2, Mkuze, Vryheid (Appendix 1). Where the soil was very clayey (% fines > 60), the amount of carbon in the soil was approximately 2.5 to 3 times greater than that in the sandy soils. The Vryheid site had almost twice as much C than similar sites in the savanna, e.g. Towoomba and Mkuze. However the difference between the amount of soil C in the two biomes was not significantly different ($F_{(1,90)} = 3.628$, $P = 0.060$). Similar trends existed for the amount of soil N.

An analysis of covariance, with the aridity index and the percentage fines at each site and land use as covariates, was used to indicate if biomes, land uses and sites differed according to the impact of land use alone. This showed that the mean site quantity of total N, total C, and light fraction N was significantly smaller for the cultivated land use ($F_{(2,31)} = 19.522$, $P = 0.000$; $F_{(2,31)} = 10.229$, $P = 0.001$; and $F_{(2,31)} = 7.802$, $P = 0.002$ respectively), but significant biome and site differences were not apparent. All other variables were insignificant with an ANCOVA using aridity index and the percentage fines in the soil as covariates.

### 2.3.3 Comparison of sites and land uses along the environmental gradients

Figure 2.3 shows that the sites were distributed along a clear moisture gradient. The driest site was that of Messina (Mes) which is located in the far north of the South African savanna. The driest site in the grassland biome was that of Bloemhof (Blm). This site is located on the western fringe of the grassland biome. Progressing from the drier west to moister east (bottom left to top right in Figure 2.3) the sites are more moist and therefore have a greater aridity index. Thus the Vryheid site (Vry) has a mean annual precipitation of 950 mm and an aridity index of 0.519 (Table 2.7).

Table 2.7 The climatic variables for the 9 locations investigated.

| Biome | Location | Tmin[1] (°C) | Tmax[2] (°C) | MAP[3] (mm) | MAE[4] (mm) | Aridity index[5] |
|---|---|---|---|---|---|---|
| savanna - | | | | | | |
| | Towoomba | 11.2 | 26.6 | 661 | 2252 | 0.294 |
| | Nylsvley | 12.0 | 26.5 | 623 | 2201 | 0.283 |
| | Messina | 15.1 | 29.6 | 309 | 2682 | 0.130 |
| | Klaserie | 14.8 | 27.0 | 562 | 2215 | 0.286 |
| | Mkuze | 14.6 | 29.0 | 578 | 2230 | 0.330 |
| Grassland - | | | | | | |
| | Bloemhof | 8.8 | 25.6 | 537 | 2280 | 0.211 |
| | Bethlehem | 7.8 | 22.0 | 650 | 1916 | 0.386 |
| | Bethal | 7.8 | 22.4 | 747 | 1702 | 0.423 |
| | Vryheid | 11.9 | 23.2 | 950 | 1671 | 0.519 |

[1]Tmin is mean annual minimum temperature at 08h00
[2] Tmax is mean annual maximum temperature at 08h00
[3] MAP is mean annual precipitation for the last twenty years
[4] MAE is mean annual potential evaporation for the last twenty years
[5] Aridity index = MAP/MAE

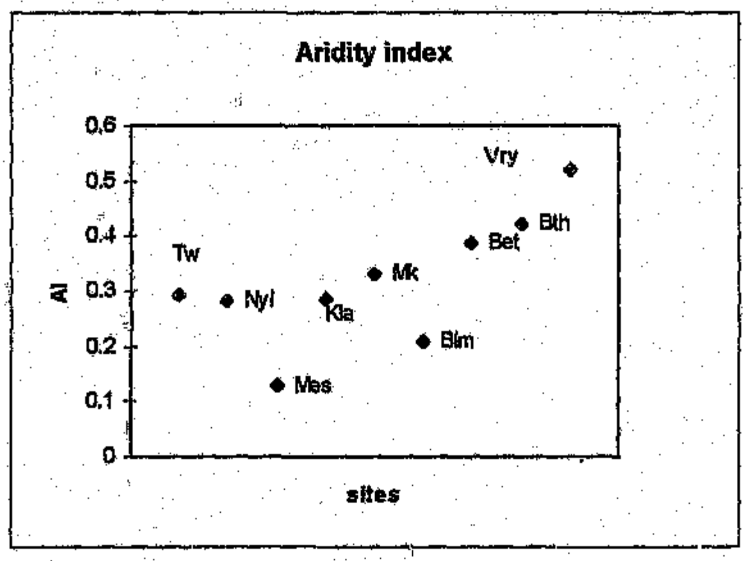


Figure 2.3 The distribution of the investigated sites according to the aridity index of each.

An examination of how the SOM fractions change along gradients of moisture and soil type (Figures 2.4 - 2.6 ) show that generally there was an increase in the amount of soil carbon and nitrogen from the drier west to the moister east (i.e. along a moisture gradient). Thus where sites received more precipitation and/or have a lower annual evaporative loss, the amount of soil total N is expected to be greater in the South African savanna and grassland biome (Figure 2.4). Similarly where sites had a high percentage of fines (silt + clay) in the soil, the amount of total soil N was also greater (Figure 2.5). Similar trends exist for the quantity of total C in the soil as the amount of total N and total C were closely correlated (Figure 2.6).

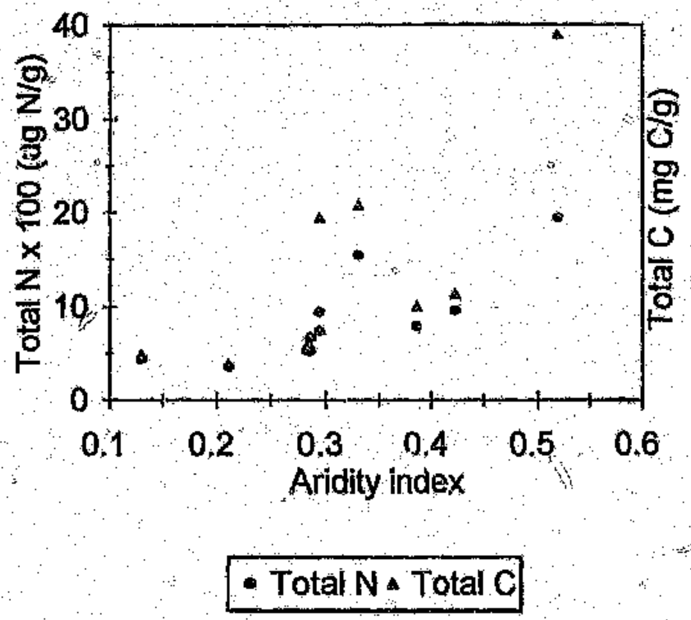


Figure 2.4 The relation between mean site (a) total nitrogen (circles) (*u*g N/g) and (b) total C (traingles) (mg C/g) for the 11 sites investigated. Each is the mean of the land uses, and shows the variation in total N and total C with site aridity.

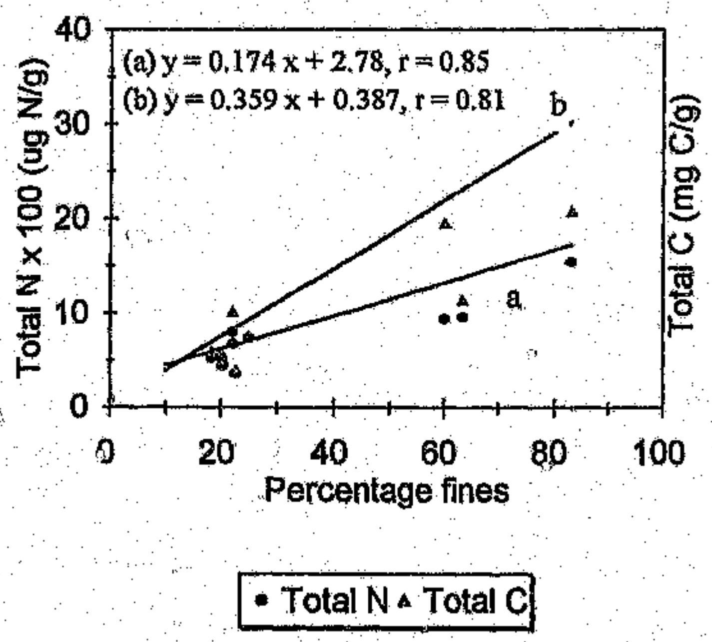


Figure 2.5 The relation between mean site (a) totrl nitrogen (*u*g N/g) and (b) total carbon (mg C/g) with the percentage fines. Means for the three land uses at a site were used.

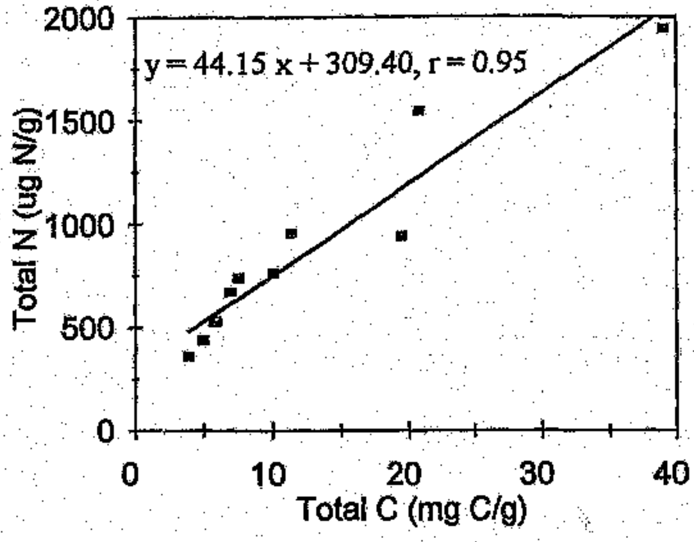


Figure 2.6 The relation between the total amount of carbon and nitrogen in the soil of the 11 sites investigated as a mean of the land uses at each site.

### 2.3.4 Soil differences between sites and land uses

The soil chemical and physical attributes for each land use at each site are presented in Appendix 1. The pH for the three land uses was not statistically different ($F_{(2,89)} = 0.963$, $P = 0.386$). The lowest pH was on the livestock land use (5.57), whilst the highest was on the cultivated land use (5.78). The pH of the soil at the Towoomba clay site was significantly greater than that of any of the other sites ($F_{(10,81)} = 17.787$, $P = 0.000$), and the Messina, Mkuze, and Bethal sites all had pH's which were statistically the same. All of these sites also had a high percentage of clay and silt (percentage fines). The percentage of fines meaned across all land uses was found to be not significantly different between the sites ($F_{(10,20)} = 1.263$, $P = 0.314$). Similarly the percentage of fines per each land use was not significantly different ($F_{(2,28)} = 0.333$, $P = 0.719$) indicating that sites did not significantly differ in pH. Table 2.4 does however show that a significant negative correlation existed for the change of pH with the AI and indicates that sites which received greater rainfall tended to have more acid soils, i.e. a lower pH.

The bulk density for the three land uses was not significantly different ($F_{(2,78)} = 0.611$, $P = 0.546$). The cultivated land use had the highest bulk density of 1.34 g/cm$^3$, while the livestock area had the lowest of 1.26 g/cm$^3$. The bulk density at particular sites was significantly different ($F_{(10,70)} = 10.831$, $P = 0.000$). The lowest bulk density was recorded at the Mkuze site, followed by the Towoomba clay site and then the Vryheid site. All of these sites had a high percentage of fines in the soil, i.e. a clayey texture and this indicates that the methodology for determining the bulk density was imprecise. This stems from the difficulty in sampling clay soils for bulk density measurements using the core technique.

Although trends exist to explain the change in soil C and N over the two environmental gradients, differences between land uses were not always apparent. However, sites of the conserved land use usually had greater levels of light fraction mass than those for the cultivated or livestock land uses (Figure 2.7). Figure 2.8 shows that the sites of the cultivated land use, and sites of the livestock land use both had significantly lower amounts of light fraction mass than sites of the conserved land use. Slopes of the regression lines for the livestock and cultivated land uses were significantly different ($F_{(9,10)} = 22.494$). This graph therefore shows that if livestock production intensified, then the light fraction mass can be expected to decline to lower levels than are in the conserved land uses.

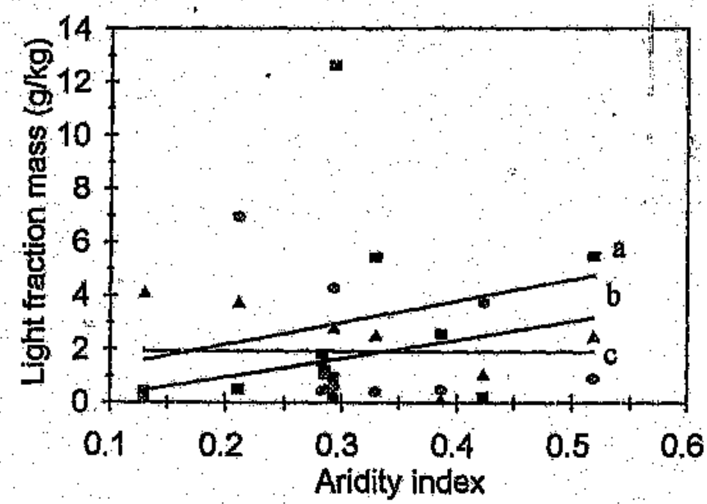


Figure 2.7 The relation between the light fraction mass and the predictor variable aridity index. Means per land use at each site have been used (n = 3) for the three land uses - conserved (a), cultivated (b) and livestock (c).

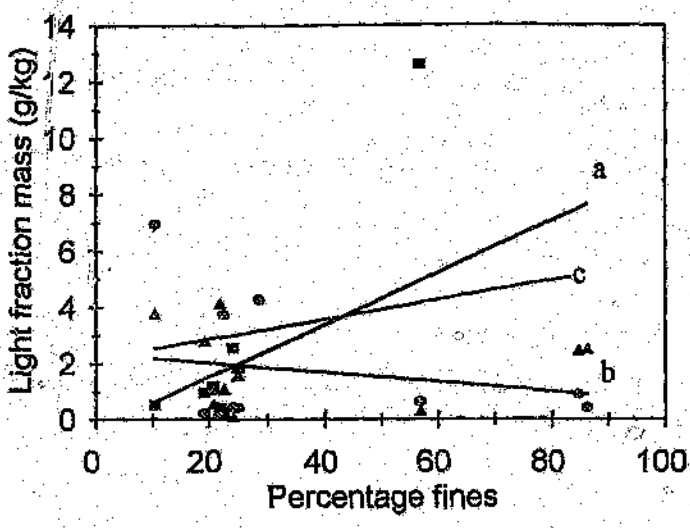


Figure 2.8 The relation between the light fraction mass and the predictor variable percentage fines (silt + clay). Means per land use at each site have been used (n = 3) for the three land uses - conserved (a), cultivated (b) and livestock (c).

The use of regression analyses to investigate the interaction between the environmental variables and the various SOM variables is not appropriate as the predictor variables do not necessarily represent a linear sequence. For example, in regressing the light fraction mass against the predictor variable % fines, the range of soil types for the 11 sites are at the two extremities. That is, soils for site tended to be either sandy or clayey, with very few sites with intermediate values between these extremes. For this reason, emphasis has been placed on the multivariate analyses as these are considered to be more powerful in analysing and interpreting the patterns in the data (Dr E.R. Robinson[1], personal communication).

### 2.3.5 Multivariate techniques to identify patterns in the similarity/dissimilarity of sites

Multivariate techniques were used to describe the patterns, or ordination, of sites and land uses according to the abundance values for "species" and environmental variables. The term species is analogous to the dependent variables Nmin, light fraction N, microbial C

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and microbial N, where as the environmental variables include the site variables bulk density, percentage fines, aridity index, light fraction mass, and pH.

A RDA performed with the species and environmental data files is shown in Figure 2.9 and Table 2.8. The species - environment correlation measures the strength of the relationship between the environment and the species for that axis. Thus axis 1 shows that there was a 57.4% correlation between the species and the environmental variables. This tapers off with additional axes. Table 2.8 shows that the RDA could only explain 28.1% of the total species variance in the first 3 axes. However 100% of the variance in the species - environment relationship was explained by the first 3 axes.

Table 2.8 Ordination summary for the RDA performed on the species and environmental data for each of the land uses at each site.

| Axes | 1 | 2 | 3 | 4 |
|---|---|---|---|---|
| **Eigenvalues** | 0.255 | 0.026 | 0.0 | 0.0 |
| **Species - environment correlations** | 0.574 | 0.340 | 0.333 | 0.0 |
| **Cumulative percentage variance of:** | | | | |
| **Species data** | 25.5 | 28.1 | 28.1 | 0.0 |
| **Species - environment relation** | 90.7 | 99.9 | 100.0 | 0.0 |

Figure 2.9 shows that little of the variance between the sites could be explained by the principal components analysis. Sites could only be separated into two broad groups. One group was positively correlated with the microbial biomass C and N (quadrant 2) where as the other group separated according to the light fraction mass at the site (quadrant 4). although the exact pattern between the distribution of sites with the $PCA_{species}$ and the RDA are not the same, there is a close resemblance. According to ter Braak and Prentice (1988) the similarity in the distribution of the sites shows that there is greater certainty that the environmental variables are appropriate in describing the way that the species relate to the environment. This is apparent from Figure 2.9 which shows how the environmental variables correlate with the distribution of the sites. The percentage of fines and soil pH were positively correlated with group 1, while the bulk density of the soil was positively correlated with group 2. Group 3 consisted of sites with a greater mass of light fraction in the soil but which had a lower soil pH and lower percentage of fines.

Investigating the correlation between the species and the environmental variables of the RDA showed that the amount of microbial biomass C and N was positively correlated with the percentage fines in the soil and soil pH. The potential rate of Nmin and light fraction N were both positively correlated with the mass of light fraction in the soil and less so with the aridity index. All variables were negatively correlated with the bulk density of the soil

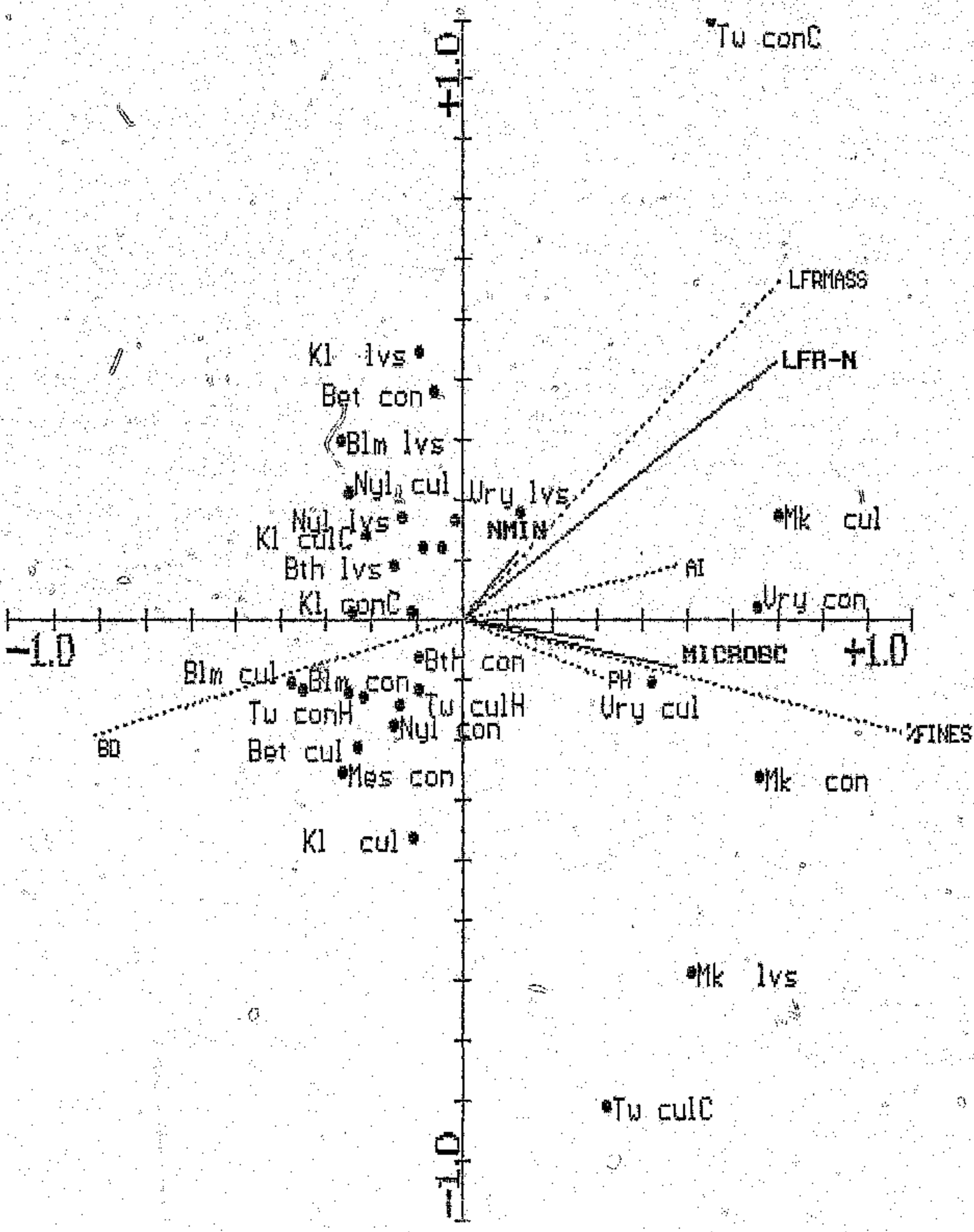



Figure 2.9 A triplot showing a redundancy analysis (RDA) for the samples, the "species" variables, and the environmental variables. ($\lambda_1 = 0.574$, $\lambda_2 = 0.340$, scaling = 2).

### 2.3.6 The proportions of N and C in the SOM fractions

The amount of N and C in the various SOM pools of each biome, land use and within the three land uses investigated in the grassland and savanna biome are presented in appendices 2 and 3.

The proportion of C and N in the three SOM pools of each land use type are shown in Figures 2.10 and 2.11. Chi-square tests of association using EPISTAT showed that no significant association between the N in the light fractions of the two biomes was evident, i.e. both the savanna and grassland biome have similar ratios of N in the microbial N pool, light fraction N pool and the passive N pool (Table 2.9 and 2.10). Similarly there was not a significant association between the proportion of N in the fractions of the land uses. That is, the proportion of N in the SOM fraction of the conserved, cultivated and livestock land uses was not significantly different. However significant differences in the proportion of N in the SOM fractions of the three land uses of each biome were evident. In the savanna biome, the cultivated land use had a greater percentage (96.77) of N in the light fraction than either the conserved or livestock land uses (65.69 and 56.64% respectively). In the grassland biome, the livestock land use had a greater percentage (75.73%) of N in the light fraction N than either the conserved or the cultivated (58.44 and 44.77% respectively) land uses.

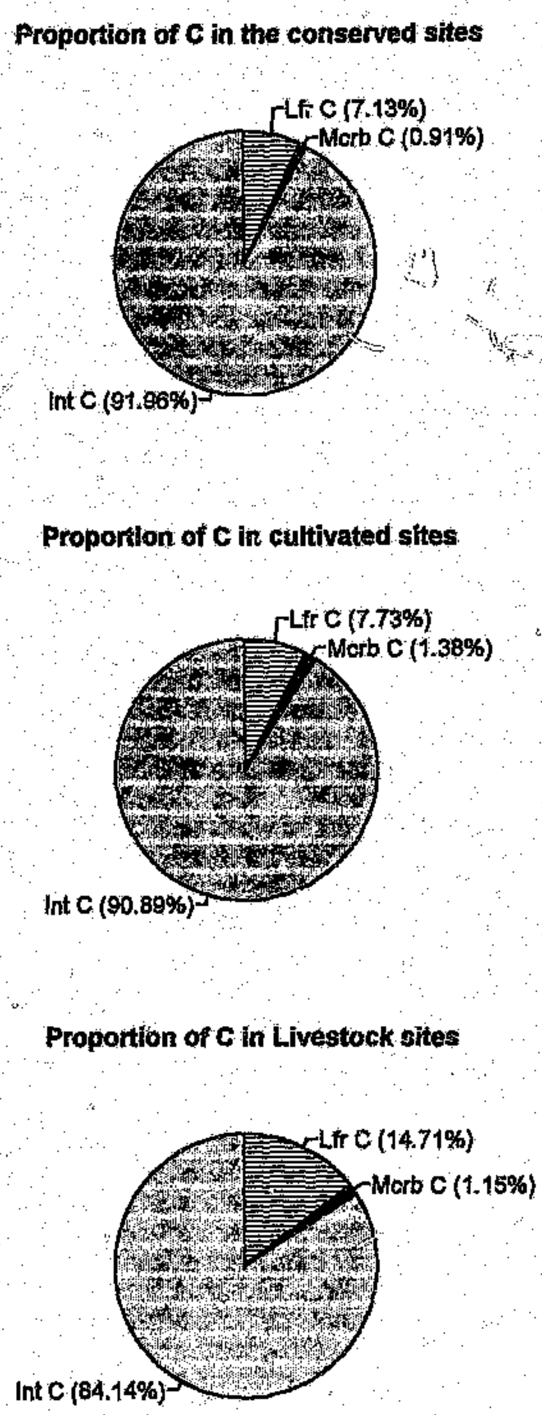



Figure 2.10 The proportion of C for the SOM fractions active pool (Mcrb C), slow pool (Lfr C) and intermediate pool (Int C).

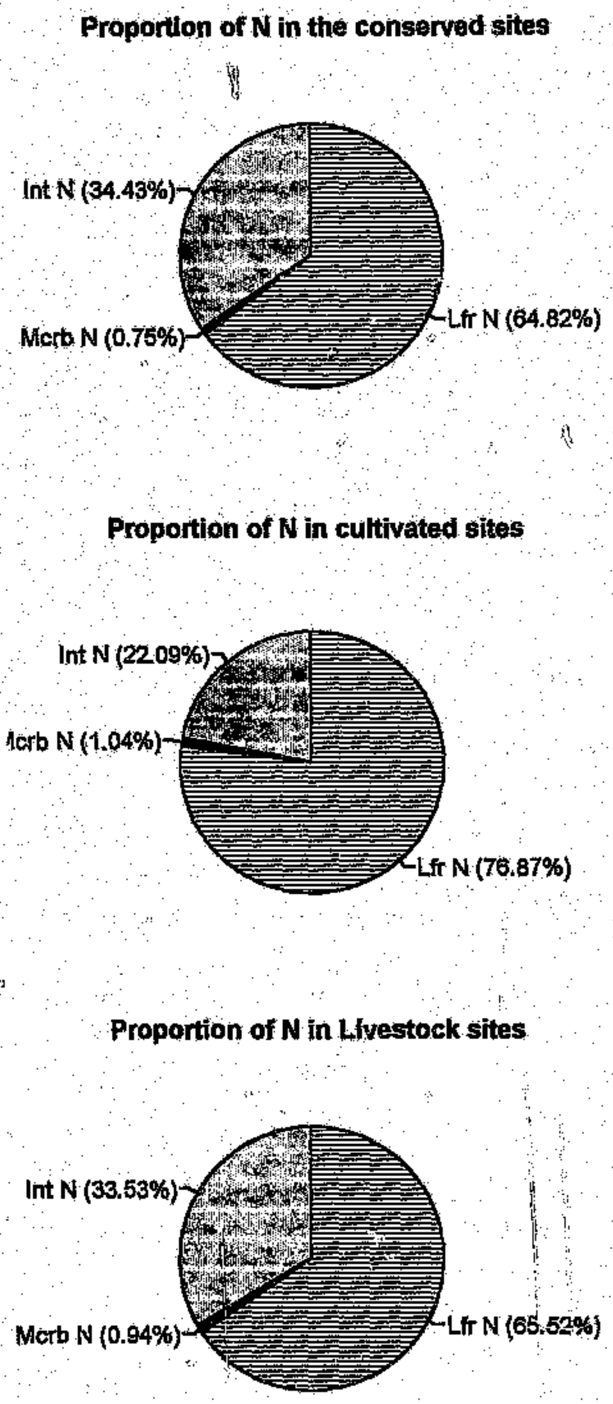



Figure 2.11 The proportion of N for the SOM fractions active pool (Mcrb N), slow pool (Lfr N) and intermediate pool (Int N).

Generally the proportion of N that is immobilised into the microbial pool of each land use ranged between 0.75% and 1.04%. Less than half the N that could be made accessible to plants by the mineralization of organic matter was locked up in the Intermediate soil pool. No significant association was evident between the carbon in the SOM fractions of the two biomes, i.e. both of the biomes had a similar proportion of C in the pools of the soil organic matter fractions. The same was true for the proportion of C in the SOM fractions of the three land uses., as well as for the proportion of C in the SOM fractions per land use in each biome.

In all land uses of the savanna and grassland biome, the proportion of C in the Intermediate pool exceeded that in the microbial or light fraction C pools by approximately 90%.

Table 2.9 A chi-square comparison of the percentages of N per soil organic matter fraction for the two biomes, three land uses, and for the three land uses in each biome. Sites and land uses have been meaned for the comparison of the savanna and grassland biome, sites have been meaned across biome type for the land use comparison, and for the land use comparison per biome.

| Comparison | Chi-square | D.F. | Significance |
|---|---|---|---|
| Savanna versus Grassland | 0.1702 | 2 | NS |
| Con V Cult V Lvstk | 2.4939 | 4 | NS |
| Con V Cult V Lvstk (Sav) | 18.8178 | 4 | * |
| Con V Cult V Lvstk (Grass) | 9.7796 | 4 | * |

Table 2.10 A chi-square comparison of the percentages of C per soil organic matter fraction for the two biomes, three land uses, and for the three land uses in each biome. Sites and land uses have been meaned for the comparison of the savanna and grassland biome, sites have been meaned across biome type for the land use comparison, and for the land use comparison per biome.

| Comparison | Chi-square | D.F. | Significance |
|---|---|---|---|
| Savanna versus Grassland | 0.2356 | 2 | NS |
| Con V Cult V Lvstk | 2.4646 | 4 | NS |
| Con V Cult V Lvstk (Sav) | 1.4856 | 4 | NS |
| Con V Cult V Lvstk (Grass) | 5.7105 | 4 | NS |

## 2.4 Discussion

### 2.4.1 Controls on the accumulation of SOM

Greenland *et al* (1992) recognise that SOM dynamics and the rate of Nmin of soil organic matter are governed by several environmental factors. These include the way that Nmin changes with temperature and rainfall, the biological activity in the soil, the composition of the organic matter entering the soil, and by how disturbance acts to alter these factors. In addition several intrinsic properties of a soil similarly determine the rate of Nmin, and therefore the dynamics of the SOM. These include the clay content, clay type, drainage, acidity, and nutritional status of the soil.

The redundancy analyses for the sites and land uses investigated in this study clearly support the above statements. Sites which were moist, and which had a high percentage of fines in the soil tended to have a greater mass of light fraction. Similarly these clayey sites and land uses had a greater microbial biomass, and greater quantities of N and C in the light fraction. The microbial biomass tended to increase with the pH, and was associated with the aridity index of the sites. Although time did not allow quantification of the fertility of the soil types that were sampled, i.e. by using growth response trials of grass production, it is possible to infer that sites and land uses which have greater percentages of fines, will be more productive. Such a conclusion is based on the greater supply of SOM and the faster rate at which it turned over, i.e. was mineralized, in the soil at these sites. Sites which have a high percentage of fines, and which are located in the more moist parts of South Africa, would therefore tend to show such responses (Figure 2. 9) on the basis of these assumptions. However quantification of the standing dead biomass at each site and land use showed that not significant relation existed between the biomass and the soil texture at a site (Figure 2.12)

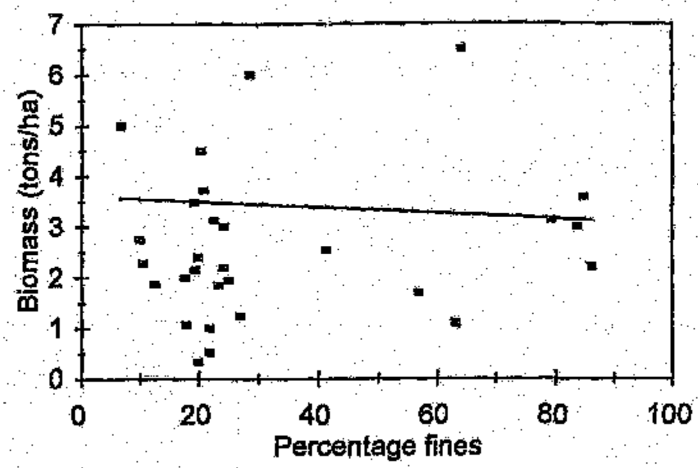



Figure 2.12 The relation between mean (n = 3) standing dead biomass and the percentage fines at a site.

Processes contributing to the formulation of such an hypothesis would include the role that the various SOM fractions, soil texture, site moisture, and other associated factors play in determining SOM turnover (Greenland *et al* 1992; ).

Micro-organisms, which mineralise SOM and thereby release nutrients, require sufficient substrate to survive. Not only must there be sufficient material, the organic matter must be of a suitable quality for decomposition to occur. That is, it must have a C:N ratio below approximately 25:1. Such a ratio is needed because micro-organisms require N in sufficient quantities that they are able to assimilate the necessary proteins and metabolites to function (Paul and Clark, 1989). If the substrate is either too limiting or if the quality of the substrate is too fibrous, i.e. it has a C:N ratio above 25:1, then micro-organisms would immobilise N. This will reduce the availability of N to plants in the short term.

The C:N ratio's of the light fraction in this study were exceedingly low ranging from 1.45 to 2.85 (results not presented). This was attributed to the method in which the light fraction C and N determinations were obtained (see methods section) as using the uncorrected data showed that the light fraction C:N ratio should have fluctuated between 12 and 25. Nevertheless the ratio's were below what is assumed to be the upper limit for N mineralization to occur, i.e. 16 to 30 (Paul and Clark 1989; Scholes and Walker 1993).

The C:N ratio for the soil, i.e. using the total C and total N quantities, ranged between 6.5 and 30 for the land uses. This is similar to C:N ratios obtained in grassland and savanna sites in South Africa (Ellery 1992, Carter, G.A 1993, Mckean 1993; Du Preez and Du Toit 1995). Du Preez and Du Toit (1995) reported C:N ratios for several agroecosystems in the grasslands of the Free State and values ranged from 9.77 to 14.21. Carter, G.A (1993) reported the C:N ratio's of several savanna sites differing in climate and vegetation and gave values in the range 8 to 30.

Havlin *et al* (1990) contend that average soil C and N contents are directly proportional to clay content. They also state that combined over all locations in eastern Kansas under differing cultivated land use regimes (conventional tillage versus no tillage), the organic C content was highly correlated to organic N content ($r = 0.98^{**}$). Similar results were shown in this study where the correlation between the total C and total N content was significant ($r = 0.92^{*}$). It would also appear that the C and N dynamics of these sites vary with the clay content and along the moisture gradient. Similar hypotheses were proposed by Du Preez and Du Toit (1995).

The supply of decomposable material on which micro-organisms can act need not only be provided by the mass of decomposing material designated as the light fraction, but is also contributed by the organic matter that is adsorbed on the silt and clay portion in the soil. Thus mineralization includes the breakdown and release of nutrients, in this case N, from all fractions of the SOM pool in the soil (Greenland *et al* 1992). Consequently it is the combined content of SOM that determines whether a particular soil type will display net positive or net negative rates of Nmin. These are designated N mineralization and immobilization respectively (Chapter 3).

The immobilization of N refers to the locking up N into the microbial biomass. Microbial biomass not only requires an adequate substrate supply, it also requires a suitable supply of energy, usually provided in the form of C; and suitable abiotic conditions. Groffman *et al* (1996) suggest that soil type is a more important controller of microbial biomass and activity, although different grass species can create significant, but small differences in the rate of Nmin. This would be evident in the way that the C:N ratio of the plant material affects the microbial biomass. Although I have been unable to assess the C:N ratios of the vegetation at each land use, the relationships suggested by Groffman *et al* (1996) appear to apply to these sites as well. That is sites which have lower ratios of C:N in the vegetation would be expected to have a greater microbial pool size if all else was the same. This needs further study to validate (but see Prasad *et al* 1994). The correlations between soil type and microbial biomass and activity appear to be a function of the percentage clay in the soil (Jones, 1973), the kind of vegetation (Hofstede 1995), and the impact of land use (Prasad *et al* 1994). These all tend to alter the SOM dynamics of the system, of which the microbial biomass is an important constituent.

*Microbial biomass*

Microbial biomass measurements provide a sensitive indication of changes brought about by soil management, long before any changes can be detected in the soil organic matter content (Powlson *et al* 1987). In this study the microbial biomass N content was significantly smaller in sites of the cultivated land use. Sites of the conserved land use and livestock land use had progressively more N in the microbial biomass.

The sensitivity of the method used to quantify the amount of C and N in the microbial biomass has received some review in the recent past. The chloroform fumigation extraction technique is accepted as a rapid, reliable and accurate method (Monz *et al* 1991). However an inconsistency in these results hints at a possible weakness in the technique. This included incubating samples for three days to prime the microbial population. Samples that had soil water contents differing from being almost 0 to being almost 100% meant that the application of a standard amount of 2.5 ml distilled water to bring the samples to a theoretical field capacity of 60%, gave that some samples had field capacities in excess of this, and approached full saturation. This would have allowed for anaerobic conditions to occur and the proliferation of anaerobic bacteria with a resultant loss of N by denitrification. Similarly the fumigation technique may have been ineffective in some of the samples because saturated soil could have protected some of the anaerobic microbial population from the effects of the fumigation. These would then have grown in size following incubation at 25°C for 5 days. C would therefore be lost due to the continued respiration of the microbial population. These factors could explain why the microbial biomass C was so variable for samples of a replicate which did not therefore yield statistical differences between sites and land uses.

Results from this study show that the microbial biomass contributed between 1 and 1.4 % of the C in the soil, and between 0.75 and 1 % of the N in the soil of the three land uses. This is the lower limit according to several authors (Paul and Clark 1989; Srivastava and Singh 1989; Scholes and Walker 1993; Prasad *et al* 1994; Hofstede 1995).

There is general agreement in the literature that soil C, and similarly soil N, in the microbial biomass ranges from 1 to 5% (mean 2 - 3%, Jenkinson and Ladd 1981). Microbial biomass C range from 10 to 200 $g/m^2$ in semi-arid savannas (Scholes and Walker 1993). Srivastava and Lal (1994) state that microbial C , N, and P were significantly related with each other and indicated that N and P concentrations in the biomass are influenced by N and P pools in the soil. Brookes *et al* (1984) and Srivastava and Singh (1988, 1989) found close relations between the microbial biomass C and N pools.

This study showed that only a weak correlation could be found for microbial biomass N and C, and this may possibly reinforce the claim that C was lost from the system owing to methodological problems with fumigating the microbial biomass.

Groffman *et al* (1996) report values in the region of 61, 92 to 100 mg C/kg soil for the microbial biomass C of three locations including sites with loamy sand soils (15 - 40% fines), silt loam soils (75 - 95% fines), and a silt loam soil (55 - 85% fines) respectively. Land uses in this study ranged from 40.21 to 236.64 mg C/kg for a sandy (< 10% silt + clay) and a sandy clay loam with 41% fines. A soil with a clay texture, i.e. Mkuze cultivated, had a percentage fines of 84% and a microbial biomass C value of 171.86 mg C/kg which is almost double that reported in Groffman *et al* (1996).

A possible explanation for the apparent accumulation of microbial C and N in fine textured soils would be that of a protective effect. Clay soils generally have greater less than 1 μm in equivalent diameter (Greenland *et al* 1992). These properties facilitate the accumulation of organic matter, of which approximately 45% is C. The greater cation and anion exchange surfaces would also facilitate the adsorption of ions such as ammonium and nitrate respectively. In addition clay soils have a greater propensity to "hold" water and therefore remain moist for longer periods. The greater cation exchange capacity of vertic soils also provide a buffer for pH changes, i.e. Figure 2.9 where the pH was positively correlated with the percentage fines; while clay soils protect micro-organisms against microherbivory. All these properties will aid the growth of microbial populations as adequate substrate and abiotic conditions exist (Wardle 1992).

*Light fraction C, N, and mass*

The mass of light fraction, i.e. the organic material in the size range of 250 μm to 2 mm in diameter, has been previously regarded as an important index of soil fertility and sustainability (Greenland and Ford 1964, Janzen 1987 in Wander *et al* 1994). Thus as the light fraction increases in the soil, the more fertile the soil is likely to be. This is because the light fraction is recognised to be an important biological substrate (Wander *et al* 1994).

Jones (1973) states that cultivation is a powerful determinant on the level of OM in the soil. He stated that soils under cultivation have a mean C content which is a little more than half that of soils which do not appear to have been cultivated. Similar conclusions were developed by Woomer (1992) who investigated carbon fluxes in woody savannas in southern Africa. The conversion of natural forests to annual field crops results in the massive and unavoidable loss of biomass C. This study also showed that significant differences occurred between the amount of C in the light fraction, but not microbial biomass, of sites and land uses. Vertic sites and those that were unaltered by land use generally had the greatest amount of soil C.

The total soil organic matter content depends on the agricultural use of the land (Quiroga *et al* 1996). Therefore for any given clay content, the level of OM was lower in soils under long term tillage, than in soils subject to rotation or in virgin soils. No significant relationships were found between the COM and clay content where COM is defined as coarse organic matter. This is roughly equivalent to the light fraction mass, and represented 16% of total OM for fine textures and 28% for coarse textures of the soils investigated in their samples.

Consequently the COM fraction showed the greatest amount of change among the soil management systems and they concluded that the combined effect of soil management and particle size distribution can result in significant differences in the distribution of OM fractions in soils with similar TOM values. The effect of land use, especially cultivation, on the light fraction mass in the soil substantiates such a claim, although a significant relationship existed between the light fraction components and the percentage of fines in the soil.

### 2.4.2 SOM *fractions and their relation with clay*

One of the principal mechanisms responsible for the preservation of organic matter (under conservative tillage systems) seems to be the formation and stabilisation of macroaggregates (Quiroga *et al* 1996). Tisdall and Oades (1982) define macroaggregates, as opposed to microaggregates, as aggregates greater than 250 *um* in diameter. Total organic matter content affects the stability of macroaggregates which are therefore susceptible to changes in soil management techniques. Microaggregate stability is mainly controlled by stable organic matter and therefore depends on intrinsic soil parameters such as texture. The water stability of such microaggregates depends on the persistent organic binding agents and appears to be a characteristic of the soil, independent of management (Tisdall and Oades 1982). However, according to Quiroga *et al* (1996) different management systems applied to soils within the same textural class lead to modifications in the proportion of the SOM fractions. Results show that total organic matter levels are strongly influenced by the clay content of the soils which confirm findings of those of Buschiazzo *et al* (1991). Wander and Traina (1996) state that the fine clay from aggregated soil had significantly higher C contents, than fine clay isolated from loose soil. It would therefore appear that macroaggregates in the soil act as potential sources and sinks for nutrients.

An evaluation of the proportion of C and N in the SOM fractions showed that land use did not significantly alter the proportions between biomes or land uses. This suggests that the relative amount of C and N in the three land uses is the same, although absolute differences between them were apparent. Thus the cultivated land use had significantly smaller amounts of C and N in the various SOM fractions than were in the SOM fractions of the conserved and livestock land uses. An explanation for these trends would include that land use reduces the amount of SOM when the input:output balance is altered. That is, the turnover of the various SOM pools of a land use or biome occur at the same rate and there are no differences in the proportions of the microbial biomass, light fraction mass, and intermediate pools. However differences in the proportions were apparent within biomes which suggests that an important interaction between land use, climate, and SOM turnover determines the C and N dynamics of a system.

The amount of C and N in the soils for the different sites and land uses showed that they were closely correlated with the percentage fines in the soil and tended to increase across an aridity gradient. Land transformations in the form of cultivation also significantly reduced the total C and N pool sizes.

Carter, G.A (1993), investigating the soil C and N contents for different savanna sites, found that the size of the total C and N pools changed with the silt and clay content of the soil at a site. Acidic nutrient poor soil sites had the lowest amounts of total C and N (3000 mg C/kg and 125.16 mg N/kg respectively), and these increased to 31 200 mg C/kg and 1316.76 mg N/kg in the basic nutrient rich soil sites. The respective percentage fines for the two kinds of soils included 21% and 84%. These results are similar to results obtained for the savanna sites in this study. Scholes and Walker (1993) report values of the order of 2500 to 7500 mg C/kg for a semi-arid savanna site (Nylsvley), which had total N contents of between 120 to 450 mg N/kg. Savanna sites in this study had a mean of 10 222 mg C/kg which is comparable to the estimates obtained in other studies (G.A. Carter 1993; McKean 1993).

Several authors, including Greyling *et al* (1990), Du Toit *et al* (1994), Prinsloo *et al* (1990), and Du Preez and Du Toit (1993) show that grasslands sites are estimated to contain from 2600 to 36 340 mg C/kg, i.e. 0.26 to 3.63% C. Estimates for the total N contain show that N can vary between 340 to 1993 mg N/kg (Du Preez and Du Toit 1993). These are very similar to the soil C and N content of the soil types investigated in this project.

Soils with high clay contents often had in excess of 2 to 3 times more C and N than soils with sandy textures. Wander and Traina (1996) suggest that soils with a higher clay content and more organic matter, as in the livestock and conserved sites of this study, appear to have a greater aggregate structure. Consequently they can be inferred to have greater quantities of total soil C and N. Cultivation significantly reduced the quantity of total C and N which would imply that the break down of macroaggregate structure has altered the C and N dynamics of the system.

Tisdall and Oades (1982) state that where soil is cultivated frequently, aggregates are exposed to physical disruption by rapid wetting and raindrop impact as well as to shearing by implements. The net effect is to expose inaccessible organic matter to micro-organisms and to stimulate oxidation and losses of organic matter. This decline in OM is usually accompanied by a decrease in the number of water-stable aggregates. This would occur because OM stabilizes the aggregate by forming and strengthening bonds between domains, i.e. parallel clay particles that are grouped closely together to behave as a unit in water (Emerson 1959, 1977). An idealised model can be built up to show that an aggregate of soil is composed of structural units of various sizes held together by various binding agents. These include those that are transient, temporary and persistent. Cambardella and Elliott (1993) state that the amount of organic C and N in an aggregate size fraction is determined by how much soil is in that fraction more than the concentration of C and N in the soil. It therefore appears that organic matter content is related to aggregate stability, and that the loss of structural stability is related, either directly or indirectly, to the loss of particulate organic matter. A loss of the light fraction mass would therefore result in losses of system C and N as the soil macroaggregate structure is changed by land use, i.e. cultivation or intense livestock production.

Greenland *et al* (1992) state that clay type as well as clay content play a role in retarding the decomposition of OM in soils with humic materials being inaccessible to microbial attack (see Tisdall and Oades 1982). While the light fraction is a more labile fraction of the organic matter than the more humified material, both fractions are involved in mineralization. According to Tisdall and Oades (1986) the relative importance of the fractions vary with the kind of soil and played an important role in interparticle bonding and aggregate stabilization.

The conversion of native forest leads to the reduction of soil structural stability, a decline in the soil organic matter content because of the lower organic substrate input, and altered microbial biomass size, composition and activity (Prasad *et al* 1994). Conversions may enhance the loss of fine soil clay particles resulting in the alteration of the soil physical structure, particularly the reduction in the soil clay fraction (Prasad *et al* 1994). Other parameters such as total soil N and P and microbial biomass also decline with the conversion of forest to cropland (Prasad *et al* 1994). Similarly alteration of the soil structure and texture often leads to a reduction in the microbial activity (Van Veen and Kuikman 1990). These alterations would reduce the availability of binding agents and increase the subsequent losses of clay and the various SOM fractions. Subsequently they may provide an expalnation for the "losses" encountered in the cultivated sites in this study.

Similarly declines in total soil organic matter content has been related to losses of organic C and N from the particulate organic matter or POM fraction (Tiessen and Stewart 1983; Cambardella and Elliott 1992, 1993). Prasad *et al* (1994) showed clearly that the OM content of a soil is the best indicator of its microbial biomass and activity. Since soil microbial parameters are indices of biological stability (Hart *et al* 1989) it can be concluded that land use conversions lead to a significant loss of biological stability.

Trudgill (1977) states that the clay content and type is a key factor related to parent material. Clay is not only a source of nutrients itself, but also determines the degree to which organically bound N and P can accumulate in the soil. There are several abiotic and biotic mechanisms that redistribute clay and nutrients in the landscape, resulting in nutrient rich and nutrient poor patches which are differently exploited by herbivores. Consequently one would expect the SOM fractions to vary according to how the clay content and climate vary over the landscape. These patterns are apparent from these results. Both total C and total soil N were greater in sites which had a greater percentage of fines in the soil, and in sites were the aridity index was high, indicating high mean annual rainfall (Jones 1973). Similarly the light fraction mass increased with the percentage fines in the soil and the aridity index. On the basis of these results (Figure 2.9) sites which have a greater percentage of fines in the soil and a higher aridity index, would have greater quantities of total soil C and N. Sandy soils and sites which have a low aridity index, require greater inputs in the form of fertilizers and/or organic matter amendments.

According to Trudgill (1977) soil characteristics are determined by parent material, age, climate, topography and vegetation. The relative importance of these factors varies greatly along an aridity gradient, but the mechanisms are frequently subtle and interactive. In semi-arid savannas, the key influence of parent material is not through elemental composition, but through clay mineralogy. The amount and type of clay produced determines the amount of N- and P-containing OM that can be accumulated in the soil, and its turnover rate. Jones (1973) however found that the OM status declined with increasing clay content in the vertisol data that was analysed for sites in tropical Africa. The vertic sites investigated in this study possessed the greater mass of light fraction and similarly had greater total C and N contents.

Jones (1973) claims that there are two important factors governing amounts of OM in well drained soils. These were the clay content and a moisture factor related to the length of the wet season, i.e. the mean annual rainfall. Findings suggest that the low levels of OM in savanna soils arise from their predominantly sandy nature and from the relatively low rainfall (Jones 1973). In addition highly significant relationships existed between soil C content and clay and rainfall. Soil C:N was also positively correlated with rainfall and soil N was highly correlated with soil clay content. Scholes (1993) claims that water acts somewhat like a switch in semi-arid systems - "when it is absent, growth ceases; but when it is present, the growth rate is determined by temperature and nutrient supply". The nutrient assimilation of plants is not limited by the inherent uptake capacity of the roots, but by the rate at which the nutrient is supplied to the root surface (Scholes 1993). This highlights the importance of the rate at which nutrients are mineralized in the soil.

### 2.4.3 The impact of land use on SOM fractions in the soil

In general, the relationships between SOM characteristics and management practises is poorly understood (Wander and Traina 1996) owing to the complexity of the continuum of organic materials that make up SOM and by the diversity of factors that regulate it's turnover. Greenland *et al* (1992) state that the rate of return of OM is much lower in cultivated sites. OM will then fall to lower equilibrium levels than in comparable soils with the composition of the OM added to soil influencing its rate of breakdown.
Cultivation generally serves to alter the rate at which the SOM turns over in the soil. Tillage leads to the soil becoming more aerobic and the addition of fertilizers and moisture by irrigation results in the microbial biomass attaining populations of size. An increased incorporation and decomposition of SOM follow after tillage and losses are more likely to increase as more N is mineralized by the microbial biomass. This results in an increase in soil fertility in the short term. However rainfall events will leach away excesses of N in the soil and the losses of N to the atmosphere as a result of nitrification and localised denitrification, will ensue. In addition the macroaggregate structure of the soil will be less stable with the removal of OM that is usually returned to the soil in the form of litter (Tisdall and Oades 1982). Such changes are postulated to increase erosive losses of clay particles, and the system could therefore become less fertile and sustainable (Figure 2.9).

The system as a whole will loose its nutrient capacity if organic matter is not returned at the same rate at which it is lost. This is generally not likely to occur because of the nature of the farming practises in many of the sites visited. Although farmers know of the importance of returning organic residues to the soil (personal communication, Mr van de Pol[2]), e.g. leaving stubble mulch on the surface, this is not often enforced and does not return an adequate substrate supply to the soil. This was apparent in this study from the significantly smaller light fraction mass in the cultivated sites. Therefore instead of farming

[2] Mr Reint van de Pol, Towoomba Agricultural Research Centre, Private Bag X1615, Warmbaths, 0480

with organic matter, farmers generally make use of fertilizers as a means of increasing soil fertility and crop production.

Cultivated sites in this study did not generally show greater quantities of total N and C even though they carry livestock in the fallow and are fertilized. According to (Whitehead 1995) grazed sward returns N in the form of urine, but this, like the addition of N in the form of fertilizers, is usually immediately taken up by the plants and so should not necessarily increase levels in the soil. Returns in the form of dung are not considered to be important in altering the absolute quantities of nutrients in the soil but they are important in redistributing nutrients in the landscape (Scholes 1990; Scholes and Walker 1993). In the longer term though, the greater return of organic N should lead to a higher level of soil organic N under grazing (Hassink and Neeteson 1991).This was not apparent from the results of this study.

# Chapter 3

## Potential rate of nitrogen mineralization

### 3.1 Introduction

The rate of nitrogen mineralization reflects how quickly nitrogen turns over in the soil. With out such turnover plants would be limited by this nutrient which is required for several metabolic processes, including the formation of nucleic acids, proteins, and amino acids, as well as enzymes required for photosynthesis. The principal source of nitrogen for plants is that obtained from the decomposition of organic matter in the soil. This process is dependent on favourable conditions of temperature, moisture, substrate supply, as well as substrate quality. Organisms including soil macro and micro fauna play an active role in decomposing the soil organic matter and mineralizing N. Consequently conditions which inhibit the activity of soil micro-organisms will in turn result in lower levels of decomposition and mineralization.

This chapter aims to quantify how the potential rate of N mineralization varies between land uses and across sites differing in aridity and soil texture.

### 3.2 Material and Method

A suitable method for quantifying the rate of nitrogen mineralization is that of the anaerobic technique proposed by Keeney (1982). The method includes submerging a 10 g soil sample in 26 ml distilled water and incubating it at 40°C for 7 days in a water bath. Soil N availability (potential mineralization) was estimated by quantifying the change in the amount of ammonium over the incubation period where ammonium ($NH_4^+$- N) levels were determined by leaching the ammonium from the sample, using a centrifuge set at 5000 revolutions per minute for 5 minutes, with 0.5 M $K_2SO_4$ and determining the amount of ammonium using the Dorich and Nelson (1983) method. This method is sensitive, precise, and an accurate colorimetric method for direct measurement of $NH_4^+$ - N in extracts (Anderson and Ingram 1993).

### 3.2.1 Data analysis

The description of how the data were analysed is provided in chapter 2 under the heading Data Analysis. One way ANOVA's were performed to determine if land uses differed and then one way ANOVA's for each land use across both biomes were used to determine if sites differed significantly. Regression lines for each land use have been compared with a F test (Sokal and Rolf 1994).

## 3.3 Results

### 3.3.1 Biomes and site differences

The potential rate of N mineralization was not significantly different in the grassland biome than in the savanna biome (-0.53 versus 1.13 mg N/kg/7 days, $F_{(1,88)} = 3.940$, $P = 0.050$). This showed that nitrogen, when averaged for all land uses of the savanna biome, was being immobilized, as indicated by the net negative rate for Nmin. In contrast the rate of Nmin was positive in the grassland biome which showed that N was being mineralized. Sites were not however different ($F_{(10,79)} = 1.557$, $P = 0.135$), which would appear to be a result of the extremely variable results obtained for Nmin (Figure 3.1).

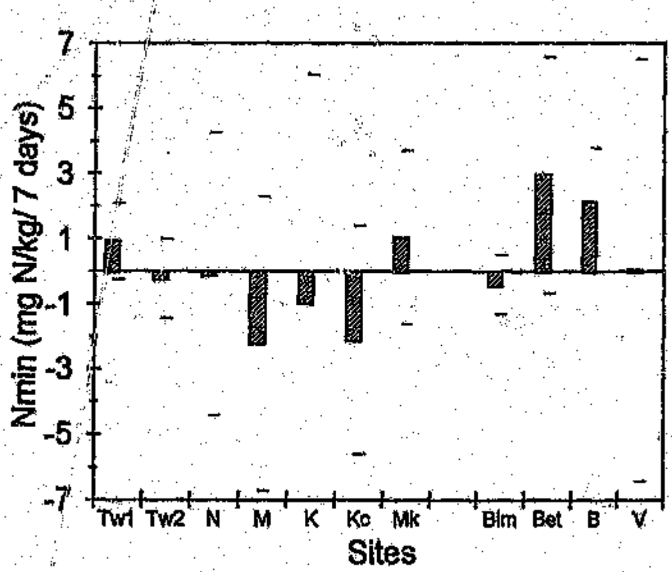



Figure 3.1 The mean (± SD) potential rate of nitrogen mineralization for the 11 sites in the savanna and grassland biome. Savanna sites include Towoomba Hutton (TW1), Towoomba clay (TW2), Nylsvley (N), Messina (M), Klaserie (K), Klaserie communal (Kc), Mkuze (Mk). The grassland sites include Bloemhof (Blm), Bethlehem (Bet), Bethal (B), and Vryheid (V).

### 3.3.2 Comparisons per land use

Figure 3.2 gives the potential rate of N mineralization for the land uses at the sites investigated in the savanna and grassland biome. Significant land use differences were apparent in the way that N was mineralized ($F_{(2,87)} = 10.823$, $P = 0.000$), with the conserved and livestock land uses mineralizing 1.86 mg N/kg/7 days and 0.95 mg N/kg/7 days respectively, and the cultivated land use immobilizing 2.19 mg N/kg/7 days. In both the savanna and grassland biome the cultivated land use had a slower rate of N mineralization and this was usually negative indicating net immobilization of N (grassland $F_{(2,33)} = 6.526$, $P < 0.05$; savanna $F_{(2,33)} = 5.491$, $P < 0.05$).

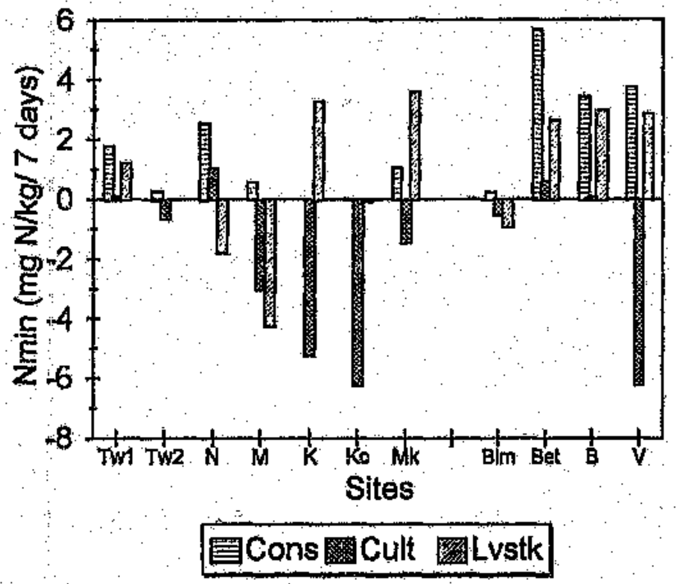



Figure 3.2 The mean (n = 3) potential rate of N mineralization for the land uses at each site of the savanna and grassland biome. Sites as per Fig 3.1.

Comparing the potential rate of Nmin within the sites of each land use type (Figure 3.2) showed that there were no significant differences for any of the land uses. Therefore sites within any one land use type had the same rate of Nmin ($F_{(9,18)} = 1.162$, $P = 0.374$ in the conserved land use; $F_{(10,21)} = 1.413$, $P = 0.241$ for the cultivated land use; and $F_{(9,20)} = 2.283$, $P = 0.060$ for the livestock land use).

A plot of the potential rate of Nmin versus total soil N for the sites of each land use indicated that a significant relation was found (Figure 3.6). This showed that the rate of

Nmin increased as the N content, and C content, in the soil increased. However, the slopes of the regression lines for each land use indicated that land use altered the way in which Nmin increased with the amount of N in the soil (Figure 3.3).

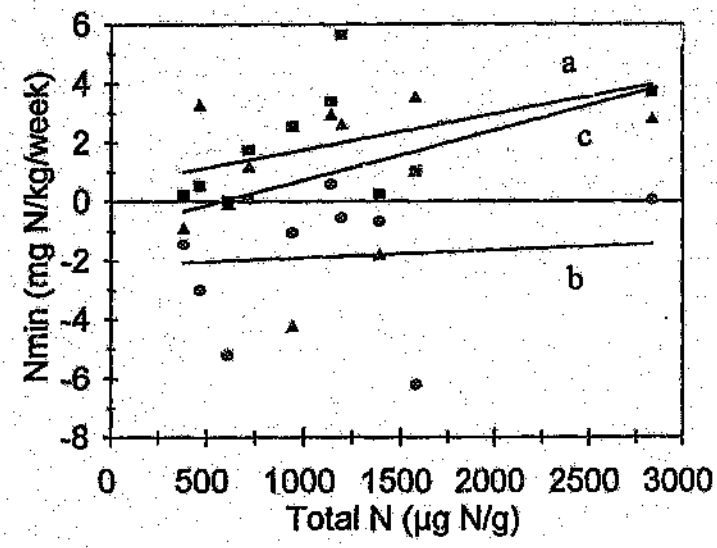



Figure 3.3 The relation between the potential rate of N mineralization and the predictor variable total N. The slope of the regression lines of the conserved (a), cultivated (b) and livestock (c) land uses were significantly different (Table 3.1). Conserved sites (squares), Cultivated sites (circles) and livestock sites (triangles).

Figure 3.3 shows that the slopes of the regression lines for the sites of each land use were all significantly different. Nmin increased more strongly with total N in the conserved sites than it did in sites of the livestock land use. Nmin in the sites of the cultivated land use showed almost no relation with the level of total N. The livestock sites at Bloemhof, Nylsvley, and Messina had increasingly greater rates of N immobilization (-0.90 to -4.21 mg N/kg/7days); and the majority of the cultivated sites also displayed the immobilization of N (-0.54 to -6.21 mg N/kg/7days). The greatest rate of immobilization was for the Klaserie communal and Vryheid cultivated lands.

The relation between the potential rate of Nmin for the three land use types along the textural and aridity gradient showed that land use altered the strength of the association (Figures 3.4 and 3.5). The conserved land use had a greater intercept than either the livestock or the cultivated. However the slope of livestock was generally greater indicating that Nmin changed more rapidly with increases in aridity and texture.

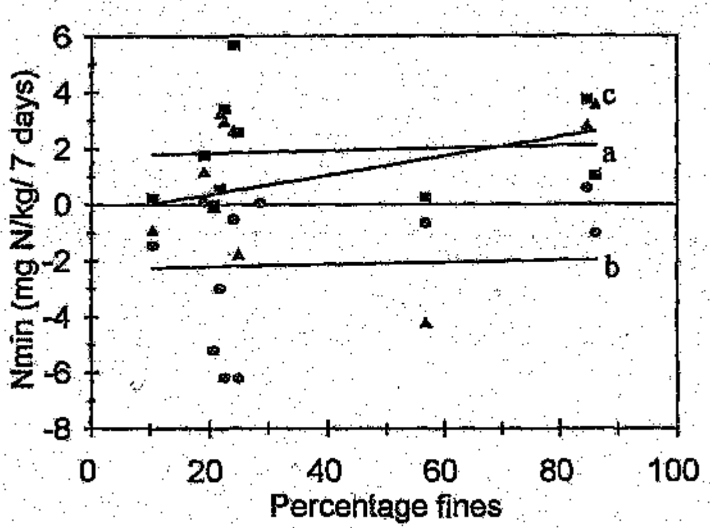



Figure 3.4 The relation between the potential rate of Nmin and the percentage fines in the soil for the sites of each land use types. The regression lines for the three land uses are conserved (a), cultivated (b) and livestock (c) and sites are indicated as conserved sites (squares), cultivated sites (circles), and livestock sites (triangles).

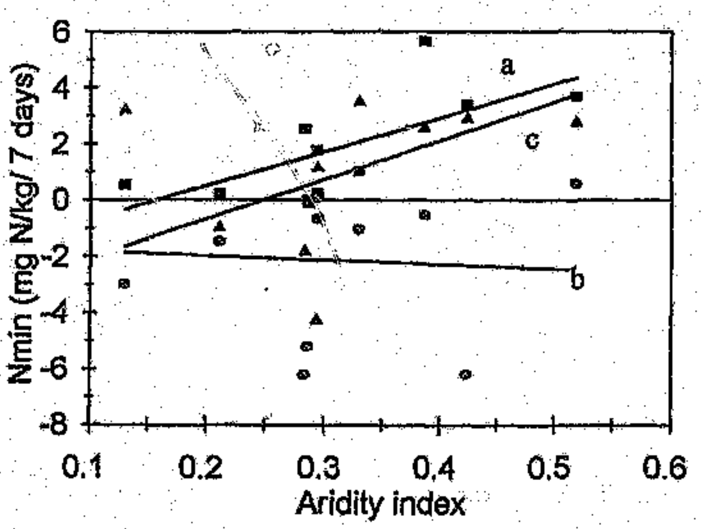



Figure 3.5 The relation between the potential rate of Nmin and the predictor variable aridity index. The regression lines for the three land uses are conserved (a), cultivated (b) and livestock (c) and sites are indicated as conserved sites (squares), cultivated sites (circles), and livestock sites (triangles).

The slopes of the regression lines for the sites of the conserved and livestock land uses were not significantly different for the aridity index, but both differed significantly with the slope of the cultivated land use. This implies that cultivation reduced the potential of the soil to mineralize N in sites which were more moist. In contrast the potential rate of Nmin increased with site moisture for the sites of the conserved and livestock land uses. A similar trend was not however apparent from Figure 3.4 which shows that the potential rate of Nmin at sites of the cultivated land use increased with an increasing percentage fines in the soil. The slope for this regression line was not different to that of the conserved land use, but differed significantly with that of the livestock land use (Table 3.1). Consequently it appears that cultivation alters how Nmin changes with an increase in the percentage fines of the soil, i.e. the potential rate of Nmin is independent of the percentage fines in the soil in sites under cultivation.

A plot of the predictor variables microbial C, light fraction N, percentage fines, aridity index and light fraction mass against the response variable Nmin showed that Nmin did not differ according to how the predictor variables changed in the three particular land use types (Table 3.1).

Table 3.1 A summary of the comparison of the slopes for the regression lines of the three land uses when testing the association between the predictor variables and the response variable Nmin. Means per site have been compared using and F test (n = 31).

| **Predictor variable** | **land use comparison** | **F ratio** | **D.F.** | **Significance** |
|---|---|---|---|---|
| Total C | | | | |
| | con V cult | 10.27 | (9,10) | * |
| | con V lvstk | 20.65 | (9,9) | * |
| | cult V lvstk | 31.97 | (9,10) | * |
| Total N | | | | |
| | con V cult | 40.54 | (9,10) | * |
| | con V lvstk | 54.33 | (9,9) | * |
| | cult V lvstk | 106.54 | (9,10) | * |
| Microbial N | | | | |
| | con V cult | 42.69 | (9,10) | * |
| | con V lvstk | 215.42 | (9,9) | * |
| | cult V lvstk | 25.81 | (9,10) | * |

These results show that Nmin could not be predicted on the basis of a single predictor variable, and implies that several such variables need simultaneously be used to predict how Nmin will change as the predictor variables change (see Chapter 2).

It is therefore not so much site location as it is soil texture, organic matter content, and allied with this, the aridity index of the site which best describes what the rate of Nmin, and the level of soil C and N will be.

### 3.3.3 Comparisons along a gradient

Regressing the potential rate of Nmin against the predictor variables, aridity index and percentage fi[illegible]n the soil, showed that no significant relation existed (Table 3.2). The only significant relations were those between potential Nmin and the predictor variables total N (Figure 3.6), total C and microbial biomass N. However the Pearson correlation coefficient for Nmin and total C was not significant (Table 3.3).

Table 3.2 A summary of the regression analyses investigating the association between the predictor variables and the response variable Nmin ($n = 90$).

| Predictor variable | P value | Coefficient of determination ($r^2$) | Correlation coefficient | Signf. |
|---|---|---|---|---|
| Total C | 0.048 | 0.128 | 0.358 | * |
| Total N | 0.024 | 0.056 | 0.237 | * |
| Soil C:N | 0.516 | 0.005 | 0.069 | NS |
| Microbial C | 0.386 | 0.006 | 0.034 | NS |
| Microbial N | 0.032 | 0.092 | 0.256 | * |
| Light fraction mass | 0.519 | 0.005 | 0.069 | NS |
| Light fraction C | 0.390 | 0.008 | 0.092 | NS |
| Light fraction N | 0.853 | 0.000 | 0.020 | NS |
| Percentage fines | 0.632 | 0.008 | 0.089 | NS |
| Aridity index | 0.067 | 0.111 | 0.333 | NS |
| pH | 0.662 | 0.002 | -0.047 | NS |

* Slope of the regression line significantly different from 0 ($P < 0.05$)

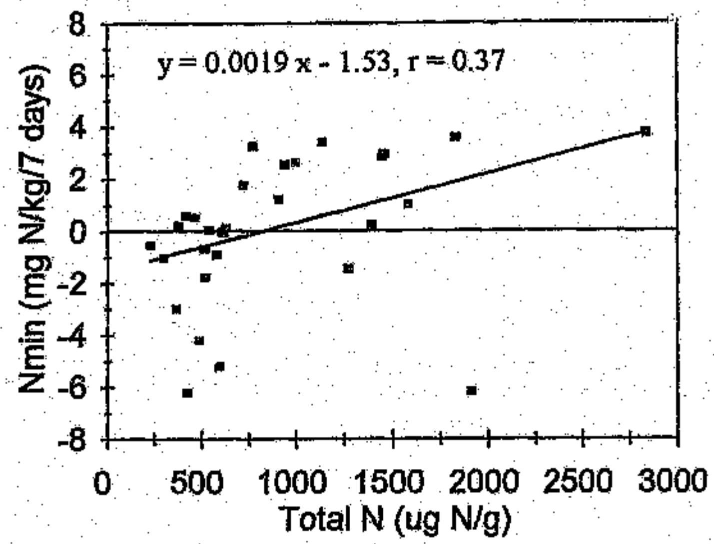



Figure 3.6 The relation between the potential rate of Nmin and the amount of N in the soil for the land uses at each site.

## 3.4 Discussion

### 3.4.1 Controls on N mineralization

The change in the potential rate of N mineralization would be expected to mirror the changes in the quantity and quality of SOM in the system. Thus as the quantity increases and the quality of the substrate increases so to should the potential rate of Nmin. Several authors however have shown that this increase need not be a simple linear response. According to Dendooven *et al* (1995) potential N mineralization ($N_o$) data give different types of curves including those which follow a negative exponential, near linear, and sigmoidal curves. They also found that the time of sampling had an important impact on the amount of N mineralized. In addition the calculation of N mineralization should be carried out with a field specific Nmin constant ($K_{exp}$) for each kind of soil rather than an averaged one valid for all soils. This would improve the estimates for Nmin and possibly show that different soils have different rates of Nmin. Clearly Nmin would therefore be expected to vary for the various sites and land uses investigated in this project.

The increase in the rate of Nmin would appear to be due to the overall increased rate of decomposition. The decomposition of soil organic matter is carried out mainly by the microbial population in the soil. Thus, as more organic matter is incorporated into the soil, i.e. by the action of tilling the soil, or if the microbial population is facilitated by the addition of fertilizers, oxygen, and water; the greater the turnover of the SOM would be and consequently the greater the release of N, i.e. the rate of N mineralization, would be.

The difference between whether a soil is mineralizing or immobilizing N can best be assessed using the MIT or Mineralization, Immobilization Turnover conceptual model. This model purports that the size of the microbial pool largely determines the direction and turnover of N in the soil. Thus when an adequate substrate supply, being of suitable quality, exists in the soil, then the microbial population can be expected to mineralize N. This results in the release of N from the organic matter and the subsequent liberation of N into the soil solution. Such N is then available for uptake by plants. However, if the substrate including components such as light fraction mass, the intermediate fraction in the soil and the organic matter adsorbed onto clay particles (passive fraction), is of limited quantity and inferior quality, i.e. a C:N ratio exceeding 30, then the microbial biomass will immobilize the N that is released by the decomposition process. Although such immobilization prevents plants gaining access to N in the short term, the N that is immobilized into the microbial pool is eventually made available once the microbial biomass undergoes decomposition. This process usually occurs when favourable conditions of moisture and temperature return. The rate of Nmin can therefore vary according to the seasons in association with how mineralization and immobilization change with the climate, but it can also vary with land use effects. This is because land transformations, primarily cultivation, tend to reduce the level of organic matter inputs into the soil (Chapter 2). This reduces the substrate supply for microbial decomposition, and reduces the rate of microbial activity, with the result being a slower rate of Nmin in the long term (Prasad *et al* 1994).

Results presented in this chapter seem to indicate that the MIT model can be used to explain the rate of Nmin for the various sites and land uses. Sites which had a high mass of light fraction associated with the soil, also tended to have a higher rate of Nmin (Figure

2.8). The mass of light fraction was closely correlated with the percentage fines in the soil (Table 2.4). This would imply that sites with a high percentage of fines in the soil would be expected to posses a faster rate of Nmin than soils that were sandy or had a low percentage fines in the soil, and also a lower mass of light fraction. Such a trend was evident (Figure 2.9), but why should this be?

A possible reason for why Nmin is generally greater in more vertic sites would be because these sites have several properties which facilitate Nmin. These include the generally greater store of organic matter that exits because of macroaggregate stability (Cambardella and Elliott 1993), the greater pool of N that occurs because of clay mineralogy (Hadas *et al* 1986; Scott *et al* 1996) and the favourable abiotic factors that enhance microbial activity (Wardle 1992). These include the interaction between soil moisture content and soil temperature (Franzluebbers *et al* 1996), and the physical and chemical buffering capacity of vertic soils which facilitate the development of large microbial populations (Greenland *et al* 1992). Consequently the turnover of SOM is rapid and more N is mineralized.

From the above one would expect the rate of Nmin to be correlated with the aridity gradient in this study. The rate of Nmin tended to increase with the aridity index for the sites indicating that drier sites had a slower rate of Nmin than moister sites. Similar trends were apparent in the way that Nmin varied with total soil C and N. The trend of increasing mineralization with increasing C and N can be explained by the increase of the above-ground, and therefore the below ground, productivity of semi-arid savannas (Scholes 1993, Scholes and Walker, 1993). Figure 3.5 also shows a similar response for Nmin with the aridity index, although the strength of the relationship is not significant. However significant relationships existed for the way that Nmin changed with site aridity of the conserved and livestock land uses, the same was not true for sites of the cultivated land use. Therefore it would appear that the relationship is significantly altered by land use. The reduced mass of light fraction in the soil of the cultivated sites may explain why this is so (Figure 2.9; Chapter 2).

This pattern does not seem to be a function of only soil type as cultivation reduced the rate of Nmin in vertic soils as well. Du Toit and Du Preez (1993) similarly found that the N content of virgin soils increased linearly with increasing fine-silt-plus-clay content ($r = 0.91$; $n = 41$) and exponentially with increasing aridity indices ($r = 0.89$; $n = 41$). They postulated that cultivation caused a significant decrease in the N fertility of soils as demonstrated by the total and mineralizable N declines, and that the decline in mineralizable N followed that of the total N decline. Consequently the impact of land use on the rate and turnover of N and C appears to be a complex process including the way that the microbial biomass, rate of Nmin, light fraction mass and change in the passive pools vary with land use, climate and soil type.

The MIT model also has utility in explaining why the land uses had different rates of Nmin. In the samples of the cultivated sites, N was probably immobilized into the microbial pool. A reason this should occur is because insufficient N was available for the microbial population to release N. The N that would normally be released into the soil solution was instead locked up into the microbial biomass which requires it for growth and metabolism. In contrast soils which displayed positive rates of Nmin indicated soils which contained sufficient SOM for the microbial biomass, that they were able to use what they needed and still leave a supply of available N in the soil solution. Soils which displayed such high rates of Nmin included the heavy textured vertic soils, and soils of the conserved and some livestock sites, where the quantity of organic matter would be sufficient to yield enough N for the microbial biomass and the plants.

According to Whitehead (1995) the rate of mineralization should be greater in sandy soils than in clay and loam soils. This is because on non-sandy soils the C:N ratio and activity of the bacterial biomass will be larger. These conclusions agree with the results of this study where the microbial biomass, and therefore by inference the bacterial activity (Paul and Clark, 1989), were greater on the vertic soils, which also displayed faster rates of potential Nmin. However the vertic soils often had greater rates of potential Nmin. Whitehead (1995) states that Nmin varies in response to soil water and temperature status, as these regulate the response of the microbial population. One would also expect the microbial biomass to be greater in sites with vertic soils owing to these soils having a greater propensity to hold soil moisture, and because they store nutrients (Scholes 1993). One can therefore conclude that increased microbial populations would result in greater rates of Nmin, assuming an adequate supply and quality of substrate. However model results (Parton *et al* $1988_a$, $1988_b$; Chapter 4) indicate that it is the slow pool, of which the light fraction mass is a primary constituent, that regulates the loss of total C and N in cultivated soils. Similar conclusions would hold for soils subject to land use disturbance.

Figure 3.1 shows that the sites investigated in this study did not fit the expected trends according to Whitehead (1995) in the rate of Nmin at each, even when the different land uses where presented individually (Figure 3.2). This may be because Nmin is a dynamic process which means that the microbial biomass has the tendency to become active only when conditions are favourable. This study also only quantified the rate of Nmin at a particular point in time. The method for quantifying the potential rate of Nmin does nevertheless provide a suitable index of the potential of the soil to mineralize N (Keeney 1982), and the method attempts to incubate the samples under favourable conditions of moisture and temperature. The reason Nmin varied so greatly for sites may be more a function of substrate supply than merely the abiotic constraints on microbial activity. Thus Figure 3.6 shows that net Nmin, when plotted with total N, was negative at values of approximately 500 *ug* N/g. These sites have a low percentage of fines, i.e. less than 15%, and may therefore have insufficient SOM for decomposition and mineralization to occur. A reason why the value of 500 *ug* N/g is highlighted is that this value may refer to the N that is tightly complexed in the clay that is present and therefore unavailable as substrate for microbial attack. The oxidative digestion used to measure total N liberates this N which can then be measured and highlights the importance why methods which deal with available forms of N, like the potential rate of Nmin, are more suitable for assessing the impacts of land use on soil fertility and sustainability.

It therefore appears that predicting the rate Nmin depends not only on knowing what the likely abiotic constraints on the microbial population are likely to be, but also on how the substrate supply and quality vary between the sites and land uses, as well as on how soil texture controls the accumulation and turnover of SOM in the soil. This is best examined by means of models which integrate the various factors to determine how the system will respond (Chapter 4).

### 3.4.2 The impact of land use on N mineralization

Mineralization rates fluctuate considerably for different land uses. Maximum daily rates for a grassland soil in the UK range from 1.01 to 3.19 kg N/ha (Gill *et al* 1995). Changes in temperature account for 35% of the variability (Gill *et al* 1995) but there was little significant effect of soil moisture at these sites. The opposite seems to apply to the semi-arid savannas where water provides the switch, and edaphic factors provide the limit to the way that the systems function (Scholes 1993).

With the addition of fertilizers at the cultivated sites, one would have expected to find greater rates of Nmin. This is because the C:N ratios decline, the soil is aerated and often also irrigated thereby increasing the microbial biomass pool and organic matter turnover. Gill *et al* (1995) state that the removal of fertilizer N input to a previously fertilized sward had no effect on mean mineralization rates, but the addition to the previously unfertilized sward, increased mineralization by two fold. Unmodified sites displayed the lowest overall mineralization rate which is in contrast to results in this study (Figure 3.2). The conserved areas sampled in this study were the only sites to exhibit positive rates of Nmin across all sites. In both the livestock and cultivated land uses, the immobilization of N seemed to predominate.

According to Hofstede (1995) the impact of grazing on the soil attributes in the Columbian paramo grasslands were restricted to physical characteristics only. That is soil chemical differences were not related to the associated changes in the above-ground vegetation. This was said even though the decomposition rate was increased by the grazing treatment which removed the above-ground vegetation and thereby increased the soil temperature as a result of vegetative losses of cover. They concluded that the extra nutrients liberated by grazing were not likely to cause measurable changes in the limited ecosystem. An examination of a broader range of habitats (Milcunas and Lauenroth 1993) showed that no relationships existed between the variables aboveground net primary production and changes in root mass, soil organic matter or soil nitrogen. All three below ground variables displayed both positive and negative values in response to grazing (Milcunas and Lauenroth 1993). Data presented here and elsewhere confirm that Nmin is closely correlated with the amount of total N in the soil, and this would suggest that Nmin would similarly be neither positively nor strongly negatively related to these below ground variables. These results both confirm and refute the data obtained in this study where the rate of Nmin was observed to be slightly slower than the rate of mineralization in the conserved land uses. Grazing did however reduce the intercept of the livestock regression line in the plot of Nmin versus the AI (Figure 3.5) and the percentage fines in the soil (Figure 3.4) which suggests that the quantity and quality of the forage will be reduced as sites become drier, i.e. with global climate change (Couteaux *et al* 1995), and as the percentage fines in the soil are reduced as a result of increased erosive losses (Scholes 1993; Scholes and Walker 1993).

# Chapter 4

## Validation of the CENTURY model

### 4.1 Introduction

Simulation models provide the researcher with the opportunity to test hypotheses over space and time. A criticism of the results presented in the previous chapters include that samples reflect system function at only a point in time. This is especially pertinent for this study as nutrient turnover, especially of nitrogen, is a dynamic process. Thus the mineralization and immobilization processes change constantly as a result of changes in substrate supply, as the environment changes from season to season, and with different forms of land use.

The dynamic changes in the mineralization and immobilization processes can often result in a loss of ecosystem sustainability. This would emerge as a result of the loss of nutrient capital typically because of the dis-equilibrium between the inputs and outputs of nutrients, either directly or indirectly in the form of soil organic matter, to the soil (Woomer 1993).

Transformations of the landscape also result in changes to the rates at which processes occur, i.e. with cultivation, the soil is made aerobic, soil organic matter is mixed to deeper layers, and as a result, mineralization rates usually increase. These changes can often make the ecosystem susceptible to losses when the natural buffers in the soil, i.e. the percentage of silt and clay, are diminished as a result of soil erosion or by the removal of ground covers. Sites with lower percentages of fines typically have reduced capacities for nutrient storage and nutrient turnover (Chapter 2).

Owing to the complexity of processes and events which tend to act in unison to change the pool size and flux rates of nutrients in the soil, models are usually employed as a means of simplifying the way in which ecosystems are thought to function. The

CENTURY model which integrates a diverse set of driver variables as a means of estimating the response of the ecosystem attributes following changes in the environment, is such an example. The CENTURY model (Parton *et al* 1987) is a plant/soil model that simulates soil organic matter and plant nutrient (N, P, and S) dynamics in grassland, forest and savanna ecosystems and is suitable for the preliminary evaluation of many land use management strategies (Woomer 1993).

The aim of this chapter is to validate the CENTURY model with reference to the sites sampled in this study. This will allow one to determine how effective the model is in predicting soil C and N dynamics. On the basis of the current management strategies used in each land use, this chapter will provide a discussion on the soil fertility and sustainability of each land use.

## 4.2 Material and Methods

The CENTURY model (Parton *et al* 1987) simulates ecosystem attributes on the basis of several driver variables. This model was selected since it makes use of three SOM fractions, is most applicable for use in different ecosystems, and has climate, soil type or texture, and vegetation type as the driving variables (Parton *et al* 1987 and 1988, Metherell *et al* 1993; Woomer 1993). The model runs on a monthly time step and the major input requirements include monthly temperature and precipitation, initial soil C and nutrient levels, plant lignin and C:nutrient ratios, soil pH and texture and additional nutrient inputs such as atmospheric deposition and biological N fixation (Woomer 1993). The model also contains an output shell that allows one to graphically interpret plots of the various output variables over time. This shell similarly allows one to save output in the form of ASCII text, which can then be used to compare output quantities.

The input of site parameters, notably the percentage of sand, silt, and clay; climatic variables including mean monthly precipitation, mean monthly maximum and minimum temperature, were used to initialize the CENTURY model (site.100 file). Climatic variables for the mean monthly precipitation, mean maximum and minimum temperature for the 6 sites investigated were obtained from the South African Weather Bureau, Pretoria. These extend back a minimum period of 20 years and therefore represent the periodic fluctuations in the climate of the subregion. The soil textural information obtained from this study, and the mean (n = 3) percentage of sand, silt and clay for each land use was used. The same applies for the mean soil pH. Instead of substituting in values for the relevant SOM pools, default values were used. Simulations extending 100 years were used, and the management schedules for each land use type (event.100 file) are shown in Appendix 4. Thus, on the basis of the site file and the event file, nutrient pool sizes will fluctuate depending on whether losses or gains are apparent.

Output variables of the soil organic matter fractions include the amount of C and N ($g/m^2$) for the active, slow and passive pools. The active pool comprises live soil microbes and microbial products with a short turnover time (residence time of ~ 1 year) and therefore is equivalent to the microbial biomass (Chapter 2).The slow pool includes plant and microbial products that are physically protected or biologically resistant to decomposition (residence time of decades) which is equivalent to the light fraction (Chapter 2), while the passive pool can be chemically recalcitrant or physically protected (residence time of 100's to 1000's of years) and includes the intermediate pool (Chapter 2). These definitions are basically equivalent to the pools that were described in Chapter 2, e.g. active pool of CENTURY equals the microbial biomass pool (Chapter 2).

Total soil C and total SOM C and N, but not total soil N as this was not available with this version of CENTURY, were also used. The difference between total soil C and total SOM C is that SOM C does not include the structural (high C:N ratio) and metabolic (low C:N ratio) below ground material, i.e. litter. This is included with total soil C.

Simulations extend a period long enough for the equilibrium level expected for that combination of climate, soil texture and management (the purpose of validation) to be reached.

Sites were classified into functional types based on the results of Chapter 2. The three functional types included sites which were vertic, i.e. had a percentage fines greater than 60%, those that were sandy with a percentage fines less than 10%, and those that fell just under midway of the two extremes and had a percentage fines of 25% (Table 4.1).

Functional types were used because there was nothing to be gained by investigating each individual site.

Table 4.1 The classification of sites into functional types based on the percentage fines or aridity index of each. The site most typical of each functional type was used to model the change in the nutrient load over time using CENTURY.

| **Functional type** | **Site** | **Percentage fines /or aridity index** |
|---|---|---|
| Clay site | Mkuze | 83.2% |
| Sandy loam site | Towoombal | 25.0% |
| Sandy site | Bloemhof | 10.0% |
| Moist site | Vryheid | 0.5 |
| Intermediate site | Bethlehem | 0.3 |
| Dry site | Messina | 0.1 |

## 4.3 Results

Figure 4.1 and 4.2 present the output obtained for four of the functional types of the conserved land use, and generally show the annual change in the variables following a 100 year simulation. The figures show that it took approximately 30 years before equilibrium levels were approached in the case of C, and that there was a continued gradual decline in the case of N. The period preceding the C equilibrium year was characterised by a fairly rapid loss of soil C as a result of the management regime. Since the management regime was standardised for functional types, the same trends are generally shown for the six functional types. However comparison of the slope of the tangent to the line representing total SOM for the first 10 years did show that the rate of change differed according to the functional type (Table 4.2).

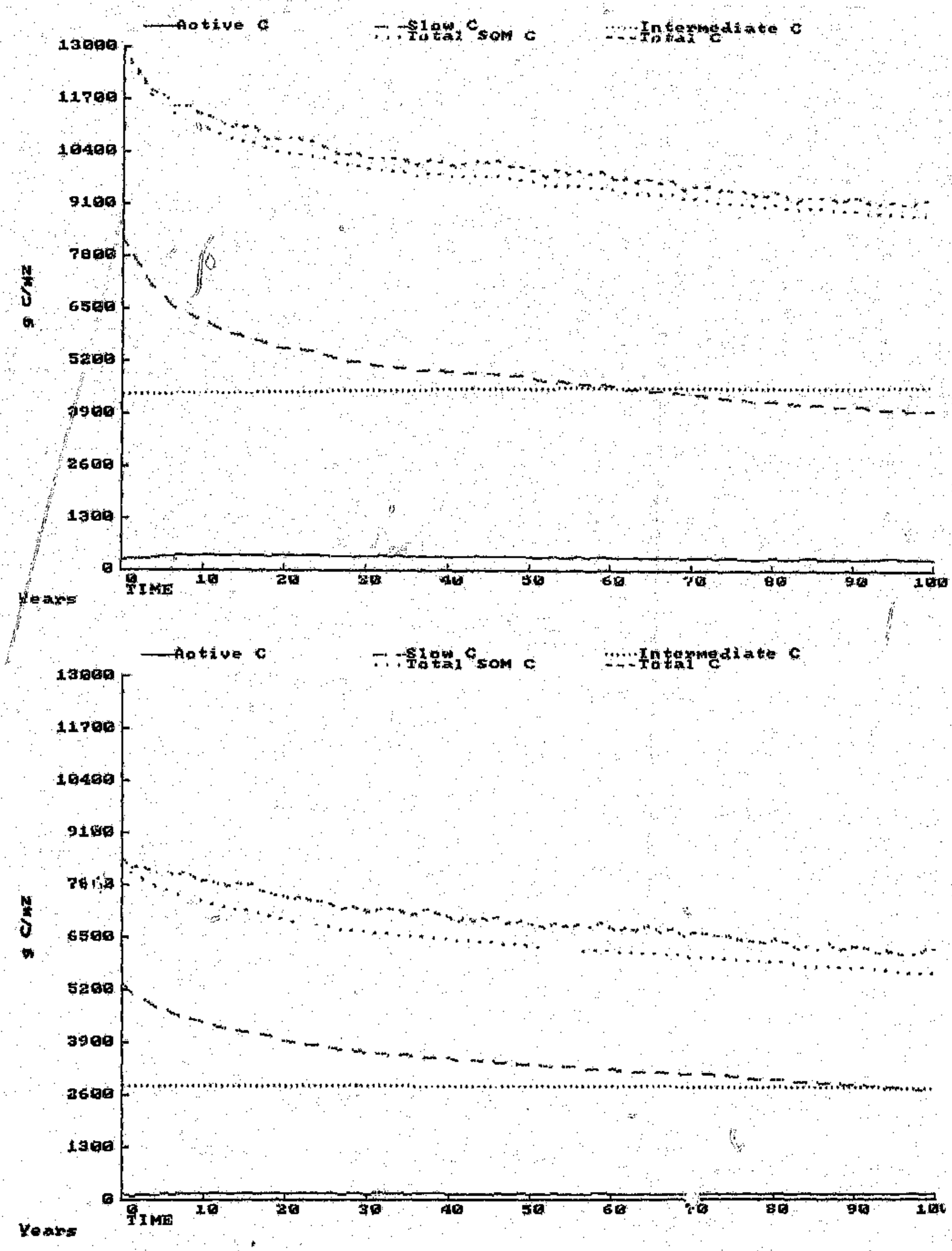



Figure 4.1 CENTURY Model simulation of the C dynamics for the high (top graph) and low (bottom graph) percentage clay functional types of the conserved land use investigated in this study.

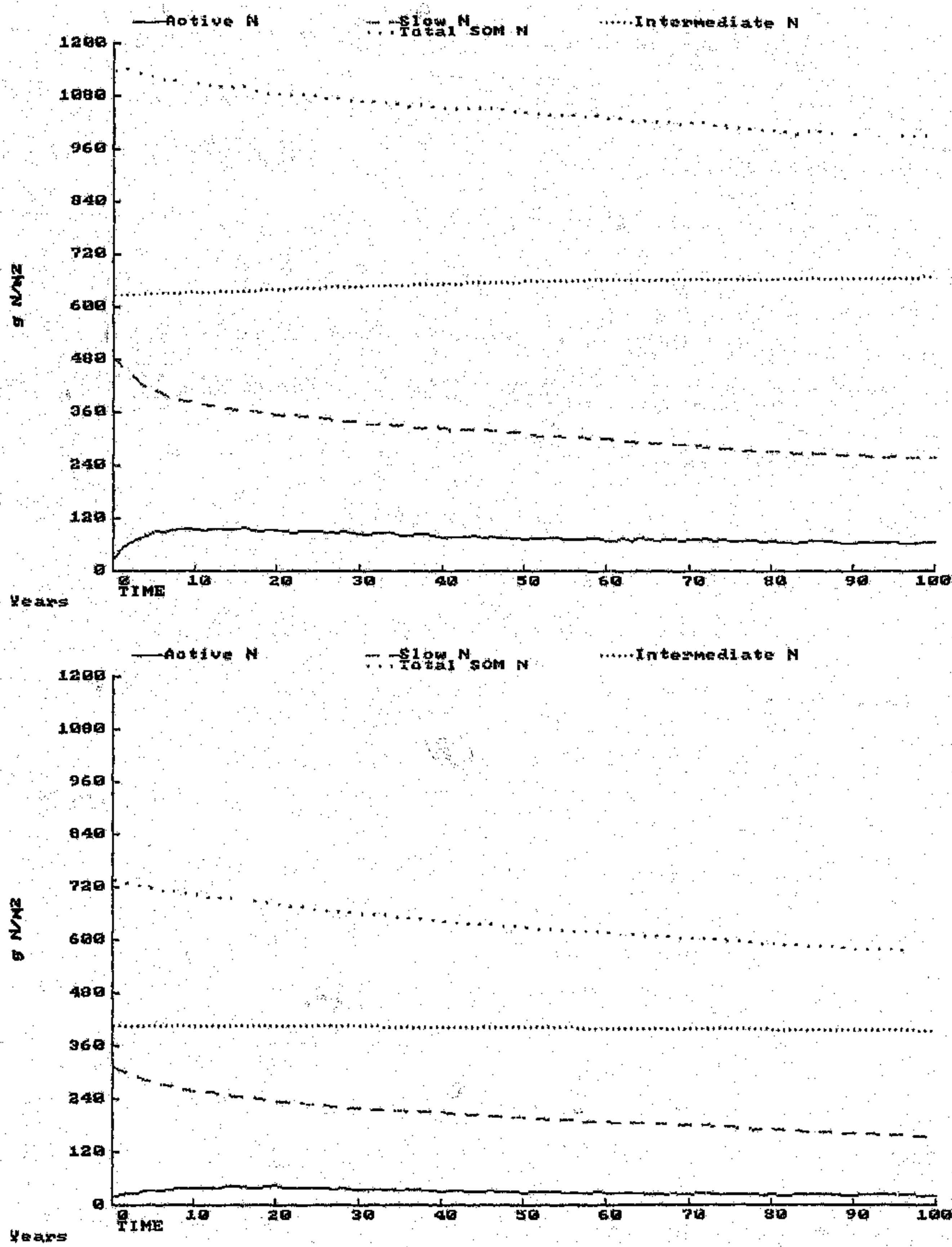



Figure 4.2 CENTURY Model simulation of the N dynamics for the high (top graph) and low (bottom graph) percentage clay functional types of the conserved land use investigated in this study.

Table 4.2 A summary of the changes in the total SOM pool size of C and N in functional types of the conserved land use over the 100 year simulation period, and the rate of nutrient loss from the slow pool during the first 10 years.

| Functional type | % loss of nutrient per 100 years | | Slope of the tangent | |
|---|---|---|---|---|
| | $C/m^2$ | $N/m^2$ | g $C/m^2$/year | g $N/m^2$/year |
| Clay site | 27 | 13 | -203 | -10.5 |
| Sandy loam site | 30 | 25 | -130 | -7.3 |
| Sandy site | 27 | 25 | -96 | -5.8 |
| Moist site | 20 | 8 | -186 | -9.9 |
| Intermediate site | 30 | 25 | -136 | -6.8 |
| Dry site | 28 | 21 | -85 | -5.2 |

**4.3.1 Comparisons of pool sizes and flux rates for the functional types of the conserved land use.**

Model output shows that total C ranged from 8 000 to 13 000 g $C/m^2$ for the conserved land use of the six functional types. The output for the various SOM fractions showed that the microbial biomass C content was always much smaller (25 - 73 times) than the carbon content of the other SOM fractions. The carbon content of the slow pool ranged from 5 200 to 8 900 g $C/m^2$ for the dry and moist functional types, respectively. The amount of C in the passive pool fell mid-way between the slow and active pool sizes at the start of the simulation. However the C content of the slow pool converged to the C content of the passive pool after approximately 60 years indicating losses of approximately 20%. The time to convergence differed for the various soil type and rainfall functional types with it being shortest for vertic soils and longest for sandy soils. Similar convergence times existed for the rainfall functional types which indicates that wet and heavily textured soils loose a greater fraction of SOM as opposed to dry and sandy sites. This would be a function of the SOM content at sites, i.e. sites with greater masses of SOM loose a higher fraction..

Figure 4.2 shows the annual simulated N output for the high and low percentage fines functional types. Total SOM N ranged from 700 to 1200 g $N/m^2$ for the six functional types. Vertic soil functional types had a larger total SOM N pool size than functional types that were sandy. The total SOM N pool gradual declined at a constant rate of approximately -1.58 g N/m$^2$/year over the 100 year simulation time, with the slow pool having the greatest effect on the rate of this decline. Thus where the clay content was greater or the aridity index greater, then the rate of decline would be almost 50%.

The simulated N content in the various SOM fractions showed that the passive pool contained the largest amounts of N (390 to 680 g N/m$^2$), and this pool remained relatively constant or increased in size with simulation time. In contrast the slow pool N content ranged from 300 to 520 g N/m$^2$. This pool however declined by 50 - 60% by the end of the simulation. For the active SOM pool the N content was lower than the slow and passive pool sizes ranging between 10 and 30 g N/m$^2$. This pool tended to increase to 44 g N/m$^2$ and 110 g N/m$^2$ for the sandy and dry functional types, and clay and moist functional types respectively, following approximately 5 - 10 years of simulation. The rate of increase was greater for the moist and clay types than for the sandy and dry types. After the initial increase the active pool remained fairly constant at the new equilibrium level, oscillating with the moisture content of the site (data not presented).

Vertic functional types had the greatest amount of total C, and similarly functional types which were moist, i.e. had a high aridity index, also had the greatest amount of total soil C. Similar trends were evident for the amount of C in the total soil organic pool, although the total SOM pool deviated slightly from the total C pool size over time. The functional types which were intermediate in aridity and clay content, exhibited a 30% loss of the total SOM C and a 30% loss of total SOM N (Table 4.2). The rate at which total C was lost during the first 10 years was greater for vertic sites and declined across all functional types being slowest for the sandy and dry sites (Table 4.2).

A comparison between the results obtained from this study and the simulated output results of the CENTURY model (Table 4.3) show that CENTURY tended to overestimate the pool size for the various SOM fractions and the amount of total C and N. The total C content of the sandy functional type was recorded at 900 g C/m$^2$ whereas the simulated C content was 9.6 times greater than this. Similarly, the total C content for the dry functional types was recorded as 1 230 g C/m$^2$ whereas the simulated C content was 6.5 times greater. Discrepancies exist where the site had a high clay content, and therefore a high light fraction mass (Table 2.3, Chapter 2), where the observed C content was almost twice as great as the simulated C content.

Table 4.3 A comparison between the observed (Obs) and simulated (Simul) results for the conserved land use pools active C, slow C, passive C, and total C. The simulated values are given as at time 0 and therefore indicate the expected pool size for that combination of climate and soil texture with minimal management losses. A bulk density of 1.3 has been used to convert mg C/g to g C/m$^2$.

| **Functional type** | **Active (g C/m$^2$)** | | **Slow (g C/m$^2$)** | | **Passive (g C/m$^2$)** | | **Total (g C/m$^2$)** | |
|---|---|---|---|---|---|---|---|---|
| | **Obs** | **Simul** | **Obs** | **Simul** | **Obs** | **Simul** | **Obs** | **Simul** |
| Clay site | 80 | 236 | 15860 | 8496 | | 4484 | 22152 | 13000 |
| Sandy loam site | 42 | 236 | 2782 | 6018 | | 2655 | 1635 | 9440 |
| Sandy site | 31 | 118 | 2574 | 5428 | | 2950 | 900 | 8614 |
| Moist site | 20 | 236 | 19378 | 8850 | | 4720 | 22152 | 13570 |
| Intermediate site | 65 | 177 | 4693 | 6372 | | 3422 | 3177 | 10030 |
| Dry site | 34 | 118 | 1253 | 5192 | | 2773 | 1230 | 8024 |

Table 4.4 shows contrasting trends for the different SOM N pool fractions. CENTURY overestimated the active pool, compared with the N contents calculated for each site.The calculated content for the slow pool, i.e. the light fraction mass, was a factor greater than that predicted by CENTURY. However the simulated N content for the total SOM was, on average, 3 times greater than the total soil N content of the functional types.

Table 4.4 A comparison between the observed (Obs) and simulated (Simul) results for the N pool sizes of the conserved land use indicating the active, slow, passive, and total SOM pool fractions of the functional types investigated. The simulated values are given as at time 0 and therefore indicate the expected pool size for that combination of climate and soil texture with minimal management losses. A bulk density of 1.3 has been used to convert *ug* N/g to g N/m².

| **Functional type** | **Active (g N/m²)** | | **Slow (g N/m²)** | | **Passive (g N/m²)** | | **Total (g N/m²)** | |
|---|---|---|---|---|---|---|---|---|
| | **Obs** | **Simul** | **Obs** | **Simul** | **Obs** | **Simul** | **Obs** | **Simul*** |
| Clay site | 1.7 | 27 | 8193 | 495 | | 643 | 412 | 1177 |
| Sandy loam site | 2.7 | 16 | 1435 | 354 | | 456 | 187 | 839 |
| Sandy site | 1.3 | 11 | 1549 | 321 | | 414 | 99 | 752 |
| Moist site | 2.3 | 27 | 11661 | 523 | | 676 | 738 | 1200 |
| Intermediate site | 1.1 | 22 | 2824 | 371 | | 480 | 311 | 877 |
| Dry site | 2.0 | 11 | 647 | 305 | | 392 | 121 | 719 |

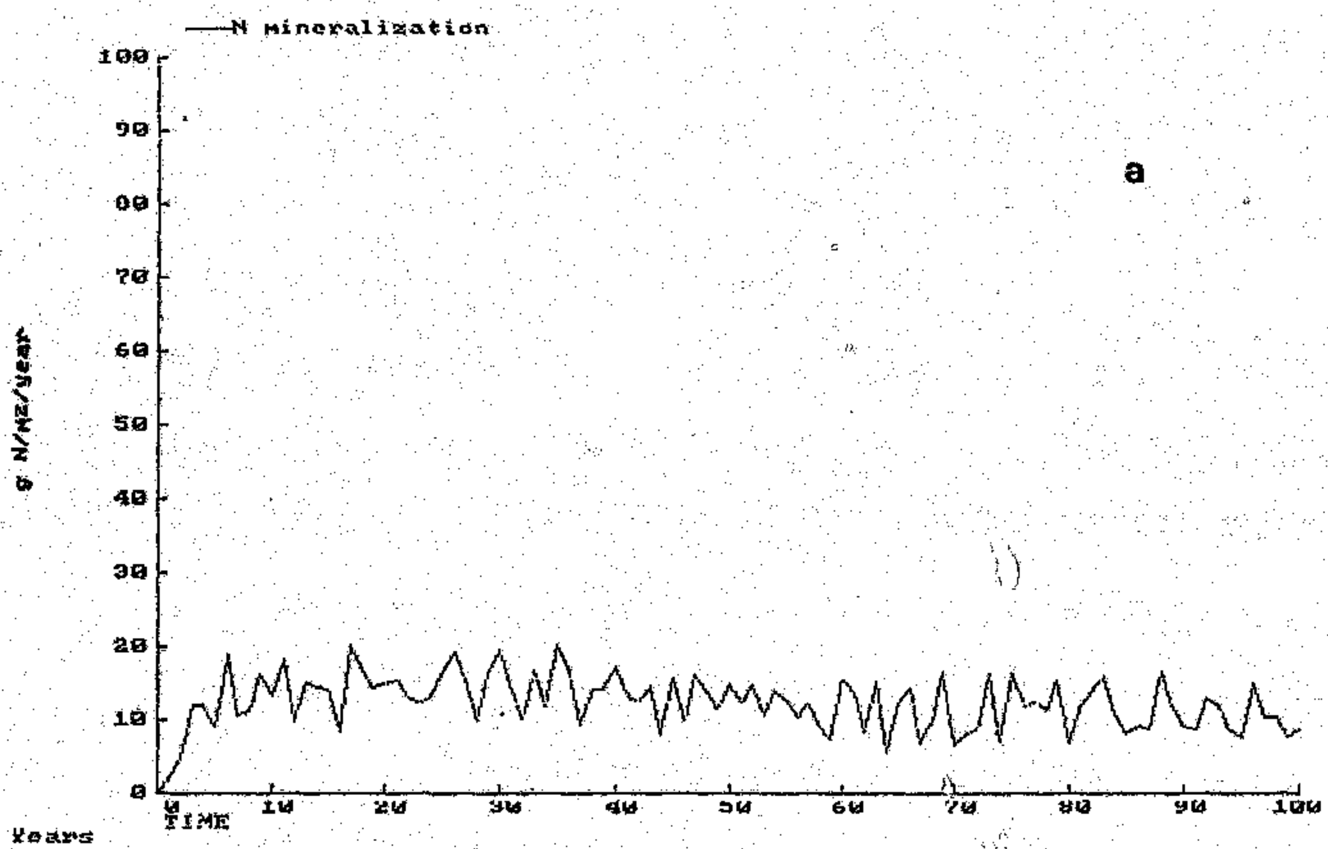
* Total SOM N used as total soil N not available as an output variable

The annual rate of N mineralization for the driest and lowest percentage fines functional types is shown in Figure 4.3. This shows that the rate of Nmin was closely correlated with the precipitation. The wet functional type and the heavier textured functional type had almost 1.5 times greater levels for Nmin. There did not appear to be any decline in the rate of Nmin over the 100 year simulation period, even though large declines in the slow pool SOM fraction were apparent.

### 4.3.3 The effects of land use

**Cultivation**

Generally land use did not alter the initial pool sizes of the various SOM fractions as expected. However at the end of the 100 year simulation period, pool sizes were reduced under the cultivated management regime as compared to the conserved management



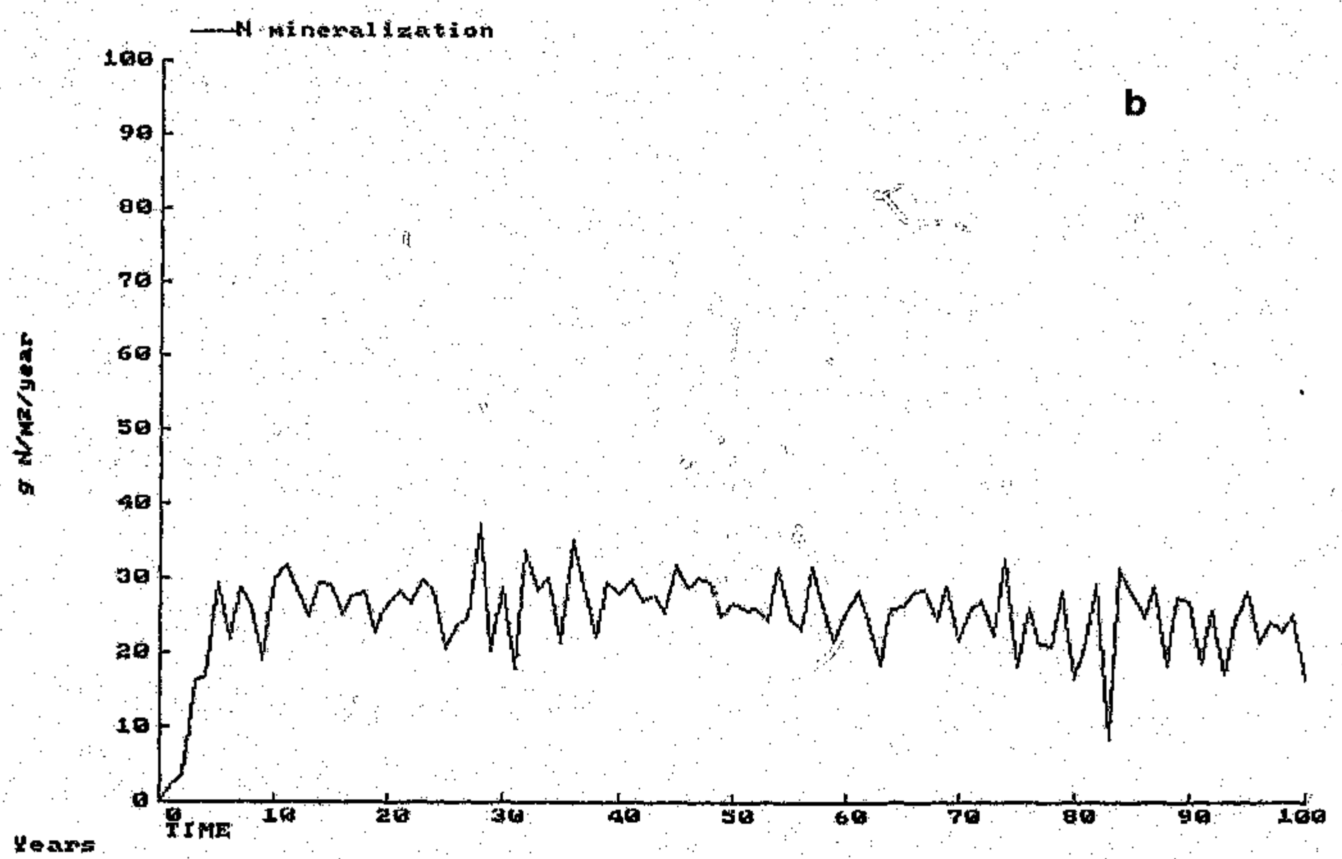
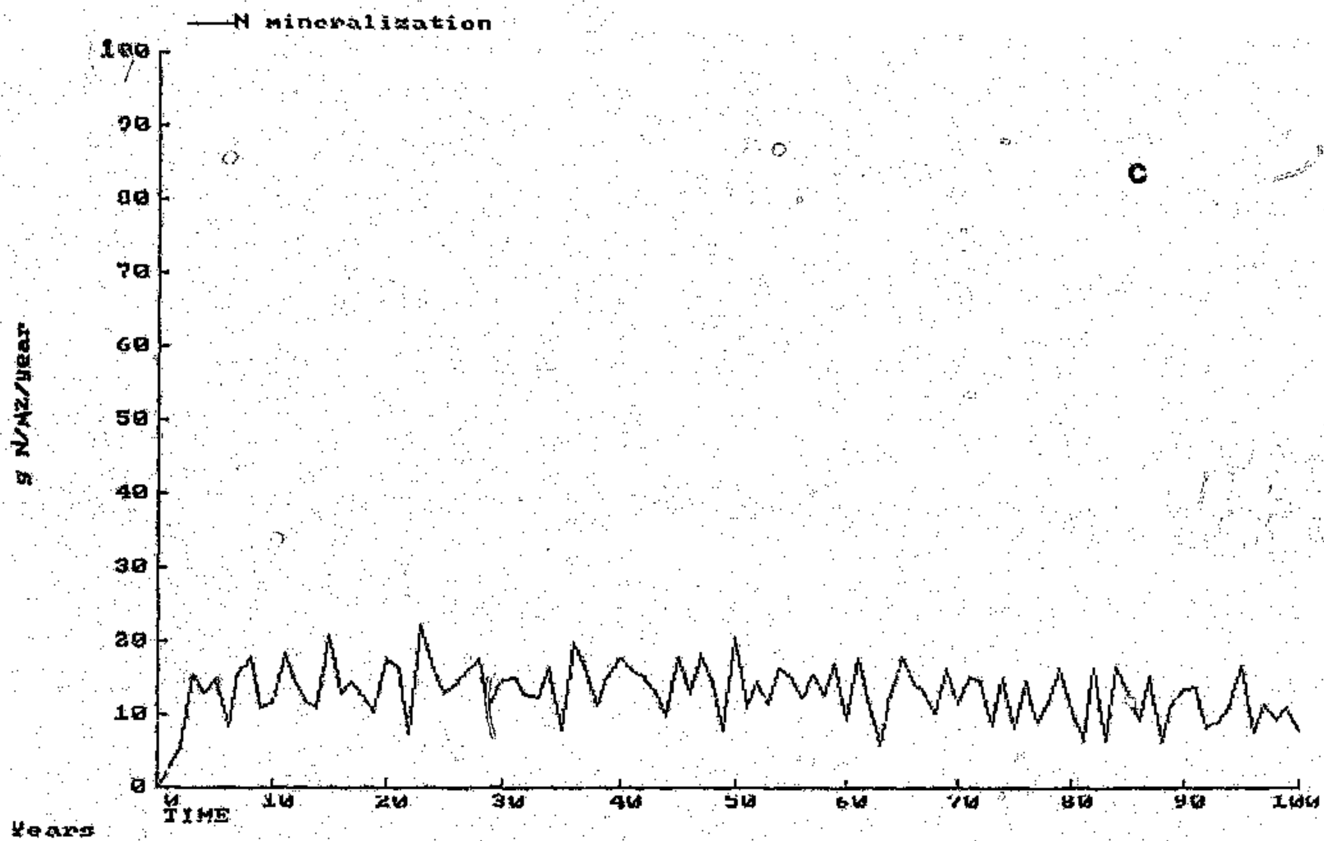
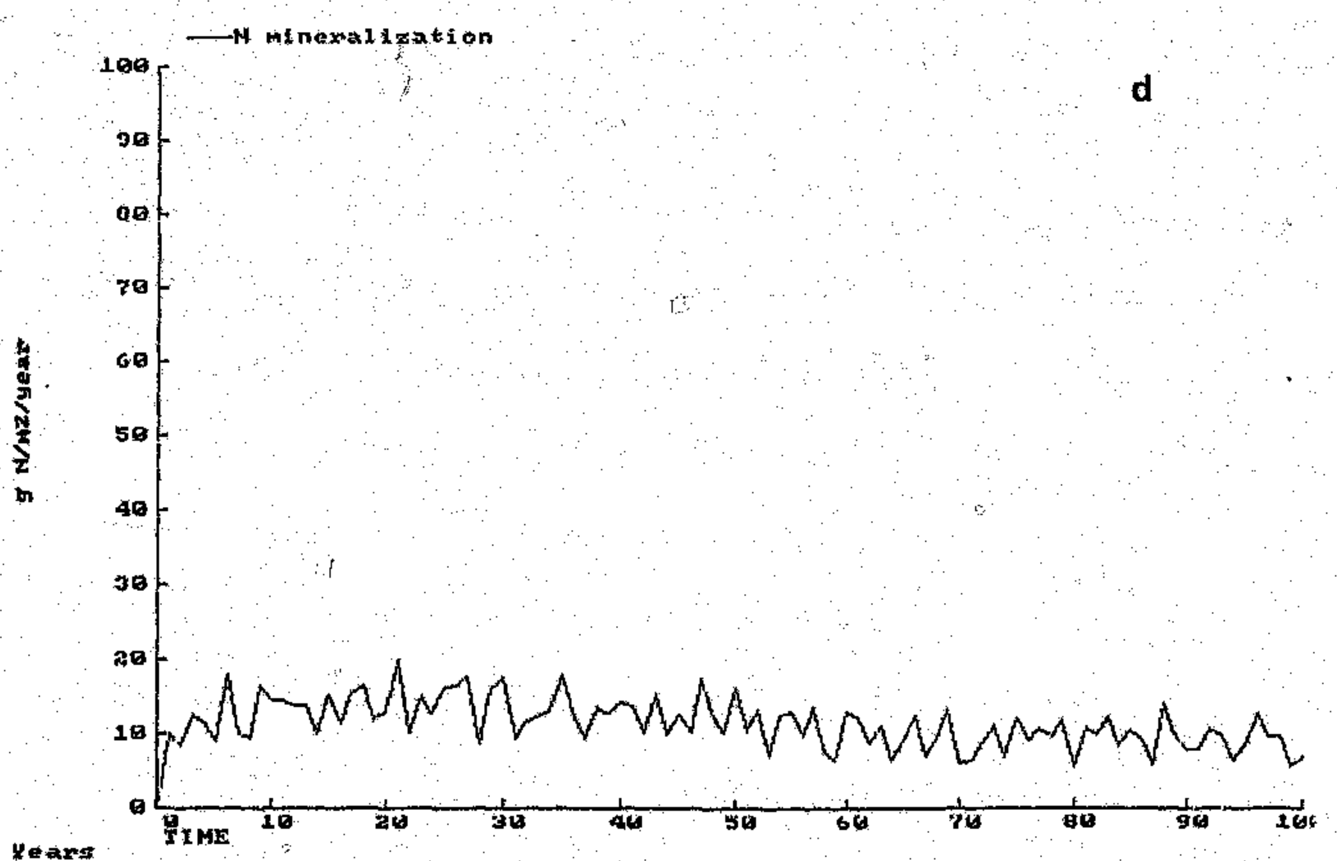




Figure 4.3 CENTURY simulations showing the rate of Nmin for the conserved (a), cultivated (b) and livestock (c) land use of the clay functional types, as well as the conserved sandy functional type (d).



N mineralization
c
g N/M2/year
0
10
20
30
40
50
60
70
80
90
100
TIME
Years

N mineralization
d
g N/M2/year
0
10
20
30
40
50
60
70
80
90
100
TIME
Years

regime (Figure 4.4). This was most clearly seen by the rapid loss of the slow C pool (Table 4.5) and narrowing between the SOM total C content and that of the total soil C content (Figure 4.4 (i)). This would infer a reduction in system C content as a result of cultivation under this management regime (see Appendix 4). A mechanism to explain this would be that cultivated systems became less diversified, or more simplified, which leads to a reduction in the possible pathways that C can be accumulated in the soil.

Table 4.5 A summary of the changes in the total SOM pool size of C and N in functional types of the cultivated land use over the 100 year simulation period, and the rate of nutrient loss from the slow pool during the first 10 years.

| **Functional type** | **% loss of nutrient per 100 years** | | **Slope of the tangent** | |
|---|---|---|---|---|
| | **C/m$^2$** | **N/m$^2$** | **g C/m$^2$/year** | **g N/m$^2$/year** |
| Clay site | 26 | 4 | -214 | -10.5 |
| Sandy loam site | 42 | 23 | -175 | -8.8 |
| Sandy site | 47 | 33 | -163 | -8.9 |
| Moist site | 34 | 13 | -192 | -8.3 |
| Intermediate site | 49 | 31 | -220 | -9.9 |
| Dry site | 37 | 22 | -113 | -6.3 |

The same trends emerge in a comparison of the results obtained from the CENTURY model versus those that were calculated in this study (Table 4.6 & Table 4.7 versus Table 4.3 & 4.4).

Table 4.6 A comparison between the observed (Obs) and simulated (Simul) results for the pools of the cultivated land use indicating the active C, slow C, passive C, and total C. The simulated values are given as at time 0 and therefore indicate the expected pool size for that combination of climate and soil texture with minimal management losses. A bulk density of 1.3 has been used to convert mg C/g to g C/m$^2$.

| Functional type | Active (g C/m$^2$) | | Slow (g C/m$^2$) | | Passive (g C/m$^2$) | | Total (g C/m$^2$) | |
|---|---|---|---|---|---|---|---|---|
| | Obs | Simul | Obs | Simul | Obs | Simul | Obs | Simul |
| Clay site | 45 | 226 | 38828 | 7910 | | 4237 | 6536 | 12543 |
| Sandy loam site | 48 | 169 | 1248 | 6102 | | 3277 | 1412 | 9605 |
| Sandy site | 20 | 169 | 2607 | 5317 | | 2825 | 395 | 8475 |
| Moist site | 48 | 226 | 13759 | 7571 | | 3955 | 8650 | 11921 |
| Intermediate site | 41 | 169 | 717 | 6158 | | 3164 | 1498 | 9266 |
| Dry site | 34 | 113 | 780 | 4915 | | 2599 | 855 | 7684 |

Table 4.7 A comparison between the observed (Obs) and simulated (Simul) results for the N pool sizes of the cultivated land use indicating the active, slow, passive, and total SOM pool fractions of the functional types investigated. The simulated values are given as at time 0 and therefore indicate the expected pool size for that combination of climate and soil texture with minimal management losses. A bulk density of 1.3 has been used to convert *ug* N/g to g N/m$^2$.

| Functional type | Active (g N/m$^2$) | | Slow (g N/m$^2$) | | Passive (g N/m$^2$) | | Total (g N/m$^2$) | |
|---|---|---|---|---|---|---|---|---|
| | Obs | Simul | Obs | Simul | Obs | Simul | Obs | Simul* |
| Clay site | 1.3 | 15 | 26621 | 480 | | 605 | 331 | 1095 |
| Sandy loam site | 1.8 | 13 | 855 | 354 | | 459 | 163 | 810 |
| Sandy site | 1.6 | 16 | 1789 | 313 | | 407 | 61 | 751 |
| Moist site | 2.0 | 26 | 9433 | 438 | | 574 | 497 | 1043 |
| Intermediate site | 1.8 | 15 | 491 | 344 | | 443 | 110 | 782 |
| Dry site | 1.8 | 15 | 536 | 292 | | 375 | 96 | 683 |

* Total SOM N used as total soil N not available as an output variable

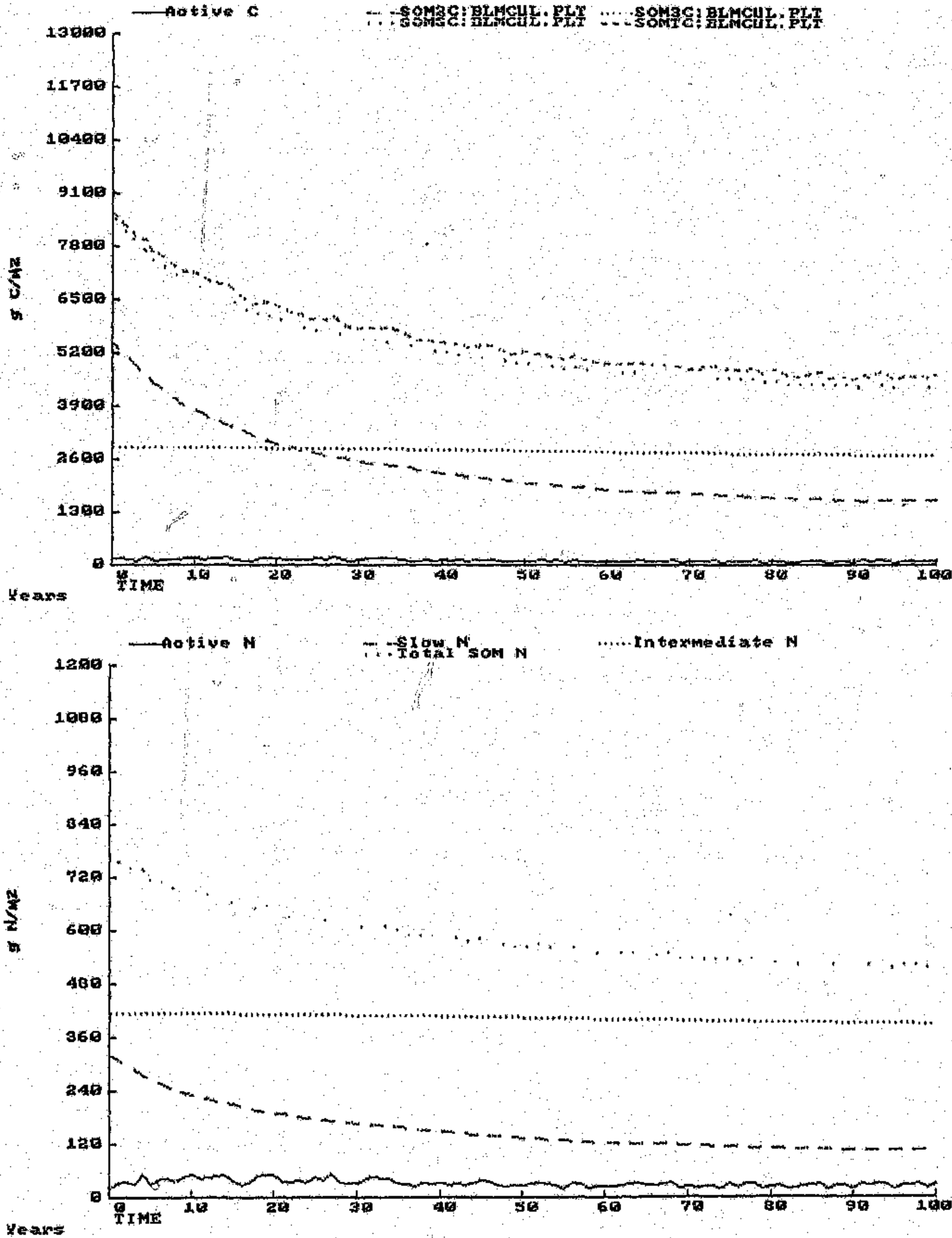



Figure 4.4 CENTURY Model simulation of the C (top graph) and N (bottom graph) dynamics for the low percentage clay functional type of the cultivated land use investigated in this study.

The greatest difference between the simulation of the conserved versus that of the cultivated land use was that the light fraction mass was reduced below the C content of the passive pool more rapidly than the conserved land use. This would be a result of the quicker rate of loss of the slow pool due to cultivation.

**Livestock production**

Figure 4.5 shows the simulated results of the livestock land use using the CENTURY model. These results follow similar trends as the conserved land use (Figures 4.1 and 4.2). The rate of change of the slow pool for the first 10 years are similar for the two land uses as are the percentage differences between the time 0 and time 100 C and N contents.

Tables 4.8 and 4.9 similarly show trends that were apparent in the conserved, and cultivated land use when comparing the simulated output with that of the observed output.

Table 4.8 A comparison between the observed (Obs) and simulated (Simul) results for the livestock pools active C, slow C, passive C, and total C. The simulated values are given as at time 0 and therefore indicate the expected pool size for that combination of climate and soil texture with minimal management losses. A bulk density of 1.3 has been used to convert mg C/g to g C/m$^2$.

| Functional type | Active (g C/m$^2$) | | Slow (g C/m$^2$) | | Passive (g C/m$^2$) | | Total (g C/m$^2$) | |
|---|---|---|---|---|---|---|---|---|
| | Obs | Simul | Obs | Simul | Obs | Simul | Obs | Simul |
| Clay site | 41 | 226 | 15850 | 7853 | | 4181 | 10169 | 12317 |
| Sandy loam site | 39 | 226 | 17685 | 5989 | | 3220 | 2166 | 9492 |
| Sandy site | 30 | 113 | 26509 | 5311 | | 2825 | 1212 | 8362 |
| Moist site | 61 | 169 | 38407 | 6723 | | 3559 | 5782 | 10509 |
| Intermediate site | 59 | 169 | 18013 | 5989 | | 3164 | 2720 | 9605 |
| Dry site | 31 | 113 | 1749 | 4859 | | 2655 | 1063 | 7684 |

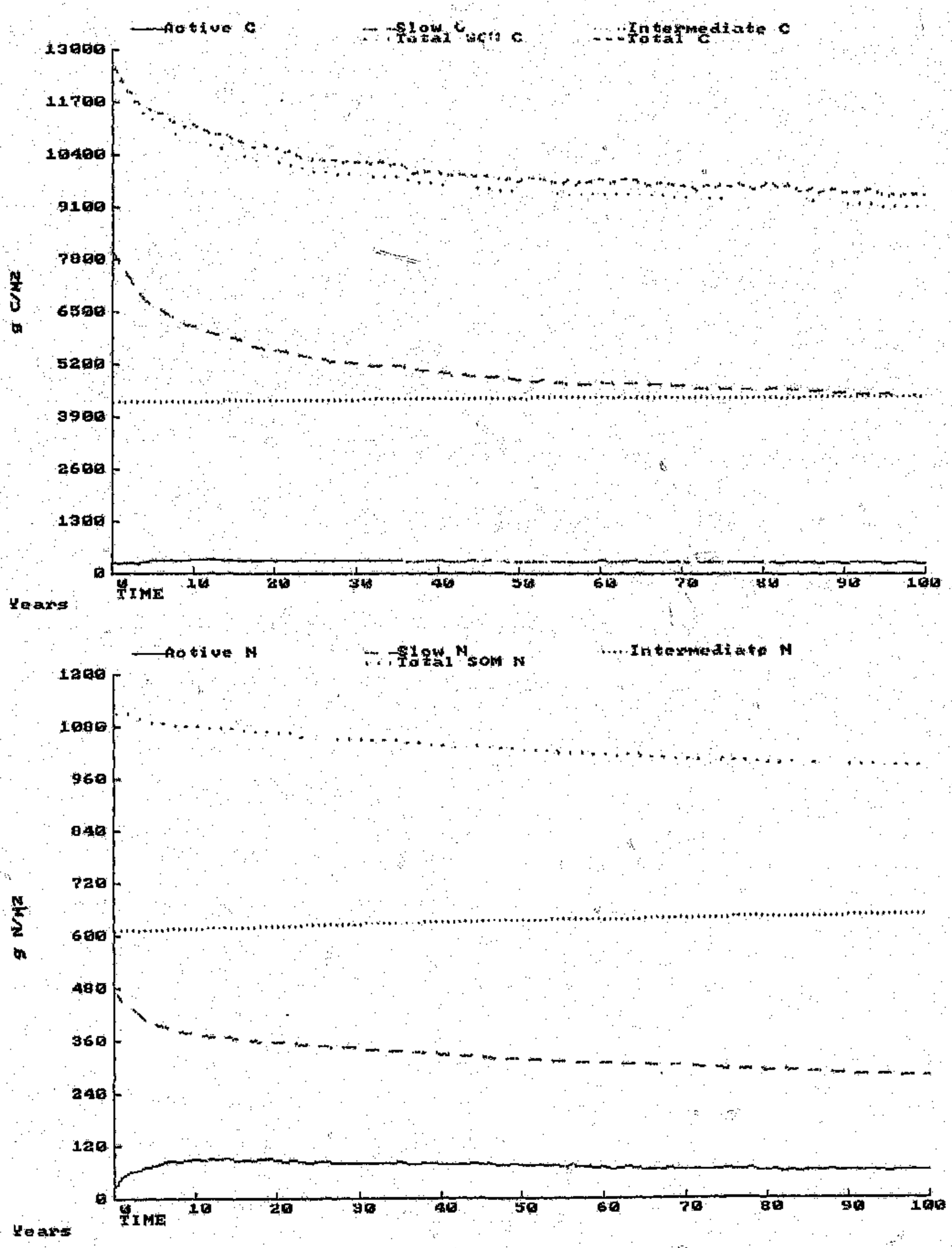



Figure 4.5 CENTURY Model simulation of the C (top graph) and N (bottom graph) dynamics for the high percentage clay functional type of the livestock land use investigated in this study.

Table 4.9 A comparison between the observed (Obs) and simulated (Simul) results for the N pool sizes of the livestock land use indicating the active, slow, passive, and total SOM pool fractions of the functional types investigated. The simulated values are given as at time 0 and therefore indicate the expected pool size for that combination of climate and soil texture with minimal management losses. A bulk density of 1.3 has been used to convert *ug* N/g to g N/m$^2$.

| Functional type | Active (g N/m$^2$) | | Slow (g N/m$^2$) | | Passive (g N/m$^2$) | | Total (g N/m$^2$) | |
|---|---|---|---|---|---|---|---|---|
| | Obs | Simul | Obs | Simul | Obs | Simul | Obs | Simul* |
| Clay site | 3.9 | 20 | 5587 | 464 | | 605 | 476 | 1090 |
| Sandy loam site | 2.3 | 15 | 6235 | 349 | | 454 | 236 | 844 |
| Sandy site | 1.8 | 10 | 8998 | 313 | | 401 | 151 | 730 |
| Moist site | 3.1 | 20 | 13039 | 391 | | 506 | 377 | 923 |
| Intermediate site | 2.6 | 15 | 6115 | 355 | | 454 | 260 | 844 |
| Dry site | 1.8 | 10 | 616 | 292 | | 375 | 128 | 677 |

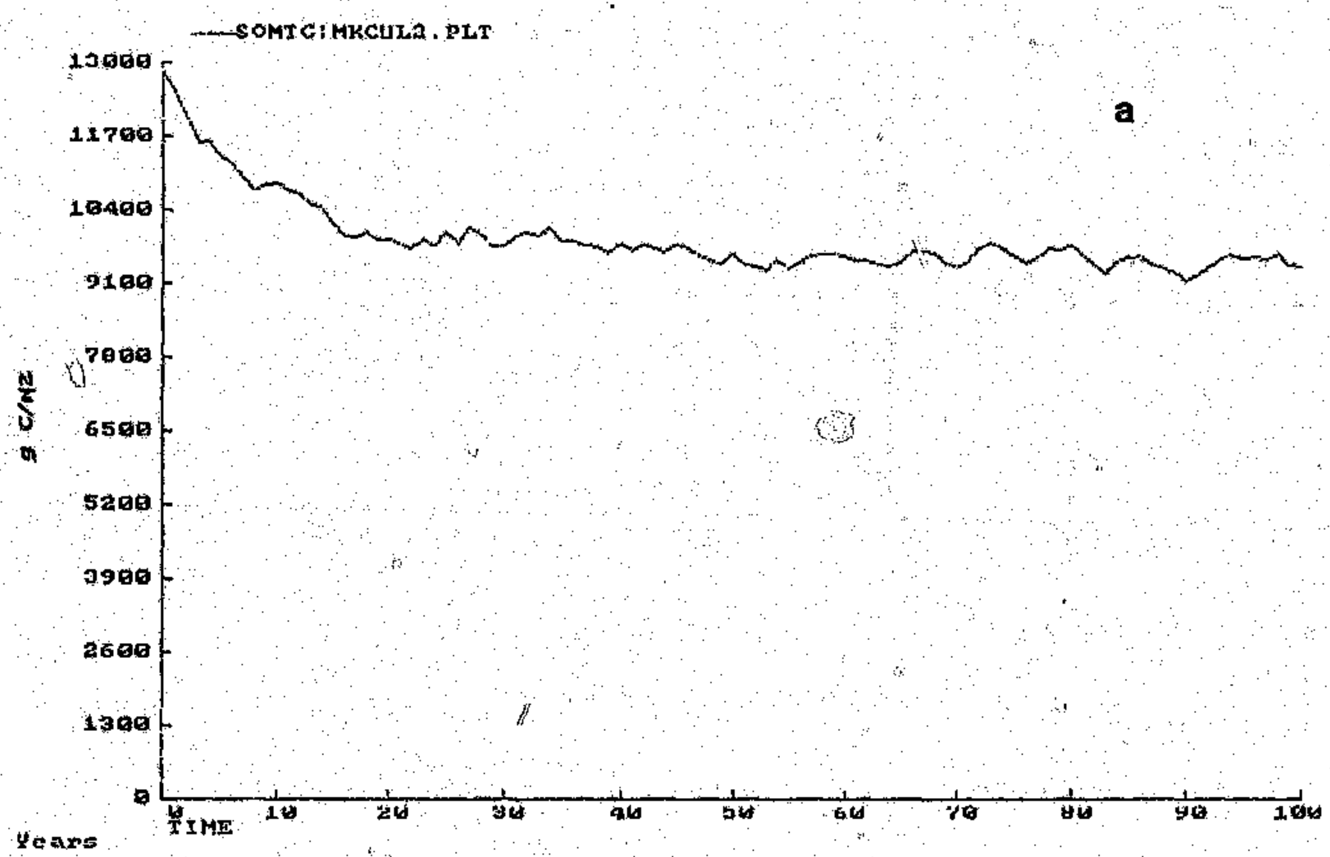
* Total SOM N used as total soil N not available as an output variable

### 4.3.4 Changes in land use intensity

Figures 4.6 and 4.7 illustrate the impact that harvesting two crops in a year, and grazing the veld at low stocking rates, has on the C and N dynamics.

Generally the effect was that two crops did not drastically change C and N dynamics in the clay functional type, but sudden changes became evident by year 70 in the sandy functional type. The reverse was true for the rainfall functional types in which sudden changes in the C and N dynamics occurred at year 70 for the wet type but not the dry type (Figure 4.6).

Figure 4.7 shows the simulation results for the low grazing management regime (similar schedule file as for the heavy grazing management, Appendix E3, but with GH substituted with GL). No differences were observed between the heavy and the light grazing simulations.



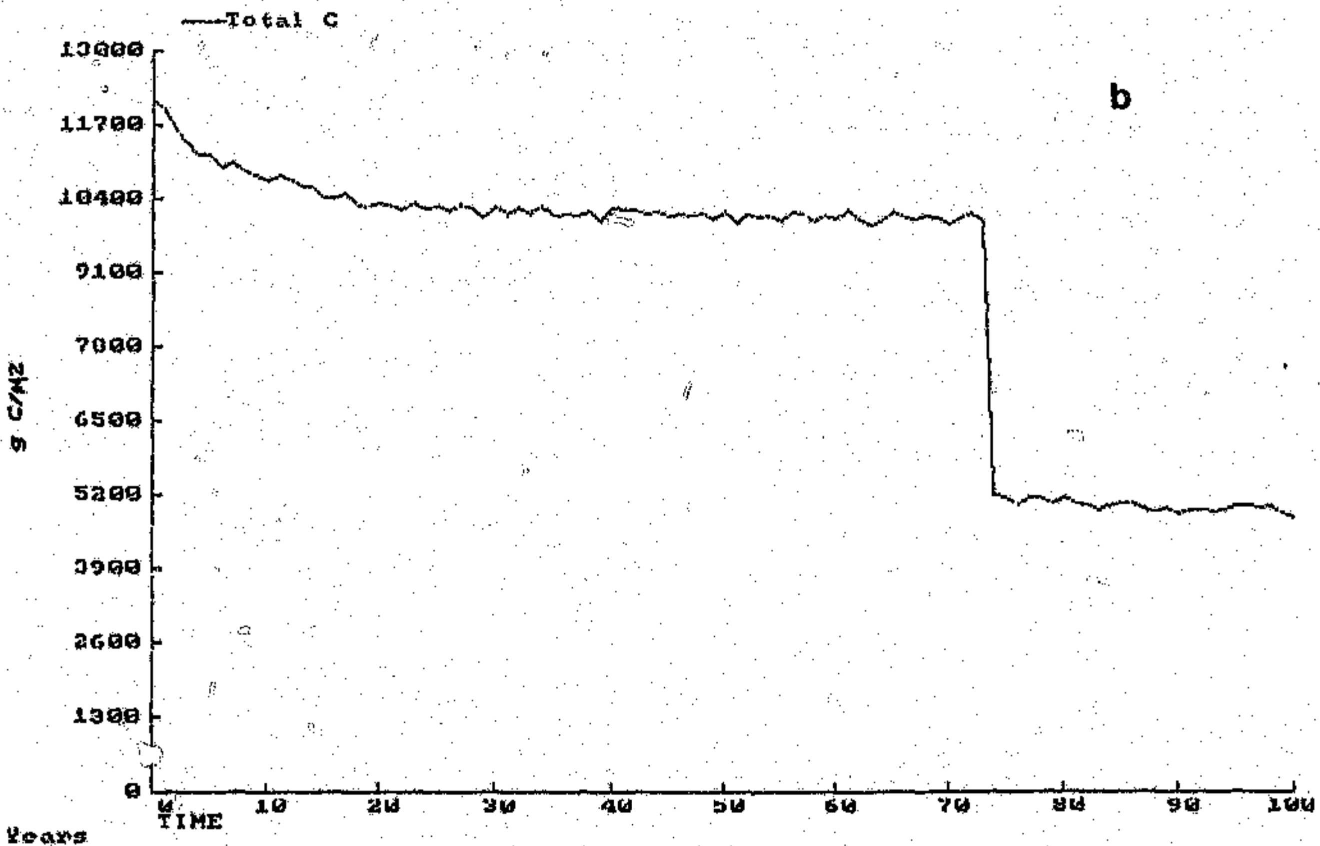
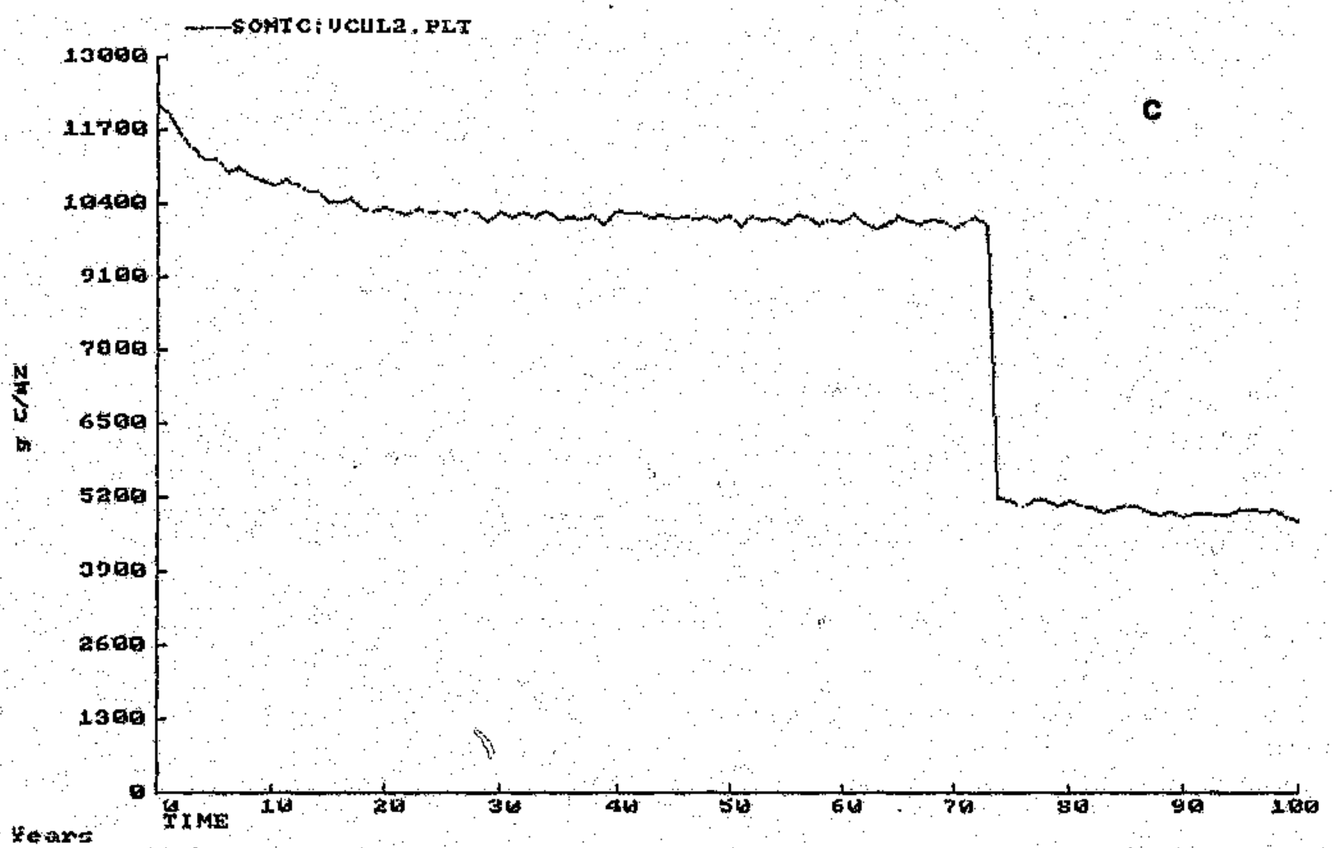




Figure 4.6 CENTURY simulations showing the effect of harvesting two crops a year on the clay (a), sandy (b), wet (c) and dry (d) functional types.



SOMTC; VCUL2.PLT
c
13000
11700
10400
9100
7800
6500
5200
3900
2600
1300
0
g C/M2
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10
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60
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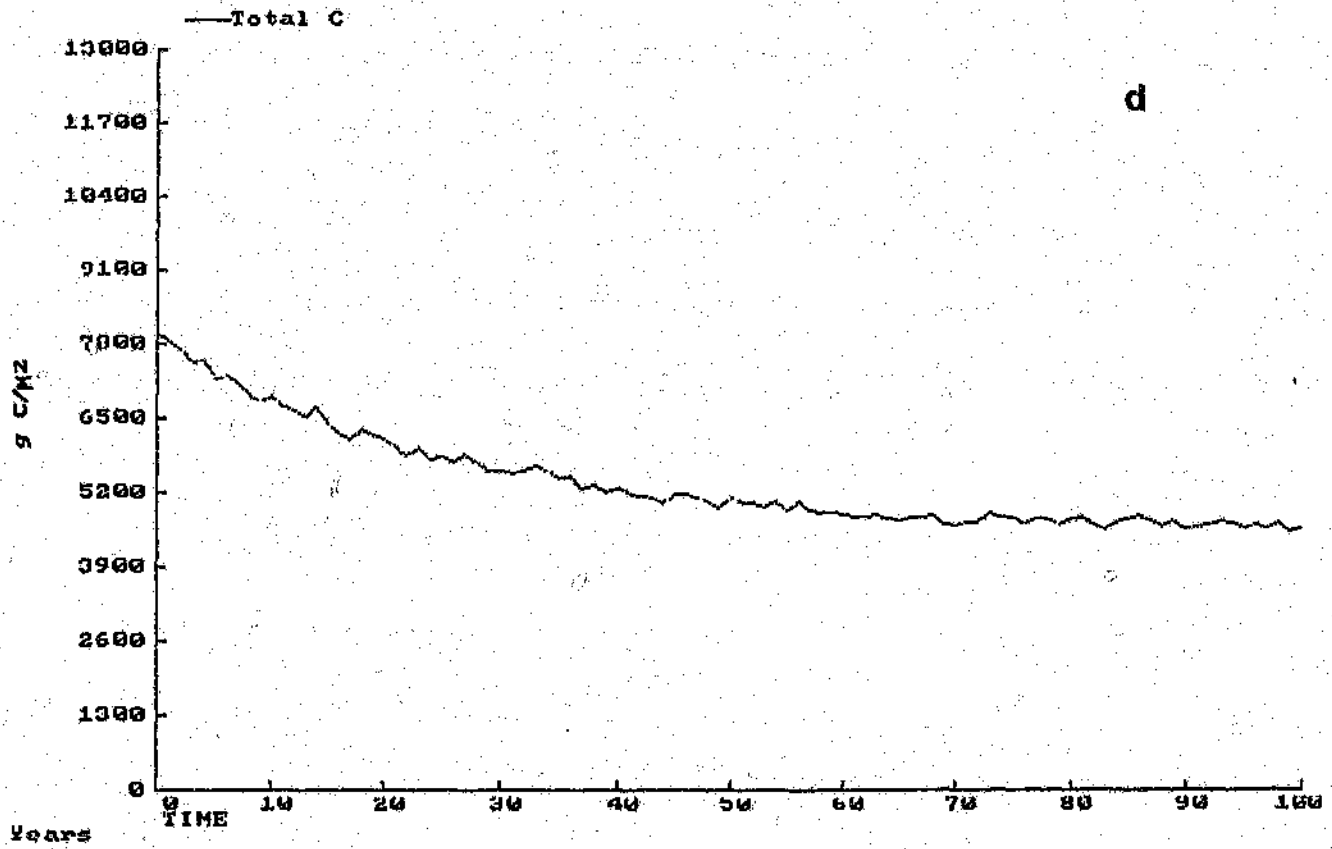
80
90
100
TIME
Years

Total C
d
13000
11700
10400
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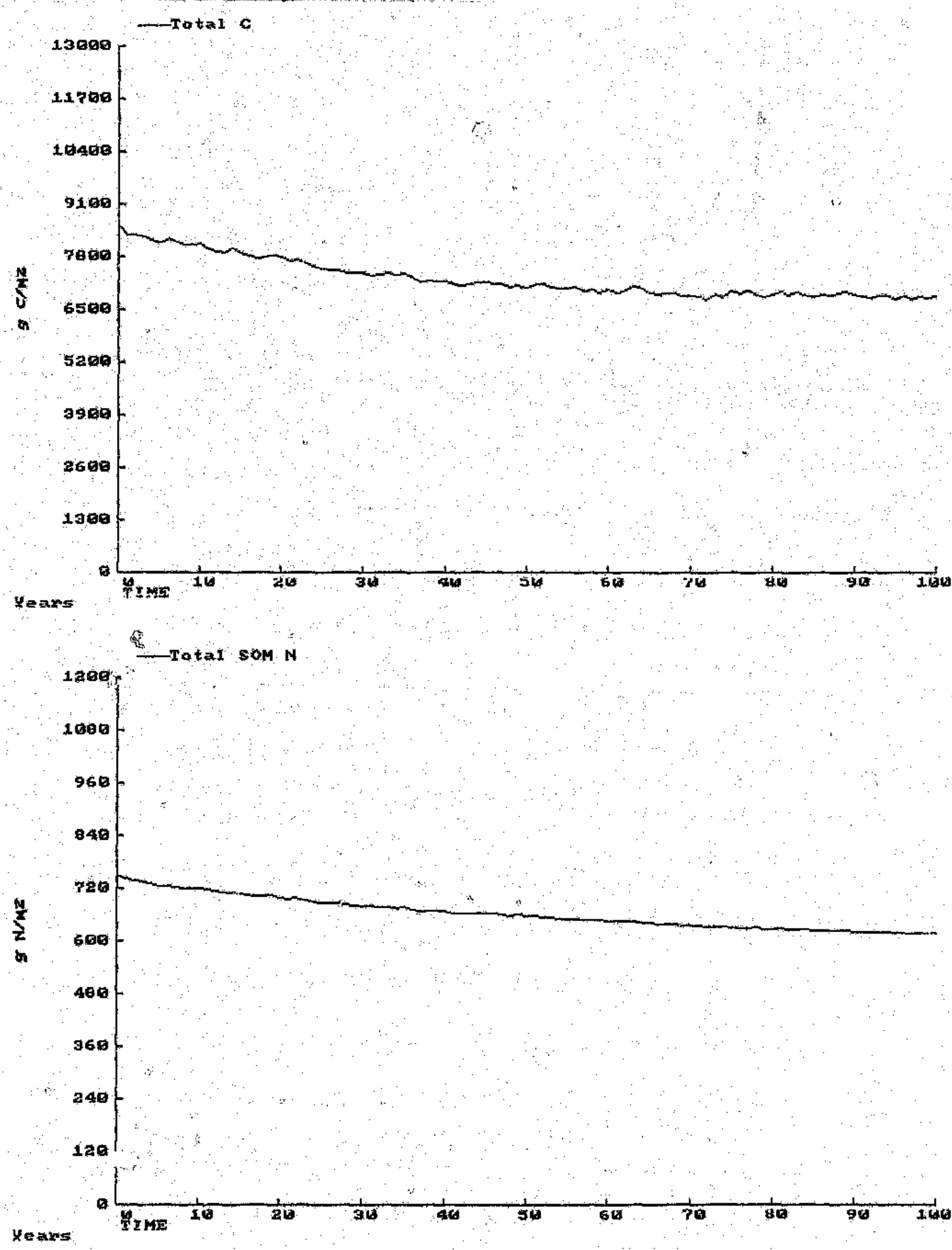


Figure 4.7 Century simulations showing the effect of reduced grazing pressure on total soil C (top graph) and SOM N (bottom graph) over 100 years for the sandy functional type.

## 4.4 Discussion

The results from the CENTURY simulations indicate that the model does not adequately reflect the absolute quantities of soil C and N that were estimated at selected sites in this study. The model did however show similar trends to those that were highlighted in Chapter 2 and 3. These included the important role that soil texture, site aridity, and land use have on the dynamics of soil C and N.

Parton *et al* (1987) in developing the first version of the CENTURY Model found that the model was able to adequately simulate the effects of climatic gradients and soil texture on SOM dynamics for 24 grassland locations in the Great Plains, USA. However the model was recognised to overestimate soil C and N levels for fine textured soils by 10 to 15%.

Model output for all the functional types of the conserved land use, including those that were sandy soils, loamy soils and clay soils, was generally 3 to 8 times greater than the observed estimates. The observed estimates for the fine soils were almost double the simulated estimates. Explanations for this could not be found, and are difficult to explain, even though soil texture is a major control of SOM dynamics in the CENTURY model. However Parton *et al* (1988a) state that the impact of clay on C stabilization will change as a function of clay mineralogy. Preliminary results suggest that the size of the passive pool, a fraction of the total pool, is larger for oxidic and allophanic mineralogy in comparison to kaolinitic and smectitic ones. Allophanic mineralogy is not present in these samples, but oxidic, kaolinitic and smectitic mineralogy is common.

The clays in this study were of the montmorillonite family indicating the presence of 2:1 lattice structures which facilitates a greater cation exchange capacity. This would equate with a greater store of C adsorbed onto the clay particles and therefore a greater pool of C. Similar statements would apply for N, but this still does not explain why such a high amount of C, and not N, was observed for the vertic functional types.

In the case of N simulated values for all functional types were greater than observed values, often by more than 6 times. This strongly contrasts with previous validations of the model, and suggests that an error may be apparent in the way that CENTURY was used. However the only possible source or error is the management regimes modelling each land use, as the appropriate soil textural and climatic variables that drive the model, were an accurate reflection for that site. Additional errors could also occur with incorrect rates of decomposition, leaching, atmospheric inputs such as atmospheric deposition and biological N fixation, C:N ratios, nitrification and denitrification etc. that would aid the accumulation of C and N in the soil.

Parton *et al* (1988b) demonstrated that the model was successful in simulating the impact of cultivation on the formation of soil organic C and N in grassland soils. Soil C and N levels stabilized at approximately 7 500 g C/m$^2$ and 750 g N/m$^2$, respectively, with differences between the simulated and observed results being less than 15%. In this study differences between simulated and observed estimates were often as great as 80%.

Parton *et al* (1988a) found good agreement between the simulated versus observed estimates when comparing the predictions from a large data base, even though CENTURY tended to overestimate losses. These were attributed to significant plant species-level deviations which could similarly apply to this study. Therefore where plants have different ratios of C:N the decomposition rates will differ and subsequently the mineralization of C and, particularly, N will change. According to Parton *et al* (1988a) individual species have quite different ways of partitioning organic matter between metabolic and structural material than the general CENTURY model would predict. It could therefore be that more emphasis was placed on partitioning C and N into the structural pools, and less into the metabolic pool, with a subsequent accumulation of C and N in the system. This is a result of the slower decomposition times and turnover rates of structural materials such as polyphenols, tannins and celluloses (Melillo *et al* 1982).

Paustian *et al* (1992) similarly claim that the CENTURY model provides a good fit to the data and that it can be adequately used to predict changes in the soil C and N pools. Thus changes in soil C and N could be explained by the rate of organic-matter input, its lignin content, C:N ratio, and by the effects of fertilizer additions on below ground C inputs. However the model could not fully explain the additional positive effects of N supply on

SOM accumulation. On the basis of these results they claim that the model supports the main functional relationships that are used to explain the changes in the pool sizes of C and N. A similar conclusion is warranted here as the changes in the C and N content, although not explicitly quantified in this study, do follow what is observed in other systems (Woomer 1993). Thus the change in the pool sizes with increasing percentage fines, increasing site moisture and decreasing land usage, show that C and N levels can be expected to be greater. These decline with increasing land use, decreasing clay content and as the sites become drier, which were the trends evident in the simulations of the various functional types.

For similar climatic regimes, SOM levels increase with increasing C addition and are highest for fine textured soils. For soils where climatic factors are most limiting to decomposition due to low temperatures, or moisture stress, SOM levels are highest per unit of C input. Conversely where decomposition is rapid, steady sate SOM levels per unit of C input are lower (Cole *et al* 1993). The overall trend is that increasing temperature and precipitation will usually enhance decomposition rates and thereby decrease soil C levels if C inputs remain unchanged. These differences were not apparent from the simulations of the wet and dry functional types. The wet site had substantially greater levels of soil C than did the dry site indicating that enhanced decomposition (Cole *et al* 1993) but not a decrease in soil C accumulation was operative. It appears that soil texture and possibly the management regime facilitated the accumulation of C in comparison to the observed C content which was approximately 3 to 8 times smaller.

Parton and Rasmussen (1994) claim that the CENTURY Model (version 2.1) was able to predict soil C and N change within ± 5% when comparing the long term effects of crop management in Wheat-Fallow agriculture. They recommended improvements to the model by incorporating a dynamic plant growth submodel to represent the effect of soil N on C and N allocation to roots, straw and grain. Soil C changes were seen as the result of aboveground C inputs, where an input of 200 $g/m^2$/year was needed to stabilize soil C. They also recommended that the mineralization of N from deeper soil layers (30 - 60 cm) needs to be considered in the N budget. Such changes are assumed to have been incorporated into the model version 4.0 that was used here, and should therefore provide an even better fit to the data, assuming the necessary parameterization is correct and that the observed data are sound. In any event the Model was unable to adequately estimate the amount of C and N in the soil at these sites.

Possible explanations why the soil C and N contents estimated for the study sites deviated so much when compared to the predicted content from CENTURY would include:

1) Different decomposition rates

Paustian *et al* (1992) stated that one of the values for which the model was most sensitive was the parameter $t_{opt}$ in the decomposition/temperature response function. $t_{opt}$ is the temperature at which the maximum specific decomposition rate is reached and is set at 35°C. Higher values tend to underestimate decomposition and therefore overestimate soil C levels. The need to alter model controls on decomposition rates ($t_{opt}$ ) is evidence that the model does not provide a fully general explanation of SOM dynamics (Paustian *et al* 1992). This means that there appeared to be site specific factors that were not explained by the model. However the relative turnover rates and production allocation between the different organic matter pools indicated that the turnover rates, soil C and N levels, and N cycling lend support to model formulations of the effect of quantity and quality of organic matter inputs on SOM dynamics. Similar conclusions could be drawn for the results obtained from this study. If the decomposition rates were modelled as being too rapid then a larger quantity of C would enter the soil, resulting in the large values that were obtained. However even after 30 years, when equilibrium levels were approached, the level of C in the soil was still vastly in excess of what was observed.

2) Incorrect management regimes

The results from these simulations indicated that the kind of management regime did not, by itself, induce substantial changes in the level or dynamics of soil C and N. It would therefore appear that the particular management regime could not be used to explain why such high C and N contents were obtained with the model.

The management regime reflects what would appear to be current practise in South Africa (personal communications and observations). However, the difficulty of modelling these management objectives for each particular functional type meant that several assumptions had to be made (see Appendix 4) and these may not have been appropriate to adequately estimate the C and N content in the soil. These included the use of default values for the C:N ratios, decomposition rate constants, inputs of N via N fixation, atmospheric depositions; losses of N by leaching, denitrification, uptake, and harvest losses, etc. Although many of these parameters have been incorporated into the model system, and global validation accounts for 93% of the variance (Parton *et al* 1988a), some factor may still be required to explain the difference of the soil C and N dynamics of these systems.

### 4.4.1 Implications of land use

Cultivation generally resulted in the greatest losses of C. These were less for functional types which had a greater percentage fines in the soil, and was largely a result of the rapid loss of C from the slow pool of the sandy and moister functional types. Similar conclusions were postulated in Chapter 3 where it was shown that cultivation significantly reduced the light fraction mass, which is a primary constituent of the slow pool. This effect was reduced for soils which were vertic. Parton *et al* (1988a) similarly contend that C losses due to cultivation occur predominantly from slow SOM and these decrease with increasing soil clay content. Field studies (Schimel *et al* 1985) support the conclusion that most cultivation SOM losses are from the slow pool.

When the cultivated management regime was intensified to include two crops within a year, opposing trends emerged from the model (Figure 4.6). The vertic functional type displayed no drastic changes in the level of soil C and N over the 100 year simulation, and it would therefore appear that such sites would be capable of producing two crops every year. In contrast the sandy functional type initially had a greater store of C in the soil, possibly as a result of the inputs of fertilizers. However after 70 years the level of C and N fell drastically to a new lower equilibrium level. This would imply that the continued production of two crops per year would not be sustained at adequate levels to prevent the collapse of the crops without substantial inputs of fertilizer N.

The wet functional type also displayed a rapid decline after 75 years. This is unexpected as the greater precipitation should facilitate a greater net primaty production. However Liebig's law of the minimum may explain that perhaps nutrients are more important.

Grazing, in contrast to other studies (Parton *et al* 1987), did not reduce the quantity of soil C and N with increases in grazing pressure. Using the management regimes for each land use, which reflect current practises in South African farming (Appendix 4), the level of soil C and N was reduced until a lower equilibrium was reached after 30 to 60 years. These losses closely mirrored the losses under the conserved management regime. However when the impact of grazing was reduced to a level which did not cause an impact on the vegetation (Figure 4.7), then the soil C and N levels fluctuated around similar levels to those under heavy grazing. That is, the plot for total SOM C and N was almost identical for the two levels of grazing pressure. The apparent stasis in the level of soil C and N under the two grazing pressures could emerge because the veld was given enough resting time. Tainton (1984) and Du P Bothma (1989) state that a rest period every fourth year facilitates the recovery of the rangeland. This occurs because the input of organic matter to the soil and subsequent decomposition and mineralization, restore the nutrients that were lost due to consumption by cattle. Importance is therefore placed on maintaining the size of the slow pool so that a balance between gains and losses can be reached. However it has recently been proposed that a High Density Grazing strategy (pers. comm[3]. C.R. Hurt, T. Morely and P.J.K. Zacharias) would not lead to drastic shifts in system function. The observation that similar levels of C and N occurred for the low as well as high intensity grazed plots may show that such a strategy can work. The debate surrounding this issue is contentious however with empirical evidence supporting both schools of thought (pers. comm[3], RWS Fynn and T.G. O'Connor; J.E. Danckwerts; K.P. Kirkman). This highlights the increasing need to use models for cross-site and -regional analyses, to compile comparable data sets including consistent measures of SOM quantity and quality, so that models can be tested and further refined.

[3] Programme and Abstracts, Congress 32 of the Grassland Society of Southern Africa, Port Elizabeth, 20 to 23 January 1997.

# Chapter 5

## General discussion and conclusions

The C and N dynamics of systems are strongly controlled by the turnover of soil organic matter. The size of SOM fractions investigated in this study shows that they were under the control of several environmental factors and that land use played an important role in modifying these sizes. This was apparent in the way that land use, particularly cultivation, altered the mass of the light fraction in the soil.

The size of the three SOM pool sizes investigated in this study was shown to be related to the texture and aridity at the site. The sizes were modified by land use, where cultivation generally reduced the size of the pools, whereas livestock production areas and conserved areas had greater levels. A mechanism postulated to explain these patterns and processes included how SOM varied with the percentage silt-plus-clay in the soil, the aridity of the site, and by how cultivation reduced the light fraction mass, otherwise designated the slow pool. Reasons for these patterns include the importance of clay in adsorbing C and N, the presence of favourable conditions for microbial decomposition, the greater plant biomass that occurs in moist sites, and the way that cultivation reduces the light fraction mass through how it may alter macroaggregate structure.

Simulations with the CENTURY model showed that similar trends were evident in the way that the SOM changed in comparison with data collected for each land use management regime, but the model markedly overestimated the pool sizes.

The six soil functional types used to model soil C and N dynamics showed that the pool size of the soil organic matter fractions was greatest for the vertic soils and the moist sites, and was lowest for the sandy soils and dry sites. Similar results were apparent from the samples that were collected from each site. CENTURY showed that the quantity of C and N in the light fraction was greater than that in the intermediate fraction with the lowest contents occurring in the microbial biomass. These results mirror the trends in the data obtained from the samples of the sites, and demonstrates that the CENTURY model would be an appropriate model to use in predicting the carbon and nitrogen dynamics of these systems.

However, large differences emerged between the CENTURY predictions for the size of the pools (almost 3 - 8 times greater), and the estimates obtained by the laboratory techniques. This requires further investigation if the model is going to be used to map regions in South Africa on the basis of the turnover of SOM and N mineralization. A possible error that needs to be addressed is that the incorrect management regime was used which allowed for the accumulation of C and N in the system. Additional errors could also include the parameterisation of variables such as decomposition rate constants, losses by nitrification and denitrification, leachate losses as well as several others. This may reflect the difficulty in obtaining accurate estimates of the rates at which organic materials decompose in the field, i.e. the light fraction, microbial biomass, intermediate fraction and passive fractions of organic materials appear to decompose at different rates according to the refractory content of the material including the content of lignins, cellulose and secondary chemicals. These different SOM fractions can not easily be separated and studied in the field where biotic and abiotic conditions change dynamically. Consequently the rate of breakdown of organic matter and release of nutrients under field conditions versus that predicted by the CENTURY model could be used to explain the differences.

In addition to the impact that land use has on reducing the input of organic matter to the soil, is how a process like N mineralization is changed by land use. The rate of N mineralization demonstrates the capacity of a soil to provide nutrients that are required for plant growth and therefore productivity. This study has shown that Nmin is closely related to the amount of N in the soil and consequently varied along the aridity and soil type gradients investigated. Land use changed the rate of N mineralization with cultivation causing a net N immobilization, compared with the conserved land use where N was mineralized. The livestock land use showed contrasting trends which can be explained by the soil texture at the sites. Simulations with the CENTURY model showed that Nmin was influenced by soil texture, precipitation, and nutrient addition. Therefore Nmin fluctuated around similar levels for the relevant sites, but the modelled estimate was much faster than the estimated rate of the cultivated land use. Nmin was therefore regarded to be a site specific attribute varying with soil texture and microclimate, as well as management regime.

Although the microbial biomass has been hypothesised as being an important index of SOM turnover and soil fertility (Greenland and Szabolcs 1994; Prasad *et al* 1994) conclusions from this study do not generally support this. No differences were observed between sites in the size of the microbial biomass, but differences were apparent between land uses. Generally less than 1.5% of the C and N was in the microbial biomass which is the lower limit of what is expected for systems such as these. This proportion did not vary between the savanna and grassland biomes or the conserved, cultivated nor livestock land uses, indicating that the turnover of SOM is similar in different land uses, even though the absolute quantities may differ. The majority of C and N was present as the intermediate fraction, which reinforces the important role that soil texture has on C and N dynamics.

In conclusion, the processes N mineralization and immobilization, as well as the physicochemical factors soil texture, pH and moisture content, could be used to explain the turnover of C and N in the sites and land uses investigated. The redundancy analysis showed that sites and land uses could be separated on the basis of the environmental variables site aridity, pH, the percentage fines (silt-plus-clay) in the soil, and the light fraction mass. Multivariate methods may therefore provide useful techniques to discriminate which sites and land uses can be sustained on the basis of how the independent variables soil texture, site aridity, soil pH and light fraction mass contribute to the turnover of C and N in the soil. By following trends in these variables over time, it would be possible to identify how the system changed and how resilient it was (Greenland and Szabolcs 1994).

Further research would include improving the fit of these data as well as additional data sets to the CENTURY model. This would allow for the development of Geographical Information System maps which can be used to delineate sustainably productive (Greenland and Szabolcs 1994) regions of South Africa on the basis of how the SOM and N mineralized in the soil. This will require the input of soil textural information and climatic data, that can be used with the CENTURY model. An important methodologically improvement for further studies would be the incorporation of soil functional types in the analysis of the C and N dynamics. As soil texture changes so subtly along the gradient, and because there are so many different soil types (both in terms of texture and mineralogy), it would be important to identify distinctly different functional types within the same climatic region so that climate would not be confounded with soil texture, and visa versa. An simple example would include categorising soils on the basis of the percentage fines in the soil, and the kind of mineralogy.

Alternatively a laboratory based experiment using soil type and soil moisture content as covariables in a factorial experiment could be used to assess how SOM turnover, Nmin and immobilization vary over time, i.e. with season. These will have to be extrapolated to field conditions, but would be cheaper to conduct. They would also allow for more accurate parameterization of the CENTURY model, as unknown parameters could be calculated.

Thus a sensitivity analysis, where various driver variables of the model are systematically altered to determine the effect each has on model output, can be used to determine suitable decomposition rates and management regimes. In addition to this, field based validations, as done with this project, can be conducted to improve model fit which will provide a robust explanation for how carbon and nitrogen differ across regions and land uses in South Africa.

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## APPENDICES

Appendix 1. The environmental variables for the land uses at each site investigated in the savanna and grassland biome.

| Biome and site | Land use | Variable: Texture | % fines (clay + silt) | $pH_{(H2O)}$ | Bulk density |
|---|---|---|---|---|---|
| *Savanna* | | | | | |
| Towoomba1 | Con | Sandy loam | 19.3 | $5.58^{de}$ | $1.56^{a}$ |
| Towoomba2 | Con | Clay loam | 56.9 | $7.30^{a}$ | $0.99^{cd}$ |
| Nylsvley | Con | Sandy loam | 25.0 | $5.30^{e}$ | $1.59^{a}$ |
| Messina | Con | Sandy loam | 21.9 | $6.14^{bc}$ | 1.25 |
| Klaserie | Con | Loamy sand | 20.8 | $5.85^{cd}$ | $1.60^{a}$ |
| Mkuze | Con | Clay | 86.3 | $6.25^{b}$ | $0.94^{cd}$ |
| *Grassland* | | | | | |
| Bloemhof | Con | Sand | 10.5 | $5.54^{de}$ | $1.65^{a}$ |
| Bethlehem | Con | Sandy loam | 24.2 | $4.95^{f}$ | $1.18^{bc}$ |
| Bethal | Con | Sandy loam | 22.6 | $5.88^{cd}$ | $1.25^{b}$ |
| Vryheid | Con | Silty clay | 84.8 | $4.79^{f}$ | $0.89^{d}$ |
| *Savanna* | | | | | |
| Towoomba1 | Cul | Sandy clay loam | 28.6 | $5.28^{d}$ | 1.26 |
| Towoomba2 | Cul | Clay | 63.3 | $7.19^{a}$ | $1.17^{de}$ |
| Nylsvley | Cul | Sand | 10.0 | $5.37^{d}$ | $1.40^{abcde}$ |
| Messina | Cul | Sandy loam | 19.6 | $6.70^{b}$ | $1.54^{abcd}$ |
| Klaserie | Cul | Sandy loam | 26.5 | $4.80^{e}$ | $1.27^{bcde}$ |
| Klaserie (C) | Cul | Loamy sand | 17.7 | $5.58^{cd}$ | $1.21^{cde}$ |
| Mkuze | Cul | Clay | 83.8 | $5.96^{c}$ | $0.73^{f}$ |
| *Grassland* | | | | | |
| Bloemhof | Cul | Sand | 6.8 | $5.88^{c}$ | $1.65^{ab}$ |
| Bethlehem | Cul | Sandy loam | 20.4 | $5.22^{d}$ | $1.72^{a}$ |
| Bethal | Cul | Sandy loam | 24.2 | $6.41^{b}$ | $1.59^{bac}$ |
| Vryheid | Cul | Clay loam | 64.3 | $5.65^{cd}$ | $1.06^{e}$ |
| *Savanna* | | | | | |
| Towoomba1 | Lvs | Sandy loam | 27.0 | $6.17^{a}$ | $1.22^{bc}$ |
| Nylsvley | Lvs | Loamy sand | 19.9 | 4.[illegible]9$^{cd}$ | $1.32^{b}$ |
| Messina | Lvs | Sandy loam | 19.9 | $6.07^{a}$ | $1.30^{b}$ |
| Klaserie | Lvs | Sandy loam | 17.8 | $6.22^{a}$ | $1.34^{b}$ |
| Klaserie (C) | Lvs | Sandy loam | 21.8 | $5.67^{ab}$ | $1.24^{bc}$ |
| Mkuze | Lvs | Clay | 79.4 | $6.11^{a}$ | $1.04^{c}$ |
| *Grassland* | | | | | |
| Bloemhof | Lvs | Loamy sand | 12.6 | $5.36^{bc}$ | $1.65^{a}$ |
| Bethlehem | Lvs | Sandy loam | 23.4 | $5.21^{bc}$ | $1.07^{c}$ |
| Bethal | Lvs | Sandy loam | [illegible]1.4 | $5.33^{bc}$ | $1.35^{b}$ |
| Vryheid | Lvs | Sandy clay loam | 41.4 | $4.64^{d}$ | $1.13^{bc}$ |

Note that significant differences ($P < 0.05$) exist between sites which have different letters, and the variable percentage fines has not been tested for significance .

Appendix 2. The amount of carbon (mg C/g soil) in the three soil organic matter fractions.

| Biome or land use | The Soil Organic Matter fractions | | | |
|---|---|---|---|---|
| | Microbial | Light fraction | Intermediate | Total |
| Savanna | 0.14 | 1.14 | 8.84 | 10.12 |
| grassland | 0.15 | 1.67 | 16.62 | 18.44 |
| | | | | |
| Conserved | 0.16 | 1.27 | 16.34 | 17.78 |
| Cultivated | 0.13 | 0.74 | 8.66 | 9.53 |
| Livestock | 0.14 | 1.83 | 10.46 | 12.44 |
| *Savanna* | | | | |
| Conserved | 0.19 | 1.21 | 9.45 | 10.85 |
| Cultivated | 0.13 | 0.87 | 7.44 | 8.44 |
| Livestock | 0.11 | 1.38 | 9.86 | 11.35 |
| *Grassland* | | | | |
| Conserved | 0.12 | 1.35 | 26.69 | 28.16 |
| Cultivated | 0.13 | 0.51 | 10.80 | 11.44 |
| Livestock | 0.18 | 2.50 | 11.37 | 14.05 |

Appendix 3. The amount of nitrogen (ug N/g soil) in the three soil organic matter fractions.

| Biome or land use | The Soil Organic Matter fractions | | | |
|---|---|---|---|---|
| | Microbial | Light fraction | Intermediate | Total |
| Savanna | 8 | 570 | 211 | 789 |
| grassland | 8 | 845 | 257 | 1110 |
| | | | | |
| Conserved | 8 | 700 | 372 | 1080 |
| Cultivated | 7 | 505 | 145 | 657 |
| Livestock | 9 | 632 | 323 | 964 |
| *Savanna* | | | | |
| Conserved | 9 | 626 | 318 | 953 |
| Cultivated | 6 | 594 | 0 | 588 |
| Livestock | 9 | 486 | 364 | 859 |
| *Grassland* | | | | |
| Conserved | 7 | 812 | 571 | 1389 |
| Cultivated | 7 | 348 | 423 | 778 |
| Livestock | 10 | 849 | 263 | 1122 |

Appendix 4 The management regimes for the land uses investigated in this study and a list of abbreviations used in tables E1 - E3 of the management regimes.

| CENTURY code | Description |
|---|---|
| X | - designates the month as the active month |
| Crop G2 | - crop of warm season species |
| Crop C-HI | - crop corn with high yield |
| Frst | - first month of crop growth |
| Last | - last month of crop growth |
| Fire M | - fire of a medium intensity removing % of the vegetation |
| Fert N1 | - fertilization with 1 $g/m^2$ |
| Fert PS3 | - fertilization with superphosphate applied at kg/ha |
| Graz GL | - grazing low meaning the grazers have no effect on grass growth |
| Graz W | - winter grazing of the standing dead grass |
| Graz GH | - grazing high meaning that grazers have a quadratic effect on grass growth |
| Harv C | - Harvest with a cultivator |
| Senm | - senescence |

Appendix E 4.1 The conserved land use management regime that is used as the event.100 file in the CENTURY model. This shows a low level of grazing as expected in conserved areas, and a one in four year fire event with a medium intensity which reflects average effects of fire over a 100 year simulation, i.e some years hot fires, some cool fires, some triennial burns, others later years.

YEAR 1 TO 3 - LOW INTENSITY "GAME" GRAZING

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | G2 | | | | | | | | G2 | | | |
| Frst | X | | | | | | | | X | | | |
| Last | | | | | X | | | | | | | X |
| Graz | GL | GL | GL | GL | W | W | W | W | GL | GL | GL | GL |
| Fire | | | | | | | | | | | | |

YEAR 4 - MEDIUM BURN EVERY FOURTH YEAR

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | G2 | | | | | | | | G2 | | | |
| Frst | X | | | | | | | | X | | | |
| Last | | | | | X | | | | | | | X |
| Graz | GL | GL | GL | GL | W | W | | | GL | GL | GL | GL |
| Fire | | | | | | | M | | | | | |

Appendix E 4.2 The cultivated land use management regime that is used as the event.100 file in the CENTURY model.

YEAR 1 - CONTINUOUS CORN CROP

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | C-HI | | | | | | | | | | | |
| Frst | X | | | | | | | | | | | |
| Last | | | | | | X | | | | | | |
| Senm | | | | | | | X | X | X | X | X | X |
| Fert | PS3 | N1 | | | | | | | | | | |
| Cult | | | | | | | | | C | | | |

Appendix E 4.3 The livestock land use management regime that is used as the event.100 file in the CENTURY model. This shows a rotational grazing schedule for a particular camp with alternating periods of heavy grazing, winter grazing, and rest.

YEAR 1 - LATE SUMMER GRAZING

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | G2 | | | | | | | | G2 | | | |
| Frst | X | | | | | | | | X | | | |
| Last | | | | | X | | | | | | | X |
| Graz | | | | | | | | | | GH | GH | GH |
| Fire | | | | | | | | | | | | |

YEAR 2 - WINTER GRAZING

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | G2 | | | | | | | | G2 | | | |
| Frst | X | | | | | | | | X | | | |
| Last | | | | | X | | | | | | | X |
| Graz | | | | | W | W | W | | | | | |
| Fire | | | | | | | | | | | | |

YEAR 3 - EARLY SUMMER GRAZING

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | G2 | | | | | | | | G2 | | | |
| Frst | X | | | | | | | | X | | | |
| Last | | | | | X | | | | | | | X |
| Graz | GH | GH | GH | | | | | | | | | |
| Fire | | | | | | | | | | | | |

YEAR 4 - FALLOW

| | Month of the year | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Activity | J | F | M | A | M | J | J | A | S | O | N | D |
| Crop | G2 | | | | | | | | G2 | | | |
| Frst | X | | | | | | | | X | | | |
| Last | | | | | X | | | | | | | X |
| Graz | | | | | | | | | | | | |
| Fire | | | | | | | | | | | | |