# FERTILITY RECOVERY IN SANDY SOILS UNDER BUSH FALLOW IN SOUTHERN MOZAMBIQUE

**BY**

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# DECLARATION

I declare that the thesis hereby submitted by me for the Philosophiae Doctor degree at the University of the Free State is my own independent work and has not previously been submitted by me at another University. I furthermore cede copyright of this thesis in favour of the University of the Free State.

Signature 30 May 2008

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## Abstract

Bush fallow under shifting cultivation is the most common practised subsistence farming system in southern Mozambique. This system is likely to persist due to the existence of large areas sparsely inhabited coupled with financial limitations preventing small scale farmers from buying fertilizers. The bush fallow is intended to recover naturally the productive capacities of soils lost during cropping. This study was conducted therefore to gain a better understanding on the composition and biomass of bush fallow vegetation, climatic factors affecting leaf litter decay of an important tree species and the dynamics of some soil fertility indicators.

Five agroecosystems representing rainfall regions of <400 mm (AE6), 400-600 mm (AE5), 600-800 mm (AE3), 800-1000 mm (AE2), >1000 mm (AE1) and a transitional agroecosystem of 400-800 mm (AE4) were selected. Within each agroecosystem, five land uses (virgin, cultivated, < 5 years fallow, 5-15 years fallow and >15 years fallow) were identified. Descriptions and comparisons of vegetation were performed between land uses within agroecosystems and similar land uses across agroecosystems, except in cultivated land; effects of soil water content and soil temperature on decomposition of *Brachystegia spiciformis* leaf litter were evaluated in recently abandoned agricultural fields cleared of any vegetation (Bare) and in >15 years fallow fields (15F) at sites in a transect that covered AE2 to AE6; and at every combination of agroecosystem and land use the dynamics of organic C, total N, CEC, pH, P, Ca, Mg and K were determined in the 0-50 mm, 50-100 mm and 100-200 mm soil layers.

A total of 204 species that including N-fixing species, belonging to 141 genera and 50 families divided into tree, shrub and herbaceous layers were identified. The tree layer was only found in virgin fields and in fields abandoned to bush fallow >15 years, whereas shrub and herbaceous layers occurred in all fields. The tree species in bush fallow fields of coastal and wetter AE1, AE2 and AE3 (dominated by *B. spiciformis* and *Julbernaldia globiflora*) outnumber those in inland and drier AE4, AE5 and AE6 (dominated by *Birchemia discolour* and *Colophospermum mopane*) and have larger diameter that result in greater biomass. Number of shrubs decreased from coastal and wetter to inland and drier agroecosystems. The herbaceous biomass declined from young to old fallow fields in coastal and wetter agroecosystems, while the converse was observed in inland and drier agroecosystems. Nitrogen-fixing species tended to occur more in bush fallow fields older than 15 years. In inland and drier agroecosystems the tree biomass in 15F fields tended to be higher than in virgin fields due to presence of succession species that differ from the original ones. In the wetter agroecosystems C loss from *B. spiciformis* leaf litter was faster, whereas in the drier ones it was more sensitive to rainfall pulses. Similarly, C loss was faster in 15F fields than in bare fields.

In coastal and wetter AE1, AE2 and AE3 there was a declining trend in organic C and total N from virgin to cultivated fields. This trend proceeds to the <5 years fallow fields and thereafter the contents of the two indicators increased in older fallow fields. A different pattern was found in the dry AE4 and AE5 where organic C and total N tended to decline gradually even with longer fallow periods. In the severely dry AE6 no clear trend was found. The pH in all agroecosystems decreased from cultivated to fallow fields, an effect attributable to a gradual decrease in the basic cations released on the soil surface by the ash produced during slash and burn. A slight increase in the silt plus clay fraction from AE4 to AE5 was found, which resulted in increased CEC, P, Ca, Mg and K. From the coastal and wetter to inland and drier agroecosystems pH, P and Ca increased, except in AE4 and AE5, which had lower pH and Ca values. The lower values of pH resulted in lower contents of P in AE4 and Ca and Mg in both agroecosystems, which have the same vegetation, suggesting that this should be the determining factor.

The results from this study showed that a bush fallow period of longer than 15 years is required for restoration of soil fertility in abandoned cultivated fields to the same level as in virgin fields. This aspect must be taken into account when strategies are developed to improve the sustainability of cropping on the sandy soils of southern Mozambique

Keywords: Agroecosystem, ecological importance, exchangeable bases, organic matter, shifting cultivation, vegetation composition

# Uittreksel

Bosbraak onder verskuiwende verbouing is die mees algemene bestaansboererystelsel wat in suidelike Mosambiek toegepas word. Die stelsel sal waarskynlik voortgaan weens die groot ylbevolkte areas en kleinboere se beperkte finansiële vermoëns om kunsmis te koop. Die bosbraak het ten doel om die produksievermoë van gronde, wat met gewasverbouing verlore gaan, natuurlik te herstel. Hierdie studie is dus gedoen om 'n beter begrip te kry van die samestelling en biomassa van bosbraakplantegroei, klimaatfaktore wat die afbraak van 'n belangrike boomspesie beïnvloed en die dinamika van sekere grondvrugbaarheidsindikatore.

Vyf agro-ekosisteme wat reënvalstreke van <400 mm (AE6), 400-600 mm (AE5), 600-800 mm (AE3), 800-1000 mm (AE2) en >1000 mm (AE1) en 'n oorgangsagro-ekosisteem van 400-800 mm (AE4) is geselekteer. Binne elke agro-ekosisteem is vyf landsgebruike (onversteurde, bewerkte, <5 jaar braak, 5-15 jaar braak en >15 jaar braak) geïdentifiseer. Beskrywings en vergelykings van die plantegroei tussen landgebruike binne agro-ekosisteme en soortgelyke landgebruike oor agro-ekosisteme is gedoen; effekte van grondwaterinhoud en grondtemperatuur op die afbraak van *Brachystecia spiciformis* blaarreste is in bewerkte lande sonder enige plantegroei en wat onlangs vir bosbraak gelos is (Kaal), en in >15 jaar braaklande (>15F) by lokaliteite wat 'n deursnit vanaf AE2 tot AE6 dek, geëvalueer; en by elke kombinasie van agro-ekosisteem en landgebruik is die dinamika van organiese C, totale N, KUK, pH, P, Ca, Mg en K in die 0-50 mm, 50-100 mm en 100-200 mm grondlae bepaal.

'n Totaal van 204 spesies wat N-bindende spesies in sluit en tot 141 genera en 50 families behoort, is geïdentifiseer en in 'n boom-, struik- en kruidlaag verdeel. Die boomlaag is slegs in onbewerkte en >15 jaar bosbraaklande gevind, terwyl die struik- en kruidlae in alle lande voorkom. Die boomspesies in bosbraaklande van die kus en natter AE1, AE2 en AE3 (gedomineer deur *B. Spiciformis* and *Julbernaldia globiflora*) is meer as die in binnelandse en droër AE4, AE5 en AE6 (gedomineer deur *Birchemia discolour* en *Colophospermum mopane*) en het 'n groter diameter en dus meer biomassa. Die aantal struike neem vanaf die kus en natter na die binnelandse en droër agro-ekosisteme af. Die kruidbiomassa neem af van die jong na ou braaklande in die

kus en natter agro-ekosisteme en die omgekeerde is in die binnelandse en droër agro-ekosisteme waargeneem. Stikstofbindende spesies neig om meer in bosbraaklande ouer as 15 jaar te wees. In die binnelandse en droër agro-ekosisteme neig die boombiomassa in die 15F lande om hoër te wees as in die onversteurde lande weens die teenwoordigheid van opvolgspesies wat verskil van die oorspronklikes. In die natter agro-ekosisteme was die verlies van C uit *B. spiciformis* blaarreste vinniger terwyl in die droër sisteme was dit meer sensitief vir reënvalbuie. Soortgelyk was koolstofverlies vinniger in die 15F lande as in die kaal lande.

In die kus en natter AE1, AE2 en AE3 is daar ‘n neiging dat organiese C en totale N vanaf onversteurde na bewerkte lande afneem. Hierdie neiging duur voort in die >5 jaar braaklande en daarna neem die inhoud van die twee indikatore toe in die ouer braaklande. ‘n Ander patroon is in die droë AE4 en AE5 gevind waar organiese C en totale N neig om geleidelik af te neem met selfs langer braak periodes. In die baie droë AE6 was daar geen duidelike patroon. Die pH in alle agro-ekosisteme het afgeneem vanaf bewerkte na braaklande en die effek word toegeskryf aan die geleidelike afname in die basiese katione wat deur die as afkomstig van kap en brand op die grondoppervlak vrygestel is. ‘n Effense toename in die slik plus klei fraksie van AE4 na AE5 is gevind wat ‘n toename in KUK, P, Ca, Mg en K tot gevolg het. Vanaf die kus en natter na binnelandse en droër agro-ekosisteme het pH, P en Ca toegeneem, behalwe in AE4 en AE5 wat laer pH en Ca waardes gehad het. Die laer waardes van pH het tot laer inhoude van P in AE4 en Ca en Mg in beide agro-ekosisteme gelei wat dieselfde plantegroei het en die is moontlik die bepalende faktor.

Die resultate van hierdie studie het getoon dat ‘n bosbraak periode van langer as 15 jaar nodig is om die grondvrugbaarheid van verlate bewerkte lande tot dieselfde vlak as die van onversteurde lande te verhoog. Hierdie aspek moet in berekening gebring word wanneer strategieë ontwikkel word om die volhoubaarheid van gewasverbouing op die sanderige gronde van suidelike Mosambiek te verbeter.

Sleutelwoorde: Agro-ekosisteme, ekologiese belangrikheid, organiese materiaal, uitruilbare basisse, verskuiwende verbouing, plantegroei samestelling.

# CHAPTER 1

# Introduction

## 1.1 Motivation

Southern Mozambique comprises three provinces, namely Inhambane, Gaza and Maputo. The total population of the three provinces is about 3.2 million whereof 80% are living in the Inhambane and Gaza provinces (Direcção de Economia, 1996). In these two provinces the people are very unevenly distributed. As usual, the population density decreases from the centre of urban areas to the rural outskirts (Folmer *et al*., 1998). A similar pattern is also observed when moving from the coastal belt inland where large areas are scarcely inhabited (Snijders, 1985; MAP, 1996).

The subsistence of 80% of the country's inhabitants depends mainly on agriculture. Based on this figure it can be estimated that about 2 million people in the provinces of Inhambane and Gaza practise agriculture as their major activity for subsistence. The most common crops produced in the two provinces are maize (*Zea mays*), cassava (*Manihot esculenta*), cowpea (*Vigna unguiculata*), groundnut (*Arachis hypogeae*), sorghum (*Sorghum bicolo*), and millet (*Penisetum typhoides*) (Reddy, 1985; Direcção de Economia, 1996). These crops are cultivated mainly in sandy soils that cover the majority of the land in southern Mozambique (Flores, 1973; MAP, 1996; Geurts, 1997). Among the listed crops the most commonly cultivated are maize and cassava. Both crops are known for their nutrient depletion of soils due to exportation through harvest (Folmer *et al*., 1998).

The sandy soils, where the majority of the population produce their crops, are generally characterized by low organic matter contents, making them prone to degradation through exploitation of their nutrients if submitted to continuous cultivation. The agricultural authorities and the population are aware of this situation. However, actual practices do not include nutrient reposition through the application of fertilizers during crop production due to social and economical constraints. Unfortunately a solution to these limitations is not expected in the short run.

A survey conducted by the Ministry of Agriculture has shown that on average a family in Mozambique cultivates 1.86 ha with annual crops. The average of 2.56 ha in Inhambane and 2.33 ha in Gaza are therefore larger compared to the rest of the country. This phenomenon is probably due to the low fertile soils that result in poor yields. The long-term yields for maize, cassava, cowpea, groundnut and sorghum are respectively 0.26, 0.42, 0.22, 0.30 and 0.40 ton $ha^{-1}$ in Inhambane and 0.21, 0.21, 0.24, 0.15 and 0.30 ton $ha^{-1}$ in Gaza (Direcção de Economia, 1996).

The most common system of land preparation used by farmers in the provinces of Inhambane and Gaza is slash and burn under shifting cultivation farming system. This cultivation method is more successful in the scarcely than in densely inhabited areas of the two provinces. In the scarcely inhabited areas cultivated land can be bush fallowed for a long enough period to recover some of its original fertility, which is not possible in the densely inhabited areas (MAP, 1996).

In Mozambique there are few studies to quantify the soil fertility dynamics and two of them were mainly concentrated in the first 20-25 km of the coastal belt (MA/FAO, 1983; Chaguala and Geurts, 1996; Folmer *et al*., 1998). Studies to quantify fertility depletion and recovery with the traditional cropping system at farm level that cover the whole country are non-existent. However, reference to two studies are worthwhile despite to the fact that one of them does not address either the depletion or recovery of soil fertility. In one study, the effects of climatic conditions and agronomic practices on crop production in Mozambique have been investigated by Reddy (1986) using meteorological data. His ultimate aim was to assist researchers to understand and /or improve rainfed crop production with the establishment of early warning system zones for Mozambique. In the other study Folmer *et al*. (1998) aimed to assess soil fertility depletion under different land uses in the country.

Folmer *et al*. (1998) used a model for their study where the macronutrients N, P and K served as indicators of soil fertility. Concerning southern Mozambique, the study resulted in two conclusions. The first is that Gaza province is one of the few provinces with high nutrient depletion. The second is that in the coastal belt of Inhambane and Gaza the buffering capacity breakdown (BCB), which is the relation between nutrient depletion and nutrient resources, is moderate to high in densely populated areas. These results are contrary with respect to the rest of the country, except for Nampula province in the north. Unfortunately, the study gives only information about soil fertility depletion for different land uses in general and not for different cropping systems within an area. Therefore, the authors concluded that the results have limited significance at farm level. Finally, they recommended that proper studies at farm level are needed.

In literature it has been stated several times that bush fallow can no longer provide a sustainable basis for farming communities in the densely populated areas of the tropics. Many studies aiming to find alternatives for slash and burn in tropical agriculture were therefore performed. The results were in general disappointing since the productivity of these soils continue to decline because of a decrease in organic matter content. However, some researches in the forest and/or savannah regions in West Africa showed that slash and burn can still be considered as the most efficient way for the accumulation of biomass and hence organic matter (Harwood, 1996). This phenomenon can be attributed to the many plant species with different type of root systems that establish usually during bush fallow (Juo *et al*., 1995).

In general, studies are scarce that deals with the dynamics of the total nutrient stock in either primary tropical forest or savannah land and the subsequent influence of cropping and fallow cycles on it. Such studies should be a central focus in low densely populated areas with small-scale farmers who cannot afford to buy fertilizers (Snapp *et al*., 1998). There is specifically a great need to quantify the restoration of nutrient stock in abandoned cultivated land by bush fallow under a range of climatic conditions. Information of this nature is required to enhance our knowledge of fertility recovery in the sandy soils used for crop production in southern Mozambique.

In summary, the following reasons motivated the study on soil fertility recovery on sandy soils in Inhambane and Gaza provinces: (i) shifting cultivation is the most common practice and will persist for a substantial period as a result of limited financial resources of farmers, (ii) the population density in the vast areas of both provinces is low (iii) there is a serious limitation of primary data, (iv) a solution should be found where farmers can better manage their lands for cropping and (v) as recommended by Lal (1996) and Nandwa and Bekunda (1998) technologies should fit the existing socio-cultural context.

The results from this study can be a valuable contribution to a database for technical decisions. As stated by Miller and Wali (1995), such type of database can be used to predict the vulnerability of soils to degradation, when subject to shifting cultivation. In addition, a database of this nature can be also of great value in the designing of cropping systems with bush fallow periods that will ensure sufficient fertility recovery of degraded soils in the Inhambane and Gaza provinces.

## 1.2 Objectives

The general objective with this study was to explain the soil fertility restoration process when cultivated land is abandoned to bush fallow by quantifying the content of selected indicators over time under different climatic conditions with virgin land serving as reference. Therefore, the following specific objectives were pursued in different land uses within an agroecosystem and similar land uses across agroecosystems:

- To compare the vegetation composition and standing biomass.
- To evaluate the effects of soil temperature and soil water content on carbon loss from leaf plant litter.
- To assess the dynamics of soil fertility indicators such as organic matter (OM), acidity, cation exchange capacity (CEC) and macronutrients.

## 1.3 Assumptions

In order to address the outlined objectives, the following assumptions were taken into account for the studied area:

- Annual rainfall is the main driving factor that influences the composition and biomass production of vegetation and for the soil restoration process and therefore a rainfall zone represents an agroecosystem.

- Composition and biomass of plant species and the dynamics of macronutrient, CEC, acidity and OM in soils of similar land uses are homogeneous in each agroecosystem.

- The soil type and dynamics of vegetation within a diameter of one km are homogeneous.

- Under local field conditions the effect of soil water content and soil temperature on carbon loss from plant leaf litter is quantifiable within one year.

- In an agroecosystem when cultivated lands were abandoned for bush fallow the level of each selected soil fertility indicator was similar.

- Five land uses, viz. virgin, cultivated and bush fallow of three ages were sufficient to describe the dynamics of the selected soil fertility indicators within an agroecosystem.

- The upper 200 mm soil layer is sufficient to study the dynamics of the selected soil fertility indicators.

## 1.4 Hypotheses

The following null hypotheses were formulated and tested:

- There are no significant differences in composition and standing biomass of vegetation among land uses within an agroecosystem and between similar land uses across agroecosystems.

- There are no significant relationships between carbon loss from plant leaf litter and soil water content or soil temperature.

- There are no significant differences in the selected soil fertility indicators among land uses within an agroecosystem and between similar land uses across agroecosystems.

# CHAPTER 2

# Literature review

## 2.1 Introduction

Almost half of the chapter is devoted to soil quality and soil degradation. In soil quality, three issues are described: The different views of its concepts, related concepts and the indicators of soil quality. In the discussion of soil degradation, beside conceptualisation, aspects of the causes of degradation, forms of its manifestation, the socio-economic implications and the role of scientists in halting and reversing it are addressed.

The remaining half of the chapter describes traditional cropping systems in the tropics, SOM as indicator of soil fertility, the restoration of soil fertility and the need for its assessment in fallow lands. In discussing traditional cropping systems, the vulnerability of the ecosystems where agricultural activities are carried out are described; the role of SOM on soil fertility as well as factors affecting its dynamics are reviewed; the importance of the restoration of soil fertility, some experiences as well as determining factors are discussed; the importance of assessment, the assessment procedure and modelling as a tool for assessment are dealt with.

## 2.2 The concept of soil quality, related aspects and indicators

### *2.2.1 The concept of soil quality*

Soil quality is a topic of interest to people in various circles. These circles include soil scientists, agriculturalists, biologists, agricultural and environmental policy decision makers, and readers of the semi-popular press. All are constantly concerned with a better understanding of soil quality (Warkentin, 1995). Therefore, many definitions of soil quality evolved that differ somewhat. Differences in these definitions result from the nature of interest or relationship somebody has with land (Shukla *et al*., 2006). The main interest of the agriculturalist is to sustain the productivity of the soil (Lal, 1998) or enhance its productivity now and in the future (Shukla *et al*., 2006). However, the conservationist may want to conserve soil while protecting the environment; the consumer may want soil to produce healthy and inexpensive food; and the environmentalist may want the soil to be capable of maintaining or enhancing biodiversity, water quality, nutrient cycling and biomass yield (Mausbach and Seybold, 1998). Hence, many attempts that resulted in different approaches over time have been made to obtain a general definition that could embrace all interests (Warkentin, 1995).

Doran and Parkin (1994) defined soil quality as " the capacity of soil to function effectively at present and in the future or as the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintaining environmental quality and promote plant and animal health". Later, Mausbach and Seybold (1998), referred to a more comprehensive definition elaborated by Karlen *et al*. (1997), which defines soil quality as "the capacity of a specific soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation".

An understanding of soil quality is one of the keys to understand natural ecological processes (Warkentin, 1995). If soil quality increases, the total productivity of natural resources also increases (Lal, 1998). The particular goal of agronomists is to maximise the productivity of conventional agricultural systems but they also envisage the improvement of soil fertility through integrated biological mechanisms that play a role in agro-ecosystems (Zinck and Farshad, 1995). To summarise, soil quality refers to the capability of soil to perform a range of functions not only in crop production, environmental protection and habitation (Scherr, 1999) but also in food safety, and animal and human health (Kennedy and Papendick, 1995).

### *2.2.2 Aspects related to soil quality*

As described in the previous section, soil quality is not a property that is easy to define. This results in the use of different names for related concepts. The health, productivity, suitability and sustainability of soil are all terms commonly used in defining soil quality (Warkentin, 1995).

#### 2.2.2.1 Soil health

On several occasions the term "soil health" is used in place of "soil quality" (Warkentin, 1995). The reason for this is that there is a relationship between soil quality and the health of plants, animals and humans (Van der Merwe and De Villiers, 1998). Warkentin (1995) cited Haberen (1992) who argued that this relationship results from the fact that animals and humans eat crops produced by soil. From an agricultural point of view, soil quality can be described as preserving health while sustaining the capacity to accept, store and recycle water, minerals and energy used to grow crops (Arshad and Coen, 1992).

#### 2.2.2.2 Soil productivity

Soil productivity is " the capability of soil to produce a specified plant or sequence of plants under a defined set of management practices" (Parr *et al.*, 1990). This capability should enhance plant and biological productivity (Van der Merwe and De Villiers, 1998). Enhancement is made possible by supplying and maintaining fertility, which is an integrated measure of the nutrient-holding capacity, microbial activity, the extent of contamination, and the rate of erosion (Smith *et al.*, 2000). Soil productivity is dependent on the capacity of the soil to fix accumulated energy (Bruce *et al.*, 1995).

#### 2.2.2.3 Soil suitability

Suitability is probably the oldest and one of the most frequently used concepts in soil quality and is related to the quantity of crops produced (Warkentin, 1995). However, to other authors, soil quality and suitability are different concepts. Miller and Wali (1995) suggest that suitability should be replaced by sustainability as the latter term reflects the fitness of a particular system of land use management.

#### 2.2.2.4 Soil sustainability

Sustainability is "the capacity of agro-ecosystems to maintain commodity production through time without threatening ecosystem structure and function" (Smith *et al.*, 2000). The concept of sustainability has different meanings at different spatial levels varying from arid areas to humid tropics, depending on which factor has to be conserved (Zinck and Farshad, 1995).

Regardless these differences, soil quality is the key to agricultural sustainability (Warkentin, 1995; Lal, 1998). The final goal of sustainability in agriculture is to develop a farming system that keeps the land physically, economically and socially suited over a long period (Miller and Wali, 1995). Therefore, a farming system should conserve natural resources, protect the environment, and enhance health and safety (Parr *et al*., 1990). At the same time, the productivity should remain efficient and should be maintained indefinitely without any negative trend (Lal, 1998). These characteristics depend largely on the long-term fertility and productivity of the land (Van der Merwe and De Villiers, 1998).

Zinck and Farshard (1995) considered the concept of soil sustainability difficult especially when it comes to putting it into practice. According to them this is because scientists from different disciplines contribute to its conceptualisation using different dimensions, which are aggravated by being temporally and spatially sensitive. To summarise, sustainability is a multifaceted concept (Syers *et al*., 1995).

### *2.2.3 Indicators of soil quality*

An important issue is the evaluation of soil quality (Halvorson *et al*., 1997). This requires that any corrective measure applied can be monitored internationally (Arshad and Coen, 1992). The monitoring could be done by selecting the most important factors, termed indicators (Miller and Wali, 1995). Indicators should measure and reflect the physical, chemical and biological status of a soil (Arshad and Coen, 1992). Hence, indicators must show whether a soil is stable, improving or deteriorating over time (Syers *et al*., 1995; Shukla *et al*., 2006). In general, they are a "pointing or directing device" (Kennedy and Papendick, 1995).

Kennedy and Papendick (1995) cited Holloway and Stork (1991), who stated that a property of soil is considered a good indicator if it satisfies the following conditions: " (i) be adequately sensitive to change; (ii) accurately reflect the functioning of the system; (iii) be universal, yet illustrate temporal or spatial pattern; (iv) be cost effective and relatively easy and practical to measure".

Arshad and Coen (1992) proposed that the following properties be used as indicators of soil quality: soil depth to a root restricting layer; available water holding capacity;

bulk density; penetration resistance; hydraulic conductivity; aggregate stability; OM; nutrient availability; pH; and, where appropriate, electrical conductivity and exchangeable sodium. In general, one property is not sufficient to indicate all the changes within a system (Kennedy and Papendick, 1995).

In reality soil quality indicators are used in different ways according to the subject of concern and its context (Syers *et al*., 1995). Although from the point of view of soil fertility, the physical, chemical and biological properties have to interact, many studies have put more emphasis on chemical properties, which are determined by pH, CEC, nutrient deficiency and toxicity (Lal *et al*., 1999). According to Havlin *et al*. (1999) pH and CEC are two of the most important chemical properties because they influence plant growth by their effect on the availability of plant nutrients.

From a biological point of view, fauna is an important indicator of soil fertility as the organisms in soil can mix inorganic and organic components and change the physical structure of the soil (Steiner, 1996). The soil organisms help to expose new surfaces to enzymatic attacks through mechanical breakdown (Berg and Ekbohm, 1991). The measurement of the abundance, diversity and number of species is specifically used as an indicator in the agro-ecosystem environment (Smith *et al*., 2000).

In general, there is no indicator that can fully characterise the status of an agro-ecosystem at multiple scales (Smith *et al*., 2000). However, some indicators may be related to more than one function (Mausbach and Seybold, 1998). Such an indicator is SOM content as it relates to many aspects of the productivity, sustainability, and environmental integrity of an agro-ecosystem (Smith *et al*., 2000). It influences many biological, chemical and physical characteristics inherent in productive soil (Stevenson and Cole, 1999). This is because the C in SOM is the most reactive component in soil (Van der Merwe and De Villiers, 1998). Hence SOM content is regarded as a very critical indicator of soil quality (Lal, 1998; Lal *et al*., 1999).

Specific levels of reference for many indicators can be identified and are termed thresholds (Miller and Wali, 1995). These thresholds are levels beyond which a system undergoes significant changes (Syers *et al*., 1995). Minor changes in the status of each one of the indicators may be early warning signals of soil degradation and can be used as tools for remediation and in the application of soil building practices (Kennedy and Papendick, 1995). Some examples of threshold values are given in section 2.3.3.2.

## 2.3 The concept of soil degradation and related aspects

### *2.3.1 The concept of soil degradation*

Similarly to soil quality, various definitions of soil degradation have been generated and the concept has changed over time depending on the area of interest of those who defined it. The evolution of the definition of soil degradation can be summarised in different phases (Dahlberg, 1994). In the beginning, soil degradation was seen as only related to desertification caused by human activities. After some time, the emphasis on human activities was reduced and climatic influences were considered as a possibility. In the following phase, the importance of the reversible and irreversible changes concept was introduced. The decline in productivity judged in relation to a specific land and the damage in relation to the respective cost of rehabilitation were the following consideration. Later on, economic considerations and the environmental criticality caused by changes were taken into account. In the end, the meaning of environmental criticality to the biophysical environment, and not simply to the people who live on it, was refined.

In general, soil degradation can be defined as a loss of, or reduction in, soil capability to satisfy a particular use (Blum, 1997). Blaikie and Brookfield (1987a) defined land degradation, in which soil is the main component, as “a reduction in capacity of the land to produce benefits from a particular land use under a specific form and land management”. In this definition, it is reflected that soil degradation is a complex phenomenon (Stewart and Robinson, 1997). It combines biophysical factors of land use and socio-economic aspects as it considers how the land is managed and the expected yield (Steiner, 1996). Therefore, soil degradation is a multidisciplinary concept (Blaikie and Brookfield, 1987a) and a major global issue (Lal, 1998). It is a subject of major concern because it affects subsequent generations (Steiner, 1996).

The process of soil degradation results in deterioration of the interaction among physical, chemical and biological processes of the soil (Steiner, 1996). Minor changes in the status of each of those interactive components may be an indication of soil degradation as they can lead to a reduction in the productive capacity of soil (Lal and Stewart, 1992; Kennedy and Papendick, 1995). A reduction in the productive capacity of soil result from some properties to be below a certain threshold, which impairs a number of functions (Warkentin, 1995).

Soil degradation develops gradually in stages (Steiner, 1996). Its magnitude depends on the net result of soil degradation and conservation processes acting on the soil (Kennedy and Papendick, 1995). The acting processes are land use, farming systems, management, climate and the resilience of the soil (Lal *et al*., 1999). According to Blaikie and Brookfield (1987a), soil degradation can be summarised as a net function of natural and human forces through the following formula: " Net degradation = (natural degrading processes + human interference) – (natural reproduction + restorative management)".

### *2.3.2 Causes of soil degradation*

Soil degradation processes can occur naturally but in many instances they are accelerated by human activities, as soils are very sensitive and vulnerable to external forces (Blum, 1997).

#### 2.3.2.1 Natural causes

Natural degradation occurs as a consequence of inherently natural factors. These are the same factors that contribute to soil formation (Mausbach and Seybold, 1998). The inherently natural factors may be climate, vegetation, topography (Arshad and Coen, 1992), and soil type of the site (Scherr, 1999). These factors can define the capacity for resilience and the sensitivity of a soil.

Climate combined with the type of soil and relief is one of the factors that determines the speed and extent of soil degradation (Steiner, 1996). The climatic factors that are active in the process of degradation are temperature, rainfall, potential evaporation, wind speed and direction and relative humidity (Arshad and Coen, 1992). In the tropics, prolonged dry seasons are followed by heavy rains with large drop diameter, which result in soil erosion when combined with accentuated relief, especially in mountainous areas and arid zones (Steiner, 1996). Higher temperatures and greater rainfall intensity, which are typical in the tropics, subject soils in most developing countries to a significant risk of climate-induced degradation (Scherr, 1999). Higher temperatures are one of the major causes that result in a decomposition rate of SOM five times higher in the tropic than temperate climates (Steiner, 1996).

The type of natural vegetation covering the soil is an important factor that determines the stability of soil against degradation (Blum, 1997). Its contribution is determined by growing plant species as they determine the amount of OM added to the soil (Lal *et al.*, 1999). Sparse vegetation can contribute to soil degradation by facilitating the initiation of the erosion process (Steiner, 1996).

The type of soil also plays an important role in its own stability. For instance, in the tropic sandy soils that are derived from infertile parent material or have been highly weathered over the millennia, leaching of soluble nutrients from soils and acidification, are common (Steiner, 1996; Scherr, 1999). In situations where expandable clays, smectite and vermiculite, are present dry aggregates may form on the surface to break down as they are wet by rain drops and, when dry and dehydrated they form a dense and hard crust (Lal *et al.*, 1999). This situation can be worsened when exchangeable sodium exceeds thresholds as this can disperse the clay fraction  (Levy *et al.*, 1994).

2.3.2.2 Anthropogenic causes

The influence of people results from their decisions and actions in management practices (Van der Merwe and De Villiers, 1998). They interfere in the soil system by land-use disturbing the soil by physical, chemical and biological means (Blum, 1997), as result of socio-economic processes (Sivakumar, 1995) or political decisions (Dahlberg, 1994).

*2.3.2.2.1 Land-use patterns*

Depending on the purpose of the land use and the associated operator skills and management practices, the results may either be a degradation or restoration in soil quality (Arshad and Coen, 1992). Human interference determines the inputs and outputs of nutrients, energy, water, and biological species in agro-ecosystems (Smith *et al*., 2000). Some examples of such interference are: poor soil management such as over-cultivation, overgrazing, poor irrigation practices and de-forestation may lead to soil degradation (Dahlberg, 1994); the use of an acid reacting mineral fertilizer such as urea or ammonium sulphate in agriculture can accelerate the acidification process in soils (Steiner, 1996); non-sustainable soil management results in a rapid decrease in OM within a few years (Steiner, 1996); the management of crop residues and tillage methods can temporarily change the quality of the soil (Lal *et al*., 1999).

*2.3.2.2.2 Socio-economic aspects*

The main cause of soil and land degradation are the demands of people for well-being coupled with an economic framework that does not include the degradation of natural resources in equations used to calculate " socio-economic progress" (Miller, 1998). Agricultural practices that are ecologically sustainable may not be profitable and are therefore not economically sustainable (Smith *et al*., 2000). Consequently, in many countries land deterioration from conventional agriculture and environmental degradation are " side-effects" of development (Zinck and Farshad, 1995). There is also often a conflict between short-term benefits and long-term consequences (Stewart and Robison, 1997). For example, increasing oil or timber exports is often encouraged without considering the possible environmental impact (Zinck and Farshad, 1995).

Poverty is a major factor in the process of the depletion of natural resources (Dahlberg, 1994). Unfavourable ratios of crop to fertiliser prices, particularly for food crops, and financial constraints are some of the key factors, which determine the current low level of nutrient replacement by use of fertilisers in food crop production (Henao and Baanante, 1999)

Population growth also contributes to soil degradation (Lal and Stewart, 1992). The increase of urban and peri-urban areas towards adjacent lands with fertile soils, where people produce food, causes the closing in of these lands (Blum, 1997). This phenomenon is a major issue in developing countries where prime agricultural land is rapidly being converted for the expansion of human habitation, industrial use, and the development of infrastructure (Lal, 1998). The situation leads to migration and use of marginal lands (Mokwunye, 1996). To compensate for the low productivity of marginal lands due to low soil fertility larger areas are used for agricultural production (Steiner, 1996).

The main reason for migration in developing countries is that mainly farmers with very poor resources undertake agriculture, which result in a high to very high probability of crop failure (Eswaran *et al.*, 1997). In Africa, where subsistence agriculture and fallow farming is practised, increasing demographic pressure compels farmers to replant in fallow land before soil fertility has been restored or to work marginal land, which is only suitable for pasture or forestry (Steiner, 1996). These low-fertility marginal lands are prone to soil degradation (Lal, 1998).

*2.3.2.2.3 Political issues*

Priorities defined by politicians associated with market elasticity vary over time, which imposes constraints on short-term adaptation of farming systems that might be incompatible with sustainability (Zinck and Farshad, 1995). Dahlberg (1994) gave two examples of political factors that contributed to soil degradation. In the first example she cited Stcjing (1992) who reported that during the colonial era, environmental research in Africa was mainly concerned with finding profitable ways to exploit natural resources. In the second example she referred to certain countries in southern Africa, including Mozambique, where political instability and environmental degradation were closely linked. Woomer *et al.* (1998) also gave an example where population growth associated with colonial policies in eastern Kenya forced the Akamba people to replace a mixture of shifting cultivation and pastoralism with permanent agriculture on defined land. This resulted in the denuding of the landscape due to the collection of wood for fuel, overgrazing and soil erosion.

### *2.3.3 Forms of soil degradation*

Soil degradation can be of physical, chemical and biological nature.

2.3.3.1 Physical degradation

The process of physical soil degradation is characterised by a disintegration of soil structure, densification and an adverse hydrothermal regime (Lal and Stewart, 1992). The disintegration of soil structure may result in pan formation (Blaikie and Brookfield, 1987a), hard-setting, compaction, crusting, drought, wetness, excessive run-off (Lal *et al.*, 1999), accelerated erosion (Miller, 1998) and terrain deformation through gully erosion (Scherr, 1999).

Soil erosion may be the major factor in the physical destructive process of degradation (Arshard and Coen, 1992). During soil erosion, water and wind remove solid particles from the surface of the soil (Steiner, 1996). It leads to the removal of the topsoil (Kayombo and Mrema, 1998), which is the layer rich in OM and nutrients (Stevenson and Cole, 1999). Erosion results in the reduction of the depth of the soil solum and the shallower soil leading to a decrease in the water and nutrients available to plants (Lal *et al.*, 1999). Ultimately, the reduction of the soil depth may also result in the sedimentation of dams (Scherr, 1999) and pollution of rivers as a result of the transport of fertiliser and pesticides by surface water runoff (Schmidt, 2000).

2.3.3.2 Chemical degradation

As a result of chemical degradation, toxicity and depletion of nutrients may occur (Dahlberg, 1994; Steiner, 1996). The loss of OM leads to many critical changes in the characteristics of soil that affect the essential biological, chemical, and physical processes influencing soil productivity (Bruce *et al.*, 1995). Its depletion results in the deterioration of the soil structure, water retention capacity and the release of nutrients (Steiner, 1996). The release of nutrients by soil is accompanied by loss of cations during the leaching process, which can lead to eutrofication of water supplies (Blaikie and Brookfield, 1987a).

Hydrogen ions released through decomposition of OM and root exudation result in acidification (Steiner, 1996). Usually, acidification develops slowly but once established it can severely damage root development of plants (Blaikie and Brookfield, 1987a). Hence, this reducing plant capacity for water and nutrient uptake (Steiner, 1996). On the other hand, as a result of acidification either a deficiency in P, Mo, Ca, Mg, and K, or a toxicity in Al, Mn and Fe can occur when soil pH drops below 5 (Van der Merwe and De Villiers, 1998). This can be harmful to plants and micro-organisms (Syers *et al*., 1995).

The buffering capacity of soil is to a large extent dependent on its CEC (Vaughan and Ord, 1985). Most of the exchangeable cations associated with CEC are plant nutrients (Havlin *et al*., 1999). Budelman and Van der Pol (1992) cited Janssen (1983) stating that, even if additional fertilizer is used, cropping ceases to be economically viable when the potential CEC of soil is less than a threshold value of 3-4 $cmol_c$ $kg^{-1}$. Under these circumstances, nutrient retention declines below the minimum necessary and leaching increases by a large margin (Steiner, 1996).

In agricultural systems, salinity is another important indicator (Pallo, 1993). Salinity increases osmotic pressure, which can have a detrimental effect on the germination of plants (Havlin *et al*., 1999). High osmotic pressure also inhibits water uptake by plants.

2.3.3.3 Biological degradation

Most of the time biological soil degradation occurs due to the depletion of the vegetation cover and OM in the soil (Steiner, 1996). The loss in vegetative cover is frequently combined with reduced biological activity and impoverishment of bio-diversity (Scherr, 1999), including microbial diversity (Miller, 1998). On the other hand, it is an indicative of the reduction of beneficial soil fauna (Steiner, 1996). There may also be a high build-up of parasitic nematodes (Lal and Stewart, 1992) and invasion of weeds (Blaikie and Brookfield, 1987a). The changes in soil quality and biota may have a catastrophic impact on the structure and function of the ecosystem as it may lead to reductions in the net primary productivity (Lal, 1998; Smith *et al.*, 2000).

### *2.3.4 Economic and social implications of soil degradation*

Soil degradation has complex economic and social implications (Blaikie and Brookfield, 1987b). Increased poverty, declining health, migration, marginalisation and a higher risk of conflict over natural resources that can lead into political instability are some of the potential implications of soil degradation (Dahlberg, 1994; Lal, 1998). Although there are no figures reporting the relationship between poverty and soil degradation, if one considers that rural communities are more dependent on agriculture, a number of factors suggests that soil degradation has a negative impact (Scherr, 1999).

People with fewer economic possibilities tend to be "pushed" onto poor land by political forces. This action aggravates soil degradation, as poor people will over-exploit the natural resources (Scherr, 1999). It feeds the so called "poverty trap" characterised by a spiral motion, where farmers move downward from high yields with low inputs to low yields with low inputs and low income (Steiner, 1996). If no actions are taken, world commodity prices and malnutrition may increase (Scherr, 1999).

Little information is available for assessing the economic effects of soil degradation, especially in developing countries. Economists tend to consider only the utilisation of the natural resource base, where natural resources are considered as providers and producers, but the deterioration of natural resources or the functional loss of ecosystem processes are ignored (Lal, 1998). However, it is known that producing crops from degraded soil requires far greater effort and cost as result of declined productivity of land and labour. These factors have been responsible for famine in agricultural areas in African countries (Blaikie and Brookfield, 1987b). On the other hand, decline in productivity can lead to a decrease in agri-based industries output, an increase in rural and urban unemployment, and a reduction in GAP and GDP (Lal, 1998). According to Scherr (1999) several authors estimated the economic effects of soil degradation. The values are crude varying annually from 26 to 28 billion dollars. Plant nutrient losses through sediment loss and N in water runoff were estimated at 5 billion dollar a year.

## 2.4 Restoration of soil fertility through fallowing

### *2.4.1 Importance of soil restoration*

The restoration and maintenance of high soil quality is important as it contributes to the improvement of the quality of the environment and economic progress (Lal, 1998). It can be achieved by abandonment of land after degradation, which allows for the possibility of soil to restore its productive attributes naturally after human-induced stress (Scherr, 1999). From the point of view of soil quality, this ability is called soil resilience (Eswaran, 1994). This process may occur in the form of forest and grass fallow (Blaikie and Brookfield, 1987a). During fallow, the soil eventually restores its physical, chemical and biological processes (Juo *et al*., 1995). According to Pekrun *et al*. (2003), an accumulation of OM in the upper layer, an increase in water infiltration after heavy rains, an increase in soil water content and a decrease in the loss of nutrients through run-off and erosion have been observed. These characteristics are the result of a lack of disturbance, which favours the maintenance of cracks and root channels, which will be converted into macropores.

From the perspective of an ecosystem, the main function of the fallow phase is essentially the transfer of mineral nutrients from soil back into the forest biomass (Juo and Manu, 1996). Then, the forest turns into the primary source of SOM through the plant residues, which act as input to the decomposer pool (Brady and Weil, 1996). Soluble nutrients released from the decomposition of litter are mainly retained in soil micropores from where they are taken up by plants (Juo and Manu, 1996). Most of the decomposition take place in the topsoil later, where activity of organisms is high. The litter quality and physio-chemical environment regulate the rate of decomposition (Swift *et al*., 1979).

### *2.4.2 Steady state condition*

Du Preez and Du Toit (1995) cited Tate (1992) who stated that a fertility equilibrium level under specific environmental conditions is reached in soil when OM inputs equal losses. This steady state level of OM depends on the site, soil and crop management practices applied (Swift *et al*., 1979; Andrén and Kätterer, 1997). When crop management practices are changed, a new OM level is attained that may be lower or higher than the previous level. This level depends entirely on the environment and quality of residues returned to the soil (Stevenson and Cole, 1999). The establishment of a new steady state level of OM can take decades (Pekrun *et al*., 2003).

Examples of SOM in a new steady state are scarce at higher levels but common at lower levels. Juo *et al*. (1995) found a decrease in soil organic C during the first 7 years of continuous maize cropping, which reached a steady state at about 65% of the level maintained by bush fallow. Du Preez and Du Toit (1995) reported a rapid rate of N fertility loss in warmer and drier regions during the first few years, whereafter the rate decreased until a new equilibrium was reached before 20 years of cultivation. They reported a similar pattern for the cooler and wetter regions but the new equilibrium was reached after 40 years of cultivation. Under similar conditions, Lobe *et al*. (2001) found a decline in the concentration of C and N as the period of cultivation increased and a new equilibrium was observed after 30 years. In semi-arid temperate conditions, Tiessen *et al*. (1994) suggested a new equilibrium, with a 50% reduction in OM only after 65 years.

### *2.4.3 Experiences of soil restoration in the tropics*

In the recovery of soil fertility, critical issues such as the following should be addressed (Zinck and Farshard, 1995): (i) What is the threshold value for the soil nutrient content to be considered sufficient to sustain crop production? (ii) What is the length of time the land needs to restore nutrients to a satisfactory threshold level? (iii) What is the process of soil nutrient recovery? Some attempts have been made to answers these questions.

The important measure during the restoration of soil fertility is the identification of soil-related constraints on crop production (Lal, 1998). Bruce *et al*. (1995) established a soil organic C range of 0.23 to 1.43 % as an indicative value for the primary limitation of soil productivity. According to Snapp *et al*. (1998) soils with less than 90% sand require a minimum 0.9% organic C. They suggested that 1.0 to 1.5% organic C will be ecologically viable in sandy soils over the long term.

Scherr (1999) stated that biological and nutrient problems can be solved over a time span of 5-10 years. However, the length of fallow needed and the ability of a system at a certain site to restore soil fertility depend on several factors: the severity of the degradation (Scherr, 1999); diversity of species and soil type (Juo and Manu, 1996); topography and climate (Brand and Pfund, 1998); population pressure (Folmer *et al*., 1998); type of land management (Zinck and Farshard, 1995) as well as the nature of the nutrient concerned (Brand and Pfund, 1998).

The process of degradation has different levels of severity varying from largely reversible to largely irreversible. Nutrient depletion which results often in imbalances are reversible processes (Scherr, 1999). However, there are systems that are able to restore soil fertility to sustainable levels while others are not. The ideal period for restoration of soil fertility in a certain system is mainly dependent on the ability of the modified system to recycle and conserve nutrients (Juo and Manu, 1996).

Woomer *et al*. (1998) found in southern Cameroon that over a period of 22 years a 100% re-establishment of 95% of the vegetation species, and that more than 60% of plant biomass and 70% of C stock were restored when using virgin forest of more than 160 years old as a reference. In other studies it was found that over 10 years of restoration, biomass production ranged from 48 to 160 ton $ha^{-1}$ (Juo and Manu, 1996; Woomer *et al*., 1998).

At Bafarona in Madagascar, the amount of litter increased with age and represented more than 25% of the above ground phytomass in the first five years of a fallow experiment (Brand and Pfund, 1998). In the same experiment, the nutrient concentration of the litter showed great variation especially with regard to exchangeable cations, as did the type of vegetation. During the referred period, a decrease in organic C and exchangeable cations in the top 200 mm soil was observed. From there onward, an increase in organic C and exchangeable cations were recorded while Al toxicity decreased.

For many soil types there is still a lack of knowledge about thresholds of soil quality indicators below which investment in restoration is uneconomic (Scherr, 1999). Juo *et al*. (1998) could find no significant changes in Ca and Mg or effective CEC, despite an increase in SOM content after 13 years of fallowing. The same authors referred to a study by Wadsworth *et al*. (1990) who noticed a similar behaviour after 50 years of fallowing an Ultisol in Mexico. Ultisols, the same as Oxisols, have limited reserve exchange bases, and nutrient uptake by fallow vegetation may lead to a decline in pH (Juo and Manu, 1996). The decline in pH may also result from the acidic exudates by root plants (Vaughan and Ord, 1985). Tiessen *et al*. (1994) found no root development below 400 mm depth of a Ferralsol in Brazil as a result of Ca and P deficiency. Ferralsols have a low capacity for resilience and easily loose OM from the topsoil. Furthermore, they have a strong acidity and low supply of available nutrients with almost no reserves of weatherable minerals (Scherr, 1999).

In a secondary forest subject to slash and burn it may take hundreds of years to produce the equivalent amount of biomass as in a primary forest (Juo and Manu, 1996). Tiessen *et al*. (1994) cited Saldarriaga (1988) who found that the basal area total biomass of a mature forest was reached only after 190 years of fallow.

Inadequate soil management has lead to the extension of agricultural cropping areas into land ecosystems that are not suitable for agriculture (Blum, 1997). The number of cycles of slash and burn, their duration, and ultimately the length of time the land remains under repeated slash and burn cultivation, are critical to the desirability and sustainability of this system (Harwood, 1996). It has been commonly stated that due to today's demographic and economic pressures, the shortened cycles of fallow for soil regeneration, especially nutrients are often not sufficient to maintain productivity of shifting cultivation. However, it is fully recognised that shifting cultivation, if properly practised, can still be sustainable (Harwood, 1996).

### *2.4.4 Factors affecting the process of restoration*

Several authors consider bush fallow as the most effective way of restoring soil fertility as it is efficient in the accumulation of biomass and in the recycling of nutrients due to the many species with different types of root systems (Juo *et al.*, 1995). Prinsloo *et al*. (1990) found that the reversion of cultivated land to pasture appeared to restore fertility only where leguminous trees were present. This agrees with Snapp *et al*. (1998) who stated that legumes with high quality residues and deep root systems are most effective in improving nutrient cycling. The same authors added that the quality of residues is also an important factor in the restoration process. High quality residues, viz. those with C/N ratios of less than 10 and low polyphenolic and lignin contents, increase soil microbial activity, P and micronutrient availability, and soil buffering capacity (Snapp *et al*., 1998). The microbiological activity can also be promoted by addition of OM (Warkentin, 1995).

The human population density determines the buffering capacity breakdown (BCB), which was defined as the relationship between nutrient depletion and nutrient resources (Folmer *et al*., 1998). In many densely populated areas of the tropics, bush fallow can no longer provide for the basic needs of farming communities (Juo and Manu, 1996) because the BCB is moderate to high (Folmer *et al*., 1998). In less densely populated areas, fallowing is still a central focus for farming, as small-scale farmers cannot afford to buy fertilisers (Snapp *et al*., 1998). However, insufficient nutrient management is supposed to be a major constraint in shifting cultivation.

Rates of nutrient cycling associated with SOM turnover under primary or secondary forest may provide a predictive tool for evaluating the potential of soils for agriculture and subsequent forest recovery (Tiessen *et al*., 1994).

## 2.5 Soil organic matter and soil fertility recovery

### *2.5.1 The role of soil organic matter*

One of the most dynamic components of soil is OM (Brady and Weil, 1996). It contributes to fertility through its influence on the physical, chemical and biological properties of the soil (Vaughan and Ord, 1985). Therefore, soil fertility depends to a large extent on its OM content (Alvarez, 2001). Organic matter improves soil structure by stabilizing soil aggregates, increases water holding capacity, buffers pH, chelates metals, interacts with xenobiotics, and retains cations and anions in the soil system (Smith *et al*., 2000). These are the reasons why quantification of SOM is important in evaluating the sustainability of an agro-ecosystem (Alvarez, 2001). Therefore, OM is a vital component in the dynamic relationship between degradation of soil and soil conservation practices (Parr *et al*., 1990).

The improvement of soil structure by OM results from its association with clay, which strengthen aggregates through the cementing of soil particles (Stevenson and Cole, 1999). This results in many benefits to the soil. It helps to minimise crusting, compaction, run-off and erosion (Van der Merwe and De Villiers, 1998). On the other hand, it lowers the bulk density, increases aeration and drainage and improves water infiltration, water holding capacity and root development (Vaughan and Ord, 1985). The root development is improved partly because SOM can hold up 20 times its weight in water (Stevenson and Cole, 1999), which is in available form for plants (Brady and Weil, 1996; Steiner, 1996). This results from colloidal properties, which create an impact that is very important especially in sandy soils (Vaughan and Ord, 1995).

SOM also acts as a source of nutrients for plants (Tiessen *et al*., 1994). It releases plant nutrients such as N, P and S slowly by a process of mineralisation, which convert them to a form available for plant growth (Steiner, 1996). This prevents their leaching to some extent. In addition, humic substances form stable complexes with some metals and thus influence their availability to plants and micro-organisms (Vaughan and Ord, 1985).

The CEC is also strongly influenced by OM, especially in soils with low sorption clay minerals (Steiner, 1996). There is usually a direct relationship between OM and CEC of soils (Vaughan and Ord, 1985). It is estimated that 20 to 70% of the CEC in many soils may be attributed to the contribution of OM (Stevenson and Cole, 1999). The CEC derived from OM contributes to the buffering capacity of soil pH as it holds cations such as H, Al, Fe, etc. (Vaughan and Ord, 1995).

Biological properties of soil are also improved by OM since it enhanced the activity of organisms usually (Smith *et al*., 2000). Soil organisms promote soil aggregation (Snapp *et al*., 1998) and their activity results in the release of plant nutrients from OM, which influences the buffering capacity of the soil and improves its water holding capacity (Vaughan and Ord, 1985).

The content of OM is related to other soil properties such as texture (Smith *et al*., 2000) and the quality of organic inputs (Snapp *et al*., 1998). It is difficult to separate the effects of OM and clay on other soil properties (Smith *et al*., 2000). An optimum OM content and a favourable C/N ratio are key factors in developing and maintaining the physical, chemical and biological properties of soil (Van der Merwe and De Villiers, 1998). High quality of organic residues inputs with a low C/N ratio are low in lignin and polyphenol, and have a high percentage of N. Both lignin and polyphenol are important factors in controlling N release from leguminous organic compounds (Snapp, *et al*., 1998).

### *2.5.2 Factors affecting organic matter and nutrient dynamics in soil*

The organic transformation of organic residues to humus in soil is very dynamic (Stevenson and Cole, 1999) and many processes determine the decomposition rate of organic residues and the rate of humus formation (Juo and Manu, 1996; Steiner, 1996). These processes depend on natural factors and management practices. Due to the purpose of this study, the focus will be on shifting cultivation.

#### 2.5.2.1 Natural factors

Natural factors that affect the dynamics of OM and nutrients in the soil are site dependent and act interactively (Du Preez and Du Toit, 1995). Stevenson and Cole (1999) cited Jenny (1930) who stated that the magnitude of their importance is as follows: climate > vegetation > topography = parent material > time.

Climate influences SOM content primarily through temperature and rainfall (Smith *et al*., 2000). These factors determine the aridity index (AI), which is defined as the ratio between the mean annual rainfall and the mean annual evaporation (Du Preez and Wiltshire, 1997). Hence, the warmer and drier regions have a low AI while the cooler and wetter regions have a high AI, characterised by the production of lower and higher biomasses respectively (Brady and Weil, 1996). These differences in biomass production play an important role in the dynamics not only of SOM but also in the cycling of nutrients (Du Preez and Du Toit, 1995). In cultivated soil OM and mineralisable N in warmer and drier agro-ecosystems reach a new equilibrium sooner than in cooler and wetter agroecosystems, namely 5-10 years versus 40-60 years.

Rainfall and temperature are important regulators of OM decomposition (Swift *et al*., 1974). Lomander *et al*. (1998) found that an increase in soil water and temperature increased the decomposition of OM. Similarly, in high rainfall regions with a high AI, nutrient depletion with significant losses of N through leaching is common (Henao and Baanante, 1999). In low rainfall regions with a low AI, OM decomposes mainly during periods of favourable soil water conditions (Stewart and Robinson, 1997).

As stated by Stevenson and Cole (1999), vegetation is the factor second in order of importance that influences the dynamics of OM and nutrients in soil. A variation in OM content can be found in virgin topsoil as a result of different botanical basal ground cover and biomass production (Du Preez and Wiltshire, 1997). Less vegetative cover produces subsequently low biomass (Steiner, 1996). In semi-arid areas in Africa, recycling of nutrients is low and nutrients tend to accumulate very slowly in soils under savannah vegetation (Henao and Baanante, 1999).

Topography and parent material are rated third and have therefore the same magnitude of influence on SOM. Topography influences SOM through its impact on the microclimate, drainage and erosion (Smith *et al*., 2000), whereas parent material determines the composition of the mineral component of the soil. In sub-humid to semi-arid tropical regions soils are usually highly weathered and rich in sesquioxides, which results in phosphate fixation (Steiner, 1996). Juo *et al*. (1995) cited several authors who stated that OM content under continuous cultivation in strongly weathered and low CEC soils often declines rapidly. Granite-derived sands have low organic C, whether in a cultivated or uncultivated state due to their limited physical and chemical protection of OM against oxidation (Snapp *et al*., 1998). Juo and Manu (1996) referred to Wong *et al.* (1987) who stated that the presence of positive charges in the surface layers of many Oxisols and Ultisols might result in a slower rate of downward movement of anions such as nitrate and sulfates. In a study conducted in a sisal plantation at Tanga in Tanzania it was found that differences in the decline of nutrient content among the major soil groups were large (Hartemink, 1997).

Soil texture plays also an important role in the cycling of OM and nutrients. A positive relationship between OM and clay is well established (Smith *et al*., 2000). There is consistency in the amount of OM with the enrichment of clay in sandy soils of sub-tropical and temperate climates (Tiessen *et al*., 1994). In a study conducted by Lobe *et al*. (2001) it was found that the concentration of OM decreased in the following order: clay>silt>sand. This agrees with Du Preez and Du Toit (1995) who found that organic C and N of virgin soils increased linearly with an increase in fine silt-plus-clay content. The effect of humus in increasing the CEC of soils is more important in sandy than clayey soils because it is difficult to maintain a high level of humus in sandy soils when cropped (Vaughan and Ord, 1978).

### 2.5.2.2 Shifting cultivation

#### *2.5.2.2.1 Process of shifting cultivation*

Shifting cultivation is the farming system mostly practised by subsistence farmers in the traditional communities of the tropics (Yemefack *et al*., 2006; Mertz *et al*., 2008). In shifting cultivation flash burning and short-term mixed intercropping follow the temporary clearing of vegetation in a primary forest or a forest in the process of regrowth, with its eventual return to natural succession (Prinsloo *et al*., 1990; Harwood, 1996). The natural succession is normally used for bush fallow or pasture (Tiessen *et al*., 1994). However, the amount of vegetation restored has been often below what is needed for soil regeneration (Mokwunye, 1996).

Shifting cultivation in tropical soils can have detrimental effects. In Burkina Faso, traditional sorghum cultivation practices changed the composition of SOM. The fine fractions were oxidised, favouring the accumulation of fractions that have undergone little decomposition (Lal *et al.,* 1999). Tiessen *et al*. (1994) found in an extremely nutrient poor Amazonian soil, no potential for agriculture beyond the three-year lifespan of the forest litter mat. The interpretation they gave was that this was caused by the interruption of biological nutrient cycles by slash and burn.

The studies referred to in the previous paragraph were conducted on tropical soils, which are similar to some soils that occur in southern Mozambique. Tropical soils have very few weatherable minerals and nutrients are therefore easily depleted. Regular nutrient inputs are required to sustain cropping (Hartemink, 1997). However, Snapp *et al*. (1998) cited Heisey and Mwangi (1996) who reported that due to current economic constraints such as the removal of subsidies, exchange rate devaluation and high inflation, most subsistence farmers in areas such as sub-Saharan Africa are still using low yield crop varieties and the addition of external inputs is low. Consequently, nutrient depletion is an important limiting factor for the sustainability of shifting cultivation (Brand and Pfund, 1998; Nandwa and Bekunda, 1998). Ultimately, the inadequate replacement of removed nutrients, coupled with the continuous loss of OM from the soil, contributes to the increasing decline in soil fertility which leads to soil degradation (Henao and Baanante, 1999).

*2.5.2.2.2 Effects of vegetation clearing*

The process of converting land for crop production by means of deforestation has severe ecological and environmental implications (Lal and Stewart, 1992). Deterioration of the vegetative cover and rapid decline of OM in the topsoil is observed after forest clearance (Steiner, 1996; Blum, 1997). These result from direct exposure of soil to the impacts of sun, wind and water (Schmidt, 2000). Then, OM stored in the soil decomposes rapidly to such an extent that during the first 5 years its content can easily be reduced by 20-50% (Steiner, 1996).

*2.5.2.2.3 Effects of biomass burning*

Burning has beneficial and detrimental effects. Beneficial effects are the ash, which contains Ca, Mg, K, P and micronutrients that can be extracted by plants (Brady and Weil, 1996). In addition there is a rapid increase in the pH and ECEC of soil (Juo and Manu, 1996). Detrimental effects are loss of C, N and S that contribute to an increase in their concentration in the atmosphere (Steiner, 1996). According to Smith *et al*. (2000) the amount of nutrients added to the soil during combustion is relatively high but they are prone to be lost through volatilisation, leaching and run-off if the soil system is unable to retain them or if plant roots do not absorb them quickly. In general cropping systems are not efficient in the use of available plant nutrients and about one third is lost during a crop cycle (Steiner, 1996). The loss of P is more prominent compared to other nutrients in certain soils. Juo and Manu (1996) cited Jordan and Szott (1991) who stated that in weathered soils, P could become the first limiting nutrient for biomass production because burning is detrimental to P mobilisation from the mycorrhyzal-root association and other P mobilising soil micro-organisms.

*2.5.2.2.4 Effects of cropping*

The OM content of soil can vary from less than 1% in young soils and those exhausted by intensive cropping, to as much as 95% in deep peats (Vaughan and Ord, 1985). When a virgin soil is cultivated, the OM content declines rapidly during the first 10 years and continues at a gradually diminishing rate for several decades (Stevenson and Cole, 1999). Du Preez and Wiltshire (1997) referred to Du Toit *et al.* (1994) who found a loss of 10-75% of OM in rainfed, cultivated topsoils in South Africa, using virgin soils as a reference. Similar results were reported by several other authors comparing OM content in the 0-50 mm soil layer, where non-tilled plots were used as reference (Pekrun *et al.*, 2003).

Juo and Manu (1996) cited Andriesse and Schelaas (1987) who stated that as a result of the practice of shifting cultivation, nutrients stored in the forest biomass are released partially by decomposition and mineralisation of plant residues. Cultivation increases both processes, which result from greater oxidative processes, due to exposure of previously inaccessible OM to micro-organisms and oxygen through physical disruption of soil aggregates (Prinsloo *et al.*, 1990). On the other hand, water and temperature tend to be more favourable for decomposing organisms in deeper than shallower soil layers, resulting in a higher rate of decomposition and thus a rapid decrease of OM when crop residues are incorporated (Pekrun *et al.*, 2003). In addition, the contact areas between soil particles and organic residues are increased (Pekrun *et al.*, 2003). This agrees with results in a report cited by Costa *et al.* (1990) that the release rate of N from residues placed on the surface was slower than when incorporated residues.

The mineralisation of OM in the plough layer results in a reduction of the organic nutrient pool which releases some plant available nutrients (Du Preez and Du Toit, 1995). These nutrients are removed from the soil by growing plants (Pekrun *et al.*, 2003). Ultimately, a portion of the nutrients contained in the plants is removed during harvesting (Juo and Manu, 1996). However, replacement of such nutrients is very little or non existent (Henao and Baanante, 1999)

**2.6 The need for research to halt and reverse soil degradation**

Soil degradation usually starts as a subtle and slow process whereof the rate depending on the buffering capacity of the soil (Warkentin, 1995). As soon as a certain threshold is exceeded the deterioration proceeds quickly (Lal and Stewart, 1992). Once this stage is reached, the process is mostly irreversible (Smith *et al.*, 2000). Then, soil degradation leads to desertification (Stewart and Robinson, 1997; Lal, 1998). Prevention of this is of utmost importance considering that soil is effectively a non-renewable resource and there is no hope that degraded soils can be restored within a time span that bears any relationship to human history (Steiner, 1996). One inch of soil, which can be lost within a year, takes thousand years to form (Thripathi and Singh, 1993). It has been estimated that between 5 to 12 million hectares of agricultural land are lost annually worldwide due to degradation, which represents approximately 0.3 to 1.0 percent of the world's arable land (Scherr, 1999). Hence, there is a need to understand the causes behind soil degradation (Dahlberg, 1994).

Soil degradation takes place all over the world but its effects are more marked in tropical soils (Steiner, 1996). For instance, the land in Africa has predominantly fragile ecosystems (Henao and Baanante, 1999). About half of the land is unsuitable for low-input agriculture because most of the soils are coarse-textured, and inherently low in OM, fertility and water holding capacity (Eswaran *et al.*, 1997). This situation is worsened by the fact that the rainfall patterns are erratic and unpredictable (Parr *et al.*, 1990). The current average application of nutrients in Africa through fertilization is annually less than 10 kg NPK $ha^{-1}$ while the estimated application needed for 1993-1995 to attain production levels was about 40 kg NPK $ha^{-1}$ every year (Henao and Baanante, 1999). Soil fertility has chiefly been maintained through shifting cultivation

(Kayombo and Mrema, 1998), which has destructive effects as described in section 2.5.2.2. Therefore, these lands are highly susceptible to desertification (Parr *et al.*, 1997). This situation raises concerns because about 65% of people depend on agriculture, which is practised basically through shifting cultivation, for their livelihood (Henao and Baanante, 1999).

According to Woomer *et al.* (1998), the current policy options aimed at reducing the destructive effects of shifting cultivation caused by slash and burn do not address the problems encountered on a daily basis by farmers who practice it. The main concern is "how can farmers better manage their lands and forests allowing for extended cropping intervals, greater biomass storage on their lands and protection of useful indigenous species"? For correct decisions to be taken, there is a need for the dissemination of information identifying regions where policy interventions are needed (Henao and Baanante, 1999). Practices should be based on the analysis of the causes in each particular case and approaches must be developed to draw solutions based on findings (Bruce *et al.*, 1995; Steiner, 1996).

The development of solutions is constrained by gaps in knowledge (Syers *et al.*, 1995). Lack of scientifically established criteria is adversely affecting the design and evaluation of meaningful soil management programmes (Seckler, 1987; Arshad and Coen, 1992). Knowledge of the stocks, balances and flow of nutrients is widely lacking (Brand and Pfund, 1998). For instance, due to a lack of information, when studying soil fertility depletion in Mozambique, Folmer *et al.* (1998) had to rely on rates of nutrient build up assumed by other researchers when they calculate nutrient flow. Therefore, more research is urgently needed, considering the fact that soil degradation is likely to become more problematic and that the majority of the population is dependent on its local environment (Dahlberg, 1994).

An extremely broad range of agricultural and agro-forestry land-use alternatives to the current shifting cultivation system exist but much scientific work needs to be done on the less fertile soils (Harwood, 1996). The prime challenge for research and development is to retard and if possible prevent loss of OM and nutrient depletion from soils in agro-ecosystems (Zinck and Farshad, 1995; Steiner, 1996). Establishing of easily measured indicators for OM turnover and nutrient cycling in natural and cultivated soils, are essential for decisions on land use and management, where measurements of static nutrient pools are inadequate (Tiessen *et al*., 1994).

Little effort has been directed towards understanding how the spatial and temporal variability of the physical, chemical and biological properties coupled to human factors influence soil quality (Halvorson *et al*., 1997). For instance, the conservationists have paid more attention to physical soil erosion, which cannot be solved without finding a solution to chemical soil erosion (Kayombo and Mrema, 1998). There is a need for better understanding of all facets of soil quality to be able to develop better indicators of trends or changes in soil quality that result from different management systems (Kennedy and Papendick, 1995). This will help to implement new agricultural practices in order to decrease pressure on the remaining forests, reclaim degraded forests and the conservation of C resources in land and soil undergoing conversion to agriculture (Woomer *et al*., 1998).

## 2.7 Assessment of soil properties in fallow lands

### *2.7.1 The aim of assessment*

The continuous assessment and monitoring of plant nutrients in agricultural soils is a way of quantifying changes in soil quality over time (Mausbach and Seybold, 1998). It is a very important tool in evaluating the sustainability of agricultural management systems (Van der Merwe and De Villiers, 1998). Assessment helps to understand the causes of the nutrient depletion process, which is important in identifying the correct measures aimed at reversing the decline in fertility as well as of degradation (Henao and Baanante, 1999). On the other hand, it can help to identify marginal and unsuitable lands, which should be managed in a specific manner to preserve their quality and avoid further degradation (Eswaran *et al*., 1997). Unfortunately, developing countries in general do not have national monitoring systems for soil quality (Scherr, 1999). This is important especially in countries with soils of variable quality where a detailed national assessment is recommended (Eswaran *et al*., 1997).

There are many levels of soil quality assessment. They may vary from visual observation to complex laboratory analysis and calculation of soil quality indices (Mausbach and Seybold, 1998). Laboratory analyses, which include sampling, appear to be very expensive but reclaiming degraded soils, when the damage exceeds a certain threshold, can be even more expensive (Crépin and Johnson, 1993).

### *2.7.2 Selection of variables, sampling and data analysis*

2.7.2.1 Variables to be selected

Van der Merwe and De Villiers (1998) cited Larson and Pierce (1994) who stated that soil quality itself cannot be monitored but it is possible through the identification and monitoring of key variables that influence the quality of the system. Hence, scientists need to press for new and continuing schemes for monitoring indicators of sustainable land management (Syers *et al*., 1995). Many soils in the tropics are poor in inorganic nutrients and maintenance of fertility is based on the recycling of nutrients from OM. Therefore, the use of inorganic nutrients alone is not a sufficiently reliable procedure (Tiessen *et al*., 1994).

All processes and interactions that affect soil quality should be considered during assessment (Blum, 1997; Halvorson *et al*., 1997). The assessment should comprise of indicators that can link ecological processes, including spatial and temporal patterns in agro-ecosystems (Smith *et al*., 2000). Temporal trends in critical indicators are more useful than one observation in time (Syers *et al*., 1995). Sensitive indicators are good predictors or early-warning tools that can be used as tools for extrapolation, such as simulation models (Zinck and Farshad, 1995). For example, in a study of shifting cultivation in southern Cameroon, Yemefack *et al*. (2006) suggested that data gathered after a 5 year cropping period are used as a baseline. However, indicators that reflect the condition of the ecosystem and its status have not yet been rigorously defined and researched (Syers *et al*., 1995). Indicators should be defined based on research into mechanisms and processes of interaction among the various components of the system (Syers *et al*., 1995).

Bruce *et al.* (1995) suggested that sustainable productivity of a site can be evaluated based on variables associated with climate, soil and hydrological characteristics. Climatic data, for instance, solar irradiation, rainfall and other factors can vary greatly (Kimble, 1998). Rainfall and temperature are important as they influence the physical, chemical and biological processes in the soil (Eswaran *et al.*, 1997). In addition, if climate determines the limits of ecological functioning of soils, then the optimal function would need to be calculated for each eco-region (Warkentin, 1995). Some of the attributes that are primarily affected by the process of soil degradation are: bulk density, OM, pH and availability and retention of nutrients (Arshard and Coen, 1992).

Yemefack *et al.* (2006) stated that, as a result of complex spatial and temporal changes in the characteristics of soil, it is difficult to obtain a comprehensive set of data for quantitative predictions of soil fertility for shifting cultivation in the tropics for the medium and long term. They cited Larson and Pierce (1991) who proposed the concept of a minimum data set (MDS) for evaluating soil quality, of which the quantitative attributes could be measured for management decisions. There are many proposals for MDS but all are subject to analysis (Kennedy and Papendick, 1995).

The MDS for the interpretation of management practices and land use changes in southern Cameroon comprise of five variables: bulk density, organic C, pH, exchangeable Ca and available P (Yemefack *et al.*, 2006). In a rain-fed agricultural system in Australia, the major indicators for sustainable agriculture are soil health, pH, nutrient balance and biota (Syers *et al.*, 1995). In a study aimed at identifying soil quality indicators using factorial analysis, it was concluded that, if only one soil quality attribute were to be used in monitoring soil quality every 3-5 years, SOM should be selected as that attribute (Shukla *et al.*, 2006). A MDS should be related to a specific research project and has to be considered the minimum information needed by the projected end user (Kimble, 1998). For a detailed assessment of land use, quantitative soil assessments can be made (Eswaran *et al.*, 1997). Such assessments should be based on standard reference values (Zinck and Farshad, 1995). However, these reference values are not always available (Syers *et al.*, 1995).

2.7.2.2 Sampling method and laboratory analysis

A sampling procedure for soil quality assessment should fit into a spatial and temporal framework (Mausbach and Seybold, 1998). Sampling areas and intervals should be of such a nature that it will facilitate the identification of trends (Halvorson *et al*., 1997). The periodic evaluation of nutrient requirements, balances, and rates of depletion in agricultural areas is a key component of this effort (Henao and Baanante, 1999). On the other hand, the sampling procedure can differ depending on the objective and specificity of the site of research and the availability of funds. According to Crépin and Johnson (1993), sampling may be exploratory sampling, simple random sampling or composite sampling. They explain that for studies on biological and chemical soil properties, where the objective is to calculate the average value, composite sampling is recommended and it minimises costs. In this type of sampling, field samples representing the soil of a certain population are thoroughly mixed to create a single composite sample from which sub-samples are derived for laboratory analysis. Laboratory analysis has the advantage of facilitating the differentiation of soil restoration and soil degradation processes (Kennedy and Papendick, 1995).

2.7.2.3 Depth of soil to be assessed

Assessment of soil degradation and soil restoration concentrate mainly on the topsoil. Most of the processes in soil take place here since the surface soil receives solar energy, rainfall and organic materials. In addition, there is a higher possibility of managing all these factors in the topsoil to reduce degradation or induce restoration (Bruce *et al*., 1995). Changes in biological indicators of soil quality such as microbial biomass, microbial activity and soil fauna are usually restricted to topsoil (Crépin and Johnson, 1993). Nutrients and plant roots are also mostly concentrated near the surface, especially in non-tilled soil (Pekrun *et al*., 2003).

The depth to which soil quality deserves special attention is in most instances prescribed by local conditions together with the aim of the investigation. Studies in this regard focussed inter alia on the upper 150 mm (Juo and Manu, 1995; Brady and Weil, 1996), 200 mm (Costa *et al*., 1994; Brand and Pfund, 1998) and 300 mm (Crépin and Johnson, 1993). Crépin and Johnson (1993) and (Govaerts, 2005) stressed that to avoid the dilution effect at deeper layers, which can increase sampling and analytical errors, it is important to assess the 0-50 mm layer. Assessment of soil quality below the root zone will be in most instances not important (Halvorson *et al*., 1997)

2.7.2.4 Quality assurance and data analysis

Depending on local characteristics and temporal variables, the assessment of land degradation cannot be done using simplified definitions because they do not link the processes involved in the different dry lands of the world (Dahlberg, 1994). Comparisons of soil quality should be made between land units and soil properties concerned of similar spatial variability and temporal scale (Halvorson *et al*., 1997). Dahlberg (1994) cited Carpenter (1984) who stated that a small scale sample survey or site-specific physical measurement could contain uncertainties. In addition, there cannot be a proper interpretation of short-term data in the absence of long-term data (Kimble, 1998). Several researchers have applied a time span of 5 years (Brand and Pfund, 1998; Smith *et al*., 2000; Yemefack *et al*. 2006).

The Environmental Protection Agency of the United States of America uses the traditional quantity to quality control concept, which includes amongst various measures the use of duplicate and calibration samples (Crépin and Johnson, 1993). This agrees with Costa *et al*. (1990) who collected 3 replicates for each of the treatments on every sampling date. The reporting of the data set should be in average values because, besides saving space, most models run on weekly or monthly averages (Kimble, 1998). For example, during quantification of SOM it is advisable to express it as mass of C or N per unit area, where the average mass of SOM per mass of soil in the sampling depth is characterised (Smith *et al*., 2000).

### *2.7.3 Modelling litter decomposition and nutrient dynamics in soil*

The conceptualisation and assessment of suitable land management should be converted to alternative models (Zinck and Farshard, 1995). The models should be calibrated and the limits of their applications should be established (Miller and Wali, 1995). The importance of land management models and strategies for predicting soil behaviour should be reviewed due to the weak link in some global ecosystem management models in the terrestrial component (Miller and Wali, 1995). For instance, although N cycles have been measured and modelled, such research focused mainly on non-cultivated soils (Warkentin, 1995).

Several models can help to predict the SOM content as a function of spatial and temporal variability: Century (Parton *et al*., 1987), SOILN (Eckersten *et al*., 1996), ICBM (Andrén and Kätterer, 1997) and Q-Model (Rolff and Ågren, 1999). To be able to run these models and others, the pattern of the rate of input and decomposition of organic material are needed and this is dependent on the environment (Snapp *et al*., 1998). Therefore, data describing the dynamics of SOM, including the quality of litter, in the system should be available (Swift *et al*., 1979).

## 2.8 Conclusions

Based on all aspects, the following conclusions may be drawn:

- Soil degradation is a process that can result from natural or land management causes. It needs to be addressed urgently as it is gradually increasing and constitutes a threat, especially in semi arid ecosystems.

- Due to socio-economical constraints, the traditional farming system of shifting cultivation is still the only alternative for the restoration of soil fertility in certain regions and, if properly practised, can still be sustainable.

- Soil scientists should consider SOM and nutrient cycling in shifting cultivation, especially in fragile ecosystems, as one of the priorities in their research agenda.

- There is a need to generate data with spatio-temporal characteristics that can be used in predictions of the fertility dynamics of soil under shifting cultivation.

- The assessment of soil fertility in fallow lands under shifting cultivation is an urgent and important issue as it can be used for future predictions.

- The assessment of soil degradation and restoration should concentrate in the upper 150 to 300 mm soil layer, with special attention to the first 50 mm because most processes occur here.

- Modelling is an important tool that can be used for predictions of soil fertility dynamics in shifting cultivation practices.

# CHAPTER 3

## Vegetation composition and biomass production in bush fallow lands in southern Mozambique

### 3.1 Introduction

Shifting cultivation is a farming system in which a plot of land under primary forest or a forest in the process of regrowth is cleared for cropping and when soil fertility declines after some years, the land is bush-fallowed (Beets, 1990; Harwood, 1996). The intention with bush fallow is to recover the fertility lost during the cropping phase through natural succession (Kayombo and Mrema, 1998; Nandwa and Bekunda, 1998). The success of this process is site dependent and therefore determined by natural factors interacting at the site (Juo and Manu, 1996).

Several authors consider bush fallow to be the most effective method of restoring soil resilience as it is efficient in the accumulation of biomass and in the recycling of nutrients due to the many species with different types of root systems (Juo *et al.*, 1995). The vegetation that grows during the fallow phase also plays a critical role in other ways: (i) It prevents direct exposure of soil to the impact of sun, wind and water, especially raindrops (Schmidt 2000); (ii) it favours the maintenance of cracks and root channels that are converted into macropores due to the absence of soil disturbance (Pekrun *et al.*, 2003); and (iii) it contributes to the organic matter content of the soil when residues of the growing plant are decomposed and mineralised (Brady and Weil 1996; Blum 1997; Lal *et al.*, 1999; Mills *et al.*, 2005).

The organic matter added to soil during bush fallow is of critical importance in restoring soil fertility (Alvarez, 2001). This is because organic matter influences many of the physical, chemical and biological characteristics contributing to a productive soil (Bruce *et al.*, 1995; Stevenson and Cole, 1999). For example, it improves soil structure by strengthening aggregates, which is beneficial for water infiltration and retention since it results in less runoff and erosion after heavy rains (Smith *et al.,* 2000). Organic matter also releases nutrients (Steiner, 1996), buffers pH, chelates metals and retains cations and anions in the soil system (Smith *et al.*, 2000). On the other hand, organic matter usually promotes the biological activity of soils since it enhances the activity (Scherr, 1999) and diversity (Warkentin, 1995; Miller, 1998) of the resident organisms.

Most small-scale farmers in southern Mozambique practise shifting cultivation as their main farming system. They usually clear a vegetated area by slashing and burning before planting crops for a period of 3 to 5 years. The cropped land is abandoned to bush fallow on account of a reduction in soil fertility to insufficient levels (MA/FAO, 1983; Geurts, 1997; DPADRI, 2002). This situation is likely to continue due to financial limitations that prevent farmers from buying fertilisers (MAP, 1996), combined with the existence of vast areas that are sparsely populated (Snijders, 1985; MAP, 1996). Hence, it is crucial to characterise and understand the changes in vegetation and in factors affecting it in the bush fallow phase. During this phase, vegetation diversity and biomass production are vital for the recovery of soil fertility.

The objectives of this study were to compare the vegetation composition and biomass production in bush fallow fields of different ages within each selected agroecosystem and in bush fallow land of similar ages across these agroecosystems.

## 3.2 Material and methods

### *3.2.1 Study area*

The study was carried out in the Inhambane and Gaza provinces located in southern Mozambique (Figure 3.1). Elevation in this area ranges from sea level in the east (coast) to 1000 m in the west (inland). Mean annual rainfall decreases from more than 1000 mm at the coast to less than 400 mm inland, whereas mean annual temperature increases from 20ºC to 26ºC. Based on moisture index four climatic zones cover the study area from the coast to inland, namely sub-humid, wet semi-arid, dry semi-arid and arid (Reddy, 1985; 1986). The natural vegetation of miombo woodlands near the coast is replaced by arid savanna inland (Campbell *et al.*, 1996) with a mean annual rainfall of 700 mm as a rough threshold (Frost, 1996).

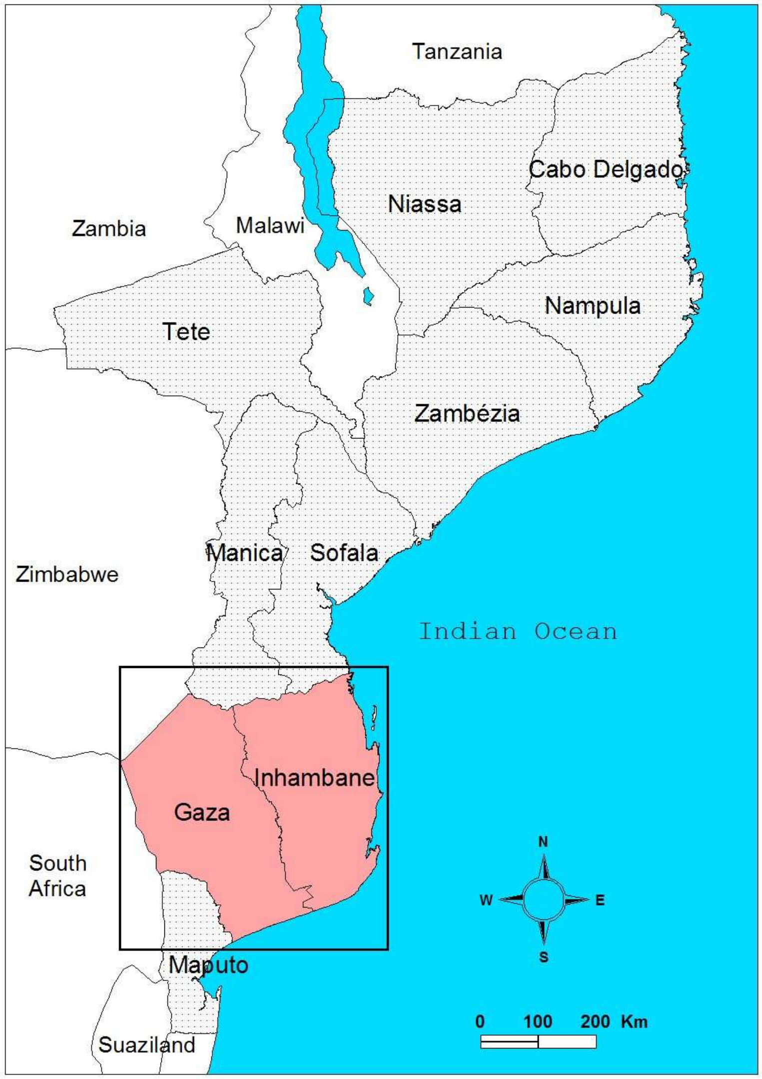


**Figure 3.1** Location of Inhambane and Gaza provinces in southern Mozambique

In terms of geology, both provinces are part of the Mozambican sedimentary basin (Flores, 1973). This basin appeared during the opening of the Indian Ocean and eastern Africa continent margin (Salman and Abdula, 1995). In both provinces, continental deposits of ancient sand dunes cover a sedimentary surface that developed over a Precambric basement and sequence of Karroo and Post-Karroo effusives (Flores, 1973). The sand material was deposited by marine transgression and regression between the Pliocene and the Holocene, around 5.4 million to 10 thousand years ago (Salman and Abdula, 1995). The resulting soils therefore range from sandy at the coast to sandy clay loam inland (INIA, 1994; 1995).

### *3.2.2 Agroecosystem and site selection*

Six agroecosystems (AE) were studied of which AE4 is a transitional agroecosystem (Table 3.1). An AE, for practical purposes, was considered to be a region where the three environmental factors that affect yield (climate, slope, soil) are homogeneous (Du Preez and Van Zyl, 1998). In southern Mozambique these three factors, as mentioned above, show a gradient from coast to inland. Based on this and recommendations by Swift *et al*. (1979) and local researchers (Geurts and Van den Berg, 1998; Reddy, 1985; 1986), every rainfall zone displayed in Figure 3.2 was regarded as an AE.

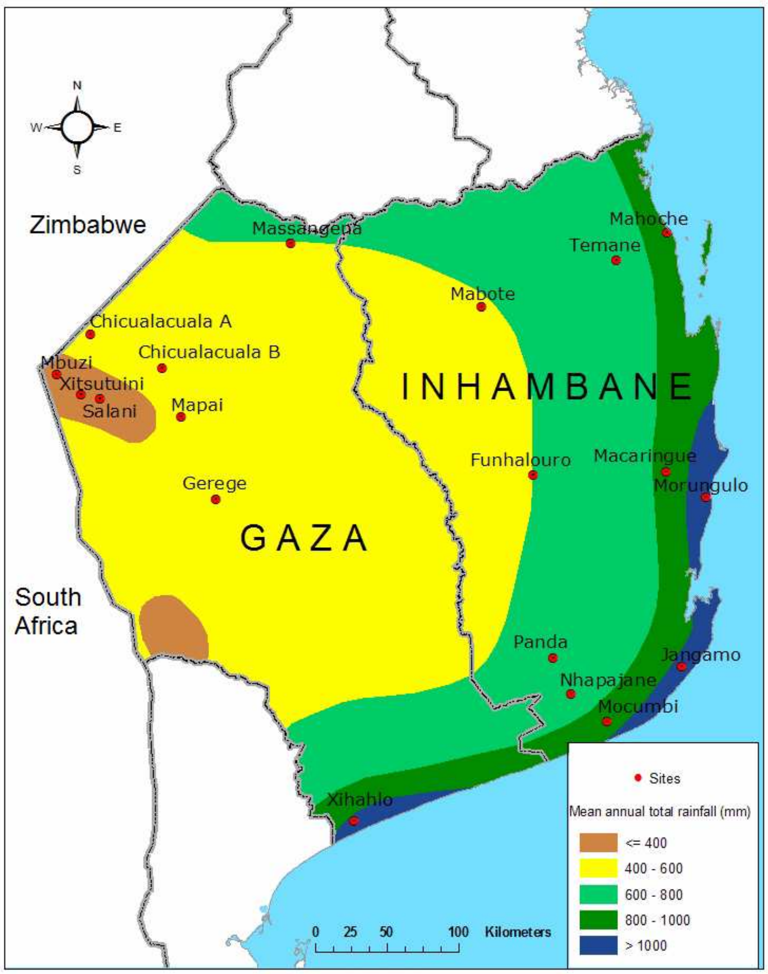


**Figure 3.2** Study area and sites with mean annual rainfall zones as adapted from FAO (2000).

**Table 3.1** Characteristics of the selected agroecosystems (adapted from Reddy, 1985; 1986 and FAO, 2000)

<table>
<tr><th colspan="2">Agroecosystem</th><th rowspan="2">Altitude (m)</th><th rowspan="2">Mean annual rainfall (mm)</th><th rowspan="2">Mean annual average temperature ($^{o}C$)</th><th rowspan="2">Mean effective rainy period (Weeks)</th><th rowspan="2">Broad climatic zone according to moisture index</th></tr>
<tr><th>Number</th><th>Code</th></tr>
<tr><td rowspan="2">1</td><td rowspan="2">AE1</td><td rowspan="6">< 200</td><td rowspan="2">> 1000</td><td>20 –23</td><td>8 – 13</td><td>Sub-humid</td></tr>
<tr><td rowspan="7">20 – 26</td><td>13 – 21</td><td rowspan="4">Wet semi-arid</td></tr>
<tr><td rowspan="2">2</td><td rowspan="2">AE2</td><td rowspan="2">800–1000</td><td>5 – 8</td></tr>
<tr><td rowspan="7"><5</td></tr>
<tr><td>3</td><td>AE3</td><td>600–800</td></tr>
<tr><td rowspan="2">4*</td><td rowspan="2">AE4</td><td rowspan="2">(400–800)</td><td rowspan="3">Dry semi-arid</td></tr>
<tr><td rowspan="4">200-500</td></tr>
<tr><td rowspan="2">5</td><td rowspan="2">AE5</td><td rowspan="2">400–600</td></tr>
<tr><td rowspan="2">23 – 26</td><td rowspan="2">Arid</td></tr>
<tr><td>6</td><td>AE6</td><td>< 400</td></tr>
</table>

* - Transitional agroecosystem since it comprises the Massangena, Mabote and Funhalouro sites (See Figure 3.2)

Three representative sites were selected in each AE except for AE5, where a fourth site was included (Table 3.2). Every site was considered a replication within an AE and comprised four fields under different land uses, namely virgin (VG), less than 5 years fallow (5F), 5-15 years fallow (10F) and more than 15 years fallow (15F). Unfortunately, at Morungulo in AE1, there was no suitable VG field available. In the selection of fields at every site, care was taken to ensure that they were located within a diameter of 1.0 km and had similar slope and soil. Another requirement was that the soils had to be apedal in nature and deeper than 1.0 m. This was verified by a profile pit in the 15F field and augering in the other fields.

### *3.2.3 Description of vegetation and calculation of biomass*

The vegetation consists of tree, shrub and herbaceous layers, so the study was performed on each layer in the relevant fields where they occurred.

For the tree layer description, quadrants of 400 $m^2$ were demarcated (Morris and Therivel, 1995). In these quadrants all plants higher than 5 m (estimated) with a stem diameter of more than 100 mm (measured with a calliper) at breast height were considered trees. These values were used to calculate tree biomass using a standard equation by assuming a mean density of 750 kg $m^{-3}$ and a correction factor of 0.6 as recommended for native species in Mozambique (Ministério de Agricultura ,1995, cited by Barbosa, 1995). The density, dominance, frequency and diversity of trees were also calculated on a relative basis using standard equations. These parameters were summed to obtain an ecological importance value (Balslev *et al.,* 1987, cited by Dallmeiner, 1992). This value shows the ecological importance of each species in the community.

The shrub layer was studied in two quadrants of 20 $m^2$ each demarcated diagonally inside the 400 $m^2$ quadrant. All individuals of ligneous species that did not fulfil the defined specifications of a tree were recorded. Those individuals with heights between 0.5 and 2 m were considered regeneration shrubs. Adult shrubs were regarded as those higher than 2 m (Halafo, 1996; Nuvunga, 1998).

**Table 3.2** Geographical location and general soil characteristics of the study sites

| Agroecosystem (MAR)* | Sites | Latitude | Longitude | Soil units** FAO 1988 | Soil units** USDA 1992 | Superficial texture** | Landform** |
|---|---|---|---|---|---|---|---|
| AE1 | Morungulo | 23° 13' | 35°28' | Haplic Arenosols | Ustic Quartzipsamments | Sandy | Coastal dunes, holocenic sand |
| (>1000) | Jangamo | 24° 16' | 35°19'' | Arenosols | Psamments | | |
| | Xihahlo | 25° 12' | 33°20'' | Ferralic Arenosols | Ustoxic Quartzipsamments | | Inner dunes |
| AE2 | Mahoche | 21° 35' | 35°13' | | | | |
| (800 – 1000) | Macaringue | 23°04' | 35°13'' | | | | Sandy plain |
| | Mocumbi | 24°37' | 34°51'' | Arenosols | Psamments | Sandy - loamy sand | |
| AE3 | Temane | 21°45' | 34°54'' | | | | |
| (600 – 800) | Panda | 24°13' | 34°31'' | | | | Inner dunes |
| | Nhapajane | 24°27' | 34°38'' | | | | |
| AE4 | Massangena | 21°39' | 32°53'' | Chromic Cambisols | Typic Ustochrepts | Sandy – sandy loam or loamy sand – sandy loam | Sandy and reddish colluvium |
| (Transitional) | Mabote | 22°03' | 34°04'' | | | | |
| | Funhalouro | 23°05'' | 34°23'' | | | | |
| AE5 | Chicualacuala A | 22°13'' | 31°38'' | | | | |
| (400 – 600) | Chicualacuala B | 22°25' | 32°05'' | Arenosols | Psamments | Sandy - loamy sand | Sandy plain |
| | Mapai | 22°44'' | 32°12'' | | | | |
| | Gerege | 23°14'' | 32°25'' | | | | |
| AE6 | Mbúzi | 22°28'' | 31°25'' | | | | |
| (<400) | Xitsutsuini | 22°35'' | 31°34'' | Calcaric Cambisols | Typic Ustochrepts | Sandy loam – sandy clay loam | Shallow soils over calcareous rocks |
| | Salani | 22°37' | 31°41'' | | | | |

* MAR – Mean annual rainfall (Reddy, 1986; FAO, 2000); **INIA (1994; 1995).

All individuals with a height of less than 0.5 m, regardless of species, were considered components of the herbaceous layer. Within the 400 $m^2$ quadrant, three quadrants of 1 $m^2$ each were demarcated randomly for determination of coverage and aerial biomass. Coverage (%) was obtained by listing the species in a quadrant and visually estimating the area covered by each species (Mueller-Dombois and Ellenberg, 1976; Kent and Coker, 1992). All individuals less than 0.5 m high were then cut about 1 cm above the soil surface with a knife. The aerial biomass was dried in an oven at 90℃ for 48 hours to obtain the dry weight (g $m^{-2}$) as described by Sutherland (1996).

### *3.2.4 Data processing*

A statistical analysis was performed at $P < 0.05$ on data of the tree and herbaceous layers with the General Linear Model Analysis of Variance (GLM ANOVA) using the statistical software package for Windows NCSS 2000 (Hintze, 1998). The variance was calculated firstly for fields with different fallow periods in each agroecosystem, and secondly for fields under the same land use in different agroecosystems. It was more appropriate to use GLM ANOVA instead of ANOVA because the number of fields per site in certain agroecosystems as well as the number of sites per agroecosystem differed. Where GLM ANOVA revealed significant differences between treatments, Fischer's least significant difference test (LSD, $P < 0.05$) was used to distinguish between treatment means.

In some instances it was impossible to calculate the frequency of trees due to absence of data and therefore no importance value for the tree species was determined. This absence of data was the result of some species not fulfilling the defined specifications for trees or certain fields having been burnt by some local community members during the period between identification and description.

Data for the shrub layer from fields under the same land use were grouped in each agroecosystem. The mean numbers of regeneration shrubs and adult plants were calculated per hectare for every land use in every agroecosystem.

## 3.3 Results

A total of 204 species were recorded, of which 10 were not identified. The identified species belonged to 141 genera and 50 families. In general, the total number of species declined from coastal and wetter to inland and drier agroecosystems, ranging from 97 in AE2 to 34 in AE6. The results are presented per vegetation layer. However, on account of their importance in restoration of soil fertility, the number of N-fixing species is reported without distinction between the layers.

### *3.3.1 Tree layer*

The total number of species in each agroecosystem and the number of species and individuals in VG and 15F fields are given in Table 3.3. Species of higher ecological importance in these fields are also listed but in some instances it was impossible to calculate importance values for the following reasons. In AE1 a suitable VG field was absent at Morungulo, whereas 15F fields at Morungulo and Jangamo did not have a tree layer as defined in this study. This was also the case in 15F fields at Massangena, Mabote and Funhalouro in AE4. No data are presented for 10F fields at any site since frequency could not be calculated as trees occurred only at Morungulo in AE1, Macaringue in AE2 and Gerege in AE5. In addition to this, community members burnt the 10F field at Funhalouro during the period between identification and description.

In terms of floristic composition, 37 species were recorded and they belonged to 27 genera and 14 families. The total number of species increased from 9 in AE1 to 12 in AE3 and then decreased to 5 in AE6 (omitting AE4, with only 2 species).

Some trends emerged when VG and 15F fields were compared: (i) number of species in VG was equal to or less than that in 15F fields in all agroecosystems, except AE3; (ii) number of tree individuals in VG was less than that in 15F fields of AE1, AE2 and AE3, whereas this trend was reversed in AE5 and AE6; (iii) size of tree (stem diameter and maximum height) in VG was larger than in 15F fields of AE1, AE2 and AE3, whereas this trend was reversed in AE5 and AE6.

**Table 3.3** Floristic composition of the most common species and species of higher ecological importance in the tree layer.

| AE | Total number of species* | Land use: Virgin – Species | Virgin – Individuals $ha^{-1}$ | Virgin – Diameter (cm) and [height (m)] | Virgin – Species of higher ecological importance** | > 15 yrs fallow – Species | > 15 yrs fallow – Individuals $ha^{-1}$ | > 15 yrs fallow – Diameter (cm) and [height (m)] | > 15 yrs fallow – Species of higher ecological importance** |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 9 | 4 | 225 | 25.67 [8.0-25.0] | - | 4 | 250 | 15.56 [11.0–14.0] | - |
| 2 | 11 | 5 | 950 | 19.36 [6.0-17.5] | *Brachystegia spiciformis* (324) *Julbernadia globiflora* (210) | 7 | 1025 | 15.90 [7.0-16.0] | *Brachystegia spiciformis* (317) |
| 3 | 12 | 8 | 775 | 17.98 [5.5-25.8] | *Brachystegia spiciformis* (244) *Brachystegia utilis* (181) | 6 | 825 | 14.88 [11.8-17.8] | *Brachystegia spiciformis* (217) *Julbernadia globiflora* (217) |
| 4 | 2 | - | 1250 | 16.23 [6.0-14.5] | *Androstachys johnsonii* (276) | - | - | - | - |
| 5 | 8 | 2 | 1625 | 13.56 [7.0-13.0] | *Androstachys johnsonii* (266) | 4 | 150 | 18.80 [4.5-15.0] | *Berchemia discolor* (181) |
| 6 | 5 | 3 | 625 | 16.30 [6.5-16.0] | *Colophospermum mopane* (246) | 3 | 175 | 18.35 [7.0-17.0] | *Colophospermum mopane* (177) |

[*] Observed in every agroecosystem; [**]Number in brackets is importance values

The species of higher ecological importance changed from the coast to inland. In VG fields *Brachystegia spiciformis* dominated in AE2 and AE3; *Androstachys johnsonii* in AE4 and AE5; and *Colophespermum mopane* in AE6. In 15F fields the dominant species were *B. spiciformis* in AE2, *B. spiciformis* and *Julbernadia globiflora* in AE3, *Berchemia discolour* in AE5 and *C. mopane* in AE6.

The mean biomass in VG and 15F fields is displayed for each agroecosystem in Figure 3.3 and across the agroecosystems in Figure 3.4. On account of high variability the GLM ANOVA test revealed significant differences in tree biomass only on three occasions. The biomass in VG of AE1 was higher than in 15F fields, whereas in AE2 and AE3 it seemed to decrease from VG to 15F fields (Figure 3.3). In contrast to this, the biomass in VG of AE5 was lower than in 15F fields and a similar trend was observed in AE6. The mean biomass of VG fields in AE1 was significantly higher than that of similar fields in the other five agroecosystems (Figure 3.4). Mean biomass of 15F fields in the different agroecosystems was almost similar when that of AE4 was not considered.

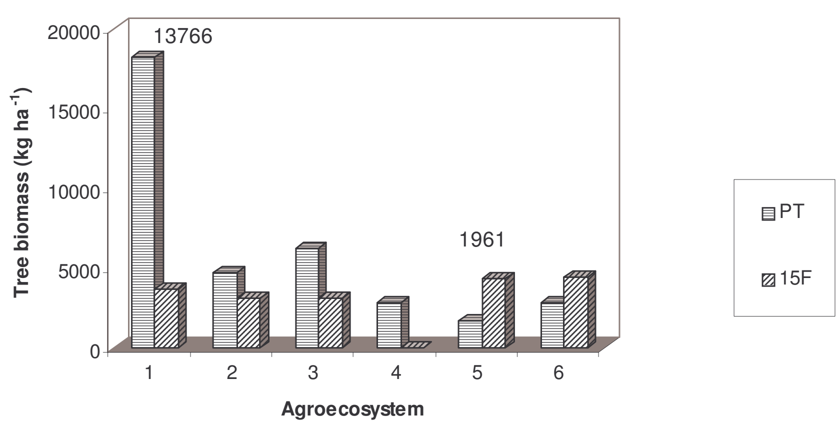


**Figure 3.3** Mean tree biomass in the VG (virgin) and 15F (> 15 years fallow) fields in the agroecosystems. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

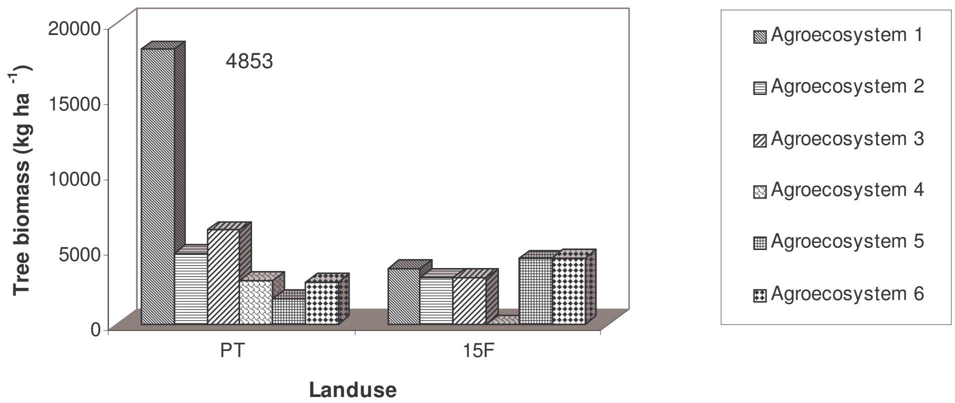


**Figure 3.4** Mean tree biomass in the VG (virgin) and 15F (> 15 years fallow) fields across agroecosystems. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ value above graphs

### *3.3.2 Shrub layer*

The number of regeneration and adult shrubs for different land uses in each agroecosystem is shown in Figure 3.5. In AE1 regeneration shrubs increased from 490 individuals ha$^{-1}$ in VG to 1750 individuals ha$^{-1}$ in 5F fields. On the other hand adult shrubs decreased from 1730 individuals ha$^{-1}$ in VG to 70 individuals ha$^{-1}$ in the 5F fields. Neither the regeneration shrubs nor the adult shrubs showed any obvious trends concerning their numbers in AE2. The numbers of both kinds of shrubs tended to decrease from VG to 5F fields in AE3, AE4, AE5 and AE6. In these four agroecosystems the adult shrubs outnumbered the regeneration shrubs slightly, except in 15F of AE3, 5F of AE4, 10F of AE5 and 10F of AE6.

The number of regeneration and adult shrubs for each land use in the different agroecosystems is illustrated in Figure 3.6. In VG fields the number of regeneration shrubs generally increased from AE1 to AE3 and then decreased to either AE5 or AE6, whereas in the 5F, 10F and 15F fields it decreased from AE1 to AE6. The number of adult shrubs showed no trend between the agroecosystems as regards 5F fields. In VG, 10F and 15F fields the number of adult shrubs tended to decline from AE1 to AE6.

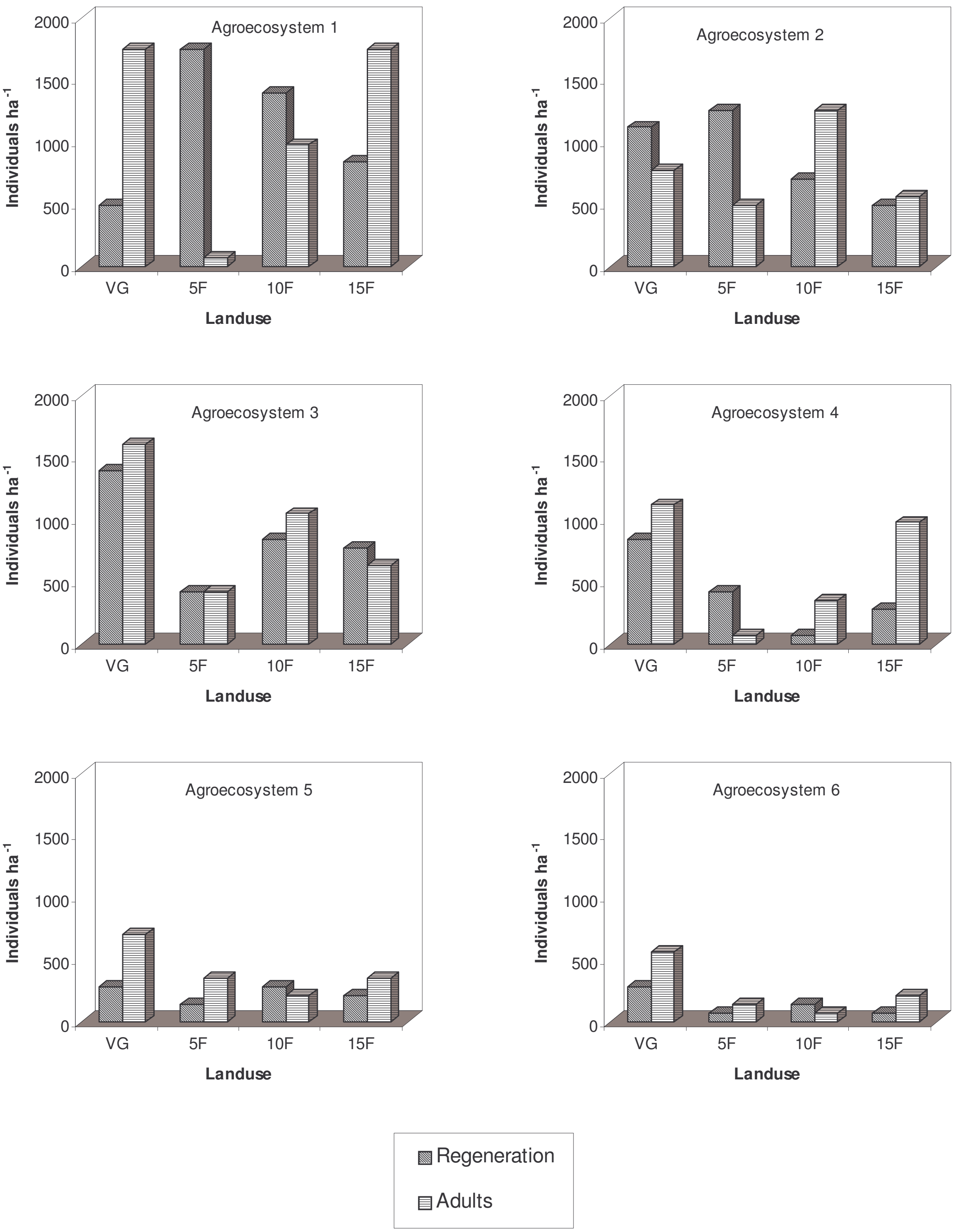


**Figure 3.5** Number of regeneration and adult shrubs for the VG (virgin), 5F (< 5 years fallow), 10F (5-15 years fallow) and 15F (> 15 years fallow) fields in the agroecosystems

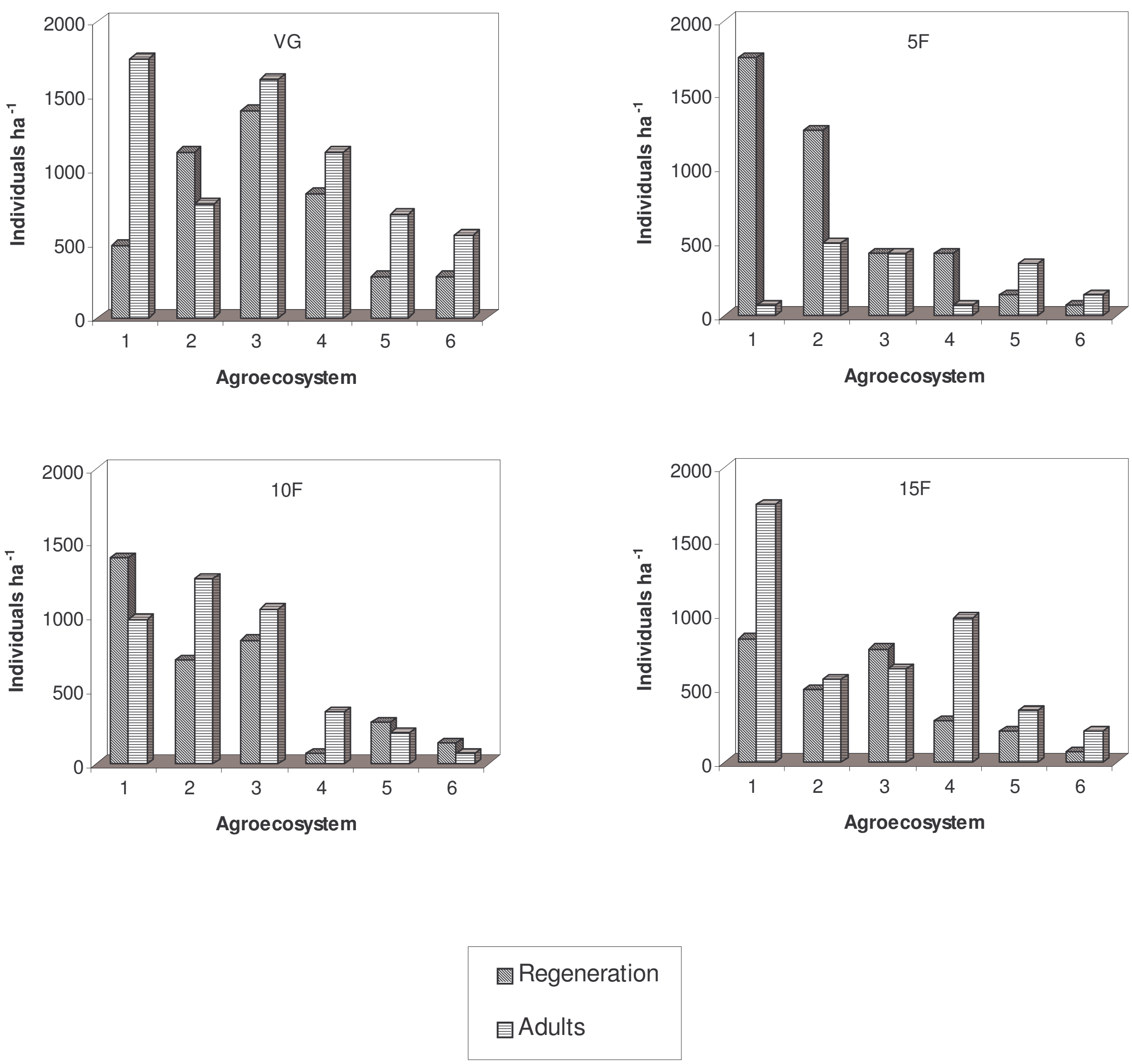


**Figure 3.6** Number of regeneration and adult shrubs for the VG (virgin), 5F (< 5 years fallow), 10F (5-15 years fallow) and 15F (> 15 years fallow) fields across agroecosystems

The most dominant shrub species also changed gradually from the coast to inland, with *J. globiflora* being dominant in AE1 and AE2, *B. spiciformis* in AE2 and AE3, *B. utilis* in AE3, and *A. johnsonii* in AE4, AE5 and AE6. In AE1, *J. globiflora* represented more individuals in regeneration than adults, while the converse was observed in AE2. In AE2 and AE3 there were more regeneration shrubs than adult shrubs of *B. spiciformis*, with a maximum of 1100 individuals ha$^{-1}$ in 5F of AE2 and 1250 individuals ha$^{-1}$ in VG fields of AE3. This trend was reversed with *B. utilis* in AE3, with a maximum of 450 individuals $ha^{-1}$ in VG fields. In AE4 and AE5, *A. johnsonii* represented more individuals in regeneration than adults, whereas in AE6 no individuals in regeneration were recorded. A maximum of 800 individuals $ha^{-1}$ in regeneration was found in VG fields of AE4.

### *3.3.3 Herbaceous layer*

The total number of species for all four land uses in each of the six agroecosystems is given in Table 3.4. These numbers did not reveal any consistent trend for either the different land uses in an agroecosystem or the same land use in the different agroecosystems. Hence, this aspect warrants no further elaboration.

The most common species generally changed among the different land uses in an agroecosystem and the same land use in the different agroecosystems. However, *Panicum maximum* (10F and 15F fields in AE2 and 5F, 15F and VG fields in AE3) and *Digitaria sp* (10F and 15F fields in AE6) dominated in more than one land use within the same agroecosystem. Four species were found in similar land uses of different agroecosystems: *Mellins repens* in 5F of AE1 and AE2; *P. maximum* in 5F of AE3 and AE5, and 15F of AE2 and AE3; *Digitaria sp* in 10F of AE5 and AE6; and *A. johnsonii* in VG of AE4 and AE5.

Percentage cover of the most common species varied from 2% (15F in AE1 and AE2) to 26% (5F in AE1). In AE1, AE2, AE3 and AE6 the highest coverage occurred in 5F fields with 26, 19, 18 and 17%, respectively. The highest coverage in the other two agroecosystems was 12% in 15F of AE4 and 14% in 10F of AE5.

Significant differences in biomass occurred between the four land uses within agroecosystems, except in AE4 (Figure 3.7). The lowest mean biomass in AE1 was in 15F, whereas in AE2, AE3, AE5 and AE6 it was in VG fields. Mean biomass decreased with longer fallow in AE2 and AE3, whereas in AE4 and AE5 the trend was opposite. Neither AE1, nor AE6 showed clear trends.

**Table 3.4** Most common species in the herbaceous layer of the different land uses

| AE* | Land use | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Virgin | | < 5 yrs fallow | | 5-15 yrs fallow | | > 15 yrs fallow | |
| | Total number of species | Most common species** | Total number of species | Most common species** | Total number of species | Most common species** | Total number of species | Most common species** |
| 1 | 11 | *Zamioculcas zamiifolia* (5) | 10 | *Mellins repens* (26) | 25 | *Oldenlandia rupicola* (13) | 10 | *Imperata cylindrica* (2) |
| 2 | 27 | *Brachystegia spiciformis* (11) | 19 | *Mellins repens* (19) | 16 | *Panicum maximum (14)* | 17 | *Panicum maximum (2)* *Dichrostachys cinerea* (2) |
| 3 | 19 | *Panicum maximum* (12) | 17 | *Panicum maximum* (18) | 17 | *Mellins repens* (11) | 18 | *Panicum maximum* (11) |
| 4 | 7 | *Androstachys johnsonii* (8) | 16 | *Tephrosia purpurea* (6) | 16 | *Setaria Pallidefusca* (8) | 17 | *Digitaria eriatha* (12) |
| 5 | 2 | *Androstachys johnsonii (3)* | 17 | *Urochloa mosambisensis* (4) *Tephrosia* sp. (4) *Panicum maximum* (4) | 9 | *Digitaria* sp. (14) | 17 | *Psydrax locuples* (9) |
| 6 | 8 | *Justicea betonica* (3) | 11 | *Tephrosia* sp. (17) | 11 | *Digitaria* sp. (14) | 11 | *Digitaria* sp. (6) |

*- Agroecosystem; ** - The number in brackets shows the percentage of coverage

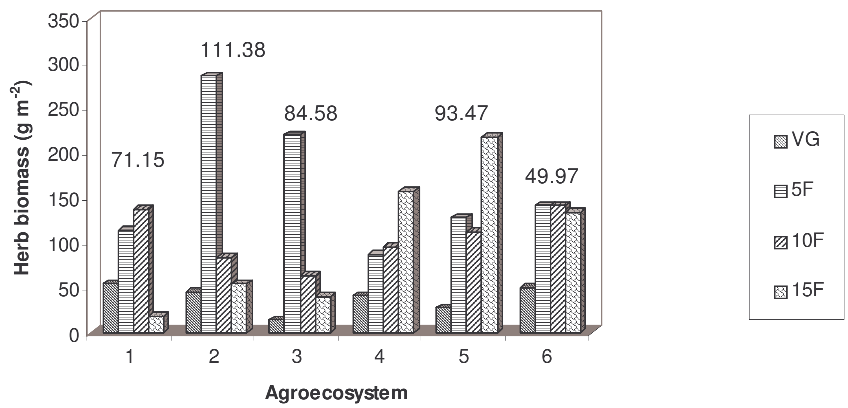


**Figure 3.7** Mean herbaceous biomass for the VG (virgin), 5F (< 5 years fallow), 10F (5-15 Years fallow) and 15F (> 15 Years fallow) fields in the agroecosystems. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

Only 15F fields showed a significant difference in biomass across agroecosystems (Figure 3.8). Generally, mean biomass of these fields increased from AE1 to AE5 and thereafter decreased to AE6. The biomass of 5F fields increased from AE1 to a maximum in AE2 and decreased thereafter

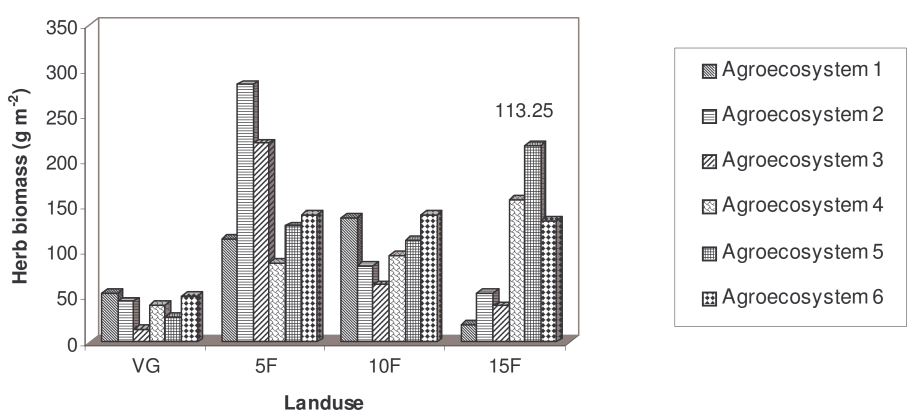


**Figure 3.8** Mean herbaceous biomass for the VG (virgin), 5F (< 5 years fallow), 10F (5-15 years fallow) and 15F (> 15 years fallow) fields across agroecosystems. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ value above graphs

**Table 3.5** Registered N-fixing species* for the different land uses in every agroecosystem

| AE** | Land use | | | |
|---|---|---|---|---|
| | Virgin | < 5 yrs fallow | 5-15 yrs fallow | >15 yrs fallow |
| 1 | *Albizia adiathifolia*<br>*Albizia versicolor*<br>*Dichrostachys cinerea* | *Albizia adianthifolia* | *Acacia* sp.<br>*Albizia adianthifolia*<br>*Albizia versicolor*<br>*Crotalaria* sp.<br>*Dichrostachys cinerea* | *Albizia adianthifolia*<br>*Dichrostachys cinerea* |
| 2 | *Albizia harveyi*<br>*Dichrostachys cinerea* | *Crotalaria monteroi*<br>*Dichrostachys cinerea*<br>*Tephrosia purpurea* | *Albizia versicolor*<br>*Tephrosia purpurea* | *Albizia adianthifolia*<br>*Albizia versicolor*<br>*Dialium* sp.<br>*Dichrostachys cinerea*<br>*Millettia stuhlmanii* |
| 3 | *Acacia xanthophloea*<br>*Albizia adianthifolia*<br>*Albizia anthelmithica*<br>*Mundulea serinea* | *Albizia versicolor*<br>*Crotalaria monteroi*<br>*Dichrostachys cinerea*<br>*Tephrosia purpurea* | *Acacia* sp.<br>*Albizia adianthifolia*<br>*Tephrosia purpurea*<br>*Tephrosia* sp. | *Acacia* sp.<br>*Acacia xanthophloea*<br>*Dichrostachys cinerea*<br>*Tephrosia* sp. |
| 4 | *Acacia Xanthophloea*<br>*Baphia massaensis* | *Dichrostachys cinerea*<br>*Tephrosia purpurea* | *Albizia anthelmithica*<br>*Tephrosia purpurea* | *Acacia* sp.<br>*Albizia adianthifolia*<br>*Crotalaria monteroi*<br>*Tephrosia purpurea*<br>*Tephrosia* sp. |
| 5 | - | *Dichrostachys cinerea*<br>*Tephrosia* sp.<br>*Tephrosia purpurea* | *Tephrosia* sp. | *Acacia grandicornuta*<br>*Baphia massaensis*<br>*Dichrostachys cinerea*<br>*Tephrosia purpurea*<br>*Tephosia* sp. |
| 6 | *Tephrosia purpurea* | *Tephrosia* sp.<br>*Tephrosia purpurea* | *Tephrosia purpurea* | *Acacia* sp.<br>*Tephrosia purpurea* |

*- (Allen and Allen, 1981); ** - Agroecosystem

to a minimum in AE4 before increasing again to AE6. Biomass in VG and 10F fields showed similar trends, decreasing from AE1 to a minimum in AE3 and it increasing again to AE6.

#### *3.3.4 Nitrogen-fixing species*

The total number of N-fixing species recorded in the study area was 16 (Table 3.5). The number varied from 0 in VG of AE5 to 5 in 10F of AE1 and 15F of AE2, AE4 and AE5. The most common species were *Acacia sp.*, *Albizia sp.*, *Crotalaria sp.*, *Dichrostachys sp.* and *Tephrosia sp*. Members of *Acacia sp*. occurred in all agroecosystems, except AE2, *Albizia sp. and Crotalaria sp*. in AE1 to AE4, *Dichrostachys sp.* in AE1 to AE5 and *Tephrosia sp* in AE2 to AE6. In AE6 only *Tephrosia sp.* was found, except in 15F fields, which also had *Acacia sp*.

### 3.4 Discussion

Some distinct trends in vegetation were observed as the habitat changed from the coastal and wetter agroecosystems (AE1, AE2, and AE3) towards the inland and drier agroecosystems (AE5 and AE6). For example, in the three vegetation layers, there was a decrease in plant diameter (14.88-25.67 to 13.56-18.80 cm), height (5.5-25.8 to 4.5-17.0 m) and total number of species (4-12 to 2-8) in both VG and 15F fields. However, the number of individuals ha$^{-1}$ increased in VG (225-950 to 625-1625) and decreased in 15F fields (250-1025 to 150-175), probably as a result of the declining trend in rainfall being associated with a change in morphology of species. This change of habitats could also explain the replacement of *Panicum maximum*/*Mellins repens* as the most common species by *Tephosia sp.*/*Digitaria sp*. in the herbaceous layer and *Albizia sp*. by *Tephrosia sp*. among the N-fixing species. These results agree with those of Styger *et al*. (2007) who reported that fallow fields tend to be characterised by species life forms, species composition, vegetation appearance or structure that are typical for each period.

Coastal and wetter agroecosystems were mainly dominated or co-dominated by *Brachystegia sp*. and *J. globiflora* as they had higher ecological importance values, which indicates species robustness for comparison between various species in both similar or different plant associations (Krebs, 1989). The decrease in tree diameter

(17.98-25.98 to 14.88-15.90 cm), height (11.5-16.5 to 11.5-14.8 m) and biomass from VG to 15F fields can be attributed to an increase in number of individuals in the latter fields, which could have led to higher competition for water and nutrients in view of the lower water-holding capacity and fertility of the sandy soils. Considering that trees were found only in 15F fields, if these fields had been split into more classes it is likely that a stronger trend could have been detected. The decrease in adult shrubs from VG to 5F and then followed by an increase in the older fallow fields, which was accompanied by a decrease in biomass in the herbaceous layer, can be explained by the fact that some of the adult shrubs later turned into trees that gradually replaced the herbaceous layer. The decline in the herbaceous layer could be related to some degree to the increase in closed woodlands, as trees have greater opportunities to intercept sunlight and use soil water for photosynthesis and other physiological activities due to higher height and deeper roots.

In inland and drier agroecosystems the observed decrease in number of trees per hectare (625-1625 to 150-175) from VG to 15F fields, which was unexpectedly accompanied by an increase in biomass, can be attributed to two causes depending on the agroecosystem. In AE5, VG fields were dominated by *A. johnsonii*, which had a smaller diameter (13.56 cm, with height of 7.0-13.0 m) than the succession species *B. discolour* (18.80 cm, with height of 4.5-15.0 m) that dominated in 15F fields. This also confirms the observations by local agricultural technicians and community members that once slashed and burnt, *A. johnsonii* never regenerates. In contrast, AE6 was dominated by *C. mopane* in both fields and similarly to AE5, the diameter increased from 16.30 cm (with height 6.5-16.0 m) in VG to 18.35 cm (with height 7.0-17.0 m) in 15F fields. The fact that most of the adult shrubs turned into trees but that the number of adult shrubs increased from 5F to 15F but did not reach the initial values found in the VG fields has two possible explanations. Firstly, the increase in number of species from VG to 15F fields could have triggered some incompatibility that affected their reproduction. Secondly, if 15F fields had been subdivided into more classes, a certain threshold would possibly have been found at which the number of shrubs was equal to that in VG fields. A different explanation can be given for the decrease in herbaceous biomass from VG to 5F, followed by an increase in 15F fields. This increase in the long-term fallow fields can be ascribed to lower competition for nutrients and water between herbaceous and shrub or tree layers, which may result from the fact that either shrubs or trees were sparsely distributed.

On account of high rainfall in AE1, the N-fixing species had a rapid establishment compared with other agroecosystems, as rainwater can increase nutrient availability throughout the year for some pulses (Menaut *et al.*, 1985, cited by House and Hall, 2008), but were soon not able to compete with other climax species. This may explain why within that agroecosystem the number of N-fixing species was highest in the 10F fields. Apparently, this limitation did not take place earlier in the other agroecosystems allowing an equilibrated diversity of N-fixing species in 15F fields, except in AE6 where only two species occurred in 5F and 15F (*Acacia sp.* and *Tephrosia purpurea*). *Acacia sp*. and *Albizia sp*. were most common in fallow fields but could become dominant as observed elsewhere in miombo woodland (White, 1986; Campbell et al., 1996). Similarly, *Dichrostachys cinerea* was also common but not dominant. On the other hand, no N-fixing species occurred in VG fields of AE5, which was dominated by *A. johnsonii*. However, in VG fields of the transitional AE4 also dominated by *A. johnsonii* N-fixing species were recorded. This suggests that in drier conditions, N-fixing species are probably suppressed by *A. johnsonii*, which tends to be the sole dominant species where it occurs (Bandeira *et al.*, 1999). In all six agroecosystems N-fixing species were not dominant. Although the objective of this study was to characterise vegetation and compare biomass production, in order to determine whether bush fallow achieves its ultimate aim of increasing soil fertility, we also documented an increase in diversity of N-fixing plants after 15 years, which boosted the agricultural potential. Fallow field management is an important tool for enhancing concentrations of nutrients (Brand and Pfund, 1998). A long fallow period combined with additional strategies such as increased economic productivity and reduced slash and burn (Burgers *et al.*, 2005) would help restore soil fertility and increase the sustainability of land use.

### 3.5 Conclusions

The number of species and composition and characteristics of vegetation in virgin and bush fallow fields of coastal and wetter agroecosystems (rainfall >600 mm) are distinct from those of inland and drier agroecosystems (rainfall <600 mm). Three layers of vegetation (trees, shrubs, herbaceous plants) occur in all virgin fields and fields abandoned to bush fallow for more than 15 years. The tree species in bush fallow fields of coastal and wetter agroecosystems not only outnumber those in inland and drier agroecosystems, but also have a larger diameter, resulting in greater biomass. In inland and drier agroecosystems the tree biomass in bush fallow fields older than 15 years tends to be higher than in virgin fields due to presence of succession species that differ from the original species. In virgin fields where *A. johnsonii* is the most dominant species, once the land is converted to cultivation through slash and burn, this species never regenerates and therefore the composition and biomass of trees in bush fallow are totally determined by succession species. The number of shrubs decreases from coastal and wetter to inland and drier agroecosystems, with the number of adult and regeneration shrubs increasing with age of fallow, except in former agroecosystems where regeneration shrubs are gradually replaced by adults. The herbaceous biomass declines from young to old fallow fields in coastal and wetter agroecosystems, whereas in inland and drier agroecosystems it increases. Bush fallow fields older than 15 years tend to have more N-fixing species than younger fallow and virgin fields. In future studies of vegetation composition it is recommended that bush fallow fields older than 15 years be split into different classes to provide a better picture of biomass changes in the tree and herbaceous layers, especially in inland and drier agroecosystems.

# CHAPTER 4

## Carbon loss from *Brachystegia spiciformis* leaf litter in the sandy soils of southern Mozambique

### 4.1 Introduction

Bush fallow still remains the main cropping practice in the less densely populated areas of sub-Sahara Africa that are dominated by small-scale farmers who cannot afford to buy fertilisers (Juo and Manu, 1996). During the bush fallow period processes occur in the soil leading to recovery of its productive attributes. These processes include the transfer of nutrients that are captured from the soil by growing vegetation through its dense root network (Snapp *et al*., 1998) and their return to soil by decay of vegetation residues (Brady and Weil, 1996). The nutrients are then retained as integral components of organic matter or adsorption on the clay-humus complex in soil (Palm *et al*., 1996).

Vegetation residues constitute a source of energy and nutrients for soil organisms during decomposition. The resulting organic matter interacts with mineral particles, forming a mineral-organic mix that determines the dynamic chemical and physical characteristics of the soil (Swift *et al*., 1979). This role of organic matter is important for the soil as a plant growth medium, especially in sandy soils, which are generally poor in plant nutrients and in ability to store water.

Decomposition of vegetation residues occurs both above- and belowground. However, the aboveground process of leaf litter decomposition dominates in most forest and some savanna systems (Frost, 1996; Herbohn and Congdon, 1998; Smith *et al*., 1998; Ostonen *et al*., 2005). In these systems decomposition and mineralisation rates of leaf litter are therefore important components of organic matter and nutrient flux (Cuevas and Lugo, 1998; Hamadi *et al*., 2000). The turnover rate depends on internal and external factors. Internal factors are determined by the plant species because this determines the morphological characteristics (Sundarapandian and Swamy, 1999; Magill and Aber, 2000) and the intrinsic genetic characteristics of plant leaves (Harmand *et al*., 2004). Hence, species influences the relationship between labile and resistant material (Andrén and Paustian, 1987; Snapp *et al*., 1998). This relationship in turn determines the activity of decomposer microorganisms. The external factors are environmental, with temperature and water content as the most influential components (King and Campbell, 1994; Smith *et al*., 1998) since they determine the presence or absence of species composition and regulate the density and activity of decomposer microorganisms (Gijsman *et al*., 1997).

A high resistance of litter to decay can lead to its accumulation on the surface and reduced incorporation into the soil, resulting in a low soil organic matter content even if the water content and temperature are optimal (Swift *et al*., 1979). Therefore, understanding the decomposition dynamics of the litter from the most dominant species could help in interpretation of the dynamics of nutrients and organic matter in soil abandoned to bush fallow (Smith *et al*., 1998). These dynamics are important for soil fertility and productivity.

Studies of decomposition and mineralisation rates of dead vegetation material have mainly been performed in humid ecosystems, with few having been carried out in tropical arid and semi-arid ecosystems (Hamadi *et al*., 2000). Although some studies have been reported in tropical ecosystems (Nyathi and Campbell, 1994; Mtambanengwe and Kirchman, 1995; Cuevas and Lugo, 1998; Hamadi *et al*., 2000; Xuluc-Tolosa *et al*., 2003), few have investigated leaf litter decomposition in agricultural fields converted to bush fallow under shifting cultivation. In these fields leaf litter decomposition is of great importance for nutrient recycling to restore soil fertility (Nyathi and Campbell, 1994).

Bush fallow under shifting cultivation is the most common practised farming system in southern Mozambique. In this area, the natural vegetation of miombo woodland near the coast is gradually replaced by arid-savanna inland (Campbell *et al*., 1996). The farmers slash and burn this vegetation for intercropping of maize, cassava, cowpea and groundnut. In addition to these crops sorghum and millet are sometimes also planted in the more arid regions. After a 3-5 years of intercropping that coincide with hand-hoe weeding once or twice a season the agricultural land is abandoned to bush fallow due to decline in soil fertility that result in low yields (MA/FAO, 1983; MAP, 1996; Geurts, 1997; DPADRI, 2002). During bush fallow *Brachystegia spiciformis* that has high ecological importance establishes as one of the succession species in the miombo woodland and to a lesser extent in the arid savanna (Chapter 3). A better understanding of the decomposition of this species' leaf litter is therefore essential to improve the farming system of bush fallow under shifting cultivation ultimately with a view of soil fertility restoration.

The objectives of the study were to evaluate the effects of soil temperature and soil water on loss of organic C from leaf litter of *Brachystegia spiciformis* in sandy soils of southern Mozambique and to fit a corresponding decomposition model.

## 4.2 Materials and methods

### *4.2.1 Study area and experimental sites*

The study was performed along a 400 km long almost east-west transect in the northern parts of Inhambane and Gaza provinces (Figure 4.1) in southern Mozambique, between 21°35'S - 35°13'E and 22°28'S - 31°25'E.

In general, there is a gradient in climate, natural vegetation and geological characteristics from east to west. Mean annual temperature is between 20 and 26°C while mean annual rainfall ranges from < 400 mm to > 1000 mm (Reddy, 1986). As pointed out earlier natural vegetation near the coast is within the miombo woodland and is gradually replaced by arid savanna inland (Campbell *et al*., 1996) with a mean annual rainfall of 700 mm as a rough threshold (Frost, 1996). From a geological standpoint, both provinces are part of the Mozambican sedimentary basin (Flores, 1973), which appeared during the opening of the Indian Ocean and eastern Africa continent margin (Salman and Adbula, 1995). The area is covered by continental deposits of ancient sand dunes that developed over a Precambric basement and sequence of Karroo and Post-Karroo effusives (Flores, 1973). Sandy material was deposited by marine transgression and regression during the Pliocene and the Holocene periods (Salman and Abdula, 1995).

Five sites, namely Mahoche, Temane, Mabote, Chicualacuala B and Mbuzi each representing an agroecosystem were selected along the east-west transect (Figure 4.1). Based on Swift *et al.* (1979) and Geurts and Van den Berg (1998), the mean annual rainfall was used to define the five agroecosystems studied. The general characteristics of the five sites are summarised in Table 4.1.

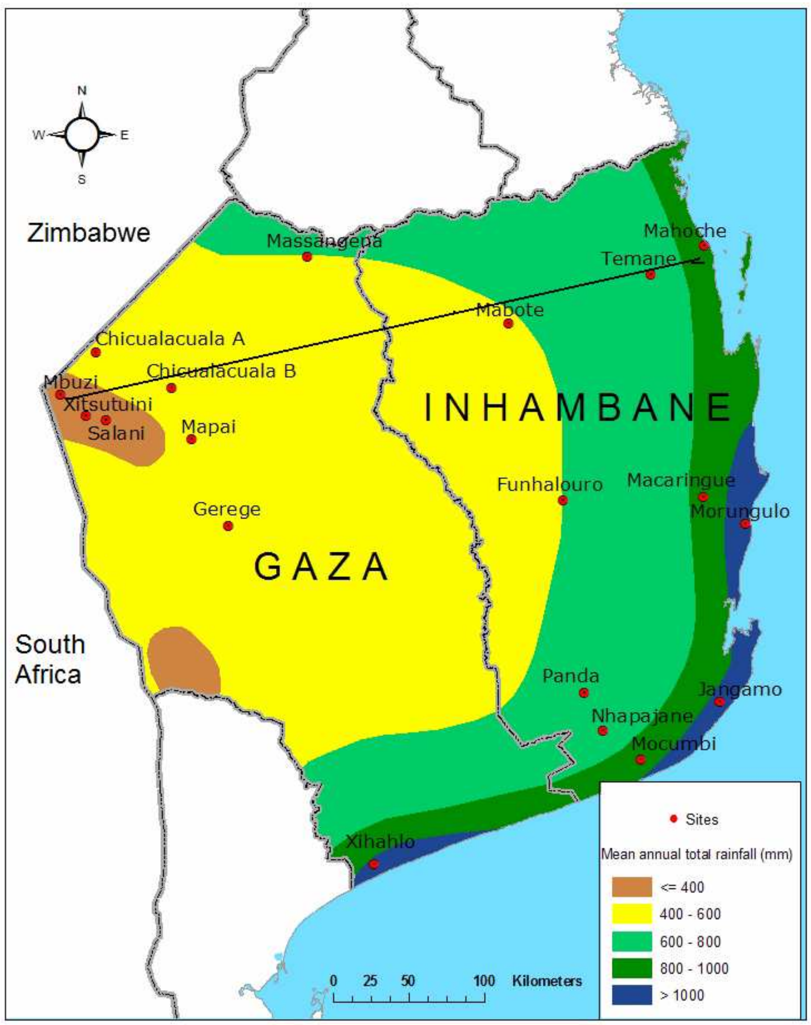


**Figure 4.1** Transect used for the study

**Table 4.1** Geographical position and climate of the five experimental sites

| Sites | Latitude | Longitude | MAT (°C) [a] | MAR (mm) [b] | Climate [c] | Soil unit[d] | Land form[d] | Vegetation[e] |
|---|---|---|---|---|---|---|---|---|
| Mahoche | 21°35’ | 35°13’ | 20-26 | 800-1000 | Wet semi-arid | Arenosols[1] Psamments[2] | Inner dunes | *Brachystegia spiciformis* |
| Temane | 21°45’ | 34°54’ | | 600-800 | | | | *Brachystegia spiciformis* *Julbernadia globiflora* |
| Mabote | 22°03’ | 34°04’ | | (400-800) | Dry semi-arid | | Sandy plain | *Androstachys johnsonii* |
| Chicualacuala B | 22°25’ | 32°05’ | 23-26 | 400-600 | | | | *Androstachys johnsonii* *Berchemia discolour* |
| Mbuzi | 22°28’ | 31°25’ | | <400 | Arid | Calcaric Cambisols[1] Typic Ustochrepts[2] | Shallow soils over calcareous rocks | *Colophospermum mopane* |

[a] MAT: mean annual temperature (Reddy, 1986).
[b] MAR: mean annual rainfall (FAO, 2000).
[c] Broad climatic zones according to moisture index (Reddy, 1986).
[d] According to INIA (1994; 1995). [1]FAO 1988, [2] USDA 1992.
[e] Most common tree species in bush fallow and virgin fields (Chapter 3).

At each site, two fields serving as treatments were selected: one recently abandoned (< 5 year) agricultural field (Bare), and one long established (>15 year) fallow (Fallow). The treatments at each site were located less than one km from each other in a flat landscape characterised by apedal sand to loamy sand soils deeper than one metre. Vegetation comprises of three layers (tree, shrub and herbaceous) in the >15 year fallow fields and two layers (shrub and herbaceous) in the <5 year fallow fields. The herbaceous biomass declines from 2750 kg $ha^{-1}$ in the <5 year fallow fields to 200 kg $ha^{-1}$ in the >15 year fallow fields. In the latter fields tree biomass is about the same throughout the study area, namely ≈ 5000 kg $ha^{-1}$. *Brachystegia spiciformis* is the common tree species though its ecological importance reducing from the miombo woodland near the coast towards the arid savanna inland where *Birchemia discolour* and *Colophospermum mopane* dominate (Chapter 3). However, where annual rainfall exceeds 600 mm a substantial fraction of the total aboveground litter in the long established bush fallow fields originates from *Brachystegia spiciformis* and that motivates the use of its leaf litter in the study.

#### *4.2.2 Soil sampling and analysis*

Three soil samples (0-50 mm layer) were randomly collected from every selected field. Each of these samples was a composite of four sub-samples collected within a diameter of 10 m. The samples were air-dried before analysed for particle size distribution (pipette method) and organic carbon (Walkley-Black). Analyses were performed by Central Agricultural Laboratories (CAL) in South Africa following standard methods (The Non-Affiliated Soil Analysis Work Committee, 1990).

#### *4.2.3 Litter sampling and decomposition experiment*

Entire mature leaves (green-yellow in colour with brown spots) from adult trees of *Brachystegia spiciformis* growing at one site were collected before abscission in the middle of winter (cold and dry season), namely June 2004. This criterion was to ensure that samples were relatively homogeneous. The quality of *Brachystegia spiciformis* leaves to decomposers varies on account of retranslocation of nutrients and synthesis of secondary plant compounds during senescence (Mtambanengwe and Kirchman, 1995; Frost, 1996). Soil fertility also influences their nutrient concentrations (Baloyi *et al.*, 1997). The chemical composition of the mature leaves used in the study was not determined. However, Baloyi *et al.* (1997) reported 12.9% crude protein, 42.7% acid detergent fibre, 24.9% lignin, 2.06% N, 0.16% P, 1.47% K, 1.20% Ca and 0.48% Mg in similar leaves. Compounds like lignin (8.2%), cellulose (21.9%) and polyphenol (4%) influence C loss from *Brachystegia spiciformis* leaflets negatively (Mtambanengwe and Kirchman, 1995).

The leaves were dried in an oven at 65°C for 72 hours and kept in a freezer. In mid-October 2004, samples of 1.0 g each were used to fill 1260 terylene litterbags (5 cm x 5 cm) of 1 mm mesh (McTiernan *et al.*, 2003; Pandey *et al.*, 2007) and subjected to decomposition under confined conditions (Swift *et al.*, 1979). It was assumed that the mesh size was sufficient to prevent losses of fragmented leaves, allow entry of important microbial decomposers (Sundarapandian and Swamy, 1999) and that overestimation of decomposition could be avoided (Palosuo *et al.*, 2007). Three lines of 42 bags each, placed at 5 m intervals, were buried at a depth of 5 cm in each of the two treatments. The bags were held by a nylon thread attached to two poles. Their contents were allowed to decompose naturally. In the Bare treatment, the soil of recently abandoned fields was hoed superficially before the bags were buried. In this treatment, the locations were hand-weeded up to a bordering distance of 5 m during the experimental period.

Rainfall was measured using a rain gauge installed 1 m above the soil surface. Soil temperature was measured at 5 cm depth every 20 min using sensors connected to Tinytag Plus – IP68 dataloggers. Volumetric soil water content was measured manually three times a day (6 a.m, 9 a.m. and 3 p.m.) with tethaprobes previously calibrated for each site according to the manufacturer's instructions (Delta-T Devices Ltd., Cambridge, England). Daily arithmetic means of soil temperature and volumetric soil water content were calculated.

The experimental setup was to retrieve three litterbags from each line in 15 days intervals, namely 9 bags per treatment. In January 2005 it was realised that the decomposition was slower than expected. As a result of this it was decided to extend the experimental period allowing for a more complete decomposition. On account of this decision and that some bags were discarded (about 10%) because they were damaged (probably by soil fauna), the number of litterbags retrieved per line was reduced to two or one and were collected in 30 days intervals. This resulting in at least 4 bags per treatment. The ultimate experimental period varied between 501 days (February 2006) at Mahoche and 514 days (March 2006) at Mbuzi.

The remaining litter in the retrieved bags was cleared of ingrowing roots where necessary, dried in an oven for 24 h at 65°C and weighed. A composite sample for each line was then prepared by mixing and milling equal amounts of litter from the relevant bags. A LECO Truspec Determinator Analyser (Leco Corporation, St Joseph, MI, USA) was used to measure the C concentrations of the composite samples. The composite samples collected on the same day were regarded as replicates for each treatment.

#### *4.2.4 Data analysis*

The single negative exponential model commonly used in decay studies of this nature was fitted to the remaining C (%) in the litterbags using soil temperature and soil water content response functions (eqs. 4.2, 4.3 and 4.4) as determinants. This model explained 81% of the variation in the total data set but the residuals, which are the differences between estimated and measured values, resulted in a negative trend. Therefore, it was decided to adopt a modified version, which assumes that a fraction, 1-$h$, of the litter decomposes according to first order kinetics and the remaining fraction, $h$, is inert within the time perspective of this experiment:

$$C_{sim} = C_0\left[h + (1-h)\exp(-k_l\tau)\right] \tag{4.1}$$

where $C_0$ is the percentage of C in the litterbags measured on the first sampling occasion, $C_{sim}$ is the remaining C content expressed as percentage of $C_0$ and $k_l$ (day$^{-1}$) is the decomposition rate constant. The variable $\tau$ is the cumulative product of daily soil water and temperature response values over time ($t$, *days)* from the first sampling to day $n$ of the experiment:

$$\tau = \sum_{t=1}^{t=n} r_\theta r_T \tag{4.2}$$

Thus, $\tau$ rescales the time axis with respect to soil water and temperature conditions (cf. Andrén and Paustian, 1987; Kätterer and Andrén, 2001).

The water content response function ($r_\theta$) was assumed to depend on the volumetric water content ($\theta$, $m^3\ m^{-3}$) of a soil according to equation 4.3:

$$r_\theta = \begin{Bmatrix} 1 & ; & \theta > \theta_{opt} \\ \dfrac{\theta - \theta_{wp}}{\theta_{opt} - \theta_{wp}} & ; & \theta_{wp} \le \theta \le \theta_{opt} \\ 0 & ; & \theta < \theta_{wp} \end{Bmatrix} \tag{4.3}$$

where $\theta_{opt}$ is the water content at which $r_\theta$ =1 and $\theta_{wp}$ is the water content at wilting point at which $r_\theta$ = 0 (Figure 4.2).

The water response function (eq. 4.3) is similar to those implemented in many C and N models (Arah, 1996; Franko, 1996; Wu and McGechan, 1998; Ma and Shaffer 2001; Shaffer *et al*., 2001). Nevertheless, non-linear response functions were tested but the $R^2$-value of the regression model did not increase. This means that a linear approximation of the water response function is a reasonable assumption. The $\theta_{wp}$ value for each site and treatment was estimated using the pedotransfer functions proposed by Rawls *et al*. (2003), which was developed from the U.S. National Soil Characterisation database using a regression tree modelling approach. The

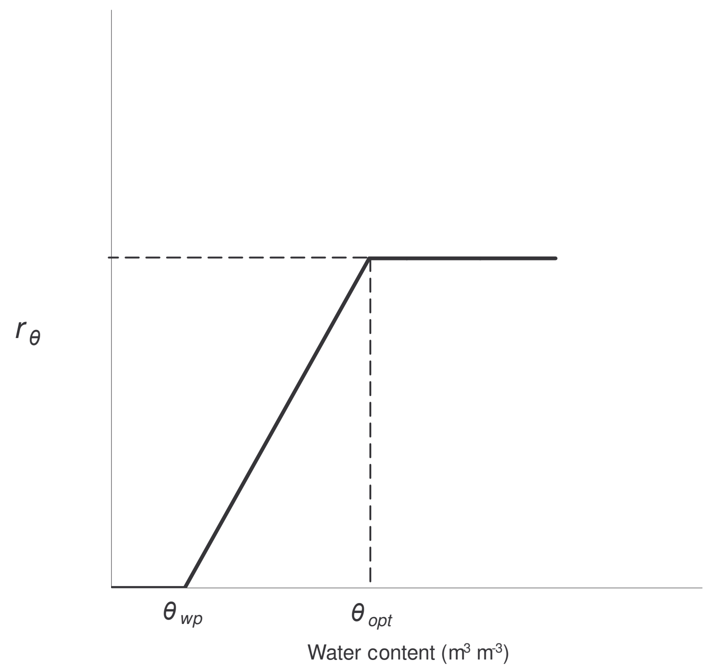


**Figure 4.2** Water response function ($r_\theta$). $\theta_{wp}$: wilting point at which $r_\theta = 0$; $\theta_{opt}$: optimal water content at which $r_\theta = 1$

input variables used were clay, sand and soil organic C measured in the treatments at all sites (Table 4.2).

**Table 4.2** Particle size distribution and organic carbon (0-50mm layer) in the five experimental sites

| Sites | Particle size distribution (%) | | | | | | Organic C (%) | |
|---|---|---|---|---|---|---|---|---|
| | Treatment | | | | | | Treatment | |
| | Bare | | | Fallow | | | Bare | Fallow |
| | Sand | Silt | Clay | Sand | Silt | Clay | | |
| Mahoche | 94 | 2 | 4 | 94 | 2 | 4 | 0.25 | 0.32 |
| Temane | 88 | 6 | 6 | 89 | 5 | 6 | 0.48 | 0.57 |
| Mabote | 92 | 4 | 4 | 92 | 3 | 5 | 0.32 | 0.19 |
| Chicualacuala B | 90 | 7 | 3 | 88 | 9 | 3 | 0.52 | 0.46 |
| Mbuzi | 86 | 9 | 5 | 87 | 7 | 6 | 0.79 | 0.65 |

Different temperature response functions ($r_T$) proposed in the literature (Kätterer *et al.*, 1998) were tested. However, all these monotonously increasing functions with temperature resulted in parameter value estimates close to zero or even negative slopes of the functions, which means that the fitted model was not sensitive to soil temperature. In this study, mean daily soil temperatures varied between 18 and 45 °C. Based on the fact that decomposition took place with some stagnant periods, it was assumed that the optimal soil temperature is between these two values. Therefore, the following simplified version of a function proposed by Parton *et al.* (1987) was tested:

$$r_T = \max\left\{0; \frac{T_{\max} - T}{T_{\max} - T_{opt}} \exp\left(1 - \frac{T_{\max} - T}{T_{\max} - T_{opt}}\right)\right\} \tag{4.4}$$

where $T$ is soil temperature, $T_{max}$ is the upper boundary above which $r_T$ =0 and $T_{opt}$ is the temperature where the response function has its maximum value, $r_T$ =1 (Figure 4.3).

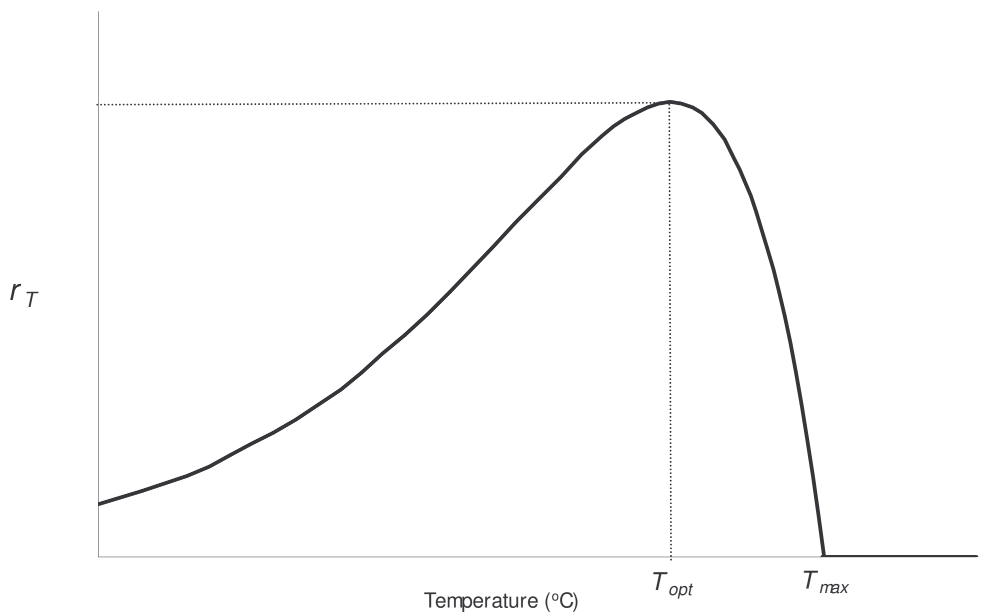


**Figure 4.3** Temperature response function ($r_T$). Tmax: upper boundary temperature at which $r_T$ = 0; Topt: optimum temperature at which $r_T$ = 1

Values of the five free parameters ($k_l$, $h$, $T_{opt}$, $T_{max}$ and $\theta_{opt}$) of the decomposition model (eqs. 4.1, 4.2, 4.3 and 4.4) were estimated for all sites and treatments simultaneously using a least squares fitting procedure. The mean values of the replicates were used. Thus, the total number of observations in the dataset was 190 (5 sites x 2 treatments x 19 measurements over time).

Residuals were analysed graphically and tested for normality using the Shapiro-Wilk and Kolmogorov-Smirnov tests (SAS software, 1999-2000, SAS Institute Inc., Cary, NC, USA). Root mean square error (RMSE) was used as a measure of model fit:

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(P_i - O_i)^2}{n}} \tag{4.5}$$

where $P_i$ and $O_i$ are predicted and observed values, respectively, and $n$ is the number of observations. The $RMSE$ reflects a magnitude of the mean difference between observations and predictions but does not indicate whether the estimate is biased.

To quantify a systematic bias, the mean bias error ($MBE$) was also calculated:

$$MBE = \frac{\sum_{i=1}^{n}(P_i - O_i)}{n} \tag{4.6}$$

A positive value of $MBE$ indicates over-prediction and a negative value indicates under-prediction.

## 4.3 Results

### 4.3.1 Soil texture and soil organic carbon

The soil texture of the selected fields varied between sand and loamy sand with ranges of 86 - 94% sand, 2-9% silt and 3-6% clay (Table 4.2). Soil organic C varied between 0.25 and 0.79% and tended to be higher in Fallow than Bare treatments in the coastal sites (Mahoche and Temane), while the opposite was observed in the transitional (Mabote) and inland sites (Chicualacuala B and Mbuzi).

### 4.3.2 Rainfall and microclimatic soil conditions

Total rainfall recorded in the first 365 days and during the whole period of the experiment, was respectively: 407 and 679 mm at Mahoche; 702 and 1363 mm at Temane; 360 and 1090 mm at Mabote; 226 and 635 mm at Chicualacuala B; and 168 and 505 mm at Mbuzi (Figure 4.4). The rainfall

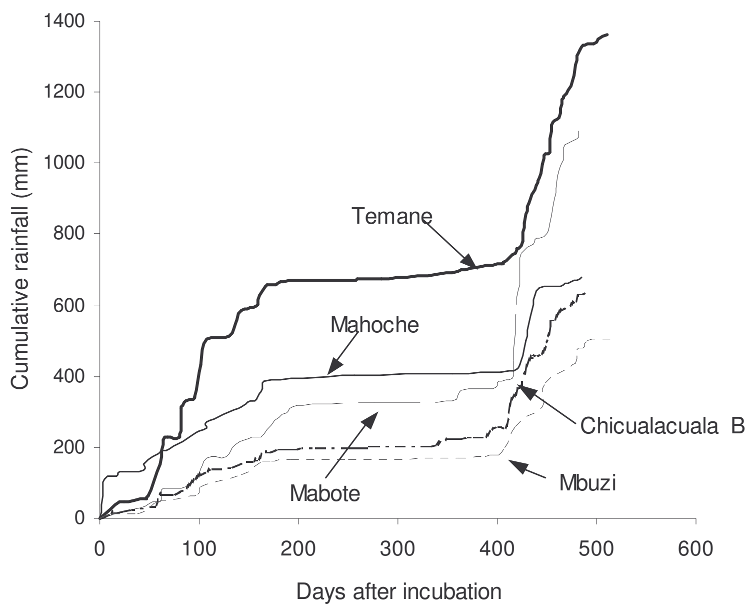


**Figure 4.4**. Cumulative rainfall (mm) during the experiment at the five sites

recorded during the first 365 days at Mahoche, Mabote and Chicualacuala B was lower than the corresponding long-term averages, i.e., 493, 240 and 274 mm, respectively (Table 4.1).

Soil water content and soil temperature followed the dynamics of rainfall, with high values recorded at the start and end of the experiment (Figures 4.5a and 4.5b). Irrespective of treatment, the mean soil water content varied between 0.039 and 0.116 $m^3$ $m^{-3}$ and mean soil temperature ranged between 27.1 and 32.6 $^oC$ (Table 4.3). Mean and maximum daily soil temperatures and also the amplitude of daily soil temperature oscillation were considerably higher in the Bare treatment than in Fallow at all sites. Short-duration peaks of high soil water content were observed at all sites. The amplitude of soil water content decreased from coastal to inland sites and ranged from 0.741 to 0.232 $m^3$ $m^{-3}$ in the Bare treatment and from 0.639 to 0.211 $m^3$ $m^{-3}$ in the Fallow treatment. On the other hand, the amplitude of the soil temperature increased from the coastal to inland sites and ranged from 39.6 to 57.6ºC in the Bare treatment and from 32.7 to 40.2°C in the Fallow treatment.

During the study, a litter layer was observed under the canopy of trees in Fallow treatment of all sites. Its thickness decreased progressively from the coast to inland.

#### *4.3.3 Carbon loss in the litter bags*

The initial C content and ash were 48.3 and 5.4% of the initial leaf litter weight, respectively. Biomass loss showed a similar pattern as C loss and therefore only data on the latter are presented. Irrespective of treatment, an increase between 0.8 and 5.8% in C content was observed at four sites between the start of the experiment and the first sampling (Figures 4.5a and 4.5b). Thereafter, a rapid decrease occurred at all five sites up to about 200 days after incubation. In the coastal and wetter sites of Mahoche and Temane, the loss continued at lower rates up to the end of the experiment. A different pattern was observed at the transitional site of Mabote and the inland and drier sites of Chicualacuala B and Mbuzi, where a period of negligible

**Table 4.3** Soil temperature and soil water content at the five experimental sites

| Site | Treat-ment | Soil temperature (°C) | | | | Soil water content ($m^3\ m^{-3}$) | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Mean | Maximum | Minimum | Amplitude [a] | Mean | Maximum | Minimum | Amplitude [a] |
| Mahoche | Bare | 32.3 | 54.7 | 15.1 | 39.6 | 0.058 | 0.759 | 0.018 | 0.741 |
| | Fallow | 28.5 | 48.2 | 15.5 | 32.7 | 0.089 | 0.641 | 0.002 | 0.639 |
| Temane | Bare | 29.7 | 47.3 | 12.5 | 34.8 | 0.110 | 0.514 | 0.007 | 0.507 |
| | Fallow | 26.2 | 38.4 | 13.3 | 25.1 | 0.116 | 0.427 | 0.024 | 0.403 |
| Mabote | Bare | 31.7 | 58.8 | 10.5 | 48.3 | 0.081 | 0.455 | 0.002 | 0.453 |
| | Fallow | 27.1 | 46.4 | 13.0 | 33.4 | 0.078 | 0.514 | 0.003 | 0.511 |
| Chicualacuala B | Bare | 30.9 | 59.5 | 11.1 | 48.4 | 0.072 | 0.314 | 0.004 | 0.310 |
| | Fallow | 27.9 | 49.2 | 14.1 | 35.1 | 0.058 | 0.268 | 0.003 | 0.265 |
| Mbuzi | Bare | 32.6 | 65.9 | 8.3 | 57.6 | 0.050 | 0.236 | 0.006 | 0.230 |
| | Fallow | 29.7 | 51.5 | 11.3 | 40.2 | 0.039 | 0.212 | 0.001 | 0.211 |

[a] differences between the observed maximum and minimum values of daily measurements.

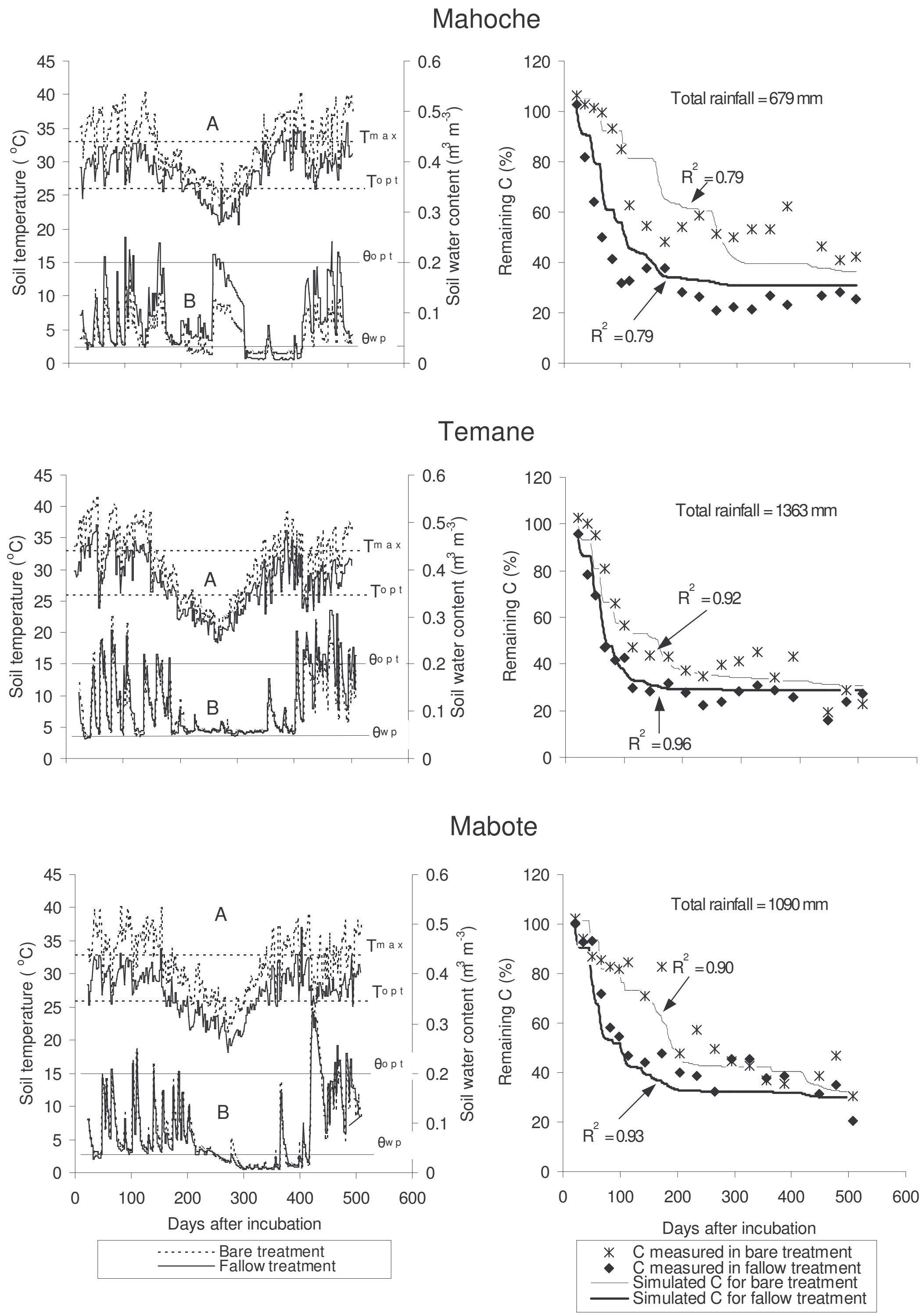


**Figure 4.5a** Soil microclimatic conditions at 5 cm depth and C change during the experimental period at the coastal and wetter/transitional sites. A: soil temperature (℃), $T_{max}$: upper boundary temperature at which the temperature response function = 0 and $T_{opt}$: optimum temperature at which temperature response function = 1. B: soil water content ($m^3$ $m^{-3}$), $\theta_{wp}$: wilting point at which the water response function = 0 and $\theta_{opt}$: optimal water content at which the water response function = 1.

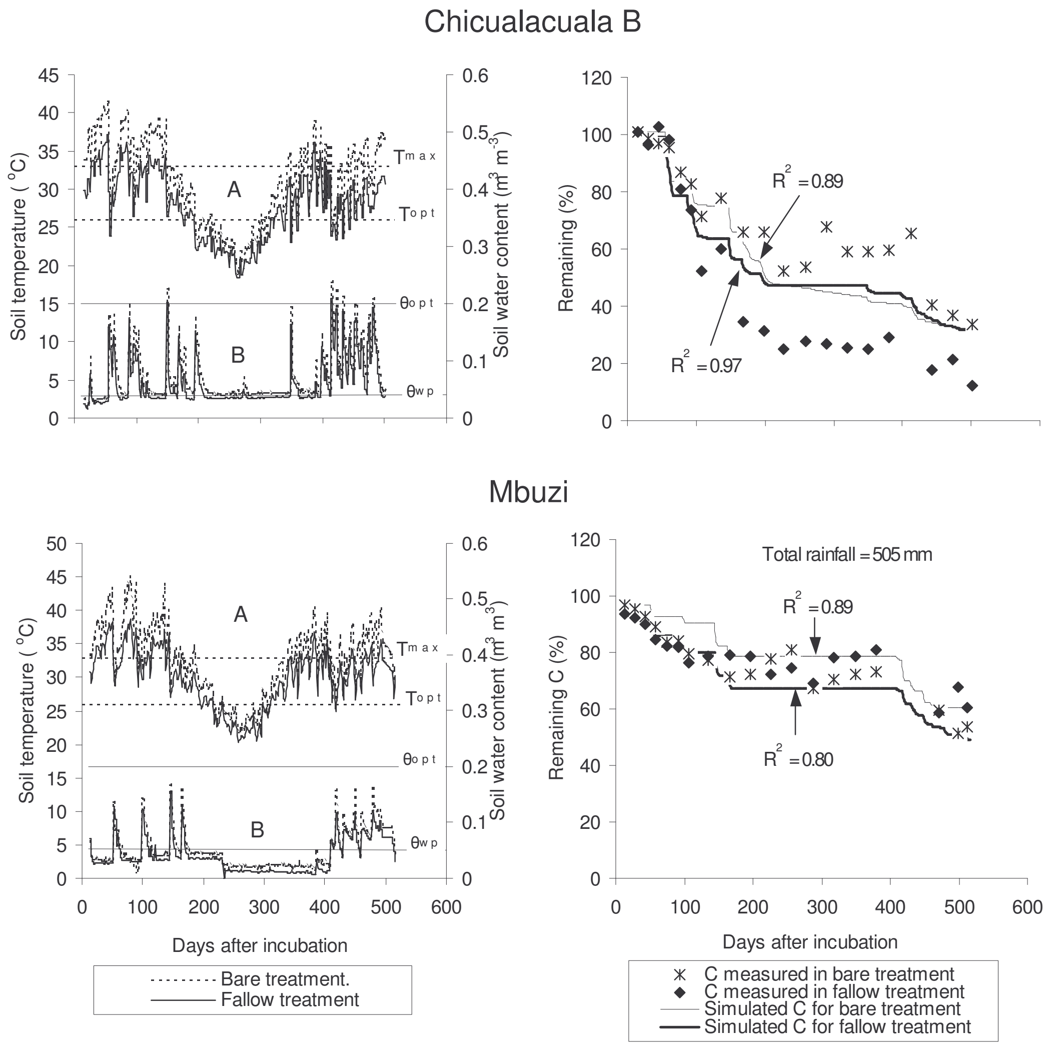


**Figure 4.5b** Soil microclimatic conditions at 5 cm depth and C change during the experimental period at the two inland and drier sites. A: soil temperature (℃), $T_{max}$: upper boundary temperature at which the temperature response function = 0 and $T_{opt}$: optimum temperature at which temperature response function = 1. B: soil water content ($m^3$ $m^{-3}$), $\theta_{wp}$: wilting point at which the water response function = 0 and $\theta_{opt}$: optimal water content at which the water response function = 1

changes in C content occurred up to about 400 days, followed by a second fairly rapid decrease at a slower rate.

The rate of decrease in C content during the first 200 days was faster at coastal sites than inland, which resulted in less C content remaining at the sites near the coast at the end of this period. Measured values relative to initial content in Fallow treatments were about 20% at Mahoche and Temane, 35% at Mabote, 55% at Chicualacuala B and 70% at Mbuzi. The values at the coastal and wetter sites did not change considerably up to the end of the experiment, whereas at the inland and drier sites further losses were observed after the second rainy season. At the drier sites, the remaining C at the end of the experiment was approximately 20% in Chicualacuala B and 50% in Mbuzi.

The rate of decrease in C content throughout the experiment tended to be higher in the Fallow than Bare treatments at all sites, except Mbuzi. At termination of the experiment the remaining C content at Mahoche and Chicualacuala B was higher in Bare than Fallow treatments, whereas at Temane, Mabote and Mbuzi the amount was about the same in both treatments.

#### *4.3.4 Parameter values and simulated carbon loss*

The estimated $\theta_{wp}$ values varied between 0.033 and 0.052 $m^3\ m^{-3}$ for the five sites, regardless of treatment. All other parameters namely $T_{max}$, $T_{opt}$, $\theta_{opt}$, $h$ and $k_1$ were optimised for the whole experiment and the results were 33 $^oC$, 26 $^oC$, 0.2 $m^3\ m^{-3}$, 0.30 and 0.063 $day^{-1}$, respectively. Both measured and simulated C was higher in Bare than Fallow treatments, except at Chicualacuala B (Figure 4.5a and 4.5b). *RMSE* and systematic bias varied between sites and treatments (Table 4.4). The magnitude of *RMSE* varied between 7.2 (Bare - Mbuzi) and 11 (Fallow - Chicualacuala B); the systematic bias varied between -0.1 (Bare - Mahoche) and 11 (Fallow - Chicualacuala B); and the coefficient of determination ($R^2$) varied between 0.79 (Bare - Mahoche) and 0.97 (Fallow - Chicualacuala B). The model explained 86% of the variation in the total data set with no trend in the residuals (Figure 4.6).

**Table 4.4** Model fit and systematic bias for the five experimental sites (% of initial carbon content)

| Site | Treatment | RMSE [a] | MBE [b] | $R^2$ |
|---|---|---|---|---|
| Mahoche | Bare | 11.8 | -0.1 | 0.79 |
| | Fallow | 11.1 | 8.7 | 0.79 |
| Temane | Bare | 7.7 | -2.5 | 0.92 |
| | Fallow | 4.8 | 2.4 | 0.96 |
| Mabote | Bare | 8.2 | -2.8 | 0.90 |
| | Fallow | 8.2 | -5.8 | 0.93 |
| Chicualacuala B | Bare | 11.2 | -7.3 | 0.89 |
| | Fallow | 14.9 | 11.0 | 0.97 |
| Mbuzi | Bare | 7.2 | 5.9 | 0.89 |
| | Fallow | 8.2 | -4.6 | 0.80 |
| All sites | | 9.7 | 0.5 | 0.86 |

[a] RMSE: root mean square error (model fit)

[b]MBE: mean bias error (systematic bias).

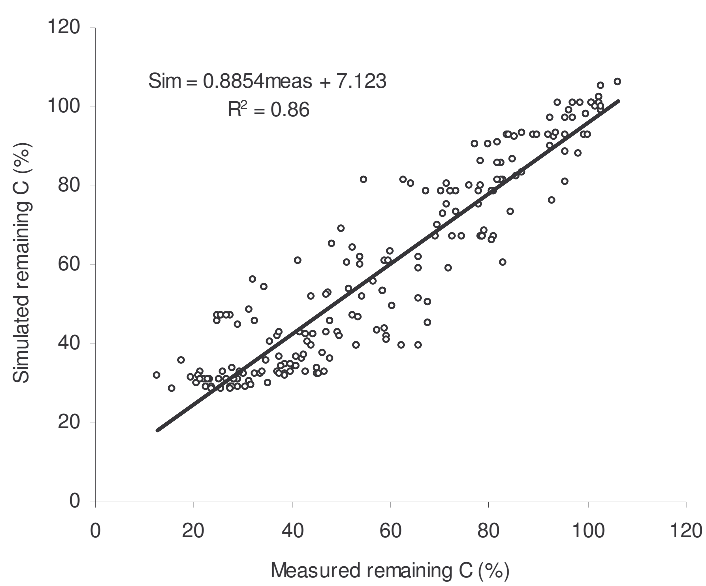


**Figure 4.6** Relationship between simulated and measured remaining organic C in the litterbags during the whole experimental period

## 4.4 Discussion

The initial increase in C content observed in this study (< 6% at four sites after 20 days of incubation) has also been observed in other studies on plant litter decomposition (Rustad, 1994; Idol *et al*., 2002). Those studies reported an increase of between 17 and 24% of the initial litter mass. Rustad (1994) ascribed the increase to contamination by bud scales, needle fragments, insect frass, pollen and other organic material unidentifiable after an incubation period lasting 54 months. In the study by Idol *et al*. (2002) incubation lasted 4 months and the increase in C content was attributed to growth of fungal hyphae and colonisation by bacteria during this period. These arguments are probably not applicable in the present study, where the increase was measured shortly after incubation. However, it is known that the nutrient content of leaves within the same tree varies (Lindsay and French, 2004). It is possible that the leaves sampled for measuring the initial C (48.3%) and ash (5.4%) content were not completely representative of the average litter quality in the litterbags, but this would not account for the large observed increase and they were probably sufficiently representative for interpretation of the results obtained.

In the study area, summer is characterised by high temperature and rainfall, factors that are favourable to the activity of soil biota and that may explain the rapid loss of C from leaf litter during the first 200 days. Several other reasons have been suggested for this finding: (i) presence of high initial content of non-structural and water-soluble material (Swift *et al*., 1979; McTiernan *et al*., 2003); (ii) breakdown of litter material increasing the total surface by exposing new surfaces to water content, electrostatic charge and chemical reactivity (Rustad, 1994); (iii) exposure of new surfaces and their susceptibility to enzymatic attack (Berg and Ekbohm, 1991); and (iv) high microbial decomposition of leached litter as a result of rainfall washing out the tannins that reduce the palatability of litter (Salamanca *et al*., 2003). These may explain the pulses of rapid decrease in remaining C content observed after raining events.

The experimental sites have all high sand contents (86-94%) and the trends observed after the first period of rapid decay are dependent on the climatic conditions at the sites:

1. At the coastal and wetter sites of Mahoche and Temane, as well as the transitional site of Mabote, a gradual slow loss from leaf litter with a final organic C content of 20 to 30% was observed in the Fallow treatment. According to Berg and Ekhbohm (1991), the gradual slow loss can result from an increasing proportion of less decomposable material.
2. At the inland and drier sites of Chicualacuala B and Mbuzi, the decomposition of leaf litter was slow with no significant loss of organic C until the last 100 days of the experiment, when a second period of rapid decay followed. This higher sensitivity of decomposition to rainfall pattern is consistent with data in Frost (1996). A possible explanation for this phenomenon is that the soil water content was generally at or below wilting point and rarely reached an optimum content, as a net result of the low rainfall during winter and low water-holding capacity of the sandy soils. In summer when the rains begin, the water supply creates temporarily favourable conditions for microbial activity in the 0-5 cm soil layer. Therefore, decomposition occurs but the decomposers often fall into stress caused by water deficit restricting or impeding their activity. Similar decomposition patterns were found by Hamadi *et al*. (2000) in other dry agroecosystems. The final C content in the leaf litter at Chicualacuala B and Mbuzi was between 20 and 60% of the initial amount.

The trends of C loss from leaf litter observed in this study agree with several other studies on leaf litter decay of miombo woodland as cited by Frost (1996), where leaf litter decay after one year was more than 90% under wet conditions but less than 40% under dry conditions. Attignon *et al*. (2004) also reported lower decay rates of leaves from one plant species in drier sites.

The Fallow treatment in general resulted in lower remaining C content than the Bare treatment throughout the course of the present experiment. This result is similar to the findings of Xuluc-Tolosa *et al*. (2003), who suggested that enhanced microbial environments in forests with more than 13 years of development could be the main reason for similar differences in their study. During the experimental period the mean daily soil temperature was higher in the Bare treatment than in Fallow. In summer the Bare treatment showed temperatures higher than 33 ºC, the estimated upper boundary for decomposition according to model optimization. The Fallow treatment only occasionally reached this boundary value. Strong winds at the coastal sites due to their proximity to the Indian Ocean could further have stimulated high soil water evaporation in the Bare treatment. These effects of wind and high temperatures were probably prevented in the Fallow treatment by the presence of a thick litter layer, which reduced amplitude of soil temperature oscillations (Facelli and Pickett, 1991) and lowered water evaporation.

Model fit and bias were generally good for the whole experiment but varied between sites. Mahoche and Chicualacuala B can be used as examples for the variation as they had almost the same total rainfall (679 and 635 mm respectively) and almost similar C content. In spite of that, the simulated curves differed. Mahoche had the lowest RMSE and $R^2$ of all sites. The higher RMSE, MBE and $R^2$ show that for Chicualacuala B, model fit was poorer, with a higher systematic bias. A possible explanation is that the simulation was based on a mathematical relationship where daily means of soil water content and soil temperature were determining factors but other environmental factors of either chemical or biological nature were not taken into account. This suggests that determinants not considered by the simulation contributed to the differences between measured and simulated remaining organic C in leaf litter at Chicualacuala B.

The model applied in this study was based on soil water content and soil temperature as driving factors of the rate of decomposition of leaf litter. However, it should be noted that the decomposition rate of fallen leaves in the field can be influenced by litterbag mesh size, which not only determines the microclimate but also restricts colonisation by certain faunal groups (Bradford *et al*., 2002). Contact between buried leaf litter and soil particles can lead to faster decomposition than for leaves falling naturally on the surface (Andrén and Paustian, 1987; Costa *et al.*, 1990) due to more favourable temperature and water content in the soil (Pekrun *et al.*, 2003) and naturally senesced leaves may decompose more slowly because of changes in their N and P content prior to abscission (Xuluc-Tolosa *et al.*, 2003). Nevertheless, the model explained 86% of the total variation, with water being the stronger determinant, explaining 75% of variation alone.

## 4.5 Conclusions

In sandy soils of southern Mozambique, soil water content has a larger effect on C loss from *Brachystegia spiciformis* leaf litter than soil temperature. Leaf litter in more than 15 year old bush fallow fields loose C faster than leaf litter in recently abandoned agricultural fields cleared of any vegetation. The modified decomposition model used explained 86% of the total variation in C loss from leaf litter with soil water content and soil temperature as driving factors.

# CHAPTER 5

## Organic matter recovery in sandy soils under bush fallow lands in southern Mozambique

### 5.1 Introduction

Shifting cultivation is the most common farming system practised in Mozambique. This system is likely to persist due to the existence of large areas with few inhabitants (Snijders, 1985) coupled with financial limitations preventing small-scale farmers from buying fertilisers. The farmers slash and burn a vegetated area, plant crops for 3-5 years and then abandon it to bush fallow due to low yields resulting from a decline in soil fertility (MAP, 1996; Geurts, 1997; DPADRI, 2002). Bush fallow is intended to restore fertility lost during the cropping phase. The success of this process is site-dependent since climate and vegetation are the main natural factors determining the dynamics of organic matter (OM) and related nutrients. Climate influences the vegetation composition and the intensity of microbial activity (Stevenson and Cole, 1999), while vegetation composition determines biomass accumulation and the amount OM added to the soil (Blum, 1997; Lal *et al.*, 1999; Mills *et al.*, 2005).

Organic matter is a vital component of the soil as it determines the dynamics of physical, chemical and biological activities. In addition to its capacity to improve soil structure and water infiltration OM also buffers pH, chelates metals and retains cations and anions in the soil system (Smith *et al.*, 2000). Thus, the ultimate result is favourable conditions for root development and improved crop yield.

In southern Mozambique, shifting cultivation is mainly practised on low fertility sandy soils because they are dominant (MAP/FAO, 1983; Geurts, 1997). The limited physical and chemical protection of carbon from oxidation leading to low OM content is the main reason for the low fertility of these tropical soils (Tiessen *et al.*, 1994; Smith *et al.*, 2000). However, studies aimed at describing and explaining the factors that determine the dynamics of OM in these soils under different periods of bush fallow are scarce. Therefore, studies of OM in soils under bush fallow in tropical conditions are crucial and current data suggest a need for refinement of the methodologies applied (Salcedo *et al.*, 1997; McGrath *et al.*, 2001). Considering that most of the processes take place in the topsoil, there is a need to assess shallow depths to avoid the dilution effect of deeper layers (Crépin and Johnson, 1993; Govaerts *et al.*, 2006).

The objectives of the present study were to quantify OM and explain the factors determining its dynamics in sandy soils under different periods of bush fallow. The hypotheses tested were: (i) OM increases with rainfall through the positive effects of rainfall on biomass production by vegetation; and (ii) OM increases with age of bush fallow.

## 5.2 Materials and methods

### *5.2.1 Study area*

The study was conducted in the Inhambane and Gaza provinces of southern Mozambique (21°00' - 25°20'S; 31°20' - 35°40'E) (Figure 5.1). From the coast to inland there is a gradient in topography, climate and natural vegetation. According to Reddy (1985; 1986) elevation increases from sea level up to 1000 m inland. Along this same gradient mean annual temperature increases from 20°C to 26°C, whereas mean annual precipitation decreases from more than 1000 mm to less than 400 mm. Reddy (1985) divided the area into four climatic zones based on moisture index as: sub-humid, wet semi-arid, dry semi-arid and arid. miombo woodland which is a characteristic feature of the coastal region is replaced by arid savannah inland (Campbell *et al.*, 1996) with mean annual rainfall of approximately 700 mm as a threshold value (Frost, 1996).

The area is mainly covered by sand material deposited by marine transgression and regression between the Pliocene and Holocene epochs, around 5.4 million to 10 thousand years ago, during the opening of the Indian Ocean and eastern Africa continent margin (Salman and Abdula, 1995). The sand was deposited over a Precambric basement and sequence of Karroo and Post-Karroo effusives (Flores, 1973).

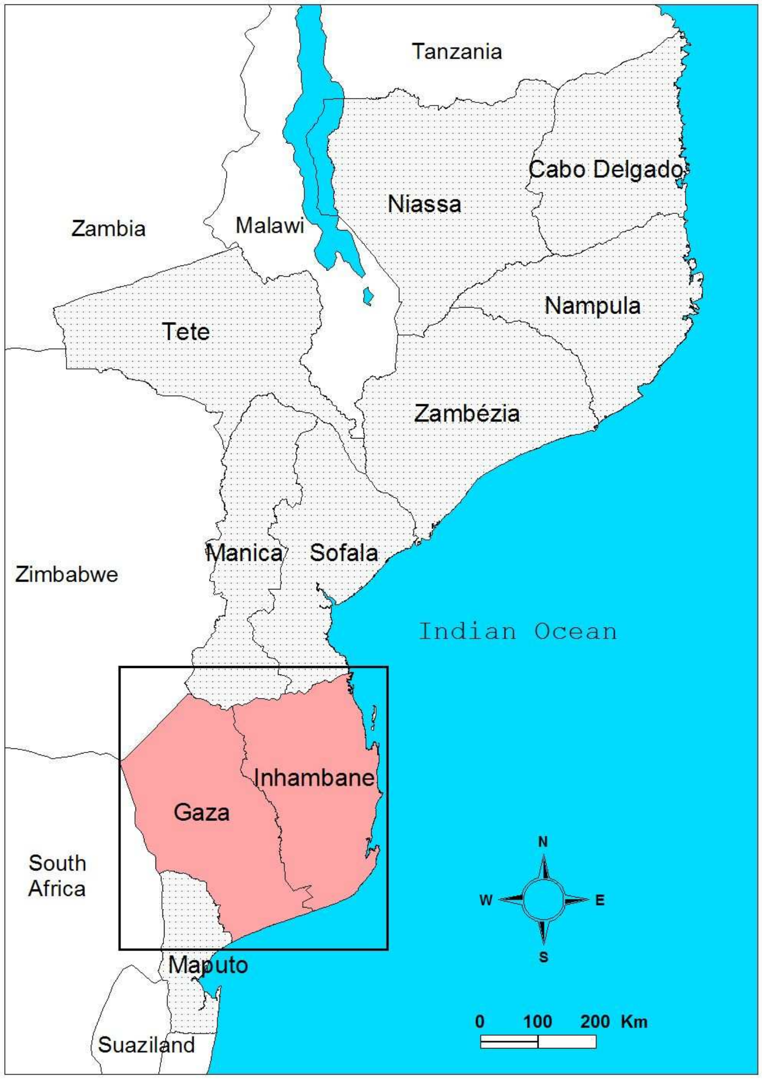


**Figure 5.1** The study area in southern Mozambique

### 5.*2.2 Agroecosystem and site selection*

Based on recommendations by Swift *et al.*, (1979) and the results of local studies (Reddy, 1986; Geurts and Van den Berg, 1998), six agroecosystems (AE 1-6) were identified for the present study (Table 5.1). Each of them represents a rainfall region one of which is transitional. An AE was defined as a region in which the three environmental factors that affect yield, namely climate, slope and soil, are similar (Du Preez and Van Zyl, 1998). Three sites were selected in each of the six AE (except

**Table 5.1** Geographical location and general climatic and soil characteristics of the sampling sites

| Agroecosystem | Sampled Sites | Lat. | Long. | Altitude* (m) | MAR** (mm) | MAT*** (°C) | Broad climatic zone according to moisture index | Soil units**** FAO 1988 | Soil units**** USDA 1992 | Landform**** |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Morungulo | 23° 13' | 35°28' | | | 20-23 | Sub humid | Haplic Arenosols | Ustic Quartzipsamments | Coastal dunes, holocenic sand |
| | Jangamo | 24° 16' | 35°19'' | | > 1000 | | | Arenosols | Psamments | |
| | Xihahlo | 25° 12' | 33°20'' | | | | | Ferralic Arenosols | Ustoxic Quartzi-psamments | Inner dunes |
| 2 | Mahoche | 21° 35' | 35°13' | | | | Wet semi-arid | Arenosols | Psamments | |
| | Macaringue | 23°04' | 35°13'' | | 800 – 1000 | | | | | Sandy plain |
| | Mocumbi | 24°37' | 34°51'' | | | | | | | |
| 3 | Temane | 21°45' | 34°54'' | < 200 | | | | Arenosols | Psamments | |
| | Panda | 24°13' | 34°31'' | | 600 – 800 | 20-26 | | | | Inner dunes |
| | Nhapajane | 24°27' | 34°38'' | | | | | | | |
| 4 | Massangena | 21°39' | 32°53'' | | | | | Chromic Cambisols | Typic Ustochrepts | Sandy and redish colluviums |
| (Transitional) | Mabote | 22°03' | 34°04'' | | (400-800) | | Dry semi-arid | | | |
| | Funhalouro | 23°05'' | 34°23'' | | | | | | | |
| 5 | Chicualacuala A | 22°13'' | 31°38'' | | | | | | | |
| | Chicualacuala B | 22°25' | 32°05'' | | | | | Arenosols | Psamments | Sandy plain |
| | Mapai | 22°44'' | 32°12'' | | 400-600 | | | | | |
| | Gerege | 23°14'' | 32°25'' | 200 - 500 | | | | | | |
| 6 | Mbúzi | 22°28'' | 31°25'' | | | 23-26 | Arid | | | |
| | Xitsutsuini | 22°35'' | 31°34'' | | < 600 | | | Calcaric Cambisols | Typic Ustochrepts | Shallow soils over calcarium rocks |
| | Salani | 22°37' | 31°41'' | | | | | | | |

Source: * Altitude (Reddy, 1985); **MAR – Mean annual rainfall (FAO, 2000); *** MAT – Mean annual temperature (Reddy 1985); **** INIA (1994; 1995).

AE5 where one additional site was included) resulting in nineteen sites (Figure 5.2). General climatic and soil characteristics of the selected sites are described in Table 5.1. Each site comprised five fields under different land uses (treatments) namely cultivated (CT), <5 years of fallow (5F), 5-15 years of fallow (10F), >15 years of fallow (15F) and virgin (VG). In Morungulo (AE1), no VG field was found. During selection of fields at each site care was taken to identify fields with similar slope and apedal soils deeper than 1.0 m within a radius of 0.5 km. The similarity of the soils was verified by digging a profile pit in the 15F fields and auger sampling the others.

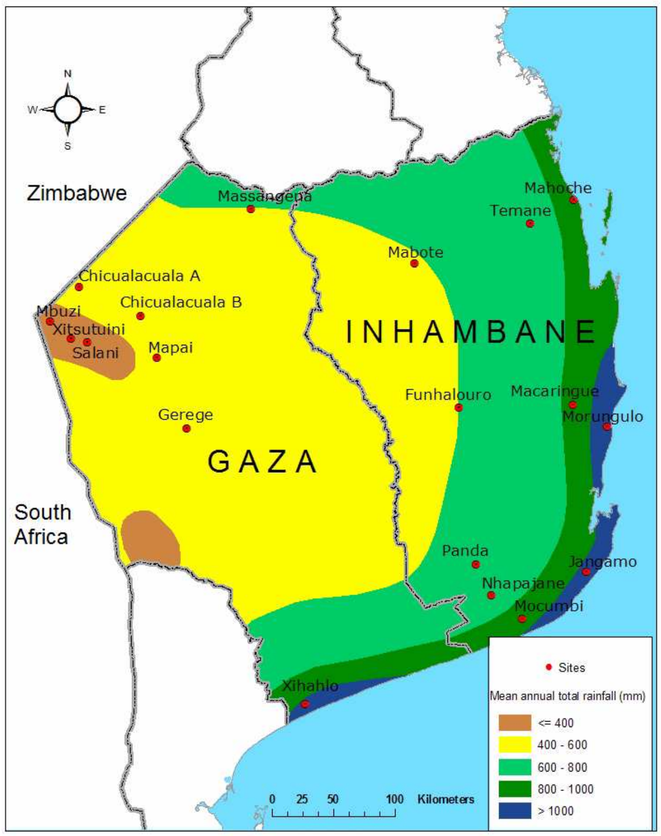


**Figure 5.2** Mean annual rainfall regions of the study area (adapted from FAO, 2000) regarded as agroecosystems and the selected sites for soil sampling

### *5.2.3 Soil sampling and laboratory analysis*

In each field three samples were collected for determination of bulk density, particle size distribution, organic carbon and total nitrogen from three depth intervals namely, 0-50 mm (L1), 50-100 mm (L2) and 100-200 mm (L3). Each of these samples was a composite of four sub-samples collected within a diameter of 10 m. Sub-samples for bulk density were collected with a standard cylinder of 100 $cm^3$ cylinder, oven-dried at 90℃ for 48 hours and weighted. Samples for particle size distribution (pipette), organic carbon (Walkley-Black) and total nitrogen (Kjeldahl) analyses were air-dried, sieved and stored to be analysed by Central Agricultural Laboratories in South Africa according to standard methods (The Non-Affiliated Soil Analysis Work Committee, 1990).

### *5.2.4 Data processing*

The bulk density of a soil layer was used to convert the content of carbon and nitrogen in that layer to mass per unit surface area before Analysis of Variance was performed using General Linear Model (GLM ANOVA) at $P<0.05$ with the statistical software package for Windows NCSS 2000 (Hintze, 1998). The reason for using GLM ANOVA was that the numbers of fields in each site and sites in each AE were unequal. Fisher's least significant difference (LSD) was calculated at $P<0.05$ where GLM ANOVA revealed significant differences. The analyses were for land uses in each AE and similar land uses across agroecosystems.

## 5.3 Results

### *5.3.1 Soil texture and bulk density*

Soil texture varied between sand in AE1, AE2, AE3 and AE4 to loamy sand in AE5 and AE6 with a weighted mean particle size distribution of 88-93% sand, 2-8% silt and 4-6% clay to 200 mm depth (Table 5.2). From coast to inland the sand content tended to decrease, whereas the silt content tended to increase with no obvious trend in the clay content. The weighted mean bulk density to 200 mm depth varied from 1.27 g $cm^{-3}$ in AE3 to 1.54 g $cm^{-3}$ in AE5.

**Table 5.2** Bulk density and particle size distribution of soils in the agroecosystems

| Agroecosystem | Depth (mm) | Bulk density (g cm$^{-3}$) | Particle size distribution (%) | | | Textural class |
|---|---|---|---|---|---|---|
| | | | Sand | Silt | Clay | |
| 1 | 0-50 | 1.28 | 92 | 4 | 4 | Sand |
| | 50-100 | 1.35 | 92 | 4 | 4 | |
| | 100-200 | 1.37 | 94 | 2 | 4 | |
| 2 | 0-50 | 1.34 | 93 | 2 | 5 | |
| | 50-100 | 1.44 | 93 | 2 | 5 | |
| | 100-200 | 1.44 | 92 | 2 | 5 | |
| 3 | 0-50 | 1.27 | 91 | 4 | 5 | |
| | 50-100 | 1.29 | 91 | 4 | 5 | |
| | 100-200 | 1.25 | 91 | 3 | 5 | |
| 4 | 0-50 | 1.45 | 91 | 4 | 6 | |
| | 50-100 | 1.48 | 90 | 4 | 6 | |
| | 100-200 | 1.49 | 90 | 4 | 6 | |
| 5 | 0-50 | 1.55 | 88 | 6 | 6 | Loamy sand |
| | 50-100 | 1.54 | 89 | 6 | 5 | |
| | 100-200 | 1.54 | 89 | 7 | 5 | |
| 6 | 0-50 | 1.50 | 89 | 7 | 4 | |
| | 50-100 | 1.52 | 88 | 8 | 5 | |
| | 100-200 | 1.51 | 87 | 8 | 5 | |

### *5.3.2 Organic carbon*

In general, the tendency observed for organic C was an increase with age of bush fallow (Figure 5.3). Significant differences in agroecosystems were only found in L1 and L2 of AE2, all layers of AE3 and L1 of AE4. In L1 of AE2 differences were found among all fields with a decrease of C content from VG to CT, followed by a further decrease to 5F and then an increase to 15F; in L2 of AE2 no differences were observed from VG to CT but a decline occurred from CT to 5F, followed by an increase to 10F and no increase thereafter. In L1 of AE3, C content declined from VG to CT and this tendency was maintained to 5F, after which no substantial increase was observed; in L2 of AE3 no significant differences were observed from VG to CT fields but a decline was found from CT to 5F followed with an increase to 10F and a decline in the 15F fields; in L3 a decline was found from VG to CT and no differences occurred between CT and 5F, thereafter the pattern was similar to L2. In L1 of AE4, C content was higher in VG than in other fields and declined almost continuously to 15F with no differences between CT, 5F and 10F.

The comparisons of organic C for similar land uses across agroecosystems are displayed in Figure 5.4. Significant differences were found in all land uses, except in L1 of CT and in L3 of 10F. The highest C contents were measured in all layers of AE1, except in 5F and in L3 of 15F. The lowest contents were recorded in AE2, except in L3 of CT and 5F and L1 of 15F. Regarding land uses, the C contents in L2 of CT fields in AE1 were only different from AE2, AE4 and AE6, whereas in L3 the tendency was a decline from AE1 to AE3 and AE4 and then an increase to the same levels as of AE1 in AE5 and AE6. In L1 and L2 of 5F fields, the C content declined from AE1 to AE2 and thereafter increased gradually reaching similar levels to AE1 in AE4 to AE6. A similar pattern was observed in L3 with a minimum C content recorded in AE3. Among 10F fields, the C content in L1 declined from AE1 to AE2, and subsequently increased in the remaining agroecosystems reaching similar levels as in AE1; in L2 the C content differed between AE1 and all other agroecosystems. In L1 and L2 of 15F fields the C content declined from AE1 to AE2, AE3 and AE4. However in AE5 and AE6 it reached similar levels as in AE1; in L3 the decline was from AE1 to AE2 whereas from AE3 to AE6 similar levels as in AE1 were reached. In L1 of VG fields the C content declined from AE1 to AE2 and thereafter showed an increase to the AE1 level in AE3 and AE6, whereas in AE2, AE4 and AE5 low contents were found. In L2 the C content declined from AE1 to AE2, AE3, AE4 and AE5 and increased to a similar level as AE1 in AE6; and in L3, the levels in AE1 were similar to those in AE3, AE5 and AE6, whereas lower values were recorded in AE2 and AE4.

**Organic C (kg ha$^{-1}$)**

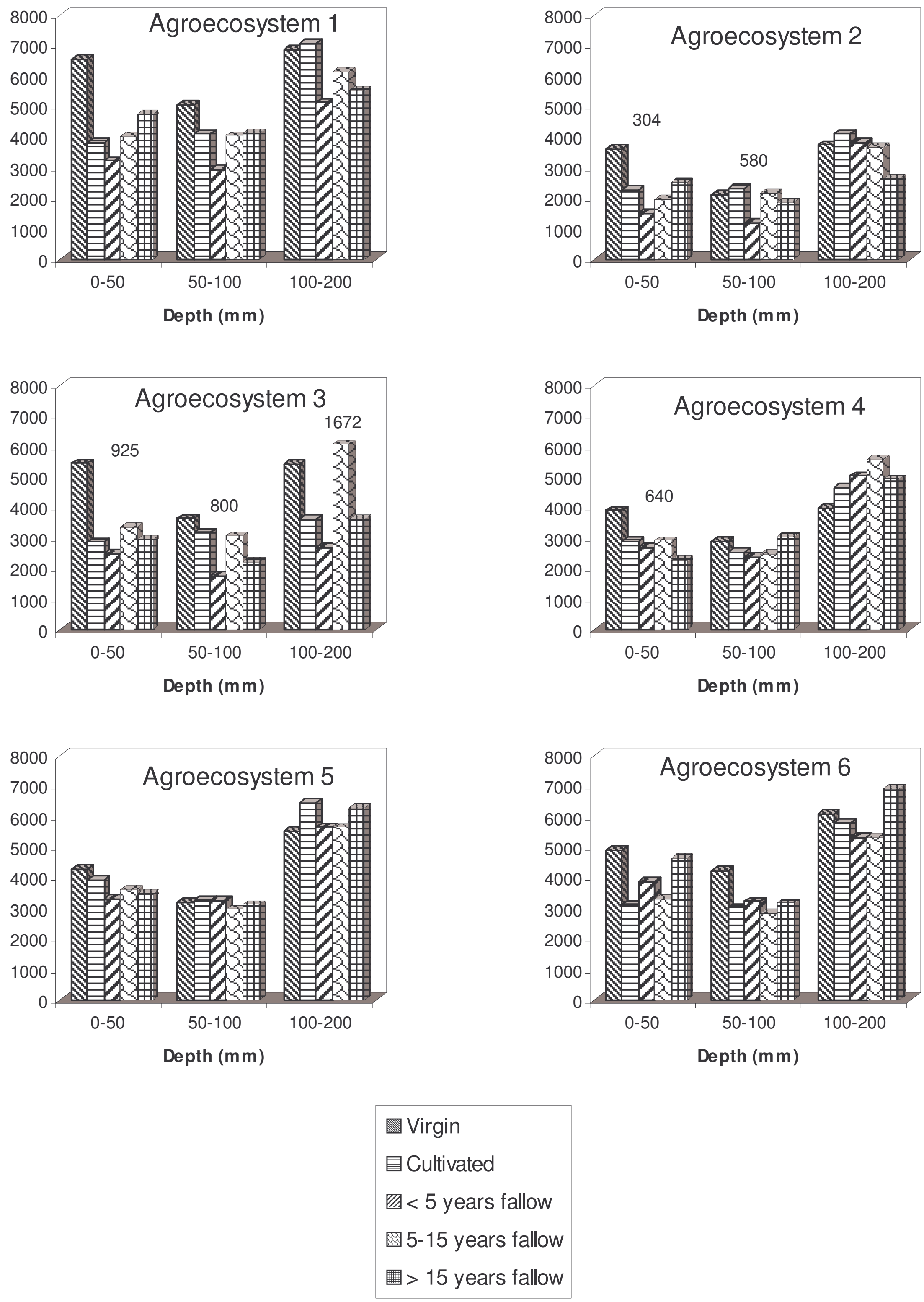


**Figure 5.3** Organic carbon in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

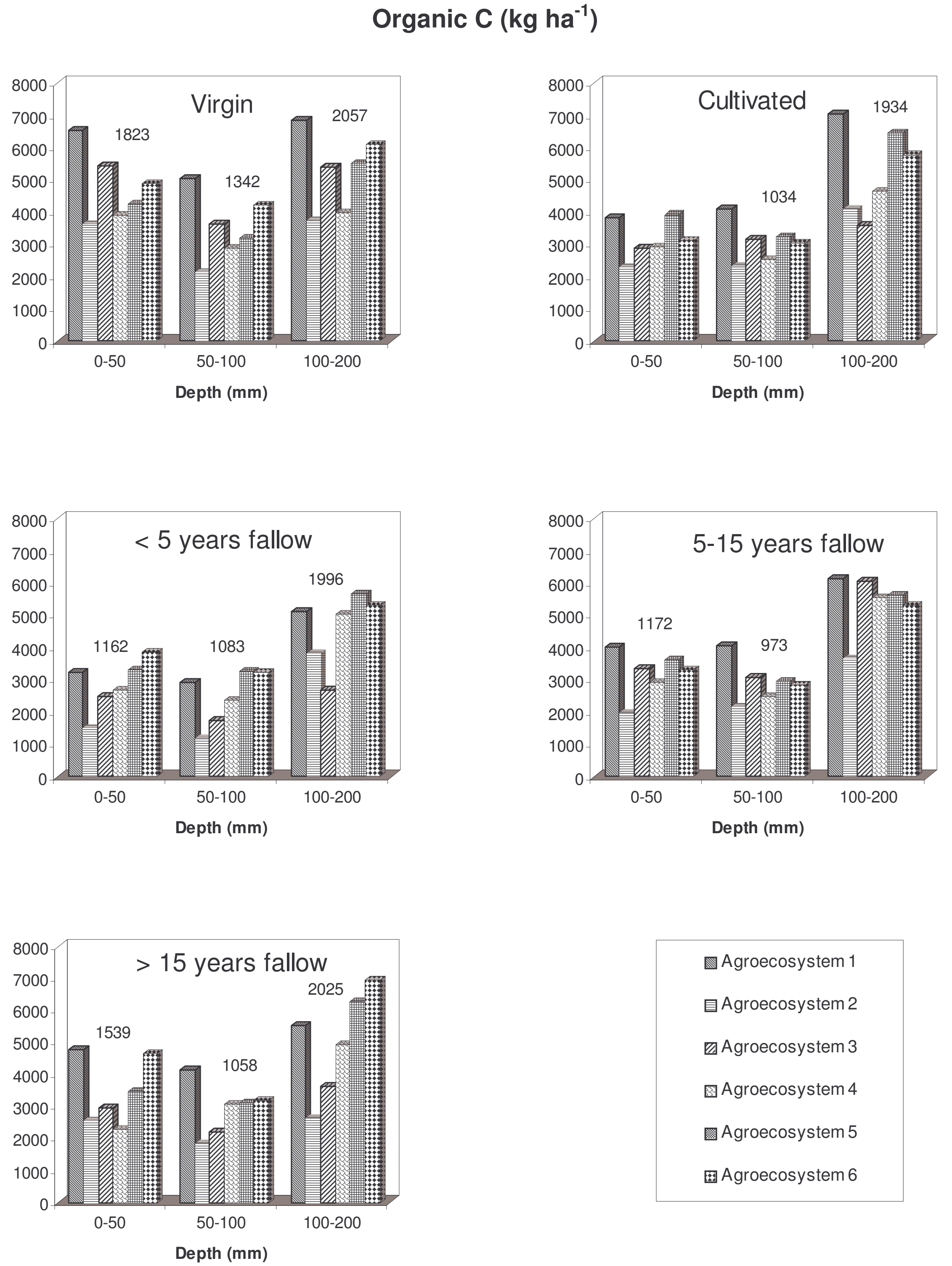


**Figure 5.4** Organic carbon in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

#### *5.3.3 Total nitrogen*

In general, no clear trend was found for total N between land uses within agroecosystems, except in AE1, AE3 and AE6 where it seemed to decline from VG to CT and then increase again in fallow fields (Figure 5.5). Differences were only found in L1 of AE2 and AE3. In both cases, higher contents were found in VG than in other fields. The content in 5F was different from 10F in AE2 and from 10F and 15F in AE3.

In similar land uses across agroecosystems, a declining trend in total N from AE1 to AE3, AE4 or AE6 was observed (Figure 5.6). Differences were only found in the L2 and L3 of CT, all layers of 10F and L1 and L3 of 15F fields. In L2 of CT, AE6 had a lower content, which was the only difference from AE1, AE2 and AE5. However, in L3 total N content declined from AE1 to AE3 and then increased to AE5 where the content was similar to that of AE1 and then declined again to AE6. In all layers of 10F fields a decrease from AE1 to AE4 followed by an increase to AE5 and another decrease to AE6 was found. In L1 differences were only recorded between AE1 and AE4/AE6, AE2 and AE4, and AE4 and AE5; in L2, AE4 was different from AE1, AE2, AE3 and AE5, whereas AE6 was different from all other agroecosystems, except AE4; in L3, differences were only found between AE4 and AE1, AE3 and AE5.

#### *5.3.4 C/N ratio*

No clear trends resulted from the conversion of VG to CT or the abandonment of the latter to bush fallow within agroecosystems (Figure 5.7). Furthermore there were no recorded differences among land uses, except L2 and L3 of AE3, L1 of AE4 and L3 of AE6. L2 of VG and CT fields in AE3 had similar C/N ratios and both differed from fallow fields (5F, 10F and 15F), which had low ratios. A similar trend was observed in L3 where VG was different from CT and 15F, and 5F was different from CT and 10F. In L1 of VG in AE4 a higher C/N ratio was found, which was different from the CT and fallow fields, whereas 10F was different from CT and the remaining fallow fields. In L3 of AE6, C/N ratio in VG was different from CT, 5F and 15F, whereas 5F differed from CT, 10F and 15F fields.

There were no clear trends among similar land uses across agroecosystems (Figure 5.8). However, significant differences were recorded in all land uses, except L3 of 5F

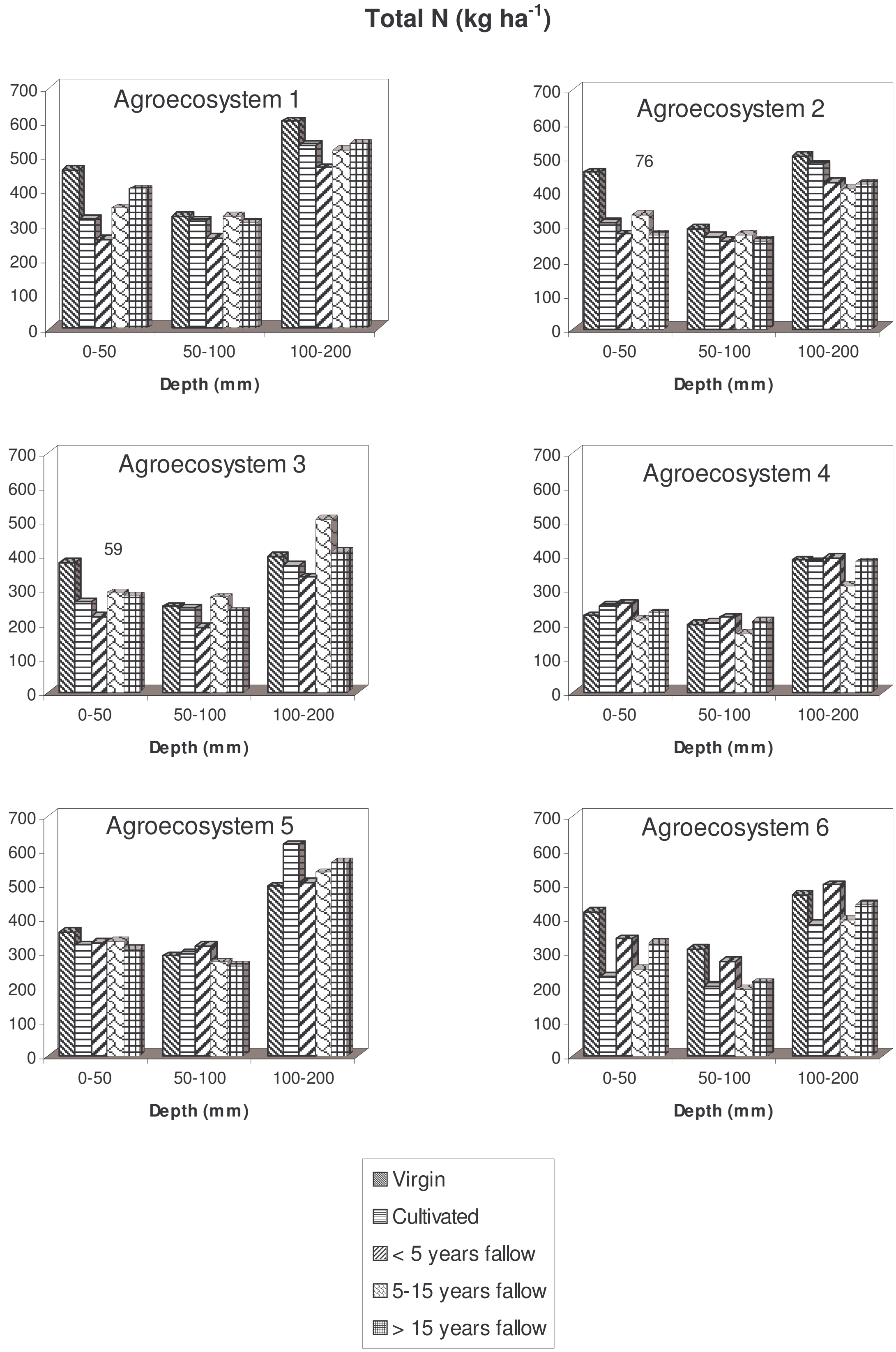


**Figure 5.5** Nitrogen in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

**Total N (kg ha$^{-1}$)**

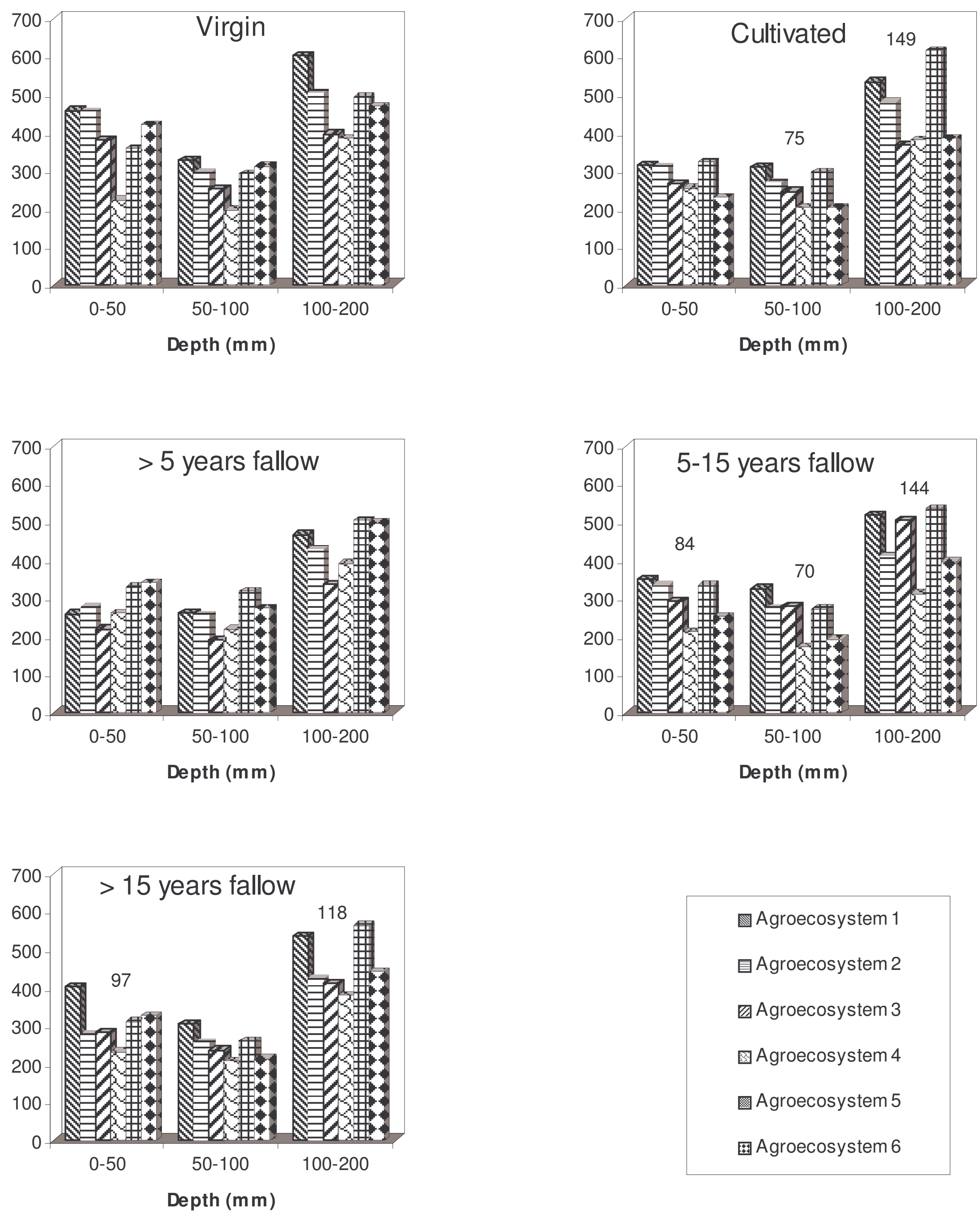


**Figure 5.6** Total nitrogen in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

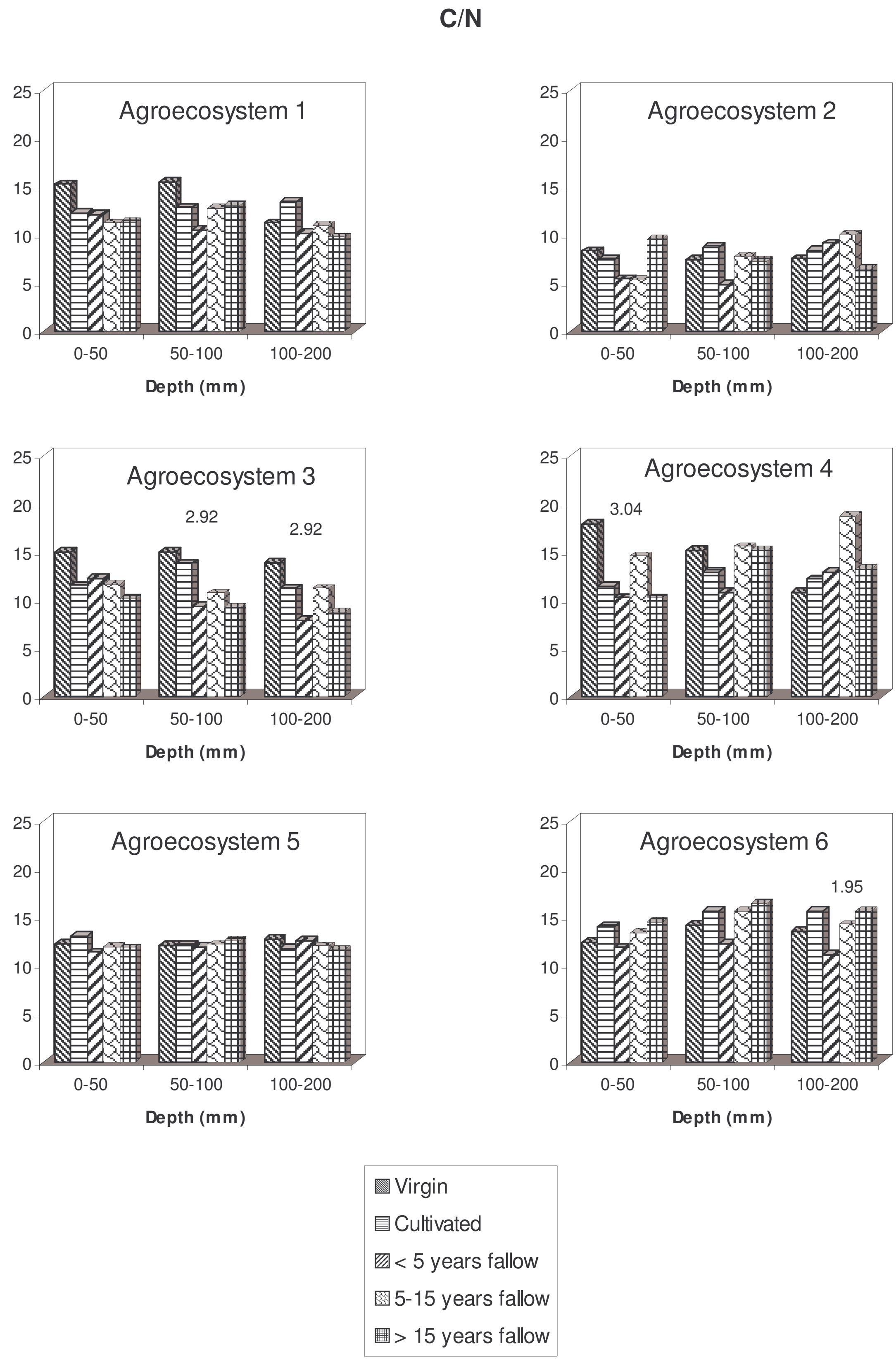


**Figure 5.7** C/N ratio in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

**C/N**

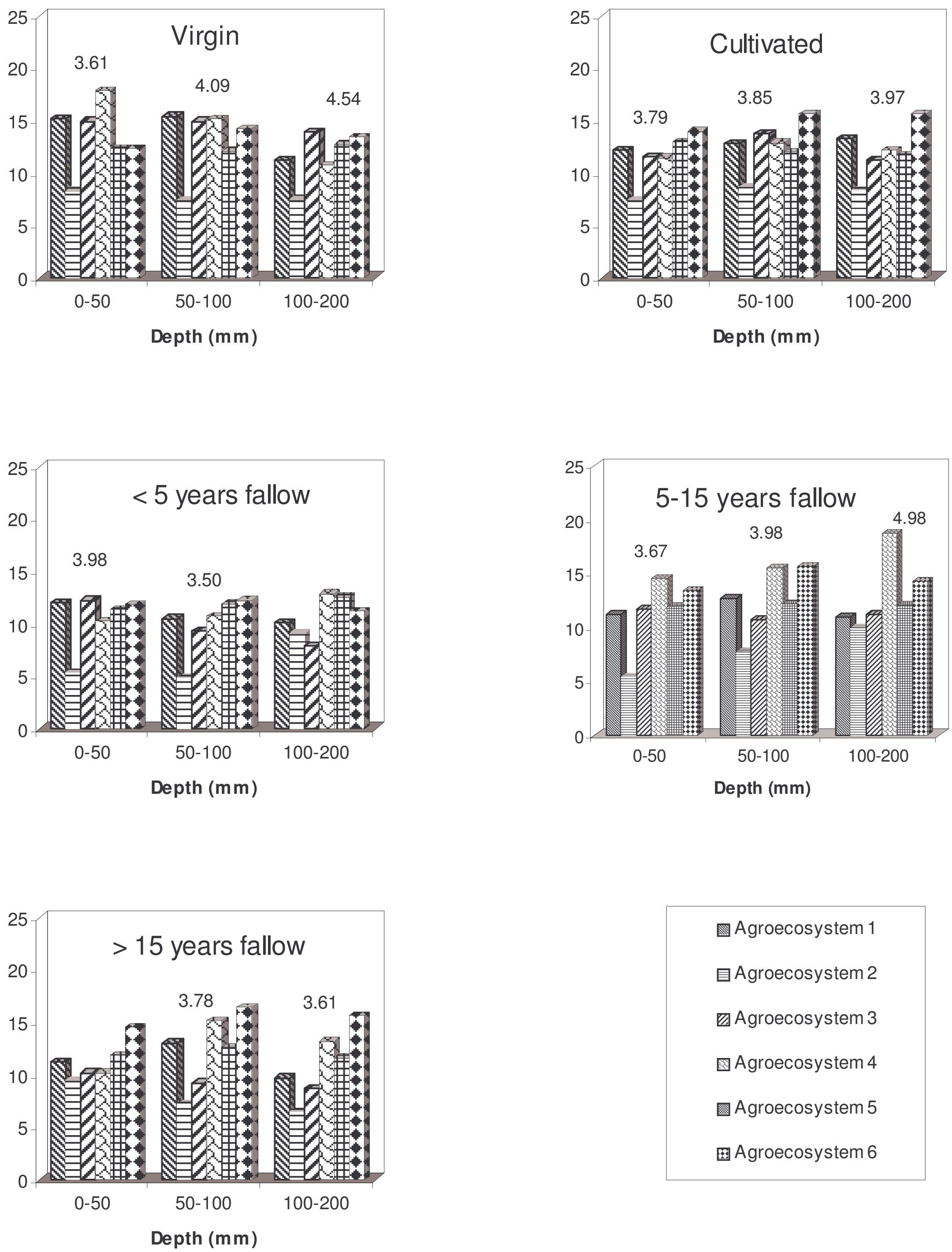


**Figure 5.8** C/N ratio in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

and L1 of 15F. In all cases where differences were found, AE2 had the lowest C/N ratio. In L1 and L2 of CT, 5F and 10F fields only AE2 was different from the other agroecosystems, whereas in L3 of CT it was different from AE6 and in L3 of 10F, AE4 had a higher ratio, which differed from all other agroecosystems. In L2 of 15F, AE2 was different from the other agroecosystems except AE3; in turn, AE3 was different from AE1, AE4 and AE6, whereas AE6 differed from AE3 and AE5. In L3, AE2 differed from AE4, AE5 and AE6, AE3 from AE4 and AE6 from all other agroecosystems except AE4. Similarly to CT, 5F and 10F, in L1, L2 and L3 of VG fields AE2 was different from the other agroecosystems except with AE1 and AE4 in L3. Additional differences were found between AE4 and AE5/AE6 in L1.

## 5.4 Discussion

This study was performed in sandy soils of adjoining agroecosystems in southern Mozambique. The agroecosystems are characterised by a decreasing rainfall and an increasing temperature from AE1 to AE6. As farmers in these areas use mainly hand hoes for land preparation only the upper 200 mm soil layer was investigated. The results for organic C, total N and C/N are discussed separately below.

### *5.4.1 Organic carbon*

The high rainfall (>1000 mm) in AE1 should have induced high biomass production resulting in high organic C input to the soil. Therefore, organic C contents in all land uses in this AE were expected to be higher than in similar fields in other agroecosystems. However, no significant differences were found, due to high variability within each land use. This suggests that the combination of high rainfall and sandy soils, which are known to have low CEC, should have resulted in leaching of dissolved organic matter containing C (Cleveland *et al.*, 2004; Vinther *et al.*, 2006) and a probably heterogeneous distribution of coarse woody debris probably resulted in the high variability in the fields (Hafner *et al.*, 2005). In AE2, the leaching effect was probably reduced due to lower rainfall (800-1000 mm) resulting in differences between land uses only in the top layers. Moreover, an eventual lower biomass production should have led to lower C content than in AE1. The lower rainfall (600-800 mm) in AE3 should have diminished the leaching effect, resulting in differences among land uses in all layers. In addition, a possible lower decomposition rate of vegetation residues could have led to accumulation of organic C resulting in a tendency for higher contents than in AE2. Moving further inland to the transitional agroecosystem (AE4) differences were found only in the first layer and no differences were found in inland and drier agroecosystems (AE5 and AE6) with rainfall less than 600 mm. High temperature and low rainfall in these agroecosystems were expected to lead to a low decomposition rate of organic residues, which combined with the slightly lower sand fraction and less weathered soils should have resulted in accumulation of organic C. In addition, based on field observations, from coast to inland, trees were gradually replaced by shrubs and grass and in the severely drier areas of AE6 vegetation consisted mainly of grass and scattered trees. These results agree with Havlin *et al.* (1999) and Stevenson and Cole (1999) who stated that in fields covered by grass there is higher accumulation of organic C.

The decline in organic C content in the first five years of fallow (5F) observed in coastal and wetter agroecosystems (AE1, AE2 and AE3) has also been found in other studies of tropical soils. Juo *et al.* (1995) cited Reich (1983) who indicated that the increase in soil temperature resulting from reduced shading is the main cause of the decrease. Brand and Pfund (1998) attributed the low input of organic matter to the soil to decomposition that occurs in this period when the succession vegetation is still building up biomass. Another possible reason for the decline can be that fields under cultivation (CT) are still benefiting from C decaying from slash and burn residues of previous vegetated bush fallow or virgin conditions. In inland and drier agroecosystems (AE5 and AE6) where regeneration of wooden vegetation is expected to be slower, any growth of grass would result in either a slow decline or gradual accumulation of organic C from CT to 5F fields.

These results suggest that rainfall is the main driver of organic C in sandy soils within the same temperature regime and that the rate of accumulation is ultimately determined by the floristic composition of vegetation, which determines the quality of residues and the pattern of input to the soil.

### *5.4.2 Total nitrogen*

Land use was not the main determinant for differences in total N content of soil in the agroecosystems. Based on field observations, N-fixing species were observed in many of the study fields, which may have caused variability that probably masked the effects of land use and agroecosystem. Mtambanengwe and Kirchman (1998) cited studies stating that the leaf litter from the miombo woodland, which is the vegetation type that covers the coastal and wetter agroecosystems of the study area decomposes slowly compared with leaf and leaflet material from N-fixing tree species. This difference in decomposition rate can also contribute to confounding the effect of land use. Therefore, it is important to study the difference in decomposition rates of vegetation residues and dynamics of N release to the soil. In AE1 in particular, high soil water content resulting from rainfall during the rainy season combined with high drainage rate of the sandy soils and soil temperature varying between 20 and 55°C favoured nitrification (Prasad and Power, 1997). Considering this, it is possible that $NO_3^-$, which is very mobile (Havlin *et al.*, 1999; Blackmer, 2000), had been produced and leached down in the soil profile resulting in high variability and no differences between land uses in the studied layers. Leaching of $NO_3^-$ is likely to be greater under bare fallow (Prasad and Power, 1997) and this can explain why the lowest values were observed in 5F fields of the coastal and wetter agroecosystems. The relatively higher total N contents found in the uppermost layer of VG fields in these agroecosystems could result from a possible low mineralization rate, which can occur when a relatively constant amount of plant residues is returned to soil, causing a relatively stable microbial population (Havlin *et al.*, 1999).

Soils under similar land uses across agroecosystems had the lowest N content in AE4. In coastal and wetter agroecosystems the higher contents may have resulted from a possible high input of biomass production, whereas in inland and drier agroecosystems the higher contents could have been due to limited decomposition caused by an unfavourable environment. In addition, based on field observations, the grass that dominated the inland and drier agroecosystems coupled with low rainfall could have contributed to less risk of N loss by leaching as grass excretes substances that inhibit nitrification (Stevenson and Cole, 1999). Considering that soil temperature in these agroecosystems reaches values above the favourable range for nitrification (Chapter 4), it is likely that limited nitrification contributes to accumulation of $NH_4^+$, which is less prone to leaching than $NO_3^-$ (Stevenson and Cole, 1999). In AE6 where it is severely dry (rainfall <400 mm) with high soil temperature evapotranspiration is likely to be high. Considering that evapotranspiration induces $NO_3^-$ uptake by vegetation (Blackmer, 2000), combined with less input from biomass, this can explain the lower total N in AE6 than AE5. The differences found among 10F fields across agroecosystems suggest that decaying residues in these fields were distributed with less variability in all the sampling layers. In the 15F fields, differences were probably more prominent in the first layer due to falling leaves and in the third layer due to established roots.

### *5.4.3 C/N ratio*

The C/N ratio was in accordance with C and N patterns previously discussed. In most fields with a similar land use across agroecosystems, organic C had the lowest values in AE2, whereas the total N was lowest in AE3 or AE4. The lowest contents were found in the 5F and 10F fields of AE3 and in the 15F and VG fields of AE4. The heavy rains that are characteristic in AE1 could have resulted in loss of N through leaching of $NO_3^-$ leading to an increase in C/N ratio. Considering that AE1 and AE2 are coastal agroecosystems with a similar type of vegetation (miombo woodland), the explanation for a lower C/N ratio in AE2 can be that AE2 has lower rainfall and thus less C input due to less biomass production and less N loss.

The C/N ratios generally tended to be higher in drier agroecosystems, with the highest values in AE6. There are two possible explanations for this phenomenon. A low soil water content can lead to less decomposition of organic material, resulting in accumulation of organic C. Residues from arid savannah species are of low quality, with high lignin and polyphenol contents, resulting in slower decomposition and less release of N.

In summary, the rainfall pattern in sandy soils determined accumulation of organic carbon and nitrogen. However, the accumulation was not linearly proportional to annual rainfall. Hence, the first null hypothesis that organic matter in the soil increases with rainfall could not be proved as higher amounts of organic C and N were found in wetter and drier agroecosystems. The null hypothesis that organic matter increases with age of bush fallow seems to apply only after five years, as a decline was observed from CT to 5F fields. Apparently, the change in vegetation from coastal to inland agroecosystems contributes in a certain manner to accumulation of organic matter. This can be confirmed by monitoring the input of C and N to the soil through the amount and quality of plant residues in bush fallow fields across agroecosystems. Therefore, in future studies it is important to investigate the role of different vegetation species in the dynamics of organic C and N in soil under bush fallow through their decomposition rate and input to the soil combined with increased number of bush fallow classes.

## 5.5 Conclusions

Rainfall proved to be the main driver of organic matter dynamics in sandy soils under semi-arid and arid conditions in southern Mozambique. In areas where rainfall was greater than 600 mm the conversion of virgin fields to cultivation decreased soil organic matter content, a trend that continued up to five years after abandonment to bush fallow after which organic matter again began to accumulate. In dry areas (rainfall 400-600 mm) organic matter tended to decline gradually even in longer fallow periods, whereas in the severely dry areas (rainfall < 400 mm) there was no clear trend. In fields under similar land use across rainfall zones, the lowest organic matter content was found in areas with annual rainfall between 800 and 1000 mm. Areas with higher and lower rainfall than this had higher organic matter accumulation probably due to higher input of biomass in the former and to lower decomposition rate of organic matter in the latter.

# CHAPTER 6

## Acidity and macronutrient recovery in sandy soils under bush fallow lands in southern Mozambique

### 6.1 Introduction

Shifting cultivation is the most commonly practised farming system in Mozambique and is the main means of subsistence for small-scale farmers (Direcção de Economia, 1996). This farming system is expected to prevail as financial constraints limit the possibility for small-scale farmers to buy fertilisers (Geurts and Chaguala, 1998), coupled with the existence of large areas that are scarcely inhabited (Snijders, 1985; MAP 1996; Folmer *et al.*, 1998). In the southern part of the country shifting cultivation is practised mainly on poorly fertile sandy soils because they are easily accessible (MAP/FAO, 1983; Geurts, 1997). Farmers slash and burn primary or regrowing forest, plant crops for 3-5 years and then abandon the land to bush fallow due to low yields induced by a decline in soil fertility (Reddy, 1985; DPADRI, 2002). The aim of abandonment to bush fallow is to restore soil production capacity lost during the cropping phase (MAP, 1996).

The ash that results from burning forest biomass has some beneficial effects on soil, e.g. an increase in pH and effective CEC (Juo and Manu, 1996), and contains essential nutrients for crops such as P, K, Ca, Mg and micronutrients (Brady and Weil, 1996). However, P could become the first limiting nutrient because the mechanism of P mobilisation from mycorrhizal-root associations and other P mobilising soil micro-organisms may be partially or completely destroyed during burning (Jordan and Szott, 1991, cited by Juo and Manu, 1996). In addition, the cropping systems are not efficient in using and storing the large amount of available plant nutrients produced by slash and burn, which can result in their loss through leaching, run-off and erosion during the cropping phase (Steiner, 1996).

Some authors have argued that shifting cultivation is no longer sustainable in the long run due to an increase in population density and consequent decline in amount of arable land per capita (Tiessen *et al.*, 1994; Steiner, 1996; Nandwa and Bekunda, 1998). However, according to Harwood (1996) this system can be sustainable if well managed. This opinion is supported by Snapp *et al*. (1998), who claim that shifting cultivation is a deserving priority for research in sparsely populated areas where small-scale farmers with financial constraints cannot afford to buy fertilisers. Mertz *et al*. (2008) emphasise that the negative opinions on the sustainability of shifting cultivation are based on empirical evidence and certain assumptions. There is thus a need for a better understanding of the system dynamics in shifting cultivation in order to determine its long-term sustainability.

The success in restoring the productive capacity of soil during bush fallow is site dependent and little is known about its progress in some climate regions. Studies aimed at describing and explaining the factors that determine the dynamics of pH, CEC and macronutrients in soils under different ages of bush fallow in tropical semiarid regions are scarce. In addition, considering that most of the processes take place in the topsoil, there is a need to assess events at shallow depths to avoid the dilution effect of deeper layers (Crépin and Johnson, 1993; Govaerts *et al*., 2006). Therefore, it is important to carry out investigations that generate data for the development of guidelines to assist farmers and land use planners at farm level on the one hand, and contribute to sustainable agricultural practices and soil conservation on the other.

The objectives of this study were to quantify acidity, CEC and macronutrients in order to explain the factors determining their dynamics in sandy soils under different bush fallow periods in southern Mozambique.

## 6.2 Materials and methods

### *6.2.1 Study area*

Southern Mozambique comprises three provinces, Inhambane, Gaza and Maputo (Figure 6.1). This study was performed only in Inhambane and Gaza Provinces (21°00'-25°20'S and 31°20'-35°40'E). From east to west there is a gradient in topography, climate and natural vegetation. According to Reddy (1985; 1986), elevation increases from sea level up to 1000 m inland. Along this gradient, mean annual temperature increases from 20℃ to 26℃ and mean annual precipitation decreases from more than 1000 mm to less than 400 mm. The area spans four

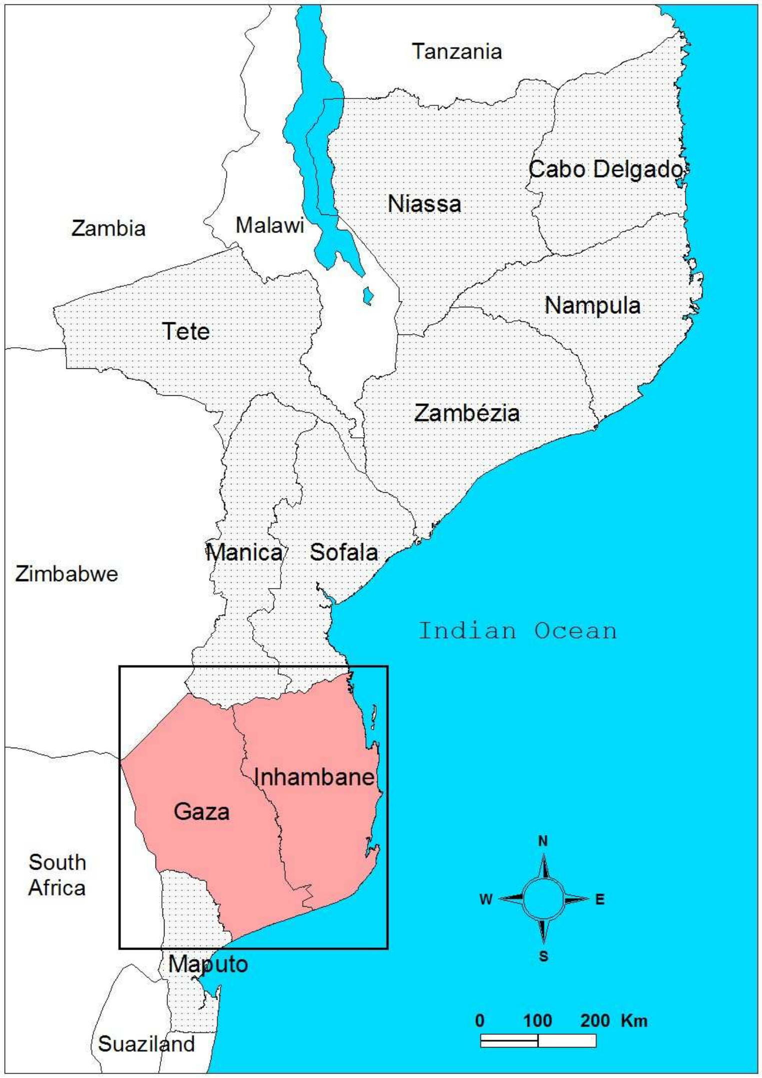


**Figure 6.1** The study area in southern Mozambique.

climatic zones based on moisture index: sub-humid, wet semi-arid, dry semi-arid and arid. The miombo woodland that covers the coastal region is gradually replaced by arid savannah inland (Campbell *et al.*, 1996), with mean annual rainfall of 700 mm as a rough threshold (Frost, 1996).

The area is mainly covered by sand material deposited by marine transgression and regression between the Pliocene and Holocene eras, around 5.4 million to 10 thousand years ago, during the opening of the Indian Ocean and east African continent margin (Salman and Abdula, 1995). The sand was deposited over a Precambric basement and sequence of Karroo and Post-Karroo effusives (Flores 1973).

### *6.2.2 Agroecosystem and site selection*

Based on recommendations by Swift *et al*. (1979) and local studies (Reddy 1986; Geurts and Van den Berg 1998), six agroecosystems (AE 1-6) were identified for this study. Each represents a rainfall region, one of which is transitional. An AE was defined as a region where the three environmental factors that affect yield, namely climate, slope and soil, were similar (Du Preez and Van Zyl, 1998). Three sites were selected in each AE, except in AE5 where an additional site was included, resulting in nineteen sites altogether (Figure 6.2). General climatic and soil characteristics of the

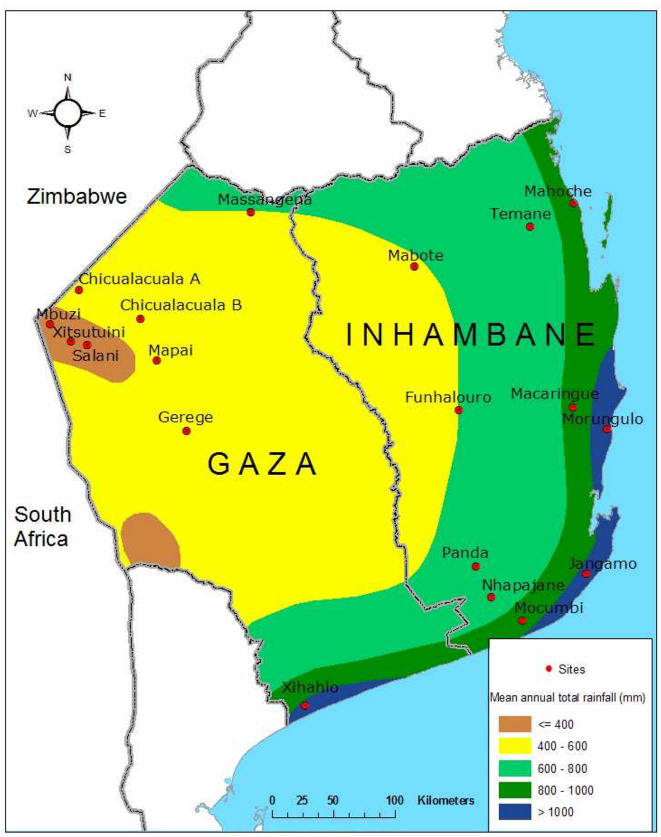


**Figure 6.2** Mean annual rainfall regions of the study area (adapted from FAO, 2000) regarded as agroecosystems and the sites selected for soil sampling.

selected sites are described in Table 6.1. Each site comprised five fields under different land uses, namely cultivated (CT), <5 years fallow (5F), 5-15 years fallow (10F), >15 years fallow (15F) and virgin (VG). In Morungulo (AE1), a VG field was not found. During the selection of fields at every site, care was taken to identify fields with similar slope and apedal soils deeper than 1.0 m within a diameter of 1.0 km. The similarity of the soils was verified by digging a profile pit in the 15F fields and augering the others.

### *6.2.3 Soil sampling and laboratory analysis*

Two sets of three soil samples were collected from the 0-50 mm (L1), 50-100 mm (L2) and 100-200 mm (L3) layers of a field, one for bulk density determination and the other for particle size distribution and some chemical analyses. Each of these samples was a composite of four sub-samples collected within a diameter of 10 m. Samples for bulk density determination were collected with a standard cylinder of 100 $cm^3$, oven-dried at 90℃ for 48 hours and weighed. Those for determination of particle size distribution (pipette), pH (KCl), P (Bray-1), Ca, Mg, K and CEC (ammonium acetate) were collected with a 70 mm diameter auger, air-dried, sieved and stored to be analysed by Central Agriculture Laboratories in South Africa according to standard methods (The Non-Affiliated Soil Analysis Work Committee 1990).

### *6.2.4 Data processing*

The content of P, K, Ca and Mg in a layer was converted to mass per unit surface area using the bulk density of that layer before Analysis of Variance. As there was variation in numbers of fields between sites and number of sites between agroecosystems, Analysis of Variance for land uses in each agroecosystem and in similar land uses across agroecosystems was carried out using the General Linear Model (GLM) procedure ($P<0.05$) in NCSS 2000 (Hintze 1998). Fisher's least significant differences (LSDs, $P<0.05$) were calculated where relevant.

**Table 6.1** Geographical location, climatic and general soil characteristics of the sampling sites

| Agroecosystem | Sampled Sites | Lat. | Long. | Altitude* (m) | MAR** (mm) | MAT*** (°C) | Broad climatic zone according to moisture index | Soil units** FAO 1988 | Soil units** USDA 1992 | Landform**** |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | Ustic Quartzipsamments | Coastal dunes, |
| 1 | Morungulo | 23° 13' | 35°28' | | | 20-23 | Sub humid | Haplic Arenosols | | holocenic sand |
| | Jangamo | 24° 16' | 35°19'' | | > 1000 | | | Arenosols | Psamments | |
| | Xihahlo | 25° 12' | 33°20'' | | | | | Ferralic Arenosols | Ustoxic Quartzi-psamments | Inner dunes |
| 2 | Mahoche | 21° 35' | 35°13' | | | | Wet semi-arid | Arenosols | Psamments | |
| | Macaringue | 23°04' | 35°13'' | | 800 - 1000 | | | | | Sandy plain |
| | Mocumbi | 24°37' | 34°51'' | | | | | | | |
| 3 | Temane | 21°45' | 34°54'' | < 200 | | | | Arenosols | Psamments | |
| | Panda | 24°13' | 34°31'' | | 600 - 800 | 20-26 | | | | Inner dunes |
| | Nhapajane | 24°27' | 34°38'' | | | | | | | |
| | | | | | | | | | | Sandy and redish |
| 4 | Massangena | 21°39' | 32°53'' | | | | | Chromic Cambisols | Typic Ustochrepts | colluvium |
| (Transitional) | Mabote | 22°03' | 34°04'' | | (400-800) | | Dry semi-arid | | | |
| | Funhalouro | 23°05'' | 34°23'' | | | | | | | |
| 5 | Chicualacuala A | 22°13'' | 31°38'' | | | | | | | |
| | Chicualacula B | 22°25' | 32°05'' | | | | | Arenosols | Psamments | Sandy plain |
| | Mapai | 22°44'' | 32°12'' | | 400-600 | | | | | |
| | Gerege | 23°14'' | 32°25'' | 200 – 500 | | | | | | |
| 6 | Mbúzi | 22°28'' | 31°25'' | | | 23-26 | Arid | | | |
| | Xitsutsuini | 22°35'' | 31°34'' | | < 600 | | | Calcaric Cambisols | Typic Ustochrepts | Shallow soils over |
| | Salani | 22°37' | 31°41'' | | | | | | | calcarium rocks |

Source: * Reddy (1985); **MAR – Mean annual rainfall (FAO, 2000); *** MAT – Mean annual temperature (Reddy, 1985); *** INIA (1994; 1995).

## 6.3 Results

### *6.3.1 Bulk density and soil texture*

The soil texture varied between sand in AE1, AE2, AE3 and AE4 to loamy sand in AE5 and AE6, with a weighted mean particle size distribution of 88-93% sand, 2-8% silt and 4-6% clay to 200 mm depth (Table 6.2). From coast to inland the sand content tended to decrease and the silt content to increase, whereas there was no obvious trend in the clay content. Weighted mean bulk density to 200 mm depth varied from 1.27 g $cm^{-3}$ in AE3 to 1.54 g $cm^{-3}$ in AE5.

**Table 6.2** Bulk density and particle size distribution of soil in the agroecosystems

| Agroecosystem | Depth (mm) | Bulk density (g $cm^{-3}$) | Particle size distribution (%) | | | Textural class |
|---|---|---|---|---|---|---|
| | | | Sand | Silt | Clay | |
| 1 | 0-50 | 1.28 | 92 | 4 | 4 | Sand |
| | 50-100 | 1.35 | 92 | 4 | 4 | |
| | 100-200 | 1.37 | 94 | 2 | 4 | |
| 2 | 0-50 | 1.34 | 93 | 2 | 5 | |
| | 50-100 | 1.44 | 93 | 2 | 5 | |
| | 100-200 | 1.44 | 92 | 2 | 5 | |
| 3 | 0-50 | 1.27 | 91 | 4 | 5 | |
| | 50-100 | 1.29 | 91 | 4 | 5 | |
| | 100-200 | 1.25 | 91 | 3 | 5 | |
| 4 | 0-50 | 1.45 | 91 | 4 | 6 | |
| | 50-100 | 1.48 | 90 | 4 | 6 | |
| | 100-200 | 1.49 | 90 | 4 | 6 | |
| 5 | 0-50 | 1.55 | 88 | 6 | 6 | Loamy sand |
| | 50-100 | 1.54 | 89 | 6 | 5 | |
| | 100-200 | 1.54 | 89 | 7 | 5 | |
| 6 | 0-50 | 1.50 | 89 | 7 | 4 | |
| | 50-100 | 1.52 | 88 | 8 | 5 | |
| | 100-200 | 1.51 | 87 | 8 | 5 | |

### *6.3.2 Cation exchange capacity*

Significant differences in CEC among fields were only found in L1 of AE2 and AE5 (Figure 6.3). In AE2 CEC decreased gradually from VG to CT and then to either 5F or 10F fields, followed by an increase to 15F fields. The CEC in AE5 increased with conversion from VG to CT, followed by a gradual decrease without significant differences between fallow fields.

The CEC of all layers for similar land uses showed significant differences across agroecosystems (Figure 6.4). In general CEC decreased from the coastal AE1 to AE2 (in VG, 5F, and 10F fields) or AE3 (in CT and 15F fields) and increased then to reach its highest values in the most inland AE6. The CECs of AE6 were significantly higher than those of the other five agroecosystems, except for L1 in the VG and CT fields of AE1. In many instances the CECs of AE1 also significantly exceeded those of AE2, AE3, AE4 and AE5. No significant differences in CEC were found among AE2, AE3, AE4 and AE5.

### *6.3.3 pH*

In all six agroecosystems the lowest pH values were recorded in the VG fields, although these were not always significantly different from those in the other fields (Figure 6.5). Without exception, slash and burn caused higher pH values in the CT fields. As bush fallow proceeded it tended to decrease pH in AE1 to AE4 but not in AE5 and AE6. In AE1, AE2, AE3 and AE4, where significant decreases were noted, it was mainly the VG fields and to a lesser extent the 15F fields that had lower pH values than the CT, 5F and 10F fields.

Moving inland across agroecosystems, pH of fields under similar land use tended to increase from AE1 to AE2 or AE3 and decreased clearly to AE4 or AE5, followed by a sharp increase to AE6 (Figure 6.6). This trend applied to all three layers. The significant differences noted were therefore mainly among AE3 and AE4 with low pH values on the one hand, and AE1 and AE6 with high pH values on the other.

### *6.3.4 Phosphorus*

In AE1 to AE5, the P in all three layers generally increased from the VG to CT or 5F fields and then decreased as bush fallow proceeded (Figure 6.7). However, significant differences in P among fields were only recorded in L1 of AE2 or AE4 and L1 to L3 of AE5. Differences in these instances were mainly restricted to the VG fields having lower P than CT and 5F fields. A quite different pattern evolved in AE6. In this agroecosystem the highest P in L1 was measured in the VG fields, followed by that in CT, 5F, 10F and 15F fields. This pattern was repeated with a few exceptions in L3.

The P in fields under similar land uses across agroecosystems tended to increase gradually from the coastal AE1 to AE5 and then sharply increased to the most inland AE6 (Figure 6.8). This implied that regardless of land use, P in all three layers of AE6 was significantly higher than that of the other five agroecosystems. The measured P in the CT, 5F and 15F fields also differed significantly in some instances between AE1 to AE5, especially in L1 and to a lesser extent in L2 and L3. No significant differences in P between these five agroecosystems were noted for the VG and 10F fields.

### *6.3.5 Calcium*

The general trend in AE1 to AE5 was that Ca increased from the VG to CT fields and then remained almost constant or decreased as bush fallow progressed (Figure 6.9). Significant differences in Ca between fields were only found in L2 and L3 of AE1 and AE2, and all three layers of AE5. In this particular case the Ca content in the CT fields exceeded that in the VG fields and also in some of the fallow fields.

Generally, the Ca of fields under similar land uses across agroecosystems tended to increase from AE1 to AE2 and then decreased to AE5, followed by a huge increase to AE6 (Figure 6.10). Thus Ca in all AE6 fields exceeded that in the other five agroecosystems significantly. The Ca in L1 and L2 of 10F fields in AE2 was also larger than that in AE4 and AE5.

### *6.3.6 Magnesium*

No general trend emerged among the fields for Mg, despite some significant differences being noted in AE2 to AE6 (Figure 6.11). However, it may be worthwhile mentioning that in AE2, AE4 and AE5, Mg tended to be lower in the VG fields than in any of the other fields.

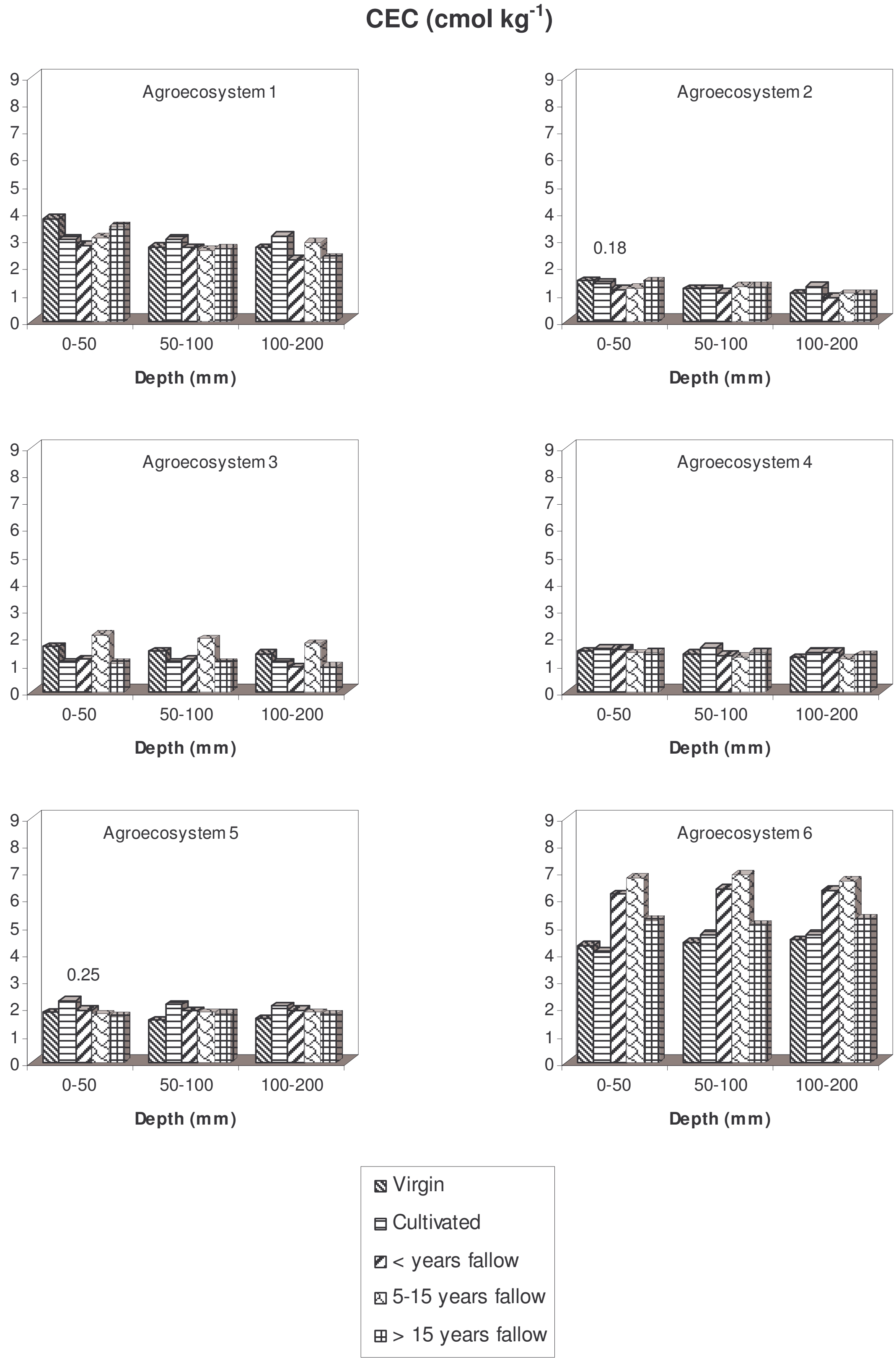


**Figure 6.3** CEC in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

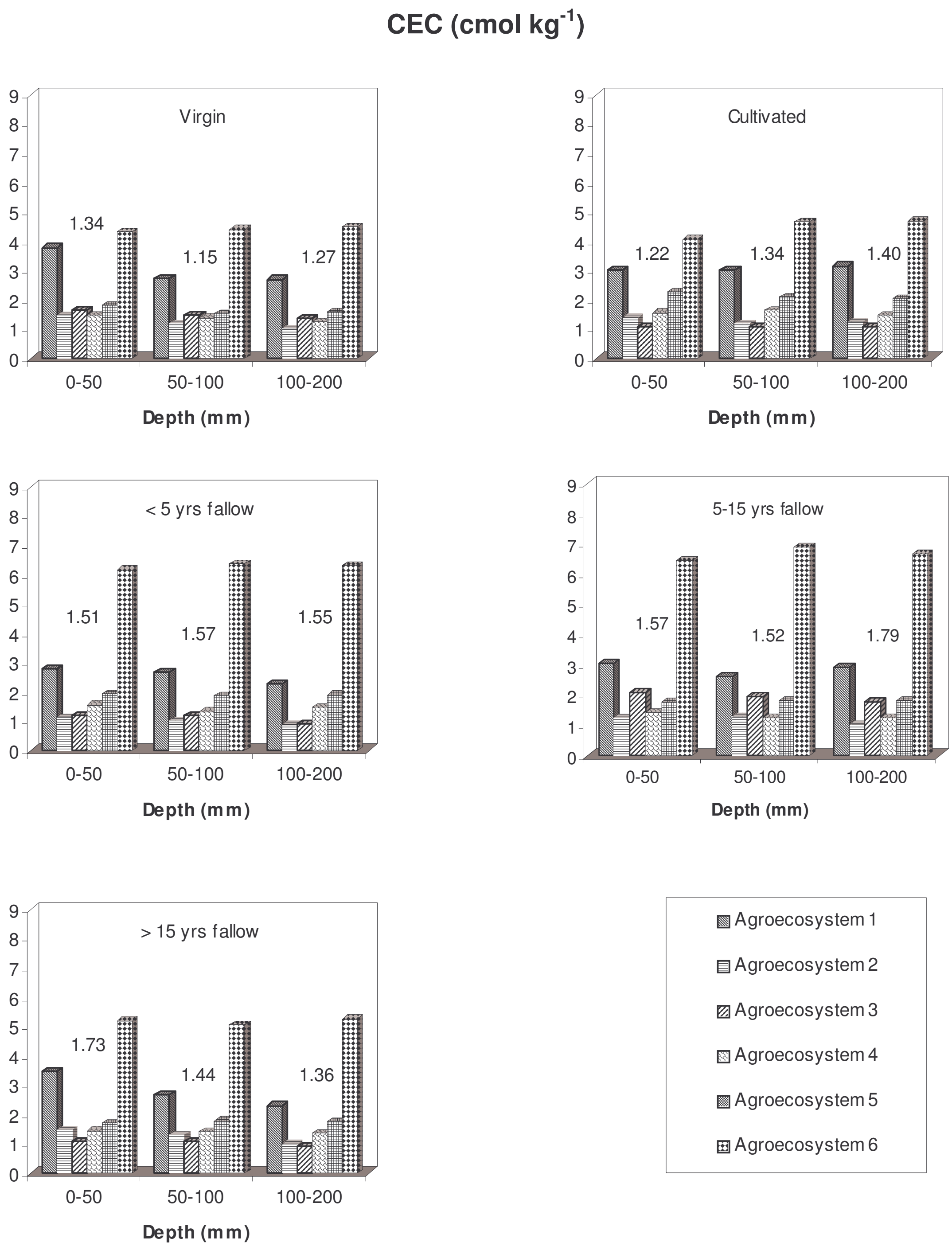


**Figure 6.4** CEC in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

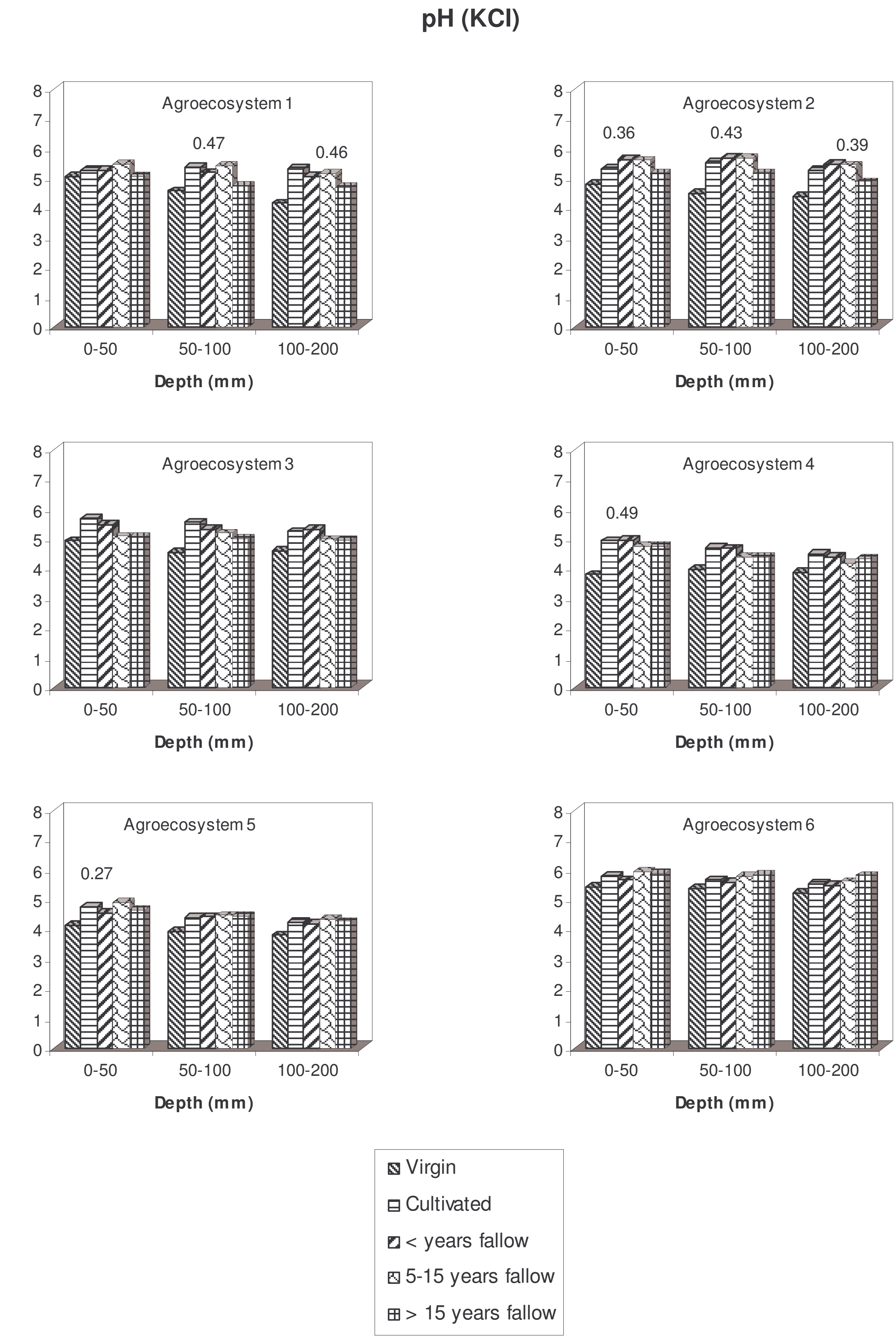


**Figure 6.5** pH (KCl) in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

**pH (KCl)**

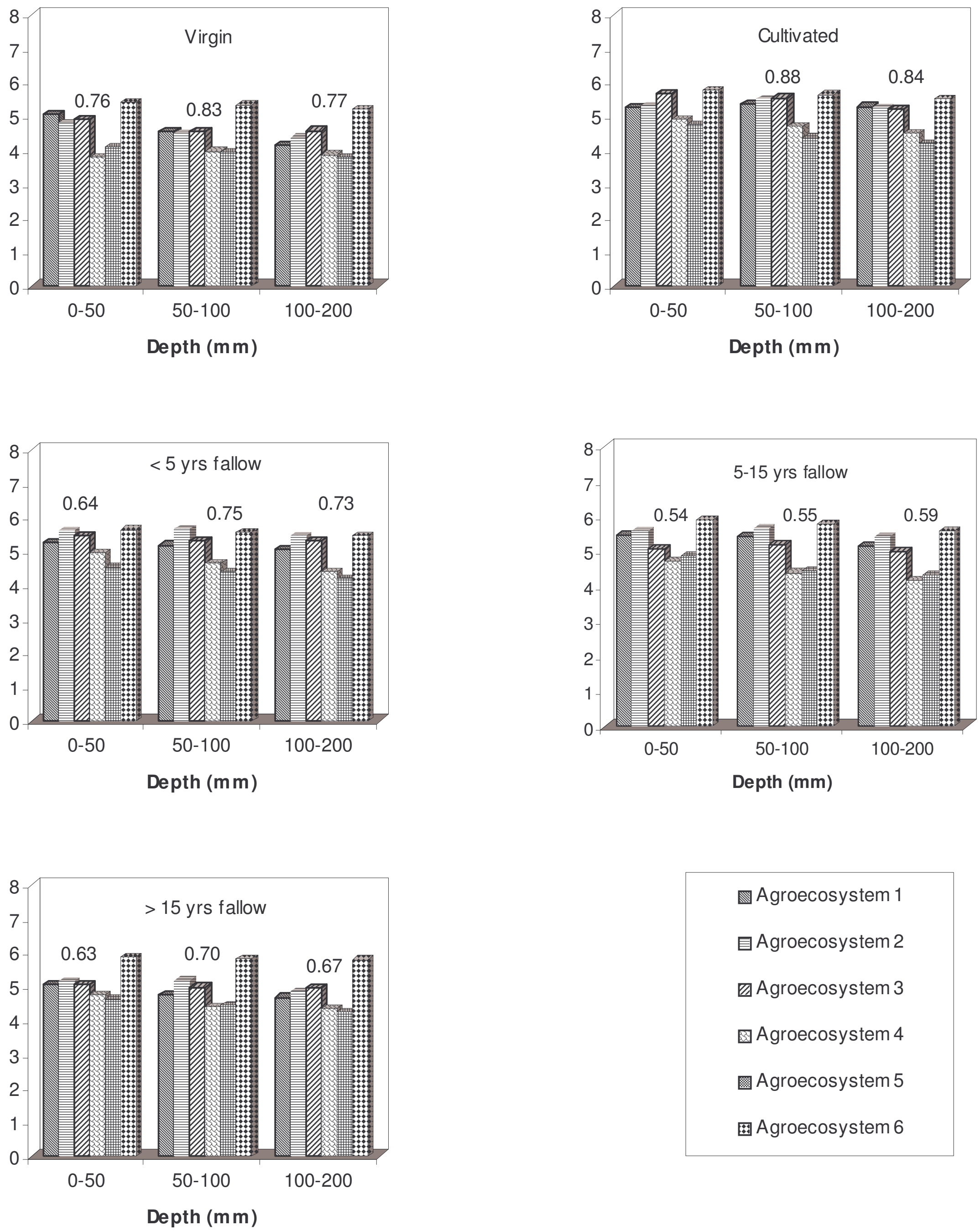


**Figure 6.6** pH (KCl) in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

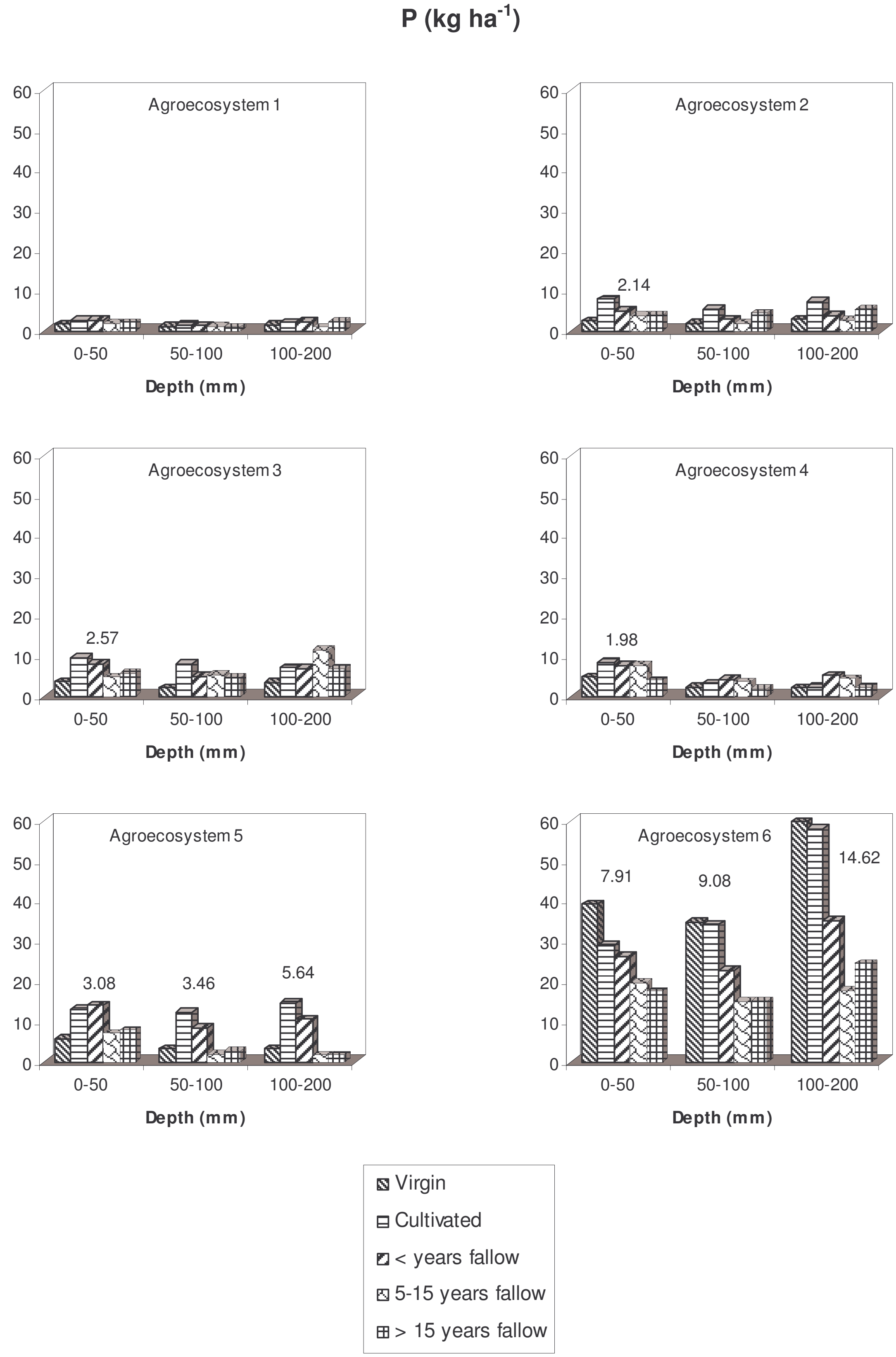


**Figure 6.7** Phosphorus in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

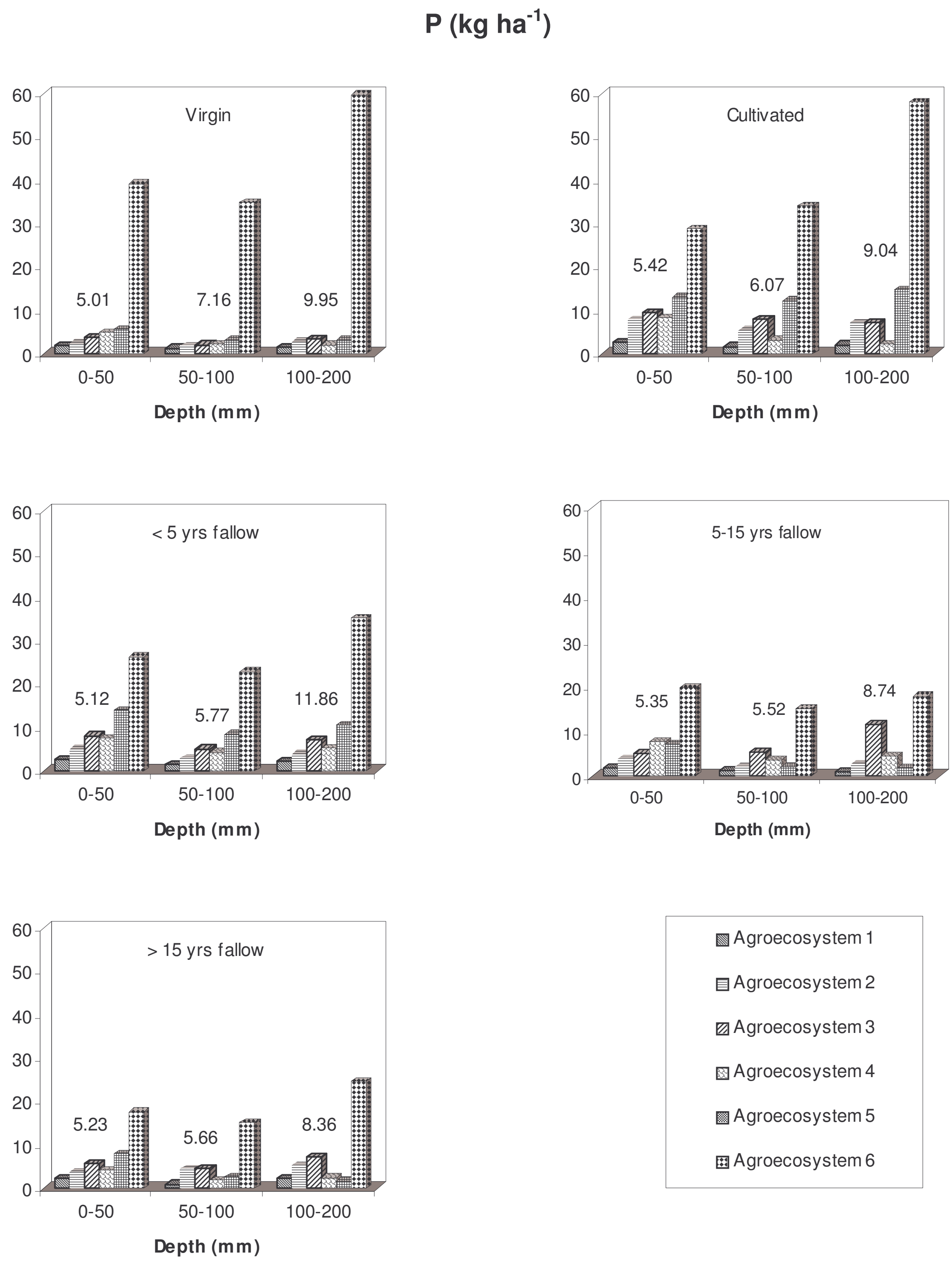


**Figure 6.8** Phosphorus in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

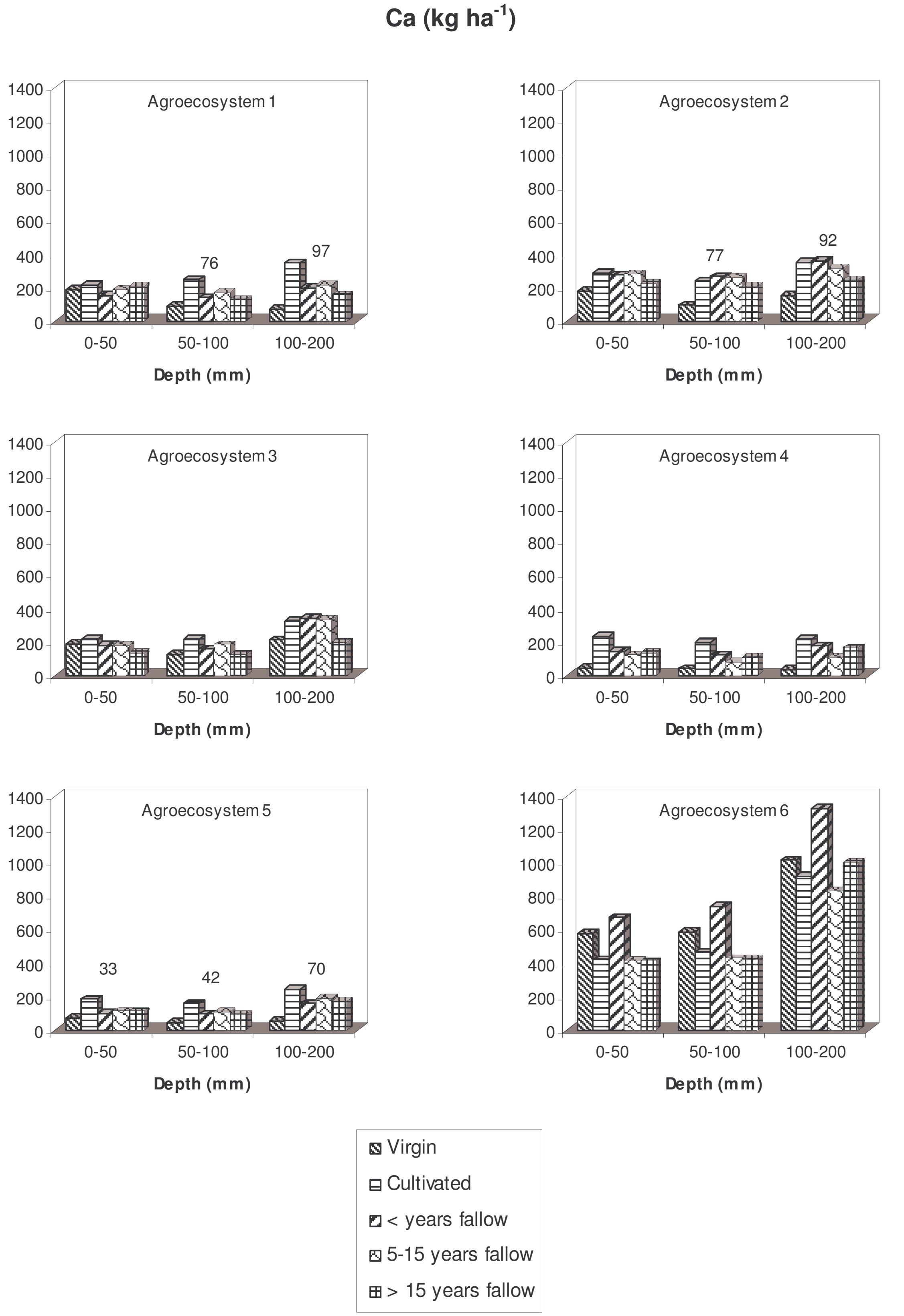


**Figure 6.9** Calcium in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

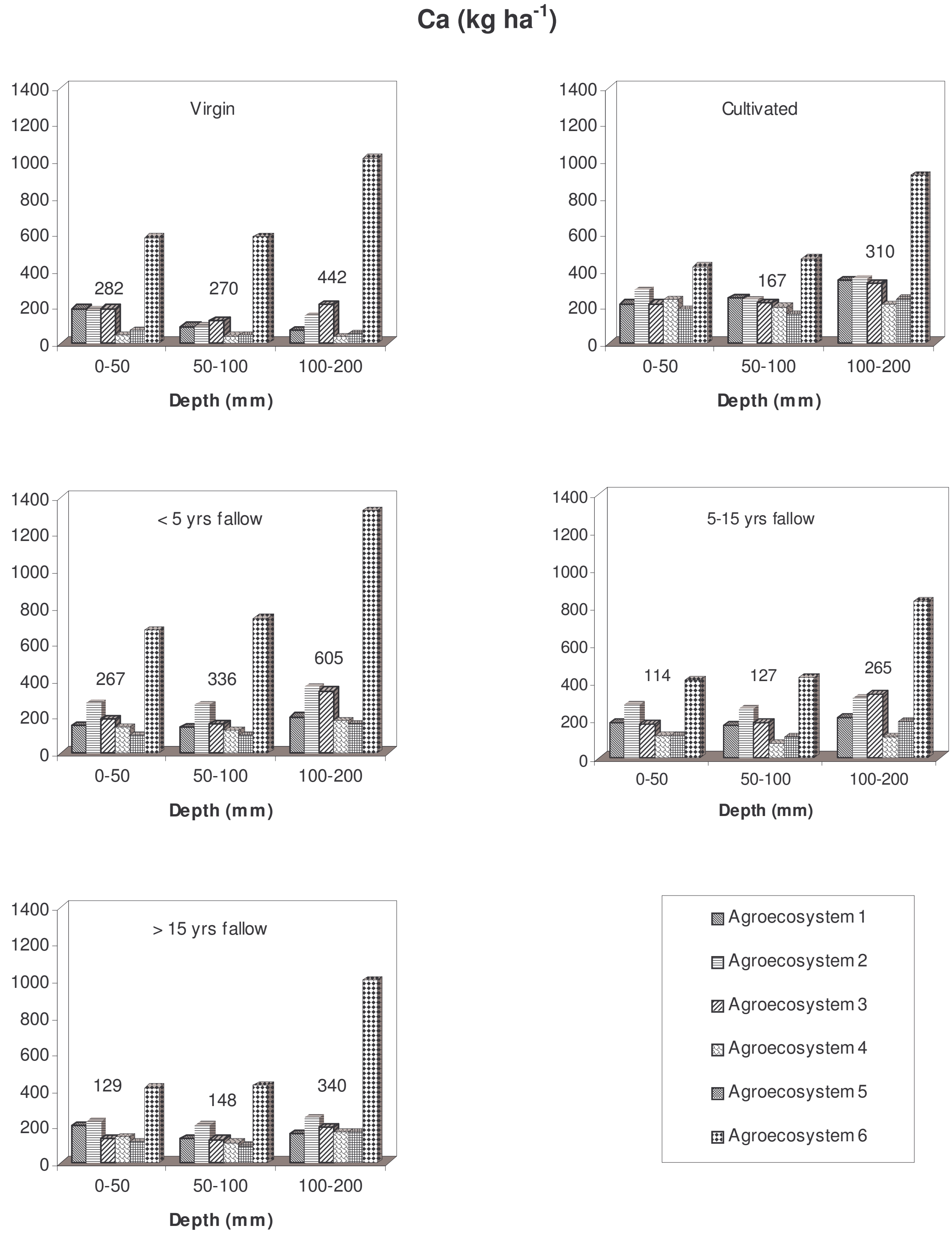


**Figure 6.10** Calcium in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

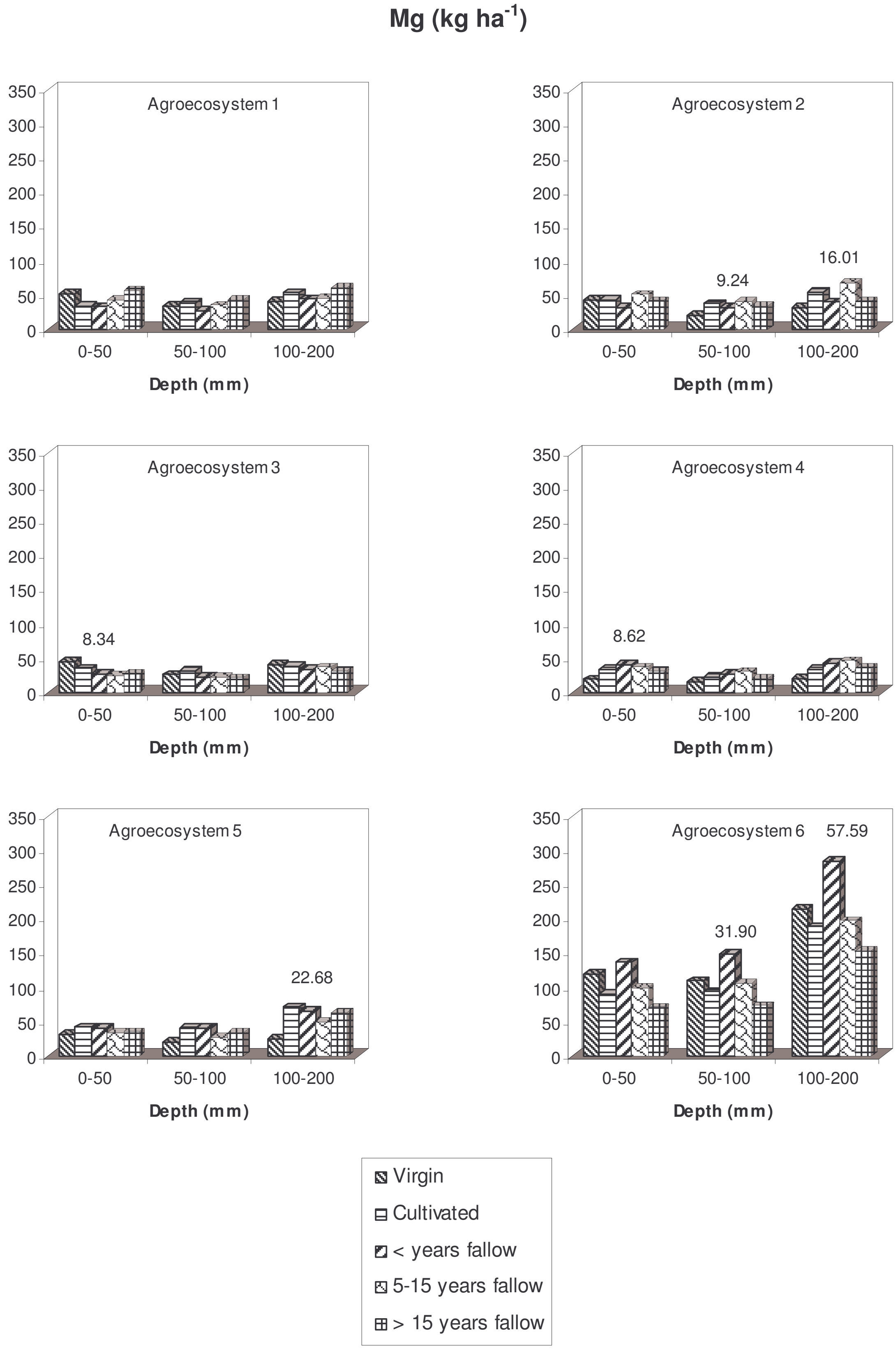


**Figure 6.11** Magnesium in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

Mg (kg ha$^{-1}$)

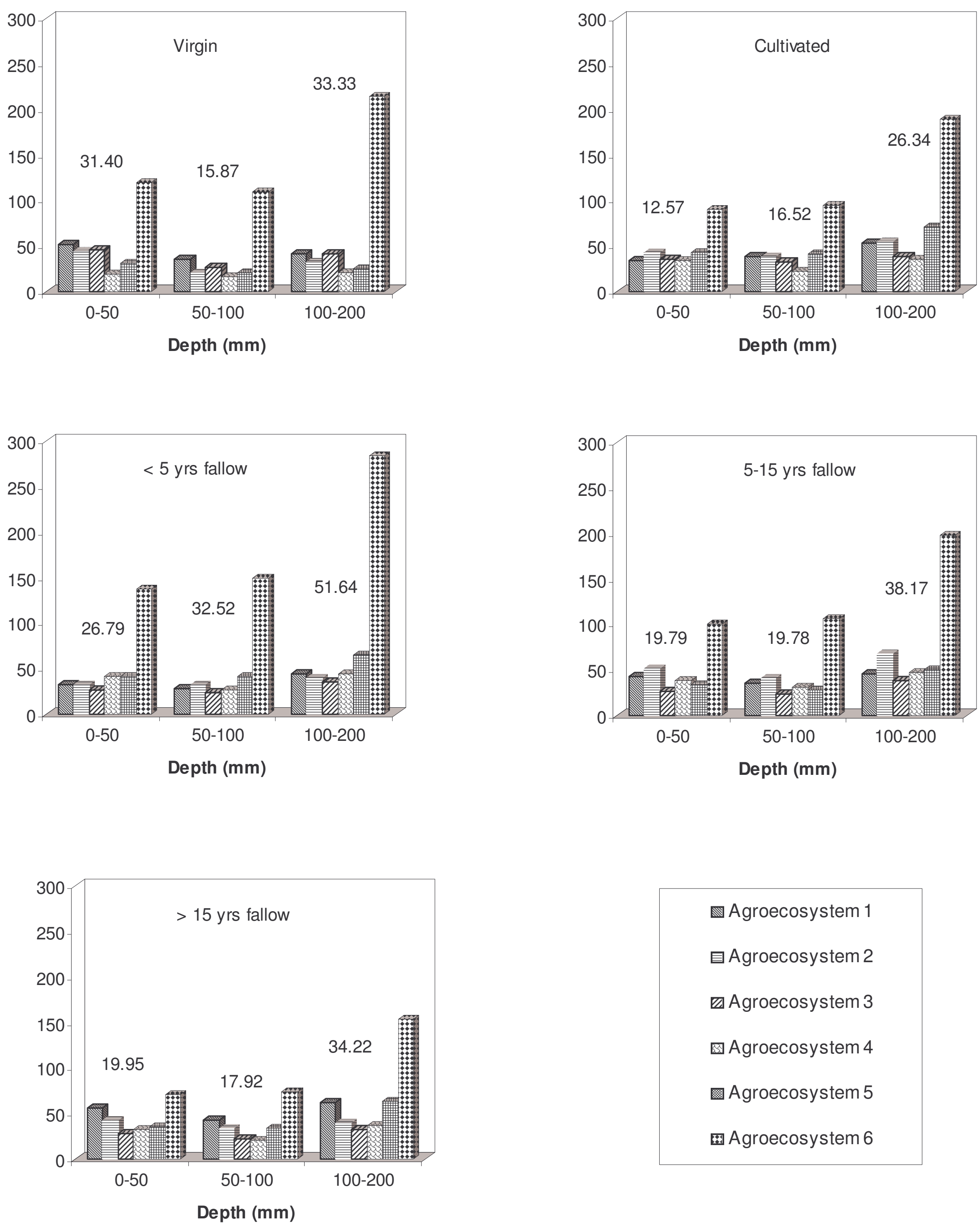


**Figure 6.12** Magnesium in three soil layers (0-50, 50-100, 100-200 mm) under different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

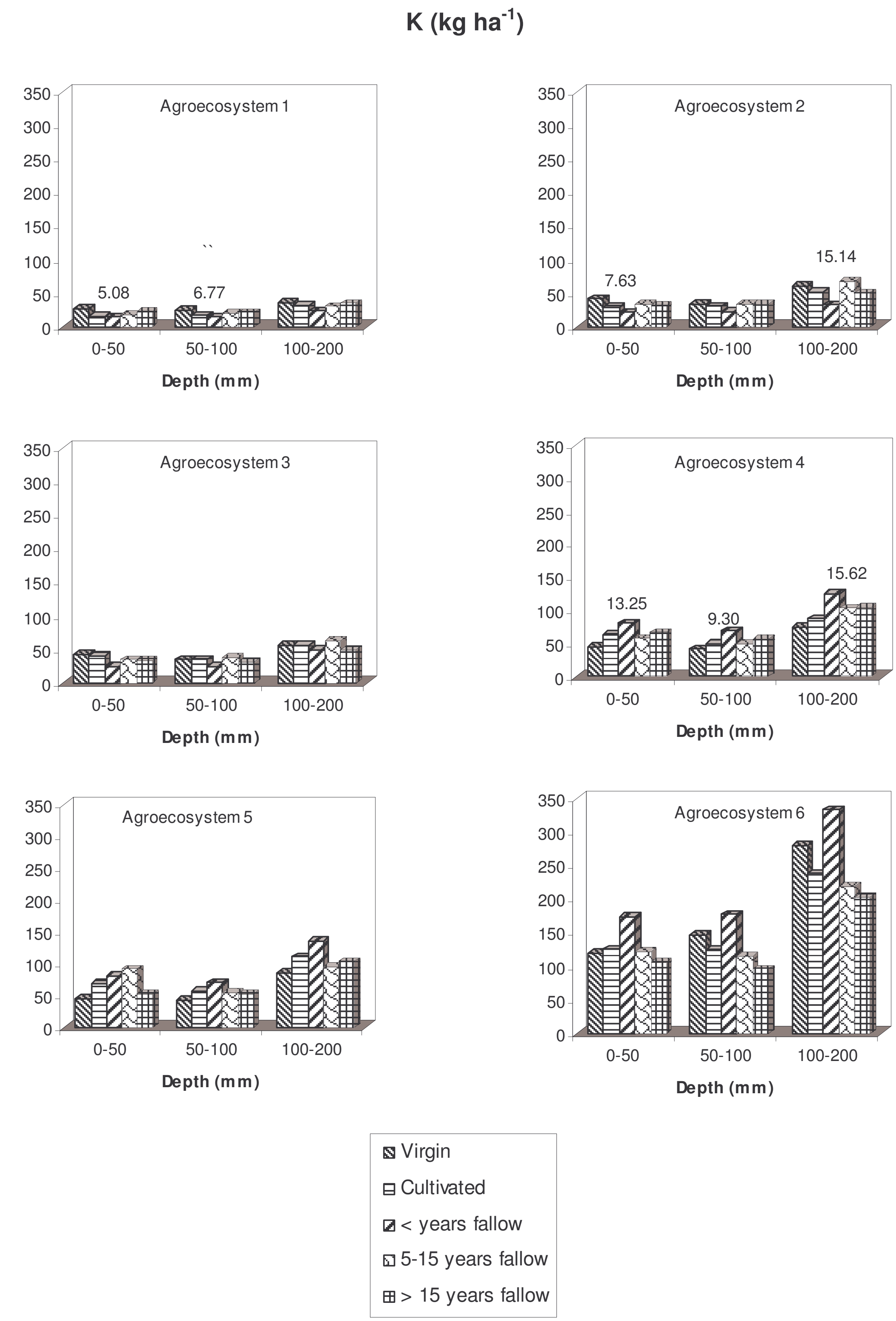


**Figure 6.13** Potassium in three soil layers (0-50, 50-100, 100-200 mm) in each agroecosystem (AE1-AE6) under different land uses. Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

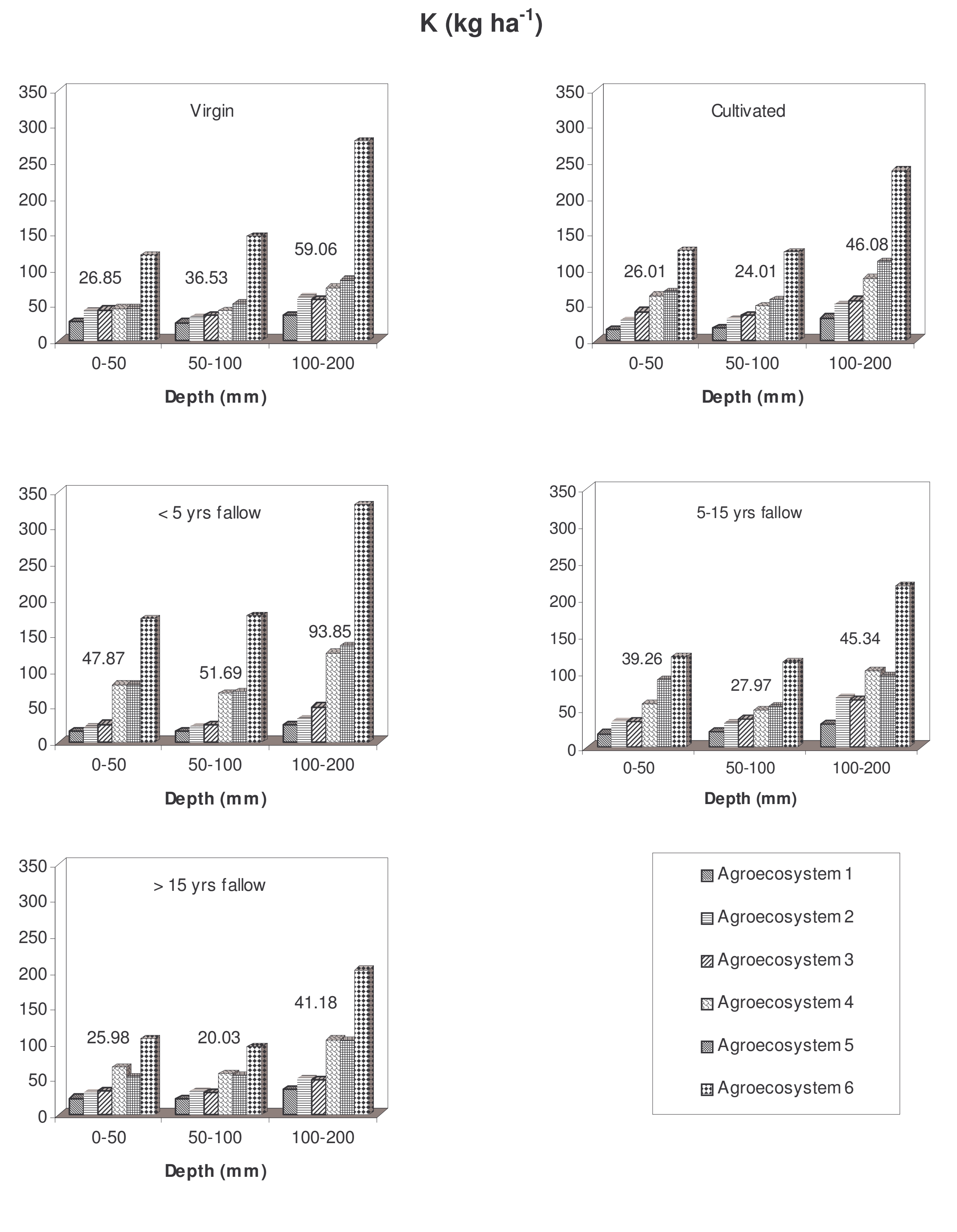


**Figure 6.14** Potassium in three soil layers (0-50, 50-100, 100-200 mm) under Different land uses across agroecosystems (AE1-AE6). Significant differences are indicated by Fischer's LSD $_{(\alpha = 0.05)}$ values above graphs

With a few exceptions, the Mg in fields under similar land use was similar across AE1 to AE5 but significantly higher in AE6 (Figure 6.12)

### *6.3.7 Potassium*

Significant differences in K among fields were only recorded in AE1, AE2 and AE4 but two general trends evolved (Figure 6.13). In AE1 to AE3, K decreased with the conversion of VG to CT fields and reached a minimum in the 5F fields before increasing in the 10F and 15F fields. This trend was reversed in AE4 and AE6 since K increased with the conversion of VG to CT fields and reached a maximum in the 5F or 10F fields before decreasing in the 10F or 15F fields.

Among fields under similar land use across agroecosystems, K increased clearly from AE1 to AE5 resulting in several significant differences between these agroecosystems (Figure 6.14). However, the K in AE1 to AE5 fields was significantly lower than that in corresponding fields of AE6 due to the sharp increase from AE5.

## 6.4 Discussion

The discussion below is based on the climate characteristics of the agroecosystems combined with the common farming practice of slash and burn, land preparation with hand hoe and vegetation succession during the bush fallow period.

### *6.4.1 Cation exchange capacity*

The trend in CEC for similar land uses across agroecosystems corresponded to a certain extent with that of organic carbon content (Chapter 5), namely a decline from the coastal AE1 to AE2 or AE3 and then an increase to the most inland AE6. The high values in AE1 could have derived from greater vegetation biomass production (Chapter 3) leading to more organic matter residues, which ultimately had a positive influence on the CEC of soil (Stevenson and Cole, 1999). Moving inland, the coastal coarse sandy dunes are gradually replaced by sandy plains (Table 6.1) with more silt plus clay (Table 6.2), which increases the CEC. This increase can also be related to a lower decomposition rate of organic matter due to an increase in dryness (Chapter 4).

### *6.4.2 pH*

The annual rainfall of more than 1000 mm in the coastal areas probably caused leaching of basic cations to deeper soil layers of AE1 and hence land use had no effect on pH in the 0-50 mm layer. As rainfall declines inland, less leaching of basic cations can be expected in AE2 and the effect of land use resulted in pH differences in all three layers. In AE3 and AE6 no differences in pH between land uses were detected. Both agroecosystems have a deep calcareous layer and their pH values tended to be higher than those of the adjoining agroecosystems AE4 and AE5, which had the lowest pH values. This suggests that basic cations have been taken up by the roots of existing vegetation from deeper soil layers and released in the shallower soil layers through decomposing litter. In AE4 and AE5 limited leaching due to low rainfall can explain why differences in pH were only found in the top layer.

Considering that the study was carried out on sandy soils, the lack of pH differences in the top layer between CT fields can be ascribed to rapid mineralisation of organic matter due to mixing by hand hoeing during land preparation. Differences in pH of fields under similar land use across agroecosystems can be attributed inter alia to the composition of the vegetation (Chapter 3). The coastal agroecosystems (AE1, AE2 and AE3) have higher numbers of plant species and biomass in each land use compared with the agroecosystems further inland (AE4, AE5 and AE6). However as mentioned, AE6 has a calcareous layer which also contributed to its higher pH values. Although an outcrop of limestone was visible in AE3, this agroecosystem differed from AE6 in vegetation composition, which seemed to overshadow the effect of the calcareous layer on its pH. Nutrient uptake by fallow vegetation (Juo and Manu, 1996) is usually associated with $H^+$ release through decomposition of plant litter and root exudation, which leads to a decline in pH (Vaughan and Ord 1985; Steiner 1996). This may explain the lower pH values in AE3 than AE6, the trend in AE3 for CT fields to have higher pH values than 5F fields and the pH decrease with increasing length of fallow period. The increase in pH observed with conversion of VG to CT fields followed by a decrease during bush fallow results from the ash that remains after burning of vegetation during land preparation. This ash increases the concentration of basic cations, especially in the 0-50 mm soil layer. These cations are gradually reduced through uptake by crops during the cropping phase, succession vegetation in fallow period and possibly leaching. This leads to a decline in pH, which is favoured by the release of acidic compounds from decomposing plant litter.

### *6.4.3 Phosphorus*

The increase in P from VG to CT fields followed by a decrease with fallow period is in accordance with the trend for pH. As explained for pH, this is due to slash and burn with land preparation and vegetation growth during fallow. The decrease in P with increased acidity results in AE1 to AE5 with pH values of 4.5-5.0, possibly from precipitation of P due to higher solubility of Fe and Al (INIA 1994; 1995; Havlin *et al.* 1999). This kind of precipitation in AE6, with pH values of 5.2-6.79, is probably countered by a higher content of basic cations. The pH range of AE6 is favourable to P availability (Havlin *et al.*, 1999; Sharpley, 2000). In addition, according to Prasad and Power (1997) the climate change from the coast to inland favours the mineralisation of P and this could also explain the gradual increase in P from the coastal and wetter to inland and drier agroecosystems. Although this study was conducted on sandy soils, the increase in finer separates and organic matter towards inland and drier agroecosystems should have contributed to higher contents of P. Organic matter is known to form complexes with Ca, resulting in more available P. The presence of high Mg probably also contributed, as it prevents adsorption of phosphate onto calcite.

### *6.4.4 Calcium*

The increase in Ca with conversion from VG to CT fields resulted from the ash produced with slash and burn for land preparation. During fallow the demand for Ca by vegetation increases and hence the decrease in Ca in soil. The adjoining agroecosystems AE4 and AE5 interchangeably had the lowest Ca content in soil (3.8 to 5.0 kg $ha^{-1}$), which suggests that the *A. Johnsonii* that dominated the vegetation may have high Ca consumption, with little being returned to the soil. Both agroecosystems had stronger acidity than the others (pH 3.80-4.96) and according to authors cited by Camberato and Pan (2000), acidic ions may displace Ca from the exchangeable complex, enabling leaching to occur. In contrast, the higher Ca contents in the drier AE6 possibly resulted from the underlying calcareous layer, as previously stated, and that may have obscured the effects of land use since the VG and 5F fields had the highest values. The high Ca in VG fields could be attributed to root uptake from deeper layers and release in shallower layers when the litter decomposed. In the 5F fields with poor vegetation cover, however, the high Ca could be attributable to upward movement of Ca during evaporation, since leaching should be limited in the very dry AE6. Although Ca is relatively immobile, it can be leached from soils (Camberato and Pan, 2000) depending on the frequency and intensity of rainfall (Prasad and Power, 1997). Taking into account the fact that from AE1 to AE6 the frequency, intensity and total amount of rainfall decreases and the CEC of soils and the presence of a herbaceous layer increases (Chapter 3), then a decline in leaching of Ca also explain the increasing gradient observed in that direction.

### *6.4.5 Magnesium*

The lack of differences between the top layer of fields in AE1 and all layers in AE2 can be explained by the fact that that these agroecosystems are characterised by heavy rainfall, since although Mg is regarded as relatively immobile, it can be lost by leaching (Camberato and Pan, 2000). Moving towards the inland and drier agroecosystems rainfall intensity decreases, allowing the effect of land use to have more impact on the Mg content of soil. This resulted in differences only in the top layers of AE3 and AE4. However, in AE4 and AE5 rainfall does not explain the tendency for an increase in Mg from VG to 5F fields followed by a decrease with age of fallow. Both agroecosystems were dominated by similar tree and herbaceous species in the VG and 5F fields and had lower pH values and Mg contents, which suggests that large amounts of Mg were leached, probably because acidifying ions released from decomposing vegetation residues displaced Mg (Camberato and Pan, 2000). The general trend of an increase in Mg content in CT and 5F fields from the coastal and wetter to the inland and drier agroecosystems, when compared with the decrease found in 10F and 15F fields, suggests that the change in vegetation species when moving inland determines the Mg content in these well vegetated fields, whereas in bare and poorly vegetated fields it is determined by rainfall. However, AE6 always had a higher Mg content, which could be attributed to the higher presence of grass compared with trees, since grass can greatly reduce leaching of Mg (Prasad and Power, 1997).

### *6.4.6 Potassium*

Potassium is relatively immobile (Camberato and Pan, 2000) but in sandy soils it is susceptible to leaching (Havlin *et al*. 1999; Sparks, 2000). The lower K content in the coastal agroecosystems with coarse sandy soils can therefore be ascribed to the characteristic heavy rainfall in these. This may also explain the decline in K from VG to CT and 5F fields, followed by a gradual increase in 10F and 15F fields of AE1 and AE2. A similar trend was observed in AE3 although the differences were not significant due to high variability. Moving inland rainfall declines, whereas silt plus clay, CEC and coverage by herbaceous vegetation increase. This may explain the gradual increase in K in all land uses. In addition, the pH in AE6 is within the range 5.5 to 7.0, where K fixation is low according to Martin *et al*. 1946 (cited by Sparks, 2000). On the other hand Havlin *et al*. (1999) reported that the high content of K found in soils of drier regions can result from concentration through evaporation. These factors certainly contributed to high K contents in AE6. Considering that AE4 is a transitional agroecosystem in terms of rainfall and vegetation (Chapter 3), the higher K content in the 5F fields (dominated by herbaceous layer) and the lower K content in the VG fields (dominated by tree layer) suggest that vegetation also played a decisive role in the amount of K in soil.

## 6.5 Conclusions

The study area is covered by sandy soils and moving from the coast to inland there is a decrease in mean annual rainfall, an increase in mean annual temperature and a slight replacement of coarser by finer soil particles. Sandy soils have a low capacity to hold cations, which in most cases leads to leaching of basic cations and results in low pH values. The pH values observed in the study varied between 3.80 and 5.91 depending on the agroecosystem and land use. In general pH increased from the coastal and wetter AE1 to inland and drier AE6, with the exception of AE4 and AE5, which had lower values. Consequently, in the same direction there was an increase in P and K contents. In AE1 possible leaching obscured significant differences ($P<0.05$) between land uses concerning measured parameters in the 0-50 mm layer except for K, which is relatively immobile. The lower values of pH in AE4 and AE5 resulted in lower contents of P in AE4 and lower contents of Ca and Mg in both agroecosystems. These two agroecosystems are covered by the same vegetation, which suggests that this vegetation should have been the determining factor. The decrease in pH from CT to 15F fields can be attributed to a gradual decrease in the basic cations released on the soil surface by the ash produced during slash and burn. These cations are either leached downward in the soil profile or absorbed by the growing vegetation. The latter is associated with the release of acidifying compounds from roots and decomposing residues, resulting in lower pH. This may have caused the decreasing trends in P and Ca as the period of bush fallow increased. However, it seems that Mg and especially K were more related to CEC, except in AE1. Therefore, pH and CEC influence the availability of Ca, Mg and K. The increase in the silt plus clay fraction from AE4 to AE6 resulted in an increase in CEC, P, Ca, Mg and K.

# CHAPTER 7

## Summary, synthesis and recommendations

### 7.1 Summary

Bush fallow under shifting cultivation is the most common practised subsistence farming system in large areas of southern Mozambique where the population is sparsely distributed. The farmers slash and burn natural or regenerated vegetation for cropping. After 3-5 years of cropping they abandon the agricultural land to bush fallow because of a decline in soil fertility that result in low yields. During bush fallow vegetation succession species establish that are responsible for soil fertility restoration. Therefore, a better understanding of the composition of bush fallow vegetation and factors affecting the decay of its litter, as well as the dynamics of OM, acidity, CEC and macronutrients in bush fallow fields is essential to improve soil fertility restoration under shifting cultivation. The results from this study can therefore contribute to a database for technical decisions, e.g. to predict the vulnerability of soils to degradation and designing a cropping system with bush fallow periods that will ensure sufficient fertility recovery of degraded soils in the Inhambane and Gaza provinces.

The region was divided into five main agroecosystems (AE's) represented by rainfall zones (<400 mm = AE6; 400-600 mm = AE5; 600-800 mm = AE3; 800-1000 mm = AE2; and >1000 mm = AE1) and one transitional zone (400-800 mm = AE4). The climate, topography and soil of each AE were for practical purposes homogeneous. In each agroecosystem, five land uses (virgin, cultivated, < 5 years fallow, 5-10 years fallow and >15 years fallow) were identified. Descriptions and comparisons of vegetation were performed between the land uses within agroecosystems and similar land uses across all the agroecosystems, except in cultivated land. *Brachystegia spiciformis* has high ecological importance and its leaves represent a substantial fraction of the total aboveground in bush fallow fields where annual rainfall exceeds 600 mm in the study area, hence the effect of natural factors (soil water content and soil temperature) on decomposition of its leaves was evaluated in recently abandoned agricultural fields cleared of any vegetation (Bare) and in more than 15 year old bush fallow fields (Fallow) of sites in a transect that covered the different agroecosystems. The dynamics of soil fertility indicators (organic C, total N, CEC, pH, P, Ca, Mg and K) were studied by analysing samples collected from three layers (0-50 mm, 50-100 mm and 100-200 mm) at every combination of agroecosystem and land use. In this study answers to the following questions were addressed:

1. **What is the vegetation composition and biomass dynamics in bush fallow lands in southern Mozambique?**

   Natural vegetation in coastal and wetter agroecosystems (rainfall >600 mm) is typical of the miombo woodlands while that in inland and drier agroecosystems (rainfall <600 mm) is typical of arid savanna. A total of 204 species that including N-fixing species, belonging to 141 genera and 50 families divided into tree, shrub and herbaceous layers were identified. The tree layer was only found in virgin fields and in fields abandoned to bush fallow >15 years, whereas shrub and herbaceous layers occurred in all fields. The tree species in bush fallow fields of coastal and wetter agroecosystems (dominated by *B. spiciformis* and *Julbernaldia globiflora*) outnumber those in inland and drier agroecosystems (dominated by *Birchemia discolour* and *Colophospermum mopane*) and have larger diameter that result in greater biomass. In inland and drier agroecosystems the tree biomass in bush fallow fields older than 15 years tends to be higher than in virgin fields due to presence of succession species that differ from the original species. In virgin fields where *A. johnsonii* is the most dominant species, the composition and biomass of trees in bush fallow are totally determined by succession species as the former species once slashed and burned during land preparation does not regenerate. Number of shrubs decreased from coastal and wetter to inland and drier agroecosystems. The herbaceous biomass declined from young to old fallow in coastal and wetter agroecosystems, while the converse was observed in inland and drier agroecosystems. Nitrogen-fixing species tended to occur more in bush fallow fields older than 15 years.

2. **How climatic factors influence carbon loss from *Brachystegia spiciformis* leaf litter in the sandy soils of southern Mozambique?**
   Two patterns of C loss were observed, one in coastal and wetter agroecosystems and the other in inland and drier agroecosystems. In the wetter agroecosystems C loss was faster, whereas in the drier ones it was more sensitive to rainfall pulses. Similarly, C loss was faster in Fallow fields than in Bare fields. During summer, Bare fields reached soil temperatures higher than the estimated upper boundary favourable for C loss from decomposing leaf litter at all sites. A simple dynamic decomposition model that described the C fraction remaining in the litterbags was developed. Coefficients of determination ($R^2$) for the individual experimental units varied between 0.79 and 0.97. The general model for all sites and fields improved the explanation of total variation from 81 to 86% when using measured soil temperature and soil water content as modifiers of the decomposition rate, compared to the single negative exponential model. Root mean square error and systematic bias were 9.7 and 0.5% of initial C, respectively. Decomposition was more strongly affected by soil water content than by soil temperature and explained 75% of the total variation. Thus, among these two factors rainfall is the main driver of C loss from leaf litter in these agroecosystems.

3. **What is the dynamics of organic matter recovery in sandy soils under bush fallow in southern Mozambique?**
   In coastal and wetter agroecosystems there was a declining trend in organic carbon and nitrogen in virgin fields converted for cultivation with a further decrease in the first 5 years of abandonment to bush fallow, which was followed by an increase in older fallows. A different pattern was found in AE4 and AE5 where organic C and N tended to decline gradually even with longer fallow periods. In the severely dry AE6 no clear trend was found. In fields under similar land use across

agroecosystems, organic carbon tended to be lower in AE2. Nitrogen in cultivated fields was lowest in AE6, whereas in <5 years fallow fields it was lowest in AE3. In >15 years bush fallow and virgin fields AE4 had the lowest content. Within agroecosystems, C/N ratio tended to be homogeneous and values among similar fields across agroecosystems tended to be lower in AE2.

4. **What is the dynamics of acidity and macronutrient recovery in sandy soils under bush fallow in southern Mozambique?**
   The pH in all agroecosystems decreased from cultivated to fallow fields, an effect attributable to a gradual decrease in the basic cations released on the soil surface by the ash produced during slash and burn. These cations are either leached downward in the soil profile or absorbed by the growing vegetation. The vegetation releases acidifying compounds from roots and decomposing plant residues. The decline in pH is related to the decreasing trends in P and Ca as the period of bush fallow increases. However, Mg and especially K were more related to CEC, except in AE1. Therefore, pH and CEC influence the availability of Ca, Mg and K. The increase in the silt plus clay fraction from AE4 to AE6 resulted in increased CEC, P, Ca, Mg and K. From the coastal and wetter to inland and drier agroecosystems pH, P and Ca increased, except in AE4 and AE5, which had lower pH and Ca values. The lower values of pH resulted in lower contents of P in AE4 and Ca and Mg in both agroecosystems, which have the same vegetation, suggesting that this should be the determining factor. In AE1 no significant differences ($P<0.05$), were observed between land uses in the 0-50 mm soil layer except for K, possibly due to leaching.

## 7.2 Synthesis

The combination of vegetation composition, leaf litter decomposition pattern and the dynamics of fertility indicators in soil of different land uses within agroecosystems and similar land uses across agroecosystems give an insight of the dynamics of fertility recovery in sandy soils under bush fallow in shifting cultivation.

The study area is mainly covered by sandy soils that have a low capacity to hold cations. Moving from the coast to inland there is a decrease in mean annual rainfall, increase in mean annual temperature, slight replacement of coarser by finer soil particles and change in vegetation composition. These factors combined to land use are determinants for the dynamics of soil fertility in bush fallow fields. The rainfall defines two distinct habitats, which obviously determines the vegetation composition, biomass production and decay rate of its litter. One habitat is of the coastal and wetter agroecosystems that has higher biomass production and higher litter decay rate, and the other is of the inland and drier agroecosystems that has lower biomass production and lower litter decay rate. The latter habitat has slightly more finer soil particles than the former habitat. Consequently, these habitats influence the dynamics of OM, acidity, CEC and macronutrients differently in soils of lands where virgin fields were converted for crop production and later abandoned to bush fallow.

The conversion of virgin into cultivated land through slash and burn results in the release of basic cations on the soil surface that are incorporated into the soil by hand hoeing. This explains the trend of increase of soil pH in cultivated fields. During cropping, the soil is more exposed to radiation and rainfall that induces favourable conditions for mineralization of organic matter. This resulting in a decrease of organic C and N. Considering that the study area is covered by sandy soils, in the coastal and wetter agroecosystems characterised by rainfall of high intensity nutrients can be easily leached downward in the soil profile. Likely, this phenomenon should occur on a smaller scale in inland and drier agroecosystems due to low intensity rainfall associated with high contents of OM and silt plus clay. In addition, the latter two factors contributed to a higher CEC that increased the capacity of soil to hold nutrients.

During the first 5 years after abandonment of cultivated land to bush fallow the soil is still exposed to radiation and rainfall as it is not properly protected by regeneration vegetation. Mineralization of OM therefore proceeds without any substantial residue inputs. As bush fallow progresses the succession vegetation establishes at rates determined by the rainfall pattern that result in higher density of plants in coastal and wetter than in inland and drier agroecosystems. During the growth of vegetation, plant roots and decaying plant litter lead to accumulation of new OM. In fields under similar land use across rainfall zones, the lowest OM content was found in AE2. Agroecosystems with higher and lower rainfall than this had a higher OM accumulation probably due to higher input from biomass in the former and to lower decomposition rate of organic matter in the latter. In inland and drier agroecosystems OM tend to decline even beyond the 5 year fallow period, which requires further investigation though it can be speculated that it may result from the type of succession species.

The decaying plant litter release acidifying compounds that lower soil pH, which in turn reduces P and Ca availability gradually with age of bush fallow. However, pH increased from the coastal and wetter AE1 to the inland and drier AE6, with the exception of AE4 and AE5 that registered lower values. Consequently, in the same direction there was an increase in P and K contents. The reduction in availability of P and Ca as consequence of low pH has larger impact in the high rainfall zones and this impact reduces towards inland and drier agroecosystems, except in AE4 and AE5. These agroecosystems share the same type of vegetation, which differ from the coastal and wetter ones. This suggests that the dominant vegetation species in these agroecosystems induces a lower pH that resulted in lower contents of P in AE4 and lower contents of Ca and Mg in both agroecosystems. The increase of acidity from cultivated to more than 15 year old fallow fields can be attributed to a gradual decrease of basic cations that were released on the soil surface by the ash produced during slash and burn. These cations are either leached downward in the soil profile or absorbed by the growing vegetation. The latter is associated with the release of acidifying compounds from roots and decomposing residues that result in lower pH. Possibly, this caused the decreasing trends of P and Ca as the period of bush fallow increases. It seems however that Mg and especially K were more related to CEC, except in AE1. Therefore, pH and CEC influence the availability of Ca, Mg and K. The increase in the silt plus clay fraction from AE4 to AE6 resulted in an increase of CEC, P, Ca, Mg and K.

Based on the fact that C loss from leaf litter was reasonably well described based on soil water content and soil temperature with the former having a larger effect than the latter, the lower decomposition due to lower rainfall and higher temperature in the inland and drier agroecosystems can explain the observed trend of increase in OM toward these agroecosystems. In regard to land use, in vegetated bush fallow fields leaves loose C faster than in abandoned agricultural fields cleared of any vegetation. Considering that soil water content under field conditions is mainly determined by rainfall implies that rainfall in the main driver of fertility restoration of sandy soils in bush fallow fields under shifting cultivation as it also determine the vegetation composition.

**7.3 Recommendations**

Two distinct habitats were included in this study. Each of them has a particular climate and vegetation. This situation obscured recommendations somewhat. Detailed studies of this nature are therefore justified in each habitat. In both habitats, the results indicated that bush fallow of more than 15 years is required to restore soil fertility of cultivated fields to the same level as in virgin fields. This suggests that with future studies bush fallow fields older than 15 years should be divided into more classes. Information obtained from such classes will enable the establishing of threshold bush fallow periods for full soil fertility recovery.

Considering the higher population density along the coastal belt and surroundings of urban areas, the study shows that in these areas crop production without external sources of nutrients can no longer be practical as there is not enough land to bush fallow for such long periods. Therefore, a strategy should be developed by the government and relevant stakeholders to identify sustainable ways to improve crop production systems other than through ordinary practised shifting cultivation. Crop production can be improved by the use of inorganic or organic sources of nutrients or a combination of both.

Most subsistence farmers will not be able to purchase manufactured inorganic fertilisers without heavy subsidy by government. Future research should therefore focus on the identification of potential organic sources of nutrients locally available. Such sources must be characterised and their performance evaluated before advocated for general use and they must be economically viable and socially acceptable. The characterization can for instance include the chemical composition and the factors that affect decomposition rate of their decaying litter.

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