LABORATORY AND FIELD SCALE EVALUATION OF AGRICULTURAL USE OF SEWAGE SLUDGE

by

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EXECUTIVE SUMMARY

1. BACKGROUND

Sewage sludge can be a valuable resource if used as a fertilizer and soil conditioner. South African farmers using sewage sludge as a fertilizer amendment reported a 20% increase in the yield of cultivated maize and 40% saving on inorganic fertilizer. The major benefits of sludge application are; increased supply of major plant nutrients; provision of some of the essential micronutrients (Zn, Cu, Mo, and Mn) and; improvement in the soil physical properties, i.e. better soil structure, increased water holding capacity, and improved soil water transmission characteristics.

Toxic compounds such as heavy metals and pathogens could compromise the beneficial use of sewage sludge. To minimise the risk of toxic effects and environmental contamination a “Guide: Permissible utilization and disposal of sewage sludge” was developed. It is therefore critical to establish the safe application rate of sewage sludge in different environmental conditions. Furthermore, with repeated sludge applications as soil conditioner, these heavy metals may accumulate in the soil to phytotoxic concentrations for crops, although at certain concentrations the metals may be deficient for crop growth.

At present, the sludge producers and legislators are hesitant to encourage the use of sewage sludge in agriculture due to lack of knowledge of the risks involved. The sludge legislation and guidelines were largely adapted from other countries and often based on theoretical risk assessments. The behaviour of sewage sludge applied to agricultural land has not been extensively researched under South African climate conditions. The quantification of the risk associated with the use of sewage sludge in agricultural practices will necessitate a long-term study under several field conditions. This research aimed to understand the behaviour of sewage sludge once applied to agricultural soil under South African conditions in the short-term.

2. OBJECTIVE OF THE STUDY

The main objectives of the study were to:

- Establish the extent of metal uptake in plants grown on sewage sludge-amended soils including winter and summer crops
- Establish the effect of soil properties on the mobility of nutrients and metals
- Establish a safe sludge load to prevent nitrogen leaching to groundwater
- Predict the persistence of sludge-borne pathogens during agricultural application
- Establish the perceptions of farmers, commercial markets and urban and rural communities for beneficial use of sewage sludge for edible crops.

3. SUMMARY OF THE MAJOR RESULTS AND CONCLUSIONS

3.1 Extent of metal uptake in plants grown on sewage sludge

Heavy metal fractions from sacrificial land disposal sites

This study did not aim to address the characteristics and behaviour of soils that have been subjected to high doses of sewage sludge for a long time. However, the short term agricultural applications used in this research indicated no toxic effects to soils or plants. Sites where sewage sludge had been applied at current guideline levels for a period of several years are extremely scarce. This is especially the case for high metal sludges. The three sites discussed were identified to determine the different metal fractions in un-limed and limed soil after long-term sludge application. The aim was to determine the influence of management practices (regular liming versus no lime additions) on the various metal fractions. In South Africa large volumes of sewage sludge with high metal levels are often disposed of on “sacrificial” lands, some of which have received prolonged regular applications of sewage in suspension form. In the Gauteng province around Pretoria three soils: a gravely sandy loam (Soil 1) on granitic parent material, a sandy clay loam (Soil 2) and a loam soil (Soil 3) on dolomitic parent material were collected for investigations into their accumulated heavy metal levels. Soils 1 and 3 had no additions of lime and were both very acid ($pH_{(Water)}$ of 4.0). Soil 2 received regular additions of lime ($pH_{(Water)}$ of 6.5) and was used as an agricultural soil. Four different extraction procedures were used namely: a saturated paste extract (water-soluble metals); $BaCl_2$ (exchangeable metals); NH_4-EDTA (potentially plant available metals) and EPA 3050 digestion (total metal content). Appreciable quantities of different heavy metals and organic material have accumulated over time in these soils. The EPA 3050 digestion indicated that Zn, Pb, Cu, Cd, Cr, Ni and V accumulated to levels above 100 mg kg^{-1} in the three soils. Cu, Pb and Zn levels in excess of 10 mg kg^{-1} were extracted with EDTA; Cu and Zn levels of more than 5 mg kg^{-1} with $BaCl_2$ and Zn and Ni levels above 0.5 mg kg^{-1} with water. In Soil 1 and 3 (pH 4) EDTA and $BaCl_2$ extracted similar levels of metals in most cases whereas EDTA extracted significantly more than $BaCl_2$ in Soil 2 (pH 6.5).

To assess the behaviour of these sacrificial soils, two sacrificial soils were twice incubated with a total lime equivalent of 45 t ha^{-1} in pots but did not attain the desired pH of 6.5 due to a very high buffer capacity. Soil samples from the pots after the incubation were extracted with NH_4-EDTA and $BaCl_2$ and the levels of Al, Fe, Mn, Cu, Zn, Pb and Cd determined by Atomic Absorption Spectrophotometry. The $BaCl_2$ extractable Mn, Pb, and Cd in Soil 1 and Mn and Cd in Soil 3 indicated increases or similar levels in extractability after liming and Al, Cu, Fe and Zn levels decreased after liming. The EDTA extractable Cu, Mn, Fe and Cd in both soils and Pb in Soil 3 increased after liming and Al, Zn and to a lesser extent Pb in Soil 1 decrease in extractability. The increased extractability of certain of these metals is contrary to what was expected from literature concerning the effect of liming and is the subject of further investigation. This aspect is a cause for concern in the short-term if liming is to be done on acidified sacrificial lands as a method of rehabilitation.

Extent of metal uptake in plants grown on sewage sludge-amended soils including winter and summer crops on three different soil types (Greenhouse trials)

The potential impact of the four main sludge-borne metals (Pb, Cd, Zn and Cu) was monitored in the research (glasshouse and field experiments) when sludge was applied to agricultural soils, taking into consideration the current S.A. guidelines.

Research was done in glasshouses on maize (*Zea mays* L.) (summer crop) and oats (*Avena sativa* L.) (winter crop), grown on different soil types (clay, loam, and sand) at a specific sewage sludge application rate (24 t ha^{-1}) using two different sludge types (low metal and high metal) over a period of 28 d. Poor sample homogenisation caused invariable results. Availability of sludge-borne metals differed between sludge types. The heavy metals were less available in the high metal industrial sludge compared to the low metal domestic sludge. The accumulation of sludge-borne metals in soil could not be proven to be in excess, even at a high application rate (24 t ha^{-1}). Furthermore, accumulation of heavy metals in seedlings did not reach phytotoxic levels. A significant increase in certain yield aspects was seen after sludge amendment to the different soil types, especially in the low metal sludge treatment.

Extent of metal uptake in plants grown on sewage sludge-amended soils including winter and summer crops (Field trials)

Field experiments on maize and oats using different total application rates (4 t ha^{-1} and 8 t ha^{-1} dry sludge for oats cultivation; and 12.5 t ha^{-1} and 25 t ha^{-1} for maize cultivation) of the low metal sludge were also completed. Difficulty in sampling was evident and possible errors in sample taking and/or analyses caused results that were difficult to interpret. No phytotoxic levels of metal accumulation were observed in the different plant parts of the crops. The sludge treatment plots compared well with plots where inorganic fertilizer (Positive Control) was added, when yield differences were calculated.

In the field experiments, no significant differences in yield were found between sludge-amended plots and the control treatments, although the number of ears per plant was significantly increased for maize plants after sludge amendment at 4 t ha^{-1}. The insignificant difference in yield between treatments was possibly due to the varying environmental conditions (e.g. hail during maize field experiment, and drought during oats field experiment) and change in soil conditions (e.g. soil pH controls availability of metals and nutrients). However, under more stable conditions in the glasshouse, a significant increase in yield (dry mass and shoot length) of crop seedlings was found. This was possibly due to the increased organic and nutrient status of the soil. A 50 and 20% increase in the yield of maize seedlings occurred when grown in the low metal and high metal sludge-amended soils, respectively, when compared to the Positive (soil amended with inorganic fertilizer) and Negative Controls (soil left unamended). However, when yield of oats seedlings was calculated on the sludge-amended soils, compared to the control treatments, the increase was 20 and 48% for the low metal sludge treatment compared to the Positive Control and Negative Control, respectively. No significant increase occurred in the yield of oats seedlings grown in the high metal sludge-amended soil compared to the controls. Insignificant differences occurred in the yield of seedlings between soil types, although the yield of seedlings in the loamy soil was higher.

The value of sludge as a soil conditioner and fertilizer was seen in the experiments although long-term experiments under field conditions still need to be done to assess possible accumulation of heavy metals in agricultural soils.

3.2 Nitrogen

To quantify the nitrogen added to soil through sludge application, it is necessary to quantify the total nitrogen applied through the sludge and the nitrogen release rate. Nitrogen losses in agricultural soils result through crop removal, leaching, surface run-off, gaseous losses through NH_3 volatilisation, denitrification and erosion (Jarvis *et al.,* 1996). The loss of nitrogen from agricultural soils is of concern due to the possible NO_3^- leaching from cultivated soils that is responsible for increasing concentrations of NO_3^- in surface and groundwaters (Cheshire *et al.,* 1999).

In this study, the nitrogen-mineralisation rate of sewage sludge, at different application rates, was quantified under South African conditions and compared to the release of inorganic nitrogen from commercial fertilizer such as limestone ammonium nitrate (LAN).

The results indicated that the addition of sludge stimulates microbial activity and the subsequent release of NH_4^+-N. The maximum extractable NH_4^+-N content was observed after 7 d when it started to decline and after day 28, the total NH_4^+-N in the soil was depleted. Autotrophic microorganisms' activity increased from day 7 as indicated by the extractable NO_3^--N. This process happens concurrently with the NH_4^+-N production. After 28 d the extractable NO_3^--N content reached a maximum, indicating that most of the readily mineralisable N was depleted. Treatments that received commercial fertilizer, showed an immediate increase in NO_3^--N content, and from there a gradual increase in NO_3^--N content. After the 63 incubation days, 32.5% of the total N content of the 10 t ha^{-1} sludge treatment were mineralised. The higher application rate gave higher values, while the lower application rate gave lower values, indicating that higher additions of sludge lead to higher microbial activity. The maximum extractable NO_3^--N content obtained on both sludge and commercial fertilizer indicated that under ideal conditions it will take up the 28 d to mineralise and nitrify approximately 30% of the organic N in sludge and the total N in commercial fertilizer. Applied NO_3^--N from fertilizer is immediately available, while organically applied N become s available over a 28 d period equivalent to a slow release fertilizer. NO_3^--N production from the sludge does not immediately take place, it is slowly released over time. More NO_3^--N can be produced from sludge during the incubation period than from fertilizer, but being slowly released, it has a lower leaching risk and has more advantages in terms of crop production.

N from sewage sludge only became available after some time, through mineralisation. This fraction of NO_3^- can be utilised much more efficiently by crops, compared to commercial fertilizer, which is an inorganic fertilizer, and immediately available.

From an agricultural point of view, the slow release of N can hold numerous advantages. When the efficiency of commercial N fertilization is limited by factors such as high NO_3^- leaching losses, or NH_4^+ volatilisation, the use of sludge as a slow release N material may decrease the N losses and increase N availability.

Different values in terms of sludge mineralisation have been obtained, due to the large number of factors that may influence the microbial activity, such as soil type, climatic factors, and sludge type.

Possible losses due to denitrification still needs to be investigated. Accurate N balances cannot be made, due to unmeasured N loss in the gas form, during the process of nitrification. It is advisable to carry out more studies on this topic, especially on the long-term. Use of lysimeters can be beneficial to study the whole N cycle including factors such as N uptake by plants, N lost due to volatilisation and leaching.

3.3 Pathogens

In South Africa, most sewage treatment plants produce a type B sludge (WRC, 1997). A type B sludge contains pathogenic organisms. It is therefore important to understand the risk of infection due to the use of sludge on agricultural soils and for a variety of crops. For this research, a comprehensive literature survey and preliminary investigations were done as well as preliminary experiments to assess the survival of sludge-borne pathogens.

The preliminary study was done on potato (*Solanum tuberosum)* as it represents a high risk crop for the use of sewage sludge and is one of the staple foods in South Africa. It has been shown that *Ascaris* and microorganisms studied, namely faecal coliforms, *E. coli* and *Salmonella* will thrive in soil for a prolonged period of time. The presence of these microorganisms on the potato peel indicates their potential hazard to public health. Due to the limitations of the techniques used, the presence or absence of microorganisms within the core of the potato could not be established conclusively. Further analysis of potatoes using PCR will be carried out, to establish whether or not these microorganisms are capable of migrating to the core of the potato. It appears that doubling the application rate from 8 t ha^{-1} to 16 t ha^{-1} does not affect the growth of microorganisms.

3.4 Public perception

The perceptions and believes of the general public towards the agricultural use of sewage sludge is as important as the scientific evidence proving the safety and/or risks associated with the practise. For this reason, a preliminary study was done to establish the extent of knowledge and perception of a group representing the general public regarding the use of sewage sludge in agricultural practices. The population polled using a questionnaire aimed to include the man on the street, supermarkets and shops selling vegetables and farmers using sewage sludge as a soil amendment. The results of 28 questionnaires illustrated the opinions of individuals representing a household earning less than R5000/month. Only 39% of the respondents were aware of what sewage sludge was before they read the information given to them. After reading the information provided, 71% indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices. Most of the respondents were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil.

The group (29 questionnaires) polled that represented the households earning above R 5000 were more informed about the source of sewage sludge. The majority, 79%, indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices and were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil.

The perception of the buyers of fruit and vegetables for the commercial markets and shops were not adequately captured in this study. The people that filled in the questionnaires were generally not at the post level required to truly reflect the opinion of the general supermarkets and vegetable shops with regards to their willingness to purchase vegetables grown on sewage sludge amended soils.

The opinion of seven farmers who are currently using sewage sludge as a soil amendment was obtained. The farmers were all in favour of using sewage sludge as a soil amendment and none of them currently exceed the recommended dosage.

4. RECOMMENDATIONS FOR FUTURE RESEARCH

This research forms part of a broader research programme where the following aspects are being investigated:

1. The short term impact of using sewage sludge in agricultural practices;
2. Establishing the impact of long term application of sewage sludge under non-beneficial conditions;
3. Establishing the status quo of sludge qualities in South Africa (Metals, nutrients and organic pollutants)
4. Documenting the technologies and their financial implications for sewage sludge treatment

Once these documents are all published, the research teams can evaluate the need for future research.

Having said that, this specific study has identified one major research need. The long term effects of the agricultural use of sewage sludge needs to be assessed. The parameters of concern would be:

Recommended dosage for different crops and different soils to obtain maximum benefit from the sludge

Protecting the environment against pollution

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The Steering Committee responsible for this project, consisted of the following persons:

Mr HM du Plessis	Water Research Commission (Chairman)
Mr JF Taljaard	Water Research Commission (Secretary)
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LIST OF ABBREVIATIONS

(Excluding SI units)

ARC	Agricultural Research Council
B	Beginning
CEC	Cation Exchange Capacity
Cv.	cultivar
d.	days
E	End
EPA	Environmental Protection Agency
exp.	experiment
fig.	figure
ISCW	Institute for Soil, Climate and Water
HI	Harvest Index
kg ha^{-1}	kilograms per hectare
kg t^{-1}	kilograms per ton
kg m^{-3}	kilograms per cubic metre
l ha^{-1}	litres per hectare
m/m	mass per mass
mg kg^{-1}	milligrams per kilogram
mol dm^{-3}	mol per cubic decimetre
MPL	maximum permissible level
N/a	Not analysed for
N/s	Not specified
$(NH_4)_2$-EDTA	Di-ammonium ethylenediaminetetra-acetic acid
PTE	Potentially Toxic Element
S.A.	South Africa, South African
SD	Standard Deviation
t ha^{-1}	tons per hectare
US EPA	United States Environmental Protection Agency
WWTP	Wastewater Treatment Plant

CHAPTER 1: INTRODUCTION

1.1 GENERAL BACKGROUND AND INTRODUCTION

Wastewater contains a mixture of organic and inorganic solids, suspended and dissolved in water (Ross *et al.*, 1992). The treatment of wastewater invariably produces a residual, which must be disposed into the environment. Most often this residual is a semisolid, odiferous, unmanageable and dangerous material commonly termed sludge (Vesilind, 1980). The ultimate disposal of the wastewater sludge continues to be one of the most difficult and expensive problems in the field of wastewater engineering (Tchobanoglous & Burton, 1991). An estimated 28% of the sludge generated at Wastewater Treatment Plants (WWTPs) in South Africa is used beneficially (Figure 1.1). Considering that the contribution of beneficial sludge amendment in other countries like Japan, the United Kingdom and the USA are 42, 50 and 35% respectively, compared to the 28% beneficial uses in S.A., sewage sludge can be used to a far greater extent in South Africa (Environmental Protection Agency, 1993). The beneficial uses of sewage sludge under S.A. conditions include agricultural application for crop cultivation, soil reclamation in areas where mining activities take place and application in gardens.

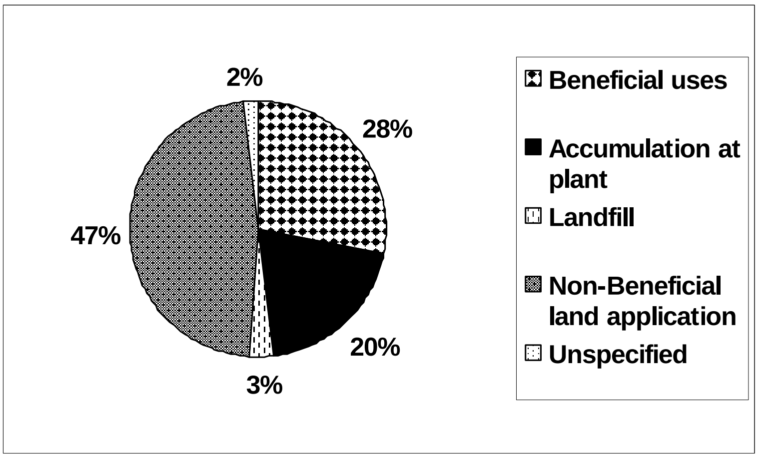

Figure 1.1. Sludge disposal in South Africa (Du Preez *et al.*, 1999).

The application of sewage sludge to agricultural soils could provide an economical way to dispose of the increasing amounts of sludge generated in the major metropolitan areas of South Africa (Korentajer, 1991), especially considering that an estimated 20% of sludges produced in S.A. is accumulated at WWTPs (Fig. 1.1).

The long-term benefits of application of sewage sludge to land are frequently limited by potentially toxic elements such as heavy metals and human pathogens (Korentajer, 1991). Concentrations of these substances originating from different sources can vary between WWTPs. Furthermore, the agricultural recycling of sewage sludge has to comply with the "Guide: Permissible Utilisation and Disposal of Sewage Sludge" (WRC, 1997) and the "Addendum to the guideline on Permissible Utilisation and Disposal of Sewage Sludge" (WRC, 2002). These guidelines deal exclusively with application of sewage sludge to land and amongst other criteria set permissible limits for concentration of heavy metals in sludge suitable for unrestricted agricultural land disposal (Snyman *et al.*, 1999).

1.2 STATEMENT OF PROBLEM

The use of sewage sludge in agriculture is limited in South Africa. The sludge producers and legislators are hesitant to encourage the use of sewage sludge in agriculture due to lack of knowledge of the risks involved. The sludge legislation and guidelines (WRC, 1997; WRC, 2002) were largely adapted from other countries and often based on theoretical risk assessments. The behaviour of sewage sludge applied to agricultural land has not been extensively researched under South African climate conditions. The quantification of the risk associated with the use of sewage sludge in agricultural practices will necessitate a long-term study under several field conditions. This research aimed to understand the behaviour of sewage sludge once applied to agricultural soil under South African conditions in the short-term.

1.3 OBJECTIVE OF THE STUDY

The main objectives of the study were to:

- Establish the extent of metal uptake in plants grown on sewage sludge-amended soils
- Establish the effect of soil properties on the mobility of nutrients and metals
- Establish the extent of metal uptake of different crops, including winter and summer crops
- Establish a safe sludge load to prevent nitrogen leaching to groundwater
- Predict the persistence of sludge-borne pathogens during agricultural application
- Establish the perceptions of farmers, commercial markets and urban and rural communities for beneficial use of sewage sludge for edible crops.

1.4 MOTIVATION

This research programme was launched to investigate: (1), the extent of metal uptake in plants grown on sewage sludge-amended soils, (2), the mobility of nitrogen, (3), the persistence of sludge-borne pathogens after the application of sewage sludge and (4), the public perception of using sewage sludge as a fertilizer. Figure 1.2 illustrates the amount of time and funds allocated to the different aspects of this research.

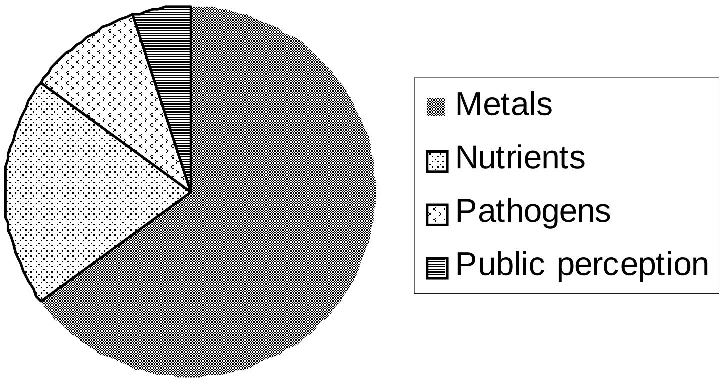

Figure 1.2. Time and funds allocated to the research aspects.

This report forms part of a larger research effort that includes the evaluation of dedicated land disposal practices for sewage sludge (K5/1209) as well as a detailed metal content survey of South African sewage sludge and an evaluation of analytical methods for metal determination (K5/1283).

1.4.1 EXTENT OF METAL UPTAKE

The research concentrated mainly on the extent of metal uptake of sludge-borne metals under South African conditions. The four sludge-borne metals of particular concern in sewage sludge in South Africa are Pb, Cd, Zn and Cu. The guideline levels (WRC, 1997) of Pb, Cd, Zn and Cu are of particular concern due to the fact that no sludge currently produced contains levels as low as those proposed (refer to WRC Report K5/1283). However, the behaviour of these metals changes under different environmental conditions. For example, different soils differ in their ability to adsorb sludge-borne heavy metals, and consequent processes like leaching or metal fixation need to be taken into consideration.

Sludges also differ between wastewater treatment plants in metal and nutrient concentration, and subsequently differ in their availability in different soil types. The ability of crops to take up these sludge-borne heavy metals also plays an important role, not only seasonally (summer and winter crops), but also through differences in specific crop cultivars (Alloway, 1995). In an effort to accommodate all these variables, the experiments incorporated three soil types (clayey, loamy or sandy), amended with sludge or commercial fertilizer. Four crops were tested; maize, sunflower, soybean and oats. The uptake of heavy metals (Cd, Cu, Pb and Zn) were measured and compared to seedlings grown in unamended soil and soil amended with commercial fertilizers. (Cadmium, Cu, Pb, and Zn were selected for the study due to the problematic nature of the guidelines – as stated earlier.) The metal transfer coefficient (Smith, 1996) between the soil and plant was calculated. These experiments were followed up with field-scale experiments on loam soil where crop yield and the fate of heavy metals were monitored. This paper does not report on the results of the field scale studies. During this study, the research focused on short-term beneficial application of sewage sludge taking into account the possible contamination risk of the four heavy metals mentioned above. However, the long-term effects of adding sewage sludge to land must also be taken into account when considering the environmental impact of sewage sludge. (The subject of metal uptake from soil applied sewage sludge is discussed in further detail in sections 5.3.1, 5.3.2, and 5.3.3).

1.4.2 NITROGEN

To quantify the nitrogen added to soil through sludge application, it is necessary to quantify the total nitrogen applied through the sludge and the nitrogen release rate. Nitrogen losses in agricultural soils result through crop removal, leaching, surface run-off, gaseous losses through NH_3 volatilisation, denitrification and erosion (Jarvis *et al.,* 1996). The loss of nitrogen from agricultural soils is of concern due to the possible NO_3^- leaching from cultivated soils that is responsible for increasing concentrations of NO_3^- in surface and groundwaters (Cheshire *et al.,* 1999).

In this study, the nitrogen-mineralisation rate of sewage sludge, at different application rates, was quantified under South African conditions and compared to the release of inorganic nitrogen from commercial fertilizer such as limestone ammonium nitrate (LAN). (This subject is discussed in further detail in section 5.3.4).

1.4.3 PATHOGENS

Conventional wastewater treatment processes do not completely remove or inactivate pathogens and parasitic organisms. In South Africa, most sewage treatment plants produce a type B sludge (WRC, 1997). A type B sludge contains pathogenic organisms. It is therefore important to understand the risk of infection due to the use of sludge on agricultural soils and for a variety of crops. For this research, a comprehensive literature survey was done as well as preliminary experiments to assess the survival of sludge-borne pathogens. (This subject is discussed in further detail in section 5.3.5).

1.4.4 PUBLIC PERCEPTION

The perceptions and believes of the general public towards the agricultural use of sewage sludge is as important as the scientific evidence proving the safety and/or risks associated with the practise. For this reason, the research project included a brief study in the form of a poll to establish the perception of the general public with regards to the use of sewage sludge in agricultural practices. (This subject is discussed in further detail in section 5.3.6).

CHAPTER 2: LITERATURE REVIEW

2.1 INTRODUCTION

Research on the use of sewage sludge in agriculture has shown that farmers recognised the importance of using organic substances to improve soil properties as early as 1862 (Parr & Hornick, 1992). Even earlier the Roman agriculturists and Asian farmers applied organic material to soil to maintain healthy and productive soils.

Today, the most serious problem in South African agriculture is the widespread rapid degradation of agricultural soils through erosion and nutrient depletion of these soils through incorrect agricultural practices. One of the readily available sources of organic material suitable for soil amendment is sewage sludge. Even though much information is available on the use of sewage sludge as an organic recycler, not much is done to protect land resources (Parr & Hornick, 1992). The research was mostly done in other countries with different climatic conditions compared to South Africa. Snyman *et al.* (1998) and Henning *et al.* (1999) have shown the beneficial use of sewage sludge under South African conditions in the short-term. To ensure the safe and sustainable use of sludge as a soil conditioner and amendment, a better understanding of different crop responses, soil, environmental conditions and sludges is needed.

The objective of this chapter was to summarise the findings of the research done internationally as well as locally on the agricultural recycling of sewage sludge. This includes the benefits and limitations of sludge as a soil conditioner, although more emphasis will be placed on the limitations which heavy metals place on the use of sewage sludge in agriculture.

2.2 SOURCES AND CHARACTERISTICS OF SEWAGE SLUDGE

In wastewater treatment, sludges are derived from various processes. Each type of sludge has its own characteristics. The moisture content of the sludge may be regarded as the most significant characteristic affecting the design and operation of the stabilisation and disposal processes (Ross *et al*., 1992). Table 2.1 shows the classification of sewage sludge to be used or disposed of on land in South Africa (WRC, 1997).

Table 2.1 Classification of sewage sludge to be disposed of on land (WRC, 1997)

Type of Sewage Sludge	Origin/ Treatment	Characteristics-Quality of Sewage Sludge
Type A Sludge	Raw sludge; Cold digested sludge; Septic tank sludge; Oxidation pond sludge	Unstable and can cause odour nuisances and fly breeding Contains pathogenic organisms and variable metal and inorganic content
Type B sludge	Anaerobic digested sludge; Surplus activated sludge; Humus tank sludge	Fully or partially stabilised - should not cause significant odour nuisance or fly-breeding Contains pathogenic organisms and variable metal and inorganic content
Type C sludge	Pasteurised sludge; Heat-treated sludge; Lime-stabilised sludge; Composted sludge; Irradiated sludge	Certified to comply with the following qualified requirement: Stabilised - Should not cause odour nuisances or fly-breeding; Contains no viable *Ascaris* ova per 10 g dry sludge; Maximum 0 *Salmonella* per 10 g dry sludge; Maximum 1000 Faecal coliform per 10 g dry sludge immediately after treatment; Variable metal and inorganic content
Type D sludge A sludge product for unrestricted use on land with or without the addition of plant nutrients or other materials. Must be registered in terms of Act 36 of 1947 if used for agricultural activities	Pasteurised sludge; Heat-treated sludge; Lime-stabilised sludge; Composted sludge; Irradiated sludge	Certified: Stabilised - Should not cause odour nuisances or fly-breeding; Contains no viable *Ascaris* ova per 10 g dry sludge; Max 0 *Salmonella* per 10 g dry sludge; Max 1000 Faecal coliform per 10 g dry sludge immediately after treatment; Maxi metal and inorganic content in dry sludge (later stipulated in review); User must be informed about the moisture and N, P and K content; User must be warned that not more than 8 t ha^{-1} $year^{-1}$ may be applied to soil and that the pH of the soil should be preferably be higher than 6.5.

Sewage sludge is basically classified as three types, A, B and C, in a decreasing order of potential to cause odour nuisances, fly breeding and transmission of pathogenic organisms to man and the environment. The type D sludge is of similar hygienic quality as type C, but since it is produced for unrestricted use on land at an application rate of 8 t ha^{-1} $year^{-1}$, the metal and inorganic contents are limited to acceptable low levels (WRC, 1997).

2.3 AGRICULTURAL UTILISATION OF SEWAGE SLUDGE

In the United Kingdom, the utilisation of sewage sludge on agricultural land is the most economic option for inland treatment plants (Davis, 1987), and it is also widely practised in Europe. However, the application of sewage sludge on agricultural soils in South Africa is still relatively unknown to farmers. Sewage sludge could play an important role in preventing soil erosion and soil degradation in South Africa.

2.4 BENEFITS OF USING SEWAGE SLUDGE IN AGRICULTURE

Sludge acts as a soil conditioner to facilitate nutrient transport, increase water retention and improve soil tilth (Ekama, 1993). Sludge also serves as a partial replacement for chemical fertilizers. The major plant nutrients (nitrogen, phosphorus and potassium) in sludges are not substantially removed during sludge processing, therefore the nutrients could improve the soil's nutritional status after sludge application. Sludge also contains essential plant micronutrients such as Cu, Zn, Mn and B (Ekama, 1993). However, the long-term benefit of sewage sludge to land as an alternative source of plant nutrients is frequently limited by potentially toxic contents of heavy metals and human pathogens (Palmer, 1993). Therefore, the benefits of sewage sludge are always weighed up against the limitations before and even during sludge application.

2.4.1 NUTRIENTS

As sludge contains high levels of organic matter and nutrients, it appears that one of the most appropriate options for re-use of sludge is use as a horticultural and agricultural soil amendment (Hu *et al*., 1996). Thus, the use of wastewater in agriculture arose from the desire to conserve nutrients to improve agriculture (Shuval, 1991). The use of sludge as soil amendment is not only beneficial in terms of providing an alternative means of sludge disposal, but also reducing the fertilizer costs. For instance, South Africa's expenditure on fertilizers in 1979 was about R400 million, and the figure soared to some R650 million in 1981 (Easton, 1983). Considering that this figure increases by R100 million each year (Easton, 1983), sludge provides a cost-effective nutritional alternative.

Sewage sludges contain appreciable amounts of N and P and have significant inorganic fertilizer replacement value for these major plant nutrients (Hall, 1986). Other inorganic nutrients such as K, Mg, Ca and certain trace elements are also beneficial (Smith & Vasiloudis, 1991). Table 2.2 shows typical macronutrient concentrations found in S.A. and United Kingdom (UK) sewage sludges. The K content of all sludges is low and not sufficient to make any significant difference to the recommended quantities of fertilizer K necessary for most cropping situations (Hall & Williams, 1984). The importance of other nutrients supplied during sludge application (e.g. Mg, Ca, Zn and Cu etc.) will depend on whether these are deficient in the soil. Thus, in considering the potential impacts of sludge nutrients on the environment, it is necessary to deal specifically with the problems attributable to N and P in agricultural soils. Epstein *et al.* (1976) found that applying sludge to soils at different rates (0, 40, 80, 120 and 240 t ha^{-1}) increased both the soil N (Nitrate-N and Ammonium-N) and P (plant available) concentrations. Sludge type, however, plays an important role to determine the extent to which the nutrients will be present and available for beneficial crop growth. From Table 2.2 it is clear that sludges in S.A. generally have higher concentrations of macronutrients compared to sewage sludges from the UK.

Studies have shown that the yield of many plant species increases following the application of sewage sludge. This can be attributed mostly to the increased supply of the nutrients as a result of sludge application. Sludge is considered to be a low-grade fertilizer and the concentration of plant nutrients in sludge is relatively low compared to the commonly used inorganic fertilizers. Associated with the low elemental concentration is the higher cost of transportation of the sludge, as compared with inorganic fertilizer (Korentajer, 1991). The response of soil to sludge application depends on many factors, such as sludge type and composition, application level and method, soil properties and climatic conditions (Metzer & Yaron, 1987).

Table 2.2 Summary of macronutrient concentrations (g kg^{-1} dry basis) found in S.A. and UK sewage sludges (mainly digested) (Smith & Vasiloudis, 1991)

Nutrient	Ranges	
	South Africa Concentration (g kg^{-1})	UK Concentration (g kg^{-1})
Nitrogen	17-58	15-25
Phosphorus	4-41	5-18
Potassium	1-11	1-3
Calcium	11-79	16-25
Magnesium	2-13	1-5

2.4.2 SLUDGE AS A SOIL AMELIORANT

An increasing problem around the world is the gradual decrease in the organic matter content of cultivated soils. In warm areas, such as southern Africa, the process of organic matter decomposition by microorganisms is high (Korentajer, 1991; Metzer & Yaron, 1987) and organic material build-up in soils to high levels is not common. The decrease in organic matter may lead to a deterioration of the soil physical properties that may in turn lead to accelerated erosion. The irreplaceable loss of agricultural soil due to soil erosion and the subsequent decrease in crop productivity is considered one of the main environmental problems on the African continent. The addition of sludge could provide a measure to maintain the organic status of soil and decrease the danger of runoff and erosion (Korentajer, 1991).

According to Act 36 of 1947 a fertilizer can only be classified as an organic fertilizer if it contains less than 20% ash and 40% water. Sludge does not comply with these criteria and can therefore not be classified as an organic fertilizer. It can still however, increase the organic status of the soil.

SLUDGE AS A N-FERTILIZER

The nitrogen in sludge is mainly present in the organic form that must first be mineralised before it can become available to plants (Singer & Munns, 1992; Trinidade *et al.*, 2001). Municipal wastewater and sludge contain variable amounts of organic N, NO_3^-, NO_2^-, and NH_4^+ (Environment Canada, 1984.). The amount of N that can be utilised depends on the type of crop, the amount of sludge added to the soil, the rainfall, etc. When the rate of plant-available NH_4^+ exceeds the rate of N uptake by the plant, the surplus NH_4^+ will be oxidised by autotrophic bacteria to NO_3^- that is very soluble and mobile. This can only move down in the soil profile and contaminate the groundwater if applied water exceeds the evapotranspiration rate (Ross *et al.,* 1992).

According to the guideline: Permissible Utilisation and Disposal of Sewage Sludge (WRC, 1997), the total amount of N in sewage sludge that can become plant available is about 30% in the first year of application, 15% in the second year and 5% in the third year. In a study done by Cripps *et al.* (1992) it was found that the mineralisation rate was 50% for the first year and 30% for the second year. Depending on the type of soil and type sludge, the mineralisation rate of digested sludge can be 20-50% (Cripps *et al.*, 1992).

Legislation in S.A. is such that the N content of the sludge determines maximum annual application rates. In principle, sludge is applied to provide equal amounts of N to inorganic N fertilizer recommendation rates. However, in the legislation, it was not taken into account that N in sludge exists predominantly in an organically bound form. It must therefore first be transformed to inorganic forms by means of mineralisation prior to plant uptake. This means that the plant availability of N from sludge at any time is considerably lower than the N from commercial inorganic fertilizers. In certain situations the low rate of release of inorganic N from sludge could be a desirable feature, which may increase its availability. For example, when the efficiency of N fertilization is limited by factors such as high NO_3^- leaching losses or high volatilisation losses of NH_4^+, the use of sludge as a slow release N material may decrease the N losses and increase N availability. It appears that sludge could also be used as a maintenance N fertilizer (Korentajer, 1991).

Although the initial amount of N in the applied sludge is high, a large amount can be lost through volatilisation (Cripps *et al.*, 1992). The fraction of NH_3 volatilised from sludge can vary between 0-50%, depending on the form of sludge applied, method of application and climatic factors (Korentajer, 1991). To be able to manage a N fertilizer programme, combined use of inorganic and organic matter inputs has been recommended as a means of maintaining high crop productivity and stable yields (Geiger *et al.*, 1992; Fernandes *et al.*, 1997).

2.4.4 SOIL PHYSICAL PROPERTIES

Sewage sludge can be utilised to improve the physical condition of soil through an increase in the organic carbon of the amended soil. The increase could also improve the water-holding capacity of the soil. Metzer & Yaron (1987) found that both the amount and composition of the sludge organic matter added to the soil affected the extent to which the soil physical properties were modified. The most significant effects of sludge addition are the increased water retention and improved water stability of the soil aggregates (soil structure). The improved soil physical status following the application of sludge frequently results in improved water transmission characteristics (e.g. higher hydraulic conductivity and increased water infiltration rate) and therefore reduced amounts of runoff and erosion (Metzer & Yaron, 1987; Korentajer, 1991). Soil chemical properties, such as cation exchange capacity, may also improve after sludge application due to the higher organic material content of the soil (Epstein *et al.,* 1976).

The improvement of the soil physical properties through the increase in organic carbon could play an important role in promoting the agricultural application of sewage sludge in the future in S.A. Generally, many soils in S.A. are low in organic matter due to rapid decomposition by microorganisms and the use of mineral fertilizers. This has led to the occurrence of widespread erosion and deterioration of soil physical status (Korentajer, 1991). It must be emphasised however, that at relatively low sludge application rates (7-25 t ha^{-1}), the changes in soil physical properties are not pronounced, and repeated application of sludge may be required to improve the soil physical properties.

Application of sludge can reduce the bulk density of a soil and studies have shown that three years after sludge incorporation, significant reductions in bulk density were still found, even at low application rates of 27 t ha^{-1} (Hall & Coker, 1983 as cited by Metzer & Yaron, 1987). This will naturally depend on the soil type.

2.5 RISKS ASSOCIATED WITH THE USE OF SEWAGE SLUDGE IN AGRICULTURE

Smith (1996) recorded an overall assessment of the potential environmental impacts caused by recycling sewage sludge to agricultural land (listed in Table 2.3). The risk factor of using sludge in terms of Potentially Toxic Elements (PTEs), organic contaminants and pathogens is relatively low, with possible risk concerning ecosystems and soil fertility. It is also clear that the main nutrients (N and P) and organic matter present in sludge are beneficial for the use of sludge on agricultural land. There are however possible risks of groundwater or surface water nitrate or phosphate contamination (Smith, 1996). The following sections deal with the contamination risks of sewage sludge applied to soil.

Table 2.3 Environmental impact risk and benefit assessment for sewage sludge recycling to agricultural land (Smith, 1996)

Environmental parameter	PTEs	Organic contaminant	Pathogens	Nitrogen	Phosphorus	Organic matter
Human health	L	P(L)	L	B	B	B
Crop yields	L	L	L	B	B	B
Animal health	L	L	L	B	B	B
Groundwater	L	L	L	P	L	L
Surface water	L	L	L	P(L)	P(L)	B
Air quality	L	L	L	P(L)	na	na
Soil fertility	P	L	L	B	B	B
Natural ecosystems	P	P	L	P	P	B

(B = beneficial effect, L = low risk, P = possible risk, na = not applicable, PTEs = potentially toxic elements)

2.5.1 HEAVY METALS

During the past 20 years a large amount of research has been done on heavy metals-soil interaction and plant uptake. This may be attributed, in part, to the relative ease of the analytical determination of the metals in sewage sludge, soil and plant tissue, as opposed to the difficulty involved with the determination of the trace amounts of organic compounds (Korentajer, 1991). Snyman *et al.* (1999) reported that not one of the wastewater treatment works in S.A. could comply with the Cu, Pb or Zn guideline concentrations present in sludge intended for unrestricted use. This problem was identified after the revision of the 1991 heavy metal guidelines with major alterations to the Cu, Pb, Zn and Cd contents allowed in sewage sludge for unrestricted use.

In areas where wastewater from metal-related and other industries is accepted into WWTPs, high levels of metals can be expected in the sludge (WRC, 1997). The Cu and Zn present in sewage sludge is mainly from domestic sources, while the major source for Cd and Pb is industrial (Smith, 1996). Reductions in the concentrations of potentially toxic elements in sludge may be anticipated in the future, however, this will become progressively more difficult as the contributions from diffuse sources gradually dominate the total heavy metal load in sewage sludge (Smith, 1996). Table 2.4 shows the mean heavy metal concentrations of sewage sludges derived from 77 of South African WWTPs (Smith & Vasiloudis, 1991), compared to the current guidelines for S.A. (WRC, 1997) and the USA [Environmental Protection Agency (EPA), 1993].

Table 2.4 Mean heavy metal content (Smith & Vasiloudis, 1991) in South African sewage sludges (mg kg^{-1} dry solids) utilised on agricultural land compared to guideline values of S.A. (WRC, 1997) and the USA (EPA, 1993)

Element	Mean	S.A. Guidelines	USA Guidelines
Zn	2 054	353.5	2 800
Cu	655	50.5	1 500
Cd	13	15.7	39
Pb	455	50.5	300

When comparing S.A. guidelines for the heavy metals in sewage sludges to the USA guidelines in Table 2.4, the South African guidelines seem overly conservative for the four heavy metals interpreted as total metal content. Furthermore, the mean concentrations of heavy metals indicated in Table 2.4 emphasise that WWTPs in S.A. cannot comply to a type D sludge in the current guidelines. This limits the unrestricted use of sewage sludge on agricultural land. The interpretation of the S.A. guidelines is not well understood and is currently under revision.

Although sludge quality has improved markedly in recent years, the concentrations of PTEs remain higher than the levels in uncontaminated soil (Tables 2.4 and 2.5). Once sewage sludge is applied to soil, heavy metals are retained indefinitely in the cultivated layers (McGrath & Lane, 1989). Therefore, repeated applications of sludge will gradually increase the trace element content of soil. However, metal leaching might take place under conditions such as low soil pH. Roy & Couillard (1998) found that there is a potential of metal leaching when sewage sludge is applied at doses equal to, or greater than 200 kg available N ha^{-1} to soils with pH lower than 4.5. Herselman and Du Preez (2000) found that high levels of heavy metals were taken up by spinach in a trial where sludge was applied to land for three years and up to application rates of 40 tons ha^{-1} y^{-1}. The also found that metal uptake by spinach in a sacrificial soil was very high and exceeded maximum permissible levels (MPL) for foodstuffs even though extent of metal leaching was restricted to the top soil layers.

Guidelines are needed to prevent contamination of soil and uptake by crops. Table 2.5 indicates the mean values of the main heavy metals in uncontaminated soils, compared to the guidelines for S.A. (WRC, 1997) and the USA (EPA, 1993) conditions. The mean background concentrations of Cu, Pb and Zn in uncontaminated soils, are higher than the permissible concentrations in soils after sludge application (S.A. guidelines), making it impossible to comply with the guidelines and therefore limiting the beneficial unrestricted use of sludge on land. In a recent study by Herselman and Steyn (2001) it was found that many South African soils have relatively high background levels of certain metals. Therefore, the guidelines need to be revised to include realistic soil background levels. Due to their role in the environment each of the metals will be discussed in further detail.

Table 2.5 Mean values of heavy metals in uncontaminated soil (mg kg^{-1} dry soil) (Smith, 1996) compared to S.A. (WRC, 1997) and USA (EPA, 1993) guidelines

Element	Mean	S. A. Guidelines (WRC, 1997)	USA Guidelines (EPA, 1993)
Zn	59.8	46.5	1 500
Cu	25.8	6.6	775
Cd	0.62	2	20
Pb	29.2	6.6	190

2.5.1.1 Zinc

Zinc is an important micronutrient for plant growth, and its presence in soil is beneficial for plants and animals (Hovmand, 1983). The Zn in sewage tends to be associated with suspended solids and is partitioned into the sludge during treatment. Sludge exhibits a wide range of Zn concentrations, which are generally higher than the background levels found in soils. The Zn content of plants also varies considerably, as a function of different soil, climate factors and plant genotype (Alloway, 1995). From previous research it is clear that the uncontrolled utilization of sewage sludge on agricultural land will lead to accumulation of Zn in the soil and consequently constitute a permanent risk for plants and crops (Lotter & Pitman, 1997).

Effect on plants and yields

Some crops are highly tolerant of Zn and can accumulate high levels in their tissue (Mengel & Kirkby, 1987) but most differ in their sensitivity to Zn toxicity (Gupta & Gupta, 1998). Zn toxicity causes a reduction in root growth and leaf expansion that is then followed by chlorosis (Mengel & Kirkby, 1987). Concentrations of 150 to 200 mg kg^{-1} in the dry matter of plant tissue are considered toxic (Sauerbeck, 1982) Grains seem to be more tolerant than beans as was shown by Vitosh *et* al. (1981). Dudka *et al*. (1994) reported that soil applications of up to 300 mg kg^{-1} did not cause a reduction in spring wheat yield and that applications of 1000 mg kg^{-1} only led to a 40 % decrease in yield. From 896 kg ha^{-1} applications only Swiss chard (*Beta vulgaris* subs. *cicla* (L.) Koch) and spinach (*Spinacia oleracea* L.) out of 12 leafy vegetables indicated toxicity symptoms (Boawn, 1971).

Livestock and humans

In both humans and livestock, acute Zn toxicity is observed only after intake of several grams (Gupta & Gupta, 1998) – 6 g is considered a lethal dose (Pais & Benton Jones, 1997). In livestock it is manifested as gastrointestinal distress, decreased food consumption, pica, decreased growth, anaemia, and poor bone mineralisation and arthritis (NRC, 1980). In humans ingestion of large amounts have led to vomiting and diarrhoea and Zn lowered the activity of Cu leading to signs of Cu deficiency (Gupta & Gupta, 1998).

2.5.1.2 Copper

Copper is also one of the important, essential elements for plants and animals. The chemistry of Cu shows that Cu is specifically adsorbed or fixed in soils (Alloway, 1995). Sewage sludges are capable of substantially increasing the soil levels of Cu (Chlopecka, 1996a). However, to the knowledge of Smith (1996), there have been no reports of plant toxicities to sludge-borne Cu when grown in fertile, limed soils. Soil organic matter appears to be the dominant factor controlling Cu retention (Alloway, 1995).

Effect on plants and yields

Cu levels of 21 mg kg^{-1} and above in plants could indicate excessive or toxic accumulation (Gupta & Gupta, 1998). Cu toxicity in plants varies greatly and Gupta (1997) reported levels ranging from 10 mg kg^{-1} in cucumber to 70 in maize. Cu is able to replace other metal ions, in particular Fe, from physiologically important centres (Mengel & Kirkby, 1987) and this leads to a commonly observed chlorosis, superficially resembling an Fe deficiency (Daniels *et al*., 1972; Pais & Benton Jones, 1997). Cu toxicity leads to reduced branching, thickening, and abnormally dark colouration in rootlets of many plants (Reuther & Labanauskas, 1966). One of the most rapid responses to Cu toxicity is the inhibition of root growth (Mengel & Kirkby, 1987) and, being particularly toxic to roots, little is translocated to the tops of plants (Pais & Benton Jones, 1997). High levels of Ca alleviate Cu toxicity (Wallace *et al*., 1966).

Livestock and humans

Copper poisoning is found in most parts of the world with sheep being affected most often and other animals also being susceptible, with the main accumulation occurring in the liver and kidneys (Fraser, 1986; Gupta & Gupta, 1998). Acute Cu toxicity in humans is rare and is associated with ingestion of gram quantities and includes haemolysis, hepatic necrosis, and renal damage (van Campen, 1991). A single dose of 0.1 to 0.2 mg Cu kg^{-1} body weight can lead to gastrointestinal disturbances in sensitive persons (Bosshard & Zimmerli, 1994).

2.5.1.3 Cadmium

Cadmium has no essential biological function, and is highly toxic to plants and animals. However, the concentrations of Cd normally encountered in the environment do not cause acute toxicity. The main inputs of Cd to the soil are fertilizer and atmospheric deposition. Crops receive 10-30% of the Cd-load directly from the atmosphere (Alloway, 1995). Sewage varies in composition and contains Cd from various sources, including human excretion, domestic products, storm waters containing particles of rubber tyres and various industrial effluents (Alloway, 1995).

Effect on plants and yields

Cadmium may depress growth at levels of 3 mg kg^{-1} but this will depend on species as well as other elements present. It interferes with photosynthesis, uptake and transport of nutrients and manifests as leaf chlorosis and necrosis followed by leaf abscission (Pais & Benton Jones, 1997). It may be accumulated in plants to high levels and not show toxic symptoms but may be harmful to humans and animals (Mengel & Kirkby, 1987; Gupta & Gupta, 1998). The Cd content in plants is highly correlated with that found in soil (Pais & Benton Jones, 1997) and it is considered to be the most mobile of the pollutant metals (McLaughlin *et al*., 2001). Soil applied Zn (Chaney *et al*., 2001) and Ca (Mengel & Kirkby, 1987) has been shown to decrease Cd uptake by plants. Cd and Zn are chemically very similar and Cd can mimic the behaviour of Zn in its uptake and metabolic functions and is also rapidly transported from the soil via the plant root to the upper parts (Mengel & Kirkby, 1987). Cd is not translocated to seed in some seed crops even though the plant may be exposed to high levels in the rooting zone (Sommer, 1979; Pais & Benton Jones, 1997). Application rates as high as 50 mg kg^{-1} did not cause any decrease in wheat yield (Dudka *et al*., 1994). The resulting grain contained Cd levels of 4 mg kg^{-1} and could prove to be toxic to livestock if used as a feed source (Gupta & Gupta, 1998).

Livestock and humans

Cd is a cumulative poison in animals in that it accumulates in the kidneys and to some extent also the liver and spleen (Mengel & Kirkby, 1987). In humans it is known to cause hypertension, cancer, immune disorders (Stewart-Pinkham, 1989), severe gastric cramps, vomiting, diarrhoea, cough, headache, brown urine, and renal failure (Berkow, 1992). Not only is Cd very toxic but it can increase the toxicity of other agents (Gupta & Gupta, 1998). The main dietary source of Cd in humans is cereal grains (Pais & Benton Jones, 1997).

2.5.1.4 Lead

Lead is neither an essential nor a beneficial element for plants and animals. When Pb is released into the environment it has a long residual time compared with most other pollutants. The Pb is bound firmly in the soil and it forms highly insoluble precipitate with phosphates (Laperche, 2000), and therefore root uptake is small. The translocation within the plant is also small, so that the Pb-content of the roots is controlled by the root uptake, while atmospheric Pb-deposition is responsible for Pb in stem, leaves and fruit. There is a general agreement that the Pb content of sludges is very variable but the typical sludge contains less than 1000 mg kg^{-1} (Alloway, 1995). Repeated applications of sludge should not lead to an accumulation in soil in excess of 50 to 300 mg kg^{-1} (Alloway, 1995), although these values are above the guideline limit for the permissible lead content in S.A. soils (6.6 mg kg^{-1}), again emphasising the conservative nature of the current guidelines.

Effect on plants and yields

Lead is considered the least mobile of the heavy metals in soils but can accumulate in some plants to levels of 350 mg kg^{-1} without visible harm (Pais & Benton Jones, 1997). Toxic effects of Pb in plants are not regularly seen in the field (Mengel & Kirkby, 1987) and almost all detailed observations of Pb toxicity in plants have been described in water culture trials (Brewer, 1966). Lead can readily be taken up by plant roots, the amount varying with species and type, but less than 3% is translocated to the tops (Pais & Benton Jones, 1997). Generally Pb levels in grain, tubers and roots are seldom affected by Pb pollution from atmospheric sources and therefore remain within normal levels of about 0.5 mg kg^{-1} (Foy *et al*., 1978). Some leafy vegetables, such as lettuce, have the ability to accumulate higher levels of Pb (Pais & Benton Jones, 1997).

Livestock and humans

Ingestion and inhalation represent the primary routes of Pb intake in animals and humans (Gupta & Gupta, 1998). It mimics the metabolic behaviour of Ca and inhibits many enzyme systems (Mengel & Kirkby, 1987) and is accumulated mainly in the skeleton (Pais & Benton Jones, 1997; Gupta & Gupta, 1998). Excessive exposure to Pb leads to pallor, gingival lead line, anaemia, and a variety of neurological symptoms (Langston & Irwin, 1989; Stewart-Pinkham, 1989) as well as possible aggressive behaviour in humans and animals (Mengel & Kirkby, 1987). It is both carcinogenic and teratogenic and impacts the intellectual development of young children (Pais & Benton Jones, 1997) through the causing of brain damage (Mengel & Kirkby, 1987).

2.5.1.5 Plant-Soil Relationships of Metal Uptake

The factors affecting the amounts of metals absorbed by a plant are those controlling:

The concentration and speciation of the metal in the soil solution

The movement of the metal from the bulk soil to the root surface

The transport of the metal from the root surface into the root

Translocation from the root to the shoot.

The uptake of metals from soils is greater in plants grown in pots of a certain soil than from the same soil in the field (Alloway, 1995). Relative differences in the uptake of metal ions between plant species and cultivars are genetically controlled. Several factors influence the bioavailability of heavy metals to crop plants. These include soil pH, CEC and metal speciation. Soil pH value has been identified as the single most important soil factor controlling the availability of heavy metals in sludge treated soil (Løbersli *et al*., 1991; Smith, 1996). Oliver *et al*. (1996) conducted field experiments to determine the effectiveness of liming to minimise uptake of Cd by wheat (*Triticum aestivum* L.) and barley (*Hordeum vulgare* L.). Raising soil pH from 4.0 to 5.0 through lime application to soil decreased Cd concentrations in the grain. However, in some cases, raising soil pH to 6.0 was not sufficient to decrease Cd concentration in grain. Raising the soil pH above this value would not be considered economically viable in many areas (Oliver *et al.*, 1996). Therefore, increasing the soil pH reduces the uptake of heavy metals up to a certain pH level, after which further increase in pH levels would not cause significant decrease in metal uptake (Smith, 1996).

The soil's CEC is also dependent on soil factors such as pH, organic matter content and soil texture (Alloway, 1995). The CEC is a measure of a soil's capacity to adsorb cations and it increases with increasing organic matter and clay content. This is due to the large quantity of negative charges provided by these soil fractions. Sandy and sandy loam soils have small CEC values due to a low clay content and little organic matter in most S.A. conditions.

The binding of heavy metals in soil by native and sludge organic matter, and the changes in metal speciation that occur over time, particularly once sludge application has ceased, are intimately linked. These are critical factors in assessing the potential long-term environmental effects of spreading sewage sludge on land (Smith, 1996). A typical application of 5 t ha^{-1} of sewage sludge per year adds about 2.5 t ha^{-1} of organic matter to the soil and would raise the soil organic matter content by 3.07% (Smith, 1996). Thus, single large applications of sludge of 50-100 t ha^{-1} $year^{-1}$ are recommended to achieve significant improvements in the physical properties of impoverished soils (Smith, 1996). However, this approach is not recommended in the sludge guidelines (WRC, 1997) which limits the sludge application rate to 8 t ha^{-1} $year^{-1}$. Nevertheless, relatively small quantities of organic matter normally applied to soil in sewage sludge can still significantly increase the adsorption capacity of soil for metals (Smith, 1996). Increasing the complexation capacity of soil and the formation of stable organo-metallic complexes by applying sludge organic matter reduces the mobility of heavy metals in soil and thus lowers their availability to plants (Alloway, 1995). The decreasing stability of organo-metallic complexes in sludge applied soil at pH 5.6 was Cu > Cd > Zn = Pb indicating the stronger binding of Cu with organic matter compared with Cd or Zn (Smith, 1996).

Scientists are increasingly emphasising the important role of sludge properties in controlling metal availability to crop plants in sludge treated soil. For instance, Steinhilber (1981) reported that Zn was eight times more plant-available in a low Zn sludge than in a high Zn sludge. The source and stabilisation method of sludges also contribute to the behaviour of the metals in sludge applied soils. For example, heavy metals in activated sludges would probably be complexed in different forms compared with anaerobically digested sludges (Smith, 1996).

The uptake of sludge-borne metals into crop plant parts plays a major role in the unrestricted use of sludge on agricultural land. Concentrations of zootoxic metals (e.g. Cd and Pb) need to be monitored in the edible parts of crops. Furthermore, the possibility of phytotoxicity of the metals (e.g. Zn and Cu) to crops might reduce yield and growth of crops if taken up in excess. Cereal crops are the main sources in the staple diet of people in southern Africa.

2.5.1.6 Effect of Sewage Sludge on Maize Cultivation (Metals)

Maize (*Zea mays* L.) cultivation occurs throughout South Africa. Currently, farmers in the vicinity of a major WWTP in the East Rand are using sewage sludge as an organic supplement with inorganic fertilizers for maize cultivation. Since the beginning of the scheme a 20% increase in the yield of maize was reported with a 40% saving on inorganic fertilizer (Du Preez *et al*., 1999). Table 2.6 shows phytotoxic and acceptable levels of different heavy metals and nutrients in maize leave tissue (Smith, 1996). The phytotoxic levels are very broadly stipulated and uptake might differ between plant genotypes and cultivars (Davis & Charlton-Smith, 1980). Uptake into different plant parts at different growth stages of maize was also not taken into consideration. (***Note:*** The effect of sewage sludge on the growth and yield of maize is discussed under section 2.5.2.2).

Table 2.6 Phytotoxic and acceptable levels of heavy metals in plant leave tissue in general (mg kg^{-1}) (Smith, 1996)

Heavy metals	Adequate	Toxic
Zn	27-150	100-400
Cu	5-30	20-100
Pb	5-10	30-300
Cd	0.05-0.2	5-30

Metal uptake by maize plants

Cd and Zn are related elements and are usually studied together as sludge-borne metals, as competition for plant uptake may occur between the elements (Pepper *et al.,* 1983). Cd and Zn can be highly toxic to plants and animals (Alloway, 1995) due to their mobility in the soil environment, and are normally the most likely to limit application of sludge on land (Korentajer, 1991). However, in a maize field experiment done by Henning *et al. (*1999), no negative effects of heavy metal contamination could be proven in either soil or plant parts, even when high sewage sludge dosages were applied. Bidwell & Dowdy (1987) did experiments to assess the availability of sludge-borne Cd and Zn over a longer period of time. Maize and soil was sampled for six years following termination of three annual applications of sewage sludge. It was found that Cd and Zn concentrations in plant tissue increased linearly within treatment levels for all years.

Logan *et al.* (1997) also reported on a four year field study to determine trace metal (including Cd, Cu, Pb and Zn) concentrations in maize tissue, as affected by a once off application of an anaerobically digested sewage sludge to loam soil. The aim was to determine the nature of the uptake response over a wide range of sludge application rates (0, 7.5, 15, 30, 60, 90, 120, 150, 188, 225, and 300 t ha^{-1} dry solids). Availability of soil metals increased linearly with total soil concentrations and generally declined over time. Cd, Cu and Zn concentrations in maize increased significantly with sludge application, while Pb levels were low compared to the control. Jarausch-Wehrheim *et al.* (1998) found that a high application of sewage sludge to a sandy soil where maize cultivation occurred, caused significantly raised Zn concentrations in all plant parts over the whole growing season, although the upper leaves and stalk parts occurred as the most important concentration and storage sites. It was found that increased Zn concentrations in sludge-treated plants coincided with a dry matter reduction of total shoots, and subsequent yield reduction. These observations provide evidence for a raised Zn exposure to animals and humans by vegetative aerial maize plant parts used in fodder and food.

Lead and cadmium being zootoxic heavy metals, need to be monitored in plant parts used by humans and animals. Lead, however, has a low solubility, mobility and bioavailability to plants and does not pose a major threat (Alloway, 1995). This was also proven by Chlopecka (1996b) who found a significant increase in maize tissue of Cu, Cd and Zn in the maize tissue grown on sludge-amended soil, but a non-significant change in plant tissue with respect to Pb concentration.

Laboratory studies of maize seedlings grown in 8, 24 and 72 t ha^{-1} amended silica sand illustrated that the inorganic fertilizer (Hoagland's solution) affected seedling growth better in terms of root, leaf mass and shoot length, but showed chlorosis. The sludge affected the growth of maize seedlings positively compared to control seedlings not receiving amendments, without heavy metal accumulation in the soil or seedling foliage, even at 72 t ha^{-1} dosages (Snyman *et al.,* 1998). Although nutrients present in sludge may increase yield, the presence of high levels of heavy metals can induce phytotoxic effects in plants and a resultant decrease in yield. The after effects of metals derived from a highly metal-polluted sludge on maize were studied by Mench *et al.* (1994) under field conditions. Plant parts were collected from plots, which received 0 (control), 50 and 300 $t_{\text{dry sludge}}$ ha^{-1} as cumulative input. The dry matter yields and plant metal concentrations were determined. Leaf yield for plants from the 300 t ha^{-1} plot was reduced by 27% and toxicity symptoms developed on shoots. Yield for plants in both sludge-treated plots were similar to the controls. Yield reduction at the high metal loading rate was probably due to the increased uptake of Cd found in the maize leaves. The effect of the metals could be detected before the appearance of any visible symptoms of phytotoxicity occurred.

Agricultural practices also play an important role in limiting the uptake of heavy metals by crops. Liming is known to control metal uptake in crops. Pepper *et al.* (1983) determined the effect of liming of sludge-applied soil on the Zn and Cd uptake of maize. Liming reduced Zn uptake in most maize tissues, whereas Cd uptake was generally unaffected. Concentrations of Zn and Cd in maize leaves from two different soils increased with increased sludge rate. For both soils, Cd and Zn concentrations followed the order: leaves > stover > cobs > kernels. The results suggested that liming to a pH of 6.5, as recommended in sludge application, would not reduce Cd uptake by maize.

The results of research on maize cultivation on sludge-applied soils are varied. It seems that the short-term application of sludge does not pose a major problem for maize cultivation. Long-term applications and application of heavy metal polluted sludges, though, could lead to the uptake of heavy metals like Zn, Cu and Cd to phytotoxic levels. Due to its low mobility in soils and subsequent low uptake, sludge-borne Pb holds no great threat for maize cultivation.

2.5.1.7 Effect of Sewage Sludge on Winter Cereal Crops (Oats, Barley and Wheat) Cultivation (Metals)

Producing high yields of high quality oats (*Avena sativa* L.) involves interactions among numerous biological factors, management strategies and climatic conditions. Oats are one of the most versatile of the cereals in regard to suitable soil type. Almost any reasonably fertile, well-drained soil is suited to oats if temperature and moisture conditions are favourable (Welch, 1995). Very little research on the direct effect of sewage sludge on oat plants has been performed, therefore other research on closely related cereal crops, e.g. wheat (*Triticum aestivum* L.) and barley (*Hordeum vulgare* L.), are included. (***Note:*** The effect of sewage sludge on the growth and yield of winter cereal crops is discussed under section 2.5.2.3).

Metal uptake

Simeoni *et al.* (1984) reported on the effect of small-scale composting of sewage sludge on heavy metal availability to oat plants. Oats were grown in pots containing composted and uncomposted sludge additions at rates equivalent to 0, 60, 120 and 240 t ha^{-1} added to an acid and neutral loam soil. Plant samples were analysed for total content of Cd, Cu and Zn. Higher application rates of compost maintained a higher pH on the acid loamy sand, which decreased Cd and Zn availability to plants, due to the metals being unavailable for plant uptake at high soil pH (Alloway, 1995). However, a 40% decrease in sludge Cu content occurred, possibly due to leaching of soluble Cu complex during composting, although no change was noted in Cu availability to crops (Simeoni *et al.*, 1984).

Moreno *et al.* (1996) studied barley grown on calcareous soil amended at different rates with sewage sludge compost, containing different heavy metal contents. When the soil was amended with high rates of contaminated compost (mainly Cd), the Cd and Zn were easily taken up by the barley plants. However, Cu in particular was retained by organic matter and was not transferred to plants. Concentrations of Cd and Zn in soils were positively correlated with Cd and Zn contents in plant material.

Unger & Fuller (1985) reported that sludge with relatively higher levels of heavy metals, particularly Cd, appeared poorly suited as fertilizer if used for an extended period of time, because of the barley plants' tendency to take up elevated levels of certain heavy metals. Some parts of barley plants are better indicators of heavy metal uptake and concentrations. Heavy metal uptake (Zn, Cu and Cd) was relatively low in a field experiment due to the neutral to slightly alkaline pH of the soil, and the presence of lime throughout the soil profile. Miller *et al.* (1995) also studied the heavy metal uptake (Cd, Cu and Zn) by barley at three application rates to three soil types. Metal contents were relatively low in barley grain, but higher in barley straw. Metal contents in plants increased with increasing sludge loading. Most plants grown on soil amended with the higher sludge rates were too high in Cd to be suitable for animal consumption. No plant materials tested exceeded the suggested maximum as seen in Table 2.6 (Smith, 1996), although the phytotoxic levels are very broadly stipulated and uptake might differ between plant genotypes and cultivars (Davis & Charlton-Smith, 1980). Uptake into different plant parts at different growth stages of oats was also not taken into consideration.

Sludges of controlled metal addition were also used in a pot trial by Sanders *et al.* (1987) to assess effects of sludge-borne Zn and Cu on the growth of barley. No phytotoxic effects of Cu and Zn on crop yields were measured for a sandy loam of pH 6.5 and with the lowest CEC of the soil types examined. It was also noted that total metal concentrations in soil, which are necessary to lead to phytotoxic levels, varied between soils because of their differences in pH and CEC.

Although reports are varied on heavy metal uptake on winter cereal crops, soil properties like soil pH and CEC play a major role in determining whether heavy metals will be available for crop uptake. Management practices like liming play an important role when sludge is amended to soils. Sludge application to soils does not seem to pose a heavy metal contamination or phytotoxicity problem when winter cereals are cultivated, although correct monitoring procedures are important.

2.5.1.8 Greenhouse Versus Field Experiments

Logan & Chaney (1983) described the potential limitations of using greenhouse pot experiments in assessing the environmental effects of toxic metals present in sewage sludge on plants. Accumulations of PTEs in plant tissues can be increased 1.5 to 5-fold compared with field studies with the same soil, sludge and crop. Greater accumulations occur for a number of reasons including:

- Use of NH_4-N fertilizers which lowers soil pH more in pots than in the field
- Higher soluble salt levels in pots than in the field due to smaller soil volume required for fertilizer nutrients
- Confinement of plant roots to the small volume of treated soil in pots
- Abnormal watering pattern and relative humidity in greenhouse pot studies.

The smaller the pot, the greater the expected error. The pots used by Sanders *et al.* (1987) were particularly small holding only 1 kg of soil. Those used by Davis & Charlton-Smith (1984) were significantly larger holding 7 kg. Chaney *et al.* (1987) considered that large pots of similar capacity were a prerequisite for trace element risk assessment using pot culture techniques. In spite of these criticisms, pot studies in controlled environments enable close control over experimental variables needed to characterise soil-plant interactions of metals. However, Logan & Chaney (1983) argued that regulations for the utilisation of sludge should be based upon field research because of the problems encountered with pot trial studies. This in spite of the large applications of sludge in field experiments that are necessary to raise soil concentrations of PTEs within the relatively short time scales that trials are normally conducted. This addition of large rates of high metal concentration sludges to soil can grossly overestimate metal availability and toxicity compared with normal, low level sludges.

2.5.2 NUTRIENTS

The loss of N and P from soil by various biological processes and physical mechanisms is a common phenomenon in agricultural activities. These losses can result in significant pollution of the aerial environment by ammonia volatilisation and denitrification, and contamination of surface and groundwater supplies through leaching of nitrate and surface run-off of both nitrate and phosphate (Smith, 1996). It was found that the application of sewage sludge to soil decreased the run-off intensity and reduced soil erosion. This led to a decrease in the total P exported through surface run-off (Van den Bossche *et al.*, 1999). Very little research has been done concerning P losses after sewage sludge applications to soil. Nitrate leaching has been investigated more thoroughly though. Increased losses of nitrate by leaching following the application of sewage sludge to soil have been frequently reported (Chang *et al.*, 1988; Wadman & Neeteson, 1992; Lotter & Pitman, 1997). The need to investigate nitrate leaching under South African conditions has been acknowledged, and future experiments on nitrate leaching under greenhouse and field conditions will be completed as part of a sewage sludge research programme.

2.5.2.1 Nitrate Leaching

The concentration of NO_3^- is often the limiting factor when disposing of sewage sludge. Worldwide, there is concern over increasing contamination of groundwater by NO_3^- due to agricultural practices (Brye *et al.*, 2001). A major problem is the possibility of NO_3^- pollution of surface and groundwater, and the potential threat to human and animal health (Ross, 1989).

Strict legislation exists in South Africa for sludge application in terms of total N application, because of the possibility of NO_3^- leaching. According to the 'Permissible Utilization and Disposal of Sewage Sludge' (WRC, 1997), the maximum loading rate of sewage sludge to agricultural land is 8 tons of dried sludge per hectare per year. No equivalent legislation exists for commercial fertilizer. Unlike sludge, which contains some N in an organic form, commercial fertilizer contains N only in the inorganic form. This means that N from fertilizer will be immediately available for plant uptake or leaching, whereas organic N from the sludge is only partly available and the remainder must be mineralised by microorganisms before it can be utilised. In sludge the plant available N is about 30% of the total N in the first year, 15% becomes available in the second year, and 5% in the third (WRC, 1997). This assures a more balanced and constant supply of N to the plant for a longer time than commercial fertilizers (Environment Canada, 1984). Commercial fertilizer can therefore contribute more to groundwater contamination than sludge.

Given the significant amount of information concerning the potential harmful effects of sludge application, municipalities need information that will allow them to apply sludge, on an ongoing basis, at maximum loading rates that will be beneficial to agricultural practices, and minimise environmental impact (Lerch *et al.,* 1990).

NO_3^- can pose a serious threat as an environmental pollutant if it is not controlled (Gaines & Gaines, 1994), and therefore the N content is often the primary factor that limits sludge application (Higgins, 1984; Douglas & Magdoff, 1991). If applied in very large amounts, excess N can cause undesirable changes in soil properties, such as a decrease in pH due to nitrification. Accumulation of NO_3^- in the soil that can lead to luxury consumption by plants (Magdoff & Amadon, 1980; Pratt & Jury, 1984 as cited by Artiola, 1991) and leaching that can create a potential for groundwater contamination (Artiola, 1991; Stamatiadis *et al.*, 1999).

N is leached mainly as NO_3^-, but in sandy soils NH_4^+ can be leached as well. Unlike heavy metals and toxic organics, NO_3^- is not adsorbed on the soil constituents and is therefore very mobile in the soil. This is a particularly serous problem in areas of shallow groundwater tables (Korentajer, 1991). Leaching of N is common where soil NO_3^- levels are high, and where the downward movement in the soil is strong. These soil conditions are largely found in humid and sub-humid regions (Stevenson, 1986). Fluctuations of NO_3^- concentrations in soil water occur throughout the year. This can be due to a low field capacity, differences in precipitation and plant uptake, etc (Hansen & Djurhuus, 1997).

Some physical soil properties, such as texture and structure, can influence NO_3^- leaching, because it has an influence on the water movement through the soil. The texture influences the water movement through the soil profile, the retention of water as well as the chemical processes (Environment Canada, 1984). The soil texture and structure have a large influence on the water movement in the soil. It determines its infiltration and permeability, and therefore the extent of NO_3^- leaching (Environment Canada, 1984). If sludge is applied to crops, the uptake by plants itself and water uptake of plants will reduce NO_3^- leaching, especially in a dry country like S.A. This is not taken into account in the legislation.
Soils with significant quantities of silt, clay and organic matter will retain more NO_3^- than soils without these fractions (Gaines & Gaines, 1994). Studies done by Gaines & Gaines (1994) show that sandy soil has a very high potential of NO_3^- leaching, whereas in clayey soils, with a higher anion exchange capacity (AEC), the risk is reduced. Higgins (1984) found that NO_3^- increases in the lower horizons are only detectible after a year of sludge application.

Environmental hazards of N

There are several environmental hazards associated with N, such as health risks, pollution of aquatic systems by NO_3^-, or the contribution to greenhouse gases by gaseous N emissions.

<u>Health hazards associated with NO_3^-</u>

NO_3^- becomes toxic to animal and humans if consumed in large concentrations. NO_3^- is reduced to NO_2^-, which is absorbed into the bloodstream, where it oxidises oxyhaemoglobin (the oxygen carrier) to methaemoglobin (which cannot carry oxygen). This leads to the disease 'methaemoglobinemia', or 'blue baby' syndrome. Infants and young animals can suffer from this and ultimately die. In healthy older animals and adults, the stomach acids and rapid absorption and excretion of unused NO_2^- makes NO_2^- poisoning unlikely (Ross, 1989; Sveda *et al.*, 1992; Miller & Gardiner, 1998). If sub-lethal concentrations are consumed by ruminant animals, they can lead to a reduction in milk production, abortions, and metabolic disorders (Die Kynoch Weidings-handleiding, 1997).

<u>Groundwater contamination</u>

NO_3^- leaching to groundwater is considered to be a major problem in the agricultural sector. One of the major contributors to groundwater pollution by NO_3^- is the use of more N than is actually needed by crops (Magdoff, 1992). NO_3^- can cause eutrophication of watercourses and the growth of algal blooms (Ross, 1989).

Because of the risks associated with NO_3^- pollution, there are strict specifications regarding NO_3^- content in drinking water (Sveda *et al.*, 1992). Drinking water standards in South Africa are 10 mg l^{-1} NO_3^--N, as legislated by the Department of National Health and Population Development (Korentajer, 1991). The levels of NO_3^--N in wastewater usually vary from 0-20 mg l^{-1}, with the maximum at 45 mg l^{-1} (Tchobanoglous & Burton, 1991).

International norms for sludge N application rates may not be applicable to the local conditions. Additional research is required to assess the rate of mineralisation and crop N uptake to ascertain the dangers of NO_3^- leaching and accumulation in plants under South African soil and climatic conditions (Korentajer, 1991).

2.5.2.2 Effect of Sewage Sludge Application on the Yield of Maize

Reports are varied on the effect of sewage sludge on yield of maize, especially comparing glasshouse and field experiments. Nitrogen loading rate has often been suggested as the basis for regulating the application of organic amendments on agricultural land. Optimal loading rates should maximize crop N-uptake. Nitrogen in sludge must be transformed to inorganic forms by the process of mineralisation prior to plant uptake. This means that in general, the availability of N from sludge is considerably lower than that from inorganic fertilizers, implying that the cost of fertilization with sludge would be higher (per unit weight N) than that of commercial inorganic fertilizers. However, in certain situations the low rate of release of inorganic-N from sludge would be a desirable feature that may increase its availability to crops (Korentajer, 1991). Hemphill *et al.* (1982) evaluated the N fertilizer value of sewage sludge on maize yield. Sludge was applied such that the total N application was equal or twice the recommendation for maize (200 kg ha^{-1}), at a high and low rate. In the first two years, the maize ear yields increased nearly proportionally to the sludge NH_4-N content at the low rate of sludge application, but were not further increased at the high rate of sludge application. However, ear yields of plants grown on sludge-applied soils were less than yields obtained with the optimal rate of $(NH_4)SO_4$ fertilizer. In the third year, however, the yields on sludge-applied soils equalled or exceeded those with optimal $(NH_4)SO_4$ fertilizer. The metal content in kernels varied less with N rate or source. However, the metal content of the leaves increased proportional to the N rate or source. Applying lime to some of the sludge-applied plots had little effect on yield of maize plants or to the metal content of kernels.

Christodoulakis & Margaris (1996) performed a greenhouse experiment and reported on the effect of final sewage effluent and sludge from a sewage treatment plant in Greece on the growth of maize. The results showed that adding sludge to the soil promoted plant growth significantly more than when commercial fertilizer was added. Plant height increased in maize individuals by 77% in the sludge application treatment compared to 25% in the case of the commercial fertilizer application.

In a short-term field experiment, Henning *et al.* (1999) evaluated the cultivation of maize on high sewage sludge dosages (12.5 t ha^{-1} and 25 t ha^{-1}) at field scale. It was found that there were no real differences in yield between treatments, although the 12.5 t ha^{-1} treatment showed significant increases in the amount of ears per plant. The application of sewage sludge as an organic fertilizer might therefore increase yield of maize plants, although the metal content of sludges together with N-loading rates will determine whether short-term or long-term application of sludge is sustainable.

2.5.2.3 Effect of Sewage Sludge Application on the Yield of Winter Cereal Crops (Oats, Barley and Wheat)

The main aspect of yield in oat plants is grain production. The harvest index (HI) and nitrogen harvest index are considered as physiological indicators of grain production. Other parameters that play an important role may be the rate of leaf appearance, floret number, grain size and grain protein content (Welch, 1995). The effect of different nitrogen application rates of sewage sludge and ammonium nitrate (inorganic fertilizer) application rates on soil profile and nitrogen uptake were studied in winter wheat by Gavi *et al.* (1997). In general, wheat yields and N uptake increased linearly with applied N as sewage sludge.

Estimated N use–efficiency using sewage sludge in grain production was 20% compared to ammonium nitrate. Estimated plant N recovery was 17% for sewage sludge and 27% for ammonium nitrate. Different sewage sludge applications (45, 180 and 540 kg N ha^{-1} $year^{-1}$) did not affect surface soil, pH, organic and total N following first and second harvest (Gavi *et al.,* 1997).

Unger & Fuller (1985) studied the differences in grain production in barley straw and grain when using low and high metal content sludges on arid land. Nitrogen responses for barley straw and grain were observed from both sludges. The low metal sludge appeared to be attractive as a potential fertilizer. Yield was higher in both grain and straw of barley when compared to inorganic fertilized plants. Weight per barley head also increased as a result of the use of sewage sludge. Yield increases also occurred in oats when compost from the small-scale composting of sewage sludge was applied (Simeoni *et al.,* 1984).

Concerning the application of sewage sludge as an organic fertilizer, it seems that sludge application does increase the yield of winter cereal crops, due to the N-loading rate. However, these results are based on short-term experiments. Therefore, more long-term experiments need to be done to assess the sustainability of sludge application to soils when cultivation of winter cereals takes place.

2.5.3 HUMAN PATHOGENS

Much has been done to minimise the potential transmission of pathogens by reducing infectivity of sludges through effective treatment processes and then matching efficiency of pathogen removal to operational restrictions on application practices and land use (Smith, 1996). Gaspard *et al.* (1995) found that nematode eggs are strongly resistant to most of the classical waste treatment processes. It is impossible to analyse for all pathogenic organisms and too costly to determine the presence of a wide range of these organisms. Therefore, only the numbers of *Ascaris* ova, *Salmonella* organisms and faecal coliforms are included as indicators of hygienic quality requirements of Type C and D sludges in S.A. (Table 2.1) (WRC, 1997).

The pathogenic organisms present in sewage sludge can be potentially dangerous to human health (Higgins, 1984; Lerch *et al.*, 1990; Ross *et al.,* 1992). There are five main types of pathogens in sludge namely: bacteria, viruses, fungi and yeasts, parasitic worms and protozoa. Humans and animals show sensitivity to some of these. The pathogens are mainly present on the soil surface or at shallow depths, where sludge has been incorporated into the soil (European Commission, 2002). The potential problems of biological pollution by sewage sludge are: contamination of surface water and groundwater by pathogens transported by runoff and percolation water (Korentajer, 1991). Even so, the majority of pathogens in sludge are rapidly inactivated in the soil system through sunlight and soil microorganisms (Snyman *et al.*, 1998).

These microorganisms derive from humans who use the sewage systems and who suffer from acute or latent infections or from known and often unknown permanent excretors of pathogens. A pathogen is an organism or substance that is capable of causing disease. These pathogens are excreted from infected individuals via faeces, urine, secretions or excretions of the nose, pharynx and skin depending on the type of infection, and reach the sewage treatment plants by sewers and sanitary installations in homes (Strauch, 1991). The spectrum and quantity of pathogens are extended by other sources connected to the system, such as hospitals, abattoirs, livestock markets and related activities (Strauch, 1991).

Infectious diseases are transmitted primarily through human and animal excreta, particularly faeces. If there are active cases or carriers in the community, then faecal contamination of water sources will result in the causative organisms being present in water. Pathogens in domestic sewage are primarily associated with insoluble solids. Many of these organisms become bound to solids following wastewater treatment and may merely be transferred to wastewater sludge (Bitton, 1994). Wastewater treatment processes concentrate these solids into sewage sludge, thus sewage sludge has higher quantities of pathogens than incoming wastewater (EPA, 1999).

Wastewater treatment processes do not completely remove or inactivate pathogenic and parasitic organisms. Although biological wastewater treatment processes such as lagoons, trickling filters and activated sludge treatment may substantially reduce the number of pathogens in the wastewater, the resulting biological sewage sludge still contains sufficient levels of pathogens to pose a public health and environmental concern (EPA, 1999). Thus, pathogenic agents can survive the treatment processes of wastewater treatment. Some of them are adsorbed to faecal particles and remain within sludge during sedimentation processes, making sewage sludge a concentrate of pathogens (Strauch, 1991).

The actual species and quantity of pathogens present in sewage sludge from a particular municipality depend on the health status of the local community and may vary substantially at different times (EPA, 1999). The four major types of human pathogenic organisms, namely bacteria, viruses, protozoa and helminths may all be present in sludge. These organisms can cause infection or disease if humans or even animals are exposed to sufficient levels. The infective dose, that is, the number of pathogenic organism to which a human must be exposed to become infected, varies depending on the organism and on the health status of the exposed individual (EPA, 1999). While some may cause infections in a susceptible host by a single organism, others may require several hundreds to be present before an infection can be evident. Pathogenicity may vary in severity from mild gastroenteritis to severe and sometimes fatal diarrhoea, dysentery, hepatitis or typhoid depending on the type of pathogen and pathogen load. Thus when reclaimed water or sludge is used on fields producing food crops, public health must be protected.

2.5.3.1 Bacteria

Bacterial pathogens of primary concern in sludge include *Salmonella*, *Shigella*, *Campylobacte*r, *Yersinia*, *Leptosporia* and the enteropathogenic *Escherichia coli*. *Escherichia coli* is abundant in human and animal faeces, where numbers may attain 10^9 per gram of faeces (Bitton, 1994). Of the *Listeria* spp*, Listeria monocytogenes* is the human and animal pathogen capable of causing severe infections like septicemia, encephalitis and meningitis especially in immunocompromised individuals, newborns and pregnant women. Several outbreaks have been associated with contaminated commercial foodstuffs, such as vegetables, milk and meat products on which these bacteria can multiply even at low temperatures (Bubert *et al*., 1999). Bubert *et al*. (1999) have pointed out that contamination of food material does not only occur during food processing, but also begins with the production of raw food materials in the environment. Both *L. monocytogenes* and *L. innocua* have been isolated from various environmental samples such as soil, vegetation and human and animal faeces (Bubert *et al*., 1999). Table 2.7 gives a list of bacteria often encountered in sewage sludge.

The survival times of pathogens in soil are affected by soil moisture, pH, temperature and organic matter (deRopp, 1999) Faecal coliforms can survive for several years under optimum conditions, while the *Salmonella* may survive for a year in rich, moist organic soil. In some instances, survival was shown to exceed a year. For instance Strauch (1991) has demonstrated that *Salmonella* could survive on and in the soil after a single application of sludge in summer for 424 to 820 days, and in winter the survival times were 104 to 350 days. Baloda *et al*. (2001) reported a survival period for *Salmonella* of 299 days in contaminated soil. Other bacteria such as *Streptococcus jaecelis*, *Clostridium botulinium*, *Clostridium tetani*, *Clostridium perfringes* and butyl-butyric clostridia were found in small numbers 7 months after sludge application (Hyde, 1976).

Table 2.7 Bacterial pathogens to be expected in sewage sludge (EPA,1999; Strauch, 1991)

Samonella spp	*Streptococcus*
Shigella spp	*Klebsiella*
Escherichia coli	*Enterobacter*
Yersinia enterocolitica	*Serratia*
Clostridium spp	*Citrobacter*
Leptospira spp	*Proteus*
Mycobacterium spp	*Providencia*
Vibrio cholerae	*Listeria monocytogenes*
Staphylococcus	

2.5.3.2 Viruses

Sludges from wastewater treatment even after anaerobic digestion may contain demonstrable amounts of viruses (Damgaard-Larsen *et al*., 1977). Even digested sludge may contain viruses and remains a potential source of infection. Human enteric viruses are excreted in faeces, and can be shed in high numbers (10^8 to 10^{10} particles per gram of faeces) by infected individuals (Abbaszadgenan *et al*., 1999). The enteric viruses include enteroviruses, rotaviruses, Norwalk and Norwalk-like viruses, adenoviruses and reoviruses. The enteroviruses namely, polioviruses, coxsackie A and B viruses and echoviruses can cause a variety of illness ranging from gastroenteritis to myocarditis and aseptic meningitis (Abbaszadegan *et al*., 1993; Abbaszadegan *et al*., 1999). Their acid stability permits limited replication in the oropharynx, transit through the stomach, and implantation in the lower intestinal tract, where they undergo extensive replication (Kopecka *et al*., 1993). The persistence of enteroviruses in sludge and sludge-amended soil was demonstrated by Damgaard-Larsen *et al*. (1977) and by Straub *et al*. (1994). The virus of greatest potential concern appears to be hepatitis A, a serious disease with appreciable potential for long-term liver damage (Pahren *et al*., 1979).

Recently, human polyomaviruses JC virus and BK virus were also indicated as being present in urban sewage obtained from widely divergent geographical areas in Europe and

Africa (Bofill-Mas *et al*., 2000). The JC virus is etiologically associated with a fatal demyelinating disease known as progressive multifocal leukoencephalopathy which has emerged as a frequent complication of AIDS in human immunodeficiency virus infected individuals. Infection with BK virus has been associated with diseases of the urinary tract including hemorrhagic cystitis and ureteral stenosis (Bofill-Mas *et al*., 2000). Table 2.8 lists viruses that can be expected in sewage sludge.

Table 2.8 Viruses that can be expected in sewage sludge (EPA, 1999; Strauch, 1991)

Enteroviruses	Hepatitis A virus
Coxsackievirus A	Rotavirus
Coxsackievirus B	Astrovirus
Echovirus	Calicivirus
Poliovirus	Coronavirus
Adenovirus	Norwalk and
Reovirus	Norwalk-like viruses

Damgaard-Larssen *et al.* (1977) have shown that virus inactivation under natural conditions is a slow process. Viruses may become eluted and travel far through the soil (Dalamgaard-Larsen *et al*., 1977) which includes both vertical and lateral migration (Straub, 1995). For instance, other enteroviruses such as the coxsackie B3 virus have been isolated 18 m below the soil surface after wastewater discharge (Straub *et al*., 1995). Rainfall and irrigation events may contribute to viral transport (Straub *et al*., 1995). Viruses readily adsorp to soil particles, and this has been reported to prolong their survival (WHO, 1979). However these viruses remain as infectious to humans as free viruses.

Viruses can generally survive for up to six months in cold weather and for three months in warm weather. Enteric viruses in loamy and sandy soil have considerable stability, with the survival times of up to 170 d. Poliovirus has been detected in soil irrigated with infected sewage sludge and effluent after 96 d in winter and 11 d in summer, and on the surface of mature vegetables 23 d after irrigation had ceased (WHO, 1979). Viral survival on crops may be shorter than in the soil if viruses on crops surfaces are directly exposed to detrimental environmental factors such as sunlight and desiccation (Pahren *et al*., 1979; WHO, 1979). However, more prolonged survival can be expected in the moist or more protected parts of plants, such as within the folds of leafy vegetables, in deep stems areas and on rough cracked surfaces of edible roots. It is also likely that viruses can penetrate damaged roots and, under certain conditions enter the stem and leafy parts of edible plants. Once crops are harvested, enteric viruses can survive for prolonged periods during commercial and household storage at low temperature. For example, polioviruses and coxsackieviruses applied to the surface of vegetables can survive for more than two months under refrigeration. The risk of human infection associated with virus-contaminated crops is greatest in the case of fruits and vegetables generally consumed raw. However, there is also a possibility that vegetables consumed after thorough cooking might become infected by contact with kitchen surfaces, utensils and hands contaminated by raw crops (WHO, 1979).

2.5.3.3 Parasites

The parasites most often found in sludge are *Ascaris* species such as *A. lumbricoides* (human intestinal roundworm) and *A. suum* (pig's roundworm) as well as some *Toxocara* and *Trichuris* species (Bitton, 1994). Table 2.9 provides a list of parasites often encountered in sewage sludge.

Table 2.9 Parasites that can be expected in sewage sludge (EPA, 1999; Strauch, 1991)

Entamoeba histolytica	*Echinococcus granulosus*
Giardia lamblia	*Ascaris* spp
Toxoplasma gondii	*Toxocara* spp
Sarcocystis spp	*Trichuris trichiura*
Taenia spp	*Hymenolepis nana*
Diphyllobothrium latum	*Cryptosporidium*

The viability of parasite eggs and larvae varies enormously. *Ascaris* eggs and certain larval stages of trichostrongylids have rather long survival times and may give rise to further infestations of animals after manure has been spread on crops or land. For instance *Ascaris* eggs can survive for two years in soil that has been irrigated with sewage. They were even found on plants that had been irrigated with chlorinated sewage (Strauch, 1991), indicating their resistance to disinfection. Generally, heminth larvae remain viable in slurry during storage and are also resistant to disinfection. Thus, soil will contain infective larvae even one year after spreading of contaminated slurry (Strauch, 1991).

Giardia, protozoan parasites, have been found in sludge in high concentrations in Western Australia where they remain the most common cause of enteric disease (Hu *et al*., 1996). Hu *et al*. (1996) reported that *Giardia* cysts were detectable in sludge that had been stored for over a year. Generally, the parasitic eggs in sludge can survive for prolonged periods. The most noxious are the *Ascaris* eggs and coccidial oocysts as they have high resistance. A single female *Ascaris* produces about 200 000 eggs daily, which are frequently very resistant to environmental effects (Pahren *et al*., 1979). Due to their extreme resistance to adverse influences, infectious eggs can live for long periods in the soil. Thus they may give rise to further infestations after manure has been spread on crops or pastureland. Helminth larvae are usually killed by composting, but often remain viable in slurry during storage. They are resistant to disinfection, and soil will still contain infective larvae one year after spreading of contaminated slurry (Strauch, 1991).

As a significant amount of animal wastes reach municipal sludges, parasites of animal origin are also of concern. These include *Toxocara* spp which are nearly universal in the pet population and have a life cycle in their normal host identical to *Ascaris* in humans. When ingested by a child these worms are unable to complete their normal developmental cycle and continue to wander through the tissues for prolonged periods before eventually succumbing. This results in a chronic and usually mild disease that can last up to a year (Pahren *et al*., 1979)

Toxoplasma gondii is the causal agent for toxoplasmosis, a zoonotic disease (Pahren *et al*., 1979). The parasite undergoes sexual reproduction only in cats, thus they appear to be the major reservoir for perpetuation of toxoplasmosis. The *T. gondii* exhibits three forms, a trophozoite which is an actively proliferating form, a cyst containing hundreds of organisms often found in the brain and muscle of chronically infected animals and the oocyst, which is an infective form found in cat faeces (Pahren *et al.,* 1979). Although in humans the acquired disease is primarily a subclinical infection and rarely comes to medical attention, uterine transmission from mother to unborn child results in severe consequences. Surviving infants often exhibit permanent central nervous system damage, blindness, and mental retardation (Pahren *et al*., 1979).

Also encountered in sludge are the organisms of the genus *Cryptosporidium* (EPA, 1999; Kuckzynska & Shelton, 1999). These are widespread protozoans that develop in epithelial cells lining the digestive and respiratory tracts of vertebrates (Champliaud *et al*., 1998). Of the *Cryptosporidium* species*, C. parvum* is the agent of clinical cryptosidiosis in humans and livestock. The *C. parvum* oocysts are shed by infected mammals and are known to be resistant to standard disinfectants (Champlaiaud *et al*., 1998). Waterborne *C. parvum* oocysts may remain viable for several months (Kuczynska & Shelton, 1999).

2.5.3.4 Applicable Technology

Cell culture assays are the most common means often applied to quantify the presence of pathogenic microorganisms (Del Rosario & Beuchat, 1995; Pillai *et al*., 1996; Janisiewicz *et al*., 1999). A major advantage of the culture-based methods is that there is no doubt about the viability of the organisms detected since they had to grow and multiply in order to be detected. However, some microorganisms have a potential to enter a viable but non-culturable state (Makino *et al*., 2000), thus making it difficult to quantify using cell culture techniques. Although viable and present in the sample, these organisms will not grow on the media used in the tests and this could lead to false negative results. Solomon *et al*. (2002) could detect *E. coli* in lettuce plant tissue using laser scanning confocal microscopy and epifluorescence microscopy.

The most popular technique used in recent years is the Polymerase Chain Reaction (PCR). This technique has received wide application in detecting microorganisms from environmental samples (Straub *et al*., 1994; Straub *et al*., 1995, Guo *et al*., 2000). Although PCR does not distinguish between the infectious and noninfectious pathogens, it nonetheless provides an opportunity to detect those microorganisms that do not replicate in cell culture (Straub *et al*., 1995). Also, both the initial and recurring costs of PCR are substantially less than the costs of cell culture techniques (Abbaszadegan *et al*., 1999). Other techniques used in microbiological monitoring include immunological techniques, and laser scanning (Venter, 2000).

2.5.3.5 Reported Outbreaks

The most common microbial disease is acute gastroenteritis. The symptoms of this condition may be caused by a number of microbial agents including bacteria, viruses and protozoans (Lee & Jones-Lee, 1993). Pathogens often implicated in outbreaks include *E. coli* and *Salmonella*. Generally fruits and vegetables contain nutrients necessary for the rapid growth of food-borne pathogens. If the barrier created by the peel and rind is broken, an opportunity is created for bacterial colonisation (Janisewicz *et al*., 1999).

Escherichia coli was shown to cause hemorrhagic colitis and gastroenteritis in the United States for the first time in 1982 (Janisewicz *et al*., 1999) and it is known to be a leading cause of childhood kidney failure (Hilborn *et al*., 1999). *E. coli* has been isolated with increasing frequency from fresh produce, including cantaloupe and watermelon (Del Rosario

& Beuchat, 1995), apple (Janisiewicz et *al*., 1999) and lettuce (Solomon *et al*., 2002). In 1996, there were four outbreaks of food-borne illness related to contaminated unpastuerized fresh apple cider, including the *E. coli* outbreak which resulted in the death of a sixteen month old girl (Janisewicz *et al*., 1999). Hilborn *et al*. (1999) reported an outbreak of *E. coli* in Illinois, USA, associated with the consumption of mesclun lettuce. In their five-year review based on the surveillance undertaken in Canada and Scotland on the infection caused by *E. coli*, Waters *et al.* (1994) noted that most of the outbreaks reported involved restaurants, retail food outlets, nursing homes, residential homes and hospitals, with the majority of infections involving family groups. They have documented 1 993 cases of *E. coli* infection with 115 cases resulting in haemolytic uraemic syndrome. Rates of infection were highest among children less than 5 years of age, while rates of resultant hospitalization were highest among the elderly (Waters *et al*., 1994). In South Africa, three cases of *E. coli* have been identified in Pretoria, while three isolates of *E. coli* were identified at the central microbiology laboratory of the South African Institute for Medical Research in Johannesburg (Koornhof, 1997).

Del Rosario & Beuchat (1995) demonstrated the ability of *E. coli* to survive and grow on cantaloupe and watermelon and on the external rind surface of these fruits. Using laser scanning confocal microscopy and epifluorescence microscopy, Solomon *et al*. (2002) demonstrated that *E. coli* could enter the lettuce plant through the root system and migrate throughout the edible portion of the plant. This suggests that edible portions of a plant can become contaminated without direct exposure to a pathogen, but rather through transport of the pathogen into a plant by a root system. Cieslak *et al*. (1993) reported that *E. coli* can survive for some time in manured soil. They reported infections of *E. coli* that occurred due to contamination of vegetables by manure. Four cases of bloody diarrhoea were identified in Maine, USA, in 1992. Cases reported included a two year old that developed haemolytic uraemic syndrome and died (Cieslak *et al*., 1993). Another incident of *E. coli* occurred in Montana, USA, in 1995 where 52 residents had bloody diarrhoea. Of these, 13 were hospitalised and 1 developed haemolytic uraemic syndrome. This outbreak was associated with the consumption of lettuce (Ackers *et al*., 1998).

For more than two decades salmonellosis (disease caused by *Salmonella*) has spread steadily among humans and animals in many countries (Strauch, 1991; Guo *et al*., 2000). In the United States, salmonellae are some of the most prevalent food-borne pathogens (Guo *et al*., 2000). They are estimated to cause approximately 1.5 million cases of infection, 15 000 hospitalisations and 500 deaths annually.

Historically, salmonellosis has most often been associated with consumption of contaminated foods of animal origin. However in more recent years changes in agronomic practices and dietary habits as well as increased importation of fresh produce have resulted in the outbreaks being associated with fruits and vegetables (Guo *et al*., 2000). For instance, consumption of fresh tomatoes was epidemiologicaly linked to 176 cases of *Salmonella javiana* infection in Illinois, Michigan, Minnesota and Winsconsin (Guo *et al*., 2000). Asplund & Nurmi (1991) have demonstrated that tomatoes can provide a favourable environment for growth of *Salmonella enteritidis*, *S. infantis* and *S. typhimurium* in spite of their low pH values. Also, several species of *Salmonella* have been implicated in gastroenteritis associated with the consumption of watermelon (Del Rosario & Beuchat, 1995). These species include *Salmonella oranienburg*, *S. javiana*, *S. bareily*, *S. poona* and *S. miami* (del Rosario & Beuchat, 1995). Outbreak due to *Salmonella* was also reported in the UK where people consumed contaminated eggs (De Louvois, 1993).

The *Campylobacter* species present in stools of infected persons are amongst the noxious microorganisms. It is estimated that in the United States *Campylobacter* strains cause more than two million cases of diarrhoea annually (Waage *et al*., 1999). In most cases, the host is a carrier that does not exhibit symptoms, but it may have acquired immunity through an earlier infection. Although these organisms are not able to grow, they may survive in the environment for several weeks at low temperatures. It is estimated that about 500 cells of, for instance, *Campylobacter jejuni* can cause human illness (Waage *et al*., 1999). As the infective dose is very small, minute numbers of *Campylobacter* cells in water or food may be a potential health hazard.

Cryptosporidum parvum is the causal agent of cryptosporidiosis, a worldwide emerging zoonotic disease (Champliaud *et al*., 1998; Kuckynska & Shelton, 1999). One of the most severe outbreaks of cryptosporidiosis was in Milwaukee, USA, where more than 400 000 people were infected (Kuckzynska & Shelton, 1999). *C. parvum* is a particularly serious health threat to immunodeficient individuals such as those suffering from AIDS and cancer patients (Kuczynska and Shelton, 1999). While immunocompetent individuals may experience short-term gastroenteritis that resolve spontaneously, immunocompromised individuals and malnourished children may suffer from chronic life-threatening diarrhoea (Champliaud *et al*., 1998).

Recent studies have shown that, human pathogens in sewage sludge introduced to soil can be conveyed to surface and groundwater used as sources of drinking water, posing a significant public health risk (Hickey & Harkin, 1998).

The agricultural use of municipal sewage sludge for growing animal feed has been a common activity in most countries. Farmers in industrialised countries are increasingly exercising restraint in the agricultural utilisation of sewage sludge, because they feel it a threat to their livestock. In view of the steady increase of latent infections with agents of multi-factorial diseases in the intensive production of poultry, pigs and calves, farmers want to seal their farms off from further danger of infection (Strauch, 1991). Also, there has been increased public scrutiny of the potential health and environmental consequences of this practice. Farmers and the food industry have expressed their concern that agricultural use of sludge may affect the safety of food products and the sustainability of agricultural land and may carry potential economic and liability risks (NRC, 1996).

Of all the pathogens that survive the wastewater purification processes, parasite ova have the longest survival times in the environment. When pathogenic sludge is applied to soil, pathogenic bacteria may continue to survive in the soil for over a year, and roundworm eggs may survive for many years, thereby maintaining the possibility of human reinfection for lengthy periods of time. *Ascaris* transmission by sludge is an extremely important consideration because of the disease’s prevalence and the extreme resistance of the eggs to adverse influences. Among the bacterial populations, the most noxious are the *Salmonella* spp and *E. coli* as there appears to be more outbreaks associated with these pathogens.

2.5.4 TOXIC ORGANIC POLLUTANTS

The role of toxic organic compounds (the most common being pesticides, polychlorinated biphenils, halogenated aliphatics, ethers and aromatic hydrocarbons) in sewage sludges must not be underestimated (Korentajer, 1991). These substances may be transferred to sludge-amended soils. Anxiety may be fuelled because less is known about the environmental consequences of trace organic pollutants in sludge-treated soil, which have been investigated only to a limited extent. This is partly explained by the fact that analytical techniques that can extract, identify and quantify small concentrations of complex mixtures present have only been developed and applied to sewage sludge in recent years (Smith, 1996). However, in contrast to the perceived concerns, the limited information available shows that organic pollutants applied to agricultural soils are unlikely to cause significant environmental problems (Wild & Jones, 1991). Litz (1999) has also developed a standardised evaluation concept based on human toxicity, ecotoxicity and soil related data to predict the relevance of organic pollutants in sewage sludge. Although these organic pollutants are outside the scope of this study future research is required for sludge disposal in South Africa.

2.6 SOCIAL ACCEPTABILITY OF USING SEWAGE SLUDGE IN AGRICULTURE

Agricultural recycling of sewage sludge is highly sensitive to the effects of adverse publicity on sludge application operations. In particular, the misinterpretation by the media of isolated reports of potential problems are frequently taken out of context, even though the overall risk to the environment from sludge recycling to agricultural land is minimal. Such adverse publicity may lead to unnecessary restrictions being imposed as the regulators adopt a highly precautionary approach when these issues are raised to avoid undue confrontation from an alarmed, but scientifically uninformed public (Smith, 1996). Evans & Atkins (1999) have documented that following a series of food scares after long-term application of sludge on land in the UK, it was necessary to change biosolids treatment and application methods to reduce contamination risk to satisfy the needs of stakeholders. This emphasised the fact that customer satisfaction and public acceptance are keys to sustainable use of sewage sludge on agricultural soils.

2.7 CONCLUSIONS

The agricultural recycling of sewage sludge is not a new concept, although very little research has been done under S.A. conditions. Sewage sludge has a potential as an organic fertilizer for agricultural practices, considering the possible improvement of the nutrient status and organic content of soils, as well as the soil physical properties (e.g. soil water holding capacity).

In spite of the proven benefits of sewage sludge application, its use in agriculture is limited by factors such as the presence of pathogenic organisms, nitrate contamination of groundwater, toxic organic pollutants and heavy metal transfer in the food chain. Of these factors the transfer of heavy metals from soil to crops appears to be the greatest potential threat. Consequently most of the existing guidelines for sludge application on land limit the amounts of sludge that can be applied on land, according to their heavy metal content (particularly the four main sludge-borne metals; Pb, Cd, Cu and Zn) (Korentajer, 1991). Previous research has shown that crop cultivation on sludge-amended soils (specifically maize and oats cultivation) can be done without contamination risk to soils or phytotoxic uptake by the crop plants. These observations were based on short-term experiments. The possible long-term heavy metal accumulation in soils and subsequent uptake by crops does pose a threat to the sustainability of sewage sludge on agricultural soils. Therefore strict monitoring procedures are necessary and the correct management practices (e.g. controlling soil pH above 6.5 through liming). Provided that these safety measures are maintained, application of sewage sludge on agricultural land can provide one of the major disposal routes for the increasing amounts of sludge being generated in S.A. This fact is further emphasised considering that the crop cultivation, especially of maize and oats, is one of the major food supplies to people in our country and that agricultural soils in S.A. are degraded and low in organic content.

CHAPTER 3: SOUTH AFRICAN BASED RESEARCH RESULTS WITH REGARD TO THE USE OF SEWAGE SLUDGE IN AGRICULTURE – RESEARCH METHODOLOGY

3.1 INTRODUCTION

Several trials were conducted to determine the effect of sewage sludge use in agriculture in South Africa. Each one of the sections below represents a trial (or set of trials) that was done to determine: 1) the mobility of metals in long-term sewage sludge-amended soils; 2) the effect of sewage sludge amendment on soil metal content and metal uptake by plants after a single application; 3) nitrogen release in soil after sewage sludge application and; 4) pathogen persistence in sewage sludge-amended soil. A thorough summary of the research methodology of trials is given in this chapter and the complete trials are listed and discussed in Appendices 1 through 13.

Two laboratory experiments were conducted on soil from three sacrificial sites to determine metal mobility after long-term continuous sludge application with and without the addition of lime. A further laboratory experiment was conducted to determine the nitrogen mineralisation rate of applied sludge compared to inorganic fertilizer N. The bioavailability of heavy metals was determined in four greenhouse trials and three (one repeated later) field trials. The greenhouse trials were conducted using maize, sunflower, oats, and soybean seedlings and the field trials using maize, sunflower and oats plants. To determine the persistence of pathogens in soil after sludge application a trial was conducted using potatoes in pots in the greenhouse.

3.2 RESEARCH METHODOLOGY

3.2.1 LITERATURE REVIEW

A literature review was done on the main aspects concerning the use of sewage sludge in agriculture in terms of its risk and benefits. These included the risks of heavy metals, nutrient pollution, utilisation and effect on yield as well as persistence of pathogens. Different scientific journals and other publications (books and pamphlets to a lesser extent) were consulted. The review was broad in terms of the scope of the subject but could by no means be extensive due purely to the volume of literature published world-wide on the subject.

3.2.2 LABORATORY SCALE EXPERIMENTS

3.2.2.1 Heavy Metal Fractions from Three Sacrificial Sites (Appendix 1)

For the purpose of this study three soils were collected in the Gauteng province around Pretoria. Soil 1 was a gravely sandy loam on granitic parent material from Rooiwal (north of Pretoria), Soil 2 was a sandy clay loam and Soil 3 a loam both on dolomitic parent material from Hartbeesfontein (ERWAT site). Soils 1 and 3 were true sacrificial soils with only a regular ploughing and no additions except regular sludge addition in suspended form. Soil 2 was used for agricultural purposes and therefore received regular additions of lime with sewage sludge also in the suspended form. These soils represent a very restricted view of the complete picture but should be adequate to illustrate some of the issues concerning rehabilitation and management of sacrificial lands.

The soils were analysed for texture, organic C and pH according to the methods prescribed by The Non-affiliated Soil Analysis Work Committee (1990). In order to determine the Effective Cation Exchange Capacity (ECEC) and extractable cations of the soil at current pH levels a $BaCl_2$ extraction was done. The Hendershot & Duquette (1986) method gave a very good indication of the ECEC of soils. The method was adapted as follows: 5 g of soil was shaken with 50 cm^3 0.1 M $BaCl_2$ in a glass bottle on a horizontal shaker for 1 h and filtered afterwards. The metals Ca, Mg, K, Na, Al, Mn, Fe, Cu, Zn, Pb, and Cd were determined through Atomic Absorption Spectrophotometry and Ni and V through ICP-MS. Due to Cl interference in the determination of Cr by ICP-MS, a 0.2 M NH_4NO_3 solution was used, with the same procedure and quantities as stated for the $BaCl_2$ extraction, to determine Cr within the soils.

The water-soluble metals were determined through a saturated paste extract according to the method described by The Non-affiliated Soil Analysis Work Committee (1990). The metals Ca, Mg, K, Na, Al, Mn, Fe, Cu, Zn, Pb, and Cd were again determined through Atomic Absorption Spectrophotometry and Ni, Cr and V through ICP-MS. Extractable acidity and Al was determined through a 1N KCl extraction and organic carbon according to the Walkley-Black method, as described by the Non-affiliated Soil Analysis Work Committee (1990).

An EPA 3050 (U.S. Environmental Protection Agency, 1986) extraction was done to determine the concentration of the total sorbed Mn, Cu, Zn, Pb, Ni, Cd, Cr, and V. All the metals were determined by ICP-MS. According to Risser & Baker (1990) this method gives a reliable indication of metals added to soils as non-silicates from industrial sources and therefore metals that are potentially mobile in the environment (Soon & Abboud, 1993).

An EDTA extraction (The Non-affiliated Soil Analysis Work Committee, 1990) was used to determine potentially plant available metals. EDTA was found or proposed by many researchers to give a very good indication of the pollution hazard of heavy metals in soils as well as being a reliable test for predicting plant available metals (Cajuste & Laird, 2000).

The results were statistically analysed using the SAS® System statistical program to obtain the Analysis of Variance (ANOVA) and the LSD – Tukey to determine significant differences.

3.2.2.2 Effect of Liming on pH and Metal Extractability (Appendix 2)

The buffer capacity of the two acid soils (1 and 3 used in the trial discussed above) was determined with a $Ca(OH)_2$ buffer (method as described in Van Der Waals & Claassens, 2002). The required amount of $CaCO_3$ (according to the buffer determination) for a pH of 6.5 was added to each soil in 7.5 kg pots (with 4 repetitions) and the soil incubated for 3 months with regular watering. After sampling the soil and finding only a slight change in pH it was decided to add an equal amount of lime ($CaCO_3$) and incubate the soil again for the same period of time. The total amount of lime ($CaCO_3$) added amounted to the equivalent of 45 t ha^{-1}.

After the second incubation period a representative sample was taken from each pot. On the limed soil and a sample from the original soil a $BaCl_2$ (method described in section 3.2.2.1) and EDTA (The Non-affiliated Soil Analysis Work Committee, 1990) extraction was done. The samples were tested at the same time to minimise experimental error differences between them. Cr, V, and Ni were not tested in this study due to the contradicting results that were found with the cations that were tested first. The reasoning was to not expend time and finances on these elements in this trial but to rather focus a new study on these contradictions and then include Cr, V, and Ni. The proposed study is discussed in Chapter 5.

3.2.2.3 Comparison of Mineralisation Rates of Sludge and Commercial Fertilizer Nitrogen Sources (Appendix 11)

A laboratory incubation trial of sewage sludge-amended soil was done under constant temperature and moisture conditions. Treatments consisted of a set of different quantities of sewage sludge and commercial fertilizer applications with three replications. The sludge was applied at the equivalent of 5, 10 and 20 t ha^{-1} dried sewage sludge per 20 cm cultivation depth. The commercial fertilizer treatments were calculated based on the assumption that only 30% of the total N content of the sludge would be available in the short-term. According to the 'Permissible Utilisation and Disposal of Sewage Sludge' (WRC, 1997), ± 30% of the total N in the sludge becomes plant available in the first year. A dark red sandy clay loam topsoil from the Hatfield Experimental farm, University of Pretoria was used. The soil was dried overnight at 40°C and sieved through a 2 mm sieve. Some of the chemical and physical properties of the soils are presented in Table 3.1. A low metal sludge, obtained from Olifantsfontein sewage plant (ERWAT), was used for the incubation trial. It was air dried, ground and sieved through a 2 mm sieve. Some of the chemical properties of the specific sludge are presented in Table 3.2.

Table 3.1 Some physical and chemical properties of the soil used in the trial

Parameter	Value
pH (H_2O)	6.4
Electric conductivity (mS m^{-1})	8.3
NO_3^--N (mg kg^{-1})	3.55
NH_4^+-N (mg kg^{-1})	1.37
C %	0.63
Textural classes	
% Clay	26
% Silt	8.2
% Sand	65.8

Table 3.2 Carbon and nitrogen quantities in the sludge used in the trial

Parameter	Value
Total N	3.14%
Total C	3.63%
C:N	1.15

After the 7 days pre-incubation, on 'Day 0', dried sewage sludge and commercial fertiliser were applied according to the different treatment levels, thoroughly mixed, brought to field capacity and placed in an incubation oven. Triplicate samples were taken at day 1, 3, 7, 14, 28, 42 and 63, and were analysed for NH_4^+ and NO_3^-.

Field capacity of the soil

Incubation studies are best done when the soil is at field capacity. To determine the field capacity (FC) of the soil, three open-end glass cylinders (40 cm $\times$ 3.5 cm) were used. One end of the cylinder was covered with filter paper, and filled with 10 cm of soil. This layer of soil was compacted until it resembled the density of the natural soil. This process was repeated until the cylinder was two thirds full. Water (25 ml) was added to each cylinder and a rubber stopper was placed loosely on top, to prevent evaporation. The cylinders were left for 24 h, until movement of the wetting front ceased. Samples were taken from the middle of each cylinder, weighed and dried at 100°C for 24 h, and weighed again. The mean value of the water content (15.29%) was assumed to be FC.

Laboratory analysis

A 1M KCl solution (100 ml) was added to the 50 g soil samples, shaken for an hour, and filtered afterwards. From this filtrate, a 50 ml aliquot was used for the analysis of NH_4^+ and NO_3^-. To this aliquot, 20 ml of a 50% NaOH solution was added to volatilise the NH_4^+-N. The solution was distilled on a Büchi distillatory and bubbled through a boric acid solution, with a colour indicator. After distillation, the, the boric acid solution was titrated with HCl. The volume of the HCl used for the titration was used to calculate the NH_4^+-N content in the incubated soil. This value is an indication of the concentration NH_4^+ in the solution. To the remaining KCl-aliquot, Davarda alloy was added to reduce the NO_3^- to NH_4^+ and the process was repeated for NO_3^- content.

Statistical analysis

Statistical analyses were carried out on the results by using the GLM statistical procedure. The different treatments, intervals, as well as treatment by interval interactions were tested. Significant differences between treatments and grouping were calculated using the Tukey test.

3.2.3 GREENHOUSE EXPERIMENTS

3.2.3.1 Plant-Soil Interactions of Sludge-Borne Heavy Metals and the Effect on Maize, Oats, Sunflower and Soybean Seedling Growth (Appendices 3, 6, 8 and 10)

Dewatered sludge samples representing a low metal sludge (Sludge 1) and high metal sludge (Sludge 2) were collected. Lead, Cd, Cu and Zn were determined in NH_4-EDTA (The Non-affiliated Soil Analysis Work Committee, 1990) and EPA3050 (U.S. Environmental Protection Agency, 1986) extracts. Three different soil types were collected in the broader Gauteng area, South Africa namely: a dark structured clayey soil (pH 8.8 – Bonnheim), a bleached sandy soil (pH 5.3 – Longlands), and a red sandy loam soil (pH 7.6 – Hutton). The experiments were done under greenhouse conditions and consisted of a randomised block design with four replications and four treatments as presented in Table 3.3. Fertilizer was applied to the soil in the form of 2:3:2 (28) (commercial grade) at an equivalent rate of 600 kg ha^{-1}.

The sludges (dried) and inorganic fertilizer was added to the different soil types one week before planting at the appropriate rates presented in Table 3.3. Five maize (*Zea mays* L., cv. Sensako 2472) seeds, nine oats seeds (*Avena sativa* L., cv. Sensako 001), five sunflower (*Helianthus annuus* L.) seeds and five soybean (*Glycine sojae* L.) seeds were planted per pot in each of the trials respectively. Soil samples were analysed at the start of the experiment after sludge application and after 28 d of growth. The Pb, Cd, Zn and Cu content in the soil was determined through the EPA 3050 method (U.S. Environmental Protection Agency, 1986) and the potentially available fraction was determined through the NH_4-EDTA method (The Non-affiliated Soil Analysis Work Committee, 1990).

The seedlings were harvested after 28 d of growth, dried at 60°C for 48 h to determine dry mass and digested to determine the total heavy metal content. The accumulation of metals in the plant parts when taken up from the soil was determined as the f factor, also known as the transfer coefficient (Smith, 1996), using the following formula:

f = $[M]_p / [M]_s$

Where $[M]_p$ = metal concentration in plant tissue (mg kg^{-1})

$[M]_s$ = metal concentration in soil (mg kg^{-1})

Table 3.3 List of the soil treatments used in the experimental design of the greenhouse study on the effect of sewage sludge on seedling growth on three soil types over 28 d

	Crop			
Treatment	Maize	Sunflower	Soybean	Oats
Sludge 1† (t_{dry} ha^{-1})	24	24	8	24
Sludge 2‡ (t_{dry} ha^{-1})	24	24	8	24
Positive control	Inorganic fertilizer applied to soil types as recommended			
Negative control	No additions to soil			
Repetitions				
Treated soil	80	80	54	80
Control	16	16	9	16

† Low metal domestic sludge

‡ High metal domestic sludge

3.2.3.2 Persistence of Human Pathogens in Crops Grown on Sewage Sludge-Treated Soils (Appendix 12)

Greenhouse experiments

Due to potato (*Solanum tuberosum*) being a staple food in South Africa it was chosen for this experiment. A low metal sludge, obtained from Olifantsfontein, and a high metal sludge from Rondebult, both these plants belonging to the East Rand Water Care Company (ERWAT), was applied to soil in pots. The trials were conducted in greenhouses on the experimental farm of the University of Pretoria under controlled conditions (temp 25 – 28^{o}C) for a three month period from May to July.

The potatoes were grown in 4 kg soil in pots. The trial comprised of 8 control pots, in which potatoes were grown in soil with no sludge. Sludge was added to the soil in other pots at an application rate of 8 t ha^{-1} for 8 pots in which low metal sludge (LMS) was added and another 8 pots that were given high metal sludge (HMS). Another set of 16 pots contained sludge at an application rate of 16 t ha^{-1} for both the LMS and HMS. Eight of these pots were dedicated for LMS while the remaining eight were used for HMS. For each of the eight pots in which potatoes were grown, there was another set of eight pots that only contained soil and sludge but no plants. These pots were subjected to the same conditions as all other pots and were used for regular sample collection. Soil samples were collected every second week. Other than water, which was added on every alternative day, no other nutrients were added to the pots. At the end of the experiment, potato samples were collected for microbial analysis. A portion of these potatoes was be subjected to Polymerase Chain Reaction.

Microbiological determinations

Procedures for analyses of faecal coliforms, *E. coli* , *Samonella* and *Ascaris* are those developed by the East Rand Water Care Company (ERWAT).

Salmonella *analysis*

All chemicals used for this analysis were purchased from Oxoid. A 1 g of sample (soil containing sludge) was placed in 10 ml buffered Peptone Water (Batch number 253802), then mixed and incubated at 35^{o}C for 18-24 h. About 0.1 ml of the mixture was transferred to 10 ml Rappaport VS Broth (Batch number 239145), and incubated at 44^{o}C for 24 h. The enrichment broth was subcultured by streaking onto Brilliant Green agar (Batch number 216866) and incubated at 35^{o}C for 18-24 h. A presumptive positive result was suspected if red colonies occured. Selected colonies were then subcultured onto Xylose-Lysine-Desoxycholate (XLD) agar (Batch number 230180), and incubated at 35^{o}C for 18-24 h. Occurrence of black colonies suggested the presence of *Salmonella*.

Analysis of faecal coliforms

About 1 g of sample was withdrawn and added to 9 ml peptone broth and incubated overnight. As the sample was too concentrated, serial dilutions were made. This mixture was then filtered by mounting the funnel onto the filter holding assembly connected to a vacuum pump. Using sterile forceps the sterile 0.45 μm gridded membrane filter (Sartorius, Batch number 0702 1/406 0102/53) was placed on the receptable of the filter holding assembly. A small volume of sterile distilled/deionised water was placed into the funnel prior to addition of the sample mixture to ensure an even distribution of the sample. When necessary, serial dilutions were made. When filtration was completed, the membrane filter was removed with sterile forceps and rolled onto MFC agar (Difco, Batch number 1162000). The petri dishes were inverted and incubated at 44.5 $\pm$ 0.5^{o}C for 18-24 h. All blue colonies were counted using a colony counter. Results were expressed as colony forming units per gram (CFU g^{-1}).

E. coli analysis

The membrane from the faecal coliform was transferred to the nutrient agar substrate containing MUG (Difco, Batch number 0249000). The plates were then incubated together with one blank at 35± 0.5°C for 4 h. Colonies were observed using a long wavelength ultraviolet light source for the fluorescence on the periphery. Results were expressed as CFU g^{-1}.

Ascaris *analysis*

For determination of *Ascaris* ova, it is essential that the moisture content be estimated. This is done by weighing 50 g of the sample into a weighed dish, and dried overnight. Another 10 g of the sample was weighed into a beaker and treated with an alkaline soap thoroughly mixing with an orange stick. The sample was then washed through a treble Visser filter (comprising mesh sizes 100 μm; 80 μm and 35 μm), by rinsing repeatedly with a strong jet of tap water. The remains in the outer filter were rinsed with tap water and centrifuged at 3000 g for 3 min. Using a Pasteur pipette, the supernatant was discarded. The pellet was resuspended in zinc sulphate (40%, 71 g/100 ml H_2O) and centrifuged further for 3 min at 3000 g. Using a Pasteur pipette, the supernatant was transferred to a vacuum filtering system, using a filter of 12 μm (Millipore, Batch number R2DN50854). The zinc sulphate was rinsed off with distilled water to avoid recrystalisation. The membrane filter was then placed in a glass petri dish and dried at 35°C. A circular weight is usually placed around the edges of the membrane to prevent curling. Once dried, the filter was sliced across its diameter and each of the halves was placed onto a slide using clear glue. Immersion oil was spread over the filter using an orange stick. The slide was then observed under the microscope to count the *Ascaris* ova.

3.2.4 FIELD EXPERIMENTS

3.2.4.1 The Cultivation of Field-Grown Maize, Sunflower and Oats on Different Sewage Sludge Dosages (Appendices 4, 5, 7 and 9)

Anaerobically digested sludge samples, dewatered to ± 20% m/m solids were collected at a Water Care Works of ERWAT, North-East Gauteng, South Africa. Analyses were done for moisture, nutrients and heavy metals (EPA3050 method – U.S. Environmental Protection Agency, 1986).

The experimental site was at Hartbeestfontein farm in the Bapsfontein area, Gauteng, South Africa. Plots (7 m x 7 m) were arranged in a randomised block design with four replications and four treatments as presented in Table 3.4. Lime was applied at 2 t ha^{-1} to the loam soil to raise the pH to the appropriate levels for the cultivation of crops. Sewage sludge and inorganic fertilizer was added to the soil one week before the desired planting date at the appropriate rates presented in Table 3.4. Seeds of the different crops were planted one week after sludge application using standard planting techniques.

Soil samples were collected to a depth of 15 cm from each of the plots at the start of the experiment, after sludge application and after four months of growth at the beginning of ear-formation. Samples were analysed for N, P, K, Ca, Mg, pH and Resistance as well as heavy metals (Mn, Zn, Cu and Pb) (EPA3050 method – U.S. Environmental Protection Agency, 1986). Cd was not tested due to low levels.

Plant material samples were collected from each of the plots. Leaf and stalk samples were randomly collected at the pre-tasseling stage of growth, while ear samples were harvested at the mature stage of ear-formation. All plant material samples were analysed for heavy metals (Zn, Pb, Cu and Mn) and the transfer coefficient determined. Yield differences between treatments were measured in terms of the average amount of ears per plant, wet mass of ears and percentage dry weight of the ears.

Table 3.4 Treatments used in the randomised block design for the field study to determine the effect of sewage sludge applied to a loam soil

Treatments	Soil application
Exp. 1	Sewage sludge applied to soil at 12.5 t_{dry} ha^{-1} (61.25 kg dewatered sludge 49 m^{-2})
Exp. 2	Sewage sludge applied to soil at 25 t_{dry} ha^{-1} (122.5 kg dewatered sludge 49 m^{-2})
Positive Control	Inorganic fertiliser (2:3:2: (28)) applied to soil at 600kg ha^{-1} (2.94 kg 49 m^{-2})
Negative control	No applications to soil

3.2.5 SOCIAL ACCEPTABILITY OF USING SEWAGE SLUDGE IN AGRICULTURAL PRACTICES (APPENDIX 13)

A questionnaire was drafted and posted to 50 randomly selected addresses selected from the Pretoria and East Gauteng telephone directory. Since only 6 responses were received, the questionnaire was e-mailed to 35 individuals and 56 interviews were done. The population surveyed aimed to include the man on the street, supermarkets and shops selling vegetables and farmers using sewage sludge as a soil amendment.

CHAPTER 4: SOUTH AFRICAN BASED RESEARCH RESULTS WITH REGARD TO THE USE OF SEWAGE SLUDGE IN AGRICULTURE – RESULTS AND DISCUSSION

4.1 INTRODUCTION

Rapid degradation of agricultural soils through erosion and nutrient depletion is one of the most serious problems in South African agriculture today. One of the readily available sources of organic material suitable for soil amendment is sewage sludge. To ensure the safe and sustainable use of sludge as a soil conditioner and amendment a better understanding of different crop responses, soil, environmental conditions and sludges is needed.

In wastewater treatment, sludges are derived from various processes and each type of sludge has its own characteristics. Sewage sludge is basically classified as three types, A, B and C, in a decreasing order of potential to cause odour nuisances, fly breeding and transmission of pathogenic organisms to man and the environment. The type D sludge is of similar hygienic quality as type C, but since it is produced for unrestricted use on land at an application rate of 8 t ha^{-1} $year^{-1}$, the metal and inorganic contents are limited to acceptably low levels (WRC, 1997).

Sludge acts as a soil conditioner to facilitate nutrient transport, increase water retention and improve soil tilth (Ekama, 1993). An increasing problem around the world is the gradual decrease in the organic matter content of cultivated soils. In warm areas, such as Southern Africa, the process of organic matter decomposition by microorganisms is high and the addition of sludge could provide a measure to maintain the organic status of soil and decrease the danger of runoff and erosion (Korentajer, 1991). Sludge also serves as a partial replacement for chemical fertilizers. The major plant nutrients (nitrogen, phosphorus and potassium) in sludges are not substantially removed during sludge processing, therefore the nutrients could improve the soil's nutritional status after sludge application. Sludge also contains essential plant micronutrients such as Cu, Zn, Mn and B (Ekama, 1993). However, the long-term benefit of sewage sludge to land as an alternative source of plant nutrients is frequently limited by potentially toxic contents of heavy metals, human pathogens (Palmer, 1993), and potential groundwater pollution due to nutrient leaching (Smith, 1996).

In areas where wastewater from metal-related and other industries is accepted into wastewater treatment plants, high levels of metals can be expected in the sludge (WRC, 1997). The Cu and Zn present in sewage sludge is mainly from domestic sources, while the major source for Cd and Pb is industrial (Smith, 1996). South African guidelines for heavy metals in sewage sludge seem very strict when compared to guidelines from other countries. This limits the unrestricted use of sewage sludge on agricultural land. The interpretation of the S.A. guidelines is not well understood and is currently under revision.

The ability of crops to take up these sludge-borne heavy metals also plays an important role, not only seasonally (summer and winter crops), but also through differences in specific crop cultivars (Alloway, 1995). Relative differences in the uptake of metal ions between plant species and cultivars are genetically controlled. Several factors influence the bioavailability of heavy metals to crop plants. These include soil pH, CEC and metal speciation. Soil pH value has been identified as the single most important soil factor controlling the availability of heavy metals in sludge-treated soil (Løbersli, *et al*., 1991; Smith, 1996; Hooda *et al*., 1997; Chaney *et al*., 2001).

The pathogenic organisms present in sewage sludge can be potentially dangerous to human health (Higgins, 1984; Ross *et al.,* 1992). There are five main types of pathogens in sludge namely: bacteria, viruses, fungi and yeasts, parasitic worms and protozoa. The potential problems of biological pollution by sewage sludge are the contamination of surface water and groundwater by pathogens transported by runoff and percolation water (Korentajer, 1991). Even so, the majority of pathogens in sludge are rapidly inactivated in the soil system through sunlight and soil microorganisms (Snyman *et al.*, 1998).

Toxic organic compounds (the most common being pesticides, polychlorinated biphenils, halogenated aliphatics, ethers and aromatic hydrocarbons) may be transferred to sludge-amended soils. However, in contrast to the perceived concerns, the limited information available shows that organic pollutants applied to agricultural soils are unlikely to cause significant environmental problems (Wild & Jones, 1991).

Agricultural recycling of sewage sludge is highly sensitive to the effects of adverse publicity on sludge application operations. In particular, the misinterpretation by the media of isolated reports of potential problems are frequently taken out of context, even though the overall risk to the environment from sludge recycling to agricultural land is minimal. Such adverse publicity may lead to unnecessary restrictions being imposed as the regulators adopt a highly precautionary approach when these issues are raised to avoid undue confrontation from an alarmed, but scientifically uninformed public (Smith, 1996).

4.2 LABORATORY SCALE EXPERIMENTS

4.2.1 HEAVY METAL FRACTIONS FROM THREE SACRIFICIAL SITES

4.2.1.1 Chemical and Physical Properties

Some of the chemical and physical properties (Appendix 1) of the three soils discussed in Chapter 3 are presented in Table 4.1. Although Soils 2 and 3 are very similar in origin it is suspected that the high organic material content was not completely oxidised after the particle size analysis and caused incomplete deflocculation of clay particles. There is a distinct difference between the pH values of Soils 1 and 3 and Soil 2. This is the result of the difference in management (regular additions of lime) of Soil 2. The organic carbon content of the soils also differed substantially and this is ascribed to the regular liming of Soil 2 that led to increased mineralisation rates.

Table 4.1 Some chemical and physical properties of the three soils

Property		Soil		
		1	2	3
Sand (%)		70.2	45.7	46.4*
Silt (%)		19.7	24.3	41.9*
Clay (%)		10.1	30.0	11.7*
Texture		SaLm	SaClLm	SaClLm
Organic C (%)		2.6	1.0	2.7
pH	Water	4.2	6.5	4.0
	$CaCl_2$	3.8	5.9	3.7
	KCl	3.3	5.5	3.5

* Values not considered accurate – see text

4.2.1.2 Exchangeable and Water Soluble Cations

The regular liming of Soil 2 is reflected in the difference in Ca and Mg levels compared to Soils 1 and 3 (Table 4.2 and Appendix 1). Of significance is the large difference concerning the exchangeable Al between Soil 1 and 3 although they have similar pH values (less than 10% for Soil 1 and more than 40% for Soil 2). The opposite is the case with exchangeable H. Coupled to the difference in Al is the difference in Fe and Mn with Soil 3 having much higher levels than Soil 1. The higher Al levels in Soil 3 indicate a much larger component of variable charge than Soil 1 as is reflected in the lower CEC of Soil 3 (finer texture) compared to Soil 1 (coarser texture). The CEC of Soil 1 is most likely dominated by the high organic material content and the CEC of Soil 3 by a combination of sesquioxides and organic material. The high clay content of Soil 2 is most likely the main contributor to the CEC of this soil due to the near neutral pH and lower organic C.

Table 4.2 Extractable, water soluble, and exchangeable cations of the three soils

Procedure and calculation	Cations ($cmol_c\ kg^{-1}$ soil)								
	Ca	Mg	K	Na	Al	H	Mn	Fe	Total
					Soil 1				
$BaCl_2$ extraction	2.16	0.54	1.56	0.08	0.39	1.03	0.06	0.02	5.84
Saturated paste	0.25	0.10	0.12	0.02	0.01	nd†	0.00	0.00	0.50
Exchangeable	1.91	0.44	1.44	0.06	0.38	1.03	0.06	0.02	5.34
					Soil 2				
$BaCl_2$ extractable	3.00	2.81	0.60	0.05	0.03	0.04	0.18	0.00	6.71
Saturated paste	0.12	0.17	0.04	0.01	0.00	nd†	0.00	0.00	0.34
Exchangeable	2.88	2.64	0.56	0.04	0.03	0.04	0.18	0.00	6.37
					Soil 3				
$BaCl_2$ extractable	0.92	0.24	0.73	0.05	1.72	0.56	0.11	0.00	4.32
Saturated paste	0.24	0.07	0.13	0.05	0.02	nd†	0.02	0.00	0.54
Exchangeable	0.68	0.16	0.60	0.00	1.70	0.56	0.09	0.00	3.78

† Not determined

4.2.1.3 Heavy Metals

Table 4.3 presents the extractable metal levels for Soils 1, 2, and 3 (Appendix 1). When the EPA 3050 values are considered, the maximum permissible levels in the guidelines (WRC, 1997) are exceeded in all the cases except Ni for Soil 1 and Cd for Soil 2. This is in line with the comments concerning the "total versus bio-available levels" argument (Beckett, 1989; McLaughlin *et al*., 2000) where "total" levels often exceed those stipulated as "maximum allowable". Plant uptake is often more readily correlated with exchangeable metal levels.

The picture changes considerably when the EDTA values are considered. Only Zn for Soil 1 and Cu for Soil 3 are higher than the values proposed by Bruemmer & Van Der Merwe (1989). In most cases the $BaCl_2$ values are much lower and the water-soluble levels are even more so. The highest values again are for Zn, Ni, and Cu in Soils 1 and 3, and Zn and Ni in Soil 2. The levels would indicate a very low immediate risk of leaching or plant uptake of these metals. Even though the EPA 3050 levels for the three soils are very similar, as well as most of the levels in the other extractants, they are lower in Soil 2 than the two acid soils indicating a smaller total load of sludge.

Table 4.3 Metals extracted from the three soils with four extractants

Soil	Metal (mg kg^{-1}) $^{\#}$						
	Cd	Cr	Cu	Ni	Pb	V‡	Zn
	EPA 3050						
1	4.02*	151.78*	176.88*	26.85	246.19*	59.10	359.10*
2	1.48	250.04*	118.18*	102.00*	39.37*	141.61	101.63*
3	2.62*	313.13*	300.55*	72.64*	76.80*	157.36	106.39*
	EDTA						
1	0.30^{bc}	0.058^{bcd}	41.13^{b}	5.68^{b}	9.85^{a}	0.65^{a}	117.03^{b}*
2	0.68^{a}	0.11^{b}	36.82^{c}	6.60^{a}	10.36^{a}	0.64^{a}	30.96^{c}
3	0.50^{ab}	0.30^{a}	80.77^{a}*	4.04^{d}	8.40^{a}	0.18^{b}	28.82^{c}
	$BaCl_2$						
1	0.30^{abc}	0.04^{cd}	3.31^{e}	4.97^{c}	1.85^{b}	0.19^{b}	151.59^{a}
2	0.24^{bc}	0.04^{cd}	0.03^{f}	1.74^{e}	0.86^{b}	0.06^{c}	5.45^{e}
3	0.30^{abc}	0.09^{bc}	14.10^{d}	4.00^{d}	1.57^{b}	0	18.49^{d}
	Water						
1	0.01^{c}	0.006^{d}	0.17^{f}	0.26^{g}	0	0.006^{dc}	4.18^{e}
2	0.004^{c}	0.007^{d}	0.05^{f}	0.03^{g}	0	0.004^{dc}	0.25^{e}
3	0.04^{c}	0.013^{d}	0.40^{f}	0.54^{f}	0.01^{b}	0.003^{dc}	5.06^{e}

‡ Vanadium is not stipulated in the guidelines
\# Values with the same letter in a specific column indicate no significant difference
* Values exceeding guideline levels (WRC, 1997).

The EDTA, $BaCl_2$, and water extractable metals are presented as a percentage of the EPA 3050 extractable levels in Figures 4.1, 4.2, and 4.3 for Soils 1, 2, and 3 respectively. These metals varied widely in the fraction extracted with water, it being the weakest extracting agent as expected. An interesting trend for most of the metals (especially in the case of Cd, Ni, and Zn and to a lesser extent Cu and Pb) is the similarity between the EDTA and $BaCl_2$ extractable metals for the two acid soils and the larger difference for the higher pH soil (Fig. 4.4). This would indicate a larger possibility of plant uptake of these metals from the acid soils than from the neutral soil. This is to be expected as a pH dependant trend even though the EDTA levels are similar for the three soils indicating similar “potential availability”. Cr and V did not exhibit such trends. Rather, the neutral pH soil (Soil 2) had a larger $BaCl_2$

extractable fraction than the acid soil (Soil 3) indicating the $BaCl_2$ extractable fractions are possibly dominated by Cr(VI) and V(V) in these soils. The extractable Cr and V expressed as a percentage of the EPA 3050 values (Figs. 4.1, 4.2 and 4.3) were very low though (compare values in Table 4.3) in all the soils, indicating a very slight pollution risk.

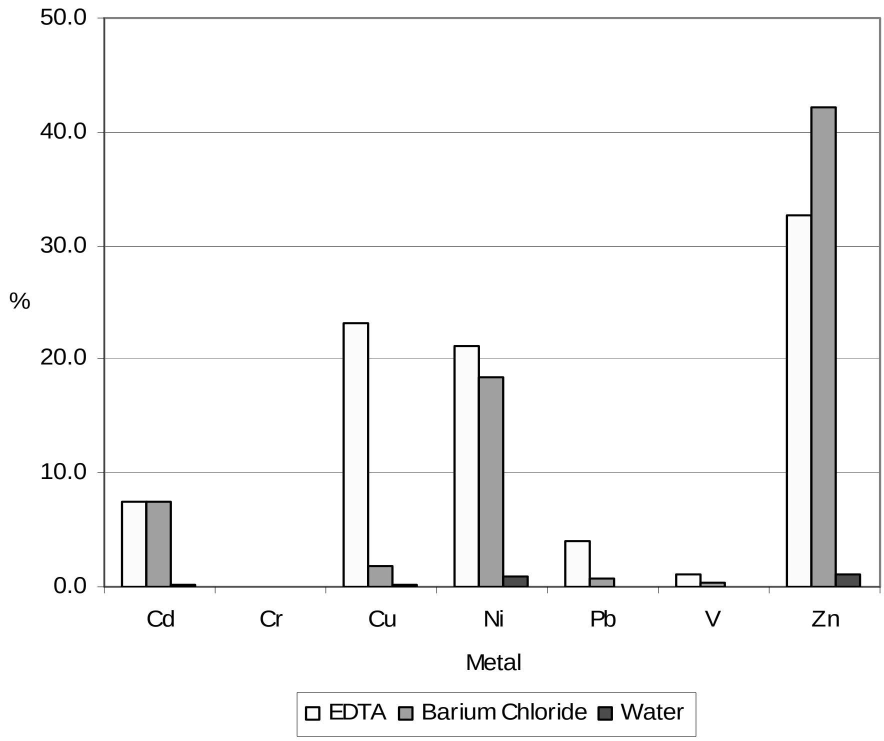

Figure 4.1. Metals extracted with three extractants expressed as a percentage of EPA 3050 extractable metals for Soil 1.

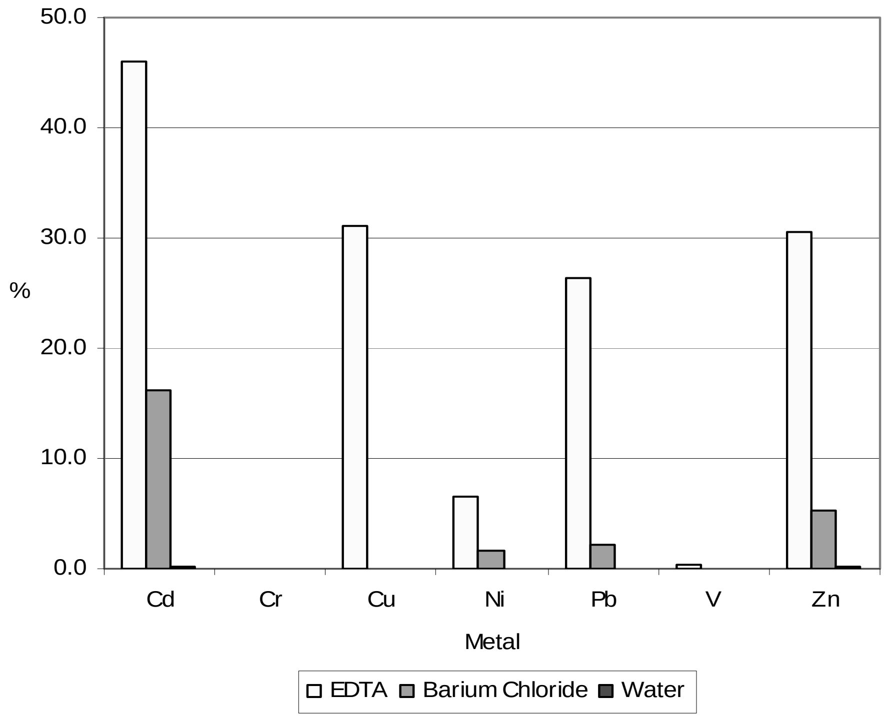

Figure 4.2. Metals extracted with three extractants expressed as a percentage of EPA 3050 extractable metals for Soil 2.

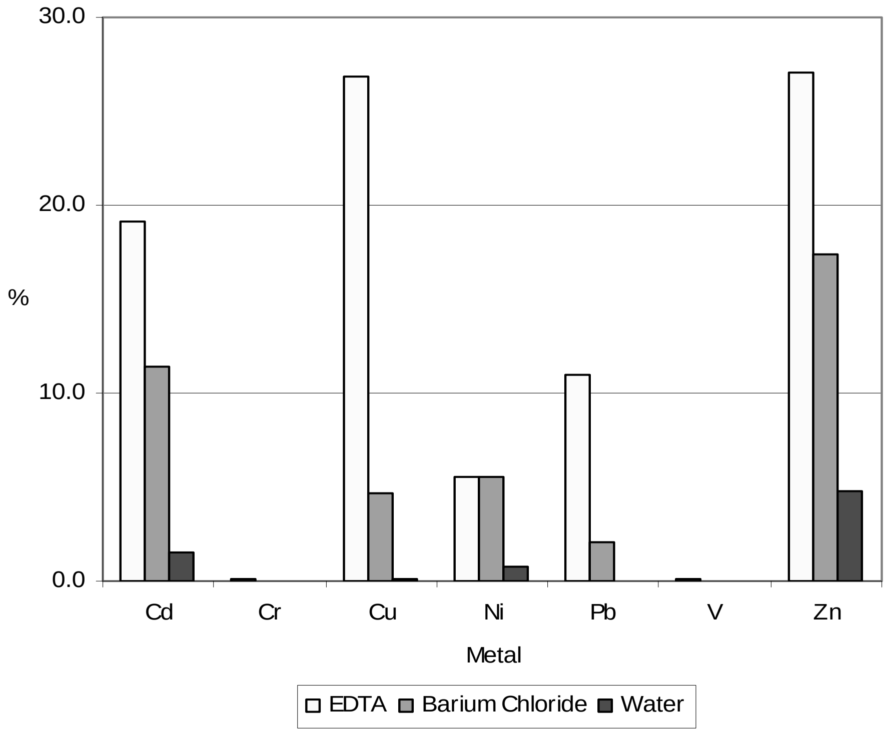

Figure 4.3. Metals extracted with three extractants expressed as a percentage of EPA 3050 extractable metals for Soil 3.

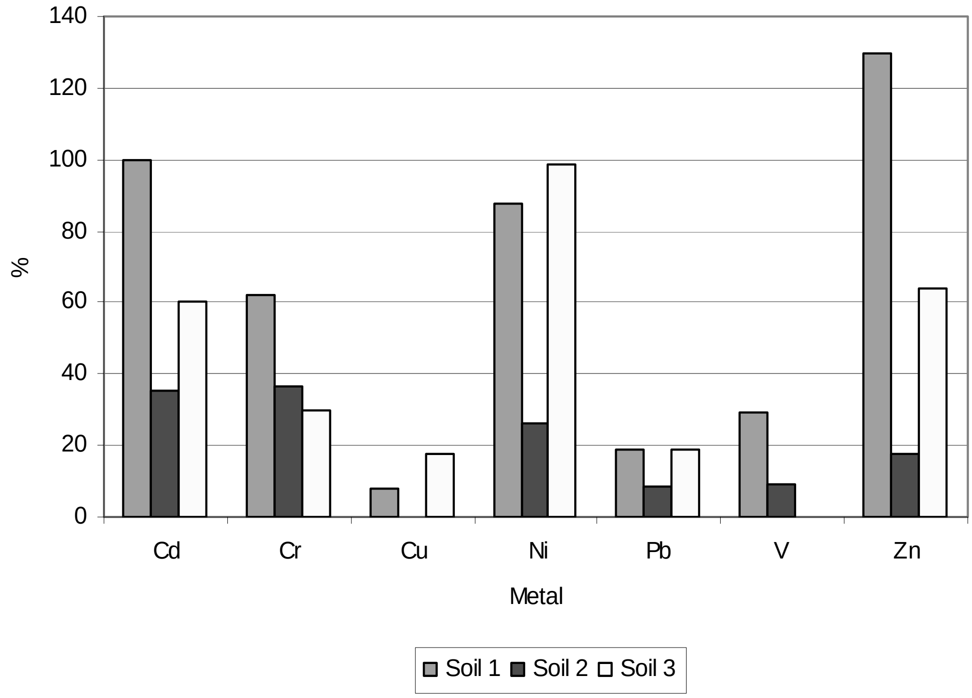

Figure 4.4. Metals extracted with $BaCl_2$ expressed as a percentage of EDTA extractable metals for the three soils.

4.2.1.4 Conclusion

The results indicate that metals have built up to very high levels in some cases. The low extractability of the metals with weak to mild extractants (water and $BaCl_2$) indicates that there is only a very slight risk of metal leaching or immediate plant uptake even though the EPA 3050 extractable levels are high. This illustrates the effect time has on the sorption of metals onto soil constituents and the fact that the longer the time allowed the less mobile a metal becomes – to a certain extent, regardless of the pH of the soil. Even though the pH of the soils differed substantially the only distinct difference was in the extractability with weak extractants and the difference was mostly around one order of magnitude. Therefore it can be concluded that if adequate time is allowed between sludge application and possible removal of metals by different agents (plant-uptake, leaching into groundwater) the mobility of the metals will be decreased substantially due to the formation of stable complexes with organic material and/or soil minerals.

4.2.2 EFFECT OF LIMING ON pH AND METAL EXTRACTABILITY

Table 4.4 gives some properties of the two soils before liming and the pH of the two soils after liming (Appendix 2). The pH values increased by 1.4 pH units for Soil 1 and 1.8 pH units for Soil 3 through the addition of the equivalent of 45 t ha^{-1}. This indicates a massive buffering capacity brought about by organic material and complexes stable at pH 4.2 and 4.0 for Soil 1 and 3 respectively. The organic C content of the soils is most uncharacteristic for South African soils in their respective areas where values seldom exceed 0.5 %. This buffering capacity could complicate efforts to bring the pH of the soil to 6.5 in line with recommendations in the field.

Table 4.5 indicates the influence of the lime addition and incubation on the $BaCl_2$ extractable cations from the two soils. The metals Mn, Pb, and Cd in Soil 1 and Mn and Cd in Soil 3 indicated increases or similar levels in extractability after liming. This is contrary to what the effect of liming was expected to be. As expected a significant decrease was found for Al, Cu, Fe, and Zn.

Table 4.6 indicates the EDTA extractable cations for the two soils before and after liming. Here most of the tested cations (Cu, Mn, Fe, and Cd in both soils and Pb in Soil 2) increased in extractability or had similar levels after liming. Al, Zn, and to a lesser extent Pb in Soil 1 indicted the expected decrease in extractability. The increased extractability of certain of these metals is contrary to what was expected from literature concerning the effect of liming. Furthermore, the possible competition of high levels of added Ca with the other metals would have seemed to also indicate a lower extractability of the metals with EDTA after liming.

Table 4.4 Some chemical and physical properties of the two soils

Property		Soil	
		1	3
Texture		SaLm	SaClLm
Organic C (%)		2.6	2.7
pH	Water	4.2 (5.6)*	4.0 (5.8)*
	$CaCl_2$	3.8 (5.3)*	3.7 (5.7)*
	KCl	3.3 (5.0)*	3.5 (5.4)*

* Values in brackets denote pH values after liming and 6 months incubation.

Table 4.5 The effect of liming on $BaCl_2$ extractable cations (mg kg^{-1} soil, n = 4)

Cation (mg kg^{-1})	Soil 1		Soil 3	
	Before	After	Before	After
Ca	431.8	1600.2	184.0	1726.4
Mg	65.1	224.2	28.5	336.65
K	610.3	412.9	284.3	260.8
Na	19.3	45.4	10.4	52.8
Al	35.0	0.2	154.7	0.2
Cu	3.3	0.2	14.1	0.8
Mn	17.5	**21.2***	29.7	12.4
Fe	6.0	0.5	0.8	0.3
Zn	151.6	11.5	18.5	3.0
Pb	1.8	1.3	1.6	0.3
Cd	0.3	**0.3**	0.3	**0.6**

* Bold values denote levels similar or higher than values determined before liming.

Table 4.6 Effect of liming on NH_4-EDTA extractable cations (mg kg^{-1} soil, n = 4)

Cation (mg kg^{-1})	Soil 1		Soil 3	
	Before	After	Before	After
Al	113.9	33.2	337.7	222.2
Cu	41.1	**37.1***	80.8	**104.6**
Mn	17.2	**49.6**	37.1	**71.3**
Fe	242.7	**383.5**	170.7	**271.7**
Zn	117.0	49.1	28.8	15.6
Pb	9.8	6.5	8.4	**8.1**
Cd	0.3	**0.8**	0.5	**1.3**

* Bold values denote levels similar or higher than values determined before liming.

4.2.2.1 Conclusion

The increased EDTA (and to a lesser extent $BaCl_2$) extractability of certain metals after liming of a sacrificial soil is a cause for concern. A possible explanation for the results includes the mineralization of organic material after liming and subsequent release of complexed metals. The results are currently the subject of further investigation. Guidelines stipulate that soils to which sewage sludge is applied should have a pH above 6.5. In the case where these soils are to be rehabilitated the mobilisation of metals through liming should lead to the search for an alternative rehabilitation method or to the description of the processes involved. Once the processes are understood adequate rehabilitation strategies can be identified.

4.2.3 COMPARISON OF MINERALISATION RATES OF SLUDGE AND COMMERCIAL FERTILIZER NITROGEN SOURCES (APPENDIX 11)

4.2.3.1 Ammonium (NH_4^+)

A sharp increase, attributed to a priming effect on the microbial activity, was observed in NH_4^+-N content in the sludge treatments one day after sludge application (Fig. 4.5) in the incubation trial. This can partly be ascribed to the free NH_4^+-N content in the sludge. The total NH_4^+ concentration increased, peaked at day 7, and decreased thereafter. After 28 d, most of the NH_4^+ was depleted and this was similar for all the treatments. After day 42 the NH_4^+-N content was depleted to such an extent that it was significantly lower than even the initial NH_4^+-N content.

4.2.3.2 Nitrate (NO_3^-)

In the trial there initially were small quantities of NO_3^--N in the sludge treated soils while the NO_3^--N in the fertilizer treatments was high due to the inorganic applied NO_3^--N (Fig. 4.6). A clear lag period was observed in the production of NO_3^- in the sludge treatments. Production of NO_3^- depended on the initial production of NH_4^+. The lag represents the time necessary for the microbial population to increase to an extent sufficient to cause an increase in the nitrification rate. The NO_3^--N content increased slowly in the early stage of incubation, while NH_4^+-N was produced and increased rapidly after 7-14 d at the expense of the NH_4^+-N. In all treatments the NO_3^- concentrations were significantly higher than the control.

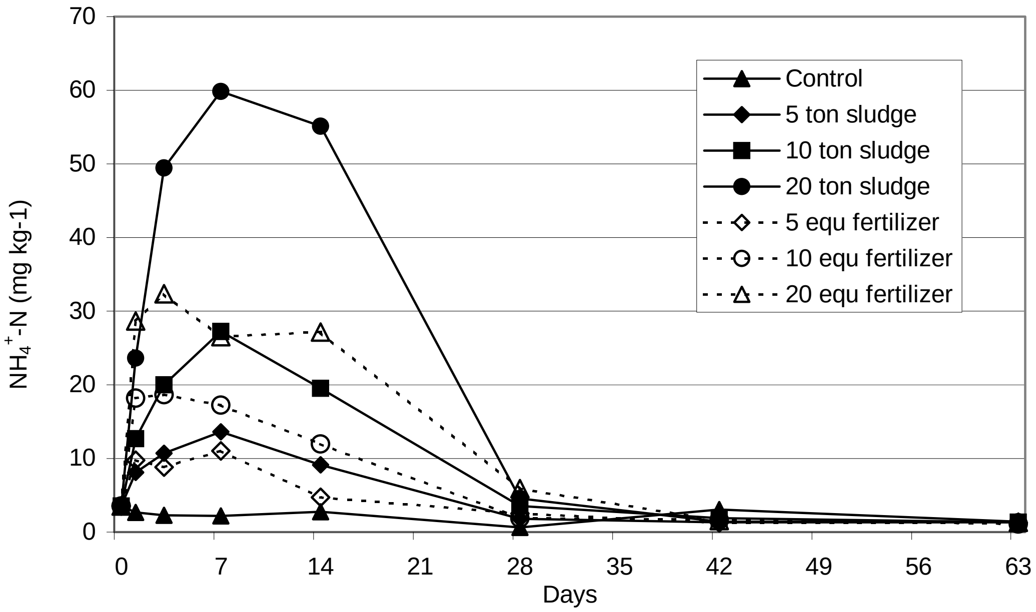

Figure 4.5. Total extractable NH_4^+-N content as a function of incubation time and differentiable N application.

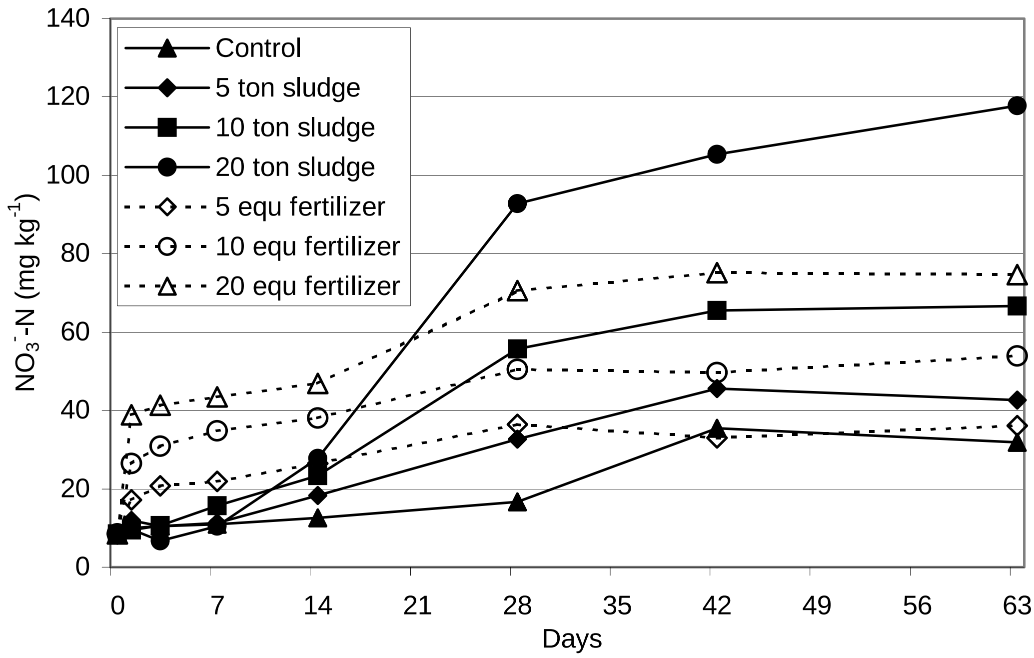

Figure 4.6. Total extractable NO_3^--N content as influenced by differential N application and incubation time.

The total extractable NO_3^--N content was higher from the inorganic N treatments than from the sludge treatments at the 10 t ha^{-1} application rate (sludge and fertilizer). For the 20 t ha^{-1}

no significant difference was obtained between the sludge and the fertilizer. This was also the case for the 5 t ha^{-1} (sludge and fertilizer) treatments.

4.2.3.3 Total Inorganic N Release

According to calculations the total organic N in the sludge that was mineralised in 63 d was 41.5% for the 20 t ha^{-1} treatment, 32.5% for the 10 t ha^{-1} treatment and 21.5% for the 5 t ha^{-1} treatment. The reason for the decrease in inorganic N release with the lower application and higher values with the high application rates can probably be ascribed to the extent that the microorganisms were stimulated. High application rates add a larger extractable energy source to the soil, which microorganisms can exploit. This increases the microbial activity and the sludge is decomposed more rapidly.

The rate of decomposition of organic matter in sludge is highly variable, depending on the sludge composition (Korentajer, 1991). Different values of mineralisation rates have been obtained in the past. Mineralisation rates of organic N found in aerobically digested sewage sludge range from 20-50 % (Cripps *et al.*, 1992) and 4-48% during a 16-week incubation trial by Magdoff & Chromec (1977) and Parker & Sommers (1983). It would therefore be wrong to generalise the inorganic N concentrations of sludges due to the preparation processes, the sampling techniques and handling prior to analysis (Parker & Sommers, 1983). Soil type, sewage sludge source and climatic factors can also have large effects on organic N mineralisation rates (Parker & Sommers, 1983; Korentajer, 1991; Cripps *et al.*, 1992).

4.2.3.4 Conclusions

In total, more N is available from sewage sludge over time, than from commercial fertilizers (depending on the application rates). The distribution at which the inorganic N becomes available, is not however equally distributed. Most of the N in sludge is organically bound, and not immediately available. N only becomes available after some time, through mineralisation. In the case of commercial fertilizer, which is an inorganic fertilizer, most of the N is immediately available.

The advantage sludge has over fertilizer, is that most of the NO_3^- only becomes available between 14-28 d of incubation. In practice it can be expected that NO_3^--N production can be lower than the incubation values, because in the incubation trial conditions were optimal for the microorganisms. This fraction of N can therefore be utilised much more efficiently by plants, which will reduce the risk of leaching. The amount of NO_3^--N that becomes available from sludge can act as a slow release N-fertilizer that can be more beneficial to crops than commercial fertilizer.

4.3 GREENHOUSE EXPERIMENTS (APPENDICES 3, 6, 8 AND 10)

4.3.1 PLANT-SOIL INTERACTIONS OF SLUDGE-BORNE HEAVY METALS AND THE EFFECT ON MAIZE, OATS, SUNFLOWER AND SOYBEAN SEEDLING GROWTH

4.3.1.1 Sludge Analysis

The Zn, Cu, and Pb levels in both the low and high metal content sludge exceeded guideline levels (Table 4.7). Cd was an exception in that the sludge levels were within current guideline levels.

Table 4.7 Metal contents of the sludges used in the greenhouse trials and limits set by the WRC (1997)

	Metal present in sludge (mg kg^{-1})			
	Cd	Cu	Pb	Zn
Sludge 1	1.8	114.3	66.0	679.0
Sludge 2	1.8	118.8	195.6	1566.2
Limit	15.7	50.5	50.5	353.5

4.3.1.2 Cd in Soil and Plants

In all the trials the Cd concentration in the soil after sludge application did not increase dramatically compared to the control treatments (Table 4.8). In fact, in some cases the determined Cd concentration was lower in the sludge treatments than in the control treatments. This would seem to indicate that standard variation in terms of experimental procedure could account for the discrepancies. Although similar values were found throughout the maize and sunflower trials for all the soils, there was a marked difference between the values for the different soils in the oats trial. The values for the soybean seedlings also differed but there were no clear trends.

The Cd concentrations in the dry plant material (Table 4.9) exhibited the same trend as the Cd concentration in the soils in that some sludge treatments led to higher concentrations in the plants than the control treatments. This trend is not consistent throughout though and again it would seem that standard experimental variation could account for the differences.

Table 4.8 Cd concentrations in the soil for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.5	0.43	0.25	0.00
Sludge 2	0.4	0.41		0.04
Positive control	0.5	0.49	0.42	0.01
Negative control	0.5	0.49	0.65	0.00
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.4	0.33	0.25	0.11
Sludge 2	0.3	0.30		0.17
Positive control	0.4	0.38	0.48	0.00
Negative control	0.4	0.38	0.81	0.04
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.5	0.38	0.63	0.5
Sludge 2	0.4	0.39		0.6
Positive control	0.4	0.44	0.86	0.4
Negative control	0.4	0.42	0.17	0.5

The same trends as mentioned above were found in the Cd transfer coefficients (Table 4.10) of the treatments with the exception of the values for the oats treatments. The high *f* factors for Cd in the oats seedlings (Sludge 2 and Positive Control) in the sandy soil is probably due to the low concentrations determined in both the soil and the seedlings. With low concentrations any difference is exaggerated and this is reflected in the low *f* factors for the Sludge 1 and Negative Control treatments. Exaggerated uptake due to pot conditions and the sandy nature of the soil could also have played a role. When the loam soil is considered the trend is reversed to a certain extent and the values indicate very little difference in the clay soil. In the case of maize and oats the transfer coefficients were much higher than those given by Korentajer (1991) but this is most probably due to the pot conditions that exaggerated metal uptake (Alloway, 1995).

Table 4.9 Cd concentrations in dry plant material for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.27 ± 0.08	0.23 ± 0.01	0.30	1.80 ± 1.1
Sludge 2	0.29 ± 0.17	0.27 ± 0.14		1.05 ± 0.1
Positive control	0.25 ± 0.10	0.24 ± 0.13	0.52	1.16 ± 0.2
Negative control	0.21 ± 0.02	0.20 ± 0.14	0.23	1.20 ± 0.3
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.18 ± 0.02	0.38 ± 0.01	0.33	1.36 ± 0.1
Sludge 2	0.17 ± 0.03	0.36 ± 0.14		1.56 ± 0.4
Positive control	0.33 ± 0.05	0.25 ± 0.02	0.32	1.02 ± 0.1
Negative control	0.28 ± 0.09	0.60 ± 0.09	0.30	1.25 ± 0.5
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.24 ± 0.09	0.32 ± 0.15	0.25	1.60 ± 0.3
Sludge 2	0.23 ± 0.09	0.20 ± 0.04		1.43 ± 0.6
Positive control	0.18 ± 0.01	0.32 ± 0.08	0.27	1.46 ± 0.8
Negative control	0.19 ± 0.05	0.18 ± 0.05	0.22	2.20 ± 0.7

Table 4.10 Cd transfer coefficients for four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.6	0.5	1.2	0
Sludge 2	0.7	0.7		24.4*
Positive control	0.5	0.5	1.2	89.2*
Negative control	0.4	0.4	0.3	0
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.5	1.1	1.3	12.4*
Sludge 2	0.6	1.2		9.2*
Positive control	0.9	0.7	0.7	0
Negative control	0.7	1.6	0.4	29.1*
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.5	0.9	0.4	3.2
Sludge 2	0.6	0.5		2.4
Positive control	0.4	0.7	0.3	3.7
Negative control	0.5	0.4	1.3	4.4
Literature values#	0.01-0.05	NA	NA	0.01-0.05

* Values skewed by low soil and average plant concentrations

Korentajer (1991).

The Cd levels remained very similar throughout experiments (soils, crops and treatments), indicating almost no risk from a once off sludge application. Cd remained under the guideline levels in the oats and sunflower trials and the levels are considered too low to come to a meaningful conclusion other than that it poses no risk after a once off application.

4.3.1.3 Cu in Soil and Plants

The Cu levels in the soil did not differ markedly between treatments for the different soils although there was a slight increase in some of the Sludge 2 treatment soils (Table 4.11). The levels were generally lower in the oats trial and the sandy soil in the soybean trial than in the maize and sunflower trials. In the case of maize and oats, the levels in the plants increased throughout most of the trials with the Sludge 2 treatments although the levels were in most cases still very similar to control levels (Table 4.12). The Cu levels remained very similar throughout experiments (soils, crops and treatments), indicating almost no risk from a once-off sludge application.

Table 4.11 Cu concentrations in the soil for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	20.5	20.4	5.3	10.8
Sludge 2	22.6	21.6		11.4
Positive control	20.7	18.0	4.2	10.9
Negative control	23.5	20.4	6.3	10.3
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	30.9	28.9	31.2	24.2
Sludge 2	29.1	30.2		23.5
Positive control	28.2	29.2	34.6	22.6
Negative control	27.8	27.2	26.7	19.5
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	33.1	33.5	36.5	19.3
Sludge 2	43.7	43.5		23.4
Positive control	37.0	39.6	45.5	21.4
Negative control	33.6	28.4	40.6	19.1

The transfer coefficients for Cu did not differ substantially between the different treatments except for the sandy soil where a slight increase was found for the sludge treatments for all the crops (Table 4.13). Although the transfer coefficients were higher than those given by Korentajer (1991), it is again considered to be the result of pot conditions as discussed for Cd.

Table 4.12 Cu concentrations in dry plant material for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	8.2 ± 2.5	12.2 ± 1.6	6.1	11.4 ± 0.9
Sludge 2	6.8 ± 1.3	10.7 ± 0.9		7.7 ± 1.7
Positive control	4.2 ± 0.5	10.6 ± 1.2	4.6	8.2 ± 0.2
Negative control	4.7 ± 0.2	13.1 ± 2.7	4.1	5.1 ± 1.1
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	12.4 ± 1.9	19.4 ± 1.8	6.1	11.7 ± 2.2
Sludge 2	8.7 ± 0.6	16.7 ± 0.8		12.4 ± 2.1
Positive control	11.3 ± 0.7	10.3 ± 0.6	5.8	11.1 ± 0.9
Negative control	11.1 ± 0.2	14.0 ± 1.7	5.3	8.7 ± 2.5
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	9.9 ± 1.1	13.8 ± 3.4	7	12.1 ± 1.2
Sludge 2	8.8 ± 0.6	14.3 ± 1.2		10.1 ± 1.2
Positive control	7.4 ± 1.5	15.3 ± 2.9	5.2	7.6 ± 0.9
Negative control	6.7 ± 1.7	13.4 ± 1.1	5.2	7.4 ± 1.1

Table 4.13 Cu transfer coefficients for four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.4	0.6	1.2	1.1
Sludge 2	0.3	0.5		0.7
Positive control	0.2	0.6	1.1	0.8
Negative control	0.2	0.6	0.7	0.5
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.4	0.7	0.2	0.4
Sludge 2	0.3	0.6		0.5
Positive control	0.4	0.4	0.2	0.5
Negative control	0.4	0.5	0.2	0.5
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.3	0.4	0.2	0.6
Sludge 2	0.2	0.3		0.4
Positive control	0.2	0.4	0.1	0.4
Negative control	0.2	0.5	0.1	0.4
Literature values#	0.01-0.05	NA	NA	0.01-0.05

Korentajer (1991).

4.3.1.4 Pb in Soil and Plants

The Pb levels in the soils of the maize and soybean seedlings do not reflect the expected increase whereas the soil of the sunflower seedlings had an increased Pb concentration with the Sludge 2 treatments (Table 4.14). This is also to a lesser extent so for some of the soils of the oats seedlings. These same trends are exhibited in the plant concentrations of Pb (Table 4.15). The oats seedlings grown on the sandy soil exhibited a significant increase in Pb content with the sludge treatments (especially Sludge 2). This is most likely due to the lack of sesquioxides in this soil and subsequent lack of sorption sites (compared to the other soils) after the Pb was released through the breakdown of the sludge. The increased Pb uptake is also reflected in the transfer coefficient for the oats Sludge 2 treatment (Table 4.16). The other treatments and crops do not exhibit any clear difference between the treatments.

Table 4.14 Pb concentrations in the soil for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	12.7	12.2	4.9	6.1
Sludge 2	13.1	15.6		6.5
Positive control	12.3	10.0	4.4	5.8
Negative control	12.2	12.9	5.4	5.9
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	21.3	23.5	54.4	66.8
Sludge 2	16.3	31.1		57.3
Positive control	22.7	22.8	58.2	70.8
Negative control	25.2	15.9	47.7	60.0
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	21.0	18.8	19.2	53.7
Sludge 2	22.6	23.8		52.8
Positive control	18.2	16.9	24.8	47.0
Negative control	19.3	15.9	23.2	35.3

Table 4.15 Pb concentrations in dry plant material for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	2.5 ± 0.5	3.2 ± 0.6	3.9	6.1 ± 2.0
Sludge 2	2.6 ± 0.3	3.0 ± 0.8		16.1 ± 6.2
Positive control	2.5 ± 0.3	2.1 ± 0.9	3.6	2.9 ± 0.04
Negative control	2.4 ± 0.5	2.5 ± 0.6	3.4	3.2 ± 0.5
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	2.1 ± 0.3	2.0 ± 0.1	3.3	4.0 ± 1.0
Sludge 2	1.6 ± 0.2	1.7 ± 0.2		3.1 ± 0.4
Positive control	2.3 ± 0.3	1.7 ± 0.2	5.9	3.6 ± 0.1
Negative control	2.5 ± 0.5	2.6 ± 0.0	3.3	3.1 ± 0.4
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	2.1 ± 0.4	2.3 ± 0.2	3.2	4.3 ± 1.2
Sludge 2	2.3 ± 0.3	2.2 ± 0.1		3.7 ± 0.7
Positive control	1.8 ± 0.1	2.2 ± 0.7	4.1	4.0 ± 0.9
Negative control	1.9 ± 0.6	2.2 ± 0.4	6.2	4.9 ± 0.8

Table 4.16 Pb transfer coefficients for four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.2	0.3	0.8	1.0
Sludge 2	0.2	0.2		2.5
Positive control	0.2	0.2	0.8	0.5
Negative control	0.2	0.2	0.6	0.5
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.1	0.1	0.1	0.1
Sludge 2	0.1	0.1		0.1
Positive control	0.1	0.1	0.1	0.1
Negative control	0.1	0.1	0.1	0.1
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.1	0.1	0.2	0.1
Sludge 2	0.1	0.1		0.1
Positive control	0.1	0.1	0.2	0.1
Negative control	0.1	0.1	0.3	0.1
Literature values[#]	0.01-0.05	NA	NA	0.01-0.05

[#] Korentajer, 1991

4.3.1.5 Zn in Soil and Plants

Of the four metals tested, Zn is the only one with a distinct increase in most of the treatments and crops for the sludge additions (Table 4.17). Levels in the clay soil were higher in the sunflower and maize trials than in the oats and sandy soil of the soybean trial. This could be ascribed to a possible higher natural background concentration in this soil compared to the sandy and loam soils. Plant-uptake of Zn by oats followed the trend of increased concentration in the soil but in the maize and sunflower trials this trend was only evident in the sandy soil and the clay soil for maize (Table 4.18). The concentration in the soybean seedlings also exhibited an increase in the sludge treatments in the sandy and clay soils. This same trend was found in the transfer coefficients of the oats and soybean trials whereas the coefficients for maize and sunflower exhibited no clear trends (Table 4.19).

Table 4.17 Zn concentrations in the soil for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	92.5	87.0	11.8	22.9
Sludge 2	72.5	72.5		26.4
Positive control	38.7	39.9	13.6	20.9
Negative control	58.5	62	10.4	19.9
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	77.2	79.3	67.6	40.2
Sludge 2	68.3	67.6		43.7
Positive control	54.2	52.9	77.6	29.6
Negative control	53.9	54.0	56.2	21.6
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	139.3	145.8	68.4	32.5
Sludge 2	182.4	197.3		59.9
Positive control	169.1	150.5	126.2	48.3
Negative control	120.9	124.4	71.1	45

Table 4.18 Zn concentrations in dry plant material for the four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	64.8 ± 7.8	66.9 ± 15.2	45.8	46.4 ± 5.4
Sludge 2	94.3 ± 15.2	104.4 ± 17.3		45.2 ± 7.9
Positive control	34.8 ± 1.0	58.1 ± 7.3	27.7	31.8 ± 5.9
Negative control	41.0 ± 3.1	78.7 ± 2.5	20.9	24.3 ± 9.8
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	92.6 ± 22.5	362.5 ± 0.0	62.5	43.6 ± 1.1
Sludge 2	123.0 ± 7.6	233.0 ± 182		45.2 ± 27.9
Positive control	75.8 ± 28.3	118.0 ± 13.4	71.6	31.8 ± 4.3
Negative control	188.5 ± 33.8	720.0 ± 88.0	49.4	24.3 ± 9.2
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	69.7 ± 2.5	83.6 ± 10.7	32.5	36.7 ± 6.2
Sludge 2	73.0 ± 7.2	130.3 ± 19.7		36.5 ± 4.8
Positive control	67.6 ± 16.0	103.6 ± 13.3	21.6	20.8 ± 5.0
Negative control	48.4 ± 17.8	99.1 ± 4.4	23.0	21.8 ± 5.1

Table 4.19 Zn transfer coefficients for four types of seedlings and four treatments in three soils used in the trial

Treatment	Sandy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.7	0.8	3.9	2.0
Sludge 2	1.3	1.4		1.7
Positive control	0.9	1.5	2.0	1.5
Negative control	0.7	1.3	2.0	1.2
	Loamy Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	1.2	4.6	0.9	1.1
Sludge 2	1.8	3.5		2.1
Positive control	1.4	2.2	0.9	1.4
Negative control	3.5	13.3	0.9	1.2
	Clayey Soil			
	Maize	Sunflower	Soybean	Oats
Sludge 1	0.5	0.6	0.5	1.1
Sludge 2	0.4	0.7		0.6
Positive control	0.4	0.7	0.2	0.4
Negative control	0.4	0.8	0.3	0.5
Literature values#	1.0-2.0	NA	NA	0.5-1.0

Korentajer (1991).

4.3.1.6 Conclusion

Overall the transfer coefficients did not vary widely (except for Cd as explained above) and a clear trend in metal uptake and metal addition through sludge is absent except for Zn (to a limited extent). This indicates that experimental conditions, such as the use of pots, predominantly led to the differences found and that the sludge treatments did not lead to detrimental conditions in terms of metal uptake. Furthermore, the trial was conducted soon after the application of the sludges and this left very little time for the metals to equilibrate with soil constituents upon its release from the sludge. The conclusion therefore is that at current levels of use there is almost no risk of metal contamination of the food chain. This situation is bound to change if long-term continuous applications and possible changes in land-use after a number of years are considered. Further research should be focused on continuous applications and land-use scenarios.

4.3.2 PERSISTENCE OF HUMAN PATHOGENS IN CROPS GROWN ON SEWAGE SLUDGE-TREATED SOILS (APPENDIX 12)

4.3.2.1 Faecal Coliforms

Faecal coliforms appeared to grow considerably in the fourth and sixth week in both the low (LMS16) and high metal sludge (HMS16) pots when sludge was applied at 16 t ha^{-1} (Fig. 4.7). Although appreciable growth was noticed in the fourth and sixth week for both the LMS and HMS, it appeared even more advanced in the LMS. Faecal coliform growth was minimal at the beginning of the experiment and also in the second, eighth, tenth, and twelfth week for both the low and the high metal sludges.

The faecal coliforms in the pots with applied low metal sludge (8 t ha^{-1}) had considerable growth in the second week, which declined markedly in the fourth week and picked up again in the sixth week (Fig. 4.8). The high metal sludge pots on the other hand only showed an appreciable growth in the sixth week and limited growth was noticed in the other weeks. Both the low and high metal sludge pots showed limited growth at zero time and the eighth to the twelfth week (week 8 to 12). More growth was observed in the pots with applications of LMS as compared to the HMS pots.

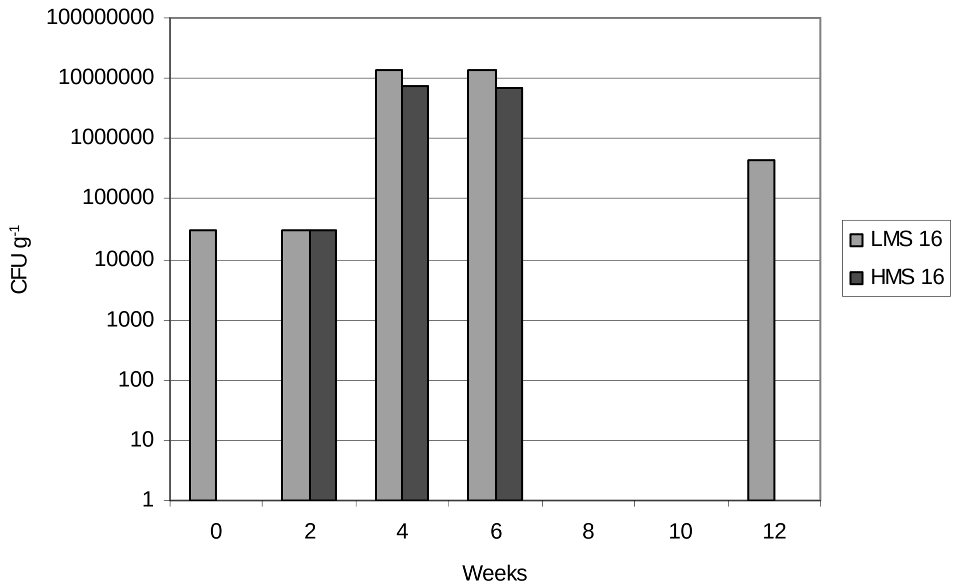

Figure 4.7. Faecal coliform growth for LMS and HMS at an application of 16 t ha^{-1}.

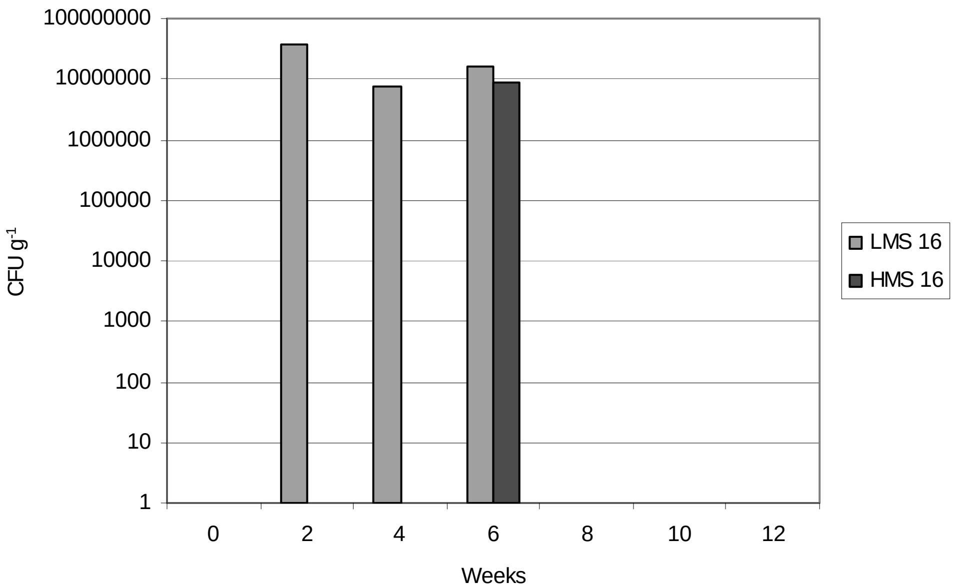

Figure 4.8. Faecal coliform growth for LMS and HMS at an application of 8 t ha^{-1}.

4.3.2.2 *E. coli*

E. coli showed a considerable growth when the low metal sludge was applied at 16 t ha^{-1} (Fig. 4.9). There appeared to be more *E. coli* in the fourth week for the LMS as compared to the HMS, although growth declined considerably in the LMS by the sixth week. A small peak was observed for the HMS in the sixth week. In general, there is hardly any noticeable growth in pots where high metal sludge was applied.

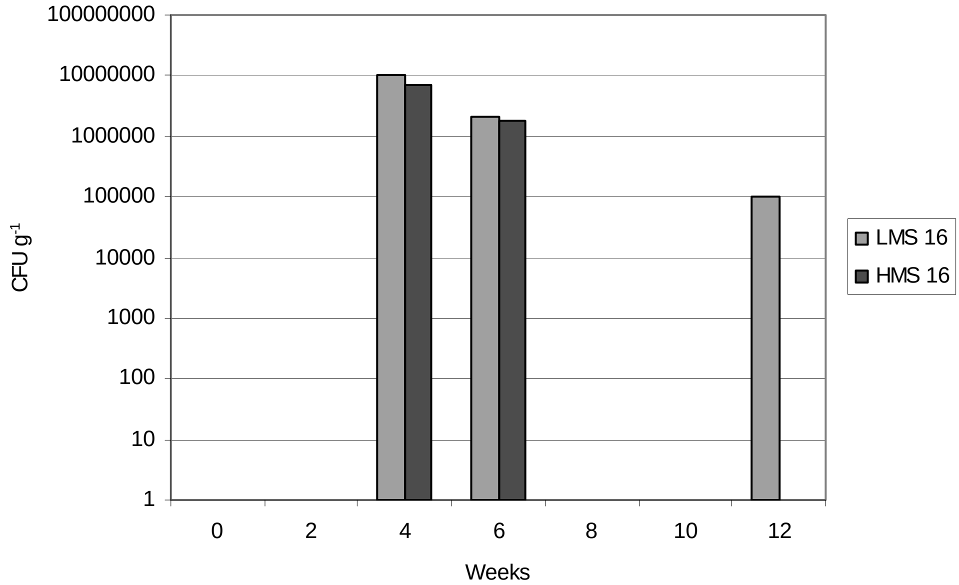

Figure 4.9. Comparison of *E. coli* growth for LMS and HMS at an application rate of 16 t ha^{-1}.

The *E. coli* growth for both the low metal and high metal sludge pots (8 t ha^{-1}) peaked at week four and declined in the sixth week (Fig. 4.10), although in the high metal sludge it is slightly lower than in the low metal sludge. There was hardly any growth for both LMS and HMS at zero time in the eighth week to the twelfth week.

4.3.2.3 *Salmonella*

Table 4.20 provides an indication of whether *Salmonella* was found in the samples at each application rate for every week sampled. The presence of *Salmonella* is indicated with positive sign, while the absence thereof is indicated with a negative sign. At zero time, *Salmonella* was only observed in the LMS at 8 t ha^{-1}. All four treatments had *Salmonella* throughout week 2 to the 10th week. However, only the 16 t ha^{-1} application rate for both LMS and HMS had *Salmonella* by the 12th week. No *Salmonella* was present in the 8 t ha^{-1} treatment for both LMS and HMS at week 12.

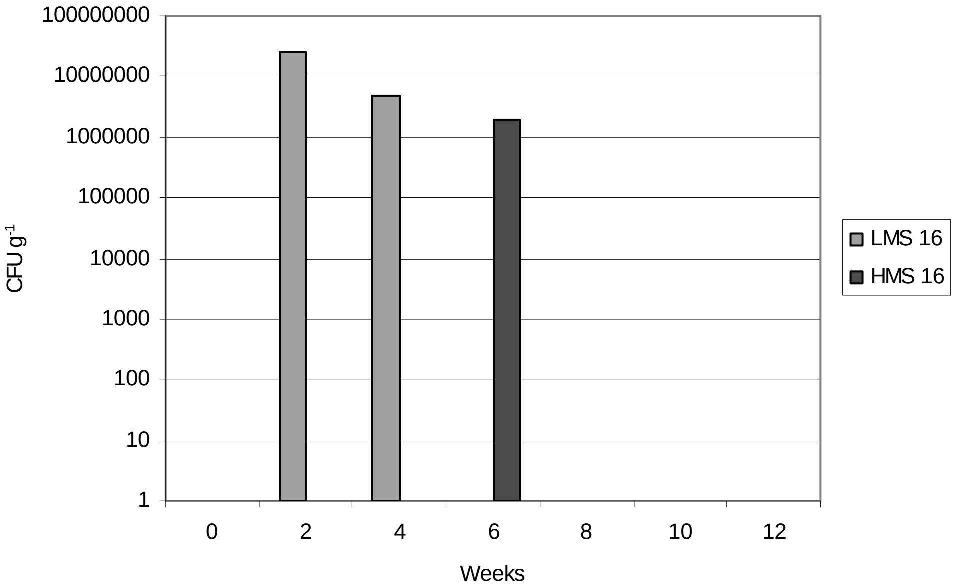

Figure 4.10. Comparison of *E. coli* growth for LMS and HMS at an application rate of 8 t ha^{-1}.

Table 4.20 *Salmonella* found in sludge-applied soil in pots

Weeks	LMS 8	LMS 16	HMS 8	HMS 16
0	+	-	-	-
2	+	+	+	+
4	+	+	+	+
6	-	+	+	+
8	+	+	+	+
10	+	+	+	-
12	-	+	-	+

(+ = presence, - = absence)

4.3.2.4 *Ascaris*

Table 4.21 indicates the total number of *Ascaris* found in all the pots for each application rate and sludge type for every week sampled. Other than at zero time and in the 4th week, there appeared to be no *Ascaris* in the soil sampled. *Ascaris* was found in large numbers in samples collected in the 4th week. For instance, a total of four (4) *Ascaris* were counted in samples collected from pots that received LMS at 16 t ha^{-1} (Table 4.21).

Table 4.21 Numbers of *Ascaris* found in sludge-applied soil in pots

Week	LMS 8	LMS 16	HMS 8	HMS 16
0	1	0	0	2
2	0	0	0	0
4	2	4	1	0
6	0	0	0	0
8	0	0	0	0
10	0	0	0	0
12	0	0	0	0

4.3.2.5 Microorganisms on Potato

Indicated in Table 4.22 are the microorganisms found on the potato peel and within the inside of the potato (potato core).

Table 4.22 Microorganisms found in potato in the 12th week of the trial

Sample	Microorganism	LMS 8	LMS 16	HMS 8	HMS 16
Potato peel	Faecal coliforms (CFU g^{-1})	0	2050	0	0
	E. coli (CFU g^{-1})	0	1800	0	0
	Salmonella	-	+	-	+
	Ascaris	0	0	0	0
Potato core	Faecal coliforms (CFU g^{-1})	0	0	0	0
	E. coli (CFU g^{-1})	0	0	0	0
	Salmonrella	-	-	-	-
	Ascaris	0	0	0	0

None of the microorganisms tested were found in the potato core. However, large numbers of faecal coliforms and *E. coli* were found on the peel in the treatment LMS at 16 t ha^{-1}. These are the mean values of eight repetitions. *Salmonella* also tested positive in this treatment. Although other microorganisms were not found in the HMS at 16 t ha^{-1}, potato peel from this treatment tested positive for *Salmonella*. None of the samples was found to contain *Ascaris*.

4.3.2.6 Conclusions

From the data presented it is clear that most of the pathogens studied decreased in number to very low or undetectable levels after six to eight weeks after sludge application. The exception is *Salmonella*, which was found until 12 weeks after sludge application. These results confirm other studies that indicated that human pathogen persistence is time dependent and that numbers decline rapidly after a few weeks after sludge application. It would therefore be safe to conclude that the longer the time allowed after sludge application the lower the risk of pathogen infection. Despite this finding, the cultivation of crops that allow for pathogen infection in humans should be avoided for a relatively long period after sludge application.

The use of sludge to grow potatoes is not permitted by the guidelines. This should remain the status quo due to the presence of pathogenic organisms on the peel of the tested potatoes. It is highly unlikely that pathogens would find their way into potatoes but the restrictions posed by the guidelines would preclude any such eventuality.

4.4 FIELD EXPERIMENTS

4.4.1 THE CULTIVATION OF FIELD-GROWN MAIZE, SUNFLOWER AND OATS ON DIFFERENT SEWAGE SLUDGE DOSAGES (APPENDICES 4, 5, 7 AND 9)

4.4.1.1 Sludge Analysis

Heavy metal concentrations (Zn, Pb and Cu) exceeded the guideline values (Table 4.23) in the maize, sunflower and oats trials, when interpreted as total metal concentration. However, the sludge still contained relatively low heavy metal concentrations compared to other sewage plants in South Africa (Snyman *et al.,* 1999).

4.4.1.2 Soil Analysis – Nutrients and Soil Chemical Conditions

Table 4.24 indicates the chemical characteristics of the soil in the different treatments of the first maize trial. The pH decreased in all the sludge treatments from the beginning to the end of the trial. In some cases the decrease was in excess of 1 pH unit. Most of the nitrogen in the anaerobically digested sludge was in the reduced form, even at the stage when it was applied to the soil, and could have contributed the decrease in pH through its mineralisation and subsequent nitrification. The resistance decreased in all of the treatments and is probably the result of leaching of free salts during the trial. In all cases the plant nutrients increased substantially. Table 4.25 presents some chemical properties of the soil in the different treatments of the oats trial.

Table 4.23 Heavy metal content in the dewatered sludge compared to guidelines (WRC, 1997)

Metal (mg kg^{-1})	Guidelines (WRC, 1997)	Concentration is sludge used in trials (mg kg^{-1})	
		Maize and sunflower	Oats
Cu	50.5	270	116
Mn	NA	988	nd
Pb	50.5	141	67
Zn	353.5	1100	780

nd = not determined.

Table 4.24 Chemical characteristics of the soil at the beginning of the experiment after sludge was applied (B) and at ear formation (E) of maize

Parameter	+ Control		- Control		Exp. 1		Exp. 2	
	B	E	B	E	B	E	B	E
pH	6.53	6.04	6.7	6.29	7.0	5.53	6.89	5.63
Resistance (Ω)	360	1 800	460	1 800	440	1 400	480	1 500
P (mg kg^{-1})	70.4	171.5	47.7	94.5	68.7	171.5	61.9	138.9
Ca (mg kg^{-1})	582	718	582	606	661	631	613	652
Mg (mg kg^{-1})	335	357	335	303	378	348	353	339
K (mg kg^{-1})	124	218	124	46	158	79	169	65
Total % N	0.213	0.27	0.16	0.25	0.16	0.304	0.231	0.28

Table 4.25 Chemical characteristics of the soil at the beginning of the oats trial

Parameter	- Control	+ Control	Exp. 1	Exp. 2
pH	6.41	5.86	6.12	5.86
Resistance (Ω)	1 400	1 600	1 600	1 600
P (mg kg^{-1})	128	191	191	192
Ca (mg kg^{-1})	1 046	866	1 056	886
Mg (mg kg^{-1})	245	203	257	230
K (mg kg^{-1})	314	421	613	649

4.4.1.3 Soil and Plant Analysis – Heavy Metals

Maize (Trial 1)

In the first maize trial all the analysed metals (Zn, Pb and Cu) exceeded the guideline levels (Table 4.26). This was due to very high background levels in the soil and the conservative nature of the South African guidelines. Manganese levels exceeded 10 000 mg kg^{-1} due to soil parent materials (dolomite) rich in Mn.

The plant concentration of the three metals is presented in Table 4.27. Previous research has shown that crops grown on sludge-amended soils exhibited significant increases in the uptake of Cu in plant tissue (Chlopecka, 1996). Cu concentrations were all below the toxic ranges of 20 to 100 mg kg^{-1} specified for plant leaves (Smith, 1996). The uptake of Cu varied between treatments and the sludge treatments had a higher uptake of Cu in the kernels and stalks compared to the control treatments. This could be due to the higher availability of the Cu in the sludge when applied to agricultural soil. The normal range for the *f* factor for Cu is between 0.01 and 0.05. In all the plant parts, in all the treatments, the transfer coefficient was higher than normal (Table 4.28). This could be due to the high soil background concentrations of Cu as evidenced from Table 4.26.

Manganese background concentrations in the soil were excessively high and therefore taken up into the plant parts at high concentrations. The leaf concentrations were below the typical toxic concentration range of 400 to 1000 mg kg^{-1} (Smith, 1996). Mn accumulated more in the maize kernels compared to other plant parts. The higher Mn concentrations in the kernels grown on the sludge-treated soil at 12.5 t ha^{-1} were possibly due to the high concentration of Mn in the sludge (988 mg kg^{-1}) being more available for plant uptake. The transfer coefficient of Mn is not included since soil concentrations of Mn were too high to be determined.

Lead was mostly taken up into the leaf tissue, although the Pb concentrations did not exceed the toxic ranges for plants of 30 to 300 mg kg^{-1} (Smith, 1996). Low uptake was found in the kernels of all treatments. The transfer coefficient was lower than the normal ranges of 0.01 to 0.05 (Smith, 1996) in the kernels (all treatments), and in the Negative Control treatment and Exp. 2 treatment in the stalks. This emphasises the immobility of lead under normal soil conditions. However, normal values and slightly higher (Negative Control and Exp. 1 treatment) transfer coefficients were found in the leaves. The low uptake and transfer coefficient of Pb in this field experiment have been found in most other field experiments with sewage sludge (Chlopecka, 1996), and this phenomenon is due to the low solubility, mobility and availability of Pb to crop plants (Chumbley & Unwin, 1982). Therefore Pb does not pose a major threat to crop plants where it is applied through sludge.

Zinc concentrations in the leaves, stalks and kernels were below the toxic ranges of 100 to 400 mg kg^{-1} (Smith, 1996). This was also found by Jarausch-Wehrheim *et al.* (1999) in earlier field studies concerning the translocation of sludge-borne Zn in field-grown maize. The higher concentration of Zn in the kernels shows increased accumulation of Zn from the leaves (sources of nutrients) to the kernels (nutrient storage sinks) (Gardner *et al.,* 1993) during the growing season. The uptake of Zn in the different plant parts between treatments did not differ much, although the Negative Control treatment had a lower uptake of Zn in the stalks compared to the other treatments. The transfer coefficient of Zn was lower than normal [between 1 and 2 (Smith, 1996)] in the plant parts.

Table 4.26 Total heavy metal content in the soil at the beginning of ear formation of maize

Metal (mg kg^{-1})	Limit*	+ Control	- Control	Exp. 1	Exp. 2
Cu	6.6	70.5	78.6	79	74.8
Pb	6.6	31.5	33.9	35.8	33.8
Zn	46.5	121.6	131.1	131.7	129.8

*Limit: Maximum soil metal content according to the "Guide: Permissible utilisation and disposal of sewage sludge" (1997).

Table 4.27 Heavy metal content in different plant parts in the first maize field trial

Metal (mg kg^{-1})	Plant part	+ Control	- Control	Exp. 1	Exp. 2
Cu	Stalks	11.3	7.3	14.6	11.8
	Leaves	12.7	11.0	9.8	10.5
	Kernels	4.2	4.8	12.4	6.9
Mn	Stalks	166.8	106.0	154.3	121.6
	Leaves	160.5	146.5	112.2	283.6
	Kernels	272.7	302.3	514.3	349.1
Pb	Stalks	0.45	0.33	0.80	0.05
	Leaves	0.82	2.82	2.78	1.45
	Kernels	0.16	0.00	0.02	0.02
Zn	Stalks	40.0	46.0	44.9	40.6
	Leaves	40.0	23.0	40.3	43.0
	Kernels	80.0	100.6	107.1	90.7

Table 4.28 Transfer coefficients for the different plant parts sampled in the first maize trial

Metal (mg kg^{-1})	Plant part	+ Control	- Control	Exp. 1	Exp. 2
Cu	Stalks	0.16	0.09	0.18	0.16
	Leaves	0.18	0.14	0.12	0.14
	Kernels	0.05	0.06	0.16	0.09
Mn	Stalks	nd	nd	Nd	nd
	Leaves	nd	nd	Nd	nd
	Kernels	nd	nd	Nd	nd
Pb	Stalks	0.01	0.01	0.02	0.00
	Leaves	0.04	0.08	0.07	0.04
	Kernels	0.01	0.00	0.00	0.00
Zn	Stalks	0.3	0.3	0.3	0.3
	Leaves	0.3	0.2	0.3	0.3
	Kernels	0.7	0.8	0.8	0.7

Maize (Trial 2)

Table 4.29 presents the concentrations of Cd, Cu, Pb, and Zn in the soil before and after application of sludge. Zn levels slightly exceeded the guideline limits but exceeded the limits by a factor of two to three in the Sludge 2 treatment. In all the treatments the extractable (EPA 3050) levels decreased from the beginning to the end of the trial. Both Pb and Cu substantially exceeded the limits but there was little difference between the control and the sludge treatments. Again the Cu levels decreased from the beginning to the end of the trial. The Cd levels were substantially below the guideline limits.

Table 4.30 presents the Cd, Cu, Pb, and Zn concentrations in the leaves and the kernels of maize. Both Zn and Cu exhibited very little difference between the control and the sludge treatments with higher levels in the leaves than in the kernels. Pb exhibited a distinct increase in leaf levels from the control to the sludge treatments although the kernel levels were very similar. This is somewhat in contrast to the findings of the first maize field trial. Little Cd was taken up by the plants (leaves and kernels) and is probably a function of the low levels found in the soil.

Table 4.29 Metal concentrations in the soil at the beginning and end of the second maize field trial

Metal (mg kg^{-1})	Stage	+ Control	- Control	Exp. 1	Exp. 2
Cd	B	0.50 ± 0.1	0.58 ± 0.1	0.87 ± 0.1	0.78 ± 0.2
(Limit = 2)	E	0.78 ± 0.2	0.75 ± 0.1	1.02 ± 0.2	1.05 ± 0.0
Cu	B	72.3 ± 2.5	69.2 ± 9.7	70.5 ± 6.5	72.9 ± 15
(Limit = 6.6)	E	50.3 ± 2.9	50.1 ± 5.1	57.6 ± 5.3	59.4 ± 2.7
Pb	B	40.7 ± 3.3	33.9 ± 5.1	33.6 ± 6.4	53.9 ± 8.1
(Limit = 6.6)	E	46.0 ± 7.4	48.1 ± 3.6	32.7 ± 7.0	55.4 ± 11.1
Zn	B	74.2 ± 2.5	80.4 ± 9.7	83.9 ± 6.5	174.4 ± 15.0
(Limit = 46.5)	E	56.2 ± 11.5	46.5 ± 16.0	71.2 ± 2.4	94.8 ± 22.6

Table 4.30 Metal concentrations in the different plant parts of the second maize field trial

Metal (mg kg^{-1})	Plant part	+ Control	- Control	Exp. 1	Exp. 2
Cd	Leaves	0.00 ± 0.0	0.00± 0.0	0.0 ± 0.0	0.1 ± 0.1
	Kernels	0.2 ± 0.1	0.1 ± 0.1	0.2 ± 0.1	0.0 ± 0.0
Cu	Leaves	64.5 ± 6.2	62.2 ± 4.4	64.4 ± 1.6	66.6 ± 0.7
	Kernels	15.0 ± 1.3	14.4 ± 1.7	13.9 ± 1.8	15.6 ± 2.5
Pb	Leaves	5.5 ± 1.3	6.1± 0.5	10.2 ± 1.4	18.9 ± 4.5
	Kernels	1.2 ± 0.2	1.0 ± 0.1	1.7 ± 0.1	2.0 ± 0.5
Zn	Leaves	41.5 ± 2.7	41.0 ± 3.7	41.5 ± 8.3	41.3 ± 4.3
	Kernels	23.8 ± 1.8	24.6 ± 3.6	24.6 ± 2.6	25.0 ± 0.9

Oats

The total concentrations of Cu, Pb, and Zn in the soil of the oats trial at the beginning and end of experiment are presented Table 4.31. Cu concentrations exceeded guideline limits at the beginning of the experiment due to high soil background levels. Pb and Zn concentrations were below guideline limits at the beginning of the experiment and the applications of sludge to the soil did not increase the total Pb and Zn concentrations significantly in the soil. The Cu, Pb, and Zn concentrations increased from the beginning to the end of the trial. The most likely reason for this is that final sewage effluent was applied to the field plots through irrigation due to extreme drought during the field trial. The increase in metal concentrations in the soil, such that the guideline limits were exceeded, could also be attributed to poor sampling techniques and/or error in analyses and sample preparation.

Table 4.31 Metal concentrations in the oats field trial soil at the beginning and end of the trial

Metal (mg kg^{-1})	Stage	+ Control	- Control	Exp. 1	Exp. 2
Cu	B	16.7	15.5	18.3	15.9
(Limit = 6.6)	E	60.7	53.8	56.5	59.7
Pb	B	2.3	2.7	2.4	2.4
(Limit = 6.6)	E	11.4	12.8	12.9	13.3
Zn	B	32.3	36.6	38.2	32.4
(Limit = 46.5)	E	85.1	89.0	114.5	105.9

The Cu, Mn, and Zn concentrations determined in the mature oats leaves are presented in Table 4.32. (It is not clear from Appendix 7 why Pb was not determined in the plant leaves). The metal concentrations did not reach phytotoxic levels in the leaves of the mature oats plants. The low uptake of Cu is due to the low mobility of Cu in soils. Manganese was not included in soil-metal concentrations since the Mn concentrations were too high in the soil as discussed under the previous trial. Higher uptake of Mn occurred in the Negative Control treatment, although uptake of Mn was mostly due to the soil background concentrations. Although there was an increase in the uptake of Zn in the sludge treatments it was not much higher than the Positive Control treatment.

Table 4.32 Heavy metal content in the oat plant leaves at plant maturity

Metal (mg kg^{-1}) [a]	Phytotoxic levels (mg kg^{-1}) (Smith, 1996)	+ Control	- Control	Exp. 1	Exp. 2
Cu	20-100	5.22	5.67	5.42	6.6
Mn	400-1000	163.8	235.2	176.9	159.6
Zn	100-400	33.0	26.8	35.8	38.9

[a] Metal concentration is expressed as mg of total extractable metals per dry kg of leaves. Each digestion was performed at least 4 times.

Sunflower

Table 4.33 indicates the Cd, Cu, Pb, and Zn concentrations in the soil during the experiment. The Cd concentrations increased slightly in the sludge treatments and also from the beginning to the end of the trial but never exceeded the guideline limits. The Cu, Pb, and Zn concentrations exceeded the guideline limits – Cu by several factors and Zn in the Exp. 2 treatment. The Cu levels increased from the beginning to the end of the trial but the Pb and Zn levels remained fairly constant with the exception of Zn mentioned above. At the beginning of the experiment the Sludge 2 treatment exhibited higher Pb levels than the other treatments as was expected. The Negative Control however exhibited higher levels than the other treatments at the end of the trial indicating the possibility of a sampling error.

Table 4.33 Metal concentrations in the soil at the beginning and end of the sunflower trial

Metal (mg kg^{-1})	Stage	+ Control	- Control	Exp. 1	Exp. 2
Cd	B	0.52 ± 0.1	0.56 ± 0.1	0.73 ± 0.2	0.81 ± 0.2
(Limit = 2)	E	0.64 ± 0.1	0.68 ± 0.1	0.95 ± 0.1	1.06 ± 0.2
Cu	B	63.9 ± 4.6	66.2 ± 8.6	69.2 ± 4.8	67.0 ± 10.8
(Limit = 6.6)	E	50.9 ± 5.6	49.5 ± 5.9	56.2 ± 6.2	44.5 ± 1.2
Pb	B	26.9 ± 4.6	29.7 ± 10.5	25.6 ± 4.6	45.8 ± 2.7
(Limit = 6.6)	E	35.7 ± 17.0	54.4 ± 10.5	28.5 ± 4.2	44.4 ± 1.2
Zn	B	66.4 ± 12.3	63.4 ± 6.8	67.0 ± 14.3	145.4 ± 24.0
(Limit = 46.5)	E	56.2 ± 4.5	48.9 ± 12.6	71.2 ± 14.8	94.8 ± 9.5

Heavy metal concentrations did not reach phytotoxic levels in the leaves of the mature oats plants (Table 4.34). Although there was an increase in the uptake of Zn, Pb and Cu in the sludge treatments, the levels were mostly of the same order of magnitude as the control treatments. Metal levels in the sunflower seed also did not indicate a significant increase from the control treatments to the sludge treatments although the Sludge 1 treatment indicated higher levels than the other treatments.

Table 4.34 Metal concentrations in the seeds and leaves of the sunflower field trial

Metal (mg kg^{-1})	Plant part	+ Control	- Control	Exp. 1	Exp. 2
Cd	Leaves	Nd	Nd	nd	nd
	Seeds	Nd	Nd	nd	nd
Cu	Leaves	32.8 ± 4.6	33.5 ± 5.2	35.9 ± 4.2	38.9 ± 2.8
	Seeds	23.7 ± 2.9	23.1 ± 3.1	30.1 ± 4.3	23.7 ± 5.6
Pb	Leaves	4.4 ± 0.3	4.0 ± 0.4	4.6 ± 0.6	9.3 ± 4.4
	Seeds	3.0 ± 0.3	2.6 ± 0.1	4.0 ± 2.2	2.5 ± 0.4
Zn	Leaves	40.0 ± 6.2	45.5 ± 11.6	47.8 ± 9.9	53.6 ± 3.4
	Seeds	73.1 ± 14.9	61.8 ± 11.6	93.7 ± 20.6	69.6 ± 6.4

4.4.1.4 Yield Results

Maize (Trial 1)

Yield differences were calculated and are shown in Table 4.35. Statistical analyses of dry mass and wet mass (t ha^{-1}) of ears did not show any significant differences between treatments, due to the varying environmental factors that might have influenced soil conditions during the different growth stages of the plant under field conditions. Another consideration is that the results are based on a single application of sewage sludge, and assumptions concerning aspects like heavy metal uptake and yield can only be made over the long-term. However, sludge amended plots still compared well with Positive Control (inorganic fertilizer) for these parameters, and 12.5 t ha^{-1} (Exp. 1) showed higher values than the Positive Control (Table 4.35). The average amount of ears plant^{-1} indicated a significant difference between the 12.5 t ha^{-1} treatment and the other treatments.

Table 4.35 The effect of sewage on yield of maize under field conditions in loam soil

Treatment	+ Control	- Control	Exp. 1	Exp. 2
Ears plant^{-1}	1.93*^{a}	1.85^{a}	2.21^{b}	1.93^{a}
Dry mass (t ha^{-1})	5.05^{a}	5.3^{a}	5.28^{a}	4.9^{a}
Wet mass (t ha^{-1})	9.9^{a}	10.26^{a}	10.2^{a}	9.46^{a}

* Each value is a mean value per plant of 5 replicates of 16 plants. Values within a row not followed by the same letter are significantly different (*P* = 0.05) according to Duncan's multiple range test.

Maize (Trial 2)

No yield data is available for the second maize field trial.

Oats

Yield differences are shown in Table 4.36. Statistical analyses of dry mass, wet mass and plant height did not show any significant differences between treatments. Sludge treatments compared well with other treatments and showed a slight increase in the dry mass (t ha^{-1}) for the low sludge application rate (Exp. 1). Yield could possibly have been influenced by the sewage irrigation water, which usually contains high phosphorus concentrations that affect soil microorganisms like mycorrhizae, which enhance plant growth.

Table 4.36 The effect of sewage sludge on yield of oats under field conditions in loam soil

Treatment	+ Control	- Control	Exp 1	Exp. 2
Wet mass (t ha^{-1})	31.6*$^{, a}$	31.6^{a}	28.8^{a}	30^{a}
Dry mass (t ha^{-1})	5.0^{a}	4.83^{a}	5.5^{a}	4.83^{a}
Plant height (cm)	126.0^{a}	122.1^{a}	127.6^{a}	126.4^{a}

* Each value is a mean value per plant of 5 replicates of plants 0.6m^2. Values within a row not followed by the same letter are significantly different. (P = 0.05).

Sunflower

Due to extensive bird damage it was not possible to accurately determine the seed yield of the sunflowers.

4.4.1.5 Pathogenic Indicator Organisms

Maize (Trial 1)

Table 4.37 indicates the average values for the pathogenic indicator organisms detected in the soil at the beginning of ear-formation and in the kernels.

Although pathogenic indicator organisms like faecal coliforms were detected, no pathogenic indicator organisms were detected when the kernels were analysed. The path for sludge-borne pathogens to move through the plant root system, into the stem, from the stem to the leaves and from there to the kernels, is difficult and as shown, any uptake of these organisms is unexpected. Therefore the further field experiments do not include the detection of sludge-borne pathogens in plant parts.

Table 4.37 Pathogenic indicator organisms detected in the soil at the beginning of ear formation and in the kernels of maize

Sample	+ Control	- Control	Exp. 1	Exp. 2
Soil:				
Total plate count CFU 100 ml^{-1}	293 000	160 000	220 000	193 000
Faecal coliform CFU 100 ml^{-1}	333	733	333	633
Salmonella	Negative	Negative	Negative	Negative
Ascaris ova	1 non-viable	1 non-viable	2 non-viable	4 non-viable
Kernels:				
Faecal coliform CFU 100 ml^{-1}	0	0	0	0
Salmonella	Negative	Negative	Negative	Negative
Ascaris ova	Negative	Negative	Negative	Negative

4.4.1.6 Conclusions

The field experiments, compared to the greenhouse experiments, showed similar results when comparing the guidelines (interpreted as total metal content) to the total metal concentrations in the sewage sludge and soil. Although heavy metal uptake did not reach phytotoxic levels during the field studies, heavy metal behaviour in the soil after sludge application, and the uptake of heavy metals into the plant parts cannot be compared to greenhouse conditions. The transfer coefficients of Zn and Pb in the different maize parts were lower than the normal range and emphasised the differences between field and greenhouse experiments. The transfer coefficients of Cu and certain treatments of Pb were higher than the normal ranges. However, these transfer coefficients were still lower compared to the seedling greenhouse experiments. In the field experiments a much larger volume of soil is under the influence of several environmental parameters that cause metal availability and behaviour of metals to differ during the growing season of crops. The accumulations of potentially toxic elements (PTEs) in plant tissues can be increased 1.5-5 fold under greenhouse conditions compared with field studies with the same soil, sludge and crop (Logan & Chaney, 1983). Greater accumulations occur for a number of reasons including the confinement of plant roots to the small volume of treated soil in pots and the abnormal watering pattern and relative humidity in greenhouse pot studies (Logan & Chaney, 1983). Therefore, field experiments provide a more practical approach to sludge-application for farmers, while greenhouse experiments give a better understanding of heavy metal behaviour in soils.

No metal toxicity levels were found in plants (maize, oats, and sunflower) but this could become a different situation with continued application of sludge as discussed in the results presented at the start of this chapter.

There was no difference in yields between treatments for the maize trial. This seems somewhat contradictory since literature regularly indicates a distinct advantage of sludge over regular inorganic fertilizer. A slight increase in yield for the lowest sludge application was found in the oats and sunflower trial.

4.5 SOCIAL ACCEPTABILITY OF USING SEWAGE SLUDGE IN AGRICULTURAL PRACTICES (APPENDIX 13)

4.5.1 THE OPINION OF INDIVIDUALS (MAN ON THE STREET)

The responses (53) were divided into two groups, those earning less than R5000 per month/household and those earning above R5000 per month/household.

4.5.1.1 Individuals in a Household Earning Less Than R5000/month

Twenty-eight (28) questionnaires represented this group. More than half (53%) of these individuals were aware that domestic wastewater went to a wastewater treatment plant for treatment. Eighteen individuals (64%) of the population was in favour of the recycling of wastewater. Only 39% of these individuals knew what sewage sludge was. Table 4.38 shows the opinions of the 28 individuals who responded on the use of sewage sludge in agricultural practices.

The survey indicated that only 39% of the respondents were aware of what sewage sludge was before they read the information given to them. Table 4.38 indicates that after they read the information provided, 71% indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices. The respondents were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. There were conflicting results regarding the purchase of vegetables grown on sewage sludge, and consuming cooked and raw vegetables grown on sewage sludge. Table 4.38 indicates that 71% of the respondents would purchase vegetables grown on sewage sludge, but only 61% would consider consuming raw vegetables grown on sewage sludge and even less of the respondents, 54%, would consume cooked vegetables grown on sewage sludge-amended soils. This indicates a responsibility to educate the broader public in the benefits of washing, peeling and cooking vegetables to prevent infection from soil borne diseases and pathogen infection.

4.5.1.2 Individuals in a Household Earning More Than R5000/month

Twenty-nine (29) questionnaires represented this group. The majority of this group (90%) were aware that domestic wastewater went to a wastewater treatment plant for treatment. Twenty individuals (69%) of the population were in favour of the recycling of wastewater. In this group, 76% of the population surveyed knew what sewage sludge was compared to 39% representing the lower income group. Table 4.39 shows the opinions of the 29 individuals who responded on the use of sewage sludge in agricultural practices.

The survey indicated that 76% of the respondents were aware of what sewage sludge was before they read the information given to them. After they read the information provided, 79% indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices. The respondents (79%) were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. The majority of the respondents (79%) indicated that they would purchase vegetables grown on sewage sludge amended soils. However, this percentage dropped to 61% when asked whether they would consider consuming cooked vegetables grown on sewage sludge amended soils. As expected a lower percentage (45%) were prepared to consume raw vegetables grown on sewage sludge amended soils compared to their willingness (61%) to consume cooked vegetables grown on sludge-amended soil.

Table 4.38 The opinion of individuals who are part of a household earning below R 5000/ month on the use of sewage sludge for agricultural purposes

Question topic	Percentage (%)	Percentage (%)	Percentage (%)
Did you know that sewage sludge could be used as a fertilizer?	Yes: 71	No: 11	No response: 18
Do you think sewage sludge should be used as a fertilizer?	Yes: 82	No: 7	No response: 11
Would you buy vegetables grown using sewage sludge as a fertilizer?	Yes: 71	No: 18	No response: 11
The use of sewage sludge as a fertilizer is not risky to human health	Agreed: 36	Disagreed: 7	Neutral: 54 No response: 4
What impact do you think the use of sewage sludge as a fertilizer can have on the soil properties?	Improves soil quality: 75	Impacts soil quality negatively: 4	No response: 21
Do you think it is safe to use sewage sludge as a fertilizer for the following applications			
Application	**Percentage (%)**	**Percentage (%)**	**Percentage (%)**
Vegetables consumed raw	Yes: 61	No: 29	No response: 11
Vegetables consumed cooked	Yes: 54	No: 36	No response: 11
Tobacco	Yes: 64	No: 25	No response: 11
Vineyards	Yes: 79	No: 10	No response: 11
Application	**Percentage (%)**	**Percentage (%)**	**Percentage (%)**
Fruits trees	Yes: 82	No: 7	No response: 11
Cereal culture	Yes: 79	No: 10	No response: 11
Sugar cane	Yes: 82	No: 7	No response: 11
Gardens and traffic islands	Yes: 75	No: 14	No response: 11
Crops which animals producing milk, meat and eggs graze.	Yes: 64	No: 21	No response: 11

Table 4.39 The opinion of individuals who is part of a household earning above R 5000/ month on the use of sewage sludge for agricultural purposes

Question topic	Percentage (%)	Percentage (%)	Percentage (%)
Did you know that sewage sludge could be used as a fertilizer?	Yes: 79	No: 14	No response: 7
Do you think sewage sludge should be used as a fertilizer?	Yes: 79	No: 14	No response: 7
Would you buy vegetables grown using sewage sludge as a fertilizer?	Yes: 79	No: 21	No response: 0
The use of sewage sludge as a fertilizer is not risky to human health	Agreed: 55	Disagreed: 17	Neutral: 24 No response: 3
What impact do you think the use of sewage sludge as a fertilizer can have on the soil properties?	Improves soil quality: 69	Impacts soil quality negatively: 0	No response: 31

Do you think it is safe to use sewage sludge as a fertilizer for the following applications			
Application	**Percentage (%)**	**Percentage (%)**	**Percentage (%)**
Vegetables consumed raw	Yes: 45	No: 55	No response: 0
Vegetables consumed cooked	Yes: 69	No: 31	No response: 0
Tobacco	Yes: 76	No: 24	No response: 0
Vineyards	Yes: 83	No: 14	No response: 3
Fruits trees	Yes: 79	No: 17	No response: 3
Cereal culture	Yes: 79	No: 21	No response: 0
Sugar cane	Yes: 83	No: 17	No response: 0
Gardens and traffic islands	Yes: 72	No: 28	No response: 0
Crops which animals producing milk, meat and eggs graze.	Yes: 69	No: 0	No response: 31

4.5.1.3 The Opinion of Supermarkets and Vegetable Shops

The opinion of the buyers of vegetable shops and major supermarkets is important for this study. They are normally required to purchase good quality vegetables and fresh produce for their enterprise at the lowest price. It is important to establish whether a buyer would consider vegetables or other fresh produce grown on sewage sludge as a risk or as inferior quality.

The responses (11 questionnaires) received back from this market segment was disappointing. The individuals who filled in the questionnaires were not the individuals who purchase the vegetables and other fresh produce for the companies. They were generally not at the post level required to truly reflect the opinion of the general supermarkets and vegetable shops with regards to their willingness to purchase vegetables grown on sewage sludge-amended soils. The posts filled by the respondents included, training and development officer, office assistant, public relations officer, receptionist, salespersons, musician, and till operators. For this reason, these results are not included in this study.

4.5.1.4 The Opinion of Farmers

The opinion of seven farmers was obtained by requesting that they fill in the questionnaire. These farmers all make use of sewage sludge as a soil amendment and operate in the east Gauteng region.

The farmers grow the following crops on sludge-amended soils: maize, dry beans, including sugarbeans and soybeans, sunflowers, fodder and fruit trees. All seven the farmers were of the opinion that sewage sludge was a good fertilizer. The reasons given for why they thought sewage sludge was a good soil amendment included:

- Any organic matter is preferred over chemical amendments
- They referred to research done by Snyman *et al.*
- It is seen to be cost effective and effective as a fertilizer
- Sewage sludge is a natural organic product and gives very good results
- It is a way of preventing pollution

The farmers receive the sludge by truck and apply the sludge mechanically using a spreader during winter and/or before planting. All the farmers that replied added less than 8 t_{dry} ha^{-1} $year^{-1}$ with an average application rate of 2 t_{dry} ha^{-1} $year^{-1}$. The farmers rated sewage sludge as a good fertilizer but mentioned that the nitrogen and potassium content was insufficient and that they needed to add more sewage sludge than commercial fertilizer. All the farmers agreed that sewage sludge used in agricultural practices did not pose a risk to human health when handled responsibly.

CHAPTER 5: SOUTH AFRICAN BASED RESEARCH RESULTS WITH REGARD TO THE USE OF SEWAGE SLUDGE IN AGRICULTURE – INTEGRATED RESULTS

5.1 INTRODUCTION

This chapter addresses the following five aspects concerning the use of sewage sludge in agriculture:

- The extent of metal uptake in plants grown on sewage sludge-amended soils
- The effect of soil properties on the mobility of nutrients and metals
- The extent of metal uptake of different crops, including winter and summer crops
- The safe sludge load to prevent nitrogen leaching to groundwater and
- The persistence of sludge-borne pathogens during agricultural application.

In order to address these five aspects it is necessary to clarify and discuss the soil physical and chemical characteristics that exert an influence on them. Thereafter the results as presented in Chapter 4 will be discussed in the context of these soil characteristics.

5.2 SOIL PHYSICAL AND CHEMICAL CHARACTERISTICS

5.2.1 SEWAGE SLUDGE APPLICATION TO SOIL – ORGANIC MATERIAL

At present the bulk of sewage sludge that is applied to soil in agriculture in South Africa is in the form of anaerobic digested sludge. The characteristics of the sludge change rapidly as soon as it is applied to soil. This is particularly true under the prevailing conditions in S.A. where sludge is to be used in agriculture and high organic matter mineralisation rates prevail due to rapid microbial decomposition (Korentajer, 1991). The organic material in the "fresh" sludge is unstable in soil and will therefore undergo extensive alteration before it becomes material resistant to mineralisation (humus). In this process a large fraction of the total C is lost mainly to the atmosphere (Stevenson, 1986).

A combination of factors influences the rate at which the microbially mediated processes lead to the decomposition of organic material in soil (Smith, 1991). As a general rule soil organic matter decomposition is curvilinearly related to soil moisture and is slow at very wet and very dry conditions (Sikora & Szmidt, 2001). An air filled porosity of 0.6 was found to be optimal for CO_2 respiration and it declined at lower and greater values in a trial by Linn & Doran (1984). Temperature affects mineralisation where increases in temperature from 10°C to 35°C (ambient temperature) also increase the breakdown rate (Sikora & Szmidt, 2001). Below and above these temperatures the rates decrease drastically.

Soil pH influences a vast range of processes in soil. Increasing soil pH from acid to near neutral levels led to an increase in the N mineralisation rate in a study by Tester *et al*. (1977). The mineralisation to NH_4^+ from organic material is less sensitive to low pH than is the nitrification phase (Tester *et al*., 1977). Present guidelines state that the pH of a soil that receives sludge should be 6.5 (US EPA, 1983). At these pH levels organic material breakdown is very rapid under favourable climatic and drainage conditions (Tester *et al*., 1977). After the application of N in an organic or reduced form the nitrification phase could lead to acidification of the soil (Hue, 1995; Brallier *et al*., 1996; Sikora & Szmidt, 2001), especially if the produced NO_3^- is not utilised by plants and leaches out of the soil.

Very little is known about the effects of heavy metals in sewage sludge on soil mineral transformation processes (Smith, 1991). Mineralisation processes are less sensitive to heavy metals than is nitrification (Smith, 1991) although these organisms can adapt to high concentrations of heavy metals (Rother *et al*., 1982). In incubation studies discussed by Smith (1991) no negative effect of metals was found on the mineralisation of organic carbon. On the other hand, a study by Brooks & McGrath (1984) indicated that both sewage sludge and farmyard manure-applied soils had the same levels of organic material and pH. The microbial action in the sludge-applied soil, though, was half of the farm yard manure-applied soil due to the presence of metals.

5.2.2 COMPLEX FORMATION

In the sludge the heavy metals that are present are probably complexed with the organic material [or biotic surfaces – Nederlof & Van Riemsdijk (1995)] in different forms or incorporated into the cell structures of dead and living microorganisms. As the sludge breaks down in the soil, the organic component of the organo-metalic complexes is either mineralised or altered into humic compounds by microbial and biochemical processes, thereby releasing the metal into the soil solution. The metals are released into the soil solution where surfaces of Fe, Mn, Al, and Si minerals and soil conditions determine the sorption rate. Sorption is a collective term for adsorption and precipitation reactions and is used due to the difficulty in distinguishing between the two processes on a macroscopic level (Sparks, 1995; Yin *et al*., 2000). In soils these two processes are not clearly differentiated but rather are facets of a continuum (Choudhry, 1984). The sorption rate is dependent on a variety of factors including the number of binding sites, metal concentration in the soil solution, dominant type or combination of minerals, soil pH, redox conditions, as well as thermodynamic aspects determining the kinetics of the reactions (reaction rate and direction).

The presence of other elements in the soil could influence the stability of the metal complexes. Divalent cations present in high concentrations in clay mineral soils could lead to a decrease in heavy metal sorption by competing binding (Zachara *et al*., 1993). Divalent cations could also lead to the decreased solubility of organic matter in acid soil, therefore leading to increased heavy metal sorption or immobilisation (Yin *et al*., 2000).

5.2.3 IRON AND MANGANESE

Iron and manganese often make up a significant and very soluble and plant available component of the metals present in sludges and composts (Chaney *et* al., 2001; He *et al*., 2001). Upon release, at pH levels close to neutral (6.5 – as required by the guidelines), the Fe and Mn precipitate due to its low solubility at these pH levels and aerobic (oxidising) conditions. Most of the metals released from the mineralising organic material could undergo co-precipitation with Fe and Mn and therefore their solubility is largely determined by the solubility of the Fe and Mn minerals. The released metals can also undergo complexation reactions with the altered organic material leading to new stabilities of the newly formed organo-metallic complexes. The stability of the new phases depends on the type of metal as well as the characteristics of the mineral or organic compound onto which it is sorbed.

5.2.4 SOIL CHARACTERISTICS

A wide range of soil characteristics determines the mobility or extractability of metals. Mineral (and metal) stabilities in soils under different conditions are determined by factors such as the pH, reduction and oxidation potentials, clay content and type, presence of elements such as phosphorus (Laperche, 2000), organic material content and type (Yin *et al*., 2000), as well as microorganism activity in soil. Many of these processes are interrelated and could influence the other such as the effect of pH on microorganisms.

5.2.4.1 Soil pH and CEC

The capacity of the soil to adsorb or precipitate metals generally increases with increasing pH and a maximum is reached at neutral or slightly alkaline conditions, with As, Mo, and Se being exceptions. Cr^{6+} is also an exception, being more mobile under alkaline conditions (Adriano, 1986; McLaughlin *et al*., 2000). As the pH decreases there is a strong increase in the solubility of heavy metal complexes, which leads to higher bioavailability and consequently a possible higher uptake by plants (Mayer, 1991). Different metals act differently and Schwarz *et al*. (1999) found that, in a trial on the heavy metal release from soils during acidification, Cd was the most and Cr the least mobile.

The effect pH has on the heavy metal bio-availability has been confirmed in many different studies that range from metal availability after sewage application (Sauerbeck, 1991; Smith, 1994; Hooda *et al*., 1997) to the adsorption of metals on clay fraction minerals (Jinadasa *et al*., 1995; Kaupenjohann & Wilcke, 1995; Straalen & Bergema, 1995; Schwarz *et al*., 1999, Yin *et al*., 2000). The reasons vary from its influence on the dominant species in solution at different pH levels (McLaughlin *et al*., 2000) to the altered stability (or solubility) of minerals containing the metals during the addition or removal of H^+ (Schwarz *et al*., 1999). For most metals plant content is positively correlated with soil solution concentration, which in turn, is directly related to soil pH (Kabata-Pendias, 2001).

The inhibitory effect of metals on soil microorganisms is to a large extent determined by the CEC and pH of a soil. An increased CEC (Doelman & Haanstra, 1979) and pH (Doelman & Haanstra, 1984) leads to lower metal availability and subsequent increase in microorganism activity. Nitrogen transformations (nitrification and mineralisation) are also influenced by CEC (Wilson, 1977; Smith, 1991) and pH (Quraishi & Cornfield, 1973; Smith, 1991). Raising of the soil pH or CEC reduces metal toxicity to nodulation process in legumes (McIlveen & Cole, 1974).

5.2.4.2 Clay Content and Type

The clay content and clay type of a soil can often be positively correlated with the amount of metals taken up by plants (Hooda *et al*., 1997; Kabata-Pendias, 2001). In many cases the horizon in which an increased content of metals is found, the metal is associated with certain clay minerals (illuvial horizon) or organic material in the A-horizon (Aubert & Pinta, 1977). Structural aspects of soils also play a role in that lithogenic metals such as Al, Fe, or Cr show lower total concentrations on aggregate surfaces than in aggregate cores, whereas ubiquitously deposited metals such as Cd, Pb, or Zn show higher total concentrations on aggregate surfaces (Wilcke & Amelung, 1996).

Different clay minerals in the soil have different affinities for metals. The extent of this affinity is determined by the type of surface of clay particles namely siloxane (all basal surfaces of layer and lattice type clay minerals) and hydrous oxide surfaces (at all “broken edges” of clay minerals as well as the surfaces of chrystalline or amorphous minerals) (Greenland and Mott, 1978). The siloxane surface possesses mainly permanent charge due to isomorphous substitution and the hydrous oxide minerals possess mainly pH dependent surface charge.

The differential affinity for metals is mainly attributed to the effects of pH on variable-charged sorption sites, which also leads to the instability of Cr^{6+} at higher pH mainly due to the form of the oxyanions (CrO_4^{2-}) in solution (McLaughlin *et al*., 2000). Jinadasa *et al*., (1995) found that the metal ion adsorption on synthetically prepared goethite was strongly pH-dependent and that Cr was more strongly adsorbed than Cd and Pb. Fendorf & Gunter (1996) found that Cr^{3+} was more stable when precipitated on goethite than on silica. When a soil system buffers the addition of acid, heavy metals (e.g. Cr) bound in silicates are released into the soil solution due to the silicate’s destruction during the buffering process (Kaupenjohann & Wilcke, 1995).

5.2.4.3 Time Elapsed After Metal Application

The time elapsed after application of heavy metals to the soils also plays a role. Extractability of Co decreased with increased sorption time in a study by Bibak *et al*. (1995). Grove & Ellis (1980) found that all extractable Cr fractions, except amorphous and crystalline forms, decreased considerably after a few days to weeks after application. This was especially so for Cr^{3+} in moderately acid to neutral soils and Cr^{6+} in acid soils. Water extractable Cr^{6+} decreased only over several weeks in moderately acidic to alkaline (pH 7.5) soils.

5.2.4.4 Partition Coefficient

All these factors, and especially pH, determine the extent of the partition coefficient (≈ distribution coefficient) of a metal in soil (Merrington & Alloway, 1997; Impellitteri *et al*., 2001). This coefficient is defined as the total concentration of the metal in the soil solution divided by the total concentration of the metal in the soil. Soil-water partitioning of organic material as well as the affinity of particulate and dissolved organic matter also determine the partitioning of metals (Yin *et al*., 2000).

5.2.5 CROP USED AND TOTAL SOIL METAL CONTENT

Hooda *et al*., (1997) conducted a trial to determine the heavy metal availability from soils treated previously with sewage sludge and in which the sludge had time to equilibrate with the soil. The results indicated that the crop used and type of metal influenced the amount that was taken up. A further factor was the total metal content in the soil. Cd, Ni, and Zn levels showed the greatest increases over background levels and Cu and Pb less so, with spinach accumulating the most, carrot intermediate levels and wheat the least.

The variation described above is caused by many differences that exist in plants with respect to plant genetic variation in metal uptake or tolerance, variation within cultivars, the soils on which these plants evolved, and the complexity involved with the plant-rhizosphere-soil interaction (McLaughlin *et al*., 2000). Steyn (1994) stresses the point that simplified assumptions concerning which plants absorb the most trace elements cannot be made and that it depends on the species of plant and the metal concerned.

5.3 DISCUSSION OF TRIALS

5.3.1 THE EXTENT OF METAL UPTAKE IN PLANTS GROWN ON SEWAGE SLUDGE-AMENDED SOILS.

The metal uptake by the plants in all the trials would seem to indicate that there was very little, if any, difference between the sludge and the control treatments. The only exception was Zn in a very limited number of cases. In the pot trials the metal uptake was higher than in the field trials. This aspect is discussed by Alloway (1995) and Tiller (1989) and is a well-known phenomenon when heavy metal uptake trials are conducted in pots. The values, however, indicate almost no difference between control and sludge treatments. This therefore indicates that the sludge metal content was not responsible for the higher uptake. Although the uptake of metals during the field trials was lower than that in the pots, the same trend of no difference between treatments was observed. Again it can be deduced that the sludge metal content did not lead to an increased metal uptake by the crops.

It is further concluded that soil properties rather than sludge properties dictated the availability of the metals to the different plants in the different trials. All the trials that were conducted with plants consisted of sludge that was applied very soon before the planting of the plants. Should more time have been allowed between sludge application and the planting it is assumed that the metal uptake in the pots would have been lower. As it is though, there was no clear trend in metal uptake from the sludge-applied soil compared to the control treatments. Although some variability was observed in many of the treatments there was no trend and the variability is rather ascribed to standard experimental error and variation.

5.3.2 THE EFFECT OF SOIL PROPERTIES ON THE MOBILITY OF NUTRIENTS AND METALS

The role that soil properties play on the mobility of metal and nutrients can only be discussed in the light of the glasshouse trials and analysis of the sacrificial site soils.

5.3.2.1 Greenhouse Trials

Although the loamy soil had a pH of 5.3 and this pH was significantly lower than that of the other two soils, it is still considered too high to cause the large-scale mobilisation of metals (Laker, personal communication). This pH did indeed contribute to the increased mobility of especially Zn but only to a limited extent. The fact that this soil has a higher Fe-oxide content than the sandy soil could have diminished the effect of the soil's pH compared to the sandy soil.

The pH of a soil should not be considered on its own when assessing the risk of metal mobility in a soil. Coupled to this is the clay, as well as sesquioxide and other mineral, content of the soil (as discussed earlier). Added to this is the role of organic material in forming stable complexes with metals. These minerals and clay particles provide adsorption and precipitation sites for heavy metals (Chaney *et al*., 2001), therefore decreasing their mobility in soil. The larger the quantity of these mineral and clay particles at a given pH, the higher the soil's ability to retain metals. This aspect is probably the main contributor to the increased mobility of Pb in the sandy soil (mainly quartz particles) although it has a higher pH than the loamy soil (with more sesquioxides).

The above mentioned factors have been extensively discussed in literature (Jones & Jarvis, 1981; Adriano, 1986; McBride, 1989; Alloway, 1995; McLaughlin *et al*., 2000; Kabata-Pendias, 2001). It is therefore safe to say that soils with a high clay content, near neutral pH and large sesquioxide and organic material content will retain a larger quantity of metals than would a sandy soil with a low pH and very small sesquioxide and organic material content. Soils that fall between these two extremes in terms of their characteristics would therefore have a metal retention capacity depending on its characteristics relative to these cases.

The reasoning above, though, is an oversimplification of a very complex combination of processes. This is illustrated by the data presented in Chapter 4 on the sacrificial soils.

5.3.2.2 Sacrificial Soils

The long-term application of sewage sludge to the three soils has resulted in a significant accumulation of several heavy metals and organic material. Although there is a difference between the extractable metal levels between the three soils discussed, most notably in $BaCl_2$ extractable metals, the potentially available fraction (EDTA) seems to be similar, regardless of pH. This data confirms the overriding importance of organic material in the determination of metal mobility. Although there is a vast difference between the sandy and clay loam soils in terms of texture, the extractability of the metals was very similar.

Although the exchangeable Al and H serve as an indication of a soil's lime requirement in certain schools, it cannot be applied under conditions such as those of these soils. The exchangeable Al differs substantially between Soil 1 and 3 whereas the levels of extractable metals are very similar. Soils that are contaminated with heavy metals, such as these, should be limed to near neutral levels as matter of principle. The condition of the three soils confirms the need for proper pH guidelines when using metal containing sewage sludge in agriculture.

As can be seen from the data presented on the pH and organic mater content, it would seem that the soils from most lands that are used as sacrificial lands for sewage sludge disposal are acidified to a large extent. This depends on the amounts disposed of and the frequency of application. The most likely cause of the acidity is the oxidation of the added reduced compounds of N and S and, to a lesser extent, reduced forms of organic carbon. The pH of Soils 1 and 3 are considered to be very acidic and therefore would require liming to pH 6.5 under the present guidelines (WRC, 1997). This is also the internationally accepted norm and is considered “good practice” by other workers (Martens & Westermann, 1991; Logan, 1992; Gupta & Gupta, 1998; Chaney *et al*., 2001).

Many remedies for toxic levels of metals in soils have been proposed and tested but in most cases metal toxicity is alleviated through the addition of lime (Hooda *et al*., 1997; Kabata-Pendias, 2001). Liming increases the soil pH and lowers the activities of the metals in solution. In soils with high organic material contents this trend is sometimes not as clear and could seem to be reversed. It is especially true in the short-term, due to the increased solubility of organic material and organo-metallic complexes with increased pH (Allen & Yin, 1996; You *et al*., 1999). This is confirmed in the trial in which large quantities of lime was added to the sacrificial soils. It is not clear from the trial when the mobility of the metals will decrease and it is therefore the subject of further investigation.

Soil is a very dynamic system and any change in terms of the alteration of certain components (such as organic material or the addition of lime) could lead to a disturbed equilibrium (or more correctly – pseudo equilibrium). In this case the addition of lime and water led to an increased rate of organic material breakdown with a subsequent release and altered stability of certain metal-complexes. Although the long-term projections would seem to indicate a lower mobility of metals after lime addition the short-term conditions could differ substantially until a new (pseudo) equilibrium is reached.

From the trials it would seem that metals are less mobile when allowed to equilibrate with soil constituents over a long time. When sludge is applied yearly it would be safe to conclude that only the metals present in the sludge would pose an immediate risk of plant-uptake. The metals that were applied through sludge the year before would have been immobilised by soil constituents and would therefore contribute very little to the immediate metal pollution risk.

In the pot and field trials with crops heavy metal concentrations were determined very soon after sludge application. The plants would have taken up some of the metals as they were being released from the organic material upon mineralisation. This could also have accounted for the higher than normal transfer coefficients as determined in the trials. If the plants had been grown on the soils after a longer period of sludge mineralisation, it could very well be that the plant metal concentrations would have been lower.

5.3.3 EXTENT OF METAL UPTAKE OF DIFFERENT CROPS, INCLUDING WINTER AND SUMMER CROPS

Cadmium uptake by the seedlings of different crops exhibited no clear trends within treatments. In some cases the soybean seedlings had slightly higher transfer coefficients than the other crops except for the oats seedlings. Transfer coefficients for the oats seedlings were highly erratic and this phenomenon is explained in Chapter 4. The uptake of Cu was very similar to Cd in that only in some treatments was a slight trend observed of higher metal uptake coupled to sludge application. The transfer coefficients only differed between the crops in the soybean and oats seedlings grown on the sandy soil in that these values were slightly higher than the other seedlings and treatments. Very similar results were found for Pb with only the oats seedlings in the sandy soil exhibiting a limited increased uptake and transfer coefficient. Zinc was somewhat of an exception in that many of the treatments and crops exhibited an increased uptake, albeit not a very pronounced trend. This was also reflected in the transfer coefficients of some of the treatments.

There were very few clear increases in metal uptake linked to sludge application in the field trials. Transfer coefficients were not calculated for the different crops (except for the first maize trial) and comparisons between crops can therefore not be made. From the first maize trial it would seem that there was a slight trend in increased Cu and Mn uptake in the maize kernels but not in the other plant parts.

The differences observed in the pot trials were linked mainly to soil type and not to the crop type with no clear trend throughout the trials. This indicates that the type of crop did not play an important role in the extent of metal uptake but rather that the soil characteristics were more important.

5.3.4 SAFE SLUDGE LOAD TO PREVENT NITROGEN LEACHING TO GROUNDWATER

From an agricultural point of view, the slow release of N as found in the trial can hold numerous advantages. When the efficiency of commercial N fertilization is limited by factors such as high NO_3^- leaching losses, or NH_4^+ volatilisation, the use of sludge as a slow release N material may decrease the N losses and increase N availability (Korentajer, 1991). In the case of commercial fertilizer, the largest fraction is immediately available. The largest fraction of NO_3^- from sludge only becomes available after 14-28 d in the trial. This fraction of

NO_3^- can be utilised much more efficiently by crops, which need more N during this period than at the germination period. Large fractions of NO_3^- in the soil at any given time are liable to leach, taken that there is adequate water. The more the NO_3^- availability can overlap with the crop N needs, the better the N efficiency, and this reduces the risk of NO_3^- leaching.

Accurate N balances cannot be made, due to unmeasured N loss in the gas forms, during the process of nitrification. It is advisable to carry out more studies on this topic, especially in the long-term. Use of lysimeters can be beneficial to N dynamics studies because the leached fraction can be collected and analysed, and field conditions can be controlled to some extent. Laboratory analysis does not always give a realistic representation due to the optimum conditions under which the trial is usually carried out.

The laboratory incubation studies indicated that a larger percentage of the organic N was mineralised when higher sludge dosages were applied. This would seem to indicate that the present 8 t ha^{-1} guidelines are sufficiently cautious. The guidelines, however do not take into account the extent of plant uptake and losses to the atmosphere of different N-compounds when sludge is used in agriculture. The present study indicated that all mineralised N was nitrified within 28 d from application under ideal conditions. These conditions seldom exist in the field so it would be safe to assume that the rate will be significantly slower under field conditions. This coupled to the N-requirement and uptake rate of most crops would indicate that the present guidelines are very conservative in terms of the sludge loading rate. Although it is impossible to predict a safe sludge load to prevent nitrogen leaching from the present data, it is recommended that extensive field trials be conducted to determine a safe load under different climatic, crop and soil conditions.

5.3.5 THE PERSISTENCE OF SLUDGE-BORNE PATHOGENS DURING AGRICULTURAL APPLICATION

From the data presented it is clear that pathogen persistence on potato (a crop that is excluded by the guidelines for production with sewage sludge) reached a peak after six weeks and decreased thereafter. Although faecal coliforms, *E. coli*, and *Salmonella* was found on the potato peel none were found in the potato cores. The risk of pathogens is well documented and crops such as potatoes, lettuce, spinach (and others) are excluded by the guidelines for production with sewage sludge. In the maize field trial pathogenic indicator organisms were found in the soil but none were found on or in the maize plants.

5.3.6 SOCIAL ACCEPTABILITY OF USING SEWAGE SLUDGE IN AGRICULTURAL PRACTICES

The surveyed group representing households earning less than R5000 per month indicated that only 39% of the respondents were aware of what sewage sludge was before they read the information given to them. After reading the information provided, 71% indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices. Most of the respondents were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. The results indicated that the respondents did not understand the risks involved in using sewage sludge as an agricultural soil amendment. The wastewater industry will need to embark on a widespread awareness of both the negative and positive impacts relating to the use of sewage sludge in agricultural practices.

The group surveyed that represented the households earning above R 5000 per month was more informed about the source of sewage sludge. The majority, 79%, indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices and were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. The majority of the respondents (79%) indicated that they would purchase vegetables grown on sewage sludge-amended soils. 61% of the respondents would consider consuming cooked vegetables grown on sewage sludge-amended soils and 45% were prepared to consume raw vegetables grown on sewage sludge-amended soils.

The perception of the buyers of fruit and vegetables for the commercial markets and shops was not adequately captured in this study. The people that filled in the questionnaires were generally not at the post level required to truly reflect the opinion of the general supermarkets and vegetable shops with regards to there willingness to purchase vegetables grown on sewage sludge-amended soils.

The opinion of the farmers who are currently using sewage sludge as a soil amendment was obtained. The farmers were all in favour of using sewage sludge as a soil amendment and none of them currently exceeded the recommended dosage of 8 t_{dry} ha^{-1} $year^{-1}$.

The results of this study give a preliminary idea of the perception of the public regarding the acceptability of using sewage sludge as a soil amendment in agricultural practices. It is recommended that a detailed survey be done to establish the opinion of the broader South African population.

5.4 RECOMMENDATIONS

5.4.1 TECHNICAL RECOMMENDATIONS

From the data presented it is clear that sacrificial sites are not sustainable in the long-term due to intensive acidification that is difficult to overcome. The chemical condition of these soils could lead to the leaching of acidity and metal contaminated water. Management of these sites in terms of regular lime addition and the establishment of a crop cover is recommended to prevent metal pollution and expensive rehabilitation measures.

The long-term impact of sludge addition and its effect on the accumulation of metals could not be ascertained from the trials. This is due to the fact that most of the plant trials were conducted very soon after sludge application. It is therefore necessary to determine metal uptake from long-term trials in which sludge is added yearly over a few years. At present it would seem as if the immediate threat of metal uptake by plants is very small. The wider but controlled use of sludge in agriculture is recommended.

Nitrogen leaching from sludge poses a smaller threat than from inorganic fertilisers due to its slower mineralisation and nitrification. The incubation trials did not ascertain the amount of N that could be taken up by plants or lost to the atmosphere. Both these “sinks” of N are substantial and it is therefore recommended that higher sludge dosages could be considered in high clay content soils and conditions where there is adequate plant cover. The application rate should be linked to the type of crop and its N requirement through the growing season.

Due to the impact of sewage sludge on soil properties such as pH, metal content, and organic carbon content, it is suggested that sites that receive sewage sludge should be monitored on a regular basis. Corrective measures such as the addition of lime or the cessation of sludge application should be taken in the light of the analysis results. Proper guidelines are however necessary if adequate monitoring is to be carried out successfully.

Pathogens could pose a problem on certain crops, especially soon after sludge application. Current crop guidelines are adequate until a thorough investigation into different crops has been carried out.

5.4.2 RECOMMENDATIONS FOR FURTHER RESEARCH

The research conducted in this study consisted of short-term trials and yielded variable results. The generated data can therefore not contribute significantly to the drawing up of new guideline metal levels in sludges and soils as well as metal application levels. The restriction lies in the short-term nature of the trials. Any future research on guideline metal levels should be conducted on a medium to long-term level to ascertain significant trends. This aspect should be noted by funding and research institutions.

Present guidelines for the use of sewage sludge in agriculture should be reviewed to take into account South African climatic and soil conditions. The approach of McLaughlin *et al.*, (2000) is recommended when drawing up guidelines. They propose a five points set of quality screening criteria for soil tests and the option of the partitioning of calibration data in regions, or according to an environmental endpoint (plant toxicity, microbial toxicity, crop metal uptake) is urged. Furthermore, pathways and test methodologies are suggested to focus and improve research efforts in the soil testing for metals and metalloids.

The five point set of quality screening criteria is quoted from McLaughlin *et al.*, (2000) as follows:

“The soil test:

1) should be relatively simple, inexpensive and robust;
2) should be calibrated under field conditions across a wide range of soil types;
3) should be independently validated;
4) should account for the major environmental factors known to affect crop metal concentrations or toxic response to plants or organisms …;
5) for prognosis, must be truly predictive, ….”

The recommendation is therefore that a number of test sites be identified in South Africa that are representative in terms of climatic and soil conditions. These sites should be identified through a pre-selection that would exclude soils with obvious risks in terms of depth, shallow water tables and hydromorphic conditions as well as other restricting criteria. Sludges of different qualities should be applied consistently for a number of years at varying levels and a number of crops should be grown. Regular monitoring of nitrogen levels in the soil as well as metal levels in soils and plants should be conducted.

5.5 FUTURE EXPENDITURE ON PROJECTS

The continuation of two projects will require further expenditure. In the first trial the losses of N from sludge-applied soil will be investigated using a set of Gas Chromatographs as well as plants. This study will be conducted in Belgium at the University of Ghent.

The investigation into the mobility of metals after liming in sacrificial soils will also be continued. An incubation trial is currently underway in which different sacrificial soils are incubated with lime over a 24-week period. At set intervals replicates will be removed and analysed for organic carbon fractions as well as EDTA extractable metals over several time intervals. With this study it is intended to determine the extent and time duration of increased metal extraction after liming of sacrificial soils and to make recommendations in terms of their rehabilitation and future use.

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Appendix 1

EXTRACTABLE HEAVY METAL FRACTIONS FROM THREE SOUTH AFRICAN SACRIFICIAL SOILS

1.1 SUMMARY

In South Africa large volumes of sewage sludge with high metal levels are often disposed of on “sacrificial” lands, some of which have received prolonged regular applications of sewage in suspension form. In the Gauteng province around Pretoria three soils: a gravely sandy loam (Soil 1) on granitic parent material, a sandy clay loam (Soil 2) and a loam soil (Soil 3) on dolomitic parent material were collected for investigations into their accumulated heavy metal levels. Soils 1 and 3 had no additions of lime and were both very acid ($pH_{(Water)}$ of 4.0). Soil 2 received regular additions of lime ($pH_{(Water)}$ of 6.5) and was used as an agricultural soil. Four different extraction procedures were used namely: a saturated paste extract (water-soluble metals); $BaCl_2$ (exchangeable metals); NH_4-EDTA (potentially plant available metals) and EPA 3050 digestion (total metal content). Appreciable quantities of different heavy metals and organic material have accumulated over time in these soils. The EPA 3050 digestion indicated that Zn, Pb, Cu, Cd, Cr, Ni and V accumulated to levels above 100 mg kg^{-1} in the three soils. Cu, Pb and Zn levels in excess of 10 mg kg^{-1} were extracted with EDTA; Cu and Zn levels of more than 5 mg kg^{-1} with $BaCl_2$ and Zn and Ni levels above 0.5 mg kg^{-1} with water. In Soil 1 and 3 (pH 4) EDTA and $BaCl_2$ extracted similar levels of metals in most cases whereas EDTA extracted significantly more than $BaCl_2$ in Soil 2 (pH 6.5).

1.2 INTRODUCTION

The generation of sewage sludge worldwide contributes to environmental problems and problems concerning adequate disposal (Hue, 1995). Application to agricultural land during its use as an ameliorant or soil conditioner is often practiced although in limited quantities in different countries. This is due to high heavy metal levels in the sludges or environmental threats posed by high N or P additions to certain soils (Dam Kofoed, 1984). Some sludge is disposed of on dedicated disposal sites on land or otherwise known as sacrificial lands. On these lands application times and rates/amounts vary widely but generally the lands are acidified (if well drained) and enriched with heavy metals and especially high levels of decomposing organic material. Soil acidification often occurs in soils due to sludge oxidation and the subsequent nitrification of N-compounds (Hue, 1995). Very little remedial action (lime, other chemical inputs) other than physical actions (ploughing, ridging, construction of paddocks, etc.) is taken. Generally the lands are left for grass or weeds to grow.

The magnitude of the elemental enrichment problems depends on the soil and underlying material properties (such as texture, depth of profile, dominant soil minerals, etc.) as well as climatic conditions such as average temperatures and rainfall (quantity, intensity and distribution) (Sommers *et al*., 1987). The acidity also poses several problems, especially concerning the mobility of the heavy metals in the soil. It is for this reason that it is considered good practice to maintain soil pH above 6 - 6.5 during the use of sludge on land (Chaney *et al*., 2001).

Addition of organic material to soil is considered a good practice especially in countries such as South Africa where organic C levels are low (Korentajer, 1991). In the case where sewage sludge is added to sacrificial lands the organic C levels can increase to such levels that the soil's pH buffer capacity is greatly enhanced. With the sewage sludge reduced compounds of especially N, P, and S are added and these lead to intense acidification upon oxidation. Under these conditions the soil therefore has a low pH with a very high buffer capacity. This leads to a very high lime requirement to ameliorate the soil to a pH of 6.5.

In order to minimise the risk of heavy metal contamination of the food chain or the environment certain guidelines have been put in place to regulate the amounts of possible pollutants to the soil. Table 1.1 represents such guidelines although most of these levels are under revision. The maximum permissible metal concentration in soils is the subject of extensive discussion with the international trend having shifted to rather quantifying the plant- or potentially plant-available metal levels (Beckett, 1989; McLaughlin *et al*., 2000). Although several extractants have been used to give an indication of the plant-availability of heavy metals (for example, Cajuste & Laird, 2000), Bruemmer & Van Der Merwe (1989) stated that the NH_4-EDTA-extractable heavy metals concentration gives a better estimate of those potentially available, and therefore suggested it to be used in the establishing of preliminary threshold values for heavy metals for South African soils.

TABLE 1.1 Maximum permissible contaminant concentration (mg kg^{-1}, dry basis) in ameliorants (Department of National Health and Population Development, 1991), maximum metal and inorganic content in soil (WRC, 1997), and suggested preliminary threshold value for NH_4-EDTA (pH 4.5) extractable heavy metals for the soils of South Africa (Bruemmer & Van Der Merwe, 1989)

Element	Maximum permissible concentration in ameliorants (mg kg^{-1})	Maximum permissible metal and inorganic content in soil (mg kg^{-1})	Suggested threshold values (NH_4-EDTA extractable) (mg kg^{-1} soil)
Cd	20	2	1
Co	100	20	10
Cr	1750	80	50
Cu	750	6.6	60
Hg	10	0.5	1
Mo	25	2.3	-
Ni	200	50	20
Pb	400	6.6	100
Zn	2750	46.5	100
As	15	2	-
Se	15	2	-

In order to determine the problems related to the judicious management of sacrificial lands, the aim of this part of the study was to investigate the heavy metal contents of three different soils from sacrificial lands in the Pretoria area of South Africa.

1.3 MATERIALS AND METHODS

For the purpose of the study three soils were collected in the Gauteng province around Pretoria. Soil 1 is a gravely sandy loam on granitic parent material, Soil 2 is a sandy clay loam and Soil 3 a loam both on dolomitic parent material. Soils 1 and 3 were true sacrificial soils with only a regular ploughing and no additions except regular sludge addition in suspended form. Soil 2 was used for agricultural purposes and therefore received regular additions of lime with sewage sludge also in the suspended form. These soils represent a very restricted view of the complete picture but should be able to illustrate some of the issues concerning rehabilitation and management of sacrificial lands.

The soils were analysed for texture, organic C and pH according to the methods prescribed by The Non-affiliated Soil Analysis Work Committee (1990). In order to determine the Effective Cation Exchange Capacity (ECEC) and extractable cations of the soil at current pH levels a $BaCl_2$ extraction was done. The Hendershot & Duquette (1986) method gave a very good indication of the ECEC of soils. The method was adapted as follows: 5 g of soil was shaken with 50 cm^3 0.1 M $BaCl_2$ in a glass bottle on a horizontal shaker for 1 h and filtered afterwards. The metals Ca, Mg, K, Na, Al, Mn, Fe, Cu, Zn, Pb, and Cd were determined through Atomic Absorption Spectrophotometry and Ni and V through ICP-MS. Due to Cl interference in the determination of Cr by ICP-MS a 0.2 M NH_4NO_3 solution was used, with the same procedure and quantities as stated for the $BaCl_2$ extraction, to determine Cr within the soils.

The water-soluble metals were determined through a saturated paste extract according to the method described by The Non-affiliated Soil Analysis Work Committee (1990). Again the metals Ca, Mg, K, Na, Al, Mn, Fe, Cu, Zn, Pb, and Cd were determined through Atomic Absorption Spectrophotometry and Ni, Cr and V through ICP-MS. Extractable acidity and Al was determined through a 1N KCl extraction and organic carbon according to the Walkley-Black method as described by the Non-affiliated Soil Analysis Work Committee (1990).

An EPA 3050 (U.S. Environmental Protection Agency, 1986) extraction was done to determine the concentration of the total sorbed Mn, Cu, Zn, Pb, Ni, Cd, Cr, and V. All the metals were determined by ICP-MS. According to Risser & Baker (1990) this method gives a reliable indication of metals added to soils as non-silicates from industrial sources and therefore metals that are potentially mobile in the environment (Soon & Abboud, 1993).

An EDTA extraction (The Non-affiliated Soil Analysis Work Committee, 1990) was used to determine potentially plant available metals. EDTA was found or proposed by many researchers to give a very good indication of the pollution hazard of heavy metals in soils as well as being a reliable test for predicting plant available metals (Cajuste & Laird, 2000).

The results were statistically analysed using the SAS® System statistical program to obtain the Analysis of Variance (ANOVA) and the LSD – Tukey to determine significant differences.

1.4 RESULTS AND DISCUSSION

Chemical and Physical Properties

Some of the chemical properties of the three soils are presented in Table 1.2. Although Soils 2 and 3 are very similar in origin it is suspected that the high organic material content contributed to the low clay percentage determination through the clumping of clay particles into larger units. Due to the high organic C content the H_2O_2 oxidation step appeared never to be long enough to oxidise all the material. There is a distinct difference between the pH values of soils 1 and 3 and soil 2. This is the result of the difference in management (regular additions of lime) of soil 2.

Table 1.2 Some chemical and physical properties of the three soils

Property		Soil		
		1	2	3
Sand (%)		70.2	45.7	46.4
Silt (%)		19.7	24.3	41.9
Clay (%)		10.1	30.0	11.7
Texture		SaLm	SaClLm	Lm
Organic C (%)		2.6	1.0	2.7
pH	Water	4.2	6.5	4.0
	$CaCl_2$	3.8	5.9	3.7
	KCl	3.3	5.5	3.5

Exchangeable and Water Soluble Cations

The regular liming of Soil 2 is reflected in the difference in Ca and Mg levels compared to Soils 1 and 3 (Figs 1.1, 1.2 and 1.3). Of significance is the large difference concerning the exchangeable Al between Soil 1 and 3 although they have a similar pH values (less than 10% for Soil 1 and more than 40% for Soil 2). The opposite is the case with exchangeable H. Coupled to the difference in Al is the difference in Fe and Mn with Soil 3 having much higher levels than Soil 1. The higher Al levels in Soil 3 indicate a much larger component of variable charge than Soil 1 as is reflected in the lower CEC of Soil 3 (finer texture) compared to Soil 1 (coarser texture). The CEC of Soil 1 is most likely dominated by the high organic material content and the CEC of Soil 3 by a combination of sesquioxides and organic material. The high clay content of Soil 2 is most likely the main contributor to the CEC of this soil due to the near neutral pH.

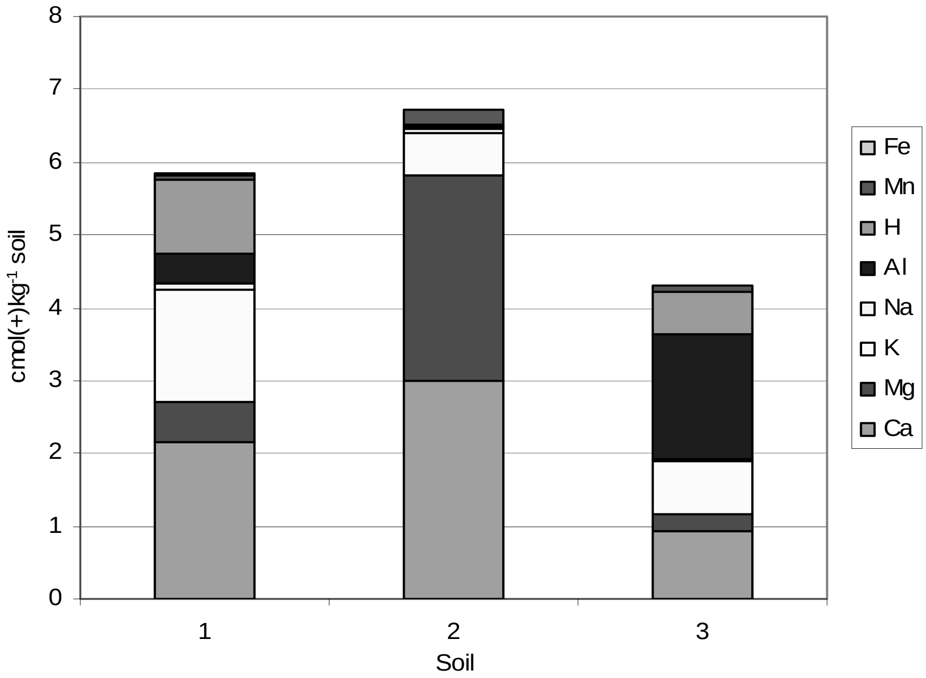

Figure 1.1. $BaCl_2$ extractable cations for the three soils. (Note: H was determined through a 1N KCl extraction).

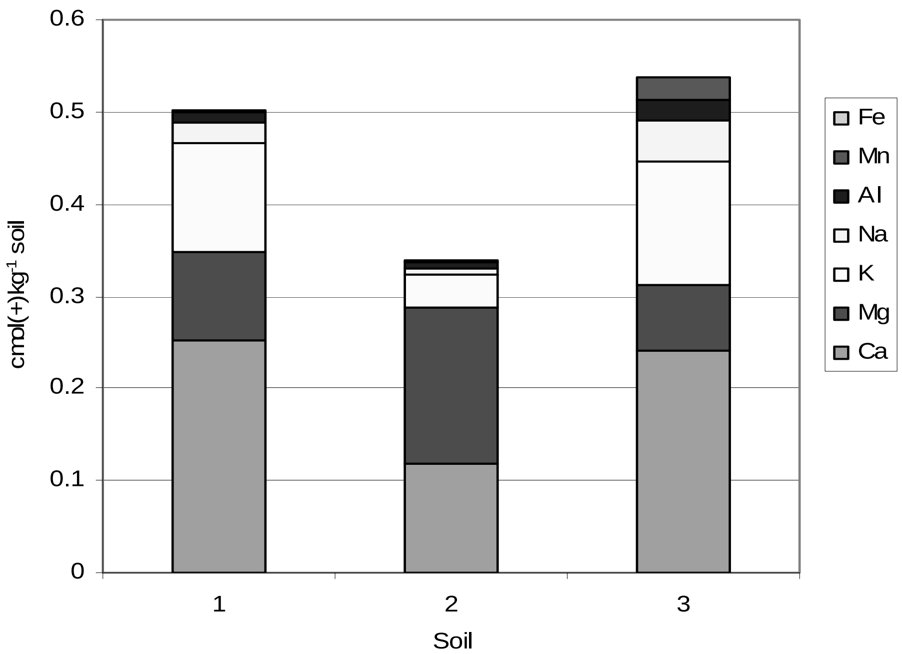

Figure 1.2. Cations determined in a saturation paste (water) extract of the three soils.

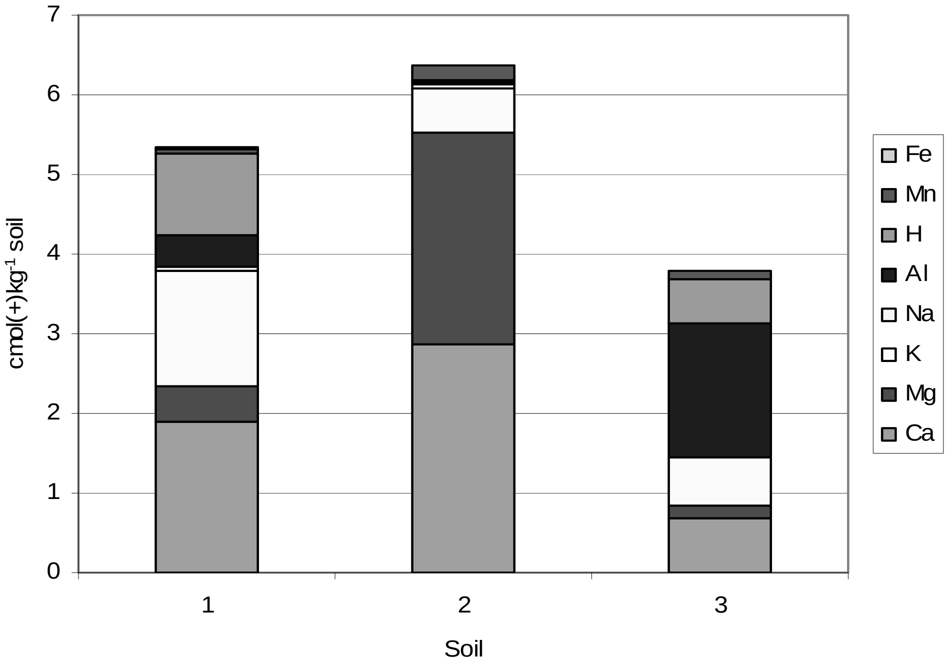

Figure 1.3. Composition of the exchange complex and resultant CEC of the different soils. [Note: This value was obtained by subtracting the water soluble cations (Fig. 1.2) from the $BaCl_2$ extractable values (Fig. 1)].

Heavy Metals

Table 1.3 presents the extractable metal levels for Soils 1, 2, and 3. When the EPA 3050 values are considered, the maximum permissible levels (Table 1.1, column 3) are exceeded in all the cases except Ni for Soil 1 and Cd for Soil 2. This is in line with the comments concerning the "total versus bio-available levels" argument. The picture changes considerably when the EDTA values are considered. Only Zn for Soil 1 and Cu for Soil 3 are higher than the proposed values. In most cases again the $BaCl_2$ values are much lower and the water-soluble levels are even more so. The highest values again are for Zn, Ni, and Cu in Soils 1 and 3, and Zn and Ni in Soil 2. The levels would indicate a very low immediate risk of leaching or plant uptake of these metals. Even though the EPA 3050 levels for the three soils are very similar, as well as most of the levels in the other extractants, they are lower in Soil 2 than the two acid soils indicating a larger total load of sludge. The EDTA, $BaCl_2$, and water extractable metals are presented as a percentage of the EPA 3050 extractable levels in Figures 1.4, 1.5 and 1.6 for Soils 1, 2, and 3 respectively.

Table 1.3 Metals extracted from the three soils with four extractants

Soil	Metal (mg kg^{-1})						
	Cd	Cr	Cu	Ni	Pb	V[‡]	Zn
				EPA 3050			
1	4.02*	151.78*	176.88*	26.85	246.19*	59.10	359.10*
2	1.48	250.04*	118.18*	102.00*	39.37*	141.61	101.63*
3	2.62*	313.13*	300.55*	72.64*	76.80*	157.36	106.39*
				EDTA			
1	0.30$^{\#bc}$	0.058^{bcd}	41.13^{b}	5.68^{b}	9.85^{a}	0.65^{a}	117.03^{b}*
2	0.68^{a}	0.11^{b}	36.82^{c}	6.60^{a}	10.36^{a}	0.64^{a}	30.96^{c}
3	0.50^{ab}	0.30^{a}	80.77^{a}*	4.04^{d}	8.40^{a}	0.18^{b}	28.82^{c}
				$BaCl_2$			
1	0.30^{abc}	0.04^{cd}	3.31^{e}	4.97^{c}	1.85^{b}	0.19^{b}	151.59^{a}
2	0.24^{bc}	0.04^{cd}	0.03^{f}	1.74^{e}	0.86^{b}	0.06^{c}	5.45^{e}
3	0.30^{abc}	0.09^{bc}	14.10^{d}	4.00^{d}	1.57^{b}	0	18.49^{d}
				Water			
1	0.01^{c}	0.006^{d}	0.17^{f}	0.26^{g}	0	0.006^{dc}	4.18^{e}
2	0.004^{c}	0.007^{d}	0.05^{f}	0.03^{g}	0	0.004^{dc}	0.25^{e}
3	0.04^{c}	0.013^{d}	0.40^{f}	0.54^{f}	0.01^{b}	0.003^{dc}	5.06^{e}

[#] Values with the same letter in a specific column indicate no significant difference
*** Values exceeding those proposed in Table 1**
[‡] Vanadium is not stipulated in the guidelines.

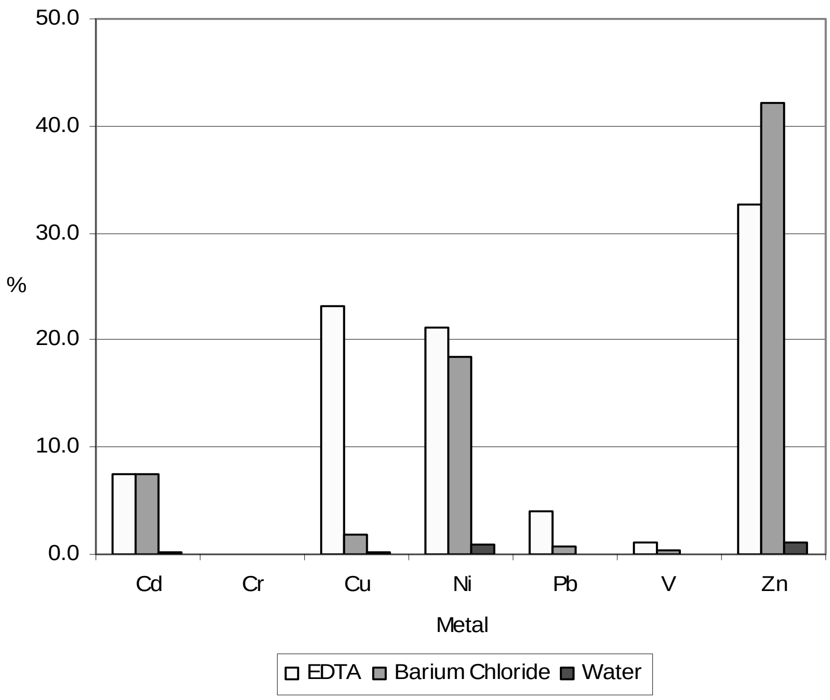

Figure 1.4. Metals extracted with three extractants expressed as a percentage of EPA 3050 extractable metals for Soil 1.

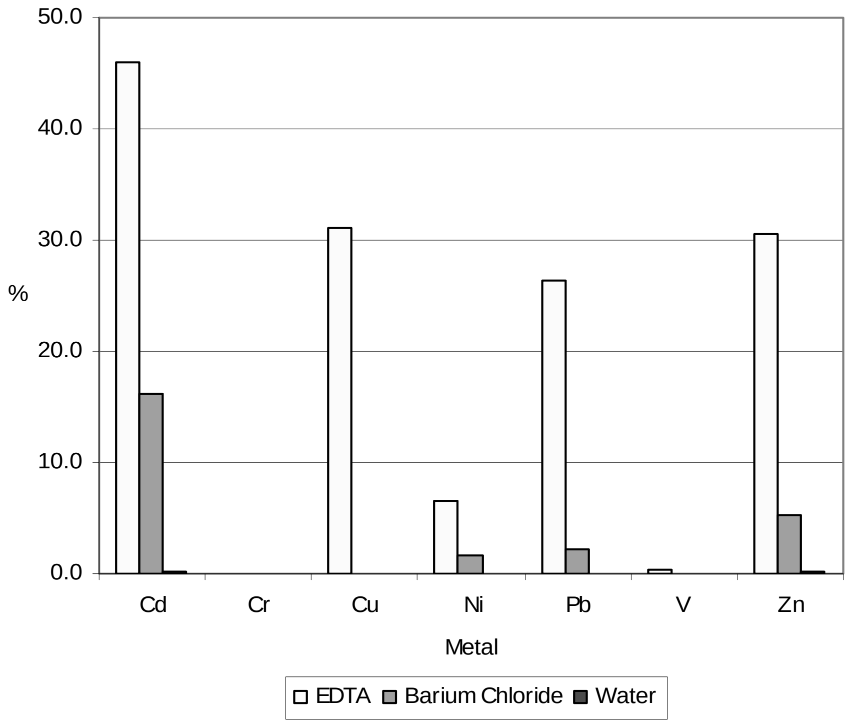

Figure 1.5. Metals extracted with three extractants expressed as a percentage of EPA 3050 extractable metals for Soil 2.

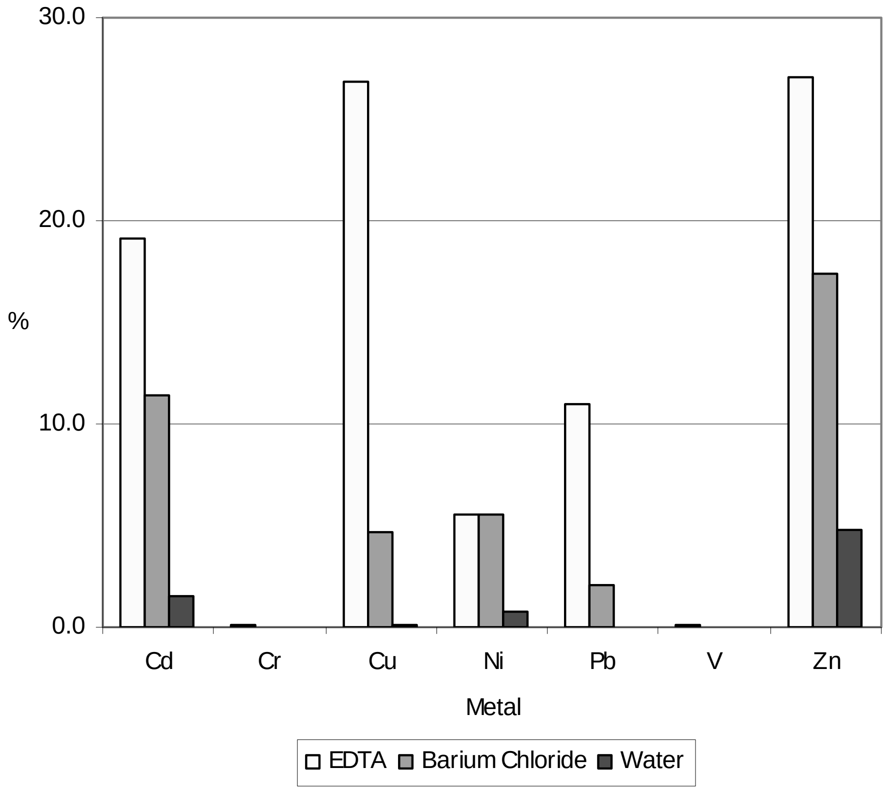

Figure 1.6. Metals extracted with three extractants expressed as a percentage of EPA 3050 extractable metals for Soil 3.

From the three figures there seems to be a trend in that the EDTA and $BaCl_2$ levels are closer for the acid soils than for the neutral pH soil – especially in the case of Cd, Ni, and Zn. This would indicate a larger possibility of plant uptake of these metals from the acid soils than from the neutral soil. This is to be expected as a pH dependant trend even though the EDTA levels are similar for the three soils indicating similar "potential availability" for the three soils. Cu and Pb indicate no such trend with EDTA and $BaCl_2$ extractable levels differing substantially regardless of the pH of the soil.

1.5 CONCLUSION AND RECOMMENDATIONS

The long-term application of sewage sludge to the three soils has resulted in a significant accumulation of several heavy metals and organic material. Although there is a difference between the extractable metal levels between the three soils, most notably in $BaCl_2$ extractable metals, the potentially available fraction (EDTA) seems to be similar, regardless of pH.

Present guidelines do not distinguish between different types of extraction agents when listing guideline values. This aspect, although receiving attention, needs further clarification as well as standardisation for South African conditions.

Due to the high levels of the metals that have built up in the soils, these soils require special attention in management to prevent further excessive increases and potential losses into other compartments of the environment. Liming to near neutral pH levels will have to be done and also included as a regular action in the management process. This aspect should, by implication, apply to all sites that are treated similarly in terms of sewage sludge disposal. Changes in metal extractability should also be monitored regularly.

On the grounds of the EDTA and $BaCl_2$ extractable metals, it would seem that Cd, Ni, and Zn are less influenced by the pH of the soil than are Pb and Cu. The metals Cr and V were the least mobile elements tested in the soils when considering the use of EDTA and $BaCl_2$ as extractants.

Although the exchangeable Al and H serve as an indication of a soil's lime requirement in certain schools it cannot be applied under conditions such as those of these soils. The exchangeable Al differs substantially between Soil 1 and 3 whereas the levels of extractable metals are very similar. Soils that are contaminated with heavy metals, such as these, should be limed to near neutral levels as matter of principle. The condition of the three soils confirms the need for proper pH guidelines when using metal containing sewage sludge in agriculture.

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Appendix 2

THE EFFECT OF LIMING ON pH AND EXTRACTABLE METALS FROM TWO SACRIFICIAL SOILS

2.1 SUMMARY

Two sacrificial soils were twice incubated with a total lime equivalent of 45 t ha^{-1} in pots but did not attain the desired pH of 6.5 due to a very high buffer capacity. Soil samples from the pots after the incubation were extracted with NH_4-EDTA and $BaCl_2$ and the levels of Al, Fe, Mn, Cu, Zn, Pb and Cd determined by Atomic Absorption Spectrophotometry. The $BaCl_2$ extractable Mn, Pb, and Cd in Soil 1 and Mn and Cd in Soil 3 indicated increases or similar levels in extractability after liming and Al, Cu, Fe and Zn levels decreased after liming. The EDTA extractable Cu, Mn, Fe and Cd in both soils and Pb in Soil 3 increased after liming and Al, Zn and to a lesser extent Pb in Soil 1 decrease in extractability. The increased extractability of certain of these metals is contrary to what was expected from literature concerning the effect of liming. This aspect is a cause for concern in the short-term if liming is to be done on acidified sacrificial lands as a method of rehabilitation.

2.2 INTRODUCTION

In Appendix 1 data concerning the extractable heavy metals and organic carbon content of three soils from sacrificial lands is presented. As can be seen from this data it would seem that the soils from most lands that are used as sacrificial lands for sewage sludge disposal are acidified to a large extent, depending of the amounts disposed of and the frequency of application. The most likely cause of the acidity is the oxidation of the added reduced compounds of N, S, and P as well as, to a lesser extent, reduced forms of organic carbon. The pH of Soils 1 and 3 are considered very acidic requiring liming to pH 6.5 under the present guidelines (WRC, 1997) and internationally accepted norms and what is considered “good practice” by other workers (Martens & Westermann, 1991; Logan, 1992; Gupta & Gupta, 1998; Chaney *et al*., 2001).

Several factors may influence the bioavailability of the heavy metals in sacrificial soils. According to Løbersli *et al*. (1991), one of these – soil acidity – is a major factor and therefore influences the pollution hazard of several heavy metals (Alloway, 1995; Kabata-Pendias, 2001). The sorption or possible precipitation capacity of the soil generally increases for most heavy metals with increasing pH, reaching a maximum at neutral or slightly alkaline levels, with As, Mo, and Se being exceptions. Cr^{6+} and V^{5+} are also exceptions, being more mobile under alkaline conditions due to their anionic form (Adriano, 1986; Alloway, 1995; McLaughlin *et al*., 2000). As the pH decreases there is a strong increase in the solubility of heavy metal complexes, which leads to higher bioavailability and consequently a possible higher uptake by plants (Mayer, 1991).

The effect pH has on the heavy metal bio-availability has been confirmed in many different studies that range from metal availability after sewage sludge application (Hooda *et al*., 1997) to the adsorption of metals on clay fraction minerals (Jinadasa *et al*., 1995; Kaupenjohann & Wilcke, 1995; Schwarz *et al*., 1999). The reasons vary from its influence on the dominant species in solution at different pH levels (McLaughlin *et al*., 2000) to the altered stability (or solubility) of minerals containing the metals during the addition or removal of H^+ (Schwarz *et al*., 1999). For most metals plant content is positively correlated with soil solution concentration, which, in turn, is directly related to soil pH (Kabata-Pendias, 2001).

Other factors also play a significant role in the immobilisation of metals applied through composts of sewage sludge. Many metals form stable complexes with organic material in soils (especially Cu) and could therefore become sparingly mobile (Hue, 1995). The contribution of Fe and Mn to the lowering of metal solubility through co-precipitation as the Fe and Mn precipitate out of solution in soils after release from compost or biosolids, has also been discussed by Hue (1995) and Chaney *et al*. (2001).

Many remedies for toxic levels of metals in soils have been proposed and tested but in most cases metal toxicity is alleviated through the addition of lime (Hooda *et al*., 1997; Kabata-Pendias, 2001), thereby increasing the soil pH and lowering the activities of the metals in solution. In soils with high organic material contents this trend is sometimes not as clear and could seem to be reversed, especially in the short-term, due to the increased solubility of organic material and organo-metallic complexes with increased pH (Allen & Yin, 1996; You *et al*., 1999).

Table 2.1 presents the current guideline levels of metals in ameliorants, soils and with the extractant EDTA. The aspect concerning the revision of these levels has been touched upon in Appendix 1. The guidelines do not include levels of metals linked to specific soil pH levels or organic material or sesquioxide contents and are therefor restricted in its application to the varying conditions encountered in South African soils.

TABLE 2.1 Maximum permissible contaminant concentration (mg kg^{-1}, dry basis) in ameliorants (Department of National Health and Population Development, 1991), maximum metal and inorganic content in soil (WRC, 1997), and suggested preliminary threshold value for NH_4-EDTA (pH 4.5) extractable heavy metals for the soils of South Africa (Bruemmer & Van Der Merwe, 1989)

Element	Maximum permissible concentration in ameliorants (mg kg^{-1})	Maximum permissible metal and inorganic content in soil (mg kg^{-1})	Suggested threshold values (NH_4-EDTA extractable) (mg kg^{-1} soil)
Cd	20	2	1
Co	100	20	10
Cr	1750	80	50
Cu	750	6.6	60
Hg	10	0.5	1
Mo	25	2.3	-
Ni	200	50	20
Pb	400	6.6	100
Zn	2750	46.5	100
As	15	2	-
Se	15	2	-

The original aim of this study was to determine the effect of different pH levels on the plant availability of the metals in the sacrificial soils. Metal extraction and the buffer capacity of the soil yielded unexpected results and these aspects will therefore be addressed here.

2.3 MATERIALS AND METHODS

The buffer capacity of the two acid soils (1 and 3) was determined with a $Ca(OH)_2$ buffer [method as described in Van Der Waals & Claassens (2002)]. The required amount of lime (according to the buffer determination) for a pH of 6.5 was added to each soil in 7.5 kg pots (with 4 repetitions) and the soil incubated for 3 months with regular watering. After sampling the soil and finding only a slight change in pH it was decided to add an equal amount of lime and incubate the soil again for the same period of time. The total amount of lime added amounted to the equivalent of 45 t ha^{-1}. The physical properties of the soils are listed in Table 2 in section 2.

After the second incubation period a representative sample was taken from each pot. On the limed soil and a sample from the original soil a $BaCl_2$ (method described in section 2) and EDTA (The Non-affiliated Soil Analysis Work Committee, 1990) extraction was done and Al, Fe, Mn, Cu, Zn, Pb and Cd determined by Atomic Absorption Spectrophotometry. The samples were tested at the same time to minimise experimental error differences between them. Cr, V and Ni were not tested in this study due to the contradicting results that were found with the cations that were tested first. The reasoning was to not expend time and finances on these elements in this trial but to rather focus a new study on these contradictions and then include Cr, V and Ni.

2.4 RESULTS AND DISCUSSION

Table 2.2 gives the pH of the two soils before and after liming. The pH values were increased by 1.4 pH units for Soil 1 and 1.8 pH units for Soil 3 by the equivalent of 45 t ha^{-1}. This indicates a massive buffering capacity brought about by organic material and complexes stable at pH 4.2 and 4.0 for Soil 1 and 3 respectively. This buffering capacity could complicate efforts to bring the pH of the soil to 6.5 in line with recommendations in the field.

Table 2.2 Soil pH before and after liming

Soil pH	Soil 1		Soil 3	
Method	Before	After	Before	After
Water	4.2	5.6	4.0	5.8
$CaCl_2$	3.8	5.3	3.7	5.7
KCl	3.3	5.0	3.5	5.4

Table 2.3 indicates the influence of the lime addition and incubation on the $BaCl_2$ extractable cations from the two soils. The metals Mn, Pb and Cd in Soil 1 and Mn and Cd in Soil 3 indicated increases or similar levels in extractability after liming. This is contrary to what the effect of liming was expected to be. As expected, a significant decrease was found for Al, Cu, Fe and Zn.

Table 2.3 The effect of liming on the $BaCl_2$ extractable cations (soil, n = 4)

Cation (mg kg^{-1})	Soil 1		Soil 3	
	Before	After	Before	After
Ca	431.8	1600.2	184.0	1726.4
Mg	65.1	224.2	28.5	336.65
K	610.3	412.9	284.3	260.8
Na	19.3	45.4	10.4	52.8
Al	35.0	0.2	154.7	0.2
Cu	3.3	0.2	14.1	0.8
Mn	17.5	**21.2**	29.7	**12.4**
Fe	6.0	0.5	0.8	0.3
Zn	151.6	11.5	18.5	3.0
Pb	1.8	**1.3**	1.6	0.3
Cd	0.3	**0.3**	0.3	**0.6**

Table 2.4 indicates the EDTA extractable cations for the two soils before and after liming. Here most of the tested cations (Cu, Mn, Fe, and Cd in both soils and Pb in Soil 3) increased in extractability or had similar levels. Al, Zn, and to a lesser extent Pb in Soil 1 indicted the expected decrease in extractability. The increased extractability of certain of these metals is contrary to what was expected from literature concerning the effect of liming. Furthermore, the possible competition of high levels of Ca with the other metals would have seemed to also indicate a lower extractability of the metals with EDTA.

Table 2.4 Effect of liming on the NH_4-EDTA extractable cations (mg kg^{-1} soil, n = 4)

Cation	Soil 1		Soil 3	
	Before	After	Before	After
Al	113.9	33.2	337.7	222.2
Cu	41.1	**37.1**	80.8	**104.6**
Mn	17.2	**49.6**	37.1	**71.3**
Fe	242.7	**383.5**	170.7	**271.7**
Zn	117.0	49.1	28.8	15.6
Pb	9.8	6.5	8.4	**8.1**
Cd	0.3	**0.8**	0.5	**1.3**

2.5 CONCLUSIONS AND RECOMMENDATIONS

The results contradicted what was expected from literature concerning the extractability of some metals. Metal adsorption on different adsorption sites in soil is mainly a pH dependent factor

Possible causes of this phenomenon are:

Breakdown of stable organic complexes (stronger than EDTA) and the formation of less stable complexes (weaker than EDTA) but still not exchangeable

This is a single once-off view into a set of reactions taking place over time in the stability of organo-metallic complexes

Increased solubility of organo-metallic complexes leading to the higher extractability with EDTA.

The results are a cause for concern in the short-term if liming is to be done on acidified sacrificial lands as a method of rehabilitation. Dedicated trials should be conducted to determine the extent and time dependency of the increased heavy metal extractable levels from such soils.

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Plant-soil interactions of sludge-borne heavy metals and the effect on maize (*Zea mays* L.) seedling growth

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Abstract

The use of sewage sludge as an organic fertiliser under South African conditions is an alternative disposal route to sacrificial land disposal. However, the lack of research done under South African conditions and the conservative nature of the heavy metal guidelines, when interpreted as total metal content is limiting the agricultural use of sludge. A glasshouse experiment, which forms part of a greater project, was conducted to characterise soil-plant interactions of the main sludge-borne heavy metals (Pb, Cd, Zn and Cu) in two sludges (low metal and high metal) to different soil types (clayey, loamy and sandy) on maize seedlings. Growth differences, heavy metal accumulation in plant parts and soil-metal concentrations (total and potentially available) were determined. The low metal sludge treatment showed the highest yield for maize seedlings when compared to controls (soil unamended and inorganic fertiliser added). The amendment of sludge to the soil did indicate higher heavy metal content, although the increase was not as predicted, owing to the difficulty of obtaining a representative sample in the soil. Except for Cd, heavy metal values in the soils (at the beginning and end of experiment) exceeded guidelines due to very high background values in the soil. No negative effects of heavy metal contamination in plant parts of the crops could be proven. Results showed that application of sludge to different soils could be useful in order to increase crop growth over a 28 d period in the glasshouse. Soil, plant and water quality monitoring, together with the prevention of metals entering the plant, is a prerequisite in order to prevent potential health hazards of sludge application to agricultural land.

Introduction

The application of sewage sludge to agricultural land as an alternative to sacrificial land disposal is not a new concept and has been practised throughout the world for the last few decades. The long-term benefits of the application of sewage sludge to land are, however, frequently limited by potentially harmful elements such as heavy metals and human pathogens. Toxic heavy metals, in particular Cd, Cu, Zn, Ni and Pb are frequently present in high concentrations in sewage sludge (Schmidt, 1997). Heavy metals may be transmitted in the food chain and, because of their high toxicity, present a threat to crop production and animal and human health (Korentejar, 1991). However, through previous research done, it appeared that adding sludge to the soil promotes plant growth significantly more than when commercial fertiliser is added. Christodoulakis and Margaris (1996) showed that plant height increased in maize individuals by 77% in the sludge amended treatment compared to 25% in the case of the commercial fertiliser amendment. Previous research done by Snyman et al. (1998) and Henning et al. (1999) has also demonstrated the short-term beneficial agricultural utilisation of sewage sludge concerning heavy metal contamination risk and the cultivation of maize under South African conditions.

According to the 1997 guidelines (WRC, 1997), the current standards for the unrestricted use of sludge on agricultural soils, cannot be attained within a reasonable framework of affordability and applied technology. Snyman et al. (1999) concluded that none of the wastewater treatment works in S.A. could comply with the Cu, Pb and Zn levels in sludge which is intended for unrestrictive use in terms of the total metal content. Investigations that illustrate the benefits of sewage sludge are extremely important, since there is still a general reluctance among agriculturists to recognise the economic value of the sewage sludge in order to improve the soil organic status without contaminating the environment (Korentejar, 1991). As part of a greater research programme, this study was proposed to assess the effect of sewage sludge on growth and yield of maize (*Zea mays* L.) seedlings under glasshouse conditions. Heavy metal concentrations (total and potentially bioavailable) were monitored in the sludge and soil to characterise plant-soil interactions of the sludge-borne heavy metals on different soil types.

Materials and methods

Collection, treatment and analysis of dewatered sewage sludge

Dewatered sludge samples were collected from two different wastewater treatment plants (WWTP) at the East Rand Water Care Company (ERWAT) representing a low metal sludge (Sludge 1) and a high metal sludge (Sludge 2). The low metal sludge (50:50 anaerobic digested sludge and thickened waste-activated domestic sludge) was collected over a period of 4 h from the beltpress facility, while the high metal sludge (anaerobic digested domestic and industrial sludge) was collected from the drying beds after the sludge was left to dry for two weeks. Analyses were done for

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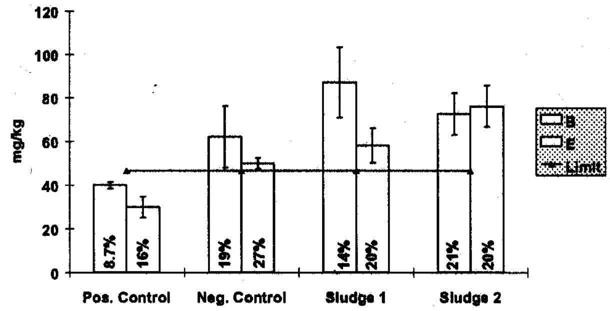
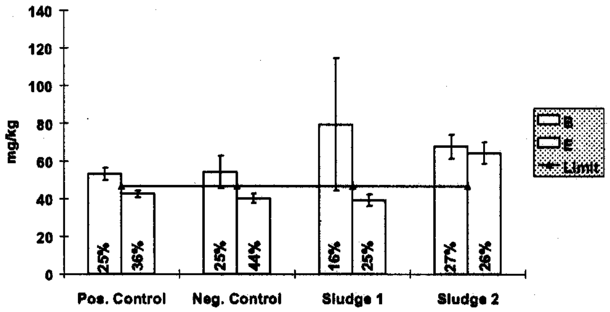
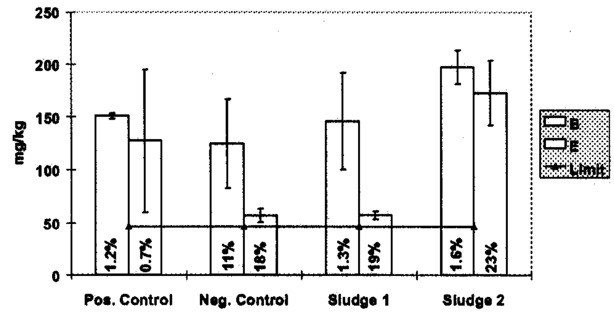
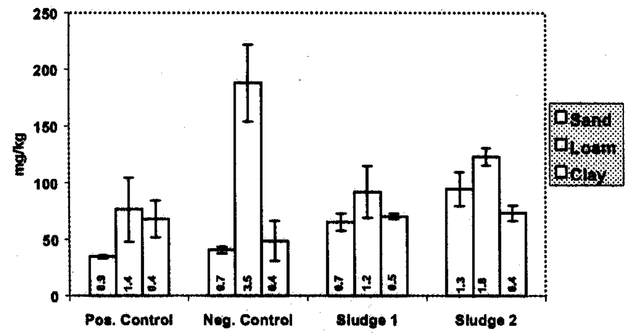
South African sludge guidelines when the metal concentrations in sludges are expressed as total metal content (EPA method 3050), and the guideline limits are interpreted as total metal concentrations. Sludge 2 showed high metal concentrations although the availability of the metals was lower than in the case of Sludge 1. This is due to the metals in activated sludges being complexed in different forms, possibly as organic complexes, compared with those present in anaerobically digested sludges (Smith, 1996).









Soil and plant material analyses

Soil pH. Soil pH is one of the major aspects controlling the availability of heavy metals in soils (Smith, 1996). Background pH values of the clayey, loamy and sandy soil types were approximately 8.8, 5.3 and 7.6 respectively.

Zinc. Zinc is a phytotoxic metal, but it is important as a micronutrient at the appropriate levels (Alloway, 1995). As seen in Figs. 2, 3 and 4, Zn concentrations were above the guideline limits in the soil types due to high background concentrations. The availability of Zn in the Sludge 2 treatment did not increase in the sandy and loamy soils, indicating the stability of the metal complexes. The availability of the Zn was more or less the same in the loamy soil compared to the sandy soil. However, the Zn in the clayey soil was far less available at the beginning of the experiment compared to the sandy and loamy soils due to the adsorption of the Zn to the clay particles which increased the cation exchange capacity (CEC) (Alloway, 1995). The increase in the potential availability of the metals in the two sludge treatments in the clayey soil (Fig. 4) was possibly due to mineralisation of the sludge-borne Zn from an organic form (complexed) to an inorganic form (more available for plant uptake). The difficulty in obtaining a representative sample in order to determine the sludge-borne Zn contribution to the soil was evident. In some cases the predicted theoretical increase of the sludge-borne Zn to the soil was different from the analytical contribution measured. For example, the theoretical increase of Zn in the loamy soil should be 6.4%. However, the increase was measured as 32% (Fig. 3).

Figure 5 shows the total Zn concentrations in maize seedling tissue after 28 d of growth. Normal transfer coefficient (f factor) of Zn in maize is between 1 to 2 (Korentejar, 1991). As seen in Fig. 5 the transfer coefficients for Zn in the sandy and clayey soils were lower than normal (except the Sludge 2 treatment in the sandy soil). The lower pH in the loamy soil caused a higher transfer coefficient in the loamy soil and subsequent higher uptake of Zn in the maize seedling tissue.

Figures from top to bottom:

Figure 2
Zinc concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate potential availability of Zn. [I = Standard deviation (STD)]

Figure 3
Zinc concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate potential availability of Zn. (I = STD)

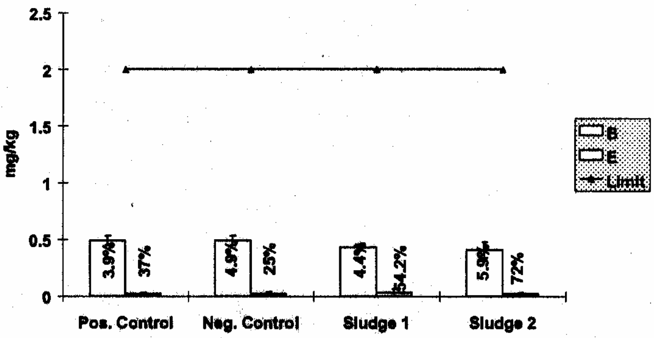
Figure 4
Zinc concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate potential availability of Zn (I = STD)

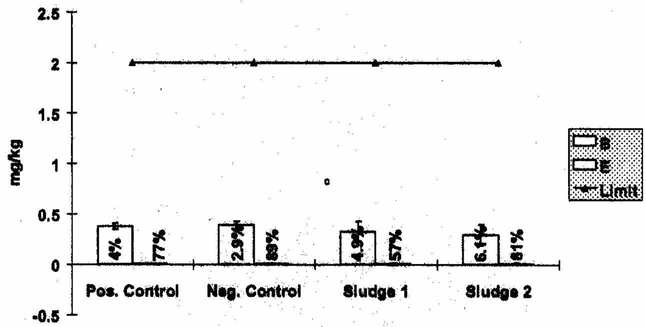
Figure 5
Total Zn concentrations in maize seedling tissue in three soil types. f factor indicated within vertical bars. (I = STD)

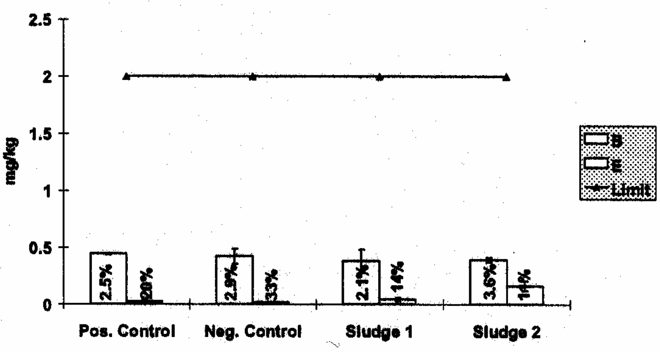
Zn concentrations reached phytotoxic levels (100 to 400 mg·kg^{-1} (Smith, 1996)) in the loamy soil in the negative control and Sludge 2 treatments.

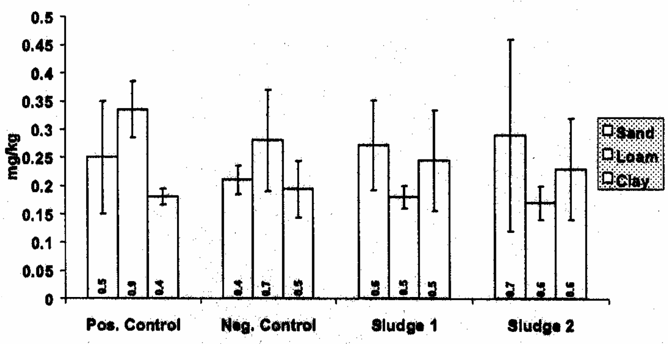
Cadmium. Cadmium is a very mobile and bioavailable metal which may accumulate in crops and humans (Alloway, 1995). Background concentrations of Cd in the sandy, loamy and clayey soils were low and did not exceed guideline limits even after sludge amendment (Figs. 6, 7, 8). There was a significant increase in availability of Cd in the soil types during the experiment, indicating the extent of the mobility of Cd when amended as a sludge-borne metal to soils. Bioavailability of Cd was lower in the clayey soil due to the CEC in clayey soils being much higher, which leads to the formation of stable complexes (Smith, 1996). Cd levels decreased significantly in the soil types over the 28 d due to accumulation of Cd in the plant tissue. The higher extent of increase in availability of Cd in the sludge treatments in the sandy soil, could be due to the mineralisation of Cd from organic complex to an inorganic form for plant uptake. The difficulty in obtaining a representative sample to determine sludge-borne Cd contribution to the soil was evident as previously shown in the sludge-borne Zn contribution. The predicted theoretical increase of the sludge-borne Cd in the sandy soil was different from the analytically contribution measured. The theoretical increase of Cd in the sandy soil would have been 2.6% and 12.5% in the Sludge 1 and Sludge 2 treatments respectively. However, no increase was measured in the sludge treatments. Even lower Cd concentrations were measured in the sludge treatments compared to the negative control (Fig. 6).

Although the bioavailability was lower in the clayey soil, Cd uptake was not lower in seedlings grown in clayey soil (Fig. 9). The plant transfer coefficient (f factor) for Cd was high in all soil types (Fig. 9) since the transfer coefficient for Cd in maize tissue is between 0.01 and 0.05. Soil pH did not play such a major role in the uptake of Cd, since higher uptake was seen in the









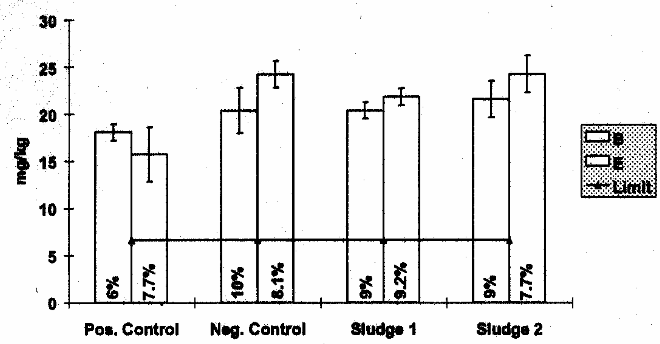
Figures from top to bottom:

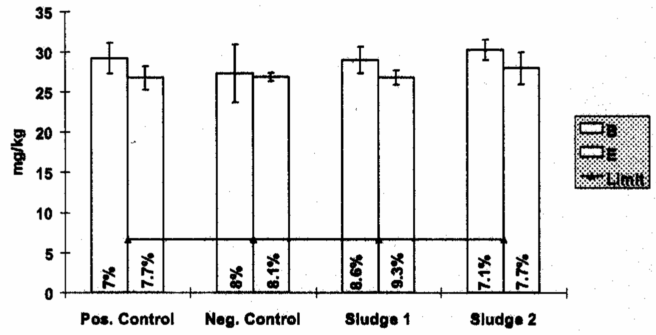
Figure 6
Cadmium concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate potential availability of Cd. (I = STD)

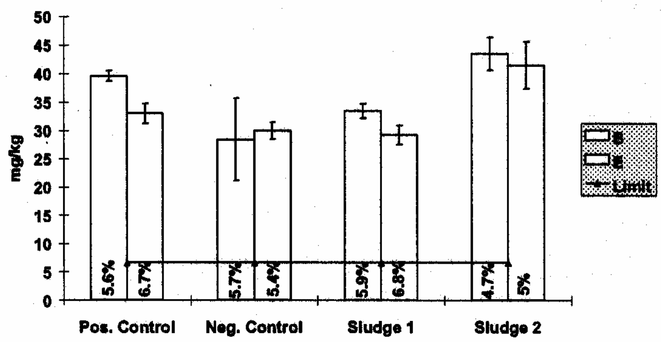
Figure 7
Cadmium concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate potential availability of Cd (I = STD).

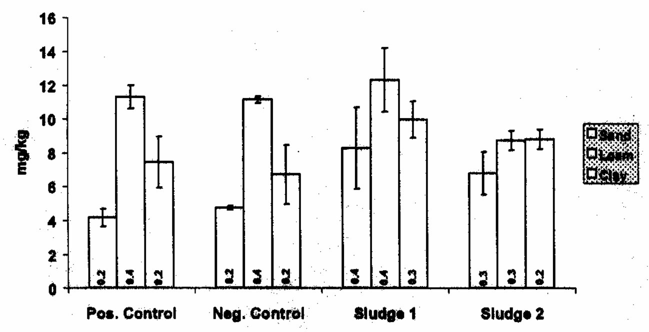
Figure 8
Cadmium concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate potential availability of Cd (I = STD).

Figure 9
Total Cd concentrations in maize seedling tissue in three soil types. f factor indicated within vertical bars. (I = STD).









two sludge treatments in both the clayey and sandy soils when compared to the loamy soil, even though calcareous pH causes metal availability to be lower in soils (Alloway, 1995). The uptake of Cd into the maize seedling tissue also did not reach phytotoxic levels of 5 to 30 mg·kg^{-1} even after a single exposure of the sludge types amended to the soil types at a rate of 24 t·ha^{-1} (Smith, 1996).

Copper. Copper is one of the most important essential elements for plants and animals (Alloway, 1995). Figures 10, 11 and 12 show that guideline limits of 6.6 mg·kg^{-1} for Cu were exceeded in the sandy, loamy and clayey soil types due to high soil background levels. Poor sample homogeneity was evident in the sandy soil since the negative control treatment showed higher Cu concentrations than the positive control treatment. Total Cu concentrations and Cu availability remained constant over the 28 d in the soil types, emphasising the fact that Cu is a relatively immobile element (Alloway, 1995). Stable sample homogeneity was evident in the loamy soil, since the predicted theoretical increase of the sludge-borne Cu in the soil was not significantly different from the analytically contribution measured. For example, the theoretical increase of Cu in the loamy soil in the Sludge 1 and Sludge 2 treatments should have been 2.5% and 13.1% respectively. The increase was measured as 5.8% (Sludge1 treatment) and 10% (Fig. 11).

Cu concentrations in seedling tissue did not reach phytotoxic levels of 20 to 100 mg·kg^{-1} (Smith, 1996) as seen in Fig. 13. The normal plant transfer coefficient (*f* factor) values for Cu in maize are between 0.01 and 0.05 (Korentejar, 1991). Therefore, the transfer coefficient for all the soil types was high. Higher transfer coefficient and uptake of Cu in the seedling tissue occurred in the loamy soil, possibly due to the low soil pH which caused a higher availability of Cu in soils for plant uptake (Alloway, 1995).

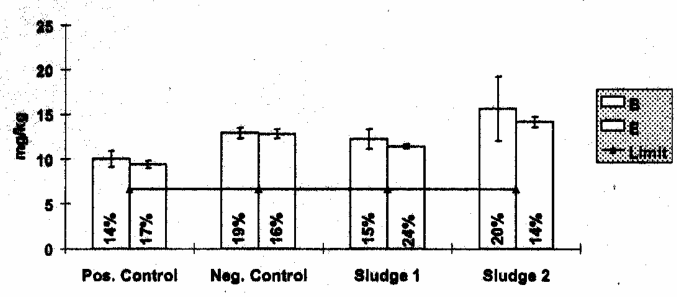
Figures from top to bottom:

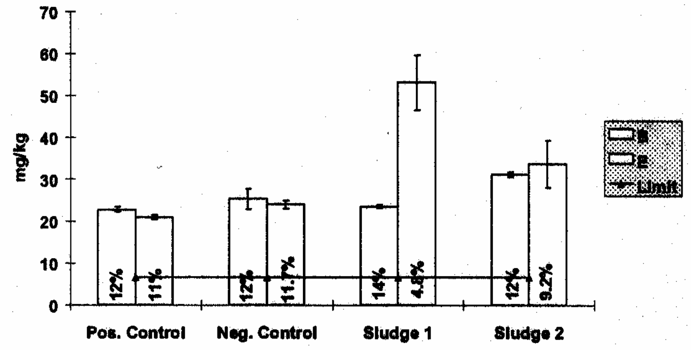
Figure 10
Copper concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate availability of Cu (I = STD).

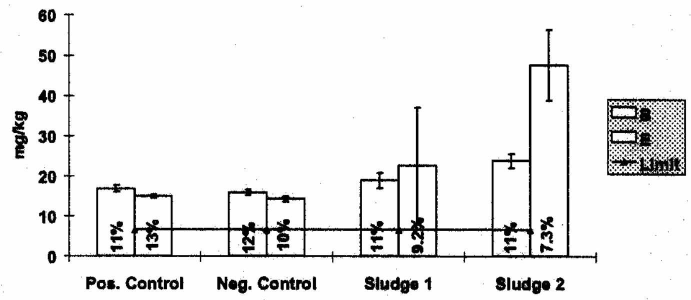
Figure 11
Copper concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate availability of Cu (I = STD).

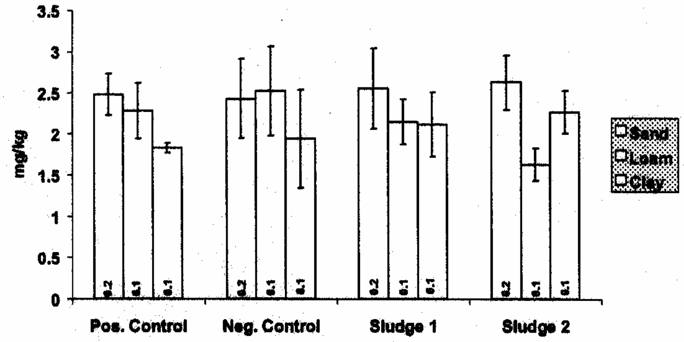
Figure 12
Copper concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate availability of Cu (I = STD)

Figure 13
Total Cu concentrations in maize seedling tissue in three soil types. f factor indicated within vertical bars (I = STD)









Lead. Lead, being a zootoxic metal, needs to be monitored in plant parts used by humans and animals (Alloway, 1995). No significant change could be found for Pb concentrations in the soil types over the 28 d period, indicating the immobility of Pb in soils. Figures 14, 15 and 16 show that the total Pb concentrations in the soil types exceeded guideline limits, mostly due to high background levels in the soils. In both sludge treatments in the clayey soil there was a slight increase in the total Pb concentrations (Fig. 16). The increase in total Pb concentrations in the sludge treatments during the experiments could be due to high root accumulation of Pb. After amendment of the high metal sludge (Sludge 2 treatment) to the sandy soil, the predicted theoretical contribution (44.1%) of the sludge-borne Pb to the total Pb content of the soil was not significantly different from the analytical contribution (36.8%), showing stable sample homogeneity. However, the difficulty in obtaining a representative sample to determine sludge-borne Pb contribution to the soil was also evident. For example, the theoretical increase of Pb in the sandy soil (Sludge 1 treatment) should have been 3.9%. However, the increase was measured as 18.1% (Fig. 14).

Uptake of Pb in seedling tissue was low and did not reach phytotoxic levels of 30 to 300 $mg \cdot kg^{-1}$ (Smith, 1996) as seen in Fig. 17. Normal plant transfer coefficient (*f* factor) values for Pb in maize is between 0.01 and 0.05 (Korentejar, 1991). Therefore, the transfer coefficient for all the soil types was high. Higher transfer coefficient and uptake of Pb in the seedling tissue occurred in the sandy soil, possibly due to the higher availability of Pb for plant uptake over the 28 d in the sandy soil (Alloway, 1995).

Growth

The average shoot length, wet and dry mass per plant are presented in Table 2. There was a definite correlation between wet mass and shoot length of maize seedlings. In the Sludge 1 treatment (all soil

Figures from top to bottom:

Figure 14
Lead concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate availability of Pb (I = STD).

Figure 15
Lead concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate availability of Pb (I = STD).

Figure 16
Lead concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages within vertical bars indicate availability of Pb (I = STD).

Figure 17
Total Pb concentrations in maize seedling tissue in three soil types. f factor indicated within vertical bars (I = STD)

TABLE 2
Average shoot length (cm), wet and dry mass (g) per plant after 28 d of growth

Parameter	Clayey			Loamy			Sandy		
	Shoot length	Wet mass	Dry mass	Shoot length	Wet mass	Dry mass	Shoot length	Wet mass	Dry mass
Positive control	32.2^{*ax}	2.256^{abx}	0.18^{ax}	36.47^{ay}	2.353^{ax}	0.23^{ax}	33.84^{bxy}	2.078^{ax}	0.19^{ax}
Negative control	31.92^{ax}	1.971^{ax}	0.15^{ax}	32.72^{ax}	2.153^{ax}	0.17^{ax}	30.85^{ax}	1.745^{ax}	0.13^{ax}
Sludge1	48.64^{cx}	4.282^{cx}	0.33^{ax}	46.95^{bx}	3.9^{bx}	0.34^{abx}	51.38^{dx}	4.548^{cx}	0.43^{ax}
Sludge2	38^{bx}	2.908^{bx}	0.16^{ax}	35.73^{ax}	2.572^{ax}	0.35^{by}	38.32^{cx}	2.811^{bx}	0.24^{axy}

*Each value is a mean value per plant of 4 replicates of 20 plants. Values within a row not followed by the same letter (x, y or z), or within a column by the same letter (a, b, c or d) are significantly different (P=0.05) according to Duncan's multiple range test.

types), the wet mass and shoot length of the seedlings were significantly higher than the other treatments. A significant increase in shoot length of seedlings grown in the Sludge 2 treatment occurred in the clayey and sandy soils when compared to the two controls. This emphasised the potential short-term beneficial effects of sludges to soils as an organic soil conditioner. This is further emphasised by the fact that no significant difference occurred between the growth of maize seedlings in the different sludge-amended soils. When comparing seedling growth among different soil types no significant differences were seen, although the shoot length of the positive control treatment seedlings grown in the loamy soil showed significant higher values than that of clayey soil.

Soil pH is probably the soil property that affects the uptake of heavy metals the most. It is commonly recommended that soil pH be maintained above 6.5 for sludge-amended soils, although some reports indicated adequate control of metal uptake at pH 6.0 (Sommers et al. 1987). In this experiment the higher uptake of sludge-borne metals in the loamy soil, for example Zn, was due to the lower pH in the loamy soil when compared to the sandy and clayey soils. However, uptake of metals never reached phytotoxic levels in the maize seedling tissue and consequently did not have an effect on plant growth and yield. However, heavy metal behaviour still differs individually and other soil physical properties, like texture, might play an important role in heavy metal behaviour in soils. Today it is commonly accepted that soil type plays an important role in heavy metal bioavailability and, therefore, toxicity (Maclean et al., 1987). This aspect was seen in the soil types used in the experiment, where the metals were often much less available in the clayey soil due to the higher CEC of clayey soils which adsorb metals.

Conclusion

In this glasshouse study, it was found that the current heavy metal guidelines for soil metal concentrations, and sludge metal concentrations, were exceeded in the soil types and sludge types for Pb, Cu and Zn, mostly due to high soil background levels. This emphasises the conservative nature of the current SA guidelines (both soil and sludge metal guidelines) when interpreted as total metal content. Predicted theoretical contribution of the sludge-borne metals, compared to the analytical contribution was different, possibly due to the difficulty of obtaining a representative sample in sludge-amended soils. Long-term experiments still need to be performed on heavy metal accumulation in soil types. However, no phytotoxic effects could be proven, because phytotoxic levels were not exceeded in maize seedling tissue. The sludge affected the yield of the maize seedlings positively compared with control plants. Although the glasshouse experiment showed the beneficial use of sludge on different soil types, the results only represent seedling growth over a 28 d period and cannot conclusively be extrapolated to the field conditions when maize is cultivated at field scale. Therefore a similar large-scale field experiment on maize cultivation will also be completed in the near future, taking into consideration the different environmental parameters. The use of sludge on agricultural soils in SA seriously needs to take into consideration the proper maintenance of soil pH values, for example through application of liming materials. In addition, adequate soil, plant and water quality monitoring procedures are required in order to prevent potential health hazards of sludge application on agricultural land. The possible revision of the SA heavy metal guidelines, when interpreted as total metal content for the unrestrictive use of sludge needs to possibly take into consideration the environmental conditions, crop planted, soil type and sludge type. This might lead to the unrestricted use of sludge on agricultural land in future in SA, causing a decrease in technological costs for wastewater treatment plants (and subsequent financial profit) to eliminate heavy metals in sludges.

Acknowledgements

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Appendix 4

THE CULTIVATION OF MAIZE (*ZEA MAYS* L.) ON HIGH SEWAGE SLUDGE DOSAGES AT FIELD SCALE (1)

4.1 SUMMARY

Maize forms part the major core of crops cultivated and exported from South Africa. A field experiment was performed in a major maize cultivating area in S.A. to assess the effect of sewage sludge on growth and yield of maize (*Zea mays* L.) under environmental conditions. Two sludge treatments were applied at higher rates to the soil at 12.5 t ha^{-1} and 25 t ha^{-1}, instead of the recommended 8 t ha^{-1}. Yield differences showed the average amount of ears $plant^{-1}$ in the 12.5 t ha^{-1} to be significantly higher than the other treatments, although no differences occurred between treatments for dry mass and wet mass (t ha^{-1}) of the ears. Heavy metals that exceeded guideline limits were due to high soil background levels. No excessive build-up of heavy metals occurred in the soil even though high sludge dosages were applied. Furthermore, no pathogenic indicator organisms could be found in the kernels and heavy metal uptake into the plant parts was not in excess.

4.2 INTRODUCTION

The impact of sewage sludge to the environment in agricultural practices within the South African context has not been investigated extensively. For example, the methodologies are not standardised to allow for site specific, soil specific and crop specific determination of the impact of sewage sludge. Henning *et al.* (2001) proved the short-term beneficial use of sewage sludge under greenhouse conditions and this was also discussed in the previous chapter. The greenhouse experiment showed higher transfer coefficients when metals were taken up from different soil types. However, field experiments are much more practical for farmers applying sludge to their lands. These experiments are performed on a larger scale, on a larger volume of soil, for a full growing season under environmental conditions, and therefore metal behaviour might be different and needs to be investigated.

The success of maize (*Zea mays* L.) cultivation under field conditions is influenced by other varying factors such as moisture, nutrient supply (especially N), weather conditions, disease and the length of the growing season (Klingman, 1957). Nitrogen loading has been used previously to determine the sewage sludge application rate for maize cultivation (Hemphill *et* *al.,* 1982). The application rates, along with other factors such as health risk and contamination aspects, should be taken into account when calculating ecologically sustainable sludge application rates at a particular disposal site (Palmer, 1993). However, as sludge has a lower macronutrient (N, P and K) content than common inorganic fertilizers, the cost per unit weight of the element is very high in comparison to inorganic fertilizers (Vesilind, 1980). Sludge can however, still be used as a fertilizer source in areas near major sewage treatment plants since sewage sludge contains appreciable amounts of N and P and has a significant inorganic fertilizer replacement value for these major plant nutrients (Hall, 1985; Coker & Carlton-Smith, 1986).

However, when sludge is applied at the appropriate nitrogen rate for maize cultivation, potentially toxic elements like heavy metals could build up excessively in the soil or be taken up by plant parts. The concentrations of the heavy metals in sewage sludge originating from different sources can vary from plant to plant, mainly due to different inputs to the sewer system (Critchley & Agg, 1986).

This study was done to assess the effect of sewage sludge on growth and yield of maize under environmental conditions in the East Rand, South Africa. The nutrient and heavy metal concentrations as well as human pathogens were monitored in the soil and plant parts during the field trial.

4.3 MATERIALS AND METHODS

Collection, treatment and analysis of dewatered sewage sludge

Anaerobically digested sludge samples, dewatered to ± 20% m/m solids were collected at a Water Care Works of ERWAT, North-East Gauteng, South Africa. Analyses were done for moisture, nutrients and heavy metals (EPA3050 method – Sims *et al*., 1991).

Experimental layout

The experimental site was at Hartbeestfontein farm in the Bapsfontein area, Gauteng, South Africa. This area is known as a major maize producing area in Gauteng. Plots (7 m x 7 m) were arranged in a randomised block design with four replications and four treatments as stated in Table 4.1. Lime was applied at 2 t ha^{-1} to the loam soil to raise the pH to the appropriate levels for maize cultivation. Sewage sludge and inorganic fertilizer were added to the soil one week before the desired planting date at the appropriate rates stated in Table

4.1. Maize kernels were planted one week after sludge application using standard planting techniques.

Table 4.1 Treatments used in the randomised block design for the field study to determine the effect of sewage sludge applied to a loam soil

Treatments	Soil application
Exp. 1	Sewage sludge applied to soil at 12.5 t_{dry} ha^{-1} (61.25 kg dewatered sludge 49 m^{-2})
Exp. 2	Sewage sludge applied to soil at 25 t_{dry} ha^{-1} (122.5 kg dewatered sludge 49 m^{-2})
Positive Control	Inorganic fertilizer (2:3:2: (28)) applied to soil at 600 kg ha^{-1} (2.94 kg 49 m^{-2})
Negative Control	No applications to soil

Soil analyses

Soil samples were collected to a depth of 15 cm from each of the plots at two different times during the experiment; at the start of the experiment after sludge application and after four months of growth at the beginning of ear-formation. Samples were analysed for the following aspects using the methods as described by the reference stated:

Nutrient content (%N, P, K, Ca, Mg, pH and Resistance)

Heavy metals (Mn, Zn, Cu and Pb) (EPA3050; Sims *et al.,* 1991). Cd was not tested due to low levels

Pathogenic indicator organisms: Faecal coliform, *Salmonella* and *Ascaris* ova (APHA-AWWA-WPCF, 1995)

Plant material analyses

Plant material samples were collected from each of the plots. Leaf and stalk samples were randomly collected at the pre-tasseling stage of growth, while ear samples were harvested at the mature stage of ear-formation. All plant material samples were analysed for heavy metals (Zn, Pb, Cu and Mn). The maize kernel samples were also analysed for the detection of pathogenic indicator organisms. Yield differences between treatments were measured in terms of the average amount of ears per plant, wet mass of ears and percentage dry weight of the ears. The determination of the transfer coefficient was also done ("Plant material analyses", Appendix 3).

4.4 RESULTS AND DISCUSSION

Analysis of dewatered sewage sludge

The dewatered sludge had a moisture content of 78.8%, solids of 21.2% and a pH of 7.06. Nutrient levels in the sludge on a dry basis were found to be 3.11% m/m P, 4.17% m/m N, 0.66% m/m K, 15 800 mg kg^{-1} Ca and 5 630 mg kg^{-1} Mg. Main heavy metals were analysed for and detected as stated in Table 4.2.

Table 4.2 Heavy metal content in the dewatered sludge compared to guidelines (WRC, 1997)

Parameter	Guidelines (WRC, 1997)	Sludge concentration
Mn (mg kg^{-1})	N/s	988
Zn (mg kg^{-1})	353.5	1 100
Pb (mg kg^{-1})	50.5	141
Cu (mg kg^{-1})	50.5	270

N/s: Not specified in the guide: Permissible utilisation and disposal of sewage sludge (WRC, 1997).

Although no limit is specified for Mn in the guidelines (WRC, 1997), the mean concentration of Mn in sewage sludge is 376 mg kg^{-1} (Smith, 1996), which shows that Mn values were high in the sludge (Table 4.2). Other heavy metal concentrations (Zn, Pb and Cu) exceeded the guideline values, when interpreted as total metal concentration. However, the sludge still contained relatively low heavy metal concentrations compared to other sewage plants in South Africa (Snyman *et al.,* 1999). This is due to a programme where the wastewater treatment plant management, local authorities and industries strive to prevent heavy metals from entering the plant and thus ultimately entering the food-chain through the application of sludge to agricultural land.

Soil analyses

Table 4.3 shows the chemical (pH and resistance) and nutrient analysis data at the beginning of the experiment and at ear-formation. The pH was at the recommended 6.5 (Korentajer, 1991) at the beginning of the experiment after sludge application, which is important since the mobility and availability of heavy metals are greatly increased at soil pH values below 6.5 (Alloway & Jackson, 1991; WRC, 1997). Soil pH values decreased during the experiment and were below 6.5 at the beginning of ear-formation. It was evident that the application of sludge did contribute to the higher rate of decrease in soil pH during the experiment, when compared to the control treatments (Table 4.3).

Table 4.3 Chemical characteristics of the soil at the beginning of the experiment after sludge was applied (B) and at ear-formation (E) of maize

Parameter	+ Control		- Control		Exp. 1		Exp. 2	
	B	E	B	E	B	E	B	E
PH	6.53	6.04	6.7	6.29	7.0	5.53	6.89	5.63
Resistance (Ω)	360	1 800	460	1 800	440	1 400	480	1 500
P (mg kg^{-1})	70.4	171.5	47.7	94.5	68.7	171.5	61.9	138.9
Ca (mg kg^{-1})	582	718	582	606	661	631	613	652
Mg (mg kg^{-1})	335	357	335	303	378	348	353	339
K (mg kg^{-1})	124	218	124	46	158	79	169	65
Total % N	0.213	0.27	0.16	0.25	0.16	0.304	0.231	0.28

As seen in Table 4.4, all the heavy metals analysed for (Zn, Pb and Cu), exceeded the guidelines and this was due to very high background limits in the soil, which is indicative of the conservative nature of the South African guidelines. This also emphasises the lack of research performed on the agricultural recycling of sewage sludge and the influence of heavy metals in sewage sludge on agricultural soils, under South African conditions. Mn levels exceeded 10 000 mg kg^{-1}.

Table 4.4 Total heavy metal content in the soil at the beginning of ear-formation of maize

Parameter	Limit*	+ Control	- Control	Exp. 1	Exp. 2
Zn (mg kg^{-1})	46.5	121.6	131.1	131.7	129.8
Pb (mg kg^{-1})	6.6	31.5	33.9	35.8	33.8
Cu (mg kg^{-1})	6.6	70.5	78.6	79	74.8

*Limit: Maximum soil metal content according to the “Guide: Permissible utilisation and disposal of sewage sludge" (WRC, 1997).

Table 4.5 shows the average values for the pathogenic indicator organism detected in the soil at the beginning of ear-formation and in the kernels.

Table 4.5 Pathogenic indicator organisms detected in the soil at the beginning of ear-formation and in the kernels of maize

Sample	+ Control	- Control	Exp. 1	Exp. 2
Soil:				
Total plate count CFU 100 ml^{-1}	293 000	160 000	220 000	193 000
Faecal coliform CFU 100 ml^{-1}	333	733	333	633
Salmonella	Negative	Negative	Negative	Negative
Ascaris ova	1 non-viable	1 non-viable	2 non-viable	4 non-viable
Kernels:				
Faecal coliform CFU 100 ml^{-1}	0	0	0	0
Salmonella	Negative	Negative	Negative	Negative
Ascaris ova	Negative	Negative	Negative	Negative

Although pathogenic indicator organisms like faecal coliforms were detected, no pathogenic indicator organisms were detected when the kernels were analysed. The path for sludge-borne pathogens to move through the plant root system, into the stem, from the stem to the leaves and from there to the kernels, is difficult and as shown, any uptake of these organisms is unexpected. Therefore the field experiment in Appendix 7 does not include the detection of sludge-borne pathogens in oat plant parts.

Plant material analyses

Manganese plays an essential role as a micronutrient for microorganisms and higher plants (Alloway, 1995). Mn background concentrations in the soil were excessively high and therefore taken up into the plant parts at high concentrations. Figure 4.1 shows the Mn concentrations of the different treatments in the different plant parts. The leaf concentrations were below the typical toxic concentration range of 400-1000 mg kg^{-1} (Smith, 1996). Mn accumulated more in the maize kernels compared to other plant parts. The higher Mn concentrations in the kernels grown on the sludge-treated soil at 12.5 t ha^{-1}, were possibly due to the high concentration of Mn in the sludge (988 mg kg^{-1}) being more available for plant uptake. The transfer coefficient of Mn is not included since soil concentrations of Mn were too high to be determined.

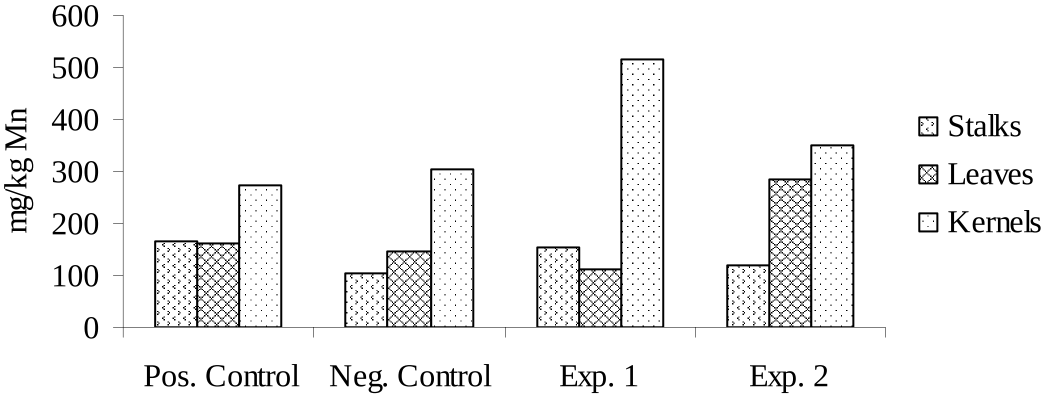

Figure 4.1. Mn concentrations in the different plant parts of maize during the field trial.

Zinc concentrations in the leaves, stalks and kernels were below the toxic ranges of 100-400 mg kg^{-1} (Smith, 1996), as was also found by Jarausch-Wehrheim *et al.* (1999) in earlier field studies concerning the translocation of sludge-borne Zn in field-grown maize. The higher concentration of Zn in the kernels shows increased accumulation of Zn from the leaves (sources of nutrients) to the kernels (nutrient storage sinks) (Gardner *et al.,* 1993) during the growing season (Fig. 4.2). The uptake of Zn in the different plant parts between treatments did not differ much, although the Negative Control treatment showed lower uptake of Zn in the stalks compared to the other treatments. The normal transfer coefficient of Zn is between 1 and 2 (Smith, 1996), and was lower in the plant parts. The same could be seen for Pb.

Figure 4.3 shows that Pb was mostly taken up into the leaf tissue, although the Pb concentrations did not exceed the toxic ranges for plants of 30-300 mg kg^{-1} (Smith, 1996). Low uptake was found in the kernels of all treatments. The transfer coefficient was lower than the normal ranges of 0.01 to 0.05 (Smith, 1996) in the kernels (all treatments), and in the Negative Control treatment and Exp. 2 treatment in the stalks, emphasising the immobility of lead. However, normal values and slightly higher (Negative Control and Exp. 1 treatment) transfer coefficients were found in the leaves. The low uptake and transfer coefficient of Pb in this field experiment have been found in most other field experiments with sewage sludge (Chlopecka, 1996), and this phenomenon is due to the low solubility, mobility and availability of Pb to crop plants (Chumbley & Unwin, 1982). Therefore Pb does not pose a major threat to crop plants.

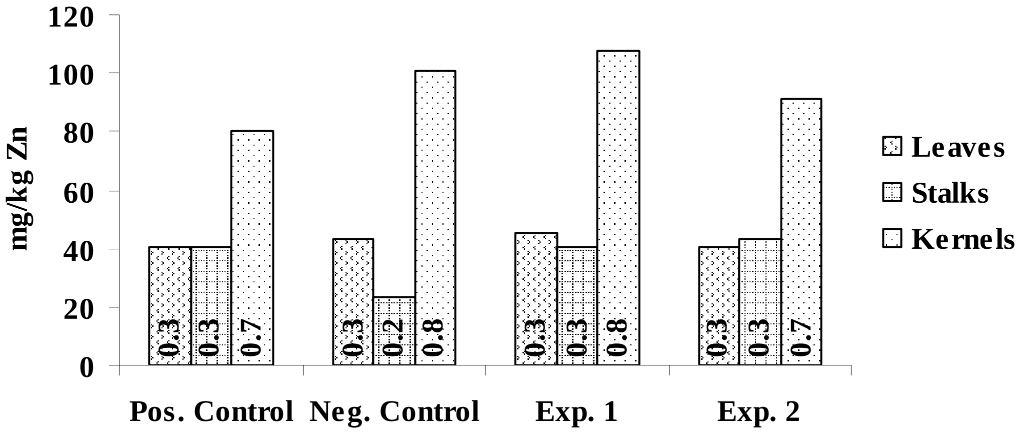

Figure 4.2. Zn concentrations in the different plant parts of maize during the field trial. *f* factor indicated inside vertical bars.

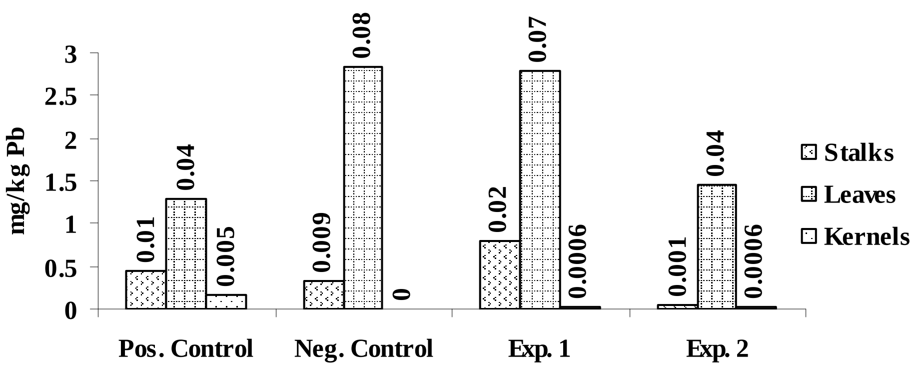

Figure 4.3. Pb concentrations in different plant parts of maize during the field trial. *f* factor indicated above vertical bars.

Previous research has shown that crops grown on sludge-amended soils exhibited significant increases in the uptake of Cu in plant tissue (Chlopecka, 1996). Cu concentrations were all below the toxic ranges of 20-100 mg kg^{-1} specified for plant leaves (Smith, 1996) as seen in Figure 4.4. The uptake of Cu varied between treatments and the sludge treatments showed higher uptake of Cu in the kernels and stalks compared to the control treatments. This could be due to the higher availability of the Cu in the sludge when applied to agricultural soil.

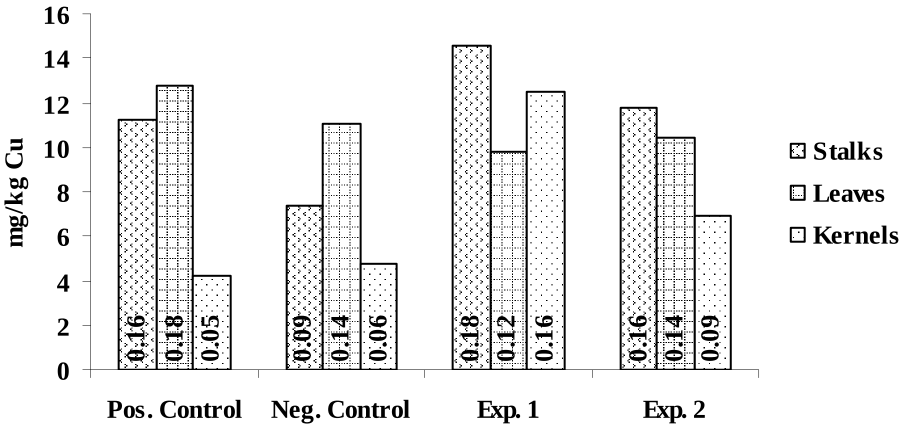

Figure 4.4. Cu concentrations in different plant parts of maize during the field trial. *f* factor indicated inside vertical bars.

The normal ranges for the *f* factor for Cu is between 0.01 and 0.05. In all the plant parts, in all the treatments, the transfer coefficient was higher than normal. This could be due to high soil background concentrations of Cu.

Yield differences were calculated and are shown in Table 4.6. Statistical analyses of dry mass and wet mass (t ha^{-1}) of ears did not show any significant differences between treatments, due to the varying environmental factors that might influence soil conditions during the different growth stages of the plant under field conditions. Another consideration is that the results are based on a single application of sewage sludge, and assumptions concerning aspects like heavy metal uptake and yield can only be made over the long-term. However, sludge-amended plots still compared well with the Positive Control (inorganic fertiliser) for these parameters, and 12.5 t ha^{-1} (Exp. 1) showed higher values than the Positive Control (Table 4.6). The average amount of ears $plant^{-1}$ showed a significant difference between the 12.5 t ha^{-1} treatment and the other treatments as shown in Table 4.6.

Table 4.6. The effect of sewage on yield of maize under field conditions in loam soil

Treatment	+ Control	- Control	Exp. 1	Exp. 2
Ears plant^{-1}	1.93*^{a}	1.85^{a}	2.21^{b}	1.93^{a}
Dry mass (t ha^{-1})	5.05^{a}	5.3^{a}	5.28^{a}	4.9^{a}
Wet mass (t ha^{-1})	9.9^{a}	10.26^{a}	10.2^{a}	9.46^{a}

* Each value is a mean value per plant of 5 replicates of 16 plants. Values within a row not followed by the same letter are significantly different (P = 0.05) according to Duncan's multiple range test.

This field experiment, compared to a glasshouse experiment done by Henning *et al.* (2001), showed similar results comparing the guidelines (interpreted as total metal content) to the total metal concentrations in the sewage sludge and soil. Although heavy metal uptake did not reach phytotoxic levels during the field study, heavy metal behaviour in the soil after sludge application, and the uptake of heavy metals into the plant parts cannot be compared to greenhouse conditions. The transfer coefficients of Zn and Pb in the different maize parts were lower than the normal range and emphasised the differences between field and greenhouse experiments. The transfer coefficients of Cu and certain treatments in Pb were higher than the normal ranges. However, these transfer coefficients were still lower compared to the seedling greenhouse experiment. In the field experiment a much larger volume of soil is under the influence of several environmental parameters, which causes metal availability and behaviour of metals to differ during the growing season of crops. The accumulations of potentially toxic elements (PTEs) in plant tissues can be increased 1.5-5 fold under greenhouse conditions compared with field studies with the same soil, sludge and crop (Logan & Chaney, 1983). Greater accumulations occur for a number of reasons including the confinement of plant roots to the small volume of treated soil in pots and the abnormal watering pattern and relative humidity in greenhouse pot studies (Logan & Chaney, 1983). Therefore, field experiments provide a more practical approach to sludge-application for farmers, while greenhouse experiments give a better understanding of heavy metal behaviour in soils.

4.5 CONCLUSIONS

Dewatered sewage sludge is used as a fertilizer and a soil conditioner by farmers in the East Rand, South Africa. Sludge can be used as a supplementary long-term nutrient releasing substance when using inorganic fertilizer, by adding extra nutrients to the soil and improving soil organic status. Yet, due to high microbial numbers and heavy metals, there is still a reluctance to acknowledge the economic value of sludge. In this study the effect of high dosages of sewage sludge applied to loam soil on the growth and yield of maize was monitored. The sludge used in this study was relatively low in heavy metals. It was found that heavy metal uptake into the plant parts was not phytotoxic. The transfer coefficients of Zn and Pb were also lower than the normal ranges. Accumulation of heavy metals in the soil did not occur, although guideline limits were exceeded due to high background values in the soil. No pathogenic indicator organisms were detected in the plant tissue.

The impact of the short-term use of sewage sludge on agricultural soils under field conditions was small, therefore metal accumulation in soils needs to be monitored over the longer term. The field experiment, compared to a similar greenhouse experiment, emphasised the different approaches used when different experiments are conducted on agricultural utilisation of sewage sludge.

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Appendix 5

THE CULTIVATION OF MAIZE (*ZEA MAYS* L.) ON HIGH SEWAGE SLUDGE DOSAGES AT FIELD SCALE (2)

5.1 MATERIALS AND METHODS

A second maize field trial was conducted with sludge application rates of 4 and 8 t ha^{-1} for "Sludge 1" and "Sludge 2" respectively. The trial layout was the same as in Appendix 4.

5.2 RESULTS AND DISCUSSION

Heavy metals in soils and plants

Figures 5.1, 5.2, 5.3 and 5.4 present the concentrations of Zn, Pb, Cu and Cd respectively in the soil before and after application of the sludge. Zn levels slightly exceeded the guideline limits but in the Sludge 2 treatment exceeded the limits by a factor of two to three. In all the treatments the extractable (EPA 3050) levels decreased from the beginning to the end of the trial. The available Zn remained fairly constant except for the 8 t ha^{-1} treatment in which it decreased markedly.

Both Pb and Cu substantially exceeded the limits but there was little difference between the control and the sludge treatments. Again the Cu levels decreased from the beginning to the end of the trial. The availability of Pb and Cu remained relatively constant as was the case with Zn. The Cd levels were substantially below the guideline limits. The availability of Cd decreased marginally from the beginning to the end of the trial but this could possibly be linked to the slight increase in the EPA 3050 extracted Cd throughout all the treatments at the end of the trial.

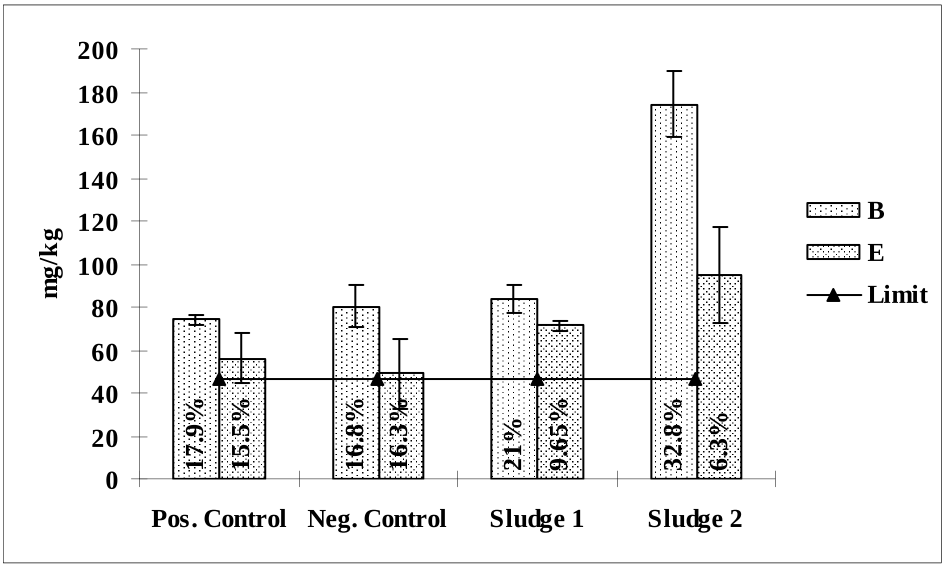

Figure 5.1. Total Zn concentrations in the soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD).

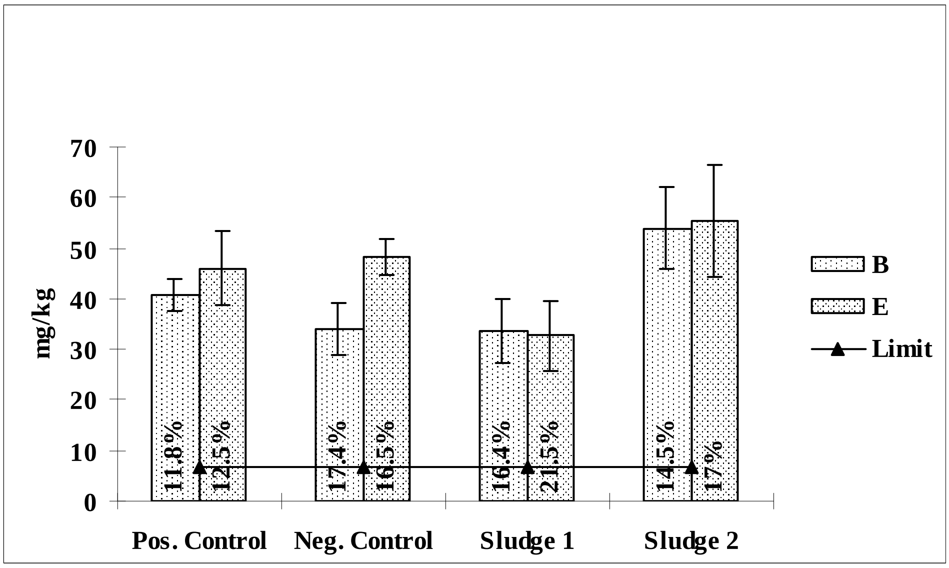

Figure 5.2. Total Pb concentrations in the soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Pb. (I = STD).

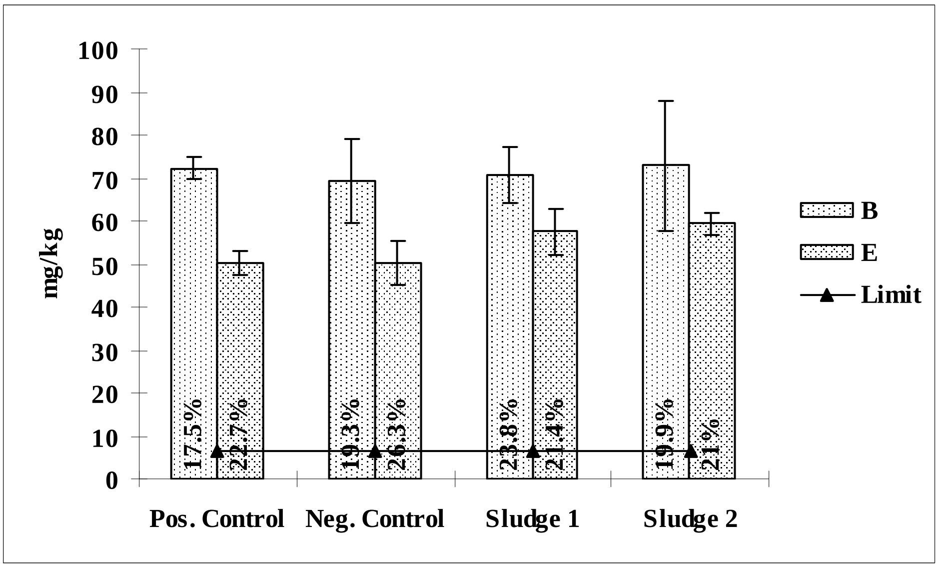

Figure 5.3. Total Cu concentrations in the soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cu. (I = STD).

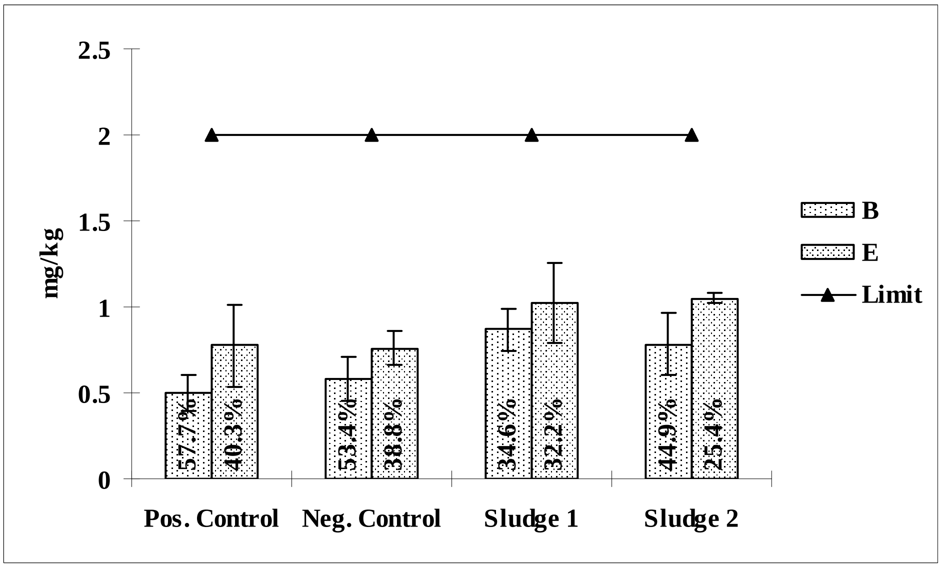

Figure 5.4. Total Cd concentrations in the soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

Figures 5.5, 5.6, 5.7 and 5.8 present the Zn, Pb, Cu and Cd concentrations respectively in the leaves and the seeds of the maize. Both Zn and Cu exhibited very little difference between the control and the sludge treatments with higher levels in the leaves than in the seeds. Pb exhibited a distinct increase in leaf levels from the control to the sludge treatments although the seed levels were very similar. The higher Pb levels in the plants are a cause for concern.

From Figure 5.8 it appears that very little Cd was taken up by the leaves and this could account for the variability in the data. It would also appear though that the seeds accumulated higher levels than the leaves in all but the highest sludge application rate. These levels are however still very low.

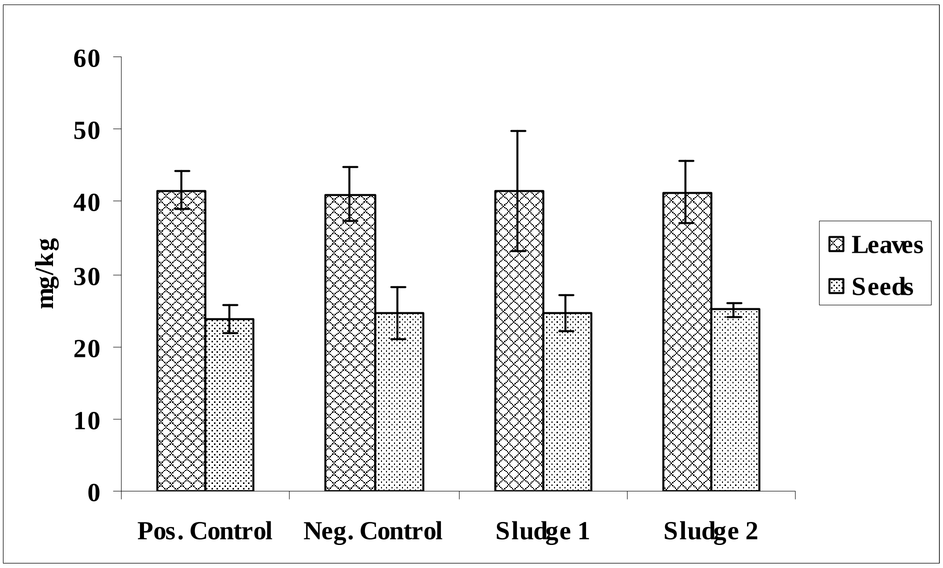

Figure 5.5. Zn concentrations in the different plant parts of maize during the field trial.

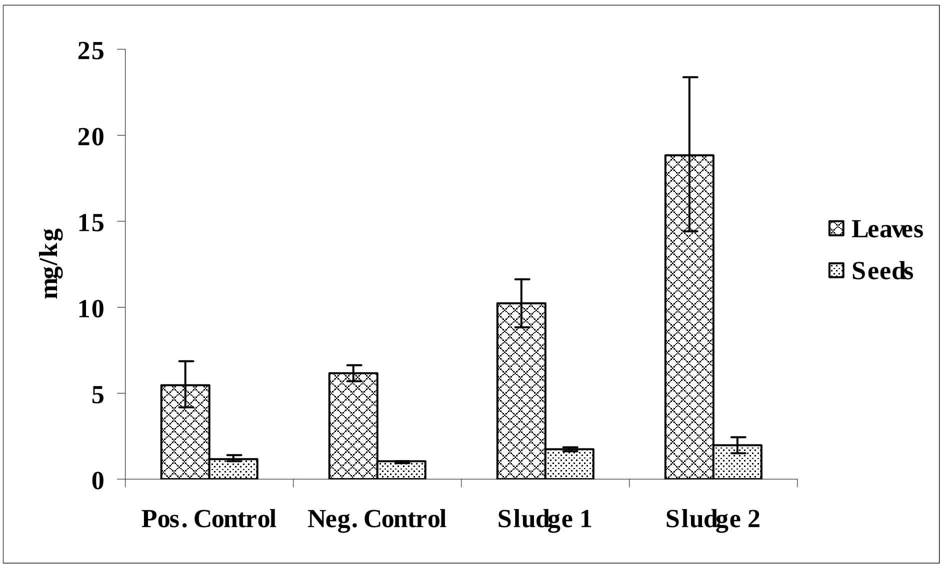

Figure 5.6. Pb concentrations in the different plant parts of maize during the field trial.

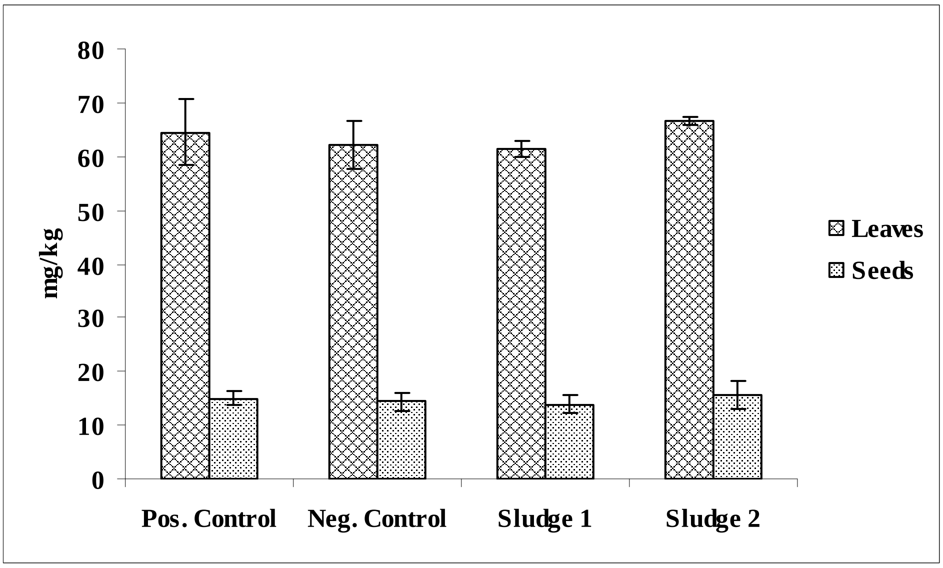

Figure 5.7. Cu concentrations in the different plant parts of maize during the field trial.

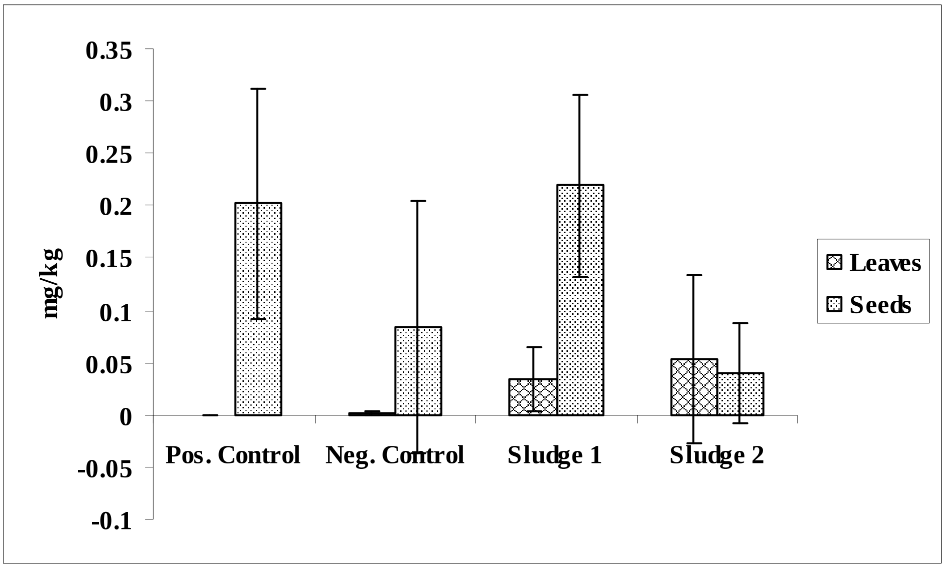

Figure 5.8. Cd concentrations in the different plant parts of maize during the field trial.

Yield

No yield data is available for the second maize field trial.

5.3 CONCLUSIONS

See Appendix 4.

REFERENCE

WRC. 1997. Guide: Permissible Utilisation and Disposal of Sewage Sludge. 1st ed. Water Research Commission, Pretoria.

Appendix 6

PLANT-SOIL INTERACTIONS OF SLUDGE-BORNE HEAVY METALS AND THE EFFECT ON OATS (*AVENA SATIVA* L.) SEEDLING GROWTH

6.1 SUMMARY

Responses of different crops cultivated on soils where sewage sludge is applied is relatively unknown under South African conditions. Oats is a major winter crop in South Africa with a different carbohydrate metabolism compared to maize. Therefore, a greenhouse experiment was done to assess the behaviour of sludge-borne heavy metals (Pb, Cu, Cd and Zn) applied through two sludges (low metal and high metal) to different soil types (clayey, loamy and sandy). Correlations on the uptake of the metals into oats seedling tissue were done and the effect on the yield of the oat plants was determined. The seedlings grown in soil treated with low metal sludge, showed significant increases in wet and dry mass when compared to control treatments. Poor sample homogenisation was evident, since the predicted metal loading of soils could not be proven analytically. Guideline limits were exceeded for Pb, Cu and Zn in the soil types, due to high background concentrations before sludge application. The lower soil pH in the loamy soil caused an increased accumulation of heavy metals in the seedling tissue. Phytotoxic levels did not occur in the plant tissue. Results showed the potential of the application of these sludges to different soils to increase growth of oats seedlings.

6.2 INTRODUCTION

Agricultural crops differ in their physiological functions and therefore also differ in certain responses. For example, nutrient requirements (N, P and K) for winter crops like oats (*Avena sativa* L.) are less than those for summer cultivated crops like maize (*Zea mays* L.) (Welch, 1995), and oats use a different carbohydrate metabolism (C3-photosynthetic pathway) compared to maize (C4 photosynthetic pathway) (Salisbury & Ross, 1991). Farmers in South Africa cultivate oats during the winter (May to September) mainly as a fodder for livestock. Oats is one of the most versatile of the cereals in regard to suitable soil type. Research on the effect of sewage sludge on growth of oats is limited, although similar experiments on other winter cereal crops such as barley (*Hordeum vulgare* L.) and wheat (*Triticum aestivum* L.) have been done (Simeoni *et al.,* 1984; Unger & Fuller, 1985). The research indicates that sewage sludge is beneficial when used as an organic fertilizer.

This study was done to assess the effect of sewage sludge on growth and yield of oats seedlings under greenhouse conditions. The uptake of four heavy metals (Pb, Cu, Cd and Zn) was monitored in the oats leaf tissue, and the heavy metal concentrations (total and available) were monitored in the sludge and soil. These values were used to draw a correlation between soil-metal concentrations and their uptake by crops.

6.3 MATERIAL AND METHODS

Collection, treatment and analysis of dewatered sewage sludge

Two sludge types [low metal sludge (Sludge 1) and high metal sludge (Sludge 2)] were collected from two different Wastewater Treatment Plants (WWTP) at the East Rand Water Care Company (ERWAT) as described in Appendix 3. The sludges were analysed by the ISCW, ARC, Pretoria, for the same aspects and using the same methods as described in Appendix 3.

Experimental layout

The experimental layout and treatments used for the greenhouse study were as previously described under "Experimental layout", Appendix 3.

Soil analyses

The collection of the soils, treatment, and analyses at the beginning and end of the experiment on the soils were done as described under "Soil analysis", Appendix 3. Nine oats (*Avena sativa* L., cv. Sensako 001) seeds were planted per pot.

Plant material analyses

Oats seedlings were allowed to grow for 28 d in the greenhouse before harvesting. The harvesting of the seedlings, determination of yield, and analyses concerning heavy metal uptake and determination of the transfer coefficient were done as described under "Plant material analyses", Appendix 3.

6.4 RESULTS AND DISCUSSION

Analysis of dewatered sewage sludge

The same sludge samples were used as in the greenhouse experiment performed in Appendix 3 on maize seedlings. Consequently, the results based on dewatered sewage sludge, which is illustrated in Figure 3.1 (Appendix 3) and the discussion thereof, are presented under "Analysis of dewatered sewage sludge", Appendix 3.

Soil and plant material analyses

Soil pH

The pH values of the soils are listed in Appendix 3.

Zinc

As seen in Figure 6.1, Zn concentrations were inside guideline limits in the sandy soil. In previous experiments done on sandy soil (collected during the summer) (Henning *et al.*, 2001) the guideline limits were exceeded. However, due to varying soil-forming factors like parent material, climate and vegetation affecting cation concentrations in soils, the background concentrations in the same soil, collected after the winter, were lower. However, in the loamy and clayey soil, concentrations approached guideline limits for Zn, due to high soil background concentrations (Figs 6.2 and 6.3). The increased Zn concentration over the 28 d period in the sandy soil is possibly due to poor sample homogenisation and/or non-representative sampling. An increase occurred in the potential availability of the Zn in the Sludge 2 treatment for all soil types (Figs 6.1, 6.2 and 6.3), possibly due to mineralisation of the sludge-borne Zn from an organic form (complexed) to an inorganic form (more soluble for plant uptake). In some cases the predicted theoretical increase of the sludge-borne Zn to the soil was different from the analytically measured contribution. For example, the theoretical increase of sludge-borne Zn (Sludge 2) in the sandy soil should be 76%. However, the increase was measured as 24.6% (Fig. 6.1). Therefore, the difficulty in obtaining a representative sample to determine sludge-borne Zn contribution to the soil was evident, and in the case of the other heavy metals (Pb, Cu and Cd) discussed in the following sections, the same difficulty occurred.

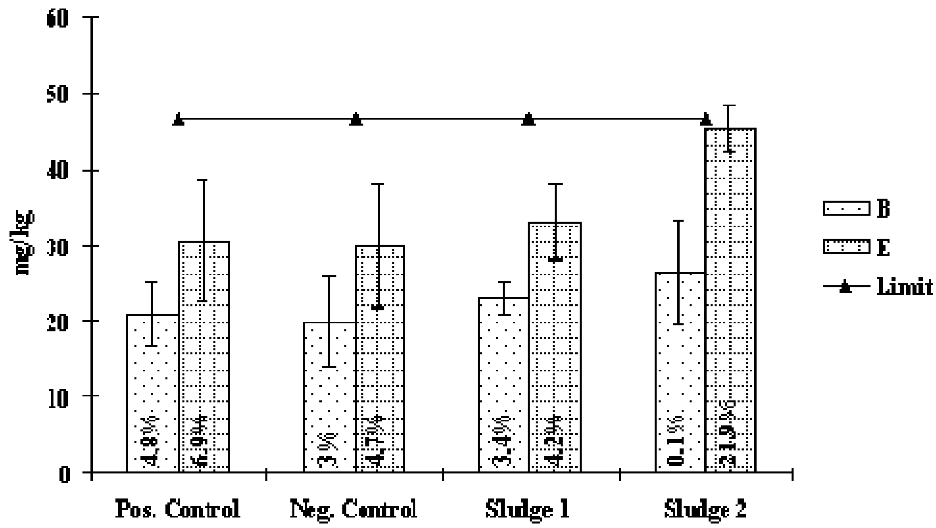

Figure 6.1. Total Zn concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD).

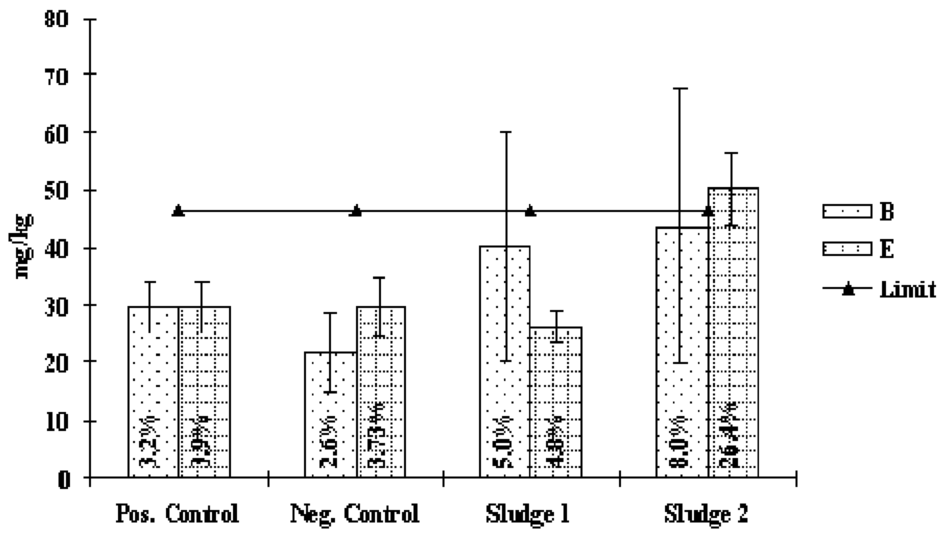

Figure 6.2. Total Zn concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD).

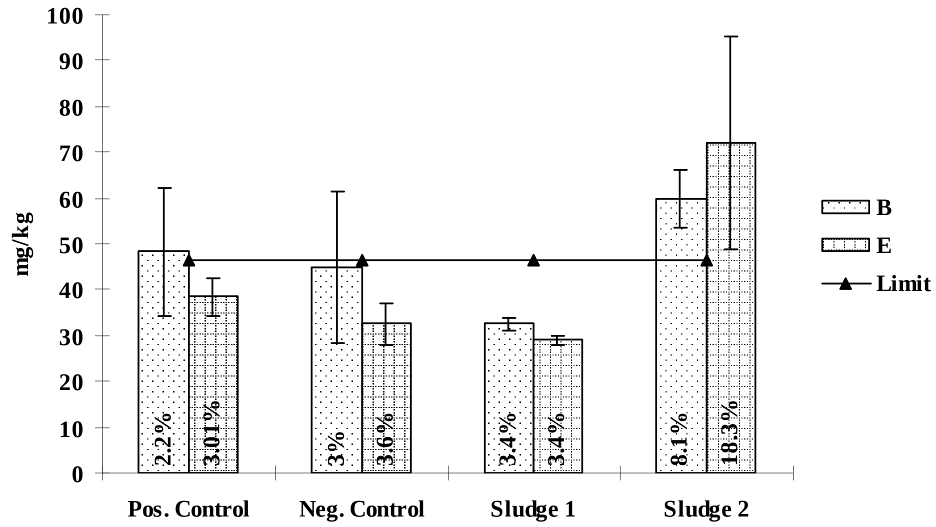

Figure 6.3. Total Zn concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD).

Figure 6.4 shows the total Zn concentrations in oats seedling tissue after 28 d of growth.

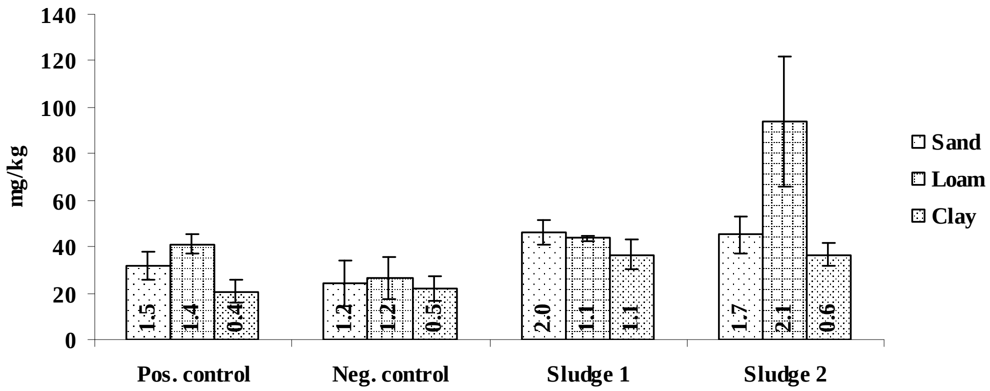

Figure 6.4. Total Zn concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

The uptake of Zn in the plant parts followed the trend of the available Zn in the soil. The transfer coefficients calculated (Fig. 6.4), were compared to the normal transfer coefficient for Zn in oats which is between 1 and 2 (Korentajer, 1991). The transfer coefficients for Zn in the clayey soils was lower than normal (except the Sludge 1 treatment) due to the strong adsorption of Zn to the clayey particles and more negative charge sites for adsorption (Alloway, 1995). The lower pH in the loamy soil (~5.3) caused a higher transfer coefficient in the loamy soil and subsequent higher uptake of Zn in the oats seedling tissue. Zn concentrations did not reach phytotoxic levels [100 - 400 mg kg^{-1} (Smith, 1996)] in the soil types.

Cadmium

The concentrations of Cd in the soil types were low and did not exceed guideline limits even after sludge application (Figs 6.5, 6.6 and 6.7). The Cd levels are very low and the variability observed in the levels throughout the three figures could be due to the sensitivity of the extraction procedure having been exceeded.

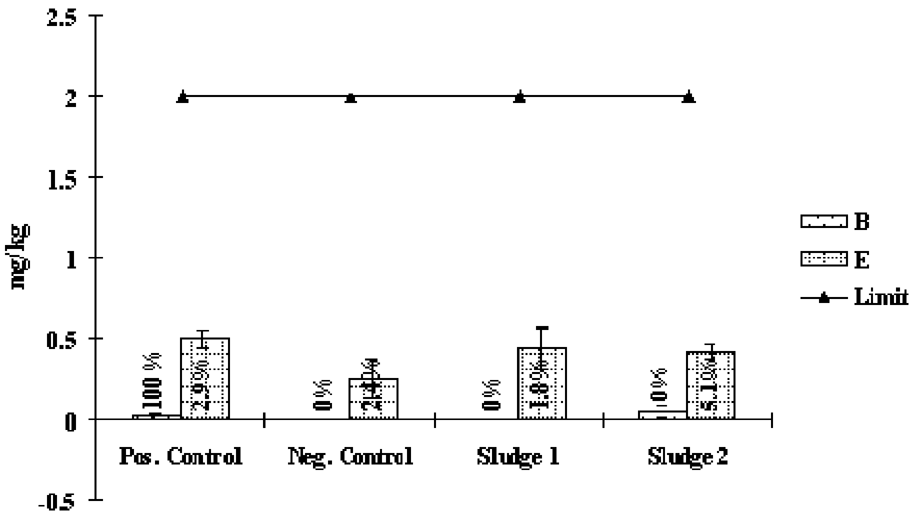

Figure 6.5. Total Cd concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

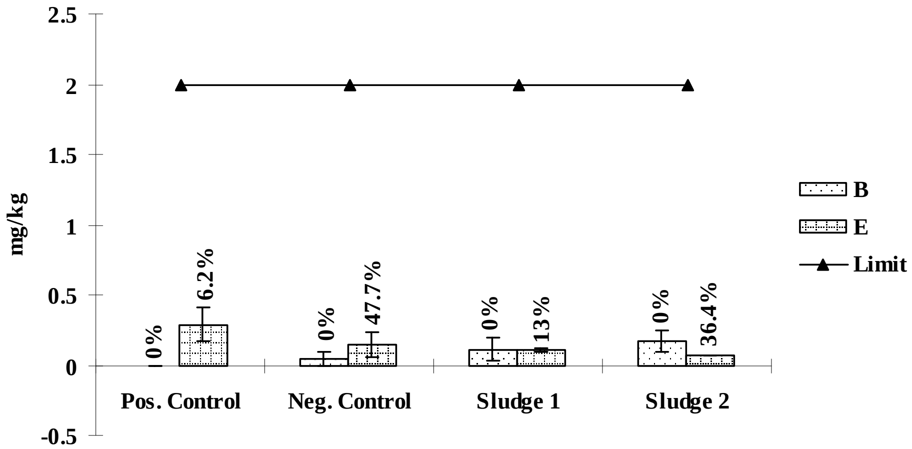

Figure 6.6. Total Cd concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

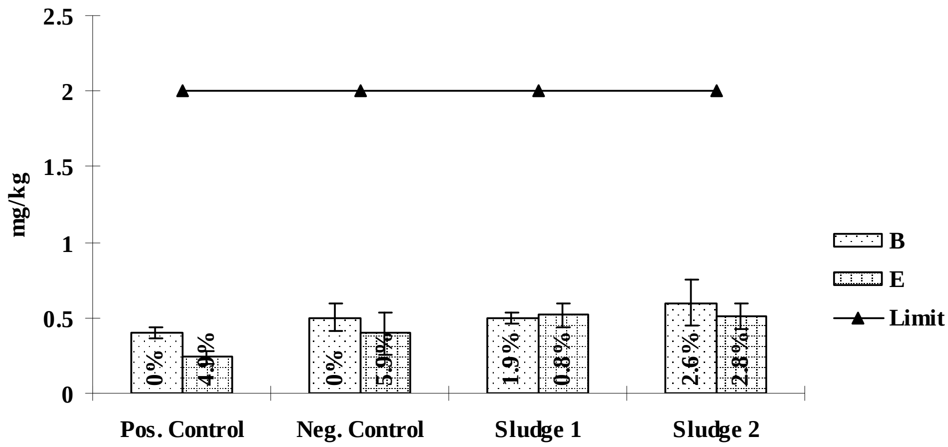

Figure 6.7. Total Cd concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

The uptake of Cd in oats seedling tissue is indicated in Figure 6.8. As in the soils data the tissue Cd content of the seedlings is highly varied as is evident from the calculated *f* factors. The tissue levels indicate no significant difference between all the treatments on all the soils. The uptake of Cd into the oats seedling tissue also did not reach phytotoxic levels of 5 - 30 mg kg^{-1} (Alloway, 1995); after a single exposure to sludge at 24 t ha^{-1}. This aspect leads to the conclusion that the Cd levels are too low to give an accurate indication of the transfer from the soils to the plants. Therefore the sludges pose no threat in terms of food chain Cd pollution if the sludges are applied at the levels indicated in this study.

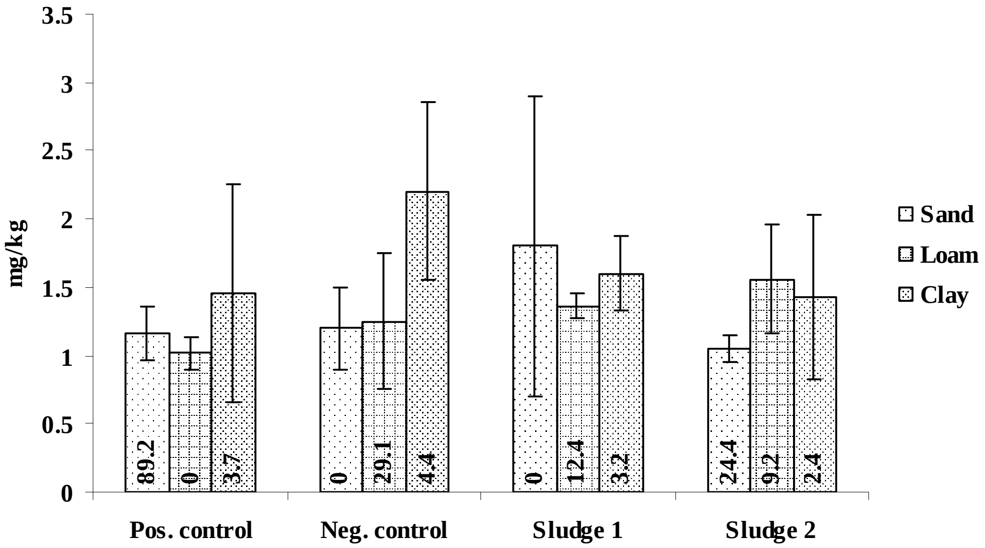

Figure 6.8. Total Cd concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

Copper

Figures 6.9, 6.10 and 6.11 show that guideline limits of 6.6 mg kg^{-1} for Cu were exceeded in the soil types due to high soil background levels. In all cases the Cu indicated a slight increase from the beginning to the end of the trial in all the soils. No difference between the treatments was evident. The Cu availability remained constant over the 28 d in the soil types emphasising the fact that Cu is a relative immobile element (Alloway, 1995). Alloway (1995) reported that much of the Cu taken up into the roots might not be translocated into the shoot.

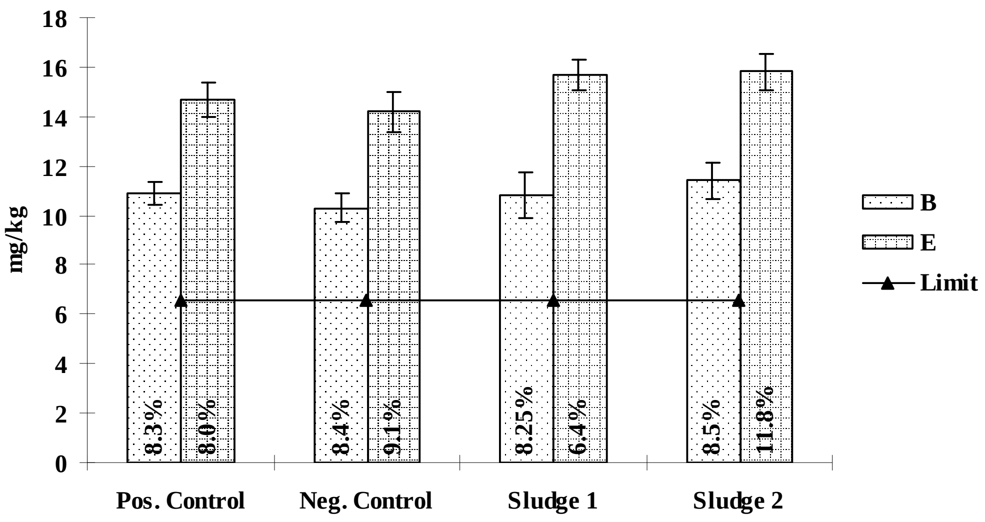

Figure 6.9. Total Cu concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

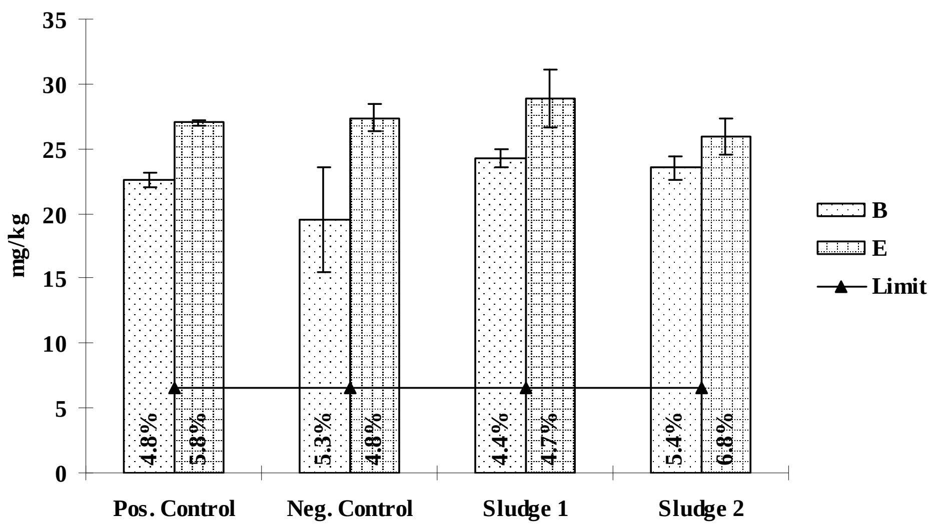

Figure 6.10. Total Cu concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

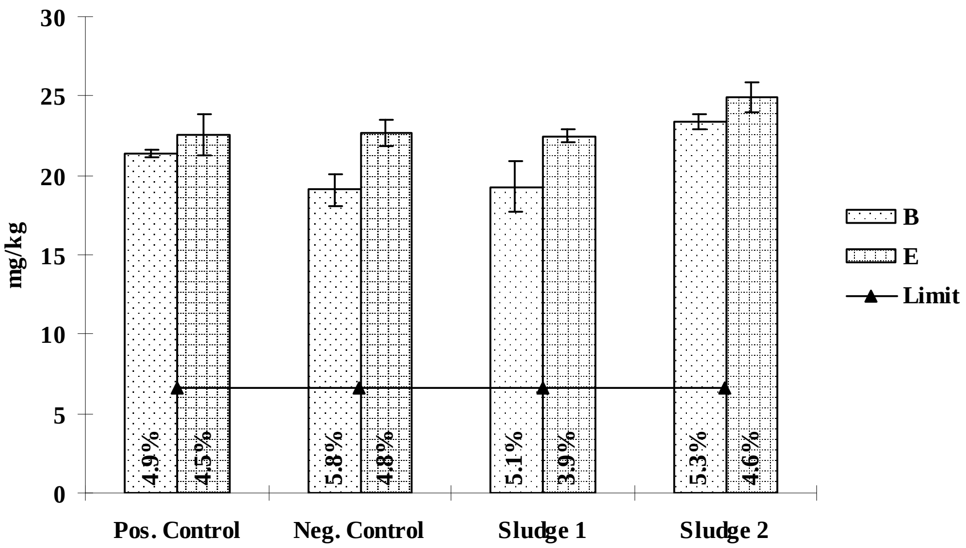

Figure 6.11. Total Cu concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

Cu concentrations in seedling tissue (Fig. 6.12) did not reach phytotoxic levels of 20 - 100 mg kg^{-1} (Smith, 1996).

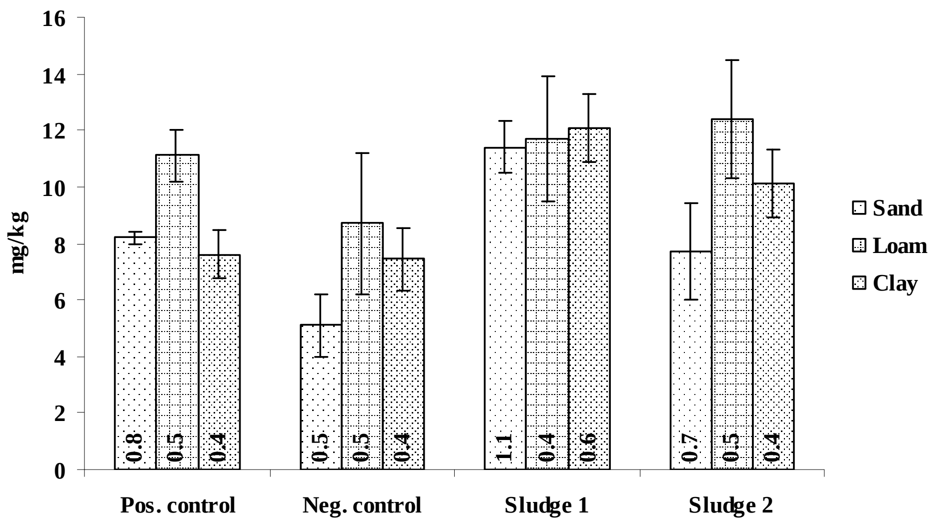

Figure 6.12. Total Cu concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

When the plant transfer coefficients for the Cu in the oats seedlings are compared to the normal plant transfer coefficient (*f* factor) values for Cu in oats [0.01 and 0.05 (Korentajer, 1991)], the transfer coefficient for all the soil types was higher than normal. Higher uptake of Cu in the seedling tissue occurred in the loamy soil, possibly due to the low soil pH, which caused higher availability of Cu in soils for plant uptake (Alloway, 1995).

Lead

Total Pb concentrations exceeded guideline limits in the three soils (Figs 6.13, 6.14 and 6.15) after the 28 d of growth in the greenhouse. The decrease in total Pb content over the 28 d in the loamy and clayey soils (Figs 6.14 and 6.15) could be ascribed to reactions with Fe and Mn oxides and hydroxides after being released from degrading organic complexes. The Longlands soil has a lack of these minerals and the Pb could therefore not be immobilised after release from organic complexes. The same factors would account for the higher *f* factors for the sandy soil in Figure 6.16. Normal plant transfer coefficient (*f* factor) values for Pb in oats are between 0.01 and 0.05 (Korentajer, 1991), therefore the transfer coefficient for all the soil types was higher than normal. Uptake of Pb in seedling tissue was low and did not reach phytotoxic levels of 30 - 300 mg kg^{-1} (Smith, 1996).

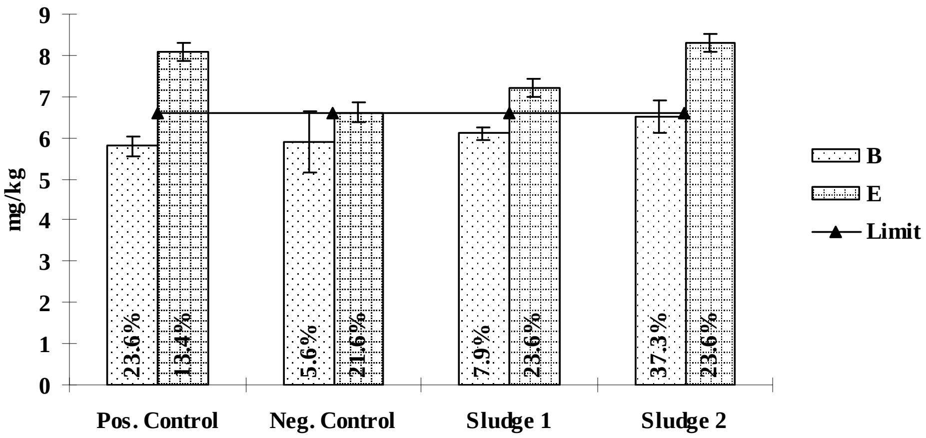

Figure 6.13. Total Pb concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

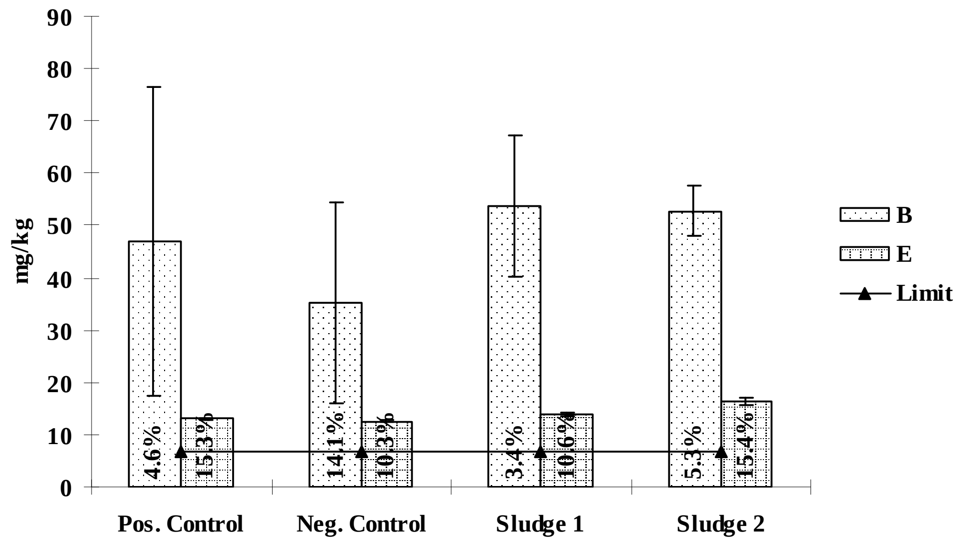

Figure 6.14. Total Pb concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

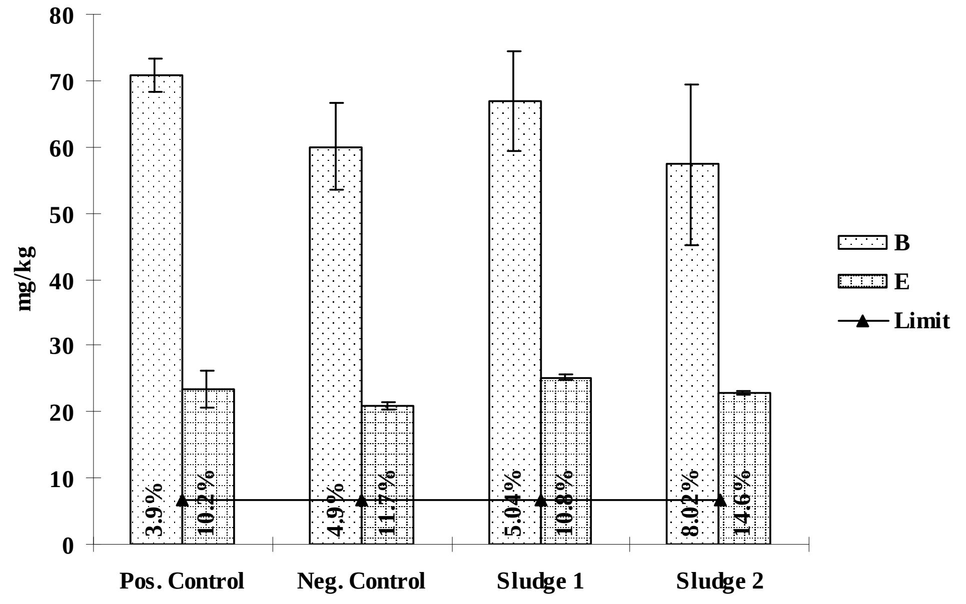

Figure 6.15. Total Pb concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

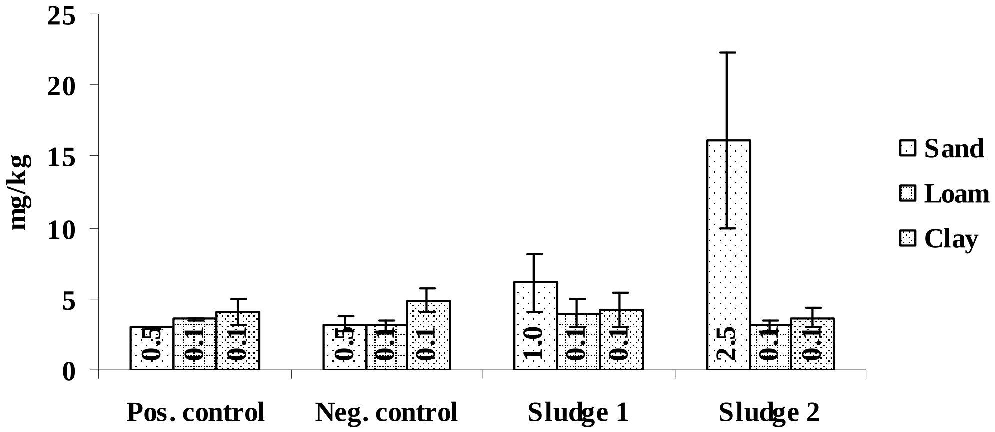

Figure 6.16. Total Pb concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

Yield

The average shoot length, wet mass and dry masses per plant are presented in Table 6.1. The general trend was that the wet mass and dry mass of the seedlings grown in the Sludge 1 treatment showed significant increases for these aspects compared to other treatments, except when compared to the wet mass of the Positive Control treatment in the loamy soil. The Sludge 1 treatment also showed a significant increase in shoot length when compared to the controls in the sandy and clayey soils. Compared to a similar greenhouse experiment done by Henning *et al.* (2001), the yield of oats seedlings grown in the Sludge 2 treatments, did not increase significantly compared to maize seedlings. The significant increases discussed above emphasised the beneficial addition of sludges to soils as an organic soil conditioner. When comparing seedling yield and growth between different soil types within treatments some significant differences were seen. The Positive Control seedlings showed significantly higher shoot lengths in the loamy soil compared to the other soils where inorganic fertilizer was added.

Table 6.1 Average shoot length, wet and dry mass per oats plant after 28 d of growth

Parameter	***Clayey***			***Loamy***			Sandy		
	Shoot length	Wet mass	Dry mass	Shoot length	Wet mass	Dry mass	Shoot length	Wet mass	Dry mass
Positive control	25.5^{ax}	0.5^{ax}	0.03^{ax}	35.7^{by}	0.6^{abx}	0.02^{ax}	32.7^{bxy}	0.6^{ax}	0.03^{ax}
Negative control	22.9^{ax}	0.4^{ax}	0.01^{ax}	26.7^{ax}	0.3^{ax}	0.02^{ax}	26.5^{ax}	0.4^{ax}	0.01^{ax}
Sludge 1	36.4^{bx}	0.9^{bx}	0.08^{bx}	38.5^{bx}	0.7^{bx}	0.05^{bx}	39.1^{cx}	1.0^{bx}	0.06^{bx}
Sludge 2	29.1^{abx}	0.5^{ax}	0.02^{ax}	26.7^{ax}	0.4^{ax}	0.02^{ax}	32.2^{bx}	0.5^{ax}	0.02^{ax}

* Each value is a mean value per plant of 4 replicates of 20 plants. Values within a row not followed by the same letter (x, y or z), or within a column by the same letter (a, b, c or d) are significantly different (*P* = 0.05) according to Duncan's multiple range test.

6.5 CONCLUSIONS

In this study the effect of sewage sludge on oats seedling growth and yield on different soil types was monitored. The two sludges used were high in metal- and low in metal content. It was found that the heavy metal guidelines were exceeded in the soil and sludge types for Pb and Cu mostly due to high soil background levels, emphasising the conservative nature of the current S.A. guidelines when interpreted as total metal content. The Zn concentrations also exceeded guidelines and it seemed that the amendment of sludge-borne Zn to the soils significantly increased the total Zn concentrations in the soil types as was also reported by Chlopecka (1996). Predicted theoretical contribution of the sludge-borne metals to total metal content were analytically different possibly due to the difficulty of obtaining a representative sample in sludge-amended soils. Long-term experiments still need to be done on heavy metal accumulation in soil types. However, as previously reported by Unger & Fuller (1985) and Miller *et al.* (1995), no phytotoxic effects could be proven, as phytotoxic levels were not exceeded in oat seedling tissue.

The increased yield of the oats seedlings, compared with control plants, was also previously reported by Simeoni *et al.* (1984). Comparing the heavy metal uptake between maize seedlings (Henning *et al.*, 2001) and the oats seedlings in the experiment, it was found that uptake varied in sludge-amended soils. For instance, higher uptake of Cd was found in oats seedlings compared to maize seedlings, while uptake of Zn was higher in the maize seedlings. The transfer coefficients of Pb, Cd and Cu were similar to the coefficients found in the maize seedlings (higher than the normal ranges), although the transfer coefficients of Zn were within the ranges. This phenomenon was possibly due to differences in seasonal soil properties (e.g. soil pH and CEC) on the availability of heavy metals, although differences could also be attributed to differences between the crops. As summer (maize) and winter cereal (oats) crops have different carbohydrate metabolisms, C4 and C3 pathways, respectively, heavy metal uptake might have been different when grown in sludge-amended agricultural soils (Salisbury & Ross, 1995).

The beneficial use of these sludges on agricultural soil could possibly play an important role in the organic material and nutrient depleted soils of South Africa. Farmers must be assured that the levels of inorganic chemicals in sludge are not sufficiently high to be toxic to plants (a possibility for Zn and Cu) or to humans and other animals, which consume the plants (a serious consideration for Cd and Pb). Monitoring soil acidity and using judicious applications of lime can prevent leaching into groundwater and can minimise uptake by plants, since these metals are generally bound by soil constituents at high pH values (Brady & Weil, 1996). The possible revision of the S.A. heavy metal guidelines to allow unrestricted use, should rather take into account the different responses of crops to sludge application and soil properties like soil pH and CEC. This might lead to the unrestricted use of sludge on agricultural land in future in S.A.

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Appendix 7

THE CULTIVATION OF FIELD-GROWN OATS (*AVENA SATIVA* L.) ON DIFFERENT SEWAGE SLUDGE DOSAGES

7.1 SUMMARY

The decrease in organic matter and nutrient-depletion of South African soils emphasises the importance of sewage sludge application to agricultural land. Oats cultivation, mainly to produce fodder, takes place during the winter months in South Africa. A field experiment was performed in an area where farmers cultivate oats. The aim was to assess the effect of different dosages of sewage sludge (4 t ha^{-1} and 8 t ha^{-1}) on growth and yield of oats (*Avena sativa* L.) as a winter crop. The following aspects were monitored during the field trial: Yield differences, soil metal concentrations in the loam soil and heavy metal accumulation in plant parts. Errors in sample taking, preparation and/or analyses probably caused incorrect soil metal concentrations at the end of the experiment, which caused the guidelines to be exceeded. Accumulation of metals in plant parts was not phytotoxic, although accumulation of metals in soil could prove a problem over the longer term. No significant differences were found between treatments for yield aspects.

7.2 INTRODUCTION

Oats is an important winter cereal crop produced throughout S.A. Oats cultivation under S.A. conditions are mainly done to produce fodder for livestock. Producing high yields of high quality oats are important and involves interactions among numerous biological factors, management strategies and climatic conditions (Welch, 1995). Although previous greenhouse experiments performed by Snyman *et al.* (1998) and Henning *et al.* (2001) have shown the beneficial use of sludge when applied to soil for seedling growth, it is important to conduct field studies to determine the actual situation in practice, due to the limitations of pot studies.

This field experiment was done to assess the effect of sewage sludge on the growth and yield of oats (*Avena sativa* L.) as a winter crop under environmental conditions on the East Rand, South Africa. The experiment was performed as a follow-up experiment after a field study performed by Henning *et al.* (1999) on maize, as previously discussed in Appendix 4.

The same aspects were monitored as discussed in Appendix 4 (monitoring for pathogenic indicator organisms was omitted).

7.3 METHODS

Collection, treatment and analysis of dewatered sewage sludge

Sludge samples were collected and analysed as discussed under "Collection, treatment and analysis of dewatered sewage sludge", Appendix 4.

Experimental layout

The same experimental site, design, and management practices (e.g. liming) were used for the experiment as previously discussed under "experimental layout" in the maize field experiment described in Appendix 4. Sludge was applied at rates differing from those listed in the Appendix 4 but the same as those listed in Appendix 5. The dewatered sludge was applied at 4 t and 8 t ha^{-1} one week prior to planting. Positive and Negative Controls were used as described in Table 4.1, Appendix 4. Oats seed were planted using standard planting techniques.

Soil analyses

Soil samples were collected at two different times during the experiment using standard sampling techniques. At the start of the experiment, after sludge application, soil samples were collected from each of the plots and analysed for nutrients. Heavy metal content (Zn, Cu and Pb) [EPA3050 method (Sims *et al.,* 1992)] was determined at the beginning of the field trial and at the mature stage of growth as described in Appendix 4.

Plant material analyses

Leaf samples were randomly collected at plant maturity from each of the plots during the field trial. All leaf samples were analysed for heavy metals (Zn, Pb, Cu and Mn). Leaf samples collected to determine yield differences were harvested as a single unit (0.6 m x 0.6 m), since individual plants could not be distinguished from other plants due to high plant density. Yield differences between treatments were measured in terms of the wet mass of harvested leaves and percentage dry weight of the leaves sampled.

7.4 RESULTS AND DISCUSSION

Analysis of dewatered sewage sludge

The inorganic macronutrient levels detected on a dry basis in the sludge were found to be 0.05% phosphorus, 4.17% nitrogen, 0.28% potassium, 1 789 mg kg^{-1} calcium and 1 465 mg kg^{-1} magnesium. Phosphorus was exceptionally low compared to the same sludge used by Henning *et al.* (1999) in another field experiment. This could be attributed to error in sample preparation or analyses. General characteristics of the sludge were as follows: Moisture content of 78.1%, solids of 21.9% and a pH of 6.71.

Main heavy metals (Pb, Cu and Zn) were analysed for and detected compared to guidelines as indicated in Table 7.1. The concentrations of Pb, Cu and Zn exceeded guideline limits.

Table 7.1 Sludge-borne metal concentrations compared to guideline limits (WRC, 1997)

Metal	Level in sludge (mg kg^{-1})	Limit (WRC, 1997) (mg kg^{-1})
Cu	116	50.5
Zn	780	353.5
Pb	66.8	50.5

Soil analyses

Table 7.2 indicates the pH and resistance as well as nutrient analysis data at the beginning of the experiment. Even though lime was applied to raise the soil pH, the pH was below the recommended 6.5 (Korentajer, 1991) at the beginning of the experiment after sludge application. pH values below 6.5 will increase the mobility and availability of heavy metals, possibly leading to higher uptake of metals in plant parts (WRC, 1997). However, maximum oat yields are achieved at a soil pH between 5.3 and 5.7 (Welch, 1995).

Resistance values were relatively normal, due to salts not being in excess in the soil (Ca and Mg in salt forms like $CaCl_2$ and $MgSO_4$). In general, phosphorus (mg kg^{-1}) and potassium concentrations increased in the soil after sludge application due to the relative high nutrient values in the sludge. This confirmed the value of sludge when used as an organic fertilizer.

Table 7.2 Chemical characteristics of the soil at the beginning of the experiment

Parameter	- Control	+ Control	Exp. 1	Exp. 2
PH	6.41	5.86	6.12	5.86
Resistance (Ω)	1 400	1 600	1 600	1 600
P (mg kg^{-1})	128	191	191	192
Ca (mg kg^{-1})	1 046	866	1 056	886
Mg (mg kg^{-1})	245	203	257	230
K (mg kg^{-1})	314	421	613	649

Zinc

Although Zn is an essential element for all living organisms, like all nutrient ions it can be toxic at high levels (Jarausch-Wehrheim *et al.*, 1999). Figure 7.1 shows the Zn concentrations in the soil during the experiment. Zn concentrations were below guideline limits at the beginning of the experiment and the applications of sludge to the soil did not increase total Zn concentrations significantly in the soil. During the field experiment final sewage effluent was irrigated to the field plots due to extreme drought. The increase in Zn concentrations in the soil, such that the guideline limits were exceeded could be attributed to poor sampling techniques and/or error in analyses and sample preparation. This emphasises the importance of taking good, representative samples and to take care in preparation and analyses of the samples.

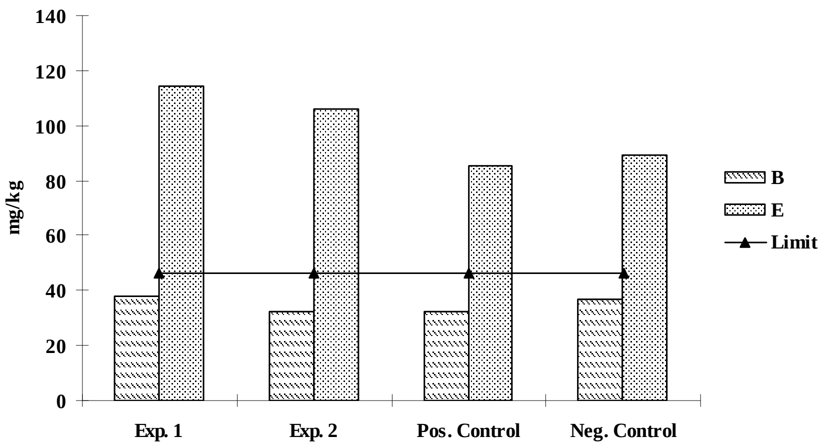

Figure 7.1. Total Zn concentrations in the loam soil from the beginning (B) to the end (E) of the experiment as compared to guidelines (WRC, 1997).

Lead

Lead is a metal with a low mobility and solubility, and hence its bioavailability is low. Pb is neither an essential nor a beneficial element for plants and animals (Alloway, 1995). Pb concentrations were below guideline limits at the beginning of the experiment. Pb concentrations increased excessively during the experiment probably due to the same reasons discussed earlier under Zn. Even though sewage effluent was applied to the soil during dry times it is unlikely that this caused the increase in total Pb concentrations seen in Figure 7.2.

Copper

The total concentrations of Cu in the soil at the beginning and end of experiment are shown in Figure 7.3. The chemistry of Cu in soils is somewhat like that of Pb in that both are adsorbed or 'fixed' in soils. Cu is still one of the most important, essential elements for plants and animals (Alloway, 1995). Cu concentrations exceeded guideline limits at the beginning of the experiment due to high soil background levels. The increase in the Cu concentration during the experiment could be attributed to the same reasons as discussed under zinc.

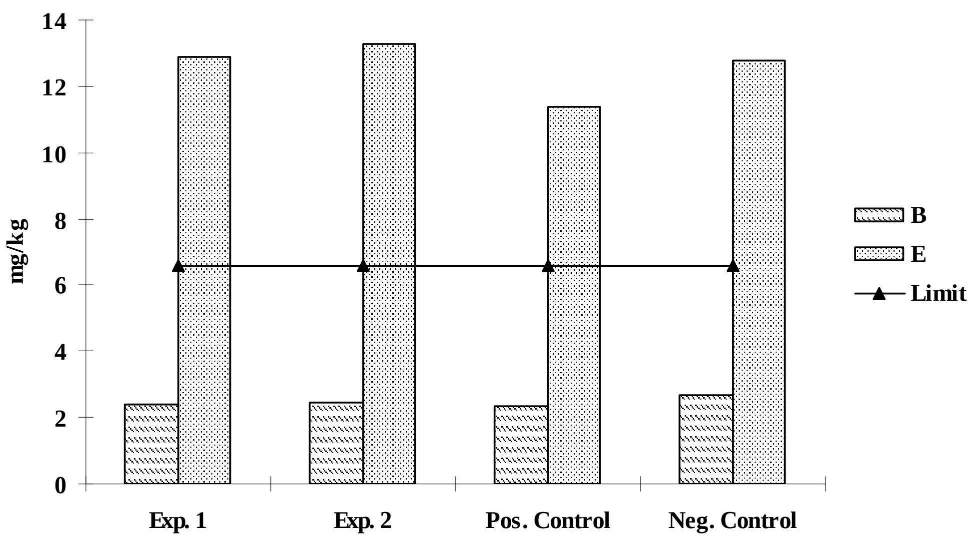

Figure 7.2. Total lead concentrations in the loam soil from the beginning (B) to the end (E) of the experiment as compared to guidelines (WRC, 1997).

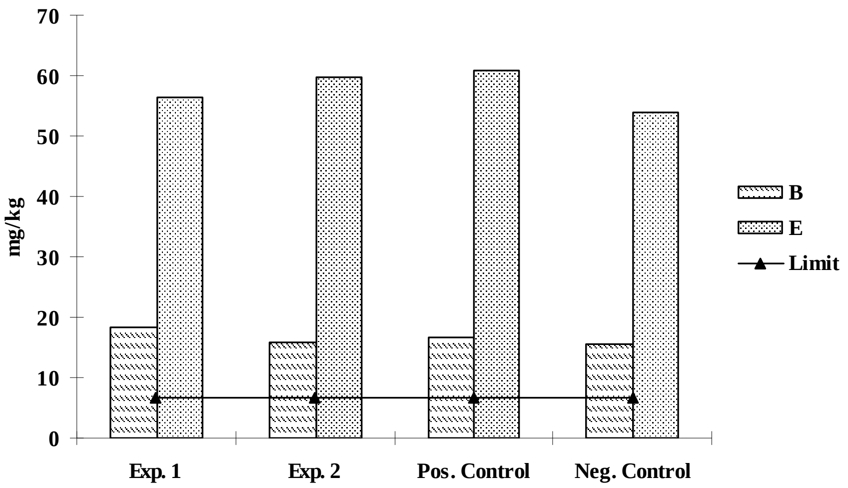

Figure 7.3. Total copper concentrations in the loam soil from the beginning (B) to the end (E) of the experiment as compared to guidelines (WRC, 1997).

Although copper exceeded guideline limits at the beginning of the experiment, and the metals increased during the experiment (which were rather due to errors in sampling), the metals analysed for did not pose a threat in terms of phytotoxicity. Soil properties like soil pH, rather than total metal content will determine whether the metals will be available for plant uptake.

Plant material analyses

Heavy metal concentrations did not reach phytotoxic levels in the leaves of the mature oats plants (Table 7.3).

Table7.3 Heavy metal content in the oat plant leaves at plant maturity

Parameter	Phytotoxic levels ($mg\ kg^{-1}$) (Smith, 1996)	+ Control	- Control	Exp. 1	Exp. 2
Zn ($mg\ kg^{-1}$)[a]	100-400	33.0	26.8	35.8	38.9
Mn ($mg\ kg^{-1}$)	400-1000	163.8	235.2	176.9	159.6
Cu ($mg\ kg^{-1}$)	20-100	5.22	5.67	5.42	6.6

[a] Metal concentration is expressed as mg of total extractable metals per dry kg of leaves. Each digestion was performed at least 4 times.

Although there was an increase in the uptake of Zn in the sludge treatments, the uptake was insignificant. Manganese was not included in soil-metal concentrations since Mn concentrations were too high to be detected in the soil as discussed in Appendix 4. Higher uptake of Mn occurred in the Negative Control treatment, although uptake of Mn was mostly due to the soil background concentrations. The low uptake of Cu is due to the low mobility of Cu in soils. However, as previously shown by Chlopecka (1996), the higher uptake of Cu in the Exp. 2 treatment (8 t ha^{-1}) indicated that the application of sewage sludge to agricultural land could lead to an increased uptake of sludge-borne Cu. Cu was possibly more available for plant uptake in the Exp. 2 treatment.

Yield differences are shown in Table 7.4. Statistical analyses of dry mass, wet mass and plant height did not show any significant differences between treatments. Sludge treatments compared well with other treatments and showed a slight increase in the dry mass (t ha^{-1}) for the low sludge application rate (Exp. 1). Yield could possibly have been influenced by the sewage irrigation water, which usually contains high phosphorus concentrations that affects soil microorganisms like mycorrhizae, which enhance plant growth.

Table 7.4 The effect of sewage sludge on yield of oats under field conditions in loam soil

Treatment	+ Control	- Control	Exp 1	Exp. 2
Wet mass (t ha^{-1})	$31.6^{*,a}$	31.6^{a}	28.8^{a}	30^{a}
Dry mass (t ha^{-1})	5.0^{a}	4.83^{a}	5.5^{a}	4.83^{a}
Plant height (cm)	126.0^{a}	122.1^{a}	127.6^{a}	126.4^{a}

* Each value is a mean value per plant of 5 replicates of plants $0.6m^2$. Values within a row not followed by the same letter are significantly different. ($P = 0.05$).

7.5 CONCLUSIONS

Farmers on the East Rand, South Africa are making use of the sludge used in the study as part of a programme with the East Rand Water Care Company. In this study the effect of sludge dosages applied to a loam soil on the growth and yield of oats was monitored. It was found that soil-metal concentrations were adequate at the beginning of the experiment, although an increase in soil-metal concentrations occurred during the experiment, possibly due to the application of final sewage effluent. Heavy metal guidelines were exceeded at the end of the experiment possibly due to error in sample preparation and/or analyses.

The influence of soil conditions (pH and CEC) and other environmental parameters vary during the growing season in field trials, making the short-term beneficial influence of the sludge on growth minimal, and the uptake of heavy metals into plant tissue to be low. However, soil properties are still the major aspect controlling the fate of sludge-borne metals applied to soils.

Long-term field experiments still need to be performed to assess the possible accumulation of heavy metals in soils after repeated yearly applications of sewage sludge. Otherwise, the yield of oat plants cultivated in sludge-applied plots, showed similar yields compared to plots where inorganic fertilizer was applied. The sludge possibly increased the soil nutrient status and organic matter. This is important since farmers are depleting the soil of valuable nutrients and micronutrients by continuous cropping leading to the loss of soil structure and soil erosion. The use of the sludge in the future in S.A. depends largely on effective, responsible monitoring procedures.

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Appendix 8

PLANT-SOIL INTERACTIONS OF SLUDGE-BORNE HEAVY METALS AND THE EFFECT ON SUNFLOWER (*HELIANTHUS ANNUUS* L.) SEEDLING GROWTH

8.1 SUMMARY

Very similar results were obtained in the sunflower seedling trial as in the maize seedling trial (see Appendix 3).

8.2 INTRODUCTION

Agricultural crops differ in their physiological functions and therefore also differ in certain responses. For example, nutrient requirements (N, P and K) for sunflower (*Helianthus annuus* L.) is generally not as high as that for maize (*Zea mays* L.) but the nutrients are also not as efficiently utilised as in the grain crops (Robinson, 1978). Sunflower is cultivated in South Africa during the summer and is often used in areas where the soil moisture is not enough for maize and where the wet season is shorter than usual (Du Toit, Loubser & Nel, nd.). It is also regularly planted in soils of heavier texture such as Vertisols (Oberholzer, 1995).

Research on the effect of sewage sludge on growth of sunflower is limited. Sunflower seed and stover generally contain high levels of chemical elements. Concentrations are highest in young plants due to rapid absorption in relation to dry matter production during early growth but decrease to plant maturity (Robinson, 1978).

This study was done to assess the effect of sewage sludge on growth and yield of sunflower seedlings under greenhouse conditions. The uptake of four heavy metals (Pb, Cu, Cd and Zn) was monitored in the sunflower leaf tissue, and the heavy metal concentrations (total and available) were monitored in the sludge and soil. These values were used to draw a correlation between soil-metal concentrations and their uptake by crops.

8.3 MATERIALS AND METHODS

Collection, treatment and analysis of dewatered sewage sludge

See Appendix 3

Experimental layout

See Appendix 3

Soil analyses

See Appendix 3

Five sunflower (*Helianthus annuus* L.) seeds were planted per pot.

Plant material analyses

See Appendix 3

8.4 RESULTS AND DISCUSSION

Analysis of dewatered sewage sludge

See Appendix 3

Soil and plant material analysis

Table 8.1 presents the results of the different determinations on the soil before and after the sunflower trial.

Table 8.1 Chemical characteristics of the soil at the beginning and end of the trial for sunflower cultivation

Parameter	+ Control		- Control		Sludge 1		Sludge 2	
	B	E	B	E	B	E	B	E
Sandy soil								
pH (Water)	7.5	7.8	7.7	8.1	7.8	7.7	7.9	7.9
Resistance (Ω)	2000	1800	3600	1600	2800	2700	2400	1800
P (mg kg^{-1})	25	19.9	5.4	7.7	13		10	11.7
% N	0.02	0.04	0.03	0.05	0.05	0.05	0.03	0.04
NH_4-N	9.04	5.96	7.11	6.33	11.6	5.67	11.9	6.33
K (mg kg^{-1})	80	66	45	39	52	37	45	35
Ca (mg kg^{-1})	538	954	755	1015	780	860	704	862
Mg (mg kg^{-1})	79	163	96	148	120	157	117	147
Na (mg kg^{-1})	0	36	9	33	14	32	7	29
Loamy soil								
pH (Water)	5.8	6.6	5.1	5.3	5.3	5.6	5.6	5.9
Resistance (Ω)	2200	2400	5200	3600	3800	2600	2200	2000
P (mg kg^{-1})	6.2	9.1	3.1	6.3	16	24.9	6.6	6.8
% N	0.07	0.05	0.06	0.07	0.06	0.08	0.08	0.06
NH_4-N	16.1	6.25	13.9	9.61	24.5	35.2	25.1	9.18
K (mg kg^{-1})	128	26	49	37	56	25	56	33
Ca (mg kg^{-1})	319	436	147	250	177	286	306	511
Mg (mg kg^{-1})	125	148	51	70	69	99	107	166
Na (mg kg^{-1})	0	28	1	43	7	45	20	35
Clay soil								
pH (Water)	8.6	8.8	8.8	8.8	8.6	8.7	8.6	8.5
Resistance (Ω)	400	380	420	350	420	440	360	320
P (mg kg^{-1})	33	6.6	3.3	4.5	4.2	13.7	6.5	17.3
% N	0.03	0.03	0.03	0.04	0.06	0.07	0.04	0.07
NH_4-N	14.5	7.86	11.9	12.3	20.6	34.3	29.6	12.4
K (mg kg^{-1})	157	134	144	105	165	115	134	178
Ca (mg kg^{-1})	6852	3467	6892	4447	6412	1727	6562	1337
Mg (mg kg^{-1})	5302	4151	5392	4202	4849	4011	4739	5060
Na (mg kg^{-1})	261	251	228	209	218	176	200	300

Zinc

In the sandy soil (Fig. 8.1) the Zn levels exceeded the levels stipulated by the guidelines except for the positive control that was below the guideline levels. In al the treatments the Zn concentration decreased from the beginning of the trial to the end. The availability of the Zn increased from the Positive Control to the Sludge 2 treatment as well as from the beginning to the end of the trial. In the loamy soil (Fig. 8.2) this trend was repeated with slightly lower Zn values throughout. The availability of the Zn displayed similar values throughout the treatments but increased slightly in the Sludge 2 treatment.

In the clay soil (Fig. 8.3) the Zn values exceeded the guideline limits three to four fold at the beginning of the trial but decreased at the end of the trial to levels very close to the guideline limits. Here the Sludge 2 treatment was an exception in that the levels were very similar before and after the trial. The Zn availability varied widely and increased drastically in the Positive Control, S1 and S2 treatments towards the end of the trial. The increase in the availability of the Zn towards the end of the trial (in all the soils) could possibly be attributed to a decrease in pH towards the end of the trial with mineralisation of organic material as well as plant cation uptake effects.

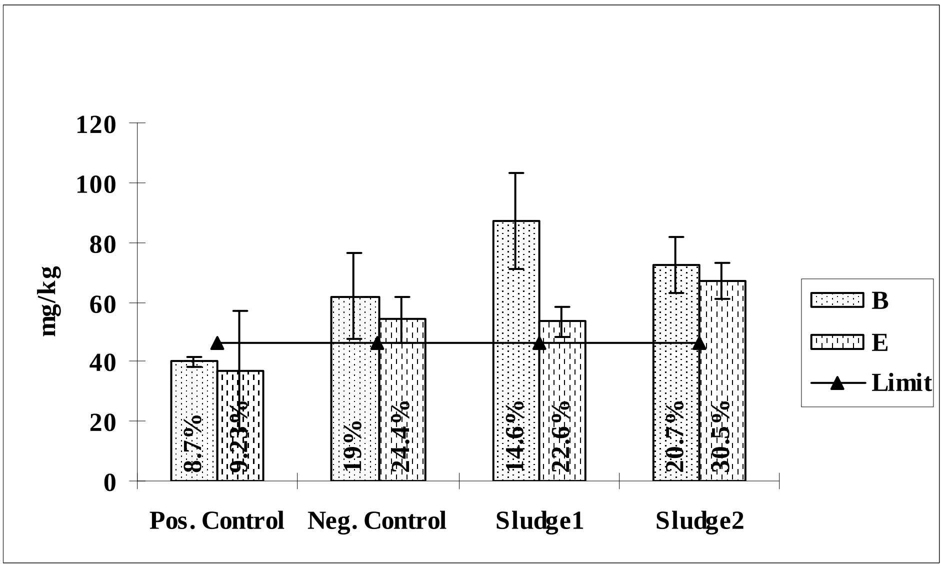

Figure 8.1. Total Zn concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD).

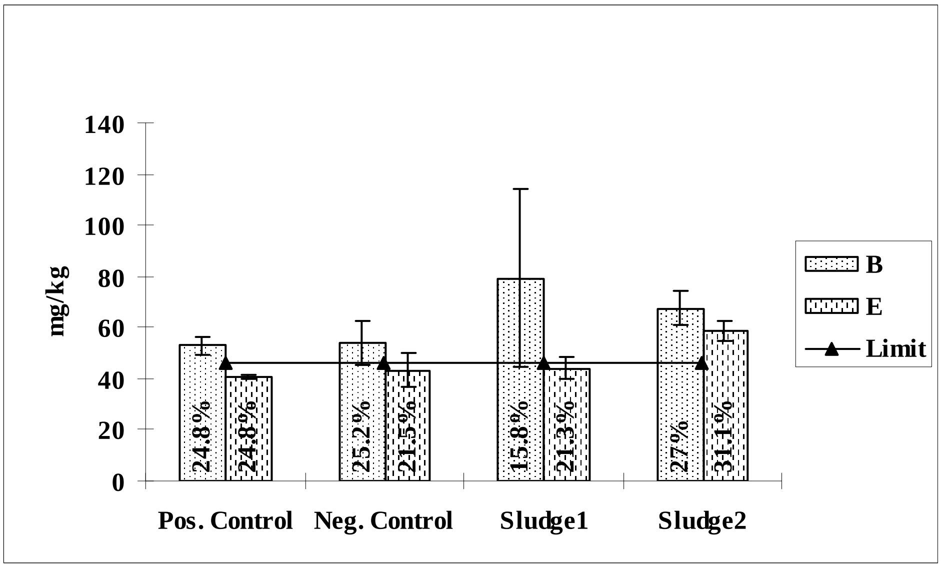

Figure 8.2. Total Zn concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD)

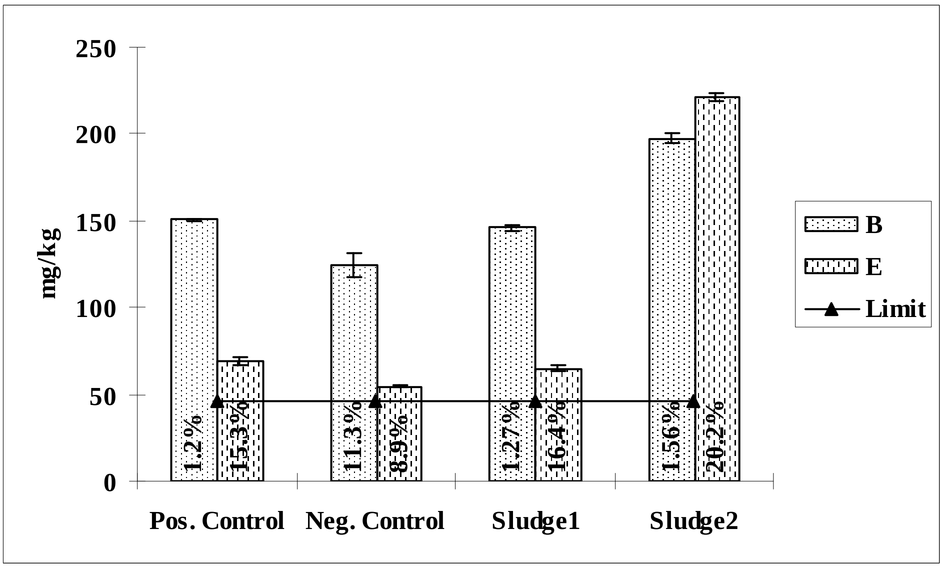

Figure 8.3. Total Zn concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Zn. (I = STD).

Figure 8.4 indicates the total Zn concentrations in sunflower seedling tissue after 28 d of growth. In all of the cases (except for the Negative Control, Sludge 1 and Sludge 2 treatments on the loamy soil) the Zn levels in the plants were below the lower toxic levels. In the case of the loamy soil the levels exceed the toxic levels substantially. These results are not consistent with the values for the other soils and would seem to indicate possible contamination of the sample or experimental error, especially in the case of the Negative Control. In the case of the two sludge treatments the low pH of the soil as well as the amounts of Zn added to the soil could play an important role. This is confirmed by the calculated *f* factors that are several orders higher than the treatments on the other soils. Although no *f* factors are presented for Zn in sunflower by Korentajer (1991), the calculated *f* factors fall within the ranges given for most crops, with the exception of the loamy soil *f* factors.

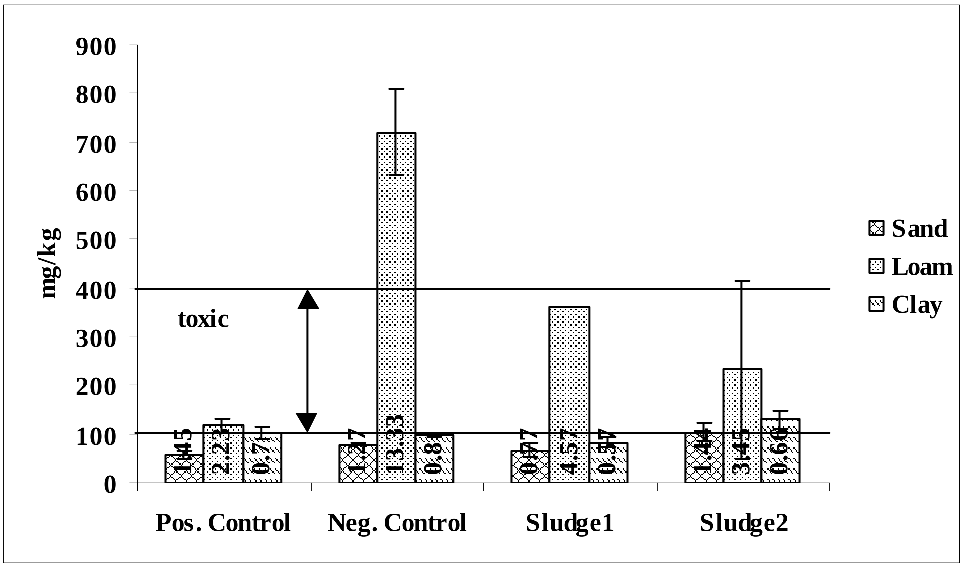

Figure 8.4. Total Zn concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

Cadmium

The concentrations of Cd in the soil types were low and did not exceed guideline limits even after sludge amendment (Figs 8.5, 8.6 and 8.7). Cd levels decreased drastically in the soil over the 28 d. This is attributed to the same factors that were discussed in Appendix 6 under Cd. In the cases where Cd was detected at the end of the trial the potential availability increased by several factors although the total levels decreased. This could be due to the calculated availability being higher percentage wise but still very similar in real terms (EDTA values at beginning and at end of the trial). Again the Cd values are considered too low to make a meaningful deduction and it is thought that the sensitivity of the testing procedure was not adequate enough anymore.

The uptake of Cd in sunflower seedling tissue is presented Figure 8.8. The plant transfer coefficient (*f* factor) for Cd was relatively high in all soil types when compared to the transfer coefficient for Cd in other crops (Korentajer, 1991). The sludge treated soil values did not differ widely from the values of the Negative and Positive Controls. This would seem to indicate that factors other than the sludge Cd content played a role in the Cd content of the seedlings.

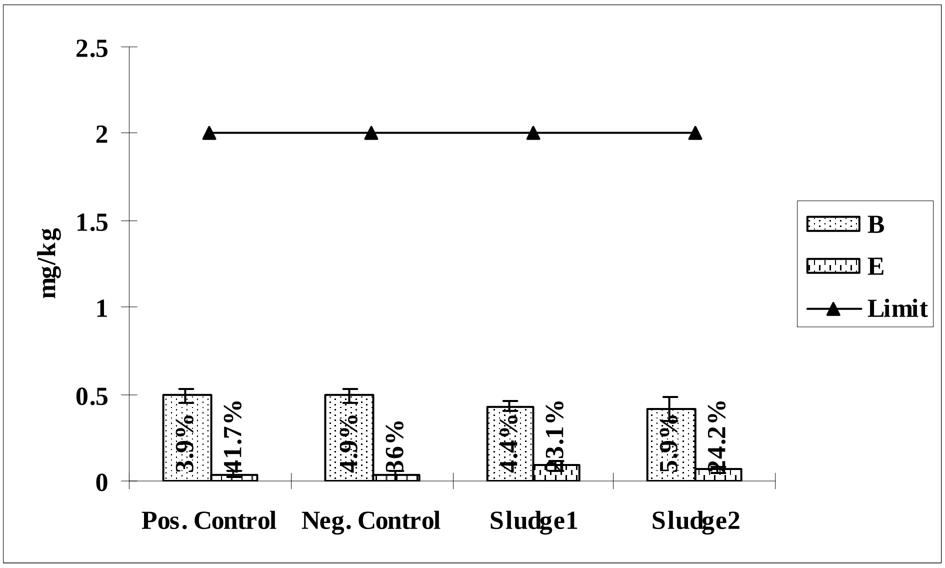

Figure 8.5. Total Cd concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

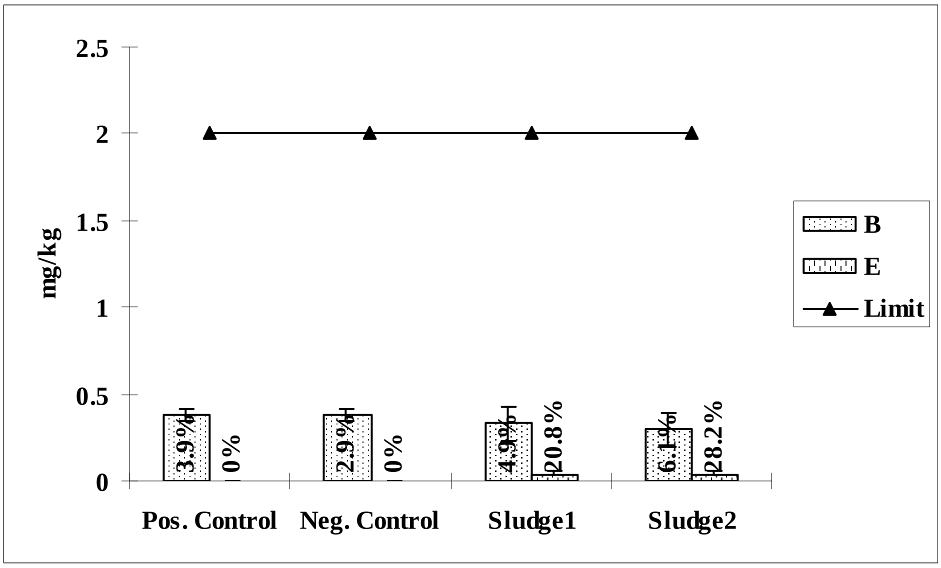

Figure 8.6. Total Cd concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

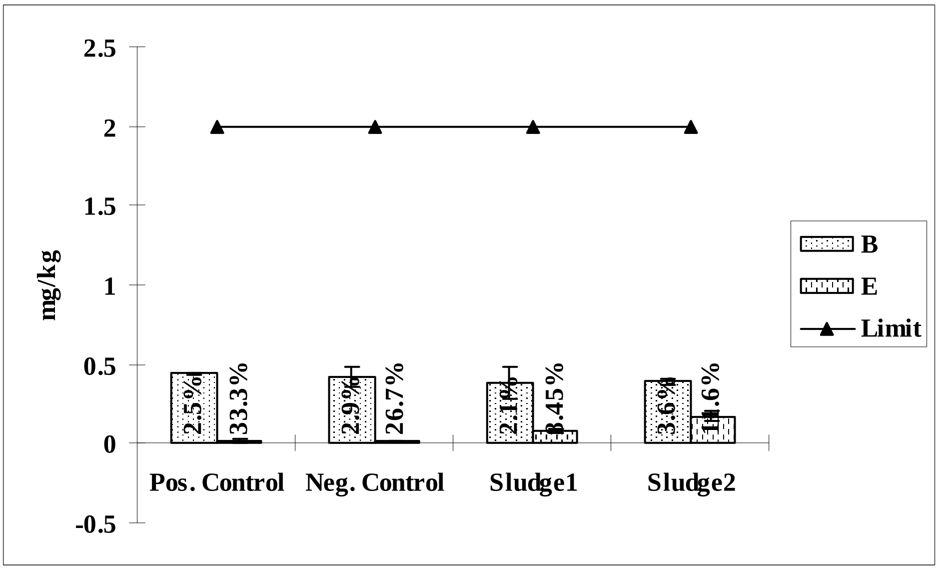

Figure 8.7. Total Cd concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate potential availability of Cd. (I = STD).

The higher transfer coefficients in the loamy soil of the sludge treatments and lower coefficients in the clayey soil were due to the effect of soil pH on metal availability as discussed earlier. The uptake of Cd into the sunflower seedling tissue did not reach phytotoxic levels of 5 - 30 mg kg^{-1} (Alloway, 1995) after a single exposure to sludge at 24 t ha^{-1}.

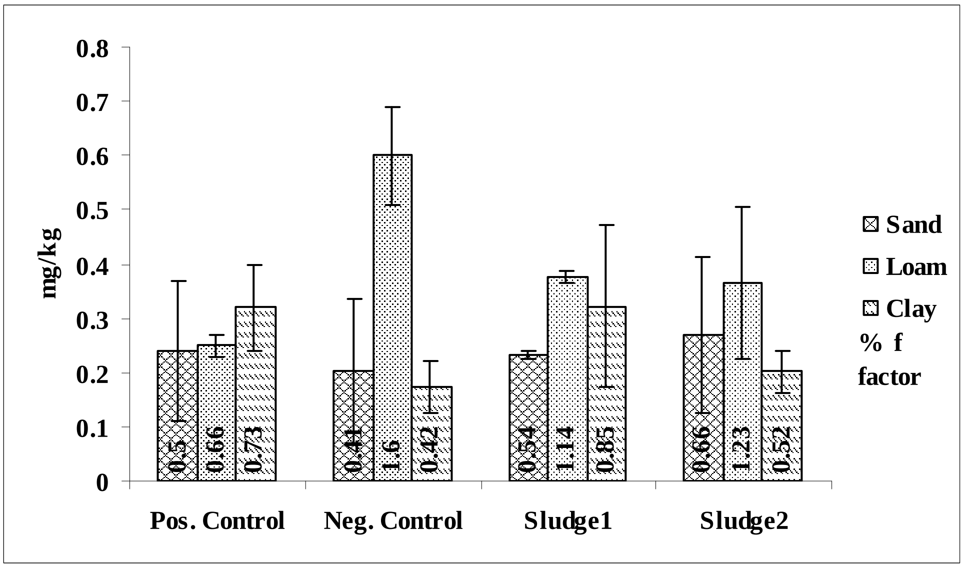

Figure 7.8. Total Cd concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

Copper

Figures 8.9, 8.10 and 8.11 indicate that guideline limits of 6.6 mg kg^{-1} for Cu were exceeded in all the soil types due to high soil background levels. Cu availability remained relatively constant over the 28 d in the different soils emphasising the fact that Cu is a relative immobile element (Alloway, 1995). Alloway (1995) reported that much of the Cu taken up into the roots might not be translocated into the shoot.

The slight differences between the controls and the sludge treatments indicate that the sludge treatments at 24 t ha^{-1} have very little influence on the Cu content of the soil when the soil has high background levels.

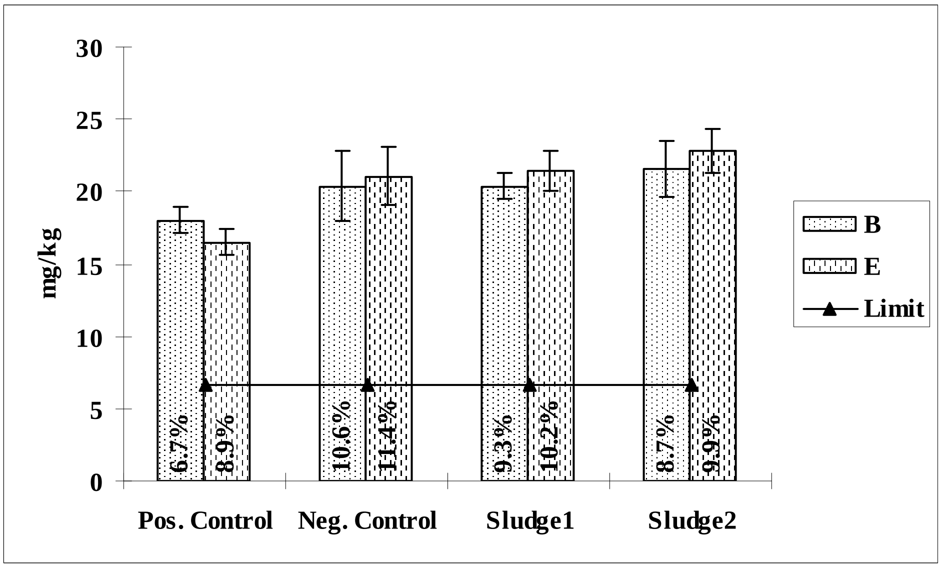

Figure 8.9. Total Cu concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

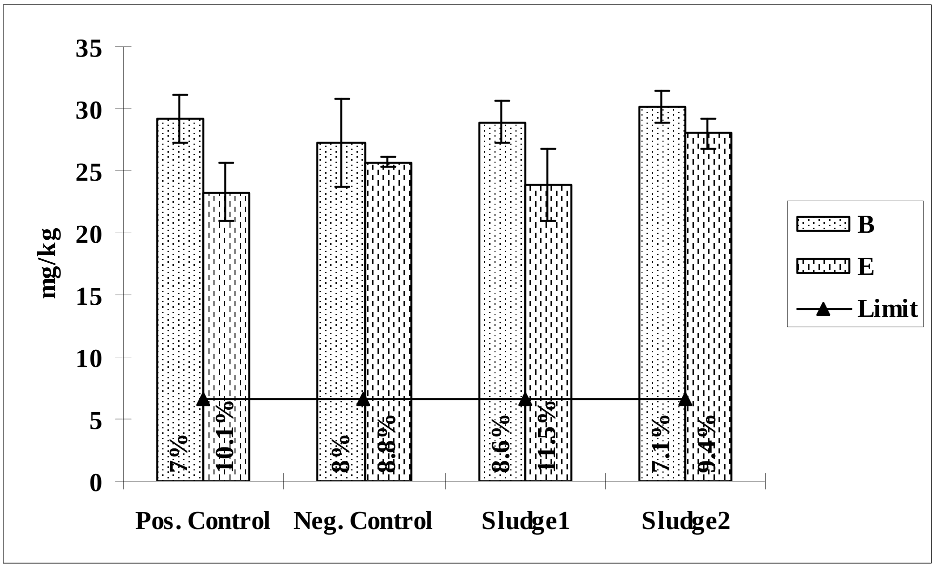

Figure 8.10. Total Cu concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

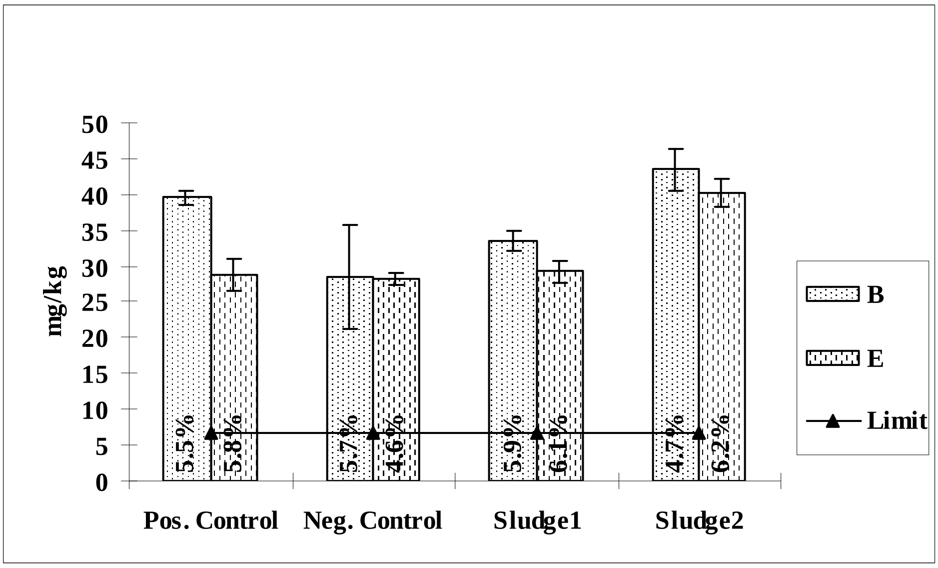

Figure 8.11. Total Cu concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

Cu concentrations in seedling tissue (Fig. 8.12) did not reach phytotoxic levels of 20 - 100 mg kg^{-1} (Smith, 1996). The *f*-values for Cu in the sunflower seedlings were slightly higher than the values reported for most other crops (Korentajer, 1991) but were consistent between the different soils and treatments. Again this indicates no increased risk from the sludge treatments compared to the control treatments. Slightly higher uptake of Cu in the seedling tissue occurred in the sludge applied loamy soil and this could be ascribed to the low pH of the soil (as discussed earlier for the other metals).

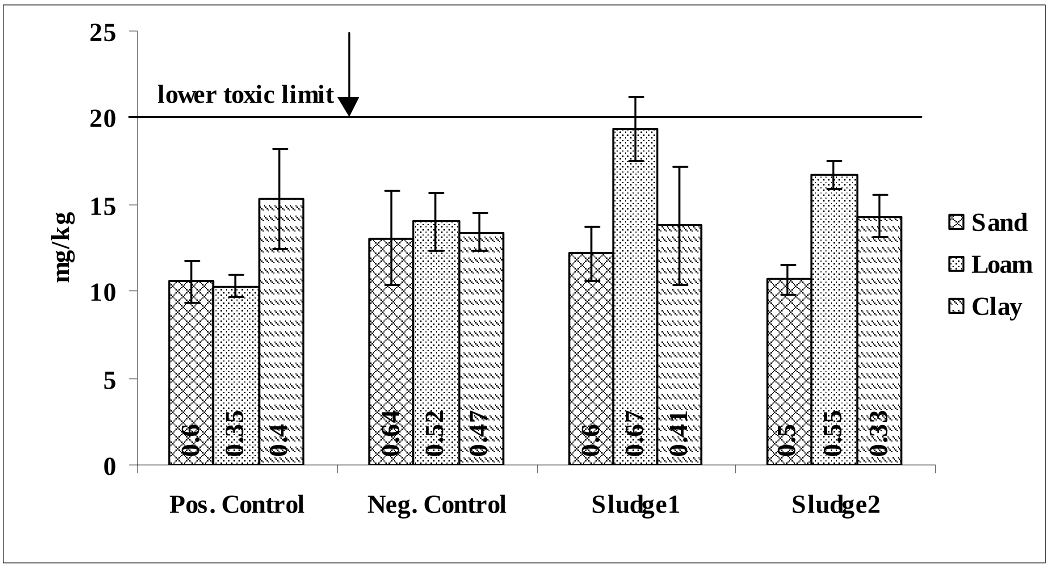

Figure 8.12.Total Cu concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

Lead

Total Pb concentrations exceeded guideline limits in the three soils (Figs 8.13, 8.14 and 8.15) before and after the 28 d of growth in the greenhouse. The levels remained relatively constant except for an increase in the Sludge 2 applied soils towards the end of the trial.

Uptake of Pb in seedling tissue was low (Fig. 8.16) and did not reach phytotoxic levels of 30 - 300 mg kg^{-1} (Smith, 1996). The *f*-values for Pb uptake by the sunflower seedlings in the different soils were relatively low for the loamy soil but slightly higher for the sandy and clay soils when compared to values for other crops (Korentajer, 1991).

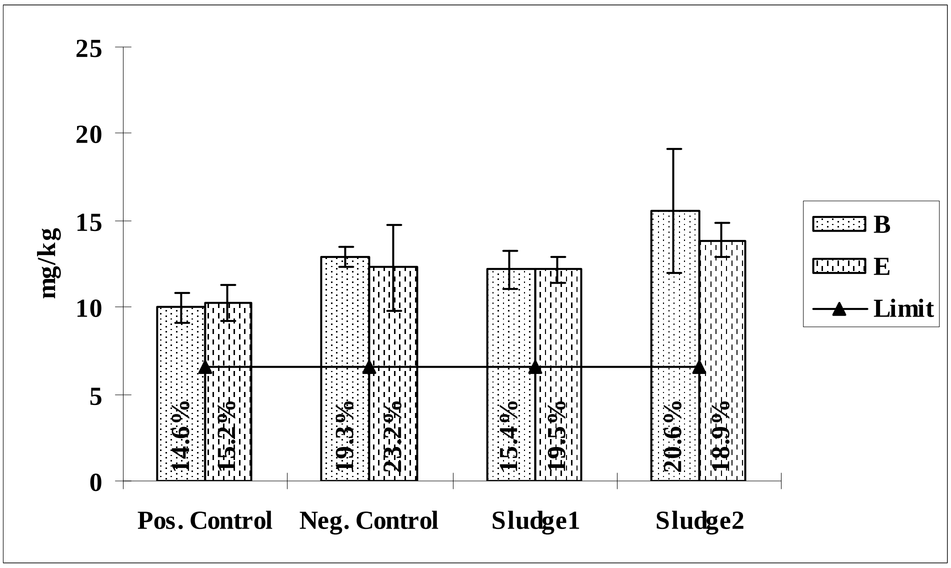

Figure 8.13. Total Pb concentrations in sandy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

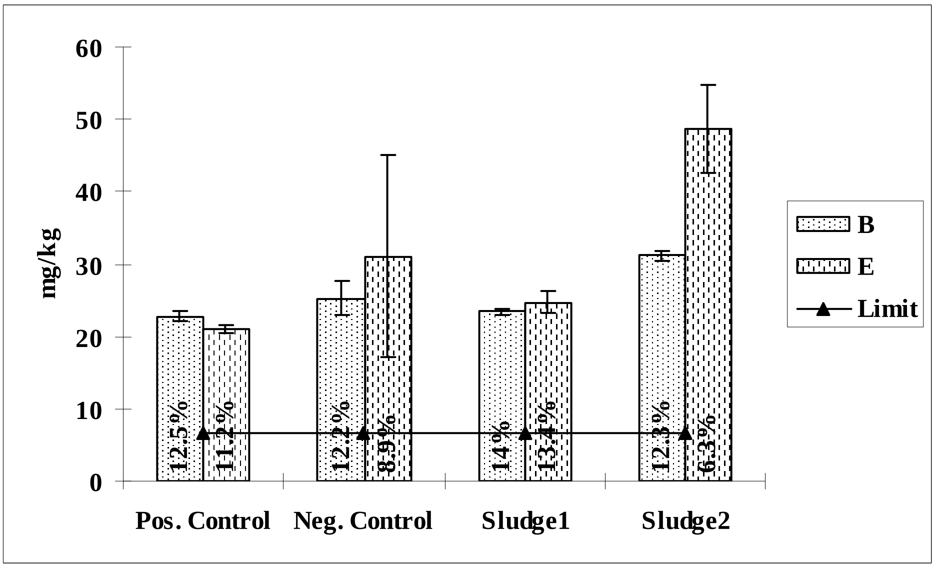

Figure 8.14. Total Pb concentrations in loamy soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

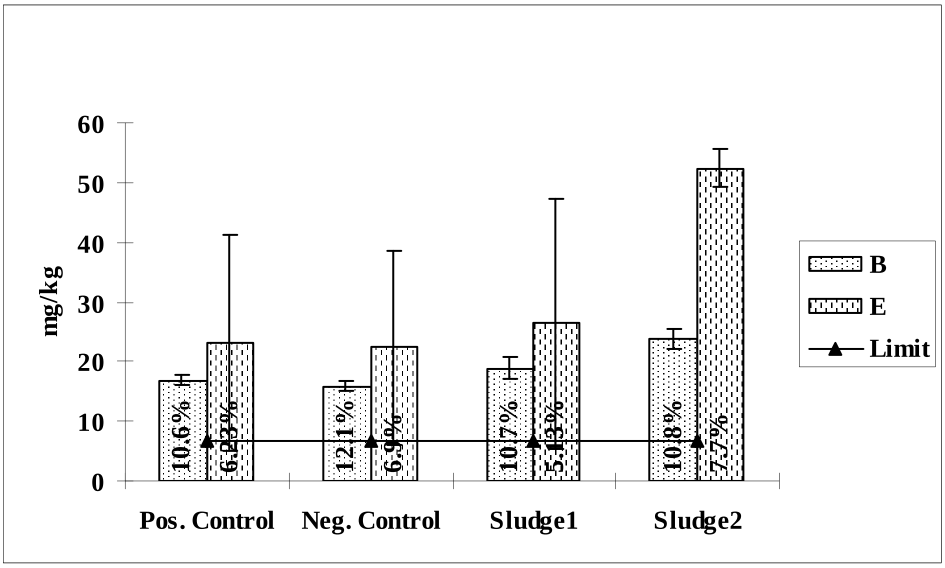

Figure 8.15. Total Pb concentrations in clayey soil at beginning (B) and end (E) of experiment compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

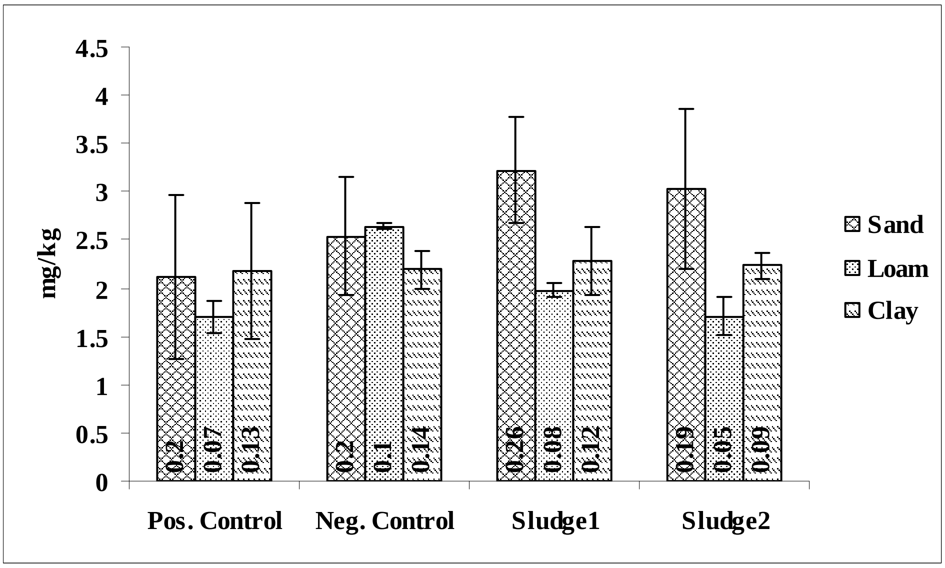

Figure 8.16. Total Pb concentrations in oats seedling tissue in three soil types. *f* factor indicated inside vertical bars. (I = STD).

Yield

Table 8.2 presents the percentage emergence of the sunflower seedlings and Table 8.3 presents the growth data of the seedlings. Statistical analyses showed that there was no difference between treatments within a soil or between soils within a treatment. An interesting aspect was between the soils in the Benoni treatment where once again the average days for emergence were less than the clay or loam soil. Although no statistical differences were found in the Olifantsfontein treatment and Positive Control between the different soils, the average days for emergence was once again much less in the loam soil than the other soils for these treatments.

Table 8.2 Percentage emergence and average days for emergence of sunflower seedlings over 14 d

Parameters	Sand		Clay		Loam	
	% Emergence	Avg. Days	% Emergence	Avg. Days	% Emergence	Avg. Days
Positive control	95^{ax}	6.26^{ax}	90^{ax}	6.25^{ax}	80^{ax}	5.8^{ax}
Benoni sludge	85^{ax}	5.92^{ax}	100^{ax}	6.65^{ax}	90^{ax}	6.7^{ax}
Olifants sludge	95^{ax}	6.35^{ax}	95^{ax}	6.25^{ax}	70^{ax}	4.94^{ax}
Negative control	90^{ax}	5.76^{ax}	100^{ax}	6.35^{ax}	85^{ax}	6.23^{ax}

Table 8.3 Average shoot length, wet mass per plant and percentage dry weight of sunflower seedlings after 28 d

Parameter	Sand			Loam			Clay		
	Shoot length	Wet mass	% Dry weight	Shoot length	Wet mass	% Dry weight	Shoot length	Wet mass	% Dry weight
Positive control	19.98^{ax}	1.758^{ax}	4.825^{ax}	16.43^{ax}	2.827^{by}	7.75^{ax}	24.35^{by}	1.895^{abx}	7.017^{ax}
Negative control	19.94^{ay}	1.403^{ax}	3.953^{ax}	16.18^{ax}	1.417^{ax}	5.54^{ax}	16.8^{axy}	1.659^{ax}	5.648^{ax}
Olifants sludge	23.76^{ax}	3.451^{bx}	6.017^{ax}	19.36^{ax}	4.314^{cx}	8.063^{ax}	25.08^{bx}	3.064^{bx}	7.807^{ax}
Benoni sludge	20.42^{ax}	1.911^{ax}	6.352^{ax}	20.8^{ax}	2.131^{abx}	6.181^{ax}	19.31^{ax}	2.334^{abx}	6.863^{ax}

From Table 8.3 it once again becomes clear that the sunflower seedlings grew significantly better in the soil types treated with Olifantsfontein sludge. Comparing growth of seedlings between soil types for each treatment the following was found:

There was a significant difference in shoot length between seedlings that grew better in clay soil than in other two soils when inorganic fertilizer was added to the soils

In the two sludge treatments there were no significant differences between the soils

In the sandy soil and loam soil the Olifantsfontein sludge treatment's biomass per plant was significantly higher than the other treatments, and in the clay soil it was significantly higher than in the Negative Control

These differences however, did not correlate with shoot length in the loam or sand but with the clay soil where shoot length was significantly longer in the Olifantsfontein sludge and Positive Control treatments.

8.5 CONCLUSIONS

In this study the effect of sewage sludge on sunflower seedling growth and yield on different soil types was monitored. The two sludges used were high in metal- and low in metal content. It was found that the heavy metal guidelines were exceeded in the soil and sludge types for Pb and Cu mostly due to high soil background levels, emphasising the conservative nature of the current S.A. guidelines when interpreted as total metal content. The Zn concentrations also exceeded guidelines and it seemed that the application of sludge-borne Zn to the soils significantly increased the total Zn concentrations in the soils as was also reported by Chlopecka (1996).

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Appendix 9

THE CULTIVATION OF FIELD-GROWN SUNFLOWER (*HELIANTHUS ANNUUS* L.) ON DIFFERENT SEWAGE SLUDGE DOSAGES

9.1 SUMMARY

Very similar results were obtained in the sunflower field trial as in the maize seedling trial (see Appendix 4 and 5).

9.2. INTRODUCTION

See Appendix 8

9.3 MATERIALS AND METHODS

Collection, treatment and analysis of dewatered sewage sludge

See Appendix 4.

Experimental layout

See Appendix 4.

Soil analyses

See Appendix 4.

Plant material analyses

See Appendix 4.

9.4 RESULTS AND DISCUSSION

Analysis of dewatered sewage sludge

See Appendix 4.

Soil analyses

See Appendix 4.

Heavy Metals

Figures 9.1, 9.2 and 9.3 indicate the Zn, Pb and Cu concentrations respectively in the soil during the experiment. Zn concentrations were slightly above guideline limits throughout the experiment except for the Sludge 2 treatment, which was considerably higher especially at the beginning. The potential available Zn fraction remained fairly constant although the total concentrations decreased somewhat to the end of the experiment, especially in the case of the Sludge 2 treatment. Cd levels were too low for determination and will therefore not be included in the discussion.

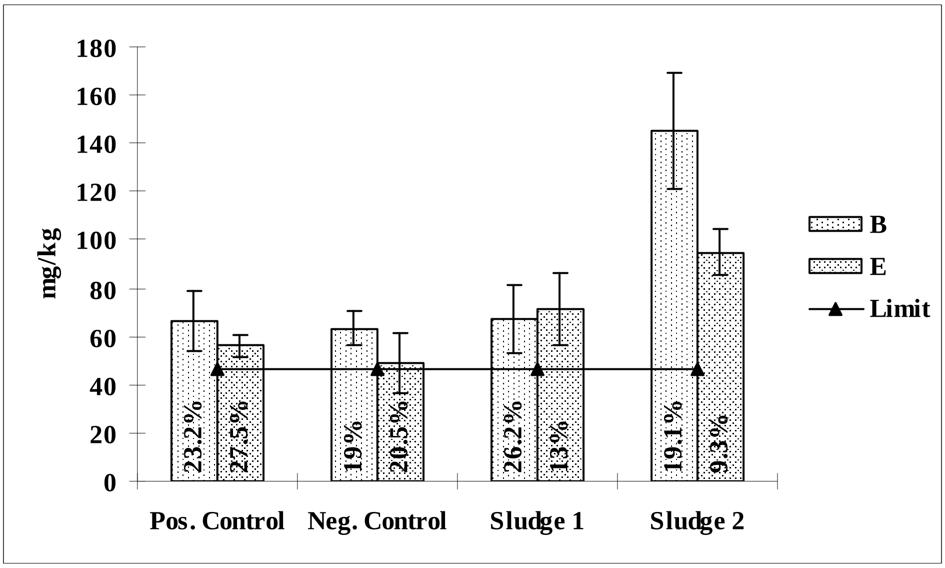

Figure 9.1. Total Zn concentrations in the loam soil from the beginning (B) to the end (E) of the experiment as compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Zn. (I = STD).

The Pb concentrations (Fig. 9.2) were higher than the guideline limits throughout all treatments. At the beginning of the experiment the Sludge 2 treatment exhibited higher levels than the other treatments as was expected. The Negative Control however exhibited higher levels than the other treatments at the end of the trial indicating the possibility of a sampling error.

The Cu concentrations (Fig. 9.3) in the soil exceeded the guideline limits by several factors but were similar throughout the treatments. In all the treatments the total concentration decreased from the beginning to the end of the trial. The potentially available Cu remained fairly constant.

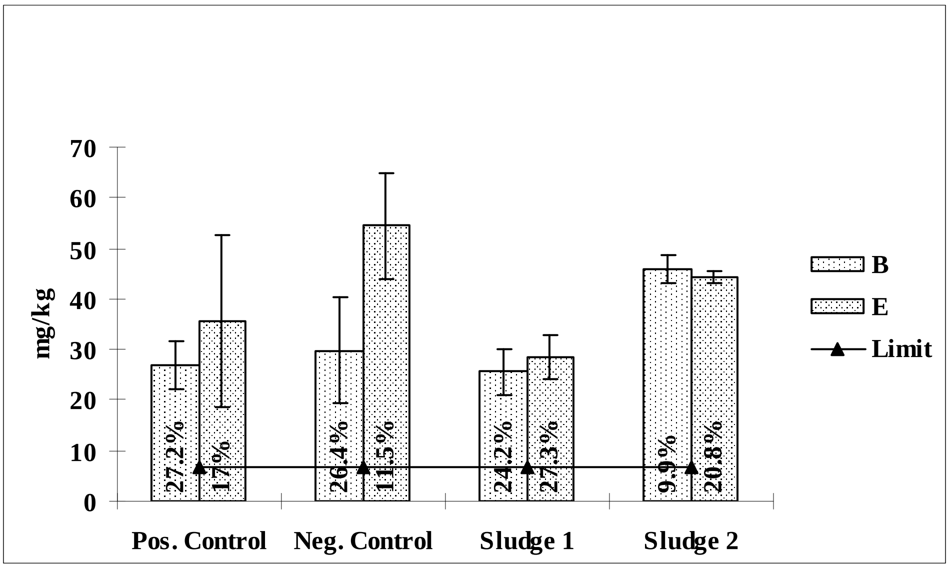

Figure 9.2. Total Pb concentrations in the loam soil from the beginning (B) to the end (E) of the experiment as compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Pb. (I = STD).

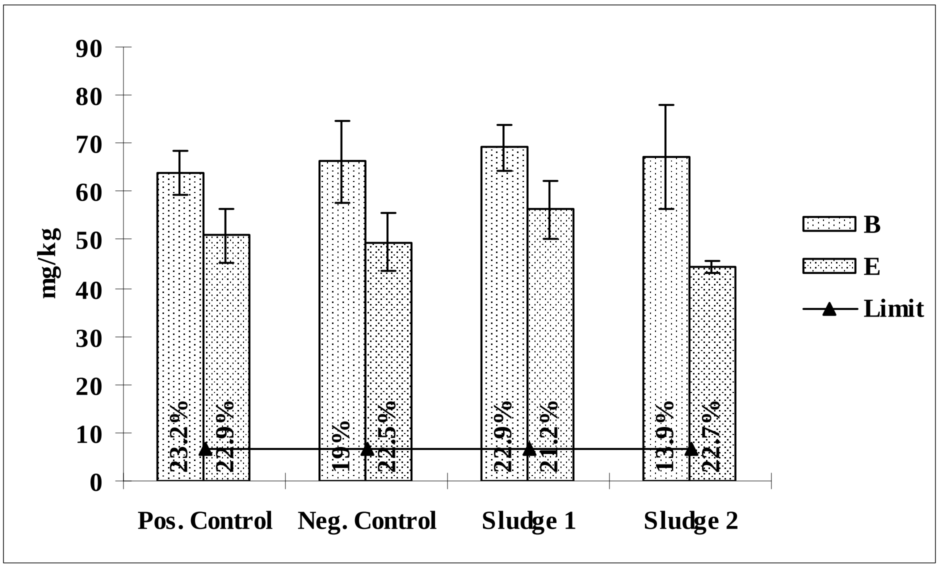

Figure 9.3. Total Cu concentrations in the loam soil from the beginning (B) to the end (E) of the experiment as compared to guidelines (WRC, 1997). Percentages inside vertical bars indicate availability of Cu. (I = STD).

Plant material analyses

Heavy metal concentrations did not reach phytotoxic levels in the leaves of the mature oats plants (Table 9.1). Although there was an increase in the uptake of Zn, Pb and Cu in the sludge treatments, the levels were mostly of the same order of magnitude as the control treatments.

Table 9.1 Heavy metal content in the sunflower plant leaves at plant maturity

Parameter	Phytotoxic levels ($mg\ kg^{-1}$) (Smith, 1996)	+ Control	- Control	Sludge 1	Sludge 2
Zn ($mg\ kg^{-1}$)[a]	100-400	40.0	45.5	47.8	53.6
Pb ($mg\ kg^{-1}$)	?	4.4	4.0	4.6	9.3
Cu ($mg\ kg^{-1}$)	20-100	33.8	33.5	35.9	38.9

[a] $mg\ kg^{-1}$ dry weight, n=4

Metal levels in the sunflower seed (Table 9.2) also did not indicate a significant increase from the control treatments to the sludge treatments although the Sludge 1 treatment indicated higher levels than the other treatments.

Table 9.2 Heavy metal content in the sunflower seeds at plant maturity

Parameter	Phytotoxic levels ($mg\ kg^{-1}$) (Smith, 1996)	+ Control	- Control	Sludge 1	Sludge 2
Zn ($mg\ kg^{-1}$)[a]	?	73.1	61.8	93.7	69.6
Pb ($mg\ kg^{-1}$)	?	3.0	2.6	4.0	2.5
Cu ($mg\ kg^{-1}$)	?	23.7	23.1	30.1	23.7

[a] $mg\ kg^{-1}$ dry weight, n=4

Yield

Due to extensive bird damage it was not possible to accurately determine the seed yield of the sunflowers.

9.5 CONCLUSIONS

The metal levels were high in the soil but only the increased Zn levels could be attributed to sludge application. Although all metals exceeded guideline limits (even in the control treatments) these levels did not lead to excessive metal uptake by the plants.

Refer to Appendix 4, 5, and 7 for a more complete explanation concerning the guideline levels.

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Smith, S.R. 1996. Agricultural Recycling of Sewage Sludge and the Environment. Biddles Ltd., Guildford.

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Appendix 10

PLANT-SOIL INTERACTIONS OF SLUDGE-BORNE HEAVY METALS AND THE EFFECT ON SOYBEAN SEEDLING GROWTH

10.1 SUMMARY

The long-term benefits of application of sewage sludge to land are frequently limited by potentially harmful elements such as heavy metals. A greenhouse experiment was done as part of a greater research project to characterise plant-soil interactions of the main sludge-borne heavy metals (Pb, Cd, Zn and Cu) to three soil types (clayey, loamy and sandy) on soybean seedlings. The experiment was a randomised block design with 3 replicates of 3 pots and 3 treatments per soil type, namely, addition of sludge at 8 $t_{dry\ mass}$ ha^{-1}, inorganic fertilizer application as recommended (Positive Control) and soil unamended (Negative Control). Soybean seedlings were harvested after 8 weeks. Wet and dry mass of the foliage was determined and analysed for heavy metal content. Wet mass of the seedlings grown in sludge-amended soil was significantly greater than the Negative and Positive Controls in all three soil types except for the Negative Control in the loamy soil. There was no increase in Cd (except in the Negative Control treatment) and Pb levels in the soybean seedlings grown in all the soil types. There was only a slight increase in Cu and Zn levels in the soybean seedlings when compared to the controls.

10.2 INTRODUCTION

Sewage sludge acts as a soil conditioner to facilitate nutrient transport, increase water retention and improve soil tilth (Ekama, 1993). The major plant nutrients (nitrogen, phosphorus and potassium) and micronutrients (Cu, Zn, Mn and B) are not removed substantially during the sludge processing and therefore may improve soil nutritional status after sludge amendment (Ekama, 1993). The long-term benefits of the application of sewage sludge to land are, however, frequently limited by potentially harmful elements such as heavy metals, in particular Cd, Cu, Zn and Pb (Schmidt, 1997). Previous research done by Christodoulakis & Margaris (1996), Snyman *et al*. (1998) and Henning *et al*. (2001) demonstrated the beneficial agricultural utilisation of sewage sludge on the cultivation of maize. The two last-mentioned authors also studied the heavy metal contamination risk of maize grown in greenhouse studies. In both studies no negative effects of heavy metal contamination in plant parts of maize seedlings could be proven.

The follow-up crop to maize in the East Rand is often a legume such as sugar beans or soybeans. The importance of legumes in maintaining soil fertility is well known and there have been several conflicting reports on the effects of heavy metals on their microsymbiont (*Rhizobium* sp.). As part of a greater research program, the research reported here outlines the follow-up to the maize greenhouse experiments to determine the effect sludge soil applications have on soybean growth under greenhouse conditions. Heavy metal concentrations (total and potentially bioavailable) were monitored in the sludge and soil to characterise plant-soil interactions of the sludge-borne metals on different soil types.

10.3 MATERIALS AND METHODS

Dewatered sewage sludge

Dewatered sewage sludge was collected at the ERWAT Olifantsfontein Water Care Works and analysed for moisture, potentially bioavailable heavy metals (Amm. EDTA method) and total heavy metal concentration (EPA3050 method) of 4 heavy metals (Cd, Cu, Pb and Zn) [Institute for Soil, Climate and Water (ISCW), Agricultural Research Council (ARC), Pretoria].

Soil preparation and analyses

Three different soil types, a Bonnheim-clayey soil, Longlands loamy soil and Hutton red sandy loam, were collected in the broader Gauteng area, South Africa, as identified and profiled by the Institute for Soil, Climate and Water, Agricultural Research Council (ISCW, ARC), Pretoria. The soil was dried for 5 d and the soil and sludge were sifted through a 5 mm sieve. The dried sludge (8 t_{dry} ha^{-1}) and inorganic fertilizer (recommended rate, Positive Control) were added to the different soil types. The treated soil samples were used to fill 54 plastic containers, which were lined with Whatman No.1 filter paper to cover the drainage holes. An additional 9 containers were filled with untreated soil (Negative Control). Soybean (*Glycine sojae* L.) seeds were planted, 5 per pot. Soil samples were analysed at the start of the experiment after sludge and fertilizer application and after 60 d. The soil samples, which were analysed at the end of the experiment, consisted of soil samples including the soybean roots. Samples were analysed for heavy metal content (total and bioavailable) of four metals (Cd, Cu, Pb and Zn) (ISCW, ARC).

Plant material analyses

Soybean seedlings were harvested 60 d after planting in the greenhouse. The above-ground parts were cut off at soil level and the wet mass was determined. The shoot length of the seedlings was measured. The foliage was then dried at 60ºC for 48 h to determine its dry mass. Above-ground dried plant samples were analysed for total heavy metal content (ISCW, ARC). The accumulation of metals in the plant parts when taken up from the soil was determined as the *f* factor, also known as the transfer coefficient (Smith, 1996), using the following formula:

$F = [M]p / [M]s$

Where:

[M] = metal concentration

p, s subscripts refer to plant and soil respectively.

Statistical analysis

There were three replicates of three pots containing 5 seedlings for each treatment. ANOVA statistical analysis was used to determine statistical differences among treatments and soil types and seedling growth.

10.4 RESULTS AND DISCUSSION

Analysis of dewatered sewage sludge

The pH of the sludge was 6.06. The concentration of the heavy metals and their percentage availability (in brackets) in the sludge was Zn 4371.2 mg kg^{-1} (27.8%), Pb 251.34 mg kg^{-1} (8.7%), Cu 542.1 mg kg^{-1} (6.1%) and Cd 52.63 mg kg^{-1} (11.2%).

Soil and plant material analyses

Soil pH

Background pH values of the sandy, loamy and clayey soils were approximately 6.14, 5.6 and 8.04 and 6.32, 5.8 and 8.03 after addition of sludge. It has been found that in soils where pH was above 6.0, there was a significant increase in shoot weight and total shoot N of alfalfa and red clover with sludge addition (Ibekwe *et al*., 1995). These authors found that nodulation was reduced, but not always completely eliminated in all low pH treatments. In our experiment the loam soil had a pH below 6.0 but almost reached pH 6 after the addition of sludge.

Zinc

The availability of the Zn in sludge-amended soil was more or less the same in the loamy soil compared to the sandy soil but was far less in the clayey soil (Figs 10.1, 10.2 and 10.3). According to Alloway (1995), this is due to the adsorption of the Zn to the clay particles. The two-fold increase in the availability of the Zn in the sludge applied clayey soil from the beginning to the end of the experiment may be due to mineralisation of the sludge-borne Zn from an organic form to an inorganic form. Zinc was also more available in the Positive and Negative Controls of the loamy soil when compared with the same treatments of the sandy and clayey soils (Figs 10.1, 10.2 and 10.3). The difficulty in obtaining representative samples in order to determine the levels of Zn in the three soil types is evident as often levels increased from the beginning to the end of the experiment instead of an expected decrease. Figure 10.4 illustrates the total Zn concentrations in soybean seedlings after 60 d of growth. There was only a slight increase in the Zn levels in the soybean seedlings in the sludge treatment when compared to the controls except for the Positive Control in the loamy soil. Korentajer (1991) gives the normal transfer coefficient (*f* factor) of Zn in cereals, celery and leeks as between 0.5 and 1.0 and of maize between 1.0 and 2.0. Unfortunately no values are given for any legume species. However, all the *f* factor values of the soybeans in this study were below 1.0 except for the Positive Control and sludge treatment of the sandy soil (Fig. 10.4). Zn concentrations never reached the phytotoxic levels of 100-400 mg kg^{-1} reported by Smith (1996).

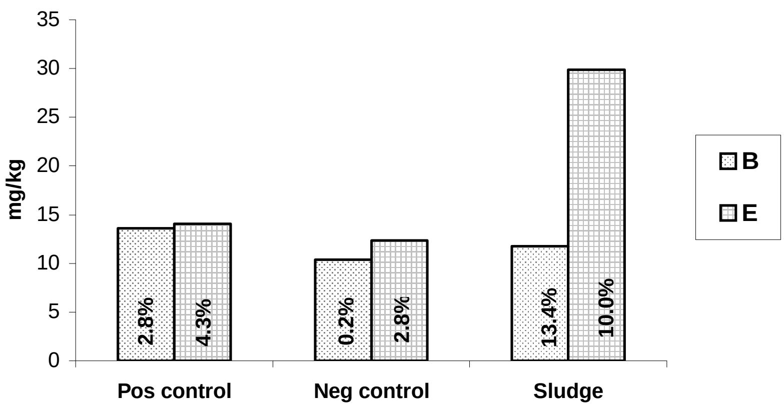

Figure 10.1. Zinc concentrations in sandy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

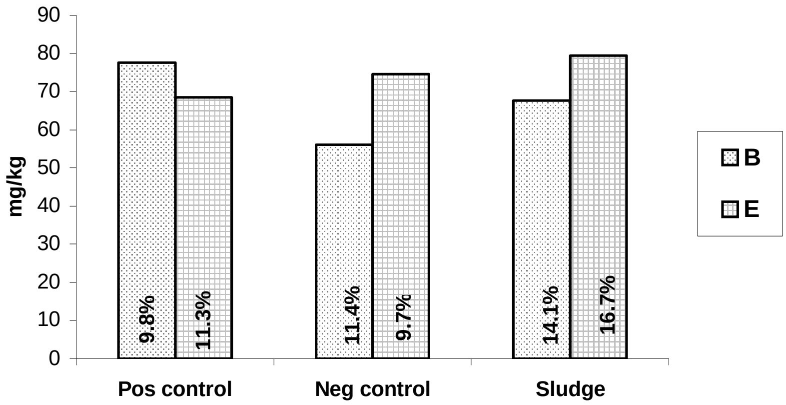

Figure 10.2. Zinc concentrations in loamy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

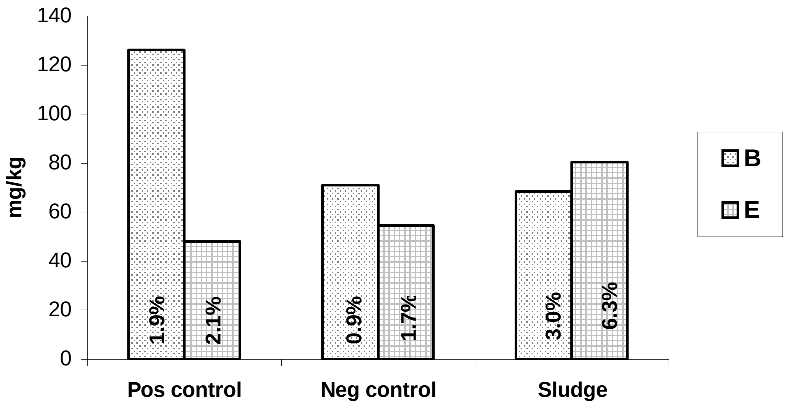

Figure 10.3. Zinc concentrations in clayey soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

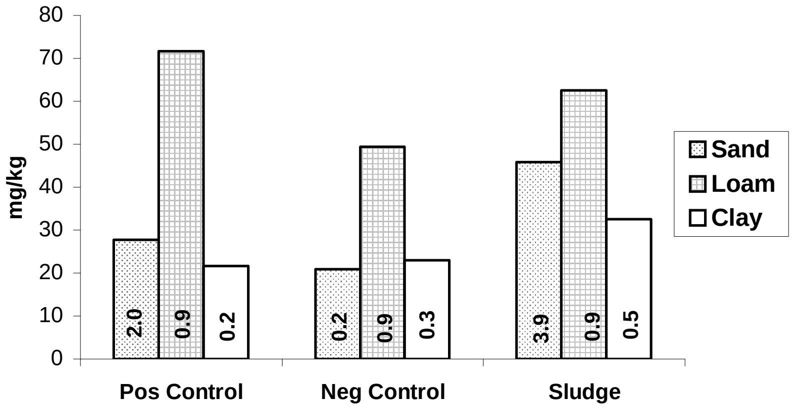

Figure 10.4. Total Zn concentrations in soybean seedling tissue in three soil types. *f* factor indicated within columns.

Lead

Lead concentrations remained either constant or decreased slightly from the beginning to the end of the experiment in all three soil types and treatments (Figs 10.5, 10.6 and 10.7). This indicates the immobility of Pb in soils as was also found by Henning *et al*. (2001). Poor sample homogeneity was evident in the sandy soil since the Negative Control showed higher Pb concentration than the Positive Control treatment. However, it must be kept in mind that the values (mg kg^{-1}) are very low for the soil treatments when compared to the other soil types. The availability of Pb at the beginning and end of the experiment of all the loamy soil treatments was greater than that of the equivalent sandy and clayey soil treatments. The Pb was least available in the clayey soil (Fig. 10.7). Levels of 30-300 mg kg^{-1} are considered phytotoxic (Smith, 1996) but were not exceeded in this experiment. There was no increase in Pb levels in the soybean seedlings grown in the sludge treatment when compared to the controls (Fig. 10.8). Soybean, although not recorded amongst the *f* values of Pb of the plant species listed by Korentajer (1991), greatly exceeded the f values (0.01-0.05) recorded by this author. As was found by Henning *et al*. (2001) studying heavy metal uptake by maize seedlings, higher transfer coefficient of Pb in soybean seedling tissue occurred in the sandy soil.

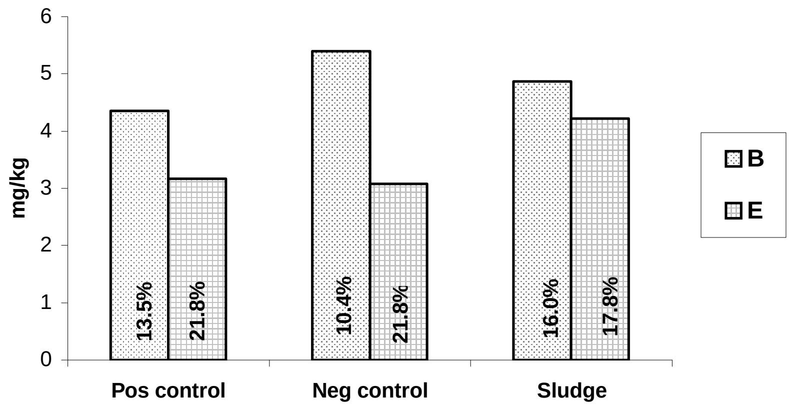

Figure 10.5. Lead concentrations in sandy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

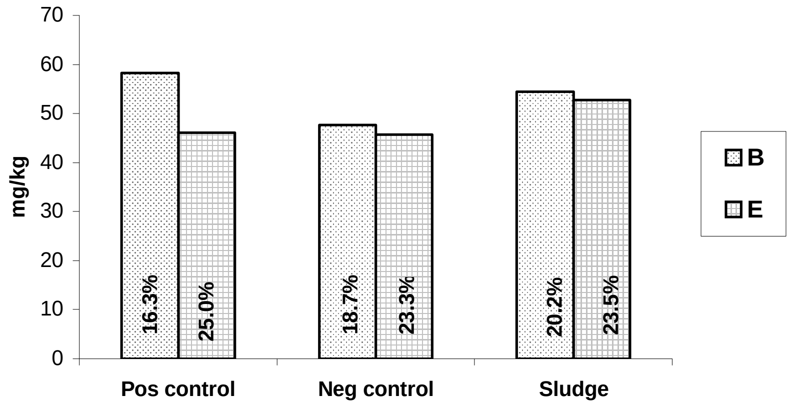

Figure 10.6. Lead concentrations in loamy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

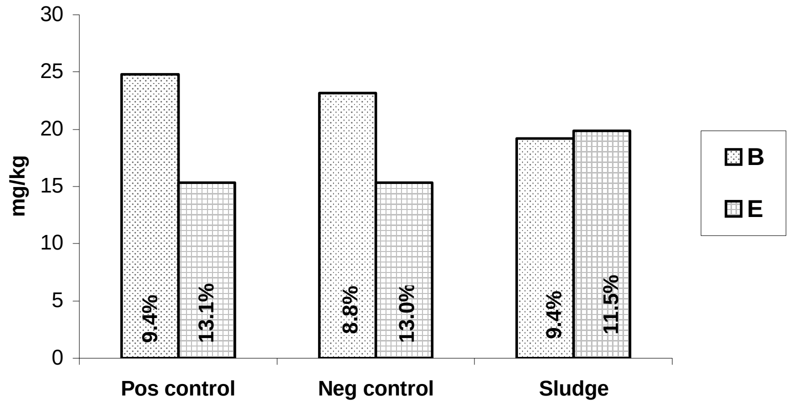

Figure 10.7. Lead concentrations in clayey soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

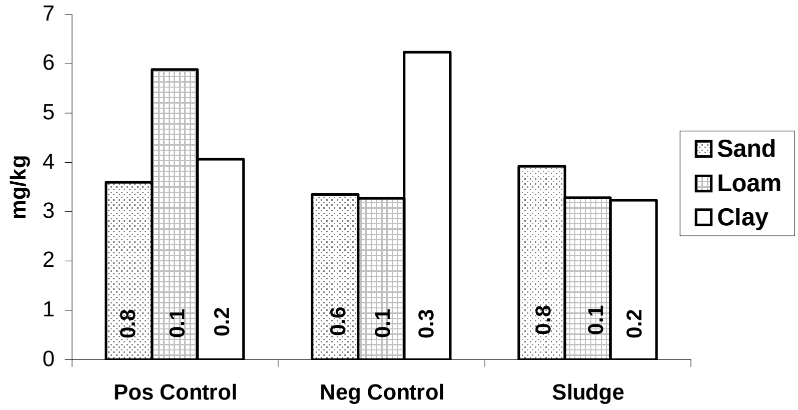

Figure 10.8. Total Pb concentrations in soybean seedling tissue in three soil types. *f* factor indicated within columns.

Copper

The concentration of Cu was very low in all treatments of the sandy soils but was similar in the loamy and clayey soils (Figs 10.9, 10.10 and 10.11). Copper was more available in all treatments of the sandy soil than in the equivalent treatments of the loamy and clayey soils. There was only a slight increase in Cu levels in the soybean seedlings grown in sludge treatment when compared to the controls (Fig. 10.12). Reddy *et al*. (1989) determined the uptake of Cu by soybeans grown on sludge-amended sandy loam soil. They found that although the Cu concentration in the soybean tissue increased under sludge treatment, the change in concentrations was relatively low. They found the same trend with Zn concentrations. Cu concentrations in seedling tissue in our experiment did not reach the toxic levels of 20-100 mg kg^{-1} (Smith, 1996) (Fig. 10.12). The *f* values for celery, cereals, leeks and maize range from 0.01-0.05 (Korentajer, 1991). Therefore, it is likely that the transfer coefficient of all the soil types in the soybean experiment are rather high, especially in the sandy soils (Fig. 10.12).

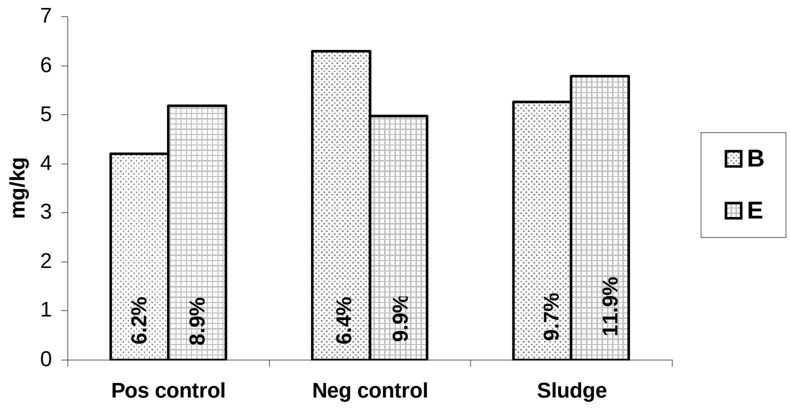

Figure 10.9. Copper concentrations in sandy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

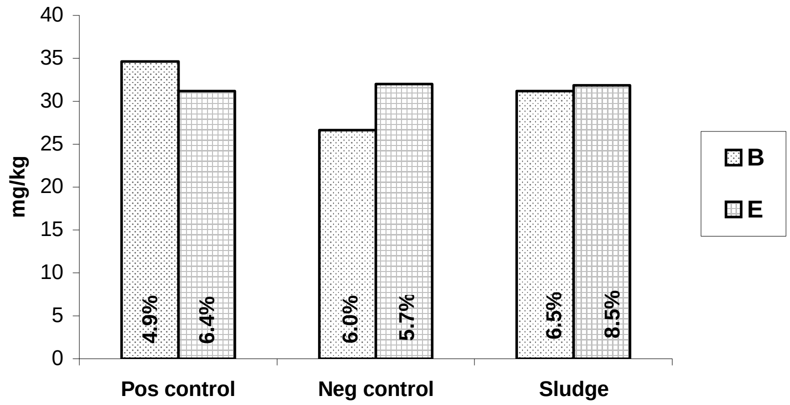

Figure 10.10. Copper concentrations in loamy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

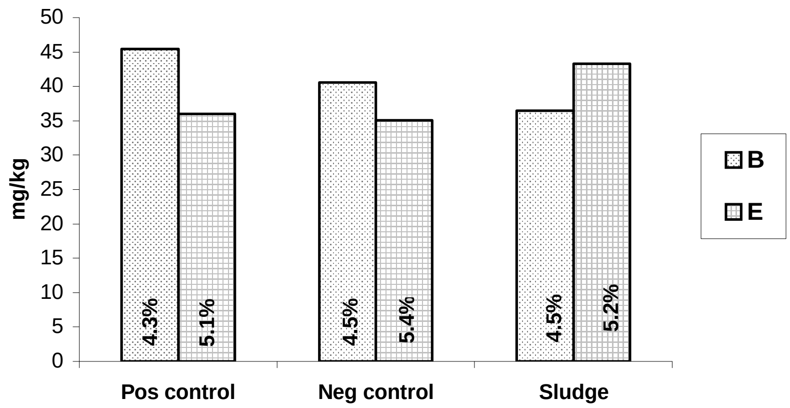

Figure 10.11. Copper concentrations in clayey soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

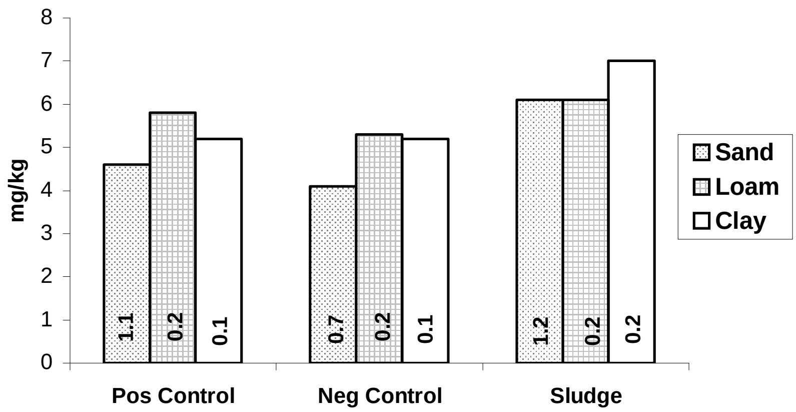

Figure 10.12. Total Cu concentrations in soybean seedling tissue in three soil types. *f* factor indicated within columns.

Cadmium

Poor sample homogeneity was again evident as levels of Cd increased from the beginning to the end in some of the treatments of all the soil types (Figs 10.13, 10.14 and 10.15). The availability of the Cd in the sludge treatment of the sandy and loamy soils decreased over the 60 d seedling growth period but increased in the clayey soil from 5.9 to 17.2%. This was also reflected in the Cd uptake by the soybean seedlings, as the transfer coefficient was lowest in the sludge treatment of the clayey soil (Fig. 10.16). Korentajer (1991) gives *f* values of 0.01-2.0 for a range of plants, which was not exceeded in this experiment. There was no increase in Cd levels in the seedlings in the sludge treatments when compared to the controls except in the Negative Control (Fig. 10.16). The uptake of Cd into the soybean seedlings did not reach the phytotoxic levels of 5-30 mg kg^{-1} (Smith, 1996).

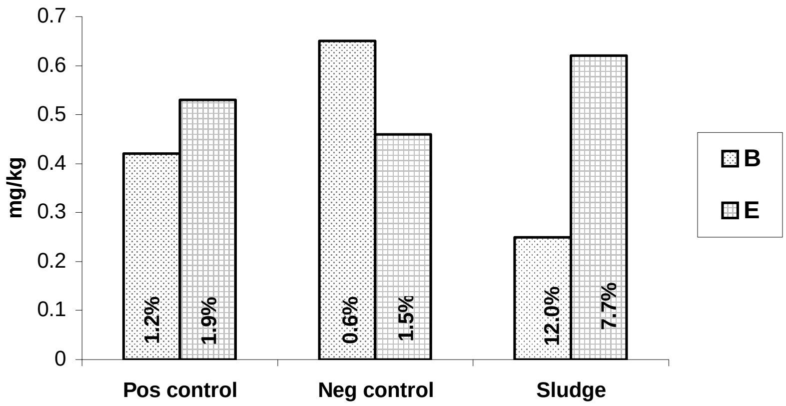

Figure 10.13. Cadmium concentrations in sandy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

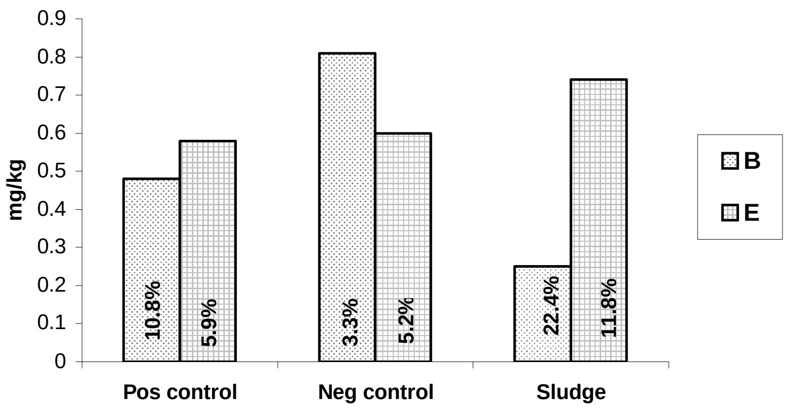

Figure 10.14. Cadmium concentrations in loamy soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

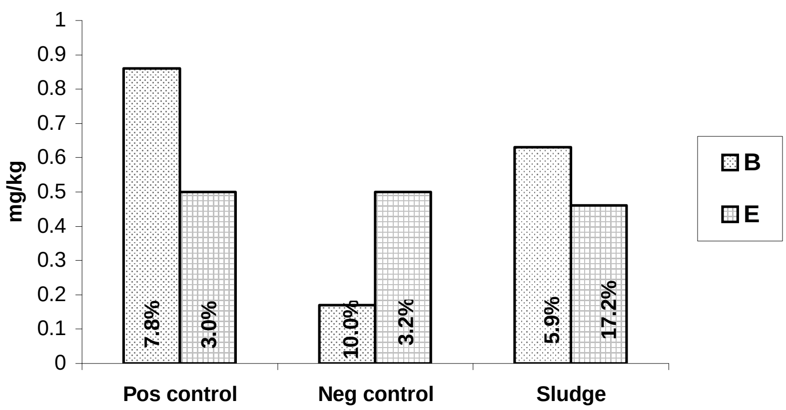

Figure 10.15. Cadmium concentrations in clayey soil at beginning (B) and end (E) of experiment. Percentages within columns indicate potential availability of the heavy metal.

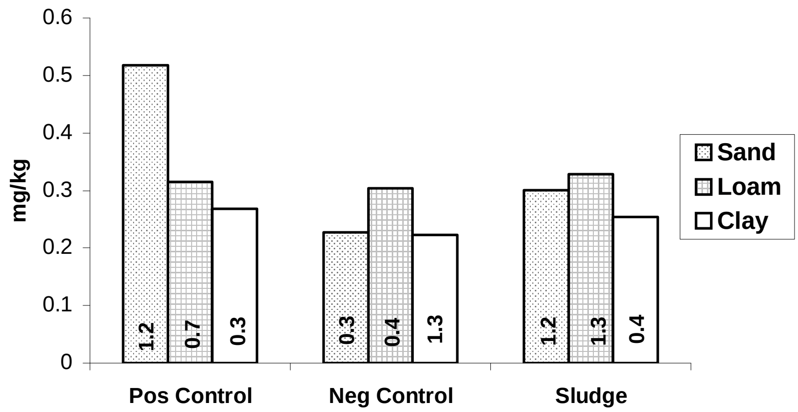

Figure 10.16. Total Cd concentrations in soybean seedling tissue in three soil types. *f* factor indicated within columns.

Growth of plants

There were no significant differences in mean seedling shoot length among the three treatments or soil types (Table 10.1). Wet mass of the seedlings grown in sludge-amended soil was significantly greater than the Negative and Positive Controls in all three soil types except for the Negative Control in the loamy soil. The dry mass showed a similar trend although the sludge treatment did not differ significantly from the Negative Control in the loamy soil and from the Positive Control in the sandy soil. There were no significant differences in the dry mass among the treatments in the clayey soil. The wet and dry mass of the Positive Control seedlings grown in clayey soil was significantly higher than those grown in sandy or loamy soils. There were no differences in the wet and dry mass of the Negative Control seedlings among the three soil types. The dry mass of seedlings was significantly higher in the sludge treatment when grown in clayey and sandy than in loamy soil. A similar trend was observed with the wet mass, however, seedlings grown in loamy soil did not have a significantly higher mass than those grown in clayey soil. Reddy *et al*. (1989) found that the addition of sludge to a sandy loam soil significantly increased grain yield of soybeans.

Table 10.1 Average shoot length (cm), wet and dry mass (g) of soybean seedlings after 60 d of growth

Parameter	Clayey			Loamy			Sandy		
	Shoot length*	Wet mass**	Dry mass**	Shoot length	Wet mass	Dry mass	Shoot length	Wet mass	Dry mass
Positive control	18.9^{ax}	6.9^{ay}	1.21^{**ay}	17.8^{ax}	4.4^{ax}	0.34^{ax}	19.0^{ax}	4.24^{ax}	0.55^{abx}
Negative control	18.2^{ax}	6.3^{ax}	0.87^{ax}	19.2^{ax}	5.4^{abx}	0.82^{bx}	18.0^{ax}	5.7^{ax}	0.91^{ax}
Sludge	21.7^{ax}	9.8^{bxy}	1.67^{ay}	19.5^{ax}	7.8^{bx}	0.87^{bx}	21.7^{ax}	10.4^{by}	1.77^{by}

*Each value is a mean value per plant of 3 replicates of 12 plants

**Each value is a mean value of 3 replicates of 12 plants

Values within a row not followed by the same letter (x,y or z), or within a column (a or b) are significantly different (*P=0.05*) according to Duncan's multiple range test.

10.5 CONCLUSION

A problem encountered during this study was the difficulty in obtaining a representative sample in sludge-amended soils. Although several replicates were used, discrepancies in concentrations of the heavy metals in the soil still occurred. No phytotoxic effects could be proven, because phytotoxic levels were never exceeded in the soybean seedlings. It is evident from the experiments that the sludge positively affected the growth of the soybean seedlings when compared with the controls of most of the soil types. It thus also appears that the sludge applied to the soil may not have had any negative effect on the rhizobia associated with the roots of the soybean seedlings. However, this aspect requires further investigation. The results in this paper only represent seedling growth over a 60 d period and cannot be extrapolated to what would occur in the field. A field-scale experiment is required to corroborate the results presented here.

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Appendix 11

COMPARISON OF MINERALISATION RATES OF SLUDGE AND COMMERCIAL FERTILIZER NITROGEN SOURCES

11.1 INTRODUCTION

N losses in agricultural soils take place through crop removal, leaching, surface run-off, gaseous losses through NH_3 volatilisation, denitrification and erosion (Jarvis *et al.*, 1996). Loss of N from agricultural soils is of concern due to the possible NO_3^- leaching from cultivated soils that is responsible for increasing concentrations of NO_3^- in surface and ground waters (Cheshire *et al.*, 1999).

Mineralisation of organic N is an important consideration when determining the appropriate rates for sludge application to cropland (Parker & Sommers, 1983). The processes of mineralisation and immobilisation are central to the control of the N flows within agricultural cycles (Jarvis *et al.*, 1996). Mineralisation of sewage sludge depends on the sludge characteristics (nature and abundance), environmental factors (temperature and humidity), and soil conditions (pH, accessibility for soil fauna, soil texture and structure), (Parker & Sommers, 1983; Janssen, 1996; Leiros *et al.*, 1999; Trindade *et al.*, 2001).

In contrast to the inorganic N content, which is immediately available to plants, and easy to quantify, the N supply from organic forms depends on the mineralisation rate, which is difficult to quantify. Quantification of organic N mineralisation rates is the first step necessary to improve the N use efficiency and to reduce the losses to the environment (Trindade *et al.*, 2001).

In South Africa strict legislation exists on the maximum allowable sludge that may be applied to the soil per annum. This however based on the information obtained from overseas literature and experience. This seems to be impractical and it would be worthwhile to look at the maximum rates again. In order to quantify the N added to soil through sludge application, it is not only necessary to quantify the total N applied through the sludge but also the N release rate. This has to be done with the mineralisation rate and N release from sewage sludge using an incubation trial.

The aim of this study is therefore to quantify the N-mineralisation rate of sewage sludge and to compare it to different levels of sludge application, and to the release of inorganic N from commercial fertilizer such as limestone ammonium nitrate (LAN).

11.2 MATERIALS AND METHODS

An incubation trial was a done in a laboratory, under constant temperature and moisture content. Soil (50 g) was placed in plastic jars and pre-incubated before the different treatments were applied. Treatments consisted of a set of different quantities of sewage sludge and commercial fertilizer applications. Each treatment was replicated three times. The sludge was applied at the equivalent of 5, 10 and 20 t ha^{-1} dried sewage sludge per 20 cm cultivation depth. The commercial fertilizer treatments were calculated based on the assumption that only 30% of the total N content of the sludge would be available in the short-term. According to the ‘Permissible Utilisation and Disposal of Sewage Sludge’ (WRC, 1997), 30% of the total N in the sludge becomes plant available in the first year.

Soil

A dark red sandy clay loam topsoil (0-30 cm) from the Hatfield Experimental farm, University of Pretoria (25° 45’S 28° 16’ E) from the same site that was later used for a field trial was used. The soil was dried overnight at 40°C and passed through a 2 mm sieve. Some of the chemical and physical properties of the soils are presented in Table 11.1.

Table 11.1 Some physical and chemical properties of the soil used in the trial

Parameter	Value
pH (H_2O)	6.4
Electric conductivity (mS m^{-1})	8.3
NO_3^--N (mg kg^{-1})	3.55
NH_4^+-N (mg kg^{-1})	1.37
C %	0.63
Textural classes	
% Clay	26
% Silt	8.2
% Sand	65.8

Sewage Sludge

The sludge that was used for the incubation trial was obtained from Olifantsfontein, a sewage plant of ERWAT. The sludge was air dried, ground and sieved through a 2 mm sieve. Generally sludge from Olifantsfontein, has very low concentrations of heavy metals, because it is from a domestic origin, and contains very little industrial waste. Some of the chemical properties of the specific sludge are presented in Table 11.2.

Table 11.2 C and N quantities of sludge

Parameter	Value
Total N	3.14%
Total C	3.63%
C:N	1.15

According to Hansen & Djurhuus (1978), during mineralisation, N-release takes place when the C:N ratio of less than approximately 25, whereas immobilisation of N takes place at a C:N ration greater than approximately 35. It can therefore be expected that due to the relatively low C:N ratio of the sludge used, low N-immobilisation would occur.

Field Capacity of the Soil

According to the literature, incubation studies are best done when the soil is at field capacity. To determine the field capacity (FC) of the soil, three open-end glass cylinders (40 cm $\times$ 3.5 cm) were used. One end of the cylinder was covered with filter paper, and filled with 10 cm of soil. This layer of soil was compacted, until it resembles the density of the natural soil. This process was repeated until the cylinder was two thirds full. Water (25 ml) was added to each cylinder and a rubber stopper was placed loosely on top, to prevent evaporation. The cylinders were left for 24 h, until the wetting front stopped moving. Samples were taken from the middle of each cylinder, weighed and dried at 100 °C for 24 h, and weighed again. The mean value of the water content (15.29%) is assumed to be FC.

Pre-incubation Trial

A pre-incubation step was carried out to ensure that the microbial population adapt to the experiment conditions. The soil was incubated at 100% FC for one week as follow: Plastic jars (300 ml) were filled with 50 g of the prepared soil and 7.65 ml of water was added to each jar, to bring the soil to 100% FC. The samples were placed in the incubator, at a constant temperature of 21°C. Every second day the jars were opened for the soil to aerate, and deionised water was added to maintain FC. This was done for 7 d and on day 7, triplicate samples were taken to be analysed for NH_4^+ and NO_3^-, which were used as the background information for the trial, and were assigned 'Day 0'.

Incubation

After the 7 d pre-incubation, on 'Day 0', dried sewage sludge and commercial fertilizer were applied according to the different treatment levels, mixed well, brought to FC and put into the incubation oven. The same procedure of aeration and water addition, which was carried out during the pre-incubation period, was repeated during the incubation trial.

Triplicate samples were taken at day 1, 3, 7, 14, 28, 42 and 63, and were analysed for NH_4^+ and NO_3^-.

Laboratory Analysis

A 1M KCl solution (100 ml) was added to the 50 g soil samples and shaken for an hour, and filtered afterwards. From this filtrate, a 50 ml aliquot was used for the analysis of NH_4^+ and NO_3^-. To this aliquot, 20 ml of a 50% NaOH solution was added to volatilise the NH_4^+-N. The solution was distilled on a Büchi distillatory and bubbled through a boric acid solution, with a colour indicator. After distillation, the boric solution was titrated with HCl. The volume of the HCl used for the titration was used to calculate the NH_4^+-N content in the incubated soil. This value is an indication of the concentration NH_4^+ in the solution. To the remaining KCl-aliquot, Davarda alloy was added to reduce the NO_3^- to NH_4^+ and the process was repeated for NO_3^- content.

Statistical Analysis

Statistical analyses were carried out on the results by using GLM statistical procedure. Comparisons between the different treatments, the intervals, as well as treatment x interval interaction were tested. Significant differences between treatments and grouping were tested by the Tukey test.

11.3 RESULTS AND DISCUSSION

In practice it is not advisable to use a single value for the N content of sludge. During the drying process prior to sludge analysis, N losses may occur due to NH_4^+ volatilisation. The Na, K, Ca, etc. concentrations are usually unaffected during the drying process (Sommers, 1977). Every sludge source used has therefore to be considered individually because the N and therefore the total N supply differ.

Ammonium (NH_4^+)

Additions of organic material, rich in readily available energy, stimulate mineralisation of organic N and C. This stimulating effect of added energy on the microorganism activity and the subsequent increase in organic matter mineralisation is called 'priming effect' (Jansson & Persson, 1982; Kuzyakov *et al.*, 2000). The priming effect may start either immediately or very shortly after the addition of organic matter to the soil. This will usually depend on the quantity and the availability of organic N in the sludge that can be used by microorganisms as well as the complexity of the organic N molecules in the sludge (Kuzyakov *et al.*, 2000). The priming effect can be observed in Figure 11.1.

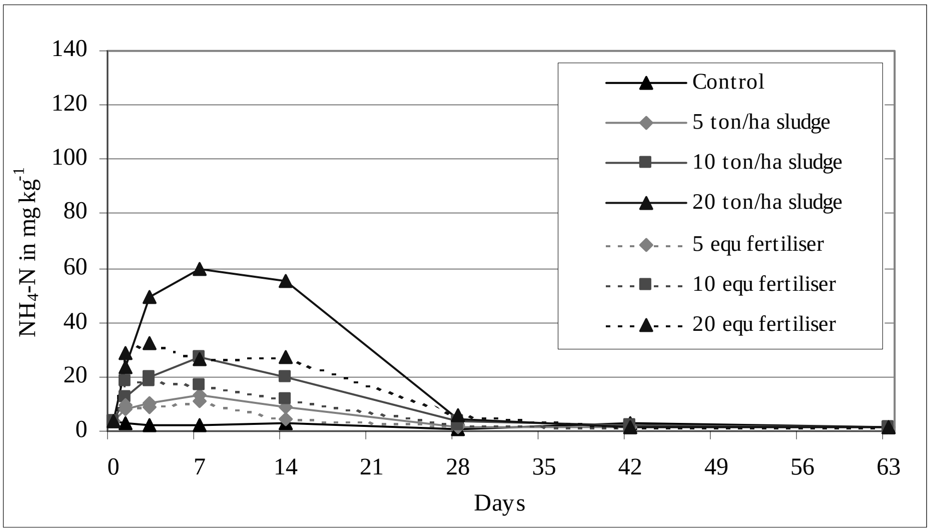


Figure 11.1. Total extractable NH_4^+-N content as a function of incubation time and differentiable N application.

Samples were taken on day 0, before the different treatments were applied, and again on day 1, 24 h after application. The sharp increase in NH_4^+-N content in the sludge treatments can partly be ascribed to the free NH_4^+-N content in the sludge, but the difference between the NH_4^+-N content of the sludge and the organic N treated soils can be ascribed to the mineralisation of organic matter.

In soils microbial activity is mostly limited by energy sources. According to studies done by Bosatta & Agren (1995), addition of inorganic fertilizer to the soil, resulted in no change in microbial activity. However, if inorganic N is available, fungi are capable of increasing their N concentration. Application of sludge though, greatly stimulates microbial activity (Jansson & Persson, 1982). The N applied in the inorganic fertilizer treatments amounts to 30% of the total N content in the corresponding sludge treatment. An increase in microbial activity because of a larger organic-N pool and readily available energy from the sludge treatments, can be possible reasons for the greater NH_4^+-N production from the sludge than from the fertilizer treatments. The total NH_4^+ concentrations continue to increase, until it peaks at day 7, after which the concentrations decrease again. After 28 d, most of the NH_4^+ was depleted and this was very similar for all the treatments.

The release of NH_4^+ only happens after the NH_4^+ requirements of the microorganisms are satisfied. The decomposition of microbial biomass therefore generally leads to the releases of NH_4^+. Most heterotrophic soil microorganisms are able to produce NH_4^+ (Coyne, 1999; Mengel & Kirkby, 2001).
The presence of NH_4^+-N in the treated soil with applied NH_4^+ or by mineralisation of organic matter can lead to different pathways in the N cycle:

It can be utilised for microbial growth

NH_3 volatilisation can occur

NH_4^+ can be utilised as an energy source by autotrophs in the nitrification process (Paul & Clark, 1989).

According to Figure 11.1 and Table 11.3, the quantity of extractable NH_4^+-N in the sludge and inorganic N applied treatments, differ significantly (P<0.0001). For all the treatments, except the control, a clear increase in NH_4^+ (priming effect) can be observed, immediately after the addition of the different treatments. The differences between the sludge and inorganic N application can mainly be attributed to the priming effect on the microbial activity. There was a significant difference between all the treatments, except between the 5 t ha^{-1} sludge and the corresponding 5 t ha^{-1} fertiliser N treatment. At this low concentration, no significant differences were obtained, probably due to the small differences in the applied inorganic N in the sludge and commercial N applications.

Table 11.3 The average total extractable NH_4^+-N for different treatments

Treatment	Total N	Available N [1]	Mean
20 t ha^{-1} sludge	628kg N ha^{-1}	188.4 kg N ha^{-1}	24.876 a
20 equ fertilizer	188.4 kg N ha^{-1}	118.4 kg N ha^{-1}	15.881 b
10 t ha^{-1} sludge	314 kg N ha^{-1}	94.2 kg N ha^{-1}	11.223 c
10 equ fertilizer	94.2 kg N ha^{-1}	94.2 kg N ha^{-1}	9.286 d
5 t ha^{-1} sludge	157 kg N ha^{-1}	47 kg N ha^{-1}	6.185 e
5 equ fertiliser	47 kg N ha^{-1}	47 kg N ha^{-1}	5.361 e
Control	0	0	2.320 f

Mean values with different letters indicate significant difference (Tukey test).

[1] Availability of N is based on values from the 'Permissible utilisation of sewage sludge' (WRC, 1997), that states only 30% of total N in sludge is available in the first year.

In terms of time intervals, as can be seen in Table 11.4, there were no significant difference between the background values and the NH_4^+-N content after 28 d. After day 42 the NH_4^+-N content was depleted to such an extent that it was significantly lower than even the initial NH_4^+-N content. At day 7, at the peak of the curve, the mean NH_4^+-N content of all treatments was significantly higher than that of the other intervals.

Table 11.4 The average total extractable NH_4^+-N content at different incubation periods

Days	Mean
7	22.530 a*
3	20.329 b
14	18.622 b
1	14.821 c
0	3.527 d
28	2.969 d, e
42	1.736 d, e
63	1.331 e

*The actual days are given in Table 6

Nitrate (NO_3^-)

In general, the production of NH_4^+-N by microorganisms causes large priming effects while production of NO_3^--N has little effect (Kuzyakow *et al.*, 2000). According to Figure 11.2, there were initially small quantities of NO_3^--N in the sludge treated soils while the NO_3^--N in the fertilizer treatments were high due to the inorganic applied NO_3^--N. Sludge that is added to the soil is readily mineralised, and increased microorganism activity releases N in the form of NH_4^+ (Alexander, 1961; Bosatta & Agren, 1995). A set of microorganisms, called *Nitrosomonas,* oxidises NH_4^+ to NO_2^-, after which another set of organisms, *Nitrobacter,* takes over and oxidises NO_2^- to NO_3^- (Paul & Clark, 1989; Singer & Munns, 1992). There is therefore an apparent lag period in the production of NO_3^- depending on the production of NH_4^+. The lag represents the time necessary for the microbial population to increase to an extent sufficient to cause an increase in the nitrification rate, as also described by Alexander (1961). The two-step nitrification process can be expressed by the following reactions (Singer & Munns, 1992):

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O$$
$$2NO_2^- + O_2 \rightarrow 2NO_3^-$$
$$\text{Overall Reaction: } NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O$$

Similar results were obtained by Tester *et al.* (1977), where the mineralisation rate of organic material was measured. A decrease in NH_4^+ was observed, with a subsequent increase in NO_3^- levels. Depending on the redox conditions in the soil, the oxidation process of NH_4^+, can also be described as follows (Schmidt, 1982):

$$NH_4^+ \xrightarrow{2e} NH_2OH \xrightarrow{2e} [NOH] \xrightarrow{2e} NO_2^- \xrightarrow{2e} NO_3$$
$$\downarrow$$
$$N_2O$$

In the oxidation of $NH_2(OH)$ to NO_2^-, an intermediate compound, [NOH] can form, that can be converted to either NO_2^- under aerobic conditions, or to N_2O under more anaerobic conditions (Schmidt, 1982). The loss of N through N_2O-gas during the nitrification process is unaccounted for in this incubation trial, and can explain the difference in N input and output. According to Figure 11.2, the NO_3^--N content increase slowly in the early stage of incubation, while NH_4^+-N is produced and increase rapidly after 7-14 d at the expense of the NH_4^+-N content.

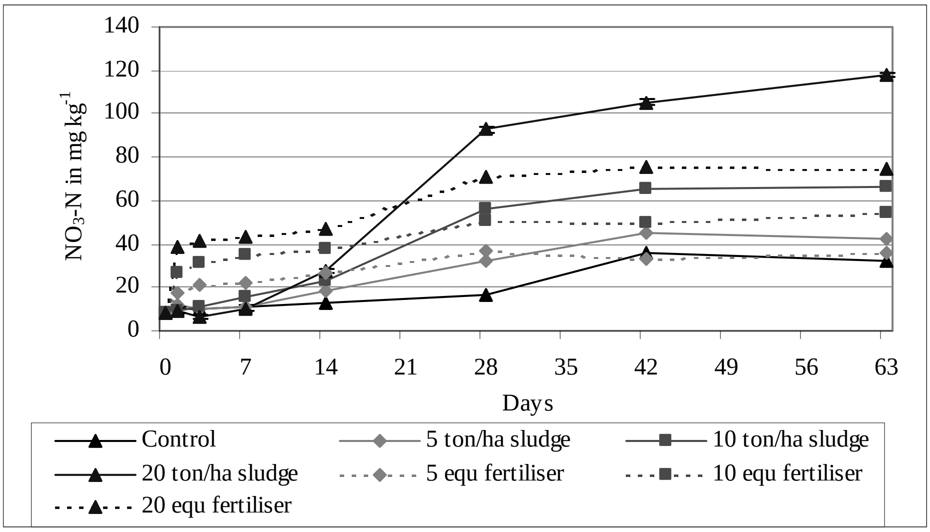


Figure 11.2. Total extractable NO_3^--N content as influence by differential N application and incubation time.

Significant differences also occur over time intervals (P<0.0001) (Table 11.5), as well as between different treatments (P<0.0001) (Table 11.6). However, at the start of the experiment, from day 1 to 7, and again at the end, day 42 to 63, there were no meaningful increases in NO_3^- concentrations. In all treatments the NO_3^- concentrations were significantly higher than the control.

Table 11.5 The average total extractable NO_3^--N at different incubation periods

Days	Mean NO_3^--N values
63	60.512 a*
42	58.516 a
28	50.764 b
14	27.627 c
7	21.193 d
3	18.734 d
1	17.626 d
0	8.457 e

*Mean values with different letters indicate significant difference (Tukey test).

Table 11.6 The average total extractable NO_3^--N for different treatments

Treatment	Mean
20 equ fertilizer	49.895 a*
20 t ha^{-1} sludge	47.347 a
10 equ fertilizer	36.585 b
10 t ha^{-1} sludge	31.947 c
5 equ fertilizer	25.042 d
5 t ha^{-1} sludge	22.650 d
Control	17.035 e

*Mean values with different letters indicate significant difference (Tukey test).

According to Table 11.6, the total extractable NO_3^--N contents were higher form the inorganic N treatments, than from the sludge treatments at the 10 t ha^{-1} application rate (sludge and fertilizer). For the 20 t ha^{-1} no significant difference were obtained between the sludge and the fertilizer. This was also the case for the 5 t ha^{-1} (sludge and fertilizer) treatments. Significant differences between the different levels of application (control, 5, 10 and 20 t ha^{-1}) were obtained.

Daily NO_3^--N production from sewage sludge

The extractable NO_3^--N production during the different incubation periods for each treatment is presented in Figures 11.3 to 11.8. These values were obtained, by calculating the difference between the extractable NO_3^--N productions between different incubation periods.

According to Figures 11.3 to 11.8, the trend in NO_3^--N production for the different N applications levels was the same. The NO_3^--N content on day 1 can be assumed to be the NO_3^--N that was either supplied through the sludge and fertilizer applications or the natural NO_3^--N content of the soil itself.

At the 5 t ha^{-1} application rate, Figure 11.3, an instant increase in the fertilizer treatment can be observed, 24 h after application. The extractable NO_3^--N from sludge only becomes available much later, and peaks at day 28.

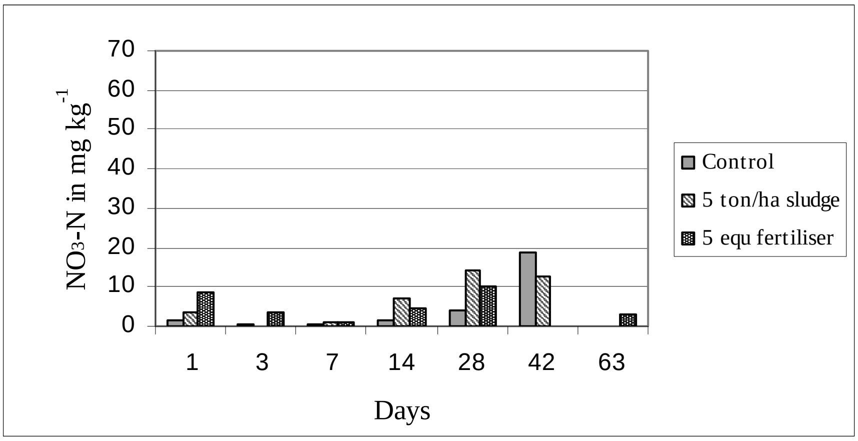


Figure 11.3. The total fractions of NO_3^--N that become available during different incubation intervals.

To put this into perspective, the daily production of NO_3^--N was calculated, and presented in Figure 11.4.

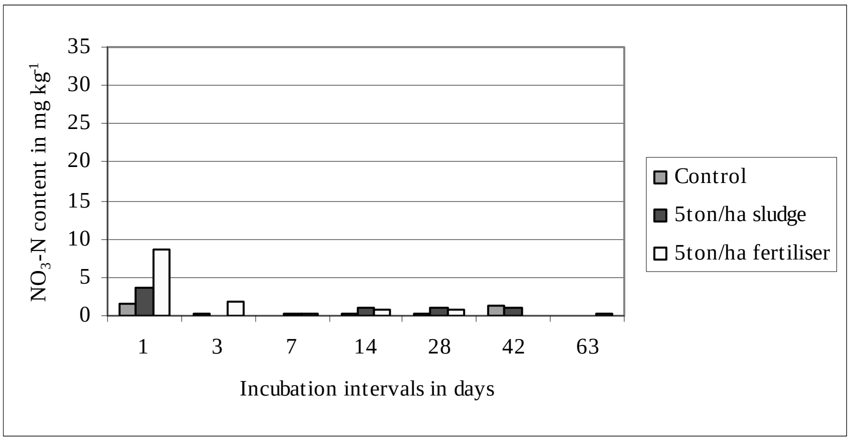


Figure 11.4. The daily NO_3^--N production as influenced by incubation time at 5 t ha^{-1} application rate.

At the sludge and fertilizer applications the trend in NO_3^--N production were the same. The NO_3^--N content on day 1 represents the applied NO_3^--N through the sludge and fertilizer applications. The daily NO_3^--N production peaks between day 28 and 42 and then decreases again. This lag of NO_3^--N production coincided with the decrease in NH_4^+-N content. This is due to the microbial oxidation of NH_4^+. A sudden increase in inorganic N from the fertilizer treatment is visible after only 24 h. NO_3^--N from the sludge only becomes available through nitrification and peaks at day 28, after which the additional NO_3^--N production decreases.

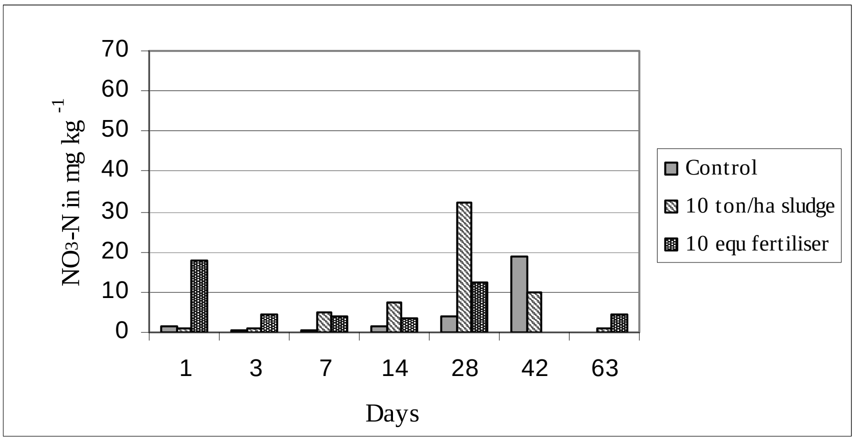


Figure 11.5. The total fractions of NO_3^--N that become available during different incubation intervals at the 10 t ha^{-1} application rate.

The daily production of NO_3^--N is expressed in Figure 11.6.

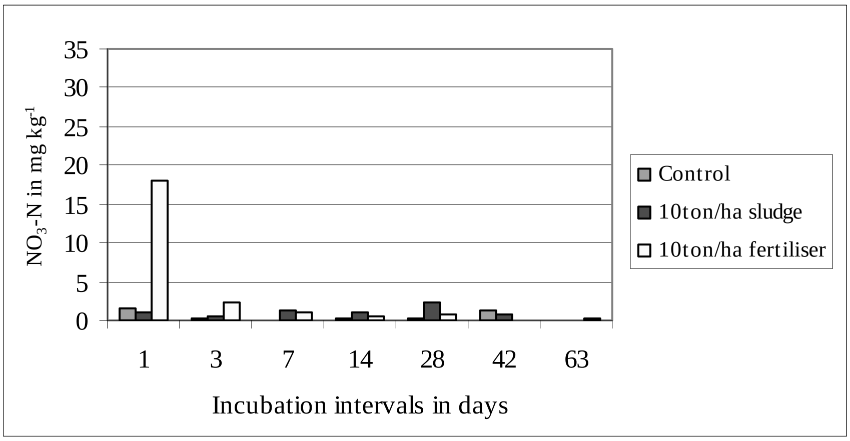


Figure 11.6. The daily NO_3^--N production as influenced by incubation time at 10 t ha^{-1} application rate.

At the 20 t ha^{-1} application rate (Figure 11.7), this effect is most enhanced. Here it can be seen, that directly after application of fertilizer and sludge treatments, more NO_3^- from commercial fertilizer is present compared to the sludge treatment. Hereafter, the additional amount available is low, but steadily increases until day 28, and decreases again.

The sewage sludge follows a different trend. Initially the concentrations of NO_3^- from sludge is extremely low, and close to zero. This is because most of the N is still present in the organic form that can be mineralised, by microorganisms, before the N can become available as also observed Korentajer (1991) and Singer & Munns (1992).

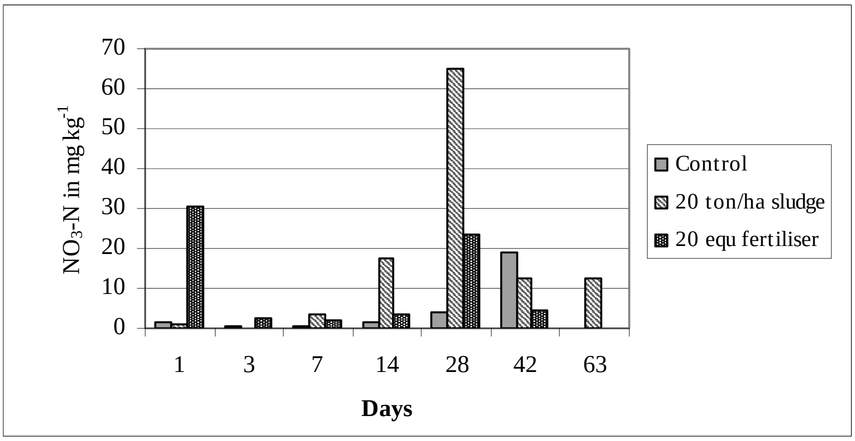


Figure 11.7. The total fractions of NO_3^--N that becomes available during different incubation intervals at the 20 t ha^{-1} application rate.

And the daily NO_3^--N production rate:

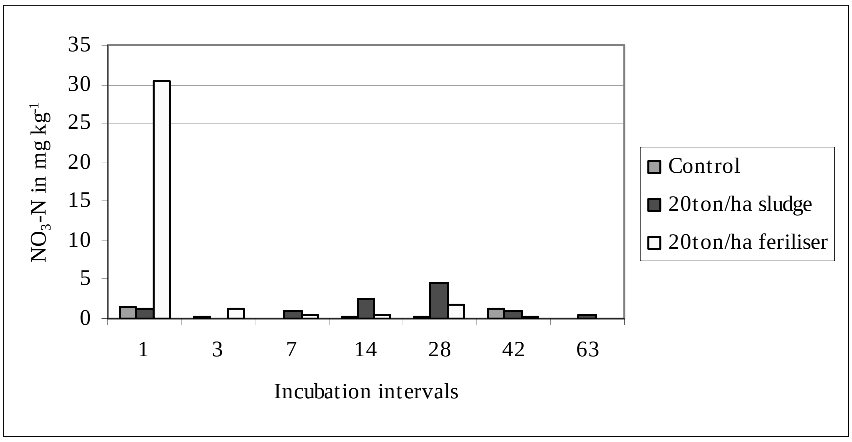


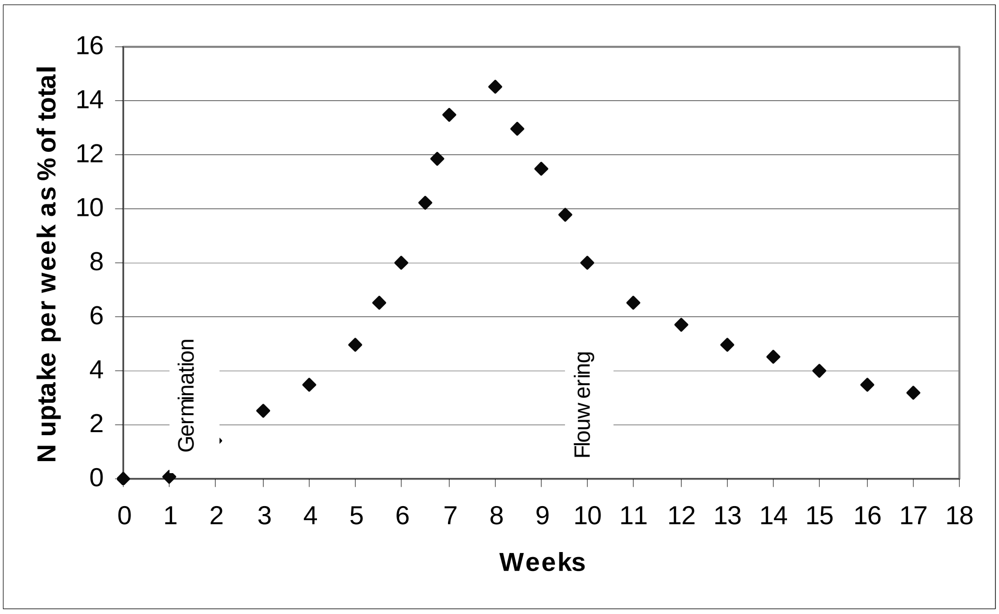
Figure 11.8. The daily NO_3^--N production as influenced by incubation time at 10 t ha^{-1} application rate.

Table 11.7. Total extractable NO_3^--N and daily production during 63 d incubation period from 20 t ha^{-1} sludge and its equivalent fertilizer treatment

Days	0	1	3	7	14	28	42	63	Total
Sludge, 20 t ha^{-1}									
Cumulative NO_3 production	8.46	9.63	6.68	10.42	27.70	92.73	105.34	117.82	
Differences between interval	8.46	1.18	-2.95	3.73	17.29	65.02	12.61	12.49	**117.8**
NO_3 production per day		1.18	0.00	0.93	2.47	4.64	0.90	0.59	
Fertilizer at 20 t ha^{-1} equivalence									
Cumulative NO_3 production	8.46	38.81	41.31	43.40	46.89	70.57	75.13	74.59	
Differences between interval	8.46	30.35	2.50	2.09	3.49	23.68	4.56	-0.54	**74.6**
NO_3 production per day		30.35	1.25	0.52	0.50	1.69	0.33	0.00	

The total amount of sludge that becomes available over 63 d is 117.82 mg kg^{-1} and 74.59 mg kg^{-1} for the commercial fertilizer application. Although the total amount of fertilizer is much less, the fractions of which become available is not equally distributed, as one can see from Figure 11.8. After day 7 the rate at which NO_3^- becomes available from sewage sludge, overtakes the rate of NO_3^-, but the *total* NO_3^--N released from sludge only overtakes the NO_3^--N release from the fertilizer treatment at day 28.

The advantage sludge has over fertilizer, is that most of the NO_3^- only becomes available between 14-28 d of incubation. In practice it can be expected that NO_3^--N production can be lower than the incubation values, because in the incubation trial conditions are always optimal for the microorganisms. This fraction of N can therefore be utilised much more efficiently by plants, reducing the risk of leaching. It is still not the ideal situation, because the maximum N utilisation by maize is at week 7 to 8 (49-56 d), just before flowering (MVSA, 1997). It is however, compared to commercial fertiliser, a much better option.



MVSA, 1997

Figure 11.9. Rate of N uptake by maize in week intervals.

This graph (Fig. 11.9) gives an indication of the NO_3^--N requirement of maize over time. The amount of NO_3^--N that becomes available from sludge can act as a slow release N-fertilizer that can be more beneficial to crops than commercial fertilizer.

Total inorganic release

Changes in inorganic N reflect the net mineralisation of organic N (Bernal *et al.*, 1998). By measuring the total release of NH_4^+ and NO_3^- over time, the mineralisation and nitrification rate of the specific sludge can be calculated. By subtracting the N release by the control treatment for every treatment, only the inorganic N release was obtained without background values. The total N release from the different treatments are presented in Figures 11.10, 11.11 and 11.12.

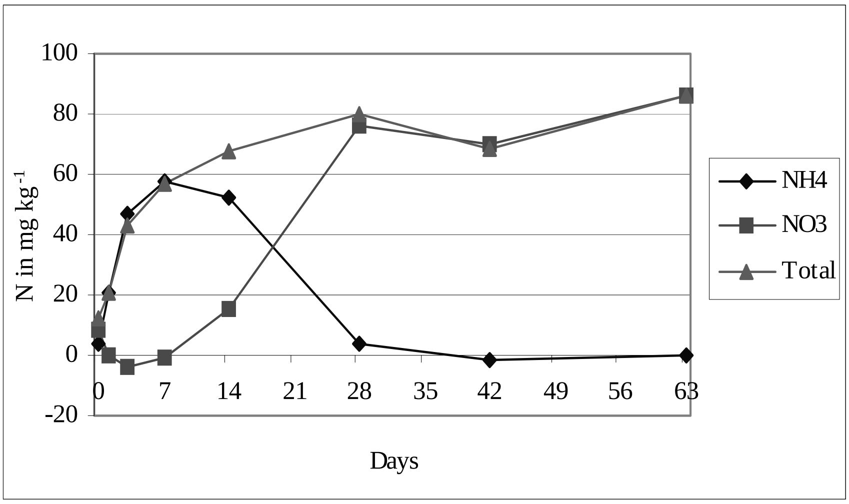


Figure 11.10. Changes in the NH_4^+-N, NO_3^--N and total N content during incubation of the treatment that received 20 t ha^{-1} sludge.

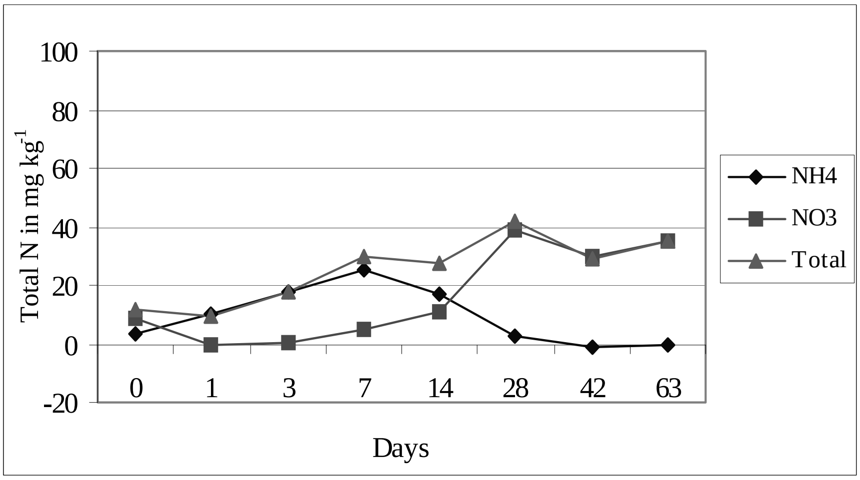


Figure 11.11. Changes in the NH_4^+-N, NO_3^--N and total N content during incubation of the treatment that received 10 t ha^{-1} sludge.

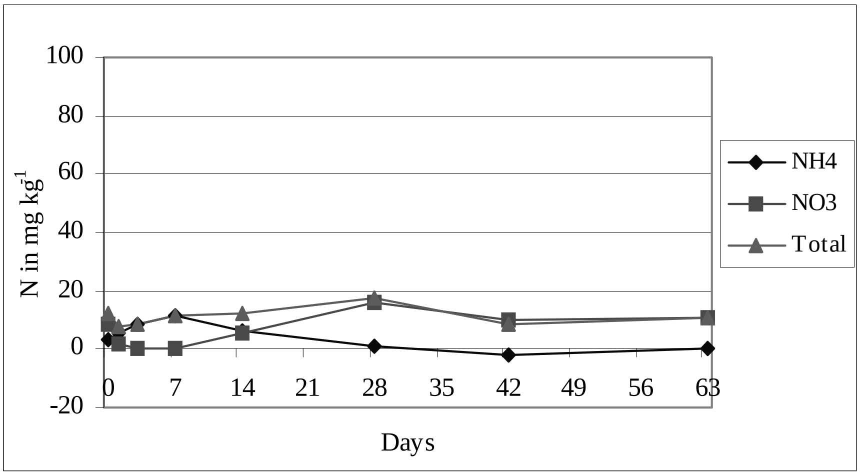


Figure 11.12. Changes in the NH_4^+-N, NO_3^--N and total N content during incubation of the treatment that received 5 t ha^{-1} sludge.

According to Figure 11.10, at the 20 t ha^{-1} treatment, the total N-content increases rapidly for the first 3 d, but N release becomes more steadily afterwards. For this treatment, the 20 t ha^{-1} represented 0.33 g of dried sludge per 50 g of soil as explained previously. This calculation is based on the assumption that the sludge is incorporated 20 cm deep in the soil. The total organic-N content of the sludge used, was 3.14%. This brings the initial N added to 10.36 mg per 50 g of soil, or 207.2 mg kg^{-1}. At the end of the incubation period (63 d), the total inorganic N produced was 86 mg kg^{-1}. This is an indication that 41.5% of the total organic N in the sludge was mineralised.

At the 10 t ha^{-1} treatment the total inorganic N after 63 d, was 34.78 mg kg^{-1}. The initial N added in the form of sludge, is 103.76 mg kg^{-1}, which indicates that 32.5% of the organic N was mineralised. This is in the same region as the 30% mineralisation in the first year, suggested by the 'Permissible utilisation and disposal of sewage sludge' (WRC, 1997).

At the 5 t ha^{-1} treatment, an inorganic N release of 21.5% was obtained. The reason for the decrease in inorganic N release with the lower application and higher values with the high application rates can probably be ascribed to what extent the microorganisms were stimulated. High application rates add a larger extractable energy source to the soil, which microorganisms can exploit. These increase the microbial activity and the sludge can be decomposed faster.

The rate of decomposition of organic matter in sludge is highly variable, depending on the sludge composition (Korentajer, 1991). Different values of mineralisation rate have been obtained in the passed. Mineralisation rates of organic N found in aerobically digested sewage sludge have been found to range from 20-50 % (Cripps *et al.*, 1992) and 4-48% during a 16-week incubation trial by Magdoff & Chromec (1977) and Parker & Sommers (1983).

It would therefore be wrong to generalise the inorganic N concentrations of sludges due to the preparation processes, the sampling techniques and handling prior to analysis (Parker & Sommers, 1983). Soil type, sewage sludge source and climatic factors can also have very large effects on organic N mineralisation rates (Parker & Sommers, 1983; Korentajer, 1991; Cripps *et al*, 1992).

11.4 CONCLUSIONS

According to the results and discussion, the following conclusions can be made:

Addition of sludge stimulates microbial activity and the subsequent release of NH_4^+-N. The maximum extractable NH_4^+-N content was observed after 7 d when it started to decline and after day 28, the total NH_4^+-N in the soil was depleted

Autotrophic microorganisms' activity increased from day 7 as indicated by the extractable NO_3^--N. This process happens concurrently with the NH_4^+-N production. After 28 d the extractable NO_3^--N content reached a maximum, indicating that most of the readily mineralisable N was depleted

Treatments that received commercial fertilizer, showed an immediate increase in NO_3^--N content, and from there a gradual increase in NO_3^--N content. After the 63 incubation days, 32.5% of the total N content of the 10 t ha^{-1} sludge treatment were mineralised. The higher application rate gave higher values, while the lower application rate gave lower values, indicating that higher additions of sludge lead to higher microbial activity

The maximum extractable NO_3^--N content obtained on both sludge and commercial fertilizer indicated that under ideal conditions it will take up the 28 d to mineralise and nitrify approximately 30% of the organic N in sludge and the total N in commercial fertilizer

Applied NO_3^--N from fertilizer is immediately available, while organically applied N become s available over a 28 d period equivalent to a slow release fertilizer

NO_3^--N production from the sludge does not immediately take place, it is slowly released over time

More NO_3^--N can be produced from sludge during the incubation period than from fertilizer, but being slowly released, it has a lower leaching risk and has more advantages in terms of crop production.

N from sewage sludge only becomes available after some time, through mineralisation. This fraction of NO_3^- can be utilised much more efficiently by crops, compared to commercial fertilizer, which is an inorganic fertilizer, and immediately available.

From an agricultural point of view, the slow release of N can hold numerous advantages. When the efficiency of commercial N fertilization is limited by factors such as high NO_3^- leaching losses, or NH_4^+ volatilisation, the use of sludge as a slow release N material may decrease the N losses and increase N availability.

Different values in terms of sludge mineralisation have been obtained, due to the large number of factors that may influence the microbial activity, such as soil type, climatic factors, and sludge type.

Possible losses due to denitrification needs to be investigated. Accurate N balances cannot be made, due to unmeasured N loss in the gas form, during the process of nitrification. It is advisable to carry out more studies on this topic, especially on the long-term. Use of lysimeters can be beneficial to study the whole N cycle including factors such as N uptake by plants, N lost due to volatilisation and leaching.

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Appendix 12

PERSISTENCE OF HUMAN PATHOGENS IN CROPS GROWN ON SEWAGE SLUDGE-TREATED SOILS

SUMMARY

The preliminary study was done on potato (*Solanum tuberosum)* as it represents a high risk crop for the use of sewage sludge and is one of the staple foods in South Africa. It has been shown that *Ascaris* and microorganisms studied, namely faecal coliforms, *E. coli* and *Salmonella* will thrive in soil for a prolonged period of time. The presence of these microorganisms on the potato peel indicates their potential hazard to public health. Due to the limitations of the techniques used, the presence or absence of microorganisms within the core of the potato could not be established conclusively. Further analysis of potatoes using PCR will be carried out, to establish whether or not these microorganisms are capable of migrating to the core of the potato. It appears that doubling the application rate from 8 t ha^{-1} to 16 t ha^{-1} does not affect the growth of microorganisms.

INTRODUCTION

The importance of sewage sludge as an aid in agricultural production has in recent years become a very popular subject (Hyde, 1976; Easton, 1983; Tester & Parr, 1983; Gagliardi & Karns, 2000). The practice of spreading sewage waste on soil has been a common way for farmers to add nutrients to soil. It is believed that when treated properly, and provided certain industrial contaminants are restricted from entering the sewage, the resultant sewage sludge can become a relatively innocuous organic fertilizer and soil conditioner of significant value for growing trees, crops and grass (WRC, 1997). In some countries such as the United States, sewage sludge is allowed to stand for up to three months before use in the hope that bacteria will die (EPA, 1999). In this study it was shown that microorganisms can persist for over three months. Although counts were minimal by the twelfth week, it is apparent that they will prevail for a period well exceeding three months. Counts of faecal coliforms also show that bacteria can survive for an elongated period of time. Strauch (1991) also indicated that faecal coliforms can survive for several years under optimal conditions.

Studies on different crops have indicated the contamination of fruits and vegetables following irrigation with sewage or wastewater (Petterson *et al*., 2001; Wachtel *et al*., 2001). Research in the US has highlighted cases where *E. coli* has been picked up by people eating raw vegetables or even peeling potatoes as well as through infected meat (Scotsman, 1998). Solomon *et al.* (2002) pointed out that *E. coli* is capable of entering the plant (lettuce) through the root system and migrating throughout the edible portion of the plant. Abdul-Raouf *et al*. (1993) demonstrated the ability of *E .coli* to grow on raw salad vegetables subjected to processing and storage conditions simulating those routinely used in commercial practice.

Although Salmonellosis has previously been associated mainly with food of animal origin, recent studies have shown that *Salmonella* contamination can be due to sewage irrigation (Melloul & Hassani, 1999), which could lead to crop contamination (Asplund & Nurmi, 1991; Guo *et al*., 2000). Several species of *Salmonella* have been implicated in gastroenteritis associated with the consumption of watermelon (Del Rosario & Beauchat, 1995).

Ascaris eggs are known to be extremely resistant to adverse condition (Strauch, 1991). Gaspard & Scwartbrod (1993) have shown that vegetables, namely, lettuce and tomato, can be contaminated with *Ascaris* following irrigation. Due to their prolonged survival, *Ascaris* pose a serious health threat. They have been reported to survive for up to two years in soil that has been irrigated with sewage (Strauch, 1991). Blumenthal *et al.* (1996) could show, doing experiments on lettuce, that the use of wastewater for irrigation causes transmission of nematode infections.

MATERIALS AND METHODS

Greenhouse experiments

Potato (*Solanum tuberosum)* was chosen for this experiment as it is one of the high risk vegetables to grow in sewage sludge. The potatoes tubers were obtained from a local farmer. A low metal sludge, obtained from Olifantsfontein, and the high metal sludge from Rondebult, both these plants belonging to the East Rand Water Care Company (ERWAT), was applied to soil in pots. The trials were conducted in greenhouses on the experimental farm of the University of Pretoria under controlled conditions (temp 25 – 28^{o}C) for a three-month period starting from May to July.

The potatoes were grown in 4 kg soil in pots. The trial comprised of 8 controls pots in which potatoes were grown in soil with no sludge. Sludge was added to the soil in other pots at an application rate of 8 t ha^{-1} for 8 pots in which low metal sludge (LMS) was added and another 8 pots that were given high metal sludge (HMS). Another set of 16 pots contained sludge at an application rate of 16 t ha^{-1} for both the LMS and HMS. Eight of these pots were used for LMS while the remaining eight were used for HMS. For each of the eight pots in which potatoes were grown, there was another set of eight pots that only contained soil and sludge. These pots were subjected to the same conditions as all other pots and were used for regular sample collection. Soil samples were collected on a bi-weekly basis. Other than water, which was added on every alternative day, no other nutrients were added to the pots. At the end of the experiment, potato samples were collected for microbial analysis. A portion of these potatoes was be subjected to Polymerase Chain Reaction (PCR).

Microbiological determinations

Procedures for analyses of faecal coliforms, *E. coli*, *Samonella* and *Ascaris* are those developed by the East Rand Water Care Company (ERWAT).

***Salmonella* analysis**

All chemicals used for this analysis were purchased from Oxoid. A 1 g of sample (soil containing sludge) was placed in a 10 ml buffered Peptone Water (Batch number 253802), mixed and incubated at 35°C for 18 – 24 h. About 0.1 ml of the mixture was transferred to 10 ml Rappaport VS Broth (Batch number 239145) and incubated at 44°C for 24 h. The enrichment broth was sub-cultured by streaking onto Brilliant Green agar (Batch number 216866) and incubated at 35°C for 18-24 h. A presumptive positive result was suspected if red colonies occured. Selected colonies were then sub-cultured onto Xylose-Lysine-Desoxycholate (XLD) agar (Batch number 230180), and incubated at 35°C for 18-24 h. Occurrence of black colonies suggested the presence of *Salmonella*.

Analysis of faecal coliforms

About 1 g of sample was withdrawn and added to 9 ml of peptone broth and incubated overnight. As the sample was too concentrated serial dilutions were made. This mixture was then filtered by mounting the funnel onto the filter holding assembly connected to a vacuum pump. Using sterile forceps the sterile 0.45 μm grid membrane filter (Sartorius, Batch number 0702 1/406 0102/53) was placed on the receptable of the filter holding assembly. A small volume of sterile distilled/deionised water was placed into the funnel prior to addition of the sample mixture to ensure an even distribution of the sample. When necessary, serial dilutions were made. When filtration was completed, the membrane filter was removed with sterile forceps and rolled onto MFC agar (Difco, Batch number 1162000). The petri dishes were inverted and incubated at 44.5 ± 0.5^{o}C for 18-24 h. All blue colonies were counted using a colony counter. Results were expressed as colony forming units per gram (CFU g^{-1}).

***E. coli* analysis**

The membrane from the faecal coliform was transferred to the nutrient agar substrate containing MUG (Difco, Batch number 0249000). The plates were then incubated together with one blank at 35± 0.5^{o}C for 4 h. Colonies were observed using a long wavelength ultraviolet light source for the fluorescence on the periphery. Results were expressed as CFU g^{-1}.

Ascaris

For determination of *Ascaris* ova, it is essential that the moisture content be estimated. This was done by weighing 50 g of the sample into a weighed dish and drying overnight. Another 10 g of the sample was weighed into a beaker and treated with an alkaline soap thoroughly mixing with an orange stick. The sample was then washed through a treble Visser filter (comprising mesh sizes 100 μm; 80 μm and 35 μm) by rinsing repeatedly with a strong jet of tap water. The remains in the outer filter were rinsed with tap water and centrifuged at 3000 g for 3 min. Using a Pasteur pipette, the supernatant was discarded. The pellet was resuspended in zinc sulphate [Saarchem (Batch number 1012122), 40%, 71 g/100ml H_2O)] and centrifuged further for 3 min at 3000 g. A Pasteur pipette was used to transfer the supernatant to a vacuum filtering system, using a filter of 12 μm (Millipore, Batch number R2DN50854). The zinc sulphate was rinsed off with distilled water to avoid recrystalization. The membrane filter was then placed in a glass petri dish and dried at 35^{o}C. A circular weight is usually placed around the edges of the membrane to prevent curling. Once dried, the filter was sliced across its diameter and each of the halves was placed onto a slide using a clear glue. Using an orange stick, immersion oil was spread over the filter. The slide was then observed under a microscope and the *Ascaris* ova were counted.

12.4 RESULTS

As shown in Figure 12.1, faecal coliforms appeared to grow considerably in the fourth and sixth week in both the low (LMS16) and high metal sludge (HMS16) pots when sludge was applied at 16 t ha^{-1}. Although appreciable growth was noticed in the fourth and sixth week for both the LMS and HMS, it appeared even more advanced in the LMS. Faecal coliforms growth was minimal at the beginning of the experiment and also in the second, eighth, tenth, and twelfth week for both the low and high metal sludges (Fig. 12.1). However, noticeable growth was observed in the twelfth week for LMS.

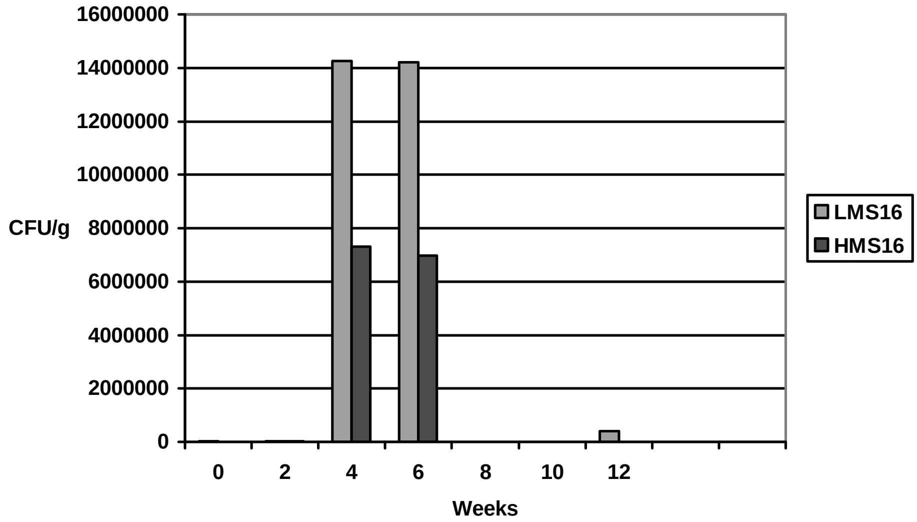

Figure 12.1. Graph of faecal coliforms for LMS and HMS at an application of 16 t ha^{-1}.

The faecal coliforms in the pots with applied low metal sludge had a considerable growth in the second week, which declined remarkably in the fourth week and picked up again in the sixth week (Fig. 12.2). The high metal sludge pots on the other hand only showed an appreciable growth in the sixth week and limited growth was noticed in the other weeks. Both the low and high metal sludge pots showed limited growth at zero time and the eighth to the twelfth week (week 8 to 12). More growth was observed in the pots with applications of LMS as compared to the HMS pots (Figs 12.1 and 12.2).

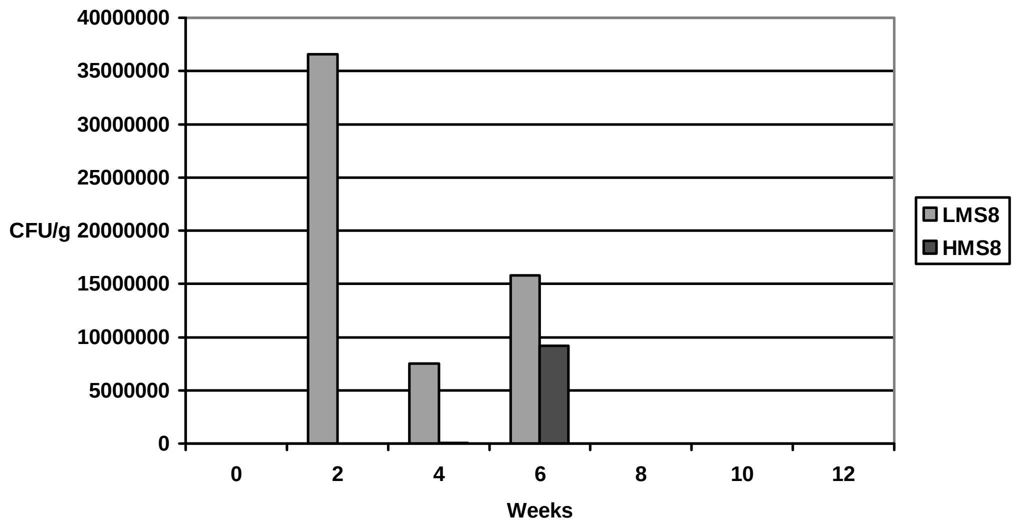

Figure 12.2. Graph of faecal coliforms for LMS and HMS at an application of 8 t ha^{-1}.

E. coli showed considerable growth when the low metal sludge was applied (Fig. 12.3). There appeared to be more *E. coli* at the fourth week for the LMS as compared to the HMS, although growth declined considerably in the LMS by the sixth week. A small peak was observed for the HMS in the sixth week. In general, there is hardly any noticeable growth in pots where high metal sludge was applied.

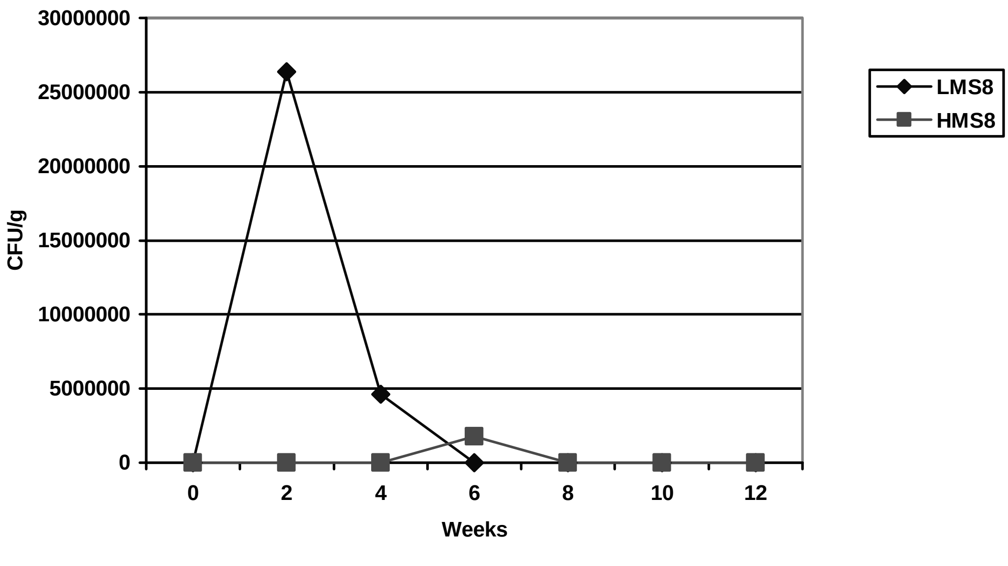

Figure 12.3. Comparison of *E. coli* growth for LMS and HMS at an application of 8 t ha^{-1}

The *E. coli* growth for both the low metal and high metal sludge pots peaked at week four and declined in the sixth week, although in the high metal sludge it is slightly lower than in the low metal sludge pots (Fig. 12.4). There was hardly any growth for both LMS and HMS at zero time in the eighth week to the twelfth week.

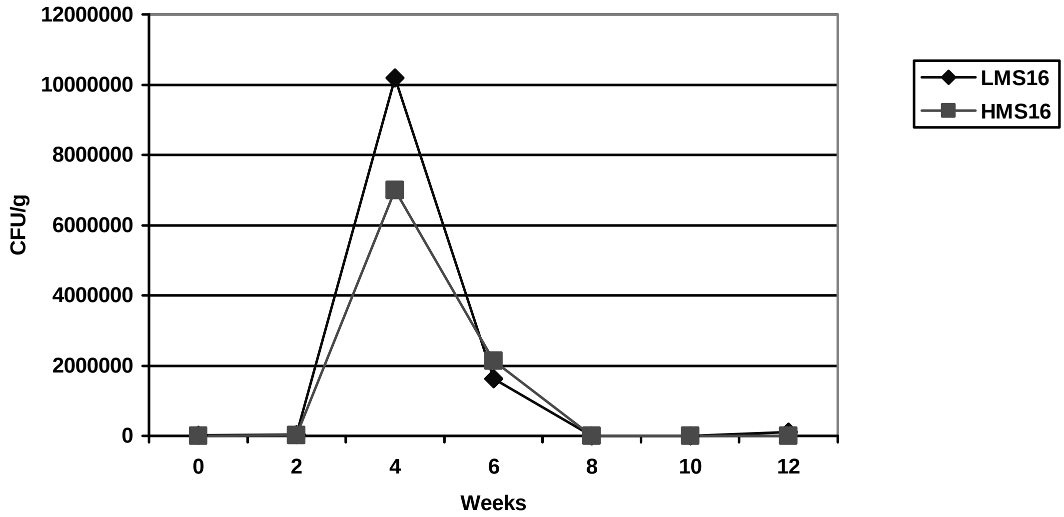

Figure 12.4. Comparison of *E.coli* growth for LMS and HMS at an application of 16 t ha^{-1}.

Table 12.1 provides an indication of whether *Salmonella* was found in the samples at each application rate for every week sampled. The presence of *Salmonella* is indicated with positive sign, while the absence thereof is indicated with a negative sign. At zero time, *Salmonella* was only observed in the LMS at 8 t ha^{-1}. All four treatments had *Salmonella* throughout week 2 to the 10th week. However, only the 16 t ha^{-1} application rate for both LMS and HMS was found to have *Salmonella* by the 12th week. No *Salmonella* was present in 8 t ha^{-1} treatment for both LMS and HMS at week 12.

Table 12.1 *Salmonella* found in sludge-applied soil in pots

Weeks	LMS 8	LMS 16	HMS 8	HMS 16
0	+	-	-	-
2	+	+	+	+
4	+	+	+	+
6	-	+	+	+
8	+	+	+	+
10	+	+	+	-
12	-	+	-	+

+ = presence, - = absence

Table 12.2 indicates the total number of *Ascaris* found in all the pots for each application rate and sludge type for every week sampled. Other than at zero time and in the 4th week, there appeared to be no *Ascaris* in the soil sampled. *Ascaris* was found in large numbers in samples collected in the 4th week. For instance, a total of four *Ascaris* were counted in samples collected from pots that received LMS at 16 t ha^{-1} (Table 12.2).

Table 12.2 Numbers of *Ascaris* found in sludge-applied soil in pots

Week	LMS 8	LMS 16	HMS 8	HMS 16
0	1	0	0	2
2	0	0	0	0
4	2	4	1	0
6	0	0	0	0
8	0	0	0	0
10	0	0	0	0
12	0	0	0	0

Figures 12.5 and 12.6 show the growth of faecal coliforms and *E. coli* for LMS at 8 t ha^{-1} and also at 16 t ha^{-1}. Despite having doubled the amount of sludge (16 t), there was not much difference in growth between the two application rates. Instead there appears to have been more growth in the 8 t ha^{-1} application than the 16 t ha^{-1}. For instance, faecal coliforms were observed in the second week to be more than 35 X 10^{6} CFU g^{-1} at an application of 8 t ha^{-1} (Fig. 12.5), whereas in the 16 t ha^{-1} by the second week, faecal coliforms were found to be only as high as 14 X 10^{6} CFU g^{-1}. This is even less than half of the value recorded for the 8 t ha^{-1} (Fig. 12.6).

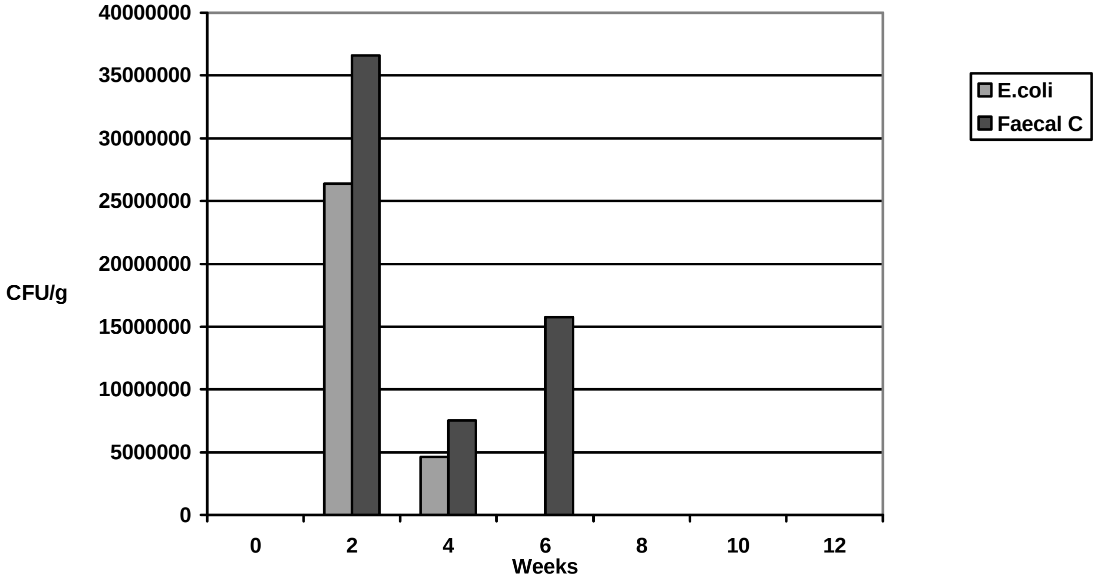

Figure 12.5. The plot of *E. coli* and faecal coliform for LMS at 8 t ha^{-1}.

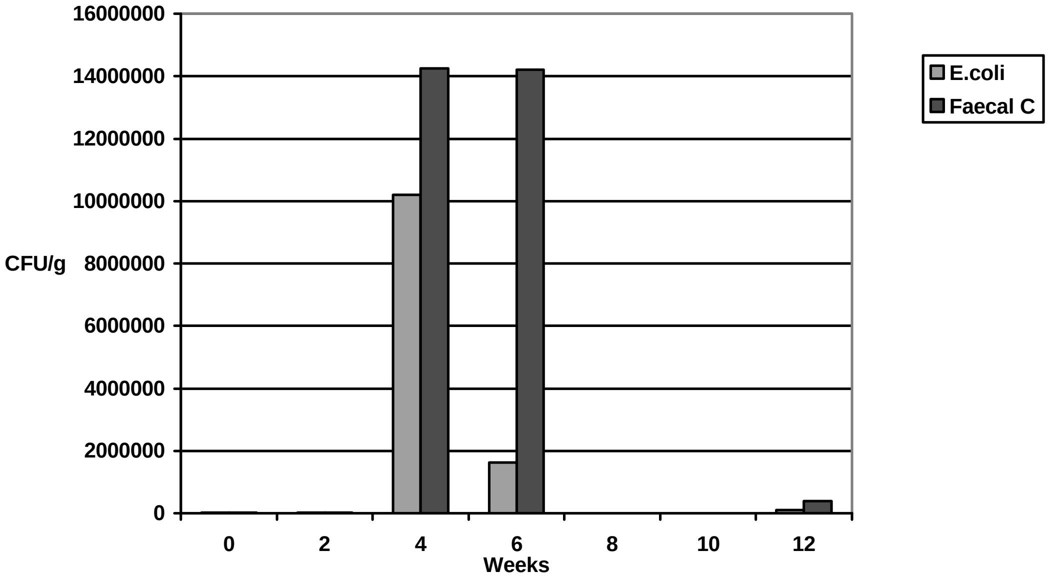

Figure 12.6. The plot of *E. coli* and faecal coliform for LMS at 16 t ha^{-1}.

Indicated in Table 12.3 are the microorganisms found on the potato peel and within the potato (potato core). None of the microorganisms tested were found in the potato core. However, large numbers of faecal coliforms and *E. coli* were found on the peel in the treatment LMS at 16 t ha^{-1}. These are the mean values of eight repetitions. *Salmonella* also tested positive in this treatment. Although other microorganisms were not found in the HMS at 16 t ha^{-1}, potato peel from this treatment tested positive for *Salmonella*. None of the samples was found to contain *Ascaris*.

Table 12.3 Microorganisms found in potato in the 12th week

Sample	Microorganism	LMS 8	LMS 16	HMS 8	HMS 16
Potato peel	Faecal coliforms (CFU g^{-1})	0	2050	0	0
	E. coli (CFU g^{-1})	0	1800	0	0
	Salmonella	-	+	-	+
	Ascaris	0	0	0	0
Potato core	Faecal coliforms (CFU g^{-1})	0	0	0	0
	E. coli (CFU g^{-1})	0	0	0	0
	Salmonrella	-	-	-	-
	Ascaris	0	0	0	0

12.5 DISCUSSION

As the results have shown, there was minimal growth in the 8th week through to the 12th week. This could be attributed to competition for food and space as resources were declining as no nutritional addition was made to any of the pots. Thus, these microorganisms were no longer able to reproduce unlimitedly as was observed in the weeks 2 to 6. Survival of bacteria is influenced by a number of factors such as optimum temperatures and availability of organic matter (Bitton, 1994)

The HMS had far less growth of both the faecal coliforms and *E. coli* as compared to the LMS. This inhibited growth is probably due to the high concentrations of metals found in the HMS. According to analysis report for metals done on the sludge from the Olifantsfontein, most of the metals analysed were found to yield high values (Fe = 4230; Al = 14824; Zn = 148) (ERWAT, 2002).

Faecal coliforms, *E. coli* and *Salmonella* were found to be present on the outside (peel) of the potato following harvest time. As the methods can only detect microorganisms if they can be cultured, it follows that no microorganisms were found on the inside (core) of the potato. Although all the microorganisms studied were found on the peel of the potato, none were found to be present in the inside of the potato.

Most of the samples collected throughout the experiment tested positive for *Salmonella*. However, in the twelfth week, only samples collected form the 16 t ha^{-1} treatment for both LMS and HMS tested positive for *Salmonella*. The persistence of *Salmonella* throughout the experiment suggests their prolonged survival in soil. *Salmonella* has been shown by other workers to survive for long periods in soil. For instance, Strauch (1991) demonstrated that *Salmonella* could survive on and in the soil after a single application of sludge in summer for 424 to 820 d. In winter the survival times were reported to be 104 to 350 d. Baloda *et al.* (2001) also confirmed the prolonged survival of *Salmonella*, which he estimated to be about 299 d in soil.

Salmonella was only found on the peel of potatoes grown in the 16 t ha^{-1} for LMS and none of the core samples tested positive.

Ascaris ova were found at zero time and also in the 4th week, but no ova were observed throughout the 6th week to the end of the experiment. *Ascaris* ova were also not found on the potato peel at the end of the experiment. However, their absence in the samples does not rule out their potential presence, as these eggs can survive for long periods of time (Strauch, 1991).

12.6 CONCLUSIONS

It has been shown that *Ascaris* and microorganisms studied, namely faecal coliforms, *E. coli* and *Salmonella* will thrive in soil for a prolonged period of time. The presence of these microorganisms on the potato peel indicates their potential hazard to public health. Due to the limitations of the techniques used, the presence or absence of microorganisms within the core of the potato could not be established conclusively. Further analysis of potatoes using PCR will be carried out, to establish whether or not these microorganisms are capable of migrating to the core of the potato.

It appears that doubling the application rate from 8 t ha^{-1} to 16 t ha^{-1} does not affect the growth of microorganisms.

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Appendix 13

PUBLIC PERCEPTION ON THE USE OF SEWAGE SLUDGE IN AGRICULTURAL PRACTICES

13.1 SUMMARY

A preliminary study was done to establish the extent of knowledge and perception of a group representing the general public regarding the use of sewage sludge in agricultural practices. The population surveyed using a questionnaire aimed to include the man on the street, supermarkets and shops selling vegetables and farmers using sewage sludge as a soil amendment. The results of 28 questionnaires illustrated the opinions of individuals representing a household earning less than R5000/month. Only 39% of the respondents were aware of what sewage sludge was before they read the information given to them. After reading the information provided, 71% indicated that they knew that sewage sludge was used as a "fertilizer" in agricultural practices. Most of the respondents were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil.

The group (29 questionnaires) surveyed that represented the households earning above R 5000 were more informed about the source of sewage sludge. The majority, 79%, indicated that they knew that sewage sludge was used as a "fertilizer" in agricultural practices and were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil.

The perception of the buyers of fruit and vegetables for the commercial markets and shops were not adequately captured in this study. The people that filled in the questionnaires were generally not at the post level required to truly reflect the opinion of the general supermarkets and vegetable shops with regards to their willingness to purchase vegetables grown on sewage sludge amended soils.

The opinion of the seven farmers who are currently using sewage sludge as a soil amendment was obtained. The farmers were all in favour of using sewage sludge as a soil amendment and none of them currently exceeded the recommended dosage of 8 t_{dry} ha^{-1} $year^{-1}$.

13.2 INTRODUCTION

The scientific evidence of both the risks and benefits associated with the application of sewage sludge to agricultural land has been well documented (Korentajer, 1991). There are probably few scientists who would argue that sewage sludge should be viewed as an unwanted by-product of sewage treatment. The literature pertaining to sewage sludge reveals that an enormous amount of research has gone into understanding and exploiting the benefits of reusing sludge and into better understanding any threat of its reuse on land. The reuse of sewage sludge on land is recognised by both government and the regulators to be the best practical environmental option (Tyson, 2002).

The South African public and the environment are protected against the risks associated with the application of sewage sludge to agricultural land by the sludge guidelines (WRC, 1997; WRC 2002). In South Africa, sewage sludge is classified in four categories, types A, B, C and D in decreasing order of potential to cause odour nuisances, fly breeding and transmission of pathogenic organisms to man and the environment (WRC, 1997). Treatment processes such as anaerobic and aerobic stabilisation, thermal treatment and lime treatment are well known and generates sludge with a quality that could be applied to agricultural land.

However, the general public's opinion on the use of treated sewage sludge as a soil amendment in agricultural practices could potentially change the current or future practices based on their support or lack thereof. The general public is not readily exposed to detailed peer reviewed scientific papers and often rely on information published in the general media. In the US, some environmental activists such as Greenpeace and the Citizens Clearinghouse on Hazardous Waste have warned about the dangers of sludge, but most groups have bought into the argument that sludge farming is the least offensive way to manage the problem of waste disposal. Some groups even support sludge farming. During the 1970s, these environmentalists worked for promulgation of the Clean Water Act. Now they find themselves in the awkward position of defending its consequence with having to face huge mountains. Sarah Clark, formerly of the Environmental Defence Fund, claims that sludge farming "is the best means of returning to the soil nutrients and organic matter that were originally removed. It is recycling a resource just as recycling newspapers or bottles is. If the right safeguards are taken, it can be environmentally protective and even beneficial."

It is speculated that there is another factor which influences public perception of sewage sludge reuse to a far greater extent then anything discussed in this research so far, and that factor is odour. Very seldom does a member of the public register concern or complain about sewage sludge reuse solely due to a fear relating to food safety; they have usually been provoked to complain by having their privacy invaded by odour from sludge spreading activity. If sewage sludge did not smell the public probably would not complain and the overall public perception of the reuse of sewage sludge would improve (Tyson, 2002).

A preliminary study was done to establish the extent of knowledge and perception of a group representing the general public regarding the use of sewage sludge in agricultural practices.

13.3 MATERIALS AND METHODS

A questionnaire was drafted and consisted of a two sections (Addendum 13.1). Section A, addressed personal information such as age group, sex, number of the people in household and the literacy level of people answering the questionnaire. Section B addressed matters on sewage sludge i.e. to establish the acceptability of the agricultural application and knowledge of sewage sludge.

The questionnaires were posted to 50 randomly selected addresses selected from the Pretoria and East Gauteng telephone directory. Since only 6 responses were received, the questionnaire was e-mailed to 35 individuals and 56 interviews were done.

The population surveyed aimed to include the man on the street, supermarkets and shops selling vegetables and farmers using sewage sludge as a soil amendment.

13.4 RESULTS AND DISCUSSION

Table 13.1 indicates the amount of responses for the different sections surveyed.

Table 13.1 Amount of responses for the different sections surveyed

Section	Responses
Individuals (man on the street)	57
Supermarkets and vegetable shops	11
Farmers	7
TOTAL	75

The distribution of the total population surveyed in this study is illustrated in Figures 13.1 to 13.7.

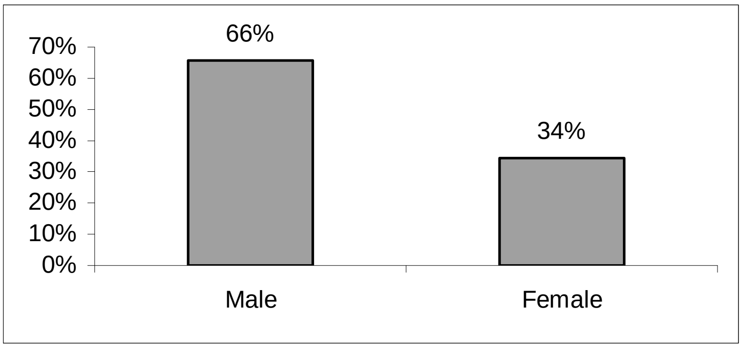

Figure 13.1. Sex of the surveyed population.

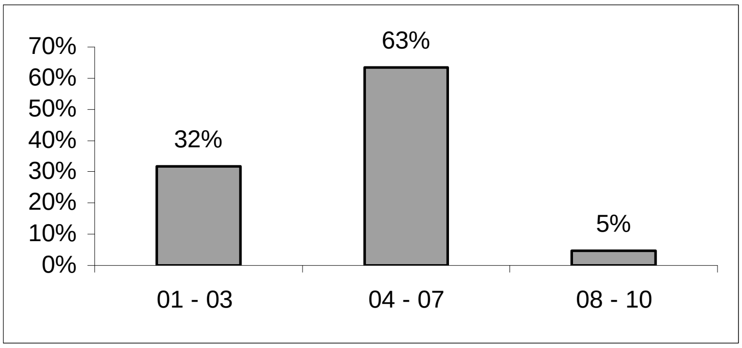

Figure 4.36. Age group of the surveyed population.

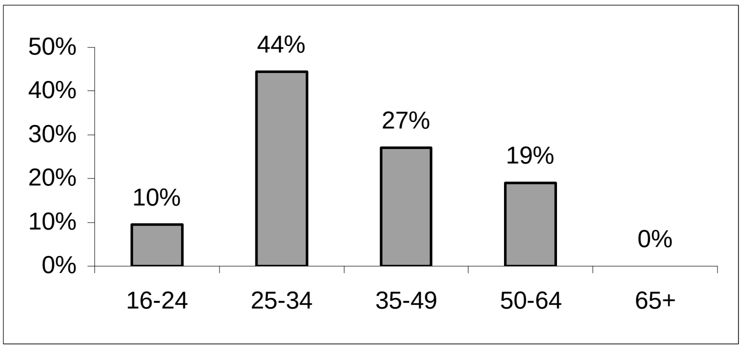

Figure 13.3. Number of people per household in the surveyed population.

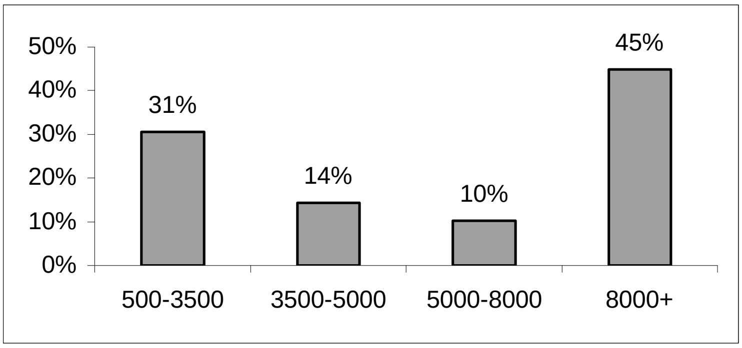

Figure 13.4. Income per household of the surveyed population.

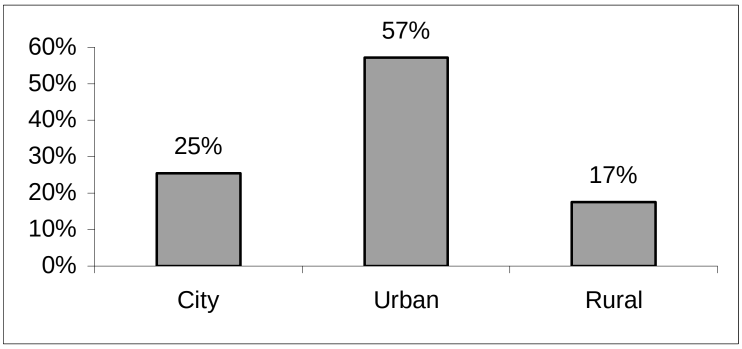

Figure 13.5. Physical distribution of the surveyed population.

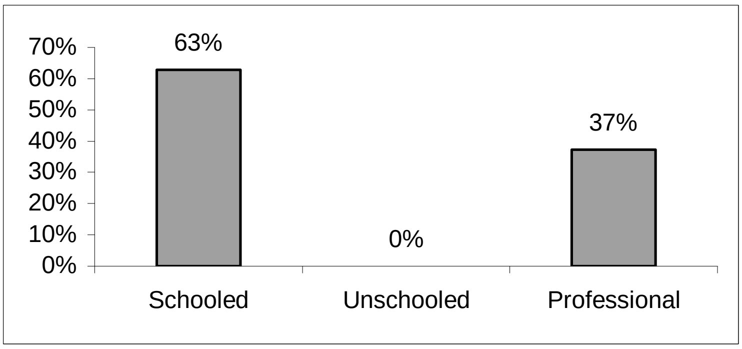

Figure 13.6. Literacy of sampled population.

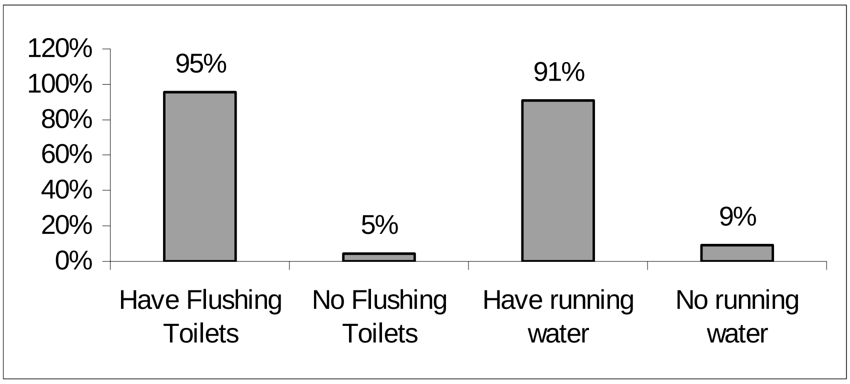

Figure 13.7. Availability of sanitation and water to the surveyed population.

The opinion of individuals (man on the street).

The responses (53) were divided into two groups, those earning less than R5000 per month/household and those earning above R5000 per month/household.

Individuals in a household earning less than R5000/month

Twenty-eight (28) questionnaires represented this group. More than half (53%) of these individuals were aware that domestic wastewater went to a wastewater treatment plant for treatment. Eighteen individuals (64%) of the population was in favour of the recycling of wastewater. Only 39% of these individuals knew what sewage sludge was. Table 4.36 shows the opinions of the 28 individuals who responded on the use of sewage sludge in agricultural practices.

The survey indicated that only 39% of the respondents were aware of what sewage sludge was before they read the information given to them. Table 4.36 indicates that after they read the information provided, 71% indicated that they knew that sewage sludge was used as a "fertilizer" in agricultural practices. The respondents were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. There were conflicting results regarding the purchase of vegetables grown on sewage sludge, and consuming cooked and raw vegetables grown on sewage sludge. Table 4.36 indicates that 71% of the respondents would purchase vegetables grown on sewage sludge, but only 61% would consider consuming raw vegetables grown on sewage sludge and even less of the respondents, 54%, would consume cooked vegetables grown on sewage sludge-amended soils. This indicates a responsibility to educate the broader public in the benefits of washing, peeling and cooking vegetables to prevent infection from soil borne diseases and pathogen infection.

Table 4.36 The opinion of individuals who are part of a household earning below R 5000/ month on the use of sewage sludge in agricultural purposes

Question topic	Percentage (%)	Percentage (%)	Percentage (%)
Did you know that sewage sludge could be used as a fertilizer?	Yes: 71	No: 11	No response: 18
Do you think sewage sludge should be used as a fertilizer?	Yes: 82	No: 7	No response: 11
Would you buy vegetables grown using sewage sludge as a fertilizer?	Yes: 71	No: 18	No response: 11
The use of sewage sludge as a fertilizer is not risky to human health	Agreed: 36	Disagreed: 7	Neutral: 54 No response: 4
What impact do you think the use of sewage sludge as a fertilizer can have on the soil properties?	Improves soil quality: 75	Impacts soil quality negatively: 4	No response: 21
Do you think it is safe to use sewage sludge as a fertilizer for the following applications			
Application	Percentage (%)	Percentage (%)	Percentage (%)
Vegetables consumed raw	Yes: 61	No: 29	No response: 11
Vegetables consumed cooked	Yes: 54	No: 36	No response: 11
Tobacco	Yes: 64	No: 25	No response: 11
Vineyards	Yes: 79	No: 10	No response: 11
Fruits trees	Yes: 82	No: 7	No response: 11
Cereal culture	Yes: 79	No: 10	No response: 11
Sugar cane	Yes: 82	No: 7	No response: 11
Gardens and traffic islands	Yes: 75	No: 14	No response: 11

Question topic	Percentage (%)	Percentage (%)	Percentage (%)
Crops which animals producing milk, meat and eggs graze.	Yes: 64	No: 21	No response: 11

Individuals in a household earning more than R5000/month

Twenty-nine (29) questionnaires represented this group. The majority of this group (90%) were aware that domestic wastewater went to a wastewater treatment plant for treatment. Twenty individuals (69%) of the population were in favour of the recycling of wastewater. In this group, 76% of the population surveyed knew what sewage sludge was compared to 39% representing the lower income group. Table 13.3 shows the opinions of the 29 individuals who responded on the use of sewage sludge in agricultural practices.

Table 13.3 The opinion of individuals who is part of a household earning above R 5000/ month on the use of sewage sludge in agricultural purposes

Question topic	Percentage (%)	Percentage (%)	Percentage (%)
Did you know that sewage sludge could be used as a fertilizer?	Yes: 79	No: 14	No response: 7
Do you think sewage sludge should be used as a fertilizer?	Yes: 79	No: 14	No response: 7
Would you buy vegetables grown using sewage sludge as a fertilizer?	Yes: 79	No: 21	No response: 0
The use of sewage sludge as a fertilizer is not risky to human health	Agreed: 55	Disagreed: 17	Neutral: 24 No response: 3
What impact do you think the use of sewage sludge as a fertilizer can have on the soil properties?	Improves soil quality: 69	Impacts soil quality negatively: 0	No response: 31

Do you think it is safe to use sewage sludge as a fertilizer for the following applications			
Application	Percentage	Percentage	Percentage (%)

Question topic	Percentage (%) (%)	Percentage (%) (%)	Percentage (%)
Vegetables consumed raw	Yes: 45	No: 55	No response: 0
Vegetables consumed cooked	Yes: 69	No: 31	No response: 0
Tobacco	Yes: 76	No: 24	No response: 0
Vineyards	Yes: 83	No: 14	No response: 3
Fruits trees	Yes: 79	No: 17	No response: 3
Cereal culture	Yes: 79	No: 21	No response: 0
Sugar cane	Yes: 83	No: 17	No response: 0
Gardens and traffic islands	Yes: 72	No: 28	No response: 0
Crops which animals producing milk, meat and eggs graze.	Yes: 69	No: 0	No response: 31

The survey indicated that 76% of the respondents were aware of what sewage sludge was before they read the information given to them. After they read the information provided, 79% indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices. The respondents (79%) were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. The majority of the respondents (79%) indicated that they would purchase vegetables grown on sewage sludge amended soils. However, this percentage dropped to 61% when asked whether they would consider consuming cooked vegetables grown on sewage sludge amended soils. As expected a lower percentage (45%) were prepared to consume raw vegetables grown on sewage sludge amended soils compared to their willingness (61%) to consume cooked vegetables grown on sludge amended soil.

The opinion of supermarkets and vegetable shops

The opinion of the buyers of vegetable shops and major supermarkets is important for this study. They are normally required to purchase good quality vegetables and fresh produce for their enterprise at the lowest price. It is important to establish whether a buyer would consider vegetables or other fresh produce grown on sewage sludge as a risk or as inferior quality.

The responses (11 questionnaires) received back from this market segment was disappointing. The individuals who filled in the questionnaires were not the individuals who purchase the vegetables and other fresh produce for the companies. They were generally not at the post level required to truly reflect the opinion of the general supermarkets and vegetable shops with regards to there willingness to purchase vegetables grown on sewage sludge-amended soils. The posts filled by the respondents included, training and development officer, office assistant, public relations officer, receptionist, salespersons, musician, and till operators. For this reason, these results are not included in this study.

The opinion of farmers

The opinion of seven farmers was obtained by requesting that they fill in the questionnaire. These farmers all make use of sewage sludge as a soil amendment and operate in the east Gauteng region.

The farmers grow the following crops on sludge-amended soils: maize, dry beans, including sugarbeans and soybeans, sunflowers, fodder and fruit trees. All seven the farmers were of the opinion that sewage sludge was a good fertilizer. The reasons given for why they thought sewage sludge was a good soil amendment included:

Any organic matter is preferred over chemical amendments
They referred to research done by Snyman *et al.*
It is seen to be cost effective and effective as a fertilizer
Sewage sludge is a natural organic product and gives very good results
It is a way of preventing pollution

The farmers receive the sludge by truck and apply the sludge mechanically using a spreader during winter and/or before planting. All the farmers that replied added less than 8 t_{dry} ha^{-1} $year^{-1}$ with an average application rate of 2 t_{dry} ha^{-1} $year^{-1}$. The farmers rated sewage sludge as a good fertilizer but mentioned that the nitrogen and potassium content was insufficient and that they needed to add more sewage sludge than commercial fertilizer. All the farmers agreed that sewage sludge used in agricultural practices did not pose a risk to human health when handled responsibly.

13.5 CONCLUSIONS AND RECOMMENDATIONS

The surveyed group representing households earning less than R5000 per month indicated that only 39% of the respondents were aware of what sewage sludge was before they read the information given to them. After reading the information provided, 71% indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices. Most of the respondents were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. The results indicated that the respondents did not understand the risks involved in using sewage sludge as an agricultural soil amendment. The wastewater industry will need to embark on a widespread awareness of both the negative and positive impacts relating to the use of sewage sludge in agricultural practices.

The group surveyed that represented the households earning above R 5000 per month was more informed about the source of sewage sludge. The majority, 79%, indicated that they knew that sewage sludge was used as a “fertilizer” in agricultural practices and were in favour of using sewage sludge as a soil amendment and agreed that it improved the properties of the soil. The majority of the respondents (79%) indicated that they would purchase vegetables grown on sewage sludge-amended soils. 61% of the respondents would consider consuming cooked vegetables grown on sewage sludge-amended soils and 45% were prepared to consume raw vegetables grown on sewage sludge-amended soils.
The perception of the buyers of fruit and vegetables for the commercial markets and shops was not adequately captured in this study. The people that filled in the questionnaires were generally not at the post level required to truly reflect the opinion of the general supermarkets and vegetable shops with regards to there willingness to purchase vegetables grown on sewage sludge-amended soils.

The opinion of the farmers who are currently using sewage sludge as a soil amendment was obtained. The farmers were all in favour of using sewage sludge as a soil amendment and none of them currently exceeded the recommended dosage of 8 t_{dry} ha^{-1} $year^{-1}$.

The results of this study give a preliminary idea of the perception of the public regarding the acceptability of using sewage sludge as a soil amendment in agricultural practices. It is recommended that a detailed survey be done to establish the opinion of the broader South African population.

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Addendum 13.1 Questionnaire to establish the public perception of the use of sewage sludge in agricultural practices

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EAST RAND WATER CARE COMPANY

RESEARCH AND DEVELOPMENT DEPARTMENT

SEWAGE SLUDGE AS FERTILIZERS

The purpose of this survey is to get the public view of using sewage sludge in agricultural practices, especially in crops cultivated for human consumption needs.

You are kindly requested to fill in the questionnaire to the best of your knowledge. A member of the project team will visit you to assist where you experience problems and to collect the questionnaire.

We advise you to read the following background information if you are not sure what sewage sludge is.

The water used for flushing toilets and washing in our houses goes into a sewer line and ends up at a sewage plant (wastewater purification plant). The water is treated using bacteria, which “eat” all the impurities such as our body and food waste out of the water. Once the bacteria have “eaten” all the impurities. The bacteria is separated from the clean water. The clean water is returned back to the environment. The bacteria that we harvest are called sewage sludge. We need to stabilise this sewage further to avoid fermentation, hence causing odour problems. The sewage sludge is then stabilised in a large reactor called an anaerobic digester. This process is similar to a composting process except that we do not add any oxygen. Once the sewage sludge is stabilised we dry it using several technologies. Some plants dry it in the sun; others use machines to dry the sludge. Once it is dry, it looks and smells like compost and we call this product biosolids. Many years of research has proven that the biosolids is very useful as a fertiliser. The biosolids contain a substantial amount of Nitrogen and Phosphorus, which is what we buy when we purchase a bag of fertiliser at the shop. The biosolids is also rich in Carbon, which is very good for the soil. We would now like to know what your opinion is regarding the use of biosolids to grow vegetables for human and animal consumption.

The questionnaire consists of two sections. Section A is about your personal information and Section B is about the use of sludge as fertilizer. Anonymity of individual people will be maintained.

Please note that it is your opinion that we want. Do not ask anyone to fill the form for you, as we would like to have a true cross section of opinions.

Note: Please do not delay even if you are not sure of all the answers.

We will be grateful for your co-operation

Forward your correspondences to:
Ms. Patricia Sibiya
ERWAT Research & Development
PO Box 13106

Norkem Park
1631
Tel: 011 929 7000
Fax: 011 929 7031
E-mail: patriciam@erwat.co.za

SECTION A

Tick the appropriate box:

Sex

Male Female

Age group

16 – 24 25 – 34 35 – 49 50 – 64 65+

How many people in your household?

…………………………………………

What is your occupation? Please give details

……………………………………………………………………………

……………………………………………………………………………

……………………………………………………………………………

What is the income level of your household?

R500 – 3500 R3500 – 5000 R5000- 8000 8000+

Where do you live?

City Urban Rural

SECTION B (Farmers)

List the types of crops you grow

…………………………………………

…………………………………………

Before reading the covering letter about the sewage sludge, did you know what sewage sludge or biosolids are?

Yes No

Do you think sewage sludge should be used as a fertilizer?

Yes No

Give reason(s)……………………………………………………………………………

……..……………………………………………………………………………………...

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Have you ever used sewage sludge as a fertilizer?

Yes No

If you use sewage sludge as fertilizer, explain

How do you apply it? ……………………………..……………………………………….

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When do you apply it? ………………………………………………………………….

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How much do you apply it? ……………………..……………………………………….

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Where do you get it? ……………………………………………………………………….

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How do you transport it to your far? ………………………………………………….

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If your answer was no in 4, would you consider using sewage sludge as a fertilizer or as a soil amendment?

Yes No

Give reason(s) ---

If your answer was yes in 4 , would you rate the sewage sludge as fertilizer the same as other commercial fertilizers?

Yes No

Give reason(s) ---

----

8. Do you think that sewage sludge pose a risk to human health.

Agree Neutral Disagree

SECTION B

Do you have flushing toilets in your house?

Yes No

2. Do you have running water in your house?

Yes No

Tell us what you thought happens to the used water that is flushed down the toilet and flows down the drain in a house.

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……………………………………………………………………………………

What effect do you think wastewater recycling has on the quality of life and the environment?

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……………………………………………………………………………………………

Before reading the covering letter about sewage sludge, did you know what sewage sludge or biosolids are?

Yes No

If yes:

Did you know that sewage sludge could be used as a fertilizer?

Yes No

Do you think sewage sludge can make a good fertilizer?

Yes No

Give reason(s)

……………………………………………...……………………………………………….

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…….

Do you think it is safe to use sewage sludge as a fertilizer for:

household vegetables consumed raw

Yes No

household vegetables consumed cooked

Yes No

tobacco

Yes No

vineyards

Yes No

fruits trees

Yes No

Cereal culture

Yes No

Sugar cane

Yes No

Gardens and traffic islands

Yes No

Crops which animals producing milk, meat and eggs graze.

Yes No

Would you buy vegetables grown using sewage sludge as a fertilizer?

Yes No

Give reason(s) .

………………………………………………………………………………………

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………………..

7. The use of sewage sludge as a fertilizer is not risky to human health.

Agree Neutral Disagree

Give reason(s)………………………………………………………………………

……………………………………………………………………..

8. What impact do you think the use of sewage sludge as a fertilizer can have on the soil properties?

………………………………………………………………………………………………