Effects of the Land Disposal of Water Treatment Sludge on Soil Physical Quality

by

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DECLARATION

I hereby certify that this research is the result of my own investigation, except as acknowledged herein, and that it has not been submitted for a higher degree in any other university.

M. Moodley

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Abstract

An essential step in producing "drinking" water is to precipitate the suspended and dissolved colloids through the addition of flocculents such as lime, ferric chloride, aluminium sulphate and/or poly-electrolytes. The by-product of this process is termed water treatment sludge (WTS) and contains mainly silt, clay and some organic matter. Previously this material was disposed of in landfill but more recently, alternative methods for its disposal are being evaluated. A potential disposal option is land treatment. In this system of waste disposal the inherent properties of the soil are used to assimilate the waste. Although the effect of the land disposal of WTS on soil chemical quality is gaining increasing research attention, few studies have investigated the effects on soil physical quality.

This study was originally commissioned by a local water utility to evaluate the effects of the land disposal of sludge produced at their works, on soil quality. At this plant organic polymers are used to both flocculate the material and to thicken the sludge in the water recovery process. Fresh sludge has a consistence approaching that of slurry but dries to angular shaped aggregates of extremely high strength. Nevertheless, sludge aggregates comprise a network of micro-pores and channels and are therefore porous. Because of these properties, the potential use of WTS as a soil conditioner was considered. Since lime, gypsum and polyacrylamide are well-recognised soil conditioners, these were included as reference treatments in the study.

Two field trials (Brookdale and Ukulinga) and laboratory experiments were designed to investigate the influence of WTS on soil in terms of water retention, hydraulic conductivity, evaporation, aeration, aggregation and strength. Seven rates of WTS are represented at the Brookdale trial but research efforts were concentrated on the 0, 80, 320 and 1280 Mg ha^{-1} treatments. WTS was also applied as a mulch (without incorporation into the soil) at the 320, 640 and 1280 Mg ha^{-1} level. Gypsum was applied at rates of 5 and 10 Mg ha^{-1}, lime at 2 and 10 Mg ha^{-1} and anionic polyacrylamide at 15 and 30 kg ha^{-1}. At the Ukulinga trial, WTS was mixed with the upper 0.2 m of the soil at rates of 0, 80, 320 and 1280 Mg ha^{-1}. Only the high rates of gypsum, lime and anionic polyacrylamide being tested at the Brookdale trial are represented at the Ukulinga trial. All treatments in this study were maintained fallow. The

laboratory study features an additional two soils to those from the field experiments, chosen to produce a range in clay contents.

WTS influenced several soil physical properties. Soil bulk density decreased following the addition of sludge to soil. This caused an increase in porosity (particularly macro-porosity) and therefore water retained at saturation, but only of statistical significance at the 1280 Mg ha^{-1} level. Equally an increase in water retention at the wilting point (-1500 kPa matric potential) also occurred, owing to the high microporosity of sludge aggregates. Despite these effects very little change in both the plant available and readily available water content occurred. Neither, gypsum nor lime caused any significant change in water retention. A slight improvement was noted on the polyacrylamide treatment at the Brookdale site but this effect did not persist for very long after the trial was established.

Although *in situ* field measurements were influenced strongly by natural spatial variability, WTS caused a marked increase in the saturated hydraulic conductivity (Ks). The reasons for this relate to the higher porosity and the inherently stable nature of the sludge aggregates, which imparts a more open structure to the soil and reduces the extent of pore blockage. This finding was corroborated in a laboratory study in which strong positive correlations between sludge content and Ks was found. The water retention curve and saturated hydraulic conductivity was used to predict the unsaturated hydraulic conductivity function (K_ψ)using the RETC computer model of van Genuchten *et al.*, 1991. The results showed a decrease in K_ψ on the sludge-amended treatments the extent of which increased with sludge content. This finding was tested in an evaporation study conducted under controlled environmental conditions. More water was conserved on the sludge-amended treatments than the control, because of its lower K_ψ. The application of the sludge as a mulch was more effective in conserving water than incorporating the sludge with soil.

The air-filled porosity at field capacity (-10 kPa matric potential) of the sludge-amended soil remained within a favourable aeration range of 10-15%, which suggests that aeration should not be a limiting factor for plant growth. Air-permeability nevertheless improved substantially.

Attempts at using the size distribution of dry soil aggregates to evaluate the influence of the sludge on aggregation proved unsuccessful. Saturated soil paste extracts for selected soil depths beneath the mulch layers at the Brookdale trial, nevertheless, showed significant increases in Ca^{2+} and Mg^{2+} concentrations, which is encouraging from a soil stability perspective. Due to the inherently strongly aggregated nature of this soil, no meaningful change in aggregate stability, however, was measured. Significant improvements in soil stability were, nevertheless, found when fresh sludge was mixed with soil. If the sludge is not allowed to dry fully beforehand the polymer that it contains remains active and available for bonding of the soil particles together. Upon drying, these polymers become irreversibly attached to the soil substrate and will not become reactivated even upon re-wetting of the soil. This also explains why sludge aggregates found below only a few centimetres of the soil surface maintained their strongly aggregated nature. This suggests that although WTS consists of mainly silt and clay, the risk of this constituent fraction becoming released and clogging water conductive soil pores are, at present, low. Despite the high strength of the sludge aggregates the penetrometer soil strength (PSS) within the tilled layer was non-significantly different from the control treatment. Below the tilled layer, however, the PSS on the sludge-amended treatments were lower owing mainly to wetter soil conditions.

The research completed to date suggests that land treatment as an environmentally acceptable disposal option for water treatment sludge shows promise since soil conditions tend to be improved.

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List of Symbols and Abbreviations

α	=	contact angle of soil water
γ	=	surface tension of water (Nm^{-1})
ψ_g	=	gravitational potential (kPa)
ψ_m	=	matric potential (-kPa)
ψ_o	=	osmotic potential (kPa)
ψ_t	=	total potential (kPa)
ε_a	=	air-filled porosity ($m^3\ m^{-3}$)
K_Ψ	=	unsaturated hydraulic conductivity (ms^{-1})
K_a	=	air permeability. The SI units are m^2 but for easier interpretation the results are presented as μm^2
K_{fs}	=	field saturated hydraulic conductivity ($mm\ h^{-1}$)
K_s	=	saturated hydraulic conductivity ($mm\ h^{-1}$)
$p_{(0.05)}$	=	probability level at 5%
CEC	=	cation exchange capacity ($cmol_C\ kg^{-1}$)
DASD	=	dry aggregate size distribution
EC	=	electrical conductivity ($mS\ m^{-1}$)
EC_e	=	electrical conductivity of the saturation extract ($mS\ m^{-1}$)
FC	=	field capacity
G5	=	treatment of gypsum applied at 5 $Mg\ ha^{-1}$
G10	=	treatment of gypsum applied at 10 $Mg\ ha^{-1}$
GMD	=	geometric mean diameter (mm)
L2	=	treatment of lime applied at 5 $Mg\ ha^{-1}$
L10	=	treatment of lime applied at 10 $Mg\ ha^{-1}$
MOR	=	modulus of rupture (kPa)
MWD	=	mean weight diameter (mm)
P15	=	treatment of polyacrylamide applied at 15 $kg\ ha^{-1}$
P30	=	treatment of polyacrylamide applied at 30 $kg\ ha^{-1}$
PAM	=	polyacrylamide
PAW	=	plant available water ($m^3\ m^{-3}$ or $m\ m^{-1}$)
PSS	=	penetrometer soil strength (kPa)
RAW	=	readily available water ($m^3\ m^{-3}$ or $m\ m^{-1}$)
SAR_e	=	sodium absorption ratio of the saturation extract
SOM	=	soil organic matter
USCP	=	unbound silt and clay percentage
WASD	=	wet aggregate size distribution
WRC	=	water retention characteristic/curve
WTS	=	water treatment sludge

Chapter 1

Introduction

1.1 The concept of land disposal

Land disposal may be regarded as the practice of applying waste to soil with the intention of assimilating the waste (Loehr, 1984). The more traditional methods of waste disposal such as landfill is rapidly becoming more costly and environmentally less acceptable. Moreover, the rapid expansion of urban peripheries has meant that existing landfill sites are facing intense pressure for health, economic and aesthetic reasons. In addition, for landfill systems to remain operational, it is important that the waste that it receives be contained within the system, usually by a clay liner. According to Lesage and Jackson (1992), however, an artificial impermeable liner is not permanent, and with time, leachates are likely to percolate through to the surrounding environment. This has obvious implications for groundwater contamination, particularly if the waste disposal site is situated on permeable ground. Historically, a major advantage of waste disposal by landfill was its low cost. In view of the more stringent regulations that now govern the practice, such as a greater intensity of monitoring, these economic gains have largely been diminished.

This modern emphasis on environmental quality and recycling has led to authorities maintaining stricter waste disposal regulations. Of importance is the protection of watercourses, lakes and the marine environment (Cameron *et al.*, 1997). Alternative waste disposal strategies to landfill such as incineration while favourable, are unfortunately applicable only to organic materials and other combustible waste of high energy content. In some cases, however, the incineration of waste can result in unacceptable levels of pollutant emissions to the atmosphere such as heavy metals, acidic gases and trace organic constituents. This merely transfers the problem and does little in the way of assimilating the waste. As a result, in recent years, land disposal as an alternative waste disposal practice has been receiving increased attention from policy makers, land managers and the scientific community at large (Karlen *et al.*, 1997).

The practice of applying wastes to soils, despite the apparent infancy of the strategy, is not a new idea. According to Miller and Miller (2000) agriculturalists from early civilisations recognised the beneficial responses of applying animal manures to farmlands. Before industrialisation these products were often used as substitutes for commercially derived fertiliser. The post industrialisation phase, however, led to a rapid expansion in both the quantity and the types of wastes being produced. This emphasised the need for sound scientifically based management guidelines to be developed for the efficient disposal of these wastes onto agricultural lands. As Cameron *et al.* (1997) points out transferring guidelines from one waste disposal system to another is both inappropriate and potentially risky due to the inherent differences in physical, chemical and biological characteristics between the wastes. As Snakin *et al.* (1996) comments, anthropogenic disturbances of soils can lead to critical changes in the biosphere that may threaten the very existence of human beings.

Most of the studies concerned with land disposal reiterate the challenges of applying wastes to soils and the importance of rigorous scientific scrutiny. In this regard, strong emphasis is placed on understanding the potential implications of applying the waste product to soils, before the commissioning of the land treatment scheme. This obviously requires that the quality of the soil to which the waste is being applied, remain unaltered or improved. The very concept of soil quality is in itself difficult to define due to the strong disparity in perception of what is good soil quality. Consequently the Soil Science Society of America has chosen to define soil quality as "the soil's fitness to support crop growth without being degraded or otherwise harming the environment" (Karlen, *et al.*, 1997). In terms of this definition some of the key indicators of soil physical quality are water retention and aeration properties, infiltration characteristics, hydraulic conductivity, soil strength and structural stability. In the present context, should water treatment sludge impact negatively on existing soil properties, for example by decreasing aggregate stability or limiting water transmission, then the quality of the soil could be compromised. On the other hand, if soil physical properties are unchanged or improved, then it may be concluded that the land disposal of water treatment sludge could be a potentially feasible practice, at least in terms of soil physical quality.

A conceptual framework for the evaluation of soil quality (Figure 1.1) was further proposed by Karlen *et al.* (1997) and seems appropriate for adoption in the present work. The conceptual model presented by Karlen *et al.* (1997) sections the process of evaluating soil quality into several levels with each level being associated with some specific objective. There are two distinct phases to the structure. In the first phase a set of parameters are identified for their potential significance in quantifying the "ability of the soil to function". In the second phase those parameters selected in phase 1 are used along with other existing information to monitor soil quality.

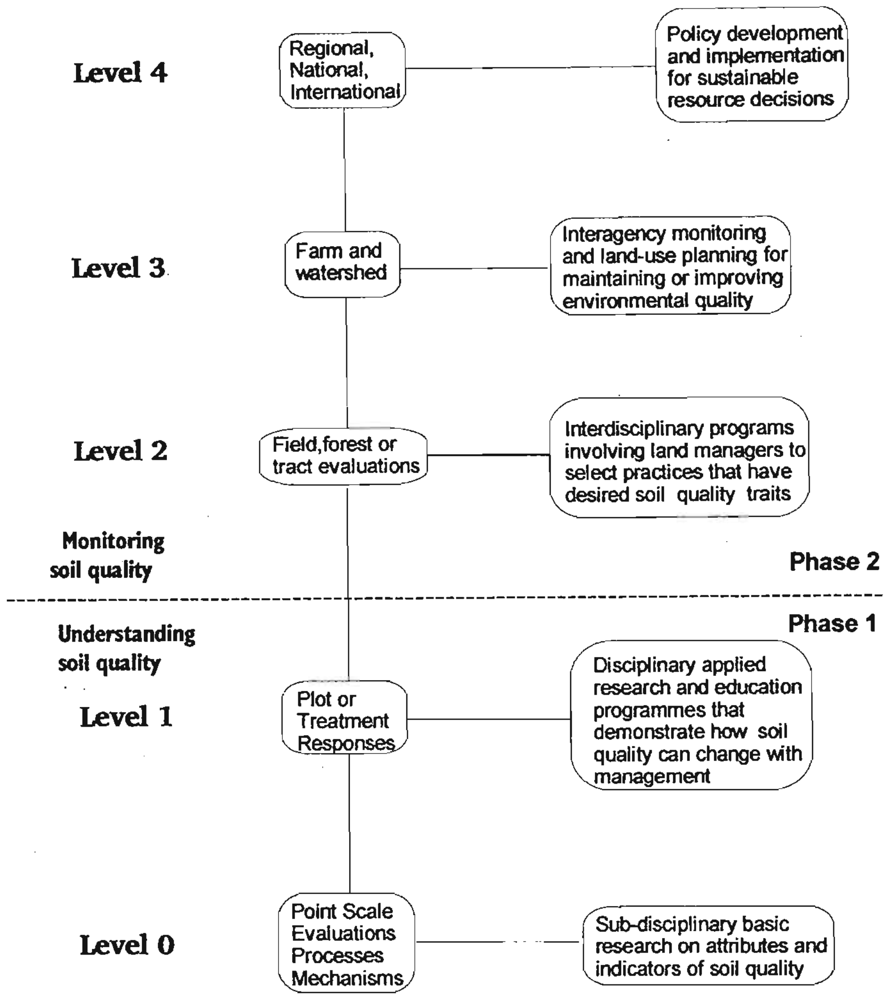

Figure 1.1 Multiple scales for soil quality evaluation (redrawn after Karlen *et al.*, 1997)

An integral part of the model according to the authors is that it offers flexibility in the choice of the parameters chosen for the assessment provided that the following conditions are met.

- The parameter must somehow influence the function for which the assessment is being made, for example biological indicators of soil quality for the function of sustaining plant growth might include parameters such as microbial biomass or respiration.

- The parameter should be measurable against some definable limit.
- The parameter should be sensitive enough to detect differences at the point scale in time and space.

Once suitable parameters which are based upon the intended aims of the investigation are selected, progression to level 1 occurs. Level 1 would be an evaluation of the potential implications on soil quality, of applying the waste product to the soil. This is essentially an experimental phase and would normally occur by way of field trials, laboratory and associated studies. It is perhaps work from this level that is most frequently reported in the literature. The major portion of the present work will fall within this category.

Ideally the knowledge gained from Level 1 will govern whether progression to level 2 is justified or not. If the results of Level 1 show that the applied waste is acceptable for land disposal then expansion from the plot scale to the field scale occurs, in other words the land treatment system becomes functional. Monitoring of the land treatment process then becomes important for essentially two main reasons, first to corroborate or add to the research findings from the plot scale and second to ensure the sustained maintenance of acceptable soil quality. During this phase, methods of waste disposal proposed earlier may be refined such that the land management practice is optimised.

Level 3 draws on the findings of the tiers below but places stronger emphasis on expanding the scale of the investigation to the farm or catchment level. Application of waste to one regional environment may not have the same responses as determined elsewhere. Consequently Level 4 will draw from larger spatial scales and integrate input from a wider range of disciplines. For example this stage of the management plan will incorporate results from Level 1 drawn from several studies conducted both in time and space and from monitoring information collected at several land treatment facilities.

Having established the feasibility of the land treatment system, its longer-term sustainability and the potential implications for environmental quality in general, the development of policies (level4) governing the land disposal practice is then possible. Such policy may be in the way of guidelines or carry some legal implication. This could be a specific limit to the amount of waste that can be applied, specific methods of applying the waste, or regulations governing the monitoring of the land disposal scheme. In this way laws can be applied more consistently without political interference. Further concepts of soil quality can then be readily appreciated by non-scientific personnel in the industries, regulatory agencies and the public (Tiller, 1992).

1.2 Motivation for this study

In 1997, Umgeni Water, the main supplier of bulk drinking water in the province of KwaZulu Natal, South Africa commenced operation of their water treatment plant. This plant effectively centralised operations in the region although a few of their satellite plants continue to operate. The unfortunate result of the now central works was that the production of sludge (a residual product of specifically the "drinking water" process) also became centralised. Water treatment plants are usually located close to the primary water source whereas landfill sites, out of necessity, are often sited in areas that pose little risk to contamination of water courses by leachate travel. Whereas previously the sludge was disposed of in landfill, this practice was regarded to be unfeasible given the anticipated volumes of sludge likely to be produced. In addition the transport of the sludge to landfill sites further afield has a host of inherent environmental drawbacks besides being economically unfavourable. In this regard alternative forms of sludge disposal were considered, one of which was land treatment.

In lieu of this, Umgeni Water purchased a property located proximally to their treatment works. Attempts at developing a suitable disposal strategy for the land treatment of the sludge were complicated by the paucity of research into water treatment sludges in general. Much of the information available was concerned with the engineering aspects of drinking water production. Where reference to the sludge *per se* was made in these works, this often referred to issues such as sludge thickening or maximising the recovery of water from the sludges, as discussed by Vesilind *et al.* (1991). In addition much of the existing literature on the subject is concerned chiefly with alum sludge. Recent advances made in high molecular weight polymer chemistry have led to polymeric flocculents becoming increasingly the preferred choice in the local water treatment industry. Consequently the legitimacy of extrapolating guidelines from abroad, directly to the present situation was in question due to the unacceptably high levels of uncertainty. Consultations with other water utilities in the country revealed the importance of researching land treatment as a potential disposal option for water treatment sludges. This decision led to the initiation of the present investigation.

1.3 Aims and Objectives

In light of the very poor research record that exists on the effects of water treatment sludge on soil physical quality, and the applied aspects of this study, of necessity, the research protocol had to adopt a broad scale approach. To ensure the applicability of the study beyond the needs of the Howick plant of Umgeni Water, this study was conducted on four soil types. These soils ranged in texture and structural stability. The aim was to evaluate any positive or negative impacts that the sludge may have on soil physical properties if applied to some of the more commonly occurring soils found within KwaZulu-Natal. Owing largely to costs, only two of these soils are represented in field studies although all of the soil types are included in the laboratory-based investigations. At the outset, the field study has been designed as a long-term evaluation, in which the soil physical, chemical and microbiological implications of the land disposal of water treatment sludge on soil quality are being addressed. In addition, importance was attached to addressing the range of aspects that would potentially determine whether land treatment as a disposal practice was a viable option or not. To maximise the research effort, each of these components is being addressed separately by individual researchers. The results from each component will then be integrated to decide whether the land disposal of water treatment sludge is an environmentally acceptable practice or not. The research to be presented herein, is an attempt at understanding the effects of the sludge on soil physical quality.

Specifically the research aims to determine:

- the effects of water treatment sludge on soil water retention and pore size distribution.
- the resulting impacts on infiltration and water transport under saturated and unsaturated conditions.
- the relative influence of water treatment sludge on soil strength.
- the effects of water treatment sludge on air permeability.
- the aggregation status of soils following application of water treatment sludge.

Based on results obtained from the investigations above, the work further aims to:

- Evaluate the feasibility of the land disposal of water treatment sludge, from a soil physical perspective.
- Derive an interim sludge handling and management policy for operation of the water treatment plant (should the findings suggests that land disposal is a feasible option).

1.4 Structure of this report

In attempting to meet the objectives given in Section 1.3, a range of aspects have been covered to evaluate the effects of WTS on soil physical quality. To communicate these findings more effectively, an outline of the thesis structure is presented here.

The first half of Chapter Two introduces the water treatment process. As will be shown in this Chapter, past research on the land disposal of water treatment sludge is at best sparse. Of the few works on the subject, most of these are concerned with the effect of the sludge on soil chemistry and fertility. The main findings from these works are reviewed in Section 2.3. Where past studies of a soil physical nature have been carried out, these are reviewed in greater detail in the relevant Chapters. Gypsum, lime and polyacrylamide are included in the experimental design to compare the effects of the sludge against these more well recognised soil conditioners. A description of these products and their potential sources are presented in Chapter Two but their significance on soil properties is reviewed in the relevant Chapters.

A description of the two field experiments and the procedures followed in establishing the trials form the focus of Chapter Three. Also included is a description of the additional soils included in the laboratory-based study and preparation of the sludge and soil mixtures. Some properties of the soils and sludge are given here.

The influence of WTS, lime, gypsum and polyacrylamide on soil water retention was monitored on several occasions at the Brookdale and Ukulinga trials. These findings are presented in Chapter Four. Owing to the inclusion of the Valsrivier and Cartref soils in the study and to reduce the variability encountered in the field studies, the effect of WTS on soil water retention (four soil types) was re-evaluated on disturbed repacked samples. The influence of WTS on plant available and readily available water concludes the Chapter.

Chapter Five is concerned with the effect of the sludge on saturated hydraulic conductivity (Ks), unsaturated hydraulic conductivity and soil water conservation. At the outset of the work concern was expressed that in adding silt and clay (WTS contains > 90% silt and clay by mass) to soil, blockage of water conductive pores could occur. Strong emphasis was therefore awarded to the testing of this theory. The results from two field approaches and a laboratory study using repacked samples are presented. Water retention curves and a measure of K_s obtained for the repacked sludge-amended treatments, formed the basis for evaluating the effect of the sludge on unsaturated hydraulic conductivity. Also discussed in Chapter 5 is the influence of the sludge on soil water conservation. The influence of the sludge on soil aeration is discussed in Chapter Six.

Chapter 7 investigates the influence of the sludge on soil aggregation. In this Chapter several approaches of realising this objective were tested to evaluate those methods best suited for the longer term monitoring of the land treatment scheme. During the investigation it became apparent that stronger improvements in soil aggregation may follow the application of fresh sludge rather than air-dry material. This hypothesis is explored in Section 7.5.

The penetrometer soil strength of the treatments, which were monitored regularly at the Brookdale and Ukulinga trials are discussed in Chapter 8. The implications of applying fresh sludge to soil, on strength development (Modulus of rupture) concludes Chapter 8. This aspect is a continuation of the work in Section 7.5. A general discussion of the main elements from this study and the potential feasibility of land disposal as a viable practice is given in Chapter 9.

In researching these various themes, some practical aspects related to the disposal of the sludge to land have come to the fore. Using Brookdale Farm as a case study, some guidelines for the handling and disposal of WTS are presented in Appendix 9.1.

Chapter 2

An overview of water treatment sludge production, its application to land and soil conditioners

2.1 The Water Treatment Process

A fundamental objective in producing potable water from turbid primary sources is the removal of the suspended and dissolved solids, organic matter and other contaminants. Primary source in the present context refers exclusively to waters contained in surface water bodies (lakes, rivers and dams) or groundwater resources. Although slight variability exists between the actual methods used in the water treatment process, the underlying principles remain the same. A typical sequence of events leading to the production of drinking water is shown in Figure 2.1.

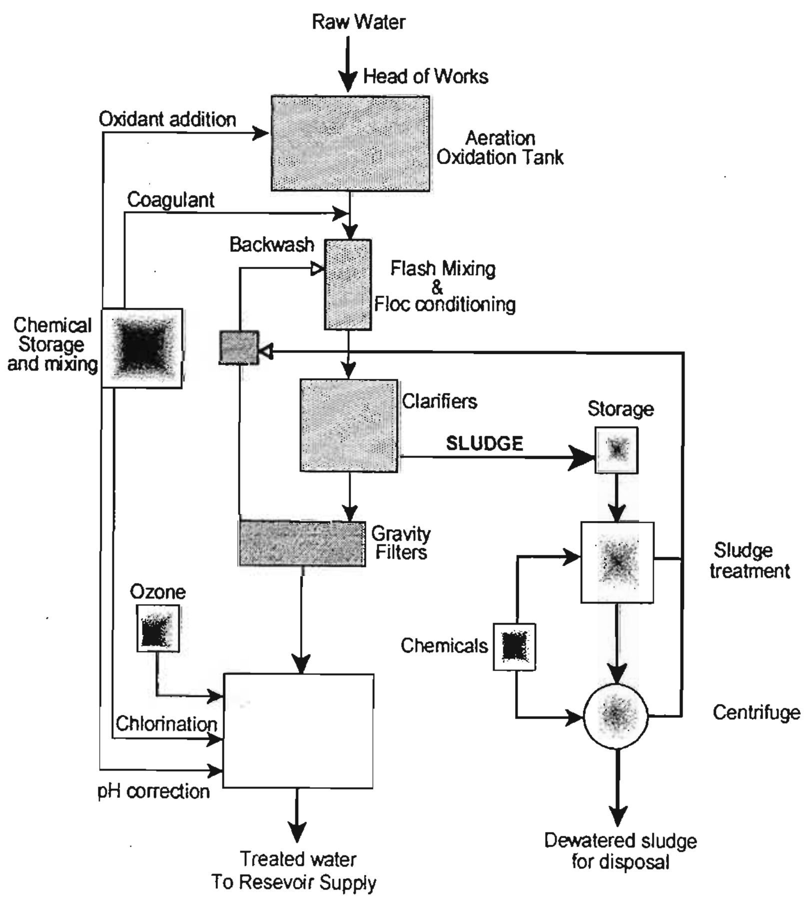

Figure 2.1 Schematic illustration of the drinking water treatment process (simplified after Umgeni Water, 2000).

The turbid water received at the inflow of the works is first passed through a filter screen to remove large objects following which the water is dosed with powdered activated carbon, sodium saturated bentonite and chlorine gas in solution. The purpose of adding these compounds is to reduce the odour and improve the taste of the final product. In addition the sodium ions from the bentonite act as an effective dispersing agent causing aggregates bonded to each other to deflocculate. The dispersed condition is favoured since it is in this state that the external surface area of the suspended particulate matter is at a maximum. Lime and chemical coagulants are then added. Typical coagulants used in the water treatment industry are aluminium sulphate ($Al_2(SO_4)_3.18H_2O$), ferric chloride (Fe_2Cl_3) and polyelectrolytes. The purpose of the lime ($Ca(OH_2)$)is to raise the pH of the water resulting in the chemical combination of cations with hydroxide ions (OH^-), sulphate ions (SO_4^{2-}) and carbonate ions (CO_3^{2-}) to form insoluble compounds (Tumeo, 1993). The water is then transferred to flocculation tanks equipped with large mechanical agitators.

Agitation of the water is necessary to increase the number of collisions between the smaller sized particles. Where cationic polymeric compounds are used as the main coagulant the nett positive charges of the polymer competes with the exchangeable cations held by the negative charge on the clay mineral surface. Bonding is therefore electrostatic (coulombic interaction) due to charge neutralisation (Aly and Letey, 1988). The effect of charge neutralisation is to compress the diffuse double layer, termed coagulation by Laird, 1997. In addition, the excess positive charge encourages the formation of cationic bridges which leads to the agglomeration of the mainly silt and clay-sized fraction (Laird, 1997). The relative adsorption of the polycation increases with decreasing valency of the exchangeable cation and decreases with increasing electrolyte concentration of the solution (Gu and Donner, 1992).

Because of the aeration process, flocs have a large surface area and can adsorb the dissolved matter from solution (Tebbutt, 1992). This results in the removal of both dissolved colour and colloidal turbidity. The concentrations of additives used in the water treatment process is dependent on the quality of the "raw" water. Water quality is influenced by factors such as the characteristics of the catchment, seasonal influences and local environmental conditions. Adjusting the dosage levels according to the quality of the water to be treated is therefore necessary. In this regard continuous monitoring of water quality at various stages of the operation process is carried out.

Following the agitation phase of the treatment process, the water is transferred to large settling tanks in which the flocculated material settles outs of suspension by gravity. The supernatant is then extracted and passed through sand filters to remove any further particulate matter and to achieve final clarification. To minimise clogging of the sand filters "backwashing" occurs in which water is introduced to the underneath of the filter at a rate much higher than the filtration rate. The rapid upward flow of water expands the filter bed causing the accumulated debris to be scoured of the surface. Water extracted from the settling tanks, although acceptable with regards to its particulate content, may still contain water-borne micro-organisms. Consequently disinfection of the water by the addition of chlorine occurs following which the water is pumped into storage tanks for distribution.

The flocculated material removed from the settling tanks comprises a solids' content in the order of 2-3 % and must be dewatered. Thickening of the sludge is important for two main reasons, first to maximise the rate of water recovery and second to convert the sludge into a manageable form for subsequent handling operations. To this end the sludge is transferred again to tanks equipped with mechanical agitators.

Sludge waters can exist in different locations relative to the particles. Vesilind *et al.* (1991) recognised essentially four main types namely free water, interstitial water, surface water and chemically bound water as shown in Figure 2.2.

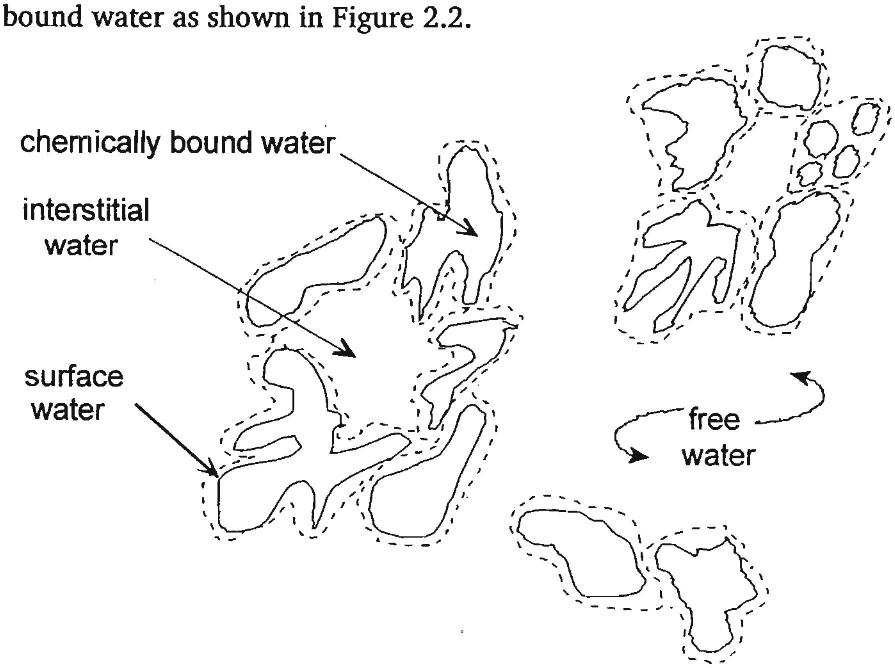

Figure 2.2 Conceptual visualisation of water in sludge (redrawn after Vesilind *et al*, 1991).

Free water is that surrounding the particles or flocs. Interstitial water is that contained within the flocs, most of which may be converted to free water by mechanical influences such as centrifugation. Surface water is the thin layer that surrounds the particle surfaces and cannot be removed by mechanical means. Chemically bound water is part of the chemistry of the solids and cannot be removed without thermo-chemical destruction of the particles. It follows therefore that much of the water recovered from sludge thickening is of essentially the free and interstitial forms although it must be noted that this in reality no physical differentiation exists between these types.

Following the mechanical agitation of the sludge, the solid fraction is passed through a high speed centrifuge after which the solids content of the sludge is in the order of 28 to 30 percent. It is in this form that the sludge is removed from the water treatment facility. The supernatant is simultaneously returned to the head of the water treatment plant.

2.2 An estimate of water treatment sludge production in South Africa

In Southern Africa 20 to 25 water treatment plants serve various regions of the country. Although accurate records of sludge produced at each of these plants are presently being sought, it is estimated that the combined production of sludge is well in excess of 200 000 m^3 per annum. The actual water content of the sludge following the dewatering process is, nevertheless, highly variable which from the previous discussion is due to the large number of parameters that regulate the water treatment process. If however, as an initial approximation, the average air-dry content of solids of 280 kg per cubic metre of wet sludge as determined at the Howick plant is extrapolated to all other water treatment sludges in the country, then the minimum estimate of 200 000 m^3 will equate to a mass of 56 000 metric tonnes of air dry material. Compared with other forms of wastes that are presently being land treated, such as flue dust and urban refuse, this is not a very large quantity. The problem is, however, that water treatment sludges under evaporative drying releases the free and interstitial water contained within the flocs rather slowly, more especially when the sludge is deposited in heaps as often happens under normal practice. It is common for several months to elapse before the sludge is completely air dry. It would therefore be in its "wet form" that the material would be disposed off to landfill. Consequently, although the mass of the sludge may be moderate, a more realistic perspective on sludge production will be to maintain the units of volume, since this is the extra landfill space needed for the material.

In April 2000 (Figure 6.2) the permeability of air was higher through the sludge amended soil cores than through the control treatment at -10 kPa matric potential, although differences between mean K_a values were statistically non-significant at the 5% level of probability ($F = 2.033$, $p = 0.163$, $DF = 3;12$). This finding was also valid when K_a was measured at a matric potential of -100 kPa except that the mean values were marginally greater due to higher air-filled porosities ($F = 0.169$, $p = 0.915$, $DF = 3;12$).

The June 2000 (Figure 6.3) investigation showed similar overall trends in K_a between the sludge-amended treatments which followed the order 1280 Mg ha^{-1} > 320 Mg ha^{-1} > 80 Mg ha^{-1} > 0 Mg ha^{-1}, although this sequence was more clearly expressed at a matric potential of -100 kPa than at FC. Nevertheless, as was found previously, differences in K_a for the sludge-amended treatments at both -10 kPa ($F = 1.267$, $p = 0.329$, $DF = 3;12$) and -100 kPa ($F = 1.121$, $p = 0.373$, $DF = 3;12$) matric potential were, however, statistically non-significant at the 5% level of probability.

Amending the Hutton soil with either lime, gypsum or polyacrylamide showed an overall non-significant influence in April 2000 on K_a at -10 kPa ($F = 1.061$, $p = 0.417$, $DF = 6;21$) and at -100 kPa matric potential ($F = 2.55$, $p = 0.052$, $DF = 6;21$). This also applied in June 2000, the data for which showed non significant differences in K_a at matric potentials of -10 kPa ($F = 1.909$, $p = 0.127$, $DF = 6;21$) and -100 kPa ($F = 2.093$, $p = 0.096$, $DF = 6;21$) when subjected to analysis of variance.

6.3.2.2 Ukulinga Field Trial

Mean K_a values (calculated from four replicates) measured on the undisturbed soil cores at matric potentials of -10 and -100 kPa collected from the Ukulinga field trial during April and August 2000 are shown in Figure 6.4 and 6.5, respectively. As was found for the Brookdale trial K_a was log-normally distributed so analysis of variance was carried out on log transformed data although actual values of K_a are reported. Irrespective of the time of the investigation or the matric potential at which K_a was measured, the general order in K_a between the sludge treatments was consistently found to be 1280 Mg ha^{-1} > 320 Mg ha^{-1} > 80 Mg ha^{-1} > 0 Mg ha^{-1}.

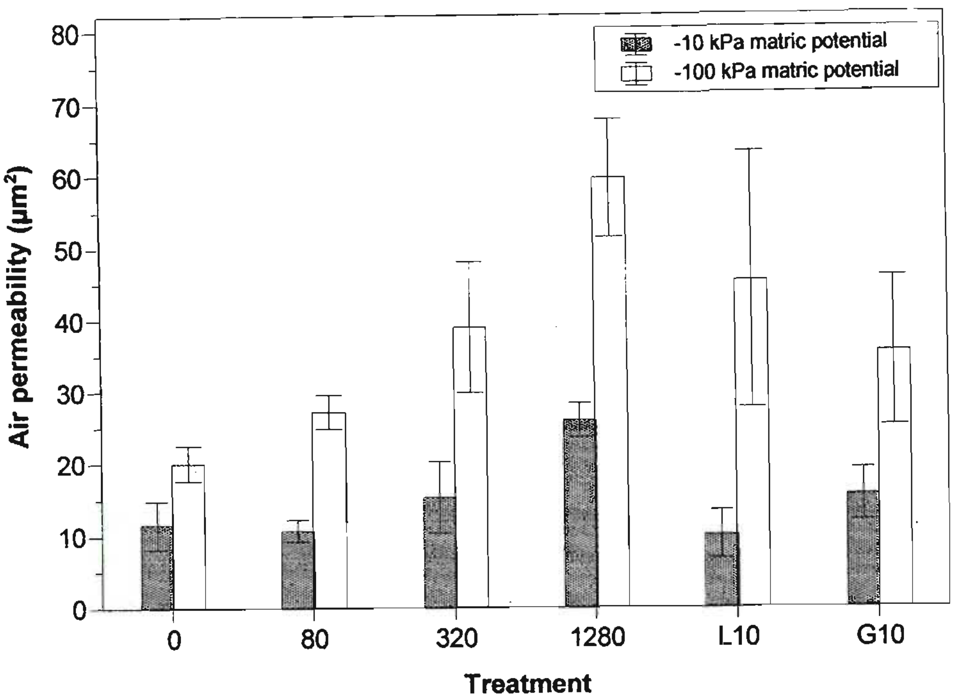

Figure 6.4 Mean air permeability (μm^2)measured on the treatments at the Ukulinga field trial in April 2000 for matric potentials of -10 and -100 kPa. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L10 and G10 are treatments of lime at 10 Mg ha^{-1} and gypsum at 10 Mg ha^{-1} respectively. The error bars represents the standard error of the mean.

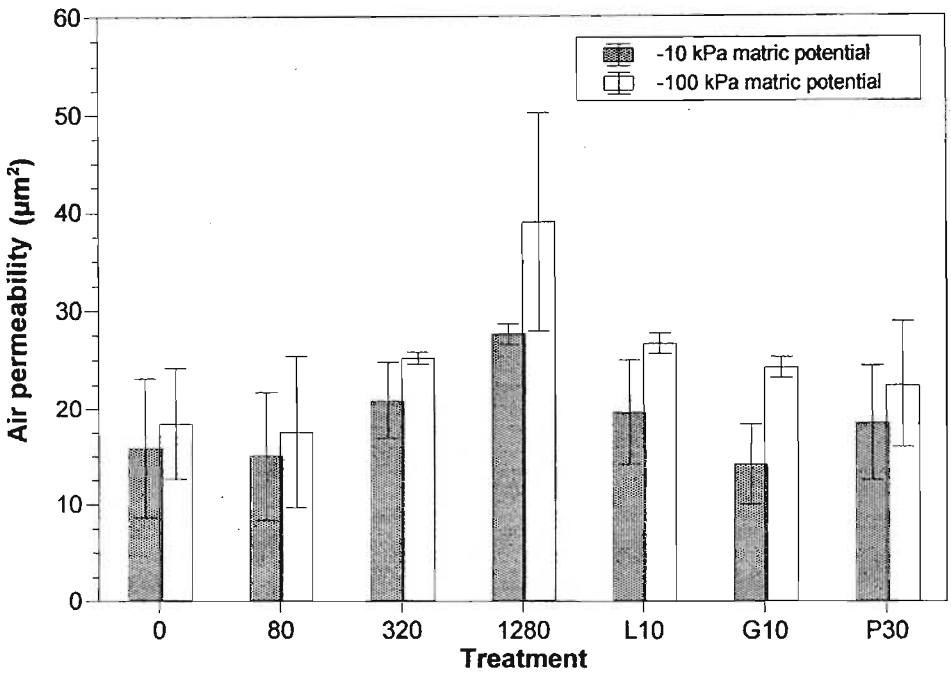

Figure 6.5 Mean air permeability (μm^2)measured on the treatments at the Ukulinga field trial in August 2000 for matric potentials of -10 and -100 kPa. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L10,G10 and P30 are treatments of lime at 10 Mg ha^{-1}, gypsum at 10 Mg ha^{-1} and polyacrylamide at 30 kg ha^{-1} respectively. The error bars represents the standard error of the mean.

In August 2000 such orders of difference in K_a between -10 kPa and -100 kPa were less distinct, In April 2000, more so for the sludge treatments than the control, there was a marked difference in K_a between FC and -100 kPa matric potential. For example, at FC the mean K_a for the 1280 Mg ha^{-1} treatment was 25.9 μm^2 but at -100 kPa this had increased to 59.6 μm^2. The reason for this is that during desorption of the sample to -100 kPa, some degree of volumetric shrinkage occurred, leading to the development of several cracks within the soil core. This effect was exacerbated particularly at the high levels of sludge application and so the air-flow rate through these samples was much higher. Nevertheless, considering that the air-filled porosity between the treatments was very similar at a matric potential of -100 kPa, it is likely that an increase in macro-pore continuity accounts for the higher mean K_a (although statistically non-significant) that was measured largely due to the more consolidated nature of the treatments at the time of sampling. Although some degree of shrinkage occurred during the desorption process this was not as extensive as was found previously. As was found in April 2000, K_a increased with increasing sludge content.

With regard to the effects of lime and gypsum on K_a in April 2000, non significant differences from the control were found at matric potentials of -10 kPa ($F = 2.118$, $p = 0.110$, $DF = 5;18$) and -100 kPa ($F = 2.227$, $p = 0.097$, $DF = 5;18$) when tested by analysis of variance. Similar findings were also encountered in June 2000 where lime, gypsum and polyacrylamide applications were found to have an air permeability non-significantly different from the control at matric potentials of -10 kPa ($F = 0.718$, $p = 0.640$, $DF = 6;21$) and -100 kPa ($F = 1.046$, $p = 0.425$, $DF = 6;21$).

6.4 Conclusions

WTS, gypsum, lime and polyacrylamide have thus far shown a statistically non-significant influence in the field on either the air-filled porosity or air-permeability of the Hutton or Westleigh soils. Air-filled porosities were highly variable both within and between the treatments, but, as explained in Chapter 4, this is perhaps due to the marked influence of tillage especially during the early phase of the investigation. The effect of tillage has, however, been gradually decreasing which should lead to an improved understanding of the changes brought about in aeration properties of the soils under study purely due to the amendments that they received.

Chapter 7

The Effect of Water Treatment Sludge on Aggregation and Soil Stability

7.1 Introduction

For the continued and successful operation of land treatment systems, it is essential that following the disposal of the waste for its assimilation, the soil remains in a well-aggregated state. Deterioration in soil structure or a decrease in its stability may hinder the capacity of the amended soil to support plant growth and maintain adequate levels of soil workability. The influence of WTS on soil stability was therefore considered important in evaluating the potential of land treatment as a disposal option.

7.2 The influence of water treatment sludge, gypsum, lime and polyacrylamide on aggregation-a review

Much has been written in the literature on the stability of soil aggregates and its associated structural expression, since this has a direct bearing on *inter alia* accelerated soil loss, permeability to air and water, strength development, cultivation practices and irrigation management. In its simplest form the stability of soil may be viewed as reflecting the balance between those forces that promote cohesion or attraction between the constituent particles and those forces acting against aggregation. A tremendous amount of effort has been expended by researchers, drawn from a variety of disciplines, to understand the mechanisms and controlling parameters under which these forces operate. A review of such works is, however, clearly beyond the scope of the present work. Consequently only a brief overview is presented here, specifically as it relates to water treatment sludge, gypsum, lime and polyacrylamide.

Water Treatment Sludge

Despite the progress in our understanding of the general mechanisms and factors that control soil aggregation, there have been limited studies on specifically the influence of WTS on this subject. In perhaps the earliest study on WTS in general, Rengasamy *et al.* (1980) applied alum sludge sampled from a water treatment facility in Adelaide, Australia, at a rate equivalent to 20 Mg ha^{-1} (as air-dried aggregates and as a suspension) to three different soil types, namely a solod, podzol and a red-brown earth. This study showed that the sludge had a beneficial effect by decreasing clay dispersion and slaking, so improving the stability of soil aggregates against the disruptive forces of water. Since then, despite comments raised by the authors that field studies were a necessary and logical extension of their research, few additional studies have been published on the subject. Skene *et al.* (1995) and Ahmed *et al.* (1997) do, however, refer to the aggregation properties of WTS, but not to their influence when incorporated with mineral soils.

Gypsum

Application of gypsum to structurally unstable soils is a common and well-accepted practice. Gypsum has been shown to improve infiltration, hydraulic conductivity and aeration and reduce soil strength, surface sealing and accelerated soil erosion (Greene and Wilson, 1984; Frenkel and Fey, 1989; Sumner *et al.*, 1990; Agassi and Ben-Hur, 1991; Illyas *et al.*, 1993; 1997; Borselli *et al.*, 1996). Gypsum contributes to soil stability by replacing sodium ions on the exchange complex with Ca^{2+} ions (Loveday, 1976; Greene and Ford, 1985). This forces a compression of the diffuse double layer that enhances clay flocculation. In addition the concentration of calcium in the soil solution prevents the disruption of soil aggregates, further adding to the stability of the soil (Shanmuganathan and Oades, 1983; Greene *et al.*, 1988; Kay and Dexter, 1990; Skidmore and Layton, 1992). For this reason, the benefits of gypsum are normally greater in sodic soils, which are characterised by a high exchangeable sodium percentage, and are usually dispersive (Arora and Coleman, 1979, Abu-Sharar *et al.*, 1987; Crescimanno *et al.*, 1995; Levy and Miller, 1997). A recent and comprehensive review of the use of gypsum for ameliorating structurally unstable soils is given by Levy and Sumner (1998).

Lime

Lime would normally be applied to acid soils to raise the pH (particularly that of the subsoil) to a target value of around 6.5 to 7 in water (Davies et al, 1993). Chan and Heenan (1998) showed that in liming a red earth from Australia (ESP < 0.5) at 1.5 Mg ha^{-1} the proportion of water stable aggregates in the size class >2 mm and < 50 μm increased after the third year. de Castro *et al.* (1999) also measured an increase in aggregate stability, three years after treating an oxisol from Brazil with 9 Mg ha^{-1} lime. Both authors ascribed an increase in Ca^{2+} levels as the main reason for their observations. According to Norton and Zhang (1998), however, at pH < 5, Al^{3+} keeps clay flocculated and aggregates stable. The replacement of Ca^{2+} in favour of Al^{3+} (Al^{3+} will precipitate as insoluble $Al(OH)_3$ as the soil pH increases) can lead to decreased aggregate stability. Smith *et al.* (1994) demonstrated this effect in treating a grey massive earth (pH in water of 5.38) with 2.5 Mg ha^{-1} lime. Six months after the trial was established, the exchangeable Al^{3+} was decreased and exchangeable Ca^{2+} increased. They, however, make no comment on the effect of lime on soil stability.

Since pH influences the electrical potential of the clay surface, changes in pH can affect the edge charge on clays and the surface charge of variable charge materials such as Fe and Al oxides. According to van Olphen (1977), edge to face bonding between clays and bonding of positively charged Fe and Al hydroxides to negative charged clay surfaces, is stronger at lower pH that at higher pH. Since lime raises the pH of the soil, it may hinder the extent of bonding and therefore potentially increase dispersion (Suarez *et al.*, 1984). Another reason offered for an improvement in soil stability after liming soils has been attributed to an increase in organic matter because of the more favourable conditions for crop growth (de Castro *et al.*, 1999). The role of organic matter in aggregate stability has been investigated by *inter alia* Haynes and Swift (1990), Piccolo and Mbagwu (1990; 1994), Angers *et al.* (1992), Beare *et al.* (1994), Barzegar *et al.* (1997) and Watts and Dexter (1997). The effect of organic matter on soil stability has been recently reviewed by Baldock and Nelson (2000) and offers useful insight into the mechanism by which these compounds operate. Despite the clearly established benefits of liming soils to crops, the effect on soil stability is less clear and mixed results have been presented in the literature.

Polyacrylamide

Many studies have shown improvements in stability following treatment of the soil with polymers, notably PAM. These works are extensively reviewed by Levy (1995) and more recently in Wallace and Terry (1998). The mechanisms by which polymers usually contribute towards soil stability are explained by Theng (1982) and remains one of the definitive publications on the subject. The extent to which PAM may increase soil stability is strongly related to its charge (positive, negative or neutral); molecular weight and density of charge

(Levy and Agassi, 1995). Excellent discussions of the relative significance of these factors on aggregate stability are provided by *inter alia* Malik and Letey (1991), Nadler *et al.* (1992) and Laird (1997) and so will not be repeated here. Nevertheless, a synopsis of some major works on the contribution of PAM in improving soil stability is given below.

Helialia and Letey (1988) showed an increase in clay flocculation when three different soils were treated with three polymers of differing molecular weight. Although all of the products that they tested decreased clay dispersion, the effects were greater for the higher molecular weight polymers than the lower weight polymers. Studies by *inter alia* Barry *et al.* (1991), Aly and Letey (1988), Chan and Sivapragasam (1996) have all shown similar findings. Bernas *et al.* (1995) applied polyDADMAC (a positively charged homopolymer of high molecular weight - polyDADMAC is used as a flocculent in the water treatment process) at 1% mass to volume to four soil types from Australia. They found that polyDADMAC increased the overall size of water stable aggregates (>2mm) and decreased the proportion of <0.125 mm aggregates in all four soils. These aspects are discussed in more detail in the sections that follow.

7.3. Aggregate size distribution of water treatment sludge-amended soils at the Brookdale and Ukulinga trials

7.3.1 Introduction

A variety of techniques for the measurement of aggregation and soil stability are documented in the literature. The exact details of these vary according to the objectives and the nature of the study. Agronomists and soil scientists have centred their efforts on tests such as hydraulic conductivity (McNeal and Coleman, 1966; Jayawardene and Blackwell,1991), the stability of individual soil aggregates against the mechanical action of water (Farres and Cousins, 1985; Loch and Foley, 1994) and clay dispersion (Lebron *et al.*, 1994). According to Loch (1994), historically, the measurement of aggregate stability has followed a sequence of:

- Wetting treatments to cause breakdown.
- Measurement of the aggregate size distributions.
- Description of the results obtained in some manner, either as a size distribution, an index of aggregation or in terms of some size distribution.

Analysis of the aggregate size distribution of soils has been used successfully in evaluating the effects of various waste products on aggregation. For example, Rasiah *et al.*, (1997) showed that when an oily waste sludge was applied to soil, a significant increase in the proportion of large aggregates at the expense of smaller aggregates occurred. This effect was linked directly to the effect of the oil in binding smaller soil fragments together. The aggregate size distribution of soils has also been used successfully in evaluating the influence of tillage on soil aggregation (Zobheck and Popham, 1990; Perfect and Kay, 1994). In view of these works, it was postulated that a similar rationale could be used to evaluate the influence of WTS on the aggregation process.

7.3.2 Materials and Methods

Changes occurring in the aggregate size distribution of the sludge amended treatments at the field trials was measured using a combination of dry and wet-sieving techniques. For convenience and to ensure a specific and roughly uniform volume of material being collected from each treatment, samples were collected using a core sampler (as described in Section 4.2.2.1). This prevented disturbing large areas of the plot, which was considered unwise especially for future measurement of water retention properties. Three cores from each plot (6 per treatment) were collected from the 0 to 50 mm soil depth at the Brookdale trial in February 1999, September 1999, November 1999 and February 2000 and at the Ukulinga trial in November 1999, April 2000 and October 2000. Three of these undisturbed cores were analysed for their dry aggregate size distribution (DASD). The remainder of the cores were used to measure their water stable aggregation properties.

Dry Aggregate Size Distribution

The soil samples on which the dry aggregate size distribution (DASD) was measured was first air dried. They were then sieved through a nest of sieves of apertures 9.5, 4.75, 2.0, 1.0, 0.5, 0.25, 0.125 and 0.053 mm for 10 minutes using a mechanical sieve shaker. The mass of the sample retained on each sieve was measured along with the length of the largest sized aggregate retained on the uppermost sieve (9.5 mm). Each aggregate size fraction was expressed as the percentage of the original mass of the soil. The mean weight diameter (MWD) was calculated as the sum of the mass fraction of soil remaining on each sieve multiplied by the mean aperture of adjacent sieves, ie

$$\mathbf{MWD} = \Sigma\, \mathbf{X_i W_i} \text{.... Eq. 7.1}$$

where Xi is the mean diameter of the size class midpoint (mm) and Wi is the proportion of the total sample retained on the sieve.

The aggregate size distribution of most soils follow a log-normal rather than a normal distribution (Kemper and Rosenau, 1986). The GMD is therefore often described by the geometric mean diameter (GMD) which is calculated as

$$\mathbf{GMD} = \exp\left(\Sigma\, W_i \log X_i \,/\, \Sigma W_i \right) \text{.....Eq. 7.2}$$

where W_i is the mass of aggregates in the size class i with an average diameter X_i and ΣW_i is the total mass of the sample. From a physical perspective the GMD may be regarded as the sieve size through which 50% of the material would pass.

Wet Aggregate Size Distribution

For the measurement of the wet aggregate size distribution (WASD) the content of each of the three cores selected for this component of the study was first passed through a 6 mm sieve, after which the sample was air dried. There is divergent opinion presented in the literature on whether aggregates used for wet-sieving should be in at its field water content or in its air dry state. According to Haynes and Swift (1990), aggregate stability as determined by wet-sieving of air-dried aggregates measures both the slaking potential of aggregates and their subsequent breakdown by mechanical action. In addition, for the sake of monitoring the change in aggregation over time, standardising the water content at which the test is carried out was considered important, which in this case is most convenient at the air-dry water content.

Once dry, the material was re-sieved to collect the 0.5 mm to 4.75 mm fraction. The equivalent of 40 g of oven dry sample was transferred to a wet-sieving apparatus similar to that described by Kemper and Rosenau (1986), which had a nest of sieves of size 4.75 mm, 2.0 mm and 1.0 mm. The water level was adjusted such that on the upstroke of the machine, the aggregates on the uppermost sieve remained just submerged at the highest point of oscillation. Before operating the machine the sample was first capillary saturated for 10 minutes after which the sieves were raised and lowered 37 mm at a rate of 29 times min^{-1} for 15 minutes. The material retained on each sieve was washed into tared 100ml beakers and dried at 105 ^{0}C for 24h, following which the sample was shaken in 0.5% sodium hexametaphosphate for 45 minutes. This was then washed through the respective sized sieve and that which was retained, transferred once more to tared 100ml beakers. The water stable fraction retained on each sieve (WSA_i) was calculated as prescribed by Angers and Mehuys (1993). The results were nevertheless expressed as the mean weight diameter calculated according to Equation 7.1. In the present context the upper and lower limit of the MWD was 3.375 mm (assuming that the entire sample was retained on the 2.0 mm sieve; and 0.75 mm (assuming that the entire sample passed through the 2.0mm and 1.0 mm sieve and was retained exclusively on the 0.5 mm sieve) respectively.

7.3.3 Results and Discussion

Dry Aggregate Size Distribution

The mean GMD values for the treatments collected from the Brookdale and Ukulinga trials are shown in Tables 7.1 and 7.2 respectively. As expected, good correlation between the MWD and GMD was obtained. The results are therefore discussed only in terms of the GMD. The MWD for the treatments is, nevertheless, presented as Appendix 7.1. In general, evaluating the influence of WTS on soil aggregation from the DASD at the Brookdale trial was disappointing. Although analysis of variance showed significant differences in the GMD between the treatments (except during September 1999) no consistent relationships between the GMD and the level of amendment could be found (Table 7.1). Regarding the effect of time on the DASD, inconsistent trends in GMD were also observed and this was independent of the treatment applied.

Table 7.1 Geometric mean diameters (mm)for the treatments investigated at the Brookdale field trial. The values quoted are the mean of three replicates.

Time	Sludge Content (Mg ha^{-1})				Lime and Gypsum Application (Mg ha^{-1})				Polyacrylamide Application (kg ha^{-1})		LSD $_{0.05}$
	0	**80**	**320**	**1280**	**L2**	**L10**	**G5**	**G10**	**P15**	**P30**	
February 1999	1.50	1.11	1.46	1.39	1.71	1.60	1.91	1.34	1.06	1.09	0.27
September 1999	1.04	1.11	1.21	1.16	0.97	1.06	1.12	1.19	1.00	1.09	N.S.
November 1999	1.62	1.59	1.60	1.59	1.48	1.33	1.42	1.31	1.14	1.15	0.22
February 2000	1.12	1.25	0.94	1.41	1.05	1.10	1.08	1.10	1.01	1.00	0.19

Table 7.2 Geometric mean diameters (mm) for the treatments investigated at the Ukulinga field trial. The values quoted are the mean of three replicates.

Time	Sludge Content (Mg ha^{-1})				Lime and Gypsum Application (Mg ha^{-1})		PAM Application (kg ha^{-1})	LSD $_{0.05}$
	0	**80**	**320**	**1280**	**L10**	**G10**	**P30**	
November 1999	1.19	1.05	1.03	1.45	1.15	1.07	ND	0.17
April 2000	1.19	1.22	1.24	1.47	1.31	1.14	ND	0.17
October 2000	1.29	1.21	1.16	1.41	1.24	1.15	1.23	NS

At the Ukulinga field trial, statistically significant differences in the GMD between the treatments were found during November 1999 and April 2000, but as before, there was no consistent trend with sludge content (Table 7.2). Nevertheless, the GMD of the 1280 Mg ha^{-1} treatment was consistently found to be higher than the other treatments. This was due more to the presence of large aggregates of WTS included in the sample than to its interaction with the soil. In time as these aggregates probably break down into a finer grade, the GMD should decrease.

These findings point to the complexities associated with evaluating the aggregation quality of soils from a measure of their *dry* aggregate size distribution and supports similar comments raised by *inter alia* Zobeck and Popham (1990), Skidmore and Layton (1992), Perfect and Kay (1997) and Rasiah *et al.* (1997). Climatic factors, management factors and inherent soil properties can have a strong influence on the dry aggregate size distribution. According to Rasiah *et al.* (1997), the log normal function (from which the GMD is determined) is based on the assumption that the probability of failure of aggregates does not depend upon its size. This assumption may not be entirely valid though, because larger sized aggregates are likely to fragment more easily than smaller aggregates. In this regard it is felt that as a monitoring index of WTS land disposal systems, a measure of the dry aggregate size distribution is of limited value in evaluating the rate of sludge breakdown.

Wet Aggregate Size Distribution : Brookdale Trial

The MWD of water stable aggregates for the treatments at the Brookdale trial is shown in Figure 7.1. Despite a uniform volume of material being collected from each treatment, the results showed high levels of variability, even when comparing the MWD within single treatments. For clarity, only the results for the February 1999 and February 2000 (the first and fourth) sampling dates are shown. The results for the intermediate sampling dates are nevertheless given as Appendix 7.2. In February 1999, analysis of variance showed significant differences ($p < 0.05$) between certain of the treatments at the Brookdale trial. In separating these means by least significant difference the 0 Mg ha^{-1}, 80 Mg ha^{-1} and 320 Mg ha^{-1} treatments were nevertheless non-significantly different from each other (Figure 7.1). WTS has high stability (MWD of 2.35 out of a possible 3.375). When added to soil at the 1280 Mg ha^{-1} level these aggregates account directly for the increased MWD. This was indeed the case since much of the material that remained on the sieves at the end of the test was composed of mainly sludge aggregates.

The results of the study carried out a year later (February 2000) showed statistically non significant differences in the MWD between the sludge treatments ($F = 2.802$, $p = 0.108$, $DF = 3;8$). Compared with the results from February 1999, the mean values in February 2000 were much lower. Inspection of the material retained on the 2 mm sieve for the 1280 Mg ha^{-1} treatment showed a far lower proportion of sludge particles than was found previously. This was due to the breakdown of these particles over time into a finer grade. From these findings it is apparent that the MWD of the sludge-amended treatments was influenced more strongly by the breakdown of sludge particles that to the interaction of the sludge in stabilising or destabilising soil aggregates.

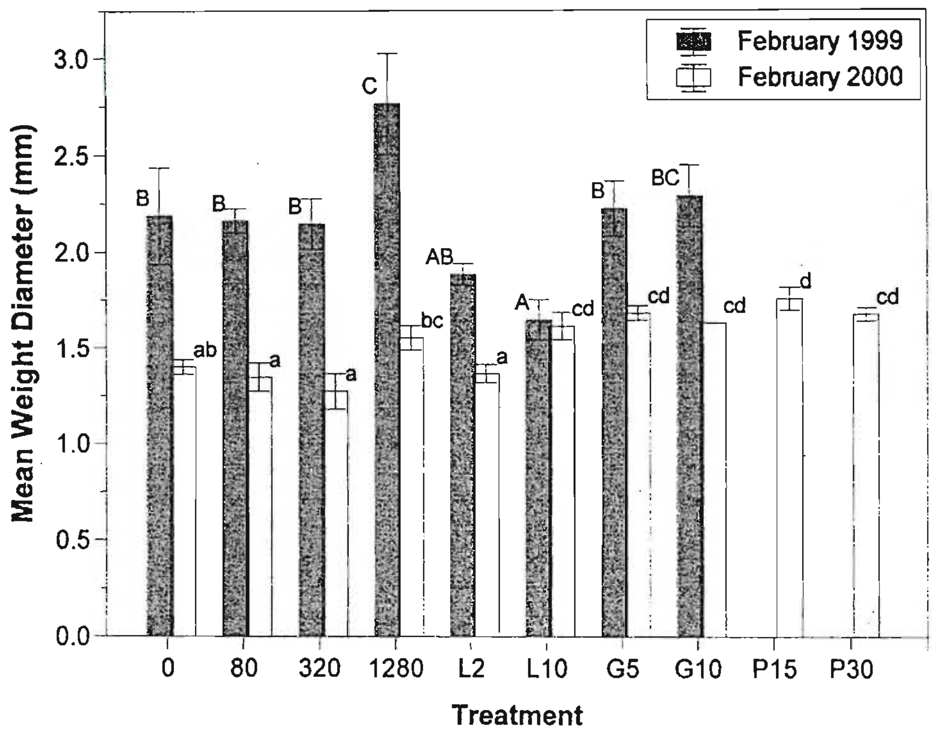

Figure 7.1 Mean weight diameters (mm) of water stable aggregates for the treatments measured at the Brookdale trial during February 1999 and February 2000. Each value represents the mean of three replicates. The treatments are as described in Figure 5.1. The error bars represents the standard error of the mean. Means followed by the same letters are statistically non-significant at $p < 0.05$ when separated by least significant difference.

With regard to the gypsum, lime and polyacrylamide (February 2000) treatments, these products did not cause any meaningful change on the aggregation of the Hutton soil. It is possible that because of the high stability shown by the Hutton soil in its unamended state (MWD in the order of 2 mm), the sensitivity of the wet-sieve method was not sufficient to detect any changes caused by these products on aggregate stability. The marginal decrease in the

MWD of the control treatment could be related to a decrease in soil organic matter (considering that the treatments were maintained fallow). Inconclusive evidence was, however, obtained in evaluating the total organic carbon of the samples collected during February 1999 and February 2000. In general the measurement of water stable aggregation for evaluating the effects of the WTS, gypsum, lime and polyacrylamide on the stability of the Hutton soil at the Brookdale trial produced unconvincing results, which was disappointing. Nevertheless, as the sludge breaks down into a more uniform grade, it is expected that the high levels of variability encountered in the findings of this study should decrease.

Wet Aggregate Size Distribution : Ukulinga Trial

Compared to the Hutton soil, the Westleigh soil (Ukulinga trial) is less strongly aggregated, as shown by the MWD in the order of 0.5 mm measured on the control treatment (Figure 7.2).

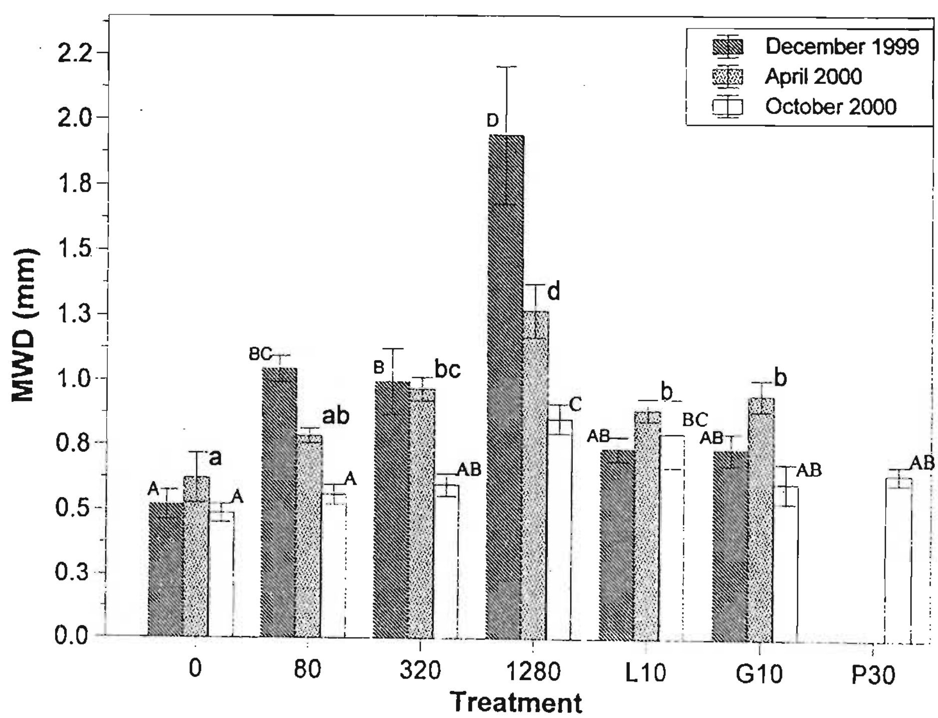

Figure 7.2 Mean weight diameters of water stable aggregates for the treatments measured at the Ukulinga trial during December 1999, April 2000 and October 2000. Each value represents the mean of three replicates. The treatments are as described in Figure 5.4. The error bars represents the standard error of the mean.

Addition of sludge to this soil caused an almost immediate increase in the MWD due to the controlling influence of the highly water-stable sludge aggregates. With regard to the sludge treatments, in December 1999, even at the 80 Mg ha^{-1} level the MWD was almost double that of the control. With time, however, as these sludge aggregates broke down, less material was retained on the 2 mm sieve and the MWD across all of the treatments decreased. As was found at the Brookdale trial, the largest decrease in the MWD was found on the 1280 Mg ha^{-1} treatment. Nevertheless, in October 2000 (a year since the trial was established), the MWD of the 1280 Mg ha^{-1} treatment was significantly different from that of the control (Figure 7.2). The 80 Mg ha^{-1} and 320 Mg ha^{-1} treatments on the other hand, were non-significantly different from the control or from one another at the 95% confidence interval ($F = 13.27$, $p = 0.002$, DF = 3;8).

With regard to the lime and gypsum treatments, both products caused a similar increase in the MWD, which in April 2000, was in the order of 0.2 mm greater than that of the control. Since then, a decrease in aggregation seems to have occurred, presumably due to the leaching of the amendment from near the soil surface. The polyacrylamide treatment sampled in October 2000 showed a MWD comparable to that of the control treatment, which implies that this product had little influence on improving the aggregation of this soil.

7.4 Electrolyte effects of water treatment sludge and implications for soil stability

7.4.1 Introduction

The electrolyte composition of the soil solution is an important factor that influences soil stability (Kay and Dexter, 1992). In soils essentially four cations are recognised as being of interest in this regard, namely calcium, magnesium, sodium and potassium (Keren, 2000). The influence of sodium in destabilising soil structure is well documented in the literature and much progress has been made in understanding the mechanisms by which this occurs (Emerson, 1994; Lebron *et al.*, 1994; Levy, 2000). The reason for this is that because of its monovalent charge and larger hydrated radii, sodium ions are less strongly absorbed on the exchange complex of soils than would be calcium or magnesium ions (which have divalent charge and smaller hydration radii) (Emerson, 1994). This causes more of these ions to move away from the clay particle surfaces into the diffuse-double layer of exchangeable cations, than would be the case for magnesium and calcium ions (Tucker, 1985). The larger hydration radii of sodium ions would also expand the diffuse double layer. This creates high swelling pressures between the clay platelets that results ultimately in their dispersion (Levy, 2000). Since sodic soils are not featured in the present study, it will not be discussed further, but a recent comprehensive review of the subject has been presented by Sumner and Naidu (1998). Since WTS sludge contains appreciable quantities of calcium and magnesium it was postulated that these ions, if released into solution, could improve aggregate stability. Because of this, evaluating the electrolyte status of the soil solution was considered important.

7.4.2 Materials and Methods.

7.4.2.1 Brookdale Trial

At the Brookdale trial the study was carried out in April 2000 at the end of the summer rainfall period on the 1280 Mg ha^{-1}, 640 Mg ha^{-1} and 320 Mg ha^{-1} mulch treatments and the gypsum and lime treatments. To differentiate between the "incorporated" treatments, the terms M_{1280}, M_{640} and M_{320} will be used henceforth to refer to the mulch treatments. The subscripts 1280, 640, and 320 refer to sludge levels of 1280 Mg ha^{-1}, 640 Mg ha^{-1} and 320 Mg ha^{-1} respectively. A small area (0.2 x 0.2m) of the sludge layer at two randomly selected positions within each of the mulch plots were first excavated to expose the soil surface underneath. Soil samples at each of these positions were then taken using a 60 mm diameter auger at 100 mm depth intervals down to a depth of 0.8 m. In addition the lime and gypsum treatments were also sampled.

The samples were transported to the laboratory, air dried, crushed to pass a 2 mm sieve and then mixed thoroughly before further use. Saturated pastes were prepared according to the guidelines of Richards (1954) and the soil solution extracted under suction using Buchner funnels. The mass water content of each paste was measured by drying a sub-sample for 24h in a force draught oven set at 105°C. For all samples the EC_e and pH of the extract were measured using a Radiometer CM 83 meter and Radiometer Ion 85 analyser, respectively. In measuring the Ca^{2+}, Mg^{2+} and Na^+ concentrations, 1.0 ml of extract was diluted with 9.0 ml of KCl solution (2500 mg L^{-1}) and in the case of the K^+ concentration, with 9.0 ml of CsCl solution (1200 mg L^{-1} Cs). The major cations in solution were analysed by atomic absorption spectroscopy.

Since the preparation of soil pastes is both time consuming and expensive it was considered prudent to investigate the possibility of meeting the objectives of the investigation by preparing a 1: 5 (soil : water) extract. Ten grams of each sample were added to centrifuge tubes and shaken in 50 ml of distilled water for 15 min on a bench shaker, allowed to stand for an hour, and then shaken for a further 5 minutes . The soil solution was then filtered through Whatman No. 2 paper after which the electrical conductivity (EC), pH and major cations were measured as for the saturated soil paste extracts.

The aggregation percentage for all samples was calculated according to Richards (1954) as

$$\frac{(\text{Total silt + clay after complete dispersion - unbound silt and clay}) \times 100}{\text{Total silt + clay after complete dispersion}} \quad \text{......Eq. 7.3}$$

In measuring the unbound silt and clay, 20g of sample was placed in 1L sedimentation cylinders to which was added distilled water in such a manner so as to favour wetting by capillarity rather than by flooding. Once the soil was wet the cylinders were brought to volume after which they were shaken end over end for 20 times over a period of approximately 40 s. The unbound silt and clay fraction was then measured by the pipette method (Kemper and Rosenau, 1986).

7.4.2.2 Ukulinga Trial

At the Ukulinga trial , from April 2000, the soil solution at a depth of 0.20 m was sampled by installing soil water samplers (Figure 7.3).

Figure 7.3 Soil water samplers used for collecting the soil solution at the Ukulinga trial.

These consists of a porous ceramic cup (although other material is sometimes used) cemented to a PVC tube. In operating these samplers a suction is created within the sampling system. Water is drawn inwards out of the pores of the cup until a corresponding capillary pressure occurs in the pores. If the capillary pressure in the suction cup is lower than in the soil, water flows from the soil into the suction cup, from where it is collected. A full review of the design and operation of these probes is given by Grossman and Udulft (1991) and Hendershot and Courchesne (1991). An advantage in using this apparatus is that it is possible to obtain a more continuous record of the soil solution composition than using extracts from soil pastes. The EC, pH and major cations of the soil solution were analysed as described previously.

7.4.3 Results and Discussion

7.4.3.1 Brookdale Trial

According to Tanton *et al.* (1988) there are two processes by which salt can move in soil, namely diffusion resulting from a concentration gradient and mass transfer with the seepage. Since diffusion is much too slow to be effective in transporting salt over long distances it can be safely inferred that large quantities of salt can only be transported by mass movement of water (Tanton et al., 1988). This assumption is extended to the discussion that follows.

In general, the concentration of solutes measured on the saturated soil pastes were in the order of ten-fold that measured on the 1:5 extract. This suggests that at least as a monitoring index, the 1:5 extract (with its associated lower cost of measurement) may be used for a rapid appraisal of the soil solution when amended with WTS. Since the trends shown by both extraction methods are similar, only the results from the saturated soil paste will be discussed further, while the data for the 1:5 extracts are presented in Appendix 7.3.

Characteristics of the saturation extract

Electrical conductivity (ECe)

When applied at the M_{320} and M_{640} levels, WTS had a marginal influence on the EC_e of the soil solution, although at equivalent soil depths the EC_e for these treatments was greater than that for the control treatment. The control treatment on the other hand, shows a near constant EC_e distribution in the 0 to 0.5 m depth range but increases slightly from 0.5 m downwards. This increase is probably related to the residual influence of the lime that was added to this field before the trial was established. The M_{320}, M_{640}, L_2 and L_{10} treatments shows strong similarity in their respective curves, all of which fall within a relatively narrow EC_e range between 25 to 60 mS m^{-1}. The slightly higher values recorded on the gypsum treatments is understood in light of the high solubility of this amendment. As expected the G_{10} treatment shows on average a larger EC_e than the G_5 treatment at most soil depths, although there are a few exceptions in this trend.

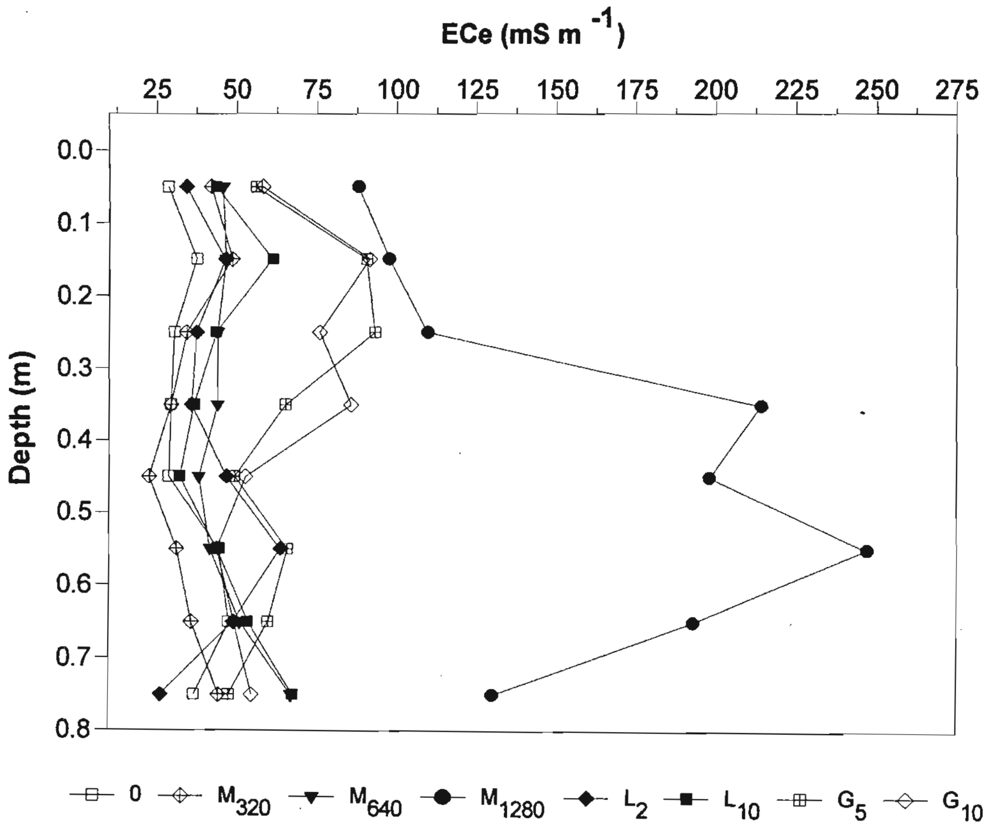

Figure 7.4 Electrical conductivity (EC_e) of the soil solution at 0.1m depth intervals for selected treatments at the Brookdale farm trial. Each data point is the mean of duplicate determinations. 0, M_{320}, M_{640} and M_{1280} refer to sludge treatments (Mg ha^{-1}) which had been applied as a mulch. L_2, L_{10}, G_5 and G_{10} are treatments of lime at 2 and 10 Mg ha^{-1} and gypsum at 5 and 10 Mg ha^{-1} respectively.

Of special interest is that the M_{1280} treatment shows substantially larger EC_e values than the rest. In the depth range 0 to 0.3 m, the EC_e curve for this treatment shows a gradual increase but below 0.3 m this increases sharply to a maximum of 260 mS m^{-1} then declines appreciably thereafter. Of additional interest is that whereas the peak (maximum EC_e) of the curves are generally at much shallower depths for the other treatments, in the case of the M_{1280} treatment this occurs at a much deeper depth (0.5 m - 0.6 m). The maximum value for this treatment is nevertheless well below the 400 mS m^{-1}, often regarded to be the threshold between saline and non-saline soil conditions (Richards, 1954).

Calcium

The concentration of Ca^{2+} in the saturation extract for the control treatment generally ranged between narrow limits of 0.7 mmol$_c$ L^{-1} to 1.75 mmol$_c$ L^{-1}. The WTS, lime and gypsum treatments increased the concentration of calcium in solution to varying degrees. As was expected the higher application rate of each amendment showed more of a response than its corresponding lower application rate.

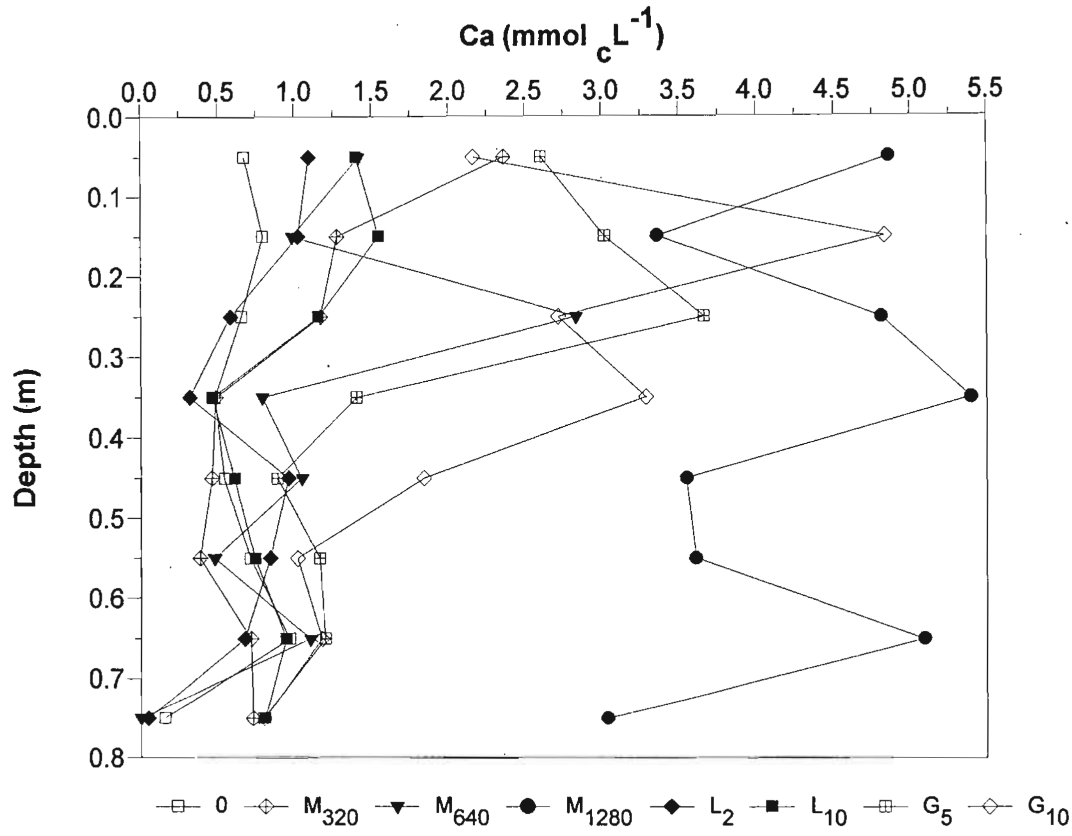

Figure 7.5 Mean calcium concentration of the saturation extract at 0.1m depth intervals for selected treatments at the Brookdale farm trial. Each data point is the mean of duplicate determinations. The treatments are as described in Figure 7.4.

Due to its low solubility, the influence of lime in increasing the concentration of calcium is accentuated nearer to the soil surface than at depth. Gypsum on the other hand, which is substantially more soluble compared with lime, increased the Ca^{2+} concentration of the saturation extract at much deeper depths. The G_{10} treatment showed a maximum value of 4.8

$mmol_c\ L^{-1}$ in the 0.1 m to 0.2 m depth increment and generally decreased thereafter. The G_5 treatments also followed approximately the same trend as for the G_{10} treatment except that the peak of the curve (3.6 $mmol_c\ L^{-1}$) occurred in the 0.2 m to 0.3 m depth increment. From a depth of 0.4 m the Ca^{2+} for the G_{10} treatment was similar to the rest of the treatments (except the M_{1280} treatment).

The addition of 320 Mg ha^{-1} and 640 Mg ha^{-1} WTS increased the Ca^{2+} concentration, which was comparable to that of the lime treatments but less effective than that of the gypsum treatments. On the other hand, when WTS was applied at the 1280 Mg ha^{-1} level, a substantial increase in the concentration of Ca^{2+} in the saturation extract occurred. For the M_{1280} treatment the calcium concentration of the extract ranged between 3.5 and 5.4 $mmol_c\ L^{-1}$, with the maximum value being recorded in the 0.3 m to 0.4 m depth increment.

Magnesium

The results of Figure 7.6 in general show higher levels of Mg^{2+} in the soil solution when compared to Ca^{2+} (Figure 7.5).

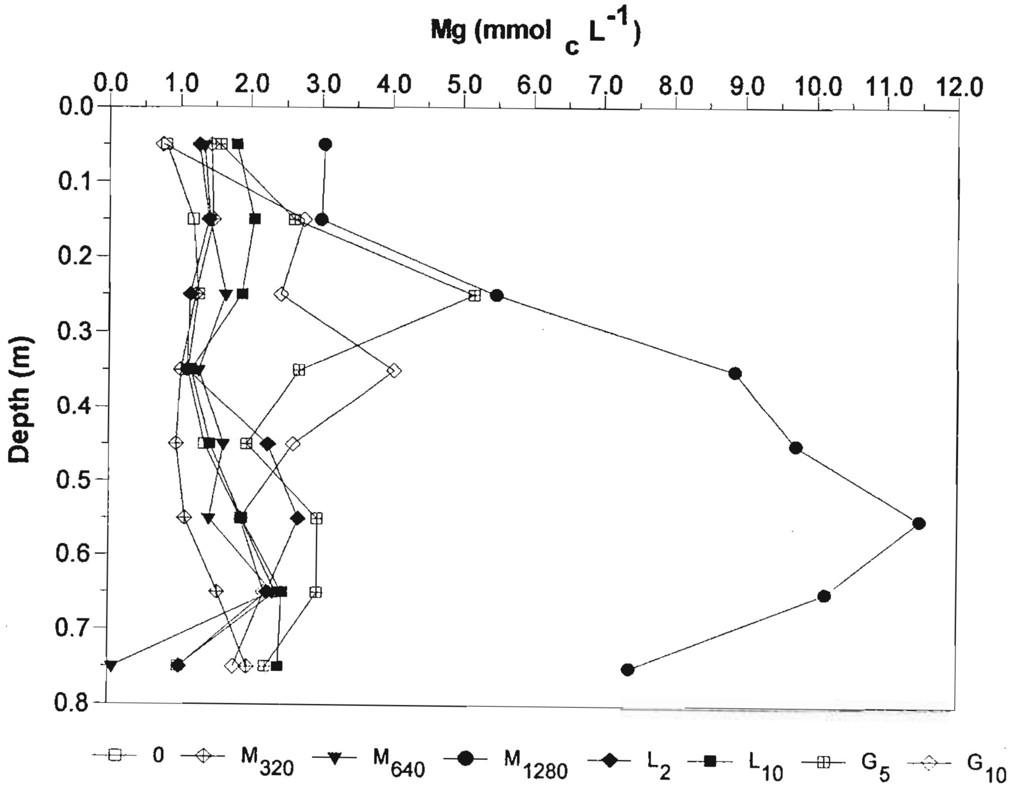

Figure 7.6 Mean magnesium concentration of the soil solution at 0.1m depth intervals for selected treatments at the Brookdale farm trial. Each data point is the mean of duplicate determinations. The treatments are as described in Figure 7.4.

It is interesting that the concentration of Mg^{2+} measured on the gypsum treatments were generally greater than that of the dolomitic lime treatments, considering that gypsum in its pure form ($CaSO_4.2H_2O$) contains no Mg^{2+}. The main reason for this is that Ca^{2+} will tend to displace Mg^{2+} from the exchange complex so driving the latter ion into solution, and encouraging its downward displacement. In addition since lime is less soluble than gypsum, its efficiency for displacing Mg^{2+} from the exchange complex is presumably lower. Compared to the rest of the treatments the M_{1280} treatment shows substantially higher Mg levels presumably due to the apparently faster rate at which the dolomitic lime is released from the sludge. With regard to the M_{320} and M_{640} treatments these showed very little difference from that of the control and generally ranged between narrow limits (1 to 2 mmol$_c$ L^{-1}) throughout the depth of interest.

Potassium

Although there was some leaching of soluble K^+ from the sludge to greater soil depths in the case of the M_{1280} treatment, in general the levels of potassium (K^+) in the saturation extract for the rest of the treatments was quite low and typically ranged between 0.01 mmol$_c$ L^{-1} to 0.035 mmol$_c$ L^{-1}(Figure 7.7).

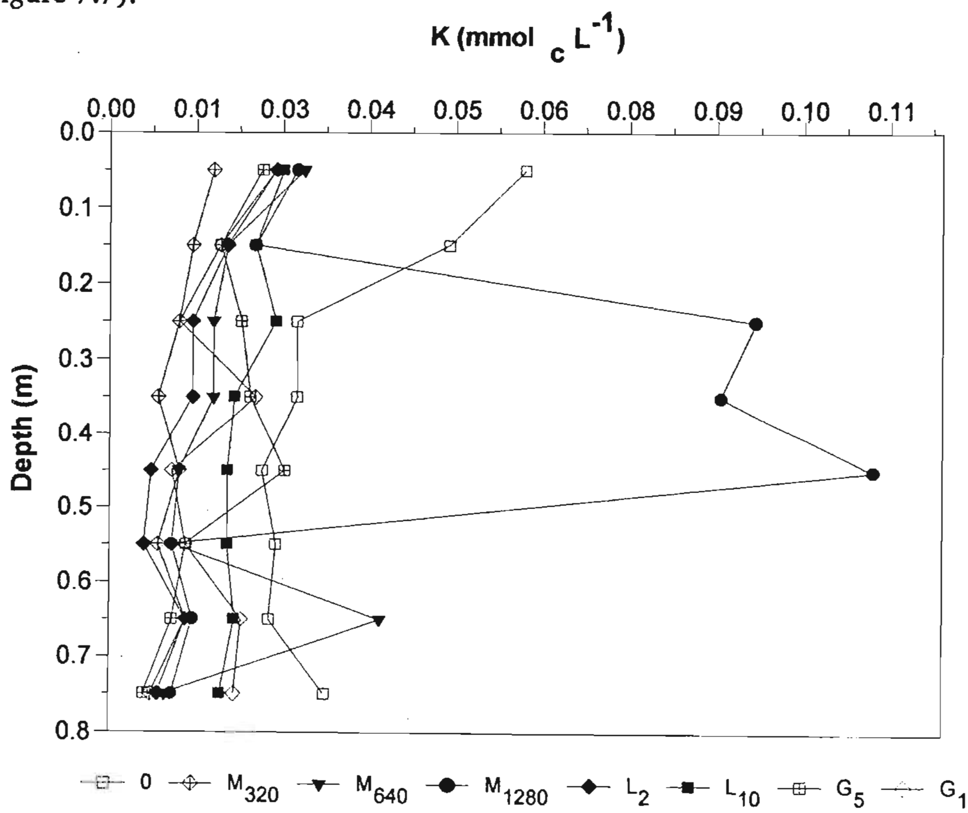

Figure 7.7 Mean potassium concentration of the saturation extract at 0.1m depth intervals for selected treatments at the Brookdale farm trial. Each curve is the mean of duplicate determinations. The treatments are as described in Figure 7.4.

For the control treatment the K^+ levels were marginally higher nearer the soil surface but declined substantially from a depth of 0.2 m and remained nearly constant thereafter. The reason for this is unclear.

Sodium

The levels of sodium for all treatments were generally low and with the exception of the M_{1280} treatment were generally < 0.7 $mmol_c\ L^{-1}$ (Figure 7.8).

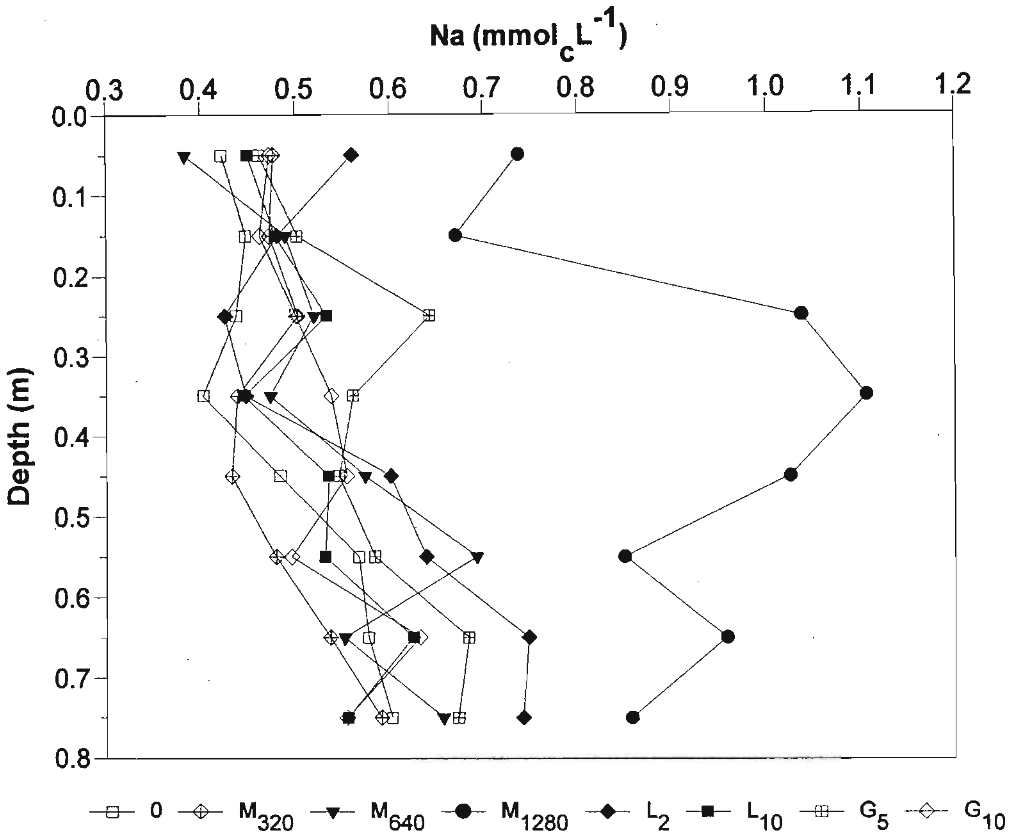

Figure 7.8 Mean sodium concentration of the soil solution at 0.1m depth intervals for selected treatments at the Brookdale farm trial. Each curve is the mean of duplicate determinations. The treatments are as described in Figure 7.4.

On the M_{1280} treatment the maximum concentration of Na^+ occurred in the 0.3 m - 0.4 m depth interval (1.1 $mmol_c\ L^{-1}$) and although this was higher than that of the control treatment it is nevertheless still quite low by normal standards. This was expected since the sludge contains low levels of exchangeable Na^+ (SAR_e = 0.96). In view of this, it is probable that much of the sodium measured in the soil solution was driven off the exchange complex by Ca^{2+} and Mg^{2+} since sodium is both monovalent and strongly hydrated with a low affinity for charged surfaces.

In addition the Hutton soil itself has rather low levels of sodium (SAR_e = 1.02) and was not expected to contribute any appreciable quantities of this cation to the soil solution. Further evidence for cation exchange being an important process in this soil may be seen in the findings from the gypsum treatments which also show slightly higher levels of Na^+ in solution compared with the control treatment . Compared to gypsum, lime showed a lower tendency to displace Na^+ because of its lower solubility. It is interesting that the M_{320} and M_{640} treatments show little difference from the control treatment in the concentration of Na^+ in solution, which supports the general trend reported for the divalent cations.

The results presented above confirm the initial hypothesis that WTS represents a potential source of divalent cations, notably magnesium and calcium. The deeper depths to which the electrolyte charged wetting front has penetrated is probably related to the much higher porosity of the mulch layer which would tend to allow for a much greater flux of electrolyte enriched water to move through the soil beneath.

Electrolyte effects on the stability of the Hutton soil

Despite the concentration of divalent cations in the saturation extract of the sludge-amended Hutton soil having increased, such influence on soil stability was difficult to evaluate. The main reason for this is that in its pristine state the Hutton soil is strongly aggregated. Consequently, non significant differences in the percentage aggregation of the samples taken from the various treatments (at equivalent depth) was found (Appendix 7.4). It may also be that the method employed in evaluating the effect of the sludge on aggregation was not sensitive enough to detect the changes that might have occurred. It is probable that where this exercise to be repeated in the presence of an inherently less stable soil type, then a stronger response to the electrolytes leached from the sludge, gypsum and lime may have occurred.

7.4.3.2 Ukulinga trial

The general changes reported in the soil solution composition of the treatments at the Brookdale trial was similar to those that occurred at the Ukulinga trial, where use was made of soil water collectors (Table 7.3) . Because of the low rainfall in the area, sufficient sample for analysis was only collected on a few occasions. Nevertheless, it was interesting that when samples were collected the largest volume was consistently obtained from the 1280 Mg ha^{-1} treatment. This is understood considering that under saturated and near saturated conditions the permeability of this treatment was much higher than the rest. Despite the advantages of this system of measurement field experience supports the comments made by Grossman and Udluft (1990) that the water content of the soil and its unsaturated hydraulic conductivity may act as limiting factors against this form of monitoring.

Table 7.3 Characteristics of the soil solution collected at the Ukulinga trial using soil water collection apparatus .

Date	Treatment	EC (mS m^{-1})	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	SAR
			mmol$_c$ L^{-1}				
18/05/2000	0	54.5	1.42	1.6	2.76	0	2.25
	80	76.1	2.39	2.3	2.78	0	1.82
	320	79.5	1.79	1.95	3.61	0	2.64
	1280	118.0	4.94	3.77	2.66	0	1.27
	L10	126.9	3.1	3.04	6.91	0.013	3.94
	G10	152.8	3.4	5.21	4.57	0.005	2.20
26/05/2000	0	49.6	1.43	1.48	3.11	0.005	2.58
	80	68.2	2.25	2.07	3.10	0.110	2.11
	320	-	-	-	-	-	-
	1280	108.7	4.7	3.29	2.75	0.008	1.38
	L10	73.3	1.84	1.53	2.5	0.225	1.93
	G10	124.0	5.32	3.03	4.04	0.19	1.98
02/10/2000	0	49.4	1.19	1.36	1.93	0.17	1.71
	80	65.6	1.91	2.09	2.10	0.263	1.49
	320	63.2	1.29	1.59	2.99	0.161	2.48
	1280	106.3	4.29	3.47	1.85	0.17	0.94
	L10	64.9	1.02	1.37	3.86	0.18	3.53
	G10	114.8	4.97	4.24	2.8	0.18	1.32

As was found for the Hutton soil at the Brookdale trial, the addition of sludge to the Westleigh soil caused an increase in the EC of the soil solution, but only of real interest at the 1280 Mg ha^{-1} level. At the 1280 Mg ha^{-1} sludge level, the EC was approximately double that of the control treatment. Both gypsum and lime also caused an increase in EC although it can be seen that gypsum was clearly more efficient because of its much higher solubility compared with lime.

The 1280 Mg ha^{-1} treatment also showed much higher levels of calcium than the control, because of the dissolution of the lime ($CaCO_3$ is added during the water treatment process) contained in the sludge. The strong similarity in Ca^{2+} concentrations between the 1280 Mg ha^{-1} sludge treatment and that of the gypsum treatment applied equally to the Mg^{2+} and Na^+ concentrations, which is also much higher than in the control. This increase in Mg^{2+} and Na^+ in the soil solution is attributed to Ca^{2+} replacement for Na^+ and Mg^{2+} (in that order) on the exchange complex. Nevertheless the sodium adsorption ratio (SAR_e) was quite low. Levels of K^+ in the soil solution were also extremely low.

With regard to the value of this measurement technique as a potential option for monitoring of land treatment systems, the consistency of the data for the control treatment is encouraging. Nevertheless because of the much greater levels of variability noted in the soil solution of the amended treatments, it is felt that more reliable information would be obtained from saturated soil pastes, and that soil water collection apparatus should essentially be used to determine the timing of these measurements.

7.5 The influence of "wet" WTS on soil stability

7.5.1 Introduction

In many respects the work reported previously has been concerned largely with the effects of <u>air - dry</u> sludge on soil stability. Although this is a convenient reference state it is also of interest to evaluate the effects of the sludge when it is applied in its "wet state" to soils. In this regard, sludge and soil (four soil types) were incubated at equivalent matric potential for a specified period and the resulting influence on clay dispersion, aggregation and soil strength studied. The discussion of the effects of WTS on soil strength is however presented in Chapter 8, although it should be noted that clay dispersion is an integral factor affecting soil strength.

7.5.2 Material and Methods

That quantity of air-dry soil (<2mm) to yield an equivalent oven dry mass of 0.4 kg (by correcting for its water content) was measured out into PVC buckets for the four soil types. Each soil was then brought to matric potentials of 0, -10 and -500 kPa, by adding the appropriate quantity of water to obtain the desired water content as determined from the retention curve. Fresh sludge with an initial water content of 6.03 g g^{-1} was equilibrated at matric potentials of 0, -10 and -500 kPa using tension tables and pressure plate apparatus. For each matric potential, the quantity of sludge required to obtain an equivalent oven dry application rate of 0, 80, 320 and 1280 Mg ha^{-1} was then calculated and added to the soil of each matric potential. Incorporation of the sludge with the soil was accomplished by churning the mixture with a flat steel bar until it was as homogenous as possible. Due to the crumbly nature of the soil and sludge at -500 kPa, slight kneading of the sample was necessary to obtain an acceptable final condition of the mixture. The buckets were then sealed and shaken end over end for five minutes before being stored in a constant temperature room maintained at 23°C for a period of eight weeks. To prevent the onset of anaerobic conditions during the incubation period, the buckets were removed once a week, shaken for five minutes and the lids left open for an equivalent period before being resealed and returned to the constant temperature facility. During this time, the buckets were also weighed so as to monitor any evaporative losses which were determined to be negligible and so no correction for this influence was made.

At the end of the incubation period a sub-sample of material was extracted from the bucket and pressed into rectangular steel moulds (0.1 m length, 0.05 m breadth, 0.01 m depth) for the measurement of soil strength, in terms of the modulus of rupture (see Section 8.3). The remaining sample was air dried and then crushed to pass a 2 mm sieve, after which the percentage aggregation of the treatments was determined as described in Section 7.3.2. The unbound silt and clay percentage for the control treatment incubated at matric potentials of -100 and -500 kPa was not measured.

In a separate but allied experiment 200 g of each soil (<2 mm , air-dry) was levelled within rectangular plastic tubs of internal dimensions 0.2m x 0.15 m. Each sample was sprayed with the equivalent of 0,10,20,30 and 40 kg ha^{-1} anionic polyacrylamide solution. Since the mass of granular PAM needed was extremely small a stock solution was prepared beforehand. The appropriate volume of stock solution was then mixed with distilled water such that when applied, each treatment was brought to field capacity. The tubs were then sealed and incubated with the remainder of the treatments. After the incubation period the samples were air-dried and gently passed through a 2 mm sieve before the percentage aggregation was measured as described in Section 7.3.2.1

7.5.3 Results and Discussion

For easier comparison between the treatments, the influence of sludge content and incubation matric potential on soil aggregation is presented in terms of the unbound silt and clay percentage (USCP) rather than the percentage aggregation (Equation 7.3). One can be derived from the other since they both should add to 100%. The results of this study for the Hutton, Westleigh, Valsrivier and Cartref soils is presented in Figures 7.9 to 7.12 respectively.

For all soil types it can be seen that both the sludge content and the incubation matric potential had a marked effect on the USCP. Increasing the content of "wet sludge" caused a decrease in the USCP. Nevertheless, because of the high inherent stability of the Hutton soil 1280 Mg ha^{-1} sludge was needed before any change in the USCP was found. Adding 1280 Mg ha^{-1} sludge (under the same conditions) to the Catref soil reduced the USCP below detectable limits (Figure 7.12). Because of their much higher USCP in their natural state, the Westleigh (Figure 7.10) and Valsrivier soils (Figure 7.11) were particularly sensitive to the influence of the sludge. An approximately 6% decrease in the USCP occurred when the Westleigh soil was incubated (at saturation) with 80 Mg ha^{-1} sludge while an equivalent value for the Valsrivier soil was in the order of 8%. Similarly at the 1280 Mg ha^{-1} sludge level (incubated at saturation), the USCP for the Westleigh soil and Valsrivier soils was in the order of 8% and 10% less than measured for the control treatment. In correlating the USCP of the treatments with sludge content (after saturating and drying), a strong negative relationship with r^2 values of 0.808, 0.860, 0.904 and 0.832 for the Westleigh, Valsrivier, Hutton and Cartref soils respectively were obtained.

Decreasing the incubation matric potential also increased the USCP for equivalent sludge content. This relationship was generally more pronounced at the 1280 Mg ha^{-1} level than at the lower sludge contents where greater variability in the findings were obtained. When the sludge is wet, the polymer that it contains remains active and is therefore capable of binding the constituent silt and clay fraction together.

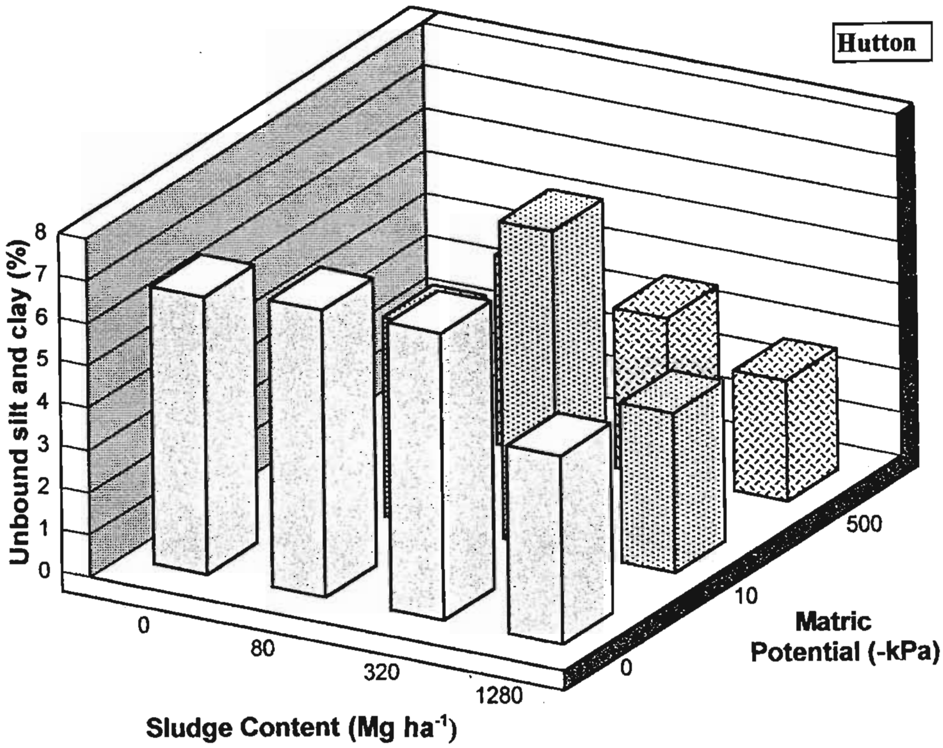

Figure 7.9 Unbound silt and clay percentage of the sludge and soil mixtures for the Hutton soil after being incubated at selected matric potentials for eight weeks.

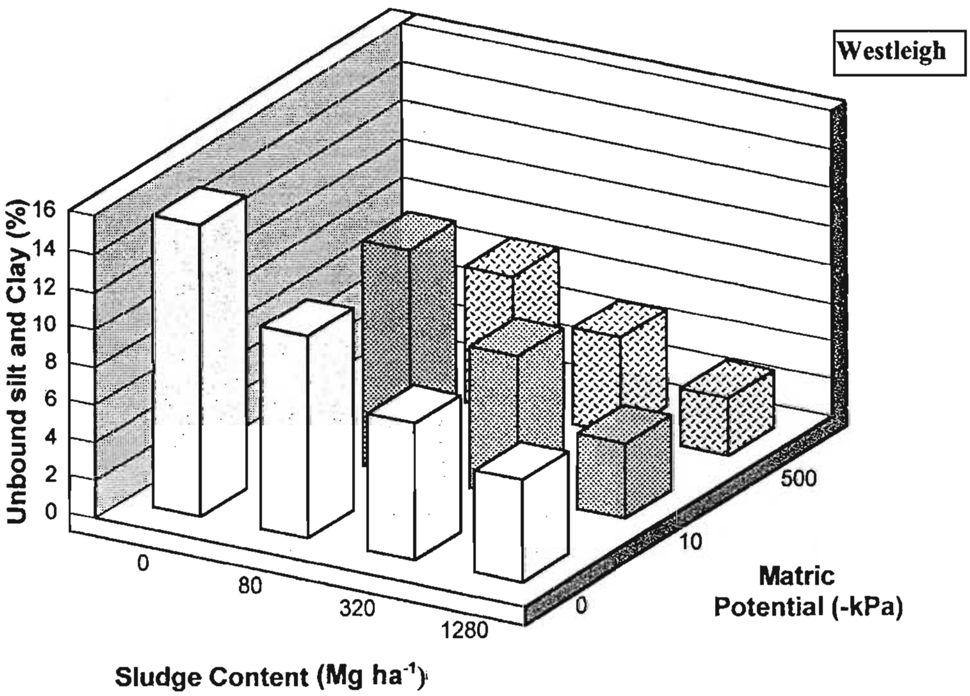

Figure 7.10 Unbound silt and clay percentage of the sludge and soil mixtures for the Westleigh soil after being incubated at selected matric potentials for eight weeks.

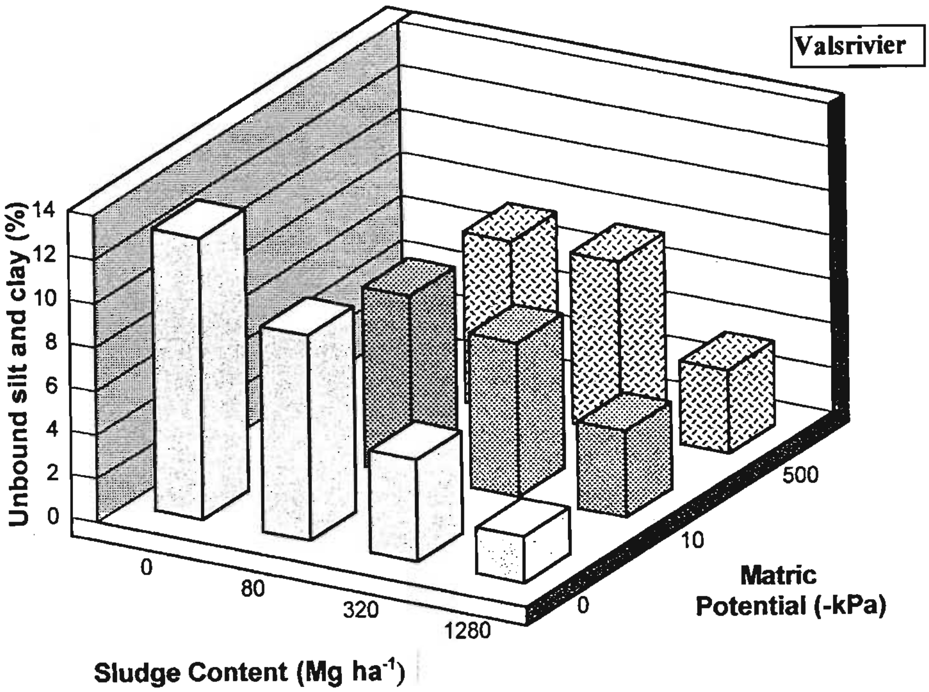

Figure 7.11 Unbound silt and clay percentage of the sludge and soil mixtures for the Valsrivier soil after being incubated at selected matric potentials for eight weeks.

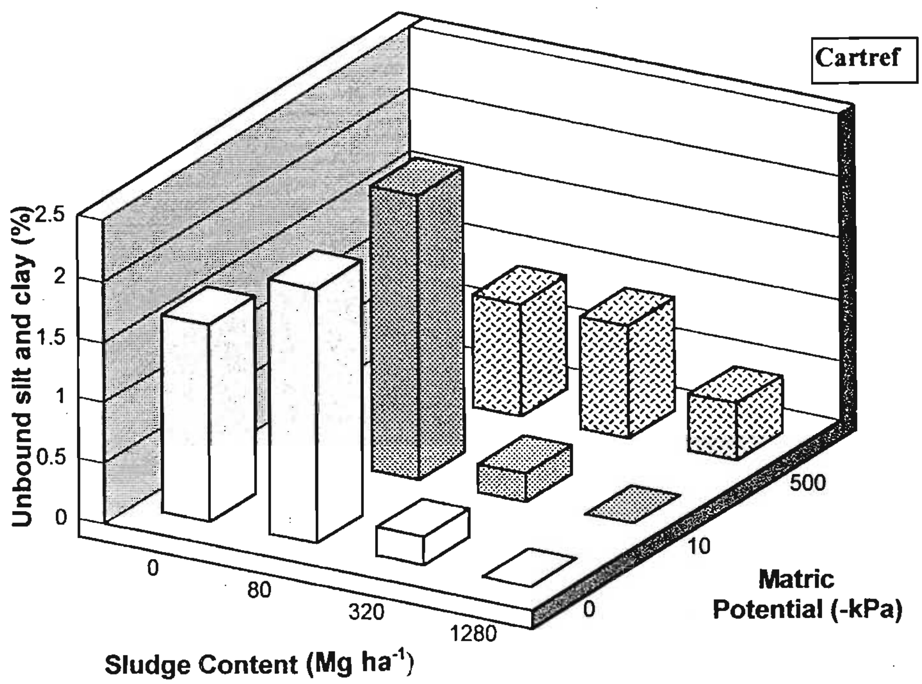

Figure 7.12. Unbound silt and clay percentage of the sludge and soil mixtures for the Cartref soil after being incubated at selected matric potentials for eight weeks.

The potential improvement in soil physical properties brought about by addition of these amendments is to a large extent regulated by the adsorptive characteristics of the polymer onto the soil surface (Levy and Ben-Hur, 1998). According to Theng (1982), polymers entering the soil solution can adopt a range of conformations which, in part, are dependent upon the ambient soil solution composition, the relative solubility and charge of the polymer, accessibility to the clay mineral surface and other steric factors. In the case of anionic high molecular weight polymers as was used to thicken the sludge, the negatively charged polyanion is repelled by the soil mineral surface and consequently only a few segments of the polymer are adsorbed (Ben-Hur and Keren, 1997). The other segments exist as long chains, loops and tails in solution (Figure 7.13). When the sludge is mixed with the soil, these long tails and loops would serve as efficient "grappling hooks" which form inter-particle bridges by mechanically interlinking to each other.

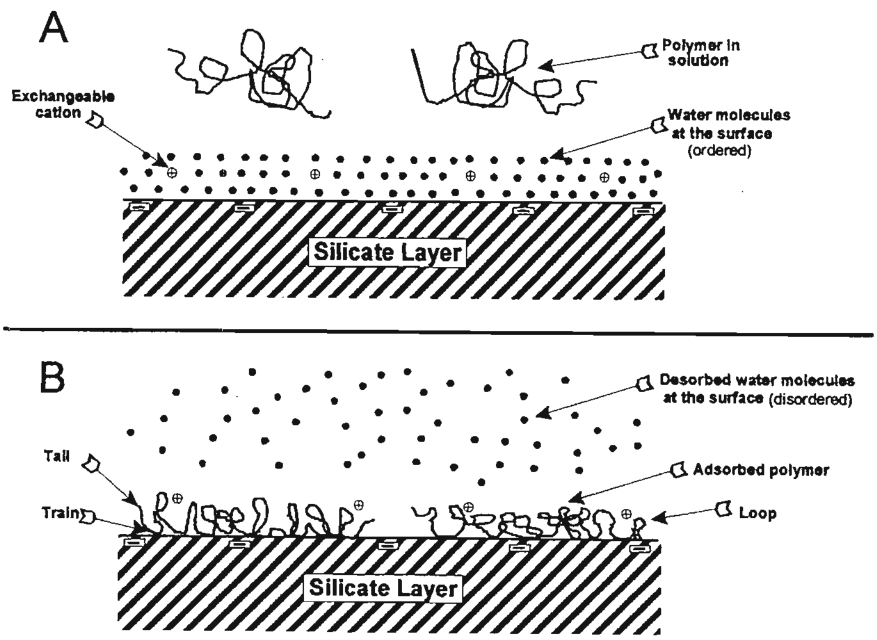

Figure 7.13 Schematic illustration of the mechanism of polymer absorption onto soil surfaces (redrawn after Theng, 1982).

Once the system is allowed to dry, short range van der Waal forces become effective and bonding of the polymer to the soil constituent particles becomes irreversible (Zhang and Miller, 1996). In addition, the probability of all of the adsorbed segments of the polymer being released simultaneously back into solution is negligible which means that once the system is dehydrated desorption of the polymer from the soil mineral surface is virtually zero (Malik and

Letey, 1991; Nadler *et al.*, 1992), leading to the decrease in the USCP. This factor also accounts for the lower USCP measured when the sludge and soil was incubated at saturation than when incubated at -500 kPa. In drying the sludge to -500 kPa matric potential, it is probable that a proportion of the polymer became irreversibly bonded to the sludge. This already bonded fraction was therefore unavailable for further reaction with the soil under incubated conditions. Nadler *et al.*, (1992) further commented that the mobility of polymers through soil is expected to be very low after adsorption and even further restricted if the soil is dried.

The effect of treating the four soils with polymer solutions of varying concentrations on the USCP is shown in Figure 7.14.

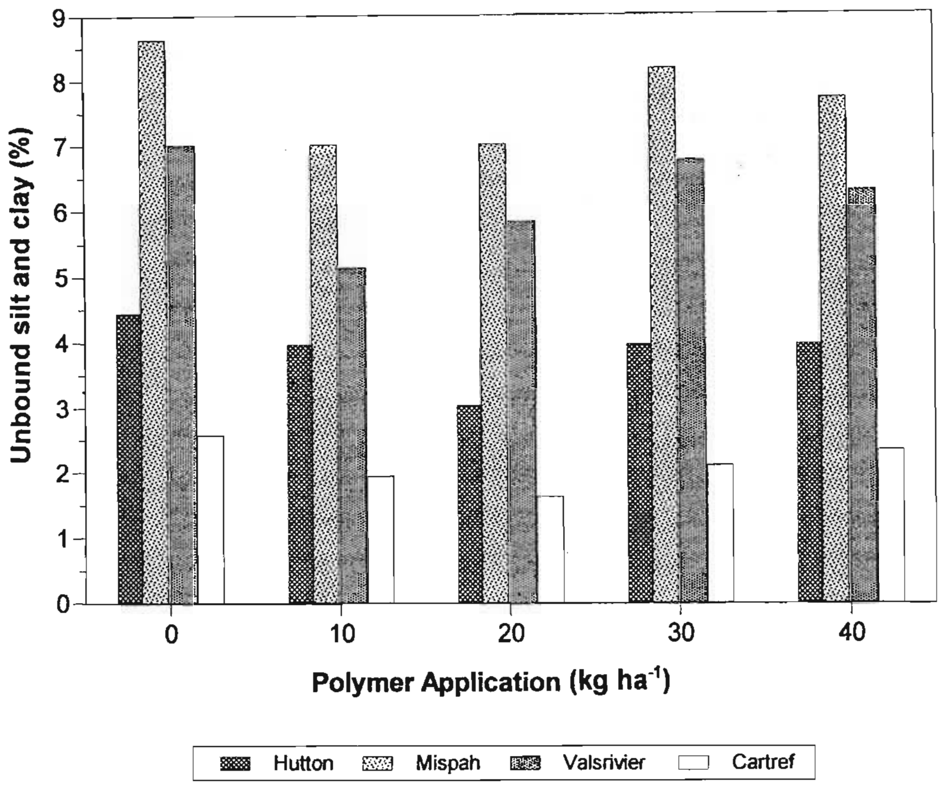

Figure 7.14 The effect on PAM application level on the unbound silt and clay percentage for the four soil types. Each value quoted is the mean of three replicates.

When applied at the 10 kg ha^{-1} and 20 kg ha^{-1} level, anionic PAM caused a progressive decrease in the USCP for all soil types but from an application level of 30 kg ha^{-1} onwards, the USCP increased, but nevertheless remained below that of the control treatment. It is probable that at PAM application levels > 20 kg ha^{-1}, a large proportion of the polymer chains would tend to interlink with one another, so decreasing the effectiveness of the polymer to become absorbed onto the soil surface. In addition, according to Malik and Letey (1991)in the case of strongly hydrolysed polymers (typically > 30 %), the long chain molecules would tend to exist as a tight coil in solution rather than be stretched out, leading to poor attachment to the soil surface. From a practical point of view this suggests that greater advantages are to be had when the polymer is applied in small dosages with a drying period between applications than if such polymers were applied in single larger dosages.

The results for the USCP that was measured on the gypsum and lime treatments were rather disappointing and no consistent relationships were obtained. This probably relates either to the complicating influence that arose from remoulding the soils or the duration of the experiment which may not have been long enough for complete dissolution of the gypsum and lime. It is also possible that the technique employed was not suited to the more subtle effects of clay flocculation.

7.6 General conclusions

Attempts at using the dry aggregate size distribution to evaluate aggregate stability produced highly variable results. It is therefore felt that in the present context, the GMD of the dry aggregate size distribution is a poor monitoring index of soil quality. At the Brookdale trial the MWD of water stable aggregates showed high levels of variability that prevented a confident evaluation of the influence of the sludge on soil aggregation. At the Ukulinga trial the investigation met with greater success. An increase in the MWD for the sludge-amended treatments compared with the control treatment was found. Despite this apparent improvement in aggregate stability as suggested by these findings much of the material retained on the sieves at the end of the test comprised mainly WTS particles. The increase in the MWD for the sludge-amended treatments is linked directly to the inclusion of these particles in the sample, than to their effects in stabilising *existing* soil aggregates.

Leaching of mainly Ca^{2+} and Mg^{2+} from the lime that is added during the water treatment process and therefore concentrated in the sludge, increased the electrolyte concentration of the soil solution at the Brookdale and to a lesser extent at the Ukulinga trials. Since calcium in particular is known to compress the diffuse double layer, increased levels of this cation may favour an improvement in soil stability. Unfortunately, because of the strongly aggregated nature of the Hutton soil, any improvement in soil stability could not be detected by the methods employed. In the Ukulinga trial especially, the replacement of Na^+ by Ca^{2+} was noted when analysing the composition of the soil solution, indicating that cation exchange is indeed an active process on the sludge-amended treatments.

The hydration state of the sludge exerts a marked influence on soil stability. When the sludge is applied to soil in a fully air-dried state the polymer which was used to flocculate the suspended material and to thicken the sludge during the water treatment process becomes strongly bonded to its constituent fraction. It is therefore unlikely to become reactivated into solution. Consequently there is little physical binding action between the sludge and soil particles. When the sludge is applied in its wet form, however, some of the polymer in the sludge solution becomes attached to the soil particles, so improving the aggregate stability of the amended soil.

In general, the results obtained in the present study show that WTS has the potential for improving soil aggregation and soil stability and that such effects may be better achieved if the sludge is incorporated with the soil, before being allowed to fully air-dry, although the practical implications of this mode of sludge disposal warrants further research. The disposal of sludge in its wet state is receiving attention and is discussed in Appendix 9.1.

Chapter 8

The Influence of Water Treatment Sludge on Soil Strength

8.1 Introduction

The mechanical strength of soil is a useful indicator of soil physical condition (Barzegar *et al.*, 1994). Soil mechanical strength provides anchorage for plants and therefore can have both a direct and indirect influence on crop growth (Young and Mullins, 1991, De Freitas *et al.* 1996). Direct influences relate to root growth and proliferation while indirect effects refer to the timing of land management practices such as tillage or cultivation (Jayawardene and Blackwell, 1990). Since water treatment sludges comprise essentially silt and clay sized material, release of this fine fraction due to a structural breakdown of the sludge (more likely to be accelerated in the field) may alter soil strength. Excessive mechanical strength, in particular, is extremely undesirable as it often leads to reduced seedling emergence, poor aeration properties, reduced hydraulic transport and increased risk of sediment scour and entrainment (Mullins *et al.*, 1987, 1990; Proffitt *et al.*, 1995). This factor is of special importance in the successful operation of the land-treatment system, since excessive soil strength development could potentially hamper the successful establishment of crops and therefore compromise the viability of the disposal scheme. To evaluate this, the influence of WTS on soil strength was studied.

The relative magnitude and distribution of strength within the soil are influenced by a range of factors *inter alia* water content, adsorbed cations, cementing agents and dispersible clays. The soil matrix nevertheless consists of a network of failure zones that arise from the presence of micro-cracks, air filled pores and other similar discontinuities (Kay and Dexter, 1992). Consequently, soil strength is a highly dynamic property.

The only published study to have directly investigated the influence of water treatment sludge on soil strength is that of Rengasamy *et al.* (1980). Although that study reported a decrease in soil strength in response to sludge application, one cannot be confident that these findings would apply to the local situation given the different type of sludge being produced by Umgeni Water. Moreover, in the original study the maximum sludge application rate tested was 30 Mg

ha^{-1} as compared with the 1280 Mg ha^{-1} being investigated at present. This reduces confidence in accurately predicting the effect of the sludge on soil strength in the current context. Indirectly, Bugbee and Frink (1985) also alluded to the reduction in soil strength with increased sludge application, although the main objectives in their study related to the potential use of the sludge as alternative potting media.

As noted in Chapter 2, most of the work concerned with the land disposal of water treatment sludge has been conducted as pot experiments. A common practice in these studies has been to sow seeds or seedlings within the amended media and then to monitor various agronomic parameters. It is interesting that none of these studies have encountered severe problems with seedling emergence which suggests, albeit it qualitatively, that the sludge had limited negative influence on soil strength for the duration of the experiments. Unfortunate aspects of these greenhouse studies, however, have been their short duration and controlled environmental conditions. In the field a more dynamic climatic regime operates. Factors such as rainfall, evaporation and biological activity among others can exert meaningful changes on soil structure and therefore soil strength. To test this scenario both field-based and laboratory-based investigations were therefore undertaken.

8.2 The measurement of soil strength in the field

8.2.1 Introduction

The measurement of soil strength in the field has been reviewed by Zimbone *et al.* (1996). Two commonly used instruments are the torvane and the penetrometer. The torvane is specially designed to measure the shear strength of the soil which relates to the frictional resistance that individual soil particles must overcome when they are forced to slide over one another or move off interlocking positions. Penetrometers are instruments consisting of a conical probe (mounted on a shaft) which is driven into the soil usually at a constant rate of entry. The penetrometer resistance is the force on the penetrometer per unit basal area of the cone (Jayawardene and Blackwell, 1990). The greater the force encountered by the probe, the larger is the penetrometer soil strength (PSS). Compared with the torvane, the penetrometer has been the preferred choice of instruments in a wide array of studies, since the former instrument is limited largely to soil surface strength estimates. While gauging soil strength at depth with the torvane is possible, the procedure is both costly and time consuming which makes it particularly difficult for application over a large area.

The applicability of the penetrometer to studies of soil strength has recently been discussed by Becher *et al.* (1997) and the principles of their operation and design by *inter alia*. Mulqueen *et al.* (1977), Bradford (1986) and Campbell and Hunter (1986). A major advantage of penetrometers is that, in many respects, closer similarity to the resistance encountered by plant roots is possible with this instrument than with the torvane. Bengough and Mullins (1991) obtained excellent correlations between root growth and PSS. In a later study Bengough *et al.*, (1997), nevertheless, commented that the frictional resistance encountered by a probe is between two to eight times greater than that encountered by roots. Moreover, plant roots are capable of deformation and will therefore favour natural failure zones or macro-pores within the soil.

Soil strength is influenced strongly by water content. As a result, interpretation of data obtained with penetrometers and torvanes are ideally interpreted with a corresponding estimate of soil water content. An unfortunate aspect, however, as explained by Jayawardene and Blackwell (1990) are that non-destructive estimates of water content such as neutron methods only provide indicators of soil water content across the total soil volume. PSS, however, is influenced by the exact location of the probe in relation to the macro-pores and the associated wetting patterns. In this regard, PSS should therefore be regarded as point measurements rather than as a bulk soil measurement (O' Sullivan *et al.*,1987). Despite the previous comment, these instruments offer useful relative information on soil strength.

8.2.2 Materials and Methods

To avoid the complications associated with the effect of soil water content, soil strength measurements were made at or near field capacity, which from the earlier discussion (section 4.1.4) is assumed to exist approximately 48 hours following a rainfall event. The instrument used throughout the investigation was of the constant recording type manufactured and marketed by Geotron Systems, South Africa (Figure 8.1). The penetrometer is operated by driving a stainless steel cone (apex angle of 30°and a basal area of 130 mm^2), mounted at the end of a 0.8 m steel shaft (10 mm diameter) vertically into the soil. Insertion of the probe is eased by a chain-driven gear mechanism (winding ratio 4.8:1) operated by turning two handles at the upper end of the instrument. A footplate ensures stability of the unit during the procedure. The rate of penetration of the probe into the soil is 1 m min^{-1} at 1 second per revolution.

A pressure loadcell (type SUB-G-200) attached to the shaft measures the resistance encountered by the probe in kilopascals at 0.01 m depth intervals. The maximum pressure and depth that can be measured are 5 MPa and 0.8 m, respectively. All readings are stored electronically in the memory of the controller unit. In addition, a convenient feature is a digital readout that allows inspection of the results while operating the unit. Usually it took approximately five minutes for an operator to obtain a single penetrometer profile. Customised software is included for the direct downloading of the data from the unit to a personal computer, following which the data can be interrogated using the software supplied or exported to other computer packages.

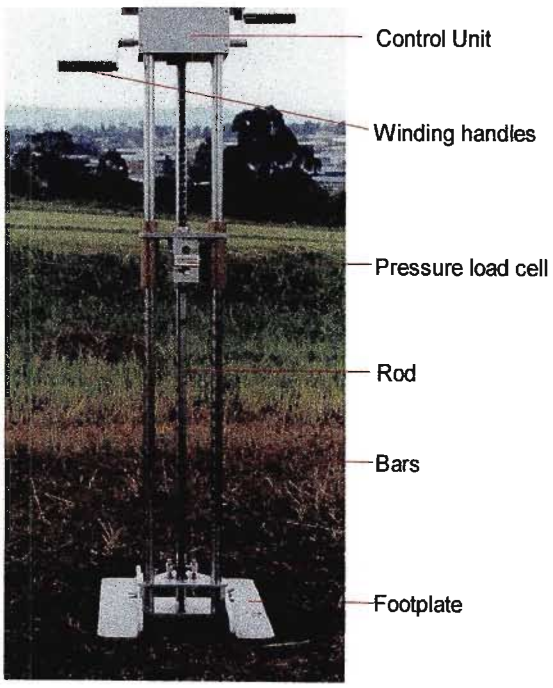

Figure 8.1 The essential components of the constant recording penetrometer of Geotron Systems used in the study.

Six PSS measurements were taken at randomly selected positions within each plot and later combined according to treatment differences. This resulted in twelve penetrometer profiles per treatment. The mulched treatments and the 40,160 and 640 Mg ha^{-1} sludge-incorporated treatments were, however, excluded from the assessment. Due to poor rainfall in the winter months it was not possible to obtain accurate estimates of soil strength. Consequently much of the investigation was undertaken from early spring to late Autumn.

8.2.3 Results and Discussion

8.2.3.1 The effect of WTS on PSS at the Brookdale Trial

Figures 8.2 to 8.5 show that within the depth of tillage (0 - 0.15m) the penetrometer profiles for the sludge-amended treatments are similar. An analysis of variance which compared the effect of sludge application level at selected soil depths on PSS was carried out on the data, the results of which are presented in Table 8.1. Since it was not certain that field capacity existed at the time of the October 2000 measurements were, however, excluded from the analysis.

Table 8.1 Summary of F-statistic significance levels for analysis of variance of the effect of WTS application rates (0, 80, 320 and 1280 Mg ha^{-1}) on PSS at the Brookdale trial.

Sampling Session	Soil Depth (m)							
	0.05	0.10	0.15	0.2	0.25	0.3	0.35	0.40
November 1999	NS	NS	NS	**	*	NS	NS	*
February 2000	NS	NS	NS	NS	NS	*	**	**
May 2000	NS	*	**	*	NS	**	**	NS

*, ** significant at $p < 0.05$ and $p < 0.01$ levels respectively. NS = not significant at $p < 0.05$

Although there are a few anomalies in the general trend, the results of Table 8.1 suggests that within the tilled layer (0 - 0.20 m), WTS had little effect on the PSS of the Hutton soil since non-significant differences in PSS were generally found. From the soil surface to a depth typically around 0.05 m, the PSS increased sharply as greater displacement of the surrounding soil by the conically shaped probe probably occurred. Below this depth and to around 0.15 m the PSS within each treatment was nearly constant. Except during November 1999, PSS values ranged between 1.0 and 1.5 MPa no matter the treatment applied.

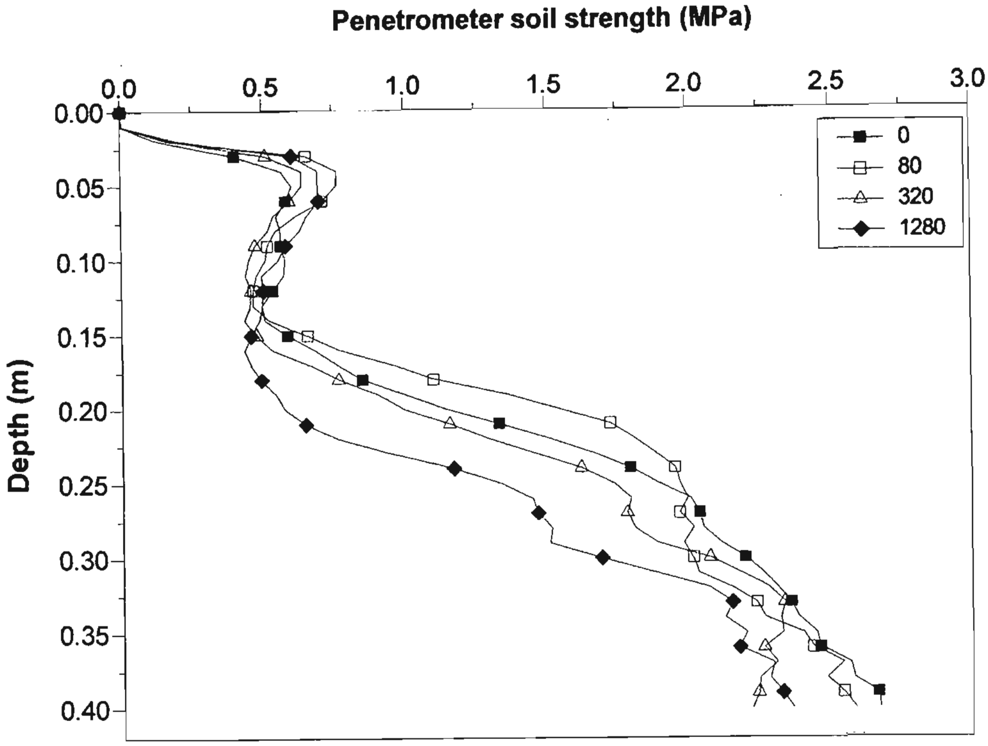

Figure 8.2 PSS profiles for the sludge-amended treatments measured at the Brookdale trial in November 1999. Each data point is the mean of 12 replicates. 0, 80, 320 and 1280 refer to sludge application rates (Mg ha^{-1}).

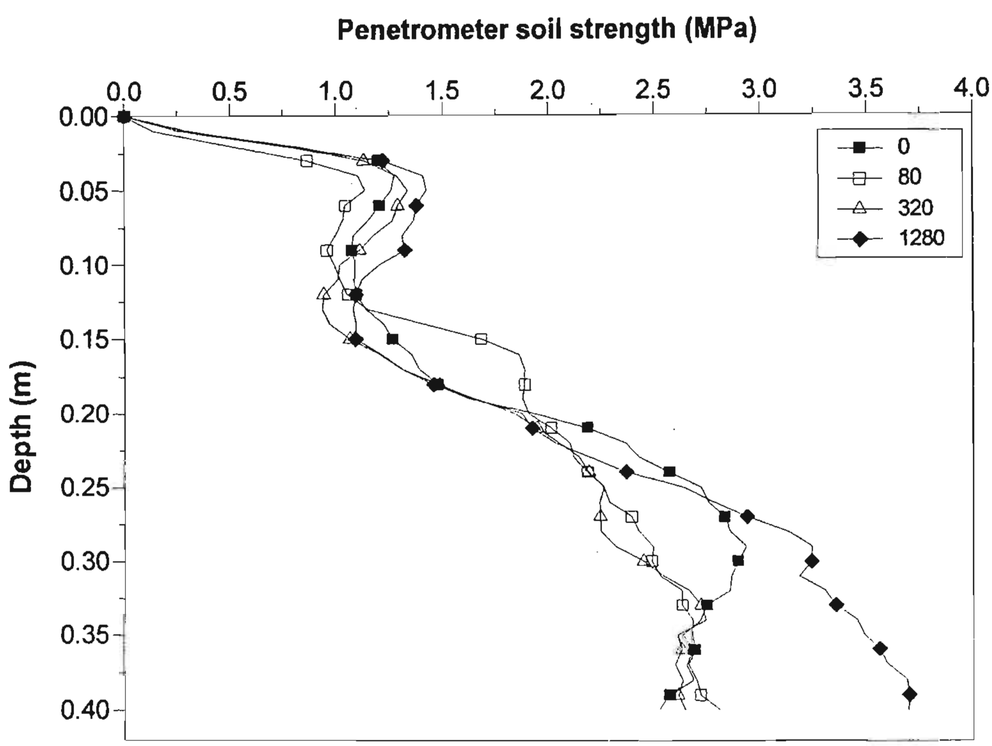

Figure 8.3 PSS profiles for the sludge-amended treatments measured at the Brookdale trial in February 2000. Each data point is the mean of 12 replicates. 0, 80, 320 and 1280 refer to sludge application rates (Mg ha^{-1}).

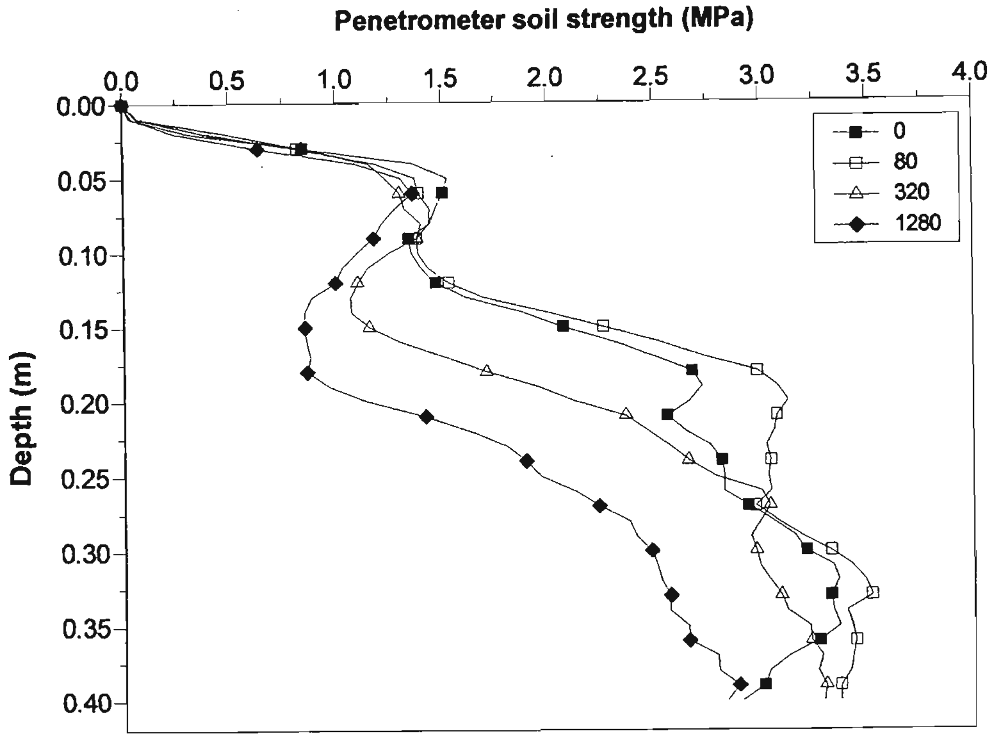

Figure 8.4 PSS profiles for the sludge amended treatments measured at the Brookdale trial in May 2000. Each data point is the mean of 12 replicates. 0, 80, 320 and 1280 refer to sludge application rates (Mg ha^{-1}).

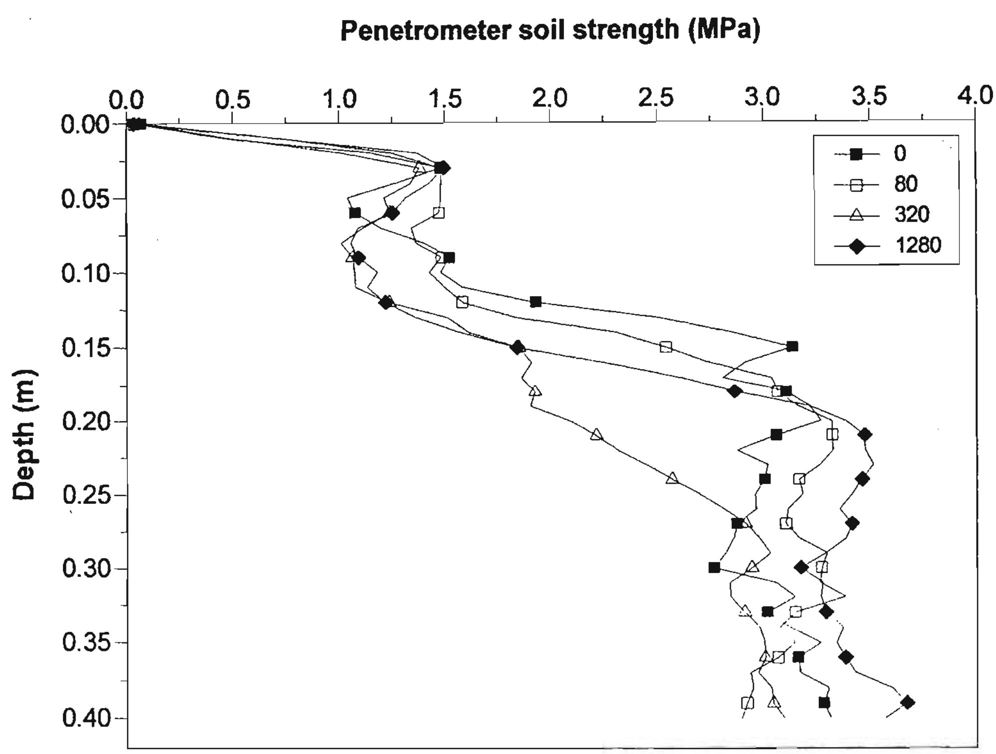

Figure 8.5 PSS profiles for the sludge-amended treatments measured at the Brookdale trial in October 2000. Each data point is the mean of 12 replicates 0, 80, 320 and 1280 refer to sludge application rates (Mg ha^{-1}).

From a soil depth of 0.15 m downwards (below the tilled layer) as would be expected, the PSS increased sharply but remained below 3.5 MPa which suggests a moderately compacted subsoil condition. Similar trends to that reported here, where PSS has increased with depth have previously been reported by 'O Sullivan *et al.* (1987). Nevertheless, statistically significant differences in PSS between the treatments imposed, were found. For equivalent soil depths the order of PSS found between the treatments were 0 Mg ha^{-1} >80 Mg ha^{-1} > 320 Mg ha^{-1} > 1280 Mg ha^{-1} (Figures 8.2 to 8.5). A potential reason for this is obtained when considering the water content of the treatments as measured in May 2000 (Table 8.2), which generally shows an increase in water content (especially below the tillage depth) with an increase in sludge content.

Table 8.2 Mass water content (kg kg^{-1}) for the sludge-amended treatments measured during May 2000. The mean of three replicates is quoted.

Soil Depth (m)	**Water Treatment Sludge Application Rate (Mg ha^{-1})**			
	0	80	320	1280
0 - 0.1	0.224	0.225	0.237	0.218
0.1 - 0.2	0.217	0.216	0.221	0.225
0.2 - 0.3	0.216	0.219	0.230	0.236
0.3 - 0.4	0.229	0.227	0.242	0.262

From the discussion presented in Sections 4.3.1.2, and 6.3.1 it was noted that WTS increased the macroporosity of the soil. In addition following from Chapter Five, it was also found that the saturated hydraulic conductivity increased and the unsaturated hydraulic conductivity decreased with increased sludge content. On sludge-amended soil these conditions would probably favour the rapid movement of water through the depth of tillage to the soil beneath. In addition because of its lower unsaturated hydraulic conductivity the loss of water by evaporation would be lower (Section 5.6.3). These factors would potentially cause the subsoil on sludge-amended soils to remain wetter for longer, as was indeed found from the results presented in Table 8.2. Consequently the PSS of the subsoil on the WTS-amended Hutton soil was lower than on the control treatment.

8.2.3.2 The effect of gypsum, lime and polyacrylamide on penetrometer soil strength at the Brookdale trial

The general trend in PSS for the gypsum, lime and polyacrylamide treatments was very similar for the November 1999, February 2000, May 2000 and October 2000 investigations. For brevity only the November 1999 data is presented here (Figure 8.6) while the data for the other sampling periods is given in Appendix 8.1.

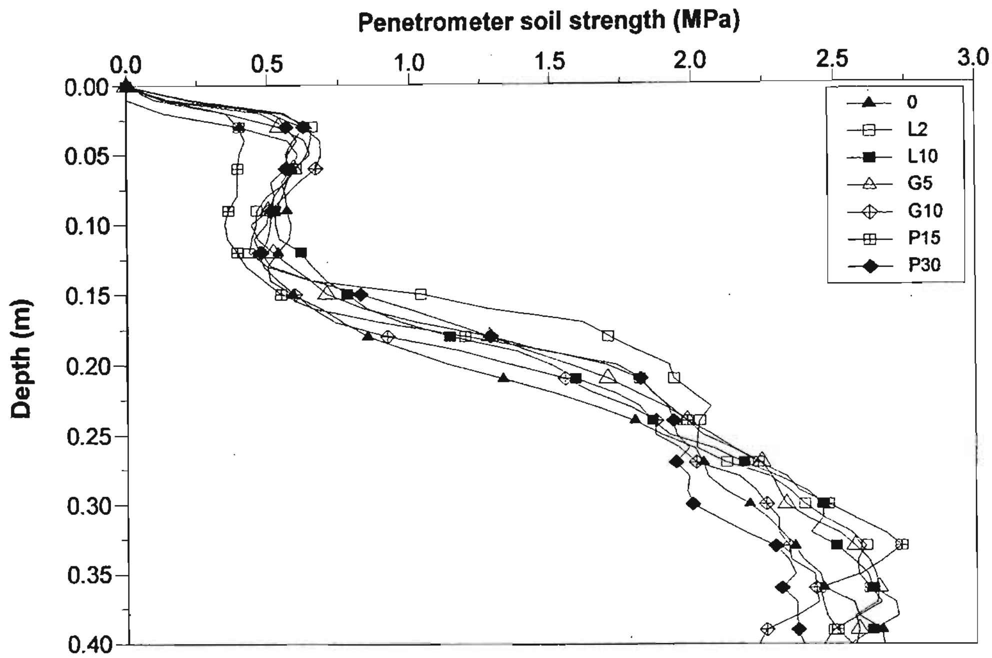

Figure 8.6 PSS profiles for the gypsum, lime and polyacrylamide treatments investigated at the Brookdales field trial during November 1999. Each data point is the mean of 12 replicates. L10,G10 and P30 are treatments of lime at 10 Mg ha^{-1}, gypsum at 10 Mg ha^{-1} and polyacrylamide at 30 kg ha^{-1} respectively.

As with the sludge-amended treatments, the PSS for the gypsum, lime and polyacrylamide treatments increased sharply to a depth of approximately 0.05 m and then remained nearly constant throughout the remaining depth of the tilled soil. At depths below 0.15 m the PSS for all of the treatments increased but (as for the sludge-amended treatments) remained below 3.5 MPa. Whereas between the sludge-amended treatments, a strong divergence in the penetrometer profiles from a depth just below the tilled zone occurred, the gypsum, lime and polyacrylamide treatments all show strong similarity in their respective patterns. Regarding the level at which the sludge was applied on soil strength, no clearly defined trends either within or between the sampling sessions was found.

8.2.3.2 The effect of WTS, lime, gypsum and PAM on penetrometer soil strength at the Ukulinga Trial

PSS profiles for all the treatments investigated at the Ukulinga field trial during April 2000 and October 2000 are shown in Figure 8.7 and 8.8 respectively. The general trend in PSS with depth shows very strong similarity to the results obtained at the Brookdale field trial, despite the difference in soil type and climate.

As in the Brookdale field trial there was a rapid increase in soil strength which peaked at approximately 1.25 MPa and then remained fairly constant around 1.1 MPa throughout the remaining depth of tillage. From a soil depth of 0.2 m downwards, there is a strong divergence between the treatments accompanied by a rapid increase in PSS. The maximum PSS across all of the treatments nevertheless remained below the 3.5 MPa level, thus indicating a moderately compact subsoil condition at the Ukulinga trial. It is interesting that the PSS for the gypsum treatment shows good agreement with that of the control. For the lime- amended treatment, however, a much stronger increase in PSS below the depth of tillage was measured. Notably, however, the 1280 Mg ha^{-1} treatment consistently shows a much lower PSS at depth, which is again probably attributed to the higher water content of this treatment.

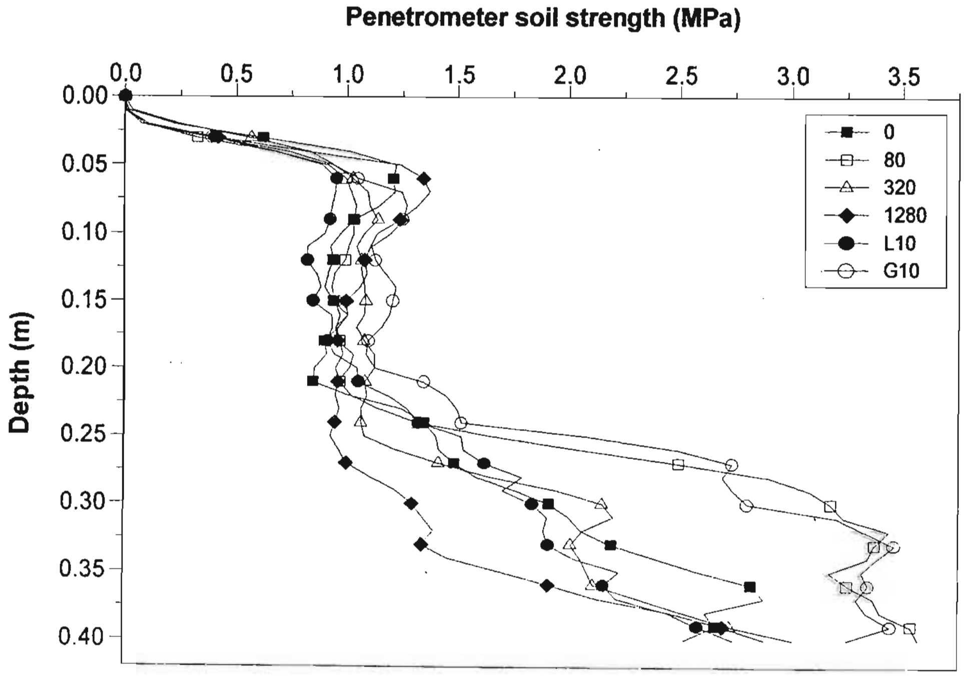

Figure 8.7 PSS profiles for the treatments investigated at the Ukulinga trial during April 2000. Each data point is the mean of 12 replicates. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L10 and G10 are treatments of lime at 10 Mg ha^{-1} and gypsum at 10 Mg ha^{-1} respectively.

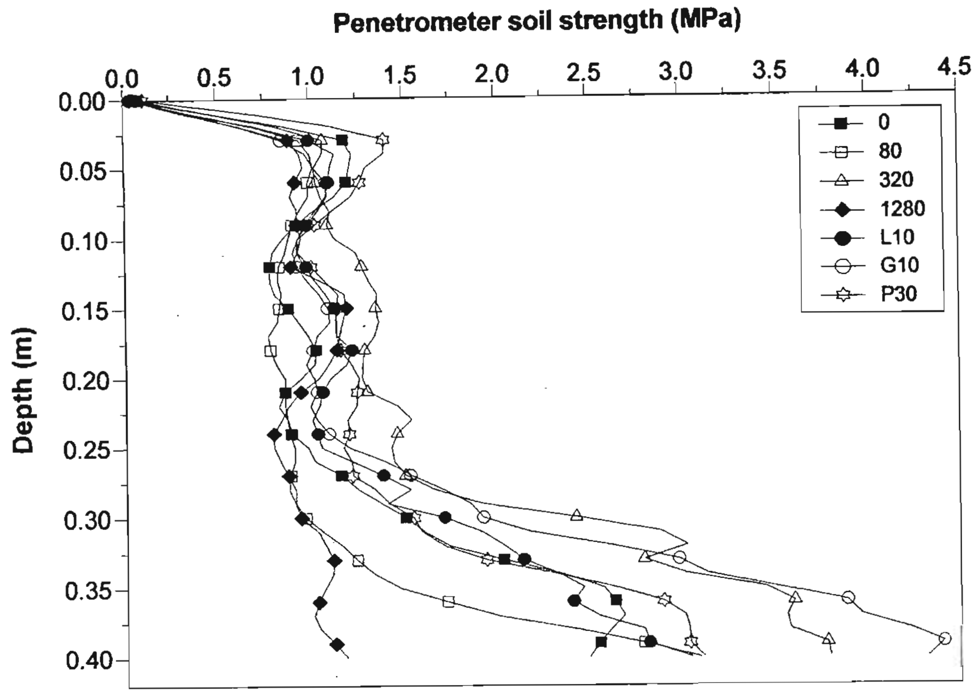

Figure 8.8 PSS profiles for the treatments investigated at the Ukulinga trial during October 2000. Each data point is the mean of 12 replicates. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L10, G10 and P30 are treatments of lime at 0 Mg ha^{-1}, gypsum at 10 Mg ha^{-1} and polyacrylamide at 30 kg ha^{-1} respectively.

Neither lime nor PAM had any meaningful influence on PSS as shown by the similarity of these curves with that of the control treatment. It is interesting that in both April and October 2000 the gypsum treatment showed a relatively higher PSS than the control treatment but that this only occurred from a depth of 0.25 m which is below the tilled soil. Within the tilled soil, however, PSS values for the G10 treatment are similar to that of the control. This implies that the differences occurring below the tilled soil, may be due more to natural subsurface variability between the plots than to the treatments that they received.

8.3 The influence of wet sludge on Modulus of Rupture

8.3.1 Introduction

The confounding influence of water content on soil strength encountered during the field studies required that an extension of the study under more controlled environmental conditions be carried out. In this regard the influence of WTS on soil strength was further evaluated using a laboratory-based technique, to clarify the previous results. Moreover, this allowed the expansion of the study to include the additional two soil types, namely the Cartref and the Valsrivier soils.

8.3.2 Materials and Methods

Details of the design and execution of the incubation experiment upon which the results and discussion that follow are based, were presented in Section 7.4 wherein the influence of wet sludge on soil aggregation was considered.

At the end of the incubation period a sub-sample of material was extracted from the bucket and pressed into rectangular steel moulds (0.0715 m length, 0.0345 m breadth, 0.01 m depth). The remaining sample was air dried and then crushed to pass a 2 mm sieve. The less than 2 mm fraction was poured gently into an equivalent set of moulds, levelled and saturated overnight following which the excess water was drained. The entire set of briquettes were then air dried for five days at room temperature and then further dried in an oven set at 50°C for an additional period of 24 h. This initial period of air drying was necessary as absence of this pre-treatment resulted in severe distortion of the briquettes and unacceptably high levels of cracking. Once dry the dimensions of the briquettes were noted. Where briquettes were formed, these were broken on a beam balance using a briquet breaking apparatus and closely following the method of Richards (1953), as outlined by Klute (1986) for measuring the Modulus of Rupture (MOR). It should, however, be noted that a strict interpretation of the MOR refers essentially to air-dried material passed through a 2mm sieve. Owing to the limited quantity of samples, only three replicates per treatment were possible.

The MOR was calculated from the following equation:

$$\mathbf{MOR = 3FL/2bd^2} \quad ....\text{Eq. 8.1}$$

where MOR is the Modulus of Rupture (Pa), F is the breaking force applied at the centre of the beam span (N), L is the length between the briquet end supports (m), b is the width of the briquette (m) and d is depth or thickness of the briquette (m).

8.3.3 Results and Discussion

Measurements of MOR for the various treatments are given in Table 8.3. Attempts at forming briquettes from the 'wet material" incubated at saturation and at field capacity generally met with good success but similar attempts using material incubated at a matric potential of -500 kPa were, however, less successful. A plausible explanation for this observation was offered by Gusli *et al.* (1994) and Becher *et al.* (1997). These authors commented that where the water content is less than a critical index, (which they termed WC_{max}) the forces generated by the water menisci are not sufficient to cohere the total soil mass, but only single grains or small aggregates. Below this critical water content the soil tended to break or crumble, as was apparent in the preparation of the briquettes at an incubation matric potential of -500 kPa.

It is further interesting that even without sludge addition, simply remoulding the soils resulted in the formation of briquettes with extremely high soil strengths. Remoulding the Hutton soil, for example, at -10 kPa matric potential increased the MOR from 10.1 kPa, to 1180 kPa. Similar responses to "remoulding" were observed for the other soil types investigated in this study. The effect of remoulding soils on strength development has been widely investigated in the literature. Several of these studies have shown that with remoulding or reworking of the soil, for example by tillage, clay dispersion increases with a concomitant increase in soil strength (Kay and Dexter, 1992; Barzegar *et al.*, 1994; Harper and Gilkes, 1994; Taboada and Lavado, 1996; Mullins, 1997; Watts and Dexter, 1997).

The main mechanisms proposed to account for the increase in soil strength following remoulding is that the clay particles when mobilised by dispersion and slaking become redistributed upon drying due to the forces exerted by the menisci (Mullins *et al.*, 1987; Kay and Dexter, 1992; Gusli *et al.*,1994; Becher *et al.* 1997). As the soil dries the effective stress increases and precipitation of soluble salts occur at zones of contact between aggregates or particles (Mullins 2000). The silt and clay once rearranged around the sand grains can lead to the formation of bridges that then interlink individual larger sized particles together (Mullins *et al.*, 1987; Harper and Gilkes, 1994). This closer packing creates more contact points between individual particles in the dry state (Barzegar *et al.*, 1994) and increases the effective soil strength. The nett result is a much more compact soil matrix (Aylmore and Sills, 1982).

Table 8.3 Modulus of Rupture (kPa) values for the treatments investigated in the incubation experiment.

Matric Potential (-kPa)		WTS Application Rate ($Mg\ ha^{-1}$)				Lime and Gypsum Application Rate ($Mg\ ha^{-1}$)			
		0	80	320	1280	L_2	L_{10}	G_5	G_{10}
Hutton	0	n.d.	3279 (56)	3133 (86)	5194 (137)	n.d.	n.d.	n.d.	n.d.
	10	1180 (474)	3498 (18)	3776 (371)	2477 (149)	2812 (59)	2582 (103)	3307 (261)	2756 (50)
	500	n.d.	0	0	0	n.d.	n.d.	n.d.	n.d.
	Air dry	10.1 (1.6)	8.9 (0.7)	0	0	n.d.	n.d.	n.d.	n.d.
Westleigh	0	n.d.	2622 (87)	3434 (241)	4642 (395)	n.d.	n.d.	n.d.	n.d.
	10	2677 (87)	3540 (191)	2278 (108)	3035 (86)	3975 (154)	3793 (75)	4808 (80)	3259 (76)
	500	n.d.	0	0	0	n.d.	n.d.	n.d.	n.d.
	Air dry	30.0 (2.5)	26.1 (4.8)	18.4 (1.5)	0	n.d.	n.d.	n.d.	n.d.
Cartref	0	n.d.	2041 (25)	1423 (56)	2445 (45)	n.d.	n.d.	n.d.	n.d.
	10	1574 (92)	2928 (153)	1911 (24)	1657 (196)	2797 (169)	3788 (58)	3625 (110)	2844 (96)
	500	n.d.	0	0	0	n.d.	n.d.	n.d.	n.d.
	Air dry	18.6 (2.5)	0	0	0	n.d.	n.d.	n.d.	n.d.
Valsrivier	0	n.d.	4052 (50)	2660 (66)	5249 (417)	n.d.	n.d.	n.d.	n.d.
	10	3358 (95)	3219 (58)	2984 (66)	1780 (81)	4491 (57)	4061 (157)	3613 (280)	3888 (185)
	500	n.d.	2243 (125)	0	0	n.d.	n.d.	n.d.	n.d.
	Air dry	368.8 (40.2)	283.9 (14.6)	153.2 (13.1)	74.7 (19.6)	n.d.	n.d.	n.d.	n.d.

n.d = not determined. For cases where briquettes could not be formed, MOR is indicated as 0. The standard error of the mean is shown in brackets.

The specific contribution of the sludge to further soil strength development is shown especially clearly for the treatments incubated at saturation. Except for the Cartref soil, the addition of 1280 $Mg\ ha^{-1}$ sludge resulted in extremely high MOR values of approximately 5 MPa. Equivalent values of soil strength for the incubation undertaken at -10 kPa, are, however, smaller.

In Section 7.4.3 it was noted that many long chained polymers contained in the sludge remain"active" in a wet state and only become irreversibly bound to the soil substrate when air dried (Zhang and Miller, 1996). It is therefore likely that these long chained polymeric compounds with their free "loops" and "tails" also become attached to basal surfaces of the soil aggregates, leading to the increased soil strength when dry. This would also explain the strong differences encountered in soil strength as a function of water content. At saturation there is a stronger likelihood that more of these polymer chains are available for bonding with the substrate than would have been the case at lower water contents.

In sharp contrast to the previous findings, MOR measurements for the <2 mm, air dried treatments shows an inverse relationship between soil strength and sludge addition. For the Cartref soil, briquettes could not be formed for any of the WTS-amended soils, whereas for the Hutton soil only the 0 and 80 Mg ha^{-1} treatments formed briquettes. Greater success in the formation of briquettes was achieved with the Westleigh and Valsrivier soil samples, which have higher soil strength values in their natural state. These soils in particular warrant further comment, as near linear negative relationships between the MOR and sludge application rate was obtained. Correlating the two parameters for the Westleigh and Valsrivier soils showed strong negative relationships with r^2 values of 0.995 and 0.863 respectively. In addition, when correlating the MOR with the dispersible clay percentage for the Westleigh and Cartref soils r^2 values of 0.948 and 0.974, respectively were obtained.

In a wet state (high matric potential) the polymer in the sludge when added to soil acts as a binding agent, which, on drying, results in high soil strength across the entire soil matrix. In crushing these soils, the dispersible clay percentage which is fundamental for cohesion of the soil matrix is radically reduced and consequently leads to a low MOR. In addition because of the tenacity with which the polymer is adsorbed onto the soil mineral surface, even further saturation of the soil did not affect its reactivation.

Regarding the comparative effectiveness of gypsum and lime on soil strength, the results show a high degree of variability. Nevertheless, there are several features of note. For briquettes dried from a matric potential of -10 kPa the higher gypsum application rate (10 Mg ha^{-1}) consistently showed much lower MOR values that the lower rate of gypsum application (5 Mg ha^{-1}) and this applied across all of the soil types. Nevertheless when compared with the equivalent remoulded control sample the final strength of these soils was much higher. As discussed previously, gypsum contributes a ready source of calcium ions to the soil solution (see Section 7.4.3.2). During reworking of the soil more negative charges on the clay mineral surface are exposed and on air drying, bonding is enhanced via exchangeable calcium ions and van der Waals forces (Kay and Dexter, 1992). Similar reasoning applies to the lime-amended treatments, although bonding by exchangeable calcium ions was probably less because of the lower solubility of lime. This hypothesis is partially supported by results for the Hutton and Westleigh soils and to a lesser extent by the Cartref soil which on average showed lower MOR values compared with that of the gypsum treatments.

8.4 Mechanisms of WTS breakdown

The incubation study has shown that the main reason for the increase in soil strength when the sludge is applied wet, is the effect of the polymer in binding the sludge to the soil (discussed in section 8.3). This factor also accounts for the inherently high strength and rigidity of the sludge when fully air-dried. Nevertheless, in Chapter 4 it was also noted that aggregates of sludge, although rigid when dry are also highly porous, consisting of a network of micro-sized pores and channels (see Figure 4.20 and 8.9). These micro-pores and channels represent potential failure zones, but appear to become activated only under conditions of rapid wetting and drying.

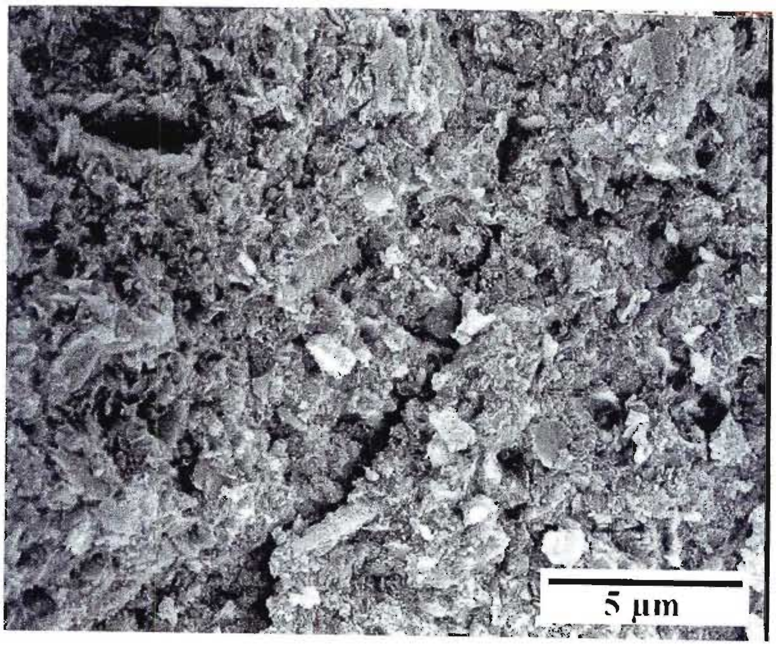

Figure 8.9 Scanning electron micrograph of a sludge aggregate showing flaws such as cracks and micro-pores which represent potential failure zones.

Because of the rigidity with which the polymer binds the constituent silt and clay fraction together, aggregates of dry sludge have little potential for swelling or plastic deformation. When subjected to rapid wetting such as flooding or immersion, because of its limited capacity for plastic deformation, shearing rather than swelling of the aggregates occurs (Braunack *et al.*, 1979). In addition, on drying, effective water stresses will tend to pull particles into a denser configuration, further increasing the potential for brittle failure (Semmel, *et al.* 1990).

Figure 8.10 shows the near surface zone of the 1280 Mg ha^{-1} mulch plot at the Brookdale trial. At the plot surface, aggregates of sludge are of a much smaller and more uniform grade compared with the much coarser material that was found within even a few centimetres depth.

Figure 8.10 The breakdown of WTS aggregates showing the much finer grade of material at the plot surface. Below a few centimetres the material is much coarser in grade.

Apart from rapid wetting, an additional mechanism accounting for the breakdown of the sludge is the impact of rainfall, that accentuates the potential failure zones and lead to further mechanical breakdown. In addition, more rapid and extreme fluctuations in temperature occurring at the soil surface would also favour the accelerated breakdown of the surficial material. Below the soil surface, the sludge is essentially protected from these influences and the rate of breakdown is therefore much less. This observation is significant in that once WTS is applied to soil, the stability of the aggregates remain high, which is probably related to the sustained performance of the polymer that was added during the water treatment process.

8.5 General Conclusions

Measurements of penetrometer soil strength (PSS) in the field, showed that in general the land disposal of WTS, at an air-dry water content, had a statistically non-significant influence on PSS for both the Hutton and Westleigh soils. Although aggregates of sludge have inherently high strength, this did not compromise the bulk strength of the soil matrix. On the sludge treatments, however, the PSS below the plough layer was generally lower than was found for the control treatment. Since the sludge treatments have a higher final infiltration rate and a lower unsaturated hydraulic conductivity, the soil below the sludge-amended zone was wetter than the control treatment. Consequently, the PSS was lower.

To evaluate the influence on soil strength when wet sludge is mixed with soil, an incubation experiment was designed (discussed in Section 8.3). The results showed that when wet sludge was applied to soil and the mixtures then air dried, soil strength increased substantially. This effect was more pronounced at higher matric potential than for drier conditions since more of the polymer was available for bonding under wetter conditions. When the mixtures were crushed and the strength subsequently evaluated, the reverse trend was noted. Because of the substantial decrease in the dispersible clay percentage and the extremely poor reactivation of the polymer when re-wet, binding of individual particles to form larger aggregates was substantially reduced. This caused a decrease in the soil strength.

When fully dry, aggregates of sludge are rigid, but the solid matrix nevertheless contains many micro-pores and fractures. Under rapid wetting, minute swelling of the sludge aggregate due to cation hydration leads to the brittle failure of these aggregates. There was evidence of this mechanism in the field as only the surface or near surface zone of the mulch layer (at the Brookdale trial) showed significant aggregate breakdown. Below the surface the sludge aggregates showed much higher levels of stability mainly because of shielding from raindrop impact and less rapid wetting and drying cycles and probably more moderate temperature fluctuations.

Chapter 9

General Summary and Conclusions

For land disposal systems to meet the main objective of assimilating the waste that it receives, it is important that the effects of such waste on soil quality is understood, preferably even before the disposal practice is commissioned. In recognition of this, a local water treatment utility expressed interest in land disposal as a potential alternative to landfill for the sludge produced at their "drinking" water treatment plant. An extensive search for guidelines against which to regulate the practice met with limited success and consequently emphasised a strong need for research to be conducted on the subject. Although a few studies have investigated the influence of water treatment sludge on soil chemistry and fertility, very little is known on the soil physical aspects of the land disposal of WTS, despite this need being strongly emphasised in early works on the topic (Rengasamy *et al.*, 1980; Skene *et al.*, 1995). Moreover, much of the research undertaken to date has been conducted as glasshouse or laboratory-based studies and usually of rather short duration, which presents problems in extrapolating these findings to the field scale. In this regard the present study was undertaken to evaluate the effects of the land disposal of WTS on soil physical quality, while an associated study is concerned with the chemical and fertility aspects.

In view of the broader objective, this study had to serve two interrelated purposes, namely to expand the present body of knowledge that is concerned with the land disposal of WTS on soil physical quality and based on these findings, to evaluate the feasibility of the land disposal of the sludge produced at the Midmar water-works. To reference the effects of WTS on selected soil physical quality indices, more recognised soil conditioners, namely gypsum, lime and polyacrylamide were also included in the investigation.

The main findings emanating from this work are as follows.

9.1 Water Retention

The influence of WTS on water retention was initially difficult to ascertain due to the complicating influence of tillage but as these effects waned over time, more clear relationships between sludge content and water retention properties became apparent. It was interesting that in general the effects of tillage persisted for much longer at the Brookdale trial (Hutton soil) than was noted at the Ukulinga trial (Westleigh soil) because of the greater degree of stability of the Hutton soil. Because of the lower bulk density of the sludge compared to the soils to which they were added, an increase in the amount of water retained at saturation was observed. In general, however, such improvement was only of statistical significance at the 1280 Mg ha^{-1} level. From near saturation to typically around -30 kPa matric potential very little difference in the volume of water retained between the treatments was noted, but from a matric potential typically <- 30 kPa, the 1280 Mg ha^{-1} sludge treatment once again retained more water. At a matric potential of -30 kPa, all pores larger than 9.7 μm in diameter are air-filled which implies that the sludge must contain pores predominantly smaller than this critical size. Scanning electron microscopy confirmed these findings and showed that apart from the presence of these pores, aggregates of sludge also comprise a series of micro-sized fractures.

With regard to the influence of lime, gypsum and to a lesser extent polyacrylamide on water retention, these products generally had no statistically significant influence on either the Hutton or Westleigh soils in the field.

In an attempt to avoid the complicating influence of tillage and to extend the study to include the Cartref and Valsrivier soils, water retention properties were re-evaluated on re-packed samples. The results of this study generally corroborated the results of the field study, although the extrapolation of these findings to the field scale should be done with caution. Nevertheless, the greatest improvement in water retention was observed at saturation which followed the order 1280 Mg ha^{-1} > 320 Mg ha^{-1} > 80 Mg ha^{-1} > 0 Mg ha^{-1}. Below -30 kPa matric potential the sludge-amended treatments held more water than the control, the relative extent of which increased with an increase in sludge content.

Because of the more controlled conditions that were afforded with using re-packed samples, an estimate of plant available and readily available water content for sludge-amended soils could be made. In general these findings showed that despite the significant improvement that the sludge has on the volume of water retained at the wilting point, both the plant available and readily available water content were statistically non-significantly different from that of the control treatment (with the exception of the Valsrivier soil). Since these indices are calculated as the volume of water retained between specific matric potentials, an increase in the volume of water retained at both ends of the matric potential range caused little change in their difference.

9.2 Aeration

The advances made in understanding the influence of WTS on water retention allow for some comment to be made with regard to soil aeration. The generally accepted view presented in the literature is that at field capacity, root development only becomes limiting if the air-filled porosity is typically less than 10 to 15 %. Neither the control treatments nor the sludge-amended treatments showed values of air-filled porosity at field capacity, below this critical index, which implies that aeration should not be a limiting factor for crop development in sludge-amended soils. When the samples were equilibrated at -10 kPa and -100 kPa matric potential and their permeability to air measured, it was found that increasing the sludge content increased the air permeability, although such differences were statistically non-significant. This increase in air permeability is probably related to an increase in *inter-aggregate* macro-porosity that occurred as a consequence of sludge application, although thin section microscopy may help to confirm this.

9.3 Aggregation

WTS is strongly aggregated due to the sustained performance of the polymer in binding the constituent silt and clay sized material. This was especially apparent in attempting to fractionate air-dried sludge for measurement of its particle size distribution, since even in the presence of a strong dispersing agent and ultrasound, many of the bonds formed by the polymer could not be broken.

Attempts at inferring the influence of the sludge on the aggregation of the soil from its dry aggregate size distribution was unsuccessful although wet sieving proved to be a more encouraging alternative. Although the mean weight diameter of the treatments at both field trials showed an increase with increasing sludge content, much of the material that remained on the sieves at the end of the test was composed primarily of sludge particles. Consequently the improvement in the mean weight diameter noted between the sludge-amended treatments was mainly because of the presence of highly water-stable sludge particles included in the sample than to the effect of the sludge in stabilising *existing* soil aggregates.

A study of the soluble salt concentration beneath the mulched treatments at the Brookdale trial showed an increase in the concentration of Ca^{2+} and Mg^{2+} as a result of leaching of these cations from the sludge. Since calcium in particular is known to promote soil stability by its effect on the diffuse double layer, these findings suggest that adsorption of these cations onto the exchange complex of soil may increase the stability of soil aggregates. This effect was clearly observed at the Ukulinga trial where suction probe apparatus were installed to monitor the soil solution composition. The results showed higher levels of Na^+ in the soil solution on the sludge amended and gypsum treatments than was found for the control treatment because of the replacement of this monovalent ion by the divalent Ca^{2+} and Mg^{2+} ions. Unfortunately due to the inherently stable nature of the Hutton soil, any improvement in aggregation made by the sludge were below the detectable limits of the methods adopted for this investigation.

The findings given above prompted an assessment of the influence that wet sludge would have on aggregation. To this end fresh sludge collected from the water-works was equilibrated to selected matric potentials and mixed with soil which was also maintained at the same matric potential. The samples were incubated for eight weeks, air-dried and the stability of the aggregates measured once more. The results showed that the incubation water content had a strong influence on the ready dispersible clay percentage. When wet, much of the polymer contained in the sludge is still in an active state and was available for bonding with the soil substrate. When incorporated with soil under drier conditions, less of these bonds remain active and consequently less of a decrease in the ready dispersible clay percentage was noted.

9.4 Soil Strength

Due to the tenacity with which the polymer binds the constituent silt and clay fractions, aggregates of sludge, although porous, display high strength. Nevertheless, when incorporated with soil in the field the PSS of the sludge-amended treatments generally showed non-significant differences from that of the control. Although measurements were made when the soil was at or near field capacity, differences in water content between the treatments influenced the results. The 1280 Mg ha^{-1} sludge treatment which was generally wetter than the rest consistently showed a lower PSS than that of the other treatments. Thus, although WTS appeared not to have a direct mechanical influence on soil strength it nevertheless decreased the PSS below the plough layer by increasing the soil water content. With regard tothe influence of gypsum, lime and polyacrylamide, these products had a non significant influence on the PSS.

The samples prepared in the incubation study for evaluating the effect of the sludge on soil aggregation were also used to determine the influence of the sludge on soil strength. Briquettes formed from wet material (without prior air drying) showed a substantial increase in soil strength due to the effect of the polymer in mechanically binding the sludge and soil particles together. An increase in the sludge content caused an increase in the MOR, since more of the polymer in the sludge was available for bonding with the soil. In addition, MOR values were much higher for samples incubated under saturation than under drier conditions. In drying the sludge before its incorporation, a portion of the polymer that would otherwise have been available for bonding becomes irreversibly bonded to the sludge particles and is therefore unavailable for further interaction with the soil. The activation state of the polymer as a fundamental factor regulating the influence of the sludge on soil strength, was further seen when briquettes were formed from mixtures that had been fully air-dried and crushed to pass a 2mm sieve. Even at the 80 Mg ha^{-1} sludge content considerable difficulty was encountered in the formation of briquettes. The central reason for this follows from comments made by Malik and Letey (1991) and Nadler *et al.*, (1992) that once the substrate to which the polymer becomes attached is air-dried, bonding is irreversible and its re-activation negligible, even if the soil is re-wetted.

9.5 Hydraulic Conductivity

During the water treatment process it is mainly silt and clay sized material that is flocculated out of suspension from the turbid waters. In view of this, one of the main concerns was that in returning this silt and clay to soils, potential problems related to reduced hydraulic conductivity may occur mainly because of the potential of the clay and silt (if mobile) to block water conductive pores. Strong emphasis was therefore placed on the effect of WTS on hydraulic conductivity.

At the field trials the saturated hydraulic conductivity was measured using double ring infiltrometers. Undisturbed soil cores were also collected for the measurement of saturated hydraulic conductivity with a view of potentially using this technique for the purpose of monitoring the land treatment scheme. Despite the significant levels of variation in K_s that were encountered in the field , even within single treatments, the results showed that WTS increased K_s . Although the estimate of K_s for each treatment differed between the double ring infiltrometry and undisturbed core methods, in general, the overall trend in K_s found *between* the treatments as evaluated from both methods showed remarkable similarity.

The findings of the field study were further corroborated in a study where sludge and soil (four types) were mixed at varying proportions and packed into permeameters. The results of this investigation showed that K_s was positively related to sludge content, as denoted by the strong linear regression relationships found between the two parameters. The principal reasons for this finding relate to:

- the increase in porosity of the sludge amended treatments, since bulk density decreased with an increase in sludge content;
- the increased macroporosity of the sludge-amended treatments allowed more water to be transmitted through these pores;
- aggregates of WTS are highly stable due to the tenacity with which the polymer binds the constituent silt and clay fraction. Consequently very little clay dispersion occurred to block water conductive pores;

- aggregates of WTS have limited potential for swelling due to the rigidity with which the polymer binds the constituent fraction. Consequently, restriction of pores was probably less on the sludge-amended treatments than on the control treatments.

Although a slight improvement in K_s on the polyacrylamide treatment (at the 30 kg ha^{-1} level) was measured, such effects did not persist for very long and in general non-significant differences from the control treatment were found. Neither lime nor gypsum had any statistically significant influence on K_s. At the Brookdale trial this finding is probably related to the inherently high stability of this soil which may have masked any improvement brought about by the addition of these products. On the other hand, on the much poorer structured and less stable Westleigh soil (Ukulinga trial) it is probable that insufficient time has elapsed for gypsum (in particular) and lime to bring about any meaningful change in soil aggregation and, therefore, K_s.

The water retention curves of the repacked samples and its corresponding saturated hydraulic conductivity were used to calculate unsaturated hydraulic conductivity (K_Ψ) using the van Genuchten (1991) model. The results showed a decrease in K_Ψ with an increase in sludge content. These results suggested that evaporation would be decreased on sludge-amended soils since evaporations tends to be partially regulated by K_Ψ. This hypothesis was tested in a study where sludge-amended soils were dried under controlled environmental conditions. The results supported the hypothesis and showed that in the presence of 1280 Mg ha^{-1} sludge, the amount of water lost was much less than that from the control. In addition it was found that mulching (where the sludge was applied on the soil surface) was a more efficient technique of conserving soil water than incorporating the sludge with the soil, although this may present some practical problems in the efficient operation of a land disposal scheme.

9.6 General Remarks

Over the past three years during which this study was conducted, there has been little evidence to suggest that the land disposal of water treatment sludge is not a feasible disposal practice. Such comments, however, stem purely from a soil physical perspective. For many of the parameters that were investigated, the addition of WTS showed an *improvement* in soil physical properties, especially concerning the movement of water through the soil under saturated conditions, which is contrary to what was initially expected. The reason for this stems mainly from the physical characteristics of the sludge that was added to the soil. Despite its porous nature WTS has high stability due to the sustained performance of the polymer that is used to bind the silt and clay sized material together. When incorporated into the soil an immediate effect is a decrease in the bulk density and a concomitant increase in porosity, particularly macro-porosity. This increase in porosity and the high stability of the sludge aggregates (aggregates of sludge have a limited capacity for swelling and dispersion) caused an increase in the final infiltration rate and saturated hydraulic conductivity of the amended medium. Moreover, owing to the addition of lime during the water treatment process, Ca^{2+} and Mg^{2+} levels in the soil saturation extract were found to increase, implying a potential increase in stability and therefore water transmission. Although the sludge was found to have little direct influence in decreasing the PSS within the amended zone (depth of tillage) it did decrease the PSS of the subsoil beneath, which compared to the control, tended to remain wetter owing to its increased infiltration. The increased macro-porosity of the sludge-amended soil also caused a decrease in the evaporation rate due mainly to a decrease in the unsaturated hydraulic conductivity. This finding could also account for the subsoil having remained wetter for longer. An additional consequence of the increased macro-porosity was an increase in air permeability particularly at the 1280 Mg ha^{-1} level. Despite these increases, however, the air-filled porosity at -10 kPa matric potential remained similar to that of the control and tended to improve aeration status. The wilting point for the sludge-amended soils was found to increase with increasing sludge application because of the large number of micro-pores that are contained within individual sludge aggregates. Water retained at -10 kPa matric potential (field capacity) also increased resulting in little change in plant available water.

With regard to the applied aspects associated with this project, the work presented herein has allowed for the development of interim management guidelines for the land disposal of WTS at Brookdale Farm (Appendix 9.1). Some additional comments on the handling of the sludge and techniques for its disposal is contained therein.

9.7 Opportunities for further research

At the onset of this study, a broad-scale approach had to be adopted in designing the research protocol, as it was considered important to address the range of aspects that would determine whether the land disposal of WTS from a soil physical perspective was a viable option or not. Despite the knowledge gained from this study, several aspects which could not be addressed or were found to be deserving of continued attention, have come to the fore, and are discussed below.

- Many of the relationships established in the present study can be linked directly or indirectly to the sustained performance of the polymer, in binding the constituent silt and clay fraction into aggregates of high strength and high stability. In the period over which this study was conducted there has been little evidence to suggest a deterioration in the functioning of this polymer. Nevertheless, it is important that a more intensive study be conducted on the mechanisms under which compounds operate with special attention being paid to the factors which could possibly bring about their degradation. The subject of polymer-clay mineral interaction is gaining increased attention, particularly in the fields of high molecular weight polymer chemistry. Close attention should be paid to research emanating from these fields.

- Findings from the present study suggest that the degradation of the sludge proceeds extremely slowly, more so when the material is protected from the impact of rainfall and rapid wetting and drying cycles. Nevertheless, it is important that longer-term effects of applying sludge to land should be established. In this regard the field trials should continue to be monitored for changes occurring particularly in aggregation and hydraulic conductivity. The intensity at which the investigations are done can probably be lengthened, to perhaps two to three times per year. In the interim, the soil water samplers should be monitored on a regular basis to assess the change in the soil solution composition over time is facilitated.

- In view of the practical objectives associated with this study, only a single sludge type drawn essentially from a single catchment was considered. It is important that, a full characterisation of the range of sludges being produced at water treatment works should be attempted. Work in this regard has already been initiated, in response to a request for water treatment sludge disposal guidelines to be established at a national level. In many respects this effectively represents progression from level 0 and 1 to level 2 and 3 in the conceptual framework (Figure 1.1) presented by Karlen *et al.* (1997).

- As outlined in the introduction of this work, a successful land treatment system is one that can sustain plant growth. Evidence from the grassed plots at both field trials suggests that crop growth is enhanced on sludge-amended soils and that the change in the soil physical environment may be an important factor in this regard.

Figure 9.1 The growth of perennial ryegrass at the Ukulinga field trial. The 1280 Mg ha^{-1} treatment is shown in the foreground.

As can be seen clearly seen from Figure 9.1 the 1280 Mg ha^{-1} WTS-incorporated treatment shown in the immediate foreground (Ukulinga field) supports a much lusher growth of perennial ryegrass compared to the rest of the treatments, despite it being well-established in the literature that WTS has a large phosphorous absorption capacity. Preliminary indicators are that root proliferation in the sludge-amended soil is a significant factor that warrants further study. Figure 9.2 shows the root system of representative ryegrass plants growing in the 0, 320 and 1280 Mg ha^{-1} sludge amended treatment at the Ukulinga trial.

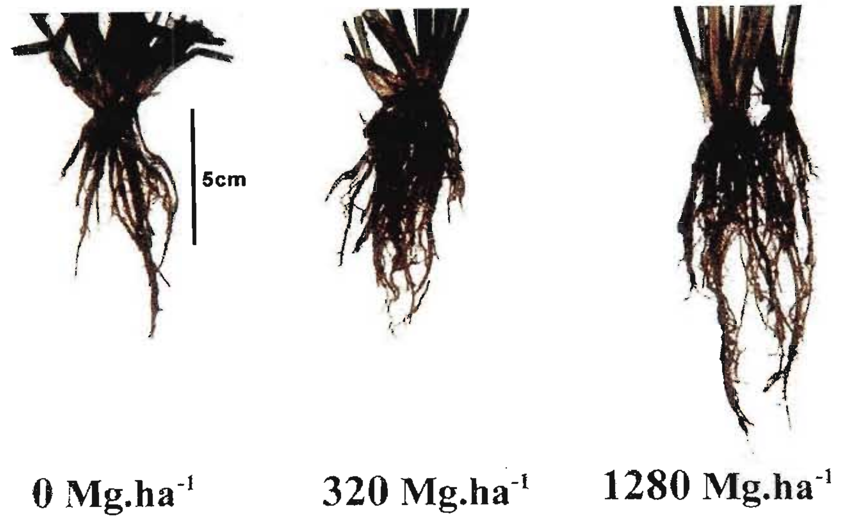

Figure 9.2 Root development of perennial ryegrass grown in sludge-amended soil at the Ukulinga field trial.

It can be clearly seen that both the both root length and root density on the 1280 Mg ha^{-1} treatment is substantially greater than that of the control treatment. The decrease in soil strength, an increase in water retention and an improvement in soil aeration could be some of the factors that account for this observation. Clearly, conclusive establishment of the reasons is dependent upon further study and should be viewed against the fertility status of WTS amended soils. Further work should aim to establish relationships between the soil physical properties and factors such as root mass, length, density, biomass production and associated plant growth parameters. Pot trials coupled with information drawn from the field trials may help greatly in achieving these objectives. Moreover, this will allow for a more integrated approach between the soil physical and chemical aspects of this study.

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List of Appendices

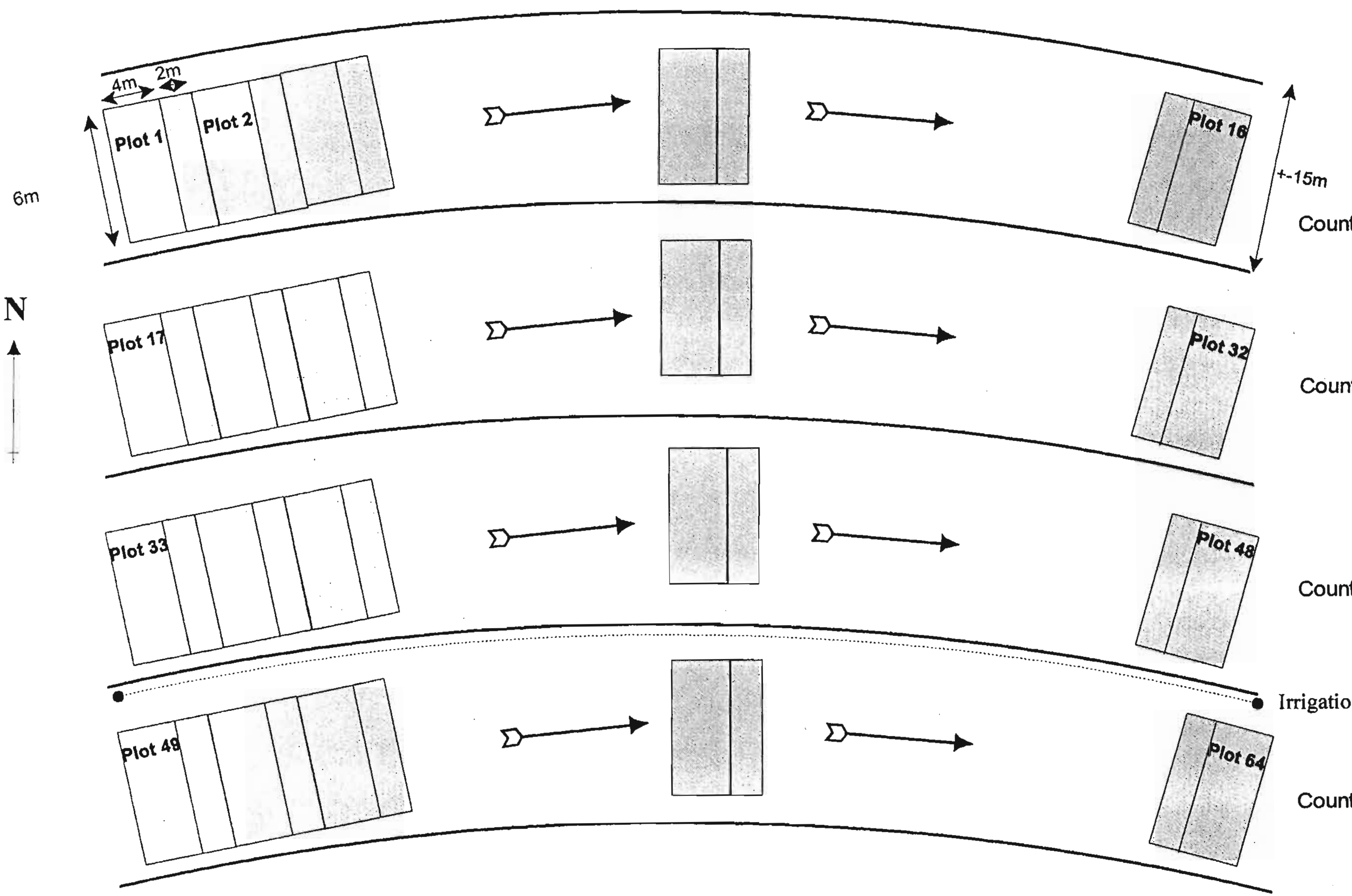
Appendix 3.1 Treatments being investigated at the Brookdale trial and layout of the experiment.

Maintained Fallow		Cropped to Perennial ryegrass	
Plot	**Treatment**	**Plot**	**Treatment**
1	P15	33	Control
2	L2	34	P15
3	1280	35	40
4	G10	36	L2
5	M_{320}	37	160
6	40	38	320
7	G5	39	M_{1280}
8	1280	40	L10
9	M_{640}	41	P30
10	M_{1280}	42	160
11	L2	43	640
12	320	44	M_{320}
13	80	45	Control
14	P30	46	P15
15	Control	47	G10
16	160	48	40
17	640	49	P30
18	80	50	320
19	160	51	G5
20	Control	52	G5
21	640	53	80
22	L2	54	L10
23	M_{640}	55	M_{320}
24	G5	56	G10
25	320	57	640
26	40	58	80
27	M_{320}	59	L2
28	P15	60	1280
29	G10	61	M_{640}
30	P30	62	M_{1280}
31	M_{1280}	63	M_{640}
32	L10	64	1280

NB The plot size is 6 m X 4 m. For a description of the treatments see the key below.

KEY

Codes used in Table above	Description of Treatment
L2	Lime applied at 2 Mg ha^{-1}. Incorporated with the top 0.2m of the soil by discing.
L10	Lime applied at 10 Mg ha^{-1}. Incorporated with the top 0.2m of the soil by discing.
G5	Gypsum applied at 5 Mg ha^{-1}. Incorporated with the top 0.2m of the soil by discing.
G10	Gypsum applied at 10 Mg ha^{-1}. Incorporated with the top 0.2m of the soil by discing.
P15	Anionic polyacrylamide applied at 15 kg ha^{-1}. Sprayed onto the plot surface as a 0.1 g L^{-1} solution.
P30	Anionic polyacrylamide applied at 30 kg ha^{-1}. Sprayed onto the plot surface as a 0.1g L^{-1} solution.
40, 80, 160, 320, 640, 1280	WTS sludge content (Mg ha^{-1}). Incorporated with the top 0.2m of the soil by discing.
M_{320}, M_{640}, M_{1280}	Subscript refers to sludge content (Mg ha^{-1}). The sludge was applied to the plot as a mulch (M) i.e. no incorporation of the sludge with the soil was undertaken.



Contours 1 and 2 are maintained fallow

Contours 3 and 4 are cropped to *Lolium perenne*.

Appendix 3.2 Manufacturers description of the polyacrylamide being used at the Brookdale and Ukulinga field trials.

Floccotan (Pty) Ltd.

Reg. No. 73/04432/07

G-26

DRY ANIONIC POLYMER

DESCRIPTION

Floccotan G - 26 is a high molecular weight, medium charge density anionic polyelectrolyte. Floccotan G-26 is a free flowing white granular product which is easily dispersed for rapid make-down. Floccotan G - 26 has been proven effective in many liquid/solids separation processes such as sedimentation, flotation, filtration and centrifugation.

TYPICAL PROPERTIES

Appearance............................	White granular solid
pH 0.1% solution (Tap water)..........	7 - 8
Charge................................	Medium anionic
Viscosity (0.5% in distilled water)....	5500 mPa.s
Viscosity (0.1% in distilled water)....	600 mPa.s

PACKAGING

Floccotan G - 26 is available in 25kg net multiwall paper bags with moisture barrier.

STORAGE

Floccotan G - 26 is slightly hygroscopic and should be stored in a dry environment.

CPW Prod L 3004

ALBERT RD. PENTRICH INDUSTRIAL ESTATE P.O. BOX 1818 PIETERMARITZBURG 3200 NATAL SOUTH AFRICA

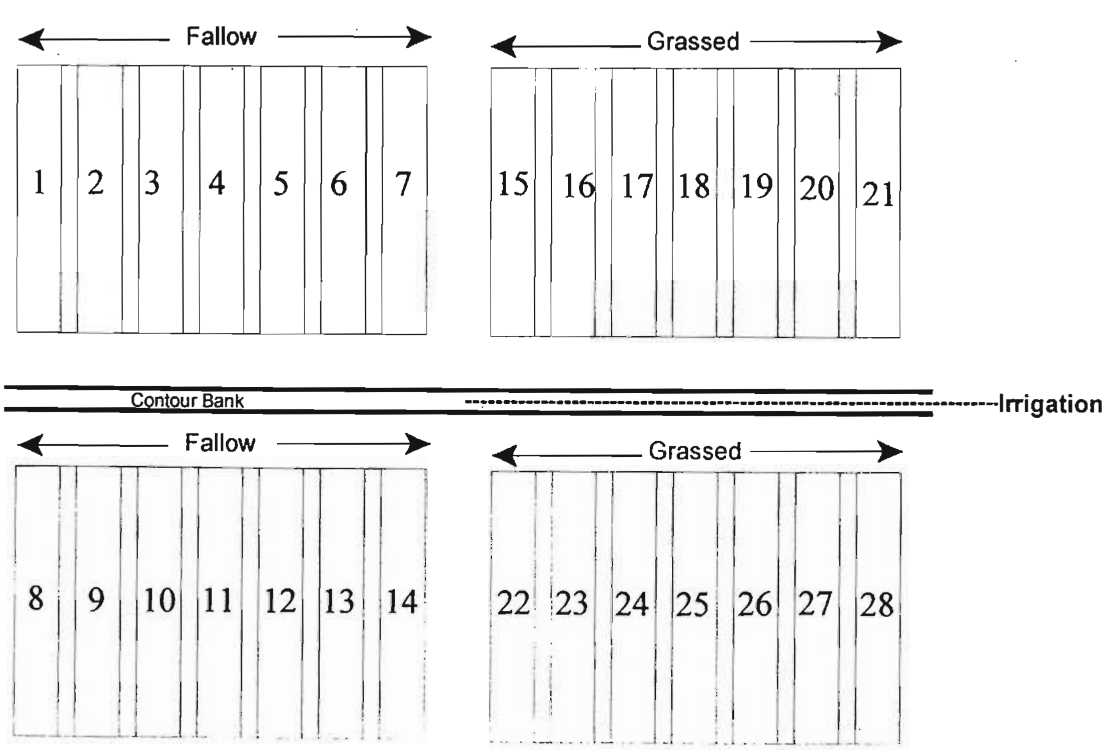
Appendix 3.3 Treatments being investigated at the Ukulinga trial and layout of the experiment.

Treatments maintained Fallow		Treatments cropped to *lolium perenne*	
Plot	**Treatment**	**Plot**	**Treatment**
1	Control	15	80
2	320	16	P30
3	G10	17	80
4	1280	18	320
5	L10	19	1280
6	320	20	G10
7	P30	21	Control
8	80	22	1280
9	G10	23	L10
10	1280	24	320
11	P30	25	L10
12	80	26	Control
13	L10	27	G10
14	Control	28	P30

For a description of the treatments see the key given below

KEY

L10	Lime applied at 10 Mg ha^{-1}. Incorporated with the top 0.2m of the soil by discing.
G10	Gypsum applied at 10 Mg ha^{-1}. Incorporated with the top 0.2m of the soil by discing.
P30	Anionic polyacrylamide applied at 30 kg ha^{-1}. Sprayed onto the plot surface as a 0.1g L^{-1} solution.
80, 320,1280	WTS sludge content (Mg ha^{-1}). Incorporated with the top 0.2m of the soil by discing.



Appendix 3.4 Major element and trace element composition of the Midmar water treatment sludge as determined by X-ray Fluorescence Spectrometry

Major Element Composition - (Weight Percentage of the oxide)

SiO_2	Al_2O_3	Fe_2O_3	MnO	MgO	CaO	Na_2O	K_2O	TiO_2	P_2O_5	Total	L.O.I
54.57	22.6	11.95	1.529	1.92	4.2	0.15	1.47	0.8568	0.24	99.4858	23.91

Trace Element Composition (mg kg^{-1})

Zn	Cu	Ni	Cr	V	La	Cd	Cl	Ba	Sc	S
84	44	53	161	154	19	4	1575	1007	33	720

Nb	Y	Rb	Zr	Sr	U	Th	Pb	Ga	Co	Ce	Nd	As
11	27	94	126	71	0	10	37	20	39	96	25	17

Appendix 3.5 Methods used for obtaining the information presented in Table 3.4

Particle Size

The particle size distribution of the samples shown in Table 3.4 were measured by the pipette method. 20g of air dried soil (<2 mm) was treated with 10 ml H_2O_2 to oxidise the organic matter. The samples were then dispersed by adding 10 ml Calgon solution (35.7g sodium hexametaphosphate and 7.9g sodium carbonate solution) in 20 ml distilled water followed by ultrasound for 3 minutes. The ultrasonic probe used was the Labsonic 2000 with an output of 350-400 W. The dispersed sample was washed through a 0.053 mm sieve into a 1 L sedimentation cylinder and made up to the mark with distilled water.

The < 0.053 mm fraction in the sedimentation cylinder was brought into suspension by agitation of the sample with a specially designed plunger. The coarse silt (0.02 to 0.05 mm), fine silt (0.002 to 0.02 mm) and clay (<0.002 mm) were determined by sedimentation and pipette sampling after the appropriate settling time according to Stokes Law (Gee and Bauder, 1986). The material retained on the sieves (sand fraction) was washed into 250 ml beakers and dried in the oven overnight. The sand fraction was sieved through a sieve stack arranged in the order of sieve apertures of 0.500 mm, 0.250 mm and 0.106 mm. The material retained on each sieve was expressed as a percentage of the oven-dry soil and classified as coarse sand, medium sand and fine sand respectively. The material passing through the 0.106 mm sieve was classified as very fine sand. All fractions were expressed as a mass percentage of oven-dry soil by correcting for the water content of the air dry sample.

Soil Texture

The sand, silt and clay percentage was used to award each soil a textural class according to the classification system of the Soil Classification Working Group, 1991.

Particle Density

Particle densities were measured in triplicate by the pycnometer method according to Blake and Hartge, 1986.

Cation exchange capacity

2.5 g of air dry (<2 mm)samples were treated with 25 ml of 0.1M $SrCl_2$ solution. The resulting slurry was shaken on a reciprocal shaker for 30 minutes and then centrifuged at 3000 r.p.m for 4 minutes. The supernatant was filtered through Whatman 541 filter paper into 100 ml flasks. The residue in the centrifuge tubes was washed a further three times with 25 ml of 0.1 M $SrCl_2$ each time shaking the sample for 15 minutes, centrifuging at 3000 r.p.m and filtering the supernatant into the 100 ml flask. The flask was then made up to the mark with 0.1 M Sr Cl_2.

For the measurement of calcium and magnesium, 8 ml of distilled water was added to 2 ml of sample. For the measurement of sodium, 9ml of 1200 mg L^{-1} potassium solution was added to 1 ml of sample and for the measurement of potassium 9ml of 1200 mg L^{-1} Caesium solution was added to 1 ml of sample. These basic cations in solution were determined using atomic absorption and flame emission spectroscopy (Varian AA 10B instrument) and expressed in terms of $cmol_c\ kg^{-1}$ soil.

Organic Carbon

Organic carbon was measured in triplicate according to the Walkley-Black method first prescribed by Walkley (1947). Air dry soil ground to pass a 0.5 mm sieve was digested in a potassium dichromate/sulphuric acid mixture to oxidise the organic carbon. Soil organic carbon content is then determined by back-titration of the excess dichromate, using a 0.5N ferrous ammonium sulphate solution.

pH

25ml of either 1M KCL or distilled water was added to 10 g of soil (air dry, <2mm) in a stoppered vial to give a soil solution ratio of 1:2.5. The samples were left to stand overnight after which the pH of the supernatant was measured with a standard glass electrode (Radiometer Ion 85 analyser).

Soil EC (1: 5)

10g of each sample was added to centrifuge tubes and shaken in 50 ml of distilled water for 15 min on a bench shaker, allowed to stand for an hour, and then shaken for a further 5 min. The soil solution was then filtered through Whatman No. 2 paper and the EC of the filtrate measured using a standard glass electrode (Radiometer CM 83 analyser).

Appendix 3.6 (a) Soil profile description for the Hutton soil at Brookdale Farm

Soil: Hu 3100
Soil Form: Hutton
Soil Family: Stella
Location: Brookdale Farm, Howick
Parent material: Dolerite
Land use: Dryland cropping

Horizon	**Depth (m)**	**Description**
A	0 - 0.38	Very dark brown (10YR 2/2); clay; weak crumb structure; dry; firm; many fine roots; gradual transition to B1.
B1	0.38 - 0.60	Dark brown (7.5 YR 3/2); clay; weak subangular blocky; moist, friable; gradual transition to B2.
B2	0.60 - 1.20+	Dark red (2.5 YR 3/6); weak subangular blocky; moist, firm.

Soil Properties

Horizon	Particle Size Analysis (%)					
	Sand			Silt		Clay
	C	M	F	C	F	
A	7	2	5	16	18	52
B1	6	1	6	20	11	56
B2	8	2	6	8	13	63

Horizon	pH		Exchangeable Cations ($cmol_c\ kg^{-1}$)				Exch. Acidity ($cmol_c\ kg^{-1}$)	Organic carbon ($g\ kg^{-1}$)
	KCl	H_2O	Ca	Mg	Na	K		
A	4.22	5.21	5.57	2.29	0.1	0.17	0.37	33.5
B1	4.14	5.50	4.38	2.60	0.24	2.17	0.40	23.2
B2	4.04	5.82	5.34	1.95	0.12	1.00	1.42	15.2

Appendix 3.6 (b) Soil profile description for the Westleigh soil at Ukulinga Farm

Soil:	We 1000
Soil Form:	Westleigh
Soil Family:	Helena
Location:	Ukulinga Farm, Mkondeni, Pietermaritzburg
Parent material:	Ecca shale

Horizon	Depth (m)	Description
A	0 -0.26	Very dark brown (10YR 2/2); silty clay loam; weak crumb structure; hard when dry; gradual transition to B.
B	0.26 - 0.55	Dark reddish gray (5YR 4/2); gravelly silty clay loam; moderate subangular blocky; very hard when dry; numerous yellowish-brown mottles; hardpan concretions.

Soil Properties

Horizon	Particle Size Analysis (%)					
	Sand			Silt		Clay
	C	M	F	C	F	
A	9	2	6	25	25	33
B	not determined					

Horizon	pH		Exchangeable Cations ($cmol_c\ kg^{-1}$)				Exch. Acidity ($cmol_c\ kg^{-1}$)	Organic Carbon ($g\ kg^{-1}$)
	KCl	H_2O	Ca	Mg	Na	K		
A	4.90	5.91	6.24	3.06	0.14	0.17	0.06	21.70
B	not determined							

Appendix 4.1a Mean bulk densities for the treatments investigated at the Brookdale trial.0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L2, L10, G5, G10, P15 and P30 are treatments of lime at 2 and 10 Mg ha^{-1}, gypsum at 5 and 10 Mg ha^{-1} and polyacrylamide at 15 and 30 kg ha^{-1} respectively.

Time	Treatments									
	0	**80**	**320**	**1280**	**L2**	**L10**	**G5**	**G10**	**P15**	**P30**
February 1999	1.113	1.014	1.110	1.026						
April 1999	1.120	1.112	1.084	1.056						
June 1999	1.106	1.095	1.097	1.051	1.077	1.113	1.088	1.125	1.080	1.065
November 1999	1.143	1.159	1.153	1.109	1.183	1.137	1.168	1.182	1.122	1.111
February 2000	1.203	1.221	1.188	1.129	1.203	1.157	1.196	1.190	1.159	1.160
April 2000	1.196	1.197	1.198	1.095	1.190	1.186	1.155	1.138	1.097	1.141
June 2000	1.163	1.175	1.132	1.106	1.138	1.178	1.140	1.118	1.146	1.126

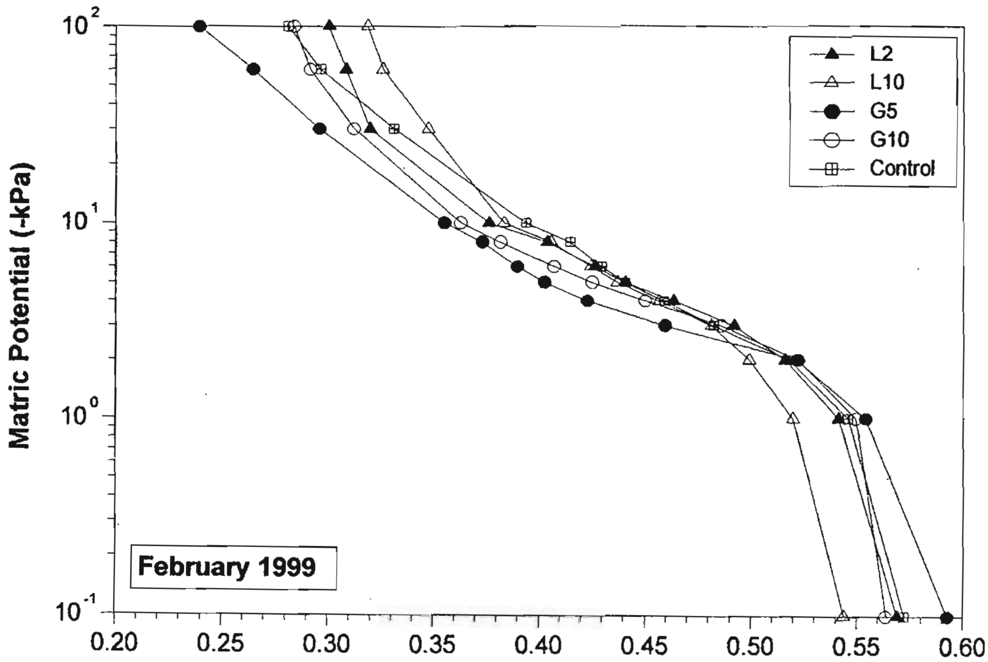
Appendix 4.1b Mean bulk densities for the treatments investigated at the Ukulinga trial.0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L10, G10, and P30 are treatments of lime at 10 Mg ha^{-1}, gypsum at 10 Mg ha^{-1} and polyacrylamide at 30 kg ha^{-1} respectively.

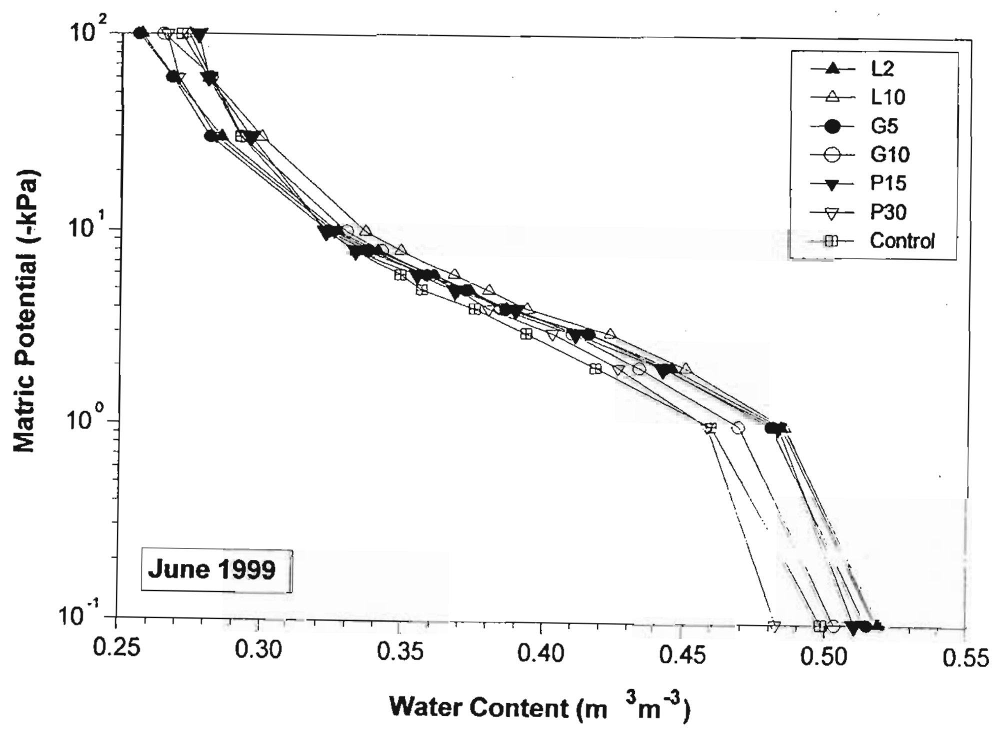
Time	Treatments						
	0	80	320	1280	L10	G10	P30
December 1999	1.167	1.170	1.108	1.099			
April 2000	1.297	1.243	1.147	0.999	1.288	1.241	
August 2000	1.252	1.179	1.091	1.034	1.212	1.185	

Appendix 4.2 Summary of analysis of variance for the effect of WTS application rates (0, 80, 320 and 1280 Mg ha^{-1}) on θ_v for selected matric potentials at the Brookdale trial.

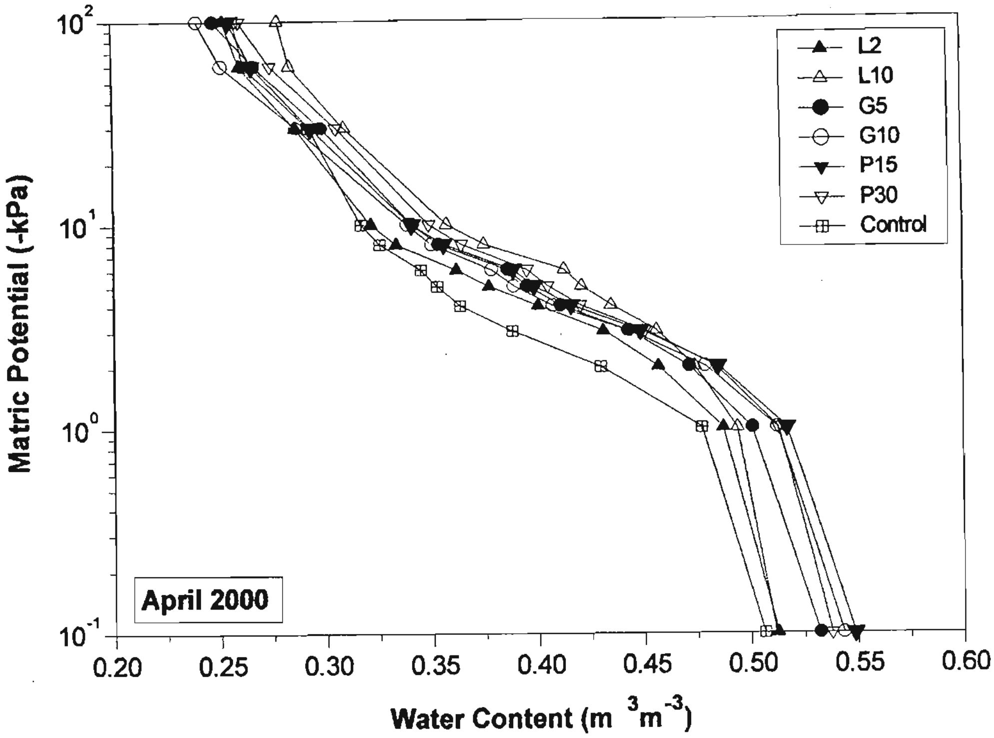
Time	Matric Potential (-kPa)	Sludge Application Level (Mg ha^{-1})				LSD $_{(0.05)}$	F	p
		0	80	320	1280			
November 1999	0	0.480A	0.503BC	0.495AB	0.529D	0.0225	8.98	0.0061
	10	0.326A	0.343B	0.348BC	0.372D	0.0096	41.13	0.0010
	30	0.289	0.302	0.302	0.308	NS	2.67	0.1180
	60	0.264	0.275	0.271	0.283	NS	1.31	0.3350
	100	0.255	0.262	0.265	0.271	NS	1.07	0.4120
April 2000	0	0.506A	0.507A	0.508AB	0.547C	0.0163	14.23	0.0003
	10	0.317	0.324	0.322	0.336	NS	2.51	0.1079
	30	0.293	0.288	0.285	0.304	NS	2.04	0.1620
	60	0.263	0.267	0.266	0.282	NS	1.46	0.2751
	100	0.258	0.256	0.254	0.266	NS	0.71	0.5642
June 2000	0	0.531	0.544	0.556	0.563	NS	3.41	0.0530
	10	0.347A	0.355A	0.359AB	0.382C	0.0146	10.10	0.0010
	30	0.316	0.318	0.326	0.337	NS	2.68	0.0940
	60	0.309	0.312	0.323	0.327	NS	0.79	0.5220
	100	0.299	0.300	0.305	0.311	NS	0.45	0.7211

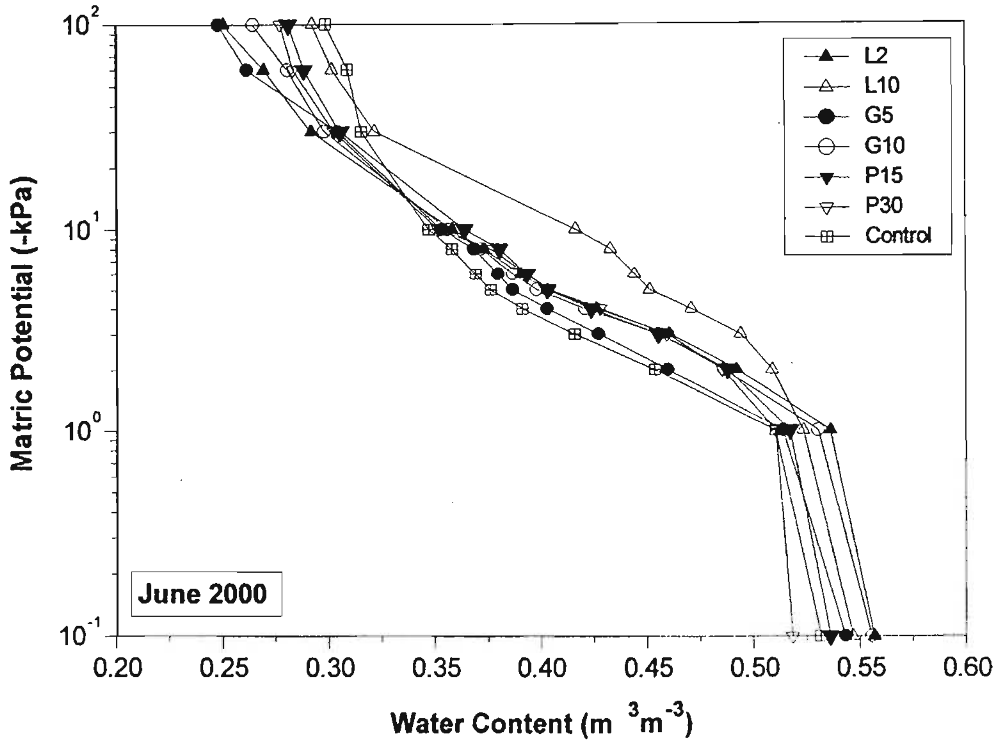
Appendix 4.3 Water retention curves for the lime, gypsum and polyacrylamide treatments at the Brookdale Trial. L2, L10, G5, G10, P15 and P30 are treatments of lime at 2 and 10 Mg ha^{-1}, gypsum at 5 and 10 Mg ha^{-1} and polyacrylamide at 15 and 30 kg ha^{-1} respectively.





Appendix 4.3 (continued)





Appendix 4.4 Summary of analysis of variance for the effect of gypsum at 5 and 10 Mg ha^{-1}, lime at 2 and 10 Mg ha^{-1}, polyacrylamide at 15 and 30 kg ha^{-1} and the control on θ_v for selected matric potentials at the Brookdale trial. The treatments are as described in Figure 5.1

Time		Treatment							LSD	F	p
		0	L2	L10	G5	G10	P15	P30	(0.05)		
November 1999	0	0.480 A	0.499 BC	0.516 D	0.512 CD	0.486 AB	0.513 CD	0.521 D	0.015	9.367	0.0003
	10	0.326	0.355	0.359	0.349	0.343	0.322	0.342	NS	1.804	0.1701
	30	0.289	0.274	0.301	0.293	0.293	0.283	0.286	NS	0.324	0.9129
	60	0.264	0.243	0.262	0.268	0.277	0.251	0.258	NS	0.374	0.883
	100	0.255	0.232	0.246	0.254	0.258	0.245	0.242	NS	0.334	09076
April 2000	0	0.506 A	0.513 AB	0.512 A	0.532 BC	0.543 C	0.549 C	0.537 C	0.204	6.070	0.0008
	10	0.317 A	0.322 AB	0.357 C	0.341 BC	0.339 BC	0.341 BC	0.349 C	0.021	3.86	0.0094
	30	0.293	0.287	0.309	0.299	0.287	0.293	0.306	NS	1.54	0.2130
	60	0.263	0.260	0.284	0.267	0.252	0.267	0.275	NS	1.769	0.1540
	100	0.258 BC	0.253 B	0.279 C	0.248 B	0.221 A	0.255 B	0.261 BC	0.023	4.849	0.0030
June 2000	0	0.531	0.557	0.547	0.543	0.555	0.536	0.518	NS	1.987	0.1133
	10	0.347 A	0.358 A	0.417 C	0.353 A	0.356 A	0.364 AB	0.352 A	0.026	7.90	0.0002
	30	0.316	0.292	0.322	0.304	0.298	0.301	0.303	NS	1.717	0.167
	60	0.309	0.270	0.302	0.262	0.281	0.289	0.284	NS	2.538	0.0524
	100	0.299 D	0.251 AB	0.293 CD	0.248 A	0.265 ABC	0.281 BCD	0.276 ABCD	0.033	3.116	0.0241

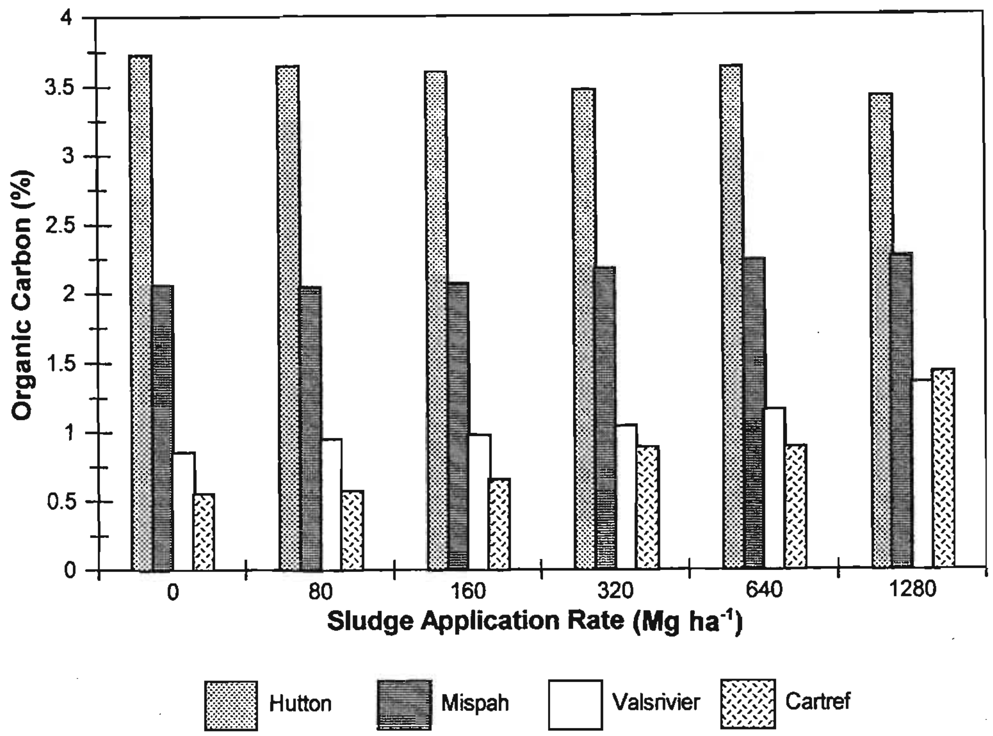
NS = non significantly different. Degrees of freedom for December 1999, April 2000 and June 2000 are 20 (3 replicates per treatment), 27 and 27 respectively.

Appendix 4.5 Summary of analysis of variance for the effect of WTS (0, 80, 320 and 1280 Mg ha^{-1}), gypsum at 10 Mg ha^{-1}, lime at 10 Mg ha^{-1} and polyacrylamide at 30 kg ha^{-1} on θ_v, for selected matric potentials at the Ukulinga trial. The treatments are as described in Figure 5.4.

Time		Treatments							LSD (0.05)	F	p
		0	80	320	1280	G10	L10	P30			
December 1999	0	0.506	0.507	0.530	0.555		n.d		NS	4.001	0.0518
	10	0.295 A	0.301 A	0.312 AB	0.344 C		n.d		0.019	17.97	0.0006
	30	0.255 A	0.256 A	0.269 B	0.320 C		n.d		0.012	66.26	0.0001
	60	0.209 A	0.214 A	0.212 B	0.292 C		n.d		0.034	14.96	0.0013
	100	0.223 A	0.235 AB	0.248 BC	0.306 D		n.d		0.022	28.47	0.0001
April 2000	0	0.487 A	0.518 B	0.553 C	0.565 C	0.503 AB	0.509 B	n.d	0.018	25.46	0.0001
	10	0.324 A	0.311 A	0.326 AB	0.360 C	0.325 A	0.312 A	n.d	0.023	5.35	0.0035
	30	0.281 A	0.271 A	0.295 AB	0.330 C	0.293 A	0.285 A	n.d	0.025	5.87	0.0022
	60	0.264 AB	0.251 A	0.284 BC	0.315 D	0.272 AB	0.252 A	n.d	0.022	10.47	0.0001
	100	0.247 A	0.235 A	0.258 A	0.309 C	0.260 AB	0.240 A	nd	0.026	8.26	0.0003
August 2000	0	0.436 A	0.468 A	0.518 C	0.589 D	0.481 B	0.471 B	0.424 A	0.022	54.97	0.0001
	10	0.325 A	0.320 A	0.337 AB	0.365 C	0.324 A	0.323 A	0.324 A	0.024	3.72	0.0112
	30	0.292 A	0.285 A	0.297 A	0.340 C	0.289 A	0.291 A	0.301 AB	0.022	6.18	0.0007
	60	0.268 A	0.265 A	0.271 A	0.324 C	0.257 A	0.264 A	0.276 AB	0.023	8.302	0.0001
	100	0.250 AB	0.251 AB	0.251 AB	0.317 D	0.243 A	0.250 AB	0.267 BC	0.023	10.43	0.0001

NS = non significantly different. n.d = not detrmined. Degrees of freedom for December 1999, April 2000 and August 2000 are 11 (3 replicates per treatment), 23 and 27 respectively.

Appendix 4.6 Mean organic carbon (%) of the sludge-amended treatments for the four soil types investigated.



Appendix 4.7(a) Mean readily available water content (m^3 m^{-3}) for the treatments being investigated at the Brookdale Trial. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L2, L10, G5, G10, P15 and P30 are treatments of lime at 2 and 10 Mg ha^{-1}, gypsum at 5 and 10 Mg ha^{-1} and polyacrylamide at 15 and 30 kg ha^{-1} respectively

Time	0	80	320	1280	L2	L10	G5	G10	P15	P30
Feb. 1999	0.112	0.096	0.075	0.068	0.076	0.064	0.115	0.079	N.D	N.D
April 1999	0.086	0.105	0.039	0.061	N.D	N.D	N.D	N.D	N.D	N.D
June 1999	0.052	0.064	0.063	0.046	0.069	0.062	0.067	0.065	0.045	0.059
Nov. 1999	0.071	0.078	0.086	0.101	0.122	0.113	0.095	0.085	0.077	0.100
April 2000	0.060	0.068	0.068	0.069	0.069	0.079	0.093	0.118	0.085	0.088
June 2000	0.048	0.055	0.054	0.071	0.107	0.124	0.105	0.091	0.083	0.074

Appendix 4.7(b) Mean readily available water content (m^3 m^{-3}) for the treatments being investigated at the Ukulinga Trial. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L10,G10 and P30 are treatments of lime at 10 Mg ha^{-1}, gypsum at 10 Mg ha^{-1} and polyacrylamide at 30 kg ha^{-1} respectively

Time	0	80	320	1280	L10	G10	P30
Dec. 1999	0.093	0.091	0.101	0.063	N.D.	N.D.	N.D.
April 2000	0.077	0.076	0.068	0.049	0.065	0.072	N.D.
August 2000	0.075	0.069	0.087	0.048	0.073	0.080	0.057

Appendix 5.1 Final non-linear least squares regression results from the RETC computer programme of van Genuchten *et al.* (1991).

Hutton Soil

Parameter	Sludge Content (Mg ha^{-1})			
	0	80	320	1280
θs	0.56064	0.57024	0.58102	0.63628
θr	0.18738	0.13162	0.20176	0.27082
α	0.97164	0.18454	1.48771	6.49058
n	1.39200	1.24269	1.36616	1.36224

Mispah Soil

Parameter	Sludge Content (Mg ha^{-1})			
	0	80	320	1280
θs	0.48436	0.48511	0.50459	0.55005
θr	<0.001	<0.001	0.06454	0.000
α	2.65312	2.73191	2.68156	4.06173
n	1.163	1.161	1.20965	1.13685

Valsrivier Soil

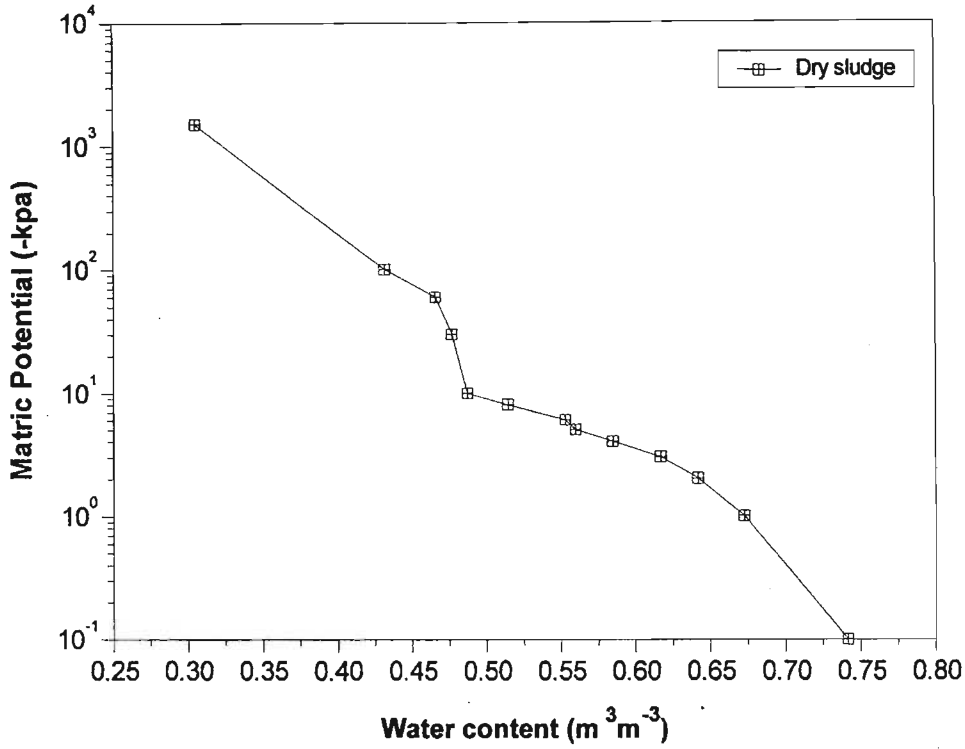
Parameter	Sludge Content (Mg ha^{-1})			
	0	80	320	1280
θs	0.44382	0.455	0.46961	0.53198
θr	<0.001	<0.001	<0.001	0.000
α	5.8977	4.32829	2.47613	6.81680
n	1.12136	1.13168	1.353	1.09322

Cartref Soil

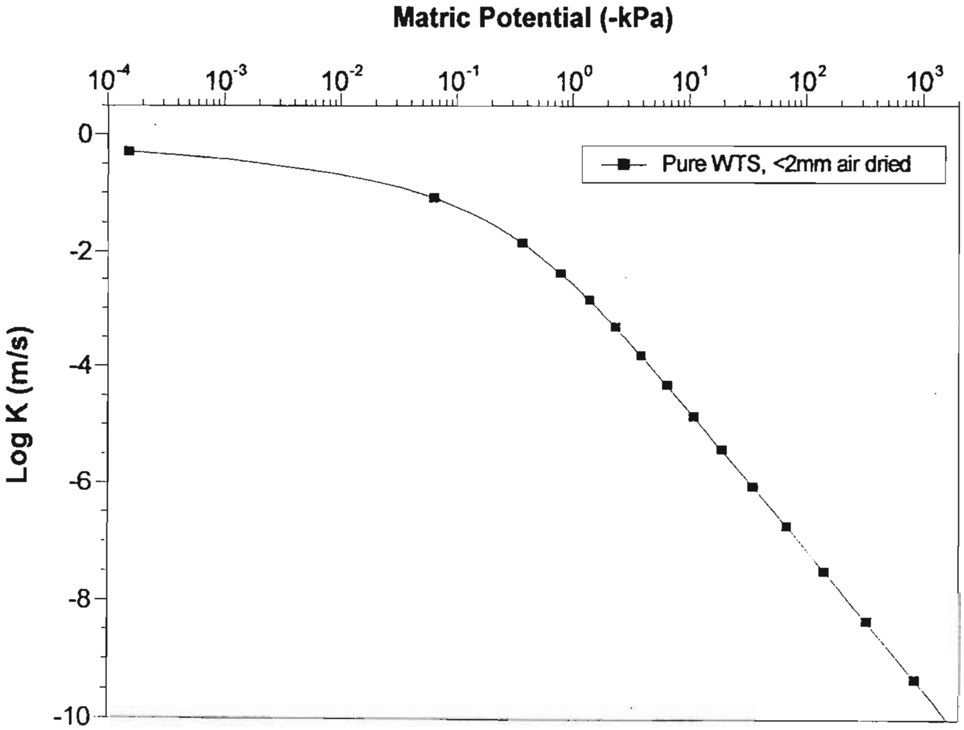
Parameter	Sludge Content (Mg ha^{-1})			
	0	80	320	1280
θs	0.39015	0.41168	0.42630	0.50069
θr	<0.001	<0.001	<0.001	0.14474
α	3.04387	4.27919	5.90402	8.51352
n	1.23215	1.24123	1.4985	1.27184

The value of <0.001 indicates that in fitting an analytical solution to the observed data the residual water content was set to 0 in the iterative sequence (van Genuchten et al., 1991).

Appendix 5.2 (a) Water retention curve measured on repacked samples of < 2mm air-dried pure sludge. Each point is the mean of four replicates.



Appendix 5.2(b) Unsaturated hydraulic conductivity function for repacked samples of < 2mm air-dried pure sludge. Each point is the mean of four replicates.



Appendix 6.1 Mean air-filled porosity of the sludge-amended treatments at the Brookdale trial. The values quoted are the mean of four replicates. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}).

Time	Matric Potential (-kPa)	Treatment 0	80	320	1280
February 1999	5	0.139	0.227	0.179	0.221
	10	0.186	0.263	0.227	0.255
	30	0.249	0.315	0.276	0.294
	60	0.283	0.344	0.293	0.309
	100	0.299	0.359	0.302	0.324

Appendix 6.2 Mean air-filled porosity of the lime, gypsum and polyacrylamide treatments at the Brookdale trial. The values quoted are the mean of four replicates. L2, L10, G5, G10, P15 and P30 are treatments of lime at 2 and 10 Mg ha^{-1}, gypsum at 5 and 10 Mg ha^{-1} and polyacrylamide at 15 and 30 kg ha^{-1} respectively

Time	Matric Potential (-kPa)	Treatment L2	L10	G5	G10	P15	P30
Feb 1999	5	0.141	0.125	0.209	0.161	N.D.	N.D.
	10	0.206	0.179	0.257	0.223	N.D.	N.D.
	30	0.262	0.214	0.316	0.274	N.D.	N.D.
	60	0.273	0.236	0.347	0.294	N.D.	N.D.
	100	0.281	0.243	0.372	0.302	N.D.	N.D.
June 1999	5	0.220	0.199	0.217	0.203	0.224	0.229
	10	0.267	0.244	0.266	0.246	0.270	0.273
	30	0.309	0.280	0.308	0.283	0.297	0.315
	60	0.326	0.299	0.322	0.294	0.312	0.328
	100	0.337	0.306	0.333	0.311	0.315	0.332
Nov 1999	5	0.171	0.165	0.164	0.179	0.214	0.205
	10	0.199	0.211	0.210	0.211	0.254	0.238
	30	0.279	0.270	0.266	0.260	0.293	0.294
	60	0.310	0.308	0.291	0.277	0.325	0.322
	100	0.498	0.325	0.305	0.295	0.332	0.338
April 2000	5	0.174	0.131	0.169	0.182	0.187	0.165
	10	0.229	0.195	0.223	0.232	0.245	0.221
	30	0.264	0.243	0.265	0.284	0.292	0.264
	60	0.290	0.268	0.297	0.319	0.320	0.294
	100	0.298	0.274	0.316	0.350	0.331	0.309
June 2000	5	0.171	0.106	0.182	0.181	0.165	0.171
	10	0.216	0.141	0.215	0.223	0.204	0.223
	30	0.282	0.236	0.264	0.281	0.262	0.272
	60	0.305	0.255	0.307	0.299	0.279	0.291
	100	0.324	0.265	0.320	0.315	0.287	0.297

Appendix 7.1 (a) Mean weight diameters (mm) calculated from the dry aggregate size distribution of the treatments investigated at the Brookdale field trial. The values quoted are the mean of three replicates.

Time	Sludge Content (Mg ha^{-1})				Lime and Gypsum Application (Mg ha^{-1})				Polyacrylamide Application (kg ha^{-1})	
	0	80	320	1280	L2	L10	G5	G10	P15	P30
February 1999	8.59	4.61	6.68	5.25	10.92	9.96	10.12	6.58	4.33	4.69
September 1999	4.19	4.12	5.47	4.33	3.54	4.02	5.20	5.43	3.53	3.75
November 1999	7.93	6.89	7.67	6.50	6.96	5.09	6.70	4.82	3.63	3.99
February 2000	3.16	4.35	2.64	4.93	3.04	3.99	3.66	3.40	2.70	2.90

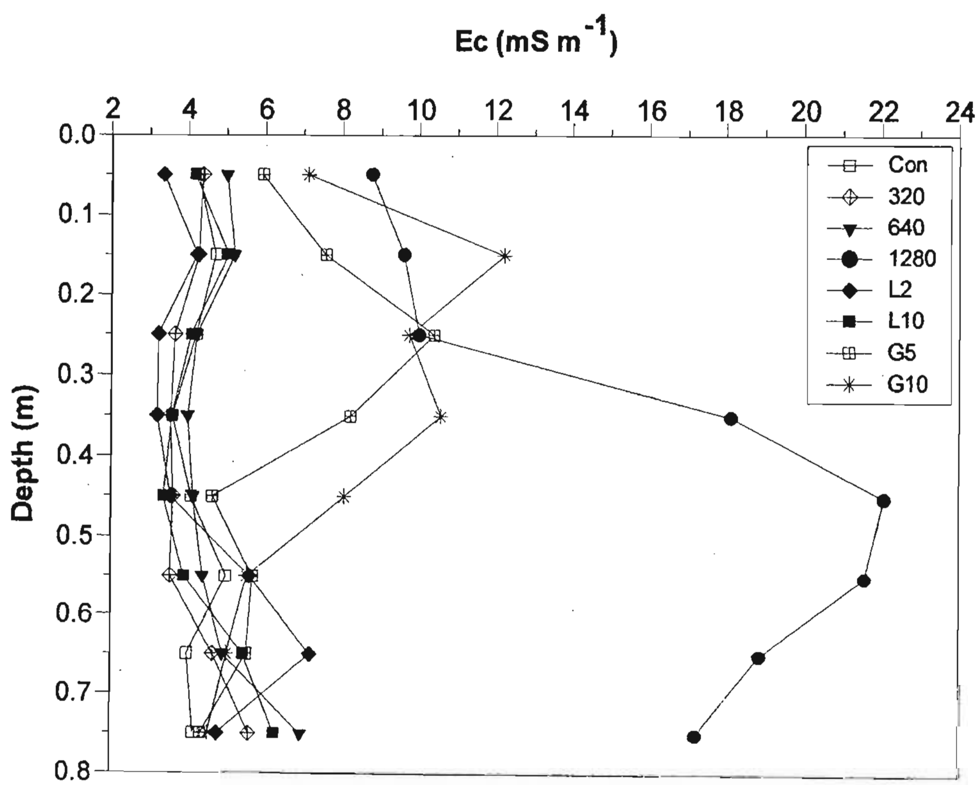
Appendix 7.1 (b) Mean weight diameters (mm) calculated from the dry aggregate size distribution of the treatments investigated at the Ukulinga field trial. The values quoted are the mean of three replicates.

Time	Sludge Content (Mg ha^{-1})				Lime and Gypsum Application (Mg ha^{-1})		PAM Application (kg ha^{-1})
	0	80	320	1280	L10	G10	P30
November 1999	4.50	3.81	3.71	5.44	3.75	3.31	N.D.
April 2000	4.02	4.30	4.44	5.52	4.78	3.43	N.D
October 2000	4.60	3.80	3.58	4.85	3.96	3.58	4.28

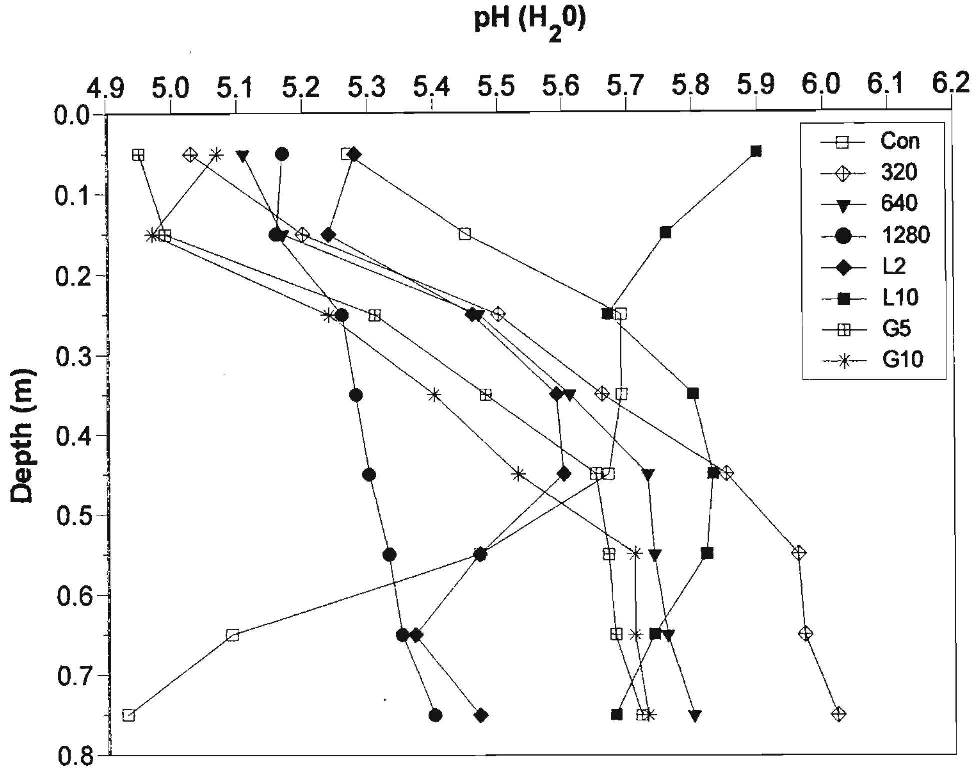
Appendix 7.2 Mean weight diameters of water stable aggregates for the treatments measured at the Brookdale trial during September 1999 and February 1999. Each value represents the mean of three replicates. 0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L2, L10, G5, G10, P15 and P30 are treatments of lime at 2 and 10 Mg ha^{-1}, gypsum at 5 and 10 Mg ha^{-1} and polyacrylamide at 15 and 30 kg ha^{-1} respectively

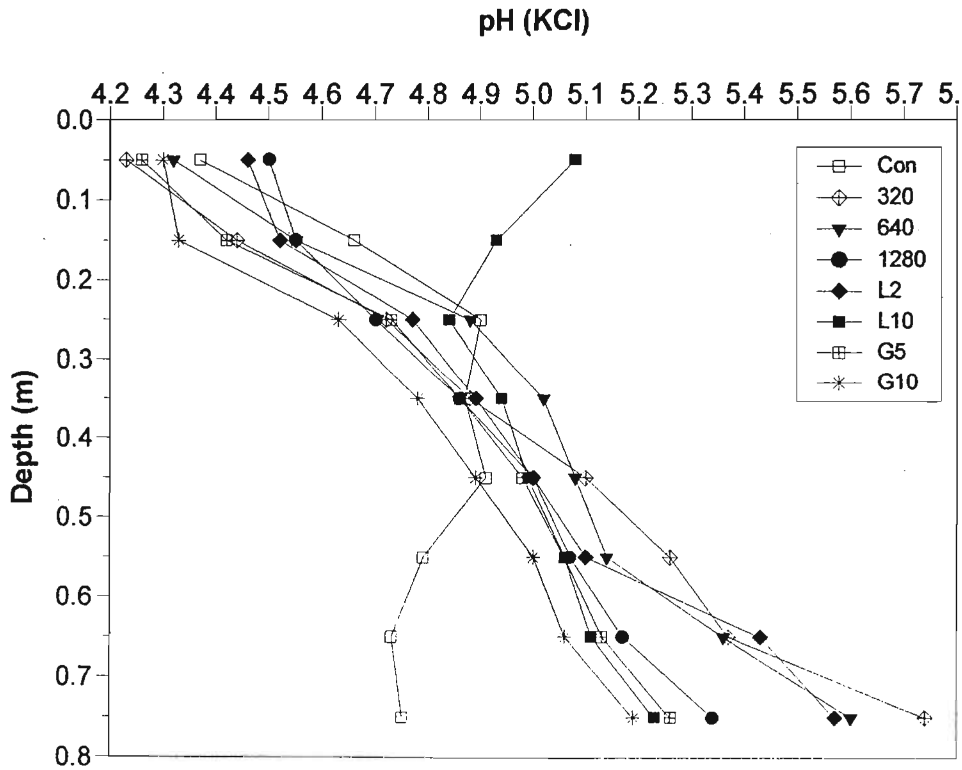
Time	Treatment									
	0	80	320	1280	L2	L10	G5	G10	P15	P30
Sep.1999	1.40	2.02	2.20	2.10	2.12	1.64	2.16	2.00	1.82	1.98
Nov. 1999	1.23	1.48	1.85	1.89	1.26	1.25	1.45	1.43	1.38	1.89

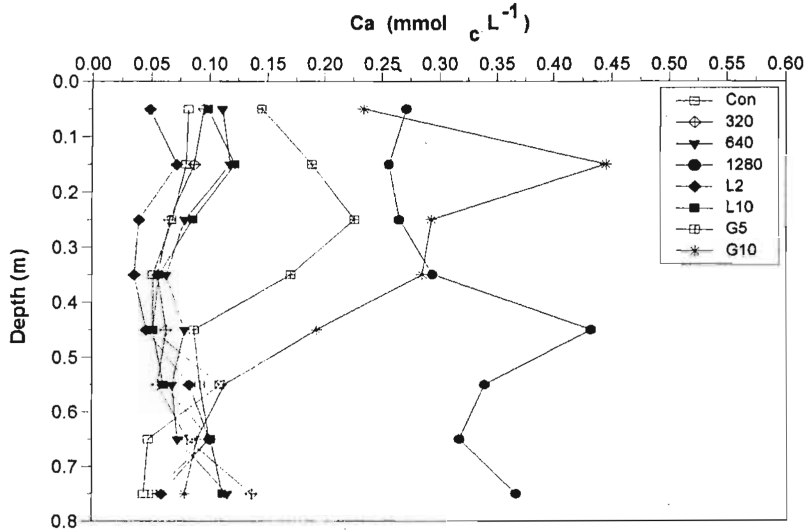
Appendix 7.3 Soil solution composition (1:5 extract) for selected treatments at the Brookdale trial.0, 80, 320 and 1280 refer to sludge levels (Mg ha^{-1}). L2, L10, G5, G10, P15 and P30 are treatments of lime at 2 and 10 Mg ha^{-1}, gypsum at 5 and 10 Mg ha^{-1} and polyacrylamide at 15 and 30 kg ha^{-1} respectively

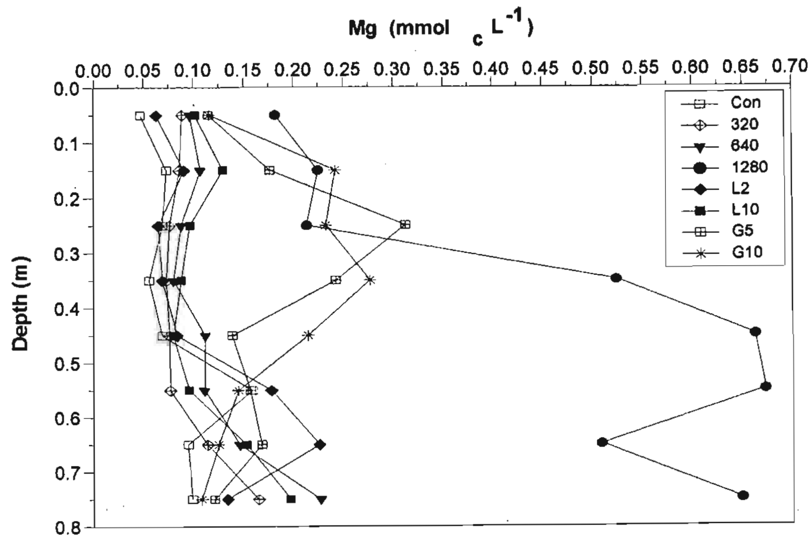


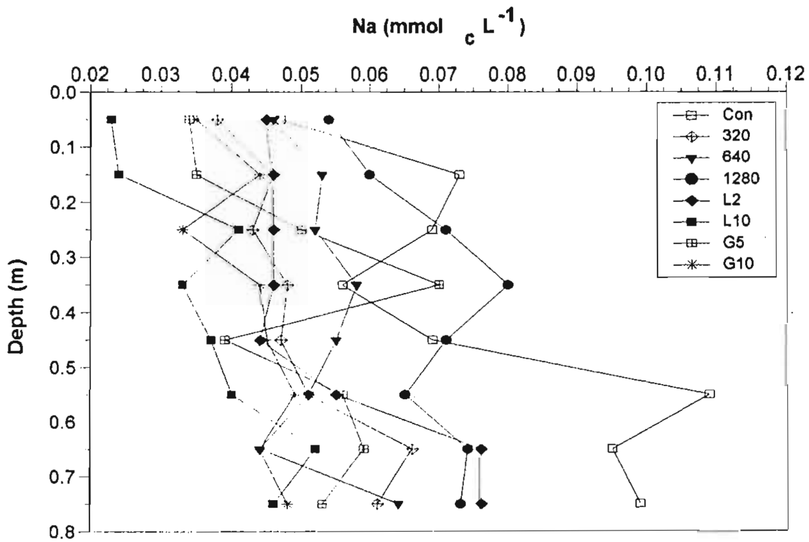
Appendix 7.3 (continued)

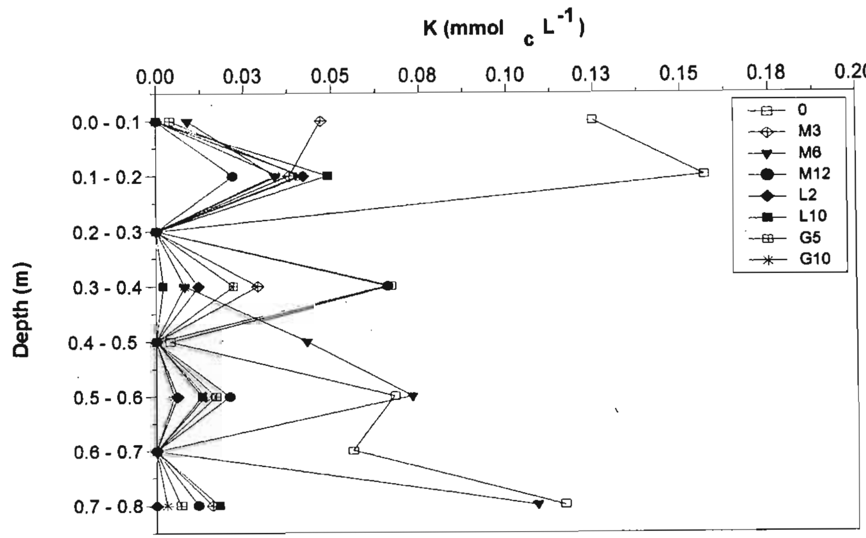




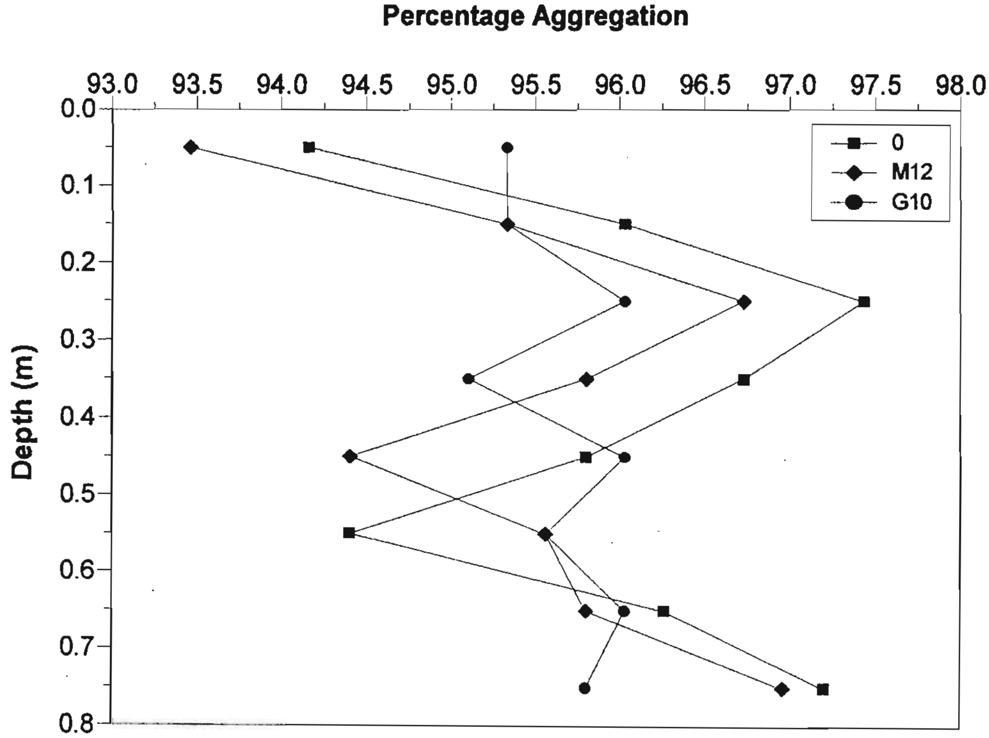

Ca (mmol c L-1)
Depth (m)
Con
320
640
1280
L2
L10
G5
G10


Mg (mmol c L-1)
Depth (m)
Con
320
640
1280
L2
L10
G5
G10

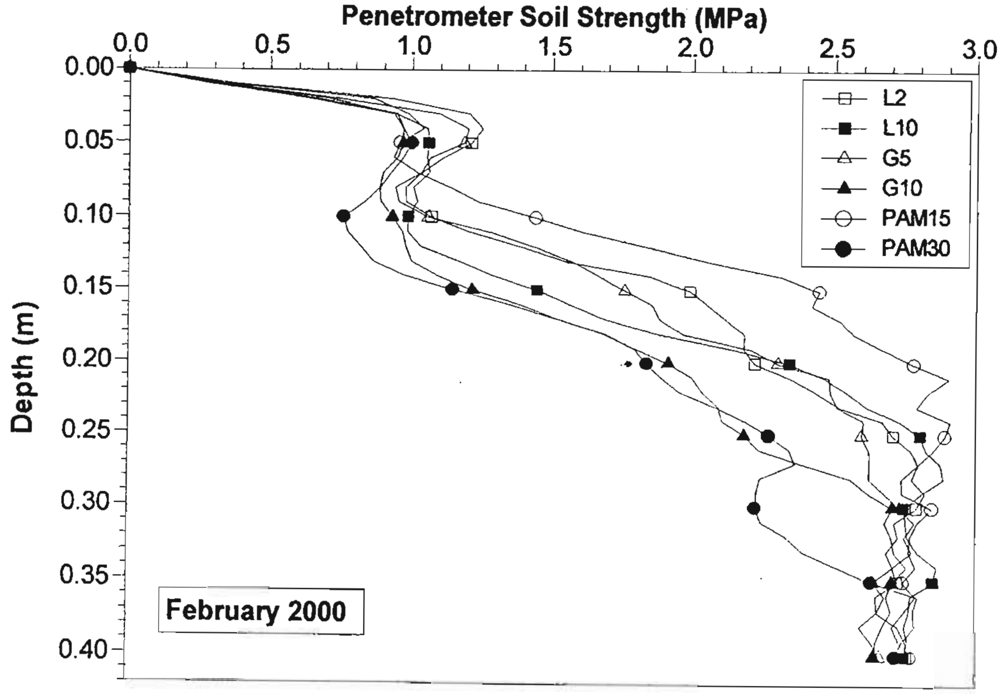

Na (mmol c L-1)
Depth (m)
Con
320
640
1280
L2
L10
G5
G10


K (mmol c L-1)
Depth (m)
0
M3
M6
M12
L2
L10
G5
G10

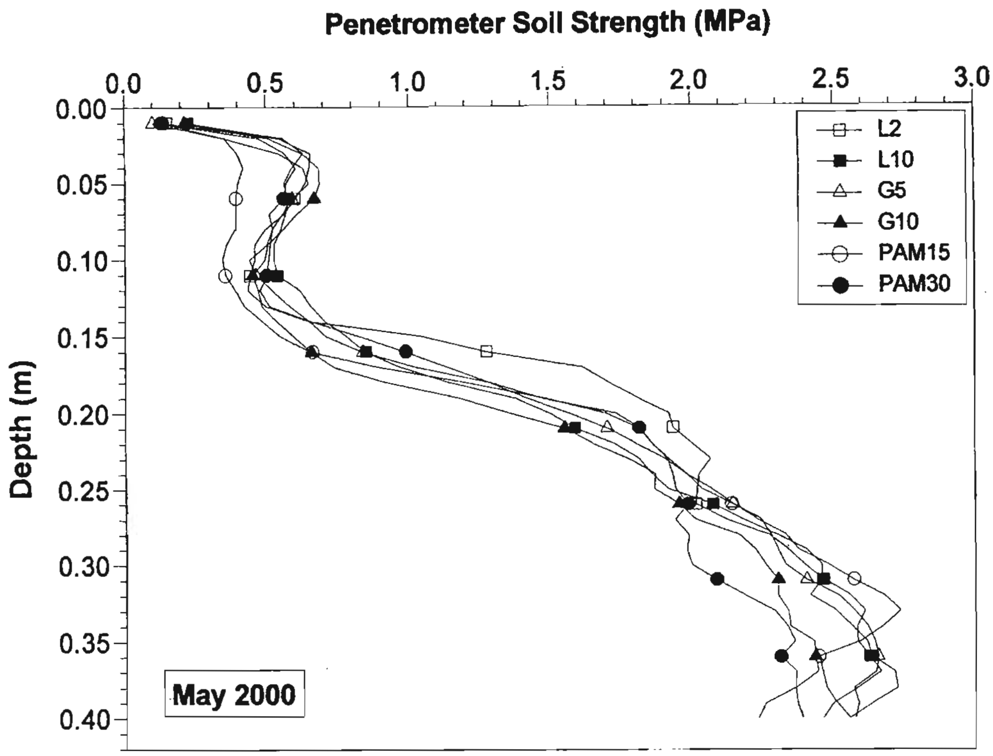
Appendix 7.4 Aggregation of the Hutton soil at the Brookdale trial as influenced by the leaching of electrolytes from the amended soil depth. The control, M12 and G10 treatments are shown.

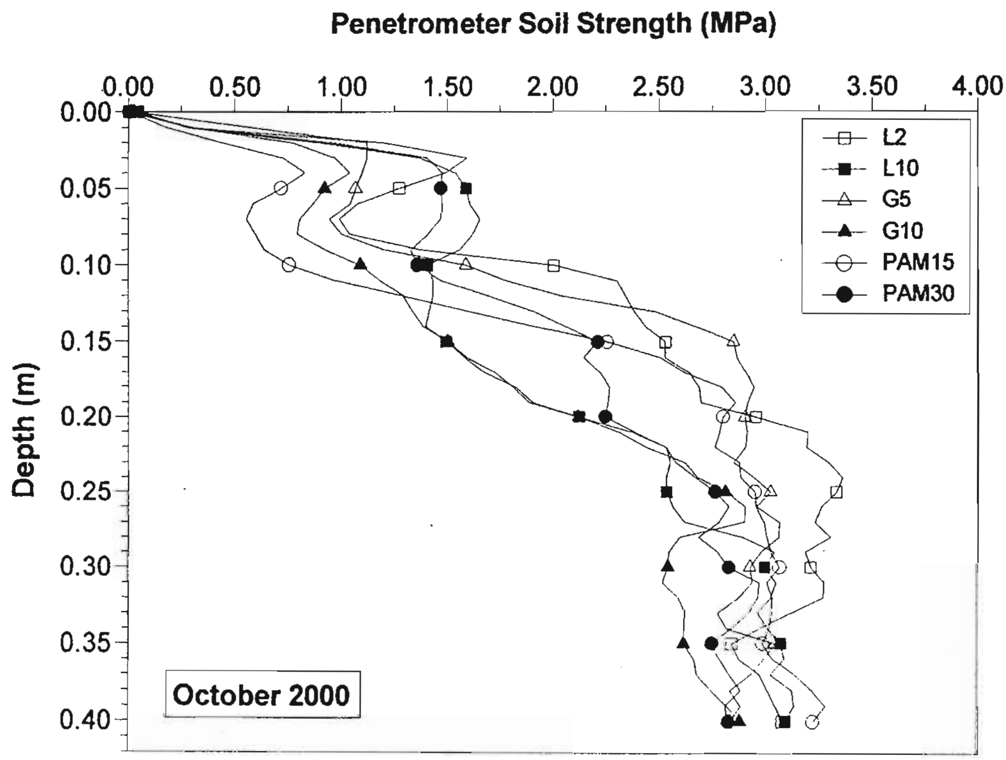


Appendix 8.1 Penetrometer soil strength profiles for the gypsum, lime and polacrylamide treatments at the Brookdale Trial.



Appendix 8.1(contd.)





Appendix 9.1

Interim Management Guidelines for the land disposal of Water Treatment Sludge at Brookdale Farm, Howick, KwaZulu-Natal

1. Introduction

Water treatment sludge (WTS) being produced at the Midmar Water Treatment Works consists of essentially flocculated silt, clay and organic matter. At its exit from the plant the sludge has a content of solid material ranging from 24 to 28% (ie 24 - 28 kg per 100 m^3 of sludge). At present the sludge is transported to Brookdale Farm in a tractor-drawn trailer and deposited in heaps for air-drying. Each trailer can carry approximately 5.25 m^3 wet material.

Initially, drying of the material is rapid but declines dramatically as a crust develops on the sludge heaps. Complete air drying of the material takes approximately four to six months at which time the sludge has approximately 0.057 kg water per kg dry solids. In this state the material is mainly of a blocky nature but with repeated wetting and drying breaks down into smaller sized aggregates. Depending upon seasonal influences the further breakdown of the sludge into a grade suitable for its further disposal can take another 3 to 5 months. This means that in the present scheme of operation, a section of the farm has to be used as a temporary sludge storage facility. If the volume of sludge produced from this plant increases, as is expected, then in the present system substantially larger areas of the property will have to be reserved for the intermediate storage of the sludge.

1.1 Alternative mode of sludge disposal

An alternative and potentially superior method of land disposal of the sludge is to apply the material "wet" directly at its point of incorporation. Were this system to be followed, advantages over the present system of operation are:

- It would eliminate the need for a temporary sludge drying area on the property as the material will be disposed off directly at the point of its incorporation with the soil.

- Double handling of the material would be avoided, and therefore lower operating costs may be expected.
- Higher levels of uniformity in the distribution of the sludge would be possible, which is an important factor in the successful operation of the land disposal scheme.

A demonstration model "muck" spreader supplied by Falcon Equipment Ltd, Howick (Figure A) has been tested for this mode of sludge disposal. During operation of the machine, flails (long chains with steel plates) attached to a central rotating shaft ejects material from the load bin (Figure B). The discharged material is usually deposited parallel to the vehicle tract in an approximately 3m wide band.

1.2 Calibration of the "muck spreader"

The rate at which the sludge is deposited by the "muck" spreader depends upon the speed of the tractor (the faster the travel, the lower is the application rate). Consequently, it was important to derive a relationship between the operating parameters of the tractor and the level of sludge application. In this way only the tractor operating speed needs to be considered in the actual disposal of the sludge, which is especially convenient from a practical point of view. To achieve this, a tractor (Ford F5000) towing a fully laden "muck spreader" was driven along a preselected route with the machine in operation. Three different tractor speeds were tested, namely 1st gear, 2nd gear and 3rd gear with the engine speed set at a constant 2000 revs min^{-1}. After each run, the mass of material deposited (on plastic sheeting suitably positioned beforehand) was measured, and adjusted to its air-dry equivalent. The results of this exercise are given in Table 1 for a single pass.

Table 1. The quantity of sludge applied at the various tractor settings tested. The engine speed was set at a constant 2000 revs min^{-1}.

Water Content	Operating parameters of the tractor		
	1st Gear	2nd Gear	3rd Gear
Wet	7.6 kg m^{-2}	4.2 kg m^{-2}	3.5 kg m^{-2}
Air dry equivalent	1.14 kg m^{-2}	0.64 kg m^{-2}	0.52 kg m^{-2}

The best coverage of sludge was obtained with the tractor operating in 1st gear. Figure C shows the type of coverage achieved immediately after the sludge was applied, and Figure D shows the same area three days later. Since a relatively thin band of material was deposited, within two weeks of its application the material was completely air dry. This is an immediate advantage over the system of letting the sludge dry in heaps because of the much faster drying time. Moreover, a more uniform grade of air-dry material is obtained which should make for much easier incorporation with the soil. A full load ($5.25m^3$) with the tractor operating at 2000 r.p.m in 1st gear will cover an area of 690 m^2 (230m x 3m band) at a sludge application rate equivalent to 11.4 Mg ha^{-1} (air dry).

2. Farm Plan

The areas within Brookdale Farm suitable for sludge disposal was identified. The Farm boundaries and contours were first digitised from a 1:5000 sheet. The soils of the area were then surveyed and classified according to soil form and family, soil wetness, depth and rockiness(Soil Classification Working Group, 1991). Thematic maps were generated for each attributes under consideration using the Arc-Info software programme (Figures E to J). Areas considered too wet, too rocky or too shallow were eliminated from the total area of the property (197.6 ha^{-1}). A final map showing only those areas suitable for sludge disposal, from now on termed the "useable area" was then generated. The area of each field within the useable area was calculated, by superimposing the existing field boundaries of the property. For convenience the useable area is divided into five main sections and the fields within each of these sections ascribed a code (Table 2). The location of these fields is shown in Figure E.

Table 2. The area of the fields (ha) suitable for sludge disposal as shown in Figure E.

Field	Section A	Field	Section B	Field	Section C	Field	Section D	Field	Section E
A1	0.5	B1	12.4	C1	10.9	D1	8.2	E1	8.9
A2	5.2	B2	0.4	C2	1.3	D2	1.4	E2	0.2
A3	1.1	B3	0.6	C3	1.9	D3	1.5	E3	0.4
A4	10.1	B4	2.0	C4	2.5	D4	1.3	E4	1.9
A5	2.3	B5	2.0	C5	0.6	D5	1.5	E5	3.7
A6	0.8	B6	2.8	C6	0.6	D6	1.7	E6	2.9
A7	1.2	B7	1.1	C7	0.6	D7	1.4		
A8	0.9	B8	2.1	C8	0.6	D8	0.8		
A9	1.8	B9	2.1	C9	0.8	D9	0.5		
A10	1.8			C10	1.5	D10	3.1		
A11	2.4								
A12	0.8								
A13	1.6								
A14	1.9								
A15	1.2								
A16	2.3								
A17	3.8								
Total (Incl. (Pasture)	39.7		25.5		21.3		21.4		18.0
Total (Excl. Pasture)	**37.4**		**13.1**		**21.3**		**21.4**		**0.0**

2.1 Sludge Disposal Plan.

<u>Option 1.</u> Using the total useable area (including pastures) and application by a "muck" spreader.

For clarity some of the calculations are repeated here.

A trailer carries 5.5 Mg of wet sludge and deposits 8 kg m^{-2} (80 Mg ha^{-1}) of this material in a 3 m band, 230m long. This means that in a single run the sludge is deposited across 690 m^2, at a rate of 11.4 Mg ha^{-1} (air-dry). If the total useable area is used (126 ha), 1826 trailer loads can be applied before the same area would have to be covered (return cycle). Given the current production level (430 trailers per year) a return cycle will take 4.2 years, assuming a single application of 11.4 Mg ha^{-1}.

Although from a soils perspective the areas cropped to pasture (A5, B1 and E) is suitable for sludge disposal, it is perhaps not advisable to use these areas until the longer term effects of the sludge are identified. Block E is also furthest from the point of entry of the sludge onto the farm and so from an economic viewpoint could be excluded. In any event, as will be shown, inclusion of these sections is not very important as yet. In this regard the useable areas (less the pastures) for each section which total 93.2 ha are given in Table 2.

<u>Option 2.</u> Using the total useable area excluding the pastures and a "muck" spreader.

The calculation proceeds as given above. The number of trailer loads that can be applied before the same area would have to be covered is 1352 and the time for a return cycle is 3.14 years.

<u>Option 3.</u> - Preferred Option

Findings to date suggests that this application rate of 11.4 Mg ha^{-1} can be safely exceeded since non-significant differences in soil physical properties were encountered even when the sludge was applied at a rate of 320 Mg ha^{-1}. It has, however, been shown that the sludge does have a high P absorption capacity and in this regard using a smaller section of the farm is perhaps more prudent until longer-term trends are forthcoming.

A convenient approach is to section the property into twenty blocks, each of 4.5 ha in area and use each block exclusively for sludge disposal over 2 years. The number of trailer loads that can be applied over a single block in a single pass is 65. The fields which comprise the 4.5 ha block should be decided upon in consultation with the farm manager.

Assuming that the production of sludge is 430 trailer loads a year, the return cycle for this block is approximately 2 months. This is sufficient time for the material to air-dry completely. In a year, the rate of sludge application will be 64 Mg ha^{-1}, and over two years 128 Mg ha^{-1}. Based on these figures it is theoretically possible to use the same block for a further 5 years. This is perhaps not advisable until further longer-term records are available (especially regarding nutrient and heavy metal concentrations). For convenience the same block being used over two consecutive years is presented but this can be varied to suit the objectives of the land manager better. Once the block has received the equivalent of 128 Mg ha^{-1} sludge it is recommended that it be returned to it's former land-used and another 4.5 ha identified for further sludge disposal. It is recommended that the material be incorporated into the soil at least once a year to ameliorate the effects of compaction associated with vehicle traffic. Should the production of the sludge increase, additional blocks or a portion thereof should be included. With any rotational system of disposal such as is being recommended here it will be of vital importance that detailed records are kept by the land manager so that the amount of sludge disposed of to any particular area is known at all times.

Figure A The muck spreader that could be used for the disposal of fresh sludge.

Figure B Fresh sludge being applied using the muck spreader. Much of the material is deposited in a 3m wide band adjacent to the path of the vehicle.

Figure C The coverage of sludge obtained with the tractor operating in 1st gear. The rate of application is 7.6 kg m^{-2}. The advantage of this method is that a more uniform distribution of sludge is obtained than if the material were distributed when air dry.

Figure D The same area shown in Figure C, three days later. Since the depth of sludge that was applied is fairly thin, evaporation of water is rapid. When fully dry the rate of application is approximately 11.4 Mg ha^{-1}.

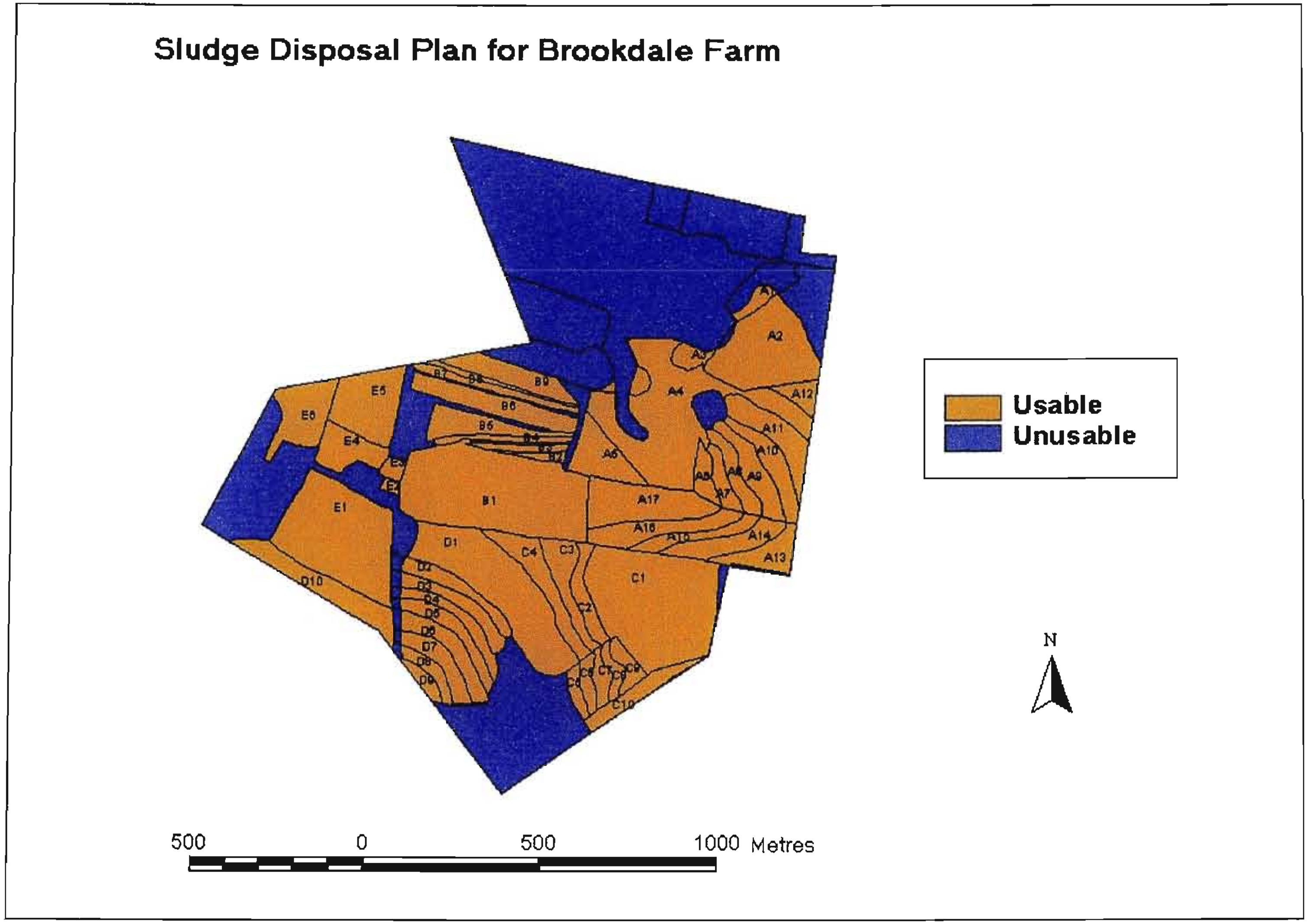


Figure E Areas at Brookdale Farm suitable for sludge disposal. The field boundaries are superimposed on the base image. For convenience each field within the useable area has been ascribed a code the area of which is given in Table 2.

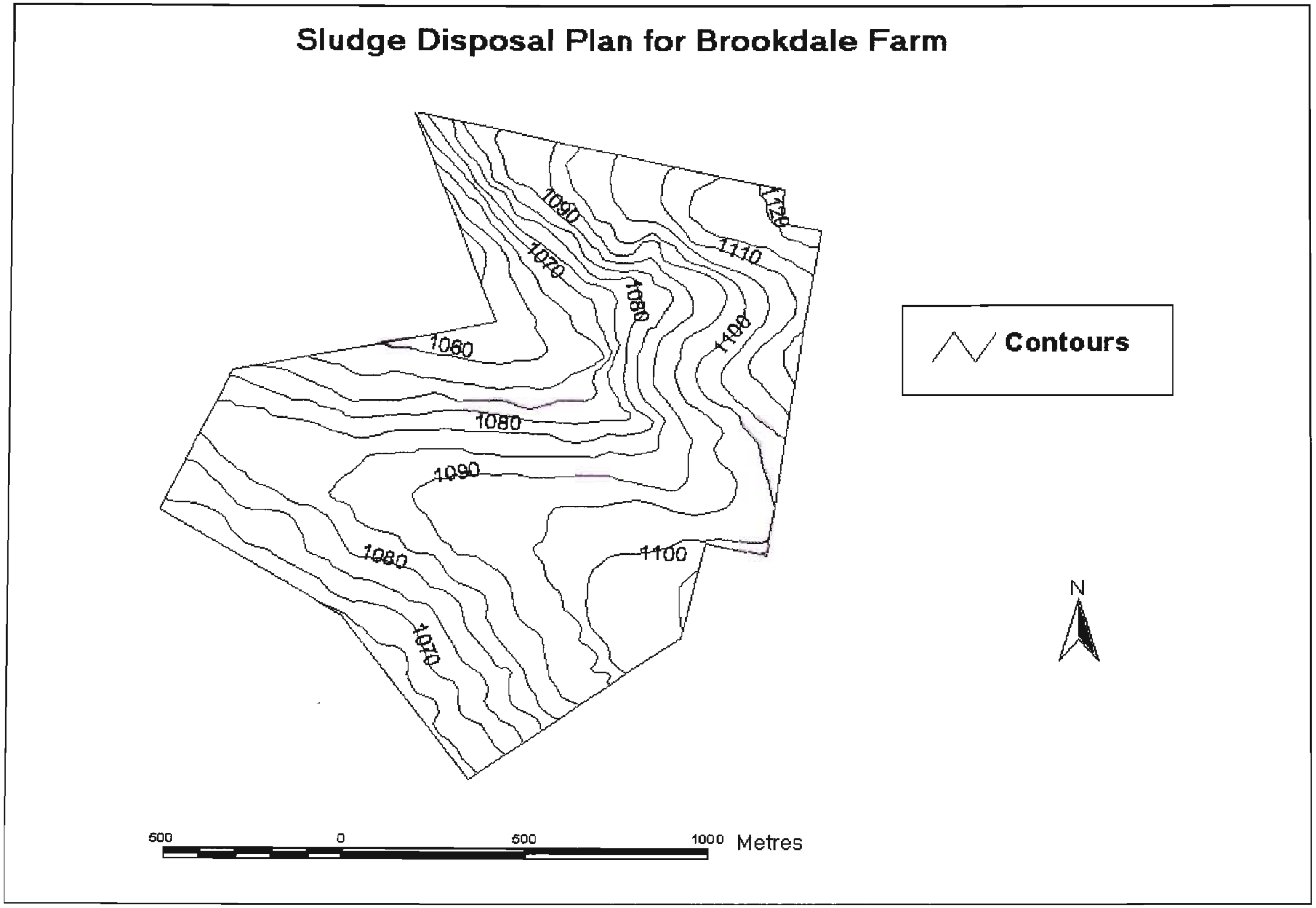


Figure F Contour map of Brookdale Farm.

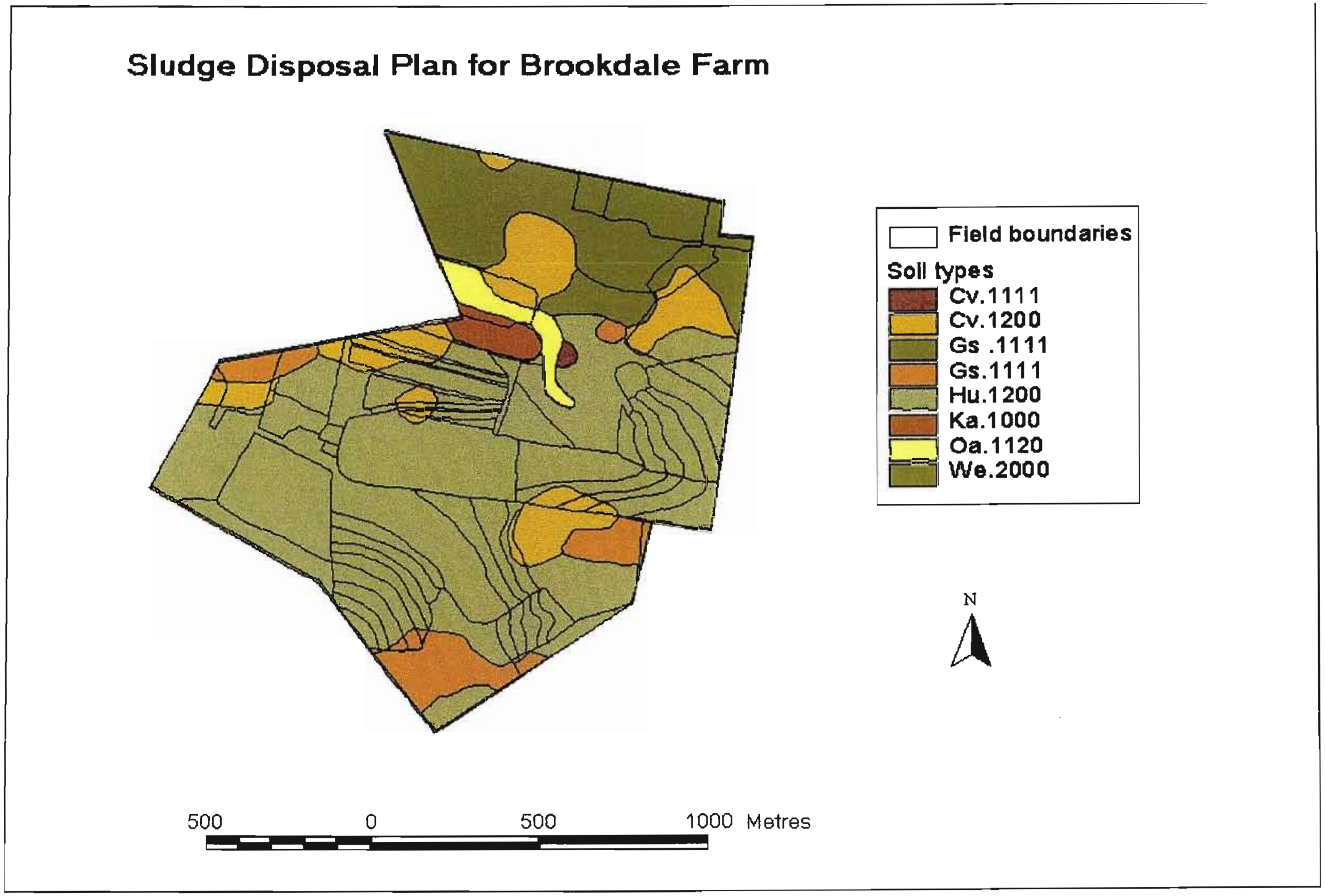


Figure G	Soils map for Brookdale Farm showing soil form and family. The field boundaries are superimposed on the base image.

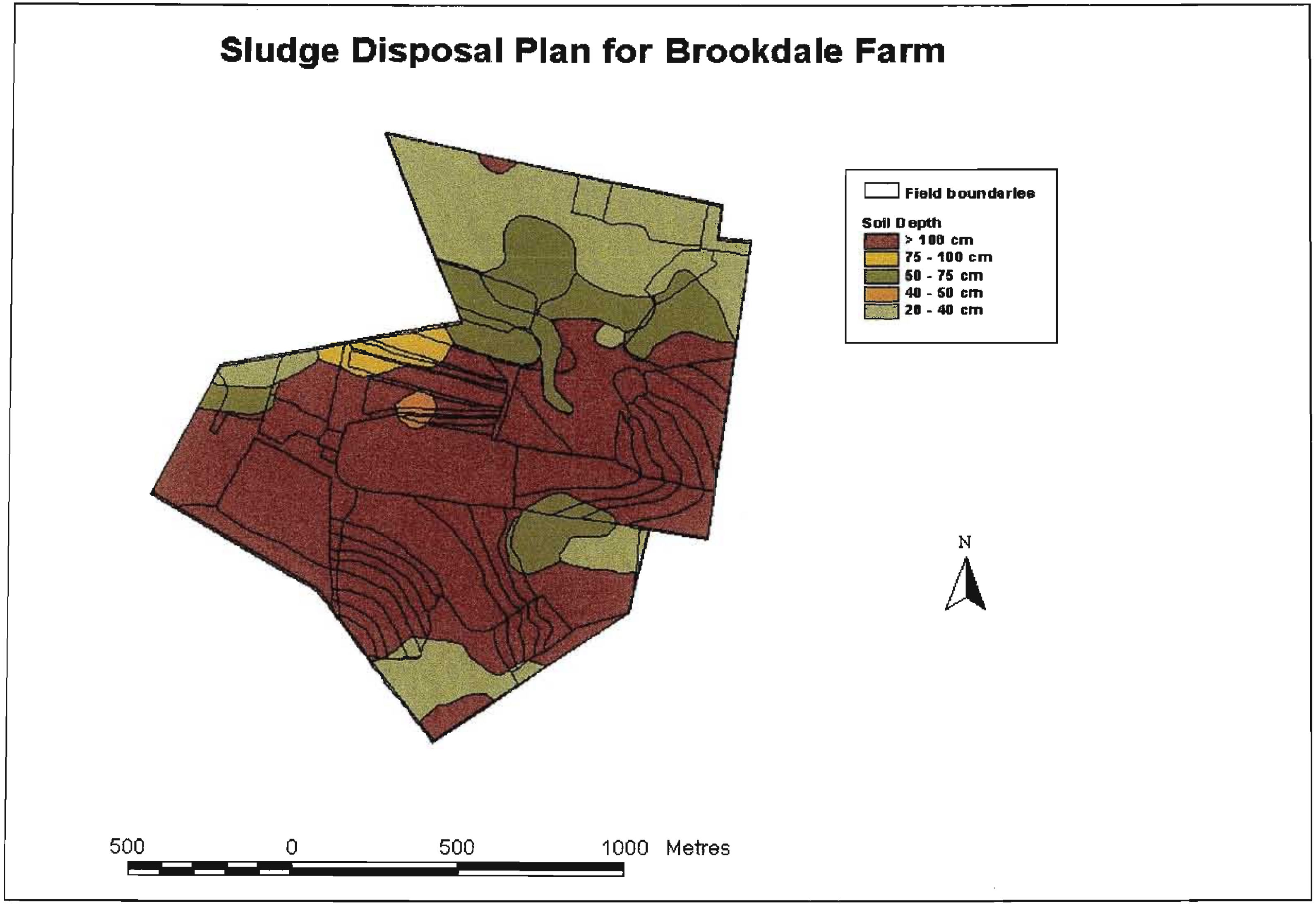


Figure H Distribution of soil depth at Brookdale Farm. The field boundaries are superimposed on the base image.

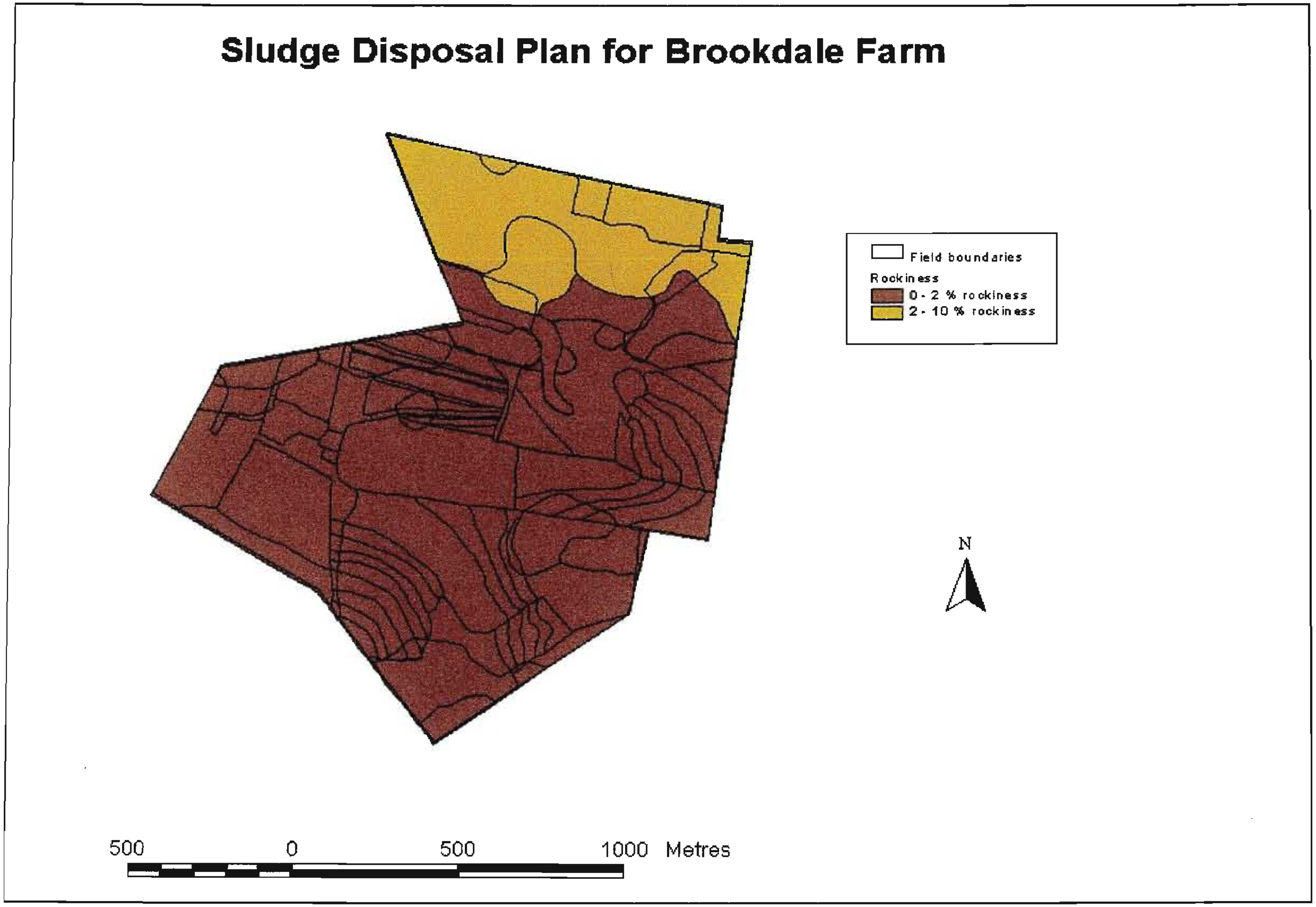


Figure I Distribution of soil rockiness at Brookdale Farm. The field boundaries are superimposed on the base image.

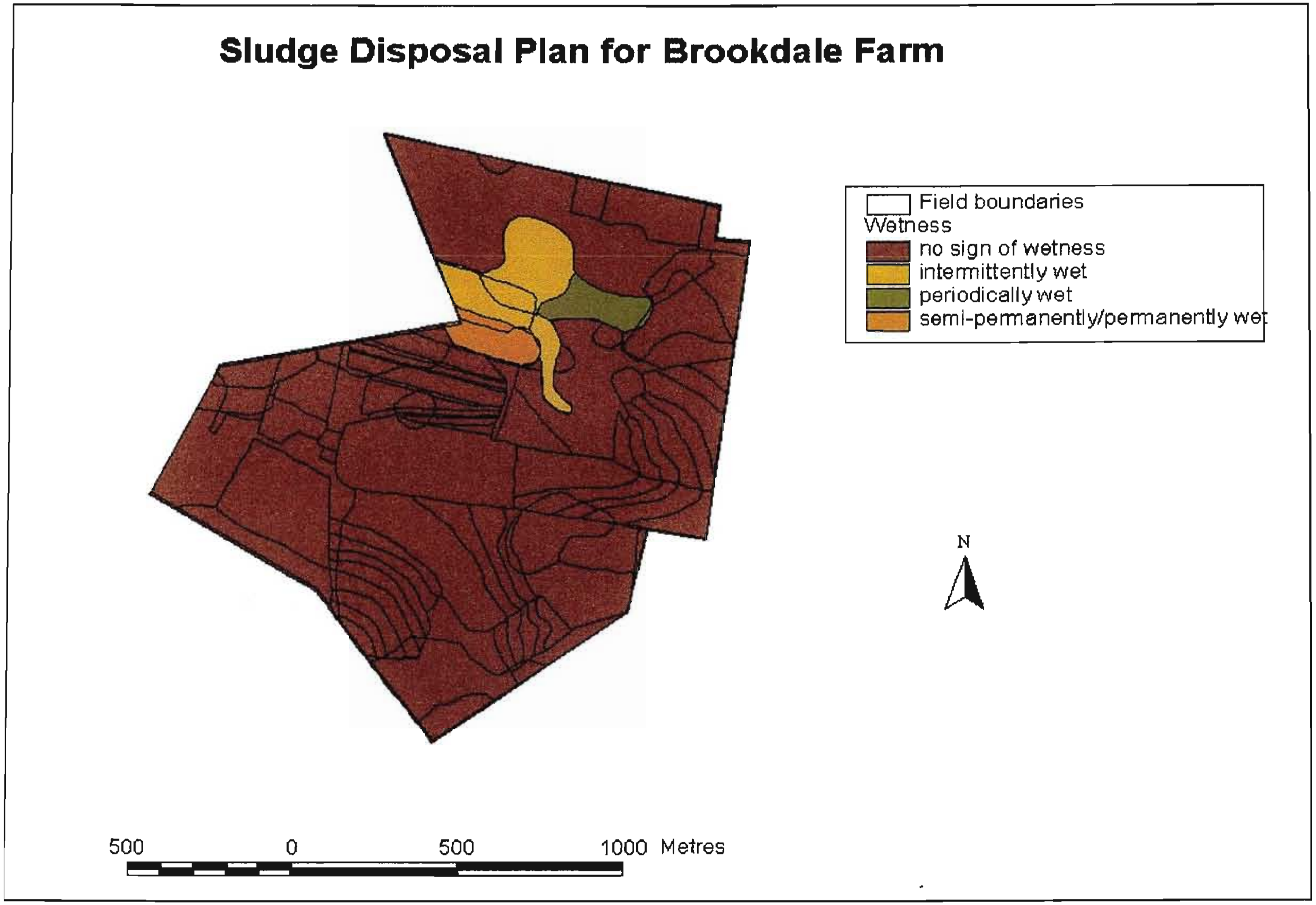


Figure J Distribution of soil wetness at Brookdale Farm. The field boundaries are superimposed on the base image.