The Impact of Alternative Grazing Methods on Soil Quality for Central Tablelands Grazing Systems

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Declaration of Originality

I hereby declare that this thesis is my own work and contains no material which has been accepted for the award of any other degree or diploma in any university or other institute of higher learning, nor material which has been previously published or written by another person, except where due acknowledgement has been made.

Neil John Southorn
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ABSTRACT

During the last decade, the concept of soil quality has evolved to help evaluate the impacts of agricultural management practices. A key feature of this approach is the potential usefulness of a measurable standard for soil quality, in much the same way as applies to water quality and air quality. However, an agreed approach to the assessment of soil quality remains problematic and elusive. The consensus view is that for it to be relevant, assessment of soil quality needs to be customised to the local circumstances of the particular investigation. This view has set the framework of this research.

The grazing of livestock is the dominant land use of much of south-eastern Australia, including the Central Tablelands of NSW, making a substantial contribution to regional economies and social fabric. There is little scope for large scale changes to alternative enterprises. The relatively high rainfall of these lands is also significant for its contribution to streamflow over much of the south east and to groundwater recharge. The land degradation that is occurring in association with agricultural management practices is reducing agricultural productivity as well as creating environmental impacts, and many of these are related to the perenniality of pastures. It is necessary to adopt management practices that change these trends, and for livestock enterprises, various types of rotational grazing strategies have been under consideration.

Soil quality is a major concern for agricultural and environmental reasons. However, assessment of soil quality requires the measurement of a number of indicators of soil quality. Soil structure is a key indicator of soil quality because of its importance to many soil functions, including water transmission, aeration, plant root penetration, habitat for soil biota, and so on, but direct measurement of soil structure is equally problematic.

In a replicated paddock-scale experiment, this research has compared two types of grazing treatments (set stocked grazing management and high intensity – short duration rotational grazing management) with a control where stock were excluded, as well as where grazing could occur without hoof pressure from livestock. A number of

surrogate measures of soil structure were deployed alongside the direct measurement of macropore characteristics using image analysis of resin impregnated soil cores, in order to evaluate their relevance in this context.

The results of this research have determined that set stocked grazing management has a greater detrimental impact on soil structure than rotational grazing management at similar long term stocking rates. It has also been demonstrated that soil structure under rotational grazing management can be maintained in a similar condition as that where livestock are removed. These results are associated with differences in livestock hoof pressure, soil structural recovery during grazing rest, maintenance of active pasture growth and soil surface litter, pasture botanical composition and favourable conditions for soil biota. Best soil structure, as evidenced by the largest unsaturated hydraulic conductivity and the greatest macroporosity throughout the 100 mm depth of sampling, was measured where pasture defoliation occurred in the absence of livestock hoof pressure. This indicates the importance of active pasture growth to soil structural condition. The research has also helped determine the strengths and weaknesses of various indicators of soil quality of relevance to Tablelands grazing systems, with image analysis most able to detect subtle differences between treatments in the nature of macropores.

These results have been used as input to a fuzzy logic model of soil quality. Compared to conventional models, fuzzy models allow the use of a wide range of data types, incomplete data and professional opinion and can be applied at different scales of investigation. This allows fuzzy models to cope with the uncertainties of complex issues such as sustainability, and can be applied to scenario testing of agricultural decision-making. Although further work is required, the model showed that such an approach has potential, by successfully discriminating between grazing treatments and deriving a soil quality 'score' for each. This approach confirmed that rotational grazing is a superior management strategy to set stocked grazing management for the maintenance of soil structure and pasture perenniality.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

1.1 Background to this research

Over the last 10 years, the concept of soil quality, its assessment and measurement, and its application to the management of agricultural (and other) systems has received widespread attention. Soil quality integrates physical, chemical and biological properties of soil for a specific land use, and is a useful concept to land managers, land use policy makers and the general public. It can complement widely adopted criteria for assessment of air and water quality in the monitoring of the natural environment.

Soil structure is nominated as a key indicator of soil quality because of its critical role in soil water dynamics, plant growth and development, and the suitability of habitat for soil biota. Soil structure dynamics will therefore influence agricultural productivity and environmental impact at the catchment scale, and vice versa - land management practices can result in rapid change in soil structural form.

Soil structural degradation has been observed since shortly after the introduction of agriculture to the Australian landscape and has been described as one of the most serious forms of land degradation (Chan and Pratley 1998). Cultivation associated with crop and pasture establishment is a common cause of soil structural decline, but the impacts of livestock are also relevant, particularly as a result of compaction from hoof pressure. In pastoral industries, mechanical intervention to relieve soil compaction is not always possible nor always cost effective and so 'biological' management of soil physical quality through better pasture management has come under scrutiny. This has created a focus on different grazing tactics – the use of livestock to manipulate the soil-plant-animal relationship.

Improvements to grazing management strategies are under consideration in the Central Tablelands of New South Wales (NSW), the geographic focus of this investigation, as elsewhere. Traditional grazing methods such as set stocking, where a fixed number of livestock are held continuously in each paddock, allow livestock to consume palatable pasture species selectively (Latham and Murray 1996). In addition to the continuous presence of hoof pressure, regeneration of perennial pasture species is compromised under such regimes, often resulting in changes to pasture botanical composition. These factors have been associated with declining pasture productivity and persistence, and sub-optimal livestock performance (Kemp and Dowling 2000).

Various strategies of 'planned grazing' have received attention from farmers and their advisors in an attempt to increase pasture productivity and consequently animal production, by enhancing the perennial species component of the pasture. One such strategy is High Intensity - Short Duration (HI-SD) grazing (Heitschmidt 1993), where large numbers of livestock are put onto pasture for a short time period when conditions are satisfactory, followed by a rest period after removal of the stock. The grazing pressure, as measured by the number and class of livestock, the timing and duration of the grazing event and the duration of the rest period, is determined by the judgement of the manager according to the stage of growth of the pasture, ground cover before and after grazing, and the expected pasture growth during the rest period. For example, a high stocking density may encourage grazing and trampling of less-palatable species, and if timed at flowering of a particular species (for example, undesirable annual species), can affect its subsequent seed set. High stocking density is also associated with rapid trampling of dead and green vegetable matter, which is likely to influence the rate of organic matter decomposition and assist the creation of surface mulch. One version of HI-SD grazing is 'cell' grazing, where larger grazing paddocks are divided into many smaller grazing cells, with the rotation of livestock between cells carried out according to a relatively strict grazing plan.

Strategies which advantage the perennial species component of pastures are likely to lead to increased dry matter production and pasture persistence, and may provide environmental benefits by increasing water use, reducing acidification, and reducing dependence on chemical control of weeds (Kemp et al. 2000, Michalk et al. 2003). Many studies have shown that perennial pasture species utilise a greater amount of stored soil water, ranging from 20 mm (White et al. 2000) to 50 mm per year (Ridley 1996) compared to annual species. In addition to potential environmental benefits, this translates into greater availability of soil water and greater utilisation of rainfall. This results from the combined effects of a generally greater root system depth and distribution and the perennial nature of these species resulting in an extension of the growing season, thus contributing to greater livestock production. Grazing tactics need to be designed to minimise detrimental impacts and maximise the rate of soil/pasture recovery.

However, much of the focus of recent research has been on the above-ground performance of such systems, which is not surprising given that this controls the short-term economic performance of the enterprise. It is suggested that HI-SD grazing strategies also influence the nature of soil structure, and therefore the long-term performance and sustainability of the enterprise, due to a number of potential impacts.

Whilst livestock hoof pressure is clearly related to soil compaction (for example, Packer 1988; Greenwood and McKenzie 2001), it has not been clear whether the high impact grazing pressure and subsequent rest period associated with HI-SD grazing are sufficient to provide decompaction benefits associated with increased organic matter turnover and the periodic removal of hoof traffic. A shift in pasture botanical composition to perennial (and often deep-rooted) species is likely to increase root activity, depth and distribution, and therefore organic matter dynamics and soil water use. Such a change may also increase the distribution and availability of substrate for soil biota (Mele et al. 1996), further impacting on soil organic matter dynamics. If an improvement in soil structure follows, as evidenced by an increase in pedality and ped stability, then improvements in soil water transport, water retention capacity, and the physical habitat for soil biota may further increase pasture productivity (Kemp et al. 2000). For example, if a small initial change in soil structure facilitates the presence or activity of burrowing soil macroorganisms or the production of deep taproots from pasture plants, then significant changes in soil physical properties are possible. In addition, Greenwood and McKenzie (2001) summarise investigations into the recovery processes for soil following grazing, including pasture root growth and decay, excretal return of nutrient and organic matter, and the presence of organic residue at the soil surface, all of which may be encouraged by HI-SD grazing management.

There have been few studies focussing on these aspects of grazing systems. Greenwood and McKenzie (2001) cite little evidence so far to support the potential benefits of HI-SD grazing on soil physical condition, finding no published research results in Australia. Proffitt et al. (1995a) state that there has been little attention in Australia to grazing management practices designed to improve topsoil structural condition. Pasture and grazing management investigations that have been linked to enterprise financial performance (for example, Michalk et al. 2003) have only recently emerged and require extrapolation to wider contexts.

1.2 Introduction to Central Tablelands grazing systems

The landscape of the Central Tablelands of New South Wales is characterised by variable topography, geology, soil types and vegetation characteristics. Kovac et al. (1990) describe the landform as undulating to low hills, with slopes typically 6% – 10%, but up to 30% with steeper rocky outcrops. Substantial areas of land have been cleared for agriculture and to a lesser extent mining, with the remaining native vegetation of the region dominated by tall woodlands in areas largely unsuitable for agriculture, although scattered remnant eucalypts are common. Kovac et al. (1990) list Chromosols, Kurosols, Dermosols, Sodosols and Calcarosols as soil types found in varying extents throughout the region, with textures varying from gravely loam to medium clay. Geological parent materials include andesite, tuff, conglomerate, greywacke and limestone.

The principal agricultural land use, historically and in present times, is the grazing of native and improved pastures by livestock for meat and wool. These enterprises make a significant contribution to the regional economy and social welfare. Mason and Kay (2000) describe the importance of the high rainfall zone of south-eastern Australia, of which the Central Tablelands of NSW are a part - permanent pasture is the dominant vegetation cover (of farmlands) producing 50% of cattle and 40% of sheep sales in southern Australia.

The climate is characterised by relatively high (annual average rainfall 750 mm) and reliable (for the temperate zones of eastern Australia) rainfall, which makes a significant contribution to streamflow of several catchments. If livestock grazing alters soil physical and hydraulic properties, this will also impact on the local water balance with consequential impacts on stream water quality and flow, with far reaching effects beyond the grazing zone. Secondary impacts associated with the transport of sediment, salt and solute nitrate will follow.

There is little commercial opportunity to change from grazing enterprises to cropping enterprises, other than horticulture, for a number of reasons: generally acidic soils, undulating to steep terrain, the presence of rock and trees, small paddock sizes, cool summer temperatures, lack of suitable crop species and lack of crop handling infrastructure. Consequently, livestock will remain part of many farming systems. Pasture species diversity, pasture productivity and grazing management become the primary tools for manipulating soil quality. The high rainfall zone of this region, by definition, should provide greater potential productivity than low rainfall zones and therefore contribute substantially to local and regional economies.

It is not possible to remove all livestock from the landscape, so management strategies need to be evaluated in the knowledge that impacts are likely to occur. Detrimental impacts can be minimised by a number of intervention strategies including confinement and supplementary feeding of livestock, allocating livestock to sacrificial areas and using amelioration tactics to repair subsequent damage, soil improvement strategies such as drainage of wet areas and pasture improvement, and temporary relocation of livestock to more resilient areas at times when soil or pasture quality may be compromised. This investigation suggests that rotational grazing strategies should also be considered, even though the evidence for this has not always been apparent.

1.3 Aims of this research

To determine the comparative effects of set stocked grazing and HI-SD grazing on soil quality, a long-term grazing experiment using merino sheep was established at Orange, in the central tablelands of NSW. This experiment focused on soil structure because of the critical importance of soil structure to soil quality, and the relatively sensitive response of soil structure to agricultural management practices. Soil structural attributes were measured in the differently managed grazing areas, in addition to a range of associated topsoil properties, to quantify differences in topsoil structure arising from alternative grazing tactics, and to elucidate the mechanisms causing soil structure change.

This research was initiated to

- identify appropriate and achievable indicators of soil quality and test their application and usefulness in the context of Central Tablelands sheep grazing systems,
- measure the long term impact of set stocked grazing management, high intensity – short duration grazing management and total grazing rest on soil quality and pasture botanical composition, at a scale that replicates a farming system,
- quantify the relationships between soil factors, pasture factors and grazing management factors in the assessment of soil quality, and
- in an objective manner, advance the concept of soil quality in the continuing debate on catchment health and the sustainability of Australian agriculture.

CHAPTER 2

THE RELATIONSHIP BETWEEN SOIL QUALITY, PASTURE MANAGEMENT AND LIVESTOCK – A REVIEW OF THE LITERATURE

2.1 Soil quality and sustainable agriculture

Most Australian farmers practice industrial agriculture. Although not universally applicable, an underlying assumption is that the purpose of agriculture is to produce a surplus of commodities to enable an agricultural business to return a profit. The environmental impacts of modern agriculture in Australia have been well documented, with sufficient data available to describe a significant proportion of the land area as degraded or at risk (Pratley and Robertson 1998, Reeves et al. 1998). The concept of sustainable agriculture accounts for these impacts. Although debate continues on the detail of the most appropriate definition, most include reference to three interacting influences: goals associated with farm production and business performance, personal and community or social goals, and environmental impact.

An individual farmer's decision and action on a particular management practice are influenced by numerous factors beyond simple measures of farm productivity or the results of scientific investigation. These factors relate to the combination of financial, social, cultural, environmental, business and political forces unique to each individual, the way these have evolved, and the perception of how they will continue to evolve. This mix of factors changes with age, circumstances and experience. These forces shape an individual's attitude to risk as well as their attitude to the quality of the information they receive. Ridley et al. (2003) found that farmers are reluctant to adopt scientific advice *per se*, particularly if considered by them to be "top down" advice, concluding that scientific advice on its own is not particularly useful in developing tools for farmers to use.

Most farmers acknowledge the importance of long term sustainability and their role as stewards of the agricultural landscape (e.g. Barr and Cary 2000, Cary et al. 2002,). However, farmers and some scientists often disagree with opinion on the impact their practices may have on the environment (e.g. Marohasy 2003) and have been shown to respond more to perceived short term economic factors than long term beliefs of environment protection as demonstrated by adoption of conservation behaviours (Cary and Wilkinson 1997, Barr and Cary 2000).

Farmers have traditionally monitored their production and business performance, and formally or informally compared their performance with benchmark values. Environmental performance is increasingly considered in a similar way (e.g. Rendell McGuckian 1996, Murray 1997), but methodologies and benchmark values are not yet fully developed. Although farm performance does not typically include the reduction in value of natural capital (Cameron and Elix 1991), Ringrose-Voase et al. (1997) propose a model where the capital value of agricultural land is linked to its productive potential, and present data to suggest that the productive potential of a particular soil–landscape type does correlate with soil quality factors such as organic matter content and soil pH.

Environmental monitoring programs have been established to monitor the impact of human activity on the natural environment, particularly in regard to air and water quality standards for human consumption. Soil quality is closely linked to these, given the role of soil in sustaining plant production, buffering hazardous materials, cycling nutrients and carbon, and partitioning infiltration and runoff from rainfall. However, because soil is a complex and variable material to deal with, and is not directly consumed by humans, soil quality has not received the same level of investigation. This is supported by National Research Council (1993), which stated that "protecting soil quality, like protecting air and water quality, should be a fundamental goal of (US) national environment policy". To achieve this goal, quantitative methods are required, and the incorporation of indicators into policy.

Sojka and Upchurch (1999) discuss the scientific weaknesses of a 'soil quality' approach. Air and water quality are defined by the concentrations of specific contaminants, compared to the pure state of the substance, as they relate to established levels of tolerance. This is not possible with soil, although it is attempted by some authors in the context of soil pollution by contamination. For example, Eijsackers (1998) describes the use of soil quality indicators for monitoring soil pollution, and the development of national standards in a number of European countries, including Denmark, Finland, Germany, the Netherlands and the United Kingdom. There is no comparable natural cycle of regeneration of soil, compared to the hydrologic cycle, even if nutrient and energy cycles are considered. Sojka and Upchurch (1999) therefore find little, if any, parallel; they do not consider a universal standard of soil quality relevant.

Soil quality, with particular reference to erosion risk and nutrient contamination of water sources, is one of 13 'issues' identified by the international Organisation for Economic Cooperation and Development requiring the development and application of indicators. This is part of a wider program of policy development relating to the impact of agriculture (and other elements of economic activity) on the environment (Parris 1996). The concept of a soil quality 'score' or index may prove popular with policy makers, because if soil quality can be classified and mapped, then formulae for allocation of government resources can be developed. Sojka and Upchurch (1999) warn of the dangers of this development, because of the inherent variability in soil quality, and because a definition of 'good' or 'bad' soil quality will depend on what the soil is used for, but economists generally prefer it. Reeves et al. (1998) summarise mechanisms for policy intervention, taking an econocentric perspective on alternate paradigms of sustainability.

Popp et al. (2000) attempt to simplify the notion of soil quality (they state that "soils are subject to natural degeneration caused primarily by erosion", which should be challenged in the Australian context) by defining an index to fit a theoretical econocentric modelling approach to derive 'policy relevant' information. Even so, they conclude that most indicators require some degree of subjectivity to complement scientific information, but that this subjectivity can be 'controlled' by appropriate technique or performance testing. Their empirical and somewhat obvious results include that optimal resource management will vary across soil type, that defining sustainability is more essential on lesser quality soils, and that uncertainty threatens long term achievement of any objective on susceptible soils.

There is a direct link between soil quality and catchment health (Walker and Reuter 1996), where a catchment may be a composite of land management units including farms. There are also similarities with the concept of rangeland health, which has been the subject of assessment for some time and which has evolved to include specific measures of soil stability and watershed function, integrity of nutrient cycles and energy flows, and the presence of functioning recovery mechanisms (National Research Council 1994).

The use of environmental indicators in assessment of farm performance is supported by Halberg (1999) for improved decision-making by farmers as well as environmental and ethical auditing. In a study of high intensity livestock farmers in Denmark, Halberg (1999) selected a number of soil quality and farm performance indicators, emphasising nutrient balance, energy balance, contamination risk and biodiversity. Although further refinement of the indicators and their interpretation is recommended, they were found useful by farmers at the farm level for including environmental factors in management decision-making.

Doran and Parkin (1994) suggest that the design of sustainable farming systems should focus on the maintenance and improvement of soil quality. Roberts (1995) nominates soil quality as one of the prime indicators of resource base sustainability, because in an agricultural context, soil quality will have a direct influence on plant productivity and soil water dynamics, and therefore catchment hydrology and water quality. Soil quality is readily manipulated by land management practices. However, McIntosh et al (1990) warn that a soil quality indicator alone is insufficient for assessing agricultural sustainability as it excludes measures of performance related to economic risk, dependence on inputs and ecosystem biodiversity.

Various authors have described a hierarchical model for sustainability, arguing that ecological characteristics of the system take priority over social and business goals. Lefroy and Hobbs (1992) and Lefroy et al. (1993), building on suggestions by Lowrance et al. (1986), argue that at the catchment level at least, ecological values and constraints should have priority, since ecological tolerances are less negotiable, and economic and social values are partly determined by human demands and expectations. Management decisions which impact on the natural resource base, upon which economic and social well-being are built, are elevated in importance, as long term sustainability is not possible with a degrading resource trend.

2.2 A definition of soil quality

Attempts have been made to define soil quality (for example, as summarised by Parr et al. (1992), Carter (1996) and Doran et al. (1994)) but a universally accepted definition has not been agreed upon, partly because of the dynamic nature and spatial variability of its inherent characteristics, the lack of objective data to quantify soil quality, and the complexity of the interactions between soil, plants, and animals. This may not be possible or necessary, given that criteria for assessment of soil quality will partly depend on the specific application and the social and economic aspects that influence it (Hamblin 1996).

An accepted approach is to focus on the functional importance of soil in the environment; that is, as a medium for the physical, chemical and biological processes that support plant growth; in the partitioning of water flow through the landscape; and as a buffer for environmental change (National Research Council 1993). The following definition of soil quality, proposed by Karlen et al. (1997) to encourage debate rather than as a final statement, has evolved from this line of thinking and includes the role of humans in the ecosystem: "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation". This definition goes some way to include the intrinsic value of soil, as recommended by Warkentin (1995). Dalal (1998) suggests that this definition be expanded to include the enhancement of soil biodiversity.

Although widely adopted as a conceptual definition, there is disagreement on how and when the definition of Karlen et al. (1997) can be applied to land management decision-making and agricultural policy. The concerns of Sojka and Upchurch (1999) arise from the difficulty in meeting this definition of soil quality from data derived from discrete measurements and applying a quantitative approach to soil quality policy in the absence of full information. In particular, they express concern about the development of soil quality indices describing "overall worth, value or condition of soil". This concern is summarised as follows; that there are risks and difficulties in "redefining the soil science paradigm away from value-neutral tradition of edaphology and specific problem solving, to a paradigm based on variable, and often subjective societal perceptions of environmental holism" (Sojka and Upchurch 1999). OECD (Parris 1996) proposes a soil risk methodology, to assess the vulnerability of a soil to degradation, rather than the allocation of a soil quality "state", within a specific agroecosystem context.

A measure of soil quality is by its nature, context specific. Halvorson et al. (1997) describe quality as a relative term, and measures of soil quality need to be tailored to the appropriate scale, and intended use of the land. For example, soil that is at risk of damage by regular cultivation may be considered to have 'poor' soil quality for the continuous production of cereal crops, but may be quite acceptable for permanent pasture. Sikora and Stott (1966) insist that farming practice and climate are necessary co-variables when interpreting soil quality data such as organic matter content. The disclosure of spatial and temporal context is therefore considered essential.

Lowrance et al. (1986) also discuss the importance of clearly defining the scale of investigation within the hierarchical scales of agricultural systems (plot - paddock - farm - catchment - region - nation), so that appropriate indicators of soil quality, and sustainability in general, can be selected for specific applications. For example, much of the work by advisory groups to Government is to assist in the development of regional and national policy, whilst recognising the need to develop farm level indicators of sustainability, particularly in relation to soil, biota and water, that can assist decision-making by land managers at the farm or paddock scale (SCARM 1993).

Debate continues on whether soil health is a better term than soil quality. 'Soil quality' seems to be preferred in objective use, and 'soil health' in subjective use. Walker et al. (1996a) also distinguish these from the term integrity when describing the landscape. 'Soil quality' is associated with measurement of attributes, and 'soil health' by personal perception or by the prevalence of pests and disease. Reginold and Palmer (1995) mix the two terms; their interest appears to be as advocates of biodynamic farming methods, where the use of the term 'soil health' is more widely used. 'Health' is frequently used to describe catchment or environmental condition, while the terms are defined separately by USDA NRCS Soil Quality Institute (http://msa.ars.usda.gov/al/auburn/nsdl/nrcs.html). The term 'soil quality' will be used throughout this investigation.

2.3 The use of indicators of soil quality

There has been much progress in the design of systems for monitoring agricultural sustainability, as a necessary step in the design of more sustainable farming systems. These include soil quality parameters. Most monitoring systems rely on the use of specific indicators; i.e. parameters that on their own do not give an assessment of sustainability, but provide data upon which a judgement can be made. An analogous approach is recommended for the assessment of rangeland health (National Research Council 1994).

The relevance of the indicators to the purpose of the investigation needs to be established. Monitoring systems designed to determine compliance with legislation may be quite different from those used for scientific results or farm management decisions, or the evaluation of benefits from public investment in policy programs. Monitoring at the plot or field scale will require a different set of criteria and sampling protocols to those used at the catchment scale. The Standing Committee on Agriculture and Resource Management compiled a suite of indicators suitable at the regional level (SCARM 1993). These comprise regional water use efficiency, regional nutrient balance based on average farm production and "average soil type" (clearly not appropriate at the farm scale), the percentage of native vegetation ground cover, and degree of vegetation fragmentation, but not including a measure of soil loss or degradation. Pilot testing of these has resulted in a number of recommendations (SCARM 1996), including further investigation of alternative regional measures of water relations, the need for more consistent data for nutrient balance studies, and improved measures of biological resilience. Rendell McGuckian (1996) used SCARM indicators as the basis of an investigation into benchmarking sustainable farming systems. A draft set of indicators has been established by NSW Department of Primary Industries; of specific relevance to soil quality are soil fertility (of which soil pH is considered critical), soil structure and soil salinity.

Because it is rarely possible to monitor all possible indicators, it is generally acknowledged that a set of key indicators will need to be identified to form part of a minimum data set (Herdt and Steiner 1995, Walker and Reuter 1996b). The outcome of this process will depend on the purpose to which the results will be put, and the resources available to sample and test. For example, Mele et al. (1996) discuss the roles of "land-user friendly" and "laboratory friendly" soil tests, and nominate their choice of biological indicators to include one that is associated with nitrate leaching and acidification, which was of particular interest in their study.

Work by Larson and Pierce (1991) has formed the foundation of this approach. They nominate the components of a Minimum Data Set for evaluation of soil quality, and describe the possible application of pedotransfer functions to predict the value of one parameter from measurement of another. They identify problems of scale (spatial and temporal), the need for benchmark or reference values, and recognise the need to retain some professional judgement. The approach is not recommended for all agroecosystems, nor all agricultural cultures. Other workers have built on this approach (Doran 1996, Doran and Parkin 1996).

There are a number of difficulties in relying on a small number of key indicators. For example, although soil microbial respiration is generally regarded as an important indicator of soil quality, it is insufficient on its own. Wardle (2002) and Reginold and Palmer (1995) differ on the interpretation of soil respiration data, and on its ability to measure ecosystem efficiency. They advocate the consideration of a suite of indicators, from which some judgement of soil quality might be interpreted. Further, they concede that guidelines for making such judgements still need to be developed.

Sojka and Upchurch (1999) are also critical of the adoption of non-scientific indicators of soil quality (e.g. smell and feel) and propose, correctly, that indicators such as total organic carbon content are not particularly useful on their own, and in the case of biological indicators, there is a general lack of hard evidence to connect a certain indicator value to a soil quality function. Franzluebbers (2002) attempts to counter some of the criticism of Sojka and Upchurch (1999), particularly in regard to the lack of evidence of the roles of soil organic matter pools to soil quality, but agrees that the measure of the total pool or organic matter is not a reliable indicator of soil function or quality.

The need for a complex of indicators is also emphasised by McIntosh et al. (1990). In a study of the effects of fertiliser application and oversowing exotic pasture species in high rainfall tussock grasslands of New Zealand, some indicators were positive (such as increased soil carbon and nitrogen contents) but others were negative (such as plant-pathogenic nematode numbers). They conclude that a simple measure of soil quality is not possible, and in the absence of an appropriate suite of indicators, soil quality interpretations may be incorrect. Sojka and Upchurch (1999) provide specific examples of misinterpretation. They cite the need for an increased rate of pesticide application for soils high in organic matter, the increased bypass flow of water and dissolved contaminants in earthworm burrows and the capacity of earthworms to transport soil borne disease organisms, the usefulness of soil compaction in wheel tracks for improved seed-soil contact, the failure of generic measures of soil microbial activity to distinguish between beneficial and harmful soil organisms, inability to determine a benchmark condition for many biological indicators and misinterpretation of their benefits (e.g. a high soil respiration rate).

To conduct a quantitative assessment of soil quality, it is necessary to assign some form of reference value to the parameters of interest, compare the measured value with the reference value, and make a judgement about the difference. For some parameters, threshold values can be based on known responses, such as plant response to soil nutrients and pH. In other cases, local information can be incorporated, such as the behaviour of the soil during wet and dry periods. Benchmarking, common in business applications, has also been applied to soil quality assessment. Despite difficulties in standardisation of methodology and spatial and temporal variability, Walker and Reuter (1996) provide guidelines for selecting reference or benchmark values for selected indicators, and a scoring system for assisting judgment on condition. Walker et al. (1996b) use a case study of a particular mixed enterprise farm to show how this approach could be applied. A different approach is taken by Murray (1997), who recommends that benchmarking for sustainability focus on processes of change, not just performance indicators, in an attempt to provide a farmer-centred approach to improvement. The Selwyn Stewardship Monitoring Scheme, described by

Wratten et al. (1997), appears quite advanced in its use of indicators to monitor sustainability, which includes a focus on soil quality. Initial use of the scheme has been reported as successful (Wratten et al. 1997).

Jodha (1995) describes the use of indicators of 'unsustainability' (for example, an inability to effectively face the impact of drought without external assistance) which may provide an alternative perspective. Soil quality concerns are not limited to "poor" soils. For example, Bridge and Bell (1994) describe significant structural degradation of Ferrosols, generally regarded as "good" soils, but with a long history of agricultural development.

Liebig and Doran (1999) included farmer perception of soil quality in a comparison with field and laboratory assessment. They concluded that farmers perceptions of soil quality were "near accurate" for 75% of comparisons for the majority of indicators used in their study, particularly for soils that were rated as exhibiting "good" soil quality. They suggest that this is an important factor to consider when planning a scientific assessment of agricultural land - that the farmer's opinion about the soil, based on crop response or its behaviour during tillage, will assist in identifying variability or the best sampling regime for testing. This is probably to be expected from farmers who have worked a field for many years, and have observed its hydraulic behaviour and other physical characteristics over time, but Liebig and Doran (1999) used data classes, such as nominal ratings (high, medium, low; ranking 1-5) rather than discrete numerical values to form their conclusions, arguing that this is appropriate when many soil quality indicators have ecologically important threshold values. Sojka and Upchurch (1999) express great concern over the use of non-scientific data, particularly when a correlation between field measurements and laboratory tests has not been established. They cite the use of soil smell as an indicator of soil quality as a case in point, where there is no objective data to support its use.

Rhoton and Lindbo (1997) suggest that effective soil depth, defined as the depth of soil to a restrictive layer, is a useful integrative measure of soil quality, based on data from one site characterised by the presence of a water-impeding layer at variable depth. Although limited in scope, the results of this study indicate a need to determine the presence of, and measure the depth to, restrictive soil horizons when comparing data within and between fields.

There is a need for reasonable standardisation of methods. MacEwan and Carter (1996) call for standardised methodologies and identification of critical limits. Minimum standards are required for sampling protocols and sample analysis. For example, Doran and Parkin (1996) and Dick et al. (1996) describe significant differences in interpretation, in this case of C and N content, between results displayed on gravimetric data compared to volumetric data. They emphasise the importance of measuring and recording soil bulk density as part of a minimum data set partly for this reason. Reginold and Palmer (1995) suggest that this may disadvantage sites with low bulk densities and greater depths of topsoil. In their study, data were compared for both gravimetric and volumetric measurements, and found that differences between treatments varied between the two methods for some soil properties. They recommend that volumetric measurements be on a per unit of soil depth basis, rather than on a standardised soil depth such as 100 mm, to avoid this bias.

If benchmark values can be applied to key indicators of soil quality, then trends in indicator properties can be monitored over time. Although Sojka and Upchurch (1999) warn of the need for caution in the application of soil quality values, the concept has intuitive appeal to those responsible for making management decisions on land use, in agriculture as well as on public land. Indeed, land management requires managers to make judgements about the suitability of a particular soil to a particular practice.

A number of papers describe the use of soil quality indicators for specific applications. For example, soil quality criteria have been used, together with other indicators of farm performance, to compare conventional farming systems with organic or biodynamic systems (Reginold et al. 1993, Penfold et al. 1995). There have been a large number of studies into the effects of tillage practices on soil quality. In many cases, key soil quality indicators are related to organic carbon contents, soil aggregate stability, penetration resistance, soil shear strength, bulk density and hydraulic characteristics, but often include measures of biological activity, such as microbial biomass. In one study, Bridge and Bell (1994) used multiple indicators to measure the impacts of continuous crop production on soil physical characteristics. Boehm and Anderson (1997) measured significant differences in soil quality after different fallow durations, and concluded that the increased fertiliser inputs and more frequent additions of crop residue resulted in improved soil quality under annual cropping compared to different durations of fallow, in Saskatchewan, Canada. Bell et al. (1997) use similar methods to determine the beneficial impact of a pasture phase in a crop rotation, compared to continuous cropping. However, such conclusions need to be considered in the circumstances applicable to each study. Indicators and their respective benchmark values that might be relevant in high input continuous cropping may not be relevant to grazing systems.

2.4 Specific indicators to describe soil quality for grazing systems

The following indicators have been drawn from a wide range of soil quality investigations. These include indicators of physical, chemical, biological and integrative properties of soil, to establish the possible indicators of relevance to grazing management – soil quality interactions.

2.4.1 Soil Texture and Structure

Soil texture refers to the proportions of sand-, silt- and clay-sized particles in a sample, as determined by standardised methods for particle size analysis (Gee and Bauder 1986). Texture can also be estimated in the field through manipulation of a bolus of soil (Northcote 1979). Soil texture provides a preliminary but essential starting point for the classification of soil. Table 2.1 summarises one view of environmental constraints linked to soil texture group.

Soil structure is more difficult to define and quantify, but generally refers to the arrangement of soil particles into aggregates, the resulting arrangement of pores within and surrounding these aggregates, and the behaviour of aggregates during wetting, drying or mechanical manipulation (Kay 1990). Letey (1991) provides the following definition of soil structure attributed to Brewer (1976): "the physical constitution of a soil material is expressed by the size, shape and arrangement of the solid particles and voids, including both the primary particles to form compound particles, and the compound particles themselves; fabric is the element of structure which deals with this arrangement". The difficulty in quantifying soil structure is implied by Letey (1991) by suggesting that the study of soil structure is both an art and a science.

Table 2.1
Environmental constraints based on soil texture group
(after Fitzpatrick 1996)

Texture group	Clay content	Restriction to root growth	Susceptibility to mechanical compaction	Water availability to plants	Drainage characteristic
Sand, loamy sand, clayey sand	5– 10%	none	moderate	low	Very rapid drainage can cause periodic soil moisture stress
Sandy loam	10-20%	none	high	Available to most crops and trees	Readily drained but not rapid
Loam, sandy clay loam, clay loam	25-35%	none	moderate	Available to most crops and trees	Very slight restriction on water movement
Light clay	35-40%	frequent	moderate	Available to most crops and trees	Restricted water flow contributes to periodic waterlogging
Medium clay, heavy clay	>45%	severe	low	high	Very slow drainage (other than self mulching clay properties)

Together, texture and structure greatly influence the physical and hydraulic characteristics of soil, including its mechanical strength, porosity, rate of infiltration, hydraulic conductivity, and water-holding capacity. These factors have a direct effect on certain plant growth factors, including mechanical impedence to root penetration, the availability of soil water, and provision for gas exchange in the root zone (Passioura 1991, Sommer et al. 1995, Tan 2000). Texture and structure will also influence soil chemistry, either directly (e.g. through the movement of solutes through the profile; Edis and White 2003) or indirectly (e.g. through the behaviour of cations in the clay fraction; Tan 2003), and soil biota (via the provision of suitable habitat; Anderson and Domsch 1995). A composite of measures is therefore needed to quantify soil structure.

Cass et al. (1996) suggest that the physical status of soil can be interpreted from the storage capacity of a soil, defined as a comparison of total plant available water and air-filled porosity at field capacity, as shown in Figure 2.1. Fitzpatrick (1996) interprets soil structural quality according to field observations of soil pedality, as shown in Table 2.2.

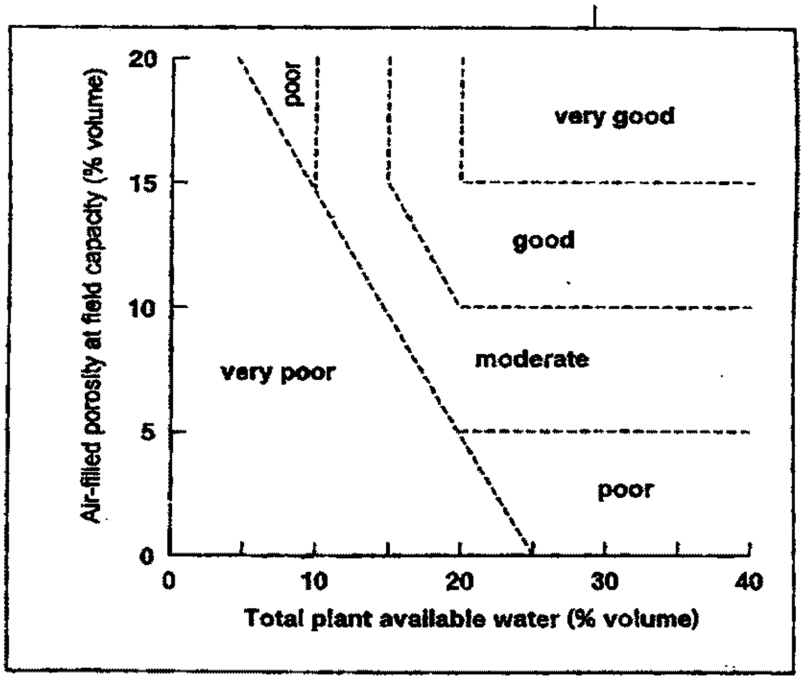


Figure 2.1
Interpreting the physical status of soils for their capacity to store air and water within soil pores for plant uptake (Cass et al. 1996).

Bulk density is a common measure of total porosity, a standard requirement for assessment of soil hydraulic characteristics, and necessary to relate volumetric to gravimetric measurement of particular parameters. For example, Thurow et al. (1986) advocate the use of bulk density as an integrating measure of soil physical quality, measuring a degree of correlation between bulk density and aggregate stability (R^2 = -0.59), soil organic matter content ($R^2 = 0.61$) and clay soil texture ($R^2 = 0.40$) in a

Structure	Soil indicator	Accessory indicator: Texture group	Accessory indicator: Consistence	Environment indication
Single grain	Single sand particles or grains	Sands	*Loose	No restriction on root growth but has a moderate susceptibility to mechanical compaction. No restriction on water movement but periodic soil moisture stress occurs because water is drained very rapidly.
Massive (No peds) (apedal)	#Massive	Sandy Loams; Loams, Clay Loams	* Soft to Firm	No restriction on root growth but has a high susceptibility to mechanical compaction. Low restriction on water movement.
Weak peds	#Poorly defined individual peds	Clay Loams	Soft to Very Hard	Moderate restriction on root growth but has a high susceptibility to mechanical compaction. Moderate restriction on water movement.
Moderate peds	Well formed peds, but not distinct	Light clays	Firm to Very hard	Strong restriction on root growth but has a moderate susceptibility to mechanical compaction. High restriction on water movement.
Strong peds Slickensides	Durable peds, evident in place; will stand displacement. Cracked, polished or grooved surfaces, ranging from 10 mm to 200 mm across	* Medium and heavy clays	Very hard to Rigid	Capacity of soil to shrink and swell and develop cracks when dry. Water drains very slowly when wet. Severe restriction on root growth.

*Self mulching clays (medium-heavy clays) generally have a loose or soft consistence.
#Sub-plastic clays (medium-heavy clays) generally have a massive structure and soft consistence.

Table 2.2
Interpreting environmental constraints to plant growth based on field assessment of soil structure (Fitzpatrick 1996)

study of grazing effects on soil properties in Texas, USA. The bulk density of a soil relative to its potential compaction is a useful indication of its degree of compaction. However, measurement of bulk density does not describe the nature of the porosity. Furthermore, the relationship between bulk density and plant growth, which may lead to a description of a limiting value of bulk density, is dependent on other soil characteristics such as pore size distribution, organic matter and soil texture (Kay and Grant, 1996).

Pore size distribution has a significant effect on plant root development (Passioura 1991), soil hydraulic characteristics and habitat for soil biota, and is therefore considered an important characteristic of soil structure, particularly when the continuity of vertical macropores is considered (Juma 1993, Kay and Grant 1996). However, direct measurement of pore sizes and their distribution and connectivity is a difficult task in the field, usually assessed by referring to disc permeameter measurements performed over a range of water tensions. Coughlan et al. (1991) provide some examples where differences in the distribution of macropore diameter have been measured on soils subject to different soil management practices. They describe this ability of a disc permeameter, and its use in the field to measure soil surface properties *in situ*, as the advantages of this method, whilst conceding that the relationship between soil structure and soil hydraulic properties is problematic in certain soil types; for example, in the case of Vertosols, which exhibit extreme shrink-swell behaviour.

An alternate approach under development is to use a digitised image of the soil pores, constructed from a soil sample impregnated with resin. Analysis of the image allows quantitative assessment of pore relations, and has been applied to comparative assessment. Douglas et al. (1992) used image analysis to detect differences in the nature of soil porosity and soil pore characteristics in a comparison of different levels of perennial grass production induced by different amounts of applied nitrogen fertilizer. Koppi et al. (1992) used image analysis to measure differences in soil physical attributes caused by wheel traffic in silage production. Proffitt et al. (1995b) also showed that image analysis can be quite sensitive compared to other measures of soil structure.

Three-dimensional characterisation of macropores has also been investigated by Perret et al. (1999) using X-ray CAT scanning and three-dimensional reconstruction software, on undisturbed soil cores extracted from the field. However, the technique, whilst providing excellent and reliable images for interpretation, requires elaborate equipment, and constructs the 3-D model from multiple 2-D scans. The use of stacked images of impregnated resin can provide similar data, and is more readily applied to larger numbers of samples using standard equipment.

The resistance of soil aggregates to crushing is a measure of soil strength related to structure as well as other factors, including organic matter content. It can be assessed by measuring shear strength or modulus of rupture, but these are likely to be most relevant to crop production.

Penetration resistance is a simple technique that can be applied in the field. A rod is pushed into the soil at a constant slow rate, and the force required is related to soil wetness, soil strength and impedence. However, simple devices require substantial judgement by the operator in both technique and interpretation, particularly in regard to dynamic friction effects. More sophisticated devices minimise the need for operator judgement and use a standard cone attachment, but still require a large amount of data to give reliable analysis. It has been reported as a useful screening technique to determine the extent of soil physical variability (e.g. Hartge et al. 1985).

Aggregate stability is an important measure of the ability of soil to maintain a structural form under cycles of drying and wetting, and is associated with the soil quality attributes of hydraulic conductivity, surface crusting, and susceptibility to erosion (Emerson 1991, Karlen and Stott 1994). Structural stability is also related to electrical conductivity and sodium adsorption ratio, as described by Cass et al. (1996) (Figure 2.2), and influenced by biological activity. Cass et al. (1996) describe the use of a simple test for structural stability, adapted by McGuiness (1991) from Emerson (1991).

Coughlan et al. (1991) compared 5 methods to measure or characterise soil structural form and stability, particularly as they might relate to Vertosols, where it is inappropriate to extrapolate results derived from rigid soils. They concluded that further work is needed to develop appropriate indices for use in soil process models, but several techniques can be selected depending on the application of the data (e.g. erosion modelling, compaction studies). Image analysis was not evaluated, but appears to satisfy many of the criteria recommended.

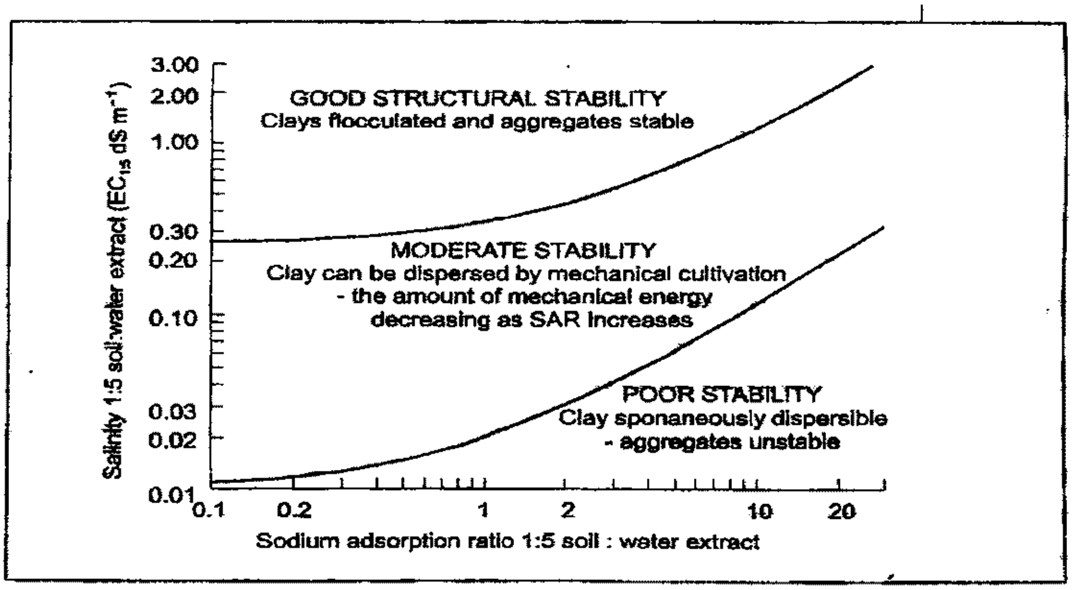


Figure 2.2
Interpreting the physical status of surface soil layers by assessment of soil salinity relative to sodium adsorption ratio (in Cass et al. 1996 modified after Sumner et al. 1996)

2.4.2 Soil Hydraulic Characteristics

The link between soil structure and fundamental soil hydraulic properties is described by Cresswell et al. (1992). Soil hydraulic characteristics include surface infiltration rate, soil water-holding capacity, hydraulic conductivity and the variation in hydraulic conductivity with depth. These characteristics influence the partitioning of rainfall between runoff and infiltration, the subsequent movement of water through and beyond the root zone, the degree of aeration within the root zone and the availability of soil water to plants, and are therefore considered important characteristics of soil quality when combined with measures of soil texture and structure. However, hydraulic properties provide only partial information about soil structural form.

Field measurement of hydraulic conductivity is described by Geering (1995), McKenzie and Cresswell (2002) and others. Techniques include single or double ring infiltrometers, and constant head, drip, well and tension disc permeameters. Kay and Grant (1996) also discuss the use of least limiting water range (LLWR) which integrates water-holding capacity with the plant growth limits of aeration and mechanical impedance. Temporal variation in soil water content, which determines the frequency that soil water content falls outside the LLWR, is one of the determinants of plant growth potential.

The surface characteristics of soil and the vegetation it supports partly determine the partitioning of water between runoff and infiltration. Runoff is associated with erosion and sediment transport, and its measurement under various management regimes is highlighted in many studies. This is accomplished by the construction of barriers surrounding experimental plots, or by measuring the other components of the local soil water balance and determining the amount of runoff by difference. With this methodology, the change in soil water store becomes an important consideration.

2.4.3 Chemical and Fertility Characteristics

Standard measures of soil fertility and chemical condition include pH, electrical conductivity (EC), cation exchange capacity (CEC), plant available nutrients (N, P, K, S), trace element nutrients, and indicator or toxic elements (e.g. Al^{3+}) (Elliot 2000). In an agricultural context, the availability of plant nutrients and the level of toxins should be non-limiting.

Soil pH is an essential parameter for soil monitoring. Soil pH is a direct measure that can be used in monitoring to detect trend of a number of processes that affect soil quality (acidification, salinity, acid sulfate conditions, alkaline sodic conditions, sources of soil profile salts, chemical restriction to root growth) as well as being an accessory measurement of soil fertility, nutrient availability and biological activity (Cresser et al. 1993). Merry (1996) judges soil pH as the single most informative chemical indicator of soil health. Soil EC is a sister measurement to soil pH, in terms of usefulness, application, relevance to plant growth and ease of measurement.

Merry (1996) nominates soil analysis for chemical fertility as a set of key indicators of soil quality. Specifically, N, P, K, exchangeable cations and trace elements are nominated for frequent measurement to determine the status of nutrients essential to plant growth. Without optimum plant growth, soil quality (and agricultural productivity) may be compromised. An oversupply of the same elements may compromise environmental values.

Soil chemical characteristics can influence surface water and groundwater quality (Jolly 1996), and the presence and activity of soil biota. These factors should be considered when assessing soil quality in the broader context, and an optimum determined between the benefit to plant growth and detrimental impact on soil and water quality. For example, nutrient transport occurs with sediment. Nitrate accumulation in excess of plant requirements can be leached through the profile, contributing to acidification, contamination of groundwater, and eutrophication of surface waters. Subsoil testing of certain parameters, such as nitrate-N and pH, should therefore be considered (Moody et al. 1996).

2.4.4 Plant Growth Characteristics

In an agricultural context, plant production (or yield) relative to potential plant production (or yield) is an indirect integrative measure of soil quality (Walker and Reuter 1996). Pasture production also contributes to soil quality via generation of organic matter. This contributes directly to soil structural condition and provides food supply for soil biota.

Specific plant growth characteristics related to soil quality include root depth and distribution, root tip condition and number of active roots, root nodulation in the case of legumes, species composition and dry matter production (Passioura 1991). For grazing enterprises, the quality of pasture will also be important, as livestock performance will vary accordingly. Consequently, the proportion of grasses, legumes and weeds, and their stage of growth, will provide additional evidence to support pasture quality assessment, and tests for digestibility, palatability, protein and metabolisable energy are widely used in detailed studies.

Maintaining at least 70% of ground cover has been accepted as a requirement to reduce the risk of soil erosion during storm events, and the relationship between ground cover and pasture composition has been nominated (Lang 1979). From the perspective of pasture recovery from grazing, a ground cover limit of 0.9 - 1.5 t DM/ha should be maintained, depending on species and stage of growth (Fitzgerald and Lodge 1997). More recently, Kemp et al. (2003) associate maintenance of species diversity and optimum pasture production with a herbage mass of between 2 and 4 t DM/ha.

Dowling et al. (2005) classify pasture species into functional groups and apply the use of a pasture species composition matrix, finding it useful to monitor the dynamics of pasture composition under grazing. Changes in species composition are likely to occur over time, which may be an important outcome of alternate grazing treatments, and may be correlated with total pasture production, increasing persistence of desirable pasture species, and differences in soil organic matter dynamics. The method requires periodic transect observation of species composition, preferably using the BOTANAL assessment described in detail by Tothill et al. (1978), and rating these according to the proportions of desirable and undesirable grasses and legumes.

2.4.5 Soil Organic Matter

Soil organic matter (SOM) plays a significant and integrative role in the assessment of soil quality. SOM is associated with soil aggregation and aggregate stability, the activity of soil biota, soil hydraulic characteristics, nutrient supply and recycling, and ion exchange and buffering capacity (Handreck 1979, Uren 1991, Moody et al. 1996).

Tisdall and Oades (1982) classify organic binding agents into three classes. Transient material is associated with the rapid decomposition of polysaccharides by microorganisms, which influences the behaviour of microaggregates. Roots and hyphae are classified as temporary binding agents, which are considered important to the formation and stability of young macroaggregates. Temporary and transient agents respond readily to management practices such as cultivation. Persistent binding agents are humic materials, associated with the stability of macroaggregates, and are considered less susceptible to soil management. Soils under pasture production normally generate a continuous addition of organic material to the soil, a condition necessary for maintenance of soil structure for many soils (Skjemstad et al. 1998), and should exhibit greater aggregate stability compared to soils under cultivation.

Organic matter provides the food supply for soil biota. In addition to the production of exudates from soil organisms, which act as binding agents themselves, their activity in the soil results in changes to the form of organic matter, and for macroorganisms, the creation of physical features such as burrows and tunnels (Papendick 1994).

SOM is usually monitored by measurement of total organic carbon content, but there is evidence that the nature and proportion of the various organic matter fractions is important. For example, the amount of labile carbon has been described as more useful than total organic carbon content as an indicator; its application in monitoring soil quality has been recommended by some authors (Blair et al. 1995, Lefroy et al. 1996) but not by others (Skjemstad et al. 1998). Particulate organic matter (POM), defined by Sikora and Stott (1996) as the fraction of SOM between 0.05 mm and 0.20 mm, has been described as a sensitive indicator of long term organic matter dynamics, and has been suggested as a soil quality indicator. The need to identify the nature of the organic matter components has been emphasised by Skjemstad et al. (1998). Using new techniques including nuclear magnetic resonance and infra-red spectroscopy, it has been shown that much of the carbon content of Australian soils is inert charcoal, and that at least part of this fraction is included in all organic carbon laboratory methods in common use. They conclude that a single measure of total organic carbon is not a useful description of the function of organic matter in soil, but there are currently no practical alternatives.

In addition to average quantity and rate of change of SOM, its distribution through the topsoil may be of interest. Franzluebbers (2002) proposes a soil organic matter stratification ratio as an indicator of soil quality (more precisely, the stratification ratio of various organic matter pools). Franzluebbers (2002) defines this ratio as the amount of soil carbon at the soil surface to soil carbon at some other depth (taken as the depth of tillage in this case). Based on data derived from comparisons of conventional tillage for crop production and direct drilling, he found that soils with a history of direct drilling had a higher stratification ratio that those under conventional tillage, and that a ratio of 2 was considered a threshold of sustainable practice. Franzeleubbers (2002) also applied stratification to various carbon and nitrogen pools, identifying soil microbial biomass stratification ratio as most sensitive to soil management. It remains uncertain how a stratification ratio might apply to untilled soils.

Although numerically small, the proportion of microbial biomass carbon appears to be a sensitive indicator of SOM dynamics (Powlson et al. 1987, Visser and Parkinson 1992, Sparling 1992, Swift 1994), but laboratory methods are required. A quantitative link between microbial biomass and soil quality remains elusive, although Chan and Heenan (1999) found that differences in aggregate stability under different crop rotations was attributed to differences in soil microbial biomass, there being no significant differences in total organic carbon or carbon contents measured by various extraction techniques.

2.4.6 Soil Biota

Soil biota include the numerous groups of micro-, meso-, and macroflora and fauna that inhabit soil and/or the soil/litter surface. Despite their relatively low biomass compared to total organic matter, their role in regulating ecosystem processes (organic matter decomposition; nutrient cycling, uptake and synchronisation of availability; contribution to soil structure and structural stability) gives them significant functional importance in the sustainability of agroecosystems (Hendrix et al. 1990, Swift 1994). These processes are detailed in Pankhurst et al. (1994). Wall and Moore (1999) summarise the mutualistic relationships between soil organisms, highlighting the importance of such relationships to ecosystem functioning, threatened by reducing soil biodiversity. Dalal (1998) allocates the roles of soil microbial biomass into three critical functions associated with carbon and nutrient transformation and pesticide degradation, in addition to its roles in soil aggregation and formation and associations with plants. The non-beneficial role of certain soil organisms such as nematodes and soil-borne plant pathogens should also be noted.

Specific indicators include the population density and activity of specific organisms, the diversity of organisms and their total biomass, the rate of specific microbial processes, or key indicators of these (Parkinson and Coleman 1991, Pankhurst 1994, Pankhurst et al. 1994, Zak et al. 1994, Stenberg 1999). King (1996) nominates microbial respiration rate as a key measure of soil quality, because of the significant contribution of soil microbes to soil ecosystem function, and their ubiquitous nature. Methods of measurement of microbial biomass are summarised by Dalal (1998), and its potential as an indicator of soil quality is discussed; a sensitive and potentially useful indicator but difficult to apply easily to comparative studies. Potter and Meyer (1990) explore the possibility of soil organism diversity as an early warning indicator of ecosystem disturbance, while Turco et al. (1994) describe it as critical. Visser and Parkinson (1992) suggest process-level studies such as carbon dioxide respiration to be the most useful for rapidly monitoring changes in soil quality, whilst providing information to assist the planning of population or community level studies, although the importance of the latter should not be overlooked (Kennedy and Smith 1995). Oberholzer et al. (1999), in an investigation of Swiss soils under various crop management regimes, determine that a universal model of soil microbial activity is not possible, and that abiotic soil properties must be defined. Comparing a number of data sets, they found best correlation (R^2 = 0.76) between substrate-induced soil respiration and the mix of abiotic factors of pH, clay content and organic carbon content. Tisdall (1991) describes the mechanisms by which fungal hyphae can create an increase in aggregate stability, including the production of binding agents such as extracellular polysaccharides.

To be a useful biological indicator, Dalal (1998) nominates the following criteria: the measurement of more than one soil function; sensitivity of measurement to change; the presence of threshhold or benchmark values; accepted interpretation; cost-effectiveness. Halberg (1999) and Stenberg (1999) make similar suggestions. Dalal (1998) states that microbial biomass meets all but one of these criteria. At the moment, there is not a clear connection between microbial biomass measurements and soil quality, because benchmark values will differ depending on soil type and land use, and there are problems with standardisation of test procedures. For example, microbial respiration can be high in contaminated or degrading soils as well as those regarded as "good", and it is likely to be more useful to investigate the type of organism that is active (such as the relative proportions of fungal biomass to bacterial biomass). Wardle (2002) confirms that soil respiration is not necessarily beneficial. Despite these limitations, microbial biomass, microbial respiration rate and their relation to soil organic matter levels remain important components of soil quality measurement, but they are not suitable in isolation.

Reuter (1998) presents work from New Zealand by Schipper and Sparling (1997) expressing a relationship between microbial biomass and the ratio of respiration rate and total C, and describing this measure as a significant indicator that soil biological processes are "in balance". This is an attempt to define an acceptable activity level for microbial processes (a higher level of microbial activity results in a higher proportion of soil carbon being metabolised) for a range of soil conditions and climate.

A cotton strip assay (CSA) technique, described in detail by Harrison et al. (1988), has been proposed as a useful indicator of microbial status, and has been shown to be sensitive to differences in microbial activity (King 1995, King et al. 1996). The results will be biased toward those microbes responding to a cellulosic substrate, but the method can be adapted to a variety of additional substrates. It has been suggested as useful in catchment studies (King and Pankhurst 1996). It therefore shows promise as a useful field technique for monitoring soil microbial activity, although Mele et al. (1996) express some reservations and Visser and Parkinson (1992) warn that laboratory approaches are preferred to minimise the effects of moisture and temperature variation on tests of microbial processes. Liebig and Doran (1999) also note the potential of CSA as a field test of microbial activity, but note uncertainty in its relationship with laboratory measurement of soil respiration and biomass characteristics. Latter et al. (1988) advise that tensile strength increases of cotton strips are possible as a result of certain biochemical reactions in soil that offset tensile strength losses due to microbial deterioration.

In comparison to CSA, which measures a global indicator of microbial activity in response to cellulosic substrate, the Biolog method uses a large number of carbon sources to characterise the range and diversity of microbes present in soil. The method has been use to compare organic production systems with non-organic systems (Bending et al. 2004), to characterise polluted soil health (Avidano et al. 2005), compare tilled with non-tilled soils (Bucher and Lanyon 2005, Liphadzi et al. 2005) and other land use comparisons. Graham and Haynes (2005) used the Biolog method and substrate induced respiration (SIR) to compare soil under different land management practices. They found that whilst Biolog was able to differentiate between classes of land use (e.g. tilled soils for the production of sugarcane and grassed soils) SIR was more effective and able to separate relatively similar land uses (e.g. native grassland from improved pasture). They concluded that the main differences were likely to relate to differences in soil organic matter content, specifically, the difference in labile carbon fraction. It was noted, however, that the undisturbed native grassland site contained a lower organic matter content than the more intensively managed improved pasture, but exhibited greater microbial diversity. It was suggested that despite the lower level of management input, the greater botanical species diversity in the native grassland site results in greater diversity in microbial habitat and more heterogeneous litter input. Banu et al. (2004) used Biolog Ecoplates to measure microbial substrate utilisation patterns in the characterisation of microbial diversity in eight soils from mostly pasture sites in NSW, noting significant differences in microbial diversity between some soils. They also noted the limitations of culture-based methods of assessment of microbial diversity, in that only a small proportion of soil microorganisms are truly culturable, recommending molecular-based methods as more successful. The Biolog method reflects the profile of the microbial community that is produced following a period of substrate utilisation and not necessarily the microbial community profile that originally existed in the soil at sampling (Graham and Haynes 2005).

Invertebrates (such as earthworms, ants, collembola) have also received interest as indicators of soil quality, because of their contribution to organic matter fragmentation and incorporation, enhancement of microbial processes, contribution to soil structural form and structural stability by construction of stable macropores, their sensitivity to certain soil conditions, and the potential of some species to incorporate surface applied soil ameliorants (Greenslade and Greenslade 1984, Lee and Foster 1991, Stork and Eggleton 1992, Baker et al. 1993). Problems with sampling, taxonomy and lack of understanding of some aspects of invertebrate ecology make it difficult to advocate their widespread use as soil quality indicators, although key indicator species can be selected for specific agroecosystems.

It may be possible to use selected species of soil/litter invertebrates for a rapid and inexpensive indication of trend, upon which more elaborate testing can be based, in a similar way that can be used for water quality assessment (Chessman 1995). Linden et al. (1994) nominates at least the burrowing soil fauna be included. For example, earthworms are easily sampled and identified (Baker and Barrett 1994) and are therefore a useful field indicator of soil quality for higher rainfall grasslands not subject to cultivation, although their distribution in all soils is not uniform. Mele et al. (1996) measure differences in earthworm abundance and diversity in a comparison of the sustainability of different pasture management systems. However, in this study the increase in earthworm activity appears to be related more to soil pH (with lime added as a treatment layer in this work) than to pasture botanical composition or other factors. Also, all plots were subjected to identical rotational grazing tactics with a fixed grazing period and interval. Fraser (1994) notes that earthworm counts are higher during the pasture phase compared to the crop phase of a crop-pasture rotation: not surprising given the impact of cropping machines on topsoil structure.

In the Central Tablelands context, the presence of earthworms will be important. Lee and Foster (1991) provide an extensive account of the benefits that earthworms create for soil hydraulic properties, particularly due to the construction of continuous and connected macropores, more so from those earthworm species that burrow to the soil surface. Ants and termites have been associated with improved soil physical quality in drier climates (Andersen 1990, Lobry de Bruyn 1999).

The relationship between soil organisms and plants is also of some interest, particularly if beneficial interactions can be advantaged. For example, root infection by vesicular-arbuscular mycorrhiza (VAM) has been linked to nutrient uptake and other processes in plants (Read 1993), and staining techniques are available to measure the extent of infection (Grace and Stribley 1991). In one study, Bethlenfalvay et al. (1985) found significant differences in VAM colonisation of a certain forage species (crested wheatgrass, *Agropyron desertorium*) in central Nevada, USA, when subjected to increased grazing pressure, and that the consequential effects on plant productivity and soil structure need to be investigated. However, it is unlikely this will be a significant indicator of soil quality in soils carrying conventionally fertilised pastures supporting introduced species.

2.5 Evaluation and selection of soil quality indicators - what is 'good' soil quality?

Walker and Reuter (1996) identify two groups of indicators – those that describe soil condition, whose values are not expected to change much from year to year, and those that describe biophysical trend, whose values are expected to vary seasonally and where more regular sampling will be required. In regard to key indicators, Walker and Reuter (1996) summarise the literature of the time, and establish the following criteria for their selection, that are useful to multiple users of the information and where a rank can be applied to each indicator (high, medium, low):

1. Ease of Capture
2. Total Cost
3. Standard Method of Estimation
4. Interpretation Criteria Available
5. Significant at this Scale
6. Low Measurement Error
7. Known Response to Intervention
8. Stable Measurement
9. Mappable Trend
10. Value as Diagnostic Tool (generic v's diagnostic)
11. Context Data Available

Similar criteria are nominated by Oakley (1995) and Doran and Parkin (1996).

Table 2.3 summarises the soil quality indicators assessed in Walker and Reuter (1996); those listed in bold are nominated by Walker and Reuter (1996) as key indicators of catchment health. Their recommendation that a morphological indicator of soil structure not be selected as a key indicator is noted. Walker and Reuter (1996) go further and summarise the expert opinion of the participants of a national workshop and define the generic threshold values for several key indicators of soil quality, listed in Table 2.4, whilst acknowledging the importance of local adjustment to these values. For two additional soil properties, Walker and Reuter (1996) provide descriptive analysis to rate soil quality criteria, including soil colour as an indicator of the drainage status of a soil (Table 2.5).

Table 2.3

Key soil quality indicator selection criteria nominated by Walker and Reuter (1996), with the usefulness of the indicator ranked as H = high, M = medium and L = low for criteria 1- 9 and 11. For criterion 10, G = generic, D = diagnostic. Those in bold type are recommended as key indicators. Refer to text p34 for criteria descriptions.

Category	Indicator	Criterion										
		1	2	3	4	5	6	7	8	9	10	11
Physical	**Slaking & Dispersion**	**H**	**L**	**H**	**H**	**H**	**L**	**H**	**H**	**H**	**D**	**H**
	Drainage status	M	M	H	H	H	L	L	H	H	G	H
	Water repellency	H	M	H	H	H	L	H	H	H	G	M
	Aeration/plant water	M	L	H	H	M	M	H	H	H	D	H
	Sodicity v's salinity	L	M	H	H	H	L	H	H	H	D	H
	Soil strength	**H**	**L**	**L**	**L**	**L**	**H**	**H**	**H**	**H**	**D**	**L**
	Water intake rate	**H**	**L**	**H**	**H**	**L**	**H**	**H**	**HL**	**L**	**D**	**H**
Morphological	**Consistence**	**H**	**L**	**H**	**H**	**H**	**L**	**M**	**H**	**H**	**GD**	**H**
	Colour	**H**	**L**	**H**	**M**	**H**	**M**	**L**	**H**	**H**	**GD**	**H**
	Roots	**H**	**L**	**H**	**H**	**H**	**L**	**M**	**H**	**H**	**GD**	**H**
	Structure	M	M	M	H	M	M	M	H	H	GD	M
	texture	**M**	**M**	**H**	**M**	**M**	**M**	**L**	**H**	**H**	**GD**	**H**
Chemical	**pH**	**H**	**L**	**H**	**H**	**H**	**H**	**M**	**H**	**H**	**GD**	**H**
	Soil redox status	L	?	H	M	H	M	M	H	M	G	L
	%C	L	L	H	M	H	H	L	H	H	G	M
	CEC	L	M	H	M	L	H	M	H	H	GD	L
	AEC	L	M	H	M	L	H	M	H	H	GD	L
	Soil oxides	L	M	H	H	M	M	L	H	M	G	L
	Clay type	L	M	H	L	H	M	L	H	L	G	L
	Total N	**L**	**M**	**H**	**L**	**M**	**H**	**L**	**H**	**H**	**G**	**L**
	Nitrate N	**L**	**L**	**H**	**M**	**M**	**H**	**H**	**H**	L	**D**	L
	Total P	L	M	H	L	M	H	L	H	H	G	L
	Extractable P	**L**	**L**	**H**	**H**	**M**	**H**	**L**	**H**	**H**	**D**	**H**
	Exchangeable Na or K	**L**	**M**	**H**	**L**	**M**	**H**	**M**	**H**	**H**	**D**	**L**
	ESP	L	M	H	M	M	H	M	H	H	GD	M
	EC	**H**	**L**	**H**	**H**	**M**	**H**	**H**	**H**	**M**	**D**	**H**
	DTPA extractable trace elements	**L**	**L**	**H**	**H**	**M**	**H**	**M**	**H**	**H**	**D**	**M**
	Extractable boron	L	M	H	H	L	H	M	H	M	D	M
	Total toxic metals	L	M	H	M	L	H	L	H	H	D	M
Biological	Respiratory activity	M	M	H	M	M	M	H	H	H	G	M
	Microbial biomass	M	M	H	M	M	M	H	H	H	G	M
	Cotton strip assay	**H**	**L**	**H**	**M**	**M**	**M**	**H**	**H**	**H**	**G**	**M**
	Nematode species assemblages	L	H	L	L	M	M	H	H	H	G	L

Table 2.4
Classification and threshold values for selected key indicators of soil quality (after Walker and Reuter 1996)

Soil property		Classification				
		very good	good	fair	poor	very poor
Soil pH		6-7	5.5–7.5	7.5-8	5–5.5 or 8–8.5	<4.5 or >8.5
Soil EC (dS/m)	sandy soil	<0.15	0.16-0.30	0.31-0.60	0.61-1.20	>1.20
	sandy clay loam	<0.25	0.26-0.45	0.46-0.90	0.91-1.75	>1.75
	heavy clay	<0.40	0.41-0.80	0.81-1.60	1.61-3.20	>3.20
Total N (%, 0-10 cm)	sandy		>0.10	0.06-0.10	<0.06	
	loamy		>0.20	0.15-0.20	<0.15	
	clayey		>0.25	0.18-0.25	<0.18	
Colwell P (ppm, 0-10 cm)	sandy		>25	20-25	<20	
	other		>40	30-40	<30	
Exchangeable K (ppm, 0-10 cm)	sandy		>150	100-150	<100	
	clayey		>300	150-300	<150	
DTPA trace elements (ppm, 0-10 cm)	Cu		>0.5	0.3-0.5	<0.3	
	Fe		>10	5-10	<5	
	Zn		>1.5	1-1.5	<1.0	
	Mn		>5	1.5-5	<1.5	
Effective root depth (m)		>1.0	0.75–1.0	0.5–0.75	0.25–0.5	<0.25
Water intake rate (mm/hr)			>70	30-10	<10	
Penetration resistance at field capacity (MPa)			<1		>1	
Penetration resistance at wilting point (MPa)			<3		>3	
Bare soil (%)		0-5	5-10	10-30	30-70	>70
Cotton strip assay (days to 50% loss of tensile strength at 20°C)			5-12	>15	>25	

In regard to soil carbon, difficulties in reliability, interpretation and large temporal and spatial variability have been discussed elsewhere, leading to reluctance to nominate threshold values of soil carbon or organic matter values. However, given the significance of soil carbon in the assessment of soil quality, and its relationship to total N and soil biological condition, some measure of soil carbon is recommended by Merry (1996) and Reuter (1998).

Table 2.5
Descriptive classification of soil colour and aggregate stability
(Walker and Reuter, 1996)

	good	fair	poor
Soil colour	Uniform; bright; red or yellow	Mottled; mixture of bright, dull and dark; red or yellow, with blotches of grey or blue or smelly-black	Uniform, mottled or stained; dull or dark; grey, blue, or smelly-black
Slaking and dispersion	Sample floats or sinks, no slaking after 2 hours; no or partial dispersion after remoulding	Sample sinks and slakes slowly; complete or partial dispersion without remoulding	Complete immediate slaking; complete or partial dispersion without remoulding

In regard to biotic indicators of soil health, King and Pankhurst (1996) summarise the features of several indicators and conclude that Cotton Strip Assay is the only indicator that meets the criteria for generic use. Additional measurements or observations are also recommended in a generic monitoring scheme: extent and duration of waterlogging, surface condition (crusting, pugging, sealing), percentage of ground cover as a measure of soil erosion risk, and actual soil loss from erosion events.

2.6 Pasture management and soil quality

The positive relationship between pasture perenniality, grazing tactics, livestock performance and soil surface protection is well known. Lang (1979) establishes the 70% rule for the percentage of ground cover to minimise erosion. Kemp et al. (1990) describe the impact of controlled grazing on pasture botanical composition from experiments in the 1980s. However, recent investigations have revealed continuing difficulty in optimising the design of grazing systems, perhaps best reflected in the long term decline in perenniality and pasture stocking rates on the central and southern tablelands of NSW (Kemp and Dowling 2000, Michalk et al. 2003, Dowling et al. 2005).

Mason and Kay (2000), in summarising the core problems underpinning the Temperate Pasture Sustainability Key Program of Meat and Livestock Australia, also describe problems with the sustainability of these pastures, stemming from a shift in research to species and cultivar selection and high input management – that most pastures are severely degraded, and that newly established pastures deteriorate faster than the time needed to repay establishment costs. Grazing tactics that improved the longevity of perennial pastures became the focus of this program.

Traditional grazing methods such as set stocking, where a fixed number of livestock are held continuously in each paddock, allow livestock to consume palatable pasture species selectively. Over time, detrimental changes to pasture botanical composition will occur, even with conservative stocking intensity. Rotational grazing will advantage the perennial component of the pasture as a result of rest between defoliation events, increasing pasture productivity and persistence and altering the local water balance (Kemp et al. 2000).

Avery (1995) recognised that grazing management will influence pasture botanical composition, but specific 'rules' for grazing management for different pastures over the range of seasons, were not determined at that time. Kemp et al. (2000) conclude that continuous grazing (i.e. set stocking) has a detrimental impact on the perennial grass component of pastures, either by a reduction in the total perennial grass component over time, or as a relative change in botanical composition. Strategic rest from grazing was described as a key management tool to manipulate pasture botanical composition, and 'rules' for timing the rest period were presented. Sheath and Clark (1996) agree that rotational grazing has some advantages during periods of pasture surplus, agreeing that higher stocking densities reduce preferential grazing. Greenwood et al. (1997) also observed large differences in botanical composition between set stocked treatments of different stocking rates but did not quantify these differences.

Kemp et al. (2000), in summarising the results from numerous experiments in the Temperate Pasture Sustainability Key Program, are clear when they assert that a strong perennial grass component in this environment increases the ecological value of pasture by reducing weed populations, increasing water use by pasture, and capturing more nitrate from depth compared to annual pasture species. They note that despite this generalisation, it is not practical or reasonable to maintain productive pastures with just perennial grasses. Further, they concede that even with a strong perennial grass component, some 'leakage' of water and nitrate still occurs, and that processes such as acidification will still occur, all-be-it at a reduced rate.

Mele et al. (1996) also describe a general reduction in pasture productivity associated with a change in botanical composition from native perennial to annual species, brought about by changes in farming systems including grazing management. In south eastern Australia, this transition has also been associated with degrading soil quality, as indicated by soil acidification, less efficient water use and nutrient limitations, observations confirmed by Kemp and Dowling (2000). This represents a difficult cycle of decline: that declining pasture productivity due to a reduction in the perennial grass component of the pasture is contributing to the factors which are accelerating the decline.

In one study, White et al. (2000) demonstrate that perennial pastures (in this case grown in a Red Kurosol in southern NSW) created a reduction in deep drainage ranging between 20 and 29 mm/year compared to annual pastures, with a corresponding reduction in solution nitrate and consequential reduction in rate of acidification. Ridley (1996) measured 50 mm more water use under perennial grasses compared to annual grasses, with deep drainage calculated to be 10 mm less per year.

Decreased water use by annual pastures makes a direct contribution to accelerated soil acidification, a significant environmental issue with resulting loss in agricultural production estimated to exceed $134m annually in the high rainfall zone of south east Australia ($100m annually in NSW; White et al. 2000). The area affected by soil acidification is expected to increase 5-fold by 2010.

Ridley (1996) also provides evidence of perennial pastures improving soil structure compared to annual pastures. In an experiment at Rutherglen, northern Victoria, on a Red Chromosol, unsaturated hydraulic conductivity at the soil surface (measured at -10 mm tension) was significantly greater under perennial pasture species (mean of 28 mm/hr) than annual species (mean of 19 mm/hr). Ridley attributes this to differences in root characteristics. Specifically, the perennial pastures had roots greater than 1.0

mm diameter to 0.7 m depth, whereas the annual pastures had no roots of this diameter, although both pastures had similar root densities.

Although not quantified, Willat and Pullar (1983) observed that grasses (mainly perennial ryegrass) dominated the botanical composition of pasture in control plots where livestock were excluded, with a mixture of clover and grass on the grazed plots. The removal of part of the grass herbage by livestock appears to have encouraged the production of clover, probably because grazing of tall herbage allows better penetration of sunlight. "Obvious" differences were also observed in the length of plant material, shorter on grazed plots due to livestock grazing and hoof impact, and this was assumed to create differences in root distribution. The amount of litter will make a significant contribution to the carbon input in pasture systems (Robertson et al. 1995, Schuman et al. 1999). Avery (1995) also says that grazing management will influence pasture water use because of its indirect effect on leaf area, temperature of the microenvironment and root distribution.

The positive effects of increased macroporosity following perennial pasture, including phalaris, are confirmed by McCallum et al. (2004). Following measurement of macropores on a Sodosol in southern NSW, they confirm the role of perennial pastures in ameliorating subsoil density by observing that subsoil (deeper than 12 cm) under phalaris contained greater total porosity than soil under annual pasture and lucerne. It is noted that although subsoil under phalaris and lucerne had similar numbers of pores > 2 mm, the subsoil under lucerne contained the greatest density of pores greater than 4 mm.

Perennial grasses produce large quantities of fine roots, capable of penetrating small fissures characteristic of hard soil compared to tap roots of certain crop and broadleaf species. For example, Cresswell and Kirkegaard (1995) cast doubt over the ability of tap-rooted crops such as canola to create new macropores in dense subsoil. Their reputation for this ability appears to be based on tap roots occupying existing macropores. They suggest that perennial species such as lucerne may be more effective because of the longer time and wider range of soil water conditions in which to establish a root system.

The density, activity and subsequent decay of pasture roots provides some resistance to soil structural decline, and opportunity for structural recovery (Greenwood and McKenzie 2001), and the associated microbial activity will encourage greater aggregate stability under pasture compared to cropping systems, particularly where tillage is employed in cropping (Tisdall and Oades 1982). Grace et al. (1994) summarise the results from several studies to show that soil total C and N, and microbial biomass C and N, all increase, or at least remain stable, under pasture compared to cropping systems, with benefits to soil aggregation and stability also noted from a pasture phase within a crop-pasture rotation (Chan and Pratley 1998). The continuous supply of organic residues, as might be expected from active pasture systems, is necessary to maintain structural integrity in soils where organic matter is the primary stabilising agent (Skjemstad et al. 1998).

A popular standard for minimum ground cover is 70%, based on the recommendations of Lang (1979), but this relates primarily to protection of soils against erosion. More recently, biomass levels of between 0.5 and 1.5 t DM/ha, depending on species, have been identified as critical lower levels for perennial species survival (Kemp et al. 2000), and levels of between 2 and 4 t DM/ha for maintenance of pasture species diversity (Kemp et al. 2003). These set the limits below which overgrazing occurs. Overgrazing will inhibit the ability of pasture plants to regenerate roots, reducing soil organic matter contributions as well as vegetative productivity.

Dorrough et al. (2004) plead that botanical biodiversity also be considered in the context of improved grazing management of grasslands. This is noted, but so too is the report by Mikhailova et al. (2000), measuring greater species diversity in certain pastures in Russia grazed and periodically cut for forage, compared to native grasslands and pasture cut yearly and grazed. The work of Kemp et al. (2003) confirms that a herbage mass greater than 4 t DM/ha may inhibit pasture species diversity and persistence.

2.7 Impacts of livestock, stocking rate and grazing strategies

Compaction pressures under livestock can be substantial, and this contributes directly to smaller macroporosity and loss of pore continuity, with increases in bulk density and soil strength commonly reported (Packer 1988, Greenwood and McKenzie 2001). Consequently, alterations to soil hydraulic properties are expected and similarly reported. Greenwood et al. (1997) found that loss in porosity after grazing by sheep was due to a decline in the number of macropores greater than 1.2 mm equivalent cylindrical diameter. Soils high in organic matter, with high aggregate stability and containing a large quantity and density of roots will be more able to resist compaction pressures and recover more quickly from them (Greenwood and McKenzie 2001). These conditions are best met with soils under active pasture growth, and prevented from overgrazing.

There are many studies demonstrating that higher stocking rates can be more detrimental to soil physical properties compared to lower stocking rates, but many of these are of short duration, often focusing on impacts at large moisture contents when soil damage is almost certain, and with set stocked plots. The reviews by Packer (1988) and Greenwood and McKenzie (2001) revealed few studies comparing alternative grazing tactics (e.g. comparing set stocked grazing management to various forms of rotational grazing, at equivalent stocking rates), particularly in Australian conditions.

Willat and Pullar (1983) showed that increased (sheep) stocking rate, albeit at what is assumed to be set stocked grazing management, resulted in an increased soil bulk density, increased penetration resistance and reduced hydraulic conductivity. These differences became significant, with results derived from two replications, when the stocking rate exceeded 15 sheep per hectare. Although not quantified, some changes to pasture botanical composition were observed, with grasses (perennial ryegrass in this case) dominant in ungrazed plots and mixed (white clover / perennial ryegrass) pastures in grazed plots. The duration of the experiment and the timing of soil measurements were not reported. The soil consisted of silty loam topsoil, with the experiment conducted in western Victoria.

Excretal returns can contribute to maintenance of soil quality, and are an integral part of pasture grazing. The grazing and camping behaviour of livestock can cause a redistribution of nutrient and organic matter contained in both dung and urine (Packer 1988), with low stocking intensity more likely to cause redistribution because of preferential grazing, and the impacts of walking tracks and camps. The location of watering points, shade, shelter and fences are also factors.

The increasing evidence that rotational grazing supports greater persistence and productivity of perennial pastures, and can contribute to beneficial changes in botanical composition, combined with the potential environmental benefits this may bring, leads to speculation that rotational grazing may also benefit soil quality. In addition, the rest period associated with rotational grazing is likely to be beneficial to soil structural recovery if plant root growth occurs, partly a function of the duration of the rest period. Michalk et al. (2003) conclude that tactical grazing (defined as deferral of grazing at critical pasture growth stages combined with changes to stocking rate), compared to continuous grazing, increases pasture perenniality and reduces annual grass weeds. There is no direct evidence to state the desirable period of rest from the perspective of soil structural recovery, although in a study of seasonal grazing effects by cattle on riparian pastures, Wheeler et al. (2002) observe that infiltration rate declined and bulk density increased following grazing, but that both soil properties returned to pre-grazing values within one year after removal of livestock.

This conclusion is not supported by certain other authors. For example, in an experiment based on rotational grazing, Warren et al. (1986) claimed that even with rotational grazing, infiltration rate decreased significantly and sediment production increased with increased stocking rate on a silty clay surface soil, and that recovery of these properties was not achieved after 30 days rest, results that are not consistent with measurements under HI-SD grazing management. However, the experiment of Warren et al. (1986) was conducted on bare soil only, with the experimental area sprayed with herbicide during the previous spring to deliberately remove vegetation and "remove the confounding effects due to variability in vegetative cover and botanical composition".

Clearly, the benefits to soil physical properties are largely derived from the growth and subsequent decay of plant roots. With the removal of plants from the grazing experiment, the results of Warren et al. (1986) only serve to emphasise the impact of livestock hoof pressure on soil properties. The rest period described by Warren et al. (1986), without the presence of pasture plants, is not comparable to the rest period associated with any type of grazing management.

Kay (1990) provides little discussion on the impact of hoof pressure on soil structure in his treatise on soil structure, which focuses on crop production management, referring to the work of Warren et al. (1986) in USA for advice, described above and despite its limitations. In another example, Proffitt et al. (1993) found that pasture root growth, as measured by the 'root length density' in the soil (m of root length per m^3 of soil) reduced under set stocked grazing as a result of trampling during the grazing season, compared to ungrazed control. However, root samples were not collected for a deferred grazing treatment, thereby limiting the comparison. The soil was one susceptible to compaction, grazed as soon as one month after cultivation, during wet winter conditions.

Grazing intensity, usually defined by stocking rate, will influence the degree of impact. Lawrie et al. (2000) averaging the data of Geeves et al. (1995) from 75 sites in southern NSW and northern Victoria, identify that 'heavy' grazing resulted in significantly lower organic carbon content and surface infiltration at -10mm tension compared to 'medium' grazing. They found that surface infiltration rates were as small and organic carbon contents were approaching those of soils under conventional tillage for crop production. Under 'light' grazing, these two soil properties approached levels of undisturbed woodland, sometimes considered a benchmark condition.

Pratley and Robertson (1998), in summarising the work of others although none from NSW, also list a number of studies comparing hydraulic properties of topsoil subject to livestock grazing to that of adjacent 'undisturbed' land. Reductions in infiltration rate or saturated hydraulic conductivity of up to an order of magnitude are reported, to as low as 5 mm/hr, although grazing does not always reduce infiltration rates. For example, Graetz and Tongway (1986) measured a three-fold increase in infiltration rate between ungrazed and grazed soil in an arid zone experiment, mainly due to the disruption of a lichen crust on the soil surface by livestock traffic.

Greenwood et al. (1997) measured significant differences in soil physical properties between grazed (by sheep) and ungrazed pasture plots, as measured by unsaturated hydraulic conductivity, bulk density (0 – 80 mm) and soil penetration resistance. They found no difference between low, medium and high stocking rates after 30 years of grazing, emphasising that the presence of livestock at even low stocking rates will impact on soil physical properties. It was concluded that livestock effects are cumulative over time, a claim that is difficult to interpret, and that soil physical properties are unaffected by stocking rate in the long term. However, the different stocking rates were all conducted by set stocked management, there were some reductions in stocking rate during the grazing period and no comparison was made with rotational grazing. They did acknowledge that maintenance of pasture 'cover' should remain a high priority for the management of soil erosion, and that stocking rate will impact on pasture botanical composition.

Packer (1988) cites several studies relating grazing intensity (taken to be stocking rate) to reduction in soil fauna activity, concluding (as do Greenwood and McKenzie 2001) that management practices which influence soil water content, soil temperature and soil organic matter, including grazing practices, will influence soil fauna diversity and abundance. The potential impacts of grazing practice on earthworms are described by Fraser (1994), indicating that earthworm density is correlated to pasture production and 'carrying capacity' (although this is not defined precisely) due to the availability of organic matter under pasture, particularly where clover species increase the availability of nitrogen. Fraser (1994) and Baker et al. (1997) claim that grazing effects that reduce pasture productivity will therefore influence earthworm density, at least in high rainfall improved pasture systems where earthworms are likely to be present. These effects include overgrazing and compaction caused by trampling. Proffitt et al. (1995b) confirm that continuous grazing reduces faunal macropores, particularly when the soil is grazed when wet and subject to plastic remoulding.

King (1996) explains that overgrazing reduces the thickness of the litter layer, therefore impacting on organic food supply for soil organisms, the habitat for organic matter decomposers and influencing soil microclimate (including the temperature of the soil surface and soil water content as a result of changed rates of evapotranspiration). These factors reduce the numbers of soil fauna in pasture soils. King (1996) believes that because moderate grazing intensity associated with 'improved' pastures (i.e. pastures with non-native species and applied fertilisers) has little impact because of the ability of the pasture to maintain organic matter levels, although 'moderate' is not quantified. King (1996) does not refer to alternative grazing strategies, although it is assumed that grazing strategies which preserve the litter layer would be preferred.

Dormaar and Willms (1998) included a large number of soil properties in their study of the effects of different grazing intensities on soil quality in fescue grasslands in Alberta, Canada. This was not a replicated experiment, but appears useful because the grazing treatments were initiated in 1949. Bulk density and water stability of aggregates were used to measure soil physical characteristics, and significant differences were measured between treatments. Of interest is the use of a number of biological measures, including root mass and distribution, soil monosaccharide content (a measure of the supply of an energy source to soil microbes) and certain enzymes associated with microbially originated monosaccharides, finding that these were sensitive to grazing intensity. Their conclusion included recommended threshold grazing intensities, above which grazing was not considered sustainable.

In one of the few studies into rotational grazing effects, Rodd et al. (1999), using penetration resistance to measure compaction, found that cattle compacted soil at a range of stocking rates, more so in the top 6 cm, compared to the same pasture/soil that was not grazed but harvested for forage (no wheel traffic). No significant difference in penetration resistance was found with increasing rest interval between grazing events, although the maximum rest interval was only 6 weeks. In this work, pastures were based on alfalfa (lucerne; *Medicago sativa*). An additional 4 week rest interval at the critical spring timing also showed no significant difference between stocking rate treatments. No effect was observed in measurements of field saturated surface and subsurface hydraulic conductivity as measured by a Guelph permeameter. The soil type under investigation was a red brown loam of low organic matter content and weak soil structure. Frost action was claimed to alleviate increased penetration resistance. Soil bulk density was found to be less sensitive as a measure of compaction than measurement of penetration resistance.

Tollner et al. (1990) also found that infiltration rate and bulk density measurements were only sometimes significantly influenced by cattle (although the class of livestock is not specified, it is assumed they were bovine from the animal weights reported). Using alfalfa (lucerne) based pastures, penetration resistance was significantly greater in grazed areas compared to soil within grazing exclosures, although the trend for bulk density was not as clear. A reduction in macropore space was observed over time, as measured by the water retention relation, but there were no significant differences at any particular sampling date between grazed and ungrazed soil. A type of rotational grazing system was deployed: 2 weeks grazing at 18.5 animals per ha then 4 weeks rest, followed by seasonal winter rest. Pastures within the exclosures were clipped to simulate pasture defoliation by livestock but without hoof pressure. Tollner et al. (1990) also believe that seasonal freeze-thaw may ameliorate compaction effects, and that increased compaction in ungrazed exclosures can be explained by "natural densification".

When soil water contents are large, soil damage associated with pugging and puddling is certain (Betteridge et al. 1999, Singleton and Addison 1999). Proffitt et al. (1995a) showed that the presence of livestock on soils with a water content near or exceeding the plastic limit can have immediate and significant impacts on topsoil structure. It appears that the plastic remoulding of wet soil under hooves will create more severe and rapid deformation than mechanical compaction of dry soil and consequential loss of macropores. Lobry de Bruyn and Kingston (1997), in a comparison of the timing of grazing of summer irrigated and non-irrigated pastures, found a reduction in surface infiltration rate under irrigated plots compared to dryland plots, obviously due to livestock hoof pressure under wet soil conditions, but also found no differences in other measures of soil quality (bulk density and percentage of water stable aggregates in this case) in the top 100 mm.

In an unreplicated experiment, Proffitt et al. (199a5) compared set stocking with a style of controlled grazing management that removed stock temporarily (varying from 2 to 7 days) following significant rainfall, defined as sufficient to increase soil water content to the plastic limit. The site was located on a Western Australian red duplex soil type (assumed to be a Chromosol) with 307 mm winter dominant annual rainfall.

Using disc permeameters on only two measurement dates, one before and one after grazing, Proffitt et al. (1995a) measured non-significant reductions in unsaturated hydraulic conductivity (at -10 mm tension) between set stocked and controlled grazing methods: from 20.3 ± 10.2 mm/hr to 11.5 ± 4.2 mm/hr in a set stocked plot, compared to a reduction from 22.1 ± 8.4 mm/hr to 15.2 ± 3.9 mm/hr under controlled grazing. For an ungrazed but mown pasture plot, the reduction was from 24.9 ± 9.3 mm/hr to 19.9 ± 5.6 mm/hr. In the same experiment, Proffitt et al. (1995a) did measure significant increases in mean bulk density, but increases were measured for all treatments including for soil under clipped but ungrazed pastures.

The duration of a grazing or pasture investigation is important. To capture the long term effects of plant root development, sufficient time is necessary to enable roots to develop through the rhizosphere, to occupy crevices and channels, then decay to form macropores. Francis and Kemp (1990) found that the macropores under pasture were mainly biogenic pores or inter-aggregate fissures, and that these increased or became more developed with time under pasture, quite rapidly after cropping. These contributed greatly to measured differences in soil hydraulic properties, with 2.5, 5 and 14 fold increases in the amount of water infiltrated under the 2, 4 and 35 year pastures respectively compared to cultivated soil. Consequently, the analysis of soil properties following a short period of grazing is unlikely to highlight all potential differences. The data reported by Proffitt et al. (1995a) follow a single grazing period of 17 weeks.

Proffitt et al. (1995b) also used image analysis (2 samples only per grazing treatment) of vertical soil surfaces to visually appraise macroporosity (defined as macropore width greater than 0.195 mm, although the selection of this pore size was not explained) at the cessation of grazing. For the upper topsoil layer (0 – 45 mm depth but excluding a dense crust of non-defined thickness that occurred for all treatments), macroporosity for the set stocked treatment was 5 ± 5 %, compared to the controlled grazing treatment at 25 ± 5 % and the ungrazed control at 20 ± 5 %. Macroporosity for the lower topsoil layer (45 – 100 mm depth) under set stocked grazing was < 2 ±

2% compared to 5 ± 2 % for soils under controlled grazing of pasture and the mown but ungrazed control, providing additional evidence of the impact of grazing tactics on macropore continuity.

2.8 Soil quality modelling

Most studies reported here specify the actual or expected change in the value of selected soil quality indicators. However, the development of a soil quality model(s) is likely to have merit. It could be applied to the assessment of land management practices, the development of land management policies, the rating of land for production or conservation purposes, and for allocation of financial resources (Parr et al. 1992). The integration of soil quality into economic models would be useful in economic analysis of agricultural systems and related policy (Jaenicke and Lengnick, 1999). The use of models for these purposes necessarily requires a judgement about soil quality that is often beyond the scope of the measurement of its component parts. Sojka and Upchurch (1999) warn of the dangers of this development, and the risks of extrapolating scientific data into policy determination.

Jaenicke and Lengnick (1999) summarise some of the research into soil quality models, categorising them into two types:

- static models, where individual soil quality attributes are aggregated into a soil quality index to model soil quality at a single point in time, and
- comparative-static models, which attempt to model the change in soil quality under some management regime over time.

They highlight the need to apply some value weighting to the component attributes, and that this has been a weakness in such approaches applied to soils. They attempt to merge these research efforts with research into economic models of system productivity and efficiency, with initial soil quality as an input and final soil quality as an output, where indices can be decomposed into various components. This provides an interesting perspective, but is based on economic theory beyond the scope of this investigation.

A number of field kits or guides are available to farmers to assist self-assessment of soil quality, often as part of a larger assessment of sustainability. NSW Department of

Primary Industries includes aspects of soil quality assessment in its home study program (NSW Agriculture n.d.). Liebig et al. (1996) evaluated a United States Department of Agriculture field kit and concluded that although there were significant differences between kit and laboratory measurements for some parameters, the kit provided a useful screening tool if repeated measures were conducted. Romig et al. (1996) describe a scorecard system for farmers to monitor trends in soil quality on their farms. Cooperation between farmers, researchers and agency personnel in the development of conservation approaches to land management is a feature of the Illinois Soil Quality Initiative, described by Wander et al. (2002).

Walker et al. (1996b) have designed their 'report card' approach such that landholders can collect data without substantial expense or sophisticated equipment, and the pooling of data, such as from members of Landcare groups, adds to the catchment database. Hulugalle et al. (1999) use the 'report card' approach combined with a simple scoring index to measure a matrix of soil physical and other properties to compare soil quality under lucerne to that under adjacent cotton crops, concluding that soil quality after 4-5 years of lucerne is better than under cotton, at least for Vertosols. The scoring index, however, applied equal weighting to all soil properties, implying that all soil properties are equally important to soil function.

The SOILpak scoring system for soil structural form (McKenzie 2001a), which has been found useful by farmers and advisors, provides a semi-quantitative approach to the assessment of various elements of soil quality. It combines field observation of the various descriptors of soil structural form with a weighting determined by professional judgement. McKenzie (2001b) found good correlation between the SOILpak score and other measures of soil structure in a Vertosol under cotton.

Some authors advocate the use of a single index for soil quality that will permit a numerically comparative and dynamic analysis. Granatstein and Bezdicek (1992) describe the use of an integrating index, to help evaluate the interactions between physical, chemical and biological parameters that determine soil quality. Larson and Pierce (1994) describe the possible application of statistical quality control procedures. Pierce (1996) proposes a soil quality control model derived from similar models in industrial contexts. Harris et al. (1996) describe the use of scoring functions, and Doran and Parkin (1994) the use of yet another soil quality index. Dexter (2004) proposes that because the shape of the water retention curve is affected by soil structure, as influenced by management of a particular soil, the slope of the retention curve at the point of inflection is a useful indicator of soil physical quality. This was tested for a number of soil types, and proved able to quantify the relative differences in soil quality according to the degree of compaction, the amount of organic matter and the ability for root penetration. However, two features of this work need consideration. Firstly, the analysis was partly based on the use of pedotransfer functions for a limited number of soil properties to predict the water retention characteristic. Secondly, the analysis was focussed on arable soils without high clay content, and likely to be limited in comparing subtle differences on soil management associated with grazing management of soils under perennial pasture.

A model which calculates a single soil quality value or score is only relevant to the context of the investigation or observation, previously established as a necessary condition for the concept of soil quality to be valid and useful. Walker and Reuter (1996) advocate comparison of individual indicators against a threshold limit in preference to the adoption of an index. This enables key indicators to be evaluated on their merits and not be diminished or disguised by incorporation into an index.

In an alternate approach, Gomez et al. (1996) describe a Framework for Evaluation of Sustainable Land Management at the farm level, and its application to comparative assessment. They allocate indicators into two groups; those that contribute to farmer satisfaction, and those that provide for resource conservation. A radar graph is used to give a visual representation of relative sustainability (Figure 2.3). This approach scores each soil quality element relative to a benchmark value, but does not presume the relative worth of each element. Such an approach may have value at the farm level.

Smith et al. (1993) use multi-variable indicator kriging to generate maps of soil quality, or more specifically, the probability of a location meeting specified soil quality criteria. Individual soil quality indicators, measured at various locations, are allocated a value of 0 or 1, depending on whether they are below or above a threshold value. The threshold value is selected to suit the situation under investigation. A new data set is derived from the combination of values for the individual indicators, and a variogram of this data is developed to establish the probability of any location meeting the combined value. The technique allows for adjustment of critical threshold values over time or with land use, the incorporation of any number of indicators, weighting according to their importance, and flexibility regarding their compliance with local soil quality 'rules'. This approach has been used in the assessment of soil contamination risk, for example Cattle et al. (2002), where threshold values are known with more precision, based on toxic limits. The approach may be useful in regional assessment of soil quality and scenario planning, and the identification of areas at risk. However, the 'yes-no' allocation of a value to soil quality parameters of interest relative to a single threshold value remains a weakness.

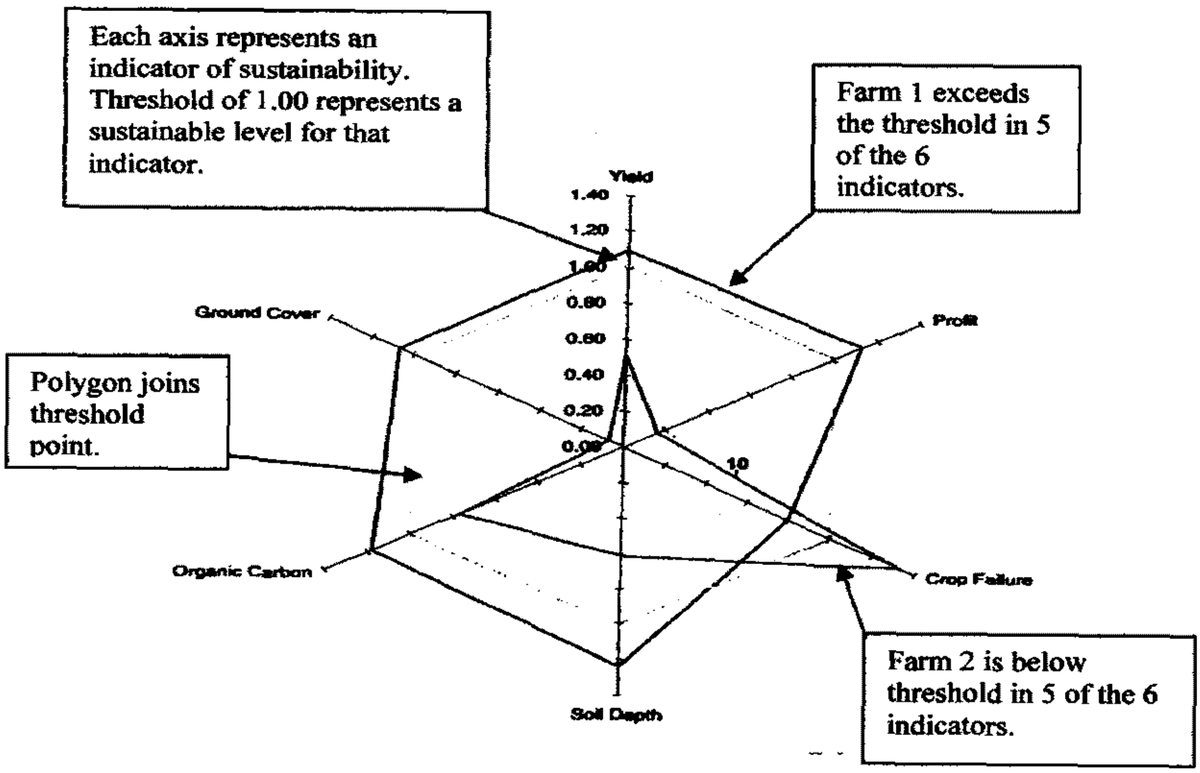


Figure 2.3
Radar graph to evaluate sustainability (Gomez et al. 1996). The shape of the polygon for a particular farm, relative to the threshold polygon, provides a visual appraisal of relative sustainability.

Fuzzy logic has been developed to assist decision making when imprecise data are subject to interpretation, and where the boundaries between data classes are indistinct; i.e. when data points can belong to more than one class to varying degrees. This is different from other forms of evaluation using multi criteria assessment. Although the name suggests otherwise, fuzzy logic is built on the sound theoretical base of fuzzy sets (Zadeh 1965). It is in widespread use in the manufacturing and process control sectors, and is beginning to be explored for agricultural applications, because it allows descriptive language to help design mathematical algorithms where professional judgment, expert opinion, and community positions can be included in addition to data from scientific experiments.

Until recently, scientific research has been built upon the principles of classical (Boolean) logic and statistical analysis. An assumption of this approach is that every proposition must be true or false (McBratney and Odeh 1997); i.e., an object can only belong to one class, and there is a distinct (crisp) boundary between classes. This assumption allows traditional statistical analysis to be performed, believing that uncertainty can be eliminated if there is sufficient knowledge of the system (Bardossy and Duckstein 1995). By taking sufficient measurements of the object, we can be confident it is or is not in a particular class.

As a simple example, a soil is considered acid if its pH is less than 7.0. Soil cannot be both acid and not-acid. However, plant response to soil pH occurs gradationally, so soil acidity is classified into grades or classes – strongly acid, slightly acid, neutral, etc.. However, the boundary between these classes is not particularly clear, because it depends to some degree on the purpose of the classification and the interpretation of the classifier, and the tolerance of the elements of the particular system (soil and plant) to acid conditions. There are also sampling issues and measurement error to contend with.

When complex 'problems' are encountered, particularly in biological and social sciences where system dynamics are complex and human values dictate decision-making (this includes soil management), applying classical methods presents some difficulties. In order to cope with variability, repeated measurements are required, so the more complex the issue, the more resources are required to maintain statistical rigour. Randomness is assumed to be the only source of uncertainty so that probabilistic analysis can be undertaken. The inherent variability in natural systems is masked or limited by the deliberate selection of repeatable treatments. This makes the results statistically valid but not necessarily relevant to the 'real' world. The temptation to modelers is to add further levels of detail to their models, in an attempt to provide a better match. However, this further increases data requirements and processing resources.

Although difficulties with the assumptions of Boolean logic have been noted since its inception (McBratney and Odeh 1997), mathematically they were difficult to realise. In recent decades, a theory of fuzzy sets (more correctly, subsets) emerged (Zadeh 1965), and the development of fuzzy logic. The theory of fuzzy sets is described elsewhere (e.g. Klir and Yuan 1995, Li and Yen 1995, Odeh and McBratney 2002).

The fundamental premise of fuzzy set theory is that an object belongs to a class to a degree, and so can therefore belong in part to different classes. The boundary between classes is no longer crisp, but fuzzy. The randomness of probability theory, where the passage of time and collection of additional data reduces uncertainty (e.g. it will rain tomorrow), is different from fuzziness, where ambiguity lies in the interpretation of words (e.g. raining heavily). Unfortunately, the choice of the term 'fuzzy' was not a particularly good one, as it tends to raise emotions amongst scientists and others more comfortable with traditional analysis. Despite this, it has been widely adopted.

The development of fuzzy logic accelerated to serve industrial applications, particularly in the field of artificial intelligence, the attempt to model human behavior and decision-making. Humans make decisions based on a range of knowledge, perceptions and values, many of which are described in linguistic terms rather than numbers, well suited to interpretation by fuzzy methods. The industrial applications of fuzzy logic, such as process control (taking the place of human operator judgment), 'smart' appliances (that 'learn' the settings to provide human satisfaction), and robotics (where machines simulate human response) are well documented (e.g. Terano et al. 1987, Yager and Filev 1994, Bardossy and Duckstein 1995). Fuzzy analysis has also found some support in management and behavioural sciences (Zimmermann et al. 1984, Smithson 1987).

In an agricultural context, 'problems' are characterised by multiple decision factors and criteria, and dynamic biophysical and business environments. Some of these have known response functions (e.g. crop response to fertiliser), others depend on attitudinal factors (e.g. approach to risk), and others require forecasting ability (e.g. commodity prices). This complexity may be best modelled with fuzzy methods. Some current agricultural applications include: image processing for weed recognition in precision farming (Yang et al. 2000); detection of crop edge by machine vision for vehicle guidance (Benson et al. 2000); assessment of fruit quality (Jahns et al. 2001); and irrigation scheduling, incorporating farmers attitude to risk (Clyma and Martin 1996). Additional agricultural applications are described by Center and Verma (1998) and Odeh and McBratney (2002).

Bardossy and Duckstein (1995) compare fuzzy and crisp mathematical modeling, and list the following limitations of crisp techniques: available data are often insufficient; non-stationary and heterogeneous factors are difficult to model; can give a false sense of accuracy; more applicable to 'sterile' or idealised problems. They suggest that reality is more complex than the models considered in science, and provide the example of water resource management – one requiring the 'hard' science of hydrology together with the 'soft' disciplines associated with social and institutional factors in the evaluation of decision consequences. They also nominate some advantages of fuzzy modeling, suggesting that these compensate for a loss in computational accuracy: simplicity; computational speed; and flexibility to adapt to the available data (quality and quantity). It is suggested here that soil management modeling is equally complex.

Terano et al. (1987) also discuss the limitations of mathematical models used in science and technology, particularly where people or societies are the objects of the model. Complexity increases the number of assumptions required, and validity can be challenged where the problem is ambiguous. The quality of these assumptions determines the acceptance of the result. Large amounts of data are difficult and costly to collect, and probability theory is difficult to use under these conditions. They compare fuzzy modeling with other approaches, including models based on physical laws, statistical analyses of data from actual measured phenomena, structural models and predicate logic. They conclude that models based on fuzzy set theory are best suited for expressing the ambiguity of meaning found in language, where boundaries between classes are gradual, and in the imitation of human judgment.

Agricultural applications may therefore include human behaviour (e.g. making soil management decisions), or societal systems (e.g. catchment management). Center and Verma (1998) describe fuzzy logic as "a powerful concept for handling non-linear, temporally dynamic adaptive systems, which permits the use of linguistic values of variables, and imprecise relationships for modeling system behavior". Despite certain limitations, Riedler and Jandl (2002) used a fuzzy logic model to quantify soil degradation under Austrian forests, and support its usefulness as an aid to expert judgment in system management. There are a number of examples where fuzzy methods have been used in soil mapping and land evaluation, based on continuous classification (Triantafilis and McBratney 1993, Dobermann and Oberthur 1997, McBratney and Odeh 1997, Chatterji 2000).

These represent one dimension only of the complexity of farm management. For example, the decision of a farm manager to apply lime to an acid soil might be described as: "I will apply 2.5 t/ha of lime because the soil pH in the topsoil is 4.8". This may be the recommendation of the agronomist or lime retailer, and is easily modeled. But the decision is more complex than this. It is more likely to be: "I will apply some lime if the soil pH in the topsoil is low, if I have enough funds available, if my tax accountant agrees, if the price for the crop next year looks like it will be OK, if the weather forecast is good, and if the contractor to do it is available". Such a decision could possibly be modeled with mathematical equations to derive an answer, but it becomes too complex for what appears to be a simple decision. Note the use of linguistic terms such as "good", the value of which changes over time and with the mix of values of other variables, and which cannot easily be modeled with traditional methods.

Li and Yen (1995) describe an example from the textile industry, where assessment of 'quality', measured by numerous measurable quality standards for a range of fabrics, needs to be matched to customer perception of 'quality', often described in vague linguistic terms such as 'softness and feel'. Such an approach may have relevance to primary producers, and assist with the evaluation of quality standards for elements of the natural catchment, such as soil. The concept of soil 'quality' or 'health' is widely held by farmers, but criticised in the past for being non-scientific (Sojka and Upchurch 1999). Fuzzy methods are likely to be helpful in matching value judgements with scientific data (McBratney and Odeh 1997).

These examples apply at the farm scale. Fuzzy methods may have application to more complex concepts and larger scales, such as sustainable catchment management, where a catchment is an aggregation of individual paddocks and farms and where human values are influential. In such situations, variables are poorly defined and measured, and the quantity and quality of hard data is limited.

Ducey and Larson (1999) apply fuzzy set theory to the issue of sustainability and ecosystem health, with particular reference to forest management and policy development. They argue that fuzzy logic is a relevant and useful approach, given the ambiguity of the meaning of sustainability and the qualitative nature of its measurement, the multitude of interested parties, and the need to match management decisions to local conditions. They propose that "sustainability" is an inherently fuzzy concept, but that fuzzy logic can add rigour to the debate. Bardossy and Duckstein (1995) also refer to sustainability principles when describing reservoir operation policy designed with the assistance of fuzzy logic.

Silvert (2000) uses fuzzy logic in the evaluation of indicators of environmental condition, with reference to coastal zone water quality, and the impact of commercial aquaculture, and list a number of advantages. The process can utilise simple but less precise observations that can be made quickly and without instruments, thereby reducing data collection costs, increasing the frequency and quantity of observations and allowing inclusion of characteristics that are difficult to measure scientifically (e.g. smell). In their study, the method revealed perturbations that might have been otherwise obscured by masses of data. They have used a technique that avoids value judgements by the modeler, and that deals effectively with missing data, both inherent characteristics of many ecosystem surveys.

Freyer et al. (2000) revisit the notion of ecological risk analysis developed in the 1970s and 1980s to develop a 'potential impact model' of agricultural activity, using fuzzy logic to build an assessment algorithm and allocate potential impact values. Where the potential impact is high, there is a potential conflict between human activity and ecosystem function. Although not yet field verified, this may be applicable at farm and landscape scales, with some limitations; in particular, the averaging of farm outputs to equate to the landscape scale.

2.9 Conclusion

The generally accepted definition of soil quality relates to the functional roles of soil in the landscape (Carter 1996); i.e. as a medium for the physical, chemical and biological processes that support plant growth; in the partitioning of water flow through the landscape; and as a buffer for environmental change (National Research Council 1993). Consequently, the concept of soil quality includes physical, chemical and biological elements and their interdependence. Although there is disagreement about the scientific rigour associated with this concept and its application (Sojka and Upchurch 1999), and difficulty in applying a generic scheme at various scales of application (catchment, farm, paddock, etc.), it remains a useful platform or framework for investigating soil management options, partly because it is this interdependence that determines optimum soil management strategies. Soil resilience, defined as the capacity of a soil to recover its functional and structural integrity after a disturbance (Kay 1990; Seybold et al. 1999), is a related concept. Resilient soils may better tolerate detrimental management practices, provided a recovery phase or intervention is available.

There have been several attempts at quantifying a soil quality index to enable numerical scoring, which appear to hold value for policy makers, mapping applications and trend analysis at the catchment scale. For land managers, a soil quality approach is most useful if soil quality criteria are linked to soil type, land use and land capability. In this way, key performance indicators of condition and trend can be identified and monitored, and threshold values of 'good' or 'bad' can be nominated in conjunction with the particular needs and capabilities of the user.

Soil structure is nominated as a key indicator of soil quality. Soil structure determines the partitioning of rainfall at the soil surface between runoff and infiltration, and the transmission of water through the profile, which in turn determines the amount of water available to plants and strongly influences the amount of soil lost to erosion. Soil structure also affects plant root growth and development, the cycling of carbon and nutrients, the exchange of gasses in the root zone, the physical habitat for soil biota, and the energy required for root penetration and ground-engaging tools (Cass et al. 1996; Chan and Pratley 1998). At the catchment scale, changes to surface hydrology are likely to be associated with increased erosion and declining catchment health. At the paddock or enterprise scale, reduced plant available water and increased management inputs associated with declining soil structure reduces profitability and sustainability (Chan and Pratley 1998).

Common indicators of soil physical quality include surface infiltration rate, saturated and unsaturated hydraulic conductivity, water-holding capacity, drainage condition, aggregate stability, bulk density, soil strength, soil consistence and penetration resistance, used in conjunction with measures of soil carbon content, exchangeable sodium percentage and soil texture. Each has its limitations and advantages. Direct measurement of soil structural form is possible by a number of techniques, including image analysis. Field assessment of soil quality attributes using visual and tactile methods is gaining popularity due to ease of use and reduced monitoring costs, provided some rigour is applied to these observations; for example, the SOILpak series, including Anderson et al. (1999). As with any suite of indicators, care must be exercised in regard to validity, reliability and repeatability, and issues relating to spatial and temporal sampling need to be resolved.

Within the soil matrix, the distribution, shape and connectivity of pores (i.e. the soil's structural form) will be of importance. For example, in studies of soil hydraulic properties in certain cropping systems, Packer et al. (1992) and Murphy et al. (1993) have shown that a macropore diameter of 0.75 mm is necessary to result in significantly different infiltration and runoff characteristics, although the presence of pasture roots is likely to influence the threshold value in a grazing system. Smaller pore sizes are important for microorganisms (1 – 6 μm) to provide accessible, habitable and protective pore spaces and the biotic interactions this facilitates (van Veen and Heijnen 1994)

Reviews by Packer (1988) and Greenwood and McKenzie (2001) describe many of the potential impacts of livestock grazing on soil quality. Livestock are a major component of farming systems across much of Australia, including high rainfall zones, Tablelands zones with steeper topography and limited opportunities for crop enterprises, and mixed farming zones where livestock co-exist with cropping enterprises. However, a definite relationship between rotational grazing and soil quality is yet to be established.

CHAPTER 3

OVERVIEW OF EXPERIMENTAL METHODS

3.1 Location and topography of the experimental site

The experimental site consists of a 6 ha field located at the campus of Charles Sturt University, Orange, NSW, at an altitude of 875 m ASL. The field has a generally easterly aspect, with a relatively uniform 3 % surface gradient, resulting in little surface run-on from adjoining higher land to the west. A 6 m buffer strip separates the site from adjoining grazed land. Figure 3.1 illustrates the site location under different seasonal conditions. An extract from the 1:10,000 orthophotomap is included in Figure 3.2 to show site location and topography.

Figure 3.1
Images of the experimental site under typical spring (top) and summer (bottom) seasonal conditions, looking south. The water trough is located at the junction of set stocked and rotationally grazed plots.

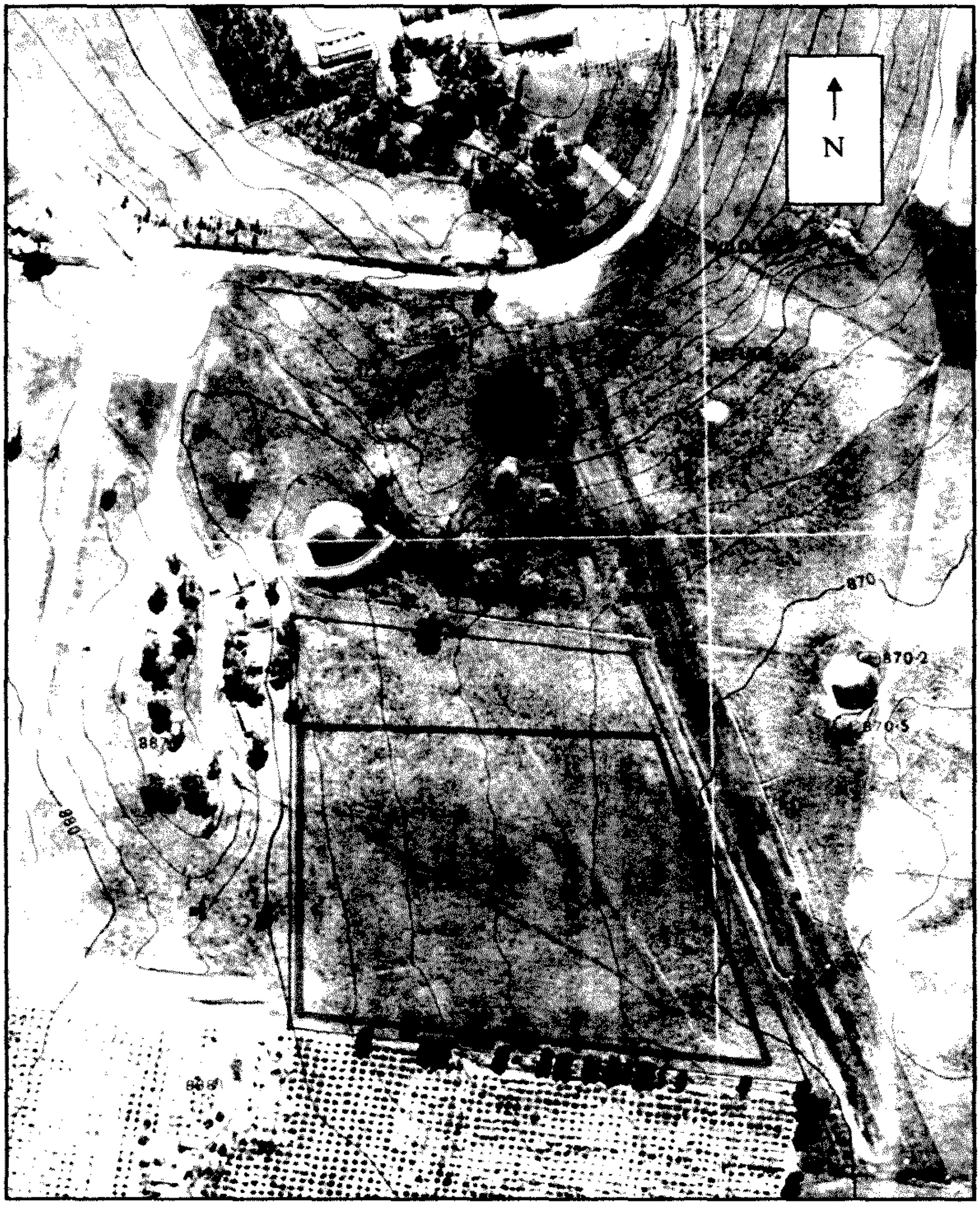


Figure 3.2
An orthophotomap extract (image from 1972) with the perimeter of the experimental site marked. The vertical contour interval is 2 m.

3.2 Site history

The site has a long history of grazing by sheep and cattle, occasional fertiliser application, and occasional machine operations associated with pasture establishment and fodder conservation but not regular cultivation for crops. The experimental site was developed during 1999, with livestock introduced in January 2000. Because of dry seasonal conditions prior to this, the field was left ungrazed during 1997 and pasture was cut for hay during 1998. Consequently, livestock had not been present on the field for more than two years prior to the commencement of this experiment.

3.3 Climate

In general terms, the climate at the experimental site is typical of the Central Tablelands of NSW; i.e. cold, wet winters and mild summers. Mean monthly minimum and maximum temperatures for the duration of the experiment are displayed in Figure 3.3. The temperature regime followed the long-term mean monthly pattern. Monthly rainfall amounts received at the experimental site during the duration of the experiment are displayed in Figure 3.4, together with long-term mean monthly rainfall. Rainfall during 2000 was wetter than average and rainfall during 2001 and 2002 was drier than average.

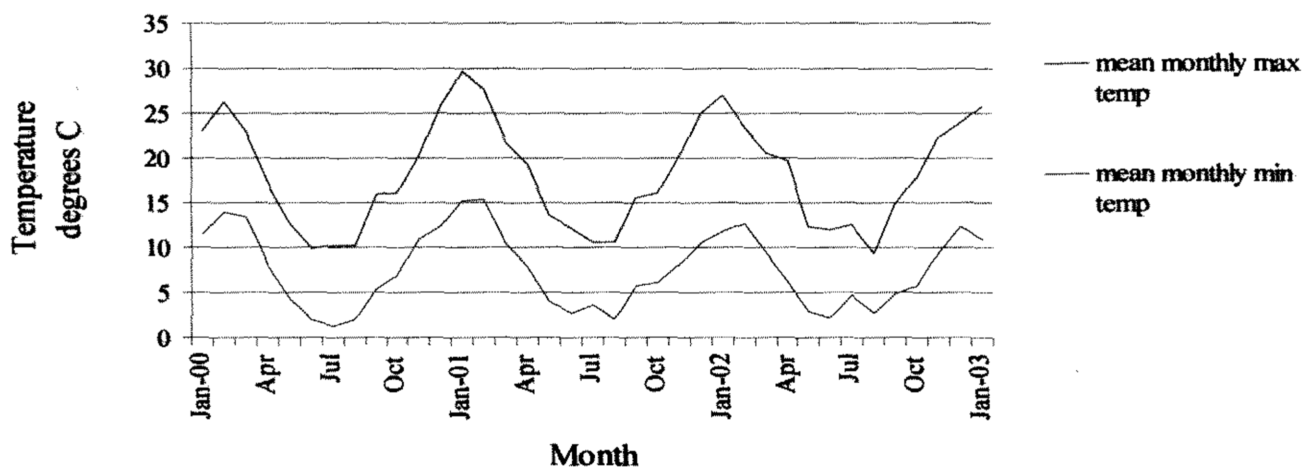


Figure 3.3
Mean minimum and mean maximum temperatures for Orange during the experiment (Bureau of Meteorology)

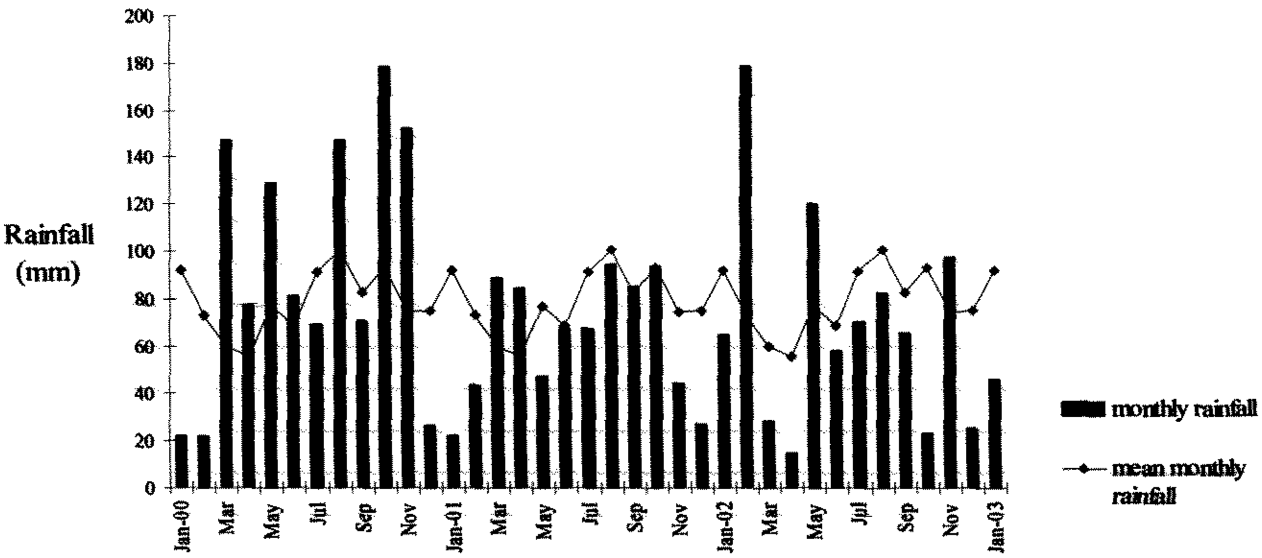


Figure 3.4
Actual and mean monthly rainfall for Orange during the experiment (Bureau of Meteorology)

3.4 Soil profile description

A 1.5 m-deep trench adjoining the experimental site was excavated. At this location, the profile is described as a shallow, well structured Brown Dermosol (Isbell 1996). Topsoil field textures are loamy, and become gradually more clayey with depth. Below about 1 m, highly weathered mudstone/siltstone dominates the profile, limiting deep drainage in wet conditions. Field observations showed a strongly pedal structure, with 30 – 50 mm sub-angular blocky peds primarily in the upper 200 mm, and 5 – 10 mm polyhedral peds throughout the profile. Abundant plant roots and frequent faunal passages were present in the upper 200 mm of the profile, decreasing with depth. A complete profile description is included in Figure 3.5.

According to Kovac et al. (1990), the site belongs to the North Orange soil landscape mapping unit, within the Molong Geanticline geological unit. The parent rock of this unit consists of medium to soft metasediments including slates, phyllites and siltstones on shale beds, largely derived from andesitic volcanics.

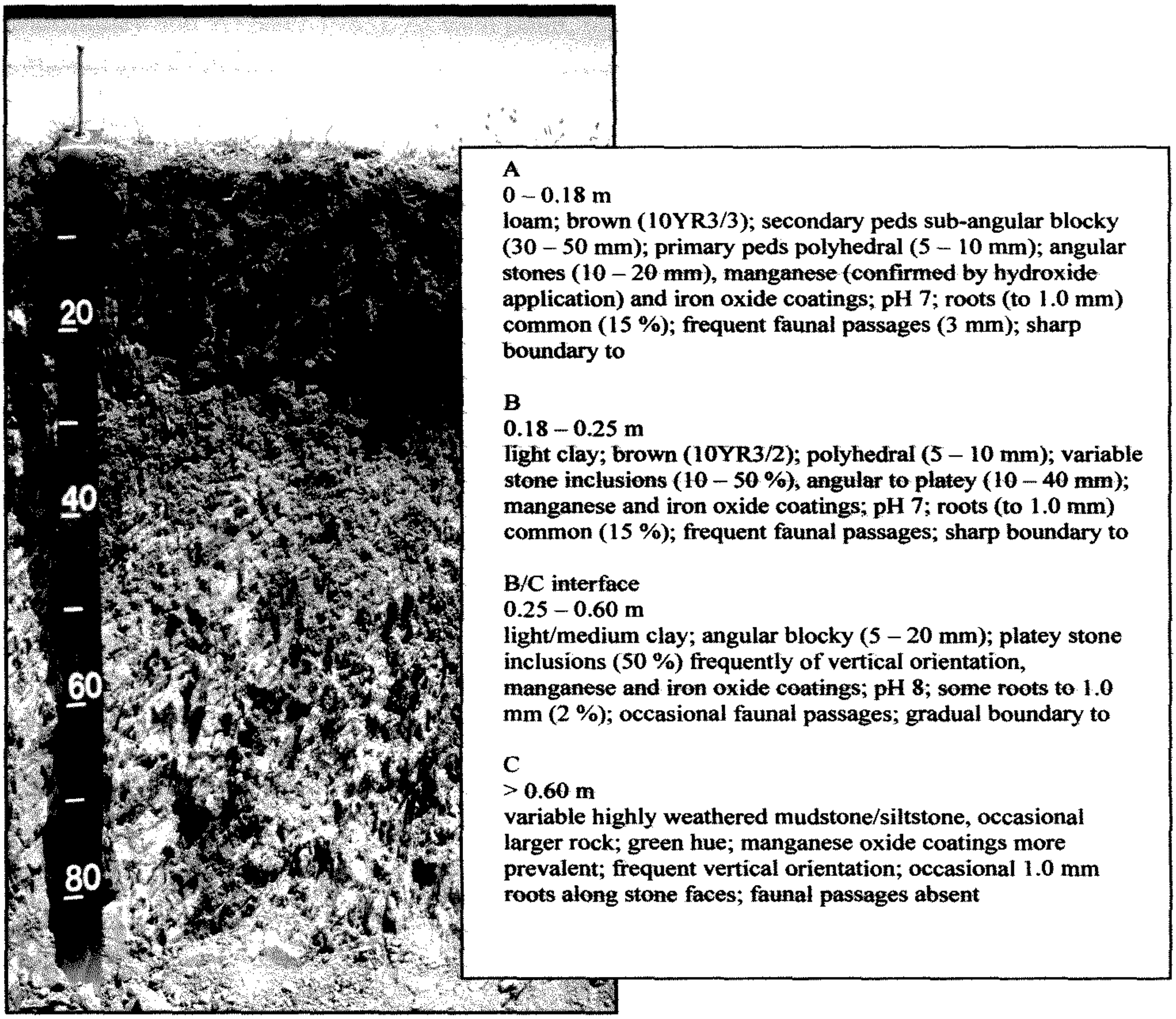


Figure 3.5
The soil profile adjoining the northern boundary of the experimental area and morphological field descriptions of horizons.

3.5 Initial soil properties

The mean value and variation of a number of soil properties were determined prior to the commencement of the experiment by analysis of 111 topsoil samples. A 25 m grid was established over the experimental area, and a sampling point randomly selected within each grid square. This was considered an appropriate sampling intensity for the size of the field, the resources available and the absence of evidence to the contrary

(McBratney and Webster 1983). At each point, topsoil samples (0 - 0.1 m depth) were collected and analysed for the following soil properties:

- bulk density (50 mm × 50 mm soil cores)
- total carbon content (LECO CHN 1000 laboratory combustion analyser; method 6B2 of Rayment and Higginson 1992)
- pH of 1:5 soil/water suspension (WPA CD500 laboratory analyzer; method 4A1 of Rayment and Higginson 1992)
- electrical conductivity of 1:5 soil/water extract (TPS 90FL laboratory analyser; method 3A1 of Rayment and Higginson 1992)
- particle size distribution (hydrometer method using chemical dispersion; Gee and Bauder 1986)

For 25 sample locations, unsaturated hydraulic conductivity at the soil surface was also measured by disc permeameter at 10 mm tension using the method described by Geering (1995). In addition, commercial soil tests conducted over the three years prior to the experiment provided data on exchangeable cation concentrations. The mean and range of values for these soil properties are displayed in Table 3.1.

These properties were evaluated individually to determine variation across the site, and to establish benchmark values. The presence of a small quantity of gravel is noted, however the reference to a mean proportion of gravel may be misleading because gravel was found in only 5 of the 111 topsoil samples, and these were located near the location of highest elevation. Samples from the western half of the field (i.e. of higher elevation) exhibited significantly smaller bulk density (mean of 1.26 g/cm^3 compared to a mean of 1.30 g/cm^3) and significantly greater organic carbon content (mean of 1.93 % compared to a mean of 1.78 %). The western half of the field also measured slightly (but not significantly) greater clay content (mean of 17.9 % compared to 16.8 %). Although some of these differences are statistically significant, they are small in absolute terms for each individual soil property.

Table 3.1
Mean and range of values of initial topsoil properties.
(Numbers in parentheses are standard error of the means)
Data for cations based on three historical analyses of bulked soil samples.

Initial topsoil property	No. samples	Mean value	Minimum	Maximum
pH	111	6.1 (0.035)	5.04	6.98
Electrical conductivity (μS/cm)	111	46.7 (1.33)	29.3	66.5
Clay content (%)	111	17.3 (0.32)	11.0	27.0
Silt content (%)	111	26.8 (0.36)	10.5	31.9
Sand content (%)	111	54.0 (0.54)	39.8	64.6
Gravel content (%)	111	1.9 (0.32)	0.0	3.8
Bulk density (g/cm^3)	111	1.29 (0.008)	1.06	1.49
Total carbon content (%)	111	1.85 (0.03)	1.28	3.18
K_{10} (mm/hr)	25	26.7 (4.84)	5.4	84.5
Cation exchange capacity ($cmol_{(+)}/kg$)	3	16.6	14.4	18.6
Exchangeable Na %	3	1.26	1.1	1.4
Exchangeable Ca/Mg ratio	3	1.86	1.6	2.4

The mean values of initial topsoil properties show no impediment to pasture growth, although the topsoil would be considered slightly saline, slightly low in organic matter, slightly low for calcium/magnesium ratio and despite the extended period with livestock removed, slightly compacted (Walker and Reuter 1996, NSW Department of Primary Industries 2004). The topsoil exhibited good to moderate aggregate stability to slaking and dispersion. Unsaturated hydraulic conductivity exhibited large variability, but from fewer samples.

3.6 Pasture botanical composition at commencement

Initial pasture botanical composition was assessed using the BOTANAL method described in Tothill et al. (1978). Prior to the introduction of livestock, dominant species consisted of perennial ryegrass (*Lolium perenne*), Yorkshire fog (*Holcus lanatus*) and subterranean clover (*Trifolium subterraneum*), but with phalaris (*Phalaris aquatica*), white clover (*Trifolium repens*), vulpia (*Vulpia bromoides*), barley grass (*Hordeum glaucum*), annual ryegrass (*Lolium rigidum*) and Patterson's curse (*Echium plantagineum*) also present. This species mix is typical of central tablelands pastures (Kemp et al. 1990).

3.7 Class of livestock

Merino wethers were selected for ease of management during the experiment. Adult sheep are more easily contained by fences, and adult merino wethers have a lower and more uniform nutritional requirement over the seasons than either young sheep or ewes, and supplementary feeding was not required during the duration of the experiment. Sheep were made available from a large mob of sheep of this class, with successive batches of sheep quickly becoming accustomed to the experimental site and readily moved to yards for occasional husbandry operations.

3.8 Grazing treatments

Two grazing treatments were deployed: set stocked grazing (treatment SS), where a fixed number of sheep were contained on each plot on a continuous basis (apart from occasional movement off site for husbandry operations), and High Intensity – Short Duration grazing (treatment HI-SD), where sheep were moved on and off the plots in accordance with pasture conditions of the day. Fenced stock exclosures enabled an ungrazed control (treatment C) to be established.

Although not replicated as an additional treatment (but labelled treatment CA), wire cages of 1 m^2 in area and a height of 100 mm were deployed at random locations within each plot (4 cages per plot). Pasture was able to be grazed from above these cages, but livestock were prevented from trafficking the soil under the cage. Supplementary information was gained from these sites. For some analyses, the data from treatments C and CA were combined (and labelled treatment U) to represent a data set where livestock were excluded. This was done as a first attempt to separate the influence of livestock hoof pressure from soil and pasture effects, but tests of significant difference between means from treatment U and treatments C and CA is not possible because of their lack of independence.

Treatment SS was set stocked at 15 dry sheep equivalents (dse) per ha, considered equivalent to average stocking rates for this region and soil type. Treatment HI-SD is managed by high intensity, short duration grazing events (for example, up to 220 dse/ha for 1-4 days, followed by some weeks rest), depending on pasture availability. For this treatment, livestock were introduced to the plots when pasture growth approached flower initiation, or after a minimum of 60 days rest from grazing, and removed when the pasture on offer declined to approximately 1.0 t DM per hectare. Livestock were removed from treatment HI-SD if surface water was present. The total grazing load was approximately equal for each of the two grazing treatments when averaged over the three year life of the experiment. In the case of treatment SS, this was 10.5 dse/ha, and in the case of treatment HI-SD, the average was 9.1 dse/ha, the difference representing only one HI-SD grazing event. However, livestock were present on treatment HI-SD for only 30 days in total, mostly during spring and summer when pasture growth was substantial.

3.9 Plot size and layout

The site was divided into plots of 0.8 ha each, regarded as sufficient to allow 'normal' grazing behaviour (Willatt and Pullar 1983, Proffitt et al. 1995). A schematic layout of plots is shown in Figure 3.6. Soil sampling was not conducted closer than 5 m to any fence, to remove the effects of stock walking tracks and camps, such as depicted in Figure 3.1 adjoining the fence and water trough.

Analysis of initial soil properties (listed in section 3.5) showed small but significant differences in soil bulk density and total carbon content, and a small but not significant difference in clay content, between the western and eastern halves of the experimental site, assumed to be due in part to a difference in elevation. There was no significant linear correlation between initial soil properties to confirm any trend in soil type or behaviour.

Soil properties in the vicinity of a buried sewer main, passing from north to south in the eastern half of the field and visible in the aerial photo of 1972 shown in Figure 3.2, were examined for the impact that soil disturbance might have had during installation of the pipe. No significant differences were observed. Even so, sampling was excluded during the experiment from within a 5 m wide band centred on the pipe location, and the layout of plots resulted in two plots from each of the primary grazing treatments spanning the area of the pipe location.

Visual inspection of the site and aerial photograph interpretation, based on vegetation characteristics of the time, also revealed the possibility that the northern extremity of the field may experience wetter conditions than other locations, probably due to occasional stormwater run-on and/or groundwater seepage from beyond the experimental area to the north. Comparison of mean values of initial soil properties showed no significant differences between the northern and southern parts of the field for any soil property.

Because of these factors, the set of initial soil properties was subjected to principal component analysis to help assess any spatial trend in soil properties when multiple soil properties were combined. The method of analysis is discussed in more detail in Chapter 7. The 111 initial soil sampling locations were provisionally allocated to a possible plot layout, based on their spatial coordinates, and principal components determined for each sampling location from the correlation matrix of the set of initial soil properties. The mean values of the first principal component, comprising 43.8% of the total variance, are listed in Table 3.2. Comparison of these means indicated a possible trend in initial soil properties, with a significant difference in mean value of the first principal component identified between plot 8, at the southern extremity of the experimental area, with some other plots.

Table 3.2

Mean values of the first principal component for initial soil properties allocated to a provisional plot layout.

Standard error of the mean in parenthesis. Treatment: SS, set stocked; HI-SD, high intensity – short duration. Location: N, north; S, south. Elevation: H, high (west); L, low (east). Significance: rows containing the same letter are not significantly different, at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference).

Plot	Treatment	Replication	Location	Elevation	Mean value of first principal component	Significance
1	SS	1	N	H	1.07 (0.26)	a
2	HI-SD	1	N	L	0.77 (0.48)	a
3	SS	2	N	L	-0.11 (0.19)	ab
4	HI-SD	2	N	H	-0.05 (0.48)	ab
5	SS	3	S	L	0.97 (0.38)	a
6	HI-SD	3	S	H	0.10 (0.49)	a
7	SS	4	S	H	-0.42 (0.25)	ab
8	HI-SD	4	S	L	-1.41 (0.29)	b

Varimax rotation of the principal component axes showed no significant differences between mean values of the rotated factors for plots allocated to the provisional layout.

Although the evidence of a trend in initial soil properties was weak, a balanced allocation of treatments to plots was selected, as shown in Figure 3.6. Treatments were allocated equally to plots of higher and lower elevation, and between northern and southern parts of the field. The replication containing the northern pair of plots fully encompassed the location of any perceived variation from differences in soil water content or initial vegetation character, and it was considered more sound to include a fourth replication of the experiment than to remove this zone from the experimental area.

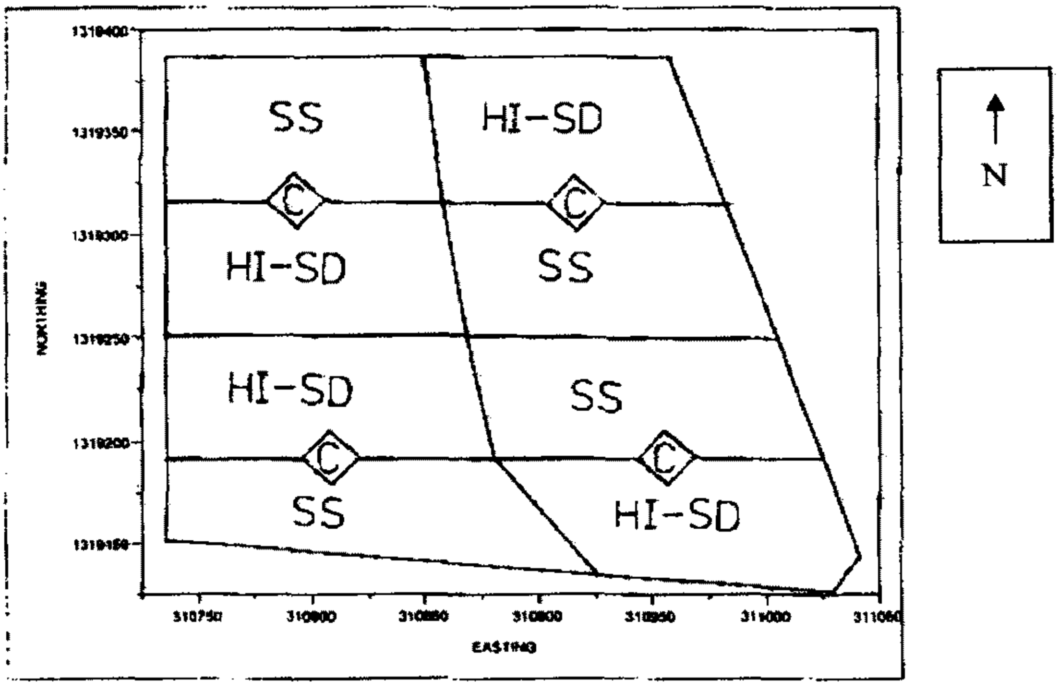


Figure 3.6
Schematic layout of plots and treatments.
Treatments: SS, set stocked; HI-SD, high intensity short duration; C, control. For treatment CA (grazed but not trampled), 4 pasture cages were located randomly within each plot.

3.10 Selection of soil quality indicators

The soil quality indicators selected for this experiment are discussed in detail in following chapters. For this experiment, 11 of the 12 key indicators recommended in Walker and Reuter (1996) are assessed, either as benchmark condition indicators or trend indicators of treatment effects. The key indicators used in this experiment were bulk density, unsaturated hydraulic conductivity, organic carbon content, penetration resistance, cotton strip assay, pasture botanical composition, soil pH, soil EC, texture, nutrient status and, by the use of image analysis of soil pores, soil structure.

3.11 Sampling dates

The term 'Year' is used throughout to mean the sampling date at the end of a nominal year of experimentation, this being the end of spring pasture growth period, and is not a precise date. Where soil properties were measured prior to commencement, this event is referred to as "Year 0". Benchmark condition was assessed at Year 0, with trend indicators measured simultaneously on three occasions on an annual basis in late spring or early summer. Materials for image analysis were not available in Year 0. Due to difficulty in using the penetrometer in dry soil conditions, measurement of penetration resistance was conducted at different dates to other soil properties.

3.12 Statistical and computational procedures

Hydraulic conductivity data were log transformed before analysis, whereas all other data met the criteria for normality and equality of variance to a reasonable degree. Outliers were deleted from two image analysis data sets; these were assumed to be caused by sampling defects in the field and resin irrigation failure. Data were evaluated by analysis of variance using JMP v.3.2.2 software (SAS 1998). Differences between means were compared using the Tukey–Kramer honestly significant difference at $P = 0.05$. Where correlations are shown, the Pearson product moment correlation coefficient was estimated, but because some properties were measured at different times, not all properties have been subjected to this analysis. Principal component analysis of the correlation matrix was also applied using JMP v.3.2.2 software (SAS 1998). Fuzzy logic models were created in Matlab v.6.5.1 (release 13) Fuzzy Logic Toolbox (The MathWorks 2003).

CHAPTER 4

SOIL BIOLOGICAL PROPERTIES & PASTURE BOTANICAL COMPOSITION

4.1 Introduction

The review of literature provides strong support for the inclusion of biological indicators of soil quality in comparing the impact of alternative land management practices, because of the importance of soil biota to soil function, soil structural form and soil structural stability. However, the selection of appropriate indicators of soil biological quality remains problematic, with constraints and limitations for each possible indicator. Mele et al. (1996) describe the limited alternative indicators for soil biological factors; few are adaptable to large scale field experiments at low cost.

In this experiment, it was assumed that the burrowing activity of soil macrofauna would be detected by resin impregnation of the resultant macropores, and so emphasis was placed on an indicator of soil microbial activity. Cotton Strip Assay (CSA) was selected, largely influenced by the report of Walker and Reuter (1996) and the low cost and non-specialised nature of this technique, despite the reservations of Mele et al. (1996). No measurements or observations of soil macrofauna were undertaken, although anecdotal observations were noted. The presence of organic matter as a supply of substrate for biotic metabolism is considered a critical element of biological condition. Total organic carbon content was used as an indicator of soil organic matter, despite the lack of information this provides on organic matter components, because of the ability to undertake these measurements with local resources. With the benefit of hindsight, and had more resources been available, additional indicators of biological condition may have been selected, and more specialised methods of assessment deployed.

For grazing systems described by this experiment, typical of much of the grazing lands of Australia, decomposition of pasture roots and litter represent the most significant source of organic matter input. Consequently, for these systems, pasture productivity becomes one of the important controls over soil organic matter content. Further, the direct effect of pasture root penetration in constructing vertical macropores should be considered, with perennial grass species having a better reputation for this characteristic than annual species. The quantitative relationship between soil quality and root depth and distribution, root tip condition and the number of active roots is well known. It was assumed that this relationship would be captured by images of root channels and other features, described in Chapter 6, and that greater value would be gained with the resources available by assessment of pasture botanical dynamics.

Greater perenniality of the pasture botanical composition will extend the productivity of a pasture, partly because of the potentially more extensive root system of perennial grass species and partly due to the delay of senescence by the regular defoliation of perennial species (Proffitt et al. 1993). This is likely to translate into greater pasture production and therefore increased water use and reduced weed control inputs, both leading to potential environmental benefits and increased livestock productivity (Kemp et al. 2000, Michalk et al. 2003). Greater stocking rates or improved enterprise outputs, such as conception rate, weaning weight, meat quality or fleece quality, are the desired outcomes of a livestock business. For the tablelands of NSW, this effectively represents a pasture harvesting system. Phalaris *(Phalaris aquatica)* is often listed as a preferred species in this context because of its productivity as a feed supply and weed competitor, although careful management is necessary to ensure an appropriate balance between grass and legume species.

The presence of different pasture species was assessed in this experiment to determine whether grazing tactics alone could be used to manipulate pasture botanical composition and degree of perenniality. According to Kemp et al. (1990), Kemp and Dowling (2000) and others, continuous grazing at average stocking rates, such as set stocking, will enable livestock to eat palatable species, and the green leaf material associated with perennial species for an extended period compared to annual species, in preference to less palatable material on offer. Consequently, the continuous defoliation of the palatable species will contribute to their decline because of the impact on root development and the competition created by the non-palatable component. Although perennial grass content therefore becomes the species group of interest, the pasture system requires some nitrogen input, which is provided via the legume content of the pasture and its ability to fix atmospheric nitrogen for many pasture systems.

4.2 Methods

4.2.1 Organic matter

For Year 0, total organic carbon was measured by a LECO CHN 1000 combustion analyser. For all subsequent measurements, the Walkley-Black method was used as described in Rayment and Higginson (1992). Consequently, Year 0 results cannot be reliably compared to results from subsequent years.

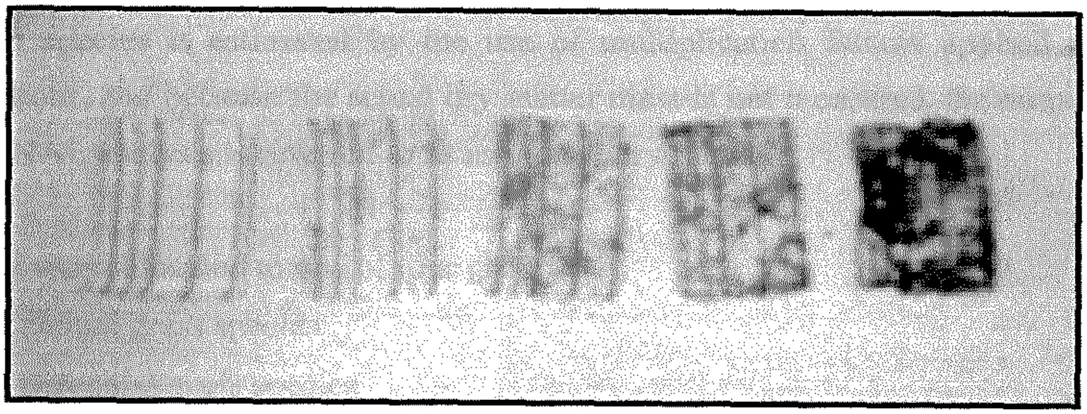
For Years 1, 2 and 3, six soil samples were taken per plot, with each sample split into two replications in the laboratory. Because of the small surface area presented in the treatment CA plots, destructive soil sampling for this treatment, necessary for OC analysis, was not performed. The Walkley-Black method was selected, despite reservations concerning the usefulness of measuring total soil organic carbon content compared to the more labile fractions of the carbon material, and the reliability of this method, for the following reasons:

- Total organic carbon is the measure of soil carbon regularly used by farmers and advisors, and has a long history of acceptance and interpretation, and
- constraints on the resources available to the experiment led to a decision to use a method that was available locally at no cost.

4.2.2 Cotton strip assay (CSA)

Although only a crude measure of microbial activity, and non-selective for specific organisms, this technique has been shown to be sensitive to differences in microbial activity for soils under pasture (King 1995; King et al. 1996) and is the only biological indicator of soil quality recommended by Walker and Reuter (1996). CSA was used in this experiment to supplement measurements of organic carbon because of its ease of use and measurement. Microbial decomposition of the fabric results in a reduction of its tensile strength, with the mean reduction in tensile strength used as an indicator of microbial activity. The method is described in detail by Harrison et al. (1988). Latter et al. (1988) describe the use of tensile strength measurements as the most satisfactory method of determining microbial deterioration of cotton strips, but also note that comparison between different soil types has limited application.

For sampling purposes, each plot was separated into three sections, and a bulked soil sample was taken from each section. Strips of Shirley® soil burial fabric were inserted into the soil samples, held in the laboratory for 12 days at 20°C and a moisture content of field capacity, then retrieved from the soil, washed and air dried. Four cotton strips were inserted into each sample, giving 12 replications per plot. The duration of 12 days was selected as being the time taken to reach approximately 50% of initial tensile strength, this being tested with trial samples. Images of cotton strips after various durations of burial are shown in Figure 4.1.



Days of burial 0 2 5 10 15

Figure 4.1
An image of typical cotton strips (40 mm × 50 mm actual size) after 0, 2, 5, 10, 15 and 20 days burial in soil at 20°C and field capacity moisture content.

Tensile strength was measured at The University of New England Animal Production Unit, with the results reported in kg for control strips (not buried) as well as buried strips. Because of the small surface area presented in the treatment CA plots, destructive soil sampling for this treatment, necessary for CSA analysis, was not performed.

4.2.3 Pasture botanical composition

Pasture botanical composition was assessed according to the BOTANAL method described by Tothill et al. (1978) and summarised as follows.

Three permanent transects were established over each of the grazed plots, with pasture botanical composition observed at 5 m intervals along each transect, amounting to between 33 and 39 observations per plot (the range is because of the unequal shape of plots resulting in different transect lengths). There was insufficient land area to conduct BOTANAL type pasture observations for treatments C and CA. At each observation, the species with the estimated maximum dry matter present was ranked 1, second most prevalent species 2 and third most prevalent species 3. For each species recorded, the number of times it is ranked first, second and third is summed, and the proportion of the total count determined. The percentage dry matter of each species is estimated by the use of multiplication factors applied to these proportions, and because the actual dry matter mass is not measured, the output of the BOTANAL analysis is referred to as estimated dry matter.

Species were allocated to one of four groups:

- annual grass species
- perennial grass species
- annual broadleaf species
- perennial broadleaf species

The allocation of species to classes is similar to the allocation of a 'perenniality' score to vegetation classification in the work of Ridley et al. (2003) on sustainability indicators. Of particular interest is the proportion of perennial grass (group PG) in the pasture botanical composition. It should be noted that Yorkshire Fog *(Holcus lanatus)* is included here as a perennial pasture species despite its reputation as a species that is unpalatable to livestock. Consequently, the data was further analysed to represent only palatable perennial grasses (group PPG) by excluding Yorkshire Fog from the calculations. Whilst Yorkshire Fog may provide environmental benefits as a perennial grass species, it is believed that its presence has distorted the botanical composition in favour of the unpalatable species.

Two additional analyses were included: a comparison of mean dry matter proportion of all palatable pasture species (group PPS) to include the proportion of clover species, and a comparison of mean dry matter proportion of Phalaris only (*Phalaris aquatica,* group P), being the preferred perennial species in this context.

4.3 Results and Discussion

4.3.1 Organic matter

The mean total organic carbon content across the whole of the experimental site (111 topsoil samples) prior to the introduction of livestock (Year 0) was 1.85% (0.03) with the mean of treatment SS plots at 1.87% (0.05) and the mean of treatment HI-SD plots at 1.85% (0.036). At Year 0, plots SS 1 and HI-SD 1 showed greater mean organic carbon content than all other plots, at 2.5 (0.2) % and 1.95 (0.2) % respectively, consistent with the findings of lower bulk density for these plots at commencement.

Results for the soil organic carbon content for all years are presented in Table 4.1. Because of a different method of measurement of total organic carbon in Year 0, Year 0 data cannot be directly compared to data from other years. The significantly greater total carbon content for the western half of the field (of higher elevation) in Year 0, reported in section 3.5, was not detected in subsequent measurements.

Table 4.1
Organic carbon content (%) under the different grazing treatments

Year	Treatment		
	SS	HI-SD	C
0	1.85 (0.03)	1.87 (0.05)	
1	2.6 (0.1)	2.4 (0.1)	2.4 (0.2)
2	2.6 (0.1)	2.8 (0.1)	2.3 (0.2)
3	2.8 (0.1) a	2.9 (0.1) a	2.5 (0.1) b

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

The total organic carbon content of soil in treatment C was significantly smaller than for both grazing treatments at the conclusion of the experiment. The reason for this is unclear, but the pasture sward in treatment C plots became dominated by senescent *Phalaris aquatica*, reducing pasture species diversity and possibly vegetative matter production. In the grazed treatments, continuous or periodic defoliation may have contributed to accelerated organic matter accumulation through increased root regeneration and decay of root and surface material (Kemp et al. 2000).

In absolute terms, the difference is of little practical significance because the total organic carbon content is similar to that of the larger values measured under pasture in other experiments (Geeves et al. 1995) and is well above the generally accepted lower limit of 1 % for maintenance of soil structure (Charman and Roper 2000). Although a generally increasing trend for total organic carbon content was observed for each treatment, there were no temporal differences ($P \leq 0.05$).

4.3.2 Cotton Strip Assay

Results for the cotton strip assay (CSA) are presented in Table 4.2. Also shown are the tensile strengths recorded on some control strips (i.e. cotton strips not buried in soil, 4 samples per batch). No significant differences between treatments were observed. The test procedure resulted in a relatively large scatter in data in Years 2 and 3, possibly due to difficulty with control of sample temperature and water content, although the tensile strength of strips in any particular soil sample were highly repeatable.

Table 4.2
Cotton strip assay as measured by residual cotton strip tensile strength (kg) and % loss of tensile strength after 12 days burial.

Year	Control strip (no burial)	Treatment					
		SS		HI-SD		C	
	kg	kg	%	kg	%	kg	%
1	66.8	37.6 (1.0)	43.7	33.4 (0.5)	50.0	30.8 (0.4)	53.9
2	67.9	42.1 (2.7)	38.0	42.3 (3.5)	37.7	43.9 (5.5)	35.3
3	66.0	42.0 (3.2)	36.4	25.4 (4.7)	61.5	39.2 (7.9)	40.6

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

In regard to soil microbial condition, these results would be rated as 'good' by King and Pankhurst (1996) in the assessment of microbial activity level for both methods of grazing. However, it is necessary to consider the sensitivity of CSA to discriminate between different grazing methods. Whilst advocating the use of CSA as an appropriate indicator of soil biotic condition, King and Pankhurst (1996) also note some limitations. Of these, the sensitivity of the results to soil temperature, water content and inherent microbial activity for the soil type are noted. In this experiment, soil sample temperature and water content were controlled during the CSA incubation period, and soil 'type' is considered uniform, so the results in Table 4.2 can be considered reliable. It is therefore assumed that the lack of difference between treatments is due to a high level of microbial activity for both treatments, consistent with the results for organic carbon content, or that changes in soil physical properties occurred but did not influence the CSA results. Evidence for the latter is provided by Shestak and Busse (2005), who found that habitable macropore space can increase in compacted soil compared to uncompacted soil and record relatively high microbial activity levels.

King (1996) states that overgrazing reduces the numbers of soil fauna. However, whilst set stocked grazing may result in differences in above ground pasture dry matter levels, this does not necessarily result in overgrazing provided pasture ground cover is maintained. The density and distribution of roots and excretal returns to the soil are likely to preserve large microbial activity.

4.3.3 Pasture botanical composition

Results for pasture botanical composition are presented in Table 4.3. For the perennial grass component (columns labelled PG), a small but significant difference was observed in Year 2 only, with treatment SS having a greater percentage of estimated perennial grass dry matter. No other significant differences were measured between treatments. The presence of a large proportion of Yorkshire Fog may have concealed more subtle effects. No differences were measured between western and eastern halves of the experimental area (i.e. of higher and lower elevation respectively).

The botanical composition measured in this experiment, whilst typical for the region, does not represent an ideal pasture mix. The substantial presence of Yorkshire Fog and the dominance of grasses compared to a balance of grass and clover species represents a sub-optimum botanical composition for both treatments. The Yorkshire Fog is unpalatable and stock may therefore overgraze remaining species, whilst the clover component is necessary to provide feed quality and soil nitrogen input.

Table 4.3
Proportion of estimated dry matter of various pasture species groups (%)

Year	PG		PPG		PPS		P	
	Treatment		Treatment		Treatment		Treatment	
	SS	HI-SD	SS	HI-SD	SS	HI-SD	SS	HI-SD
1	76.1 (2.0)	80.9 (2.7)	35.7 (11.1)	36.9 (5.0)	93.6 (0.6)	95.0 (2.6)	0.42 (0.36)	1.48 (1.21)
2	85.3 (1.4) a	71.9 (2.7) b	32.5 (6.3)	28.2 (7.9)	94.4 (0.8)	95.7 (1.5)	0.12 (0.13)	5.8 (4.2)
3	53.7 (3.3)	63.3 (3.7)	51.5 (3.7)	59.8 (3.4)	70.9 (5.6)	75.9 (4.4)	2.6 (0.4)	7.5 (2.7)

Pasture groups are as follows: PG, perennial grass content; PPG, palatable perennial grass content; PPS, all palatable pasture species; P, Phalaris content. Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

The apparent changes in botanical composition over time appear inconsistent, with treatment SS recording an increase in PG in Year 2 followed by a sharp decrease, compared to treatment HI-SD which measured a small decrease each year, with a large decrease in PPS in Year 3 for both treatments. Treatment HI-SD exhibited an increase in P each year whereas treatment SS exhibited a small decline in P in Year 2 then a small increase in Year 3. However, a single annual assessment of botanical composition measured by the BOTANAL method, based on presence or absence of species, masks the inherent seasonality of pasture growth patterns, particularly as pasture species approach senescence. Further, the use of grazing tactics alone to influence botanical composition needs to be considered. Other intervention strategies normally available in a commercial grazing context, such as herbicide application to manage a sudden weed outbreak at a particular time, were not undertaken. Consequently, the significance of comparisons between years provides limited insight compared to between year observations.

4.3.4 Anecdotal observations

For treatments C and CA, a large number of earthworms were repeatedly observed compared to grazing treatments, but no measurements of abundance were made. This was assumed to be as a result of the more continuous presence of pasture vegetation, creating the possible combined effects of cooler soil temperatures and larger soil water content.

4.4 Conclusion

Whilst supporting the importance of measuring indicators of soil biological activity, the results from this experiment showed no significant differences between treatments. This may be due to the narrow choice of indicator, and the relatively small differences to be expected in biological function under the two grazing treatments unless some biological threshold was crossed (e.g. if a small increase in surface compaction under hoof pressure resulted in a significant difference in water infiltration). In this experiment, CSA data showed no significant differences between treatments. An initial conclusion may be that this test method, in isolation, is not sensitive enough to detect what will be shown to be subtle differences in soil properties as a result of different grazing tactics, or that the differences in soil properties are not resulting from differences in soil microbial activity. This is consistent with the observation of no differences in soil carbon content between grazing treatments, and the amount of soil carbon being above any lower critical threshold for all treatments. However, final conclusions will be based on multivariate analysis where the influence of soil biological properties may be more strongly exhibited.

Although no differences ($P \leq 0.05$) in pasture botanical composition were measured between treatments in this experiment, it was observed that by the conclusion of the experiment, treatment HI-SD plots carried greater estimated perennial grass dry matter, greater estimated dry matter of palatable perennial grasses, greater estimated dry matter of palatable pasture species in total and greater estimated dry matter of phalaris *(Phalaris aquatica)*, the preferred and dominant perennial grass species, compared to treatment SS. Whilst it is likely that a longer time is necessary for significant changes in pasture botanical composition to be realised by the use of grazing tactics alone, this trend after three years of grazing is encouraging, and consistent with the conclusions of Michalk et al. (2003).

CHAPTER 5

SOIL STRUCTURE ASSESSMENT BY SELECTED SURROGATE METHODS

5.1 Introduction

Traditional practice by soil investigators in the field has been to assess soil structure by measurement of surrogate soil properties; those that bear some relationship to the nature of a soil's porosity. For a particular soil texture, differences in soil bulk density, penetration resistance, hydraulic conductivity and/or some other indicator are often used to classify and compare differences in soil structure. Such methods may be easier and less expensive to deploy, but provide only indirect evidence of the nature of a soil's porosity. In this experiment, three field methods of indirect assessment of soil structure (soil bulk density, penetration resistance and unsaturated hydraulic conductivity) are used to determine their ability to detect differences between grazing methods. These methods are recommended by Walker and Reuter (1966) as suitable indicators of soil quality, and because they have been widely adopted in experiments by others, are expected to allow comparisons between this experiment and others. They offer relatively simple application in the field, allowing a larger number of data to be collected with the resources available. These results will be compared to direct assessment of soil structure by image analysis in Chapter 6.

In the context of tablelands grazing systems, compaction pressures under livestock can be substantial (Packer 1988, Greenwood et al. 1997, Greenwood and McKenzie 2001), and this contributes directly to a reduction in macroporosity and loss of pore continuity (Cresswell et al 1992, Betteridge et al. 1999, Lawrie et al. 2000, Greenwood and McKenzie 2001). Increases in bulk density and soil strength and a decrease in hydraulic conductivity under high stocking rates are commonly reported (Proffitt et al. 1995a, Proffitt et al. 1995b, Greenwood et al. 1997, Dormaar and Willms 1998, Greenwood and McKenzie 2001). However, the influence of alternative grazing methods is less well known. If livestock are removed from a pasture for some time, such as with rotational grazing, subsequent pasture growth may result in a recovery from compaction stress. Increases in water infiltration may occur, resulting in increased soil water content. These factors, and the increased productivity of the grazing livestock, will contribute to more sustainable production.

In addition, the impact of set stocked grazing on pasture botanical composition, and any subsequent impact on soil properties, should be considered. Under set stocked grazing management, livestock are continuously able to consume the more palatable species in preference to less palatable species. This will inhibit the ability of desirable pasture species to compete with undesirable species by a direct impact on root production and seed set. When pasture dry matter levels are low, the continued presence of livestock under set stocked grazing management will also reduce litter accumulation and expose the soil surface to erosion and raindrop compaction. These factors may influence soil physical properties because of differences in root growth, root channel development and organic matter decomposition.

5.2 Methods

5.2.1 Bulk density

Cylindrical steel cores 50 mm × 50 mm were driven into the topsoil (6 samples per plot to 50 mm depth). The oven-dry weight of soil was used to calculate bulk density. Samples were collected at the same time as sampling for unsaturated hydraulic conductivity and image analysis; i.e. at a single sampling date per year during late spring.

5.2.2 Penetration resistance

A Rimik® recording cone penetrometer (10 mm cone diameter, cone length 25 mm, 30° included cone angle, recessed shaft) was used to measure penetration resistance, as per the method described by Hignett (2002), using 30 insertions per plot to a depth of 600 mm recording at 15 mm depth intervals. It was intended to take penetrometer readings at the same time as sampling for other soil properties, however on some occasions resistance exceeded the capacity of the device so its use was limited to conditions where the soil water content was near field capacity. As a consequence, penetrometer readings in Year 0 and Year 1 were obtained on different dates than for other soil properties, and for Year 2, penetrometer readings were abandoned. For Year 3, contemporaneous sampling for all soil properties was possible. Soil water content was measured at that time to determine if penetrometer data required correction for soil water content. There were no significant differences in mean soil water content between treatments at this time (17.5%, 17.3%, 18.4% and 18.8% for treatments SS, HI-SD, C and CA respectively) so a correction was not applied. For some insertions, small subsurface stones stopped insertion before reaching full depth. These data were deleted and additional insertions were performed, however it is suspected that the presence of stones, and possibly the presence of dense root mats, may have influenced other penetrometer readings that were not detected at the time of insertion.

Penetration resistance was analysed using two scales of measurement:

- the average of penetrometer readings between the soil surface and a depth of 105 mm, considered the depth of major influence from livestock hoof pressure and considered equivalent to the depth of sampling of other soil properties, and
- the average of penetrometer readings between the soil surface and 300 mm.

It was not possible to effectively measure penetration resistance in treatment CA because the small footprint of the cages prevented sufficient insertions.

5.2.3 Unsaturated hydraulic conductivity

A CSIRO tension disc permeameter was used to estimate the hydraulic conductivity of the soil surface (6 measurements per plot), using the method described by Geering (1995) and McKenzie and Cresswell (2002). The water tension was set at 10 mm, which was found necessary to provide repeatable results due to surface cracking when partially dry, but this practice, and the tension at which measurements were taken, was consistent with the practice of Ridley (1996) and Murphy (1998). At this tension, it is assumed that pores greater than 1.5 mm radius are excluded from contributing to hydraulic conductivity.

5.2.4 Dates of measurement for surrogate methods

For all three surrogate methods, measurements were taken just prior to the reintroduction of livestock to treatment HI-SD plots. This date was variable depending on the rate of pasture growth since the previous grazing event, but represents the end of the grazing rest period associated with HI-SD management. Under this regime, any benefits to soil condition arising from the rest period of HI-SD management associated with the removal of livestock hoof pressure, or accelerated pasture root growth during pasture recovery from grazing, should have been maximised. Differences between treatments should be considered in this context.

5.3 Results and Discussion

5.3.1 Bulk density

At commencement (Year 0, prior to the introduction of livestock), the bulk density of the top 50 mm of soil across the experimental area (mean of 111 samples) was 1.28 g/cm^3. The mean bulk density of all treatment SS plots was 1.28 g/cm^3 and the mean bulk density of all treatment HI-SD plots was 1.29 g/cm^3. At the time of these measurements, the location of CA and C treatment plots was not decided. Although plot SS 1 and HI-SD 1 (located as shown in Figure 3.6) were of lower initial mean BD than all other plots, at 1.17 g/cm^3 and 1.25 g/cm^3 respectively, this was considered acceptable because their adjoining location enabled a fourth replication of the grazing treatments whilst maintaining 0.8 ha per plot. By Year 1, the mean bulk density for plots SS 1 and HI-SD 1 were not different to other plots of treatments SS and HI-SD respectively ($P \geq 0.05$).

Results for bulk density are presented in Table 5.1. Mean bulk densities measured in this experiment are generally at the smaller end of the range of results summarised by Greenwood and McKenzie (2001) from many experiments and various soil types carrying pastures grazed by sheep; soil bulk densities between 0.83 g/cm^3 for ungrazed soils to 1.57 g/cm^3 for soils with "heavily grazed" pastures are reported by Greenwood and McKenzie (2001), and as high as 1.88 g/cm^3 by Packer (1988), although the it should be noted that bulk density under equivalent loads for different soils is texture dependent.

No differences were observed between treatments in any year. Bulk density of the HI-SD and SS treatment plots was significantly higher at Year 0 than at any other time. Although the implication is that the introduction of livestock created an immediate decrease in mean bulk density, this is not supported by the CA and C treatment results for later years, where bulk density was measured to be similar to other treatments even with the exclusion of livestock hoof pressure. Consequently, it is assumed that the greater soil bulk density measured in Year 0 may have been as a result of past management practices on the experimental site, which included vehicle traffic associated with hay making, and/or reduced pasture root activity because of dry seasonal conditions in previous years.

Table 5.1
Mean soil bulk density (g/cm^3) under the different grazing treatments

Year	Treatment			
	SS	HI-SD	C	CA
0	1.28 (0.01)	1.29 (0.01)		
1	1.08 (0.03)	1.03 (0.03)	1.12 (0.04)	1.01 (0.05)
2	1.14 (0.02)	1.16 (0.03)	1.20 (0.03)	1.15 (0.08)
3	1.05 (0.03)	0.99 (0.03)	1.11 (0.03)	1.05 (0.04)

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

For all treatments, mean soil bulk density was significantly greater in Year 2 than in Years 1 and 3. Sampling for Year 2 occurred at the end of a period of below average rainfall, with soil conditions sufficiently dry to prevent satisfactory use of a penetrometer. Reduced pasture growth prior to the sampling date may have limited macropore recovery from pasture root growth. When combined with possible soil shrinkage due to low soil water content, and additional difficulty in extracting soil cores, increased soil bulk densities could be expected. Although not containing a high clay content, it is possible that shrinkage effects may have contributed to the differences in mean soil bulk density from year to year.

5.3.2 Penetration resistance

Penetration resistance was analysed using two scales of measurement. Firstly, the average of penetrometer readings between the soil surface and a depth of 105 mm was determined. This depth was selected to correspond with the sampling depth of other soil properties and is assumed to be the maximum depth of influence of livestock hoof compaction (Proffitt et al. 1995a, Greenwood et al. 1997), lying within the A horizon of this soil (refer to Figure 3.5). Secondly, the average of penetrometer readings between the soil surface and 300 mm was determined. This depth more closely corresponds with the primary depth of influence of pasture roots and faunal passages, and is approximately the depth of the interface between the B and C horizons for this soil (refer to Figure 3.5).

A summary of results is presented in Tables 5.2 and 5.3. Because of soil water content differences between measurement dates, the analysis is presented for comparison between treatment means at any measurement date only, and no attempt is made to identify any temporal trends. For the same reason, correlation between penetration resistance and other soil properties was not attempted.

Table 5.2
Mean penetration resistance (kPa) at an insertion depth of 0 – 105 mm under the different grazing treatments.

Year/date	SS	HI-SD	C
0/13-08-99	996 (20.8) a	1098 (25.6) b	
1/13-01-00	2091(53.3) a	2365 (61.3) b	1783 (99.6) c
3/08-01-02	982 (41.5) a	917 (31.9) a	584 (31.6) b

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Table 5.3
Mean penetration resistance (kPa) at an insertion depth of 0 – 300 mm under the different grazing treatments.

Year/date	SS	HI-SD	C
0/13-08-99	1236 (24.6)	1295 (25.4)	
1/13-01-00	1853 (45.8) a	2072 (50.8) b	1914 (80.5) ab
3/08-01-02	1124 (45.3) a	1159 (37.1) a	858 (31.7) b

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Penetrometer readings taken shortly after establishment of the experimental layout but before the introduction of livestock (Year 0 in Tables 5.2 and 5.3) showed a small but significant difference between plots allocated to each treatment for 0 – 105 mm only, associated with plots SS 2 and SS 3 (location shown in Figure 3.6) having a smaller mean penetration resistance (0 – 105 mm). These two plots were located at slightly lower elevation than the remaining plots. Although soil water content measurements did not reveal a significant difference in mean water content for these two plots, within-plot soil water content variation may have been responsible. Because of the sensitivity of penetration resistance measurements to soil water content, a large difference in penetration resistance may have been generated from a small difference in soil water content. Whilst this was noted, it was considered insufficient difference to prejudice experimental layout, because no significant differences were observed for mean penetration resistance 0 – 300 mm, and the mean penetration resistance across all plots was smaller than the generally accepted 3000 kPa necessary to inhibit plant root growth (Walker and Reuter 1996).

After a period of 5 months grazing, which covered the significant pasture growth and heavy grazing seasons of spring and early summer, penetration resistance was again measured. Although well short of a full calendar year (drying soil conditions would have prevented use of the penetrometer in subsequent months) these data are labelled as Year 1 in Tables 5.2 and 5.3.

This analysis revealed that for Year 1, treatment HI-SD exhibited significantly larger mean penetration resistance at both scales of measurement, with treatment C having significantly smaller mean penetration resistance for 0 – 105 mm soil depth only. The reduction in mean penetration resistance with soil depth for both grazing treatments is indicative of seasonal drying conditions, with topsoil penetration resistance values approaching the nominal limit of 3000 kPa.

Despite these differences being statistically significant, the difference in absolute value is small, and it is not clear (in fact it is unlikely) that these differences alone are sufficient to affect plant growth and soil water behaviour. Greenwood et al. (1997) measured significantly greater penetration resistance (25 – 50 mm depth only) of ungrazed soil (mean penetration resistance 1070 kPa) compared to a grazed soil at various stocking rates (mean penetration resistance of 1710 kPa – 1970 kPa for grazed soils) and a soil water content of 10%. Differences were smaller at higher soil water contents. They also concede that whilst significant differences in penetration resistance were observed, at low soil water contents, their practical significance is questionable when compared to the effects created by seasonal conditions and pasture botanical composition; in their experiment, pasture productivity was greatest at 'medium' stocking rates despite a reduction in porosity.

Profitt et al. (1995a) compare penetration resistance measurements under high, low and zero stocking rates of sheep in an unreplicated experiment consisting of a single grazing event on soils previously tilled. However, mean data are not tabulated and they provide no direct comparison between treatments, preferring to summarise the treatment effect with soil water content. Greenwood and McKenzie (2001) cite this data as partial evidence that soil strength is higher under high stocking rates, particularly at low soil water contents. Although from a different context (the grazing of cattle on loam-textured soil in Canada), Rodd et al (1999) found that mean penetration resistance at 0 – 60 mm depth (1340 kPa) was significantly greater than that of ungrazed soil (780 kPa), and for 165 – 300 mm depth, the resistances were 2060 kPa and 1520 kPa respectively. They concluded that pasture root growth may have been affected, although no measurement of root growth was taken.

Penetration resistance was again measured at Orange after two years of livestock grazing. There was no significant difference in mean penetration resistance between grazing treatments at either scale of measurement. However, mean penetration resistance for the control plots was significantly smaller at both scales of measurement, with the absolute difference more substantial than at previous measurement dates and more consistent with data reported by others and summarised above. This date of measurement represents more than 3 years without the presence of livestock to cause compaction or defoliation, enabling unrestricted pasture root growth to assist with the creation of macropores. Frequent observation of earthworms in treatment C plots provides additional potential for macropore development. Under these conditions, soil water content and temperature differences more favourable to soil macrofauna are likely, and penetration resistance reduced accordingly. Unfortunately, the small size of treatment CA plots prevented the collection of useful penetration resistance data for this treatment, which prevents a comparison of pasture defoliation without hoof pressure for this soil property.

Further analysis also revealed small but significant differences in penetration resistance between plots located at higher elevation within the experimental site (plots SS1, SS4, HI-SD2 and HI-SD3; refer to Figure 3.6) compared to the plots located at lower elevation (plots SS2, SS3, HI-SD1 and HI-SD4) for Years 1 and 3. These differences were not measured at the commencement of the experiment (Year 0). Analysis of data from individual plots showed that mean penetration resistance in plot HI-SD3 (0 – 105 mm) was significantly larger than plot HI-SD4 as well as some other plots, indicating significant within-treatment differences as large as differences between treatments, as shown in Tables 5.4 and 5.5.

Table 5.4
Mean penetration resistance (kPa) at an insertion depth of 0 – 105 mm at different elevations.
H = plots of higher elevation
L = plots of lower elevation

Year/date	H	L
0/13-08-99	1042 (20.1)	1051 (27.0)
1/13-01-00	2408 (57.6) a	2045 (55.0) b
3/08-01-02	1058 (43.4) a	840 (26.0) b

Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Table 5.5
Mean penetration resistance (kPa) at an insertion depth of 0 – 300 mm at different elevations.
H = plots of higher elevation
L = plots of lower elevation

Year/date	H	L
0/130899	1237 (25.8)	1294 (24.3)
1/130100	2056 (46.3) a	1867 (50.8) b
3/080102	1227 (46.8) a	1055 (33.4) b

Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

It was at first thought that this may have been caused by a difference in soil water content, possibly as a result of natural soil water drainage through the landscape. It was expected that the plots exhibiting larger penetration resistance at 0 – 105 mm, in this case the plots located at higher elevation, would be of smaller soil water content. This was tested by returning to the field the following day to extract soil samples (0 – 100 mm) for gravimetric measurement of soil water content. Interestingly, a small but significant difference was observed in soil water content, but with the plots located at lower elevation being drier (mean soil water content of 14.5 %) compared to plots at higher elevation (mean soil water content of 19.6 %).

Similar differences were also detected in data sets of penetration resistance in subsequent measurements, but not for other soil properties. Consequently, it is believed that the difference in penetration resistance may have been caused by the presence of small stones in some parts of the higher plots, occasionally observed during soil sampling. The presence of small stones would allow the penetrometer probe to continue down the soil profile, but would increase friction and create occasional obstacles to the penetrometer tip (Proffitt et al. 1995a, Greenwood et al. 1997).

5.3.3 Unsaturated hydraulic conductivity

Mean unsaturated hydraulic conductivity (K_{10}) at commencement (Year 0), measured across the whole of the experimental site, was 26.7 mm/hr. Only 25 measurements were taken prior to the introduction of livestock, which was insufficient to compare means of treatment plots. Results for the logarithm of hydraulic conductivity (log K_{10}, log mm/hr) for the duration of the experiment are presented in Table 5.6. The results are also presented in mm/hr after back transformation, for ease of interpretation. Although some significant differences are observed between treatments during early years of the experiment, with treatment SS exhibiting the smallest unsaturated hydraulic conductivity and treatment CA the largest, these differences were not sustained to Year 3. At the conclusion of the experiment, no significant differences were observed between treatments, and no apparent trend with time was observed. All values were within the range of values for K_{10} reported by Lawrie et al. (2000) from grazed pastures on a large number of experimental sites across south-eastern Australia.

5.4 General discussion of surrogate measures of soil structure

In this experiment, no significant differences in soil bulk density were measured between treatments, nor between years after Year 0. The measurement of significantly larger bulk density at Year 0 only for treatments SS and HI-SD does not lead to any

Table 5.6
Unsaturated hydraulic conductivity, K_{10} (mm/hr), under the different grazing treatments.

Year	Property	Treatment			
		SS	HI-SD	C	CA
1	Log K_{10} (log mm/hr)	1.24 (0.03) a	1.28 (0.05) ab	1.41 (0.06) ab	1.48 (0.10) b
	K_{10} (mm/hr)	18.7	21.3	27.8	35.5
2	Log K_{10} (log mm/hr)	1.17 (0.04) a	1.34 (0.05) b	1.28 (0.07) ab	1.41 (0.09) b
	K_{10} (mm/hr)	14.8	21.9	19.0	25.9
3	Log K_{10} (log mm/hr)	1.37 (0.06)	1.39 (0.04)	1.35 (0.19)	1.42 (0.07)
	K_{10} (mm/hr)	27.9	26.5	26.9	28.9

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$). Mean K_{10} at Year 0 was 26.7 (4.84) mm/hr.

robust conclusion because similar measurements were not taken for treatments C and CA, and results from all subsequent years for treatments C and CA do not support this initial response.

Willatt and Pullar (1983) measured soil bulk densities ranging from 0.89 g/cm^3 to 1.05 g/cm^3 under increasing stocking rate, but these differences were not significant even though the soil was described as conducive to compaction. Proffitt et al. (1993) also found no significant differences in soil bulk density when comparing continuous grazing to deferred grazing tactics (i.e. with livestock removed when the soil water content exceeded the plastic limit) with sheep, between any treatment at any time throughout the sampling period. This was attributed to the depth of sampling for bulk density, 0 – 40 mm, possibly being greater than the depth of compaction suspected in their experiment. Whilst this is noted, an increase in bulk density occurring in the top

20 mm of a sample will result in an increase in bulk density for a 40 mm depth of sampling. Abel-Magid et al. (1987) also found no consistent differences in soil bulk density between continuous and rotational grazing systems. Although Greenwood and McKenzie (2001) investigated a large number of grazing experiments to conclude that soil bulk density is a surprisingly good indicator of soil changes due to grazing, this appears to be a simplification when attempting to compare grazing management strategies.

In this experiment, differences in penetration resistance followed the expected trend for differences between grazing methods. The extended period in the absence of livestock appears to have been important in reducing penetration resistance. However, within-treatment differences related to plot elevation appear to cast some doubt over this interpretation and/or the usefulness of this method in this and other experiments, despite the widespread use of penetration resistance as an indicator of soil quality (for example, Proffit et al. 1995a, Greenwood et al. 1997, Rodd et al. 1999).

McKenzie (2001a), following a comparison of various measures of soil structure in Vertosols, is critical of the use of penetrometers to measure soil structure because of: (i) the sensitivity of the results to soil water content, (ii) the lack of correction of penetration resistance measurements for differences in soil water content in many published data sets and the difficulty in doing so, (iii) the need for careful operator technique to avoid inclusion of dynamic distortion of results, and (iv) the risk of soil build-up on the penetrometer tip in wet soils. The potential of small stones or other sources of obstruction is also problematic, contributing to occasional extreme values in penetration resistance data sets. Some authors prefer the comparison of median instead of mean values for this reason (Proffitt et al. 1995a, Greenwood et al. 1997). Consequently, whilst a potentially useful tool for rapid detection of areas of substantial compaction such as hard pans, the use of penetrometers for detecting the subtle differences in soil structure likely under different grazing management regimes, and using this data to compare grazing management regimes at different sites, cannot be recommended.

For SS grazing management, a significantly smaller unsaturated hydraulic conductivity (at 10 mm tension) was measured after Year 2 of this experiment (14.8

mm/hr), compared to both HI-SD rotational grazing (21.9 mm/hr) and grazing over soil cages (CA, 25.9 mm/hr), with the ungrazed control (C) not significantly larger at 19.0 mm/hr. Significant differences were not sustained to the end of Year 3.

Inconsistent results for soil hydraulic properties have also been a feature of other grazing experiments. Abdel-Magid et al. (1987) found inconsistent trends in results for ponded water infiltration rate under different stocking rates of cattle. Although the heavy stocking rate treatment showed decreased water infiltration rate compared to light and moderate stocking rates at the end of a grazing period in each of two years, there was no clear relationship between grazing method and soil hydraulic properties. In their experiment, based on two treatment replications, sampling locations were selected to ensure similarity of vegetation cover and not selected at random.

Willatt and Pullar (1983) measured a significant reduction in hydraulic conductivity from 110 mm/hr to 21.6 mm/hr under high stocking rates of sheep, on a soil they describe as conducive to compaction. However, they do not describe the duration of grazing or the timing of these measurements, but do describe changes in pasture botanical composition. It is therefore difficult to attribute the change in soil hydraulic properties to the treatment in their experiment. Rodd et al. (1999) found no differences in surface or subsurface saturated hydraulic conductivity with the grazing rest interval (of cattle) or sward type, finding such measurements highly variable and less able to confirm differences between treatments. In contrast, Greenwood et al. (1997) found that a reduction in saturated hydraulic conductivity was due to the loss of macropores greater than 1.2 mm diameter.

Proffitt et al. (1993) found that deferral of grazing (of sheep) for 6 weeks or more when soil water contents exceeded the plastic limit had a significant beneficial effect on unsaturated hydraulic conductivity (at 10 mm tension), compared to continuous grazing, after the first year but not in subsequent years. However, their experimental site was cultivated only one month prior to the introduction of livestock, to establish pasture, and the initial impact of livestock under these conditions is to be expected. After the third year of their experiment, differences in unsaturated hydraulic conductivity were observed only if the deferral of grazing exceeded 6 weeks. The reduction in K_{10} was attributed to the reduction in pore volume as a result of compaction by livestock hoof pressure. This period of deferral is consistent with that typical of HI-SD grazing management. However, Proffitt et al. (1993) also measured a reduction in K_{10} for the ungrazed treatment. This was attributed to the 'natural densification' of the soil of their experiment, which had been cultivated only months before the commencement of their experiment. Consequently, the reduction in K_{10} in the grazed treatments of their experiment should be anticipated.

In a replicated experiment with sheep, Nie et al. (1997) compared grazed plots to those where livestock were removed for seven months for 'pastoral fallow' in preparation for oversowing. They measured a 67% increase in unsaturated hydraulic conductivity at 20 mm tension and a 26 % increase in saturated hydraulic conductivity in topsoil (0 – 50 mm), but not at other depths. They also measured an 11% decrease in soil bulk density. This provides positive evidence of the ability of pasture rest to stimulate soil structure recovery, in this case in high rainfall (1160 mm annual average) conditions where soil pugging from livestock is prevalent.

The nature of the soil surface under pasture may contribute to the difficulty in measuring consistent results for soil hydraulic properties. Koppi et al. (1992) and Greenwood et al. (1997) discuss the microtopography of the surface of soils under pasture, describing the non-uniform distribution of macropores associated with the crowns and growing points of pasture plants, particularly perennial species, as likely to create non-uniform hydraulic properties. Significant differences in K_{10} were measured by Ridley (1996) for different pasture types, with K_{10} under *Phalaris* measuring 28 mm/hr, compared to 19 mm/hr under annual ryegrass, in an experiment to determine the water balance of different pasture types. Despite the advice of Viera et al. (1981), it is therefore likely that a larger number of measurements is necessary under pasture conditions than would be necessary for soils devoid of vegetation and provided with a uniform surface by cultivation.

The duration of pasture growth may be important. Using laboratory measurement of soil cores, Francis and Kemp (1990) measured 2.5, 5 and 14-fold increases in the amount of water infiltrated under 2, 4 and 35 year old pastures respectively, compared to cultivated soil. They observed a more well-developed sub-angular blocky soil structure with time under pasture even though the number of biogenic pores, as counted by visual interpretation of surface images, was similar.

In summary, inconsistent results for soil hydraulic properties has been the case for some time. A review of the impact of grazing methods on water infiltration in rangelands by Gifford and Hawkins (1978) was inconclusive, and is not a surprise to Packer (1988), who lists the many interacting factors that influence the inherent variability of water infiltration rate as follows: (i) variability in compaction behaviour; (ii) distribution of surface litter and standing dry matter; organic matter content; (iii) spatially non-uniform excretal return; (iv) animal impact on litter decomposition and surface crusts; (v) presence or absence of a few vertically continuous macropores; (vi) pasture type and its contribution to macroporosity; (vii) variable pasture density resulting in variable evapotranspiration rates and subsequent variability in soil water content in the vadose zone. Proffitt et al. (1995b) add raindrop impact on bare soil patches and the blockage of existing macropores by new plant growth to this list of factors.

5.5 Conclusion

The surrogate measures of soil structure deployed in this experiment provide inconsistent evidence of an association between grazing strategies and soil quality. It is possible that a longer time is required for differences to become manifest, that a single measurement date per year is inadequate to detect temporal trends, that a larger number of replications is necessary (for example, Proffitt et al. (1993) found significant differences in unsaturated hydraulic conductivity between grazing treatments with 24 samples per treatment, double the sampling intensity used in this experiment), or that the nature of these measurements is insensitive to differences in macropore geometry and density.

Despite these reservations, surrogate measures of soil structure are frequently used to pass judgement on the impact of grazing methods on soil quality, and are therefore considered unreliable conclusions. These have principally been in the comparison of continuous, or set stocked, grazing at different stocking rates, whereas this experiment aimed to test the effect of grazing rest. In various experiments, soil types that are susceptible to compaction, large soil water contents that exceed the plastic limit, vegetation removal, prior cultivation of the soil, and selective measurement of soil and pasture attributes are reported. Consequently, direct comparison between experiments is problematic. The need to fully describe experimental conditions upon which such judgements are made in detail has been acknowledged by Proffitt et al. (1995a) and is not new (Gifford and Hawkins 1978).

CHAPTER 6

SOIL STRUCTURE ASSESSMENT BY IMAGE ANALYSIS

6.1 Introduction

Direct measurement of soil pore attributes is likely to provide superior information about soil structure than surrogate methods, such as those described in Chapter 5. Image analysis of soil core sections, from either horizontal or vertical planes, has proven useful in various studies in recent years. The results reported by Douglas et al. (1992) and Koppi et al. (1992) are important because of the use of image analysis in soils growing perennial grass, although their treatments considered wheel track compaction and fertiliser application respectively, and were both conducted in Scotland. The analysis reported by McKenzie (2001b) provides a useful comparison between image analysis and other measures of soil structure, although this work was conducted on a Vertosol cropped with cotton. Image analysis enables direct measurement of pore size, shape and surface area, and when multiple stacked sections are analysed, indirect assessment of pore connectedness. Surrogate methods for assessment of soil structure may replicate certain important and relevant soil functions associated with soil structure (for example, hydraulic transmission at the macro-scale) but are not able to provide the additional measurements of pore shape and surface area, and so are less able to explain the mechanisms of macropore dynamics.

Measurement of pore shape, combined with knowledge of the size distribution of pores, provides evidence of the mechanisms of pore development, such as root channel construction, soil faunal burrows, and shrink-swell cracking. Measurement of pore surface area has relevance to soil gas and hydraulic transmission properties at the micro-scale, and may offer greater opportunities for the observation of root growth pathways (McKenzie 2001b). The distribution of pores throughout the profile can assist in the assessment of limits to plant growth and of the influence of negative impacts such as compaction from livestock hoof pressure. The method can detect small differences in pore geometry, and is therefore more sensitive to differences in soil management than indirect methods. Disadvantages of the method include the need for destructive sampling, the use of specific and relatively expensive consumable materials, the inability to detect micropores and the need for appropriate imaging apparatus and digitising software.

6.2 Methods

In this experiment, the maximum depth of influence of livestock hoof pressure, and therefore the minimum depth of sampling for image analysis, was assumed to be 100 mm, consistent with the findings of Singleton and Addison (1999). The grazing management treatments are described as follows:

SS:	set stocked grazing management
HI-SD:	high intensity, short duration grazing management
C:	control; livestock permanently excluded
CA:	soil protected from hoof pressure by wire cages, but defoliation by livestock possible
U:	the combination of data from treatments C and CA

Note that because of the lack of independence, tests of significance are not appropriate for comparison of results between treatment U and treatments C and CA, and are included only for comparison to treatments SS and HI-SD.

Cylindrical cores of undisturbed topsoil were excavated by hand using 160 mm diameter PVC tube (depth 140 mm, 3 samples per plot for treatments SS and HI-SD, and 8 samples for treatments C and CA). Samples were taken in late spring/early summer each year, at the same time as sampling for other soil properties. The method of excavation involved removing soil from outside the PVC tube, pushing it vertically downward as excavation progressed, then excavating under the tube to remove the core. In this way, the soil core remained undisturbed. Image analysis materials were not available for use at Year 0, so porosity characteristics are not known at commencement.

In the laboratory, soil cores were saturated with Araldite® resin containing diluent and fluorescent dye, based on the method described by Moran et al. (1989). The following Ciba-Geigy components and proportions were used: base resin LC191 (100 g); diluent DY026 (100 g); Hardener LC249 (70 g); opacifier DWO131 (3 g); fluorescent dye *Oracet Yellow* (0.25 g) or *Solvent Red* (1.0 g). The use of diluent and the method of irrigation allow the resin to displace water and air, and occupy soil macropores. Given the advice by Greenwood and McKenzie (2001) based on a review of a large number of experiments, that changes in porosity following grazing are mainly due to decreased macroporosity, this was deemed to be the class of pore of primary interest. Pre-treatment of the soil sample to remove soil water was contemplated, but according to Moran et al. (1989), the question of appropriate soil water content is answered as follows: that "the choice of resin viscosity is such that resin infiltrates and replaces resident soil water, thereby minimising problems reported elsewhere such as the need for acetone replacement, brittle handling of dehydrated specimens, and shrinkage deformations".

Once set hard, the top surface of each core was ground to a level surface on account of natural surface irregularities, and the cores were then sectioned to prepare horizontal surfaces at nominal depths of 10, 50 and 100 mm. It was considered that the use of stacked horizontal images would reveal more information on the continuity of macropores than vertical images, because vertical images are no better than horizontal images in detecting the continuity of tortuous macropores. Consequently, limited resources were applied to increasing the number of horizontal images. Under ultraviolet light, soil macropores can easily be differentiated from solids due to the fluorescent dye in the resin, and a binary digital image can be generated. These were taken over a 150 mm diameter area to avoid edge effects. The resolution of the images allowed resin-filled macropores of 0.065 mm and larger to be detected.

Each image was analysed using the SOLICON® computer program (Cattle et al. 2000) to provide data on macropore attributes at each nominal depth and combinations of these depths. In this experiment, the following macropore attributes were measured using the SOLICON ® program:

- macroporosity (the proportion of macropore space in the total soil volume, mm3/mm3, or %)
- macropore surface area (mm^2/mm^3)
- macropore count density (the number of macropores/mm^2)

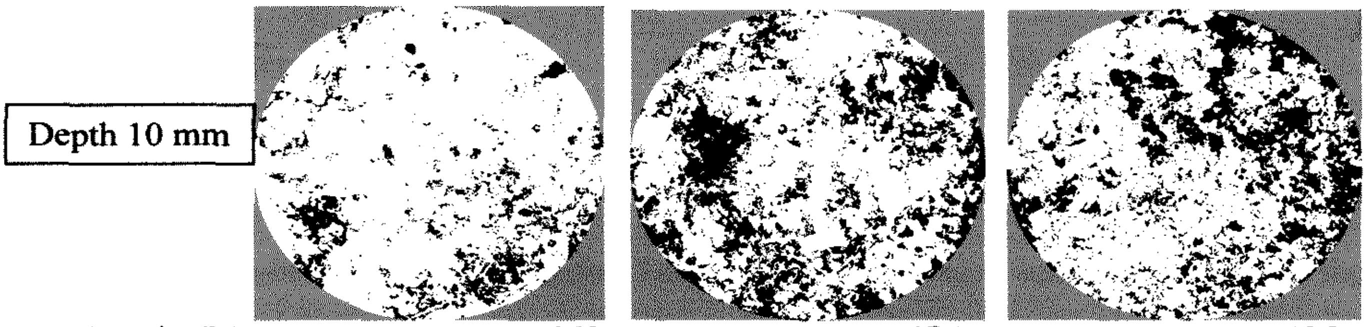
- macropore shape (as determined by the degree of roundness of pores, = 1.0 for circular pores and approaches 0.0 for pores with the shape of a thin needle)
- macropore sieve (the mean diameter of pores if they were all spherical in shape, mm)

Some example images and corresponding pore attributes are shown in Figure 6.1, with pores showing as black, together with corresponding SOLICON® output. The example images clearly show the potential for measurable visual differences between treatments, and with depth for any treatment, with the shape and size distribution of pores revealed.

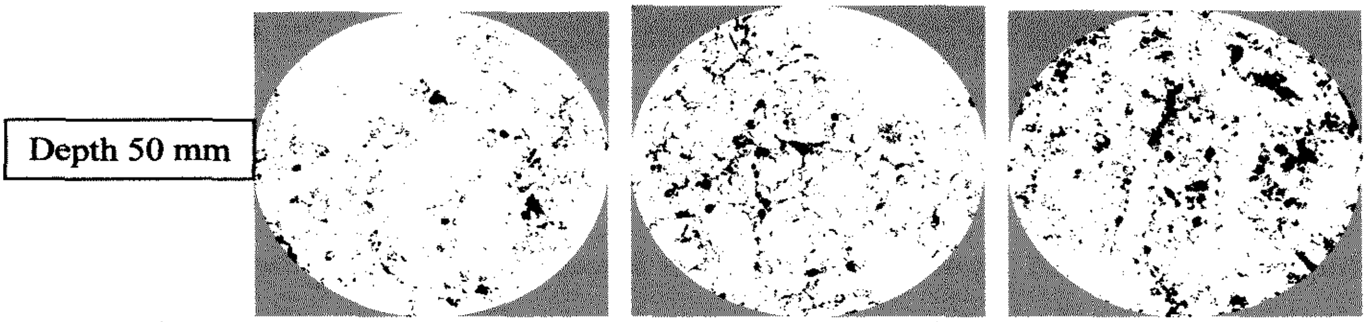
In the following results, for each macropore attribute the average over all three depths of measurement is labelled as 'average' of that attribute. In an attempt to improve correlation between unsaturated hydraulic conductivity and image analysis data, results for macroporosity were adjusted to remove macropores greater than 1.5 mm radius, corresponding to the macropore size limit excluded in the measurement of unsaturated hydraulic conductivity at 10 mm tension. This adjusted data is labelled 'mesoporosity'.

Additional adjustments of the porosity data were undertaken. Because much of the compaction effect of livestock is expected to occur in the top 50 mm of soil, macroporosity data for 10 mm and 50 mm depths were averaged to more closely measure this effect, and this data is labelled 'topsoil macroporosity'. In an attempt to provide even better correlation with unsaturated hydraulic conductivity data, mesoporosity was also averaged over these two depths, and this is labelled 'topsoil mesoporosity'. To separate the effects of large pores, the difference between macroporosity and mesoporosity was also measured for both average and topsoil depths.

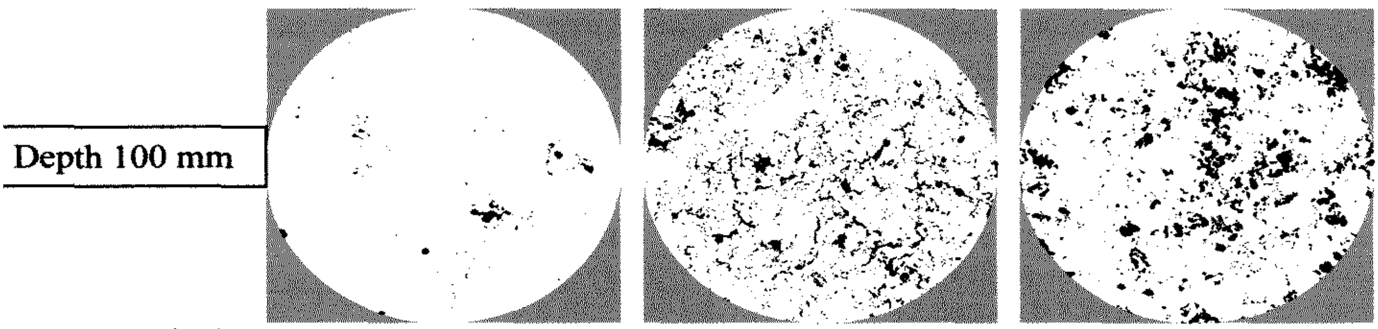
	TREATMENT SS	TREATMENT HI-SD	TREATMENT C



Porosity (%)	6.58	17.4	18.8
Pore surface area (mm^2/mm^3)	0.43	0.87	0.81
Pore shape factor	0.53	0.46	0.47
Pore count (no./mm^2)	0.12	0.18	0.18
Pore sieve (mm)	0.44	0.56	0.71



Porosity (%)	2.65	6.26	8.43
Pore surface area (mm^2/mm^3)	0.19	0.33	0.38
Pore shape factor	0.57	0.57	0.58
Pore count (no./mm^2)	0.07	0.17	0.12
Pore sieve (mm)	0.45	0.44	0.70



Porosity (%)	0.60	4.70	7.06
Pore surface area (mm^2/mm^3)	0.03	0.49	0.43
Pore shape factor	0.57	0.58	0.59
Pore count (no./mm^2)	0.01	0.11	0.09
Pore sieve (mm)	0.54	0.37	0.55

Figure 6.1

Example binary images of horizontal soil sections taken at three depths from three grazing management treatments, with corresponding Solicon® estimates of pore attributes. Actual size is 150 mm diameter. Pores are shown as black pixels.

6.3 Results

6.3.1 Macroporosity

Results for average macroporosity, defined here as the porosity from macropores greater than 0.065 mm diameter, are summarised in Table 6.1 with respect to comparison of treatment means and Table 6.2 with respect to temporal trends. At the conclusion of the experiment, average macroporosity for treatment SS was significantly smaller than treatments HI-SD and CA (and treatment U), with this trend becoming established in Year 2. Under treatment SS, average macroporosity showed a continuous reduction, with macroporosity significantly smaller in Year 3 compared to Year 1, whilst average macroporosity for treatments HI-SD and CA (and treatment U) showed a stable or increasing trend. The largest average macroporosity in Years 2 and 3 was consistently measured at locations where pasture was grazed through cages, which prevented hoof pressure on the soil surface. The trend for treatment C, under ungrazed soil where pasture became rank, was inconsistent. For treatment SS, and surprisingly for treatment C as well, average macroporosity declined to significantly smaller values by Year 2, whereas average macroporosity for treatments HI-SD, CA and U maintained a stable temporal trend.

Table 6.1
Average macroporosity (proportion of total porosity from pores > 0.065 mm diameter; %); comparison of treatment means in any year.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	10.4 (1.2)	11.0 (1.2)	16.1 (0.8)	13.9 (2.0)	14.6 (1.4)
2	7.3 (0.3) a	9.8 (0.5) ab	8.7 (1.0) ab	12.0 (1.0) b	11.4 (1.0) b
3	5.6 (0.7) a	12.4 (0.7) b	9.4 (0.8) ab	12.7 (0.9) b	11.8 (0.8) b

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Table 6.2
Average macroporosity (proportion of total porosity from pores > 0.065 mm diameter; %); comparison of within-treatment means across years.

treatment	year		
	1	2	3
SS	10.4 (1.2) a	7.3 (0.3) b	5.6 (0.7) b
HI-SD	11.0 (1.2)	9.8 (0.5)	12.4 (0.7)
C	16.1 (0.8) a	8.7 (1.0) b	9.4 (0.8) b
CA	13.9 (2.0)	12.0 (1.0)	12.7 (0.9)
U	14.6 (1.4)	11.4 (1.0)	11.8 (0.8)

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Results for soil macroporosity at each depth of measurement are shown in Table 6.3. Even after one year of grazing, significant differences between the grazing treatments and treatment U emerged, with macroporosity under treatment U being significantly larger at 50 mm and 100 mm depths of measurement than for treatment SS, and at 100 mm depth for treatment HI-SD. By Year 2, treatment SS had smaller macroporosity than all other treatments at each depth, with these differences significant with treatments CA and U. At the conclusion of the experiment, differences with treatment HI-SD also became significant at all sampling depths, with macroporosity under treatment HI-SD remaining similar to macroporosity under treatments CA and U at all depths. This indicates that rotational grazing of livestock can sustain as much macroporosity as where livestock hoof pressure is permanently removed.

Data for macroporosity were analysed to measure the stratification ratio of macroporosity (i.e. the ratio of macroporosity at 10 mm to macroporosity at 100 mm) as suggested by Franzeluebbers (2002) in relation to other soil attributes. Although treatment SS had the highest ratio (3.02 ± 0.65) and treatment HI-SD the lowest (2.63 ± 0.67) in Year 3, indicating a more uniform distribution of macroporosity with depth for treatment HI-SD, differences were not significant. In any case, a stratification ratio takes no account of the absolute value of macroporosity in any treatment.

Table 6.3
Macroporosity at each depth of measurement (%) for different grazing treatments.

Macroporosity (%)	Depth (mm)	Treatment SS	HI-SD	C	CA	U
Year 1	10	21.5 (2.6)	22.2 (3.0)	30.2 (3.0)	26.3 (4.1)	27.6 (2.9)
	50	6.0 (1.0) a	6.9 (0.7) ab	10.2 (01.6) b	8.7 (1.3) b	9.2 (1.0) b
	100	3.7 (0.6) a	4.1 (0.6) a	7.9 (2.2) b	6.8 (1.1) b	7.1 (1.0) b
Year 2	10	14.3 (0.7) a	17.2 (1.2) ab	15.1 (2.2) ab	22.5 (1.9) b	20.0 (1.7) b
	50	4.7 (0.4) a	5.8 (0.8) ab	6.5 (0.8) ab	8.7 (1.1) b	7.9 (0.8) b
	100	2.9 (0.3) a	4.2 (0.6) ab	4.7 (0.9) ab	7.0 (1.0) b	6.2 (0.8) b
Year 3	10	9.3 (1.2) a	18.1 (2.1) b	15.9 (1.6) ab	22.7 (2.4) b	20.4 (1.9) b
	50	4.4 (0.7) a	7.6 (1.2) b	7.5 (0.8) ab	8.0 (0.10) b	7.8 (0.7) b
	100	3.8 (0.6) a	8.0 (0.7) b	4.7 (0.3) ab	7.5 (1.1) b	6.7 (0.9) b

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Results for topsoil macroporosity, defined as the average of macroporosities at 10 mm and 50 mm depths of measurement, are derived from Table 6.3 and listed in Table 6.4.

Table 6.4
Topsoil macroporosity (%) for different grazing treatments.

Year	Treatment SS	HI-SD	C	CA	U
1	13.7 (1.7)	14.6 (1.7)	20.2 (1.2)	17.5 (2.6)	18.4 (1.8)
2	9.5 (0.4) a	11.9 (0.7) ac	10.8 (1.3) a	15.6 (1.4) b	14.0 (1.2) c
3	6.7 (0.9) a	14.5 (1.2) b	11.7 (0.8) b	15.3 (1.4) b	14.1 (1.1) b

Refer to text for treatment descriptions. Treatment U unable to be compared to treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

By Year 1, both grazing treatments exhibited smaller topsoil macroporosity, and by Year 2, topsoil macroporosity was significantly smaller in treatment SS than for treatments HI-SD, CA and U. This trend became more pronounced by Year 3, with treatment SS having significantly smaller topsoil macroporosity than any other treatment, maintaining a declining trend over time whilst all other treatments maintained a stable trend.

6.3.2 *Mesoporosity*

Results for average mesoporosity, defined as the porosity derived from pores 1.5 mm – 0.065 mm in diameter, are summarised in Table 6.5.

Table 6.5
Average mesoporosity (porosity from pores 1.5 mm - 0.065 mm diameter; %); comparison of grazing treatment means in any year.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	10.1 (1.2)	10.4 (1.1)	14.4 (1.0)	10.7 (1.2)	12.0 (1.0)
2	6.9 (0.3) a	8.9 (0.7) ab	7.1 (0.6) ab	10.2 (0.7) b	9.5 (0.8) b
3	4.6 (0.7) a	10.3 (0.6) b	8.0 (0.7) ab	10.0 (0.7) b	9.5 (0.6) b

Refer to text for treatment descriptions. Treatment U unable to be compared to treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

The results are similar to those for average macroporosity, with treatment SS having significantly smaller average mesoporosity after year 2, with this trend continuing to the conclusion of the experiment. Average mesoporosity was largest in Year 1 for all treatments, with average mesoporosity remaining stable throughout the experiment in treatments HI-SD and CA.

Results for topsoil mesoporosity, defined as the average of mesoporosities at 10 mm and 50 mm depths of measurement, are summarised in Table 6.6. Topsoil mesoporosity was largest in Year 1 for all treatments. The relatively stable trend in topsoil mesoporosity throughout the experiment is noted for treatments HI-SD and CA. In Year 2, topsoil mesoporosity was significantly smaller in treatments SS and C than treatments CA and U. By Year 3, topsoil mesoporosity in treatment SS was significantly smaller than all other treatments.

Table 6.6
Topsoil mesoporosity (%) for different grazing treatments; comparison of treatment means.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	13.2 (1.7)	13.7 (1.6)	18.0 (0.9)	13.5 (1.6)	15.0 (1.2)
2	9.0 (0.4) a	10.9 (0.7) ab	8.8 (0.9) a	13.0 (1.0) b	11.6 (0.9) ab
3	5.9 (0.8) a	12.1 (0.7) b	9.9 (0.7) b	12.1 (0.9) b	11.3 (0.7) b

Refer to text for treatment descriptions. Treatment U unable to be compared to treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

6.3.3 Porosity from large pores

Whilst mesoporosity represents a large proportion of measured macroporosity, the difference represents the contribution to total porosity from pores larger than 1.5 mm. This class of macropore is likely to be produced from root channels, soil shrinkage cracking and the burrowing activity of soil macrofauna, and is associated with rapid water transmission (Packer 1988, Roberts and Packer 2000, Greenwood and McKenzie 2001). Although a relatively small contribution to total porosity, the differences between treatments are revealing, as summarised in Tables 6.7 and 6.8.

Table 6.7
Difference between average macroporosity and average mesoporosity (%), representing that part of average porosity due to macropores > 1.5 mm diameter; comparison of treatment means.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	0.35 (0.09) a	0.66 (0.13) a	1.65 (0.45) ab	3.20 (0.84) b	2.68 (0.61) b
2	0.37 (0.05) a	0.86 (0.15) ab	1.61 (0.38) bc	1.98 (0.34) c	1.85 (0.25) c
3	0.88 (0.17) a	2.09 (1.60) ac	1.43 (0.02) ab	2.67 (0.41) b	2.33 (0.34) c

Refer to text for treatment descriptions. Treatment U unable to be compared to treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Table 6.8
Difference between topsoil macroporosity and topsoil mesoporosity (%), representing that part of topsoil porosity due to macropores > 1.5 mm diameter; comparison of treatment means.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	0.49 (0.12) a	0.81 (0.17) a	2.14 (0.75) b	4.02 (1.19) b	3.39 (0.85) b
2	0.47 (0.07) a	1.01 (0.11) ab	2.00 (0.50) bc	2.57 (0.47) c	2.38 (0.35) c
3	0.85 (0.19) a	2.43 (0.98) ab	1.78 (0.12) ab	3.28 (0.65) b	2.78 (0.47) b

Refer to text for treatment descriptions. Treatment U unable to be compared to treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

In Year 1, both grazing treatments exhibited a significantly smaller proportion of large pores (> 1.5 mm) than treatment CA when averaged over the full 100 mm depth of sampling, and both treatments C and CA when averaged over the uppermost 50 mm depth. Similar differences between treatments were observed in Year 2. By Year 3, treatment differences were more obscure, with treatment SS being significantly smaller than treatment CA. For both grazing treatments, the maximum contribution to porosity from pores > 1.5 mm was observed in Year 3, which was not the case for the ungrazed treatments, although the value for treatment SS remained significantly smaller.

6.3.4 Macropore surface area

Results for macropore surface area are summarised in Table 6.9. By Year 3, average pore surface area was significantly smaller for treatment SS compared to treatments HI-SD, CA and U.

Results for macropore surface area at each depth of measurement are shown in Table 6.10. The significantly smaller pore surface area under treatment SS was observed at each depth of measurement at the end of Year 3, a trend that mirrors results for macroporosity, with significantly smaller macropore surface area for treatment SS.

Table 6.9
Average macropore surface area (mm^2/mm^3) for different grazing treatments.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	0.59 (0.07)	0.62 (0.06)	0.75 0.09)	0.47 0.04)	0.57 (0.05)
2	0.40 (0.02) ab	0.48 (0.03) ab	0.33 (0.03) a	0.52 (0.03) b	0.46 (0.03) ab
3	0.23 (0.04) a	0.45 (0.04) b	0.35 (0.05) ab	0.42 (0.04) b	0.40 (0.03) b

Refer to text for treatment descriptions. Treatment U unable to be compared to treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Table 6.10
Macropore surface area at each depth of measurement (mm^2/mm^3) for different grazing treatments.

Pore surface area (mm^2/mm^3)	Depth (mm)	SS	HI-SD	Treatment C	CA	U
Year 1	10	1.05 (0.12)	1.05 (0.12)	1.09 (0.04)	0.72 (0.03)	0.84 (0.06)
	50	0.42 (0.07)	0.50 (0.06)	0.67 (0.09)	0.42 (0.07)	0.50 (0.06)
	100	0.28 (0.05)	0.31 (0.06)	0.49 (0.14)	0.28 (0.03)	0.35 (0.06)
Year 2	10	0.69 (0.03) ab	0.71 (0.04) ab	0.50 (0.06) a	0.74 (0.038) b	0.66 (0.04) ab
	50	0.30 (0.03)	0.35 (0.04)	0.29 (0.03)	0.45 (0.03)	0.39 (0.03)
	100	0.21 (0.02) a	0.25 (0.04) a	0.21 (0.03) a	0.38 (0.03) b	0.32 (0.03) b
Year 3	10	0.40 (0.05) a	0.60 (0.05) b	0.48 (0.05) ab	0.62 (0.04) b	0.57 (0.04) b
	50	0.18 (0.03) a	0.33 (0.06) ab	0.35 (0.05) ab	0.33 (0.05) b	0.34 (0.03) b
	100	0.15 (0.03) a	0.34 (0.04) b	0.22 (0.02) ab	0.31 (0.06) ab	0.28 (0.04) ab

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

There are few other studies that have measured macropore surface area for comparison. Douglas et al. (1992), in soil growing perennial grass subject to different levels of fertiliser application, used image analysis to measure macropore surface areas ranging from 0.8 – 0.6 mm^2/mm^3 at 10 mm depth, 0.38 – 0.16 mm^2/mm^3 at 50 mm depth and 0.3 – 0.2 mm^2/mm^3 at 100 mm depth. In a similar comparison of perennial grassland subject to different levels of tractor wheel traffic, Koppi et al. (1992) measured macropore surface areas ranging from 0.6 to greater than 1.0

mm^2/mm^3 at 10 mm depth, 0.42 – 0.78 mm^2/mm^3 at 50 mm depth and 0.46 – 0.9 mm^2/mm^3 at 100 mm depth. These values are similar to those measured in this experiment.

McKenzie (2001b), in a Vertosol growing cotton, used image analysis to compare macropore attributes in soil from the cropped ridge compared to soil located in wheeltrack furrows. The macropore surface area in soil from ridges decreased from 1.05 mm^2/mm^3 at 50 mm depth to 0.486 mm^2/mm^3 at 250 mm depth, increasing again to 0.833 mm^2/mm^3 at 350 mm depth. In the samples from the furrow, subject to irrigation flows and tractor wheel traffic, the macropore surface area decreased from 1.34 mm^2/mm^3 at 50 mm depth to 0.211 mm^2/mm^3 at 150 mm depth, increasing to 0.475 mm^2/mm^3 at 350 mm depth. The changes in macropore surface area with depth were assumed to be associated with soil compaction, with the wider range in values to be expected in cultivated soils subject to compaction.

6.3.5 Macropore count

Results for average pore count density are summarised in Table 6.11. Significant differences were observed between treatments HI-SD and C in Year 2 only and between treatments SS and HI-SD in Year 3, with significantly fewer macropores in treatment SS compared to treatment HI-SD in Year 3. A generally declining trend in average pore count with time was observed for all treatments. Results for pore count density at each depth are presented in Table 6.12. A generally declining trend for macropore count is also observed with depth, with this being most apparent for treatment SS.

Although some significant differences in macropore count were measured between treatments, the practical significance may not be as relevant. The range in values of macropore count is equivalent to 1 macropore per 5.7 mm^3 (for treatment HI-SD at 10 mm depth in Year 1) to 1 macropore per 26.3 mm^3 (for treatment SS at 100 mm depth in Year 3). These values are unlikely to be a constraint or opportunity in the context of plant root growth. There are no comparable studies that report macropore count.

Table 6.11
Average pore count density (no./mm³) for the different grazing treatments.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	0.124 (0.013)	0.140 (0.012)	0.148 (0.025)	0.108 (0.021)	0.124 (0.017)
2	0.106 (0.007) ab	0.110 (0.008) a	0.070 (0.004) b	0.110 (0.016) ab	0.100 (0.007)
3	0.050 (0.007) a	0.083 (0.010) b	0.066 (0.011) ab	0.072 (0.008) ab	0.071 (0.007) ab

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Table 6.12
Pore count density (no./mm³) at each depth of measurement for different grazing treatments.

Pore count density (no./mm²)	Depth (mm)	SS	Treatment HI-SD	C	CA	U
Year 1	10	0.165 (0.013)	0.175 (0.014)	0.136 (0.018)	0.127 (0.013)	0.130 (0.010)
	50	0.124 (0.020)	0.143 (0.016)	0.175 (0.027)	0.093 (0.015)	0.121 (0.017)
	100	0.083 (0.013)	0.103 (0.021)	0.131 (0.034)	0.103 (0.046)	0.113 (0.032)
Year 2	10	0.142 (0.007) a	0.130 (0.007) a	0.079 (0.007) b	0.117 (0.005) ab	0.104 (0.007) b
	50	0.097 (0.008) ab	0.104 (0.010 ab	0.071 (0.006) a	0.119 (0.007) b	0.103 (0.008) ab
	100	0.078 (0.010)	0.086 (0.014)	0.060 (0.007)	0.094 (0.010)	0.082 (0.010)
Year 3	10	0.079 (0.010)	0.090 (0.007)	0.070 (0.007)	0.076 (0.008)	0.079 (0.010)
	50	0.045 (0.007)	0.073 (0.012)	0.079 (0.013)	0.071 (0.010)	0.074 (0.008)
	100	0.038 (0.007) a	0.080 (0.009) b	0.052 (0.010) ab	0.069 (0.013) b	0.064 (0.010) ab

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

6.3.6 Macropore shape

Results for pore shape factor are summarised in Table 6.13. Although some significant differences were observed at Year 2, with a significantly more rounded macropore shape factor with treatment HI-SD compared to treatment C, the differences in absolute terms appear small. A generally increasing (more rounded) trend in average pore shape factor is observed with time. There are no comparable studies that have measured macropore shape factor.

Table 6.13
Average macropore shape factor for the different grazing treatments.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	0.519 (0.010)	0.496 (0.021)	0.485 (0.025)	0.533 (0.014)	0.517 (0.014)
2	0.554 (0.006) ab	0.561 (0.008) a	0.562 (0.013) ab	0.526 (0.009) b	0.538 (0.009) ab
3	0.584 (0.014)	0.563 (0.021)	0.562 (0.011)	0.556 (0.008)	0.558 (0.007)

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

The results for pore shape factor at each depth are presented in Table 6.14. A significantly larger pore shape factor at the soil surface under treatment SS for Years 2 and 3 is noted. Also noted is the generally increasing macropore shape factor (more rounded macropores) with depth. More rounded pores may be associated with biological activity such as development of root channels or soil faunal activity, and less rounded macropores at the soil surface may be associated with aggregate disruption from hoof pressure or soil drying.

Although some significant differences between treatments were measured for macropore shape factor, the range in values of macropore shape factor is 0.407 to 0.645, unlikely to have practical significance. In non-cracking soils such as those of the experimental site, and under the influence of permanent pasture, the degree of roundness is well developed, and clearly visible in the sample images of Figure 6.1.

Table 6.14
Macropore shape factor at each depth of measurement for different grazing treatments.

Pore shape factor	Depth (mm)	Treatment SS	HI-SD	C	CA	U
Year 1	10	0.449 (0.014)	0.448 (0.012)	0.407 (0.022)	0.449 (0.027)	0.435 (0.020)
	50	0.553 (0.011)	0.521 (0.021)	0.484 (0.067)	0.572 (0.012)	0.543 (0.025)
	100	0.556 (0.010)	0.519 (0.040)	0.565 (0.019)	0.578 (0.014)	0.574 (0.011)
Year 2	10	0.505 (0.007) a	0.486 (0.007) ab	0.499 (0.025) ab	0.455 (0.010) b	0.470 (0.012) b
	50	0.578 (0.009)	0.588 (0.008)	0.570 (0.015)	0.555 (0.008)	0.560 (0.007)
	100	0.580 (0.010)	0.645 (0.026)	0.616 (0.024)	0.568 (0.015)	0.584 (0.014)
Year 3	10	0.529 (0.014) a	0.473 (0.015) b	0.486 (0.029) ab	0.495 (0.020) ab	0.492 (0.016) ab
	50	0.607 (0.014)	0.575 (0.010)	0.580 (0.005)	0.563 (0.010)	0.569 (0.007)
	100	0.602 (0.025)	0.643 (0.058)	0.609 (0.014)	0.611 (0.019)	0.611 (0.014)

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

6.3.7 Macropore sieve

Results for pore sieve are summarised in Table 6.15. For Years 1 and 2, treatments CA and U had significantly greater average pore sieve than both grazing treatments. A greater average pore sieve was measured for treatments SS, HI-SD and CA in Year 3, but no significant differences were measured at the conclusion of the experiment. This is perhaps surprising, given that smaller pores may be expected as a result of the impact of livestock hoof pressure, but the relationship between pore size and pore count needs to be considered and is discussed elsewhere.

The results for pore sieve at each depth are presented in Table 6.16. In both Year 1 and Year 2, treatment SS had significantly smaller pore sieve than treatments CA and U at most depths. In Year 2, significant differences between treatment SS and all other treatments were observed at various depths. By Year 3, the difference between treatment SS and treatment CA was maintained as significant only at a depth of 10 mm, and at 100 mm depth, macropore sieve was similar for all treatments including treatment SS.

Table 6.15
Average macropore sieve (mm) for the different grazing treatments.

Year	Treatment				
	SS	HI-SD	C	CA	U
1	0.46 (0.03) a	0.53 (0.05) a	0.60 (0.05) ab	0.89 (0.09) b	0.79 (0.08) b
2	0.52 (0.03) a	0.64 (0.03) b	0.85 (0.08) c	0.76 (0.04) bc	0.79 (0.04) b
3	0.92 (0.07)	0.89 (0.09)	0.80 (0.06)	1.02 (0.08)	0.96 (0.06)

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at P ≤ 0.05 (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences (P≤ 0.05).

Table 6.16
Macropore sieve (mm) at each depth of measurement for different grazing treatments.

Pore sieve (mm)	Depth (mm)	Treatment SS	HI-SD	C	CA	U
Year 1	10	0.51 (0.03) a	0.57 (0.02) a	0.73 (0.09) a	1.18 (0.19) b	1.03 (0.14) b
	50	0.46 (0.04) a	0.42 (0.02) a	0.50 (0.07) ab	0.67 (0.08) b	0.62 (0.06) b
	100	0.40 (0.05) a	0.59 (0.14) ab	0.56 (0.08) ab	0.81 (0.08) b	0.73 (0.07)
Year 2	10	0.58 (0.03) a	0.67 (0.02) ab	0.94 (0.21) bc	0.93 (0.07) bc	0.93 (0.08) b
	50	0.49 (0.03) a	0.60 (0.06) ab	0.81 (0.06) b	0.70 (0.06) ab	0.74 (0.04) b
	100	0.50 (0.04) a	0.92 (0.18) b	0.81 (0.10) ab	0.66 (0.07) ab	0.71 (0.06) ab
Year 3	10	0.70 (0.06) a	0.85 (0.11) ab	0.95 (0.04) ab	1.14 (0.15) b	1.08 (0.10) b
	50	0.96 (0.17)	0.80 (0.05)	0.68 (0.08)	0.86 (0.08)	0.80 (0.06)
	100	1.07 (0.15)	0.89 (0.10)	0.75 (0.15)	1.05 (0.11)	0.97 (0.10)

Refer to text for treatment descriptions. Treatment U unable to be compared to Treatments C and CA. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at P ≤ 0.05 (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences (P≤ 0.05).

6.4 Discussion

Important information was gained by direct measurement of macropore characteristics using image analysis. Of greatest significance were the results for topsoil macroporosity (average of 10 mm and 50 mm depths) and topsoil mesoporosity, which showed that after 3 years of grazing, treatment SS resulted in significantly smaller pores and less topsoil macroporosity and mesoporosity compared to all other treatments. Average macroporosity (total porosity > 65 μm) was shown to be generally increasing or stable with treatment HI-SD, and generally decreasing with treatment SS, with these differences significant after 3 years of grazing. Furthermore these differences were maintained to at least 100 mm from the soil surface, indicating greater vertical continuity of macropores under HI-SD grazing and greater disruption of macropore continuity under SS grazing. At the conclusion of this experiment, average macroporosity and macroporosity at each depth of measurement under treatment HI-SD were similar to the maximum macroporosity recorded, under the CA treatment.

Similar trends were observed for mesoporosity, defined here as the proportion of macroporosity from macropores between 1.5 mm and 0.065 mm diameter, and for pores greater than 1.5 mm diameter, providing evidence that differences in pore density exist across a range of pore sizes. Whilst large pores have a critical influence over soil hydraulic properties (for example, according to Packer et al. (1992), formation of macropores greater than 0.75 mm diameter is necessary to create significantly larger surface infiltration rate and reduced soil erosion in certain soils), smaller pore sizes (1 – 30 μm) are necessary for micro-organisms to provide accessible, habitable and protective pore spaces and the biotic interactions this facilitates (van Veen and Heijnen 1994, Shestak and Busse 2005). It is clear that grazing of pastures, whether by set stocked grazing management or by rotational grazing, impacts on macroporosity by reducing the number of macropores, especially those larger than 1.5 mm in diameter, as evidenced by the measured differences between macroporosity and mesoporosity in this experiment (Tables 6.1 – 6.9).

This is consistent with the findings of other researchers. Packer (1988), Proffitt et al. (1995a) and Greenwood and McKenzie (2001) summarise the possible mechanisms for the development and disruption of soil aggregates and macropores when the soil is subject to livestock grazing. When soil water content is high, remoulding of the soil surface occurs, to beyond the immediate depth of penetration of livestock hooves, and that repeated grazing with large soil water contents will continue to degrade soil structure. Proffitt (1995a) uses the plastic limit water content of the soil as a benchmark for the risk of soil remoulding under these conditions. Below this water content, compaction occurs as a result of the combined effects of axial and shear stresses associated with the dynamic forces of livestock standing, grazing and walking. Such stresses cause compaction (in the general sense of a reduction in macroporosity) because of the disruption of aggregates into smaller particles and the repacking of smaller particles to fill existing voids. Root growth resulting in the occupation of macropores is also likely to reduce measured macroporosity (Kay 1990, Greenwood and McKenzie 2001)

The extent of compaction depends not only on the applied force, but also on soil water content, soil texture, soil organic matter content, the stability of aggregates and the initial state of porosity (Packer 1988). It should be noted that some decompaction mechanisms are also possible in a grazing-pasture system. A grazing system by definition requires the soil to be growing pasture, and active pasture growth is likely to help create macropores by the construction of root channels and an increase in soil organic matter (Douglas et al. 1992, Greenwood and McKenzie 2001). Although not observed during the experiment at Orange, it is possible that the action of livestock hooves can disrupt crusts that might occur on the soil surface, thereby improving water infiltration and other soil properties (Greenwood and McKenzie 2001). Greenwood and McKenzie (2001) also discuss the rebound of soil subject to temporary compression, such as following the treading by livestock.

The relationship between total macroporosity, macropore count and macropore size is an interesting one. Whilst the soil under treatment HI-SD showed an increase in total macroporosity, and soil under treatment SS a decrease, macropore count declined for both (more so for treatment SS; Tables 6.11 and 6.12) but macropore sieve increased for both (Tables 6.15 and 6.16). Macropore shape became slightly more rounded for both treatments (Tables 6.13 and 6.14), but macropore surface area declined for both (more so for treatment SS; Tables 6.9 and 6.10). This indicates a subtle shift in the nature of the macroporosity over time, with the macropore population of the soil becoming smaller but the average macropore size becoming slightly larger. The impact of grazing is apparent for both grazing treatments, more so for treatment SS, with treatment HI-SD retaining a somewhat greater number of larger macropores. This appears to be consistent with the establishment of macropores along plant root channels or as a result of soil macrofauna activity. Given that both treatments contained similar soil types, pasture species and class of livestock, these differences can be attributed to grazing method. It is possible these changes are associated with changes in pasture botanical composition; for example, a reduction in pore population associated with compaction by livestock is partly offset by the retention of root channels associated with perennial grasses. These relationships are explored further in Chapter 7.

In an experiment to compare the effects of vehicle wheel pressure on soil structure under perennial grass during fodder conservation operations, and therefore in the absence of livestock, Koppi et al. (1992) used the image analysis method to measure greater macroporosity, greater macropore surface area and smaller aggregate size in a zero wheel traffic treatment compared to a reduced wheel pressure treatment and a conventional traffic treatment. This result is reported here because differences were attributed to differences in soil faunal population and activity as well as differences in compaction pressure, a factor likely to be present in the experiment at Orange. Differences in surface roughness were also measured, and the effects this may have on soil microclimate, and consequently soil condition, were also discussed.

Compared to treatment C, treatment CA recorded larger macroporosity and mesoporosity in Years 2 and 3 at all depths of measurement, larger proportion of average and topsoil macroporosity from pores > 1.5 mm diameter in all years, larger average macropore surface area and macropore count in Years 2 and 3, and larger macropore sieve in Years 1 and 3 at all depths of measurement. It is therefore appropriate to further consider the effect of pasture defoliation in the absence of livestock hoof pressure.

Although not subject to the grazing of livestock, Douglas et al. (1992) used the image analysis method to measure macropore (> 0.195 mm) characteristics in soil carrying perennial grass subject to different application rates of fertiliser. They found that the soil with the greatest application rate of fertiliser and the most vigorous pasture growth contained the largest number, size and volume of macropores, particularly within the depth range of 40 – 80 mm. This provides additional evidence of the effects of pasture growth on soil macroporosity. The treatment with the smallest fertiliser application rate and the least pasture production was measured to have the greatest porosity but with a larger number of smaller pores, and this was particularly noticeable at the soil surface (0 – 40 mm depth). The authors postulated that the lesser fertiliser rate created shallower pasture root production, thereby contributing to this effect. They concluded that more vigorous pasture growth rates contribute to more porous and stable soil structure. This is important evidence in the context of a grazing experiment because of the separation of pasture and livestock factors. Douglas et al. (1992) found no significant differences in soil bulk density in their experiment, with the authors claiming this method of soil structure assessment has insensitive depth resolution.

Tisdall & Oades (1980) measured some interesting differences in aggregate stability of a red-brown earth when the ryegrass grown in it was managed under different (simulated) grazing treatments. Aggregate stability was greatest when the ryegrass was provided with ample water, so as not to limit growth potential, and was top clipped at monthly intervals. The increased aggregate stability of soil containing pasture that was regularly clipped was associated with the presence of vesicular-arbuscular mycorrhizal (fungal) hyphae. It was hypothesised that the clipping led to cycles of root decay and regeneration, stimulating hyphae production and leading to the additional release of water soluble root exudates. Interestingly, clipping at two week intervals was not as beneficial, probably because the time interval was insufficient to allow root recovery from the stress of clipping. This observation is consistent with the intention and practice of HI-SD grazing management. Two additional features should be noted. The experiment was conducted in pots, eliminating the equivalent of livestock hoof pressure and therefore resembling the intention of treatment CA, and the effect of the presence of hyphae persisted for several months after the plants had died. This latter observation implies that although the hyphae may not have been viable, they retained an ability to bind soil particles.

In this experiment, although treatment SS created some significant impacts on the volume and nature of macropores at all depths of measurement, these were less pronounced at 100 mm depth. This is likely because the compaction effects of livestock hooves do not extend to this depth in this soil type, and that the effects of aggregate disruption and repacking are unlikely at this depth.

6.5 Conclusion

These results confirm that under the conditions of this experiment, set stocked grazing has a detrimental effect on soil macropores, particularly at the soil surface, and that high intensity rotational grazing appears to create macropore characteristics more similar to ungrazed pastures throughout the full depth of measurement. The mechanisms contributing to these differences are a combination of:

- consolidation at the soil surface caused by livestock hoof pressure,
- aggregate disruption and repacking at the soil surface, also caused by the action of livestock hooves, and
- macropore construction by flora and fauna, possibly aided by a non-significant change in pasture botanical composition.

In this experiment, similar trends to Proffitt et al. (1995b) were measured between set stocked grazing and HI-SD and CA treatments, but not with treatment C. Interestingly, treatment CA showed no significant differences over time in any measurement of average soil properties, indicating that the nature of macropores was unchanged with regular defoliation of pasture combined with removal of livestock hoof pressure. This leads to the conclusion that planned grazing may be a useful tool in managing soil physical quality, perhaps more so than total pasture rest.

CHAPTER 7

RELATIONSHIPS BETWEEN SOIL PROPERTIES

7.1 Introduction

The relationships between soil properties are explored further here, using linear correlation and principal component analysis of the results presented in previous chapters. Few studies have attempted correlation between soil properties in grazing or soil quality investigations, despite the collection of multivariate data sets, so the work by McKenzie (2001b) and Brejda et al. (2000a and 2000b) become important comparators in this context.

Two levels of investigation are described. For some soil properties (bulk density, unsaturated hydraulic conductivity, and the results derived from image analysis), data was collected on identical sampling dates and from the same (within 0.5 m radius) point locations within plots. These data are compared together, with principal components derived from the correlation matrix. The remaining pasture and soil properties (perennial grass content, palatable perennial grass content, Phalaris content, organic carbon content, penetration resistance and cotton strip assay) were measured using different sampling methods and different sampling locations within the plots. In the case of pasture assessment, the BOTANAL method used here required multiple observations along multiple transects in each plot, which prevented site-specific measurements. For organic carbon content and CSA measurements, bulked soil samples were used, and penetration resistance was measured on different dates to other measurements. Consequently, the plot averages for these data have been compared to the plot averages of all other data in a separate investigation of correlation and principal components.

Principal component analysis was selected because of its potential usefulness in interpreting pedological meaning from multivariate data sets (Webster and Oliver 1990) according to the strength of alignment of soil attribute clusters with yet-to-be-determined factors (in this case, soil quality factors). It is a common and valuable method of factor analysis applied to measurements of soil attributes, although it is a mathematical construct that carries no direct physical meaning, because it requires no prior estimates of the amount of variation in each soil attribute explained by the factors (Webster and Oliver 1990, Brejda 2000a). The analysis finds the principal axes of the multidimensional attribute space and determines the coordinates of each data point relative to these. When there is a large number of soil attributes with a diverse range of measurement scales, it is likely that certain attributes will have variances much larger than others and bias the analysis. Under these circumstances, the scales of measurement are standardised so that all attributes have a unit standard deviation and the analysis is applied to the correlation matrix.

If correlation is present, it is often the case that there is clustering of correlated soil attributes associated with the first few principle axes (SAS 1995); if there is little correlation between soil attributes, then clustering is weak. The eigenvalue of each principal component represents the proportion of the variance explained by that component, with eigenvalues greater than 1.0 explaining more total variation in the data than individual soil attributes (Brejda 2000a). If correlation is present, the majority of the variance will be explained by the first few components, and the cumulative explained variance will be high. The eigenvector of a soil attribute is the simple correlation between the attribute and the principal component. A soil attribute with an eigenvector with a value approaching 1 indicates a heavy loading on a particular component. Where multiple soil properties demonstrate similar loading, some relationship is implied.

Analytical rotation of the first few principal components attempts to improve the strength of alignment of the attributes, with simple orthogonal rotation preserving the integrity of the structure of the data. With the widely used Varimax rotation (Webster and Oliver 1990, SAS 1995), each original axis is aligned as closely as possible to one of the new axes by maximising the vectors along these axes. This emphasises the clustering of attributes so that each attribute contributes as strongly as possible to one of the rotated components and as little as possible to the others. In this way, the strength by which groups of soil attributes can be associated with each rotated component, which can now be referred to as a factor, can be measured by the factor score. A large communality for a soil attribute indicates that a large proportion of its variance is explained by the factor, with a small communality indicating that much of the variance of that soil attribute remains unexplained (Webster and Oliver 1990, Brejda 2000a). In the following analyses, attribute communalities are presented in conjunction with factor scores, and the importance of attributes with low communality are noted in subsequent interpretation of factors.

As a final step in the analysis, the factor scores for each soil attribute are applied to the mean values of the soil attributes for each treatment. In this way, analysis of variance of the differences in treatment factor scores can detect treatment effects for each factor.

7.2 Comparison of soil attributes measured at similar locations and dates

7.2.1 Correlation between selected site-specific soil properties

Pearson product moment correlation coefficients within each year are shown for selected soil attributes in Tables 7.1, 7.2 and 7.3. These soil attributes were collected from the same (within 1 m) location and at the same time, so are nominated as 'site-specific' data. Correlation between image analysis results for average and topsoil macroporosity and average and topsoil mesoporosity was generally high, as expected given the algorithms used to estimate these variables and their co-dependency. Consequently, it is the correlation between these properties and the more traditional surrogate measures of soil structure (bulk density and unsaturated hydraulic conductivity) that is of most interest.

Some correlation was noted between some of the image analysis results and the other measures of soil structure, most notably between porosity measurements and unsaturated hydraulic conductivity in Year 2, but perhaps surprisingly, only weak correlation was exhibited between macropore count, shape or sieve with bulk density or unsaturated hydraulic conductivity. Strong correlation between porosity measurements and results for macropore count and macropore surface area is also apparent, less so for macropore shape factor, and least for macropore sieve. Significant negative correlation between macropore shape factor with macroporosity and other macropore attributes is to be noted, implying that more rounded pores are associated with smaller values of macroporosity. The negative correlations between macropore sieve, surface area and count is also of interest; a higher macropore population appears to be associated with smaller macropores whilst providing a larger macropore surface area.

Table 7.1
Pearson product moment correlation coefficients for selected site-specific soil attributes from Year 1. Values marked * are significant at $P \leq 0.05$.

	Log K10	BD	Ave. macro-porosity	Pore count	Pore surface area	Pore shape
BD	-0.16					
OC	0.03	-0.10				
Ave. macroporosity	0.33	-0.35*				
Pore count	0.18	-0.20	0.46*			
Pore surface area	0.25	-0.20	0.68*	0.84*		
Pore shape	-0.06	0.12	-0.56*	-0.47*	-0.62*	
Pore sieve	0.02	-0.21	0.31	-0.40*	-0.40*	0.16

Table 7.2
Pearson product moment correlation coefficients for selected site-specific soil attributes from Year 2. Values marked * are significant at $P \leq 0.05$.

	Log K10	BD	Ave. macro-porosity	Ave. meso-porosity	Topsoil macro-porosity	Topsoil meso-porosity	Macro-porosity @ 100	Pore count	Pore surface area	Pore shape
BD	-0.29									
OC	-0.29	-0.29								
Ave. macroporosity	0.40*	0.13								
Ave. mesoporosity	0.43*	0.02	0.98*							
Topsoil macroporosity	0.41*	-0.03	0.94*	0.97*						
Topsoil mesoporosity	0.39*	-0.03	0.90*	0.97*	0.97*					
Macroporosity @ 100	0.39*	0.29	0.80*	0.74*	0.66*	0.56*				
Pore count	0.16	-0.12	0.30	0.48*	0.37*	0.53*	0.12			
Pore surface area	0.37*	0.04	0.81*	0.90*	0.77*	0.86*	0.60*	0.80*		
Pore shape	-0.14	-0.22	-0.67*	-0.69*	-0.58*	-0.58*	-0.67*	-0.49*	-0.70*	
Pore sieve	0.18	0.01	0.33	0.17	0.31	0.12	0.35	-0.58*	-0.20	0.19

Table 7.3
Pearson product moment correlation coefficients for selected site-specific soil attributes from Year 3. Values marked * are significant at $P \leq 0.05$.

	Log K10	BD	Ave. macro-porosity	Ave. meso-porosity	Topsoil macro-porosity	Topsoil meso-porosity	Macro-porosity @ 100	Pore count	Pore surface area	Pore shape
BD	-0.06									
OC	0.10	-0.21								
Ave. macroporosity	0.09	-0.10								
Ave. mesoporosity	0.15	0.00	0.96*							
Topsoil macroporosity	0.00	-0.16	0.98*	0.91*						
Topsoil mesoporosity	0.07	-0.03	0.97*	0.98*	0.96*					
Macroporosity @ 100	0.32	0.10	0.66*	0.75*	0.48*	0.60*				
Pore count	0.24	0.11	0.57*	0.74*	0.45*	0.63*	0.74*			
Pore surface area	0.20	0.05	0.84*	0.94*	0.75*	0.88*	0.80*	0.91*		
Pore shape	-0.14	-0.17	-0.43*	-0.56*	-0.36	-0.52*	-0.51*	-0.64*	-0.63*	
Pore sieve	-0.33	-0.13	-0.02	-0.23	0.08	-0.12	-0.37	-0.66*	-0.43	0.52*

7.2.2 Principal Component Analysis

Principal component analysis revealed that in Year 2, the first three components accounted for 86.9% of the variance, and in Year 3 the first three components accounted for 83.1% of the variance, with data derived from image analysis contributing strongly to the first component in both cases. This is shown in the summaries in Tables 7.4 and 7.5 of the first three eigenvalues and the eigenvectors for the correlation matrices.

Table 7.4
Eigenvalues and eigenvectors for the first 3 components using site-specific data from Year 2.

Statistic	Component		
	1	2	3
Eigenvalue	6.04	2.05	1.46
% of variance	54.9	18.7	13.3
Cumulative %	54.9	73.6	86.9
Soil attribute	Eigenvector		
BD	0.018	0.129	-0.759
log K_{10}	0.157	0.226	0.521
topsoil macroporosity	0.385	0.108	0.077
topsoil mesoporosity	0.382	-0.034	0.085
average macroporosity	0.315	0.228	-0.221
average mesoporosity	0.395	0.150	-0.000
macroporosity at 100 mm	0.403	0.020	-0.005
macropore surface area	0.368	-0.267	0.010
macropore count	0.174	-0.592	0.099
macropore shape	-0.306	0.164	0.276
macropore sieve	0.072	0.629	0.065

Table 7.5
Eigenvalues and eigenvectors for the first 3 components using site-specific data from Year 3.

Statistic	Component		
	1	2	3
Eigenvalue	6.01	2.05	1.09
% of variance	54.6	18.6	9.9
Cumulative %	54.6	73.2	83.1
Soil property	Eigenvector		
BD	0.016	-0.251	-0.741
log K_{10}	0.106	-0.302	0.614
topsoil macroporosity	0.325	0.403	0.009
topsoil mesoporosity	0.368	0.265	-0.064
average macroporosity	0.341	-0.145	0.056
average mesoporosity	0.362	0.311	0.021
macroporosity at 100 mm	0.394	0.165	-0.033
macropore surface area	0.355	-0.249	0.066
macropore count	0.349	-0.258	-0.038
macropore shape	-0.268	0.209	0.238
macropore sieve	-0.159	0.544	-0.043

Varimax rotation of the principal components was investigated to enable the initial interpretation to be refined, with the relative contributions to related components showing greatest discrimination when the first three components were rotated. For Year 2, these results are summarised in Table 7.6, which lists the rotated factor scores and communalities for each of the site-specific soil attributes.

Table 7.6
Rotated factor scores and communalities for site-specific soil attributes using a 3 factor model of Year 2 data. The maximum factor score for each soil attribute is shaded.

Soil attribute	Rotated factor scores			Communalities
	Factor 1	Factor 2	Factor 3	
BD	0.146	0.197	-0.904	0.878
log K_{10}	0.344	0.286	0.672	0.652
topsoil macroporosity	0.940	0.089	0.187	0.927
topsoil mesoporosity	0.919	-0.114	0.191	0.894
average macroporosity	0.816	0.279	-0.184	0.778
average mesoporosity	0.977	0.149	0.098	0.987
macroporosity at 100 mm	0.986	-0.038	0.089	0.981
macropore surface area	0.873	-0.443	0.090	0.965
macropore count	0.357	-0.877	0.140	0.916
macropore shape	-0.764	0.280	0.266	0.732
macropore sieve	0.228	0.886	0.117	0.851

In Table 7.6, the shaded cells in any column represent groups of soil attributes that demonstrate the strongest alignment with each factor; i.e. the largest factor score for each soil attribute. There is strong alignment of topsoil macroporosity, topsoil mesoporosity, average macroporosity, average mesoporosity, macropore surface area and macropore shape with Factor 1, although the lower communality for macropore shape is noted. Macropore count and macropore sieve align most strongly with Factor 2, and bulk density and unsaturated hydraulic conductivity align most strongly with Factor 3 (although the low communality for unsaturated hydraulic conductivity is noted here also).

Factor scores have been calculated for each treatment, with mean factor scores from site-specific data of Year 2 presented in Table 7.7. Treatment CA is significantly different from all other treatments for Factor 1, and treatment SS is significantly different from all other treatments for Factor 2.

Table 7.7
Mean factor scores for each treatment derived from site-specific data of Year 2.

Factor	Treatment			
	SS	HI-SD	C	CA
Factor 1	-0.580 (0.132) a	-0.018 (0.227) a	-0.377 (0.335) a	1.225 (0.428) b
Factor 2	-0.626 (0.232) a	0.166 (0.455) b	1.236 (0.187) b	0.225 (0.264) b
Factor 3	-0.183 (0.157)	0.331 (0.333)	-0.239 (0.292)	0.166 (0.671)

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Results for Year 3 site-specific data are summarised in Tables 7.8 and 7.9.

Table 7.8
Rotated factor scores and communalities for site-specific soil attributes using a 3 Factor model of Year 3 data. The maximum factor score for each soil attribute (other than macropore shape) is shaded.

Soil attribute	Rotated factor scores			Communalities
	Factor 1	Factor 2	Factor 3	
BD	-0.124	0.218	**-0.816**	0.729
log K_{10}	-0.078	0.570	**0.578**	0.666
topsoil macroporosity	**0.980**	-0.006	0.076	0.966
topsoil mesoporosity	**0.959**	0.204	-0.028	0.962
average macroporosity	0.560	**0.656**	0.016	0.745
average mesoporosity	**0.979**	0.154	0.069	0.987
macroporosity at 100 mm	**0.927**	0.362	-0.018	0.990
macropore surface area	0.502	**0.798**	0.005	0.888
macropore count	0.490	**0.787**	-0.104	0.870
macropore shape	-0.382	-0.590	0.297	0.582
macropore sieve	0.131	**-0.859**	0.070	0.760

A similar pattern of factor alignment emerges from Table 7.8, although macropore surface area is more strongly aligned with Factor 2, and the low communality of macropore shape indicates weak alignment with any of the three factors in this model. Table 7.9 compares factor scores for each treatment from Year 3 data, with treatment SS being significantly different from all other treatments in Factor 1. There were no significant differences between treatments for Factor 2 or Factor 3.

Table 7.9
Mean factor scores for each treatment derived from site-specific data of Year 3.

Factor	Treatment			
	SS	HI-SD	C	CA
Factor 1	-0.984 (0.178) a	0.732 (0.202) b	-0.076 (0.295) b	0.810 (0.263) b
Factor 2	-0.142 (0.336)	0.309 (0.475)	0.137 (0.050)	-0.121 (0.413)
Factor 3	0.042 (0.388)	0.264 (0.408)	-0.805 (0.395)	0.059 (0.272)

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

Factor 1 is associated with macroporosity and mesoporosity data derived from image analysis. Factor 2 is associated with the nature of macropores, and Factor 3 appears to be associated with bulk density. The results summarised in Tables 7.7 and 7.9 provide evidence that differences between treatments become more apparent when multiple soil attributes are considered. Treatments CA and SS are clearly different from other treatments for Factors 1 and 2. The inclusion of multiple image analysis measurements using factor analysis may be a more convenient method to distinguish treatment effects. The lack of differences between treatments for Factor 3 confirms the superiority of image analysis over surrogate measures of soil structure to distinguish between treatments.

7.3 Comparison of plot average data

7.3.1 Correlation between soil and pasture properties

Correlation coefficients for all soil and pasture properties estimated from the average of values measured from each plot for Year 2 are shown in Tables 7.10a and 7.10b, with significant correlation noted. Apart from correlation between certain image analysis attributes, expected given the algorithms to determine porosity attributes by image analysis, significant correlations between pasture botanical composition and various macropore characteristics, unsaturated hydraulic conductivity and penetration resistance 0 – 105 mm, and for macropore surface area with other macropore characteristics, are apparent. The lack of correlation between macropore count, macropore shape and other soil attributes is also noted.

Table 7.10a
Correlation coefficients for soil and pasture properties estimated from the average of values measured from each plot for Year 2. Values marked * are significant at P ≥ 0.05.

	Phalaris content	Perennial grass content	Palatable perennial grass content	Pen. Resist. 0-105	Pen. Resist. 0-300	CSA	BD	OC	Log K10
Perennial grass content	-0.16								
Palatable perennial grass content	0.60	0.50							
Pen. Resist. 0-105	0.13	-0.79*	-0.47						
Pen. Resist. 0-300	0.53	-0.73*	-0.30	0.79*					
CSA	-0.53	-0.17	-0.26	-0.05	-0.41				
BD	0.60	-0.22	-0.08	0.28	0.60	-0.46			
OC	0.25	0.42	0.58	0.09	-0.09	-0.37	-0.18		
Log K10	-0.11	-0.84*	-0.42	0.51	0.26	0.52	-0.06	-0.51	
Topsoil macroporosity	0.38	-0.75*	-0.19	0.53	0.69	-0.28	0.32	-0.33	0.58
Topsoil mesoporosity	0.37	-0.64	-0.11	0.44	0.62	-0.37	0.27	-0.27	0.49
Average macroporosity	0.50	-0.72*	-0.10	0.35	0.68	-0.27	0.37	-0.46	0.53
Average mesoporosity	0.62	-0.62	0.05	0.37	0.71*	-0.47	0.38	-0.22	0.37
Macroporosity @ 100	0.35	-0.60	0.00	0.15	0.34	0.13	0.24	-0.56	0.66
macropore surface area	0.28	-0.49	0.06	0.19	0.33	-0.23	0.03	-0.26	0.51
macropore count	0.09	-0.11	0.26	-0.09	-0.04	-0.24	-0.27	-0.02	0.25
macropore shape	-0.24	-0.23	-0.25	0.18	-0.08	0.57	-0.27	-0.05	0.27
macropore sieve	0.08	-0.84*	-0.57	0.81*	0.63	0.28	0.35	-0.29	0.69

Table 7.10b
Correlation coefficients for soil and pasture properties estimated from the average of values measured from each plot for Year 2 (cont'd.). Values marked * are significant at P ≥ 0.05.

	Topsoil macro-porosity	Topsoil meso-porosity	Average macro-porosity	Average meso-porosity	Macro-porosity @ 100	Pore surface area	Pore count	Pore shape
Topsoil mesoporosity	0.99*							
Average macroporosity	0.93*	0.91*						
Average mesoporosity	0.93*	0.94*	0.96*					
Macroporosity @ 100	0.81*	0.77*	0.84*	0.74*				
Macropore surface area	0.90*	0.94*	0.84*	0.85*	0.82*			
Macropore count	0.63	0.72	0.52	0.60	0.57	0.89*		
Macropore shape	-0.44	-0.55	-0.33	-0.45	-0.36	-0.57	-0.66	
Macropore sieve	0.35	0.20	0.32	0.21	0.22	0.00	-0.38	0.54

Correlation coefficients for all soil and pasture properties estimated from the average of values measured from each plot for Year 3, representing the conclusion of the experiment, are shown in Tables 7.11a and 7.11b. Significant correlation between cotton strip assay and the following soil and pasture properties is observed: Phalaris content, average macroporosity and average and topsoil mesoporosity, macroporosity at 100 mm depth, macropore surface area and macropore count. Note that negative correlation is expected, with a low value for cotton strip assay in this data set representing a low cloth tensile strength and therefore increased microbial activity. Perennial grass content is negatively correlated with macropore sieve and macropore shape but positively correlated with macropore count, as was macropore count with phalaris content.

Table 7.11a

Correlation coefficients for soil and pasture properties estimated from the average of values measured from each plot for Year 3. Values marked * are significant at P ≥ 0.05.

	Phalaris content	Perennial grass content	Palatable perennial grass content	Pen. Resist. 0-105	Pen. Resist. 0-300	CSA	BD	OC	Log K10
Perennial grass content	0.67								
Palatable perennial grass content	0.55	0.96*							
Pen. Resist. 0-105	-0.00	-0.23	-0.15						
Pen. Resist. 0-300	0.18	-0.04	0.06	0.87*					
CSA	-0.76*	-0.61	-0.52	0.36	-0.04				
BD	-0.03	-0.04	-0.02	0.61	0.46	0.17			
OC	0.40	0.30	0.10	-0.29	-0.38	-0.28	-0.23		
Log K10	0.24	0.27	0.16	-0.43	-0.67	0.13	-0.44	0.48	
Topsoil macroporosity	0.31	0.32	0.23	-0.54	-0.13	-0.75	-0.40	0.08	-0.21
Topsoil mesoporosity	0.53	0.42	0.27	-0.43	-0.06	-0.83*	-0.30	0.29	-0.13
Average macroporosity	0.41	0.48	0.39	-0.55	-0.14	-0.83*	-0.32	0.12	-0.19
Average mesoporosity	0.56	0.63	0.52	-0.54	-0.16	-0.89*	-0.26	0.20	-0.07
Macroporosity @ 100	0.67	0.63	0.50	-0.40	-0.02	-0.91*	-0.22	0.25	-0.08
macropore surface area	0.70	0.56	0.38	-0.32	0.00	-0.91*	-0.15	0.50	-0.10
macropore count	0.92*	0.84*	0.71	-0.36	-0.14	-0.83*	-0.29	0.44	0.39
macropore shape	-0.46	-0.81*	-0.74	0.33	0.29	0.41	-0.31	-0.12	-0.30
macropore sieve	-0.38	-0.77*	-0.68	0.68	0.69	0.38	0.13	-0.37	-0.62

Table 7.11b
Correlation coefficients for soil and pasture properties estimated from the average of values measured from each plot for Year 3 (cont'd.). Values marked * are significant at P ≥ 0.05.

	Topsoil macro-porosity	Topsoil meso-porosity	Average macro-porosity	Average meso-porosity	Macro-porosity @ 100	Pore surface area	Pore count	Pore shape
Topsoil mesoporosity	0.95*							
Average macroporosity	0.98*	0.96*						
Average mesoporosity	0.93*	0.95*	0.98*					
Macroporosity @ 100	0.90*	0.97*	0.95*	0.98*				
Macropore surface area	0.80*	0.94*	0.85*	0.98*	0.94*			
Macropore count	0.53	0.68	0.62	0.75	0.80*	0.77*		
Macropore shape	-0.24	-0.31	-0.37	-0.52	-0.47	-0.37	-0.58	
Macropore sieve	-0.17	-0.20	-0.30	-0.43	-0.33	-0.17	-0.60	0.82*

7.3.2 Principal component analysis

Eigenvalues and eigenvectors for the first 3 components, accounting for 83.1% of the variance derived from results for the plot average of soil and pasture data from Year 2, are shown in Table 7.12. The factor pattern from rotation of three components is summarised in Table 7.13, with the maximum factor scores for each soil attribute shaded to better identify alignment with the factors. Topsoil macroporosity, topsoil mesoporosity, average macroporosity, average mesoporosity, macroporosity at 100 mm depth and macropore surface area associated with Factor 1. The soil and pasture attributes of unsaturated hydraulic conductivity, palatable grass content, perennial grass content, penetration resistance 0 – 105 mm and macropore count, shape and sieve are associated with Factor 2. Bulk density, CSA, penetration resistance 0 – 300 mm and Phalaris content are associated with Factor 3. Low communalities are recorded for organic carbon content and palatable perennial grass content. Calculation of factor scores for the two grazing treatments showed no significant differences for any factor.

Table 7.12
Eigenvalues and eigenvectors for the first 3 components using the plot average of soil and pasture data from Year 2.

Statistic	Component		
	1	2	3
Eigenvalue	8.15	4.11	2.69
% of variance	45.3	22.8	14.9
Cumulative %	45.3	68.1	83.1
Attribute	**Eigenvector**		
phalaris content	0.155	0.206	0.078
perennial grass content	-0.282	0.281	-0.139
palatable perennial grass content	-0.068	0.360	0.304
penetration resistance, 0-105	0.193	-0.256	0.330
penetration resistance, 0-300	0.259	-0.099	0.010
CSA	-0.084	-0.317	0.069
BD	0.138	0.013	-0.472
OC	-0.131	0.194	0.608
log K10	0.217	-0.284	0.169
topsoil macroporosity	0.346	0.039	0.027
topsoil mesoporosity	0.334	0.108	0.037
average macroporosity	0.336	0.044	-0.100
average mesoporosity	0.328	0.136	0.023
macroporosity @ 100	0.289	0.033	-0.128
macropore surface area	0.297	0.170	0.107
macropore count	0.185	0.286	0.217
macropore shape	-0.122	-0.381	0.227
macropore sieve	0.155	-0.406	0.089

Table 7.13
Rotated component factor scores and communalities for the average of soil and pasture properties using Year 2 data and a 3 factor model. The maximum factor score for each soil attribute is shaded (except for organic carbon content).

Attribute	Rotated factor score			Communality
	Factor 1	Factor 2	Factor 3	
phalaris content	0.253	0.152	**0.792**	0.715
perennial grass content	-0.475	**0.845**	-0.177	0.971
palatable perennial grass content	0.033	**0.733**	0.202	0.580
penetration resistance, 0-105	0.115	**-0.730**	0.434	0.736
penetration resistance, 0-300	0.296	-0.533	**0.752**	0.936
CSA	-0.166	-0.438	**-0.760**	0.797
BD	0.048	-0.193	**0.750**	0.602
OC	-0.377	0.471	0.357	0.491
log K10	0.510	**-0.739**	-0.285	0.887
topsoil macroporosity	**0.881**	-0.322	0.327	0.986
topsoil mesoporosity	**0.915**	-0.177	0.317	0.969
average macroporosity	**0.849**	-0.302	0.343	0.930
average mesoporosity	**0.835**	-0.136	0.497	0.963
macroporosity @ 100	**0.871**	-0.241	-0.006	0.818
macropore surface area	**0.988**	0.003	0.060	0.979
macropore count	0.361	**0.868**	-0.144	0.903
macropore shape	-0.581	**-0.569**	-0.248	0.723
macropore sieve	-0.030	**-0.954**	0.226	0.962

For Year 3 results, eigenvalues and eigenvectors for the first 3 components, accounting for 85.3% of the variance derived from results for the plot average of soil and pasture data, are shown in Table 7.14.

The factor pattern from rotation of three components is summarised in Table 7.15. This shows strong alignment of topsoil macroporosity, topsoil mesoporosity, average macroporosity, average mesoporosity, macroporosity at 100 mm depth, macropore surface area, and CSA with Factor 1. This factor will be nominated as the macroporosity factor. Table 7.15 also shows strong alignment of penetration resistance, bulk density and unsaturated hydraulic conductivity with Factor 2, nominated as the soil strength factor. There is strong alignment of pasture botanical composition and macropore count, shape and sieve with Factor 3, nominated as the pasture factor. The communality of organic carbon content prevents its useful inclusion in any of the three factors.

Table 7.14
Eigenvalues and eigenvectors for the first 3 components using the plot average of soil and pasture data from Year 3

Statistic	Component		
	1	2	3
Eigenvalue	9.37	3.31	2.67
% of variance	52.1	18.4	14.8
Cumulative %	52.1	70.5	85.3
Attribute	Eigenvector		
Phalaris content	0.240	0.019	0.275
perennial grass content	0.259	-0.119	0.328
palatable perennial grass content	0.216	-0.086	0.370
penetration resistance, 0-105	-0.182	0.287	0.339
penetration resistance, 0-300	-0.075	0.445	0.295
CSA	-0.301	-0.139	-0.048
BD	-0.090	0.147	0.436
OC	0.123	-0.204	-0.083
log K10	0.051	-0.485	-0.027
topsoil macroporosity	0.258	0.799	-0.271
topsoil mesoporosity	0.283	0.197	-0.180
average macroporosity	0.285	0.171	-0.186
average mesoporosity	0.310	0.115	-0.092
macroporosity @ 100	0.308	0.164	-0.050
macropore surface area	0.290	0.169	-0.050
macropore count	0.294	-0.074	0.128
macropore shape	-0.202	0.192	-0.298
macropore sieve	-0.198	0.397	-0.121

Table 7.15
Rotated component factor scores and communalities for the average of soil and pasture properties using Year 3 data and a 3 factor model. The maximum factor score for each soil attribute is shaded (except for organic carbon content).

Attribute	Rotated factor score			Communality
	Factor 1	Factor 2	Factor 3	
Phalaris content	0.426	-0.067	0.747	0.744
perennial grass content	0.315	0.108	0.926	0.969
palatable perennial grass content	0.214	-0.013	0.885	0.830
penetration resistance, 0-105	-0.383	-0.863	-0.050	0.894
penetration resistance, 0-300	0.043	-0.970	-0.011	0.942
CSA	-0.821	-0.001	-0.501	0.924
BD	-0.348	-0.664	0.310	0.658
OC	0.162	0.474	0.224	0.301
log K10	-0.301	0.780	0.327	0.806
topsoil macroporosity	0.966	0.151	-0.009	0.956
topsoil mesoporosity	0.968	0.096	0.150	0.969
average macroporosity	0.952	0.140	0.160	0.952
average mesoporosity	0.905	0.163	0.352	0.969
macroporosity @ 100	0.919	0.053	0.374	0.988
macropore surface area	0.882	0.031	0.342	0.895
macropore count	0.560	0.241	0.709	0.875
macropore shape	-0.129	-0.194	-0.831	0.746
macropore sieve	-0.042	-0.644	-0.719	0.933

Mean factor scores calculated from each grazing treatment are summarised in Table 7.16. A number of soil and pasture attributes were not measured in treatments C and CA so factor score calculation was not possible for these treatments. The two grazing treatments were significantly different for Factor 1, the macroporosity factor, but not the other factors.

Table 7.16
Mean factor scores for each treatment derived from site-specific data of Year 3.

Factor	Treatment	
	SS	HI-SD
Factor 1	0.984 (0.284) a	-0.738 (0.189) b
Factor 2	-0.031 (0.303)	0.024 (0.673)
Factor 3	0.278 (0.815)	-0.208 (0.367)

Refer to text for treatment descriptions. Standard error of the mean shown in parentheses. In any row, pairs of values with the same letter are not significantly different at $P \leq 0.05$ (Tukey–Kramer test of honestly significant difference). In rows where no letters are shown, there are no differences ($P \leq 0.05$).

7.4 Discussion

These analyses confirm the importance of image analysis as a useful tool to provide sensitive discrimination between soil treatments in this experiment, with many of the primary image analysis results strongly aligned with the first principal component. However, the separation of the soil properties of macropore count, shape and sieve, also derived from image analysis, into a different factor requires some explanation.

It is noted that in this experiment there were only small differences in macropore shape between soil treatments, with average macropore shape factor in Year 3 ranging from 0.562 to 0.585 across all treatments, and from 0.485 to 0.585 across the full data set of all years and all treatments. Given that the maximum possible range of macropore shape factor is between 0 and 1, this indicates a very small variation in this property. Two further points can be made. Firstly, the moderate degree of macropore roundness in these results is unlikely to create any limitation to soil processes, with the degree of roundness likely to influence soil quality only at extreme values. Secondly, a large degree of roundness does not necessarily imply high soil quality. For example, a large degree of roundness cannot improve soil quality if the size, number or continuity of macropores is limiting. To illustrate this point, the sample images in Figure 6.1 display a wide range of porosities but a narrow range of macropore shape factor. Macropore sieve, defined as the average spherical equivalent of macropores, may be a useful single measure of macropore size, but takes no account of the number of macropores or the total macropore volume, shape or surface area. For many soil processes, macropore surface area may represent a superior indicator of soil quality as a measure of the exchange surface for water and gasses, as long as transmission characteristics provided by macropore size, shape and continuity are satisfactory. Macropore sieve, defined as the average spherical equivalent of macropores, may be a useful single measure of macropore size, but takes no account of total macropore volume, shape or surface area.

The relationship between pasture botanical composition and soil structural properties takes a number of forms. Positive correlation is demonstrated between pasture perennial grass content (palatable and total), particularly for Phalaris, and macroporosity, mesoporosity, macroporosity at 100 mm depth, macropore surface area and macropore count. Negative correlation is recorded with macropore shape and sieve. These trends are confirmed by principal component analysis, where the pasture factor identified in Year 3 shows similar relationships.

Although some caution may be required before establishing a causal relationship between pasture botanical composition and soil quality, the role of perennial grasses in improving soil quality in general, and macroporosity in particular, has been noted by other authors. Ridley (1996) speculates that soil hydraulic properties under Phalaris (compared to annual grasses) are better (increased infiltration, increased water storage and increased water usage) because of the larger number of Phalaris roots greater than 1 mm at a soil depth of 0-10 cm, and the large number of Phalaris roots greater than 1 mm diameter to at least 0.7 m soil depth, with annual ryegrass having zero roots of this size at this depth. McCallum et al. (2004) measured a subsoil macropore count (greater than 2 mm diameter) of $190/m^2$ at 120 mm depth in a Sodosol after 10 years of Phalaris growth, compared to a mean of $68/m^2$ after annual crops.

The negative correlation between perennial grass content (PG) and macropore sieve and shape remains of interest. Note that correlation between palatable perennial grass content (PPG) and these two macropore properties is present but not significant. The difference between 'perennial grass content' and 'palatable perennial grass content' is the inclusion of Yorkshire Fog in the former, indicating that the presence of Yorkshire Fog in the pasture has had some influence over macropore characteristics. Whilst the direct impact of this species remains unclear, although possibly related to soil densification, it confirms that pasture botanical composition is an important consideration.

Total organic carbon content consistently exhibited weak correlation or uncertain alignment in principal component investigations. This may be attributable to the relatively high and stable organic carbon content in all plots, the similarity of treatments in their impact on soil carbon and the inability of the Walkley-Black method used in this experiment to separate and compare the labile soil carbon fractions. By contrast, Brejda et al. (2000a and 2000b) found that total organic carbon was the single most useful indicator in discriminating between soil management practices in a study to identify those properties most useful as regional indicators of soil quality, although they also acknowledge that over short time intervals, more labile fractions of soil carbon may need to be evaluated. It should be noted that the investigations of Brejda et al (2000a and 2000b) were regional in scale, covering widely different soil types, with mean organic carbon contents varying from 0.65% to 3.7% for the different land uses, including conservation reserves and forests.

McKenzie (2001b) undertook an extensive comparison of measures of soil structure in a Vertosol under cotton. He included image analysis measurements of macroporosity, macropore surface area, macropore star length and macrosolid star length (indices of pore and clod diameters) in conjunction with the following surrogate measures of soil structure: shear strength, bulk density, air-filled porosity, clod shrinkage parameters and visual/tactile assessment using the SOILpak score (McKenzie 2001a). Correlation coefficients for these soil attributes are summarised in Table 7.17, with McKenzie (2001b) finding that all these indicators, including bulk density, distinguished compacted soil from well structured soil in a Vertosol growing cotton. Contrary to the correlations found in the experiment at Orange reported here, McKenzie found significant correlation between soil bulk density and macroporosity as determined by image analysis, as well as finding best correlation between soil bulk density and shear strength. Importantly, McKenzie found significant correlation between the SOILpak score and all other measures of soil structure, including strong correlation between SOILpak score and bulk density. This emphasises the absence of a single superior field measure of soil structure, and the usefulness of visual and tactile assessment by skilled observers in the field.

Coughlan et al. (1991) warn against transferring conclusions from Vertosols to other soil types, so the results reported by McKenzie for a non-rigid Vertosol may not be transferable to rigid soil types or those with distinct horizons. The wide range of soil compaction conditions should also be noted (the comparison of within-row ridge soil to the wheel trafficked furrow of a cotton field) as should the high clay content and compaction susceptibility of the soil type compared to the soil conditions for grazing systems reported here. Hydraulic properties were not measured. Consequently, the results reported by McKenzie are relevant to diagnosis of widely different soil compaction conditions and may not be as conclusive in detecting subtle differences in soil structure imposed by grazing management tactics.

Table 7.17
Pearson product-moment correlation coefficients for selected soil attributes measured in a Vertosol under cotton (McKenzie 2001b). Values marked * are significant at P ≥ 0.05

1. soil shear strength
2. Macroporosity
3. macropore surface area
4. macropore star length
5. macrosolid star length (an index of clod diameter)
6. BD
7. Air filled porosity
8. Volume of air filled pores at zero water content
9. SOILpak score

Soil attribute	1	2	3	4	5	6	7	8
2	-0.50							
3	-0.49	0.62*						
4	-0.35	0.60*	0.49					
5	0.40	-0.83*	-0.84*	-0.64*				
6	0.81*	-0.58*	-0.55*	-0.32	0.52			
7	-0.76*	0.62*	0.63*	0.23	-0.60*	-0.95*		
8	-0.59*	0.55*	0.57*	0.30	-0.35	-0.49	0.61*	
9	-0.77*	0.73*	0.71*	0.56*	-0.65*	-0.80*	0.76*	0.65*

Although doubt has been cast on the usefulness of the cotton strip assay compared to more sophisticated methods to measure soil microbial activity, this experiment has measured strong correlation between cotton strip assay results and many soil structural properties. This has been hypothesised by several researchers (for example, King 1996) but via comparison of CSA results with other measures of microbial activity. The strength of alignment of CSA with Factor 1, the macroporosity factor, in Year 3 implies some correlation between macroporosity characteristics and soil microbial activity. It has been suggested that the nature of the macroporosity in soil partly determines the physical habitat of soil flora and fauna (Juma 1993, Pankhurst et al. 1994, Turco et al. 1994) and this relationship may be demonstrated with these results. It may also be derived from a secondary relationship, in that macroporosity characteristics will partly determine soil water transmission and content, thereby further influencing soil microbial populations. The alignment of macropore surface area with Factor 1 is also of interest - the surface area of macropores may be a more useful indicator of the total microbial habitat space than the number of pores and their sieve dimension.

Shestak and Busse (2005) subjected soil cores from two different soil types to compaction in the laboratory to represent increases of 20% and 40% in soil bulk density. A large number of soil microbial indices (microbial biomass, microbial respiration, total and culturable bacteria and fungi, N mineralisation, surface CO_2 efflux, carbon use using the Biolog method, and phospholipid fatty acids) and soil physical indices (bulk density, pore size distribution, water holding capacity and gas diffusion) were measured over a 67 day period. Microbial measures were either unaffected by simulated compaction or showed inconsistent increases across sampling dates and soil types. They found that whilst macropores greater than 30 μm were reduced in number by between 50% and 90% in compacted soil cores, habitable sized pores for microorganisms (defined by Shestak and Busse (2005) as the range in macropores 0.2 μm – 30 μm) increased by at least 40%. Further tests of microbial respiration and biomass in field soils (in this case under a mixed conifer plantation) showed no significant difference across soil strengths ranging from 75 to 3800 kPa. They concluded that biological and physical indices of soil quality, specifically soil bulk density and penetration resistance, were poorly linked. Abdel-Magid et al. (1987) also found inconsistent trends for bulk density and ponded water infiltration rate under different stocking rates. Arshad et al. (1999) note that the distribution of pore sizes can alter water retention characteristics without changing bulk density, further evidence of the inability of bulk density measurements to provide meaningful interpretation.

7.5 Conclusion

Average macroporosity as determined by image analysis is considered a superior measure of porosity than bulk density, and macroporosity at 100 mm depth is considered a useful measure of macropore continuity. The nature of macropores, as defined by their shape, number and surface area, has influenced the comparison of treatments in this experiment.

Based on the results of this and other experiments, the relative importance of bulk density measurements, and to a lesser extent, hydraulic properties and penetration resistance, should be discounted in preference to direct assessment of soil structure in the modelling of grazing strategies and their impact on soil quality, where the impact may be much less than, for example, the impact of tillage systems on similar soil types.

The results from this experiment provide evidence to confirm the importance of perennial grass to the management of soil structure, and support the conclusions of Michalk et al. (2003). The apparent importance of soil microbial activity, as described by the correlation between CSA results and other macropore characteristics, is noted but not quantified. The more continuous presence of active pasture roots provided by perennial species, the greater range of pore spaces provided by a soil with greater macroporosity, and possible changes to soil water relations appears to have had an influence in this experiment.

Consequently, it is concluded from the results of this experiment that alternative grazing tactics should be considered as one of the strategies for protecting soil structural condition.

CHAPTER 8

A FUZZY MODEL APPROACH TO SOIL QUALITY ASSESSMENT

8.1 Introduction

Fuzzy logic has potential as a methodology for modeling soil quality interactions, because it can combine data and observations from diverse classes of information, and allows the expert judgement of the investigator, based on linguistic interpretation if necessary, to be incorporated into a decision tool. This chapter describes this potential and its application to the results of this experiment. A complete description of fuzzy methodology is provided by McBratney and Odeh (1997), Center and Verma (1998), Odeh and McBratney (2002), Kantardzic (2003) and others.

8.2 Membership Functions – Fuzzy versus Crisp

Assessment of soil quality relies on the measurement of key indicators and the allocation of specific values to a quality class; for example, classes with linguistic names such as 'good', 'fair' or 'poor' following the methodology of Walker and Reuter (1996). Such an approach provides for a comparison between measured values and preferred values and a level of satisfaction to be determined. If the level of satisfaction is sufficiently low, a decision can be made on intervention.

The degree that an object fits a particular class is defined by its membership function; more specifically, the grade of membership. In crisp sets, the grade of membership can only be bi-valued (1 or 0). In fuzzy sets, the grade of membership (or degree of fulfillment) can hold any value between or including 1 and 0. The shape of the membership function can be determined by a range of techniques, including experimental data, expert or community opinion, and survey results.

By way of example, consider the allocation of soil pH into classes. Figure 8.1 uses the interpretation of Walker and Reuter (1996), earlier described in Table 2.3, to establish a subjective classification of 'moderately acid' soil pH. Note the use of language ('moderately') to apply meaning to this classification, subject to a range of interpretations and of relevance to the specific goals of an investigation or action, yet necessary to make judgement of quality and to decide subsequent management intervention. Under a crisp classification, soil of pH less than 5.0 or more than 5.5 cannot be a member of this set and must be classified differently; a soil with pH of 4.95 is 'not moderately acid'. Figure 8.2 illustrates the same classification using alternative membership functions (bell shaped and trapezoidal) where the boundary of the classes extends across a larger range of values.

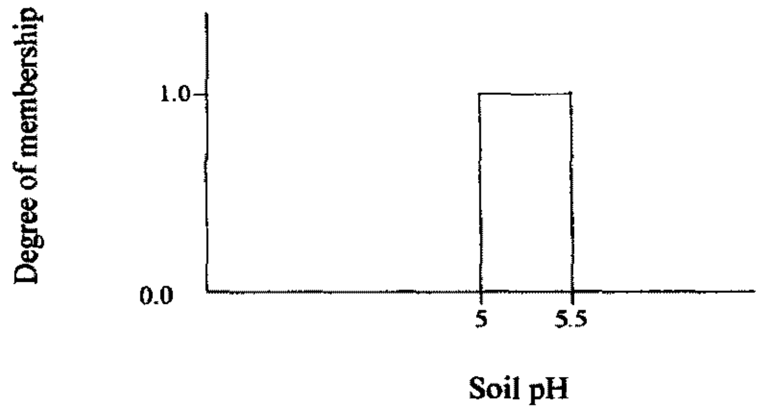


Figure 8.1
Crisp set membership function for 'moderately acid' soil pH according to Walker and Reuter (1996).

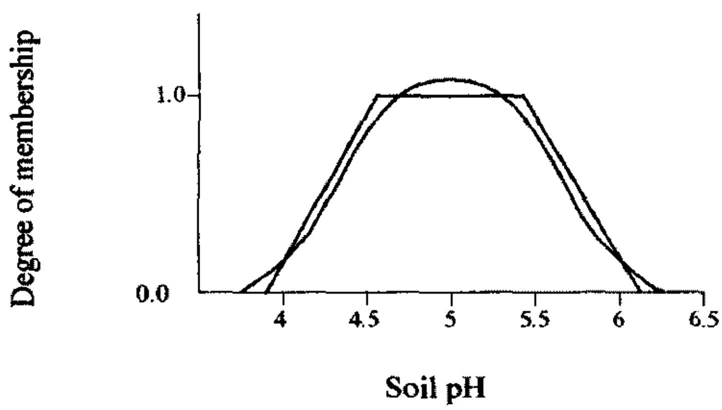


Figure 8.2
Bell and trapezoidal shaped fuzzy membership functions for data of Figure 8.1.

Figure 8.3 extends this comparison to multiple classes of soil pH, again using the classification of Walker and Reuter (1996). Immediately, one of the limitations of a crisp approach to soil quality classification becomes apparent: that soil with a pH > 4.5 but < 5.0 is not allocated to a class. Obviously, a different boundary condition can be decided, but additional difficulties are also present. For example, a soil which has a pH of 5.49 is classed as 'poor', whereas soil with a pH of 5.50 is classed as 'good'. Such boundaries are inevitable in a crisp approach but do not reflect real phenomena such as plant response or accuracy of measurement.

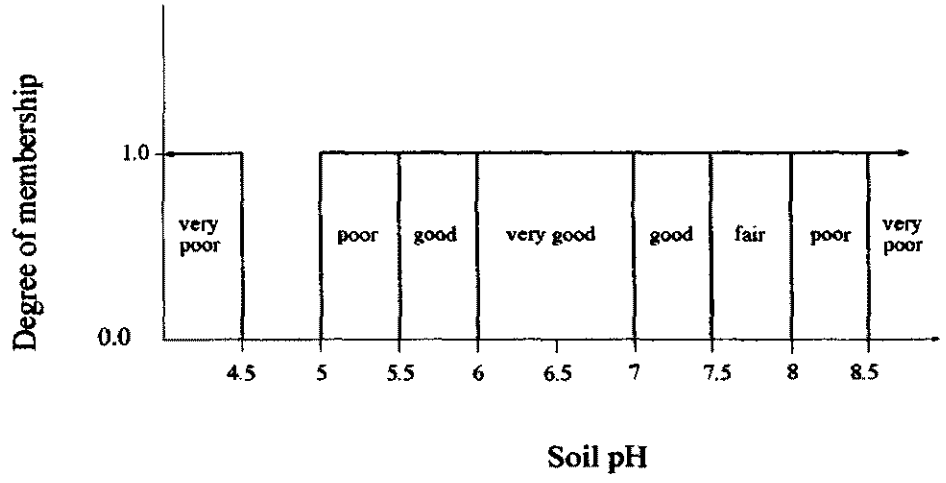


Figure 8.3
Allocation of soil pH classes from Walker and Reuter (1996) using crisp membership.

Figure 8.4 illustrates the same classification purpose using fuzzy set membership based on trapezoidal membership functions. Under this scheme, a soil with a pH between 4.5 and 5.0 would be partly 'poor' and partly 'very poor', and a soil with pH 4.95 would be mostly 'poor', thereby eliminating the problem of class boundaries with crisp membership. Figures 8.5 to 8.8 compare two additional soil properties using crisp and fuzzy approaches based on the classification boundaries of Walker and Reuter (1996). Discontinuous class boundaries are again problematic, and in Figure 8.7, a large range of values are not allocated to a crisp class at all.

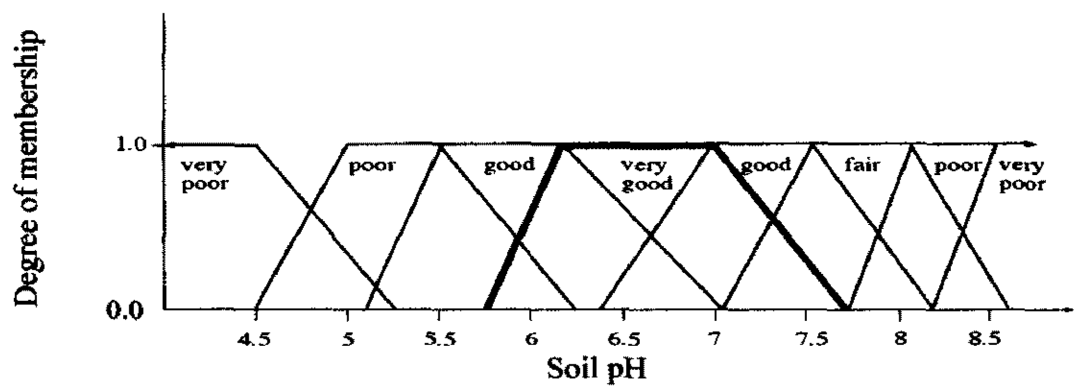


Figure 8.4
Description of soil pH class by fuzzy membership function.

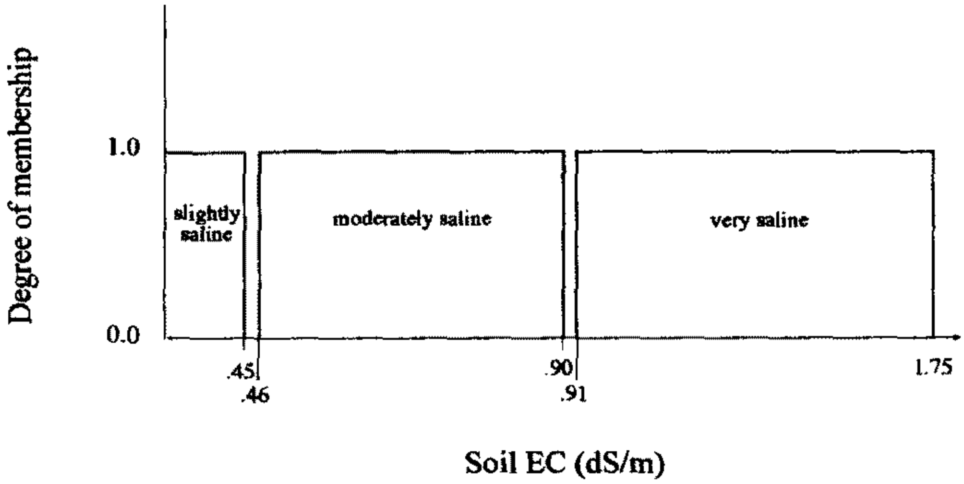


Figure 8.5
Classification of soil EC for a sandy clay loam using crisp membership classes for the recommendations of Walker and Reuter (1996).

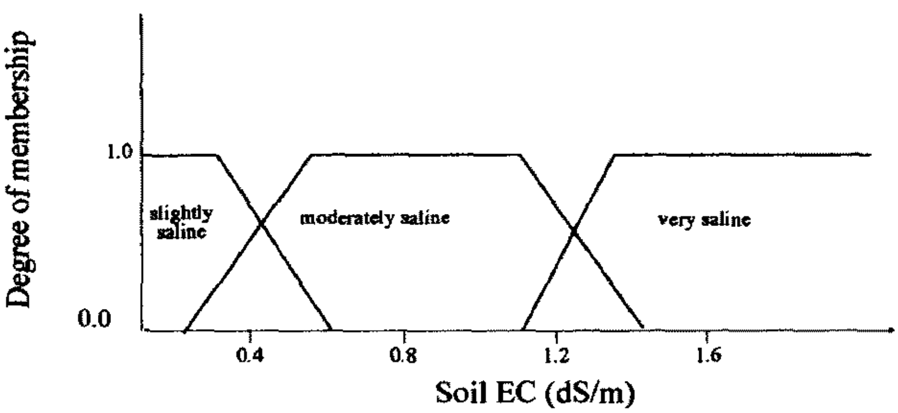


Figure 8.6
The same data as Figure 8.5 using fuzzy set membership.

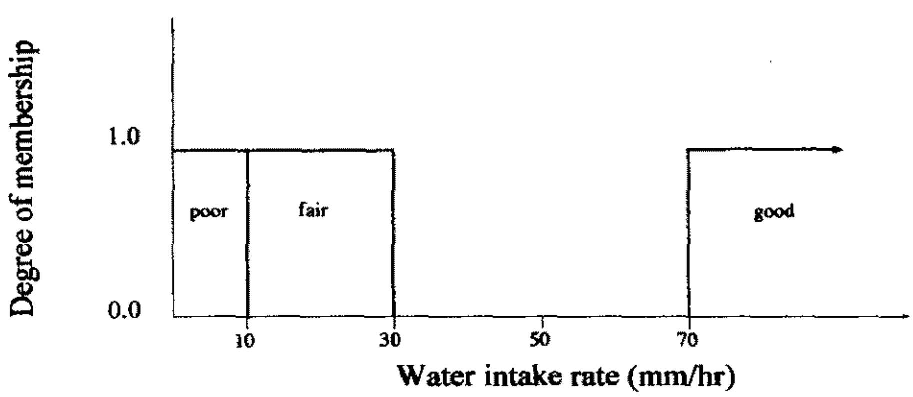


Figure 8.7
Soil surface infiltration rate using crisp membership functions for the recommendations of Walker and Reuter (1996).

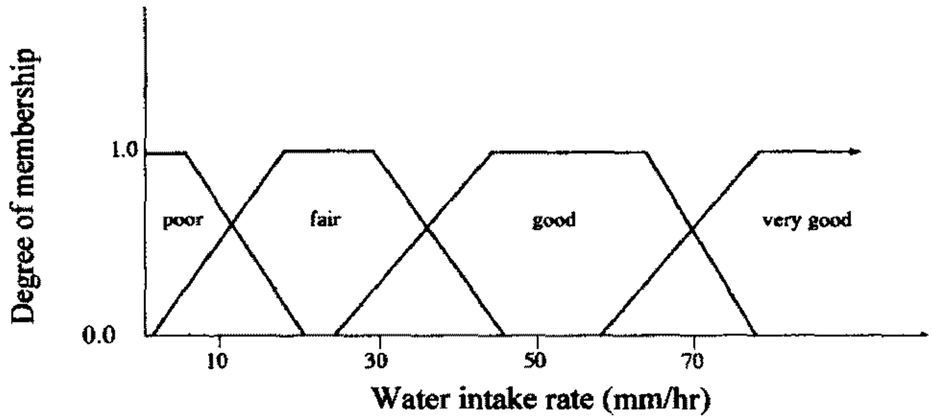


Figure 8.8
The same data as Figure 8.7 using trapezoidal fuzzy set membership.

8.3 Fuzzy Logic operations and rules

The fuzzy membership functions define the degree of membership of a particular input value (e.g. soil pH of 4.95) to a fuzzy set. Inference rules resolve multiple inputs into multiple fuzzy linguistic sets. When multiple inputs provide the antecedent to a particular rule, logical operators are applied to determine the single value that represents the result of the antecedent for that rule. Examples of some operators, those used in the remainder of this discussion, are summarized in Figures 8.9 to 8.11.

Consider two fuzzy sets, A and B (Figure 8.9). The union of these sets (Figure 8.10) is the maximum of A or B created by application of the operator 'or', and the intersection of these sets (Figure 8.11) is the minimum of A and B created by application of the operator 'and'. Such operations enable output information to be constructed following the fuzzification of crisp data for linguistic interpretation.

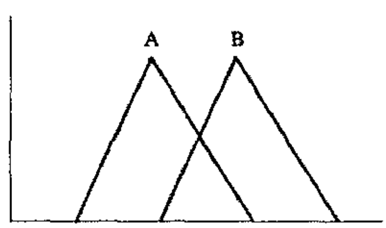


Figure 8.9
Possible fuzzy set membership functions of two hypothetical soil properties.

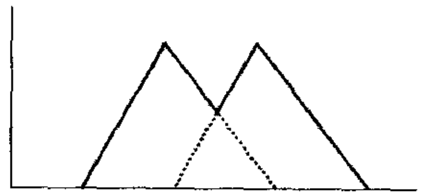
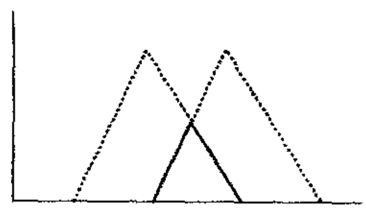


Figure 8.10
Union of sets; operator "or"; maximum of A or B.



Figures 8.11
Intersection of sets; operator "and"; minimum of A and B.

'If – then' inference rules form the conditional statements that comprise fuzzy logic; for example, the first rule of those described in Figure 8.12: 'If soil pH is poor and EC is very saline and hydraulic conductivity is poor then soil quality is low'. Each rule generates a fuzzy output set for the output variable, with multiple sets converted by commutative aggregation into a single set. This enables the output set to be defuzzified to generate a single value for the output variable.

The application of inference rules and the resultant output set manipulation is better illustrated by the simplified example in Figure 8.12, where the purpose is to generate a soil quality evaluation or score with a value between 0 and 10. This example uses the fuzzy set membership functions of the three soil properties used earlier – soil pH from Figure 8.4, soil EC from Figure 8.6 and water intake rate from Figure 8.8 – assuming that these soil properties determine soil quality for this purpose. Expert opinion determines the classification of these soil properties as 'good' or 'bad' to define the shape of the membership functions of the three input soil properties.

It is also necessary to define the membership function of the output variable, in this case, the soil quality score. For simplification in this example, the range of scores is arbitrarily set at 0 – 10, and divisions within this range (low, medium and high) set as trapezoidal functions. The true range and shape of membership function will be determined by known or deduced limits to soil quality, plant response to key indicators of soil quality, environmental constraints such as runoff and erosion threshold, or other factors.

The inference rules applied in this example are:

- If soil pH is poor and EC is very saline and intake rate is poor then soil quality is low. (By way of explanation, the use of the "and" operator implies that if any one of these soil properties is limiting, then the soil quality score can only be low).
- If soil pH is fair and EC is moderate and intake rate is fair then soil quality is medium.
- If soil pH is good and EC is slightly saline and intake rate is good then soil quality is high (i.e. for the soil quality score to be high, then all three soil properties must be good).

In Figure 8.12, the first nine elements represent the input data matrix, with the three columns representing the three soil properties of interest, and the three rows the three inference rules. The fourth column represents the output data. The first three elements of the output data are the contributions to the output variable, in this case the soil quality score, from the three inference rules respectively. Each inference rule is applied sequentially to determine the contribution of that rule to the output variable. For example, a vertical dashed line is shown on the pH scale for the pH value of 5.7. The intersection of this line with each membership function in the first column determines the weight each gives to the output variable. Where this line intersects each membership function, a horizontal line through to the output variable membership function quantifies that weighting. This is repeated for the next two soil properties, with the intersection of the vertical dashed lines corresponding to a soil EC of 0.3 and a water intake rate of 42 leading to the weight of those soil properties to the output variable. The minimum of those contributions is selected, with each contribution to the soil quality membership function shaded.

The final element of the output column (in the bottom right corner of Figure 8.12) is the aggregation of the separate contributions from each rule, which represents a fuzzy measure of the soil quality index that emerges. One way to numerically defuzzify the aggregated output fuzzy set is to determine the numerical centroid of this space; in this example, a soil quality index of 7.1 is derived from these soil properties. A further refinement is that each rule can be weighted according to its relative importance to the output variable, although this refinement is not applied in Figure 8.12.

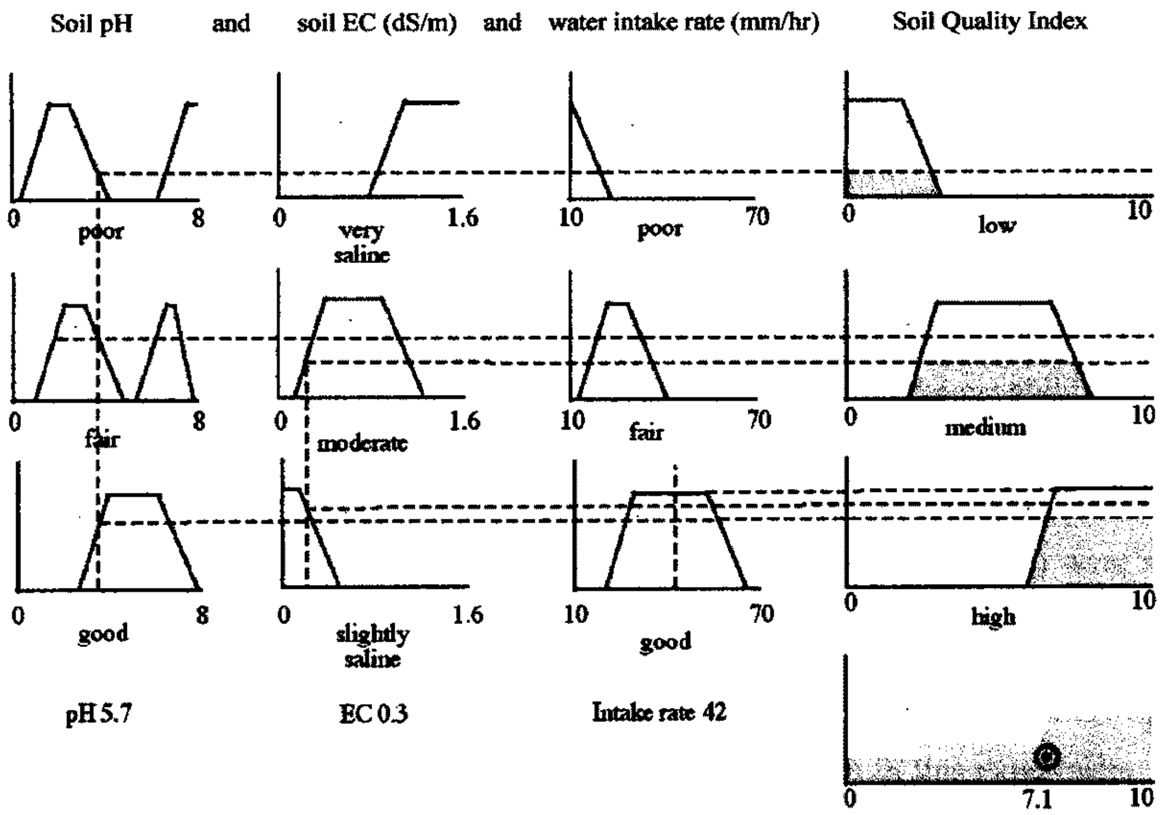


Figure 8.12

A conceptual example of fuzzy analysis of a simplified soil quality score. Membership functions are shown schematically for illustration. A soil quality score of 7.1 occurs with soil pH of 5.7, soil EC of 0.3 and water intake rate of 42.

A large number of input variables derived from diverse measurement sets can be applied to derivation of a single output variable, such as soil quality, in a simple graphical interpretation of fuzzy sets. The potential diversity of data sets is a key feature of this approach – scientific data can be included with social and economic data when linguistic value classification and scales are employed. This approach is extended in the following section to capture the results of this experiment and to compare the results of specific treatments.

8.4 A Fuzzy Logic model of soil physical quality under grazing

8.4.1 Selection of key indicators and membership functions

The selection of indicators was informed by the results of the experiment and principal component analysis of correlation matrices. Variables were selected if analysis showed them to be sensitive to grazing management practices or showed strong alignment with principal components.

A large number of additional indicators are possible, such as those summarized in Table 2.1, but the selection for this model did not include variables known to be stable for this experiment. The purpose of this investigation was to test the capabilities of a fuzzy model to detect the impact of grazing tactics on soil properties at a particular site, so variables assumed to be common to the treatments (climate, soil type, etc.) and variables not likely to change in the life of this experiment (pH, EC, etc.) were not included here.

Twelve indicators of soil quality were selected from those measured in this experiment, listed in Table 8.1. The range of values measured in this experiment for each selected variable was used to determine the range of values for membership functions, also listed in Table 8.1. The mean values of the selected indicators for each treatment in Years 2 and 3 are listed in Table 8.2.

Table 8.1
Key indicators selected for fuzzy modeling, and the range of values applied to membership functions.

Variable	Abbreviation	Unit of measurement	Model range	
			Minimum value	Maximum value
macropore shape factor	shape		0.0	1.0
macropore sieve	sieve	mm	0.0	2.0
average macroporosity	macro	mm^3/mm^3	0.0	0.2
macroporosity at 100 mm depth	macro100	mm^3/mm^3	0.0	0.15
unsaturated hydraulic conductivity	K_{10}	mm/hr	0	100
cotton strip assay	CSA	kg	10	70
macropore surface area	surface	mm^2/mm^3	0.2	1.0
macropore count	count	no./mm^2	0.0	0.25
perennial grass content	PG	%	30	100
organic carbon content	OC	%	0.0	4.0
bulk density	BD	g/cm^3	0.5	1.5
penetration resistance 0-105 mm	penresist	kPa	0	5000

Within each range, allocation to membership classes was based on known experimental evidence and expert advice obtained from the literature, including that contained in Table 2.3. The shape of the input variable membership functions was determined somewhat arbitrarily, using linear, trapezoidal or bell shaped functions in the absence of more specific advice. By way of example, the membership function for macropore shape is shown in Figure 8.13, with the linear shape of the membership function selected in the absence of more specific information. The range of values of macropore shape can only be from 0 to 1, and given the narrow range of measured values in this experiment for the soil type in question, the interpretation of 'flattened' and 'rounded' as linguistic descriptions of macropore shape have been defined as less than 0.3 and more than 0.7 respectively.

Table 8.2
Mean values for selected soil quality indicators for each treatment.

Variable	Abbreviation	Unit of measurement	Year 2 treatment mean				Year 3 treatment mean			
			SS	HI-SD	C	CA	SS	HI-SD	C	CA
macropore shape factor	shape		0.554	0.561	0.562	0.526	0.584	0.563	0.562	0.556
macropore sieve	sieve	mm	0.52	0.64	0.85	0.76	0.92	0.89	0.80	1.02
average macroporosity	macro	mm^3/mm^3	0.073	0.098	0.087	0.120	0.056	0.124	0.094	0.127
macroporosity at 100 mm depth	macro100	mm^3/mm^3	0.029	0.042	0.047	0.070	0.030	0.08	0.047	0.075
unsaturated hydraulic conductivity	K_{10}	mm/hr	14.8	21.9	19.0	25.9	27.9	26.5	26.9	28.9
cotton strip assay	CSA	kg	42.1	42.3	43.9	43.9	42.0	25.4	39.2	39.2
macropore surface area	surface	mm^2/mm^3	0.40	0.48	0.33	0.52	0.23	0.45	0.35	0.42
macropore count	count	no./mm^2	0.106	0.110	0.070	0.110	0.05	0.083	0.066	0.072
perennial grass content	PG	%	85.3	71.9	70.0	71.9	53.7	63.6	70.0	63.6
organic carbon content	OC	%	2.6	2.8	2.3	2.8	2.8	2.9	2.5	2.9
bulk density	BD	g/cm^3	1.14	1.16	1.20	1.15	1.05	0.99	1.11	1.05
penetration resistance 0-105 mm	penresist	kPa	2091	2365	1783	1783	982	917	584	584

The membership function of the output variable, 'Soil Quality' (abbreviated to SQ), is shown in Figure 8.14 (as exported from MatLab Fuzzy Logic Toolbox). The output variable is a constructed index of soil quality with a dimensionless score between 0 and 10. Again, the shape of the membership functions was selected somewhat arbitrarily, but sufficient to test the model capability. A soil quality score approaching the value 10 is described as 'best' soil quality, a score greater than about 7.5 good, and so on, with fuzzy boundaries between these classifications.

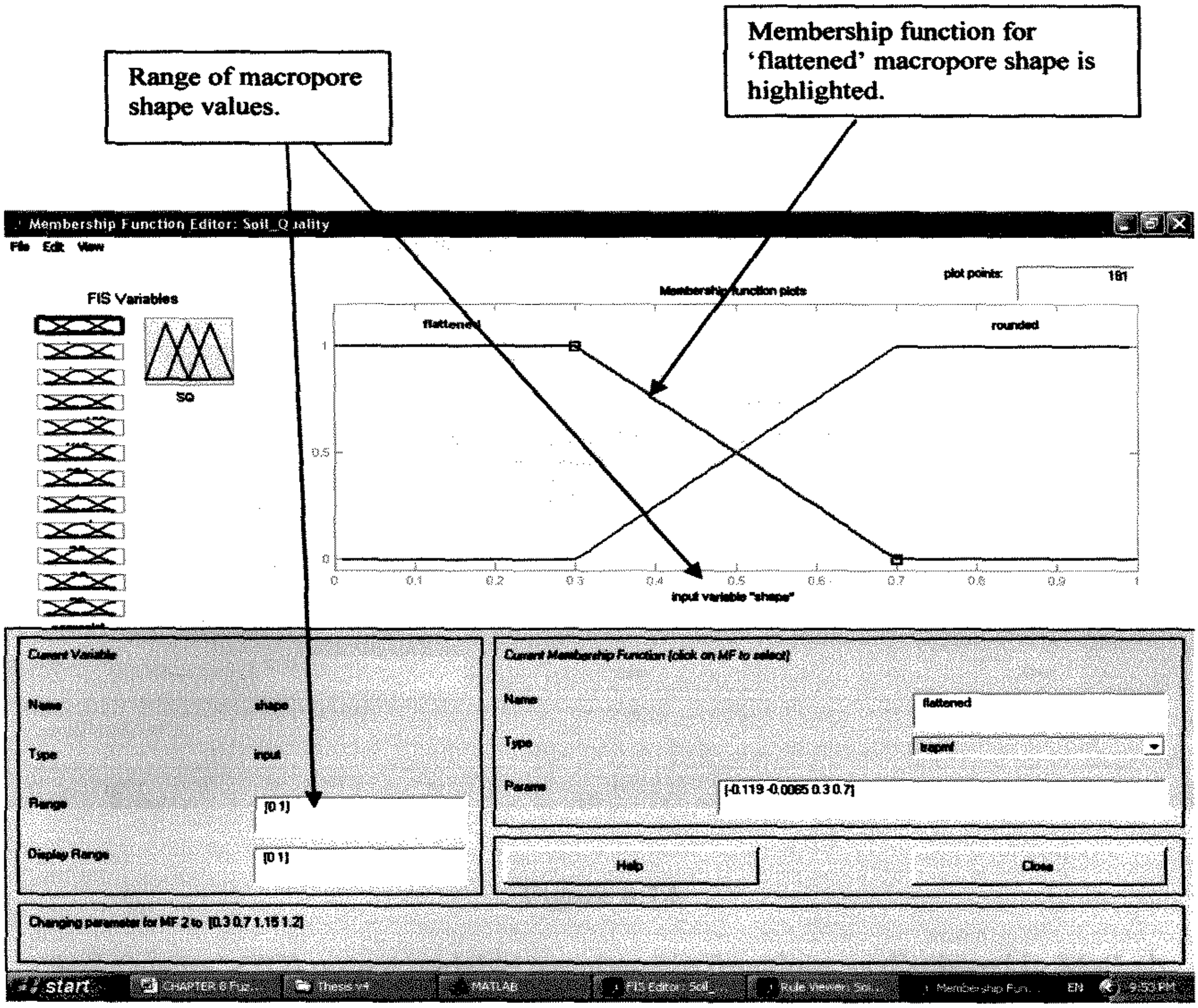


Figure 8.13
Membership function for the input variable 'Macropore Shape' (abbreviated to 'shape'), as displayed in MatLab Fuzzy Logic Toolbox computational package with notation. The function for 'flattened' macropore shape is highlighted.

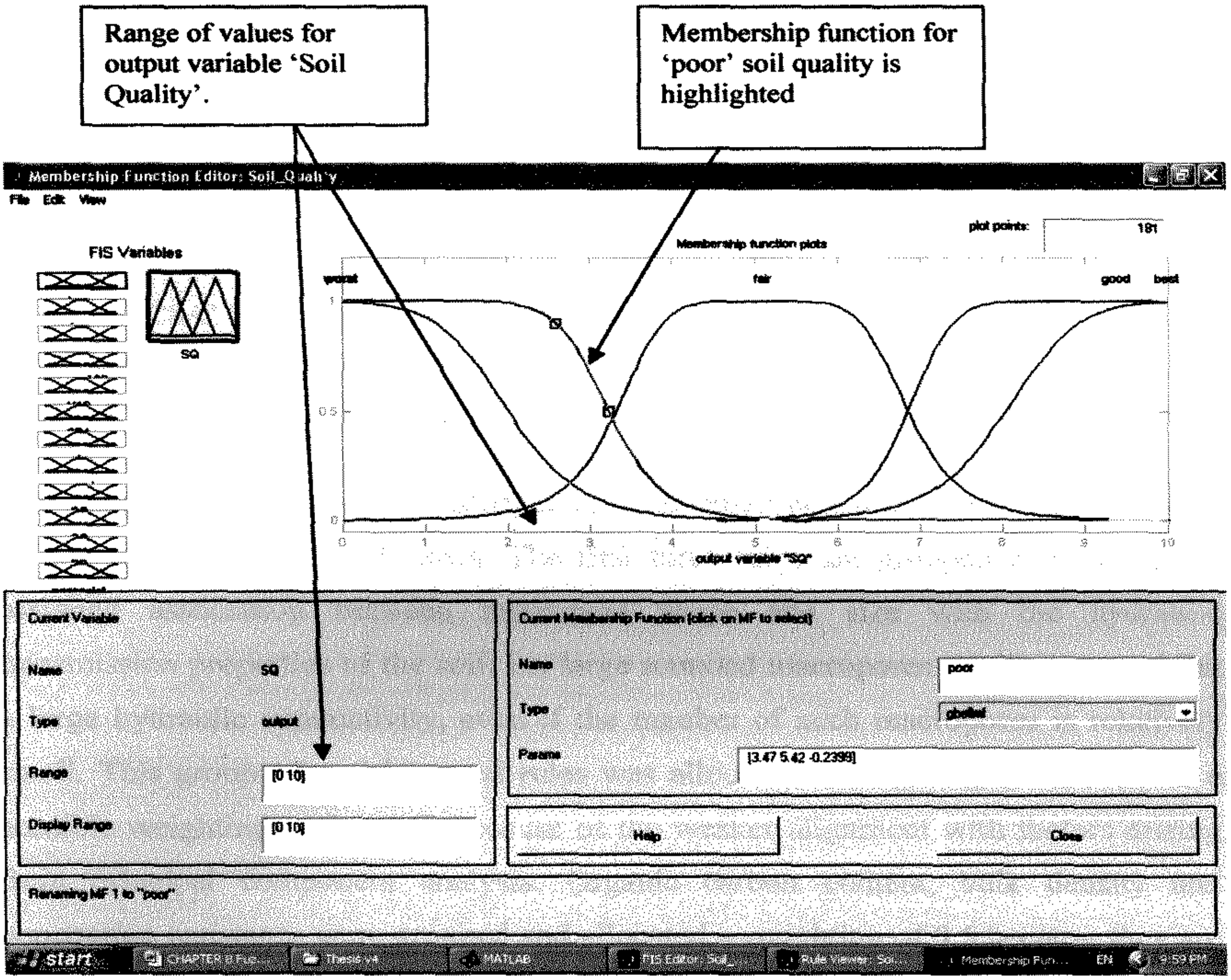


Figure 8.14

'Soil Quality' (SQ) output variable membership functions as displayed in MatLab Fuzzy Logic Toolbox with notation. The membership function for 'poor' soil quality, as measured on a scale of 0 – 10, is highlighted.

8.4.2 Inference rules

The following inference rules were adopted in this model, with the weighting factor in brackets (refer to Table 8.1 for the full terminology for each variable):

1. If 'shape' is rounded and 'sieve' is large and 'K10' is good then 'SQ' is best (0.75).
2. If 'shape' is flattened and 'sieve' is small and 'K10' is poor then 'SQ' is worst (0.75).
3. If 'shape' is rounded and 'sieve' is medium and 'K10' is fair then 'SQ' is fair (0.75).
4. If 'OC' is good and 'BD' is good and 'penresist' is good then 'SQ' is good (0.5).
5. If 'OC' is fair and 'BD' is fair and 'penresist' is fair then 'SQ' is fair (0.5).
6. If 'OC' is poor and 'BD' is poor and 'penresist' is poor then 'SQ' is poor (0.5).

7. If 'macro' is small and 'macro100' is small and 'surface' is poor and 'count' is poor then 'SQ' is poor (1.0).
8. If 'macro' is medium and 'macro100' is medium and 'surface' is fair and 'count' is fair then 'SQ' is fair (1.0).
9. If 'macro' is large and 'macro100' is large and 'surface' is good and 'count' is good then 'SQ' is good (1.0).
10. If 'CSA' is poor and 'pasture' is poor then 'SQ' is worst (1.0)
11. If 'CSA' is fair and 'pasture' is fair then 'SQ' is fair (1.0)
12. If 'CSA' is good and 'pasture' is good then 'SQ' is best (1.0)

The design of these rules was influenced by the experimental evidence and the results of statistical analysis, as follows. The first three rules are grouped because of the assumed association between macropore shape and size with the hydraulic transmission properties of the soil; i.e. large rounded macropores are likely to exhibit a large hydraulic conductivity, even if the number of such macropores is relatively small. This group of variables and rules was allocated a weighting of 0.75 (from a possible weighting of 0 – 1.0) because of the weaker alignment with factors arising from principal component analysis. Organic carbon content, bulk density and penetration resistance were grouped and allocated a weighting of 0.5 because they are all relatively weak indicators of differences in soil quality under the conditions of the experiment reported here. Average macroporosity, macroporosity at 100 mm depth, macropore surface area and macropore count were grouped as being derived from image analysis and weighted at 1.0 because of their strong association with factors arising from principal component analysis. Cotton strip assay and pasture botanical composition were grouped because they are both associated with biological activity, with the inference rule given a weighting of 1.0 also because of their strong association with factors arising from principal component analysis.

8.4.3 Model results

The model was tested by computing the Soil Quality score derived from Year 2 and Year 3 treatment means for each variable. These values are listed in Table 8.1, although a number of adjustments to certain values are to be noted. For treatments C and CA, it was not possible to use BOTANAL assessment of pasture botanical composition, so treatment C was given 70% perennial grass content and treatment CA was given the same perennial grass content as treatment HI-SD, with these adjustments supported by anecdotal observation. For treatment CA, where sampling for cotton strip assay, organic carbon content and penetration resistance was not possible, the mean values for cotton strip assay and penetration resistance were taken to be the same as treatment C, and organic carbon content was taken to be the same as treatment HI-SD.

A sample of the output membership function for treatment C is shown in Figure 8.15 (again, as exported from MatLab Fuzzy Logic Toolbox).

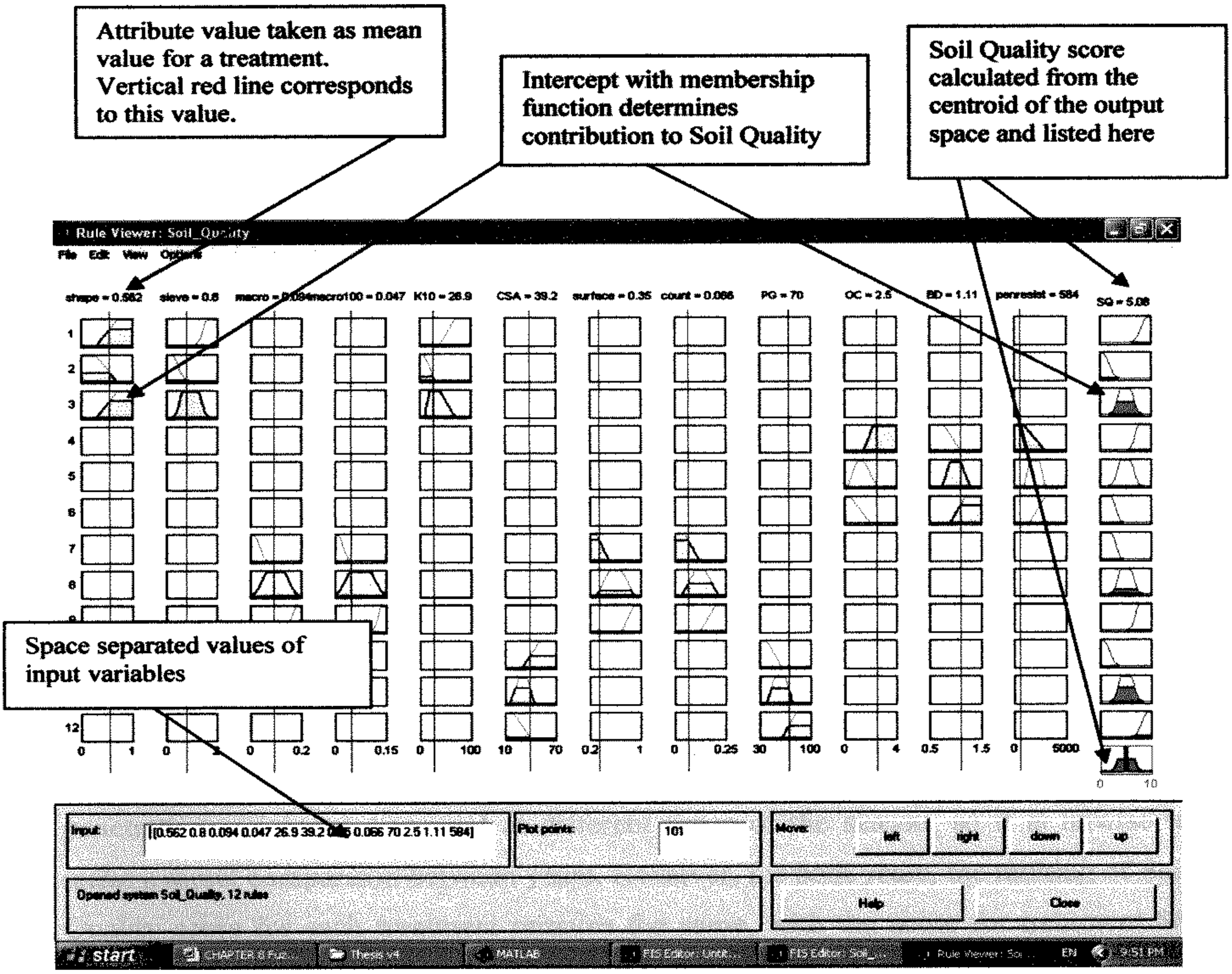


Figure 8.15
Sample fuzzy model output for treatment C, Year 3, as exported from MatLab Fuzzy Logic Toolbox with notation.

Figure 8.15 has been created using the same process as described for the simplified example in Figure 8.12. The input variable data consists of 144 elements in a 12 × 12 matrix, with the rows representing the inference rules and the columns representing the input variables. The range of values is shown on the X-axis for the column, and is that listed in Table 8.1. The vertical red lines in the columns are located by the value selected for each input variable, as listed at the top of each column; 0.562 for macropore shape factor, and so on. In Figure 8.15, the values selected are the mean values for Treatment C for Year 3, as shown in Table 8.2, adjusted for missing values. The membership functions of the relevant input variables are shown in each element of the matrix. The highlighted part of each membership function is that part associated with the selected input variable value. In any row, the minimum of these is used to determine its contribution to the output variable, the Soil Quality (SQ) function in the right hand column. The element at the bottom right corner of Figure 8.15 is the aggregation of each contribution, with the centroid value listed as SQ = 5.08. The results from the model for all treatments are summarized in Table 8.3.

Table 8.3
Soil Quality scores for each treatment, for Year 2 and Year 3, derived from the Fuzzy Logic model.

Treatment	Year 2 Soil Quality score	Year 3 Soil Quality score
SS	4.19	4.37
HI-SD	4.74	5.45
C	5.05	5.08
CA	5.00	5.28

For each treatment, the Soil Quality score increases (i.e. better soil quality) from Year 2 to Year 3. The model includes data from a wide range of indicators, including penetration resistance measured at different times of the season, so it is unwise to make a direct comparison between years. However, for both years, the ranking of Soil Quality score for each treatment remains the same, with the score for treatment SS lowest (worst soil quality) treatment CA largest (best), and treatment HI-SD approaching that of treatment CA. This is consistent with expectations.

8.5 Discussion

The model demonstrates sensitivity in discriminating between grazing treatments, consistent with the expected scores for these treatments, with HI-SD grazing expected to result in better soil quality than SS grazing. With appropriate rest periods, treatment HI-SD soil quality might be expected to approach that where livestock hoof pressure is removed. These results add evidence to the conclusion that rotational grazing strategies should be considered as one aspect of soil quality protection.

However, by including the data from image analysis, the model includes soil structure attributes which are difficult and time-consuming to measure in the field compared to traditional bulk density measurements; as a consequence, these structural attributes are not often reported. Table 8.4 lists results from the model with image analysis data excluded, to simulate a field experiment where image analysis data is not conducted.

Table 8.4
Soil Quality scores for treatments, Year 2 and Year 3, derived from the Fuzzy Logic model without image analysis data.

Treatment	Year 2 Soil Quality score	Year 3 Soil Quality score
SS	3.43	4.49
HI-SD	4.35	5.16
C	4.10	4.85
CA	4.70	5.17

With image analysis data excluded, the Soil Quality scores predicted by the model are all smaller (i.e. implying that soil quality is not as good) than those scores predicted where image analysis data are included. However, the ranking of treatments remains the same for each year of measurement, the Soil Quality score for treatment SS remains smallest for both years, with treatment SS having the smallest (i.e. worst) Soil Quality score in both years, and treatment CA the largest (i.e. best soil quality). With image analysis excluded, the ranking of treatment SS remains lowest (worst), but the rankings of the other treatments are different when the image analysis data is included. The explanation for theses differences requires further investigation, but may be related to the high weighting given to image analysis data in the model compared to surrogate measures of soil quality. It is suggested that because the inclusion of image analysis data improves the quantity of data available, that a superior assessment of soil quality is provided.

Although the generation of a score or index from the output space may be a convenient measure of Soil Quality for comparative purposes, as with any index it may also limit the usefulness of the output information. The shape of the output membership function may be of greater interest because it indicates the relative influence of each rule; different output membership functions can have the same centroid, so the causes of 'good' or 'bad' soil quality will be different. Further, the use of the centroid to measure the shape of the output function reduces the influence of extreme scores; the maximum and minimum Soil Quality scores that are possible in the model used in this example are the centroids of the 'best' and 'worst' Soil Quality membership functions, not their extreme values. It is not possible to have a Soil Quality score of 0 or 1 when using the centroid of the output variable membership functions to summarise the value of the output variable.

An extension of the approach used here would allow for a hierarchy of rules to be developed. Sets of rules could be established that capture soil physical quality, soil chemical quality and soil biological quality to provide a greater level of sophistication. These could be used as input to a higher order estimation of Soil Quality. If this is combined with indicators of enterprise performance (for example, the measures of enterprise productivity and profitability proposed by Walker and Reuter (1996)), an index of farm sustainability could emerge. At a higher level still, an index of farm sustainability can be integrated with measures of landscape health and integrity to inform decisions associated with catchment management. Fuzzy models allow the nesting of highly diverse elements of complex systems. For example, Michalk et al. (2003) have recently determined the relative profit as well as measurements of water balance, nutrient balance, soil quality, pasture productivity and pasture species diversity from a sheep grazing enterprise at Carcoar, NSW. Simple fuzzy models have the potential to incorporate socio-economic variables such as 'profit' in a multi-disciplinary approach to sustainable farming systems.

8.6 Conclusion

In capturing the results from this experiment, the fuzzy logic model provided additional evidence that set stocked grazing is likely to have a greater negative impact on soil quality than rotational grazing, that rotational grazing may have no greater impact than total grazing rest and the benefits of grazing rest are associated with maintenance of active pasture growth. If further work can demonstrate a critical threshold score for soil quality in particular grazing systems, then the model can determine the relative impacts of alternative grazing strategies. By measuring the closeness of the 'Soil Quality' score to the threshold value, and its rate of change, the model can predict if corrective management action is required. The model runs successfully with limited data; for example, if only traditional or surrogate measures of soil structure are available, the model is adaptable to use by farmers and advisors as well as scientific reseachers.

Fuzzy logic provides a convenient modeling approach that combines expert opinion with sometimes limited objective data on soil properties to deliver quantitative assessment of alternative land management practices. It is emphasised that for fuzzy logic applications, the inclusion of additional variables is a simple procedure, enabling customization of models to suit the purpose of the investigation, including comparison of different soil types and climate regimes across many sites. Fuzzy modeling is well suited to such applications. Fuzzy models are flexible, able to accommodate many variables and rules with a fast run time and interactive 'what if?' capability. They provide a convenient methodology to combine common sense and professional judgement with scientific and socio-economic data.

However, this usefulness must be balanced by the sensitivity of the process to the opinion and bias of the model designer, the expertness of the opinion that might be included and the interpretation of the resultant output. In the approach described here, output scores are not accompanied by a statement of confidence, and this alone requires a different attitude to the modeling process; that the purpose of this approach is to inform a decision-making process, explore relationships between variables and test scenarios, and is not an end in itself.

CHAPTER 9

GENERAL DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER INVESTIGATION

9.1 General discussion

This investigation has confirmed that the continuous presence of livestock, under set stocked management, is likely to alter topsoil structural attributes not only at the soil surface, but to at least 100 mm below the soil surface. HI-SD rotational grazing management maintained soil structure over the duration of this experiment. The duration of this experiment is considered important. Many studies of soil – pasture – livestock interactions are of short duration, often only a single grazing event, and do not capture the potential benefits of grazing rest from rotational grazing management. Although Greenwood et al. (1997) argue that the impacts of grazing are cumulative, and all grazing tactics are likely to have similar impacts in the long term, their study focused on differences in stocking rate where all treatments were set stocked, and did not consider rotational grazing management strategies.

The use of image analysis in this experiment showed that the soil under treatment HI-SD maintained superior structure throughout the duration of this experiment, compared to the soil under treatment SS, according to the following summary:

- greater macroporosity at all depths of measurement, and greater average mesoporosity, at all three sampling dates over three years, with the differences being significant in Year 3;
- greater average and topsoil porosity from large (> 1.5 mm diameter) pores at all three sampling dates;
- larger macropore surface area at all depths of measurement at all three sampling dates over three years, with the differences being significant in Year 3;
- larger macropore count at all depths of measurement at all three sampling dates over three years (except at 10 mm depth in Year 2);
- larger macropore sieve in Years 1 and 2 (but not Year 3).

During the three years of the experiment, soil structure under treatment HI-SD maintained a stable trend, whereas soil structure under treatment SS showed a decline in soil structural quality to create the differences listed above. It was not determined if these changes in soil structure were sufficient to alter soil function (e.g. soil water retention, pasture root depth or density) and therefore pasture productivity (an economic measure of soil function for a grazing system). However, the declining trend in macroporosity under set stocked grazing management indicates that soil function was likely to be compromised at some point in time.

These results not only confirm the relative impacts of the two grazing treatments, but also demonstrate the sensitivity of image analysis to detect subtle differences in porosity. However, the method does have limitations in measuring microporosity. In this study, the smallest pore size that could be measured was 0.065 mm, and this depended on exposure to the resin flow path. It is possible that whilst treatment SS demonstrated a reduction in macroporosity, an undetected increase in microporosity may have occurred. The microporosity of soil is likely to contribute significantly to many soil functions such as water, nutrient and gas retention and exchange, and to the physical arrangement of microbial habitat (Shestak and Busse 2005).

The selection of key indicators in this experiment was biased toward soil structural condition, and to a lesser extent, soil biological factors. It was assumed that these soil attributes would respond more sensitively to grazing management practices than chemical indicators. The latter were benchmarked at the commencement of the experiment but not retested. Consequently, this assumption cannot be investigated further. The application of the cotton strip assay proved more useful when considered as part of a multivariate analysis than when considered in isolation, providing evidence that indicators of soil biological activity should be considered in investigations of this type.

Measurements of bulk density and organic carbon content, whilst suitable for benchmarking against other sites, did not prove as useful for discriminating grazing treatments in this study as direct measurement of macropore characteristics, although it is acknowledged that the inclusion of additional key soil properties such as soil shrinkage or resistance to compaction may have improved interpretation. In the case of bulk density measurements, all that can be ascertained of the structural condition is the total porosity; bulk density alone indicates nothing of the size, shape and continuity of pores. Although Proffitt et al. (1993) claim that differences in bulk density may have been masked by a relatively shallow zone of compaction compared to the depth of sampling for bulk density, differences should still be measured if enough samples are taken. It is more likely that sampling for bulk density is itself a source of error, insensitive to the small differences that may be present in the amount and nature of the macroporosity. The method used here to measure organic carbon content is not suitable for the assessment of the labile component of organic matter, more likely to be affected by grazing tactics than the total carbon content, and the soil-pasture systems under investigation here also have a relatively robust organic matter cycle. No significant differences in unsaturated hydraulic conductivity were observed between treatments, but a greater density of sampling may have been more revealing.

Overgrazing and the presence of livestock during periods when soil water contents exceed the plastic limit have already been shown to be detrimental (Proffitt et al. 1995b). However, other factors are also likely to be important, such as plant root and soil biota dynamics. Secondary effects associated with grazing management tactics are also likely; for example, because HI-SD grazing has potential to maintain soil structure compared to the detrimental effects of set stocked grazing, it is likely to make more soil water available to plants, thereby contributing to increasing plant productivity. Because HI-SD grazing management can preserve a greater quantity of surface litter, the impacts of treading events are unlikely to be as severe.

Field observations during sampling, although not quantified in this study, indicated many more earthworms under both the control plots and caged areas compared to the grazed treatments. This was probably due to a combination of factors, including a better substrate supply from organic matter, increased soil water availability, reduced summer soil temperatures, reduced hoof pressure and more hospitable physical habitat. HI-SD grazing management will advantage these factors compared to set stocked grazing management. It was hoped this would be confirmed by measurement of macropore shape and size characteristics, but although some differences were measured, no trend could be determined. The potential usefulness of the Cotton Strip

Assay as an indicator of soil biological activity, foreshadowed by King (1996), was confirmed.

The differences in soil structural properties between treatment SS and treatment HI-SD were not always significant. This may be attributed to a number of factors. In particular, even set stocked grazing can have a relatively benign impact on soil structural condition (compared to cultivation for cropping), given that the soil under set stocked grazing still has perennial pasture species present. During this experiment, the pastures were not subject to severe grazing, nor subject to extended dry or wet conditions. Passioura (1991) explains that hard soil can still contain biopores which are readily occupied by plant roots. Because these spaces are often widely separated, and with a relatively slow rate of migration through the intermediate space, crop roots cannot as easily access the water and nutrients that might be contained there and suffer greater stress than perennial plants when soil water conditions become marginal. However, little effect may be observed when soil water content is non-limiting.

Passioura (1991) also explains that the non-homogenous distribution of root material and associated organic compounds can distort measurements of soil organic and microbial properties, possibly at play in this experiment. The term 'natural densification' has been applied to the observation of increased soil bulk density under pastures, even with conservative (or no) grazing (Tollner et al. 1990). This can be partly explained by the action of plant roots, in two ways: the occupation of existing macropores by active plant roots, and the radial (but not longitudinal) compressive pressure created by active root growth (Kay 1990). In an early analysis of this issue, Barley et al. (1965) commented that soil strength has a general influence on root elongation, rather than as a limiting condition encountered in unusual soils.

The greater macroporosity observed with HI-SD grazing management may not mean that soil water drainage is greater compared to SS grazing management. The greater macroporosity is most evident at the soil surface, which may increase water intake rates and soil water storage, but not necessarily increasing deep percolation. The presence of a larger proportion of perennial grasses is a contributing factor. Perennial species have a longer growing season than annual species, and are likely to have a deeper denser root system, increasing water use under these species. At the root-soil interface, however, other mechanisms are at play. Kay (1990) describes a loss of porosity but increase of soil strength and stability in soil adjacent to roots as a result of root growth. If this soil remains undisturbed, the macropore that is created when the root decays remains an important site for water movement as well as a site for future root growth. In addition to these mechanical effects, the dehydration resulting from plant transpiration will also influence soil structural properties. Root exudates are also likely to be a factor. Quick and Murray (1991) suggest that the contribution of these mechanisms to soil stability is in strengthening the walls of coarse pores, those most responsible for the transmission properties of soil. Disruption of these macropores by the actions of livestock will interfere with water transmission.

Warren et al. (1986) claim rotational grazing with increased stocking rates also decreases infiltration rates significantly and increases sediment production. These results are not consistent with measurements of macroporosity under HI-SD grazing management. Note that the experimental design of Warren et al. (1986) was flawed, conducted on bare soil only and with a rest period of only 30 days. In contrast, Michalk et al. (2003) show that deferral of grazing at certain growth stages, combined with strategic grazing tactics, will increase perenniality and reduce annual weed populations in pastures of the Central Tablelands, and that the abundance of perennial grass is related to increased water use, increased pasture productivity and improved environmental outcomes. Although Dowling et al. (2005) found no apparent medium-term benefit of the specific grazing strategy of 'time-controlled' rotational grazing on pasture botanical composition, based on experimental data from a number of sites in southern Australia, they had little doubt that strategic grazing rest of pastures does have the potential to improve grassland composition and that other benefits may accrue, including potential benefits to soil condition.

It may be interesting to note that the 'best' soil structure, as indicated by the largest values of unsaturated hydraulic conductivity and macroporosity throughout the full depth of measurement, was in the caged areas, where pasture cover was removed by grazing but the soil was not subject to the detrimental effects of hoof traffic. In the ungrazed control plots, pasture composition quickly became dominated by mature Phalaris (*Phalaris aquatica*), whereas the caged areas contained continuously active mixed pasture species of around 75 mm in height. It is likely that this resulted in different root system dynamics, and consequently differences in organic matter cycling and soil biota activity. This provides further evidence of the potential of active pasture growth to restore soil structure during rest periods from grazing, consistent with the experimental findings of Tisdall and Oades (1980) on perennial ryegrass grown and clipped in pots to simulate defoliation.

This proposition, at the centre of the debate on rotational grazing and conservation of grasslands, is also supported by other researchers. Ganjegunte et al. (2005) compared different grazing management practices on the amount and nature of soil organic carbon in the 0 – 50 mm topsoil layer of a sandy loam soil. Light grazing created significantly greater soil total organic carbon and nitrogen than heavy grazing or continuous rest, and the lignin content of soil organic carbon was significantly greater for continuous rest than either grazing treatment. They concluded that light grazing is the more sustainable management practice for these grazing systems compared to heavy grazing and non-grazed pastures. Reeder and Schuman (2002) measured significantly greater soil carbon (0 – 300 mm soil depth) in grazed pastures compared to non-grazed exclosures. They found that immobilization of the soil carbon in above-ground litter, and the trend toward annual species with less dense root systems in the pasture botanical composition for the non-grazed pastures, contributed to this difference, and regular defoliation of grazed pastures contributed to a greater shoot turnover and subsequent redistribution of soil carbon. Proffitt et al. (1993) confirm the potential usefulness of pasture grazing to delay senescence (by reducing leaf area and therefore evaporative water losses) and therefore maintain active pasture growth for an extended period of time. The increased perenniality of pastures subject to planned grazing tactics, whether planned rotational grazing or tactical deferral of grazing, provides further evidence (Kemp et al. 2000, Michalk et al. 2003). Mikhailova et al. (2000) also found that botanical species diversity was greatest under a periodically-cut grazed field, compared to regular grazed field and more again than native grassland.

Whilst recovery of soil structure will be assisted by active pasture growth during rest from grazing, the time necessary to restore soil physical conditions has been shown to be highly variable, from some months to many years (Greenwood and McKenzie

2001). For example, Wheeler et al. (2002) found that infiltration and bulk density returned to pre-grazing values after 1 year. This variability is to be expected given the complex factors that determine pasture growth rates, none more variable than rainfall. However, to satisfy the needs of an intensive rotational grazing management system, some certainty is required over soil restoration patterns, preferably on a seasonal or annual basis to match seasonal management decisions on pasture management and livestock husbandry. This certainty is not yet available. However, in this experiment it was shown that macroporosity under HI-SD grazing management was maintained throughout the duration of the experiment, and a continuous decline in macroporosity occurred under set stocked management. This indicates that the rest periods used under this management regime, and under the climatic variation that occurred, may have been appropriate.

It is possible that if a soil is subject to a lesser amount of degradation, perhaps because of the use of rotational grazing management, then the time to recover will also be less. It is also possible that the time to recover from grazing impact may need to be measured from the date of a particular seasonal rainfall trigger rather than the date of cessation of grazing. Further, if a soil is already rated as 'satisfactory' or better in terms of soil quality, then recovery from a single grazing event is likely to be faster compared to a soil which is in a degraded condition at the time of grazing. It is therefore necessary to consider a different set of grazing and pasture management tactics during the transition period from 'degraded' to 'satisfactory' before considering optimisation of grazing management thereafter. The duration of this transition period will also be site-specific and rainfall dependent, although likely to be accelerated if soil conditions are conducive to the activity of soil biota, and intervention with herbicide and fertiliser application is considered.

Questions remain regarding the relative impact of livestock on soil quality properties. These can be partly answered by comparison of treatments C and CA with results from grazing treatments. In this experiment, treatment CA had significantly greater unsaturated hydraulic conductivity at –10 mm tension than both grazing treatments, and significantly greater macroporosity, mesoporosity, pore surface area and pore sieve at all depths of measurement compared to the set stocked grazing treatment. Given that all plots had the same pasture botanical composition, at least insofar as its assessment by the BOTANAL method, it can be concluded that the continuous presence of livestock will damage soil structure, more so than for high intensity – short duration rotational grazing, and that the presence of the compaction effects of livestock hooves is a contributing factor. Further, it can be asserted that the presence of actively growing pasture (as demonstrated by the CA treatment) may impart greater benefits to soil structure than 'locking up' pasture land (as simulated by treatment C), because root matter replenishment and the development of root channels is likely to be beneficial.

Increasing the perenniality of a pasture will increase the duration of green herbage and should increase livestock performance. However, a greater amount and seasonal duration of green herbage appears to be insufficient on its own to make an enterprise more sustainable when economic criteria are included in the assessment of sustainability. For example, Michalk et al. (2003) found that the deferred grazing necessary to improve perenniality of grazed pastures at Carcoar, NSW, resulted in a smaller net cash flow in the medium term compared to continuous grazing, due to the reduced grazing days available. Even so, neither of these grazing systems on pasture was able to finish lambs to market specifications – a supplementary cropping system or other feed supplement was required. Although it was found that continuously grazed pastures had a higher net cash flow than tactically grazed pastures (such as HI-SD type grazing management), continuous grazing had more negative environmental impacts in the long term, associated with reduced water use and subsequent increased risk of acidification and salinisation.

The implications of these results at the catchment scale should be considered. The impacts on catchment hydrology, soil erosion and water quality associated with overgrazing and severe climatic events such as drought and high intensity rainfall are well known. However, it appears likely that minor changes in the way stocking rates and grazing pressure are managed has the potential to bring changes to the water balance of grazing land. Grazing strategies such as HI-SD grazing management may have potential to contribute to improved catchment health as well as enterprise productivity. They will also require a greater level of management expertise and flexibility, and it is necessary to consider how they are integrated with other farm activities, stock health issues and farm layout.

9.2 Conclusions

It is not possible to remove livestock all together from an extensive grazing enterprise, other than from specified areas for rehabilitation or fodder conservation, so it is necessary to manage grazing activity to minimize detrimental impacts. This factor is of critical relevance to Central Tablelands farms, where livestock grazing is the dominant agricultural land use. With this underlying philosophy, the following conclusions can be made from this experiment:

1. **Set stocked grazing management is likely to reduce soil macroporosity, and high intensity – short duration rotational grazing is a strategy that has potential to protect soil macroporosity.**

Recommendations to the contrary arising from certain other work should be discounted on the basis of failure to include rotational grazing in experiments, departure from commercial application, and deliberate antagonism to the claims made in support of 'cell' grazing. The need to analyse the assumptions and experimental practice of other pasture–grazing–soil investigations before adopting their recommendations is apparent.

2. **The nature of soil porosity is important in detecting differences in soil condition between alternative grazing treatments.**

This research has provided more sophisticated detail and greater sensitivity than conclusions based on bulk density measurements, hydraulic characteristics, organic matter investigations and other surrogate measures of soil structure under the conditions of this experiment. An understanding of the nature of macropores has provided greater information about the functionality of soil than surrogate data. In turn, this has enabled recommendations based only on surrogate measures of soil structure to be questioned. In particular, the density of sampling, quality of field practice and discrimination between sampling depths needs further consideration for surrogate methods. The importance of biological indicators of soil condition has been highlighted. Consequently, this work has contributed to an improved capacity to select and correctly apply key indicators of soil quality for Central Tablelands grazing systems.

3. **If rejuvenation of soil structure is an objective in a grazing system with exotic pasture species, then occasional planned grazing at an appropriate time of the season may be more beneficial than continuous rest.**

In addition to the benefits accruing from rotational grazing, this work has demonstrated that best soil structure under the conditions of this experiment was measured where pasture defoliation could occur in the absence of livestock hoof pressure. In this experiment, some evidence of 'natural densification' was observed, with treatment C generally exhibiting less macroporosity, less mesoporosity, smaller pore surface area, smaller pore count and smaller pore sieve compared to treatment CA, although few of these differences were significant.

4. **Management of livestock alone can contribute to both economic and environmental goals of sustainable grazing systems by maintenance of soil structure and pasture perenniality.**

The short term economic performance of grazing enterprises on Central Tablelands farms centres on the quantity and value of livestock products, which is dependent on the amount and quality of pasture produced per dollar of input costs. Balancing these measures of farm productivity with the long term environmental sustainability of a catchment remains a challenge, but maintenance of active perennial pasture growth has been shown to be a critical factor. HI-SD grazing management has been shown to protect soil structure and is likely to advantage the perenniality of pastures in Tablelands grazing systems.

5. **Simple fuzzy modeling techniques can be deployed to explore relationships between the factors at play and test scenarios at multiple scales of investigation.**

This work has enabled the relationships between soil, pasture and grazing management factors, and their influence over the objective measurement of soil quality, to be explored in a way that has not been done before, using fuzzy methods. Furthermore, these methods are relevant at plot, paddock, farm and catchment scales. Fuzzy methods provide for mixed data types, incomplete data and the inclusion of professional opinion, necessary to cope with the complexity of questions pertaining to sustainability. At the same time, fuzzy methods are sufficiently robust to enable otherwise complex or ambiguous concepts to be reliably quantified.

9.3 Recommendations for further investigations

Whilst image analysis has proven to be an essential component in the methods used in this research, a number of operational issues remain. These include the risk of retained soil water, differences in the amount of retained soil water at the time of sampling, sampling intensity (spatial and temporal) and the destructive nature of the sampling. Further work is recommended to continue improvement of image analysis methods.

More importantly, the interpretation of the data that is generated requires further investigation, to link the nature of macropores to soil quality. Even if techniques such as image analysis can quantify the nature of macropores, the precise relationships between macroporosity, soil quality and soil function remain elusive. Relevant soil functions include hydraulic behaviour, healthy plant growth and resistance to compaction and erosion. Further research is required to 'fine tune' quantitative measurements that support the assessment of these functions, because the results reported here do not determine the change in soil structure that is necessary to create a change in soil function. In particular, the use of BOTANAL should be supplemented with additional measures of pasture performance and production. Soil structural assessment based on Solicon® calculations on vertical images of soil should also be included, as a complement to assessment of horizontal images.

Although the research reported here has made some significant findings, developed methods for quantifying certain elements of the soil quality matrix of indicators and tested a new approach to soil quality modelling, it has still not finally answered the larger question – what is a good Soil Quality score for grazing systems? For example, Shustak and Busse (2005) found poor correlation between physical and biological indicators of soil quality, so what is the nature of soil porosity that is conducive to soil biological activity, and are soil water content and temperature more important? What is the nature of soil porosity that determines unacceptable hydraulic characteristics? What is the density of vertically continuous macropores to ensure 'good' water transmission? If a soil property is considered 'good' or 'bad' across a particular threshold value, then what is the sensitivity of measurement that is required? Is the absolute value of the Soil Quality score more important than the change in value over time, or the rate at which the score approaches a critical threshold of sustainability? Can a 'good' Soil Quality score create complacency in the eyes of the farm manger? How sensitive is a Soil Quality score to farm management practices? For example, soil recovery from grazing may take some considerable time, yet soil degradation from grazing may only take a matter of hours if heavy stocking rates are applied to a soil that is conducive to compaction and having excessive soil water content. More research is required to create more sophisticated analyses, to integrate findings on soil quality with the other elements of sustainability, and to determine farm manager attitude to such approaches to management.

Farm manager attitude is likely to be influenced by productivity (and therefore profit) improvements as a result of some change to grazing tactics. The research reported here avoided this issue by applying similar levels of production as measured by grazing days. This does not provide evidence of any potential difference in productivity between grazing treatments, nor whether a change to grazing tactics is more beneficial than, say, application of additional fertiliser. It is recommended that future research include suitable measurements of livestock and pasture productivity to the limit of good pasture management.

Future work should also centre on a higher level of discrimination of measurement, more frequent measurement to assess within-season variability, and testing of optimum rest period for soil structural recovery. Further investigation into the relationship between grazing management and soil fauna in grazed pastures is also recommended.

The initial condition of the soil investigated here has not been compared to other soils to determine a soil quality rating at the commencement of the experiment.

Consequently, further work is required to benchmark 'starting' conditions in a variety of catchment settings, which will assist determine priorities for intervention.

Questions of sustainability, which include issues associated with farm business performance, can be modelled by fuzzy methods. These are suitable when objective data on system performance is incomplete, and where professional and personal judgement is necessary; for example, in setting an acceptable level of profit, where increasing profit may require activity which is detrimental to environmental outcomes in the long term. However, further research is required to develop these models in the continuing quest for a sustainable agriculture.

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APPENDIX

The enclosed compact disc contains the soil core images used for analysis in this research and text output from the SOLICON® analysis.

The images are located in separate folders for each of the 3 years of measurement. Please consider the following file notation.

Each image is labelled according to the plot, replication, nominal sampling depth and year of measurement. Treatment SS is abbreviated to A, with each plot numbered 1 to 4. For each plot, the three sample locations are referred to as 01, 02 and 03. For each sample location, the depth of measurement is noted (10 mm, 50 mm or 100 mm). Each year is denoted by Y1, Y2 or Y3. For example, the image labelled A101@10-Y1 is from treatment SS, plot 1 of 4, replication 1 of 3, 10 mm depth of sampling from Year 1. Treatment HI-SD is abbreviated to B. Treatment C is identified by the two grazed plots it is located between; for example, image A1B2 is from the control plot located between grazing plots A1 and B2. Images labelled CAGE are from sites where pasture cages allowed defoliation by grazing but prevented livestock hoof pressure.