SUSTAINABLE AGRICULTURAL USE OF MUNICIPAL WASTEWATER SLUDGE: MATCHING NUTRIENT SUPPLY AND DEMAND

Report to the
Water Research Commission

by

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This report forms part of a series of two reports. The other report in the series is Sustainable Sludge Land Application and Co-disposal Strategies (WRC Report No.1724/2/12).

PREFACE

Management of sludge from both industrial and domestic origins is an ever growing challenge. The beneficial utilization of sludge is no doubt a better long-term waste management strategy than disposal. Municipal sludge is a source of essential nutrients and it is believed that its agricultural use will play an ever increasing role in future strategies. On the other hand, petro-chemically derived biological sludge emanating from Sasol Secunda also has favourable properties that can aid in the vegetation of the large gasification ash dumps at the Sasol Petrochemical Plant. The beneficial utilization of sludge in both an agricultural and environmental context was therefore the main impetus behind this project titled "Sustainable Sludge Land Application and Co-disposal Strategies". Over the course of three years two focuses evolved: 1) Strategies for sustainable agricultural use of municipal sewage sludge as a source of nutrients, and; 2) The potential of sludge amended combustion coal ash residues as plant growth media.

As a result, the findings of this study are also compiled in two separate reports. The report "Sustainable Agricultural Use of Municipal Wastewater Sludge: Matching Nutrient Supply and Demand" details the quantification of responsible sludge application rates in an attempt to match nutrient supply from sludge to actual crop uptake for sustainable agricultural use. It also reports on the development of a mechanistic crop water balance model (SWB) to which N routines were added to provide a reasoning support tool to this end. The P fertilizer value of several sludges, and N mineralisation as influenced by sludge stability and water potential, are also investigated and reported on.

The second report focuses on the potential use of a petro-chemically derived sludge for reclamation of a coarse ash dump and is entitled "The potential of sludge amended combustion coal ash residues as artificial plant growth media: A laboratory column study to assess the influence of weathering on elemental release". It includes a laboratory column study in which weathering, elemental release and water holding characteristics of co-disposed industrial sludge, fine and gasification mixtures are investigated to assess chemical and physical suitability as potential growing media for plants. The report highlights the benefits and risks of mixing various waste streams in certain ratios, as well as the expected end products as these mixes weather over time.

EXECUTIVE SUMMARY

Agricultural use of sewage sludge is a well-known practice around the world. Sludge acts as a source of plant macro and micronutrients, improves soil structure, and minimizes soil erosion and runoff. In view of these potential agronomic benefits, many countries have developed guidelines to optimize sludge use without compromising environmental sustainability.

Sludge has been used on agricultural lands mainly as a source of the macro elements nitrogen (N) and phosphorus (P). In many parts of the world, sludge is applied according to the N requirements of crops. A large fraction of the N in sewage sludge is, however, in organic form, and needs to be converted into inorganic forms for plant utilization. Hence, N availability for crop uptake from sludge-amended soils depends on the rate of mineralization, and subsequent inorganic N losses via volatilization, denitrification, and leaching. These processes are influenced by type of soil, availability of water, soil temperature, soil pH, and sludge treatment processes at Water Care Works. Therefore, direct extrapolation from specific studies regardless of variability in soil type, climate, and cropping system could compromise both agricultural and environmental sustainability.

The majority of previous studies on beneficial agricultural use of sludge were conducted in the northern hemisphere, where soil-climate combinations are quite different to our southern African conditions. Since processes regulating nitrogen availability for crop uptake from sludge-amended soils are governed by climatic (temperature and water availability) and edaphic factors (soil pH, CEC, texture), locally based trials are essential.

Furthermore, nutrient uptake by a given cropping system is a function of crop species, water availability, climate, soil type, management practice, and farming intensity. The infinite number of combinations among the above-mentioned factors could never be adequately covered by field trials. Therefore, there is a clear need for a mechanistic reasoning support tool (model) to determine environmentally responsible sludge application rates that attempt to match nutrient supply to crop demand under widely ranging conditions.

The current South African sludge utilization guideline permits an annual upper sludge application rate of 10 t ha^{-1}. The guideline, however, encourages field scale local research to fine-tune these stipulations.

The aims and objectives of this study were to:

a) investigate the dynamic nature of the ideal sludge application rate across a range of management practices and cropping systems;
b) investigate the agronomic benefits and potential environmental impacts through nitrate leaching of N based sludge land application approaches;
c) evaluate the sustainability of high surface application rates above crop requirements for turfgrass sod production;
d) quantify the influence of water potential, sludge stability and temperature on N release in order to generate parameters for the Soil Water Balance (SWB) model;
e) add a nitrogen sub-routine to the SWB model;
f) test the reliability of the model as a reasoning support tool for sludge management on agricultural lands;
g) determine the influence of soil and sludge properties on P release/mineralization and phyto-availability;
h) develop a simple method to determine the phosphate fertilizer value of sludge.

Results from this study showed that dryland maize was able to use twice as much and intensive irrigated double cropping systems three times as much N as the supply from sludge at the current South African upper limit. This assumes a sludge N content of 3%. Crop growth, yield, and nitrogen uptake varied across years under dryland maize production in response to the availability of rainfall, especially at high sludge application rates, implying the need for a dynamic nutrient management system. This variation is an indication of the complex interaction between climate, N mineralization and crop response. In contrast, under intensive irrigated systems, crop growth, yield, and N uptake remained relatively stable. It was also evident from this study that N exported with forage from intensive irrigated systems was at least three times higher than similar treatments under dryland maize forage production. Nevertheless, sludge applied at all rates, including the lowest sludge application rate of 4 t ha^{-1} yr^{-1}, resulted in the accumulation of excess P, both under dryland and irrigated cropping systems.

Weeping lovegrass hay yield was highest at high sludge application rates of 16 t ha^{-1}, but such high application rates resulted in the application of N above crop N demand. Sludge application at all rates was associated with excess P accumulation in the soil profile. This calls for monitoring under N based sludge application management for potential P mobility over time, especially in low P fixing soils and soils which received P for an extended period from various sources.

Under turfgrass sod production, increasing sludge application rates up to 67 t ha^{-1} significantly improved turf establishment rate and colour. The ability of sods to remain intact during handling improved as sludge application rate increased to 33 t ha^{-1}, but deteriorated at higher rates. High rates of 100 t ha^{-1} minimized soil loss from the site but caused unacceptably high leaching losses. According to this study, anaerobically digested paddy dried sludge could be applied as high as 33 t ha^{-1} for turfgrass sod production – four times higher than the initial 8 t ha^{-1} limit. This rate improved turf growth and quality with minimal environmental impact through nitrate leaching.

The fertilizer value of sludge is a trade-off between the concentration and availability of the macro nutrients N and P, and the transportation costs from wastewater care works to farms. Ideally sludge should be applied according to crop N demand. But ultimately the maximum amount of sludge applied to a site will depend on the accumulation of total and available P and the risk this poses for pollution, as long as the threat from other pollutants is minimal.

It was evident from this study that N removal varied across a range of cropping systems, seasons, and management practices. In addition, N content varies across a range of sludge types and N mineralization from sludge could take several years before reaching a stable state. Therefore, a dynamic, mechanistic decision support tool is needed in order to accommodate the interaction between the various factors involved.

The N sub-routine incorporated into the existing SWB model was successfully calibrated with acceptable accuracy for dryland maize, an irrigated maize-oat rotation, and dryland pasture. The model was tested against independent sets of data and predicted leaf area index, aboveground biomass, grain yield, aboveground biomass N uptake, and grain N uptake with acceptable accuracy. The model predicted the above mentioned variables for dryland pasture with low accuracy due to the non-mechanistic approach followed by the model when simulating perennial grasses. The N model looks promising, especially for dryland maize and irrigated maize-oat rotation, to be used as reasoning supporting tool for sustainable sludge use on agricultural lands.

It is also imperative that field research is supported with laboratory incubation trials. Model parameters, such as mineralization rates of varying sludge types are generated through incubation trials. The accuracy of model prediction heavily relies on the accuracy of the input parameters.

Results from incubation studies showed that release of N from sludge increased with an increase in percent volatile solids content. In other words, the more stable the sludge, the less inorganic N was released. While release of N is desirable for crop growth, less stable sludges may have a higher potential for nitrate leaching. Moreover, gradual release of nitrates characteristic of more stable sludges is more advantageous for crop uptake.

N mineralization seemed to reach a steady rate at about three weeks after sludge application. The release of N from sludge was influenced by soil temperature and water potential. Greater mineralization was observed at higher temperatures of 45^{o}C and increased with a decrease in water tension; nevertheless, greater nitrification was observed at 25^{o}C. These observations suggest that nitrate accumulation and leaching from sludge-amended soils may be reduced in warm and dry soil conditions.

Incubation and phyto-availability studies showed that sludge products like Agriman and Activated Sludge Pasteurization process (ASP) have similar phosphate plant availability as commercial fertilizers. These products are therefore agronomically viable sources of P.

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CHAPTER 1

INTRODUCTION

1.1 General Background

Population growth in cities and the expansion of industries are causing enormous pressure on wastewater care works (WCW) as a result of the high volume of waste products that needs to be either beneficially used or disposed of in an environmentally sustainable way.

Beneficial use of sewage sludge on agricultural lands is a well-known practice around the world. The benefits include: a source of essential crop nutrients (Muse *et al*., 1991), improves soil structure (Ojeda *et al*., 2007), and minimizes soil erosion and runoff (Muse *et al*., 1991; Ojeda *et al*., 2003). Nutrients applied above crop requirements, however, can be detrimental to plant growth (Brady, 1974) and will ultimately pollute water bodies (Neal *et al*., 2002). In addition, waste products from cities and industrial areas contain pathogens, toxic elements and organic contaminants which can pose a serious health hazard. Therefore, many countries have developed sewage sludge guidelines to optimize agricultural benefits without compromising sustainability.

Beneficial agricultural use accounts for 28% of the total sludge produced from South African WCWs (Du Preez *et al*., 2000). This is despite the enormous pressure on South African WCWs to dispose of or utilize their sludge in an environmentally sustainable way. The poor public perception of sludge use in agriculture as well as the lack of thorough local field scale studies contributes to the low fraction of sludge generated being used in this beneficial way.

1.2 Problem Statement

A large fraction of the N in sewage sludge is organic and must be converted into inorganic forms before plants can utilize it. Hence, N availability for crop uptake from sludge-amended soils depends on the rate of mineralization (Kelley *et al*., 1984), and the subsequent inorganic N losses via volatilization, denitrification, and leaching. These processes are influenced by the type of soil, availability of water, soil temperature, soil pH, and sludge treatment processes at WCWs. Therefore, direct extrapolation of local studies on specific cropping systems using limited sludge types could compromise both the environment and crop production in the long run.

Crop nutrient demand is also species dependent. Actual crop nutrient uptake is regulated by other external factors including water and nutrient availability, climatic, and edaphic factors. Hence, the overall annual crop nutrient uptake from a given field is a function of the complex interaction between the crop type, nutrient availability, water availability, management practice, and farming intensity. There is a plea for local field scale studies in the recently published South African Sludge guideline in order to fine-tune current guidelines based on local findings (Snyman and Herselman, 2006).

1.3 Objectives of the study

The objectives of the study were to:

- investigate the dynamic nature of ideal sludge application rate across a range of management practices and cropping systems.
- investigate the agronomic benefits and potential environmental impacts through nitrate leaching of N based sludge land application systems.
- evaluate the sustainability of high surface application rates above crop requirements for turfgrass sod production.
- quantify the influence of water potential, sludge stability and temperature on N release in order to generate parameters for the SWB model;
- add a nitrogen sub-routine to the Soil Water Balance (SWB) model.
- test reliability of the model as a reasoning support tool for sludge management on agricultural lands.
- determine the influence of soil and sludge properties on P release/mineralization and phyto-availability;
- develop a simple method to determine the phosphate fertilizer value of sludge.

1.4 Motivation

The majority of previous studies on beneficial agricultural use of sludge were conducted in the northern hemisphere, where soil-climate combinations are quite different to our Southern African conditions. Since processes regulating nitrogen availability for crop uptake from sludge-amended soils are governed by climatic (temperature and water availability) and edaphic factors (soil pH, CEC, texture), locally based trials are essential.

Furthermore, nutrient uptake by a given cropping system is a function of crop species, water availability, climate, soil type, management practice, and farming intensity. The vast number of combinations among the above mentioned factors could never be adequately covered by field trials. Hence, there is a need for a mechanistic decision support tool (model).

It is also imperative that field research is supported with laboratory incubation trials. Model parameters, such as mineralization rates of varying sludge types are generated through incubation trial. The accuracy of model prediction heavily relies on the accuracy of these input parameters.

The report is presented in the following order:
Chapter 2 is the literature review. It provides a general background followed by discussion on sewage sludge types and characteristics, as well as beneficial agricultural use. It presents a brief description of the model before concluding with motivation and rationale of the study.

Chapter 3 presents the effect of sludge stability, temperature, and soil water potential on nitrogen mineralization in sludge-amended soils. This chapter details the net inorganic N release by various sludge types across a range of temperature and soil water potentials.

Chapter 4 investigates phosphate release and the relative phosphate fertilizer value of various agricultural sludge products.

Chapters 5, 6 and 7 present local field scale cropping systems studies on municipal sludge amended soils. These chapters highlight the contrast between agronomic benefits and environmental impacts of varying sludge application rates across a range of cropping systems. The varying cropping systems include an intensive irrigated maize-oat rotation, dryland maize, dryland pasture, and turfgrass sod production.

Chapter 8 covers incorporation of N and P routines into a local generic crop model. This chapter presents a detailed description of the model.

Chapter 9 deals with model calibration, corroboration, and practical applications. Initially the model is calibrated using data collected from field studies and literature. The calibrated model is then tested against independent data sets collected in this study. Finally, the model is used to assess long-term environmental and agronomic implications of using similar sludge application rates on various cropping systems.

Chapter 10 summarizes important findings and forwards recommendations for further studies.

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CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

Beneficial agricultural use of municipal sludge is a common practice in various regions of the world. Beneficial agricultural use of sludge, however, only accounted for 28% of the total sludge produced in South African wastewater treatment plants (Du Preez et al., 2000). A combination of factors including low sludge quality due to pathogen and pollutants as well as lack of sufficient local information on the beneficial agricultural use of sludge contributed to this relatively low usage in agriculture.

Sludge is often used in agriculture as a source of macro nutrients especially N and P. A large fraction of the N in sewage sludge is organic and must first be transformed to inorganic N before it can be available for crop uptake. The processes involved in the transformation of N from organic to inorganic forms are influenced by factors such as soil pH, soil texture, bulk density, soil water, and temperature (Gilmour and Gilmour, 1980; Sims and Boswell, 1980; Parker and Sommers, 1983; Artiola and Pepper, 1992; Barbarick *et al*., 1996; Janssen, 1996; Leiros *et al*., 1999). In addition, all major processes involved in the nitrogen cycle of agricultural lands including crop nutrient uptake are influenced by water availability, soil temperature, climatic factors, soil type, management practices, crop type, and farming intensity.

Understanding the complex interactions between the various factors involved in this nutrient cycle is pivotal for safe and sustainable agricultural use of sludge. Therefore direct extrapolation of field studies across a range of agro-climatic conditions could have negative agricultural and environmental impacts. On the other hand, conducting medium to long-term field studies across various ecological zones, soil types, sludge types, and cropping systems is not only expensive but also logistically impractical. A validated model could, however, play a significant role in extrapolating field results across a range of climatic conditions, soil types, and cropping systems.

The aim of this chapter is to highlight up-to-date knowledge regarding the overall processes involved during beneficial agricultural use of sludge. This comprises the various N transformations in sewage sludge-amended soils, P availability of these soils, pros and cons of beneficial agricultural use of sludge, and description of N and P models.

2.2. Sewage Sludge Types, Characteristics, and Agricultural Use

2.2.1 Sewage sludge types

The characteristics of sewage sludge vary depending on its treatment. *Primary sewage sludge* is the result of the primary settling of solids from wastewater, and which has not yet undergone any treatment process. It has unpleasant characteristics, is full of pathogens and is not recommended for land application (US EPA, 1995; Spinosa and Vesilind, 2002).

Secondary sewage sludge has undergone substantial degradation and it involves a primary clarification process followed by biological treatment and secondary clarification (US EPA, 1995). *Tertiary sewage sludge* is generated through further processing by chemical precipitation (using aluminium, iron, lime, or organic polymers) and filtration (US EPA, 1995). The most common sludge treatment processes with their corresponding effects on sludge properties and land application practices are summarized in Table 2.1.

Table 2.1. Effects of sewage sludge treatment processes on sludge properties and land application practices (Adapted from US EPA, 1984).

Sludge category	Treatment process	Effect on sewage sludge properties	Effect on land application practice
Primary	Thickening	Increases solid concentration by removing water, thereby lowering sewage sludge volume	Lowers sewage sludge transportation costs.
Secondary	Digestion (Anaerobic and Aerobic)	Reduces the volatile and biodegradable organic content and the mass of sewage sludge by converting it to soluble material and gas. Reduces pathogen levels and odour.	Reduces sewage sludge volume. Therefore lowers sludge transportation costs
Tertiary	Alkali stabilization	Raises sewage sludge pH. Temporarily decreases biological activity. Reduces pathogen levels and controls putrescibility and odour.	High pH immobilizes metals as long as pH levels are maintained, thus reducing heavy metal leaching and crop uptake
	Conditioning	Improves sewage sludge dewatering characteristics. May increase the mass of dry solids to be handled. May improve sewage sludge compatibility and stabilization.	Polymer-treated sewage sludge may require special operational considerations at the land application site.
	Dewatering	Increases solid concentration of organic sewage sludges to 15-40%, and for some inorganic sewage sludges to 45% or more. Improves ease of handling by converting liquid sewage sludge to damp cake. Some nitrogen and other soluble materials are removed with the water.	Lowers sewage sludge transportation costs.
	Composting	Lowers biological activity. Can destroy most pathogens. Degrades sewage sludge to humus-like material. Increases sewage sludge mass due to addition of bulking agent.	Excellent soil conditioning properties. Significant storage space usually needed. May contain lower nutrient levels than less processed sewage sludge.
	Heat drying	Disinfects sewage sludge. Destroys most pathogens. Slightly lowers potential for odour and biological activity.	Greatly reduces volume of sewage sludge. Therefore lowers transportation costs.

2.2.2 Benefits of using sewage sludge in agricultural lands

The organic matter in sewage sludge applied to agricultural lands improves the soil's *physical characteristics* and structure, thereby improving soil porosity, stability of aggregates and water retention (Ojeda *et al*., 2007). It plays a significant role in improving the water holding capacity of sandy soils, aeration and water movement in clay soils, eases plant root penetration, and reduces runoff and soil erosion (Muse *et al*., 1991). It also improves soil *chemical characteristics* by increasing cation exchange capacity and also acts as a source of macro and micro nutrients for plants (Mininni and Santori, 1987).

2.2.3 N in sewage sludge

A large fraction of the N in sewage sludge is in organic form, and needs to be converted into NH_4 and NO_3 before it could be taken up by plants. Other processes that affect plant N availability include ammonia volatilization and denitrification losses. Sewage sludge treatment and handling affects the characteristics of N in various ways as indicated below:

i) Dewatering reduces the amount of water soluble inorganic N (NH_4 and NO_3),
ii) Heat or air drying and lime treatment reduce NH_4 through volatilization in the form NH_3, but does not affect NO_3,
iii) Aerobic treatment facilitates the conversion NH_4 to NO_3, while anaerobic conditions inhibit oxidation of NH_4 to NO_3.

a) Nitrogen mineralization

Mineralization is the transformation of N from its organic state to the inorganic forms of NH_4^+ or NH_3 by heterotrophic soil organisms consuming nitrogenous organic substances as a source of energy (Lutz, 1965; Jansson and Persson, 1982). The release of ammonium during mineralization is driven by the need of micro-organisms for carbon (Sprent, 1987).

Organic N in biosolids is believed to be made up of various organic components that differ in their rate of mineralization. These components are usually called pools. In most cases, researchers categorise organic N into three pools: i) a rapid turnover pool made up of amino acids and proteins, ii) a slow turnover pool, and iii) a highly resistant pool that is often not detected in short period soil incubation trials (Henry *et* *al*., 1999; Smith *et al*., 1997). The size of each pool depends on the type of sludge and sludge treatment process. The mineralisation of each pool is influenced by climatic (temperature and water content) and edaphic factors (soil pH, texture, structure, and soil fauna) (Leiros *et al*., 1999; Janssen, 1996; Parker and Sommers, 1983; Trindade *et al*., 2001).

N mineralization is enhanced as the soil water content increases from permanent wilting point to field capacity and as the temperature increases from cold to warm (Vinten and Smith, 1993). Under controlled warm and moist laboratory conditions, the rapidly mineralizable pool decomposes in two to four weeks. In situations where either the soil water or temperature is not ideal, this extends from two to six months. The slowly mineralizable and highly resistant pools, however, take months or even years for complete decomposition (Henry *et al*., 1999).

b) Ammonia volatilization

Ammonia volatilization is the gaseous loss of NH_3 from the soil surface to the atmosphere (Haynes and Sherlock, 1986). Volatilization takes place as a result of the difference in concentration gradient between ammonia in solution and the ambient air (Freney *et al*., 1983) and can be very large. Fine *et al*. (1989) reported a loss of 87% of the mineralised nitrogen through ammonia volatilisation from activated sewage sludge. Other studies conducted by Beauchamp *et al*. (1978), indicate that about 20% of total nitrogen applied was lost as ammonia during the first week of decomposition from anaerobically digested liquid sludge.

Ammonia volatilization is highly dependent upon the sludge treatment process, and whether the sludge is left on the surface or incorporated into the soil. Volatilization is enhanced in the presence of higher soil water content, air temperature, and wind speed (Henry *et al*. 1999) and suppressed at low pH, cool temperatures, and low wind speeds (Freney *et al*., 1983).

c) Denitrification

Denitrification is the gaseous loss of nitrogen in the form of nitrous oxide (N_2O) and dinitrogen (N_2), mediated by microbial action (Delwiche, 1981; Tisdale *et al*., 1985). The microorganisms responsible for denitrification need nitrogen in the form of nitrate, soluble organic carbon and water-logged (anaerobic) conditions (Henry *et al*., 1999). Two of the above conditions (nitrate and soluble carbon) are prominent in sludge-amended soils which could enhance denitrification under water-logged conditions.

2.2.4 Nature of P in sewage sludge

Phosphorus in municipal wastewater is mainly composed of inorganic (orthophosphates and polyphosphates) and organic components (Ekama *et al*., 1984). Orthophosphates comprises 70 to 90% of total P both in raw and settled municipal wastewater (Ekama *et al*., 1984). Orthophosphates are available for biological metabolism without further breakdown while polyphosphates usually need to undergo hydrolysis and revert to orthophosphates (Crites *et al*., 2005). Generally the concentration of total P in municipal wastewaters may range between 4 to 12 mg L^{-1} of which a third is organic. Secondary wastewater treatment can only remove 1 to 2 mg L^{-1} P (Tchobanoglous *et al*., 2003). Considering the soil solution P concentration benchmark of 1 mg L^{-1} for wastewater discharge to rivers and streams (Sims and Pierzynski, 2000), direct discharge of wastewater effluents after secondary treatments without further P removal processes could cause eutrophication of the receiving water bodies.

Wastewater treatment plants use either chemical or biological phosphorus removal methods to reduce the concentration of P below the benchmark before they can discharge to rivers and streams. Chemical precipitation is used to remove the inorganic forms of phosphate by addition of coagulants. The most commonly used coagulants are calcium, aluminium, and iron (Tchobanoglous *et al*., 2003). With biological phosphorus removal, P accumulating microorganisms are encouraged to grow and consume P and are subsequently removed (Ekama *et al*., 1984; Storm *et al*., 2004).

Sewage sludge treatment methods which include chemical or biological nutrient removal processes influence the availability of P for crop uptake (Frossard *et al*., 1996; Maguire *et al*., 2001; Penn and Sims, 2002; Kirkham, 1982; McCoy *et al*., 1986). The predominant form of P in sludges that underwent tertiary treatment is inorganic P (McLaughlin, 1984). Chemicals used in tertiary treatments such as Al or Fe salts, decrease the crop available P fraction (Elliott *et al*., 2002; Häni, *et al*., 1981; Kyle and McClintock, 1995). The addition of lime to the Fe or Fe + Al treated sludges, however, increases the concentration of the easily soluble P fraction (Penn and Sims, 2002).

A significant build-up of total (Penn and Sims, 2002; Tesfamariam *et al*., 2009) and Bray- 1P (Herselman *et al*., 2005) was reported in sludge amended soils as opposed to their corresponding control treatments, which did not receive sludge. The labile fraction of soil P (M3-P, M1-P, FeO-P, and water soluble P) was especially high for soils that received biological nutrient removal sludge, compared to the control and treatments that received Fe and lime or Fe and Al treated sludge (Penn and Sims, 2002). This shows that detailed knowledge of the sludge treatment processes is crucial for sustainable agricultural use of sludges.

2.2.5 Classification of sludge for use on agricultural lands

Sludge products from cities and industrial areas contain pathogens, toxic elements and organic contaminants which can pose serious health hazards. Heavy metals of concern that can be found in sewage sludge include As, Cd, Cu, Pb, Hg, Mo, Ni, Se, Zn, and Cr. Organic contaminants that can be found in sewage sludge include pesticides (chlordane, endrin etc.), herbicides (2.4-D, 2.4.5-TP Silvex), volatiles (benzene, carbon tetrachloride etc.), and semi volatiles (O-Cresol, m-Cresol etc.) (U.S. EPA, 1995). Sewage sludge must therefore be used within certain guidelines, and many countries have published sewage sludge regulations for land application (U.S. EPA, 1995; Wang, 1997; Krogmann *et al*., 2001; Snyman and Herselman, 2006).

The current South African sludge guideline (Snyman and Herselman, 2006) considers three categories (*microbiological, stability*, and *pollutant* classes), to assess the appropriateness of a given sludge for agricultural use. According to the *microbiological* classification, a sludge with *microbiological* class “A” can be applied without restriction at agronomic rates, class “B” may not be appropriate to use for some crops with edible parts below the soil surface, and class “C” can only be used for agricultural application if *stability* class 1 or 2 is achieved and this is restricted for certain crops (Snyman and Herselman, 2006).

According to the *stability* classification, sludge with *stability* class “1” can be applied without restriction at agronomic rates, class “2” can also be applied at agronomic rates but additional management systems need to be in place. However, class “3” cannot be used for agricultural application (Snyman and Herselman, 2006).

According to the *pollutant* classification, sludge of class “a” can be applied without restriction at agronomic rates; class “b” can also be applied at agronomic rates if analyses of the receiving soil prove it can accommodate the load. Class “c”, however, cannot be used for agricultural purposes (Snyman and Herselman, 2006).

2.2.6 Sludge application rates on agricultural lands

Generally, sewage sludge application rates on agricultural lands are based on crop N requirements (Barbarick *et al*., 2010). In most US States, agricultural sludge application rates are based on the N requirements of crops and the concentrations and application rates of other trace elements, as defined in the US EPA 503 rule (US EPA, 1994). However, phosphorus based management is considered in areas with high soil P test (Sims *et al*., 2000). In South Africa sludge application rate is based on crop N requirements and the concentration and application rates of trace elements, as defined in the guideline (Snyman and Herselman, 2006). Studies conducted by Herselman *et al*. (2005) on selected sites where sludge has been applied for long periods indicated high levels of P in the form of Bray-1P to depths of 0.5 m. Hence, the potential environmental impacts that may occur on water bodies over time from P carried with percolation water below the active root zone or surface runoff needs attention.

Previous research by Kelling *et al*., (1977); Pierzynski (1994), Peterson *et al*. (1994) and Maguire *et al*. (2000a,b) indicates that continuous sludge applications based on nitrogen demand will cause soil P to accumulate to levels above those needed for optimum crop production. Phosphorus availability from sludge-amended soils, however, depends on the processes the sludge underwent in the wastewater care works (Kyle and McClintock, 1995; Maguire *et al*., 2001; Jokinen, 1990; Soon *et al*., 1978b; Kirkham, 1982; McCoy *et al*., 1986; Frossard *et al*., 1996).

A typical sludge guideline that considers the potential future impacts of P accumulation in the soil is that of the Province of Ontario, Canada. This guideline requires soil monitoring for possible P accumulation. They recommend that bicarbonate extractable P levels should not exceed 60 kg P ha^{-1}, the concern being P contamination of surface waters by soil erosion or runoff (McLaughlin, 1984).

2.2.7 Sludge application timing and methods

It is advisable to schedule sludge applications on agricultural lands around the time of tillage or planting. However, it depends on the type of sludge, type of soil, crop, climate, and method of application (Shepherd, 1996). Correct sludge application timing is essential for efficient use of nutrients and to minimise possible ground or surface water pollution (Evanylo, 1999).

Sludges are commonly applied to agricultural lands either on the soil surface (through spraying or spreading) or incorporated into the soil (through injection or agricultural implements). Surface application is most commonly used on pastures, range and forest lands. Sludge incorporation, on the other hand, is most commonly used for agronomic crops (Evanylo, 1999).

Surface application of liquid sludge is conducted using tractor drawn tank wagons, special applicator vehicles equipped with flotation tyres, or irrigation systems. It is usually restricted for use in areas with slopes less than 7 percent. The disadvantages of spraying liquid sludge on the surface are mainly potential odour problems and the reduction in the aesthetic value of the application site. To avoid the risk of runoff losses and excess leaching below the root zone, liquid sludge is advised to be split applied rather than single large volume applications (Evanylo, 1999).

Liquid sludge can also be injected below the soil surface. This method minimizes odour problems, reduces ammonia volatilization, minimizes runoff losses and can be used in areas with slopes of up to 15 percent. Liquid sludge injection can be conducted using tractor-drawn tank wagons with injection shanks or tank trucks fitted with flotation tyres and injection shanks. Dewatered sludge is usually surface applied to crop lands using equipment similar to that used for applying limestone, or animal manures. The sludge is then incorporated into the soil by ploughing (Evanylo, 1999).

2.2.8 International and local experiences with beneficial agricultural use of sludge

a. Grain crops

A range of studies have shown that maize stover and grain yield increased with increase in sludge application rate, regardless of the soil type (Cunningham *et al*.

1975; Kelling *et al*. 1977; Binder *et al*. 2002; Lavado *et al*. 2006; Bozkurt 2006). In other independent studies, the response was nonlinear, with relatively small yield increases reported at high application rates (Soon *et al*., 1978a; Binder *et al*., 2002). The cause for the decline in grain yield at very high sludge application rates was reported to be salt toxicity (Cunningham *et al*., 1975).

Crop response to an increase in sludge application rate is controlled by water availability. This has been well demonstrated by Soon *et al*. (1978a), where crop response to sludge rates of more than 200 kg N ha^{-1} was minimal under dryland conditions. In contrast, under irrigated conditions, maize grain yield increased as N supply from sludge increased up until 400 kg ha^{-1} N (Binder *et al*., 2002). Significant accumulation of nitrate in the top 0.75 m of the soil profile was reported in the former case (Soon *et al*., 1978a). This is in contrast to the irrigated system where there was barely any nitrate accumulation due to high crop N uptake (Binder *et al*., 2002). In the latter case, significant nitrate accumulation was evident only at sludge application rates that supplied 950 kg N ha^{-1} mainly as a result of the decline in the crop nitrogen use efficiency.

b. Pastures

It is well documented that sludge application improves dry matter yield of pasture grasses (Kelling *et al*., 1977; Soon *et al*., 1978a; Michalk *et al*., 1996; Sullivan *et al*., 1997; Zebarth *et al*., 2000; Cogger *et al*., 2001). Similar to agronomic crops, pasture response to sludge application rate depends on initial soil background nutrient concentration, crop nutrient demand, climate, and management practices. For instance, a significant increase in tall fescue (*Festuca arundinacea Schreb.*) dry matter yield was reported as sludge application rate increased to 20 t ha^{-1} (905 kg N ha^{-1}) in an area receiving 1020 mm rainfall supplemented with irrigation as required (Cogger *et al*., 2001). In other studies conducted by Sullivan *et al*. (1997), prairie grass (*Bromus uniloides (Willd)*) dry matter yield increased as sludge application rate increased to 26.9 t ha^{-1} (1064 kg N ha^{-1}) in an area receiving 1232 mm rainfall. In contrast, rye (*Secale cereale* L.) yield response to an increase in sludge application rate was positive only up to rates of 7.5 t ha^{-1} in a silty loam soil receiving about 500 mm (Kelling *et al*., (1977).

A high sludge application rate of 20 t ha^{-1} yr^{-1} (>800 kg N ha^{-1}) that improved tall fescue dry matter yield, however, caused high nitrate accumulation in the soil profile

(Cogger *et al*., 2001). In addition to the obvious negative environmental impacts associated with high sludge application rates, there are also some concerns related to fodder quality (nitrate concentration in grass). These concerns are extended to the health of workers harvesting forage and animals consuming the forage. Studies conducted by Soon *et al* (1978a) showed that high sludge application rates that supplied 1600 kg N ha^{-1} increased the concentration of NO_3-N in the grass to hazardous levels for livestock consumption (>3 mg g^{-1}). Regarding the health of workers and foraging animals, studies conducted by King and Morris (1972) showed that the forage was essentially coliform-free at harvest time, indicating that there should be little danger of disease transmission to workers handling the harvested forage and the animals consuming the forage. This, however, depends on the microbiological classification of the sludge and time interval between sludge application and harvest.

c. Turf production

Application of composted sludge to various turfgrass species planted on different soil types has improved turf growth and quality. For instance, the incorporation of composted biosolid in a disturbed urban soil hastened the establishment of two turfgrass species, Kentucky bluegrass (*Poa pratensis* L.) and perennial ryegrass (*Lolium perenne* L.) (Loschinkohl and Boehm, 2001). Similarly, composted biosolid incorporated into a sandy loam B-horizon improved kentucky bluegrass turf vegetative cover, colour, and density (Linde and Hepner, 2005). It was also evident from studies conducted by Landschoot and McNitt (1994) that increasing composted biosolid application rate increased kentucky bluegrass turf cover dramatically (greater than 80% one month after seeding). In addition to improving turfgrass quality, the addition of composted sludge increased soil organic matter content, bulk density, and increased water infiltration rate compared with an unamended control (Landschoot and McNitt, 1994; Cheng *et al*., 2007).

2.3 Nutrient Modelling

Modelling the dynamics of nutrients in the soil system dates back to the early 70s (Shaffer *et al*., 2001). The first integrated soil system N models include that of Dutt *et al*. (1972) in the USA and Beek and Frissel (1973) in the Netherlands. Since then, many N models have been developed for varying objectives with different levels of complexity and accuracy. Nitrogen models vary from purely research oriented to management and screening tools (Shaffer *et al*., 2001).

The development of various N and P models with differing levels of complexities has made significant contribution in understanding the N and P cycles in crop-soil systems. N and P models have been developed to simulate N and P cycles in the soil-crop system under irrigated conditions, dryland conditions or both. At the same time, some N and P models are crop specific while others are generic and could simulate crop growth and N and P cycles under crop rotation.

This project required a cropping systems model that could simulate N and P cycles to guide responsible sludge management. The reason for this is that nutrient management, salt management and irrigation are inextricably linked. There is also a need to explore trade-offs between nutrient management, optimal yields and pollution through nitrate leaching. In addition there was no generic crop model available with the option for perennial grass management practices such as cutting hay or forage at a selected growth stage or day interval.

For the reasons above, it was decided to add N and P modules to the existing SWB model – a crop growth / irrigation scheduling model that has already been validated for various crops and grasses under irrigation and dryland conditions in South Africa. The N simulation approaches and algorithms were obtained primarily from CropSyst (Cropping Systems Simulation Model) (Stöckle *et al*., 2003), since CropSyst and SWB have similar backgrounds and approaches, having both grown out of work done by Prof. Gaylon Campbell from Washington State University. The P simulation approach was obtained and followed from the GLEAMS and SWAT models.

Generally, mechanistic models provide a better understanding of the nutrient cycle by mechanistically describing the major processes involved in the nutrient cycle. The main difference between models is thus the difference in the degree of sophistication in the description of these major processes (Frissel and Van Veen, 1982). Some are used for short term seasonal soil nutrient dynamics simulations while others for predicting long-term nutrient dynamics of decades and longer. Details of the processes involved and the equation used in the model are presented in Chapter 8 under “Model description”.

2.4 Conclusion

Municipal sewage sludge is used as a source of nutrients in agriculture worldwide. Nutrients applied above crop demand, however, could be detrimental to plant growth

(Brady, 1974) and will ultimately pollute water bodies (Neal *et al*., 2002). Hence, nutrient management that aims to match supply and demand as closely as possible is vital for sustainable agricultural use of sludge. Crop nutrient uptake is a function of crop species, water availability, climate, management practice, and farming intensity. Ideally, site and crop specific studies ought to be conducted. However, this is largely impractical.

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CHAPTER 3

EFFECT OF SLUDGE STABILITY, TEMPERATURE AND SOIL WATER POTENTIAL ON NITROGEN MINERALIZATION IN SLUDGE-AMENDED SOIL

3.1 Introduction

Proper handling of sewage sludge is particularly problematic not only because of the bulky nature of the large amounts generated every year, but also because improper handling could present certain health, environmental and social challenges. For many decades, research has focused on proper disposal or reuse of sewage sludge. Particularly, utilization of sludge on croplands has been evaluated and has become a common practice because of the many benefits associated with its agronomic use. Land application of sludge is seen as a panacea for waste management problems and has been shown in many cases to improve soil physical characteristics, stimulate microbial activity and add plant nutrients to the soil (Boyle and Paul, 1989; Webber *et al*., 1996; Peterson *et al*., 2003). Moreover, the high organic nitrogen contents of sludge makes it an important contributor to plant-available inorganic forms of nitrogen, and have reduced farmers' dependence on nitrogenous fertilizers.

Much of the nitrogen in sludge, however, is bound in organic forms, and can only become available for plant uptake after these have been mineralized by soil micro-organisms. Determining pattern and rate of N mineralization is crucial in deciding amount and timing of sludge additions to fields in order to match nutrient supply to crop needs. Uncertainties about N-mineralization rates from sludge could cause either a deficiency of N or excessive nitrate levels in soil, leading to problems such as poor crop growth in the former, and nitrate leaching and groundwater contamination in the latter. Mineralization of organic N in sewage sludge is a complex process, dependent on environmental factors such as moisture and temperature, and substrate factors such as the physical and chemical properties of the sludge itself and the medium in which it is applied (Serna and Pomares, 1992).

Information about the dynamics of N release from sludge is an important aspect of sludge characterization (Peterson *et al*., 2003) and because rate of N mineralization varies with individual sludges (Serna and Pomares, 1992), it is pertinent to establish individual mineralization rates for sludges commonly applied to crop fields. Such wide-spectrum N-release studies are critical in these times when land application of sludge is gaining impetus. This will serve as blueprint in assisting farmers and policy makers in making informed decisions on how much of a particular sludge to apply and at what frequencies for specific cropping systems.

Several studies have been conducted by our team to evaluate the dynamics of nitrogen in sludge-amended soils. One of the early endeavours was a study to compare nitrogen mineralization from sludge and commercial nitrogen fertilizer, the results of which have been reported in Van Niekerk (2004) and Snyman and van der Waals (2004). That study aimed at evaluating the pollution hazards of nitrates from sludge and commercial fertilizer. Sludge was applied to soil at 5, 10 and 20 tons ha^{-1} and commercial fertilizer (limestone ammonium nitrate (28% N)) was applied at three levels equivalent to 30% of the N content of the corresponding sludge treatments. This approach was based on the assumption that about 30% of the N in sludge would be available in the short term (WRC, 1997). These trials found that in total, more nitrate was available from sewage sludge over time, than from "equivalent" commercial fertilizer doses (Fig. 3.1). The research brought to light the fact that sludge can act as a slow release nitrogen source thereby enhancing the efficiency of plant nutrient uptake and reducing the risk of nitrate leaching.

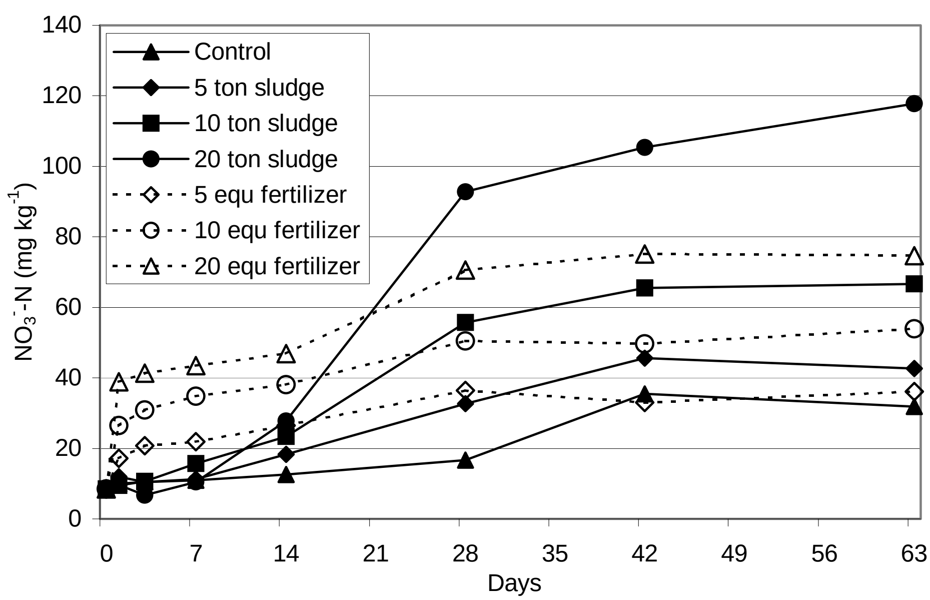


Figure 3.1 Total extractable NO_3^--N content as influenced by incubation time and different rates of sludge and inorganic N fertilizer (Van Niekerk, 2004)

It became imperative, therefore, to improve our understanding of nitrogen release dynamics from sludges. Investigation into the effect of environmental conditions on release of nitrogen from sludge was a logical next step. This report presents a brief account of research approaches and key findings from these recent studies.

The following aims were pursued:

i) To assess the influence of sewage sludge stabilization on nitrogen release from sludge;

ii) To determine the net inorganic nitrogen release from sludge-amended soil as a function of temperature and water potential;

iii) To estimate sludge nitrogen mineralization rate constant and half-life.

3.2 Materials and Methods

3.2.1 Soil and sludges

The soil used for incubation was a red sandy clay loam, taken from a 0-20 cm depth of a soil profile on the Hatfield Experimental Farm of the University of Pretoria located at 25° 45’ S and 28° 16’ E. The soil sample was air-dried, ground and passed through a 2 mm mesh sieve, then analyzed for selected soil physico-chemical properties shown in Table 3.1.

Table 3.1 Selected physical and chemical properties of experimental soil

Parameters analysed	**Values**	**Parameters analysed**	**Values**
pH	5.80	P (mg kg^{-1})	34.4
EC (mS m^{-1})	0.12	Ca (mg kg^{-1})	940
Clay (%)	30.0	Mg (mg kg^{-1})	270
Silt (%)	19.2	Na (mg kg^{-1})	8.0
Sand (%)	50.8	K (mg kg^{-1})	75.0
Total N (%)	0.03	NH_4-N (mg kg^{-1})	8.64
Org C (%)	0.85	NO_3-N (mg kg^{-1})	52.3

Three different sludges were used for soil amendment:

1. Vlakplaas anaerobically digested and paddy dried sewage sludge
2. Olifantsfontein activated, partially digested and belt pressed sewage sludge
3. Sasol activated, anaerobically digested industrial sludge

In terms of stability, Vlakplaas sludge was more stable than Olifantsfontein sludge, which was more stable than Sasol sludge as indicated by their volatile solid percentages (Table 3.2).

The sludges were dried, ground to pass through a 2 mm sieve and analysed for selected chemical properties. Results are shown in Table 3.2, while Table 3.3 shows a detailed distribution of nitrogen forms in the different sludges used.

Table 3.2 Selected chemical properties of sludge used in incubation trial

Parameters analysed	**Source of sludge**		
	Vlakplaas	**Olifantsfontein**	**Sasol**
pH	6.3	6.7	5.8
Total N (%)	1.93	5.33	7.91
Total C (%)	11.6	29.9	38.9
C : N	6	5.6	4.9
Total P (%)	2.43	3.97	0.64
Ca (%)	1.58	3.32	1.07
K (%)	0.11	0.45	0.35
Na (%)	0.14	0.13	0.27
Mg (%)	0.17	0.57	0.35
NH_4-N (%)	0.66	0.12	0.47
NO_3-N (%)	0.24	0.01	0.01
Org N (%)[a]	1.03	5.2	7.43
Total solids (%)[b]	48	13	10
Volatile solids (%)[c]	44	73	91

[a]*Org N calculated by difference: Total N – (NH_4-N+ NO_3-N)*

[b]*Total solids: Mass left behind after drying samples in an oven at about 103°C.*

[c]*Volatile solids: Portion of total solids lost after subjecting dry samples to heating in a furnace at about 550°C.*

Table 3.3 Distribution of nitrogen forms in sludge

	mmol kg^{-1}			
Source of sludge	**Total N**	**NH_4^+**	**NO_3^-**	**Organic N**
Vlakplaas	1379	367	38.7	973.3
Olifantsfontein	3807	66.7	1.6	3738.7
Sasol	5650	261	1.6	5387.4

3.2.2 Treatments

Treatments consisted of sludge-amended soil subjected to three levels of temperature $T_1 = 10^oC$, $T_2 = 25^oC$, $T_3 = 45^oC$; and three water potentials W_1 = -10 kPa,

W_2 = -100 kPa, W_3 = -1000 kPa for each temperature. Treatments were replicated three times.

The quantity of water corresponding to a particular water potential was established gravimetrically based on a water retention curve determination using a ceramic pressure plate extractor (Dane and Hopmans, 1999).

3.2.3 Incubation procedure and sample collection

The amount of sludge at a rate equivalent to 10 ton ha^{-1} on a dry mass basis (0.31g, 0.23g and 0.22g) for Vlakplaas, Olifantsfontein and Sasol sludges respectively was added and thoroughly mixed with pre-incubated 50 g soil samples. These amounts were calculated assuming a bulk density of 1.19 g cm^{-3}, a plough layer depth of 20 cm, and including a moisture correction factor that differed slightly among the sludges. Water potentials were adjusted and then samples incubated at selected temperatures for 56 days. A cold chamber with adjustable temperature was used for the 10^{o}C setting. For the 25^{o}C treatment a controlled incubation room was used, and for 45^{o}C, samples were kept in an oven. Water potential was monitored by weighing the treatments and if necessary, adding an amount of distilled water to keep the treatments at the required water potential. Samples incubated at 10^{o}C and 25^{o}C were monitored for water potential every two days while samples incubated at 45^{o}C were monitored daily because of the latter’s tendency to lose water more rapidly. During this time of monitoring water potential, samples were also aerated with an electrical fan for 1 minute. The three replicate samples of each treatment were sacrificed after incubation times of 0, 1, 7, 14, 28 and 56 days for determination of inorganic N by micro Kjeldahl steam distillation (Mulvaney, 1996) after extraction with a 1M potassium chloride solution.

3.2.4 Generation of parameters for modelling N dynamics

N mineralization was described using first-order kinetics (Stanford and Smith, 1972):

$$N_{(t)} = N_0 (1- e^{-kt})$$

Where $\mathbf{N_{(t)}}$ = net N mineralized at time t;

$\mathbf{N_0}$ = potentially mineralizable N;

k = first-order rate constant and

t = incubation time.

The half-life $t_{1/2}$ is given by (ln 2)/k.

3.2.5 Statistical analysis

A general linear model procedure of the SAS statistical program (SAS Institute, 1990) was used to test for significance of differences among treatments. Treatment means for NH_4^+, NO_3^- and net N release were listed from high to low and compared using the least significant difference (LSD) test.

3.3 Results and Discussion

3.3.1 Net inorganic N released after a 56-day incubation period

Fig. 3.2 (a) shows net N mineralized from sludge organic N and Fig. 3.2 (b) shows N liberated as a result of the mineralization of organic N plus the inorganic N already in the sludge.

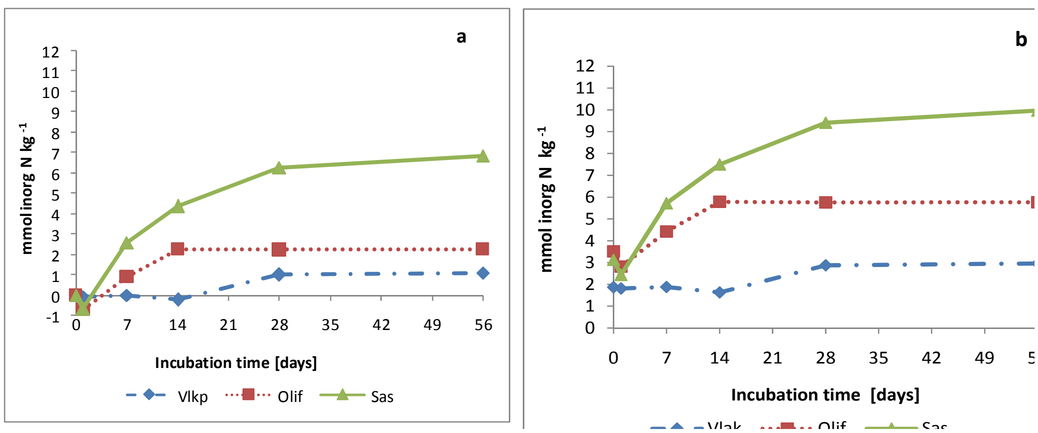


Figure 3.2 (a) Net N mineralized; (b) net N released (mineralization plus inorganic N in sludge) from Vlakplass (Vlkp), Olifantsfontein (Olif) and Sasol (Sas) sludges at 25°C and -10 kPa

For determination of the differences in N mineralization from the sludges, the 25°C temperature level and -10 kPa water potential were chosen since from the literature, these represent conditions for optimal N mineralization. Release of total inorganic N from sludge was in the order Sasol>Olifantsfontein>Vlakplaas. These results suggest that release of N from sludge increased with an increase in percent volatile solid content. In other words, the more stable the sludge, the less inorganic N was released. Even though Vlakplaas had higher inorganic N compared to Olifantsfontein and Sasol (Table 3.3), mineralization of N was higher from the two less stable sludges.

A subsequent study by our team corroborates these findings. In a three week incubation study of nitrogen release from sludge materials under two temperature levels (25 and 45°C) and three soil water potentials (-100, -50, and -20 kPa) we observed that apart from the ammonium-enriched commercial ASP sludge, inorganic N released closely followed the volatile solid content of the sludges (Figure 3.3). The other sludges used in that experiment were Vlakplaas and Daspoort municipal sludges, Agriman commercial sludge, and Sasol sludge of petrochemical origin.

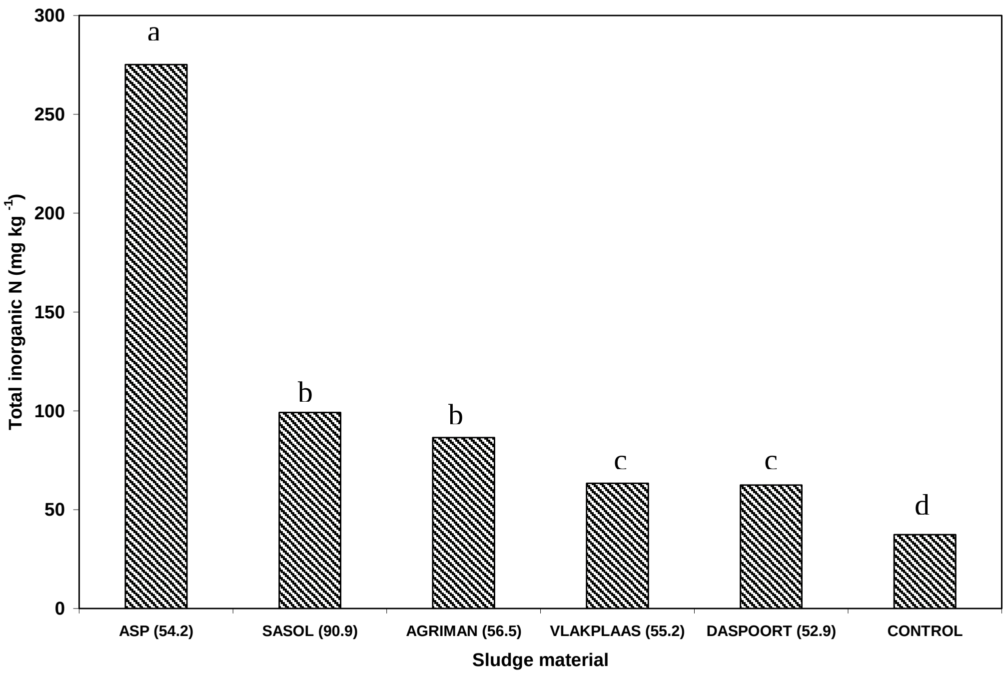


Figure 3.3 Total inorganic nitrogen in soil amended with various sludge materials, and an unamended control after 21 days of incubation. (Percent volatile solid content of sludges in brackets; letters show significant differences at 5% probability level)

3.3.2 Temperature and water potential effects

Statistical results indicated that there was a significant interaction between temperature and water potential. Figures 3.4 to 3.6 and Tables 3.4 to 3.6 show results from the 56-day incubation for the different sludges.

(i) Vlakplaas

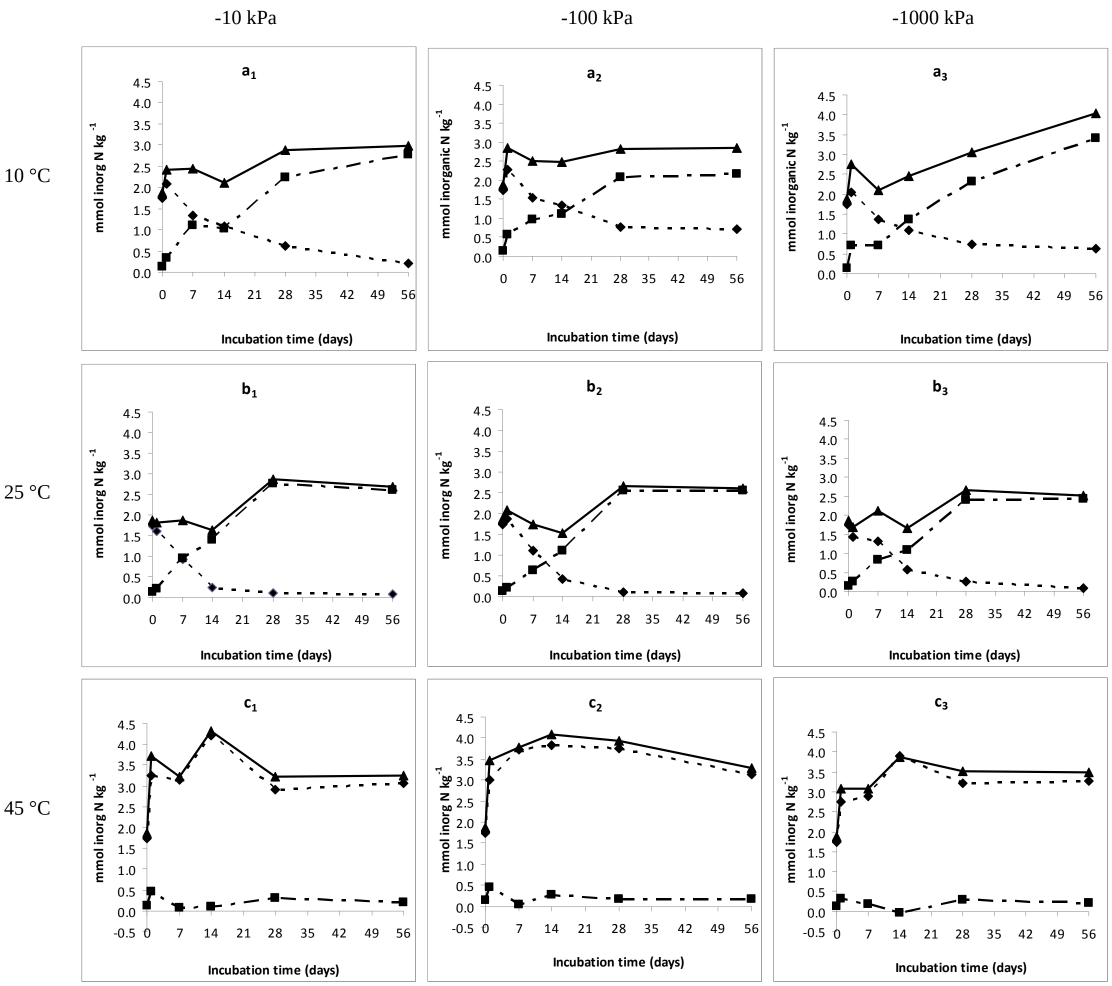


Figure 3.4 Inorganic N released from Vlakplaas sludge during a 56-day laboratory incubation at different temperature and water potential levels (---◆--- = NH_4; ---■--- = NO_3; —▲— = Total inorganic N).

$a_1 = T_1W_1$, $a_2 = T_1W_2$, $a_3 = T_1W_3$;
$b_1 = T_2W_1$, $b_2 = T_2W_2$, $b_3 = T_2W_3$;
$c_1 = T_3W_1$, $c_2 = T_3W_2$, $c_3 = T_3W_3$;
where T_1 = 10°C, T_2 = 25°C, T_3 = 45°C,
W_1 = -10 kPa, W_2 = -100 kPa, W_3 = -1000 kPa

Table 3.4 Ranking and treatment mean comparison of NH_4^+, NO_3^- and net N released, for Vlakplaas sludge-amended soil after 56 days of laboratory incubation

Treatment	**NH_4^+**	**Treatment**	**NO_3^-**	**Treatment**	**Net N released**
	[mmol kg^{-1}]				
T_3W_3	3.28 (0.04)^{a}	**T_1W_3**	2.99 (0.07)^{a}	**T_3W_3**	3.50 (0.17)^{a}
T_3W_2	3.13 (0.34)^{ab}	**T_2W_1**	2.93 (0.07)^{a}	**T_1W_3**	3.36 (0.12)^{ab}
T_3W_1	3.05 (0.06)^{b}	**T_2W_2**	2.63 (0.02)^{b}	**T_3W_2**	3.29 (0.40)^{ab}
T_1W_3	0. 80 (0.02)^{c}	**T_2W_3**	2.60 (0.04)^{bc}	**T_3W_1**	3.25 (0.07)^{b}
T_1W_2	0.70 (0.12)^{cd}	**T_1W_1**	2.52 (0.10)^{c}	**T_2W_1**	2.90 (0.04)^{c}
T_1W_1	0.60 (0.05)^{d}	**T_1W_2**	1.99 (0.06)^{d}	**T_1W_1**	2.77 (0.13)^{c}
T_2W_3	0.11 (0.00)^{e}	**T_3W_3**	0.22 (0.04)^{e}	**T_2W_2**	2.46 (0.04)^{d}
T_2W_1	0.07 (0.00)^{e}	**T_3W_1**	0.20 (0.12)^{e}	**T_2W_3**	2.25 (0.04)^{d}
T_2W_2	0.03 (0.00)^{e}	**T_3W_2**	0.16 (0.02)^{e}	**T_1W_2**	2.03 (0.07)^{f}

Treatment means in column followed by the same letter are not statistically different at α = 5%;

LSD for NH_4^+ = 0.15;

LSD for NO_3^- = 0.08 and

LSD for Net N released =0.21;

Figures in brackets denote standard errors.

(ii) Olifantsfontein

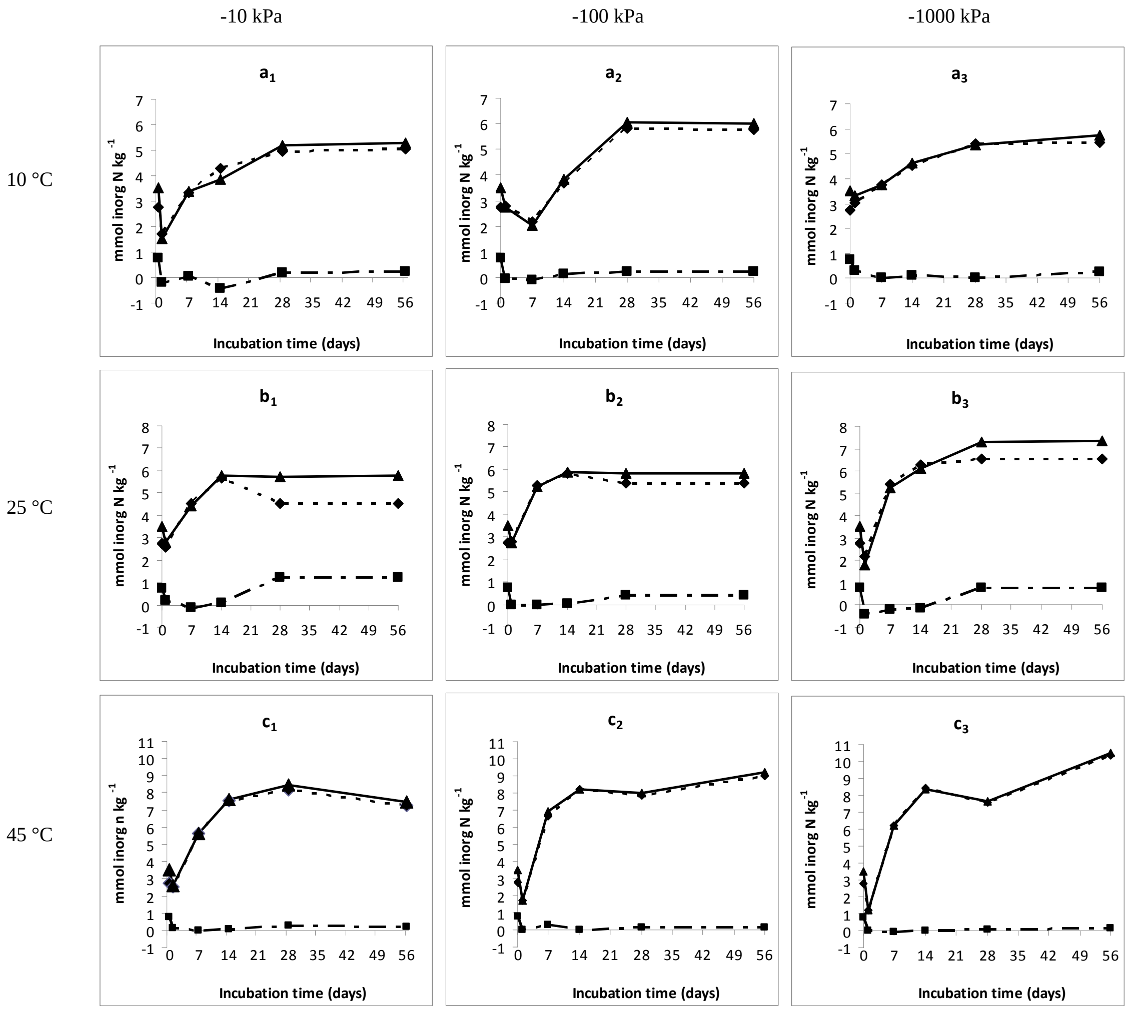


Figure 3.5 Inorganic N released from Olifantsfontein sludge during a 56-day laboratory incubation at different temperature and water potential levels (---◆--- = NH_4; ---■--- = NO_3; —▲— = Total inorganic N)

$a_1 = T_1W_1$, $a_2 = T_1W_2$, $a_3 = T_1W_3$;
$b_1 = T_2W_1$, $b_2 = T_2W_2$, $b_3 = T_2W_3$;
$c_1 = T_3W_1$, $c_2 = T_3W_2$, $c_3 = T_3W_3$;
where T_1 = 10°C, T_2 = 25°C, T_3 = 45°C,
W_1 = -10 kPa, W_2 = -100 kPa, W_3 = -1000 kPa

Table 3.5 Ranking and mean comparison of NH_4^+, NO_3^- and net N released, for unstable Olifantsfontein sludge-amended soil after 56 days of laboratory incubation

Treatment	**NH_4^+**	**Treatment**	**NO_3^-**	**Treatment**	**Net N released**
	[mmol kg^{-1}]				
T_3W_3	10.4 *(0.45)*^{a}	**T_2W_1**	1.23 *(0.06)* ^{a}	**T_3W_3**	10.5 *(0.45)*^{a}
T_3W_2	9.03 *(0.30)*^{b}	**T_2W_3**	0.78 *(0.06)* ^{b}	**T_3W_2**	9.19 *(0.29)*^{b}
T_3W_1	7.25 *(0.22)*^{c}	**T_2W_2**	0.44 *(0.05)* ^{c}	**T_3W_1**	7.44 *(0.18)*^{c}
T_2W_3	6.55 *(0.08)*^{d}	**T_1W_3**	0.28 *(0.06)* ^{d}	**T_2W_3**	7.33 *(0.21)*^{c}
T_1W_1	6.03 *(0.18)*^{e}	**T_1W_1**	0.24 *(0.05)* ^{de}	**T_1W_1**	6.27 *(0.24)*^{d}
T_1W_2	5.78 *(0.02)*^{e}	**T_3W_1**	0.19 *(0.04)* ^{e}	**T_1W_2**	5.92 *(0.09)*^{de}
T_1W_3	5.44 *(0.05)*^{f}	**T_3W_2**	0.16 *(0.02)* ^{ef}	**T_2W_2**	5.83 *(0.06)*^{e}
T_2W_2	5.39 *(0.02)*^{f}	**T_3W_3**	0.12 *(0.02)* ^{f}	**T_2W_1**	5.76 *(0.61)*^{e}
T_2W_1	4.52 *(0.51)*^{g}	**T_1W_2**	0.12 *(0.07)* ^{f}	**T_1W_3**	5.72 *(0.04)*^{e}

Treatment means in column followed by the same letter are not significantly different at α = 5%;

LSD for NH_4^+=0.32;

LSD for NO_3^- = 0.06 and

LSD for Net N released =0.36;

Figures in brackets denote standard errors

(iii) Sasol

-10 kPa -100 kPa -1000 kPa

10 °C

25 °C

45 °C

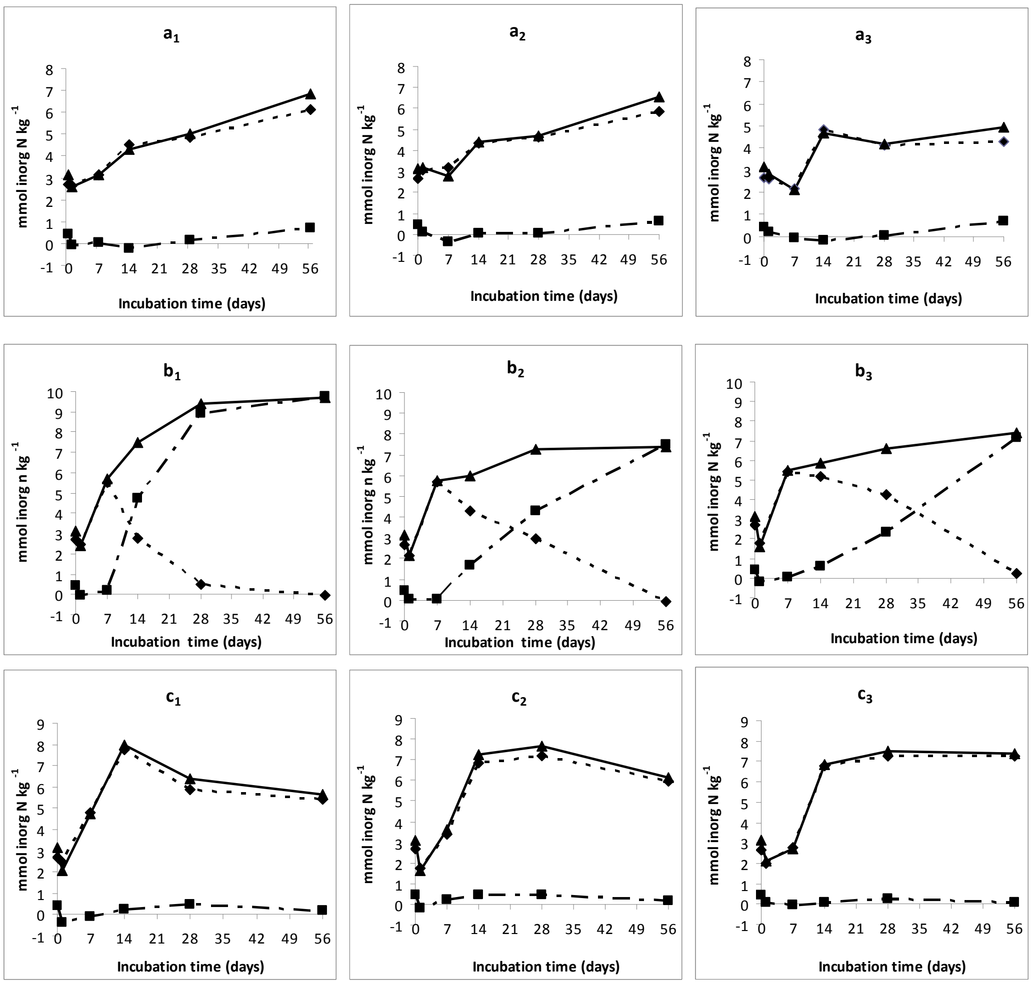


Figure 3.6 Inorganic N released from Sasol sludge during a 56-day laboratory incubation at different temperature and water potential levels (---◆--- = NH_4; ---■--- = NO_3; —▲— = Total inorganic N)

$a_1 = T_1W_1$, $a_2 = T_1W_2$, $a_3 = T_1W_3$;
$b_1 = T_2W_1$, $b_2 = T_2W_2$, $b_3 = T_2W_3$;
$c_1 = T_3W_1$, $c_2 = T_3W_2$, $c_3 = T_3W_3$;
where T_1 = 10°C, T_2 = 25°C, T_3 = 45°C,
W_1 = -10 kPa, W_2 = -100 kPa, W_3 = -1000 kPa

Table 3.6 Ranking and mean comparison of NH_4^+, NO_3^- and net N released, for SASOL sludge-amended soil after 56 days of laboratory incubation

Treatment	**NH_4^+**	**Treatment**	**NO_3^-**	**Treatment**	**Net N released**
	[mmol kg^{-1}]				
T_3W_3	7.28 *(0.09)*^{a}	**T_2W_1**	9.75 *(0.55)*^{a}	**T_2W_1**	9.75 *(0.38)*^{a}
T_1W_1	6.15 *(0.04)*^{b}	**T_2W_2**	7.47 *(0.25)*^{b}	**T_3W_3**	7.47 *(0.10)*^{b}
T_3W_2	5.97 *(0.08)*^{c}	**T_2W_3**	7.18 *(0.24)*^{b}	**T_2W_2**	7.10 *(0.26)*^{c}
T_1W_2	5.87 *(0.13)*^{c}	**T_1W_3**	0.69 *(0.03)*^{c}	**T_2W_3**	7.10 *(0.26)*^{c}
T_3W_1	5.44 *(0.12)*^{d}	**T_1W_1**	0.69 *(0.10)*^{c}	**T_1W_1**	6.83 *(0.21)*^{c}
T_1W_3	4.28 *(0.10)*^{e}	**T_1W_2**	0.65 *(0.02)* ^{c}	**T_1W_2**	6.50 *(0.10)*^{d}
T_2W_1	0.21 *(0.03)*^{f}	**T_3W_1**	0.20 *(0.02)*^{d}	**T_3W_2**	6.13 *(0.12)*^{e}
T_2W_2	0.00 *(0.00)*^{g}	**T_3W_2**	0.19 *(0.02)*^{d}	**T_3W_1**	5.63 *(0.12)*^{f}
T_2W_3	0.00 *(0.00)*^{g}	**T_3W_3**	0.10 *(0.02)* ^{d}	**T_1W_3**	5.00 *(0.20)*^{g}

Treatment means in column followed by the same letter are not statistically different at α = 5%;

LSD for NH_4^+= 0.09;

LSD for NO_3^- = 0.28 and

LSD for Net N released=0.26;

Figures in brackets denote standard errors

There was an initial boost in mineralization during the first few days of incubation. This initial nitrogen flush has been attributed to mineralization of the microbial biomass that is killed during drying of the soil used in laboratory incubation (Benbi and Richter, 2002). Knowledge of this phenomenon is critical in order to avoid overestimation of mineralization rates in the field.

In general, total inorganic N released from sludge increased with temperature. Specifically, however, more NH_4^+ was observed from treatments kept at 45^{o}C while NO_3^- was elevated at 25^{o}C. This increase in mineralization at high temperatures was also observed by Zaman and Chang (2004), who reported greater N mineralization at 45 than at 25 and 5^{o}C. These results do not necessarily contradict the optimum temperature range for mineralization (25^{o}C-35^{o}C) suggested by several authors such as Doran and Smith (1987), however, they underpin the importance of studying N mineralization at the different stages of mineralization, namely ammonification and nitrification, or by looking at the different forms of N released. NH_4^+ may continue to be released at temperatures as high as 45^{o}C, while conversion of NH_4^+ to NO_3^- may be limited at such high temperatures. Whichever form of N predominates in the soil has practical implications both for the farmer and for the environmentalist. In a situation where NH_4^+ is the dominant N form in the soil, plants will take up this cation, thereby making room for the soil exchange sites to be resaturated by H^+, with the detrimental consequence of acidification of the soil. On the other hand, a dominance of NO_3^- in the soil will lead to a higher tendency of nitrate leaching and contamination of the groundwater beneath.

There were low nitrate levels observed for sludge-amended soils kept at 10 and 45^oC. The inhibition of nitrification may be as a result of the sensitivity of nitrifying bacteria to extreme temperatures and to NH_4^+ toxicity (Sierra *et al.,* 2001; Zaman and Chang, 2004).

A high concentration of heavy metals may be another cause of low nitrate levels observed, especially for Sasol sludge. Wilson (1977) reported that industrial sludge reduced nitrification, and that at the highest rate of 16 mg g^{-1}, industrial sludge completely inhibited nitrification for the first two weeks of their experiment, and attributed this to the high levels of metals, particularly Zn, Cd and Pb in the industrial sludge. High concentrations of metals can cause a reduction in microbial biomass (Chander and Brookes, 1991; Bardgett and Saggar, 1994) leading to reduced mineralization of organic matter. Sandaa *et al*. (1999) reported a decrease in bacterial diversity in soil amended with sewage sludge with high metal content. High metal content may therefore have inhibited nitrification, especially for Sasol sludge.

Net N release had different trends during the initial stage of incubation. The negative period observed, especially for the less stable sludges, was a direct result of their initial low inorganic N content. For example, the less stable Olifantsfontein sludge contained 2% inorganic and easily metabolizable N, while Vlakplaas sludge contained 26% inorganic N. Therefore, in order to mineralize the less stable sludge, microorganisms may have required more nitrogen than is contained in the sludge and will therefore, incorporate the available inorganic N from the soil into their cellular components (Brady and Weil, 1999). Probert *et al*. (2005) also observed that in the initial stage of soil incorporation of organic sources, inorganic N was immobilized, even with substrates having C:N ratios less than 20.

Water potential effect did not show a consistent trend and this may be attributed to a difficulty in maintaining the treatments at the specific water potentials, especially the treatments kept at -1000 kPa and 45°C which tended to dry out rather rapidly. Upon rewetting these samples, the mineralization flush phenomenon may have been activated thereby masking any treatment effects.

Nevertheless, results from a subsequent experiment using five sludge materials, three water potential levels (-100, -50, -20 kPa) and two temperature regimes (25, 45°C) show the effects of water potential on N mineralization from sludge (Fig. 3.7). From those results, NH_4^+ was not significantly affected by water potential, however, NO_3^- levels decreased with dryness of soil. Water potential less than -20 kPa seemed to inhibit nitrate formation. The sensitivity of nitrifiers to extreme moisture conditions has already been highlighted.

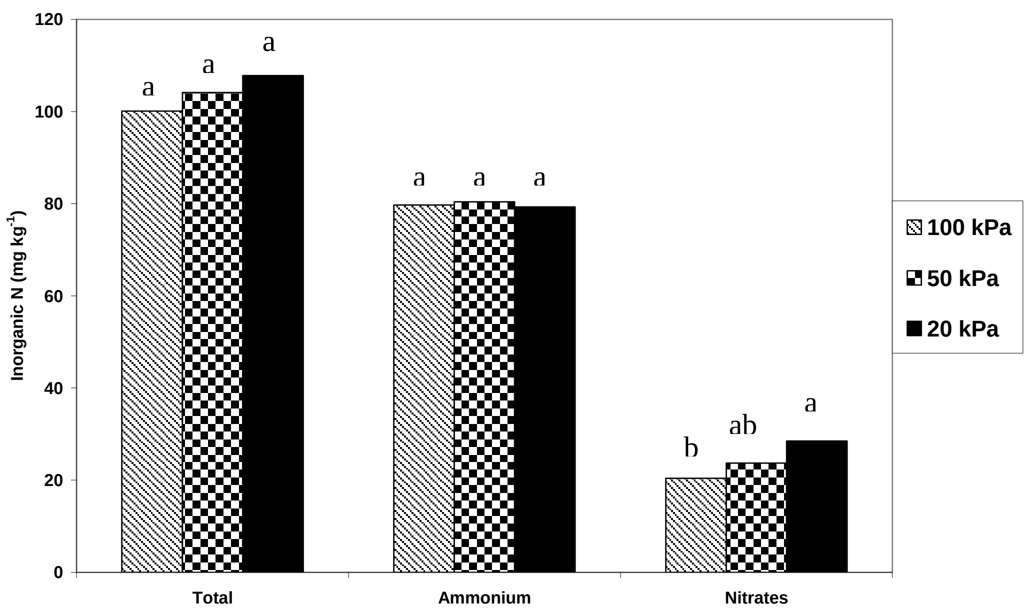


Figure 3.7 Water potential effect on N mineralization in sludge-amended soil. Graphs show average for treatments in a factorial experiment using 5 sludges, water potentials of -100, -50, -20 kPa and temperature levels of 25 and 45°C. (Letters show significant differences at 5% probability level for each of the nitrogen forms)

3.3.3 Mineralization rate constant and half-life

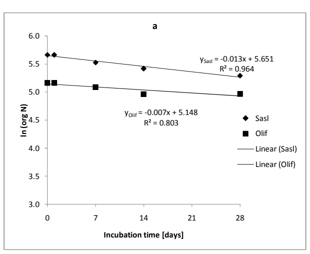
A common approach in N modelling is to generate N release and mineralization parameters under optimum conditions (Serna and Pomares, 1992; Zaman *et al.,* 1999; Hernandez *et al*., 2002). The same approach was followed here. Data collected at 25°C and water potential of approximately -10 kPa was used to assess the kinetics of N release from sludge-amended soils. In order to determine the rate order of N release, data was presented in terms of the natural logarithm of organic nitrogen decay as a function of time. The plot was used to assess the linearity of this relationship, which is the diagnostic test for first order reactions (Fig. 3.8). The organic N was taken as the difference between initially existing organic N and the net N released at each incubation period. Thereafter the organic N values were converted into natural logarithm to provide data to compute the rate constants from the equation:

$$N_{(t)} = N_o \,(1- e^{-kt})$$

where $N_{(t)}$ is the net N mineralized at time t; N_o the potentially mineralizable N; k is the mineralization rate constant and **t** the incubation time.
The rate constant k is therefore,

$$k = \ln (N_t/N_o)/ t;$$

The linear regression coefficient showed that N mineralization for Olifantsfontein and Sasol were reasonably well approximated during the first 28 days using first order kinetics (Fig. 3.8 a). The organic N decay in the Vlakplaas amended soil seemed to follow first order kinetics during the whole incubation period (Fig. 3.8 b). These results support the findings of various other authors (Serna and Pomares, 1992; Benbi and Richter, 2002; Hernandez *et al*., 2002; Hseu and Huang, 2005).



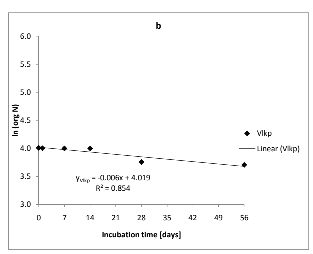


Figure 3.8 The natural logarithm of organic N and estimated rate constants (slope of graphs) for Sasol and Olifantsfontein sludge (a) compared to that of Vlakplaas (b)

The Vlakplaas sludge-amended soil had the lowest mineralization rate constant (0.042 week^{-1}) and longest corresponding half life (116 days) while Sasol sludge amended soil exhibited the highest rate constant (0.093 week $^{-1}$) and shortest half life (53 days) (Table 3.7).

There was only a small difference between the Vlakplaas and Olifantsfontein sludge-amended soil. The Olifantsfontein sludge amended soil had a rate of (0.049 week $^{-1}$) and half life of (99 days). The reason could be that both are municipal sludges and therefore had similar organic composition.

Table 3.7 C/N ratios, total organic N, N mineralization rate constants and half lives of the fast cycling pool of sludges

Sludge types	**C/N**	**Total Org N [mg kg $^{-1}$]**	**k (Day $^{-1}$)**	**k [Week $^{-1}$]**	**Half life [days]**
Vlakplaas	6.0	55.0	-0.006	-0.042	116
Olifantsfontein	4.5	175	-0.007	-0.049	99
Sasol	5.0	288	-0.013	-0.093	53

These are similar to rates found by Stanford and Smith (1972), which ranged from (0.035-0.095 week^{-1}) for 39 samples studied. Also, Hseu and Huang (2005), reported that anaerobically digested sludges had rates from 0.047-0.075 week^{-1}, and 0.047-0.105 week^{-1} for aerobically digested sludges. Higher mineralization rates have also been reported: 0.089-0.883 week^{-1} (Serna and Pomares, 1992) and 0.228-1.140 week^{-1} (Hernandez *et al*., 2002).

Our results are supported by findings of Dou *et al.* (1996), who reported the goodness of fit of different kinetic models to depend on incubation period, and that a single first-order model provided good data fit for incubation periods ≤ 15 weeks; while double first-order models provided better results for incubation period > 15 weeks.

The incubation period used for the determination of mineralization rate and half-life was adopted based on the previously mentioned study done on Olifantsfontein sludge (van Niekerk, 2004). Her study showed that after 42 days, the rate of N mineralization approached zero and ammonium formation was practically negligible after 28 days (Fig. 3.9).

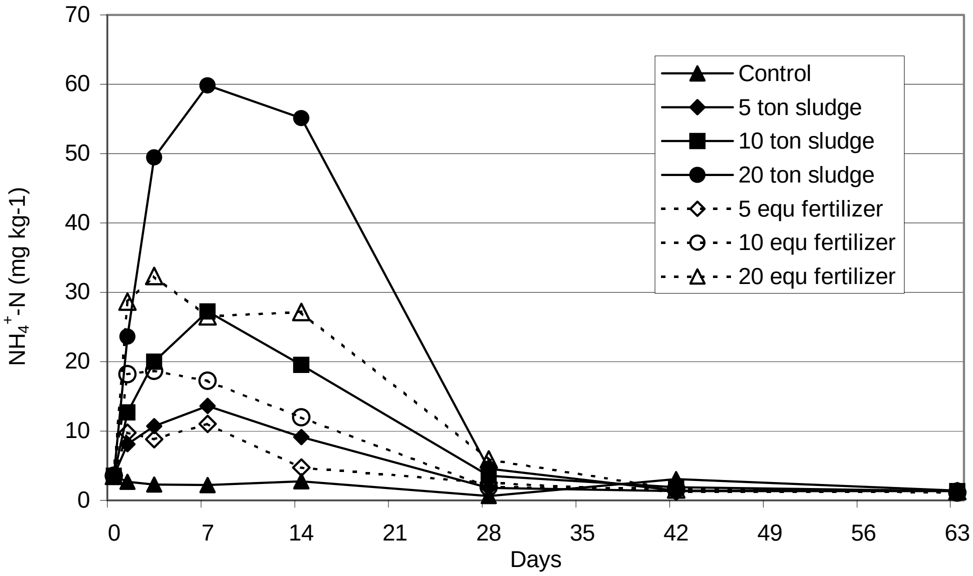


Figure 3.9 Total extractable NH_4^+-N content as a function of incubation time and different rates of sludge and inorganic N fertilizer (van Niekerk, 2004)

In addition Figures. 3.3 to 3.5 indicate that after about 28 days mineralization had reached a significantly slower release pool. However, to get some quantification on the potential sizes of slower cycling N pools the approach of Smith *et al*. (1998) was adopted. N pool sizes were estimated at 26% of total sludge organic N for the rapid release pool, and 42% of total organic N for the slower release pool. The remainder is designated as the resistant pool. Results are shown in Table 3.8.

Table 3 8 Estimated sizes of N pools of different types of sludge investigated

	(mg kg $^{-1}$)			
Sludge source	**Tot Org N**	**Rapid release pool**	**Slow release pool**	**Resistant pool**
Vlakplaas	55.0	14.3	23.1	17.6
Olifantsfontein	175	45.5	73.5	56.0
Sasol	288	74.9	121	92.3

It is reasonable to expect that a double first order kinetic model (Molina *et al*., 1980) would better estimate the rate constant to account for the existence of different organic N pools (Dou *et al.,* 1996; Hseu and Huang, 2005; Smith *et al*., 1998; Benbi and Richter, 2002). However, it was not possible from this study to obtain rate constants of slower cycling pools.

3.4 Conclusions and recommendations

Adequate knowledge of the factors that influence release of nitrogen from sludge is crucial for effective management of sludge application to agricultural soils. This study has made a number of observations that are of importance both to the wastewater treatment plant manager and to the farmer who uses the final waste product.

Release of nitrogen from sludge was found to differ from sludge to sludge. Apart from differing organic and inorganic N contents, a probable trend identified by this study is that N release from sludge may be related to sludge volatile solid content. This relationship was apparent in this study of few sludge materials. It will be necessary to investigate this trend further, using more sludge materials. If indeed, a relationship does exist, a simple correlation equation can subsequently be used to predict N release from sludge instead of tedious and time-consuming incubation experiments. Inclusion of volatile solid content as an input parameter may then enhance the predictive capabilities of existing N release models such as the SWBSci (Annandale *et al*., 1999).

Because of the differing total nitrogen and volatile solid contents of sludges, it is recommended that regular characterization of sludge from WWTPs be carried out. The frequency of characterization as outlined in the Guidelines for the Utilization and Disposal of Wastewater Sludge Volume 2 (Snyman and Herselman, 2006) should be followed. The simple laboratory determination method of Croll *et al*. (1985) which was also recommended by Snyman and Herselman (2006) may be used. Alternatively, a quick and easy determination of volatile solid content may suffice if more investigations establish a definite relationship between volatile solid content and N release.

This study found that N release from sludge is influenced by the soil's temperature and water potential. Dry soil conditions of water potential less than -20 kPa and temperatures below 10°C may inhibit mineralization. Consequently, mineralization of N from sludge will be slow during dry winter months. At such periods, it may be advisable to augment soil N status with inorganic N fertilizers, or with inorganic N-enriched sludge products such as ASP.

Although mineralization of organic N to ammonium may continue at high temperatures, conversion of ammonium to nitrates may be inhibited at these temperatures, thus leading to a build-up of ammonium in the soil. While plants may take up ammonium, significant soil acidification may occur when H^+ replaces the absorbed ammonium ions from the soil's exchange sites. Moreover, when temperature suddenly becomes suitable for nitrification, rapid conversion of ammonium to nitrates may ensue, leading to increased potential for nitrate leaching. It is advisable therefore, to apply sludge in areas and at periods when soil temperature and water potential are optimum for nitrification. From this study a temperature of 25°C and water potential in the range of field capacity may be adequate.

This study has also calculated mineralization rate constants and half-lives for three locally available sludges. These values are necessary input parameters needed for computer model simulations. However, it has to be said that the half-lives calculated were only for the fast cycling pool of sludge. There is still need to determine rate constants and half-lives of the slower cycling pools. For this, longer-running experiments are needed. Furthermore, determination of rate constants for more sludges should also be pursued.

Lastly, because of a possible initial immobilization of soil inorganic N following sludge application to soil, it is recommended that sludge be applied to soils at least three weeks before crops reach their stage of highest nitrogen demand. That way, problems with nitrogen deficiency and nitrate leaching will be substantially reduced.

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CHAPTER 4

P AVAILABILITY AND FERTILIZER VALUE OF VARIOUS TYPES OF SLUDGE

4.1 Introduction

4.1.1 The P fertilizer value of sewage sludge

Phosphorus (P) is a scarce natural resource and the global reserves of primary source, such as rock phosphate, is steadily declining. According to recent estimate, reserves will be depleted in about 90 years at the current rate of usage (Stewart, *et al*., 2005). On the other hand, sewage sludge production is globally on the increase and the fertilizer value of sewage sludge has long been recognised (USA EPA, 1999). However, apart from potential health risks like pathogens, heavy metals and organic pollutants, the nutrient availability from sludge can also vary considerably (Smith *et al*., 1998; Petersen, 2003).

The USEPA manual for land application of sewage and domestic septage, reports that sewage sludge is 50% as “effective” as inorganic P fertilizer. However, this value has been a point of some debate. Scientists have been highly critical of this sweeping generalisation because of the lack of scientific evidences supporting this claim (O’ Connor *et al*., 2004). The P availability from sludge is highly dependent on the treatment it underwent. The origin and treatment method of sewage wastewater therefore determine the P fertilizer of sludge (Petersen, 2003). Phosphate removed by chemical precipitation using poly-aluminium salts, aluminium-sulphate or ferric chloride, results in the precipitation of sparingly soluble, aluminium phosphates (Al-P) and ferric phosphates (Fe(III)-P). As a result, the P in the sludge has low water extractability and plant availability. However, it poses low environmental risk (Maguire *et al.,* 2000; Samie and Römer, 2001; Elloitt *et al*., 2002; Hyde and Morris, 2004; O’ Connor *et al*., 2004; Krogstad *et al*., 2005). On the other hand, sludge that underwent biological phosphate removal (BPR-sludge) has a P fertilizer value, in terms of plant availability, similar to that of manure and inorganic fertilizer (Stratful *et al*., 1999, O’ Connor *et al*., 2004).

O’ Connor *et al*. (2004) proposed three general phyto-availability classes relative to triple super phosphate (TSP). The proposed classes were: High (> 75% of TSP), moderate (25-75%) and low (< 25% of TSP). This proposal was based on a two-year greenhouse study, using 12 different types of sludge using Bahia grass (*Paspalumnotatum* Flugge) as a test crop. Results from this study showed that BPR-sludge fell in the high category. However, Al / Fe-P-sludges fell in the moderate to low category. Sludges with total Fe and Al content > 50 g kg^{-1} and sludge processed to high solids content (> 60%) all fell in the lowest class.

4.1.2. Phosphate removal from sewage wastewater

Phosphate is removed from sewage wastewater chemically, biologically (using micro-organisms) or both methods (NRC, 1996). The removed inorganic and/or organic phosphate, is concentrated in the sewage sludge leaving an almost phosphate free wastewater (<1 mg L^{-1}) P to be discharged.

Phosphate removal is a tertiary wastewater treatment process. Conventional inorganic phosphate removal refers to the treatment of phosphate-rich sewage wastewater with ferric chloride, aluminium-sulphate or calcium hydroxide (Marx *et al.*, 2004). Assuming the dominant form of orthophosphate is H_2PO_4 – (between a pH between 4 and 6).The basic aqueous reactions involving aluminium and ferric iron responsible for the precipitation of phosphate can be summarized as follows:

Variscite precipitation:

$Al^{3+} + 2H_2O + H_2PO_4^{-} \leftrightarrow AlPO_4.\ 2H_2O_{(s)} + 2\ H^{+}$ log K = 2.50

Strengite precipitation:

$Fe^{3+} + 2H_2O + H_2PO_4^{-} \leftrightarrow FePO_4.\ 2H_2O_{(s)} + 2\ H^{+}$ log K = 6.85

(Lindsay, 1979).

Struvite ($MgNH_4PO_4.\ 6H_2O$) precipitation is another method of phosphate removal using magnesium hydroxide and sodium hydroxide:

$Mg^{2+} + NH_4^{+} + H_2PO_4^{-} + 6\ H_2O \leftrightarrow MgNH_4PO_4.\ 6H_2O_{(s)} + 2\ H^{+}$ log K = 6.40

(Lindsay, 1979).

Precipitation with lime gives a very fine initial precipitate of calcium phosphate, for example brushite:

$Ca^{2+} + 2H_2O + H_2PO_4^- \leftrightarrow CaHPO_4 . 2H_2O_{(s)} + H^+$ log K = -0.63

Calcium phosphate is generally considered more soluble than Al-P and Fe(III)-P. However, with time the solubility of Ca-P can be significantly decreased as a result of the formation of octacalcium phosphate:

Using aluminium as aluminium salts or poly-aluminium chloride (and / or ferric chloride) can cause co-precipitation of metals (USA EPA, 1999).

Biological removal of phosphate from sewage wastewater is done with bacteria known as phosphorus accumulating organisms (PAO). These bacteria are aerobic heterotrophs that only thrive under certain conditions. They are mainly filamentous bacteria (Wagner and Loy, 2002; Crocetti *et al*., 2000; Hesselman *et al*., 1999).

The amount of phosphate removed from wastewater is strongly correlated with the P concentration in the sewage wastewater and the number of PAO probe-binding cells. Bacteria closely related to *Rhodocyclus, Acinetobacter* and *Propionibacter* are all known as PAO in sewage wastewater sludge. Biological phosphate removal in sewage wastewater treatment was first observed in India (Srinath *et al.*, 1959; Marx *et al.*, 2004). The process has been developed from an engineering perspective and not a microbiological perspective. Therefore, despite the fact that PAO have not yet been isolated and cultured, the process of biological phosphate removal is already widely applied in wastewater treatment.

Biological removal of phosphate is considered a better process than chemical precipitation of phosphate because there are no counter ions (like chloride) in the water that can increase salinity. The effectiveness of P removal from wastewater sludge can be improved by combining biological phosphate removal with the addition of, for example, ferric chloride accompanied by nitrogen removal (Van Loosdrecht *et al.*, 1997).

4.1.3. Phosphate dynamics in sludge amended soils

The plant availability of P from wastewater sludge is not only a function of the methods employed to remove phosphate but also determined by soil properties and application strategies. Sewage sludge is applied in agricultural lands in either dry or liquid forms. Similar to inorganic fertilizer, it is usually spread on the soil surface or incorporated or injected in bands (USA EPA, 1999). When placed in bands this is done at a depth of 0.1 m to 0.3 m, to reduce run-off of and the development of odour problems (Brady and Weil, 2002). Band placement is usually a strategy employed in high phosphate fixing soils.

P could be fixed in the soil both biologically and chemically. However, the irreversible sorption of P rendering it unavailable to plants is largely attributed to chemical fixation reactions. The most important soil factors that determine the soil's ability to attenuate phosphate are: The presence of ferric, aluminium and manganese (oxy) hydroxides, soil pH, organic matter content and texture which in turn determines the total surface area (Brady and Weil, 2002).

Phosphate in the soil solution is mainly made unavailable because of sorption by Fe and Al (oxy) hydroxides. If the phosphate does not desorb over time, the sorbed phosphate will be changed into sparingly soluble aluminium phosphate, namely variscite ($AlPO_4.2H_2O$) and strengite ($FePO_4.2H_2O$).

Goethite and gibbsite are the most common surfaces on which phosphate sorption can occur. These mineral surfaces have high point of zero charge (pH ranging from 8.9-9.0) and exhibit a net positive charge in the pH range of most soils as well as a high affinity for orthophosphate (Essington, 2004). The sorption reaction involves strong inner sphere complexation of phosphate and its low plant availability is attributed to this. At a pH of below five, sorption is more pronounced on goethite because of its higher point of zero charge and the greater solubility of gibbsite at pH < 5. At pH > 7 phosphate sorption is increasingly associated with calcium (Brady and Weil, 2002).

P is not considered a mobile ion, however, significant P leaching can take place when the P fixation capacity is exceeded, or when bypass flow of P occurs through biological or physical macro pores in times of excessive percolation (Sims *et al*., 1998).

It is generally accepted that plants take up phosphate from the soil solution in the inorganic orthophosphate forms, for example, $H_2PO_4^-$and HPO_4^{2-} (Beck and Sanchez, 1994). Therefore, the organic P needs to be mineralized before it can be available for plant uptake (Pietersen *et al.*, 2003). Organic phosphate sources in the soil can be divided into three groups, namely inositol phosphate, nucleic acids and phospholipids. Inositol P is the most abundant organic P form because of its greater resistance against microbial degradation. Nucleic acid comes from the degradation of plant and animal remains by micro-organisms and generally occurs at low concentrations in the soil because it is more easily broken down by micro-organisms. Phospholipids are the products of lipid degradation and are also easily degradable. Consequently, phospholipids generally occur at low concentrations in the soil. Furthermore, humates and fluvates released from sludge can also form chelates with Al and Fe ions in solution increasing the solubility and plant availability of Al-P and Fe(III)-P (Brady and Weil, 2002).

Different enzymes, for example, phosphatase and phytase, catalyse the mineralization of organic P in the soil, which transforms it to plant available inorganic P forms (He and Honeycutt, 2001).

Maximum net N and P mineralization occurs when soil water content is near field capacity and soil temperature is at 25°C (Stanford *et al*., 1973; Cassman and Munns, 1980; Kladivko and Keeney, 1987; Goncalves and Carlyle, 1994). However, in the field, these optimum conditions are transient. In contrast the drying of soil to permanent wilting point of plants (-1.5 MPa) reduces microbial activity and cumulative mineralization of carbon (C), nitrogen and phosphorus (Mikha *et al*., 2004). The mineralization process will therefore be negatively influenced due to the drying cycle and the onset of less favourable conditions. Upon rewetting microbial populations have to recover to optimum levels before pre-drying mineralization rates can be re-established. Van Gestel *et al*. (1993) found that microbial mass can decrease by 58% when soils are subject to a drying and rewetting cycle. Higher mineralization is also associated with higher soil C content (Eghball *et al*., 2005).

The influence of wetting and drying on P availability is not only the result of biological activity but also of physico-chemical effects (Nguyen and Marschner, 2005). Drying is known to decrease the availability of P and this is mainly attributed to the precipitation of P from solution onto mineral surfaces, and the irreversible dehydration of sorbed P and P minerals (Fe, Al, Mn and Ca phosphates) (Wiklander and Koutler-Andersson,1966).

4.2. Problem statement

Agricultural use is potentially an infinite sink for sludge use as a low grade fertilizer. There is a growing realization that a better understanding of the influence of treatment processes on the fertilizer value of sludge is crucial. This, in turn, will help in ensuring long term sustainable agricultural use. In light of this, various agricultural sludge products have emerged, for example, the Activated Sludge Pasteurization (ASP) and the Agriman products. However, it is reasonable to expect that N and P release from these commercial sludge products will be different from that of activated or anaerobically digested sludge. The fertilizer value of these products has not been investigated in South Africa and more information is needed to guide the commercial use of sludge products.

4.3. Research approach

This chapter formed part of an M.Sc. study and a multi-scale approach was followed that included : 1) Laboratory incubation study to investigate phosphate release from various types of sludge products; 2) Greenhouse pot trial and 3) A field trial to assess actual phyto-availability.

4.4. Merits of open and closed mineralization experimental systems to investigate P release.

4.4.1. Background

It is imperative that field scale research and modelling be supported with in-depth laboratory studies, such as incubation mineralization studies. Laboratory scale experiments are traditionally used to elucidate mechanisms and factors influencing certain processes to parameterize models. The conventional experimental approach followed in laboratory mineralization studies are incubation methods. These systems are open with respect to air movement and volatilisation, however, these systems do not allow the leaching of water. Soils are incubated with organic material for a certain period, at constant temperature. The containers should have airtight lids in order to maintain a constant water content or water potential. A headspace is left above the soil to ventilate the system regularly to keep the system oxidized. The purpose of ventilation is to facilitate nitrification, which is mediated by strictly aerobic micro-organisms. Incubation systems are, therefore, only ‘open’ with respect to gas movement and not to leaching.

Many variables and interplay between variables exist in open systems with respect to water movement. The interplay between aeration, water content and water potential varies temporally and spatially in the soil. This has a potential impact on microbial dynamics, organic matter mineralization and chemical reaction. The challenge is therefore to generate scenarios under laboratory conditions that could reflect the reality out on the field. For example, from a chemical point of view the movement or leaching of water through the soil under field conditions constantly changes the composition of the soil solution. Leaching can moderate soil conditions by removing compounds or elements released from the sludge that potentially can inhibit microbial activity. In conventional incubation systems water movement is not allowed, consequently elements and compounds, if not volatilized will accumulate. This in turn can alter mineralization rates and cumulative N and P release.

Decision on the most appropriate approach to investigate P release from sludge amended soil systems should therefore be established before an extensive investigation commences.

4.4.2 Aim

The aim of this study was to compare P release from sewage sludge and sludge fertilizer products in an open-type incubation system, subjected to leaching as well as wetting and drying, to a conventional closed system incubation set-up.

4.4.3. Materials and Methods

Sludge characterization

For this preliminary study on the merits and demerits of open and closed systems, only three sludge types were selected: two agricultural sludge products and an industrial sludge:

- ASP (Activated Sludge Pasteurization), from Daspoort Water Care Works (WCW), is a sludge product enriched with both nitrogen and phosphorus. The ASP product is manufactured by injecting anhydrous ammonia and phosphoric acid into an 11 to 18% solid content raw sludge. The product is then dried to produce a granular product with a diameter of 3 to 5 mm. This is a sludge that contains a soluble inorganic form of P,

- Sasol biological sludge, from the petrochemical industry is an aerobic activated sludge and less stable than the agricultural sludge products. Sasol sludge was an example of a sludge that underwent biological P removal;
- Agriman is an agricultural sludge product from Sutherland ridge WCW. Activated sewage sludge is dewatered, dried and then granularized and sold commercially. This sludge underwent both biological and chemical P removal.

The analyses of the ASP, Sasol and Agriman sludges are presented in Table 4.1.

Table 4.1. Selected physical & chemical properties of the different types of sludge and MAP

	Agriman	Daspoort	Vlakplaas	Sasol	ASP	MAP	Units
pH	7.1	6.5	7.0	6.0	7.7	4.70	
Water content [a]	8.89	5.6	5.3	6.74	9.72	2.08	%
Solids content[a]	91.11	94.5	94.7	93.3	90.3	97.9	
Ash content	39.6	44.4	42.3	8.9	40.3	37.6	
Volatile Matter	51.5	50.1	52.4	84.4	50.0	60.3	
Total N	4.03	2.95	3.12	7.50	10.7	11.0	
Total C	27.5	26.3	26.5	45.5	20.5	1.1	
C/N Ratio	6.8	8.9	8.5	6.1	1.9	0.1	
Ca	3.28	3.18	2.84	0.65	1.23	0.52	%
Mg	0.80	0.33	0.26	0.24	0.92	0.82	
P	3.74	3.84	2.81	0.79	11.5	23.8	
K	0.46	0.10	0.17	0.28	0.17	0.13	
Na	0.10	0.06	0.19	0.08	0.04	0.09	
Fe	4.65	10.5	11.4	0.75	1.61	0.62	
Cu	899	467	628	37	315	78	mg kg $^{-1}$
Mn	691	381	2504	147	309	389	
Zn	1216	1480	4067	235	768	6683	
S	0.87	0.92	2.78	1.11	0.74	2.67	%
Al	1210	6610	8896	2299	5630	1223	mg kg $^{-1}$
Water soluble Cl	0.27	0.21	0.37	0.18	0.45	1.83	%
Water soluble P	2.25	0.10	0.03	1.94	83.3	193	g kg $^{-1}$

- a) Air-dried sludge samples

Soil characterization

Two soils with varying pH, clay content and P fixing capabilities (Table 4.2) were selected for this incubation study. These soils were collected from two different sites: a. one from the Experimental Farm of the University of Pretoria in Hatfield (HAT), b. the experimental farm near Bronkhorstspruit, Miertjie le Roux (MLER). The pH of the HAT soil was adjusted to 6.5 using calcium oxide (CaO), before the pre-incubation commenced. The amount of CaO used was equivalent to 2445 kg CaO per ha.

Table 4.2.Selected chemical and physical properties of MLER and HAT soils.

	MLER (Low P fixing soil)	**HAT (high P fixing soil)**	**Units**
pH (water)	6.6	4.5[a]	
Total N	570	580	mg kg $^{-1}$
Total C	0.7	0.68	%
C/N Ratio	12.28	11.7	
Total P	221	295	mg kg $^{-1}$
P Bray 1	9.0	4.6	
Total K	789	862	
Total Ca	864	458	
Total Mg	473	362	
Ammonium acetate extractable			
K	115	66	
Ca	511	61	
Mg	146	17	
Na	5	2	
Texture			%
Sand	84.4	67.8	
Silt	3.8	5	
Clay	11.3	25	
Texture	Loamy sand	Sandy clay loam	
CEC	7.49	11.8	$cmol_c kg^{-1}$

a) pH before adjustment to 6.5.

Field capacities of the selected soils were determined from retention curve developed using the pressure plate apparatus. The corresponding gravimetric water contents were 0.11 and 0.16 kg kg^{-1} for the MLER and HAT soils respectively.

Sludge – soil treatments

The HAT and MLER received the equivalent of the 10 tons per hectare (on dry mass basis) of dried sludge, the maximum allowable annual sludge application rate on agricultural lands (Snyman and Herselman, 2006). The application rates were calculated by assuming a dry bulk density of 1200 kg m^{-3} for the MLER and HAT and a hypothetical incorporation depth of 0.3 m. Considering the gradient in P concentrations across the sludge types used, the 10 ton ha^{-1} sludge application was expected to create variation among the three sludge types (Table 4.3).

Table 4.3.Calculated N, P and K application rates.

Sludge	P content (%)	P loading rate	
		kg ha $^{-1}$	mg kg $^{-1}$
ASP	11.2	1013	282
Sasol	0.8	74.7	20.8
Agriman	3.7	337	93.6

Experimental layout

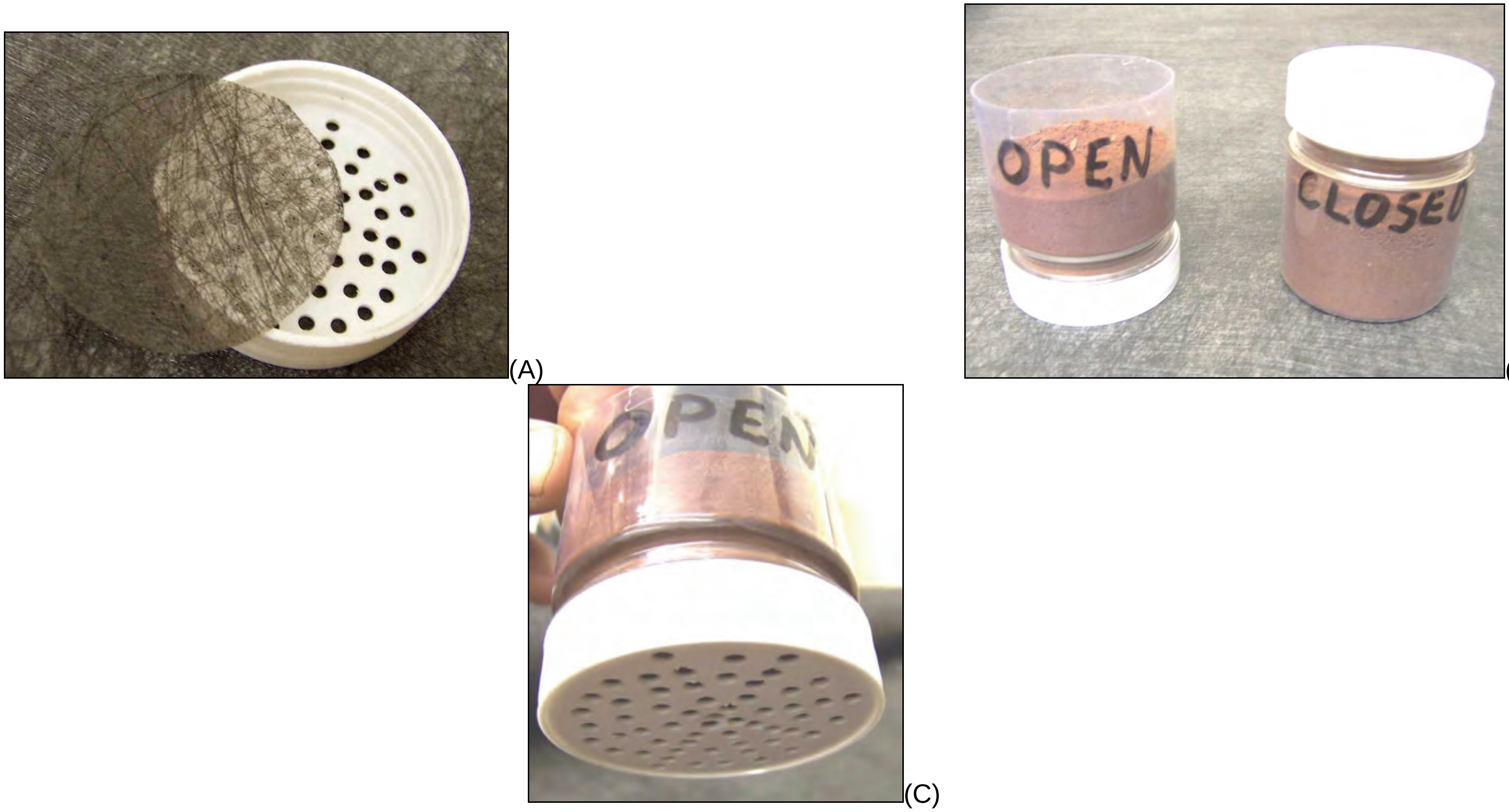
Each soil was incubated with the three sludge types as well as a zero control (no sludge). The incubation trial consisted of 8 separate incubations, each incubated for different periods (four replicates for each period). The different incubation times were 1, 3, 7, 14, 21, 28, 35 and 42 days. Initially, each soil was pre-incubated at field capacity for seven days. This was followed by the application of treatments. Similar procedures were followed for the open incubation system. Both the closed and open system incubations were conducted at a constant temperature of 21^{o} C.

The main differences between the open and closed systems were that the open system was allowed to dry and the soil solution entrained in the pore volume was periodically removed under vacuum. After the drying cycle, 30 cm^3 of de-ionized water was added and vacuum applied to collect the leachate for N and P analyses.

The containers used (volume 200 cm^3, height: 65 mm and diameter 53 mm) are illustrated in Figure 4.1. The open system containers were modified by removing the bottom-end and perforating the lids (diameter of holes drilled: 3 mm) (Figure 4.1B and C). The lids were lined with two layers of geo-textile (made by Du Pont) to prevent soil loss when leaching the soil (Figure 4.1A).

Extractions and analyses

The closed and open systems were sacrificed after their respective incubation times and extracted with 200 ml 1 *M* potassium chloride (KCl) solution. The samples were then shaken on a reciprocal shaker at 180 rpm for 1 hour, after which the samples were filtered through a No. 2 Whatman filter paper. Extracts were then analysed for ammonium, nitrate and phosphorus. Phosphorus was analysed by means of ICP-AES (Smart Analyzer Vision 2.11.0629 Spectro Analytical Instruments GmbH and Co. KG). Ammonium and nitrate were determined colourimetrically using an automated flow system.



(A) (B) (C)

Figure 4.1. The perforated cap with geo-textile (A), comparison of the containers used for the open and closed system (B), and the open system container with perforated base attached (C).

4.4.4 Results and discussion

In general, the differences in P release between the open and closed systems were lower in contrast to N. In a high P fixing soil, the type of incubation system will have a negligible effect on P release. The mobility of P was generally very low for both soils. This is attributed to the fact that the source of P in the sludge, or the solid phase it forms in the soil, is sparingly soluble. Under these conditions P release is not transport controlled. Thus a normal closed incubation system will be adequate to determine P dynamics because it is driven by chemical properties of the sludge and the soil and not environmental conditions as in N dynamics.

4.5. Laboratory scale investigation of P release

4.5.1. Aim

To assess the fertilizer value of selected sludge types.

4.5.2. Materials and Methods

This follow-up study was a much larger experiment than the previous preliminary study and it included more soils and sludges. Soils differing in P sorption capacities and texture (clay content: 7, 12 30, and 45%), were used in a 168-day incubation study. In this study all the treatments received the same amount of total P (equivalent to 280 kg P ha^{-1}).Three control treatments were included: Two positive controls using potassium phosphate and commercial mono-ammonium phosphate (MAP) and one negative control (no phosphate added).

Sludge characterization

The following two sludge types were added to the previous three used in the open and closed system incubation trial:

- Vlakplaas dry bed sludge, from Vlakplaas WCW where P is chemically precipitated with ferric chloride.
- Daspoort sewage sludge, from Daspoort WCW where P is removed using a combination of biological and chemical methods.

The candidate sludges were analysed at the Institute for Soil Climate and Water (ISCW) as follows:

- Total nitrogen (N), and carbon (C) content using a Carlo Erba NA1500 C/N analyzer (Carlo Erba Strumentazione, Milan, Italy);
- Total elemental analyses of Ca, Mg, K, Na, Fe, Cu, Zn, Mn, Al, S, and P was conducted using an ICP-AES after a microwave acid digestion (EPA 3050);
- pH was measured in a 1:5 sludge to water ratio;
- Soluble P was determined using an ICP-AES after water extraction (1:20 solution to sludge ratio);
- Cl was determined using Ion Chromatography after extracting in a 1:200 solution ;
- Ash content was estimated as the percentage of the mass of solids that did not volatilise when combusted at 550°C;
- The volatile matter was taken as the percentage of the mass of solids that did volatilise at 550°C.

Soil characterization

Initially, eleven soils were collected and analysed (Table 4.4) in order to find four soils with gradient in P sorption capacities (PSC). Soils A, D, E and H were selected according to the criteria. The selected soils had marked differences in clay content (7-45%), similar pH, low Bray-1 extractable P and a gradient in P sorption capacity. There was a strong linear correlation between the clay content and P sorption capacity of the 11 soils (Fig 4.2).

Table 4.4 Soil texture and selected chemical properties of the initial 11 soils. Highlighted soils A, D, E and H were selected for further study.

Soil	Colour	pH		P Bray-1	Sand	Silt	Clay	Texture	PSC
		(H_2O)	(KCl)	(mg kg^{-1})	%				(mg kg^{-1})
A	light brown	5.5	4.4	6	76	12	12	Sandy loam	100
B	yellow	4.9	4.0	71	79	8	13	Sandy loam	100
C	dark brown	5.5	4.5	15	48	22	30	Sandy clay loam	300
D	dark red	5.4	4.4	15	48	22	30	Sandy clay loam	225
E	red	5.8	5.1	2	25	30	45	Clay	475
F	light red	6.6	6.0	10	85	4	11	Loamy sand	75
G	red	4.4	3.8	5	65	10	25	Sandy clay loam	225
H	yellow	5.8	5.0	12	83	10	7	Loamy sand	2.5
I	yellow	6.3	5.2	9	83	5	12	Loamy sand	25
J	red	6.1	4.8	3	82	6	12	Loamy sand	75
K	red	5.5	4.5	6	84	5	11	Loamy sand	100

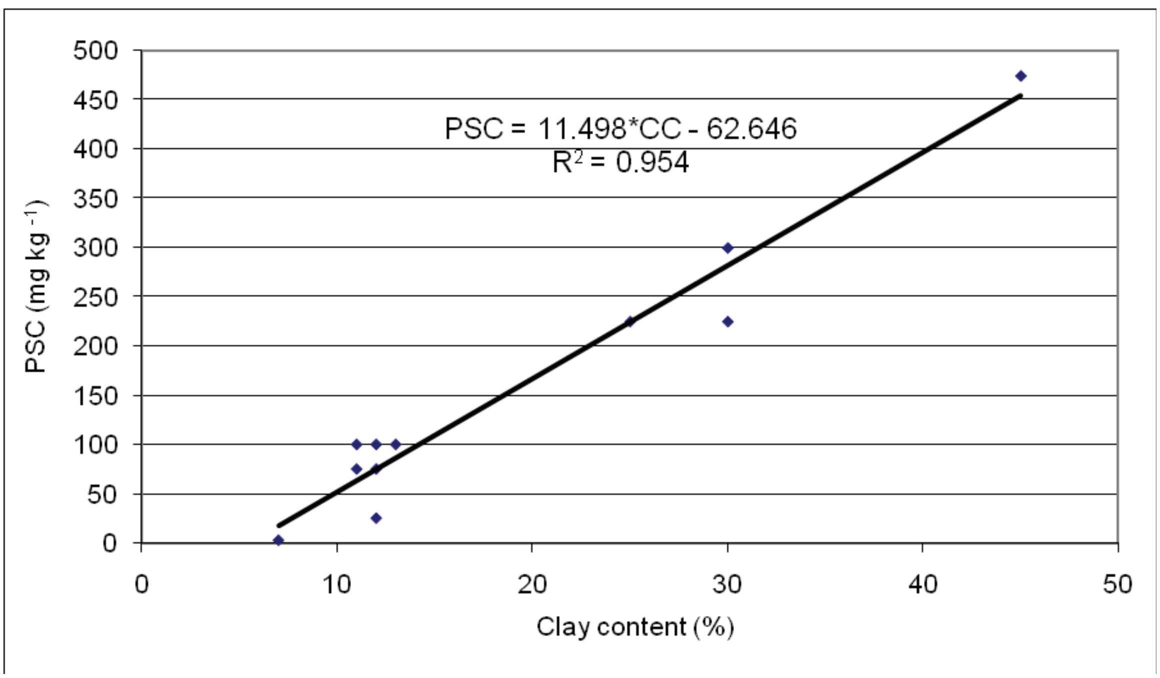


Figure 4.2 Correlation between the phosphate sorption capacity (PSC) and the clay content (CC) of 11 initially screened soils.

Soil extractions and analyses

Bray 1 (0.03 *M* NH_4F + 0.025 *M* HCl) a commonly used extractant to assess plant available P was used. Extracts were analysed using an axially viewed ICP – AES (The Non-Affiliated Soil Analysis Work Committee, 1990).

The relative phosphate fertilizer value (RPFV) of sludge was estimated after 168 days of incubation.

RPFV = Bray (sludge amended soil) / Bray (MAP amended soil) X 100

4.5.3. Results and discussion

General sludge properties

The Sasol sludge had the lowest Fe (0.75%), P (0.79%) and Ca (0.65%) than the other four sludges (Table 4.4). Considering the general criteria set for BPR sludges (Fe < 1%), the Sasol sludge falls within this category. Elemental ratio of P:Fe has been commonly used to evaluate the P fertilizer value of a sludge. For instance, Samie and Römer, (2001), recommended that sludges with a P:Fe ratio of >1:5 not to be considered for agricultural use. The P to Fe molar ratio indicates the unity for the solid phase of $FePO_4$. $2H_2O$, the same also applies for the P:Al ratio of the aluminium phosphate analogue ($AlPO_4$. $2H_2O$). A P:Fe ratio approaching unity indicates that the P predominantly resides in $FePO_4$. $2H_2O$. The same logic applies for the P:Al ratio. The P:Fe ratio of Sasol sludge was 1.9, that means on mole basis there was twice as much P in the sludge than Fe (Table 4.5). In fact, the P content of Sasol sludge was higher than the sum of the Fe, Al & Ca concentration. This indicates that an appreciable fraction of the Sasol sludge resides in organic form.

PWEP is often used as an indicator for P plant availability as well as the potential for environmental risk (Elloitt and O' Connor, 2007). Of the sludges tested, Sasol sludge had the second highest PWEP (24.6%), after the enriched ASP sludge (72.4%). This was slightly higher than an average PWEP value reported by Elloitt and O' Connor (2007) for chicken manure (20.7%).

Table 4.5 The molar ratios of P:[Fe+ Al], P:Fe, P :Al, and P:Ca, and the Percentage of Water soluble P (PWEP) for four sludge and MAP.

Sludge	P : [Fe + Al]	P : Fe	P : Al	P : Ca	PWEP[a] (%)
Sasol	1.2	1.9	3	1.6	24.6
Agriman	1.4	1.5	26.9	1.5	6.02
ASP	7.5	12.9	17.8	12.1	72.4
Vlakplaas	0.4	0.4	2.8	1.3	0.11
Daspoort	0.6	0.7	5.1	1.6	0.26
MAP	49.2	69.2	169.5	59.2	81.1

a. PWEP = (Water Soluble P / Total P) × 100

The N and P enrichment of the ASP sludge resulted in the highest PWEP. This is indirectly revealed by the large P:Fe and P:Al ratios of the ASP sludge which indicate that only a small amount of P resided in either ferric or aluminium phosphates.

Vlakplaas and Daspoort sludges had much higher Fe contents (11.5% & 10.4% respectively) than the other three. The effect of such high Fe content on P availability is clearly indicated by the low PWEP and a P:Fe ratio of less than 1. In addition the P:Fe ratios of both Vlakplaas and Daspoort were less than unity. This signifies the presence of “excess” Fe than P in the sludge, which could potentially fix the soluble P in a soil. Therefore, Vlakplaas and Daspoort sludges have lower P fertilizer values, but pose little environmental risk through leaching or runoff losses.

Influence of sludge amendment on P availability

Many soil based coefficients have also been proposed to predict plant available P. Acid oxalate extractable P expressed as a molar ratio of acid oxalate extractable Al and Fe, known as the phosphate sorption index (PSI), is commonly used in Europe as a predictive tool to assess potential mobility and plant availability of P. In USA a similar index using Mehlich 1, standard P extractant used in the United States, has been developed (O’ Connor *et al*., 2004; Krogstad *et al*., 2005). However, these approaches are not commonly used in South Africa. Bray-1 extractability is a familiar reference and commonly used as a measure of plant available P for non-calcareous soils of the Eastern Highveld. Therefore Bray-1 was used as the basis for RPFV estimation.

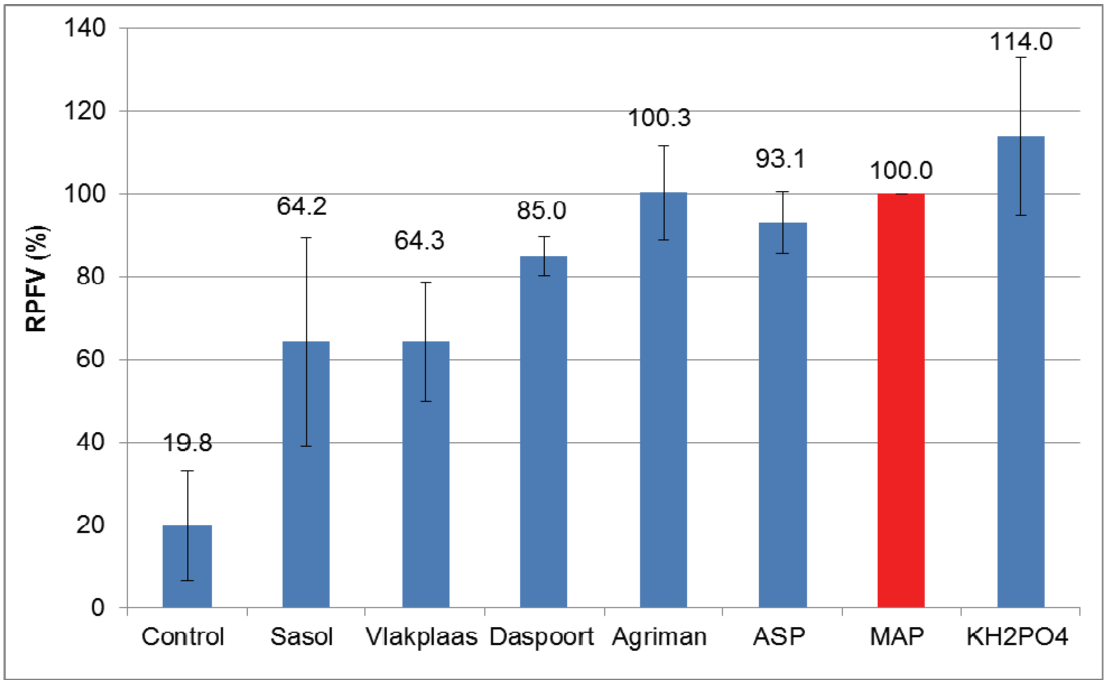


Figure 4.3 The mean relative phosphate fertilizer value (RPFV) of various sludges, estimated from the P Bray extractability after 168 days of incubation. (error bars indicate the standard deviation)

RPFV index predicted that Vlakplaas and the Sasol sludge had the lowest P fertilizer value due to the high Fe content (Fig 4.3). Nonetheless, the RPFV remained still higher than the control treatment and the excess Fe in the sludge did not reduce P Bray extractability. It was interesting to note that, the RPFV of the Sasol sludge was relatively similar to that of Vlakplaas, after 168 days of incubation. This was despite the fact that Sasol sludge being biologically treated and had low Fe content. Total elemental analysis of the Sasol sludge showed high levels of elements like selenium, vanadium, and various forms of organic compounds that could negatively affect the micro-organisms responsible for mineralization. In addition, previous research on composting using Sasol sludge also showed an initial lack of heat generation, which is associated with the microbially mediated exothermic oxidation of organic material. However, the limiting conditions were overcome and the normal heat generation occurred later.

No real trend was observed for the Vlakplaas, Daspoort, Agriman and ASP over time. Vlakplaas had a mean RPFV of 70.2 ± 9.08% for all the incubation times, Daspoort (76.0 ± 11.3%), Agriman (93.2 ± 9.57%) and ASP (98.7 ± 7.63%).

The influence of phosphorus sorption capacity (PSC) on the Bray extractability of soils is well illustrated in Fig 4.4. An increase in the PSC from 2.5 mg kg^{-1} to 225 mg kg^{-1}, reduced plant available P by half. However, at sludge application rates of 10 t ha^{-1}, even the Sasol and Vlakplaas sludges were able to maintain Bray 1-extractable P levels of above 30 mg kg^{-1}(optimum for most crops) for soils with PSC of 2.5, 100, and 225 mg kg^{-1}. The concentration of Bray-1 extractable P, however, remained below 10 for soils with a PSC of 475 mg kg^{-1}. Therefore, soil properties could play dominant role in determining the availability of P in the presence of high P fixing soils.

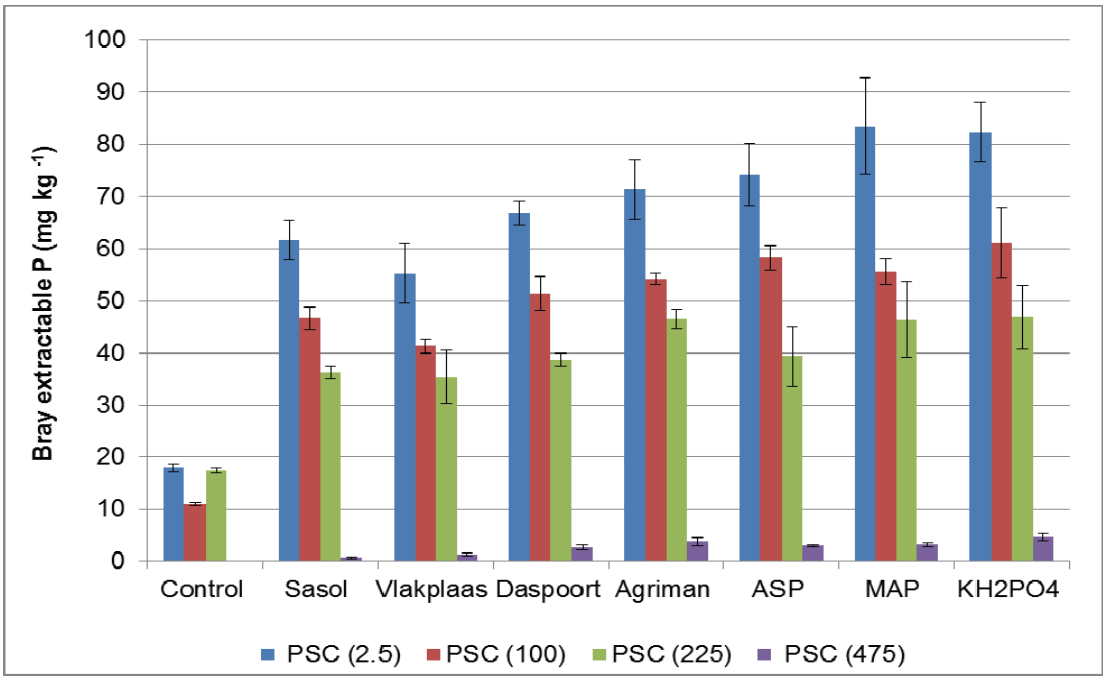


Figure 4.4The influence of phosphate sorption capacity (PSC, in mg kg^{-1}) on the Bray-1 extractability of P after 168 days of incubation.

On average, the RPFV index predicted that Agriman and ASP were similar to MAP. These products are potentially agronomically viable phosphate sources. However, it must be kept in mind that the P content is still appreciably lower than that of MAP. The overall fertilizer value of sludges, needs to take into account the economic viability (content, availability and transport cost).

4.6. Phyto-availability studies: Greenhouse Pot trial and field trial

4.6.1 Background

The incubation study was followed up by a greenhouse pot trial as well as a field trial, to assess P phyto-availability from sludge.

4.6.2 Aim

To assess actual P phyto-availability from selected sludges in a greenhouse pot trial and under field conditions.

4.6.3. Material and methods

Greenhouse pot trial

The pot trial was conducted on one soil type (soil A (PSC 100), Table 4.4) treated with similar sludge types used in the incubation studies, potassium phosphate, commercial mono-ammonium phosphate (MAP), and a zero control. The treatments were replicated four times. The test crop was an open pollinated maize cultivar (Sahara) from K_2-Agri. The trial was duplicated in such a way that similar trial was conducted without maize. In this trail the treatments received similar P rates (equivalent to 280 kg Pha^{-1}), N (350 kg Nha^{-1}), and K (300 kg Kha^{-1}). The plants were harvested 42 days after planting and both wet and dry above ground biomass were determined. Harvested plants were dried, pulverized and acid digested for P determination using an axially viewed ICP – ES. A Bray 1 (0.03 *M* NH_4F + 0.025 *M* HCl) extraction was also performed on all the soils.

Field trial

The field trial was conducted in Leandra, Mpumalanga on a virgin soil not previously used for agriculture. This site was the source for soil A (PSC=100 mg kg^{-1}) which was previously used in the pot trial. Two sludges (Agriman and ASP) and a commercial fertiliser (MAP) were applied at a rate of 280 kg total P ha^{-1} and incorporated into a soil depth of 0.25 m. The treatments were laid out in a randomized block design with four blocks. The size of each plot was 20 x 20 m. Soil samples were collected before and after sludge application until harvest. The same maize cultivar was used for both the pot and field trials. Bray-1 extractable P was used to correlate plant available soil P with the P concentration of maize above ground biomass. Both maize grain yield and above ground biomass P uptake were measured. The collected maize aboveground biomass was pulverized and acid digested for P determination using an axially viewed ICP – AES.

4.6.4 Results and discussion

Greenhouse pot trial

Maize P uptake from Agriman and ASP treated soils were significantly higher ($p \leq 0.05$) than Vlakplaas and Daspoort treatments (Fig 4.5). The trend was similar to the prediction made by RPFV index (Fig 4.3), except for the Sasol sludge treatment which showed greater plant availability than the RPFV index. There were no significant differences in P uptake between the commercial fertilizers, the Agriman and ASP sludge products.

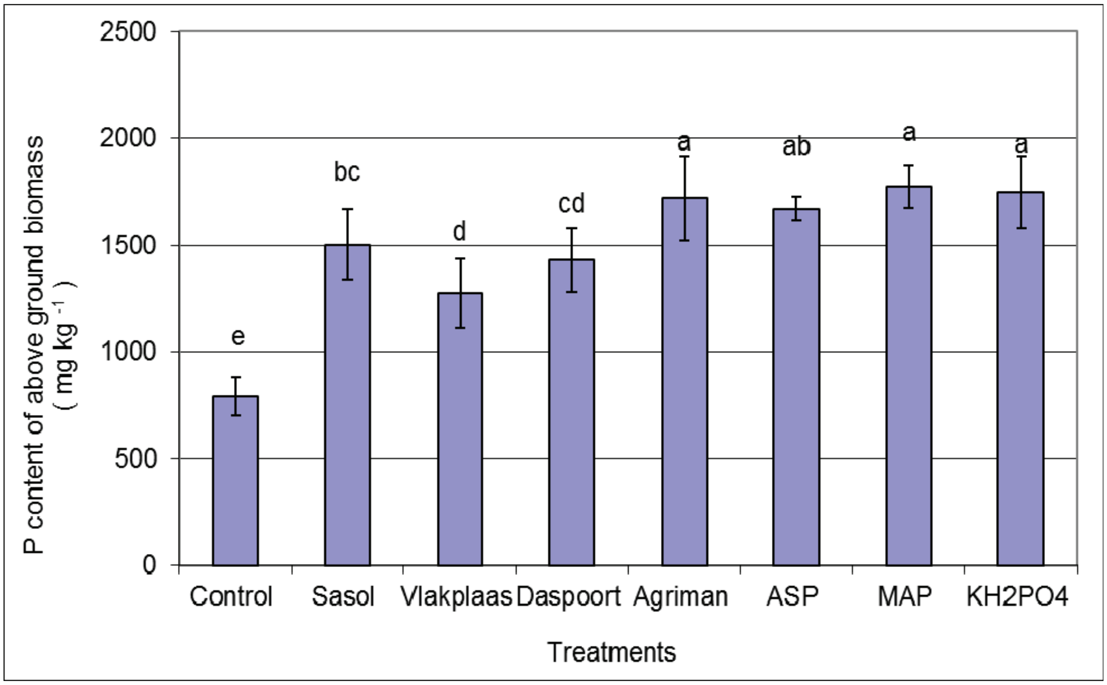


Figure 4.5 P content of maize above ground biomass (42 days after planting) grown in a soil treated with various sludges, a commercial inorganic fertilizer and a zero P control treatment (P content expressed as mass of P per mass of dry plant material).

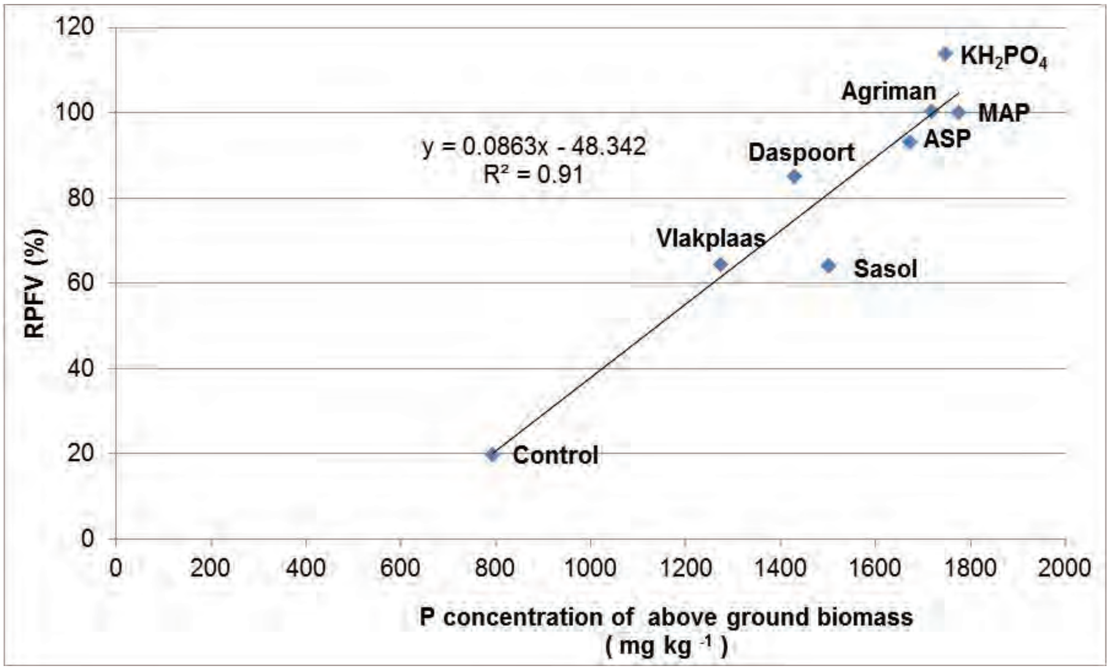


Figure 4.6 Correlation between the predicted RPFV (Bray 1-P from the 168 days of incubation) and P concentration of maize above ground biomass from the pot trial.

There was a strong correlation (R^2= 0.91) between the calculated RPFV and above ground biomass P concentration (Fig. 4.6). This indicates the potential of RPFV as a useful index to predict plant availability of P from sludge amended soils.

Field trial

The field trial data did not show any significant difference in grain yield ($p \leq 0.05$) between treatments. This was mainly attributed to the fact that the soil P level was sufficient and therefore P was not a limiting factor for crop growth (Fig 4.7). However, there was a significant grain P concentration difference between the zero P control treatment and other treatments (Fig 4.8.).

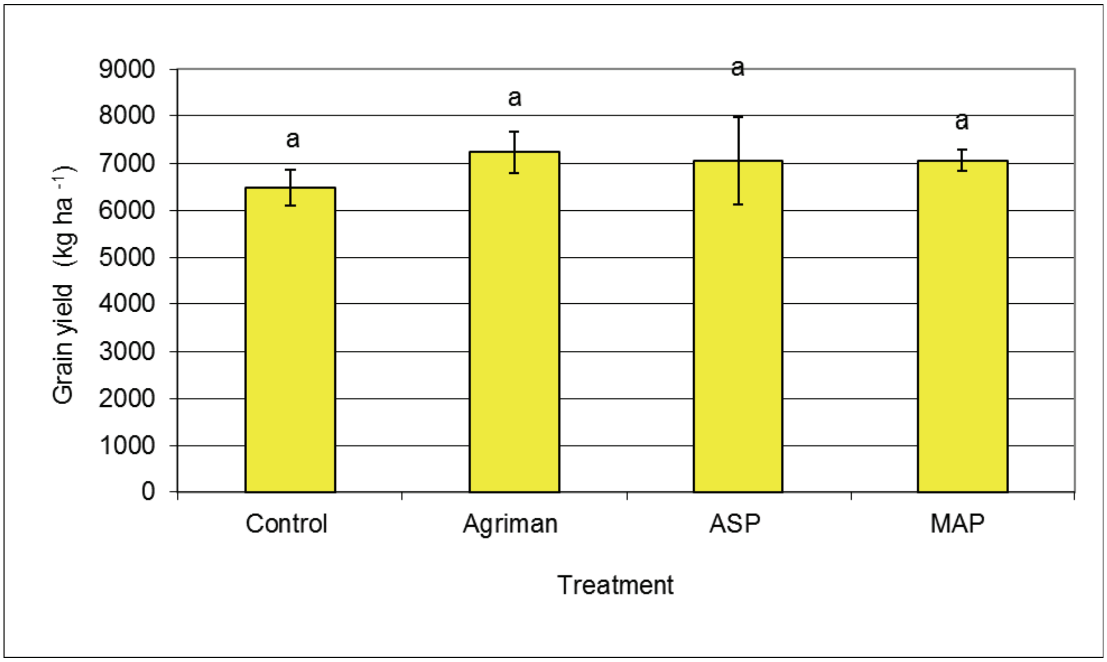


Figure 4.7 Maize grain yield grown in a soil treated with two sludge types, an inorganic commercial fertilizer and a zero P control treatments (field trail).

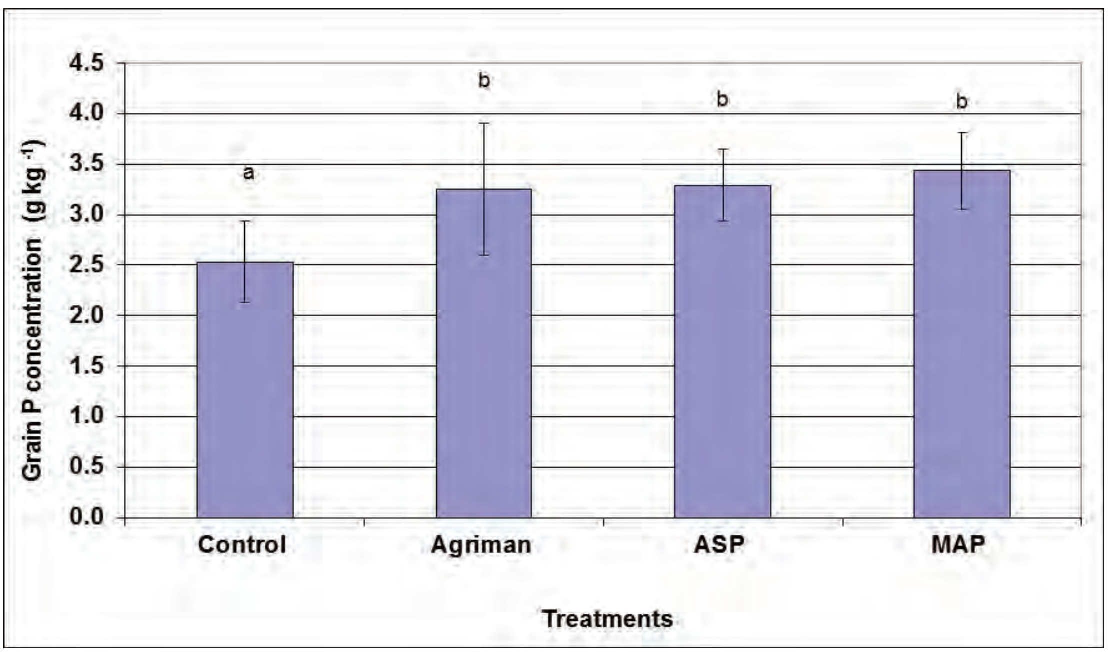


Figure 4.8 P concentration of maize grain grown in a soil treated with two sludges, an inorganic fertilizer, and a zero P control treatment (field trial).

Although Agriman sludge was mot enriched with P as in the case for ASP, it had similar RPFV values with ASP. Nevertheless the total P content of Agriman sludge (3.74%) is third of ASP (11.5%). This has transport cost implications because the amount of sludge required to get a given amount of P from Agriman is three times than ASP.

Apart from being a source of phosphate, the addition of organic material like sludge is also expected to decrease phosphate sorption of soils. This is attributed to the release of ligands during the break down of sludges. For instance, fluvic acid is often isolated from sludge amended soils (Sposito *et al*., 1978). The conjugated bases of these acids will compete with phosphate for adsorption sites. Furthermore, these ligands are also strong complexing and reducing agents that can increase the solubility of ferric (oxy) hydroxides and reduce the sorption surface area.

4.7 Conclusion and recommendations

Both open and closed system incubation studies have their own merits and demerits. From model parameter generation perspective, the closed system is preferred because of the control that one can have on the system. The open system could better be used for model validation.

Sludge products like Agriman and ASP showed similar P plant availability as the commercial fertilizer MAP, indicating their viability as low grade P fertilizer sources. The concentration of P in Agriman was, however, a third of the concentration in ASP. The P fertilizer value of the various sludges was influenced by the wastewater treatment processes. Sludges treated with Fe, such as Vlakplaas, experienced low P phyto-availability. This is in contrast to sludges that have low Fe content such as Agriman.

Field scale studies showed that there was a strong correlation between RPFV and maize above ground biomass P concentration. It also apparent from this one season field study that plant availability of P from sludge amended soils with high P fixing capacity is mainly influenced by the soil properties. Therefore, at high sludge application rates, where the amount of P added exceeds the PSC of a soil, sludge characteristics are expected to influence P phyto-availability.

From a famer's perspective, the P fertilizer value of sludge is a function of the content and availability of the nutrients as well as the cost of implication related to transport from wastewater treatment plants to farms and the on farm sludge application costs. Therefore, it is of at most importance to dry sludges and improve the plant availability of P as well as other nutrients to minimize the negative implication on transport costs.

Future research

More detailed sludge characterization needs to be conducted in order to better understand the long term release characteristics of various sludges treated both chemically and biologically. Proper sludge characterization will help to improve the parameterization of the SWBSci model and give a better picture of the fertilizer value of a given sludge type for those who want to do quick simple estimations.

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CHAPTER 5

LOCAL FIELD SCALE STUDIES ON AGRONOMIC CROPPING SYSTEMS UTILIZING MUNICIPAL SLUDGE AS NUTRIENT SOURCE

5.1 Introduction

Several studies have shown that maize stover and grain yield increased as sludge application rate increased, regardless of soil type (Cunningham *et al*. 1975; Kelling *et al*. 1977; Binder *et al*. 2002; Lavado *et al*. 2006; Bozkurt 2006). In certain cases, crop response was non-linear at high application rates (Soon *et al*., 1978; Binder *et al*., 2006), while in others, high sludge application rates resulted in rapid grain yield decline due to salt toxicity (Cunningham *et al*., 1975).

Crop response to nutrient supply is largely influenced by the availability of water. Under dryland farming, Soon *et al*. (1978) reported hardly any crop response for N supplies exceeding 200 kg ha^{-1}, regardless of soil type. In contrast, Binder *et al*. (2002) reported significant improvement in maize grain yield under irrigated farming as N supply from sludge increased to 400 kg ha^{-1}.

When using sludge in agricultural lands at high rates, there are potential environmental risks that need to be considered. Referring back to the previous two examples, under dry land conditions, high N supplies of 400 kg ha^{-1} resulted in significant accumulation of nitrate in the top 0.75 m soil layer (Soon *et al*., 1978). In contrast, the irrigated system that received a similar amount of N barely showed any nitrate accumulation, mainly due to the higher nitrogen use efficiency of the same crop (Binder *et al*., 2002). Significant accumulation of nitrate was reported in the latter case as the N supply increased to 950 kg ha^{-1} due to the decline in the nitrogen use efficiency (N uptake/N supply) of the crop. The main concern with nitrate accumulation in a soil profile is ground or surface water pollution, especially during high rainfall events.

The other challenge when using sludge in agricultural lands is the nature of N from sludge. A large fraction of the N in sludge is organic, and needs to be mineralized into an inorganic form before it can be utilized by plants. The rate at which the organic N is released into an inorganic form is influenced by external factors besides the inherent chemical compositions of the sludge. These include temperature, water, and pH, among others. Consequently, there might be significant nutrient carry over effects in subsequent years after application, depending on the nature of the sludge and climatic and edaphic factors (Cogger *et al*., 2001; Binder *et al*., 2002). Therefore, one should consider the amount of nutrients that were carried over from previous years before deciding how much to apply. Such good practice improves crop production and is a precondition for sustainable agricultural use of sludge. Most of the previous medium to long term controlled field studies on agricultural use of sludge have been conducted overseas under varying soil and climatic conditions. Direct extrapolation of those studies to South African conditions could compromise both agricultural production and the environment. It is imperative, therefore, that local studies be conducted to inform production practices that will ensure for agricultural and environmental sustainability.

The objectives of this particular study were:

- To investigate the dynamic nature of ideal sludge application rate across a range of management practices and cropping systems, and
- To investigate the agronomic benefits and resilience of using municipal sludge according to crop N demand

5.2 Materials and methods

5.2.1 Field site description

Field experiments were conducted at the East Rand Water Care Works (ERWAT), Johannesburg, Gauteng, South Africa. The study site is situated at an elevation of 1577 m above sea level, latitude 26^{o} 01' 01" S and longitude of 28^{o} 16' 55" E (Figs. 5.1, 5.2, and 5.3). The soil of the experimental site is a clay loam (Hutton, Soil Classification Working Group, 1991) having a clay content of 36-46%, and pH (H_2O) of 5.3 to 6.1. In the beginning of the study, the cation exchange capacity of the soil (ammonium acetate extract) was 10-12 cmol kg^{-1} and the electrical conductivity of a saturated paste extract ranged from 8 mS m^{-1} at 1.2 m depth to 36 mS m^{-1} in the top 0.3 m soil layer.

Figure 5.1 Satellite image of the study site (a) enhanced satellite image (b)

5.2.2 Weather data

Weather data was collected from an automatic weather station located about 100 m from the experimental site. The automatic weather station consisted of an LI 200X pyranometer model PY 47034 (LiCor, Lincoln, Nebraska, USA) for measuring solar radiation, an electronic relative humidity and temperature sensor installed in a gill screen (R.M. Young, Minnesota, USA), an electronic cup anemometer to measure wind speed (R.M. Young, Minnesota, USA), an electronic rain gauge Model TR-525M-R2 (Texas Electronics Inc., Dallas, Texas, USA), and a CR10X data-logger (Campbell Scientific Inc., Utah, USA). The area has a long term annual average rainfall of 700 mm, mainly during the months of October to March. Rainfall recorded between 2007 and 2009 is presented in Appendix A.

Figure 5.2 Panoramic view of the study site during the summer season with dryland and irrigated maize, dryland pasture, and lawn for sod production

5.2.3 Sludge characteristics

The sludge used in this study was anaerobically digested and air dried in concrete drying beds (Fig. 5.4). According to the current South African sludge guideline (Snyman and Herselman, 2006), this sludge is classified as pollutant class “a” due to its low heavy metal content (Table 5.1). Based on the microbiological report from the ERWAT laboratory, the sludge can also be classified as microbiological class “A”.

Considering the low odour and vector attraction characteristics of the sludge, it can be classified as stability class 1. The current South African sludge guideline (Snyman and Herselman, 2006) allow such sludges to be utilized in agriculture without restriction, as long as the N applied does not exceed crop demand, with the upper limit set at 10 t ha^{-1} yr^{-1}. Selected chemical compositions of the sludge are presented in Table 5.1.

Table 5.1 Selected characteristics and chemical constituents of anaerobically digested, paddy dried sludge used during the 2007/08 and 2008/09 growing seasons

Constituents	Unit	Growing seasons	
		2007/08	2008/09
N	%	3.09	2.74
NH_4_N	mg kg^{-1}	7660	6440
NO_3_N	mg kg^{-1}	11.1	9.45
Total P	%	2.24	3.43
P – Bray1	mg kg^{-1}	66.04	50.12
Total C	%	20	21.2
Water content	%	38	33
EC_e	mS m^{-1}	3110	1043
K	mg kg^{-1}	1356	1720
Ca	% (mass base)	1.00	2.70
Mg	mg kg^{-1}	1145	3410
pH (H_2O)		6.08	8.1
Cd	mg kg^{-1}	18.91	19
Hg	mg kg^{-1}	1.81	1.08
Cr	mg kg^{-1}	503.8	419
As	mg kg^{-1}	17.94	6.5
Pb	mg kg^{-1}	102	75
Zn	mg kg^{-1}	2325	4920
Ni	mg kg^{-1}	144.5	152
Cu	mg kg^{-1}	526.8	681
Fe	%	15.01	14.60
Al	%	1.61	1.45

Figure 5.3 Panoramic view of the study site during a winter season with irrigated oats and lawn for sod production

5.2.4 Field trial and treatments

Plots of 25 m^2 were arranged in a randomized complete block design comprising four replications of five treatments (Figs. 5.2 and 5.3). The trial was laid out to accommodate widely different levels of bio-solid application to high and low productivity-agronomic cropping systems. It consisted of two agronomic cropping systems namely: irrigated maize-oat rotation and dryland maize.

The treatments for dryland maize and the irrigated maize-oats rotation consisted of three sludge application rates (4, 8, and 16 t ha^{-1} yr^{-1}), a control, and an inorganic fertilizer treatment. The value of 8 t ha^{-1} yr^{-1} represents the annual agricultural upper limit of the former (1997) South African sludge guideline (Water Research Commission, 1997), which has recently been raised to 10 t ha^{-1} yr^{-1} (Snyman and Herselman, 2006). Sludge rates of 4 and 16 t ha^{-1} yr^{-1} represent half and double the former norm. Total N and P applied from sludge during the study period are presented in Table 5.2. Municipal sludge is a poor source of K, therefore all sludge treatments received additional K fertilizer equivalent to

the inorganic fertilizer treatment. For the irrigated maize-oats rotation, the annual sludge application was split into two, with half applied to both crops at planting. For dryland maize, however, the entire amount was applied at the beginning of the season before planting. This trial was initiated in 2005, before the publication of the current sludge guideline.

Table 5.2 Total N and P added from sludge and inorganic K added from inorganic fertilizer to all sludge treatments of dryland maize and irrigated maize-oat rotation.

Time of fertilizer application	(Sludge application rate (t ha^{-1}))	2007/08			2008/09		
		N	P	†K	N	P	†K
		kg ha^{-1}					
Dryland maize	0	0	0	0	0	0	0
	4	124	90	60	110	137	60
	8	248	180	60	220	274	60
	16	495	358	60	438	548	60
Irrigated maize-oat rotation	0	0	0	0	0	0	0
	4	124	90	186	110	137	186
	8	248	180	186	220	274	186
	16	495	358	186	438	548	186

† K was topped up from inorganic fertilizer as sludge is a poor source of K

Figure 5.4 Concrete drying beds at Vlakplaas from which sludge was taken for this study

Applications of the inorganic fertilizer maize treatment were conducted according to the recommendations of the OMNIA Fertilizer Company, with some modification to the amount of K recommended (Table 5.3). The amount of inorganic fertilizer applied to irrigated oats was based on the Fertilizer Handbook (Fertilizer Society of South Africa, 2000) for a target forage yield of 11 t ha^{-1} (Table 5.3).

Table 5.3 Inorganic fertilizer application timing and type of fertilizer applied to dryland maize, irrigated maize, and irrigated oats during the 2007/08 and 2008/09 growing seasons at ERWAT, Ekurhuleni district, South Africa.

Time of fertilizer application	Fertilizer type	Dryland maize			Irrigated maize			Irrigated oats		
		N	P	K	N	P	K	N	P	K
		kg ha^{-1}								
At planting time	NPK 2:3:2(22%)	13	20	13	6	9	6	6	9	6
	LAN (28% N)	17								
	KCl (50% K)			7			14			
Three weeks after planting	Limestone ammonium nitrate (LAN)				90			84		
	Super phosphate (18.5% P)					31			11	
Five weeks after emergence	LAN				64			55		
	Super phosphate								10	
	KCl						40			40
Seven weeks after emergence	LAN	66			66			55		
	Super phosphate								10	
	KCl			40			40			40
Cumulative annual application		**96**	**20**	**60**	**226**	**40**	**100**	**200**	**40**	**86**

Sludge and inorganic fertilizer were broadcast (Fig. 5.5) and immediately incorporated into the soil with a manually operated, diesel powered rotovator (Agria) (Fig. 5.6). After sludge incorporation, the soil was levelled using rakes and maize (PAN 6966) was planted in 0.9 m rows at rates of 80000 seeds per hectare under irrigation and half this rate under dryland conditions. Each plot consisted of 6 rows, 5 m in length, with the outer row on either side taken as border rows.

After harvesting irrigated maize, each treatment received the corresponding sludge and inorganic fertilizer rates, and these were incorporated before planting oats. Oats were planted at a rate of 90 kg ha^{-1} using a hand-drawn planter with double disk openers. Each oat plot consisted of 15 rows, 5 m in length spaced 0.3 m apart, of which two rows on either side are border rows. Maize and oats planting and harvesting dates are presented in Table 5.4.

Figure 5.5 Manual sludge broadcasting over the experimental plots

Figure 5.6 Sludge incorporation using a manually operated, diesel powered rotovator (Agria)

Table 5.4 Planting and harvesting dates for dryland maize, irrigated maize, and irrigated oats during the 2007/08 and 2008/09 growing seasons at ERWAT, Ekurhuleni district, South Africa.

	Dryland maize		Irrigated maize		Irrigated oats	
Growing season	Planting date	Harvesting date	Planting date	Harvesting date	Planting date	Harvesting date
2007/08	18 Nov. 2007	15 Apr. 2008	18 Nov. 2007	15 Apr. 2008	02 May 2008	15 Nov. 2008
2008/09	18 Nov. 2008	15 Apr. 2009	18 Nov. 2008	15 Apr. 2009	02 May 2009	15 Nov. 2009

5.2.5 Irrigation

The irrigated maize-oats rotation experiment was planted under drip irrigation (Fig. 5.7).

Figure 5.7 Drip irrigation system and wetting front detectors installed in the maize-oat rotation cropping system

The lateral spacing between dripper lines was 0.3 m. The distance between the drippers in a line was 0.3 m. The system was operated at a pressure of 100 to 150 kPa with an average drip rate of 1.3 l hr^{-1} per emitter (13.8 mm hr^{-1}). In the absence of rainfall, maize was irrigated 10 mm every three days for the first four weeks after planting. This was followed by a weekly irrigation according to neutron probe (Model 503 DR CPN Hydroprobe, Campbell Pacific Nuclear, California, USA) deficit readings, to fill the profile to field capacity until harvest.

During the winter seasons of each year, oats were irrigated 10 mm every three days for the first four weeks. During the rest of the season, the crop was irrigated on a weekly basis using a site calibrated neutron probe soil water deficit readings for the first week followed by next irrigation schedule according to SWB model recommendation.

5.2.6 Plant sampling

Plant samples were collected at the soft dough stages of both dryland and irrigated maize for forage yield (total above ground biomass) determination from 2 m lengths of the three middle rows (typically 22 plants under dryland and 43 plants under irrigation). Additional plant samples were collected for crop N and P uptake determinations. At crop maturity, plant samples were taken for grain yield determination from 2 m lengths of the four middle rows. Two additional plant samples per plot were taken from the dryland maize and four from the irrigated maize for crop N and P uptake determinations.

Oats were harvested at the hard dough stage to minimize damage by birds. Samples during harvesting were collected from 2 m lengths of the six middle rows. Additional plant samples were collected for N and P uptake determination.

5.2.7 Growth analyses

Plant samples collected for growth analyses from dryland maize and the irrigated maize-oats rotation were partitioned into leaves, stems, and grain (maize and oats). One sided leaf area was measured with an LI-3100 belt driven leaf area meter (LI-COR, Lincoln, NE, USA). Leaf area index (LAI) was computed from the measured one-sided leaf area divided by the ground sampling area. The components (leaves, stems, and grain) were then dried in a forced-air oven at 60^{o}C for 48 hours to determine above-ground biomass. Above-ground biomass was calculated as the sum of leaf, stem, and seed.

5.3 Results and discussion

5.3.1 Irrigated maize forage yield

Irrigated maize forage yield increased significantly with doubling the sludge-application rate (Fig. 5.8). There was also a similar response pattern for the grain yield (data not shown). This was mainly attributed to the relatively higher availability of nutrients, especially nitrogen, compared with the lower sludge application rates under water non-limited conditions. Highest forage yield was harvested from the 16 t ha^{-1} yr^{-1} sludge treatment and lowest from the control. It was also interesting to note that forage yield was lower in 2008/09 than 2007/08 growing season. This was mainly due to the decline in the soil N status under these water non-limiting conditions (Tesfamariam, 2009).

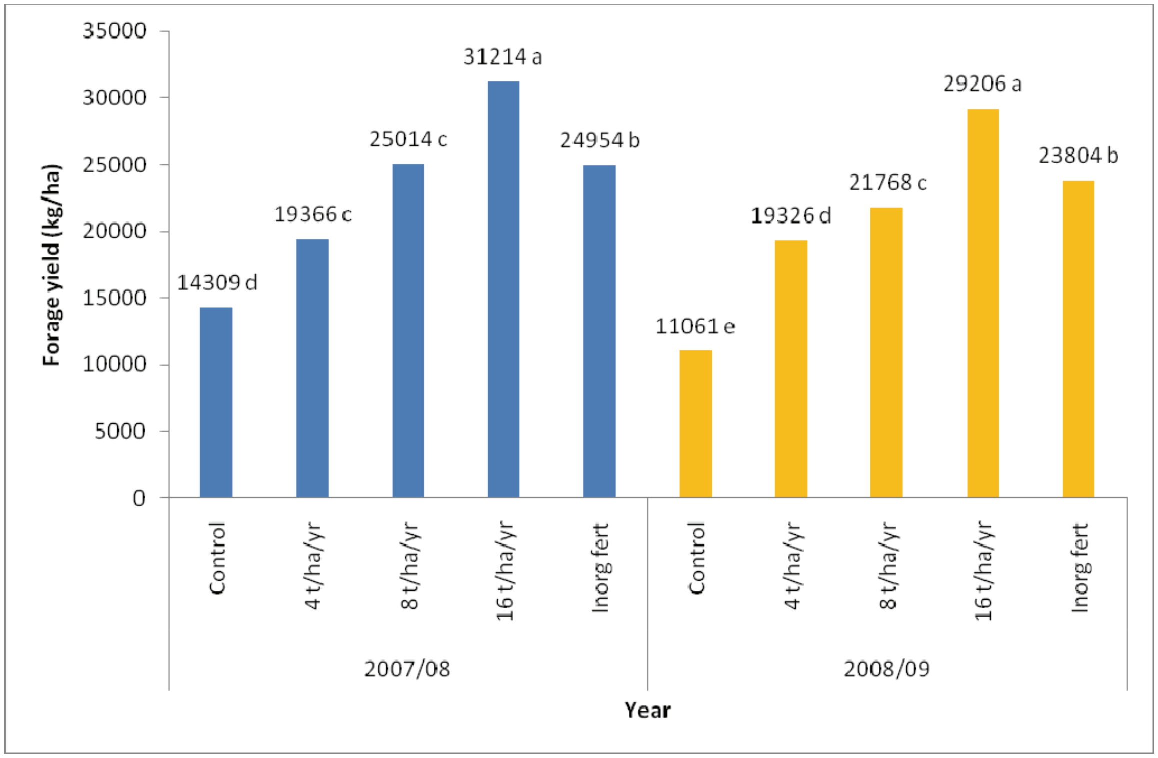

Figure 5.8 Irrigated maize forage yield for three sludge application rates (4, 8, and 16 t ha^{-1} yr^{-1}), an inorganic fertilizer treatment, and a control (no sludge or inorganic fertilizer) on a clay loam soil during the 2007/08 and 2008/09 growing seasons (bar graphs of the same year having different letters indicate statistical significance).

In the tropics, water and nitrogen are the limiting factors for crop growth and dry matter production (Moser *et al*., 2005). Under water non-limiting irrigated farming systems, however, nutrients, especially nitrogen are limiting. It is well documented that crop growth, dry matter production and forage yield increases with the availability of N (Bozkurt *et al*., 2006) mainly due to the supply of N and other nutrients, including C (Christie *et al*., 2001).

5.3.2 Dryland Maize forage yield

Doubling the upper limit sludge norm (16 t ha^{-1} yr^{-1}) increased dryland maize forage yield but this was not statistically significant (Fig. 5.9). Highest forage yield was harvested from the 16 t ha^{-1} treatment compared with lower sludge application rates. The inorganic fertilizer treatment produced similar forage yield as the 16 t ha^{-1} sludge treatment. This was despite twice the total N supply from the 16 t ha^{-1} sludge treatment.

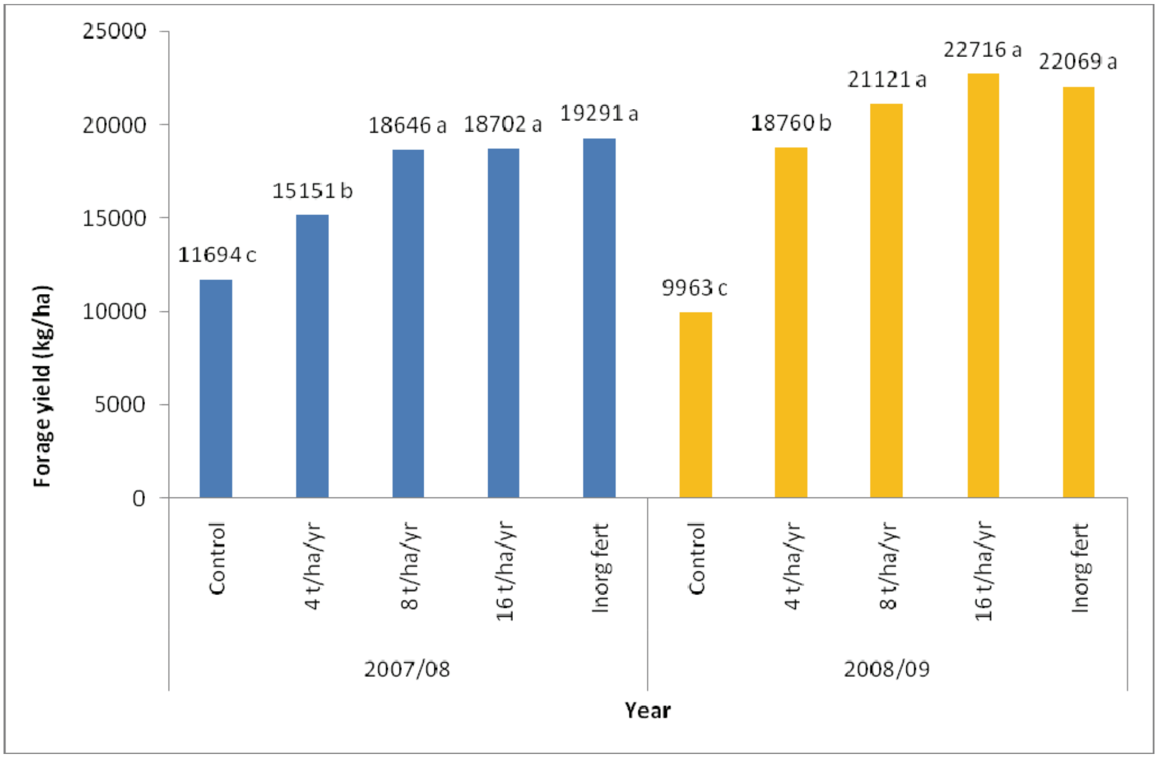

Figure 5.9 Dryland maize forage yield for three sludge application rates, an inorganic fertilizer treatment, and a zero application control on a clay loam soil during the 2007/08 and 2008/09 growing seasons (bar graphs of the same year with different letters indicate statistical significance).

Forage yield was relatively lower in 2007/08 than 2008/09 growing season, despite the higher rainfall experienced during the 2007/08 season (Appendix A). This was mainly due to the late planting of the maize crop in the 2007/08 season. Generally forage yield was far lower under dryland than similarly fertilized irrigated treatments due to the shortage of water under dryland conditions indicating the dynamic nature of sludge application rate. According to Tisdale *et al*. (1985), crop response to nutrient supply depends on the soil water status and crop yield reduction is inevitable for every stress event (Rhoads and Bennett, 1990). Therefore, a dynamic reasoning support tool is vital for resilient agronomic use of sludge.

5.3.3 Oat forage yield

Oat forage yield increased consistently with increase in sludge application rate (Fig. 5.10). Oat grain yield also followed similar trends to that of the forage (data not presented). This is in line with studies conducted in other countries (Wang *et al*., 2000; Collins *et al*., 1990). Oat forage yield from the inorganic fertiliser treatment was higher than the 8 t ha^{-1} sludge treatment, despite similar total N supply. This is attributed to the higher plant availability of N from the inorganic fertilizer treatment as opposed to sludge, that has a large organic N fraction.

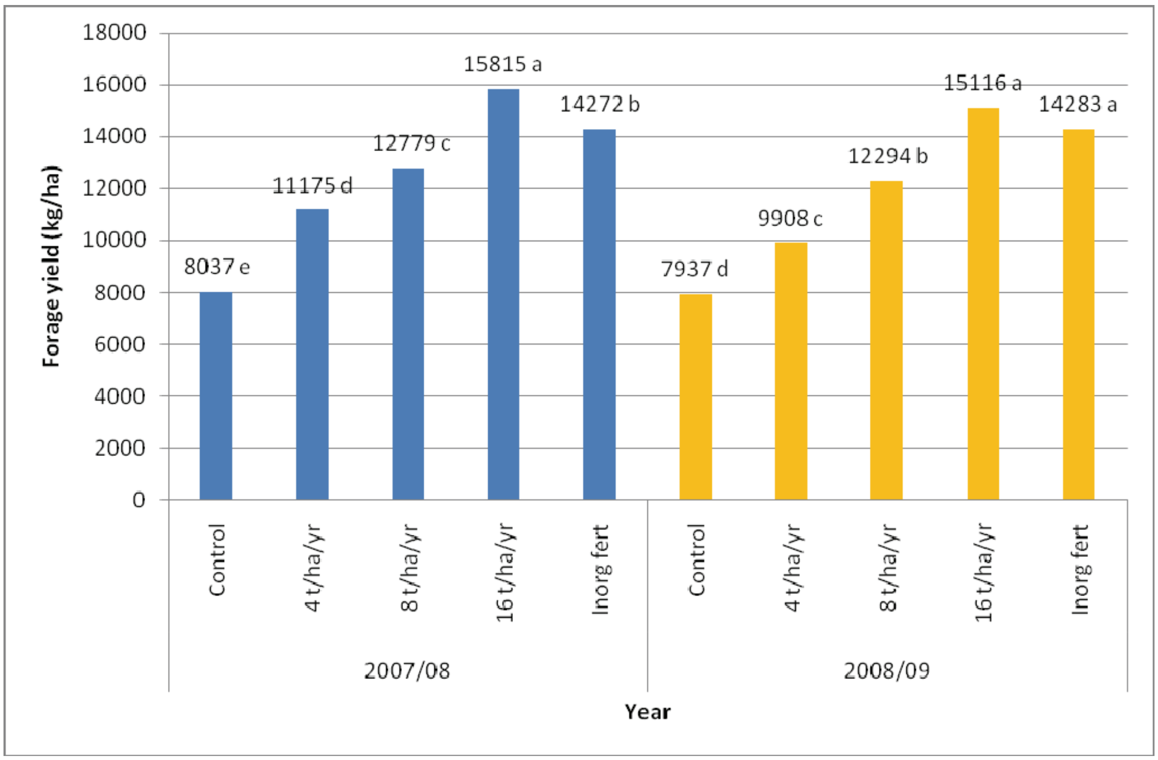

Figure 5.10 Irrigated oat forage yield for three sludge application rates, an inorganic fertilizer treatment, and a zero application control on a clay loam soil during the 2007/08 and 2008/09 growing seasons (bar graphs of the same year with different letters indicate statistical significance).

5.3.4 Nitrogen and phosphorus mass balances

Dryland maize

Dryland maize forage N uptake increased consistently with increase in sludge application rates (Table 5.5). Under this crop system, the soil was able to supply 77 kg N ha^{-1} during the 2007/08 growing season and twice higher during 2008/09 to the zero control treatment (zero sludge or inorganic fertilizer applied). This is equivalent to 50% of the total N uptake by the 16 t ha^{-1}. The amount of N supplied from the soil organic matter for this study site was twice higher than the average value estimated for South African soils (40 kg ha^{-1} yr^{-1}) by Van Biljon et al. (2008). This is to be expected considering the previous history of the study site, which was natural grassland and only cultivated during the past six years as opposed to the decades, if not, a century of cultivation for most South African agricultural lands.

The two year study showed that forage N uptake was 16% times higher on average than grain N uptake under dryland maize production. It was also interesting to note that forage and grain N uptake was twice higher on average in 2008/09 than 2007/08 due to the late planting during the 2007/08 growing season indicating the effect of planting time on crop nutrient uptake. According to Tesfamariam (2009), the lowest maize forage and grain yield for the study site was recorded during the 2006/07 growing season, which received 426 mm rainfall. Grain N uptake during the 2008/09 growing season was 8 times higher on average than the dry 2006/07 season (Table 5.5, last column) because of the relatively higher rainfall of 664.5 mm experienced during the 2008/09 season (Appendix A). These facts indicate the dynamic nature of sludge application rates across management practices, planting time, and water availability. Consequently, sludge application rates need to be adjusted accordingly using a dynamic reasoning support tool.

Similar to N uptake, dryland maize forage P uptake increased with increase in sludge application rate (Table 5.6). Based on the mass balance calculation of total P imported with sludge less forage P export, total P supply was far higher than crop demand, even at the lowest sludge application rate of 4 t ha^{-1} yr^{-1} under the prevailing climatic conditions.

It was quite clear from this and previous studies conducted in other countries (Maguire *et al*., 2000; Kelling *et al*., 1977; Peterson *et al*., 1994) that sludge application according to crop N demand could result in excess P addition more than what the crops can utilize. This in the long-term might contribute to an increase in the risk for non-point sources of pollution through runoff (Maguire *et al*., 2000). Local studies also indicated that both total and Bray-1P increased in the top soil of areas that received sludge for extended periods compared with surrounding fields that received no sludge (Herselman *et al*., 2005).

Table 5.5 Seasonal N mass balance of dryland maize grown at three sludge application rates, an inorganic fertilizer treatment, and a zero application control on a clay loam soil.

Treatment	N applied		Forage N uptake		Cumulative N applied less forage uptake	Grain N uptake		Cumulative N applied less grain uptake	Grain N uptake
	2007/08	2008/09	2007/08	2008/09		2007/08	2008/09		2006/07
Control	0	0	126	164	-290	61	128	-189	22
4 t ha^{-1} sludge	124	110	162	257	-185	92	243	-101	24
8 t ha^{-1} sludge	248	219	238	265	-36	117	220	130	28
16 t ha^{-1} sludge	496	438	244	336	354	131	290	513	33
Inorganic fertilizer	96	96	254	338	-400	138	289	-235	40

Table 5.6 Seasonal P mass balance of dryland maize grown at three sludge application rates, an inorganic fertilizer treatment, and a zero application control on a clay loam soil.

Treatment	P applied		Forage P uptake		Cumulative P applied less forage uptake	Grain P uptake		Cumulative P applied less grain uptake
	2007/08	2008/09	2007/08	2008/09		2007/08	2008/09	
Control	0	0	13	23	-36	10	19	-29
4 t ha^{-1} sludge	90	137	21	35	171	14	23	190
8 t ha^{-1} sludge	179	274	29	40	384	15	25	413
16 t ha^{-1} sludge	358	549	30	44	833	18	26	863
Inorganic fertilizer	20	20	22	49	-31	17	40	-17

Irrigated maize/oat rotation

Under this rotation system, the soil organic matter was able to supply 177 and 231 kg N ha^{-1} to the zero control treatment during the 2007/08 and 2008/09 growing seasons, respectively. This was twice higher than under dryland maize production, which is mainly attributed to the higher water availability under irrigated systems creating favourable environment for the microbes to decompose more soil organic matter.

In general, forage N uptake under intensive irrigated systems was twice higher on average than dryland maize. Similarly, irrigated maize-oat rotation grain production was able to utilise 1.4 times higher N than dryland maize grain production, proving the dynamic nature of sludge application rate and the need for a dynamic reasoning support tool.

According to the two year cumulative data (Table 5.8), a sludge application rate of 6.8 t ha^{-1} would suffice to satisfy the maximum P uptake by the 16 t ha^{-1} yr^{-1} sludge treatment in the study site. The assumptions are, sludge P content of 2.8% (average value during the study period), and a 60% plant availability of P (according to incubation trial – Chapter 4). Nonetheless, 40% of the crop P uptake in this study was contributed from the soil reserve as could be seen from the zero control treatment.

Finally, it is vital to note that crop N uptake (Table 5.7) by the irrigated maize-oat rotation was 4 times higher than P uptake (Table 5.8). This is in contrast to the 1:1 ratio of N:P in sewage sludge. Therefore, it is of utmost importance to consider the potential P build up which is associated with N based sludge applications. Although crop availability of this P and its potential environmental impact is influenced by the sludge treatment process (biological or chemical). Hence, sludge application rates should be synchronized with the background concentration of plant available P in the soil, sludge treatment process, and not only to the crop N demand.

Table 5.7 Seasonal N mass balance of irrigated maize-oat rotation grown at three sludge application rates, an inorganic fertilizer treatment, and a zero application control on a clay loam soil.

Treatment	N applied		Forage N uptake		Cumulative N applied less forage uptake	Grain N uptake		Cumulative N applied less grain uptake
	2007/08	2008/09	2007/08	2008/09		2007/08	2008/09	
Control	0	0	177	231	-408	102	138	-240
4 t ha^{-1} sludge	124	110	247	395	-408	138	219	-123
8 t ha^{-1} sludge	248	219	319	438	-290	194	276	-3
16 t ha^{-1} sludge	496	438	474	578	-118	291	367	276
Inorganic fertilizer	426	426	438	486	-72	244	293	315

Table 5.8 Seasonal P mass balance of irrigated maize-oat rotation grown at three sludge application rates, an inorganic fertilizer treatment, and a zero application control on a clay loam soil.

Treatment	P applied		Forage P uptake		Cumulative P applied less forage uptake	Grain P uptake		Cumulative P applied less grain uptake
	2007/08	2008/09	2007/08	2008/09		2007/08	2008/09	
Control	0	0	43	53	-96	12	20	-32
4 t ha^{-1} sludge	90	137	60	78	89	18	21	188
8 t ha^{-1} sludge	179	274	92	99	262	28	40	385
16 t ha^{-1} sludge	358	549	108	125	674	35	53	819
Inorganic fertilizer	80	80	76	97	-13	30	60	70

5.3.5 Conclusions

The annual forage and grain production per unit area was much higher under an intensive irrigated maize-oat rotation than under dryland maize production. So was the crop nutrient uptake and therefore the corresponding sludge application rates. It was also evident that crop nutrient requirement was much higher for forage production than grain both under dryland and irrigated maize-oat rotation. Considering the negative potential environmental impacts associated with high sludge application rates and poor agronomic yields of low rates, it is vital to match crop nutrient demand with supply. Crop nutrient demand is a function of crop cultivar and weather variables especially temperature and water availability. Nitrogen availability from sewage sludge is also influenced by its composition, soil, and weather variables. Therefore, a dynamic reasoning support tool is needed to solve the complex interaction between factors involved in crop production, environmental impacts and sustainability. It was also evident from this and other studies that sludge application according to crop N demand could result in the build-up of P in the soil profile. Therefore, it is of utmost importance to set certain field monitoring mechanisms in agricultural lands receiving sludge, to assess the long term availability and mobility of P and possibly, heavy metals.

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CHAPTER 6

LOCAL FIELD SCALE STUDIES ON PERENIAL DRYLAND PASTURE UTILIZING MUNICIPAL SLUDGE AS NUTRIENT SOURCE

6.1 Introduction

It is well documented that sludge application improves dry matter yield of pasture grasses (Kelling *et al*., 1977; Soon *et al*., 1978; Kiemnec *et al*., 1987; Michalk *et al*., 1996; Sullivan *et al*., 1997; Zebarth *et al*., 2000; Cogger *et al*., 2001). This, however, is dependent on climate, management regime, and grass N requirement. For instance, tall fescue (*Festuca arundinacea Schreb.*) dry matter yield increased significantly with increase in sludge application up to rates of 20 t ha^{-1} (905 kg N ha^{-1}) in an area receiving 1020 mm rainfall supplemented with irrigation (Cogger *et al*., 2001). In other studies conducted by Sullivan *et al*. (1997), prairie grass (*Bromus uniloides (Willd)*) dry matter yield increased as sludge application rate increased to 26.9 t ha^{-1} (1064 kg N ha^{-1}) in a 1232 mm rainfall area. In contrast, Kelling *et al*. (1977) reported positive rye (*Secale cereale* L.) response only for sludge rates of up to 7.5 t ha^{-1} at Arlington, Wisconsin, USA.

Although high sludge application rates are reported to improve pasture grass yields, there have been serious concerns about the agronomic and environmental sustainability of such rates. Studies similar to those of Cogger *et al*. (2001), which improved tall fescue yield at sludge application rates of 20 t ha^{-1}, resulted in high accumulation of NO_3 in the soil profile. Similar findings were reported from earlier studies conducted by Soon *et al*. (1978) on a similar soil type planted to Brome grass. According to the latter study, nitrate accumulated in the top 0.9 m of the soil profile for soils that received 800 kg N ha^{-1} and more.

The accumulation of plant available P in the soil profile above that which crops could utilize has also been a major environmental concern associated with high sludge application rates. Michalk *et al*. (1996) report plant available P values exceeding 300 mg kg^{-1} in the top few cm of sludge amended soils. Such high concentrations could definitely pose a risk to surface water bodies through runoff in solution form (Sims and Pierzynski, 2000). Most of the studies on agricultural use of sludge have been conducted overseas under varying soil and climatic conditions. Direct extrapolation of those studies to the

South African condition could compromise both agricultural production and the environment.

The objectives of the study were:

- To investigate the dynamic nature of ideal sludge application, and
- To investigate the agronomic benefits and sustainability of applying municipal sludge according to crop N demand

6.2 Materials and methods

6.2.1 Field trial and treatments

The study was conducted in the same location as for the agronomic crop study described in Chapter 5. The treatments for the dryland pasture (weeping love grass) (Fig. 6.1) consisted of three sludge application rates (4, 8, and 16 t ha^{-1} yr^{-1}), a zero control, and an inorganic fertilizer treatment. The annual sludge application rate for all weeping love grass treatments was split into two, so that half was applied in the beginning of the season and the remaining half following the first cut. Nitrogen, phosphorus, and potassium were applied to the inorganic fertilizer treatments according to the Kynoch pasture handbook (Dickinson *et al*., 2004) for a target annual hay yield of 10 t ha^{-1}.

Both sludge and inorganic fertilizer were broadcast over the soil surface of already established weeping love grass plots. Unlike maize and oats, however, both sludge and inorganic fertilizers were left on the soil surface (not incorporated). Sludge application and hay cutting dates are presented in Table 6.1. The amount and timing of inorganic fertilizer applications are presented in Table 6.2 and the amount of sludge application in Table 6.3.

Figure 6.1 Perennial dryland pasture (weeping love grass) planted under three sludge application rates, an inorganic fertilizer treatment and a zero control (0 sludge and inorganic fertilizer) at ERWAT

Table 6.1 Sludge application and hay cutting dates for Weeping love grass during 2007/08 and 2008/09 growing seasons at ERWAT, Ekurhuleni district, South Africa.

Growing season	Sludge application date		Hay cutting date	
	First	Second	First	Second
2007/08	14/11/2007	20/02/2008	17/02/2008	22/05/2008
2008/09	01/11/2008	20/02/2009	31/01/2009	11/05/2009

Table 6.2 Fertilizer type and application time of a weeping love grass experiment during the 2007/08 and 2008/09 growing seasons at ERWAT, Ekurhuleni district, South Africa.

Time of fertilizer application	Fertilizer type	Amount of nutrient added		
		N	P	K
		kg ha^{-1}		
In spring	NPK 2:3:2(22%)	13	20	13
	Limestone ammonium nitrate (LAN) (28% N)	87		
	KCl (50% K)			27
After first cut	LAN	100		
	Super phosphate (18.5% P)		20	
	KCl			47
Cumulative annual application		**200**	**40**	**87**

Table 6.3 Sludge application rate for Weeping Love grass during 2007/08 and 2008/09 growing seasons at ERWAT, Ekurhuleni District, South Africa.

Treatment (sludge application rate t ha^{-1})	Sludge application rate (kg ha^{-1})			
	2007/08		2007/08	
	First cut	Second cut	First cut	Second cut
0	0	0	0	0
4	62	62	55	55
8	124	124	110	110
16	247.5	247.5	219	219

6.2.2 Plant sampling

Plant samples were taken on a monthly basis from an area of 0.5 m^2. A 0.5 m border was left around all sides of each plot during sampling. Additional plant samples were taken randomly twice during the growing season from each plot for N and P uptake determination. Above-ground samples for hay yield determination were collected 0.05 m above the soil surface at flowering from a 1 m^2 area. Additional samples were collected at harvest from each plot for grass N and P uptake determination.

6.2.3. Soil sampling

At the end of each growing season, three soil samples were collected diagonally from each plot using an auger. The soil samples were collected from the 0-0.1m, 0.1-0.3m, 0.3-0.5m, and 0.5-0.8m layers. The three samples from each layer of a plot were combined and mixed to make a single homogenous composite soil sample per layer.

6.3 Results and Discussion

6.3.1. Hay yield

Hay yield improved with increase in sludge application rate, with the highest yield being harvested from the 16 t ha^{-1} yr^{-1} treatment (Fig. 6.2). This was mainly attributed to the relatively higher plant N availability because N is considered the key element for dry matter production (Miles and Manson, 2000). It has also been well explained by Rethman *et al*. (1984) that dry matter production of weeping love grass increases with an increase in the availability of N.

Generally, hay yield was 23% higher on average during the relatively wet 2007/08 growing season (766.5 mm rain) than the relatively dry 2006/07 season (405 mm rain) (Tesfamariam, 2009). Similarly, the 2008/09 growing season (664.5 mm rain) produced 19% higher hay yield than the dry 2006/07 season, from the same study site, indicating the effect of water availability on hay production. It is also interesting to note that Weeping love grass hay yield from all treatments was much higher than the long-term average values reported by Dickinson *et al*. (2004) from deep Hutton soils receiving an average annual rainfall of 600 mm (6 t ha^{-1}). The high yield harvested from the current study is most probably due to a better nutrient status and relatively higher rainfall.

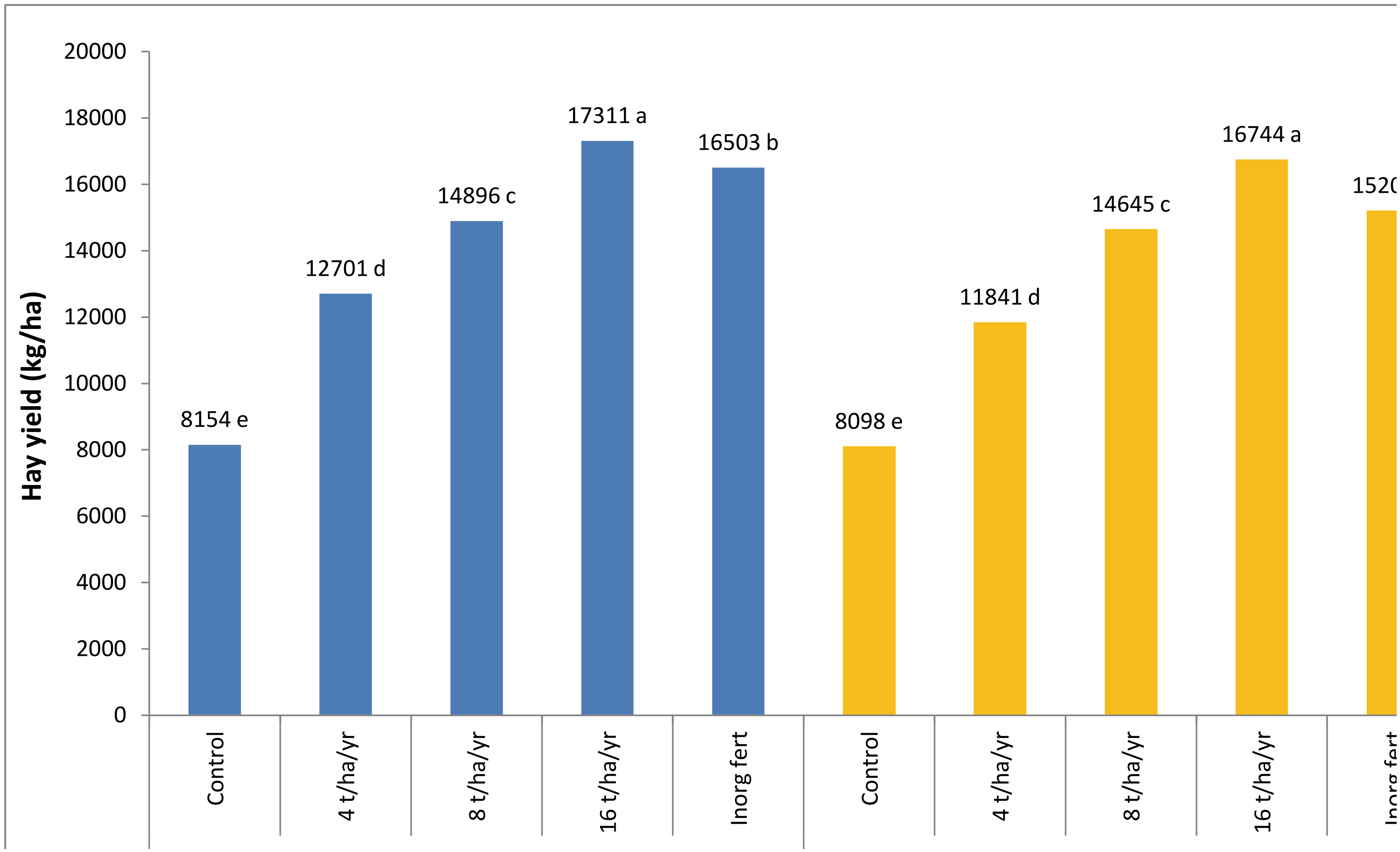

Figure 6.2 Weeping love grass hay yield during the 2007/08 and 2008/09 growing seasons (bar graphs of the same year having different letters indicate statistical significance)

6.3.2. Nitrogen mass balance

Weeping love grass hay N uptake was highest from the 16 t ha^{-1} sludge treatment (Table 6.4). The soil organic matter was able to supply 91 and 92 kg N ha^{-1} to the zero control treatment during the 2007/08 and 2008/09 growing seasons, respectively. This was twice higher than the average value reported by Van Biljon *et al*. (2008) for South African soils due to the reasons explained in Chapter 5.

The mean annual N uptake by the 16 t ha^{-1} sludge treatment during the 2007/08 and 2008/09 growing seasons was 18 and 16% higher than similar treatment during the dry 2006/07 growing season (Table 6.4, last column), respectively. This is mainly attributed to the low rainfall experienced during the 2006/07 growing season, which infers the effect of water availability on crop N uptake and therefore the sludge application rate. The low N content of sludge applied during the 2006/07 growing season (2.219%) relative to the 2007/08 (3.09%) and the 2008/09 (2.74%) seasons might have contributed to some extent to this difference in N uptake. Therefore, sludge application rates need to be adjusted according to the availability of water (rainfall) and should also take into account the amount of volatilization losses especially from surface applied sludge on perennial pastures.

Table 6.4 Seasonal N mass balance of weeping love grass planted on a clay loam soil treated with three sludge application rates, an inorganic fertilizer, and a zero control.

	N applied		Hay N uptake		Cumulative N applied less hay uptake	Hay N uptake
Treatment	2007/08	2008/09	2007/08	2008/09		2006/07
Control	0	0	91e	92e	-183	109
4 t ha^{-1} sludge	124	110	182d	157d	-105	160
8 t ha^{-1} sludge	248	219	210c	182c	75	188
16 t ha^{-1} sludge	495	438	245a	239a	449	202
Inorganic fertilizer	200	200	245b	214b	-59	189

Previous studies conducted by Adamsen and Sabey (1987) showed that 40.3% of the NH_4-N from surface applied sludge was lost as NH_3 during the first two weeks of its application. This was in contrast to a recorded loss of 0.35% from an incorporated sludge from the same study. Similarly, other studies have also shown that 51-127% of the initial ammonium could be lost in the form of ammonia gas from a surface applied anaerobically digested sludge (Henry *et al*., 1999). Other losses, which need to be taken into account to adjust sludge application rates, include denitrification and leaching losses. These complex combinations of factors suggest the necessity for a mechanistic reasoning support tool.

According to the two year cumulative average data (Table 6.5), a annual sludge application rate of 1.6 t ha^{-1} would suffice to satisfy the uptake by the 16 t ha^{-1} sludge treatment. The assumptions are, sludge P content of 2.8% (average value for the study period), and 60% plant availability of P (according to incubation trial – Chapter 4). It is also utmost importance to consider the contribution from the soil reserve, which in this case was 65% of the total P uptake by the 16 t ha^{-1} sludge treatment.

Table 6.5 Seasonal P mass balance of weeping love grass planted on a clay loam soil treated with three sludge application rates, an inorganic fertilizer, and a zero control.

Treatment	P applied		Hay P uptake		Cumulative P applied less hay uptake
	2008	2009	2008	2009	
Control	0	0	12	23	-35
4 t ha^{-1} sludge	90	137	20	30	177
8 t ha^{-1} sludge	179	274	23	27	403
16 t ha^{-1} sludge	358	549	26	29	852
Inorganic fertilizer	80	80	24	25	111

Phosphorus can be transported by runoff from a site in dissolved form or attached to soil particles and could end up in water bodies enhancing the rate of eutrophication (Carpenter *et al*., 1998). The rate of phosphorus loss (particulate or dissolved) from surface applied sludge through runoff, however, depends on the type of sludge (Elliott and O'connor, 2007). According to these authors, highest concentrations of dissolved and particulate P was reported in runoff collected from plots treated with sludge produced by biological phosphorus removal methods. The concentration of total and dissolved P in runoff from Al and Fe treated sludges, however, did not differ significantly from that of a zero control. Nevertheless, studies conducted by Tesfamariam (2009), show that the mobility of P from Fe treated sludges could be affected by the soil background concentration and application rates. Therefore, the long-term mobility characteristics of Fe and Al precipitated P in sludge-amended soils needs to be investigated under varying soil, climate and application rates.

6.4 Conclusions

Weeping love grass hay yield and N uptake increased significantly with increase in sludge application rate. Hay yield and N uptake, however, varied across seasons in response to the availability of water and to some extent to the variation in sludge N content. Considering the complex interaction between sludge N content, availability of water, and other unaccounted factors that affect N availability from sludge and crop response, it is crucial to have a mechanistic reasoning support tool both from agronomic and environmental perspectives. It is also at most importance to consider the status of plant available P in soils before sludge could be applied according to crop N requirements. Therefore, site, sludge (type and contents), and season specific application rates are essential for resilient use of sludge on dryland pasture production.

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CHAPTER 7

LOCAL FIELD SCALE STUDIES IN TURFGRASS SOD PRODUCTION PLANTED TO MUNICIPAL SLUDGE AMENDED SOILS

7.1 Introduction

The application of municipal sewage sludge with low contaminant levels (heavy metals and organic pollutants) to agricultural lands for crop production has been successfully deployed in many countries (Shepherd, 1996; Goldstein, 1997; Jarausch-Wehrheim *et al*., 1999; Kao *et al*., 2005). The nutrient content of sludge produced by municipal water treatment works often exceeds the requirements of nearby crops. Transporting sludge further afield is not always economically viable. In such situations, excess sludge could be exported through turfgrass sod production.

Turfgrass sod production provides the opportunity to export large volumes of sludge from limited areas because a layer of soil is removed with the harvested turf. The sludge removed along with the sod has the added advantage of being a slow-release nutrient source for the grass at the final establishment location (Muse *et al*., 1991). In addition, sludge application has the potential to minimize soil loss from the turfgrass production site, which on average is 1 cm depth per harvest (Charbonneau, 2003), by minimizing the thickness of soil exported with the sod.

The impact of composted sewage sludge on turf production has been well studied. Studies conducted by Cisar and Snyder (1992), Angle (1994), and Loschinkohl and Boehm (2001) show that the application of composted sludge improved turfgrass establishment rate and quality. Linde and Hepner (2005) and Cheng *et al*. (2007) also reported the benefits of incorporating composted sludge in improving soil water retention, bulk density, and cation exchange capacity. Other research includes turfgrass response to manure application and the quality of runoff water (Gaudreau *et al*., 2002; Vietor *et al*., 2004). However, little is known about turf growth and environmental consequences of surface application rates that are well in excess of plant nutrient requirements. Furthermore, the current South African sludge guideline has an upper limit of 10 t ha^{-1} yr^{-1} (Snyman and Herselman, 2006). According to this guideline, farming systems that can responsibly exceed this upper limit norm can be approved upon successful motivation.

The objectives of the study were to test the following four hypotheses: that high sludge surface application rates well above recommendations based on crop nutrient removal

- are possible without reducing turf growth and quality
- do not cause an accumulation of N and P below the active root zone (0.45m)
- can minimize soil loss through sod harvesting, and
- do not cause unacceptably high nitrate and salt leaching

7.2 Materials and methods

7.2.1. Field trial and treatments

The five treatments of the turfgrass experiment consisted of 0, 8, 33, 67, and 100 t ha^{-1} sludge per sod harvest (on oven-dry mass basis). The value of 8 t ha^{-1} represents the annual agricultural upper limit of the 1997 guideline (Water Research Commission, 1997). Rates of 33, 67, and 100 t ha^{-1} are equivalent to depths of 5, 10, and 15 mm sludge respectively, assuming a sludge bulk density of 666 kg m^{-3}.

Data presented in this report does not include from the 2007/08 and 2008/09 growing seasons due to technical problems encountered during data collection. It rather presents data collected successfully during the 2005 and 2006 growing seasons. On 3 March 2005, kikuyu (*Pennisetum clandestinum* Hochst. ex Chiov.) turfgrass was established vegetatively using sods which completely covered the soil surface. Sod was harvested on 15 June 2005, leaving 0.02-0.03 m wide ribbons of sod, 0.5 m apart, after which sludge was spread uniformly on the soil surface according to the treatments, whereafter the trial commenced. Turf then re-established itself vegetatively from these ribbons and the first trial sod harvest occurred 8 months later (14 February 2006). The same procedure was followed for the second season trial (second cutting), which started on 20 February 2006 and sods were again harvested on 29 October 2006.

7.2.2. Irrigation

Adjustable nozzle micro-sprayers (Hunter, California, USA) at the four corners of each plot were used to irrigate the turfgrass using municipal drinking water (Fig. 7.1). The micro-sprayers were adjusted to a 90^{o} spray pattern and had a delivery rate of 72 mm hr^{-1}. In the absence of rainfall, 10 mm of irrigation was applied every third day for the first

30 days after sludge application. During the rest of the growing season, until sod harvest, turf was irrigated twice a week (first irrigation 15 mm, followed three days later by irrigation sufficient to cause a Wetting Front Detector (WFD, www.fullstop.com.au) buried 0.3 m below the original pre-harvest soil surface to collect a water sample). A WFD is a funnel shaped instrument that converges the downward flow of water, producing saturation when the surrounding soil reaches 2-3 kPa suction (Stirzaker, 2003; Stirzaker and Hutchinson, 2005). It is essentially a passive lysimeter, which signals the passing of a wetting front and stores a soil water sample as the front passes. The WFDs were installed in all but the 67 t ha^{-1} treatment.

Figure 7.1 Microsprayers at corners of each plot used for irrigation

7.2.3. Mowing and plant sampling

Turf was cut with a reel mower (Protea Turf, Germiston, South Africa) to a height of 0.03 m each week (Fig. 7.2). Clippings were removed according to common practice in the area, which also assists in exporting additional nutrients from the field. Clipping samples were collected from a 0.5 m^2 area during each mowing for dry matter determination.

Additional clipping samples were taken during the season for N and P analyses. The plant samples were dried in a forced-air oven at 60^{o}C. After drying, plant samples were ground in a rotary mill (Arthur H. Thomas Co., Philadelphia, PA) to pass a 2 mm screen, whereafter samples were analyzed.

Figure 7.2 Real mower used to cut the grass on a weekly basis to a height of 0.03 m

7.2.4. Sod harvest

Sods were harvested to a depth of 0.02 m from all treatments when the slowest growing treatment had reached full cover (Fig. 7.3). A hand operated, light-weight, petrol powered sod cutter (Protea Turf, Germiston, South Africa) (Fig. 7.4), used by commercial sod growers in the area, was used. Each sod had a width of 0.5 m and length of 1 m. A measure of sod integrity was determined by picking up and holding each sod vertically from one of the narrow edges and reporting the percentage that remained intact. Sod samples of 0.25 m^2 were collected from each plot for dry-matter determination. Plant material was separated from the soil in the sod, oven dried, and weighed. The soil was dried separately for 24 hours at 105^{o}C and weighed. Additional sod samples were taken randomly, combined into one sample for each plot and separated into plant and soil components for N and P analyses. The plant samples were washed free of soil before analysis, using deionised water. The wash solution was combined with soil, and plant components were dried in a forced-air oven at 60^{o}C. After drying, plant samples were ground in a rotary mill (Arthur H. Thomas Co., Philadelphia, PA) to pass a 2 mm screen and soil samples pulverized, whereafter samples were analyzed.

Figure 7.3 Sods harvested to a soil depth of 0.02 m once the slowest treatment had reached full cover

7.2.5. Soil sampling

Soil samples were collected on 16 June 2005 before treatment application from all lawn plots using an auger. The samples were collected from the following layers: 0-0.15m, 0.15-0.45 m, 0.45-0.75 m, and 0.75-1.05 m. The samples from each layer of the turfgrass plots in the same block (replication) were combined and mixed to make a single composite soil sample per layer.

Figure 7.4 A hand operated, light-weight, petrol powered sod cutter

After two consecutive sludge applications and sod harvests, three soil samples were taken diagonally from each plot using an auger. The samples were collected from the same depths originally sampled. The three samples from each layer of a plot were combined and mixed to make a single homogenous soil sample per layer. The soil samples were dried and pulverized. The dried and ground soil samples were digested and analyzed for total N, total P, NO_3^-, NH_4^+, and Bray -1P.

7.2.6. Soil loss through sod harvest

Three 0.5 m long steel pegs of 0.010 m diameter were inserted vertically into each plot, with 0.05 m exposed above the surface to determine soil loss during sod harvest. Two pegs were located at adjacent corners of each plot, and the third in the middle of a plot. Each was marked with a single groove 0.04 m above the original soil surface to indicate the position at which to tie a length of string. After each sod harvest, a piece of string was stretched between the two adjacent corner pegs across the peg in the middle of the plot, creating a "V" shaped transect. Distance to soil surface readings were then taken every 0.7 m along this transect, using a vernier-calliper. The difference in soil surface elevation before and after each sod harvest was recorded as the depth of soil exported with the sod. This assumes negligible soil disturbance, as a light-weight sod cutter was used and there was no tillage after harvest.

7.2.7. Turfgrass establishment rate

Rate of turf establishment was evaluated 34 and 65 days after sludge application as percent basal coverage, using the 10 pin bridge method. The bridge had 4 legs supporting a 0.5 m long horizontal frame with 10 pins located every 0.05 m. The bridge was placed randomly five times in each plot and the number of pins touching grass counted. Visual ratings of turf colour were conducted according to a scale of 1 to 9, where 1 was straw brown and 9 represented dark green (Linde and Hepner, 2005), with a rating of 6 considered minimally acceptable (Morris, 2007). Visual evaluations were made monthly and averaged over the growing season.

7.3 Results and Discussion

7.3.1. Establishment rate

Establishment rate was estimated from mean percent basal cover. Sludge application rates above the 1997 South African upper limit norm (8 t ha^{-1}) significantly improved turf establishment rates (Table 7.1). The rapid rate of establishment in the high sludge application treatments can be attributed to the increase in the availability of essential nutrients, especially N and P and/or the low bulk density of the growing medium. Sludge application rates above agronomic limits, therefore, could have the advantage of increasing the number of sod harvests per season by speeding up the rate of turfgrass growth. This opens up an opportunity to export even more sludge.

7.3.2. Turfgrass colour

Turfgrass colour rating significantly improved with increase in sludge application rate (Table 7.1). Turfgrass which received sludge rates higher than the former South African upper limit norm exceeded the ‘acceptable’ colour ratings (7-9). The highest colour ratings were scored by the 67 and 100 t ha^{-1} treatments, and the lowest by the zero sludge treatment. The increase in colour rating observed with increase in sludge application rate was most likely as a result of the increase in plant available N, as reported by Bilgili and

Acikgoz (2005). Nitrate concentration data from the WFDs installed at 0.3 m (Fig. 7.5) also clearly illustrated that there was an increase in the availability of N with increase in sludge application rate during most of the growing season.

7.3.3. Sod integrity

The percentage of sods that did not break during handling increased as the sludge application rate increased to 33 t ha^{-1}, but decreased at higher rates (Table 7.1). The 0 t ha^{-1} sludge treatment produced heavy sods, up to 54 t ha^{-1} heavier than the treatment receiving 100 t ha^{-1} sludge. An increase in the sludge application rate to 33 t ha^{-1} reduced sod mass and the more vigorous turf was able to bind the soil/sludge mix effectively. However, as the sludge application rate increased to 67 and 100 t ha^{-1}, the sod became weaker and a greater proportion fell apart during handling. Nevertheless, sod integrity was still good at 67 t ha^{-1}, and was not significantly different from the 33 t ha^{-1} treatment in 2005 (93 vs. 90%). Although significant in 2006 (96 vs. 88%), the difference was only 8%. Highest percentages of strong intact sods were harvested from the 33 t ha^{-1} treatment, and the lowest from the 100 t ha^{-1} treatment for both seasons.

One may expect that the strategy of harvesting sods once the slowest growing treatments had reached full cover may also have contributed to differences in sod integrity among the treatments as a result of differences in maturity. However, it is interesting to note that the slowest growing treatments (0 and 8 t ha^{-1}) did not have the weakest sods, as the high fraction of soil in the sod matrix, increased their integrity relative to the 100 t ha^{-1} treatment, which had a very high fraction of sludge in the sod.

The growth study showed that establishment and colour continued to improve up to an application of 67 to 100 t ha^{-1}, well above the maximum recommended limit, but that sod integrity decreased after 33 t ha^{-1}.

Table 7.1 Kikuyu (Pennisetum clandestinum Hochst. ex Chiov.) turfgrass sod quality (establishment rates (% mean vegetative cover), visual colour ratings, and sod integrity) as affected by five sludge application rates during the 2005 and 2006 growing seasons at East Rand Water Care Works, Johannesburg, South Africa.

Sludge application	Days after sludge application				Seasonal mean visual colour rating†		Sod integrity (percent harvestable sod per unit area)	
	2005		2006					
Rate	34	65	34	65	2005	2006	2005	2006
	Mean vegetative cover, %				Colour ratings		%	
Control	34	58	36	68	4.0	3.8	72	71
8 t ha^{-1} per cut	41	74	43	75	4.8	5.3	72	73
33 t ha^{-1} per cut	56	88	59	87	7.0	7.3	93	96
67 t ha^{-1} per cut	63	95	68	95	8.5	8.5	90	88
100 t ha^{-1} per cut	71	98	71	98	8.5	9.0	58	62
LSD (5%)	6	5	6	6	0.7	0.7	7	7
CV‡, %	7	4	7	5	6	7	6	6

† Visually rated on a scale from 1 to 9 (1 = straw brown; 9 = dark green).

‡ Coefficient of variation

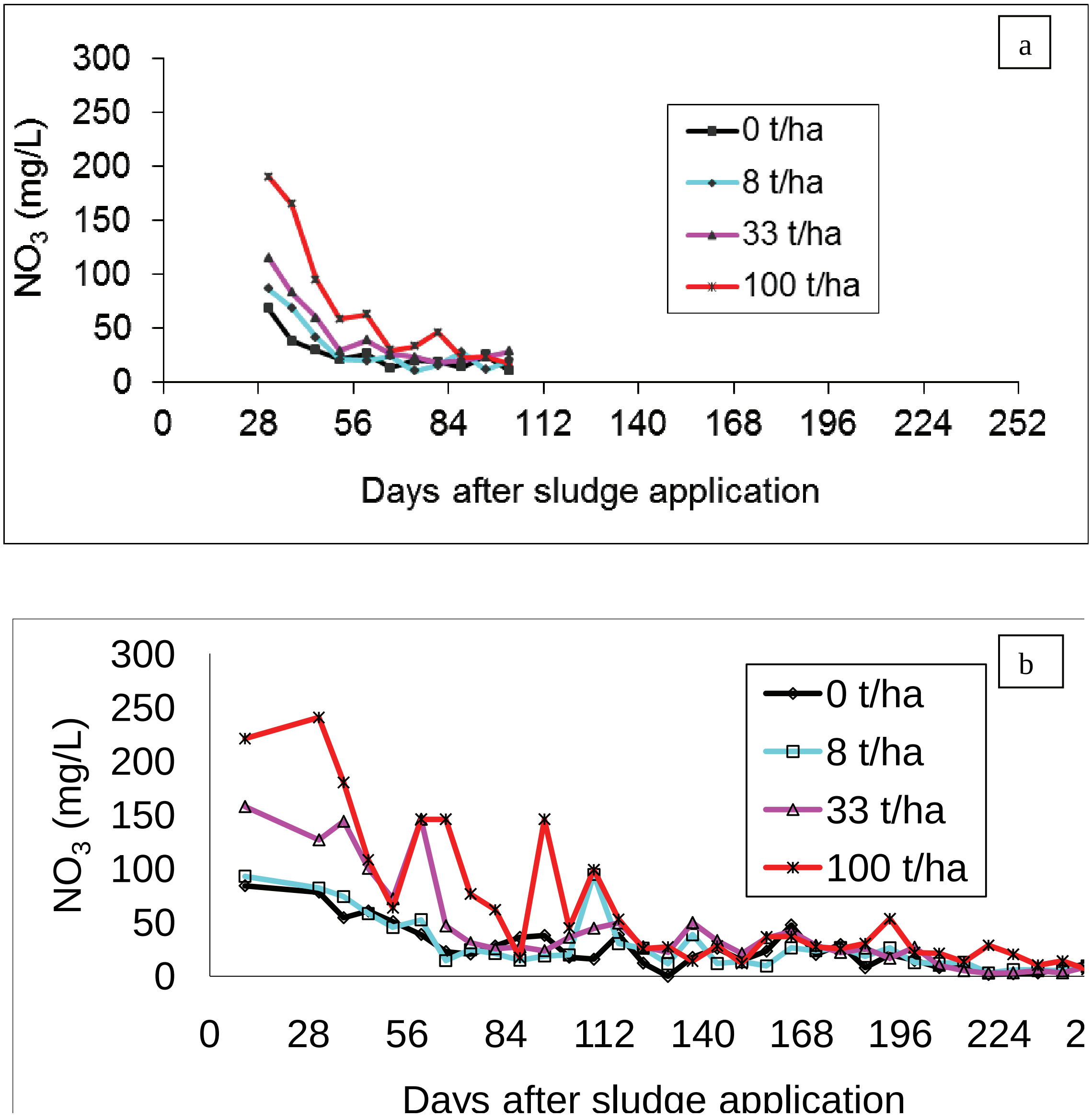


Figure 7.5 Concentration of nitrate in soil solution samples collected from wetting front detectors installed at 0.30 m under a turfgrass sod (*Pennisetum clandestinum*) trial for four sludge application rates (0 t ha^{-1}, 8 t ha^{-1}, 33 t ha^{-1}, and 100 t ha^{-1}) during (a) year 2005 and (b) 2006.

7.3.4. Accumulation of N in soil below the active root zone

The mass of N exported with the sod increased significantly with increase in sludge application rate (Table 7.2), with 86 to 91% in the substrate fraction of the sod. The amount of N in the plant component of the sod also increased significantly with increase in application rate, although plant N contributed only 10-15% of the total sod N. Similarly, Vietor *et al*. (2002) reported the export of a large N fraction in the soil component of the sod. Excess N exported with the sod can provide nutrition for turfgrass at the new site of establishment, where the original topsoil has often been removed due to construction. It was also evident that a significant amount of N was removed with clippings.

The total N content within the top metre of soil declined over the two year period in all treatments receiving less than 100 t ha^{-1} sludge (Table 7.3). Based on soil sampling over this depth (change in soil storage), sludge applications of around 67 t ha^{-1} were approximately in balance for nitrogen (Table 7.3). Generally a mass balance difference was observed in the top 1 m soil layer for all treatments. This could be attributed to leaching (Sierra *et al*., 2001; Samaras *et al*., 2008), volatilization (Beauchamp *et al*., 1978; Robinson and Polglase, 2000), and denitrification losses (Monnett *et al*., 1995). Considering the high leaching fraction experienced in this study, the largest loss was most likely through leaching. Sampling errors, N content variation within the sludge matrix, and soil heterogeneity may also have contributed to the differences.

Total soil N content decreased with depth for all treatments (Fig. 7.6a). After two years, total N of the top 0.15 m layer was less than the initial value of the same layer for all treatments. This could be due to grass uptake, leaching to lower layers, and the removal of a thin surface layer of soil during sod harvesting, which is rich in organic matter. Significant accumulation of N was evident in the 0.15-0.75 m layers of the 100 t ha^{-1} and the 0.45-0.75 m layers of the 67 t ha^{-1} treatments (Fig. 7.6a), despite the decline in the top 0.15 m layer. This indicates that N was leached to the lower layers because of the high leaching fraction experienced during the study period.

Soil nitrate concentration is a highly dynamic property, reflecting the size of the labile N pool and the antecedent conditions favouring mineralization and leaching. Sludge application rates lower than 33 t ha^{-1} showed a marked depletion in nitrate levels by the end of the study (Fig. 7.6b). Rates of 33 t ha^{-1} and above, maintained the nitrate concentration of the soil profile close to initial values, which as a benchmark, were within the ranges reported for golf courses from a lysimeter study (Wong *et al*., 1998).

Table 7.2. Total N imported with sludge, vs. exported with sods and clippings during the 2005 and 2006 growing seasons at East Rand Water Care Works, Johannesburg, South Africa.

	N imported with sludge vs. exported with sod and clippings									
	Imported		Exported							
			2005				2006			
			Sod				Sod			
Sludge application rate	2005	2006	Soil	Plant	Clippings	Total	Soil	Plant	Clippings	Total
t ha^{-1}	kg ha^{-1}									
0	0	0	400	66	174	640	297	37	154	488
8	243	151	466	77	268	811	358	47	255	660
33	1001	622	773	96	487	1356	655	68	419	1142
67	2031	1262	1050	141	620	1811	798	100	582	1480
100	3032	1884	1104	143	797	2044	1094	143	712	1949
LSD (5%)			70	10	30	78	56	4	30	61
CV‡, %			6	7	6	5	6	3	5	5

† Means in the same column followed by the same letter are not significantly different at $P < 0.05$ level.

‡ Coefficient of variation

Table 7.3 Total nitrogen and phosphorus mass balances after two years of sludge application and sod harvest for five sludge application rates during the 2005 and 2006 growing seasons

Sludge application rate	Nitrogen				Phosphorus			
	Imports	Exports	Change in soil storage†	Mass balance difference‡	Imports	Exports	Change in soil storage†	Mass balance difference‡
$t\ ha^{-1}$	$kg\ ha^{-1}$							
0	0	1128	-1295	-167	0	459	-465	-6
8	394	1471	-1318	-241	305	713	-452	-44
33	1623	2498	-1116	-241	1257	1389	-165	-33
67	3293	3291	-254	-256	2551	2349	175	-27
100	4916	3993	740	-183	3806	3479	291	-36

† Estimated by subtracting the initial soil profile N or P content before treatment application from the soil profile N or P content at the end of the two year study period (2006).

‡ Estimated by subtracting the change in soil storage from the (Imports – Exports).

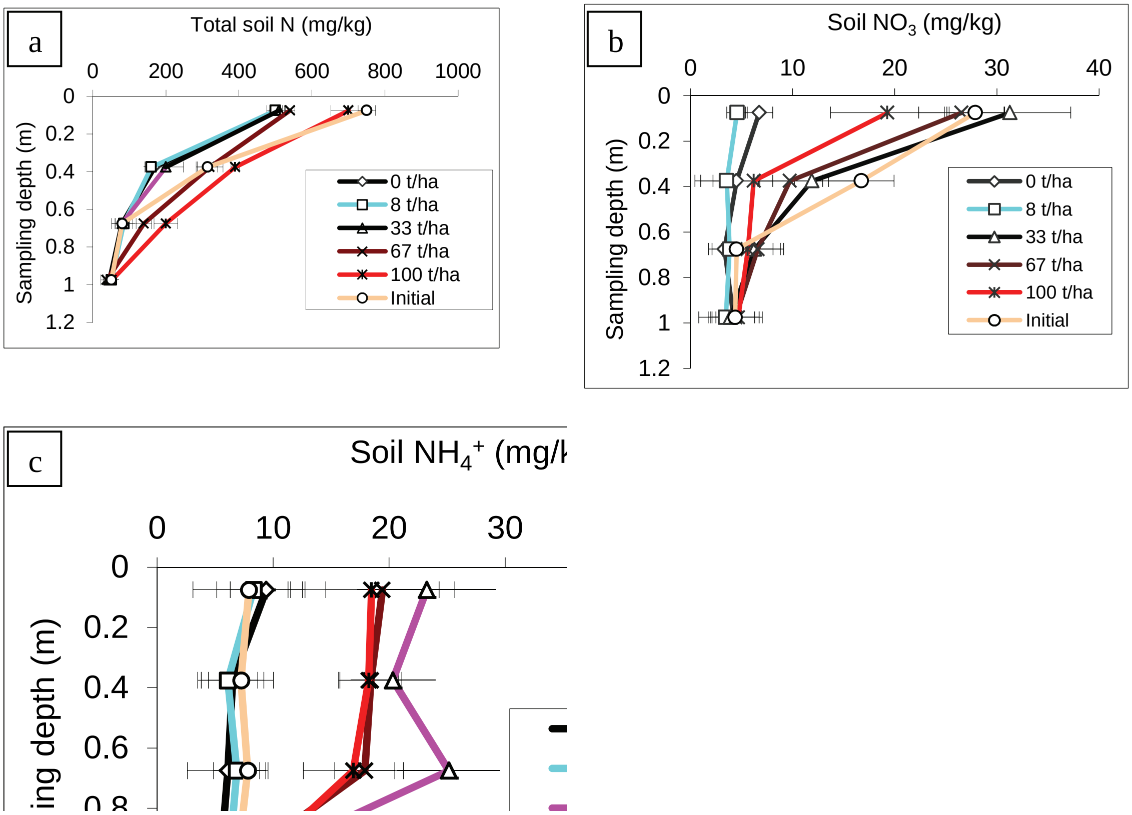


Figure 7.6 Soil profile (a) total N (b) nitrate, and (c) ammonium as affected by two consecutive years of sludge application at five rates (0, 8, 33, 67, and 100 t ha^{-1}) in a turfgrass sod (*Pennisetum clandestinum*) field trial, sampled before treatment application in 2005 (initial) and after two sod harvests in 2006.

The concentration of ammonium was elevated at applications greater than 33 t ha^{-1} (Fig. 7.6c). This was probably due to adsorption to CEC sites, as the soil had a high and fairly uniform clay content (34%-40%) to a depth of 1.2 m.

This study shows that single high application rates could pose detrimental environmental risks through nitrate and other solute leaching. The negative environmental impacts could even get worse under poor irrigation management practices. Therefore, the option of “once-off high sludge application” provided in Volume 4 of the current South African Sludge Guidelines should be revisited.

7.3.5. Phosphorus

The mass of P exported with the sod increased significantly as the sludge application rate increased (Table 7.4), with 93% to 99% in the substrate fraction of the sod. Phosphorus stored in the soil profile declined over the two year period in all treatments receiving less than 67 t ha^{-1} of sludge, based on soil sampling over the 1.0 m depth (Table 7.3). Based on this sampling depth (change in soil storage), sludge application rates of 33 to 67 t ha^{-1} were approximately in balance for phosphorus. However, the 67 t ha^{-1} treatment was associated with a net positive P accumulation (175 kg ha^{-1}) within two years. A mass balance difference was evident for all treatments and could most probably be from sampling errors, P content variation within the sludge matrix, and soil heterogeneity.

Soil profile total P (mg kg^{-1}) and Bray-1 extractable P (mg kg^{-1}) decreased with depth for all treatments (Figs. 7.7a and 7.7b). Sludge application rates higher than 33 t ha^{-1} resulted in the build-up of total P in the top 0.15 m soil layer. However, the concentrations of Bray-1 extractable P for all treatments were less than the initial soil profile concentration (before treatment application) (Fig. 7.7b). The build-up of total P in the top 0.15 m soil layer of the 67 and 100 t ha^{-1} rates may be from some sludge remaining after sod lifting or from preferential flow of particulate P (Jensen *et al*., 2000; Brock *et al*., 2007).

The Bray-1 extractable P concentrations of all treatments were below the optimum concentration of 25 to 30 mg kg^{-1} required for most crops (Sims and Pierzynski, 2000) and below the 'low sufficiency' range of 6-12 mg kg^{-1} for the common soil Bray-1P test reported by Havlin *et al*. (2005). The decline in Bray-1 extractable P from all sludge applications of less than 67 t ha^{-1} could most probably be due to plant uptake. For treatments receiving sludge rates higher than 67 t ha^{-1}, however, it might also be caused by the Fe added with the sludge. This is because the sludge used in this study has gone through tertiary treatment, and is rich in insoluble Fe-P compounds. Chemicals used in tertiary treatments such as Al or Fe salts, decrease the labile P fraction (Elliott *et al*., 2002; Häni, *et al*., 1981; Kyle and McClintock, 1995).

In summary, total N within the top metre of soil decreased over the duration of the trial for rates not exceeding 67 t ha^{-1}. Total P content within the same depth also declined for rates not exceeding 33 t ha^{-1}. Bray-1 extractable P, however, decreased for all treatments. Based on soil sampling over a depth of 1.0 m, sludge applications of around 67 t ha^{-1} approximately preserved N mass balance, but was associated with significant N leaching to the lower 0.45-0.75 m soil layer. The P mass balance was, however, preserved at application rates of 33 t ha^{-1}. Total P accumulation was evident mainly in the top 0.15 m of treatments which received sludge application rates higher than 33 t ha^{-1} under the prevailing management practices and climate.

Table 7.4 Total phosphorus imported with sludge, vs. exported with sods and clippings during the 2005 and 2006 growing seasons at East Rand Water Care Works, Johannesburg, South Africa.

Sludge application rate	P imported with sludge vs. exported with sod and clippings									
	Imported		Exported							
			2005				2006			
			Sod				Sod			
	2005	2006	Soil	Plant	Clippings	Total	Soil	Plant	Clippings	Total
t ha^{-1}	kg ha^{-1}									
0	0	0	229	16	13	258	170	19	12	201
8	157	148	358	18	21	397	275	21	20	316
33	648	609	661	18	48	727	598	22	42	662
67	1315	1236	1148	18	53	1219	1056	25	49	1130
100	1962	1844	1782	21	112	1915	1440	26	98	1564
LSD (5%)			108	1	3	108	59	1	3	59
CV†, %			6	5	4	4	6	5	6	5

† Coefficient of variation

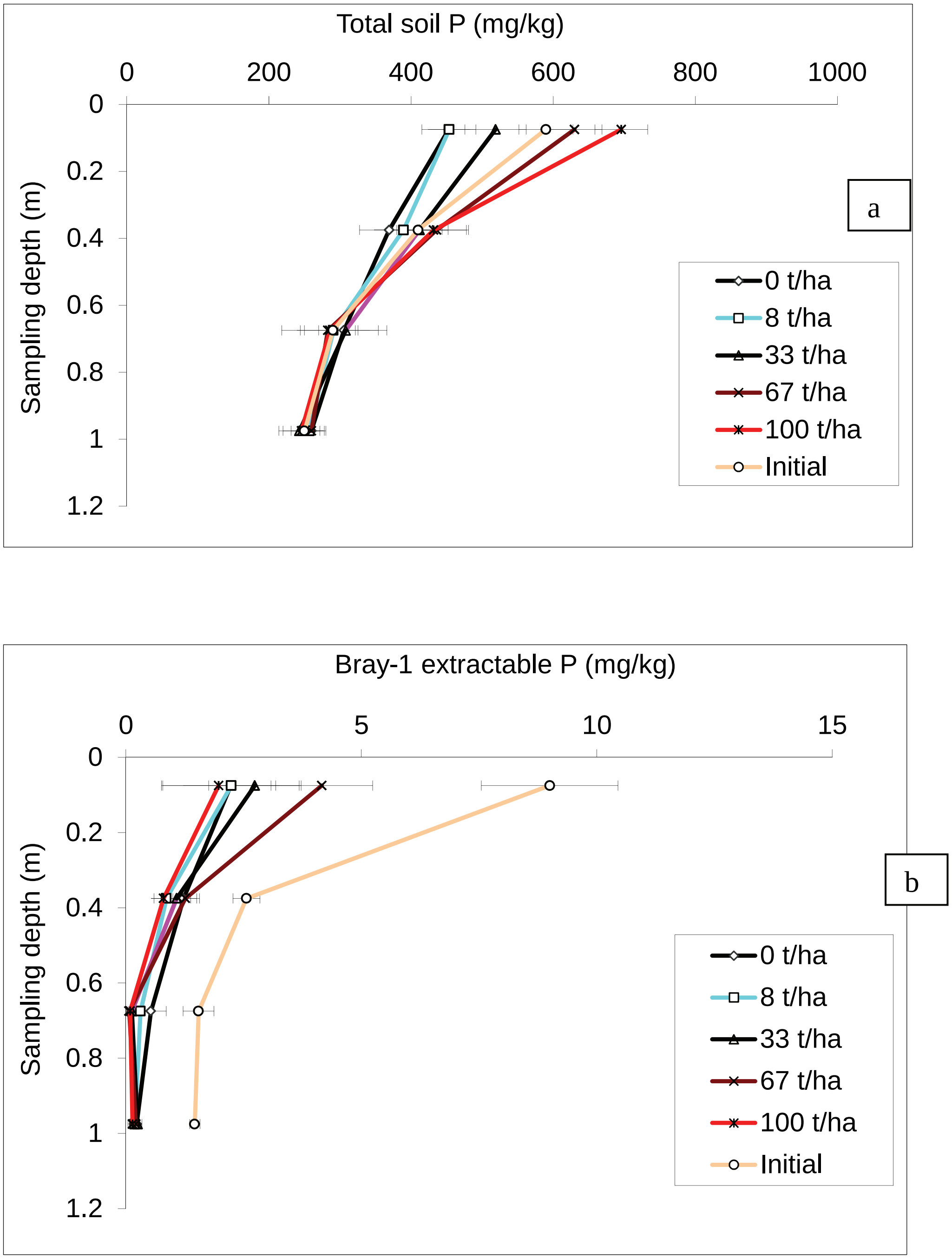


Figure 7.7 Soil profile (a) total P (b) Bray-1 extractable P as affected by two consecutive years of sludge application at five rates (0, 8, 33, 67, and 100 t ha^{-1}) in a turfgrass sod (*Pennisetum clandestinum*) field trial, sampled before treatment application in 2005 (initial) and after two sod harvests in 2006.

7.3.6. Soil loss through sod harvesting

Soil loss from the site was estimated from the soil depth exported with the sod. Results from this study show that severe soil loss takes place in turfgrass sod production areas where little or no amendment is applied (Table 7.5). The depth of soil exported from the site decreased significantly with increasing sludge application rates above 8 t ha^{-1}. The application of 33, 67, and 100 t ha^{-1} saved 120, 217, and 290 t ha^{-1} soil respectively from being exported from the site within two years (assuming a soil bulk density of 1380 kg m^{-3}). Considering the very slow average global rate of soil formation, 700 kg ha^{-1} yr^{-1}, (Wakatsuki and Rasyidin, 1992) the 33, 67, and 100 t ha^{-1} treatments saved soil from being exported which could have taken 86, 155, and 207 years to form, respectively.

Table 7.5. Sod mass and cumulative soil thickness exported with turfgrass sods as affected by five sludge application rates after two consecutive sludge application and sod harvest events at East Rand Water Care Works, Johannesburg, South Africa.

Sludge application rate	Sod mass		Cumulative Soil thickness exported (2005-2006)
	2005	2006	
	t ha^{-1}		mm
0	149	156	22.0
8	147	155	21.5
33	141	148	13.3
67	112	128	6.3
100	99	102	1.0
LSD (5%)	9	10	1.6
CV†, %	4	5	8

† Coefficient of variation

Sod mass decreased for every increment in sludge application rate because of the low bulk density of sludge (666 kg m^{-3}) compared with soil (Table 7.5). This also has a direct financial implication for sod transportation. Thus sludge applied above agronomic limits, will also reduce sod transportation costs.

In areas where sludge is surface applied for turfgrass sod production, the probability of heavy metal accumulation is expected to be significantly lower than those where sludge is incorporated into the soil. This is because the surface applied sludge is removed along with a thin layer of soil during sod harvest. This will, however, depend on the mobility of the elements as well as irrigation management practices. Therefore, the equations used to estimate the permissible application rate (PAR) in Volume 4 of the current South African Sludge Guidelines should be reconsidered under such circumstances.

In summary, the application of sludge did help to reduce the rate of soil loss through sod harvesting and should reduce transportation costs through lowering sod mass. Rates as high as 100 t ha^{-1} were required to largely avoid soil loss. However, such high application rates deleteriously affected sod integrity.

7.3.7. Nitrate and salt leaching

Soil solution nitrate and EC measurements were made on samples collected by the 0.3 m deep WFDs during the first three months after sludge application in 2005, and throughout the 2006 study period. Because the 67 t ha^{-1} treatment did not have WFDs, this treatment is excluded from the discussion that follows.

Nitrate leaching

In the beginning of the season, soil solution nitrate concentration increased with sludge application rate (Fig. 7.5). This is to be expected as nutrient supply far exceeds demand directly after sod harvest, as explained in detail by Geron *et al*. (1993). Later during the season, the concentration of nitrate remained at low levels (Fig. 7.5), presumably because the greater demand from the turf matched the mineralization rate from the sludge. Similar results and trends were recorded both during the 2005 (Fig. 7.5a) and 2006 (Fig. 7.5b) growing seasons.

Generally, the concentrations of nitrate in the soil solution from all treatments were within the ranges reported by Biró *et al*. (2005) for leachate from organic and conventionally managed horticultural lands (0-255 mg L^{-1} NO_3^-). It was also less than the maximum nitrate concentration in leachate from a simulated golf green (376 mg L^{-1} NO_3^-) reported by Shuman (2001).

The concentration of nitrate leachate from all treatments remained higher than South African drinking water standards (44 mg NO_3^- L^{-1}) (Korentajer, 1991) for the first two to three months in the 100 t ha^{-1} sludge treatment (Fig. 7.5). The wetting front detectors installed at a 0.6 m depth, however, did not collect soil solution samples from any of the treatments. Therefore, it was not clear whether the nitrate that passed the WFD at 0.3 m, had leached below the WFD at 0.6 m, perhaps in a weak front below the detection level of the WFD, or was stored between the two depths.

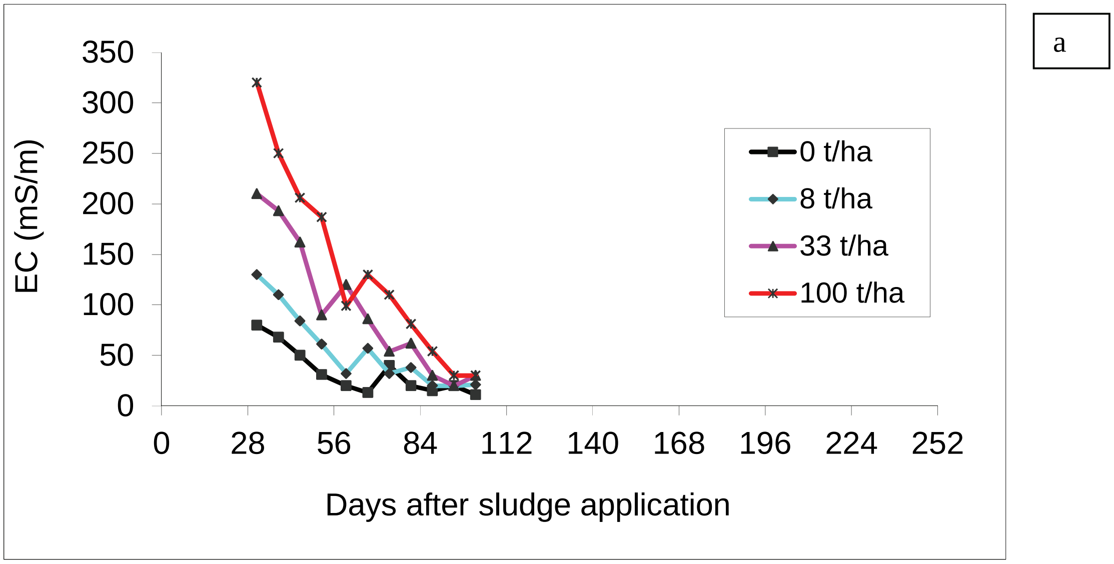
The seasonal average nitrate leachate concentrations for the 0, 8, and 33 t ha^{-1} sludge treatments (26, 29, and 43 mg NO_3^- L^{-1} respectively) were less than the South African drinking water standard (44 mg L^{-1}) (Korentajer, 1991) and the EU nitrate concentration limit for groundwater (50 mg NO_3^- L^{-1}) (Vlassak and Agenbag, 1999). The seasonal average nitrate concentration for the 100 t ha^{-1} treatment (63 mg NO_3^- L^{-1}), however, exceeded both limits. Compared with the zero sludge treatment, the addition of 8, 33, and 100 t ha^{-1} sludge increased the average seasonal nitrate concentration of the leachate by 3, 17 and 36 mg NO_3^- L^{-1} respectively.

Salt leaching

In the beginning of the season soil solution EC increased with sludge application rate (Fig. 7.8). The EC of soil solution samples from the 100 t ha^{-1} treatment collected 10 days after sludge application were only slightly higher than the published threshold value, with a 10% yield reduction predicted (300 mS m^{-1}) for kikuyu (Yiasoumi *et al*., 2005). Nevertheless, highest mean percent vegetative cover was recorded for this treatment (Table 7.1). Treatments receiving less than 100 t ha^{-1} sludge, however, had lower soil solution EC values than this threshold (Figs. 7.8a and b).

For soil solution samples from the 33 and 100 t ha^{-1} treatments, EC dropped very fast at the beginning of the season. This indicates that most of the salts added through the sludge were leached below the active root zone during the first 60 to 84 days after application. This was mainly because of the high leaching fraction (0.27 in 2004/05 and 0.3 in 2005/06) experienced during those periods from irrigation and rainfall.

An increase in soil salinity following sludge application is inevitable and was also observed in the studies conducted by Navas *et al*. (1998) and Stamatiadis *et al*. (1999). Nonetheless, the soil salinity levels of all the treatments at the end of the trial (Fig. 7.9) were much lower than the threshold value of 300 mS m^{-1} for kikuyu production (Yiasoumi *et al*., 2005). At the end of the trial, application rates higher than 33 t ha^{-1} significantly increased soil salinity, but the levels were still acceptably low (Fig 7.9).



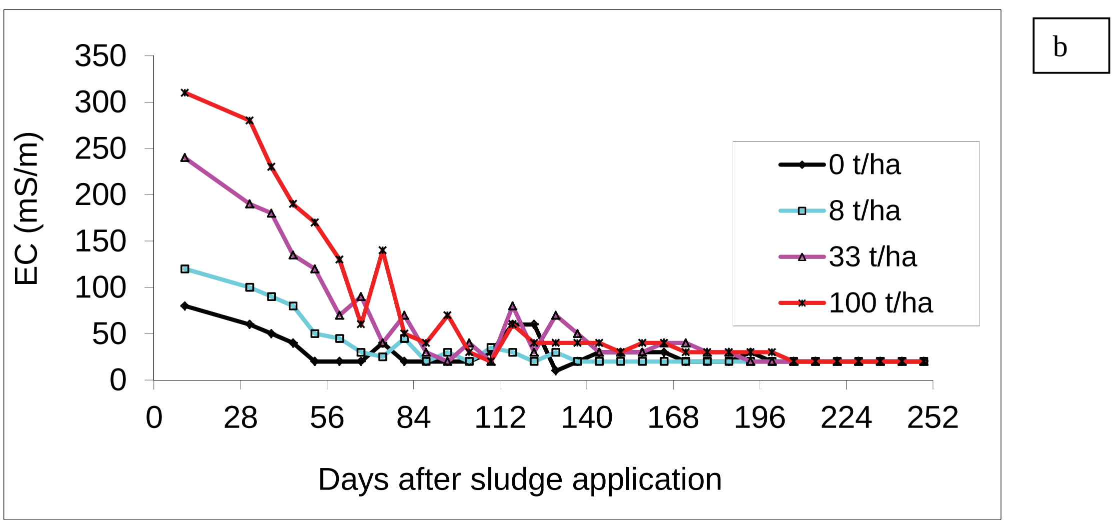


Figure 7.8 Electrical conductivity of soil solution samples collected from wetting front detectors installed at 0.30 m in a turfgrass sod (*Pennisetum clandestinum*) trial for four sludge application rates (0 t ha^{-1}, 8 t ha^{-1}, 33 t ha^{-1}, and 100 t ha^{-1}) during the (a) 2005 and (b) 2006 growing seasons.

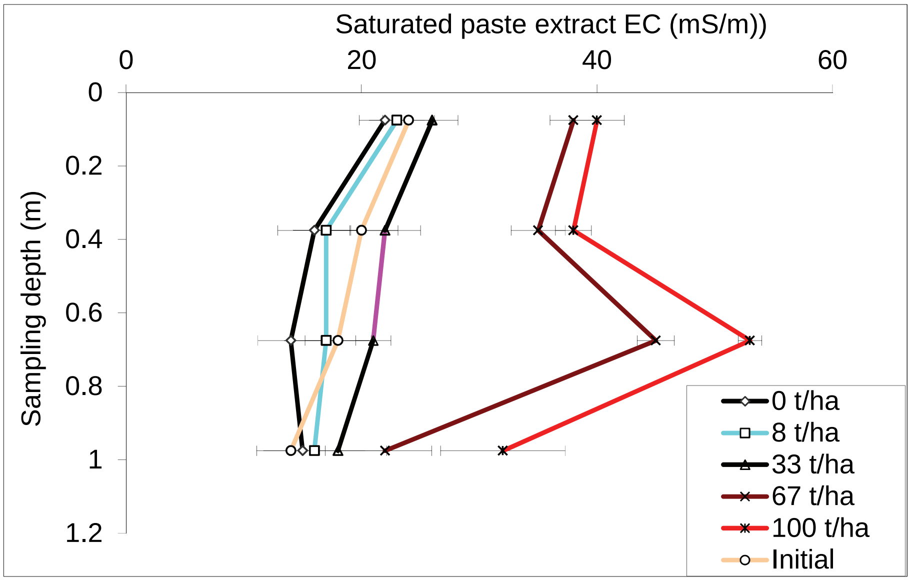


Figure 7.9 Soil profile electrical conductivity as affected by two consecutive years of sludge application at five rates (0, 8, 33, 67, and 100 t ha^{-1}) in a turfgrass sod (*Pennisetum clandestinum*) field trial, sampled before treatment application in 2005 (initial) and after two sod harvests in 2006.

In summary, the requirement to leach salts is the most difficult aspect of managing large volume sludge applications (Tesfamariam *et al*., 2009). Nitrate leaching could not be quantified in this study, but based on the solutions collected at 0.3 m depth and the change in soil N over the two seasons (Table 7.3), it would have been significant. Based on the EC of soil water and the growth studies, the leaching fraction could have been reduced with better irrigation management. It may also be necessary to apply the sludge in two applications, and delay the second application until low nitrate was measured in wetting front detectors. This high leaching fraction, however, could simulate worst case scenarios.

7.4. Conclusions

The study demonstrated several advantages of high volume sludge application to turf for sod production. Establishment rate and turf colour improved up to 67 t ha^{-1}. Very high applications of 100 t ha^{-1} had no deleterious effect on the turf growth, despite the salt content of the sludge. High applications also produced lighter sods, which could reduce transport costs, and minimize the loss of soil from the site. However, very high rates also drastically reduced sod integrity.

Soil sampling before and after the two year trial showed that N and P did not accumulate below a depth of 0.45 m until more than 33 t ha^{-1} sludge was applied. At the end of the study, available N and P were relatively low, and would not constitute a threat to groundwater in this specific soil type.

The disadvantages of excess sludge application are apparent when comparing the N and P mass balances with the actual change as measured by soil sampling. Clearly there are large losses of N and P which are hard to manage. In particular, the requirement to leach salt makes it difficult to control the leaching of nitrate, although in this study a lower leaching fraction would have been possible. Application rates higher than 67 t ha^{-1} would be unacceptable on environmental grounds, but they were also unacceptable for reasons of turf quality, as the sods tended to break apart during handling.

An application rate of 33 t ha^{-1}, four times higher than the 1997 South African recommended upper limit, or more than three times the new guideline, would be an acceptable compromise. This rate improved growth and sod quality over the zero and 8 t ha^{-1} application treatments. Great care would be required to minimize nitrate leaching during the first two months using proper irrigation management practices. Leaching could be managed by allowing the EC of leachate captured by the wetting front detectors not to exceed the threshold value of 300 mS m^{-1}. In other words, the crop coefficient could be reduced, and only increased when high salt levels were recorded. Applications greater than 33 t ha^{-1} may be possible if applications are split by delaying the second application until low nitrate is measured in wetting front detectors, as long as turf quality is not negatively affected. The implications of findings from this study on Volume 4 of the current South African Sludge Guidelines are that the option of a "once-off high sludge application rate" should be reconsidered due to the potential environmental impacts from nitrate leaching. In addition, the equation used to estimate the permissible application rate (PAR) in surface applied sludge for turfgrass sod production needs revisiting.

Therefore, all hypotheses were accepted for application rates not exceeding 33 t ha^{-1}, on the proviso that some soil loss was acceptable and that the leaching fraction was carefully managed during the first two months after sludge application.

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CHAPTER 8

DEVELOPMENT OF NITROGEN AND PHOSPHORUS MODELLING CAPABILITIES IN SWB-SCI

8.1 Introduction

To enable the simulation of the nutrient balances associated with sludge applications to land in South Africa, the locally developed SWB model was selected for inclusion of nitrogen (N) and phosphorus (P) subroutines. SWB is a generic, mechanistic crop growth model that was first developed in the early 90s and has since been well calibrated and validated with independent data sets (Annandale *et al*. 1999a, 1999b). In 2001 the chemical equilibrium model of Robbins (1991) was included into SWB. This has also been validated and is successfully being used by students and consultants to model salt movement and gypsum precipitation in soils.

This decision to include N and P was made despite the existence of models such as EPIC which simulate similar processes (Williams *et al*. 1984), because such models often have limitations due to the simplicity of their crop growth and biophysical processes (Stöckle *et al*. 2003) and there is a lack of opportunity to improve processes and algorithms contained in these models as they are not 'in-house'.

N and P simulation approaches and algorithms were obtained primarily from CropSyst (Cropping Systems Simulation Model) (Stöckle *et al*. 2003) for N, and GLEAMS (Groundwater Leaching Effects of Agricultural Management Systems) (Muller and Gregory, 2003) for P. SWAT (Soil Water Assessment Tool) (Neitsche *et al*. 2002) was also consulted to a limited extent. CropSyst was developed by C. Stöckle and R. Nelson from Washington State University and M. Donatelli from ISCI, Italy. It is described by Stöckle *et al*. (2003) as a multi-year, multi-crop, daily time-step crop simulation model, and was designed to draw from the conceptual strengths of EPIC (Erosion Productivity Impact Calculator) but with a more process orientated approach. GLEAMS was developed by S. Muller and J. Gregory at the University of Florida. The model is based on CREAMS, which was developed by the U.S. Department of Agriculture's Research Service to evaluate agricultural non-point source pollution from field-scale catchment areas. CropSyst and GLEAMS are written in the Visual Basic and Fortran programming languages, respectively, while SWB is written in Delphi.

The N subroutines of CropSyst were selected because both models have similar backgrounds and approaches, having both grown out of work done by Prof. Gaylon Campbell from Washington State University. In addition, CropSyst partitions organic matter into three pools: fast, slow, and lignified pools, which is ideal for sludge characterisation, because, according to Smith *et al.* (1997), sludges have at least three organic pools.

SWB continues to be improved with growing needs and most recently an independent weather database and quick set-up simulation capabilities have been included in the model.

8.2 Model Description

8.2.1 Model interface

Five new interface windows have been included in SWB-Sci model as various additional inputs are required for SWB to simulate N and P processes at local scale. Additional inputs, together with how these are used in processes in the model are discussed below.

a. Nutrient initialization

Inorganic N and P soil concentrations can be entered into the 'Initial N & P' interface (Fig. 8.1). Organic N and P values are calculated from the 'OM%' (organic matter percentage) value using C:N and C:P ratios for the various OM pools. If certain 'Initial N & P' inputs are not entered they will be estimated by the model using the OM% of the soil. Algorithms to estimate initial soil nutrients are taken from GLEAMS as well as from SWAT. This will be helpful to users who do not have all the input values. Details of the equation used can be found in Van der Laan (2009).

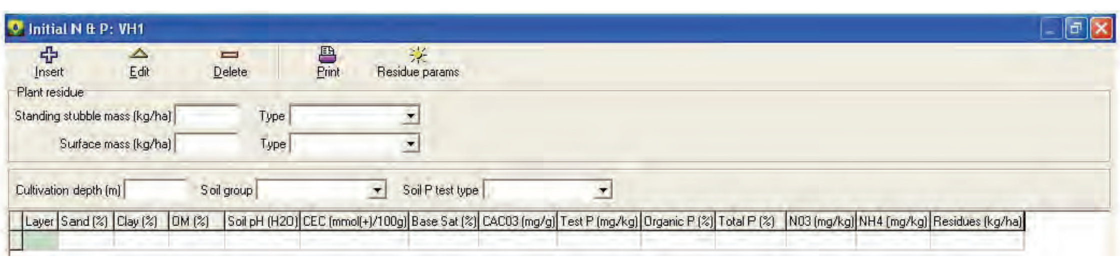

Figure 8. 1 Nutrient initialization input screen

b. Fertilization

The model accounts for both organic and inorganic fertilizer applications (Fig. 8.2). A wide range of pre-defined organic and inorganic fertilizers with respective N and P concentrations are provided.

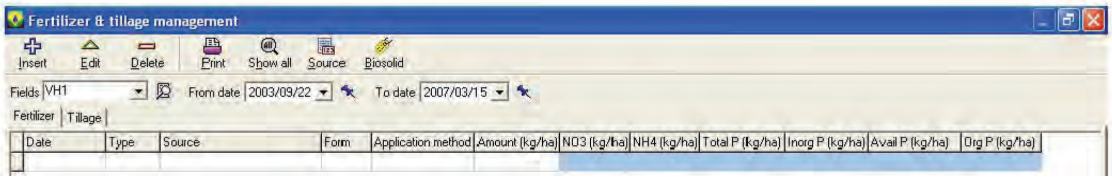

Figure 8. 2 Fertilizer management input screen

If a pre-defined fertilizer is selected (Fig. 8.3), the user is only required to enter the amount being applied and application method. Users are also able to specify user defined values for the fertilizer being applied. In some cases, values can be entered as concentrations and the model will convert to kg ha^{-1} to increase user-friendliness. Fertilizer can be either 'Broadcast' or 'Incorporated' under 'Application method'. When inorganic fertilizer is broadcast, the inorganic N and P remains on the soil surface until a rainfall/irrigation event after which it is added to the surface layer.

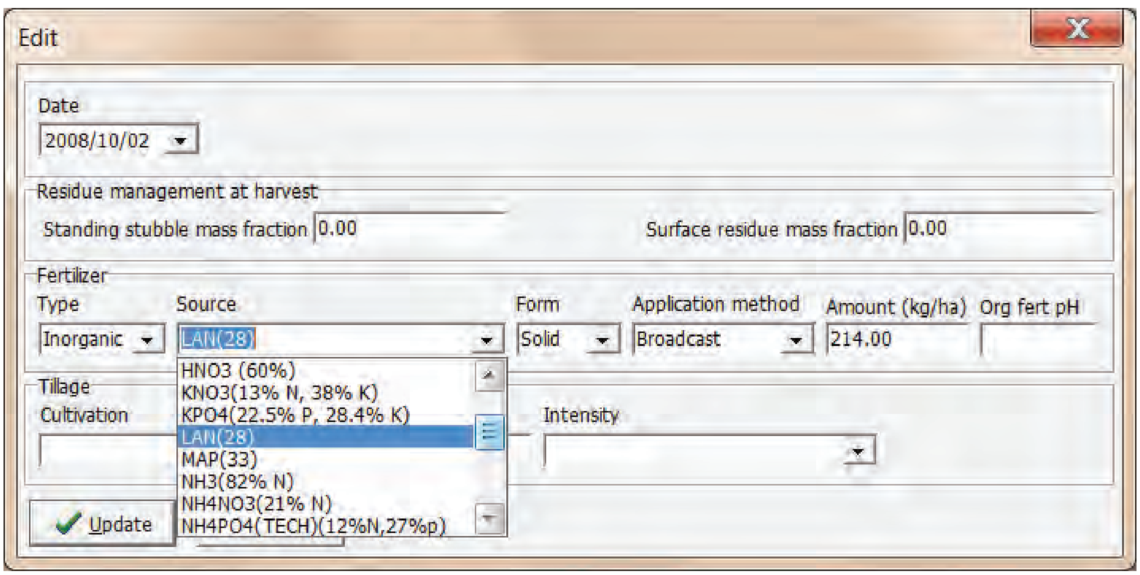

Figure 8. 3 Pre-defined fertilizer selection screen

Regarding sludge applications, it is advisable to parameterise for each batch used. This is because both the N and P concentrations, as well as the fractions of the different pools, vary from batch to batch. The parameters which need to be analysed include: fraction of each pool (fast-, slow-cycling, and lignified) and their corresponding half-life values in days, C fraction of the sludge, C:N ratio, ammonium content, water content, and the area to mass ratio as presented in Fig. 8.4.

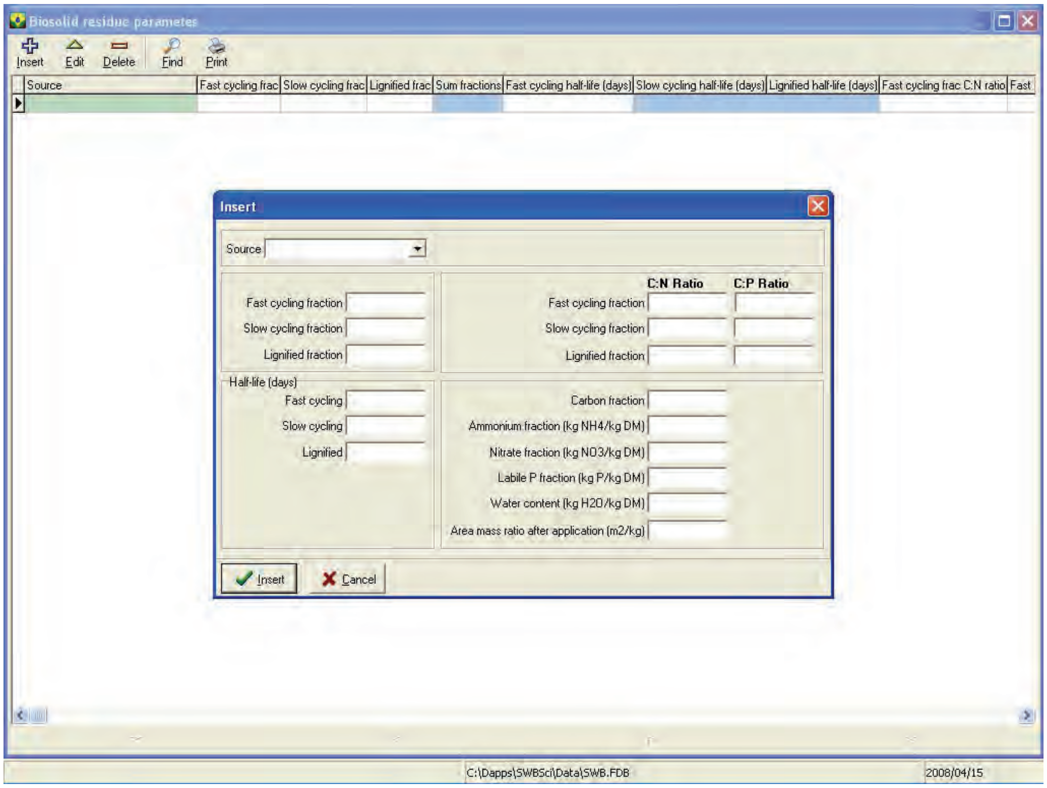

Figure 8. 4 Sludge and other organic N and P sources parameterisation screen

c. Addition of N and P through rainfall/irrigation

The model also accounts for N and P additions through rainfall and irrigation (Fig. 8.5). This is done by entering the concentrations of N and P in rainfall/irrigation. Different concentrations can be entered for each rainfall/irrigation event, otherwise the model will use the most recent concentration entered. This method can be used to account for fertigation as well. Runoff and leaching losses of nutrients can occur as a result of over-irrigation, or excessive rainfall.

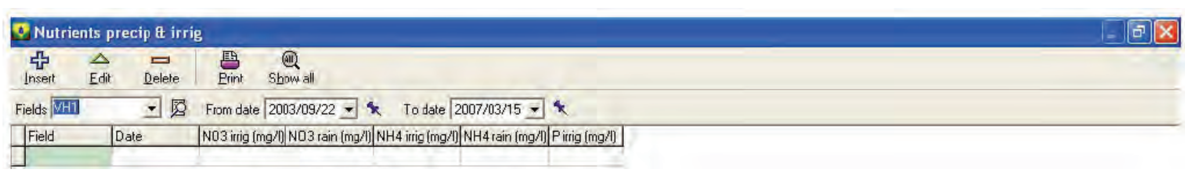

Figure 8. 5 Rainfall and irrigation nutrient additions input screen

d. Tillage management

Tillage is simulated using GLEAMS' algorithms. Different tillage implements are assigned with different incorporation and mixing efficiencies, which determine the incorporation of residues and manure into the soil and the redistribution of the various organic matter pools and inorganic N and P pools between the soil layers. Burning events can also be simulated. In the event of burning, 95% of N and 5% of P is removed from the surface (Du Preez, 2006 – personal communication). This is a modification of the GLEAMS model, which removes 95% of both N and P. These values might need to be re-visited in future. User defined inputs for incorporation and mixing efficiencies are permitted, and depth of tillage must be specified. Incorporation and mixing efficiencies of various tillage implements are included as an inputs (Appendix b).

8.2.2 Processes simulated

a. Mineralization

Mineralization is the transformation of N from its organic state to the inorganic forms of NH_4^+ or NH_3 by heterotrophic soil organisms consuming nitrogenous organic substances as a source of energy (Lutz, 1965; Jansson and Persson, 1982). The release of ammonium during mineralization is driven by the need of micro-organisms for carbon (Sprent, 1987).

Mineralization in SWB-Sci closely follows the methods used by CropSyst. Soil organic matter (SOM) is divided into microbial biomass, labile SOM, metastable SOM, and passive SOM. Each pool has its own decomposition constant. The C fraction in all organic matter pools has a constant value of 0.58. Both plant residue and biosolid are divided into three pools: fast-cycling, slow-cycling and lignified material. Each pool has its own C to CO_2 fraction which is hard-coded.

For standing and surface stubble crop residue, a contact fraction is used to account for surface residue contact with the soil during decomposition. Potential C decomposition of all organic matter pools is calculated as follows:

$$\begin{aligned} \text{PotCarbonDecomp} &= \text{CarbonMassInorganic\_Residue} \times \text{ContactFract} \\ &\times (1 - \text{Exp}(-\text{DecompConstant} \times \text{TempFunction})) \times \text{MoistFunction} \end{aligned} \tag{1}$$

b. Immobilization

Immobilization is the net incorporation of inorganic nitrogen, mainly NH_4^+ or NO_3^-, into organic forms (microbial tissue) during the decomposition process (Vinten and Smith, 1993).

It is a process which works in the opposite direction to mineralization. The decomposition of residue with a high C:N ratio results in a negative net residue nitrogen mineralization referred to as immobilization. This, however, changes in due time with the growth and stabilization of the microbial population resulting in a decline of the C:N ratio and the release of inorganic nitrogen. Most models set a C:N ratio beyond which immobilization takes place. This ratio is set to 24 by CropSyst.

When calculating immobilization, net N mineralization is calculated first. If N mineralization does take place from a pool, then the N immobilization demand is assumed to be zero. If the calculated mineralization is below zero, however, then the absolute value of this amount becomes the N immobilization demand and net N mineralization is set to zero. This is done for each SOM pool and accumulated to form a total N immobilization demand. N immobilization firstly takes place from the NH_4^+ pool. If there is not enough NH_4^+ to satisfy the total immobilization demand, N from the NO_3 pool will also be immobilized. If there is not enough N from both pools to satisfy demand, this deficit will carry over to the next day. This deficit will further contribute to decreasing the decomposition rate through its effect on the decomposition reduction factor which is calculated as follows:

$$\text{DecompReducFact} = (\text{NImmobDemand} - \text{DeficitForImmob})/\text{NImmobDemand} \quad .. \quad (2)$$

Total cumulative C lost in the form of CO_2 is accounted for.

Phosphorus mineralization and immobilization

As CropSyst does not currently simulate SOM mineralization of P, new code was written to accomplish this. C:P ratios of the various organic matter pools are used to obtain the quantity of P mineralized directly from the amount of C mineralized from SOM (Eq. 1). In the same way, P mineralization from crop residue is obtained from N mineralization quantities using crop N:P ratios. The calculation of P immobilization by the microbial biomass is calculated following the same strategy.

c. Nitrogen transformation processes

i. Nitrogen volatilization

Ammonia volatilization is the gaseous loss of NH_3 from the soil surface to the atmosphere (Haynes and Sherlock, 1986). Volatilization takes place as a result of the difference in concentration gradient between ammonia in solution and the ambient air (Freney *et al*.,

1983) and can be very large. Whether the applied NH_4^+ fertilizer is broadcast or incorporated has a primary role in the amount of volatilization that takes place. Soil pH and cation exchange capacity (CEC) further influence the fraction of applied ammonium fertilizer which is available for volatilization. A turbulent transfer coefficient value is calculated making use of wind speed at 2 m and soil, residue and/or crop friction velocities, as well as the leaf area index (LAI) of the crop. The algorithm for volatilization is too long to present. It is available in Tesfamariam (2009).

ii. Nitrification

Nitrification is the oxidation of the ammonium ion or ammonia to nitrate or nitrite (Delwiche, 1981). Cropsyst simulates nitrification only in the presence of ammonium in the soil profile. In cases where there is ammonium, the model further checks for the nitrate to ammonium ratio of the soil. If the soil nitrate to ammonium ratio is less than 8, the model assumes no nitrification. However, if the nitrate to ammonium ratio exceeds 8, the model uses Eq. 3 to estimate the amount of nitrification.

$$N_{nitrified} = (NH_4 – NO_3 / 8) * (1 - \exp(-0.2 * pHF * TF)) * WF \qquad (3)$$

Where

$N_{nitrified}$ – is the mass of nitrified N

NH_4 – is the mass of ammonium N

NO_3 – is the mass of nitrate N

TF – is the soil temperature function (Eq. 4)

WF – is the soil water function (Eqs. 5-7)

pHF – is the soil pH function (estimated using Eq. 9)

$$TF = ((T – T_{min}) \wedge (Q) * (T_{max} – T)) / ((T_{opt} – T_{min}) \wedge (Q) * (T_{max} – T_{opt})) \qquad (4)$$

Where

T – is real time soil temperature

T_{min} – minimum temperature below which there is no microbial activity (-5 ^oC)

T_{max} – maximum temperature above which there is no microbial activity (50 ^oC)

T_{opt} – optimum temperature for microbial activity (35 ^oC)

Q – is constant (2) estimated using $((T_{min} – T_{opt}) / (T_{opt} – T_{max}))$

$$\text{If } 0.1 \le WFP < 0.5 \qquad WF = ((WFP – WFP_{min}) / (WFP_{low} – WFP_{min})) \qquad (5)$$

$$\text{If } 0.5 \le WFP \le 0.7 \qquad WF = 1 \qquad (6)$$

$$\text{If } 0.7 \le WFP \le 1 \qquad WF = WF_{sat} + (1- WF_{sat}) * ((1 - WFP) / (1-WFP_{high})) \wedge 2 \qquad (7)$$

Where

WFP – is the water filled porosity (Eq. 8)

WFP_{min} – is the least amount of water filled porosity (0.1)

WFP_{low} – is the low water filled porosity (0.5)

WFP_{high} – is the highest level of water filled porosity (0.7)

WF_{sat} – is moisture function value at saturation (0.6)

$$WFP = WC/Saturation\ WC \tag{8}$$

$$pHF = (pH - pH_{min}) / (pH_{max} - pH_{min}) \tag{9}$$

where

pHF – is pH function

pH – is measured soil pH

pH_{max} – is the maximum temperature for nitrification (6.5)

pH_{min} – is the minimum temperature for nitrification (3.5)

iii. Denitrification

Denitrification is the gaseous loss of nitrogen in the form of nitrous oxide (N_2O) and dinitrogen (N_2), mediated through microbial action (Delwiche, 1981; Tisdale *et al*., 1985). According to Ma and Shaffer (2001), the simulation of denitrification by models is empirical due to both the unknown nature of the process and the spatial and temporal variability of anaerobic conditions in the soil.

When estimating denitrification, Cropsyst first estimates a respiration response function (RRF) (algorithm too long to present) based on the soil temperature function (TF) Eq. 4. The model then puts forward two preconditions which should be fulfilled if denitrification is to take place. These preconditions are: primarily there must be nitrate_N in the soil profile and, secondly, the denitrification soil water function (DWF) (Eqs. 10-12) must exceed zero. Finally, the denitrified N mass is computed using Eq. 13.

$$\text{If } WFP < WFP_{NR} \qquad DWF = 0 \tag{10}$$

$$\text{If } WFP_{NR} < WFP < WFP_{infl} \qquad DWF = 0.9 - 0.9 * (WFP_{infl} - WFP) / (WFP_{infl} - WFP_{NR}) \tag{11}$$

$$\text{If } WFP_{infl} < WFP \qquad DWF = 1 - 0.1 * (1 - WFP) / (1 - WFP_{infl}) \tag{12}$$

Where

WFP – water filled porosity

WFP_{NR} – water filled porosity without response to denitrification (0.6)

DWF – denitrification soil water function

WFP_{inf} – water filled porosity where inflection is observed in denitrification (0.8)

$$N_{denitrified} = k * soil_{mass} * \text{minimum}(DWF, RRF, (NO_{3\text{-}dry}/(NO_{3\text{-}dry}+c)) \quad (13)$$

Where

$N_{denitrified}$ – denitrified N mass

k – potential denitrification constant (0.000032)

$soil_{mass}$ – dry soil mass of the profile (layer)

RRF – respiratory response function

$NO_{3\text{-}dry}$ – nitrate concentration of the dry soil

c – denitrification half rate (0.00006 kg N / kg Soil)

iv. Nitrogen fixation

Certain crops are able to fix N and this capability has been included in the model, based on the approach by Bouniols *et al.* (1991). Daily N fixation is calculated as follows:

Nitrogen fixation = Minimum (Crop N demand*N fixation factor, Min daily N fixation mass) (14)

Where Minimum daily N fixation mass = 6kg N ha^{-1} d^{-1}

N fixation factor is the minimum of the following factors:

N fixation temperature factor:

1 for air temperatures >36^{o}C

0.7 for temperatures between 0-36^{o}C

0 for temperatures <0^{o}C

Soil N factor: 0 for root zone N masses > 300 kg ha^{-1}

1 for root zone N masses < 100 kg ha^{-1}

1-[(Root zone N mass – 100)/300] for root zone N masses between 100-300 kg ha^{-1} (15)

N fix moisture function (MF) = (*PAW top 30* – 0.5)/0.5 (16)

Where *PAW top 30,* is the plant available water in the top 30 cm of the soil profile.

For crops that are able to fix N, the N demand of the crop is reduced by an amount that can be supplied by N_2-fixing bacteria i.e. after crop already taken up N fixed.

v. Crop nitrogen uptake

Nitrogen uptake is based on CropSyst algorithms which follow the approach of Godwin and Jones (1991). N uptake is determined by first calculating reference plant N concentration and crop N demand, followed by actual crop uptake. Before this, however, total potential N and P uptake is calculated according to the amount of available N and P in the soil, and uses adsorption coefficients of 0, 5.6 and 60 for NO_3, NH_4 and labile P, respectively. For the calculation of reference plant N concentration, different critical, minimum and maximum N concentration parameters are hard-coded for C3 and C4 plants. Finally N uptake is determined from the minimum of crop N demand and potential N uptake. When N supply does not meet crop N requirement, crop growth is reduced using an N-limited growth factor.

vi. Crop phosphorus uptake

Phosphorus uptake is determined as the minimum between crop demand and potential uptake. A crop specific *Active Uptake Factor* must be specified by the user, and using this factor, the amount of plant available P in the soil layer and the *Moisture function*, a daily *Crop P uptake factor* is determined. Two options are currently available to estimate crop P demand:

Option 1

In this simpler approach, crop uptake of P is linked to crop N uptake and is determined using N:P ratios for various crops. The effects of P deficiencies on the crop are not simulated when using this option.

Option 2

For this approach, users specify the crop P concentration at emergence, as well as optimal P concentrations for the vegetative and reproductive growth phases. Root P concentration can also be specified or else taken as 1/6 of root N concentration. The model then uses these concentrations to calculate daily crop P demand. P that has been taken up is first assigned to the roots. If available P does not meet root or aboveground P demand, stress effects on crop growth are determined.

d. Phosphorus transformation processes

The modelling of P processes in soil is generally accepted to be highly challenging and problematic. The nutrient component of GLEAMS is very empirical and only code for selected processes were included. It was decided to simulate P mineralization and immobilization in a similar way to N in CropSyst, making use of C:P ratios. Crop P uptake is related to N uptake (as modelled in CropSyst) using crop specific N:P ratios. N:P ratios were obtained from GLEAMS and other literature. These ratios can be altered as users see fit in the 'Crop parameters' list. Currently N:P ratios for crops remain constant throughout the growing season, but this may be changed at a later stage if found necessary.

Inorganic phosphorus flow

Movement of inorganic P between Labile P and Active P pools is determined by the following equation:

$$\text{Labile Active P Flux} = 0.1WF\, e^{(0.115ST-2.88)}\text{Labile P} - [\text{Active P} \times PAI/(1-PAI)] \qquad (17)$$

Where ST is soil temperature

PAI is phosphorus availability index (Eq. 18-20)

Slightly weathered $\quad PAI = 0.0054 * \text{Base sat\%} + 0.116 * pH(H_2O) - 0.73 \qquad (18)$

Highly weathered $\quad PAI = 0.46 - 0.0916 * \ln(\text{Clay\%})$…………………..(19)

Calcareous soils $\quad PAI = 0.58 - 0.061 * (CaCO_3)$… $\qquad (20)$

As can be seen from the above equation, soil water content and temperature will influence the flux rate. If the flux is positive it indicates movement from the Labile P to the Active P pool, while if the flux is negative, it indicates movement from the Active P pool to the Labile P pool. Also, if the flux is negative, the equation is recalculated using a rate factor of 0.6 instead of 0.1. The Stable P pool is always four times larger than the Active P pool, so movement between these two pools will be determined by the following equation:

$$\text{Active Stable P Flux} = \text{P Flux Coeff} \times (4 \times \text{ActiveP} - \text{StableP}) \qquad (21)$$

where P Flux Coeff = P flux coefficient, with a value of 0.00076 for calcareous soils, and calculated using equation (22) for weathered soils:

$$\text{P Flux Coeff} = \exp^{[-1.77 \times (PAI-7.05)]} \qquad (22)$$

8.2.3 Mass balances

Several balances have been built into to the model and form part of the output. These will be an indication of model performance if matter (water, salt, N, P) has been 'created' or 'destroyed'.

8.3 Conclusions

A significant amount of progress has been made regarding the development of a local scale N and P model. Complete seasons can now be simulated and favourable outputs achieved with the N model but the P model needs improvement especially for use with sludge treated with Fe and Al salts to precipitate P. A primary objective of this model will be to use it as a reasoning support tool for agricultural sludge application under varying cropping systems, sludge types, and climatic conditions to attempt to match nutrient supply at demand and thereby maximize the value of sludges as nutrient sources, and minimize their potential liability as environmental pollutants. This will reduce the need to conduct costly long-term field trials and information generated could be used in the development of updated guidelines for sludge application. The model is being developed with the intention that it will not only be used for research, but can also be useful to consultants, extension officers, economists and even farmers in improved nutrient management planning.

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CHAPTER 9

MODEL CALIBRATION, CORROBORATION, AND LONG TERM ENVIRONMENTAL AND AGRONOMIC IMPACT ASSESSMENT

9.1 Introduction

As described in chapter 8, Nitrogen and phosphorus subroutines have been included in the locally developed SWB-Sci model in order to simulate nutrient balances of fields to which sludge is applied. As SWB is a generic, mechanistic crop growth model, specific parameters for each crop need to be determined. One of the approaches for the determination of specific crop growth parameters is to conduct field trials.

This chapter will focus on N modelling, to quantify N balances for designed sludge application rates and managing leaching. Although P accumulation with N based agricultural sludge application strategies is causing concern, the complex nature of P in Fe or/and Al salt treated sludge makes it difficult to model P. In addition, the long term availability of P from sewage sludge treated with Al and Fe salts is still a subject that remains contentious in the scientific literature.

This report presents selected crop variables for model calibration against measured values for maize, oats, and weeping love grass crops. The model was calibrated under water and nutrient non limiting conditions using data collected during the 2004/05 growing season. It was then evaluated against independent data sets representing the following three conditions: (1) water limited but nutrient non limited conditions (Agronomic crops and dryland pasture) (2) water non limited but nutrient limited conditions (Agronomic crops), and (3) water and nutrient limited conditions (Agronomic crops). Model accuracy was assessed based on the statistical parameters proposed by De Jager (1994). These statistical parameters with their reliability criteria are given in Table 9.1.

Table 9.1. Model evaluation statistical parameters with reliability criteria (after De Jager, 1994).

Statistical parameter abbreviation	Extended meaning of abbreviation	Reliability criteria
r^2	Coefficient of determination	>0.8
D	Willmot (1982) index of agreement	>0.8
MAE (%)	Mean absolute error expressed as a percentage of the mean of the measured values	<20

9.2 Model Calibration

Model parameters like specific leaf area, leaf stem partitioning factor, thermal time requirements, maximum root depth, maximum crop height, dry matter water ratio, radiation use efficiency, extinction coefficient, stem to grain translocation, top dry matter at emergence, and maximum grain N concentration were determined from field data. The remaining parameters were obtained from literature.

9.2.1 Agronomic crops

Generally the model was calibrated successfully for maize and oats. Most of the simulated values for selected variables agreed closely to measured values under water and nutrient non limiting conditions (Fig. 9.1). Although maize aboveground biomass was underestimated towards the end of the season, the difference was not significant and all the statistical parameters were within the ranges prescribed by De Jager (1994) (Table 9.2). Similarly, maize above ground N uptake was underestimated towards the end of the season, which is understandable considering the underestimation of aboveground biomass. This was also indicated by a high RMSE (52.74 kg N ha^{-1}).

9.2.2 Dryland pasture (*Eragrostis curvula*)

The model was calibrated successfully for weeping love grass and most of the simulated values for selected variables agreed closely to measured values (Fig. 9.1). Weeping love grass hay yield simulations were close to the measured values for the first cut of the year. The model, however, underestimated weeping love grass hay yield during the second cut. Nonetheless, all the variables were within the acceptable ranges of statistical accuracy (Table 9.2).

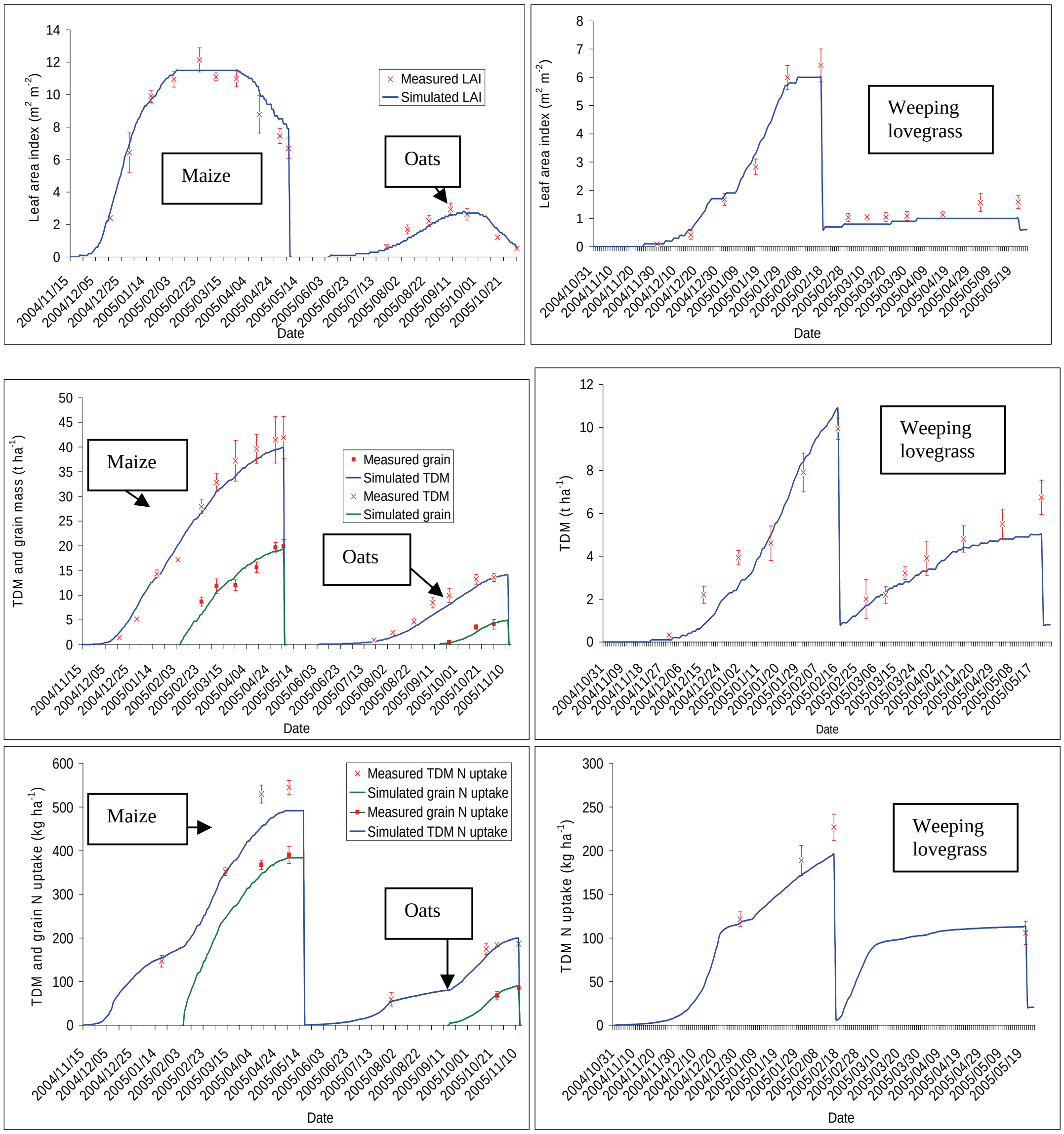

Figure 9.1 Simulated (solid lines) and measured values (symbols with standard deviation) from top to bottom of leaf area index, grain yield, aboveground biomass (TDM), and aboveground biomass N uptake for maize, oats and weeping lovegrass

Table 9.2 Statistical parameters of the SWB model calibration simulations for maize, oats, and weeping love grass during the 2004/05 growing season.

Variable	n	D	RMSE	MAE (%)	R^2
Maize					
LAI	10	0.94	0.82	7.79	0.98
Aboveground biomass	10	0.98	2.18	7.40	0.99
Aboveground biomass N uptake	4	0.92	52.74	8.47	0.99
Grain yield	6	0.88	1.72	9.72	0.95
Grain N uptake (combined irrigated maize-oats)†	4	0.99	12.82	3.71	0.99
Oats					
LAI	7	0.92	0.38	17.06	0.93
Aboveground biomass	7	0.92	1.52	15.18	0.99
Aboveground biomass N uptake	4	0.95	13.19	6.73	0.98
Grain yield	3	0.90	0.77	17.39	0.95
Weeping love grass					
LAI	13	0.98	0.34	14.24	0.99
Aboveground biomass	13	0.92	0.89	15.81	0.97
Aboveground biomass N uptake	4	0.87	20.84	9.15	0.99

†Grain N uptake by irrigated maize and oats for the irrigated maize-oat rotation double cropping system was combined for statistical analyses due to few data points measured for oats.

9.3 Model Corroboration

Model corroboration was conducted using independent data set collected in 2008/09 for maize, oats and weeping love grass. The variables used to evaluate the accuracy of the model were LAI, aboveground biomass, grain (agronomic crops), and aboveground biomass and grain N uptake.

9.3.1 Agronomic crops

The model estimated top dry matter and grain yield (Fig. 9.2) as well as crop N uptake (Fig. 9.3) with acceptable accuracy under water and nutrient non-limiting conditions. Model estimations of similar variables under water limiting but nutrient non-limiting (Figs. 9.4 and 9.5) and water and nutrient limiting (Fig. 9.6) conditions were also acceptably accurate. The only exception was N uptake under water and nutrient limiting conditions, where crop N uptake was overestimated (Fig. 9.7).

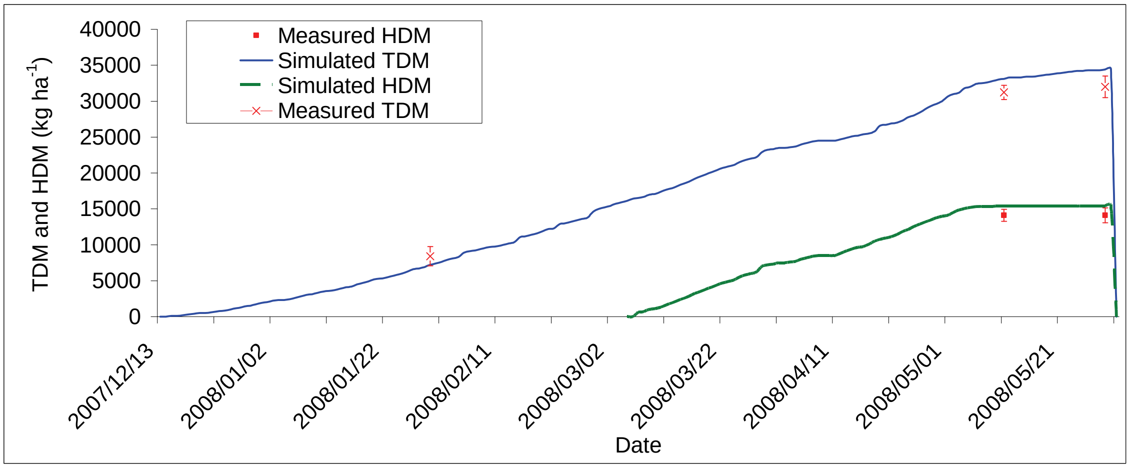

Figure 9.2 Simulated (solid lines) and measured values (symbols) of top (TDM), and harvestable dry matter (HDM) under water and nutrient non-limiting conditions.

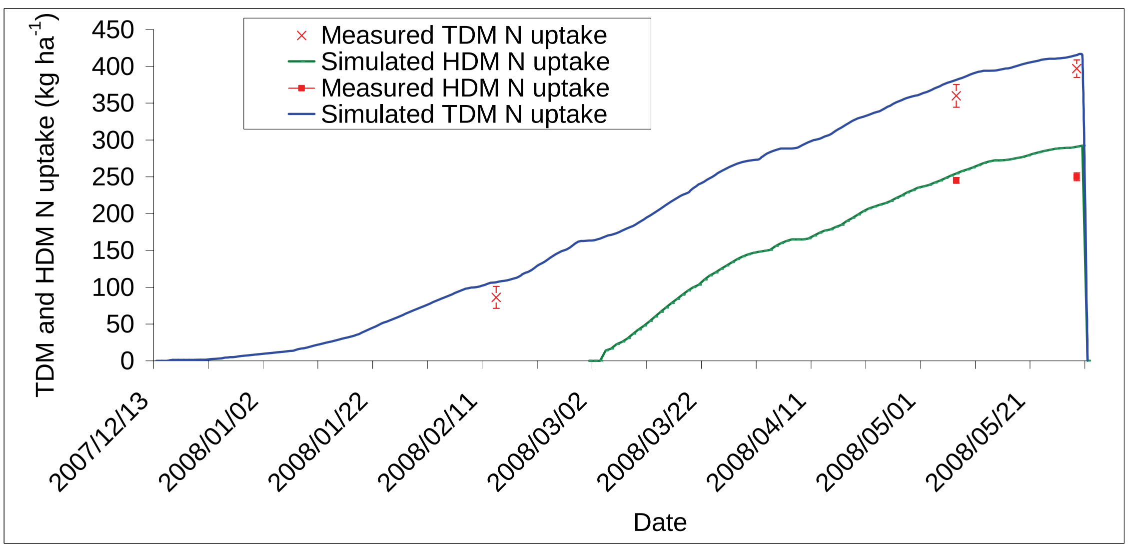

Figure 9.3 Simulated (solid lines) and measured values (symbols) of top (TDM), harvestable dry matter (HDM), N uptake under water and nutrient non-limiting conditions.

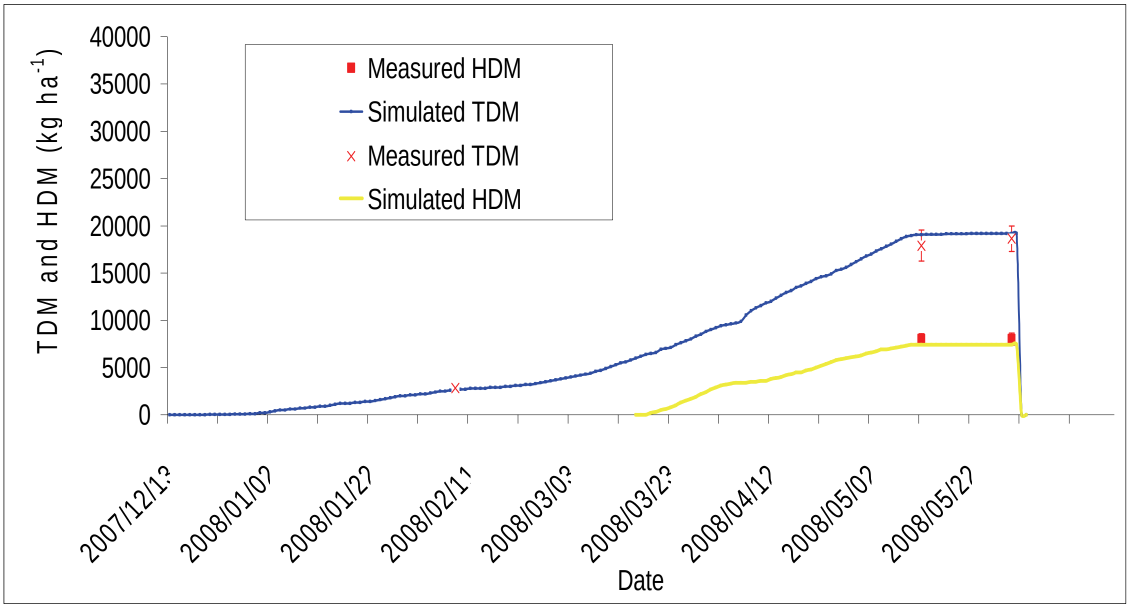

Figure 9.4 Simulated (solid lines) and measured values (symbols) of top (TDM) and harvestable dry matter (HDM) under water limiting but nutrient non-limiting conditions.

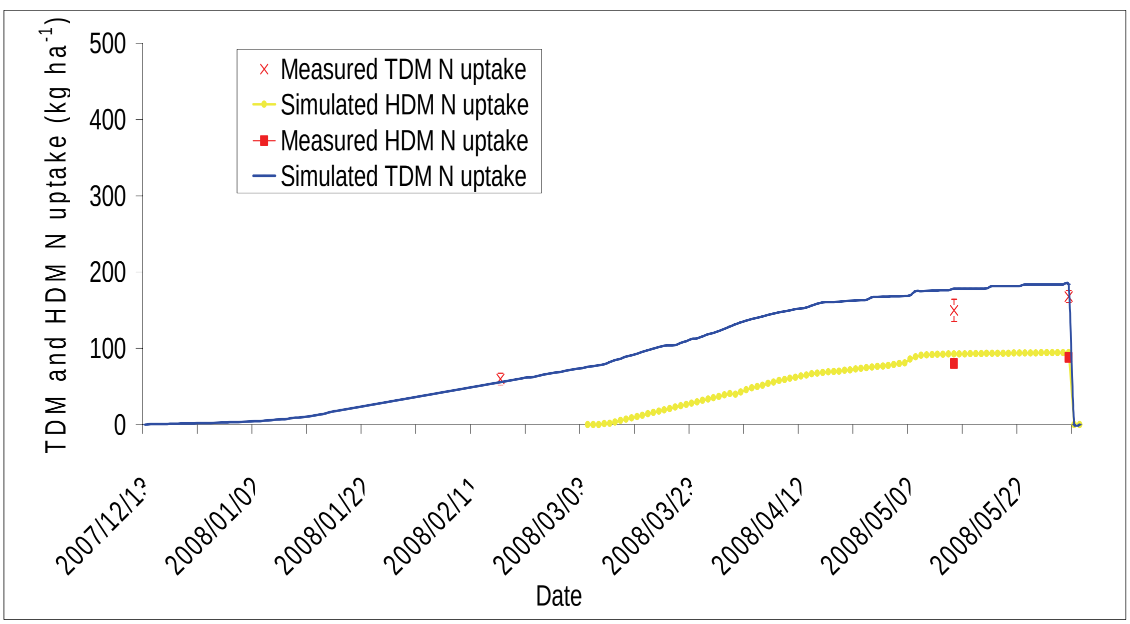

Figure 9.5 Simulated (solid lines) and measured values (symbols) of top (TDM) and harvestable dry matter (HDM) N uptake under water limiting but nutrient non-limiting conditions.

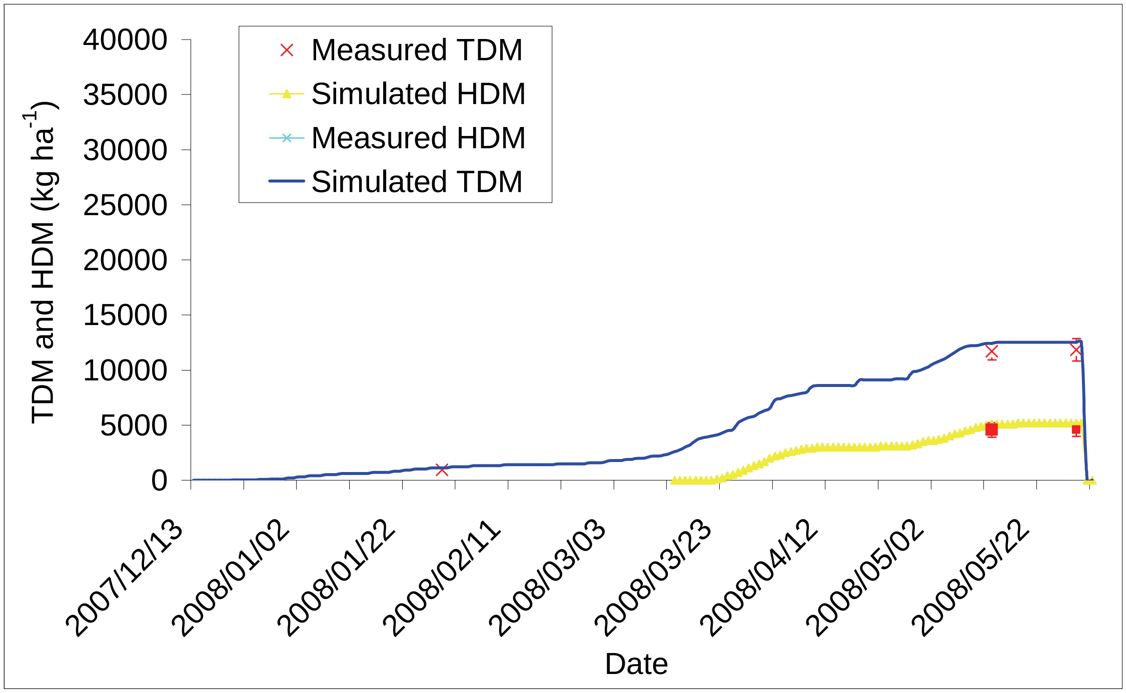

Figure 9.6 Simulated (solid lines) and measured values (symbols) of top dry matter (TDM) and harvestable dry matter (HDM) under water and nutrient limiting conditions.

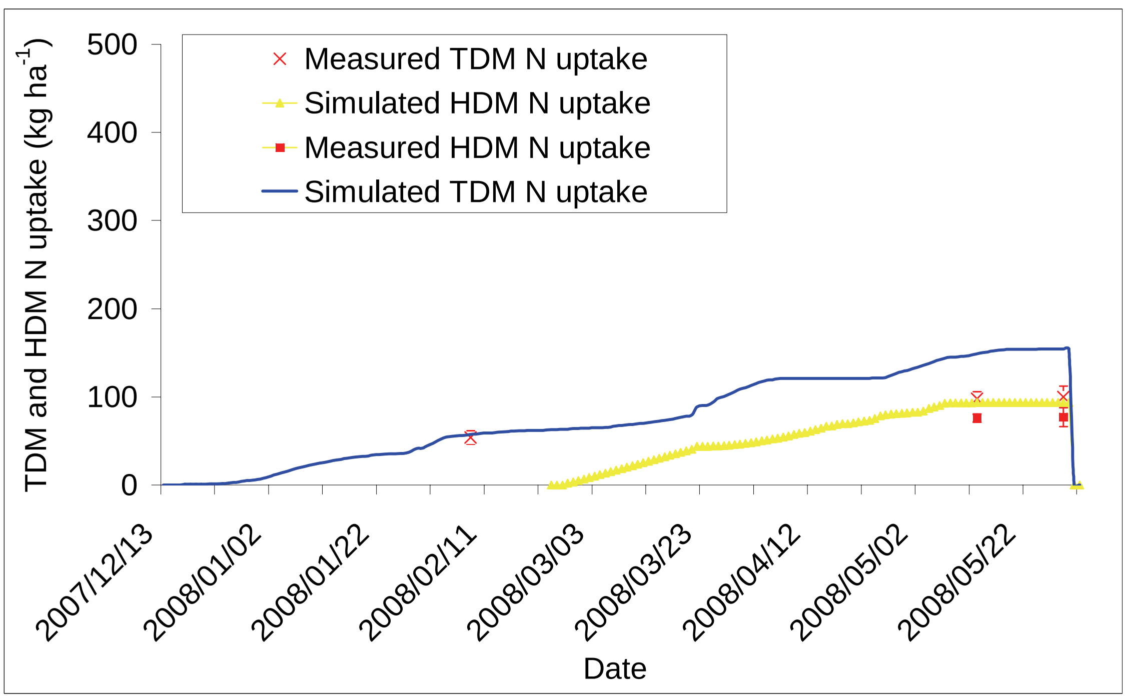

Figure 9.7 Simulated (solid lines) and measured values (symbols) of top (TDM) and harvestable dry matter (HDM) N uptake under water and nutrient limiting conditions.

9.3.2 Weeping love grass

The model predicted weeping love grass hay yield and hay N uptake with relatively lower accuracy (Figs. 9.8 and 9.9) in contrast to the agronomic crop predictions (Figs. 9.2-9.7). Nevertheless, the model closely followed patters of the variables. The relatively poor prediction capability of the model for perennial dryland pasture production was mainly due to the non-mechanistic way of simulating dormancy during winter and re-growth in spring.

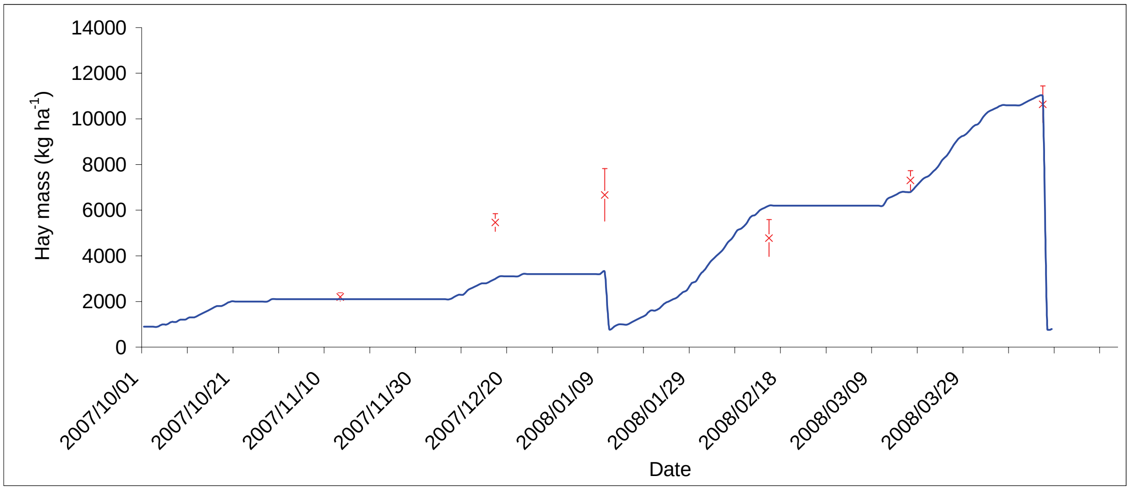

Figure 9.8 Simulated (solid lines) and measured values (symbols) of weeping love grass top dry mass (hay) under water limiting but nutrient non-limiting conditions.

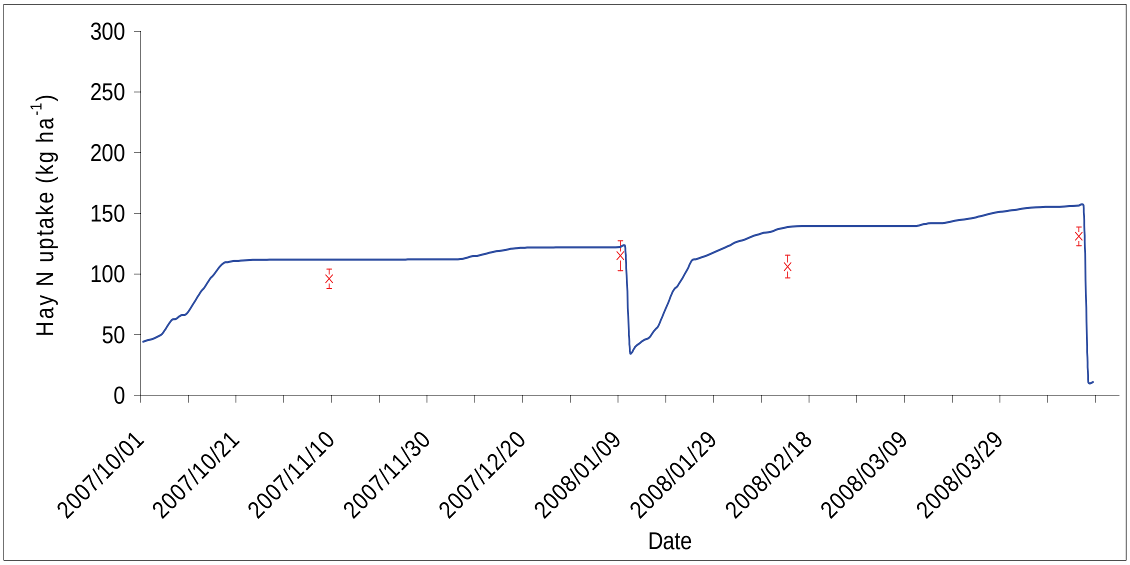

Figure 9.9 Simulated (solid lines) and measured values (symbols) of weeping love grass top dry mass (hay) N uptake under water limiting but nutrient non-limiting conditions.

9.4 Scenario Modelling for Selected Cropping Systems

Scenario simulations were conducted to highlight the long-term agronomic and environmental implications of water availability and farming intensity on sludge application rates. Weather data from OR Tambo international airport was used for the long-term simulation.

9.4.1 Rainfall/irrigation application implications on sludge application rates

Previous studies have shown that crop growth and nutrient uptake response to sludge application rate increases with the availability of water (Binder *et al*., 2002). This was also well demonstrated by model simulations of 38 years, where maize forage and grain yield was higher under irrigation (Fig. 9.10) than dryland farming (Fig. 9.11). The only exception was the wet 1986/87 season, where yield was similar for both dryland and irrigated systems. It was interesting to note that crop yield under irrigated cropping system remained consistent across years. In contrast, forage and grain yield of dryland maize varied across years with the lowest yield being harvested during the 2006/07 growing season (Fig. 9.11) mainly due to low rainfall.

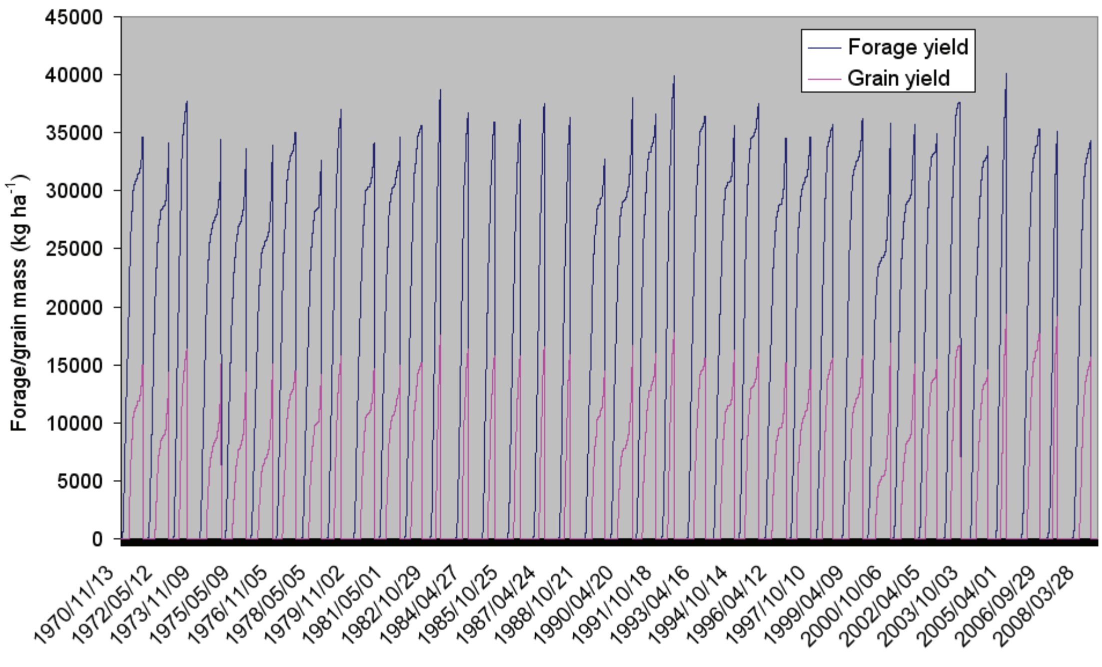

Figure 9.10 Simulated forage and grain yield of irrigated maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

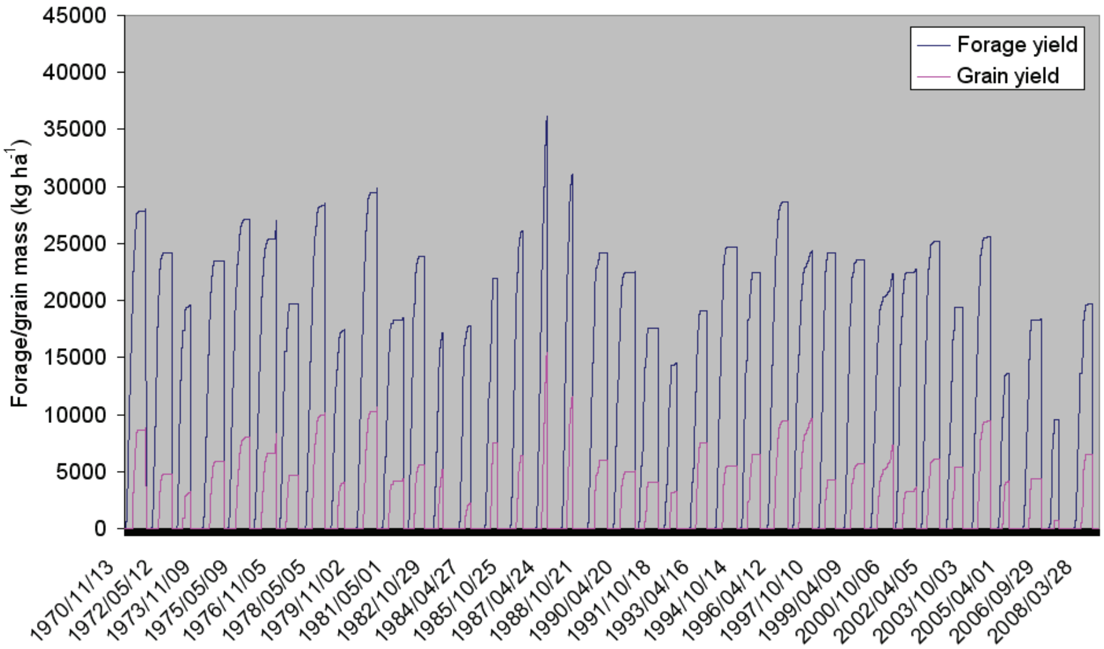

Figure 9.11 Simulated forage and grain yield of dryland maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

Generally, maize forage and grain N uptake was higher under irrigation (Fig. 9.12) than dryland farming (Fig. 9.13). Similarly, dryland maize N uptake was high during high rainfall years and low in low rainfall seasons (Fig. 9.13). It was also evident from the simulation results that N mineralization was lower from dryland farming than irrigated systems (Fig. 9.14). This was mainly because N mineralization is influenced by the availability of water.

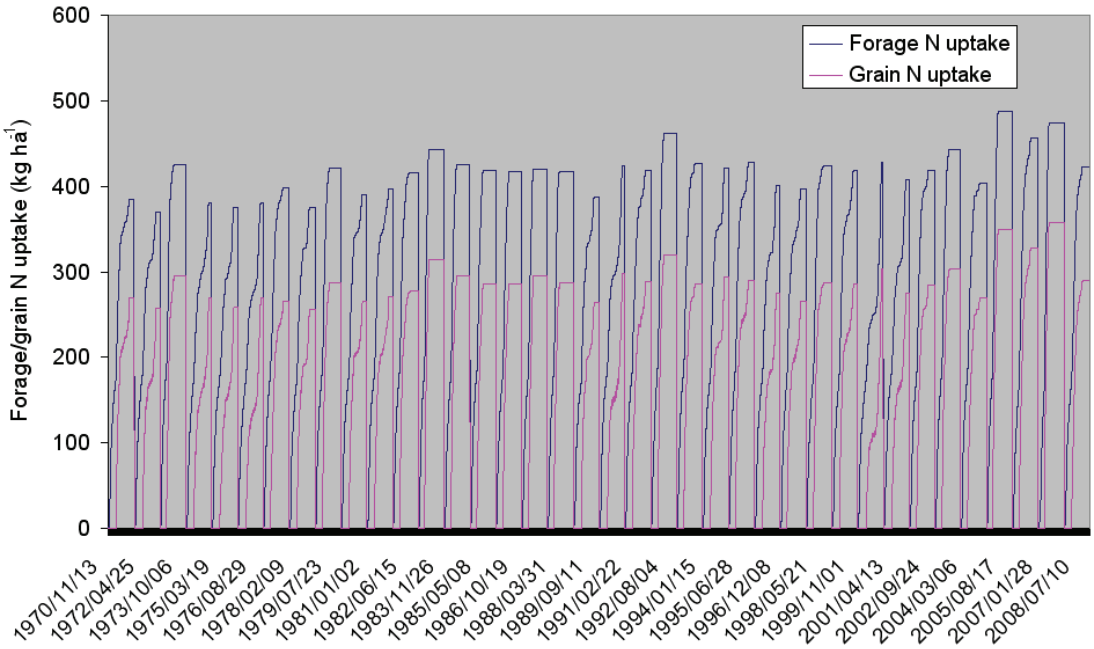

Figure 9.12 Simulated forage and grain N uptake of irrigated maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

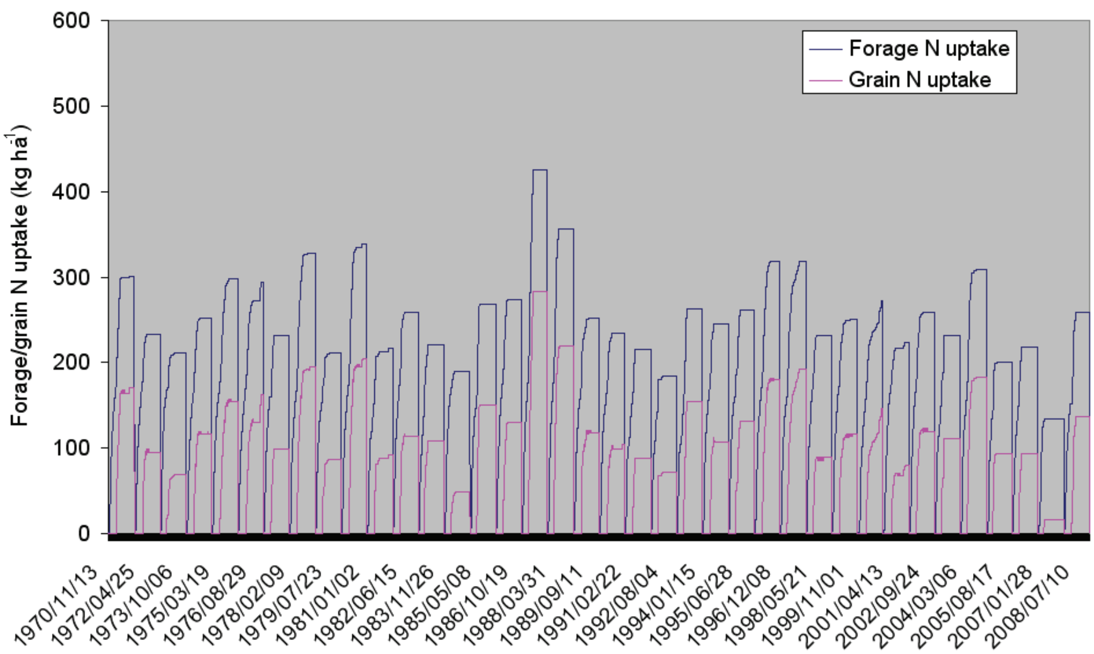

Figure 9.13 Simulated forage and grain N uptake of dryland maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

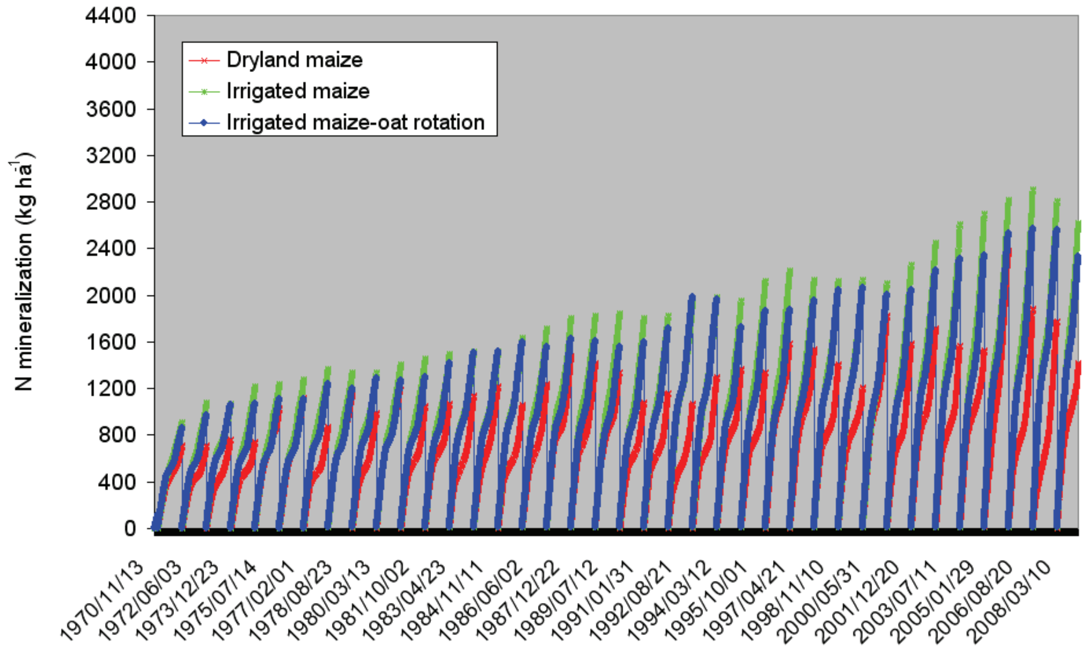

Figure 9.14 Simulated N mineralization from dryland maize, irrigated maize, and irrigated maize-oat rotation planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

It is evident from the above simulation graphs that it might be easier to match nutrient demand and supply of crops under irrigation than dryland conditions. Under dryland cropping systems, where rainfall is erratic, the amount of nutrients that could be utilized by a crop varies depending on the availability of rainfall. At the same time, the availability of rainfall influences the release of nutrients from organic matter. A model could play a significant role to estimate carry-over nutrients from previous applications, potential losses through leaching and volatilization, and, therefore schedule fertilization.

Although the availability of nutrients is essential for optimal crop production, amounts exceeding crop requirements and/or poor irrigation management strategies could cause severe environmental impacts through leaching or runoff. For instance, irrigating at specific intervals according to crop requirement to fill the profile to field capacity resulted in frequent leaching from maize fields (Fig. 9.15). In this specific scenario, the main cause was rainfall events which followed irrigation. There were also a few other instances where single high rainfall events caused leaching of nitrate below the active root zone. The frequent nitrate leaching reported under irrigated maize could, however, be minimized through the adoption of better irrigation strategies, such as leaving room for rain during each irrigation event.

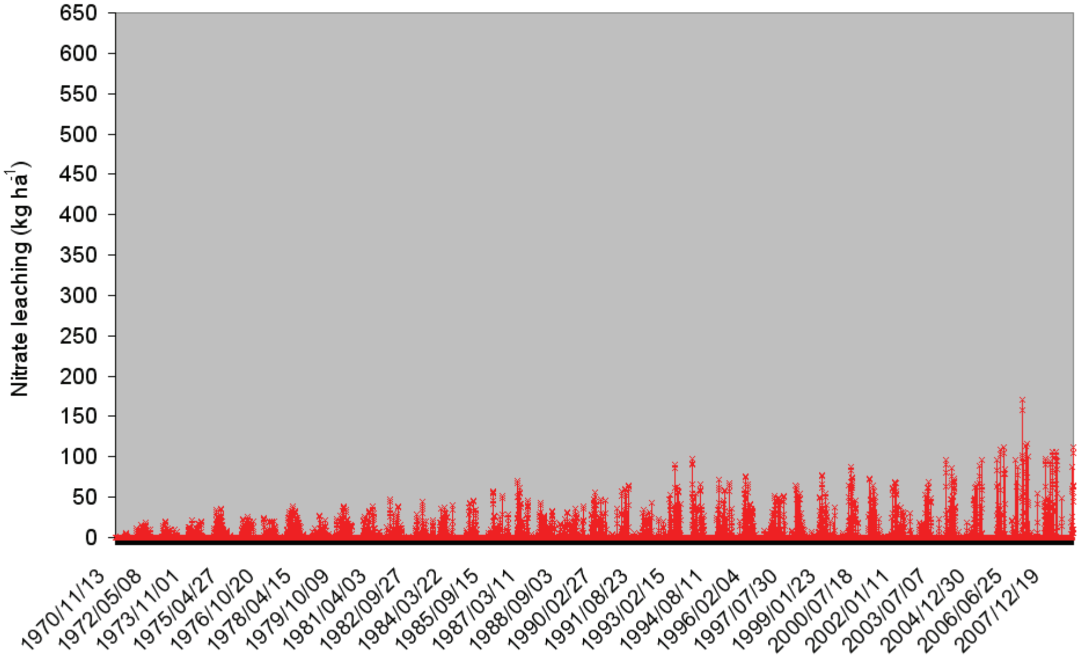

Figure 9.15 Simulated nitrate leaching from irrigated maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge.

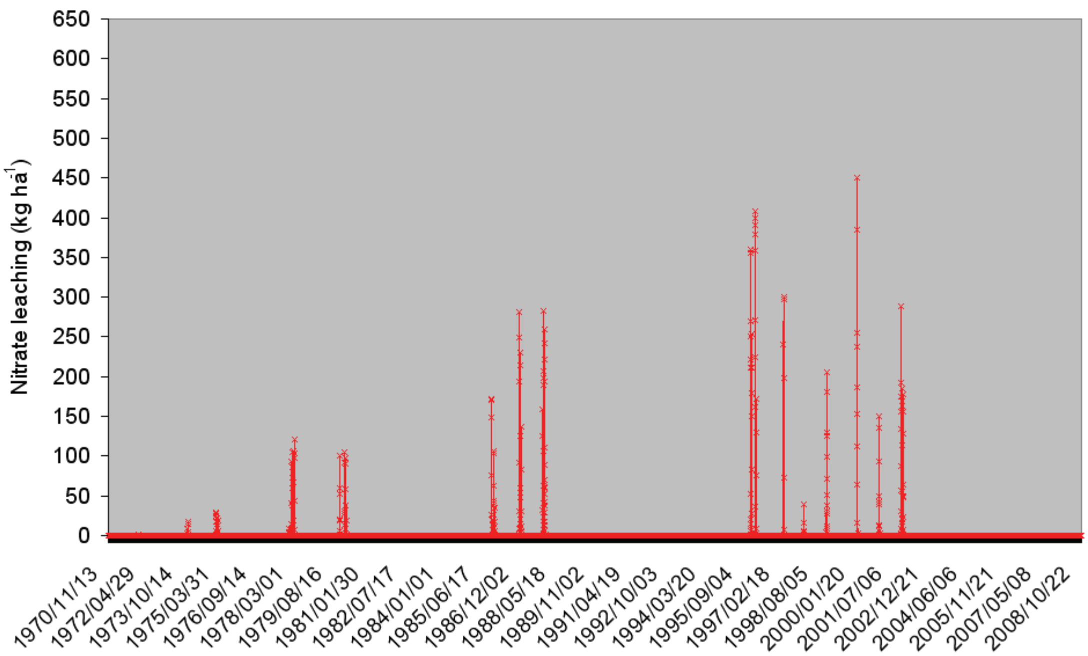

Figure 9.16 Simulated nitrate leaching from dryland maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge.

In contrast, there were very few nitrate leaching events under dryland maize production during the same time period. The N fluxes during those few leaching events were, however, enormous (about 400 kg ha^{-1} d^{-1}) (Fig. 9.16).

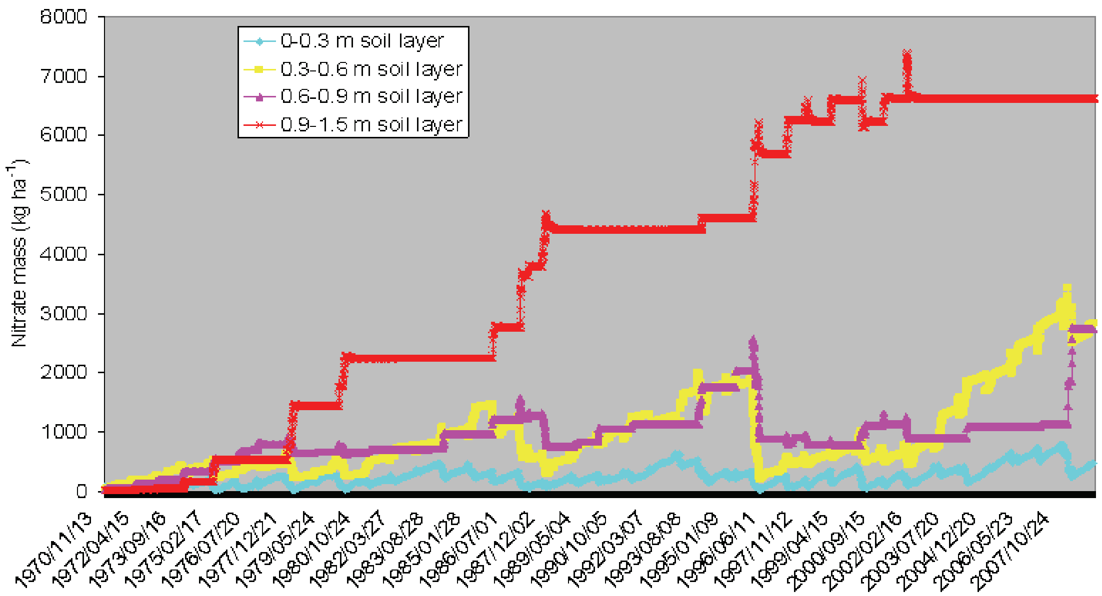

Figure 9.17 Simulated nitrate accumulation in the soil profile of dryland maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge.

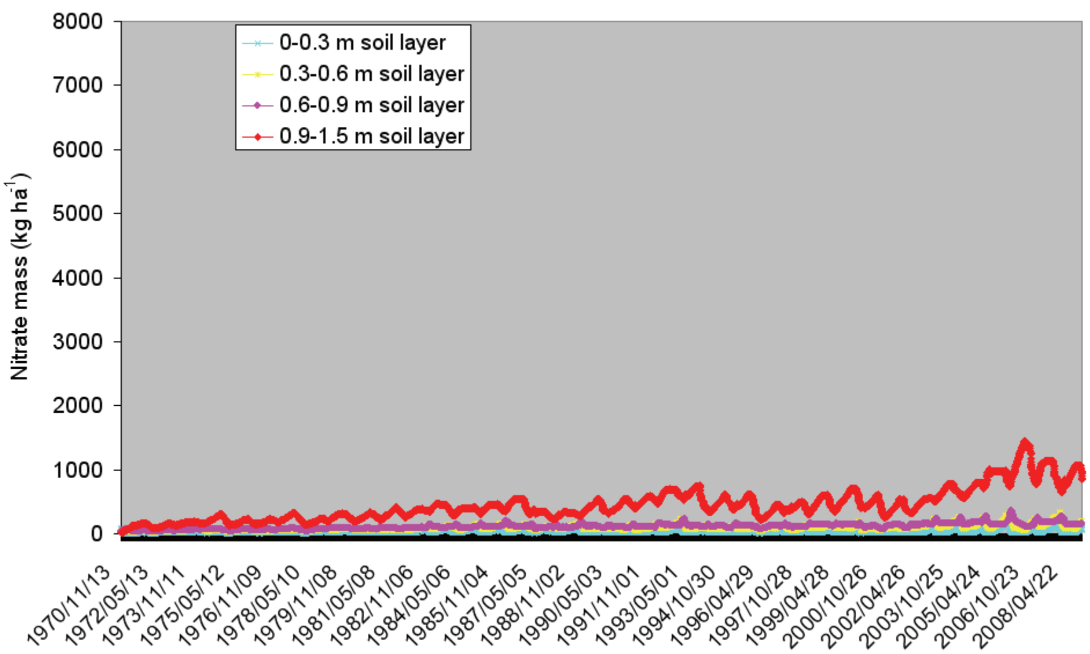

Figure 9.18 Simulated nitrate accumulation in the soil profile of irrigated maize planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge.

It was interesting to note that nitrate accumulation in the 0.9-1.5 m soil profile of the dryland maize increased over time indicating a potential future environmental threat through leaching (Fig. 9.17). This is in contrast to the irrigated maize, which had relatively low nitrate accumulation throughout the profile (Fig. 9.18). The high accumulation of nitrate under dryland farming also indicates that nitrogen was not limiting for crop production.

9.4.2. Farming intensity implications on sludge application rates

Intensive cropping systems improve land productivity by increasing the number of harvests per year as long as the water and nutrient demand of crops are satisfied. Hence, sludge guidelines which do not consider these complex matrices could either affect crop yield or environment or both. For instance, maize yield was lower under an intensive irrigated maize-oat rotation (Fig. 9.19) as opposed to irrigated maize alone (Fig. 9.10) in 24 out of 38 years when 10 t ha^{-1} yr^{-1} sludge was applied to both cropping systems. This was mainly attributed to the deficiency of nutrients, especially N, under the irrigated maize-oat rotation (Fig. 9.20) as opposed to the irrigated maize alone (Fig. 9.12). Hence, the recommended sludge application rate should be dynamic and needs to consider the intensity of the cropping system for optimal crop production with minimal environmental impacts.

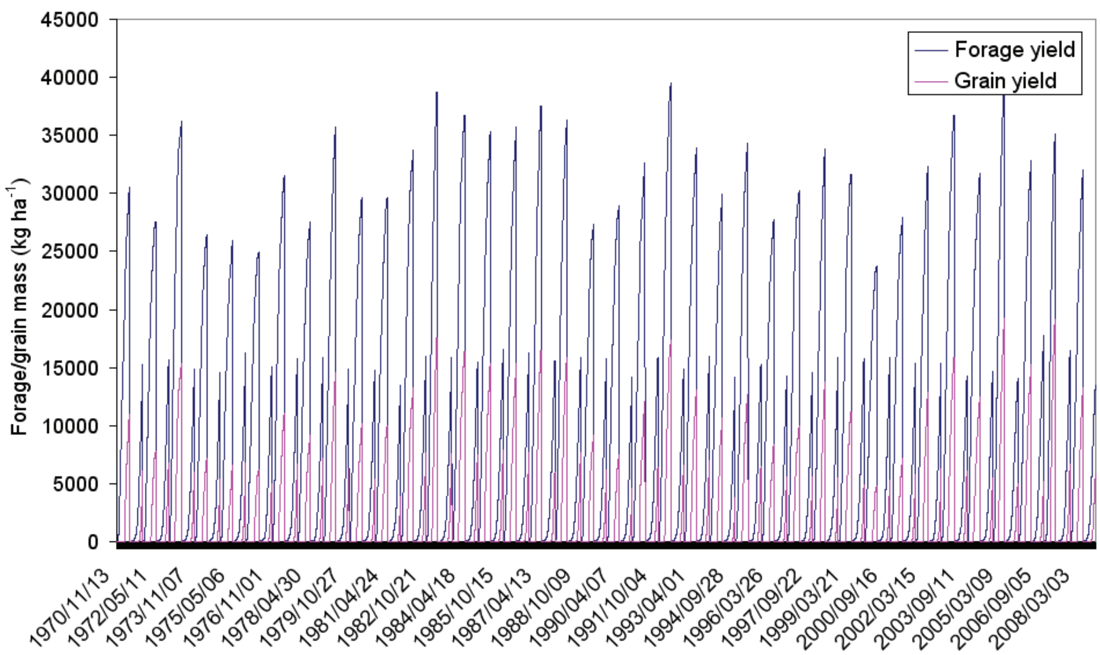

Figure 9.19 Simulated forage and grain yield of an irrigated maize-oat rotation planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

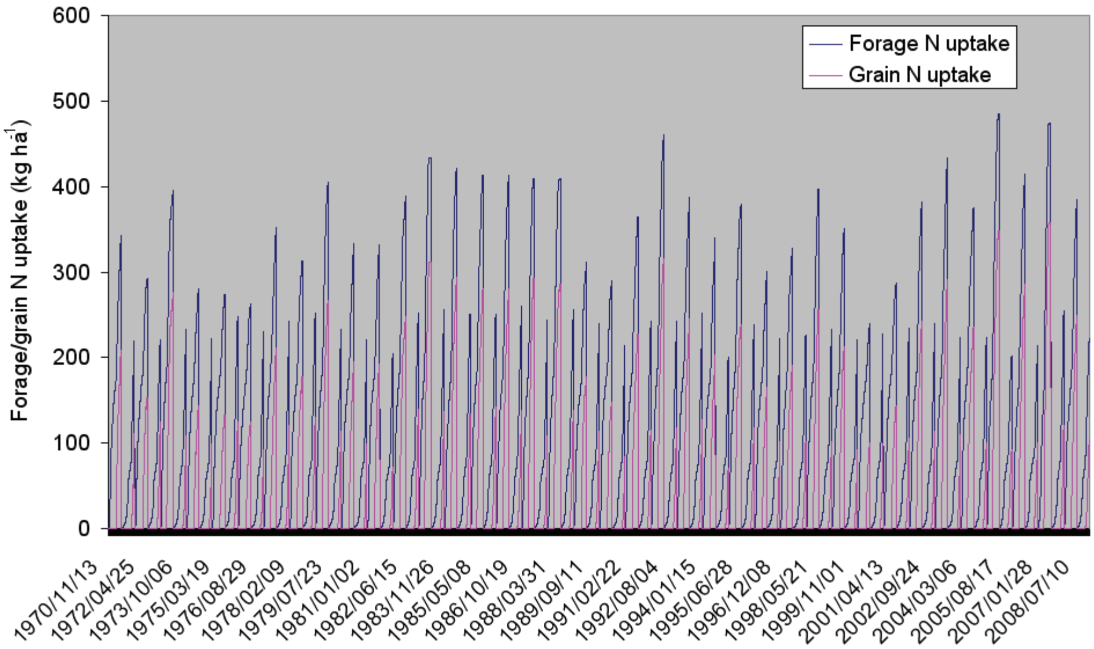

Figure 9.20 Simulated forage and grain N uptake of an irrigated maize-oat rotation planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge

Generally, N mineralization from the irrigated maize-oat rotation was similar to irrigated maize and was higher than dryland maize and dryland pasture (Fig. 9.14). Nevertheless, in most cases, nitrate leaching (Fig. 9.21) was higher under irrigated maize alone than for an irrigated maize-oat rotation. This was mainly due to extra N uptake by oats during the winter season from the rotation, which minimised the potential for nitrate leaching in early summer that could occur before active maize root development.

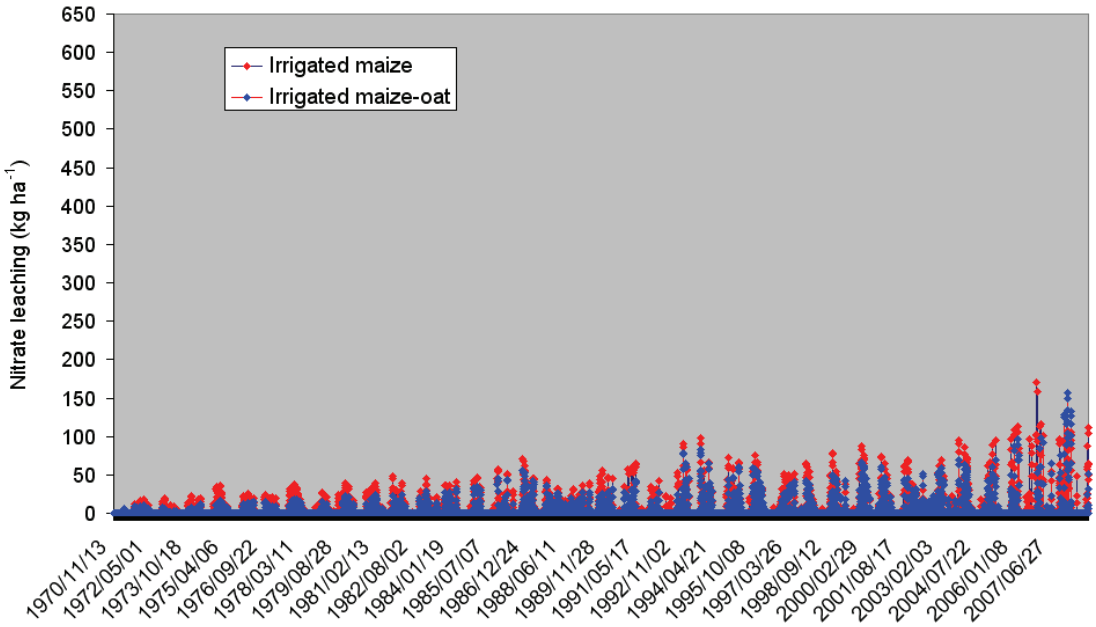

Figure 9.21 Simulated nitrate leaching from irrigated maize and irrigated maize-oat rotation planted to a clay loam soil treated with 10 t ha^{-1} yr^{-1} sludge.

9.5 Conclusions

The model was successfully calibrated for maize, oats, and weeping love grass. The model simulated both dry matter production and crop N uptake with acceptable accuracy. Model corroboration conducted using independent data sets also proved the accuracy of the model in simulating the above mentioned variables for agronomic crops under varying nutrient and water supply conditions. Although, the model showed relatively low accuracy under certain conditions for agronomic crops and dryland pasture, with some improvement in the processes involved, it shows great promise as a reasoning support tool, and can already be used through scenario simulations to inform prudent sludge management practices.

Long-term model simulation results show that agricultural sludge application rates are influenced by water availability and farming intensity. From the model simulation results it can also be deduced that the choice of irrigation strategies could have a significant impact on the level of nitrate leaching under irrigated systems. The scenario simulations also show that dryland farming systems planted under erratic and unpredictable rainfall conditions are prone to high environmental risk from nitrate leaching if sludge application rates are not properly estimated. Irrigated systems could also pose similar risks if proper irrigation management practices are not implemented.

A useful tool has been developed, and industry can now guide the research team in running industry useful simulations to address specific concerns they may have. There should also be opportunities to add value to the South African sludge guidelines using the model.

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CHAPTER 10

DISCUSSION AND RECOMMENDATIONS

The annual cumulative crop nutrient removal from agricultural systems is a function of the crop cultivar, water and nutrient availability, cropping intensity, and management practices. So it is apparent that fertilization programmes of either commercial inorganic or organic sources, including municipal sludges should be adjusted accordingly for sustainable crop production. Management of most commercial inorganic fertilizers is relatively simpler than for organic sources, because the nutrients are readily available for crop uptake in most cases. Large fraction of nutrients, especially N, from sludge are organically bound and are not readily available for uptake by plants. In addition, some wastewater treatment plants add Fe and Al salts to precipitate P from sludge, which affects P availability for uptake by plants. The availability of such nutrients is influenced by a combination of factors including sludge characteristics, climatic, and edaphic factors.

Nitrogen mineralization incubation studies showed that release of N differed from sludge to sludge, and seemed to follow sludge volatile solid content. Consequently, regular characterization of sludge used on agricultural land is imperative in order to ensure adequate management of N in sludge-amended soil. This study also found that N release from sludge is influenced by the soil's temperature and water potential. Dry soil conditions of water potential less than -20 kPa and temperatures below 10°C may inhibit mineralization. Consequently, mineralization of N from sludge will be slow during dry winter months, leading to the recommendation that sludge be applied when conditions are optimum for nitrification (25°C, field capacity) in order to avoid crop N deficiency and a build-up of ammonium in the soil. The N mineralization rate constants calculated from this study are important model input parameters, however, determination of rate constants of N release from the slower release pools is still needed to improve model prediction.

The incubation and phyto-availability studies showed that sludge products like Agriman and that of the Activated Sludge Pasteurization process (ASP) have similar phosphate plant availability as commercial fertilizers. These products are therefore agronomically viable sources of P. The RPFV value of Vlakplaas was, however, the lowest of the municipal sludges tested. This was attributed to the high iron content of the sludge, which was most likely the result of chemical removal of phosphate in the wastewater by means of ferric phosphate precipitation. The negative effect of this tertiary treatment process on phosphate plant availability from sludge is well documented.

Good correlations were obtained between the Relative Phosphate Fertilizer Value (RPFV) index developed in this study and the P content of above ground biomass of maize. This index was found to be a better tool to predict actual plant availability than using only Bray 1 extractability as an indicator. The RPFV approach is therefore recommended as a routine test to determine the phosphate fertilizer value of sludge.

Various crops have varying nutrient requirements. For instance, under water and nutrient non limiting conditions, the nutrient demand of maize is much higher than wheat and barley. It is also apparent that the potential crop nutrient uptake could vary across cultivar of a given crop type. Typical examples are the contrasting nutrient demands of dwarf short season maize and tall long season maize cultivars. This brings forth the need for careful nutrient management practices which include sludge application rates.

Cropping intensity is one of the major factors that influence the cumulative nutrient requirement of a given cropping system per unit area per year. According to this study, forage production under an intensive irrigated maize-oat rotation was able to utilize, on average, two times the N and three times the P of that of dryland maize. This indicates the need for sludge application rates to be adjusted according to the intensity of farming system, as cumulative annual nutrient uptake is dependent up on it. The Management practices of either harvesting the grain only or removing the whole crop from a field also plays a significant role in the field nutrient mass balance. Results from this study showed that forage N uptake was 9-21% and 37-44% higher than grain N uptake by dryland maize and irrigated maize-oat rotation. Consequently, the sludge application rate should also be adjusted accordingly to optimize crop production while promoting environmental resilience.

It was also evident from this study that sludge application at all rates including the lowest rate of 4 t ha^{-1} caused excess accumulation of total P under dryland maize, intensive irrigated maize-oat rotation, and dryland pasture. This implies that certain monitoring mechanisms should be set in areas where sludge is applied according to crop N requirement for potential mobile P accumulation. Precautions should be taken when sludge is applied on the soil surface as is done for perennial pastures. This is because the rate of build-up of heavy metals and other elements is faster on the top few centimetres of the soil profile if sludge is not incorporated into the soil. Therefore, a time line should be set to determine how often to incorporate the sludge into the soil. Simple equations such as the one used in Volume 4 of the current South African Sludge Guidelines to estimate permissible application rate (PAR) could be modified to estimate the cumulative application rate above which any further application can only be done if the sludge is going to be incorporated into the soil.

Under turfgrass sod production, increasing sludge application rates up to 67 t ha^{-1} significantly improved turf establishment rate and colour. The ability of sods to remain intact during handling improved as the sludge application rate increased to 33 t ha^{-1}, but deteriorated at higher rates. High application rates of 100 t ha^{-1} minimized soil loss from the site but caused unacceptably high leaching losses. According to this study, anaerobically digested paddy dried sludge could be applied as high as 33 t ha^{-1} for turfgrass sod production – four times higher than the old 8 t ha^{-1} limit. This rate improved turf growth and quality with minimal environmental impact through nitrate leaching. Under turfgrass sod production, where sludge is surface applied, the equation used in Volume 4 of the current South African Sludge Guidelines to estimate the permissible application rate (PAR) may not be applicable because during sod harvesting the sludge, along with a few millimetres of soil is removed. The environmental risks that may arise due to potential leaching of some of the heavy metals should, however, be monitored onsite at certain time intervals. In addition, the option of single high volume sludge application (60 or 120 t ha^{-1}) provided in Volume 4 of the current guide should be reconsidered for environmental reasons. This is because results from this study have shown that high application rates exceeding 33 t ha^{-1} result in high nitrate leaching losses below the active plant root zone.

It was evident from this study that N uptake varied across a range of cropping systems, seasons, and management practices. In addition, N content varies across a range of sludge types and N mineralization from sludge could take several years before reaching steady state conditions. Therefore, a dynamic, mechanistic reasoning support tool is needed in order to accommodate the interaction between the various factors involved.

The N sub-routine incorporated into the existing SWB model was successfully calibrated with acceptable accuracy for dryland maize, the irrigated maize-oat rotation, and dryland pasture. The model was tested against independent sets of data and predicted leaf area index, aboveground biomass, grain yield, aboveground biomass N uptake, and grain N uptake with acceptable accuracy. The model predicted the above mentioned variables for dryland pasture with somewhat lower accuracy due to the non-mechanistic approach followed by the model when simulating perennial grasses but should not be a problem to improve pasture simulation. The N model looks promising, however, especially for dryland maize and irrigated maize-oat rotation to be used as a reasoning support tool for sustainable sludge use in agricultural lands.

Long term model simulation for the study site under the prevailing climatic conditions showed heavy nitrate leaching events when a dry season is followed by a wet season due to unused nutrients being carried over to the next season. Hence, nutrient management under dryland farming will still remain a challenge due to the unpredictable nature of rainfall under arid and semi-arid climatic conditions. This will influence the annual sludge application rates. The potential environmental risks of nitrate leaching could, however, be minimized without compromising agricultural production by monitoring fields and using models as reasoning support tool.

Ideally sludge should be applied to satisfy crop N demand, and this is clearly dynamic. Ultimately, the maximum amount of sludge applied to a site will depend on the accumulation of total and available P and the risk this poses for pollution, as long as the threat from other pollutants is minimal.

It is anticipated that this study will help to improve the local knowledge of the benefits and disadvantages of sludge application to agricultural lands. Moreover, results from this research will contribute to fine-tune the current guideline on agricultural use of wastewater sludge (Volume 2) and beneficial use of sludge at high application rates (Volume 4).

Recommendations for future research include:

- A monitoring protocol to complement the reasoning support system, to ensure that leaching losses are kept within reasonable limits.
- The maximum number of years for which sludge could be left without incorporation under perennial pasture grasses with minimal environmental impact through runoff nutrient losses needs investigation.
- The potential for pollution through nitrate leaching as well as P and nitrate runoff losses from turfgrass established at new sites from sods treated with high sludge rates needs investigation.
- Updating the SWB-Sci model to simulate perennial grasses mechanistically.
- Scenario modelling for real life situations.
- Managing irrigation to optimise salt leaching while minimizing nitrate leaching from sludge-amended soils.
- Developing a user-friendly reasoning tool for use by wastewater treatment plant managers based on long term scenario simulations using the SWB-Sci model.
- Testing SWB-Sci model for P.

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APPENDIX A

Two weekly cumulative rainfall data at ERWAT during the 2007-2009 growing seasons

Year	Weeks	Rain (mm)											
		Jan	Feb	Mar	Apr.	May	June	July	Aug.	Sep.	Oct.	Nov.	Dec.
2007	1-2										71.6	20.6	69.3
	3-4									31.9	32.3	35.1	2.8
2008	1-2	58.8	56.3	23	17.1	32.9	14.3				0.2	77.6	69.8
	3-4	147.6	3.8	136.3	0.6	10.2	2.6				72.4	21.8	43.6
2009	1-2	38.2	92.8	34.3	6.8	4.2							
	3-4	95.6	43.9	62.7	0.6								

APPENDIX B

Incorporation and mixing efficiencies of various tillage implements

Tillage implement	Incorporation efficiency	Mixing efficiency
Anhydrous ammonia applicator	0.05	0.05
Bedder--lister	0.95	0.05
Burn	0	0
Chisel	0.1	0.05
Cultivator--field	0.1	0.1
Cultivator--row	0.1	0.1
Digger--peanut	0.05	0.05
Digger--potato	0.15	0.05
Disk harrow--offset	0.85	0.6
Disk harrow--tandem	0.75	0.5
Disk tiller	0.3	0.05
Disk plough	0.8	0.4
Disk plough--one way	0.5	0.5
Do-all	0.1	0.25
Drill--deep furrow (dempster)	0.3	0.05
Drill--small grain	0.05	0.05
Harrow--spike tooth	0.05	0.05
Harrow--spring tooth	0.05	0.05
Moldboard plough	1	0.25
Paraplough	0.05	0.05
Planter--in-row chisel	0.05	0.05
Planter--knife, disk, sweep	0.05	0.05