

**Sustainable use of sewage sludge as a source of nitrogen and
phosphorus in cropping systems**

by

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DECLARATION

I the undersigned, declare that the thesis, which I hereby submit for the degree of Doctor of Philosophy at the University of Pretoria, is my own work, except where acknowledged in the text, and has not previously been submitted by me for a degree at this or any other tertiary institution.

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1 Sep. 2009

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LIST OF ACRONYMS AND ABBREVIATIONS

Al	Aluminium
As	Arsenic
Cd	Cadmium
Cr	Chromium
Cu	Copper
CFU	colony forming units
ERWAT	East Rand Water Care Works
EU	European Union
Fe	Iron
FeO	Iron oxide
Hg	Mercury
LAI	Leaf area index ($\text{m}^2 \text{m}^{-2}$)
MAE	Mean absolute error
M1-P	Mehlick-1 extractable phosphorus
M3-P	Mehlick-3 extractable phosphorus
N	Nitrogen
P	Phosphorus
Ni	Nickel
NO ₃	Nitrate
NH ₄	Ammonium
Pb	Lead
PFU	plaque-forming units
r^2	Coefficient of determination
Se	Selenium
US EPA	United States Environmental Protection Agency
WUE	Water use efficiency ($\text{kg ha}^{-1} \text{mm}^{-1}$)
Zn	Zinc

ABSTRACT

Municipal sewage sludge is used as source of plant nutrients world wide for agriculture. However, many countries do not make full use of this opportunity. A lack of local knowledge about the benefits and disadvantages of sludge contributes to this low utilisation. For instance, only 28% of the sludge produced in South Africa is beneficially utilized on agricultural lands. The overall objectives of this study were 1) to determine responsible sludge loading rates for a range of cropping systems 2) to investigate the agronomic benefits and sustainability of using municipal sludge according to crop N demand, and 3) to develop a tool to enable extrapolation of these results to other regions (soils, climates) and other cropping systems. Field experiments were conducted on a wide range of cropping systems including dryland maize, irrigated maize-oat rotation, dryland pasture, and turfgrass sod production. An 8 Mg ha⁻¹ control (South African old annual upper limit norm) was compared with sludge rates of 0, 4, and 16 Mg ha⁻¹ for the agronomic crops and dryland pasture. Under the turfgrass sod production, the aim was to export large volumes of sludge with the sod without compromising the environment. Therefore, an 8 Mg ha⁻¹ control treatment was compared with sludge rates of 33, 67, and 100 Mg ha⁻¹ which are equivalent to depths of 5, 10, and 15 mm sludge, respectively. Doubling of the old annual upper limit significantly increased grain and forage yield of both the dryland maize and the irrigated maize-oat rotation. This rate also improved weeping lovegrass hay yield, water use efficiency and crude protein content. Residual nitrate in the soil profile after harvest, and solution samples collected from wetting front detectors were

used as indicators of groundwater pollution through nitrate leaching in the medium term. For the irrigated maize-oat rotation and dryland pasture, a low leaching risk was indicated even at high sludge loading rates of 16 Mg ha⁻¹ in this clay loam soil. In contrast, residual nitrate for similar sludge rates under dryland maize cropping did reveal the potential for pollution through leaching. Sludge loading at all rates resulted in the accumulation of total P and loading rates of 16 Mg ha⁻¹ increased Bray-1P in all agronomic and pasture cropping systems. In the case of turfgrass for sod production, sludge loading rates up to 67 Mg ha⁻¹ significantly improved turfgrass establishment rate and colour. The ability of sods to remain intact during handling and transportation improved as the sludge loading rate increased to 33 Mg ha⁻¹, but deteriorated at higher rates. A sludge loading rate of 100 Mg ha⁻¹ was needed to eliminate soil loss at harvest, but this rate was associated with unacceptably high N leaching losses and poor sod strength. The variation in sludge quality, crop nutrient removal across a range of cropping systems, and seasons indicates that a dynamic, mechanistic decision support tool is needed to estimate responsible sludge loading rates. A mechanistic N module was adapted and incorporated into an existing soil water balance/crop growth model (SWB). The model was calibrated with statistically acceptable accuracy for dryland maize, irrigated maize-oat rotation, and dryland pasture. The model was tested against independent data sets and was able to predict the measured variables of interest with acceptable accuracy for dryland maize, irrigated maize and oats. For dryland pasture, the model predicted similar variables of interest with lower accuracy for medium-term simulations, but this

improved with updating the profile water content after every hay cut. The ideal sludge loading rate to satisfy crop N demand is dynamic and should be adjusted according to cropping systems, seasonal rainfall variability, sludge N concentration, and sludge application strategy (N or P based). The ultimate cumulative sludge loading of an area will depend on the accumulation of total and Bray-1P, and the risk this poses for pollution, as long as the risk from other pollutants remains minimal. The SWB model shows promise as a decision support tool for sludge management in agricultural lands.

CHAPTER 1

GENERAL INTRODUCTION

Beneficial use of sewage sludge on agricultural lands is a very well known practice around the world. The benefits include; a source of essential crop nutrients (Muse et al., 1991), improvements in soil structure (Ojeda et al., 2007), and minimisation of soil erosion and runoff (Muse et al., 1991; Ojeda et al., 2003). Nutrients applied above a crop's nutrient requirement, 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 wastewater treatment plants. This is despite the enormous pressure on South African wastewater treatment plants to dispose of or utilize their sludge in an environmentally sustainable way. The poor public perception of sludge use and the lack of thorough local field scale studies have resulted in a rigid guideline with single upper limit for all cropping systems. This situation exacerbated the current low usage rates in agriculture.

Generally, sludge with acceptable quality for agricultural use is applied according to crop N requirements (Mile and Graveland, 1972; Dotson, 1973). However, sludge application according to crop N demand causes P build up in the soil profile (Kelling et al., 1977; Pierzynski, 1994; Maguire et al., 2000a,b), although its availability can be affected by sludge treatment processes in the water care works (Soon et al., 1978a; Kirkham, 1982; McCoy et al., 1986; Jokinen, 1990; Kyle and McClintock, 1995; Frossard., 1996; Maguire et al., 2001).

A large fraction of the N in sewage sludge is in the organic form, whereas plants absorb N in the form of NH_4 and NO_3 . The availability of N for plants depends therefore on the rate of mineralization of the sludge (Kelley et al., 1984) and the losses of inorganic nitrogen through volatilization, denitrification, and leaching. All the above mentioned processes are influenced by soil type, availability of water, soil temperature, and soil pH. Furthermore, the type of sludge treatment process affects both mineralization and volatilization, while sludge application methods largely influence ammonia volatilization (Henry et al., 1999). Therefore, direct extrapolation of studies conducted in a specific soil type, climate, and sludge type can compromise both the environment and crop yield.

A range of studies have shown that an increase in sludge application rate increased grain and stover yield of agronomic crops (Binder et al., 2002, Cooper, 2005; Lavado et al., 2006; Bozkurt, 2006) as well as dry matter yield of pasture grasses (Sullivan et al., 1997; Zebarth et al., 2000; Cogger et al., 2001). Crop

response to sludge application rate is, however, influenced by the availability of water. For instance studies conducted by Soon et al. (1978b) showed a positive maize growth response up until a sludge N supply of 200 kg ha^{-1} under dryland as opposed to 400 kg N ha^{-1} under irrigation (Binder et al., 2002). Sludge applications of 400 kg N ha^{-1} under dryland cropping resulted in the build up of nitrate in the soil profile compared with similar rates under irrigation. In contrast, pasture grass response to sludge application rate increased until a total N supply of just above 800 kg ha^{-1} , but such a high rate was associated with nitrate and P accumulation in the soil profile (Cogger et al., 2001). The above shows that crop nutrient demand is dynamic and is influenced by the cropping system and the availability of water.

Despite this, the former South African sewage sludge guideline had a single upper limit for all cropping systems of $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ which was based on studies conducted in other countries. It was recently updated to $10 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ based on local laboratory incubation studies and very few short-term (1- 2 years) local field trials on specific crops. Such a guideline may be too inflexible for a range of different cropping systems, sludge types, nutrient contents, climates, and soil types. Both guidelines called for local field scale medium to long-term studies to be carried out (Water Research Commission, 1997; Snyman and Herselman, 2006). This study was initiated after the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ guideline was released, and was the first of its kind in South Africa to include agronomic benefits and environmental impacts of different sludge rates on contrasting cropping systems.

The **objectives** of this study were

- To investigate the dynamic nature of sludge application rate across a range of management practices and cropping systems.
- To investigate the agronomic benefits and sustainability of using municipal sludge according to crop N demand
- To evaluate the sustainability of high surface loading rates above crop requirements for turfgrass sod production.
- To adapt the N sub routine from the CropSyst model and include it into the SWB model, and to test its reliability as a decision support tool for sewage sludge management on agricultural lands.

In order to meet these objectives the following hypotheses were tested.

Agronomic crops:

1. Sludge application above the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ limit will increase dryland maize and irrigated maize-oat rotation grain and forage yield.
2. Under both dryland and irrigated conditions, more N can be exported in forage than grain.
3. Sludge applications that produce the highest yield under irrigated and dryland conditions will not cause an accumulation of N in the soil profile and are not susceptible to excessive nitrate leaching.
4. Sludge application according to crop N demand results in the accumulation of total and plant available P in the soil profile.

Dryland pasture (Weeping lovegrass)

1. Sludge application above the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ limit will increase weeping lovegrass hay yield, crude protein content, and water use efficiency.
2. The ideal sludge application rate to satisfy weeping lovegrass N demand is dynamic and could exceed the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge limit.
3. Under high hay yield production conditions, N supply from double the norm can fully be utilized and such systems are not prone to excessive nitrate leaching.
4. Sludge application according to crop N demand results in the accumulation of total and plant available P in the soil profile.

Turfgrass

1. High sludge surface loading rates well above recommendations based on crop removal
 - a. Are possible without reducing turf growth and quality.
 - b. Do not cause an accumulation of N and P below the active root zone.
 - c. Can minimize soil loss through sod harvesting, and
 - d. Do not cause unacceptably high nitrate and salt leaching.

Nitrogen and phosphorus availability from sludge is influenced by sewage sludge treatment processes, climatic and edaphic factors. At the same time, crop nutrient uptake is dependent on crop type, water availability, nutrient availability, climate, soil type, management practice, and farming intensity. Ideally, specific studies would need to be conducted for each sludge type, cropping system,

management practice, and climate, but this is impractical. Therefore this project seeks to add a mechanistic N sub routine into an existing mechanistic soil water balance and crop growth decision support model. The model is tested for accuracy against the independent data sets collected from various cropping systems described in this study. The predicting capability of the Model appear promising and could be utilized by agronomists and sludge guideline developers as a decision support tool for sustainable use of sludge across various types of cropping systems, agro-ecological zones, and soil types. Phosphorus dynamics from sludge, however, is not modelled in this study due to the complex nature of P availability from sludge treated with Fe and Al salts, a subject that remains contentious in the scientific literature.

The thesis 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 contains a detailed description of nitrogen modelling before concluding with motivation and rationale for the study.

Chapter 3 the Materials and Methods, presents information on field site description, sludge characteristics, cropping systems and treatments, rainfall and irrigation, plant and soil sampling, chemical analysis and statistical methods.

Chapter 4 presents the effect of various sludge application rates on grain and forage yield under dryland (maize) and irrigated conditions (maize-oats rotation). This is followed by measurement of grain and forage N uptake, soil profile N mass balance, residual nitrate and ammonium, and nitrate leaching. Finally the total P mass balance and residual Bray-1P of the cropping systems under investigation is presented.

Chapter 5 covers dryland pasture (weeping lovegrass) and the effect of various sludge application rates on hay yield, crude protein content, N uptake and water use efficiency. Soil profile N mass balance, nitrate leaching, residual nitrate and ammonium, total P mass balance and residual Bray-1P are presented.

Chapter 6 is on turfgrass. This chapter addresses the agronomic benefits and disadvantage of very high sludge application rates and the associated negative environmental impacts such as N and P accumulation below active root zone, nitrate and salt leaching and soil loss through sod harvesting.

Chapter 7 deals with modelling. In this chapter the model is initially calibrated using data collected from field studies and the literature. The calibrated model is tested against independent data sets collected during the study period.

Chapter 8 summarizes important findings and forwards recommendations for further studies.

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CHAPTER 2

LITERATURE REVIEW

2.1 Background information

Population growth in cities and the expansion of industries is resulting in a rapid increase in the volume of waste products that need to be either beneficially used or disposed of in some way. Similar to other countries, South African wastewater treatment plants are under enormous pressure to dispose of or utilize their sludges and effluents in environmentally sustainable ways. The daily total wastewater flow emanating from South African wastewater treatment plants was estimated at 5400 Ml d⁻¹ (Marx et al., 2004). Considering the potential benefits of using sludge on agricultural lands, South Africa, like all other countries, have developed sewage sludge guidelines for beneficial agricultural use (WATER RESEARCH COMMISSION, 1997; Snyman and Herselman, 2006). Despite this, only 28% of the total South African sludge is used for beneficial agricultural purposes.

The first South African sludge guideline (WATER RESEARCH COMMISSION, 1997) was developed based on studies conducted in other countries and it allowed a maximum annual application rate of 8 Mg ha⁻¹ yr⁻¹. The guideline was updated in 2006 (Snyman and Herselman, 2006) based on local sludge-soil mix incubation studies and few single year field scale trials on specific crops and allows a maximum annual application rate of 10 Mg ha⁻¹. These studies showed

the ideal mineralization rates and the potential for nitrate leaching under optimal soil water and temperature conditions (Snyman and Van der Waals, 2004), which is rare under field conditions. Actual sludge mineralization and ammonium nitrification on the field is, however, variable depending on the sludge composition, edaphic and climatic factors. Consequently, due to the lack of sufficient local medium to long term field scale studies, there has been calls for further study by both the former and current South African sewage sludge guidelines. This study investigates the agronomic benefits of various sludge application rates and the possible potential environmental impacts associated with various sludge application rates on four major cropping systems: irrigated maize-oat rotation, dryland maize, dryland pasture, and turfgrass.

In general South African sludges have mean N content of 3.85% and organic matter content of 55% (Snyman and Herselman, 2006). Large fraction of the N in sewage sludge is organic and should first be transformed to inorganic N before it could be available for crop uptake. The processes involved in the transformation of N from organic to inorganic are influenced by factors such as soil pH, soil texture, bulk density, soil water, and soil 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.

The complex interaction between the different factors involved in the nitrogen cycle of the soil – plant system makes it difficult to extrapolate field studies across different agro-climatic conditions and soil types. Moreover, 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. Nevertheless, a validated model could play a significant role in extrapolating field data. Nitrogen subroutine was added to the existing SWB model as part of the study. The potential of the new SWB model with N subroutine to extrapolate to other locations will be tested using data collected from this study.

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 it 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 is generated through the biological treatment of primary sludge followed by clarification. *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 solids 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 quantity. 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 compactability 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 cost.

2.2.2 Nitrogen and sewage sludge

Most of the nitrogen in sewage sludge is in organic form, whereas plants absorb N in the form of NH_4 and NO_3 . The availability of N for plants depends therefore on the rate of mineralization of the sludge (Kelley et al., 1984) and the losses of inorganic nitrogen through volatilization and denitrification (Fig. 2.1).

Sewage sludge treatment and handling affects the characteristics of N in various ways.

- 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 it 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).

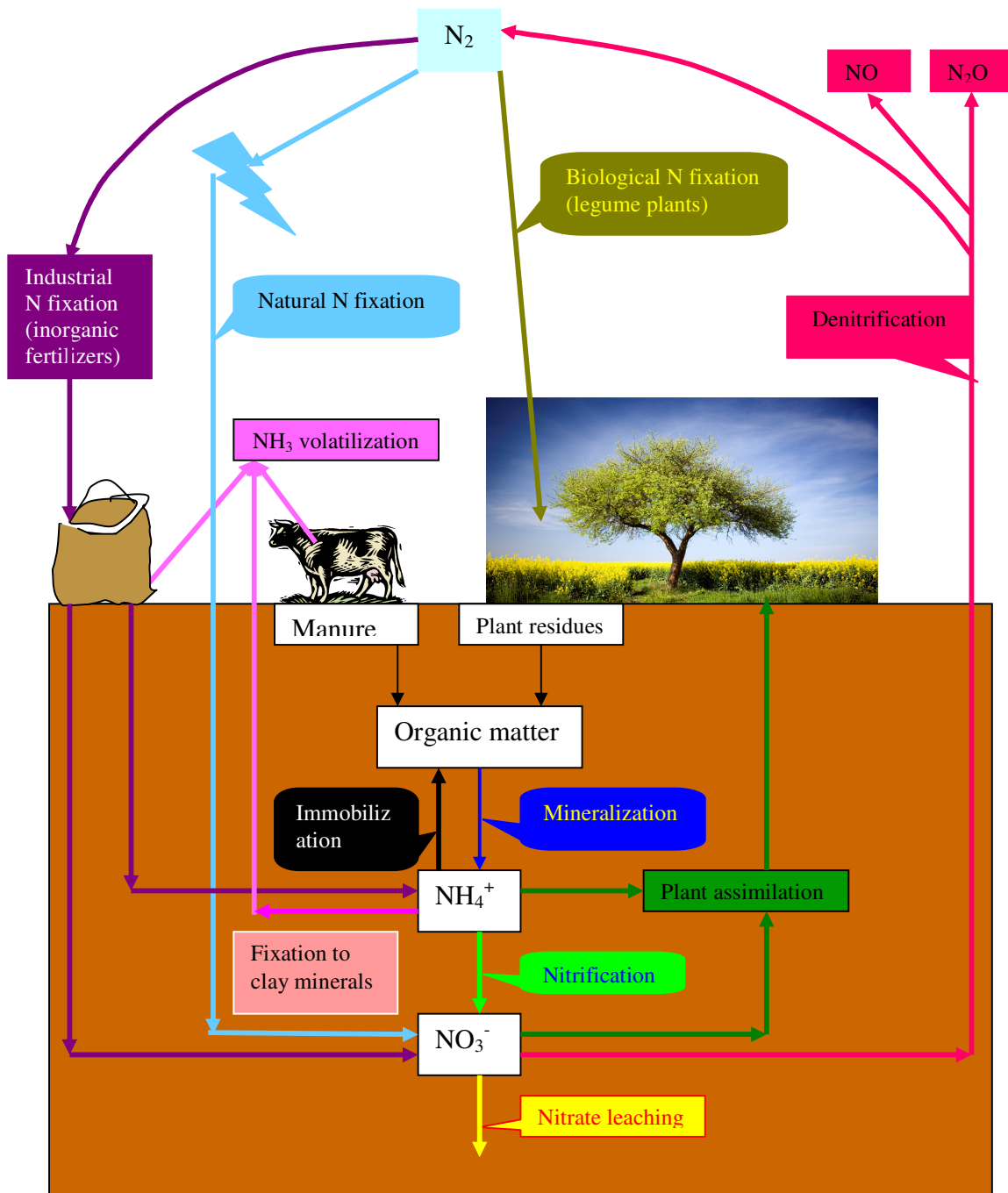


Figure 2.1 Simplified nitrogen cycle in terrestrial plant-soil system.

The organic N pool is usually divided into three pools based on the potential rate of mineralisation. These are 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 its treatment. The mineralisation of each pool is influenced by environmental factors (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).

The variation in mineralisation rates between different sludge types during the year of application is mainly as a result of the size of the rapidly mineralisable organic nitrogen pool, assuming similar environmental and edaphic factors (Henry et al., 1999, Table 2.2). Mineralisation also continues at a slower rate for the following 3 years, after which it is considered negligible (Table 2.3). Results from many long-term studies have shown that four or more years after sludge application, 20 to 50 percent of the organic nitrogen remains in the soil as stable organic matter (Henry et al., 1999).

Organic matter decomposition (nitrogen mineralization) increases 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 mineralisable pool decomposes in two to four weeks. In situations where either the soil water or temperature is not ideal, it extends from two to six months. The slowly

mineralisable and highly resistant pools, however, take months or even years for complete decomposition (Henry et al., 1999).

Table 2.2 Estimates of nitrogen mineralization for various sludge treatment methods in the year of application (percent of initial organic N) (adapted from Henry et al., 1999).

Treatment method		Mineralization (% of initial organic N)
Anaerobically digested	Liquid	20 – 40
	Dewatered	25 – 45
	Heat-dried	25 – 45
Aerobically digested		30 – 50
Lagooned		10 – 30
Lime-stabilized		30 – 60
Composted		0 – 30
Drying bed		15 – 40
Oxidation ditch		30 - 50

Table 2.3 Nitrogen mineralization rate estimate ranges for all types of sludge for years following the application year (percent of the remaining organic N) (adapted from Henry et al., 1999).

Year following application year	Mineralization rate (% of remaining organic nitrogen)
1	5 – 12
2	2 – 6
3	1 - 2

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 vapour pressure 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 dependant upon the sludge treatment process, and whether the sludge is left on the surface or incorporated (Table 2.4). Volatilization is increased by 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).

Ammonia volatilization is high from initially wet soils (below saturation), which are allowed to dry slowly (Fenn and Escarzaga, 1977). This is because evaporative loss of water promotes ammonia volatilization by increasing or maintaining the concentration of ammonia in the soil solution over time (Haynes and Sherlock, 1986). On the other hand, there is no ammonia volatilization from very low soil water content soils (Nelson, 1982). It is also well documented that ammonia volatilization is high from surface application of NH_4^+ compound fertilizers

compared with incorporated fertilizers (Fenn and Kissel, 1976; Quemada et al., 1998).

Table 2.4 Ammonia volatilization rates from Northwest Biosolids applied in western Washington (maritime climate) (adapted from Henry et al., 1999).

Treatment method				Volatilization rate (% of initial ammonia lost)
Anaerobically digested	Liquid (incorporated)			20 – 40
	Dewatered	Incorporated	Agronomic rate	14 – 50
			Double agronomic rate	25 – 49
			Lime amended	45 – 134
			Reduced pH	12 – 28
		Surface applied		51 - 127
Aerobically digested (incorporated)				6 – 12
Lagooned (incorporated)				4 – 20
Lime-stabilized (incorporated)				14 – 22
Drying bed (incorporated)				2 – 5
Oxidation ditch (incorporated)				9 - 23

c) Denitrification

Denitrification is the gaseous loss of nitrogen in the form of nitrous oxide (N_2O) and dinitrogen (N_2), mediated through microbial action (Fig. 2.1) (Delwiche, 1981; Tisdale et al., 1985). The microorganisms responsible for the denitrification need nitrogen in the form of nitrate, soluble organic carbon and water-logged (anaerobic) conditions (Henry et al., 1999). Sewage sludge land application

provides two of these three conditions (nitrate and soluble carbon) thereby making denitrification highly susceptible to wet conditions. It was clearly illustrated by Henry et al. (1999) that denitrification is greater under irrigated systems compared with dryland conditions (Table 2.5).

Table 2.5 Suggested denitrification values for sludges applied to agricultural lands in the Pacific Northwest, USA (adapted from Henry et al., 1999).

Farming system	Denitrification rate (% of inorganic N lost)
Non-irrigated	0
Irrigated	0 -15

2.2.3 Phosphorus and sewage sludge

The most common forms of inorganic P in wastewater aqueous solutions are orthophosphates and polyphosphates. Orthophosphates are available for biological metabolism without further breakdown while polyphosphates usually should 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 1 to 4 mg L⁻¹ is organic. Secondary wastewater treatments can only remove 1 to 2 mg L⁻¹ P (Metcalf and Eddy, 2003). This is in contrast to the soil solution P concentration benchmark of 1 mg L⁻¹ for wastewater discharge to rivers and streams (Sims and Pierzynski, 2000). Therefore, direct discharge of wastewater effluents after secondary treatments without further P removal processes could cause eutrophication of surface waters. Wastewater treatment plants, therefore, use either chemical or biological

phosphorus removal methods to bring down the concentration below the benchmark before they could discharge it to rivers and streams. Chemical precipitation is used to remove the inorganic forms of phosphate by the addition of coagulants and mixing of wastewater and coagulant. The most commonly used coagulants are calcium, aluminium, and iron (Tchobanoglous et al., 2003). In the biological phosphorus removal method, the P accumulating organisms are encouraged to grow and consume P and are subsequently removed (Storm et al., 2004).

Sewage sludge treatment methods and chemical or biological nutrient removal processes influence the availability of P (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 have undergone tertiary treatment is inorganic P (McLaughlin, 1984). Chemicals used in tertiary treatments such as Al or Fe salts, decrease the plant available P fraction (Elliott et al., 2002; Häni, et al., 1981; Kyle and McClintock, 1995). Therefore, the percentage of total P found in the easily soluble fraction is higher in sludges not treated with Fe or Fe + Al. The application of lime to the Fe or Fe + Al treated sludges, however, increased the concentration of the easily soluble P fraction (Penn and Sims, 2002).

In work done by Penn and Sims (2002), a significant increase in total soil P was observed in sludge amended silty clay loam and sandy loam soils compared with their control (no sludge). The labile forms of soil P (M3-P, M1-P, FeO-P, and

water soluble P) were especially high for soils that received biological nutrient removal sludge, compared to the control and the soils receiving Fe and lime treated sludge treatments. Soils that received sludge treated with Fe and Al salts, slightly increased the labile soil P, but soils which received sludge treated with Fe and lime resulted in an intermediate labile P increment (Penn and Sims, 2002).

Long term studies conducted on wastewater applied to a sandy soil for 30 to 50 years, showed an increase in the inorganic form of P, that was dominated by Al-bound phosphate (Beek et al., 1977). Other studies conducted by Soon and Bates (1982) on soils amended with Ca, Al, and Fe treated sludges show that significant increase in soil P was in the Al- and Fe-bound P fraction, irrespective of the sludge type used. Therefore, it is of the utmost importance to consider the type of sludge used when quantifying sludge application rates. This can help to optimise crop harvests by minimising the environmental impacts.

2.2.4 Utilising sewage sludge on agricultural lands

Beneficial use of sewage sludge on agricultural lands is a common practice in various regions of the world. This can be clearly observed from Table 2.6 which presents the percentage of sewage sludge applied to agricultural lands for 15 European Countries, USA, Australia, and South Africa. Beneficial use accounts for 28% of the total sludge produced from South African wastewater treatment plants, which places South Africa 11th in the list of Table 2.6. The remaining 72% is accounted for through land fill (3%), non beneficial land application (47%),

accumulation at plant (20%), and unspecified (2%) (Du Preez, et al. 2000). A combination of factors including low sludge quality due to pathogen and pollutants as well as lack of sufficient information on the beneficial use of sludge contributes to the relatively low sludge usage in agriculture.

Table 2.6 Annual sewage sludge produced and the percentage applied to agricultural lands for 15 European Countries and USA. (USA and EU (AEA Technology Environment, 2002); Australia (Priestley, 1991); South Africa (Lötter and Pitman, 1997))

Country	Annual dry sludge production (Mega tons)	% of sludge produced applied to agricultural lands
Austria	0.320	13
Belgium	0.075	31
Denmark	0.130	37
France	0.700	50
Germany	2.500	25
Greece	0.015	3
Ireland	0.024	28
Italy	0.800	34
Luxembourg	0.015	81
Holland	0.282	44
Portugal	0.200	80
Spain	0.280	10
Sweden	0.180	45
United Kingdom	1.107	55
Switzerland	0.215	50
USA	6.900	41
Australia	0.300	9
South Africa	0.310	28

The major reason for applying sludge is as a source of plant nutrients. Optimal agricultural production with minimal environmental impacts can be achieved through proper sludge application and management practices. These practices include sludge application according to cropping system nutrient demand and proper sludge application timing and methods. Since most of the nitrogen in

sewage sludge is bound in organic form, it is released slowly compared to inorganic fertilizers, as can be seen in Figure 2.2. The organic matter in sewage sludge is also a good source of energy for soil *micro flora* (Muse et al., 1991).

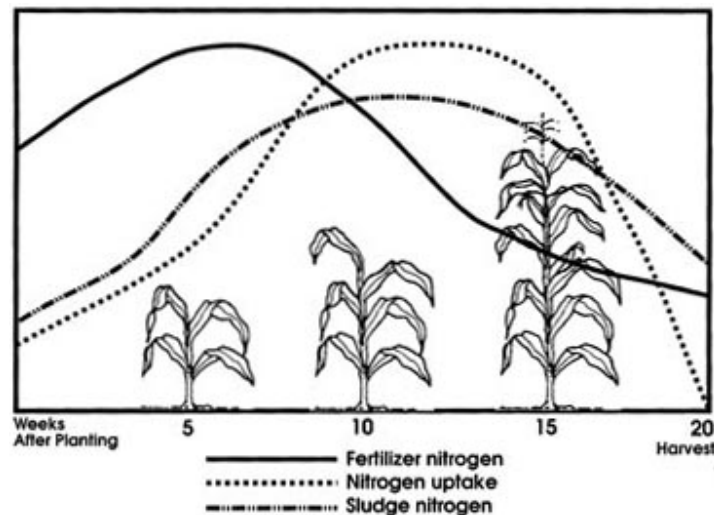


Figure 2.2 Nitrogen requirement of maize during the growing season and nitrogen availability from fertilizer compared with sludge (adapted from Muse et al., 1991).

The organic matter in sewage sludge applied to agricultural lands also improves the soil's *physical characteristics* and structure, thereby improving soil porosity, stability of aggregates and water retention (Ojeda et al., 2007). It can play 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 as a source of macro and micro nutrients for plants (Mininni and Santori, 1987).

2.2.5 Sludge application rates on agricultural lands

Generally, application rates of sewage sludge to agricultural lands are dictated by the nitrogen content of the sludge (Mile and Graveland, 1972; Dotson, 1973). In most US States, the sewage sludge application rate to agricultural land is based on the N requirements of crops and the concentrations and loading 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).

Previous research by Kelling et al., (1977); Pierzynski (1994), Peterson et al. (1994) and Maguire et al. (2000a,b) indicate 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 type of treatment and processes which the sludge went through in the water care works (Kyle and McClintock, 1995; Maguire et al., 2001; Jokinen, 1990; Soon et al., 1978a; Kirkham, 1982; McCoy et al., 1986; Frossard et al., 1996).

The province of Ontario, Canada, also monitors the soil for possible P accumulation at sludge disposal sites. They recommend that bicarbonate extractable P levels should not exceed 60 kg P ha^{-1} , the concern being P contamination of waters by soil erosion or runoff (McLaughlin, 1984). Sims and Gartley (1996), classified soil P status based on M1-P analyses as follows: 0 –12

mg kg^{-1} = low, 13 –24 mg kg^{-1} = medium, 25 – 50 mg kg^{-1} = optimum and > 50 mg kg^{-1} = excessive. Considering such a classification for agricultural lands receiving sludge could minimise ground and surface water contamination through leaching and runoff, thus improving the sustainability of sludge use on agricultural lands.

Similar to most US States, the current South African guideline is based on the nitrogen needs of crops (sludge suitable for agricultural use) (Snyman and Herselman, 2006). However, unlike the US guideline, it does not consider phosphorus based management.

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, time, 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 should preferably be applied in split rather than a single big application (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 sludges are 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).

Consequences of very high sludge applications

Excess nutrients are detrimental to plant growth (Brady 1974), and pollute water bodies (Kumm, 1976; Keeney, 1973). As water moves over the soil surface it carries along decomposing organic matter, fertilizers, pesticides, as well as

sediments. The nutrient loads of such runoff waters result in nutrient enrichment of water bodies such as lakes and dams. This nutrient enrichment is the cause of undesirable proliferation of aquatic plants (eutrophication). Nitrogen in water bodies is oxidized into nitrite and nitrate, which depletes levels of dissolved oxygen (Neal et al., 2002; Sparks, 1995) resulting in the death of fish and other aquatic creatures (Cameron and Haynes, 1986).

Water that percolates beyond the root zone may also carry nutrients in a dissolved form, thereby contaminating groundwater. Contaminated water presents a hazard for humans and animals, as well as for plants. A consequence of groundwater nitrate pollution is methemoglobinemia or blue baby syndrome, which can cause death in infants (Croll and Hayes, 1988).

2.2.6 Classification of sludge for use on agricultural lands

Waste products from cities and industrial areas contain pathogens, toxic elements and organic contaminants which can pose serious health hazards. Some of the pathogens that can possibly be found in sewage sludge before its treatment and the diseases they cause are presented in Table 2.7 (U.S. EPA, 1995).

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).

Table 2.7 A few of the pathogens that could potentially be present in municipal sewage sludge and the diseases or symptoms they cause (adapted from U.S. EPA, 1995).

Pathogen	Disease/symptoms
Salmonella sp.	Salmonellosis (food poisoning), Typhoid fever
Escherichia coli	Gastroenteritis
Shigella sp.	Bacillary dysentery, severe gastroenteritis
Hepatitis A virus	Infectious hepatitis
Echoviruses	Meningitis, paralysis, encephalitis, fever, “flu-like” symptoms, diarrhoea, etc.
Entamoeba histolytica	Amoebic dysentery
Giardia lamblia	Diarrhoea, abdominal cramps, weight loss
Ascaris sp.	Digestive and nutritional disturbances, abdominal pain, vomiting, restlessness, coughing, chest pain, and fever
Trichuris trichiura	Abdominal pain, diarrhoea, anaemia, weight loss
Toxocara canis	Fever, muscle aches, neurological symptoms

The South African Sewage Sludge Guideline classifies sludge under microbiological, stability, and pollutant classes (Snyman and Herselman, 2006). Tables 2.8 and 2.9 present the microbiological and pollutant classes of the current South African guideline, compared with those of the USA and EU and Table 2.10 presents the stability classes of the current South African guideline.

Table 2.8 South African preliminary classification: microbiological class (Snyman and Herselman, 2006) compared with the USA (US EPA, 1995); (US EPA, 2003).

Organism	South African microbiological class				US microbiological class	
	A	B	C		A	B
	All three samples comply	Two of three samples comply	Sample that failed doesn't exceed the following	One or more of the samples exceed	Representative sample at time of use or disposal	Geometric mean of 7 samples at time of use/disposal
Faecal coliforms (CFU/g _{dry})	< 1000	< 1 × 10 ⁶	< 1 × 10 ⁷	> 1 × 10 ⁷	< 1000	< 2 × 10 ⁶
Helminth ova (Total viable ova/g _{dry})	< 0.25 or (one viable ova/4g _{dry})	< 1	4	> 4	< 1	N/A
Salmonella sp. (CFU/4g _{dry})	N/A	N/A	N/A	N/A	< 3	N/A
Enteric viruses (PFU/4g _{dry})	N/A	N/A	N/A	N/A	< 1	N/A

CFU - colony forming units PFU - plaque-forming units N/A – not applicable

The current South African sewage sludge guideline has similar pollutant class ranges with US guideline for Cr, Cu, Pb, Ni, Zn, As, Cd, and Hg. The EU guideline is, however, very conservative compared to both the South African and US guidelines. The US guideline adds Se to the list of pollutants, which is not considered in either the South African or EU guidelines. The South African and US microbiological class “A” ranges for faecal coliforms and total viable helminth ova are similar. The US guideline, however, adds salmonella and enteric viruses to the list.

Table 2.9 South African preliminary classification: pollutant class (Snyman and Herselman, 2006) compared with the US land application pollutant limits (US EPA, 1995) and proposed EU maximum permissible limits in sludge in mg kg^{-1} (IC Consultants, 2001).

Element	Pollutant limits (mg kg^{-1})					
	South African pollutant class			US land application pollutant limit		EU proposed maximum limit
	a	b	c	Concentration in bulk and bagged sewage sludge	Ceiling concentration in all sludge that is land applied	
As	<40	40 – 75	>75	41	75	N/A
Cd	<40	40 – 85	>85	39	85	10
Cr	<1200	1200 – 3000	>3000	1200	3000	1000
Cu	<1500	1500 – 4300	>4300	1500	4300	1000
Pb	<300	300 – 840	>840	300	840	750
Hg	<15	15 – 55	>55	17	57	10
Ni	<420	420	>420	420	420	300
Zn	<2800	2800 - 7500	>7500	2800	7500	2500
Se	N/A	N/A	N/A	36	100	N/A

Bagged sewage sludge is sold or given away in a bag or other container for application to land (US EPA, 1995)

N/A - not applicable

The current South African sewage sludge guideline considers all three categories (*microbiological*, *stability*, and *pollutant* classes), together to assess the appropriateness of using a given type of sludge for agricultural purposes. 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).

Table 2.10 South African preliminary classification: Stability class (Snyman and Herselman, 2006).

Stability class	1	2	3
	Plan/design to comply with one of the options listed below on a 90 percentile basis.	Plan/design to comply with one of the options listed below on a 75 percentile basis.	No stabilisation or vector attraction reduction options required.
Vector attraction reduction options (applicable to stability class 1 and 2 only)			
Option 1	Reduce the mass of volatile solids by a minimum of 38 percent		
Option 2	Demonstrate vector attraction reduction with additional anaerobic digestion in a bench-scale unit.		
Option 3	Demonstrate vector attraction reduction with additional aerobic digestion in a bench-scale unit.		
Option 4	Meet a specific oxygen uptake rate for aerobically treated sludge		
Option 5	Use aerobic processes at a temperature greater than 40°C (average temperature 45°C) for 14 days or longer (eg. during sludge composting)		
Option 6	Add alkaline material to raise the pH under specific conditions		
Option 7	Reduce moisture content of sludge that do not contain unstabilised solids (from treatment processes other than primary treatment) to at least 75 percent solids		
Option 8	Reduce moisture content of sludge with unstabilized solids to at least 90 percent solids		
Option 9	Inject sludge beneath the soil surface within a specified time, depending on the level of pathogen treatment		
Option 10	Incorporate sludge applied to or placed on the surface of the land within specified time periods after application to or placement on the surface of the land		

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 conducted, however, class “3” can not be used for agricultural application.

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, however, class “c” can not be used for agricultural purposes (Snyman and Herselman, 2006).

Table 2.11 Permissible utilisation of sludge in agricultural applications based on the South African sludge classification system (adapted from Snyman and Herselman, 2006)

South African sludge classification		Is agricultural use an option?	Any additional restrictions and requirements?	Notes
Microbiological class	A	Yes	No	Could potentially be used as a saleable product.
	B	Qualified yes	Yes	General restrictions/ requirements apply.
	C	May be	Yes	Only permissible if stability class 1 or 2 is achieved. (General restrictions/ requirements apply)
Stability class	1	Yes	No	Could potentially be used as a saleable product.
	2	Qualified yes	Yes	Additional management actions required to encourage compliance with class 1.
	3	No	Not applicable	Stability class 3 may not be used in agricultural practices.
Pollutant class	a	Yes	No	Could potentially be used as a saleable product.
	b	Qualified yes	Yes	If the soil analyses is favourable.
	c	No	Not applicable	Pollutant class c may not be used in agricultural practices.

2.2.7 Experiences with sewage sludge on cropping systems

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 some cases the response was non linear, with smaller yield increases at high application rates (Soon et al., 1978b; Binder et al., 2006). At very high application rates, grain yield has been reduced by salt toxicity (Cunningham et al., 1975).

The optimal sludge application rate for nitrogen depends on the availability of water. Soon et al. (1978b), found no crop response for sludge rates containing above 200 kg N ha⁻¹ under dryland conditions, both in a clay loam and loamy sand soils. In those studies the mean above ground biomass N uptake was 85 kg ha⁻¹. In other studies conducted by Binder et al. (2002) under irrigated conditions, however, maize grain yield increased significantly with increase in sludge application rates which supplied N up until 400 kg ha⁻¹. The mean above ground biomass N uptake in this irrigated system was just above 300 kg ha⁻¹.

Significant accumulation of nitrate in the top 0.75 m soil layer was reported by Soon et al., (1978b) following three years of sludge application at rates of 400 kg N ha⁻¹ under dryland maize production. In other studies conducted by Binder et al. (2002) under irrigation, however, there was no significant accumulation of N at this rate of 400 kg N ha⁻¹ mainly due to the higher nitrogen use efficiency of similar crop under irrigation. The latter study, however, reported a significant nitrate accumulation when sludge was applied at rates of 950 kg N ha⁻¹ due to the decline in the nitrogen use efficiency. The accumulation of nitrate in the soil

profile could lead to ground water pollution especially in the beginning of the season if heavy rain falls before the active crop root development.

Since a large fraction of N in sludge is organic pool, there may be significant carry over effects in the subsequent years after application (Cogger et al., 2001; Binder et al., 2002). It is therefore important to quantifying the amount of carry over effects each year before new sludge applications.

The yield of small grain cereals such as wheat (*Triticum aestivum*) and triticale (*Tritico-secale*) increased with sludge application (Oloya and Tagirwa, 1996; Cooper, 2005; Lavado et al., 2006) up to a certain point. After this, the effects of high salt dominated (Simeoni et al. 1984).

b. Pastures

It is well documented that sludge application improves dry matter yield of pasture grasses (Kelling et al., 1977; Soon et al., 1978b; Kiemenec et al., 1987; Michalk et al., 1996; Sullivan et al., 1997; Zebarth et al., 2000; Cogger et al., 2001), although responses are dependant on climate and management regimes. For instance, tall fescue (*Festuca arundinacea Schreb.*) dry matter yield increased significantly with increase in sludge application up to rates of 20 Mg ha⁻¹ (905 kg N ha⁻¹) in an area receiving 1020 mm rainfall and supplemented with irrigation as required (Cogger et al., 2001) during summer season. 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 Mg ha^{-1} ($1064 \text{ kg N ha}^{-1}$) in a 1232 mm rainfall area. Kelling et al. (1977) reported positive rye grass (*Secale cereale* L.) response for sludge rates of up to 7.5 Mg ha^{-1} in a silty loam soil at Arlington but up to 15 Mg ha^{-1} in a sandy loam soil at Janesville, Wisconsin, USA. He argued that this was most probably due to the combination of factors including the better initial nutrient status of the soil in Arlington.

Although high sludge applications of $20 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ ($>800 \text{ kg N ha}^{-1}$) significantly improved tall fescue dry matter yield, this rate was associated with the accumulation of NO_3 across time (Cogger et al., 2001). Surprisingly, similar findings were reported from earlier studies conducted on Brome grass in a similar soil type (Soon et al., 1978b). According to this study, nitrate accumulated in the soil profile to a depth of 0.9 m at higher sludge application rates supplying 800 kg N ha^{-1} and above. Other negative environmental impacts of very high sludge application rates are the accumulation of plant available P above the concentration required for optimal crop production of $20\text{-}50 \text{ mg kg}^{-1}$. This has been reported in studies conducted by Michalk et al. (1996), where the plant available P exceeded 300 mg kg^{-1} on the soil surface. Such high rates could definitely pose risk to surface water bodies through runoff in solution form (Sims and Pierzynski, 2000).

In addition to the obvious negative environmental impacts associated with high sludge loading rates, there are also some concerns related to fodder quality

(nitrate concentration in grass). These concerns are extended to the health issues of workers harvesting forage and animals consuming the forage. Studies conducted by Soon et al (1978b) showed that high sludge loading rates that supplied $1600 \text{ kg N ha}^{-1}$ increased the concentration of $\text{NO}_3\text{-N}$ in the grass to hazardous levels for livestock consumption ($>3 \text{ mg g}^{-1}$). Regarding the health issue 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.

c. Lawn sod production

The application of composted sludge on various turfgrass species planted to different soil types has improved turf growth and quality. For instance the incorporation of composted biosolid in a disturbed urban soil facilitated the establishment rate 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 increases water infiltration rate compared with an unamended control (Landschoot and McNitt, 1994; Cheng et al., 2007).

2.3 Nitrogen modelling

Sewage sludge treatment and handling processes affect the characteristics and availability of nutrients from sludge. At the same time, crop nutrient uptake is dependent on crop type, water availability, nutrient availability, climate, soil type, management practice, and farming intensity. Therefore studies conducted in a specific crop under specific conditions can only serve to produce guidelines for that specific situation.

Direct application of such guidelines to different situations can compromise both the environment and crop yield. A validated mechanistic model can play a significant role to extrapolate results responsibly, by conducting scenario simulations for various types of crops grown on various soil types and under different climatic regimes. This will reduce the requirements of performing costly long-term site specific field trials and information generated can be used in the development of updated site specific guidelines for sludge application where needed.

Modelling the dynamics of nitrogen in the soil system dates back to an early 70s (Shaffer et al., 2001). The first integrated soil system N models include that of by

Dutt et al. (1972) in the USA and Beek and Frissel (1973) in 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 oriented tools (Shaffer et al., 2001).

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

This project required an N model that could simulate various irrigation management practices i.e. irrigating the profile to field capacity, leaving room for rain, or irrigating with a leaching fraction, all of which are critical for long-term scenarios. The reason for this is that N management, salt management and irrigation are inextricably linked. There is also a need to explore trade-offs between nutrient management, optimal yields and pollution such as through leaching. In addition there was no generic 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 a nitrogen module to the SWB model - a crop growth / irrigation scheduling model that has already been validated for various crops and grasses under irrigation and dryland conditions. 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.

Generally, mechanistic models provide a better understanding of the nitrogen cycle by mechanistically describing the major processes involved in the N cycle such as volatilization, mineralization, immobilization, nitrification, denitrification, crop uptake, and leaching. The main difference between models is thus the difference in the degree of sophistication of these major processes (Frissel and Van Veen, 1982). Some of the models are used for short term seasonal soil N monitoring while others for predicting long-term N dynamics of decades and longer.

Shaffer et al. (2001) have detailed the differences in the major nitrogen transformation processes between various models. They also elaborate about similarities between models regarding the major nitrogen transformation processes: mineralization, immobilization, nitrification, and volatilization. Most nitrogen transformation processes in the soil profile including mineralization,

nitrification, and denitrification are quantified using first order kinetics (Ma and Shaffer, 2001). The rate constants are then modified based on the soil temperature, water content, and other factors.

2.3.1 Mineralization

The mineralization of organic substances and residues is greatly influenced by the C:N ratio of the material which influences the net mineralization or immobilization rate (Vinten and Smith, 1993). Carbon to nitrogen ratio (C:N ratio) is an approximation of the important parameter energy:nitrogen (E:N) ratio. Low C:N ratio organic residues have faster mineralization rates and result in net mineralization. According to Haynes (1986a), the lowest C:N ratio is 8:1 and is found in microbial biomass. The C:N ratio of clover, beans and lucerne is in the range of 13:1 to 23:1 where as cereal straw and other mature plant stalks have 60:1 to 80:1 C:N ratios. The C:N ratios of some plant materials with N free lignin and residual substances from many peat soils with high C:N ratios are poor sources of energy for most micro organisms (Jansson and Persson, 1982).

Various organic materials do have variable decomposition rates depending on their resistance to microbial activity. Consequently many models simulate this difference by classifying the organic matter into pools. This classification, as explained by Ma and Shaffer (2001), is based on physical (mobile or immobile carbon and nitrogen), chemical (organic with different C:N ratios and

compositions or inorganic) and biological characteristics (capable of transforming to other forms by various types of microbes at different rates).

Most multiple pool N models include soil microbial biomass pool but only the RZWQM model simulates the growth and death of microbial populations (Ma and Shaffer, 2001). Models differ in the approach they follow in simulating organic matter decomposition. For instance models such as NTRM use regression equation. On the other hand, models such as RZWQM, CropSyst, NLEAP, CERES, EPIC, GLEAMS, LEACHM, CENTURY, SOILN, ANIMO, DAISY, SUNDIAL, and CANDY use first order kinetics. Other models including NCSOIL use Monod kinetics (Ma and Shaffer, 2001).

Decomposition rates for most of the models such as NLEAP, CROPSYST, RZWQM, CERES, EPIC, GLEAMS, CENTURY, NCSOIL, LEACHM, SOILN, ANIMO, DAISY, SUNDIAL, and CANDY are adjusted after the soil temperature and soil water content. Some of these models consider additional factors which do affect organic decomposition rates. These factors include soil pH, soil ionic strength, soil microbial population, C:N ratio, bulk density, soil texture, and lignin content. Details of the mineralization functions used by various models and the additional factors used by each model could be found from Ma and Shaffer (2001) and McGechan and Wu (2001).

The N model in SWB which is adopted from Cropsyst partitions organic residue into three pools (fast, slow and lignified). The model simulates carbon decomposition from each pool using Eq. 1. Net mineralization is then computed using Eq. 2. The model, however, estimates soil organic matter decomposition separately using Eq. 3.

$$\text{Carbon decomposition} = C_{\text{res}} * CF * (1 - \exp(-k * TF)) * WF \quad [1]$$

Where

C_{res} – is carbon mass in residue or manure

CF – is residue or manure contact fraction (from literature)

k – is residue or manure decomposition constant

TF – is soil temperature function (computed using Eq. 4)

WF – is soil water function (computed using Eqs. 5).

$$\text{Net N mineralization} = (1 / \text{CN ratio}_{\text{decomp}} - C_{\text{trans}} / \text{CNratio}_{\text{pool}}) * C_{\text{decomposition}} \quad [2]$$

Where

$\text{CN ratio}_{\text{decomp}}$ – is CN ratio of the decomposing pool

C_{trans} – is carbon fraction transferred from the decomposing pool to another pool.

$\text{CNratio}_{\text{pool}}$ – is the CN ration of the receiving pool.

$$\text{Carbon decomposition} = C \text{ mass} * \min(TF, WF) * \text{Tillage}_{\text{factor}} * k \quad [3]$$

Where

C mass - is pool carbon mass

min (TF, WF) - is the minimum of the two (soil temperature and water functions)

Tillage_{factor} - is tillage decomposition rate adjustment factor (computed by the model (Stöckle et al., 2003))

k - is carbon decomposition constant.

Temperature and water functions are estimated by the model using Eqs. 4 and 5.

$$TF = ((T - T_{min}) ^ (Q) * (T_{max} - T)) / ((T_{opt} - T_{min}) ^ (Q) * (T_{max} - T_{opt})) \quad [4]$$

Where

T - is real time soil temperature

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

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

T_{opt} – optimum temperature for microbial activity (35 °C)

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

Water function (WF) is computed differently for different soil water filled porosity ranges. The model partitions the water filled porosity (WFP) into three groups and computes the WF for each group separately as presented in Eqs. 5 - 7.

$$\text{If } 0.1 \leq \text{WFP} < 0.5 \quad \text{WF} = ((\text{WFP} - \text{WFP}_{\min}) / (\text{WFP}_{\text{low}} - \text{WFP}_{\min})) \quad (5)$$

$$\text{If } 0.5 \leq \text{WFP} \leq 0.7 \quad \text{WF} = 1 \quad (6)$$

$$\text{If } 0.7 \leq \text{WFP} \leq 1 \quad \text{WF} = \text{WF}_{\text{sat}} + (1 - \text{WF}_{\text{sat}}) * ((1 - \text{WFP}) / (1 - \text{WFP}_{\text{high}}))^2 \quad (7)$$

Where

WFP – is the water filled porosity

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)

2.3.2 Immobilization

Immobilization is the net incorporation of mineral nitrogen, mainly NH_4^+ , into organic forms (microbial tissue) during the decomposition process (Vinten and Smith, 1993). It is a process which works in opposite direction to mineralization. The decomposition of residues with high C:N ratio results in a negative net residue nitrogen mineralization or 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 mineralizable nitrogen. Most models set a C:N ratio beyond which immobilization takes place. This ratio is set to 24 by CropSyst model.

A periodic decline in the amount of plant available nitrogen is observed after the incorporation and decomposition of cereal straw and other mature plant stalks due to immobilization (Powlson et al., 1987). Nevertheless, mineralization is improved with time after the continuous incorporation of organic residues.

Nitrogen immobilization is simulated using residue C:N ratio (NTRM and NLEAP), decomposition rates (CERES and EPIC), or both decomposition rate and C:N ratios (SOILN, ANIMO, DAISY, SUNDIAL, CropSyst, and CANDY). During immobilization, some models such as NCSOIL, NLEAP, and CROPSYST take up inorganic N from the NH_4 and NO_3 pools, whereas other models such as EPIC immobilize only NO_3 (Ma and Shaffer, 2001).

Cropsyst model estimates immobilization only when the net mineralization is less than zero using Eq. 8.

$$\text{Nitrogen immobilization} = -\text{Nitrogen net mineralization} \quad [8]$$

2.3.3 Nitrification

Nitrification is the oxidation of ammonium ion or ammonia to nitrate or nitrite ion (Delwiche, 1981). Howard-Williams and Downes (1993) classified the processes responsible for nitrification into 2: namely autotrophic and heterotrophic nitrification. Autotrophic nitrification is the process of transforming ammonium or ammonia to nitrite by a group of bacteria called *Nitrosomonas* sp. and

Nitrosolobus sp. followed by the transformation of nitrite to nitrate by a separate group of bacteria called *Nitrobacter* sp. Whereas, heterotrophic nitrification is the oxidation of reduced organic nitrogen compounds to oxidised nitrogen species.

The nitrification process is described by three types of kinetics: Monod, first-order, and zero order models (Hansen et al., 1995). Models such as RZWQM, EPIC, LEACHM, SOILN, SUNDIAL and CROPSYST follow first-order kinetics to simulate nitrification. However, models such as GLEAMS, NLEAP and NCSOIL use zero-order kinetics. On the other hand, NTRM uses regression equation to estimate the rate of nitrification (Shaffer, 1985) whereas CERES, DAISY, and CANDY models follow the Michaelis-Menten equation (Godwin and Singh, 1998; Godwin and Jones, 1991).

Factors which influence the rate of nitrification include soil water, soil temperature, soil pH. Models such as NLEAP, RZWQM, CERES, EPIC, GLEAMS, NCSWAP, LEACHM, SOILN, and SUNDIAL consider the impact of soil water and temperature on the rate of nitrification (Shaffer et al., 1991; MA et al., 2001; Godwin and Singh, 1998; Williams, 1995; Knisel, 1993; Molina et al., 1983; Hutson, 2000; Eckersten et al., 1998; Bradbury et al., 1993; Stöckle and Nelson, 2000). RZWQM, CERES, EPIC, SOILN, and Cropsyst consider one additional factor, soil pH. Models such as RZWQM consider more factors which could impact the rate of nitrification such as oxygen concentration, and population of autotrophs (Ma et al., 2001).

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 estimates nitrification as zero. However, if the nitrate to ammonium ratio exceeds 8, the model uses Eq. 9 to estimate the amount of nitrification.

$$N_{\text{nitrified}} = (NH_4 - NO_3 / 8) * (1 - \exp(-0.2 * pHF * TF)) * MF \quad (9)$$

Where

$N_{\text{nitrified}}$ – 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)

MF – is the soil water function (Eq. 5 – 6 and Eq. 10)

pHF – is the soil pH function (estimated using Eq. 11)

$$\text{If } 0.7 \leq WFP \leq 1 \quad WF = 1 * ((1 - WFP) / (0.3)) ^ 2 \quad (10)$$

$$pHF = (pH - pH_{\min}) / (pH_{\max} - pH_{\min}) \quad (11)$$

where

pHF - is pH function

pH – is the measured soil pH

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

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

2.3.4 Denitrification

According to Ma and Shaffer (2001), the simulation of denitrification process 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. Similar to mineralization and nitrification, models describe the denitrification process as Michaelis-Menten, zero order, or first order kinetics (Hansen et al., 1995). Denitrification is simulated as zero order by SOILN and first order by models such as NLEAP, RZWQM, CERES, EPIC, GLEAMS (Hansen et al., 1995), and Cropsyst. LEACHM model simulates denitrification using Michaelis-Menten kinetics (Hutson, 2000). Models such as NTRM, however, does not simulate denitrification (Ma and Shaffer, 2001).

The effects of both soil temperature and soil water on the rate of denitrification are considered by models such as NLEAP, RZWQM, CERES, EPIC, GLEAMS, LEACHM, SOILN, CANDY, and Cropsyst (Ma and Shaffer, 2001; Stöckle and Nelson, 2000). Some of the models consider additional factors such as water extractable soil carbon content (CERES), percentage of soil organic carbon

content (EPIC and GLEAMS), and soil pH, soil carbon substrate, and population of denitrifiers (RZWQM) (Ma and Shaffer, 2001).

When estimating denitrification, Cropsyst first estimates respiration response function (RRF) (algorithm too long to present) based on the soil temperature function (TF) Eq. 4. The model then presents forward the following 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, secondly the denitrification soil water function (DWF) (Eqs. 12-14) must exceed zero. Finally, the denitrified N mass is computed using Eq. 15.

$$\text{If } WFP < WFP_{NR} \quad DWF = 0 \quad (12)$$

$$\text{If } WFP_{NR} < WFP < WFP_{infl} \quad DWF = 0.9 - 0.9 * (WFP_{infl} - WFP) / (WFP_{infl} - WFP_{NR}) \quad (13)$$

$$\text{If } WFP_{infl} < WFP \quad DWF = 1 - 0.1 * (1 - WFP) / (1 - WFP_{infl}) \quad (14)$$

Where

WFP – water filled porosity

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

DWF – denitrification soil water function

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

$$N_{denitrified} = k * soil_{mass} * \text{minimum} (DWF, RRF, (NO_{3-dry}/(NO_{3-dry}+c)) \quad (15)$$

Where

$N_{\text{denitrified}}$ – denitrified N mass

k – potential denitrification constant (0.000032)

$\text{soil}_{\text{mass}}$ – dry soil mass of the profile (layer)

RRF – respiratory response function

$\text{NO}_{3\text{-dry}}$ – nitrate concentration of the dry soil

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

2.3.5 Ammonia volatilization

The sources of NH_3 for volatilization include inorganic fertilizers applied to the soil in the form of NH_4^+ compounds and the decomposition of organic nitrogenous sources. The rate of ammonia volatilization is affected by factors such as soil pH, soil temperature, soil CEC, soil water content, wind speed, fertilizer application method (surface, incorporation), and pH of fertilizer (Nelson, 1982; Freney et al., 1983). Ammonia volatilization increases with increase in soil and fertilizer pH, wind speed, and soil temperature (until 45°C) (Nelson, 1982).

Ammonia volatilization is simulated by most models as first order kinetics (Ma and Shaffer, 2001). Some models such as NTRM and NCSOIL, however, do not simulate ammonia volatilization (Ma and Shaffer, 2001). Different models consider different factors to adjust ammonia volatilization rate. For instance, NLEAP, EPIC, GLEAMS, Cropsyst, and RZWQM use soil temperature. Some of these models such as NLEAP (Shaffer et al., 1991) consider additional factors

such as residue cover, soil cation exchange capacity, and fertilizer application method (surface application or incorporation). RZQWM and Cropsyst consider additional factors such as wind speed and ammonium pressure gradient between soil and air.

Ammonia volatilization could take place from the inorganic fertilizer, biosolids, plant residue, or soil ammonium (McGechan and Wu, 2001). Most models (Ma and Shaffer, 2001) including Cropsyst simulate volatilization from the top soil layer. Similar to most N models, Cropsyst does not estimate ammonia volatilization from soil ammonium, because generally soils and crops are not considered as the major source of NH_3 (Bussink and Oenema, 1998). Ammonia loss from crops is estimated to be below $2 \text{ kg NH}_3\text{-N ha}^{-1} \text{ yr}^{-1}$ (Holtan-Hartwig and Bøckman, 1994).

Cropsyst simulates ammonia volatilization from inorganic fertilizer (liquid or solid) and organic fertilizers (liquid or solid). Cropsyst primarily estimates ammonia N mass available for volatilization ($\text{NH}_4\text{-AV}$) as a function of soil-CEC and fertilization application method (broadcast or incorporated) before estimating ammonia volatilization. The model estimates ammonia volatilization only if the $\text{NH}_4\text{-AV}$ is greater than zero. If the $\text{NH}_4\text{-AV}$ is greater than zero, Cropsyst estimates ammonia volatilization using Eq. 16.

$$\text{NH}_3\text{-volatilized} = ((K_H * \text{NH}_3\text{-conc})/p_{\text{atm}}) * \text{TTC} \quad (16)$$

Where

NH_3 -volatilized – is volatilized ammonia mass

K_H – is Henrys constant (estimated according to the air temperature)

NH_3 -conc – is ammonium concentration

p_{atm} – is atmospheric pressure

TTC – is turbulent transfer coefficient

2.3.6 Crop nitrogen uptake

Nitrogen is mostly taken up by plants in the form of NO_3^- during a convective flow of soil water to plants in response to transpiration from the canopy (Olson and Kurtz, 1982). This is mainly because of the low attraction of NO_3^- to the soil colloid compared with NH_4^+ which has a higher force of attraction to the soil colloids. This low force of attraction between the NO_3^- and soil colloids makes it easier for the NO_3^- to be carried by mass flow to the plant roots. In addition, the rapid nitrification of NH_4^+ in most soil conditions play a significant role in minimizing its availability for crop uptake (Haynes, 1986b). Nevertheless, NH_4^+ could be the major source of N to crops under soil conditions which are not favourable for nitrification such as anaerobic and acidic soil conditions (Haynes, 1986b).

Crop N uptake simulation in most models is driven by plant N demand (Ma and Shaffer, 2001). These authors stressed that crop N uptake depends on the

availability of NH_4^+ and NO_3^- , transpiration rate, and diffusion of soil N to root surface. Most models estimate crop N uptake as passive uptake (estimated from the rate of transpiration). Models such as RZWQM, however, consider active N uptake using Michaelis-Menten kinetics if N supply through passive uptake is not sufficient to satisfy the crop demand (Hanson, 2000). The N model added to SWB was adopted to simulate crop N uptake as a passive and active processes. Crop nitrogen uptake is estimated as the minimum value between crop demand and soil supply by the following models CERES, EPIC, GLEAMS, and CropSyst (Godwin and Jones, 1991; Williams, 1995; Knisel et al., 1993; Stöckle and Nelson, 2000).

2.4 Motivation for this study

Previous studies conducted in South Africa were laboratory incubation trials and single year field scale studies on specific crops (oats and maize). The majority of studies relating to beneficial sludge use on agricultural lands have been conducted in the northern hemisphere, under different soil-climate combinations than those experienced in South Africa. Since processes which determine the availability of nitrogen from sewage sludge (mineralization, immobilization, volatilization, nitrification, and denitrification) are influenced by climatic (temperature and water availability) and edaphic factors (soil pH, CEC, texture), locally based trials are essential.

This study was initiated in 2005 while the 1997 South African sewage sludge guideline was in place (Water Research Commission, 1997). This sludge guideline had an upper limit of $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ and was recently increased to an upper limit of $10 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, based on short-term laboratory incubation studies (Snyman and Van der Waals, 2004). According to the current guideline, farming systems which can responsibly exceed this upper limit norm can be approved upon successful motivation.

Sewage sludge treatment and handling processes affect the characteristics and availability of nutrients from sludge. Demand for nutrients by crops is dependent on crop type, water availability, climate, soil type, management practice, and farming intensity. The vast number of combinations among the above factors could never be adequately covered by field trials. The strategy in this project was therefore to use four widely differing farming systems in order to develop and test a mechanistic decision support model.

This study investigates four major cropping systems as presented below.

1. Dryland maize (*Zea mays* L.)

Maize is the staple food for the majority of the South African population and accounts for 51% of the cultivated land (FAO, 2005). About 8.0 million tons are produced each year, of which half is white maize for human consumption and half yellow maize for animal feed (Du Plessis, 2003; Vermeulen, 2005). The

rationale for selecting maize as a test crop was because it is the most commonly grown crop around the study area.

According to the Fertilizer Society of South Africa (2000), the mean N removal by maize accounts to 15 kg N per ton of grain only or 27 kg N per ton of grain plus stover. This is similar to the estimation made by O’Gara (2007) who reported that the total N removal by a whole maize plant is 1.6 times the amount taken off in the grain. The mean export of P by one ton maize grain was estimated as 3 kg (Fertilizer Society of South Africa, 2000).

Rainfall is the limiting factor for the productivity of dryland maize, making it difficult to predict the amount of sludge required. There is likely to be insufficient N for maximum yields in wet seasons and an excess of N in soil following dry years. The latter situation would predispose the system to nitrate leaching in the beginning of the new season, before a deep root system had developed.

2. Irrigated maize-oats rotation (*Zea mays* L. and *Avena sativa* L.)

Access to irrigation makes it easier to predict crop yields and hence the appropriate sludge application rates. Irrigation also allows for double cropping, and hence the potential to remove more nutrients from the soil. At the same time the continually wet conditions are more conducive to leaching, should high levels of N build up in the soil.

A substantial proportion of the maize crop is irrigated: 36% in the Free State, 20% in Mpumalanga, and 28% in North West is irrigated (FAO, 2005). Oats can be grown between two maize crops. Oats are promoted as a healthy grain for human beings (Peterson, 1992) and a high quality fodder for animals (Schrickel et al., 1992). According to the Fertilizer Society of South Africa (2000), the mean N removal by oats grain alone accounts to 33 kg per ton of grain. The mean export of P by one ton of oats grain was estimated as 4 kg (Fertilizer Society of South Africa, 2000).

3. Perennial dryland pasture (*Eragrostis curvula* (Schrad.) Nees)

Dryland pastures occupy large areas, need high amounts of fertilizer for optimal growth and are common around built up areas (Muchovej and Rechcigl, 1998). Perennial grasses have the potential to reduce nitrate leaching compared to annual crops due to their established root system. They are considered a good choice for repeated sludge applications because of their efficient nitrogen utilization under intensive management practices and because a number of harvests can be made in a year (Cogger et al., 2001).

Weeping love grass is a summer growing perennial, which is native to South Africa (Skerman and Riveros, 2008) and the most widely cultivated grass in South Africa (Dickinson et al., 2004). Weeping love grass is used for animal feed, either grazed or as hay. In the drier farming regions it is recommended for leys with Alfalfa (Duke, 1983). Besides animal feed, weeping love grass is used to

stabilize terraces and roads and control wind and water erosion (Duke, 1983; Cook et al., 2005).

Generally biomass production is poor under conditions of low rainfall and soil fertility. However, under favourable environmental conditions and adequate N and water supply, biomass yield could increase to 20 - 30 Mg ha⁻¹ (Cook et al., 2005). Nitrogen is the key element to a high quality fodder production. Nitrogen application rate depends on the quantity and quality of fodder needed, rainfall of the region, and soil production potential. As a rule of thumb weeping love grass requires 20 kg of N and 2 kg of P for every ton of dry matter produced (Dickinson et al., 2004).

4. Lawn sod production (*Pennisetum clandestinum*)

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, by minimizing the thickness of soil exported with the sod (Charbonneau, 2003).

Kikuyu is a warm season perennial grass, which was introduced to South Africa 80 years ago. It is an important pasture grass in the high rainfall areas and under irrigation (Dickinson, 2004), due to its good nutrition values (Butler and Bailey 1973; Marais et al., 1992). It has the capability to regenerate easily following repeated mowing and thus is used as lawn. Kikuyu is a low maintenance grass and is able to recover quickly from moderate to severe wear injury (American lawns, 2008). The rationale for the selection of kikuyu as a test crop for sod production was because it is the main turfgrass species in South Africa. In addition, it has the ability to grow under acidic soils, establishes easily in soils rich with organic matter, is salt tolerant (Dickinson, 2004), and has the ability to take up large amounts of N (FAO, 2008) and P (Hanna et al. 2004).

According to Williams (quoted by Dickinson, 2004), kikuyu requires about 23 kg of N and 2.9 kg of P per ton of dry matter produced. Generally, kikuyu is less sensitive to P deficiency except for extremely deficient soils (Miles, 1997; Mathews et al. 2001). In this study, however, the main purpose of sod production was to export sludge with sod as a growing medium.

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CHAPTER 3

MATERIALS AND METHODS

3.1 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° 01' 01" S and longitude of 28° 16' 55" E. The area has a long term annual average rainfall of 700 mm, mainly during the months of October to March. 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 (H₂O) 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 ranged from 8 mS m⁻¹ at 1.2 m depth to 36 mS m⁻¹ in the top 0.3 m soil layer.

3.2 Sludge characteristics

The sludge used in this study was anaerobically digested and paddy-dried. 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 3.1). Based on the microbiological report from the ERWAT laboratory, the sludge can also be classified as microbiological class "A".

Table 3.1 Chemical characteristics of anaerobically digested, paddy dried sludge used during the 2004/05 – 2007/08 growing seasons (source Vlakplaats wastewater treatment plant)

Element	Unit	Year			
		2004/05	2005/06	2006/07	2007/08
N	%	3.032	1.884	2.219	3.094
NH ₄ _N	mg kg ⁻¹	2018	4362	4064	7660
NO ₃ _N	mg kg ⁻¹	183	6	40	11
P	%	1.962	1.844	2.76	2.24
P – Bray1	mg kg ⁻¹	166	154	40	66
Total C	%	23	20	21	20
Water content	%	46	30	28	38
EC _e	mS m ⁻¹	1814	1212	1412	3110
K	mg kg ⁻¹	3804	710	689	1356
Ca	mg kg ⁻¹	25116	13062	17450	10042
Mg	mg kg ⁻¹	5358	591	829	1145
pH (H ₂ O)		6.01	6.2	6.02	6.08
Cd	mg kg ⁻¹	1.63	0.07	0.15	18.91
Hg	mg kg ⁻¹	1.70	0.02	0.03	1.81
Cr	mg kg ⁻¹	51.93	1.50	2.92	503.8
As	mg kg ⁻¹	7.08	0.18	0.23	17.94
Pb	mg kg ⁻¹	54.46	9.41	1.37	102.0
Zn	mg kg ⁻¹	459.9	4.33	20.85	2325
Ni	mg kg ⁻¹	23.81	1.37	0.97	144.5
Cu	mg kg ⁻¹	97.2	3.21	4.59	526.8

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) allows such quality 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 $10 \text{ Mg ha}^{-1} \text{ yr}^{-1}$. Selected chemical characteristics of this A1a sludge are presented in Table 3.1. The total amount of N and P added from sludge to each cropping system is presented in the corresponding chapters while other macro elements and heavy metals are presented in Appendixes 1 and 2.

3.3 Field trial and treatments

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

3.3.1 Dryland maize and irrigated maize/oats rotation

The treatments for dryland maize and irrigated maize-oats rotation consisted of three sludge rates ($4, 8, \text{ and } 16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), a control, and an inorganic fertilizer

treatment. The value of $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ represents the annual agricultural upper limit of the 1997 South African sludge guideline (Water Research Commission, 1997), which has recently been raised to $10 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ (Snyman and Herselman, 2006). Sludge rates of 4 and $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ represent half and double the former norm.

Applications to the inorganic fertilizer maize treatment were conducted according to the recommendations of the OMNIA Fertilizer Company in 2005, with some modification to the amount of K recommended (K application increased due to deficiency symptoms) since the 2005/06 growing season for all crops (Table 3.2). As municipal sludge is a poor source of K, a similar amount to that applied to the inorganic fertilizer treatments was applied to all sludge treatments. 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 Mg ha^{-1} (Table 3.2). 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.

Sludge and inorganic fertilizer were broadcast and immediately incorporated into the soil with a manually operated, diesel powered rotovator (Agria). After sludge incorporation, the soil was levelled using rakes and maize (PAN 6966) was planted in 0.9m rows at rates of 80000 seeds per hectare under irrigation and

half this rate under dryland conditions. Each plot consisted of 6 rows of 5 m in length, with the outer one row on either side taken as boarder row.

After harvesting irrigated maize, sludge and inorganic fertilizer was applied accordingly and again incorporated, after which oats were planted. Oats were planted at a rate of 90 kg ha⁻¹ using a hand-drawn planter with double disk openers. Each oats plot consisted of 15 rows, spaced 0.3 m apart and 5 m in length, with two border rows on either side. Maize and oats planting and harvesting dates are presented in Table 3.3. The time and amount of inorganic fertilizer applications for dryland maize and irrigated maize/oat rotation are presented in Table 3.2.

Table 3.2 Inorganic fertilizer application timing and type of fertilizer applied to dryland maize, irrigated maize, and irrigated oats during the 2004/05 to 2007/08 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

NB. During the 2005/06, 2006/07 and 2007/08 growing seasons K application was increased to 20 kg ha⁻¹ at planting (both dryland and irrigated maize), to 40 kg ha⁻¹ five weeks after germination (irrigated maize), and 40 kg ha⁻¹ at seven weeks after germination (both dryland and irrigated maize) due to deficiency symptoms observed in leaves.

Table 3.3 Planting and harvesting dates for dryland maize, irrigated maize, and irrigated oats experiment conducted during the 2004/05-2007/08 growing seasons at ERAWAT, Ekurhuleni district, South Africa.

Growing season	Dryland maize		Irrigated maize		Irrigated oats	
	Planting date	Harvesting date	Planting date	Harvesting date	Planting date	Harvesting date
2004/05	13 Dec. 2004	03 May 2005	13 Dec. 2004	03 May 2005	27 June 2005	15 Dec. 2005
2005/06	16 Dec. 2004	†	16 Dec. 2004	†	12 Apr. 2006	28 Nov. 2006
2006/07	20 Dec. 2006	10 May 2007	20 Dec. 2006	10 May 2007	‡	‡
2007/08	18 Nov. 2007	15 Apr. 2008	18 Nov. 2007	15 Apr. 2008	02 May 2008	15 Nov. 2008

† Maize was damaged by herbicide and was harvested before maturity.

‡ Oats were not planted during this season.

3.3.2 Weeping lovegrass (*Eragrostis curvula*)

The treatments for the dryland pasture (weeping lovegrass) consisted of three sludge rates (4, 8, and 16 Mg ha⁻¹ yr⁻¹), a control, and an inorganic fertilizer treatment. The annual sludge application rate for all weeping lovegrass 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 weeping lovegrass treatments

based on the Kynoch pasture Handbook (Dickinson et al., 2004) for a target annual hay yield of 10 Mg ha⁻¹.

In the beginning of the study, during the 2004/05 growing season both sludge and inorganic fertilizers were broadcast over the soil surface and immediately incorporated into the top 10cm soil layer with a manually operated, diesel powered rotovator (Agria). After sludge incorporation, the soil was levelled using rakes. A mixture of weeping lovegrass (10 kg ha⁻¹) and teff (*Eragrostis teff*) (6 kg ha⁻¹) were planted on 15 Nov. 2004 and a hand drawn roller was used to ensure good seed-soil contact for better germination. During the rest of the study period, however, both sludge and inorganic fertilizer were broadcast over the soil surface of the already established weeping lovegrass plots. Unlike maize and oats, however, both sludge and inorganic fertilizers were left on the soil surface without incorporation. Sludge application and hay cutting dates are presented in Table 3.4. The amount and timing of inorganic fertilizer applications are presented in Table 3.5.

Table 3.4 Sludge applications and hay cutting dates for Weeping lovegrass during the 2004/05 to 2007/08 growing seasons at ERWAT, Ekurhuleni district, South Africa.

Growing season	Sludge application date		Hay cutting date	
	First	Second	First	Second
2004/05	14/11/2004	20/02/2005	17/02/2005	22/05/2005
2005/06	01/11/2005	20/02/2006	31/01/2006	11/05/2006
2006/07	20/10/2006	10/01/2007	09/01/2007	16/04/2007
2007/08	26/09/2007	14/01/2008	11/01/2008	16/04/2008

Table 3.5 Type of fertilizer applied and application timing for a weeping lovegrass experiment during the 2004/05 to 2007/08 growing seasons at ERWAT, Ekurhuleni district, South Africa.

Time of fertilizer application	Fertilizer type	Amount of Individual nutrient added		
		N	P	K
		kg ha ⁻¹		
At planting time, or beginning of sprouting, and after every first cut in a season	NPK 2:3:2(22%)	13	20	13
	Limestone ammonium nitrate (LAN) (28% N)	87		
	KCl (50% K)			27
Four weeks after first application	LAN	100		
	Super phosphate (18.5% P)		20	
	KCl			47
Cumulative annual application		200	40	87

3.3.3 Turfgrass (Kikuyu, *Pennisetum clandestinum* Hochst. ex Chiov.)

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

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.

3.4 Rainfall and irrigation

3.4.1 Rainfall

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 (Thermistor Humitter 50Y) 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 Electronic Inc., Dallas, Texas, USA), and a CR10X data-logger (Campbell Scientific Inc., Utah, USA). Monthly rainfall distribution during the study period is presented in Table 3.6.

3.4.2 Irrigation

Irrigated maize/oats rotation

The irrigated maize-oats rotation experiment was planted under drip irrigation. The lateral spacing between dripper lines was 0.5 m during summer 2005, but was reduced to 0.3 m since the 2005 winter for the rest of the study period. The distance between the drippers in a line was 0.3 m. The drip system was operated at a pressure of 100-150 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. In 2005, this was followed by irrigation

according to the FAO crop factor method in the SWB model once every five days until harvest in the absence of rainfall. In 2007 and 2008, however, maize was irrigated according to neutron probe deficit readings to fill the profile to field capacity once a week in the absence of rainfall.

During the winter seasons of each year, oats were irrigated every three days (9 mm in 2005 and 10 mm in 2006 and 2008) for the first four weeks. This was followed by irrigations according to a site calibrated neutron water meter (Model 503 DR CPN Hydroprobe, Campbell Pacific Nuclear, California, USA) deficit readings to a depth of 1.2 m for the rest of the season until harvest, mostly twice a week in 2005. In 2006 and 2008, however, the crop was irrigated weekly based on site calibrated neutron probe soil water deficit readings followed by next irrigation schedule according to SWB model recommendation.

Table 3.6 Monthly rainfall distributions during the 2004/05 to 2007/08 growing seasons at ERWAT, Ekurhuleni district, South Africa.

Months	Growing seasons				
	2004	2005	2006	2007	2008
January		168	108	70	207
February		94	145	38	60
March		70	78	31	159
April		63	18	19	21
May			4	1	
June					
July					
August					
September				18	
October		12	18	117	73
November	32	129	117	56	101
December	89	84	132	72	112

Turfgrass

Adjustable nozzle micro-sprayers (Hunter, California, USA) at the four corners of each plot were used to irrigate the turfgrass using municipal drinking water. The micro-sprayers were adjusted to a 90° 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 Mg ha⁻¹ treatment.

3.5 Soil solution sampling and analyses

Nitrate and salt leaching was monitored by wetting front detectors (WFDs) installed both at 0.3 and 0.6 m depths located in the middle of a plot. The detectors were installed in all treatments of the irrigated maize/oats rotation, in the control, 8 and 16 Mg ha⁻¹ yr⁻¹ treatments of the dryland maize and dryland pasture. The detectors were also installed in the control, 8, 33, and 100 Mg ha⁻¹ sludge treatments of the turfgrass trial.

Soil solution samples collected from the wetting front detectors were analyzed for electrical conductivity (EC) using an ECScan-High EC meter (Eutech Instruments, Malaysia) and nitrate concentration was analyzed using a C99 Multiparameter Bench Photometer (Hanna Instruments, Italy).

3.6 Plant sampling

3.6.1 Dryland and irrigated maize

In 2004/05 growing season, above-ground samples were collected approximately every two weeks for growth analyses from an area of 0.5 m² (typically two plants under dryland and four under irrigation). In 2006/07 and 2007/08 growing seasons, however, plant samples were taken only three times during the growing season: First sample taken at the eight leaf growth stage, second at the soft dough stage and the third at crop maturity. At the soft dough stage, plant samples were taken for forage yield 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 at the eight leaf growth stage, at the soft dough stage, and at crop maturity from both dryland and irrigated maize for crop N and P uptake determinations. At crop maturity, plant samples were taken for grain yield determination from 2 m lengths of the three middle rows in 2004/05 and 2 m length of the middle four rows in 2006/07 and 2007/08. Two additional plant samples per plot were taken from dryland maize and four plants from irrigated maize for crop N and P uptake determinations.

3.6.2 Irrigated oats

Whole plant (above-ground) samples were collected every two weeks from all plots for growth analyses from an area of 0.45 m² (typically three rows of 0.5 m length). A hand grab of additional plant samples were collected randomly from all plots twice during the growing season before the soft dough stage for N and P analyses. At the soft dough stage plant samples were collected for forage yield determination from 2 m lengths of the 6 middle rows. At the same time a hand grab of additional plant samples were collected for N and P uptake determination.

The crop was harvested once the crop matured during the 2005 and 2006 growing seasons. In 2008, however, the crop was 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. In addition, a hand grab of plant samples were collected for N and P uptake determination.

3.6.3 Weeping lovegrass

In 2004/05, four replications of whole plant (above-ground) samples were harvested 0.05 m above the soil surface approximately every two weeks for growth analyses from an area of 0.5 m². In the years which followed (2005/06, 2006/07, 2007/08), however, plant samples were taken approximately once a month from an area of 0.5 m². A 0.5m border was left around all sides of each plot during sampling. A hand grab of 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 m² area. In addition, a hand grab of grass samples were collected at harvest from each plot for grass N and P uptake determination.

Growth analyses

Plant samples collected for growth analyses from dryland maize, the irrigated maize-oats rotation, and weeping lovegrass 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 °C for 48 hours to determine above-ground biomass. Above-ground biomass was calculated as the sum of leaf, stem, and seed.

3.6.4 Turfgrass

Clipping samples were collected from a 0.5 m² area during each mowing for dry matter determination. Additional clipping samples were taken during the season for N and P analyses. 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 m² 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 °C and weighed. Additional sod samples were taken randomly, combined into one sample for each plot and separated into plant and soil 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 °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.

3.7 Soil sampling

Soil samples for chemical analyses from the top soil layer were collected from differing depths for the various farming systems, depending on the impact of tillage in the distribution of nutrients within the soil profile. Consequently, soil samples from the top layer were collected from a depth of 0.3m in dryland maize and irrigated maize-oats rotation because the soil was rotovated each season to a depth of 0.3m. Soil samples from dryland pasture and turfgrass, however, were collected from the top 0.1 and 0.15m layers respectively because the sludge was surface applied in both farming systems.

3.7.1 Dryland maize and irrigated maize/oats rotation

At the beginning of the study, soil samples were collected on 11 Dec. 2004 before treatment application from the dryland maize and irrigated maize/oats rotation plots using an auger. The samples were collected from the 0-0.3 m, 0.3-0.6 m, 0.6-0.9 m, and 0.9-1.2 m layers. The samples from each layer of the dryland maize and irrigated maize/oats rotation plots in a block (replication) were combined and mixed to make a single homogenous soil sample per layer.

At the end of each growing season, three soil samples were collected diagonally from each plot of a treatment using an auger for the dryland maize. The samples were collected from the 0-0.3 m, 0.3-0.6 m, 0.6-0.9 m, and 0.9-1.2 m layers. The three samples from each layer of a plot were combined and mixed to make a single homogenous soil sample per layer. Similar procedures were followed during soil sampling and sample preparation of the irrigated maize-oats rotation farming system.

3.7.2 Dryland pasture

At the beginning of the study, soil samples were collected on 09 Nov. 2004 before treatment application from the dryland pasture plots using an auger. The samples were collected from the following layers: 0-0.1m, 0.1-0.3m, 0.3-0.5m,

and 0.5-0.8 m. The samples from each layer of the dryland pasture in the same block (replication) were combined and mixed to make a single homogeneous soil sample per layer.

At the end of each growing season, three soil samples were collected diagonally from each plot using an auger from the dryland pasture treatment plots. 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 soil sample per layer.

3.7.3 Turfgrass

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 homogeneous soil sample per layer.

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 0-0.15 m, 0.15-0.45 m, 0.45-0.75 m, and 0.75-1.05 m layers. The three samples from each layer of a plot were combined and mixed to make a single homogenous soil sample per layer.

All soil samples from all cropping systems were dried and pulverized. The dried and ground soil samples were digested and analyzed for N, P, NO_3^- , NH_4^+ , and Bray -1P.

3.8 Plant and soil chemical analyses

Total P in sludge, soil, and plant samples as well as plant heavy metal uptake were determined after wet acid digestion using an Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES) (SpectroFlame Modula; Spectro, Kleve, Germany). Plant extractable soil P was analyzed using the Bray-1 extraction method, following standard procedures (Non-affiliated Soil Analyses Work Committee, 1990). Soil and plant samples were ground to pass through a 150 μm screen and analyzed for total C and N using a Carlo Erba NA1500 C/N analyzer (Carlo Erba Strumentazione, Milan, Italy). Soil samples were extracted in 1:5 1 M KCl and tested for ammonium and nitrate with the Lachat Autoanalyzer (Lachat Quick Chem Systems, Milwaukee, WI, USA). Soil salinity was estimated from saturation extracts (Non-affiliated Soil Analyses Work Committee, 1990). Extracts were analyzed for electrical conductivity using a Beckman conductivity bridge and reported in units of mS m^{-1} . Sludge electrical conductivity was also analyzed on a 1:2 sludge to water ratio extract following similar procedures.

3.9 Additional methods involved in turfgrass trial

3.9.1 Mowing and sod harvest

Turf was cut with a reel mower (Protea Turf, Germiston, South Africa) to a height of 0.03 m each week. Clippings were removed according to common practice in the area, which also assists in exporting additional nutrients from the field. Sods were harvested to a depth of 0.02 m from all treatments when the slowest growing treatment had reached full cover. A hand operated, light-weight, gasoline powered sod cutter (Protea Turf, Germiston, South Africa), used by commercial sod growers in the area, was used. Each sod had a width of 0.5 m and length of 1 m.

3.9.2 Soil loss through sod lifting

Three 0.5 m long iron 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 harvesting. Two pegs were located at adjacent corners of each plot, and the third was in the middle of the plot. Each was marked with a single groove 0.04 m above the soil surface to indicate the position at which to tie a length of string. After each sod harvest, the string was stretched between the two adjacent corner pegs across the peg in the middle of the plot, giving 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.

3.9.3 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.

3.10 Model parameter description

The new SWB model requires specific crop parameters as well as management, weather and soil data as an input to run both the crop growth and nutrient models. The input parameters required by the model namely, specific crop parameters, soil data, crop data, and nutrient module parameters and data are listed below. Details on parameters and their determination can be obtained from

Annandale et al. (1999) for the crop growth model and Stöckle and Nelson (2000) for the nitrogen model.

3.10.1 Crop growth model

Specific crop parameters

- a. Canopy extinction coefficient for solar radiation (K);
- b. Dry matter: water ratio (DWR) in kPa;
- c. Radiation use efficiency (E_c) in kg MJ^{-1} ;
- d. Base temperature (T_b) in $^{\circ}\text{C}$;
- e. Optimum temperature (T_{lo}) in $^{\circ}\text{C}$;
- f. Maximum temperature ($T_{cut-off}$) in $^{\circ}\text{C}$;
- g. Thermal time: emergence ($EMDD$) in $\text{d }^{\circ}\text{C}$;
- h. Thermal time: reproductive phase ($FLDD$) in $\text{d }^{\circ}\text{C}$;
- i. Thermal time: maturity ($MTDD$) in $\text{d }^{\circ}\text{C}$;
- j. Thermal time: transition ($TransDD$) in $\text{d }^{\circ}\text{C}$;
- k. Thermal time: leaf senescence in ($MaxLeafAge$) in $\text{d }^{\circ}\text{C}$;
- l. Leaf water potential at maximum transpiration rate ($psilm$) in kPa or J kg^{-1} ;
- m. Maximum transpiration rate (Tr_{max}) in mm d^{-1} ;
- n. Specific leaf area (SLA) in $\text{m}^2 \text{kg}^{-1}$;
- o. Leaf stem partitioning factor ($PART$) in $\text{m}^2 \text{kg}^{-1}$;
- p. Total dry matter at emergence ($TDMstart$) in kg m^{-2} ;
- q. Root fraction (f_r);
- r. Stem translocation ($TransI$);
- s. Root growth rate (RGR) in $\text{m}^2 \text{kg}^{-0.5}$;
- t. Maximum canopy height (H_{Cmax}) in m, and
- u. Stress index (SI).

Soil data:

- a. Runoff curve number in mm;
- b. Matric potential at field capacity and permanent wilting point in kPa or J kg^{-1} ;
- c. Maximum drainage rate in mm d^{-1} and drainage factor;
- d. Soil layer:
- e. Thickness in m;
 - Volumetric soil water content at field capacity and permanent wilting point in m m^{-1} ;
 - Initial volumetric water content in m m^{-1} , and
 - Bulk density in Mg m^{-3} of each layer.

Crop data:

- a. Model type;
- b. Name of the crop;
- c. Planting date;
- d. Starting date of the simulation;
- e. Weather ID;
- f. Area of the field in ha;
- g. Irrigation timing options (amount in mm, interval in days, depletion in %);
- h. Irrigation system (type and design details), and
- i. Management (root zone or profile).

Weather data:

- a. Latitude ($^{\circ}\text{N}$ or $^{\circ}\text{S}$);
- b. Hemisphere;
- c. Wind speed in m s^{-1} and its height of measurement;
- d. Maximum and minimum daily temperature in $^{\circ}\text{C}$;
- e. Solar radiation in $\text{MJ m}^{-2} \text{d}^{-1}$;
- f. Precipitation in mm, and

- g. Maximum and minimum relative humidity in % or vapour pressure in kPa or else dry and wet bulb temperatures in °C.

3.10.2 Nitrogen model

Initial soil profile data for each layer:

- a. Organic matter (%)
- b. Soil pH (H₂O)
- c. CEC (mmol/100g)
- d. Nitrate (mg kg⁻¹)
- e. Ammonium (mg kg⁻¹)
- f. Residue mass (kg ha⁻¹)

Fertilizer and tillage:

- a. Fertilizer type (organic, inorganic)
- b. Fertilizer form (Solid, liquid)
- c. Application method (Incorporated, surface applied)
- d. Amount applied (kg ha⁻¹)
- e. Depth of cultivation
- f. Residue management after harvest (fraction of standing and surface residue left after harvest)

Organic fertilizers (biosolids, manure) parameters:

- a. Carbon fraction

- b. Ammonium fraction
- c. Nitrate fraction
- d. Water content
- e. Fast cycling fraction, half life, and C:N ratio
- f. Slow cycling fraction, half life, and C:N ratio
- g. Lignified fraction, half life, and C:N ratio

3.11 Statistical analyses

Statistical analyses were performed to evaluate the impact of various sludge application rates on the following measured traits: Forage, grain yield and crude protein contents of dryland maize, irrigated maize, and oats; hay yield and crude protein contents of weeping lovegrass; turfgrass establishment rate, colour, sod strength, thickness of soil and sod mass exported, sod N and P, and plant N and P exported with the sod. The statistical analyses were conducted using Analysis of Variance (ANOVA) and General Linear Model (GLM) procedures of Windows SAS version 9.0 (SAS Institute, 2002), to determine significant treatment effects on measured response variables. When treatment effects were found to be significant, Fisher's protected LSD at the 0.05 level was used to separate means.

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CHAPTER 4

AGRONOMIC CROPS

The following hypotheses are tested in this chapter.

HYPOTHESIS 1

Sludge application above the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ limit will increase:

- a. Dryland maize and irrigated maize-oat rotation grain yield.
- b. Dryland maize and irrigated maize-oat rotation forage yield.

HYPOTHESIS 2

- a. More N can be exported in either grain or forage under irrigated rotation than dryland cropping.
- b. Under both dryland and irrigated conditions, more N can be exported in forage than grain.

HYPOTHESIS 3

- a. High yielding irrigated and dryland cropping systems can fully utilize N supply from double the norm and are not susceptible to excessive nitrate leaching.

HYPOTHESIS 4

Sludge application according to crop N demand results in the accumulation of total and plant available P in the soil profile.

4.1 Grain and forage yield

4.1.1 Grain yield

a. Dryland maize (*Zea mays* L.)

Dryland maize grain yield was significantly increased by doubling the recommended annual sludge limit of 8 Mg ha⁻¹ yr⁻¹ for the first two crops, but not the third. Inorganic fertiliser gave higher yields than the 8 Mg ha⁻¹ yr⁻¹ limit in two of the three years (Table 4.1).

Table 4.1 Dryland maize grain yield response to three sludge application rates, a control, and an inorganic fertilizer treatment during the 2004/05 to 2007/08 growing season.

Treatment	Dryland maize grain yield		
	2004/05	2006/07	2007/08
	Mg ha ⁻¹		
Control†	4.98	1.33	4.92
4 Mg ha ⁻¹ yr ⁻¹	5.86	1.52	6.47
8 Mg ha ⁻¹ yr ⁻¹	6.17	1.74	8.00
16 Mg ha ⁻¹ yr ⁻¹	6.87	2.04	8.29
Inorganic fertilizer	6.35	2.17	8.56
LSD (5%)	0.32	0.09	0.37
CV, %	3	3	3

† zero sludge and inorganic fertilizer applied

The rainfall was 564, 405, and 710 mm for the three seasons respectively, and the low yield for the 2006/07 growing season was a consequence of drought. Water stress during the silking, tasseling and pollination stages results in severe yield losses by inhibiting pollination, reducing silk elongation as well as delaying silk formation (Rhoads and Bennett, 1990). Even though crop response to nutrient supply is influenced by the availability of water (Tisdale et al., 1985), there was a clear response to increasing nutrition in dry (2006/07) as well as wet years (2007/08).

b. Irrigated maize-oat (*Avena sativa* L.) rotation

Irrigated maize grain yield was significantly improved by doubling of the 8 Mg ha⁻¹ yr⁻¹ sludge limit in all three seasons (Table 4.2). Yields under irrigation were much higher than dryland, ranging from 65% greater in a wet year to 7 fold greater in a dry year. The 16 Mg ha⁻¹ yr⁻¹ sludge application always gave a significantly higher yield than inorganic fertiliser. Irrigated oats grain yield also benefited from double the recommended sludge in all three seasons. There was no clear difference between the highest sludge application and the inorganic fertiliser treatments on oat grain yield.

The most likely explanation for the significant response to sludge was N supply, since the plant available background soil P of 29 mg kg⁻¹ remained within or higher than the optimum range required for most crops, (25-30 mg kg⁻¹), as defined by Sims and Pierzynski (2000). The role of N in the development of

kernels in maize is to establish and maintain the crop's photosynthetic ability (Below, 1995 and 1997), as well as facilitating the development of reproductive sinks (Below et al., 2000). Thus N is generally the limiting factor in irrigated maize production through its effect on kernel number and mass (Rhoads and Bennett, 1990; Lemcoff and Loomies, 1986; Jacobs and Pearson, 1992).

Table 4.2 Irrigated maize-oat rotation grain yield response to three sludge application rates, a control, and an inorganic fertilizer treatment during the 2004/05 to 2007/08 growing seasons.

Treatment	Irrigated maize-oat rotation grain yield					
	Maize			Oats		
	2004/05	2006/07	2007/08	2005	2006	2008
	Mg ha ⁻¹					
Control†	13.61	5.00	4.82	2.75	2.69	2.32
4 Mg ha ⁻¹ yr ⁻¹	14.80	7.47	7.06	3.32	3.78	3.08
8 Mg ha ⁻¹ yr ⁻¹	16.10	10.40	10.31	3.89	4.34	3.60
16 Mg ha ⁻¹ yr ⁻¹	19.94	15.21	14.10	4.09	4.93	4.04
Inorganic fertilizer	15.06	11.18	11.05	4.01	4.51	4.30
LSD	1.11	0.72	0.38	0.18	0.13	0.22
CV, %	5	5	3	3	2	4

† zero fertilizer and inorganic fertilizer applied

NB: During the 2005/06 growing season maize was damaged by herbicide at early stage and oats was not planted during the winter season of 2006/07 due to technical problems.

Similarly, nitrogen influences the grain yield of oats by affecting the various yield components, such as panicle density, kernel number, dry matter production and

harvest index. N deficiency results in the decline of yield by causing abortion of kernels located at the end of the ear, or by interrupting dry matter accumulation prematurely (Pearson and Jacobs, 1987).

In summary, doubling the annual upper limit of sludge from 8 to 16 Mg ha⁻¹ improved dryland maize yield in two out of three seasons. The yield of irrigated maize and oats was increased each year. Consequently, Hypotheses 1a was accepted for irrigation conditions, and in the majority case for dryland conditions.

4.1.2 Forage yield

a. Dryland maize

Sludge applied at double the annual 8 Mg ha⁻¹ limit significantly improved dryland maize forage yield, except during the third growing season (Table 4.3). As was the case for grain, forage yield was reduced in the drier than average 2006/07 season. Dryland maize forage yield from the 8 Mg ha⁻¹ yr⁻¹ sludge treatment was similar to the inorganic fertilizer, apart from the dry 2006/07 growing season.

The increase in forage yield with higher sludge applications is most likely a consequence of the increasing N supply. However, it is not clear why there was still a significant response to increasing sludge in the dry as well as wet seasons, since the demand for N in season two would only have been about one third of that for season three.

Table 4.3 Dryland maize forage yield response to three sludge application rates, a control, and an inorganic fertilizer treatment during the 2004/05 to 2007/08 growing seasons.

Treatment	Dryland maize forage yield		
	2004/05	2006/07	2007/08
	Mg ha ⁻¹		
Control†	11.67	4.50	11.69
4 Mg ha ⁻¹ yr ⁻¹	13.73	4.79	15.15
8 Mg ha ⁻¹ yr ⁻¹	14.89	5.44	18.65
16 Mg ha ⁻¹ yr ⁻¹	15.78	6.18	18.70
Inorganic fertilizer	14.44	6.42	19.29
LSD (5%)	0.57	0.27	1.69
CV, %	3	3	7

† Zero sludge and inorganic fertilizer applied.

b. Irrigated maize-oat rotation

Doubling of the recommended annual upper sludge limit significantly improved irrigated maize as oat forage yield in all three seasons (Table 4.4). There was a noticeable decline in irrigated maize forage yield over time, especially in the control and low sludge treatments. The highest sludge application of 16 Mg ha⁻¹ produced greater yields than inorganic fertiliser for maize forage in all seasons, and for oats in two of the three seasons.

The forage yield response to sludge is again attributed to N supply (Christie et al., 2001; Moser et al., 2005) and is consistent with previous studies conducted in

other countries (Collins et al., 1990; Wang et al., 2000; Bozkurt et al., 2006). The inorganic fertiliser and 16 Mg ha⁻¹ sludge treatments received similar N applications of 1278 kg and 1309 kg respectively over the three seasons. Nevertheless the irrigated maize forage yield was greater under the 16 Mg ha⁻¹ of sludge, suggesting that the beneficial effects of sludge were more than just N supply.

Table 4.4 Irrigated maize-oat rotation forage yield response to three sludge application rates, a control, and an inorganic fertilizer treatment during the 2004/05 to 2007/08 growing seasons.

Treatments	Irrigated maize-oat rotation forage yield					
	Maize			Oats		
	2004/05	2006/07	2007/08	2004/05	2005/06	2007/08
	Mg ha ⁻¹					
Control†	31.56	14.66	14.31	8.24	8.05	8.04
4 Mg ha ⁻¹ yr ⁻¹	32.45	19.53	19.37	10.32	11.83	11.18
8 Mg ha ⁻¹ yr ⁻¹	34.30	25.18	25.01	11.35	12.86	12.78
16 Mg ha ⁻¹ yr ⁻¹	41.88	33.45	31.22	13.64	16.44	15.82
Inorganic fertilizer	32.81	25.00	24.95	12.96	14.57	14.27
LSD	1.89	1.27	2.20	0.62	0.42	1.5
CV, %	4	4	6	4	2	8

† Zero sludge and inorganic fertilizer applied

The yield of oats was higher with 16 Mg ha⁻¹ sludge than fertiliser in two of the three seasons, although both received similar levels of N. However, oats forage

yield was higher from the inorganic fertiliser than the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatment, due most likely to the lower total N applied in the sludge and the slower mineralisation rate during the winter period.

Hypotheses 1 b was accepted for irrigated conditions, namely that sludge application above the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ limit will increase the forage yields of maize and oats in rotation. Hypothesis 1b was accepted in the majority cases for dryland maize forage yield.

4.2 Crop N uptake

4.2.1 Grain N uptake

a. Dryland maize

Dryland maize grain N uptake increased significantly with increase in sludge application rate (Table 4.5) for sludge rates ranging between 0 and $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$. Grain N uptake was lower during the 2006/07 growing season and higher during 2007/08 in response to rainfall patterns. Grain N uptake was highest from the inorganic fertilizer treatment. This occurred despite higher total N supply from the 16 Mg ha^{-1} sludge treatment (485, 355, and 495 kg N ha^{-1} as opposed to 96 kg N ha^{-1} of inorganic fertilizer each year during 2004/05, 2006/07, and 2007/08 respectively).

Table 4.5 Dryland maize grain N uptake from a clay loam soil treated with three sludge application rates, an inorganic fertilizer, and a control during the 2004/05 to 2007/08 growing seasons.

Treatments	Dryland maize grain N uptake		
	2004/05	2006/07	2007/08
	kg ha ⁻¹		
Control†	72	22	61
4 Mg ha ⁻¹ yr ⁻¹	88	24	92
8 Mg ha ⁻¹ yr ⁻¹	102	28	117
16 Mg ha ⁻¹ yr ⁻¹	117	33	131
Inorganic fertilizer	123	40	138
LSD (5%)	3	1	3
CV, %	2	2	2

† Zero sludge and inorganic fertilizer applied

NB. In 2005/06 maize was harvested at early stage due to damage by herbicide.

b. Irrigated maize

Irrigated maize grain N uptake increased significantly as the sludge application rate increased, with highest uptake recorded from the 16 Mg ha⁻¹ sludge treatment (Table 4.6). Similarly, oats grain N uptake increased as sludge application rate increased to the maximum rate, except in 2004/05. Irrigated maize grain N uptake decreased over time, and the effect was greatest at lower application rates of sludge.

Table 4.6 Irrigated maize-oat rotation grain N uptake from a clay loam soil treated with three sludge application rates, an inorganic fertilizer, and a control during the 2004 to 2007/08 growing seasons.

Treatments	Irrigated maize-oat rotation grain N uptake					
	Maize			Oats		
	2004/05	2006/07	2007/08	2004/05	2005/06	2007/08
	kg ha ⁻¹					
Control†	195	74	60	50	29	42
4 Mg ha ⁻¹ yr ⁻¹	219	103	93	62	54	45
8 Mg ha ⁻¹ yr ⁻¹	282	143	132	76	57	63
16 Mg ha ⁻¹ yr ⁻¹	391	204	220	76	73	70
Inorganic fertilizer	292	160	161	86	76	83
LSD	8	7	12	1	2	4
CV, %	2	3	6	1	2	4

† Zero sludge and inorganic fertilizer application

NB. In 2005/06 maize was harvested at early stage due to damage by herbicide.

In general, grain N uptake was higher under irrigated maize compared with similar treatments under dryland conditions, except the control which was slightly lower during the 2007/08 growing season. In addition, the overall annual grain N uptake by irrigated maize-oat rotation was at least three times higher than similar treatments under dryland conditions during the 2004/05 growing season and twice higher in 2007/08. This indicates that nitrogen demand of grain production under intensive cropping systems was much higher than dryland conditions and sludge application rates should be adjusted accordingly.

4.2.2 Forage N uptake

a. Dryland maize

Dryland maize forage N uptake increased with increase in sludge application rate throughout the study period except in 2005/06 where maize was damaged by herbicide (Table 4.7). Similar to grain N uptake, dryland maize forage N uptake was lower in 2006/07 but higher in 2007/08. Dryland maize forage N uptake was highest from the inorganic fertilizer treatment except in 2004/05.

Table 4.7 Dryland maize forage N uptake from a clay loam soil treated with three sludge application rates, an inorganic fertilizer, and a control during the 2004/05 to 2007/08 growing seasons.

Treatments	Dryland maize forage N uptake			
	2004/05	2005/06	2006/07	2007/08
	kg ha ⁻¹			
Control†	123	20	66	126
4 Mg ha ⁻¹ yr ⁻¹	148	25	69	162
8 Mg ha ⁻¹ yr ⁻¹	188	24	79	238
16 Mg ha ⁻¹ yr ⁻¹	201	20	92	244
Inorganic fertilizer	197	64	103	254
LSD (5%)	8	8	2	14
CV, %	3	2	2	5

† Zero sludge and inorganic fertilizer applied

NB. In 2005/06 maize was harvested at early stage due to damage by herbicide, thus forage N uptake was very low.

This study showed that the amount of nitrogen exported with forage (Table 4.7) was higher than the amount exported with the grain (Table 4.5). According to the three year mean data, N exported with forage under dryland condition was twice higher than the amount exported with the grain. This indicates that N exported during forage production is much higher than grain production.

b. Irrigated maize

Irrigated maize and oats forage N uptake significantly increased with increase in sludge application rate (Table 4.8). Irrigated maize forage N uptake decreased across time for all treatments (excluding the 2005/06, where maize was harvested at early stage due to damage by herbicide). Oat forage N uptake, however, did not have similar trend of decreasing across years. Irrigated maize forage N uptake by the 8 Mg ha⁻¹ yr⁻¹ treatment was similar to the uptake by the inorganic fertilizer during the 2004/05 and 2006/07 growing seasons.

Forage N uptake was higher under irrigated systems compared with similar treatments under dryland conditions. According to the three year mean results from this study, forage N uptake was at least twice higher under irrigation compared with similar treatments under dryland conditions. In addition, the overall forage N uptake by the irrigated maize-oat rotation was at least three times higher than similar treatments under dryland conditions. On the other hand, the three year mean irrigated maize-oat rotation forage N uptake was twice

higher than grain uptake for all treatments. This was also similar under the dryland maize production system.

Table 4.8 Irrigated maize-oat rotation forage N uptake from a clay loam soil treated with three sludge application rates, an inorganic fertilizer, and a control during the 2004/05 to 2007/08 growing seasons.

Treatment	Irrigated maize-oat rotation forage N uptake						
	Maize				Oats		
	2004/05	2005/06	2006/07	2007/08	2005	2006	2008
	kg ha ⁻¹						
Control†	318	32	158	141	103	56	101
4 Mg ha ⁻¹	343	43	210	179	111	109	136
8 Mg ha ⁻¹	390	85	299	268	144	122	157
16 Mg ha ⁻¹	545	92	401	397	187	171	217
Inorganic fertilizer	397	84	310	304	272	158	198
LSD	9	6	22	19	7	4	18
CV, %	2	4	5	5	3	2	7

† Zero sludge and inorganic fertilizer applied

NB. In 2005/06 maize was harvested at early stage due to damage by herbicide, thus forage N uptake was very low.

Forage N uptake was higher under irrigated systems compared with similar treatments under dryland conditions. According to the three year mean results from this study, forage N uptake was at least twice higher under irrigation compared with similar treatments under dryland conditions. In addition, the overall forage N uptake by the irrigated maize-oat rotation was at least three

times higher than similar treatments under dryland conditions. On the other hand, the three year mean irrigated maize-oat rotation forage N uptake was twice higher than grain uptake for all treatments. This was also similar under the dryland maize production system.

According to the three year mean data for the irrigated maize-oat rotation given 16 Mg ha⁻¹ yr⁻¹ sludge, grain N export and forage N export was equivalent to the total N contained in 25 and 13.5 Mg ha⁻¹ yr⁻¹ sludge, respectively. This assumes an average sludge N content during the study period (2.56%). Under dryland maize production, however, the N uptake was equivalent to a total N supply from 7 Mg ha⁻¹ yr⁻¹ sludge for forage production and a 3.7 Mg ha⁻¹ yr⁻¹ for grain production (herbicide damaged maize data of 2005/06 excluded). If the N content of the sludge was 3.85%, the amount of sludge needed to satisfy the N demand of irrigated maize-oat rotation would be 16.6 Mg ha⁻¹ yr⁻¹ for forage production and 9 Mg ha⁻¹ yr⁻¹ for grain production. Therefore, this indicates that the upper limit sludge norm is dynamic and is influenced by farming intensity (intensive irrigated vs. dryland), management practices (forage vs. grain production), and N content of sludge. This is clearly presented in Fig. 4.1).

To summarize, under an annual sludge regime of 8 Mg ha⁻¹ on maize, 247 kg, 505 kg, 557 kg and 957 kg of N ha⁻¹ was exported from dryland grain, dryland forage, irrigated grain and irrigated forage respectively (Table 4.9). When the sludge limit was doubled, dryland grain N export increased by 14% and forage

export by just 6%. The increase was much larger under irrigated conditions. Grain N export was increased by 46% and forage export by 40% when the 8 Mg ha⁻¹ sludge limit was doubled.

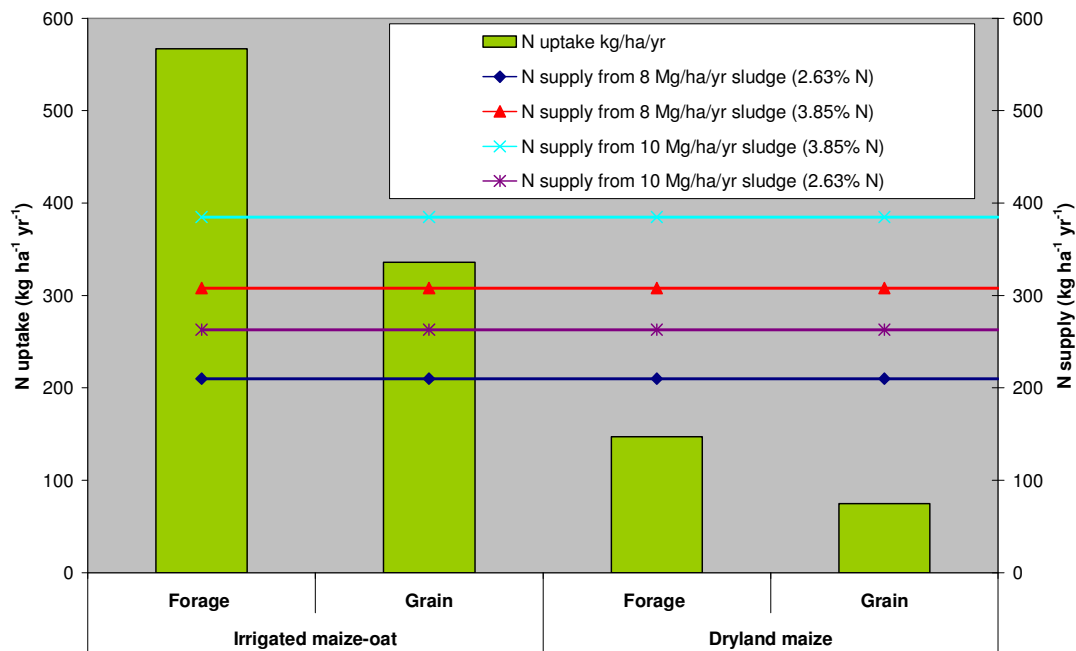


Figure 4.5 Three year cumulative mean N uptake by the 16 Mg ha⁻¹ yr⁻¹ sludge treated irrigated maize-oat rotation and dryland maize (bars) versus total N supply from sludge 8 Mg ha⁻¹ yr⁻¹ (former norm) and 10 Mg ha⁻¹ yr⁻¹ (current norm) with variable N contents (2.56% mean value in his study vs. 3.85% South African sludge mean value (Snyman and Herselman, 2006)).

Clearly, N uptake is higher under irrigated rotation than dryland cropping and under both dryland and irrigated conditions, more N can be exported in forage than grain, so hypothesis 2 was accepted. The ability to add a second crop to

the irrigated system allows for more N export, viz. 27-40% for N export as grain or 42-62% more as forage.

Table 4.9 Cumulative grain and forage N uptake by dryland maize and irrigated maize oat rotation during the 2004-2008 study period.

Cropping system	Treatment	N removed in maize		N removed in oats	
		Grain	Forage	Grain	Forage
		kg ha ⁻¹			
Dryland maize	Control†	155	315		
	4 Mg ha ⁻¹	204	379		
	8 Mg ha ⁻¹	247	505		
	16 Mg ha ⁻¹	281	537		
	Inorganic fertilizer	301	554		
	LSD (5%)	5	17		
	CV, %	1	2		
Irrigated maize-oats rotation	Control†	329	617	121	260
	4 Mg ha ⁻¹	415	732	161	356
	8 Mg ha ⁻¹	557	957	195	423
	16 Mg ha ⁻¹	815	1343	219	575
	Inorganic fertilizer	613	1011	245	628
	LSD (5%)	14	29	4	19
	CV, %	2	2	3	3

† Zero sludge and inorganic fertilizer applied

4.3 Soil profile total N mass balance, residual nitrate and ammonium, and nitrate leaching

4.3.1 Total N mass balance

Mass balance was calculated by subtracting the N exported over the 2004-2008 period from the total applied in fertiliser or sludge. Under dryland conditions, the fertiliser application of 96 kg N ha⁻¹ was close to that exported in the grain, but insufficient to satisfy N export in forage. The appropriate level of sludge would be about 3 Mg ha⁻¹ for dryland grain and just over 5 Mg ha⁻¹ for dryland forage, substantially less than the 8 Mg ha⁻¹ norm (assuming N supply from the 8 Mg ha⁻¹ non-limiting). Grain N uptake in the maize-oat rotation was close to the supply from more than 8 Mg ha⁻¹ sludge, but even double this quantity was insufficient to satisfy N removal in forage (Table 4.10).

Nitrogen mass balance based on sludge input – crop output is not the full story because there are other sources and sinks for N, in particular losses due to leaching, denitrification and volatilisation. Thus it is necessary to measure the change in N store in the soil profile. Under dryland conditions, the changes in total N are small, although accumulation is observable in the 8 and 16 Mg ha⁻¹ sludge treatments (Fig. 4.2). This build up of N should be expected, because crop response to nutrient supply depends on the soil water status (Tisdale et al., 1985). Under water limited conditions crop N recovery is low (Ma et al., 1995), which results in the build up of residual N in the soil profile (Rimski-korsakov et al., 2009).

Table 4.10 Cumulative N mass balances (N supply less uptake) of dryland maize and irrigated maize-oat rotation for the 2004/05 to 2007/08 growing seasons.

Treatment	Cumulative crop N mass balance					
	Dryland maize			Irrigated maize-oat		
	N applied	Grain only	Forage	N applied	Grain only	Forage
	kg ha ⁻¹					
Control†	0	-155	-315	0	-450	-877
4 Mg ha ⁻¹ yr ⁻¹	334	130	-45	327	-249	-761
8 Mg ha ⁻¹ yr ⁻¹	669	422	164	655	-98	-725
16 Mg ha ⁻¹ yr ⁻¹	1336	1055	799	1309	275	-609
Inorganic fertilizer	288	-13	-266	1278	420	-361

† Zero sludge and inorganic fertilizer application

NB

- Nitrogen supply and uptake by maize during the summer of 2005/06 growing season was excluded from the mass balance because crop was harvested at early stage due to damage by herbicide.
- Nitrogen applied and uptake by oats during the 2005/06 is included in the mass balance of irrigated maize-oat rotation.
- Oats was not planted during the winter season of 2006/07 growing season, therefore, the N mass balance of irrigated maize-oat rotation for the same year is only for irrigated maize.
- Negative mass balance indicates that the total N supply was lower than the crop demand.

Under irrigated conditions, all treatments show a drop in total N compared to the initial values in the top 0.6m. There is little change below this depth, although the inorganic fertiliser treatment show an increase in total N. The decline in soil profile N suggests that the supply was insufficient to satisfy the demand of the crop under non-limiting water supply and is likely to be the cause behind maize grain and forage yield reduction across time. (Tables 4.2 and 4.4).

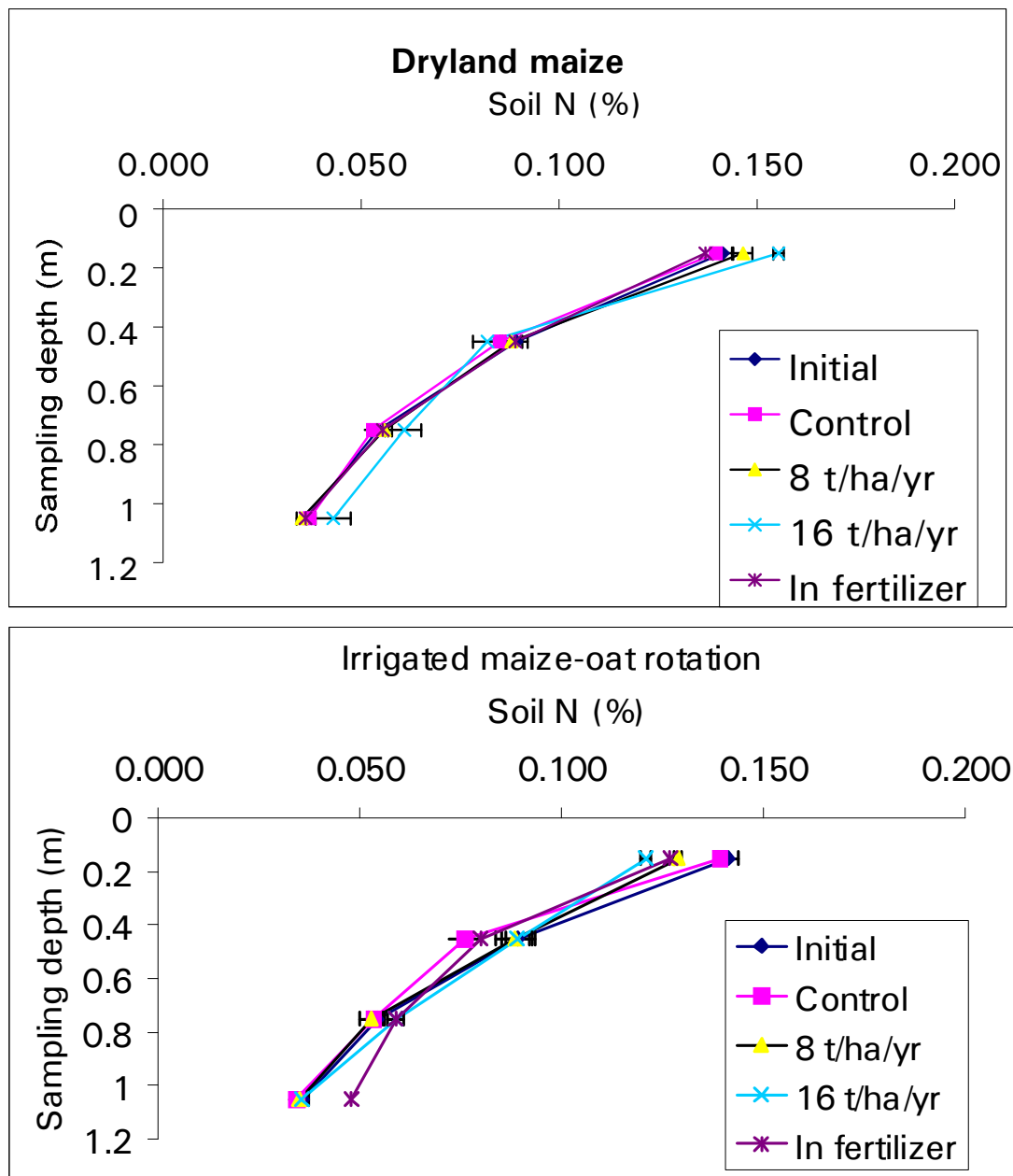


Figure 4.6 Soil profile total N content at the beginning of the study before treatment application (initial) and at the end of three years of study (2006/07).

Although the change in total N in Figure 4.2 look small, they have a major impact on the N mass balance calculation shown in Table 4.11. Total N fell over the study period in most treatments and especially under irrigated conditions. It was

only under dryland conditions receiving sludge according to the norm and above which resulted in a net positive N accumulation over the study period.

Table 4.11 Cumulative N applied less forage N uptake, soil profile N change in storage and mass balance difference between the supply less forage uptake and change in storage for the 2004/05 to 2006/07 growing seasons.

† N applied from sludge and removed with the herbicide damaged maize during the 2005/06 are included in the computation.

Treatment	Cumulative N applied less forage N uptake†		Soil profile N change in storage (final less initial)		Mass balance difference ((Applied less forage uptake) less (change in storage))	
	Irrigated					
	Dryland maize	maize-oat	Dryland maize	Irrigated maize-oat	Dryland maize	Irrigated maize-oat
	kg ha ⁻¹					
Control‡	-209	-666	-244	-772	35	106
8 Mg ha ⁻¹ yr ⁻¹	280	-558	234	-637	46	79
16 Mg ha ⁻¹ yr ⁻¹	828	-431	869	-660	-41	229
Inorganic fertilizer	-76	-143	-137	-267	61	124

‡ Zero sludge and inorganic fertilizer applied

NB

The soil profile N change in storage and mass balance difference are computed for the years 2004/05 to 2006/07 only, because soil profile samples were not taken in 2007/08.

If there were no losses from the system, the sum of N supply in sludge less forage export and change in N storage should sum to zero. The fact that 7 of the 8 treatments show a net positive N mass balance means that there must be unaccounted losses. These losses could be attributed largely to volatilization (Robinson and Polglase, 2000) with leaching (Samaras et al., 2008; Moreno et

al., 1996), and denitrification (Monnett et al., 1978) contributing under the irrigated maize-oat rotation.

Denitrification losses could account as high as 15% of the available inorganic N under irrigated systems (Henry et al., 1999). Similarly, few high rainfall events which followed an irrigation event are likely to have caused leaching losses below the depth of measurement. Sampling errors, N content variation within the sludge matrix, and soil heterogeneity may also have contributed to the observed differences.

4.3.2. Residual nitrate

Table 4.12 shows the nitrate in the soil after the oat harvest and before the sludge or fertiliser was applied to the following maize crop. This residual nitrate from the previous crop is present in the soil prior to the onset of the rainy season, a time when the N demand by the maize crop will be low. Doubling of the upper sludge limit under dryland maize production gives an almost four-fold residual nitrate in the top 0.6 m of soil, making the system more susceptible to leaching. Residual nitrate concentrations were substantially lower under irrigated maize-oat production, due to the greater demand for N combined with potential leaching losses (Table 4.12).

Van Biljon et al. (2008) suggest a lower threshold value for $\text{NO}_3\text{-N}$ for maize production of 70 kg ha^{-1} and an optimum value of $280 \text{ kg NO}_3\text{-N ha}^{-1}$. Thus the 16

Mg ha⁻¹ sludge and the inorganic fertiliser treatments would have sufficient N in the soil for dryland production after two years of the corresponding treatment applications.

Table 4.12 Residual nitrate mass after crop harvest in the top 0.6 m soil stratum of dryland maize and irrigated maize-oat rotation during the 2004/05 to 2006/07 growing seasons

Cropping system	Treatment	Initial	After 2 nd crop harvest in 2004/05	After 2 nd crop harvest in 2005/06	After maize harvest in 2006/07
			kg ha ⁻¹		
Dryland maize	Control†	183	95	172	99
	8 Mg ha ⁻¹	183	69	114	95
	16 Mg ha ⁻¹	183	188	424	355
	Inorganic fertilizer	183	140	289	282
Irrigated maize-oat rotation	Control†	183	81	31	38
	8 Mg ha ⁻¹	183	66	52	51
	16 Mg ha ⁻¹	183	106	22	72
	Inorganic fertilizer	183	150	59	56

† Zero sludge and inorganic fertilizer applied

4.3.3 Residual ammonium

Soil profile ammonium concentration remained similar or less than the initial values even with doubling of the upper limit sludge norm. The only exception was that of the 16 Mg ha⁻¹ yr⁻¹ sludge treatment at the end of the 2004/05 growing season for both dryland and irrigated maize-oat rotation (Table 4.13). Generally, the concentration of ammonium remained more or less similar throughout the soil profile monitored (data not shown). This was mainly due to the adsorption to CEC

sites, as the soil had high and fairly uniform clay content (34-40%) to a depth of 1.2 m (Tesfamariam et al., 2009).

Table 4.13 Residual ammonium mass after crop harvest in the top 0.6 m soil profile of dryland maize and irrigated maize-oat rotation during the 2004/05 to 2006/07 growing seasons

Cropping system	Treatment	Initial	After 2 nd crop harvest in 2004/05	After 2 nd crop harvest in 2005/06	After maize harvest in 2006/07
			kg ha ⁻¹		
Dryland maize	Control†	62	28	52	43
	8 Mg ha ⁻¹	62	56	52	52
	16 Mg ha ⁻¹	62	112	61	61
	Inorganic fertilizer	62	58	54	61
Irrigated maize-oat rotation	Control†	62	58	54	61
	8 Mg ha ⁻¹	62	44	54	61
	16 Mg ha ⁻¹	62	114	34	56
	Inorganic fertilizer	62	62	54	47

† Zero sludge and inorganic fertilizer applied.

4.3.4 Nitrate leaching

Sludge applied at double the annual upper sludge limit slightly increased nitrate concentration in soil solution collected from 0.3 m deep WFDs during the first two months after sludge application (Fig. 4.3a). However the nitrate concentration from the 16 Mg ha⁻¹ yr⁻¹ treatment was substantially lower than the concentration collected from the inorganic fertilizer treatment. Nitrate concentration in soil solution collected from the 0.6 m deep WFDs installed in all sludge treatments

(Fig. 4.3b) were also less than the South African drinking water standards of 44 mg NO₃ L⁻¹ (Korentajer, 1991).

Generally, nitrate concentration in soil solution collected from both 0.3 and 0.6 m WFDs in the first 40 to 70 days was higher from plots which received inorganic N fertilizer. The sudden rise of nitrate concentration in soil solution collected from both 0.3 and 0.6 m deep WFDs under the inorganic fertilizer treatment were linked to fertilizer application events. The first soil solution samples from the 0.6 m deep WFDs were collected once the irrigation was applied according to neutron probe deficit readings to a depth of 1.2 m, 41 days after planting for maize and 80 days for oats. None of the WFDs installed in the dryland maize treatments, however, collected soil solution.

The higher nitrate concentration in soil solution from the inorganic fertilizer treatment compared with sludge treatment should be expected because 50% of the N in the inorganic fertilizer was in nitrate form and remaining 50% as ammonium. In contrast, 75% of the total N in the sludge was in the organic form. The potential for nitrate leaching during the first two months after sludge and inorganic fertilizer application must be minimized by proper irrigation management practices.

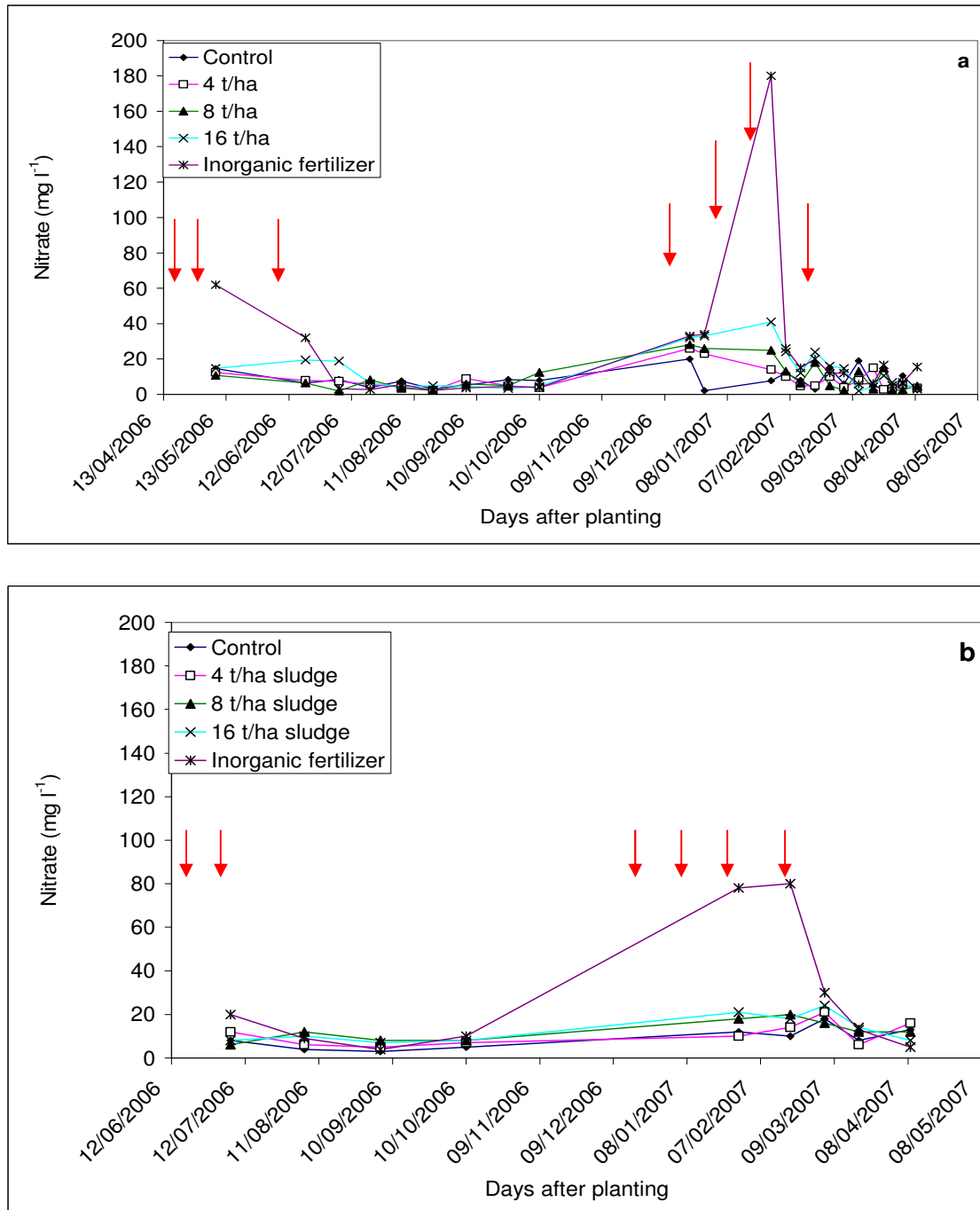


Figure 4.7 Nitrate concentration of leachate collected from 0.3 m (a) and 0.6 m (b) depth wetting front detectors in an irrigated maize-oat rotation during the 2006/07 growing season (arrows indicate inorganic fertilizer application events).

The following hypothesis was accepted:

- High yielding intensive irrigated cropping systems can fully utilize N supply from double the norm and does not seem susceptible to excessive nitrate leaching in the medium term.

The following hypothesis was rejected for dryland maize cropping system during dry years:

- High yielding dryland maize cropping system can fully utilize N supply from double the norm and is not susceptible to excessive nitrate leaching

4.4 Total P mass balance and residual Bray-1P

4.4.1 Total P mass balance

Based on the mass balance calculation of sludge imported less forage exported, total P supply from sludge rates of about $4 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ were approximately similar to the uptake by the irrigated maize-oat rotation (Table 4.14). The 16 Mg ha^{-1} treatment, which was unable to satisfy the N demand of irrigated maize-oat rotation forage production, supplied 655 kg ha^{-1} extra P during the four year study period. This clearly shows that sustainable crop production using sludge must consider both N and P.

Table 4.14 Cumulative P mass balances (supply less forage uptake) of dryland maize and irrigated maize-oat rotation during the 2004/05 to 2007/08 growing seasons.

Treatment	Cumulative crop P balance			
	Dryland maize		Irrigated maize-oat	
	Grain only	Forage	Grain only	Forage
	kg ha ⁻¹			
Control†	-32	-42	-86	-166
4 Mg ha ⁻¹ yr ⁻¹	233	230	152	45
8 Mg ha ⁻¹ yr ⁻¹	500	488	379	222
16 Mg ha ⁻¹ yr ⁻¹	1058	1049	850	655
Inorganic fertilizer	-2	-5	99	-16

† Zero sludge and inorganic fertilizer applied

NB

- Phosphorus supply and forage P uptake by maize during the 2005/06 growing season was excluded from the mass balance because crop was harvested at early stage due to damage by herbicide.
- Phosphorus supply and uptake by oats during the 2005/06 is included in the mass balance of irrigated maize-oat rotation.
- Oats was not planted during the winter season of 2006/07 growing season; therefore, the P mass balance for the same year was only for irrigated maize.
- Negative mass balance indicates that the total P supply was lower than the crop demand.

Based on soil sampling to a depth of 1.2 m, the total P increased across time during the three year study period for all treatments except the control (Table 4.15). Total soil P significantly increased in the top 0.3 m layer of the 8 and 16 Mg ha⁻¹ sludge treatments for both dryland maize and irrigated maize-oat rotation. There was slight increase of P in the 0.3-0.6 m layers of the 8 and 16 Mg ha⁻¹ treatments under the irrigated maize-oat rotation. However, the difference was not significant (Fig. 4.4).

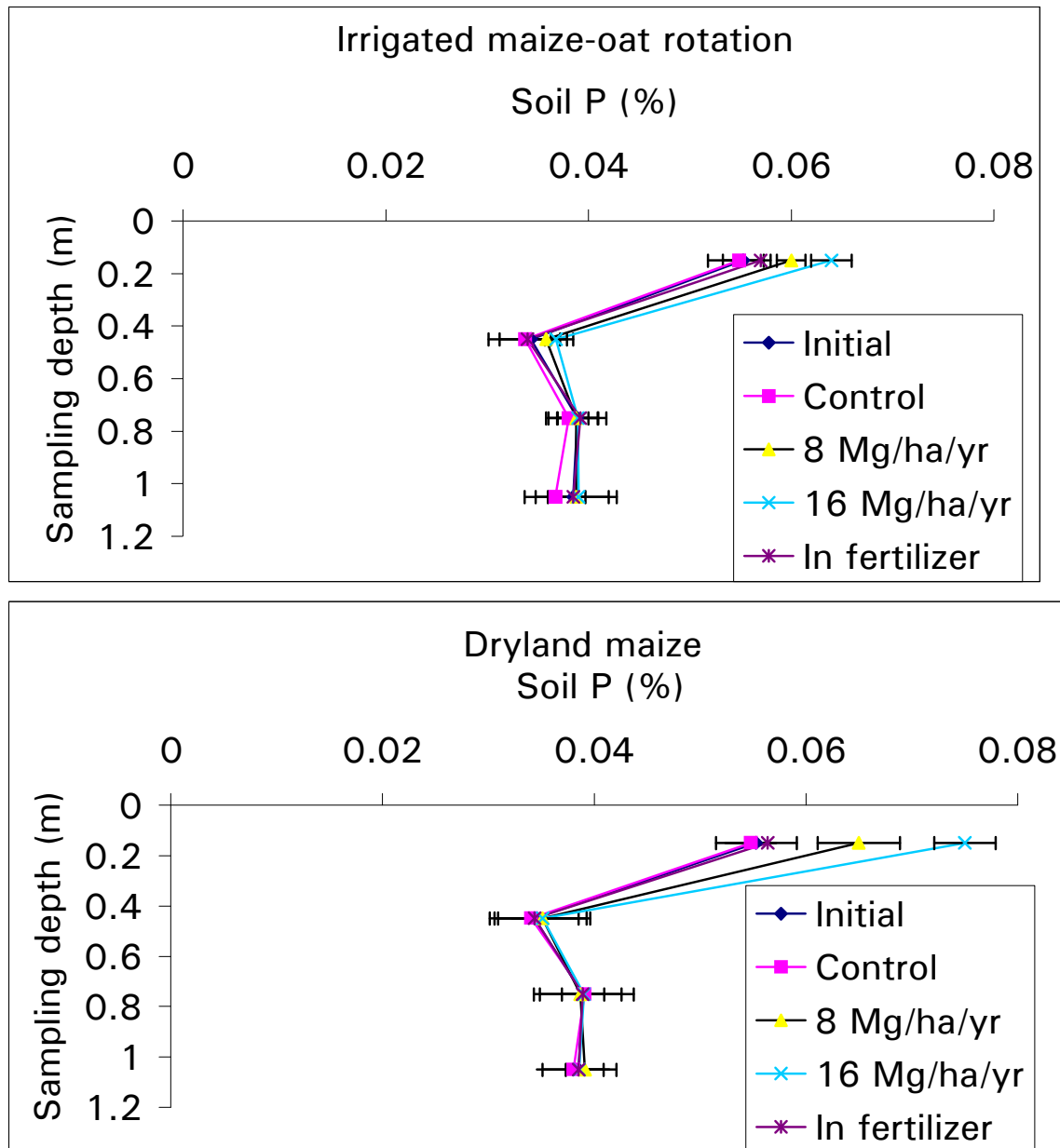


Figure 4.8 Soil profile initial total P content in contrast to P content change following three years of study with three sludge application rates, inorganic fertilizer treatment, and a control (zero sludge and inorganic fertilizer applied).

A mass balance difference was evident between the two mass balance methods (import less export versus final less initial soil profile total P mass) for all

treatments (Table 4.15). Possible sources contributing to the difference include sampling errors, P content variation within the sludge matrix and soil heterogeneity.

Table 4.15 Cumulative soil profile P change in storage and mass balance difference between the supply less forage uptake mass balance and change in storage.

Treatment	Soil profile P change in storage (final less initial)		Mass balance difference ((Supply less uptake) less (change in storage))	
	Dryland maize	Irrigated maize-oat	Dryland maize	Irrigated maize-oat
	kg ha ⁻¹			
Control†	-51	-159	16	37
8 Mg ha ⁻¹ yr ⁻¹	451	266	23	-50
16 Mg ha ⁻¹ yr ⁻¹	916	491	70	63
Inorganic fertilizer	29	67	-24	-62

† Zero sludge and inorganic fertilizer application

NB

- The soil profile N change in storage and mass balance difference are computed for the years 2004/05 to 2006/07 only because soil profile samples were not taken in 2007/08.
- N applied from sludge and removed with the herbicide damaged maize during the 2005/06 are included in the computation.

Therefore, it is clear from this study and previous studies conducted in other countries (Kelling et al., 1977; Peterson et al., 1994; Maguire et al., 2000) that sludge application to agricultural lands according to crop N demand will result in excess total P accumulation in the soil profile.

4.4.2 Soil profile residual Bray-1 extractable P

Under the irrigated maize-oat rotation, Bray-1P remained similar or less than the initial amount in the top 0.6 m soil stratum of the 8 Mg ha⁻¹ sludge treatment. In contrast, Bray-1P mass in the 0.6 m soil stratum of the same treatment under dryland maize production increased across years (Table 4.16). At the end of the 2006/07 growing season, the amount of Bray-1P in the top 0.6 m soil stratum of the 16 Mg ha⁻¹ yr⁻¹ sludge treatment was 2.2 and 2.4 times higher than the initial mass for irrigated maize-oat rotation and dryland maize, respectively. It was interesting to note that the amount of Bray-1P at the end of the 2006/07 growing season for the irrigated maize-oat rotation and dryland maize control treatments was 1.17 and 1.99 times higher than the initial mass, respectively. This shows that there was also release of Bray-1P from the soil organic matter due to mineralization and/or desorption from the soil particles.

Generally, the background concentration of Bray-1 P in the top 0.3 m soil depth of the study site (29 mg kg⁻¹) was within the optimum range required for most crops, (25-30 mg kg⁻¹), as defined by Sims and Pierzynski (2000). It was, however, higher than the concentration required for optimal maize production under South African soils for similar soil types as described by Schmidt et al. (2007) (20.1 mg kg⁻¹). At the end of the three year study period, the concentration of Bray-1P in all dryland maize treatments (> 61 mg kg⁻¹) and the 16 Mg ha⁻¹ yr⁻¹ sludge treated irrigated maize (75 mg kg⁻¹) was above the optimum concentration as described by Sims and Pierzynski (2000) (50 mg kg⁻¹). Concentrations

exceeding the optimum have negative environmental impacts due to an increase in soluble P concentration in runoff losses. According to Andraski and Bundy (2003), the concentration of dissolved P in runoff has a linear relationship with top-soil test P. A Bray-1P concentration of 250 mg kg^{-1} could yield 1 mg P L^{-1} in soil solution, which is often used as a bench mark for water pollution (Sims and Pierzynski, 2000).

Table 4.16 Residual Bray-1P mass after crop harvest in the top 0.6 m soil profile of dryland maize and irrigated maize-oat rotation during the 2004/05 to 2006/07 growing seasons

Cropping system	Treatment	Initial	After 2 nd crop harvest in 2004/05	After 2 nd crop harvest in 2005/06	After maize harvest in 2006/07
kg ha^{-1}					
Dryland maize	Control†	147	186	209	272
	8 Mg ha^{-1}	147	178	240	292
	16 Mg ha^{-1}	147	167	383	357
	Inorganic fertilizer	147	136	264	332
Irrigated maize-oat rotation	Control†	147	222	144	172
	8 Mg ha^{-1}	147	146	137	143
	16 Mg ha^{-1}	147	104	172	324
	Inorganic fertilizer	147	142	117	87

† Zero sludge and inorganic fertilizer applied

The amount of Bray-1P under the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treated irrigated maize-oat rotation remained similar or less than the initial amount throughout the study period. This is in contrast to the net positive total P mass balance of the 8 Mg ha^{-1} treatment (Table 4.14). The same treatment under dryland maize, however, had

twice higher Bray-1P in the 0-0.6 m soil stratum at the end of the three year study period. It was interesting to note that the difference in three year cumulative crop P uptake between these two treatments (148 kg ha^{-1}) was close to the difference in residual Bray-1P mass in the top 0.6 m soil stratum (149 kg ha^{-1}). Similarly, the difference in three year cumulative crop P uptake between irrigated maize-oat rotation and dryland maize control treatment (87 kg ha^{-1}) was close the difference in residual Bray-1P mass in the top 0.6 m soil stratus (100 kg ha^{-1}). This shows that the total mass of plant available P from the control and $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ treated irrigated maize-oat rotation was similar to that of the dryland maize during the study period.

The $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treated irrigated maize-oat rotation, however, had 1.4 times higher plant available P (plant uptake plus residual Bray-1P) than the same treatment under dryland maize production. This indicates that at higher sludge application rates plant available P could be affected by the availability of water.

The percentage of plant available P (Bray-1P) as opposed to the total P applied during the 2004/05 to 2006/07 growing seasons (Table 4.17, column 6) was higher under irrigated sludge treatments compared with similar treatments under dryland maize production. Bray-1P accounted only 7 and 12% of the total P from the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatments under the dryland maize and irrigated maize oat rotation, respectively. Similarly, Bray-1P was 11 and 37% of the total P from the $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatments under dryland maize and irrigated maize-

oat rotation, respectively. The remaining fraction is either in organic form or precipitated in the form of Fe-P because the sludge used in this study was treated with FeCl_3 at the wastewater treatment plant. The addition of FeCl_3 is responsible for the direct interaction and precipitation of a Fe-P mineral phase and adsorption onto a newly formed Fe hydroxide surface reducing the solubility of P (Huang et al., 2007; O'Connor et al., 2004; Huang and Shenker, 2004; Elliott et al., 2002; Maguire et al., 2000).

Table 4.17 Cumulative P applied, total plant available P, normalized plant available P, and the percentage of Bray-1P in contrast to the total P applied during the 2004/05 to 2006/07 growing seasons.

Cropping system	Treatment	Total P applied†	Total plant available P‡	Normalized plant available P§ kg ha ⁻¹	Bray-1P % of total P applied¶
Dryland maize	Control	0	307		
	8 Mg ha ⁻¹ yr ⁻¹	526	343	36	7
	16 Mg ha ⁻¹ yr ⁻¹	1051	422	115	11
	Inorganic fertilizer	60	387	80	133
Irrigated maize-oat rotation	Control	0	294		
	8 Mg ha ⁻¹ yr ⁻¹	415	342	48	12
	16 Mg ha ⁻¹ yr ⁻¹	830	600	306	37
	Inorganic fertilizer	200	282	-12	-6

† Total P applied is the sum of all P applied to the various treatments during the 2004/05 to 2006/07 growing seasons.

‡ Total plant available P is computed as the sum of crop P uptake during the 2004/05 to 2006/07 growing seasons and residual Bray-1P at the end of 2006/07 growing season.

§ Normalized plant available P is computed by subtracting the total plant available P of the control treatment from the total plant available P of each treatment.

¶ Bray-1P % is the percentage of Bray-1P in contrast to the total P applied to the various treatments.

In summary, soil response to various sludge rates during the three year field study show that continuous sludge applications at high rates to satisfy crop N demand resulted in the accumulation of total P and Bray-1P. Farina and Channon (1987) recommend P fertilization to build up the soil P status regardless of seasonal variation in crop yield, while Venter (1984) is in support of fertilizing crops according to the expected target yield. In both cases some standardized monitoring needs to be implemented in order to evaluate the soil test P status so that it would not exceed the environmental loss threshold value.

Therefore, the following hypothesis was accepted:

- Sludge application according to crop N demand results in the accumulation of total and plant available P in the soil profile across time.

4.5 Conclusions

Doubling of the annual 8 Mg ha^{-1} sludge limit significantly increased grain and forage yield of both dryland maize and the irrigated maize-oat rotation. The increase in yield was small in the case of dryland crops, and only a small percentage of the extra N applied in the sludge was exported. Grain N export was increased by 46% and forage export by 40% when the 8 Mg ha^{-1} sludge limit was doubled.

Grain and forage yield of the irrigated maize-oat rotation displayed a decreasing trend during the study period. This may have been due to lack of N. In all sludge

treatments the amount of N exported was greater than that applied, except for grain export at 16 Mg ha⁻¹ sludge. This resulted in crop nutrient consumption from soil reserve, causing a decline of total N. In contrast, despite the increase in maize forage and grain yield reported with increase in sludge rate under dryland maize production, a net positive total N accumulation was evident for sludge applications of 8 Mg ha⁻¹ yr⁻¹ and higher.

Export of N in forage was approximately double that exported in the grain. Nitrogen exported in the forage from the irrigated rotation was at least three times higher than similar treatments under dryland maize production. Mass balance calculations involving the N applied, N exported in product and change in N storage revealed substantial losses from the system, particularly under irrigated conditions. These losses were large even in the control and fertiliser treatments, but highest in the 16 Mg ha⁻¹ sludge treatment.

Soil solution samples collected from 0.3 and 0.6 m deep wetting front detectors under the irrigated maize-oat rotation indicated higher risk of ground water pollution through nitrate leaching from the inorganic fertiliser treatment compared sludge treatments. The residual nitrate left in the soil after the maize crops was higher under dryland than irrigated conditions, especially for the 16 Mg ha⁻¹ application rate. The potential for nitrate leaching was clear, although the drainage volumes under dryland conditions were unknown.

Sludge applied according to crop N demand would result in total and Bray-1P accumulation in the soil profile and could cause a threat to surface water body pollution through time. Ideally the upper sludge limit to satisfy crop N demand should be dynamic because it depends on the sludge N content, the intensity of cropping, and the availability of water. But ultimately the maximum amount of sludge applied to an area will depend on the accumulation of P and the risk this poses for pollution as long as the threat from other contaminants is minimal.

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CHAPTER 5

PERENNIAL DRYLAND PASTURE - WEEPING LOVEGRASS

(*Eragrostis curvula* L.)

The following hypotheses are tested in this chapter.

HYPOTHESIS 1

- Sludge loading above the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ norm could improve weeping lovegrass hay yield, crude protein content, and water use efficiency.

HYPOTHESIS 2

- The ideal sludge loading rate to satisfy weeping lovegrass N demand is dynamic and could exceed the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge norm.

HYPOTHESIS 3

- a. Under high hay yield production conditions, N supply from double the norm can be fully utilized and are not prone to excessive nitrate leaching.

HYPOTHESIS 4

- Sludge application according to weeping lovegrass N demand results in the accumulation of total and Bray-1P in the soil profile.

5.1 Hay yield, crude protein content, and water use efficiency

5.1.1 Hay yield

Doubling of the sludge upper limit norm significantly improved weeping lovegrass annual hay yield except during 2005/06, where the difference was not statistically significant (Table 5.1). Hay yield was generally higher for the first cut compared with the second each year, except for the 2007/08 growing season, where the second cut was higher than the first (Table 5.2). It was interesting to note that weeping lovegrass annual hay yield from the 8 Mg ha⁻¹ yr⁻¹ sludge treatment was similar to that of the inorganic fertilizer. The only exception was during the wet 2007/08 growing season, where hay yield from the inorganic fertilizer was significantly higher than the 8 Mg ha⁻¹ treatment.

Table 5.1 Annual hay yield of weeping lovegrass as affected by three sludge application rates, inorganic fertilizer, and control.

Treatments	Weeping lovegrass hay yield			
	2004/05	2005/06	2006/07	2007/08
	Mg ha ⁻¹			
Control†	10.18	9.08	6.97	8.15
4 Mg ha ⁻¹ yr ⁻¹	10.74	9.64	10.05	12.70
8 Mg ha ⁻¹ yr ⁻¹	14.10	12.50	11.87	14.90
16 Mg ha ⁻¹ yr ⁻¹	15.08	12.78	12.36	17.31
Inorganic fertilizer ‡	13.78	12.56	11.78	16.50
LSD (5%)	0.61	0.43	0.46	0.73
CV, %	3	3	3	3

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N and 80 kg P per hectare per annum applied

In general, hay yield throughout the study period was much higher than the long term average values of 6 Mg ha⁻¹ reported by Dickinson et al. (2004) from Hutton soils receiving similar annual rainfall. This is most probably due to a better nutrient status of the soil from this study compared with the soils reported by Dickinson et al. (2004).

The relatively low hay yield from the second cut compared with the first cut of each year was mainly attributed to the relatively low rainfall experienced during the second half of the season (Fig. 5.1). This is because a relatively higher rainfall experienced during the second cut of the 2007/08 growing season rendered higher forage yield than the first cut. The amount of rainfall and weeping lovegrass hay yield of a control and three sludge rate treatments are presented in a 3 D graph (Fig. 5.2). The graph vividly shows that weeping lovegrass hay yield response to sludge rate increases with increase in rainfall. This coincides with the explanation by Miles and Manson (2000) and Dickinson et al. (2004), who report that weeping lovegrass response to N increases with increase in the availability of water (rainfall).

Table 5.2 Weeping lovegrass hay yield per cut of three sludge application rates, an inorganic fertilizer, and a control during the 2004/05 to 2007/08 growing seasons.

† Zero sludge and inorganic fertilizer applied

Treatments	Weeping lovegrass hay yield per cut							
	2004/05		2005/06		2006/07		2007/08	
	First cut	Second cut	First cut	Second cut	First cut	Second cut	First cut	Second cut
	Mg ha ⁻¹							
Control†	6.69	3.49	4.96	4.12	4.09	2.88	4.00	4.15
4 Mg ha ⁻¹ yr ⁻¹	6.97	3.77	5.51	4.13	5.96	4.09	5.94	6.76
8 Mg ha ⁻¹ yr ⁻¹	8.65	5.45	7.39	5.11	7.34	4.53	6.12	8.78
16 Mg ha ⁻¹ yr ⁻¹	9.14	5.94	7.66	5.21	7.74	4.62	6.67	10.64
Inorganic fertilizer‡	8.49	5.29	7.44	5.12	7.16	4.62	7.16	9.34
LSD (5%)	0.41	0.25	0.26	0.26	0.26	0.31	0.38	0.64
CV, %	3	3	3	4	3	5	4	5

‡ 100 kg N and 40 kg P per hectare per annum per cut applied

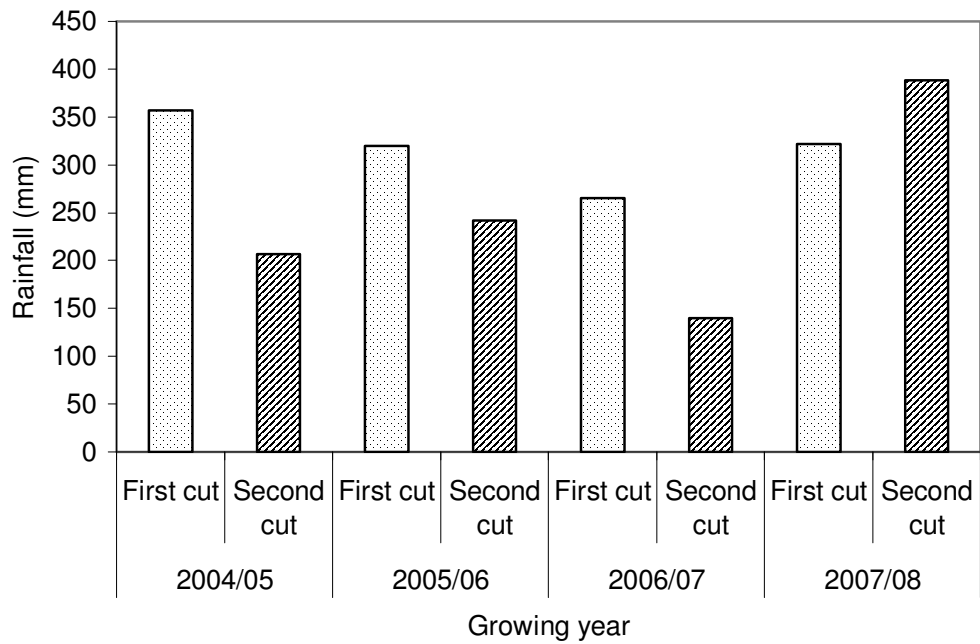


Figure 5.1 Rainfall distribution during the first and second cuts of weeping lovegrass planted during the 2004/05 to 2007/08 growing seasons, at ERWAT, Ekurhuleni district, South Africa.

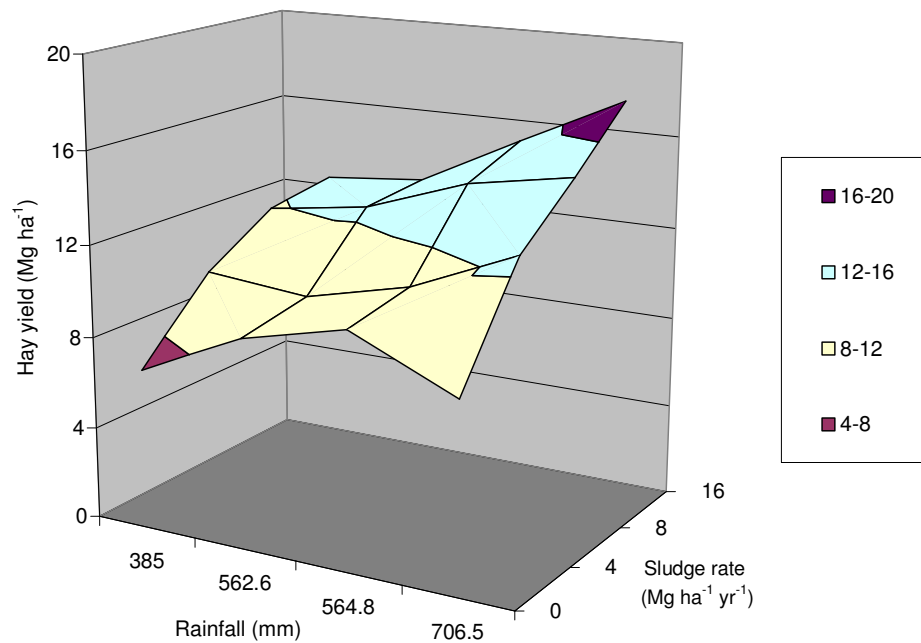


Figure 5.2 Weeping lovegrass hay yield as affected by rainfall amount and sludge application rate.

5.1.2 Crude protein content

Doubling of the annual upper sludge limit improved weeping lovegrass crude protein content, except for the second cuts of the 2006/07 and 2007/08 growing seasons (Table 5.3). It was also evident that the level of crude protein from the second cut of each year was lower than that of the first. Crude protein content is the most limiting constituent for animal performance feeding on pasture and was estimated by multiplying hay N content by a factor of 6.25 as reported by Meissner et al. (2000).

The increase in the level of crude protein with doubling of the 8 Mg ha^{-1} sludge limit was mainly attributed to the increase in the availability of N, because crude protein of weeping lovegrass increases with fertilization under nutrient limiting conditions (Masters and Britton, 1990). The low crude protein observed during the second cut of the first three years was mainly due to the low rainfall, because weeping lovegrass crude protein content decreases under drought conditions (McFarland, 1999). This is most probably due to the translocation of N from leaves to roots, which is common in warm season grasses during drought periods (Heckathron and Delucia, 1999). The low crude protein content observed during the second cut of 2008 was, however, mainly attributed to a dilution effect as a result of the high biomass production (Table 5.2, column 9).

Generally, the crude protein levels of all the treatments were within the ranges reported by Strickland (1973) for weeping lovegrass ($63\text{-}175 \text{ g kg}^{-1}$). It was also above the minimum crude protein requirements of ruminants ($70\text{-}80 \text{ g kg}^{-1}$) Meissner et al., (2000), except for the second cut of the control treatment during the 2007/08 growing season.

Table 5.3 Crude protein content of weeping lovegrass as affected by three sludge application rates, an inorganic fertilizer treatment, and a control.

† Zero sludge and inorganic fertilizer applied

Treatments	Weeping lovegrass crude protein content							
	2004/05		2005/06		2006/07		2007/08	
	First cut	Second cut	First cut	Second cut	First cut	Second cut	First cut	Second cut
	g kg ⁻¹							
Control†	121	94	96	79	106	85	77	62
4 Mg ha ⁻¹ yr ⁻¹	126	98	100	86	107	89	104	77
8 Mg ha ⁻¹ yr ⁻¹	128	102	100	89	108	85	104	77
16 Mg ha ⁻¹ yr ⁻¹	155	112	102	99	112	85	107	77
Inorganic fertilizer‡	136	106	104	94	109	85	107	82
LSD (5%)	4	3	3	4	6	1	3	1
CV, %	2	2	2	3	2	2	2	1

‡ 200 kg N and 80 kg P per hectare per annum applied

5.1.3 Effect of sludge application rate on rainfall use efficiency

Sludge applied at double of the upper limit norm significantly improved the annual rainfall use efficiency compared with lower rates (Table 5.4). The significant improvement in annual water use efficiency was due to the significant increase in hay yield per unit water used for each cut, or for one of the two cuts in a season (Table 5.5). Rainfall use efficiency (RUE) is a factor which indicates the productivity of an ecosystem (Guevara et al., 2005). This depends on soil and vegetation condition and its dynamic status (Le Hou  rou, 1984).

During the 2006/07 growing season, the water use efficiency of treatments that received 8 and 16 Mg ha⁻¹ sludge and an inorganic fertilizer was highest compared with similar treatments during the other three growing seasons, which experienced higher rainfall and hay yield. Nevertheless, each treatment had relatively lower hay yield compared with similar treatment due to the low rainfall experienced during the specified year. Generally, the rainfall use efficiency from this study was much higher than the ranges reported by Guevara et al. (2005) for weeping lovegrass (3.7-10 kg DM ha⁻¹ year⁻¹ mm⁻¹) in Argentina. This is most probably due to higher water holding capacity and nutrient status of the soil from this study among other factors.

Table 5.4 Annual rainfall use efficiency of weeping lovegrass as affected by three sludge application rates, an inorganic fertilizer, and a control.

Treatments	Annual rainfall use efficiency			
	2004/05	2005/06	2006/07	2007/08
	kg ha ⁻¹ mm ⁻¹			
Control†	18	16	18	12
4 Mg ha ⁻¹ yr ⁻¹	19	17	26	18
8 Mg ha ⁻¹ yr ⁻¹	25	21	31	21
16 Mg ha ⁻¹ yr ⁻¹	27	23	33	24
Inorganic fertilizer‡	24	22	31	23
LSD (5%)	1	1	1	1
CV, %	3	2	3	4

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N and 80 kg P per hectare per annum applied

To summarize, sludge applied at double the norm improved weeping lovegrass hay yield, crude protein content, and water use efficiency compared with lower rates under the prevailing climatic conditions. Therefore, the following hypothesis was accepted:

- Sludge application above the 8 Mg ha⁻¹ yr⁻¹ norm could improve dryland pasture (weeping lovegrass) hay yield, crude protein content, and water use efficiency.

Table 5.5 Rainfall use efficiency of weeping lovegrass per cut as affected by three sludge application rates, an inorganic fertilizer, and a control.

† Zero sludge and inorganic fertilizer applied

Treatments	Weeping lovegrass seasonal rainfall use efficiency							
	2004/05		2005/06		2006/07		2007/08	
	First cut	Second cut	First cut	Second cut	First cut	Second cut	First cut	Second cut
	kg ha ⁻¹ mm ⁻¹							
Control†	19	17	16	17	17	21	13	11
4 Mg ha ⁻¹ yr ⁻¹	19	18	17	17	25	29	19	17
8 Mg ha ⁻¹ yr ⁻¹	24	26	22	21	30	33	19	23
16 Mg ha ⁻¹ yr ⁻¹	26	28	24	22	32	33	21	27
Inorganic fertilizer‡	24	25	23	21	30	33	22	24
LSD (5%)	1	1	1	1	1	2	1	2
CV, %	3	4	3	4	3	4	5	6

‡ 200 kg N and 80 kg P per hectare per annum applied

5.2 Hay N uptake

Weeping lovegrass annual hay N uptake increased significantly with doubling of the annual upper sludge application limit (Table 5.6). Annual hay N uptake from the control treatment tended to decrease over time. Other treatments, however, did not show a specific trend across years. Crop N uptake was higher during the first cut than the second, except in 2008, where the second cut was higher for the 8 and 16 Mg ha⁻¹ yr⁻¹ sludge treatments (Table 5.7).

Table 5.6 Annual weeping lovegrass N uptake from three sludge application rates, inorganic fertilizer treatment, and a control during the 2004/05 to 2007/08 growing seasons.

Treatments	Weeping lovegrass hay N uptake			
	2004/05	2005/06	2006/07	2007/08
	kg ha ⁻¹			
Control†	182	129	109	91
4 Mg ha ⁻¹ yr ⁻¹	200	145	160	182
8 Mg ha ⁻¹ yr ⁻¹	266	192	188	210
16 Mg ha ⁻¹ yr ⁻¹	333	208	202	245
Inorganic fertilizer‡	275	201	189	245
LSD (5%)	10	8	8	10
CV, %	3	3	3	3

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N and 80 kg P per hectare per annum applied

1 Table 5.7 Weeping lovegrass hay N uptake per cut from three sludge application rates, inorganic fertilizer
2 treatment, and a control

Treatment	Weeping lovegrass hay N uptake							
	2004/05		2005/06		2006/07		2007/08	
	First season	Second season	First season	Second season	First season	Second season	First season	Second season
	kg ha ⁻¹							
Control†	130	52	76	53	70	40	50	41
4 Mg ha ⁻¹ yr ⁻¹	141	60	88	57	102	58	99	84
8 Mg ha ⁻¹ yr ⁻¹	177	89	118	73	127	62	102	108
16 Mg ha ⁻¹ yr ⁻¹	227	106	125	83	139	63	115	131
Inorganic fertilizer‡	185	90	124	77	126	63	123	123
LSD (5%)	8	3	5	5	6	4	7	8
CV, %	3	3	3	5	4	5	5	6

3 † Zero sludge and inorganic fertilizer applied
4 ‡ 200 kg N and 80 kg P per hectare per annum applied
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The reported increase in N uptake for every increment in sludge rate depicts the increase in the availability of nitrogen. This is because nitrogen uptake is a more sensitive indicator of nitrogen availability from nitrogenous fertilizer sources (Kiemnec et al., 1987). This increase in plant N availability enhanced weeping lovegrass hay yield, because N is considered the key element for dry matter production (Miles and Manson, 2000) and dry matter production of weeping lovegrass increases with the availability of nitrogen (Rethman et al., 1984). In contrast, the decline in N uptake over time for the control treatment was mainly due to the decline in soil N availability.

Assuming that 50 percent of the organic N is released during the first year of sludge application (Snyman and Herselman, 2006) followed by 8, 3, 1, and 1 percent of the organic N applied during the first year being released during the second, third, fourth, and fifth years, respectively (Sullivan and Cogger, 2000). The four year mean N uptake by the 16 Mg ha⁻¹ sludge treatment was equivalent to an N supply from a 15.8 Mg sludge ha⁻¹ during the first year of application (Fig. 5.3). The rate decreased during the following 8 years and reached an equilibrium application rate of 13.2 Mg ha⁻¹ from the 10th year on. If the sludge N content was 3.85%, the annual sludge application rate could have equilibrated at 8.8 Mg ha⁻¹ from the 10th year on (Fig. 5.3). The decline in sludge application rate across years was mainly due to the N carry over effects from previous year's applications.

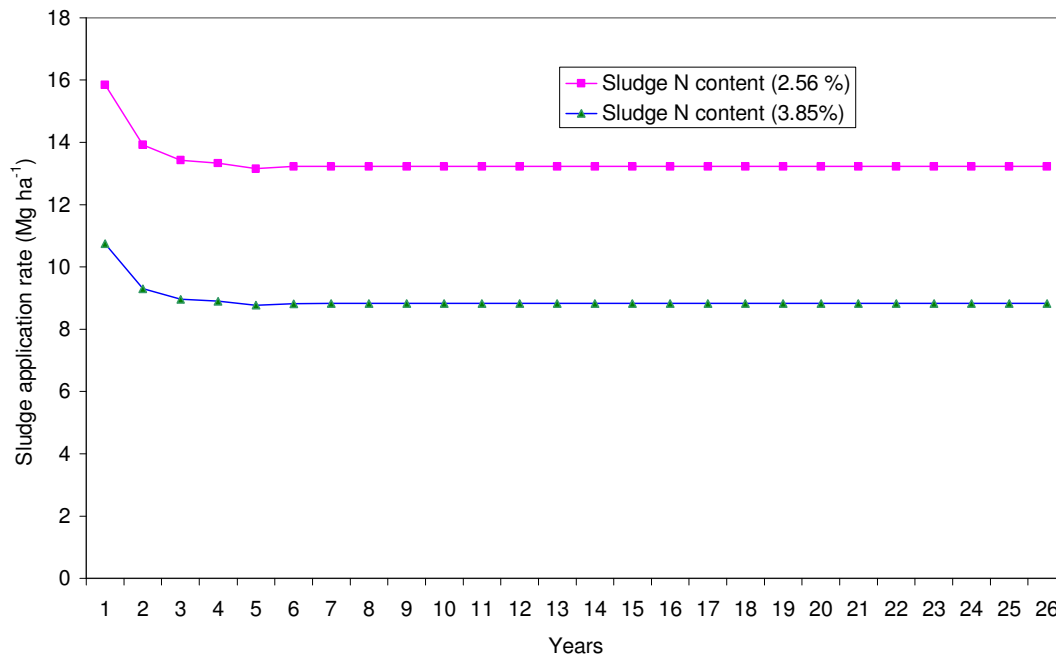


Figure 5.3 Sludge application rate to satisfy four year mean weeping lovegrass N demand (247 kg N ha^{-1}) as affected by sludge N content and N carry over effects.

The higher hay N uptake during the first cut compared with the second was mainly attributed to the availability of higher rainfall during the first cut of the year (Fig. 5.1). The event during the 2007/08 growing season, where higher rainfall during the first cut from the 8 and 16 Mg ha^{-1} sludge treatments produced higher yield than the second proved this reality. The availability of water plays a significant role in the mineralization of organic nitrogen (Jansson and Persson, 1982; Vinten and Smith, 1993), nitrification of ammonium (Breuer et al., 2002) and crop uptake (Mackay and Barber, 1985).

Therefore, the following hypothesis was accepted:

- The ideal sludge application rate to satisfy dryland pasture (Weeping lovegrass) N demand is dynamic and could exceed the 8 Mg ha⁻¹ yr⁻¹ sludge norm.

5.3 Soil profile total N mass balance, nitrate leaching, residual nitrate and ammonium

5.3.1 Total N mass balance

Based on the mass balance calculation of N imported with sludge less N exported with hay, total N from sludge applied according to the 8 Mg ha⁻¹ limit was close to being but not quite sufficient to satisfy weeping lovegrass N demand (net negative mass balance) (Table 5.8). Negative mass balances of all but the 16 Mg ha⁻¹ treatment shows that the crop utilized N from the soil reserve. Therefore, total N supply from sludge with 2.56% mean N applied according to the 8 Mg ha⁻¹ norm is not sufficient for optimal weeping lovegrass hay production under the prevailing climatic and soil conditions. Doubling of the upper limit sludge norm, however, resulted in a net positive mass balance. This indicates that the total N supply surpassed hay N uptake. The excess N from double the norm was accumulated mainly in the 0-0.1 m soil layer, because the sludge was surface applied (Fig. 5.4).

Table 5.8 Cumulative N supply (CUM NS)), uptake (CUM NU), and mass balance of a weeping lovegrass treated with three sludge application rates, inorganic fertilizer, and a control.

Treatment	Four years cumulative mass balance				
	CUM-NS	CUM-NU	CUM-NS less CUM-NU	Change in soil N storage	Mass balance difference
	kg ha ⁻¹				
Control†	0	511	-511	-600	89
4 Mg ha ⁻¹ yr ⁻¹	409	687	-278	-381	103
8 Mg ha ⁻¹ yr ⁻¹	818	856	-38	-203	165
16 Mg ha ⁻¹ yr ⁻¹	1637	988	649	330	319
Inorganic fertilizer‡	800	910	-110	-271	161

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N per hectare per annum applied

Soil profile sampling to a depth of 0.8 m (change in soil storage) (Table 5.8) showed similar trends to the supply less uptake mass balance. According to this profile analyses, doubling of the annual sludge upper limit resulted in a net positive mass balance while sludge applied according to the norm had a net negative mass balance. Interestingly, however, there was a net positive difference between the two mass balances: supply less uptake (CUM_NS less CUM-NU) and change in storage between final and initial soil profile N contents (Change in soil N storage). This difference increased with increase in sludge application rate (Mass balance difference) (Table 5.8).

The most probable cause for the mass balance difference was ammonia volatilization losses, which is not accounted for in the N import less export mass

balance. This is because the sludge used in this study was anaerobically digested with about 20-25% of total N in ammonium form. In addition, the sludge was surface applied. Previous studies conducted by Adamsen and Sabey (1987) showed that 40.3% of the $\text{NH}_4\text{-N}$ from surface applied sludge could be lost as NH_3 gas during the first two weeks of its application. This was in contrast to 0.35% loss from an incorporated sludge reported in the same study. Similarly, other studies have also shown that on average 89% 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 possible sources of the differences include sampling errors, N content variation within the sludge matrix, soil heterogeneity, and probably denitrification (which is not dominant under dryland cropping), and leaching (which could have been insignificant due to the low rainfall experienced).

Comparison of mass balance differences was conducted between dryland maize forage production, where sludge was incorporated immediately after application, (Table 4.10) and weeping lovegrass, sludge surface applied, (Table 5.8). The mass balance difference of weeping lovegrass for the 8 and 16 Mg ha^{-1} sludge treatments was more than four times higher than similar treatments under dryland maize conditions. In addition, the wetting front detectors buried at 0.3 m depth responded only four times during the four year study period, each event with nitrate concentrations of less than 14 mg L^{-1} . This implies the presence of high ammonia volatilization losses from surface application under the weeping

lovegrass production, supporting the previous argument on the most probable reason for the mass balance difference.

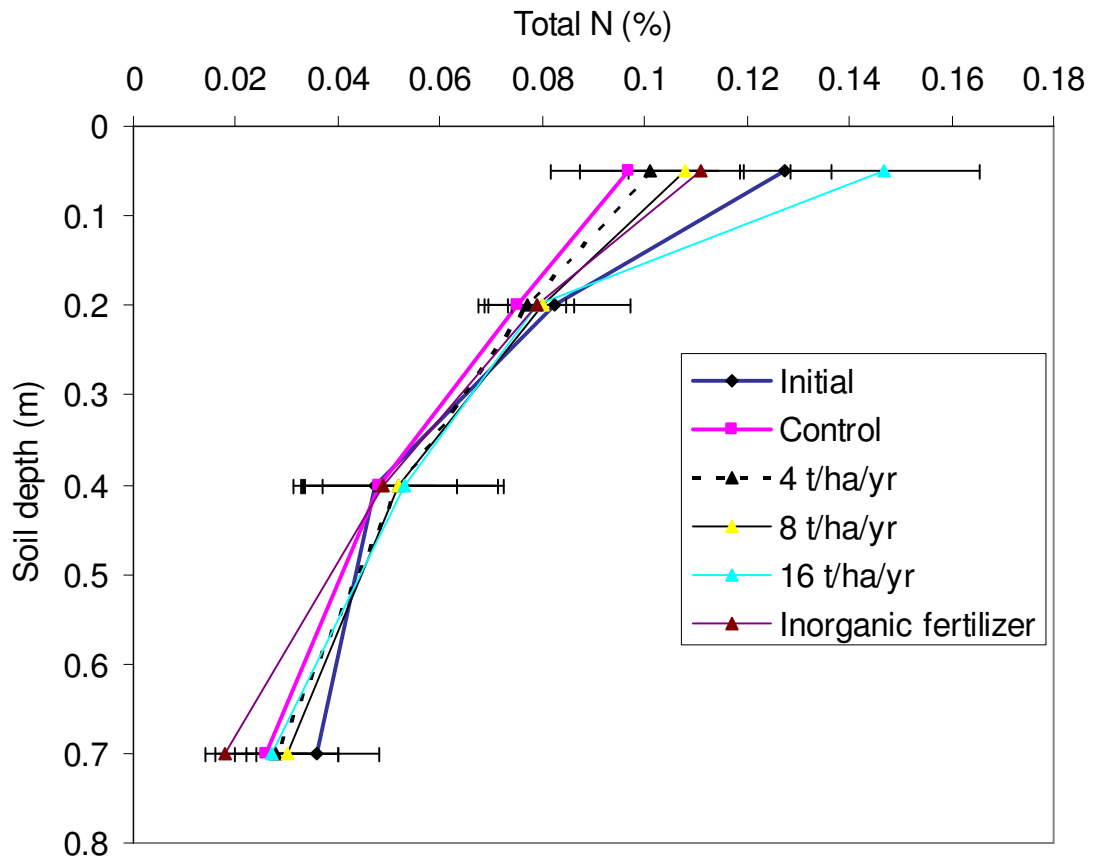


Figure 5.4 Initial soil profile total N and after four years of study with three sludge rates (4, 8, and 16 Mg ha⁻¹ yr⁻¹), an inorganic fertilizer (200 kg N ha⁻¹ yr⁻¹), and a control.

5.3.2 Residual nitrate and nitrate leaching

Residual nitrate in the top 0.5 m soil layer of all treatments including sludge applied at double the norm remained less than the initial amount throughout the study period (Table 5.9). The reduction in nitrate content of all treatments was evident in the 0-0.3 m soil stratum, while the content in the 0.3-0.8 m layer remained similar to initial values (Fig. 5.5). It was also evident that sludge applied at double the norm had relatively lower residual nitrate than lower sludge rates in 2005/06, but higher at the end of 2007/08 growing season growing season.

Table 5.9 Residual nitrate mass in the top 0.5 m soil stratum of weeping lovegrass plots treated with three sludge rates, an inorganic fertilizer, and a control treatment.

Treatment	Initial	After 2nd hay cut in 2005/06	After 2nd hay cut in 2007/08
		kg ha ⁻¹	
Control†	96	60	18
4 Mg ha ⁻¹ yr ⁻¹	96	62	18
8 Mg ha ⁻¹ yr ⁻¹	96	52	16
16 Mg ha ⁻¹ yr ⁻¹	96	51	58
Inorganic fertilizer‡	96	33	51

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N and 40 kg P per hectare per annum applied

Although residual nitrate at the end of the study was relatively higher for the 16 Mg ha⁻¹ yr⁻¹ sludge treatment than other treatments, it was still low and was almost half of the initial mass. The main reason for monitoring residual nitrate after harvest was that nitrate leaching under dryland conditions usually takes place in the beginning of the rainy season, especially before active root nutrient uptake in

the presence of high rainfall and residual soil nitrate. Nitrate is susceptible to diffusion and transport through mass flow with soil water because there is little tendency for soil colloids to absorb nitrate, which is negatively charged, as they mostly possess a net negative charge (Cameron and Haynes, 1986).

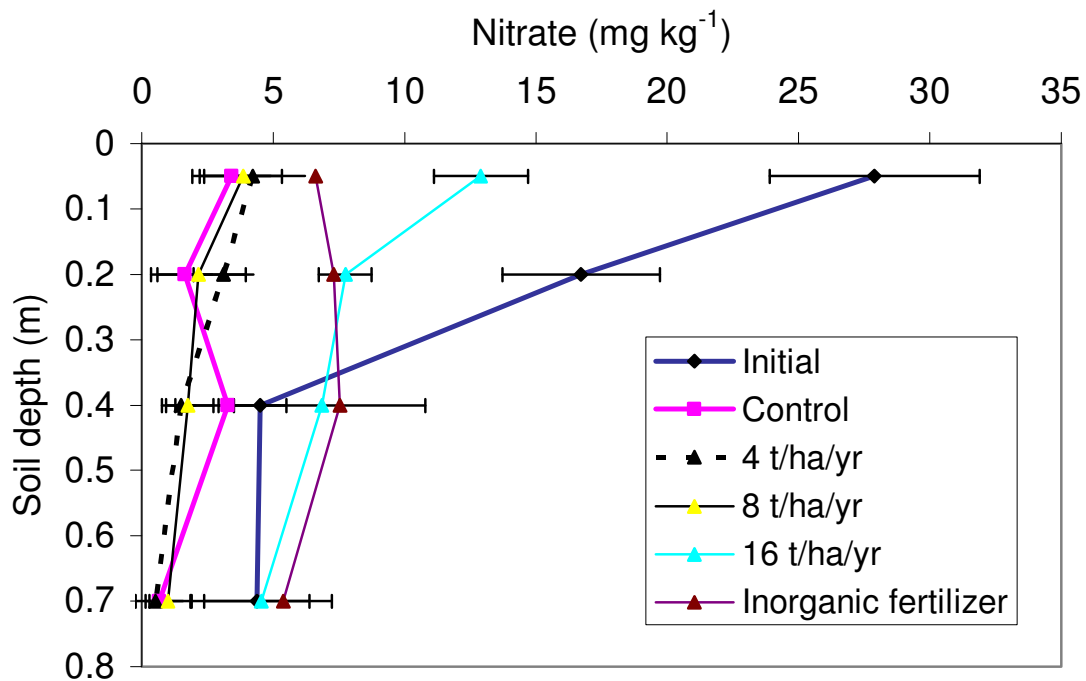


Figure 5.5 Residual nitrate before treatment application (initial) and after four consecutive years of treatment application (three sludge rates, inorganic fertilizer and control).

In this specific study the mean annual rainfall was low (405 mm in 2007) to moderate (710 mm in 2008). The wetting front detectors buried at 0.3 m depth responded only four times during the four year study period. The dates of response and the corresponding nitrate concentrations from the 16 Mg ha⁻¹ yr⁻¹

sludge treatment were: 21/02/2005 ($10 \text{ mg L}^{-1} \text{ NO}_3$), 01/03/2006 ($14 \text{ mg L}^{-1} \text{ NO}_3$), 22/01/2008 ($8 \text{ mg L}^{-1} \text{ NO}_3$), and 16/03/2008 ($11 \text{ mg L}^{-1} \text{ NO}_3$). The control and $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatments had similar or lower nitrate concentrations during the same time period. None of the WFDs buried at 0.6 m, however, responded. Therefore, considering the low to moderate rainfall experienced during the study period, low residual nitrate, and low nitrate concentrations collected during very few events might indicate that nitrate leaching was minimal during the study period.

5.3.3 Residual ammonium

Residual ammonium in the top 0.5 m soil stratum remained equal or less than initial values for all treatments (Table 5.10). Residual ammonium increased with increase in sludge application rate at the end of the 2007/08 growing season, though it did not have any specific trend in 2005/06.

The low residual ammonium mass reported at the end of the study, despite a net positive total N mass balance, might indicate that the ammonium added from the sludge was either nitrified or taken up by the plants. The possibility for ammonium to leach below the active root zone (0-0.6 m) in this specific soil type under the prevailing rainfall and sludge rates is unlikely, considering the following facts. Primarily the predominantly negatively charged soil clay and organic matter particles can fix ammonium through the process of cation exchange. Secondly

ammonium can easily be immobilized by microbial biomass, and lastly, ammonium is unstable and is readily nitrified (Cameron and Haynes, 1986).

Table 5.10 Residual ammonium mass in the top 0.5 m soil stratum after every second weeping lovegrass hay cut during the 2004/05 to 2007/08 growing seasons.

Treatment	Initial	After 2 nd hay cut	After 2 nd hay cut in
		in 2005/06	2007/08
		kg ha ⁻¹	
Control†	65	16	20
4 Mg ha ⁻¹ yr ⁻¹	65	41	38
8 Mg ha ⁻¹ yr ⁻¹	65	31	46
16 Mg ha ⁻¹ yr ⁻¹	65	36	65
Inorganic fertilizer‡	65	59	47

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N and 40 kg P per hectare per annum applied

Despite the net positive total N mass balance of the 16 Mg ha⁻¹ yr⁻¹ sludge treatment (Table 5.8), residual ammonium and nitrate remained similar or less than initial values. This indicates that a large fraction of the N in the soil is present in organic form.

In conclusion, sludge applications that produced the highest weeping lovegrass hay yield are not necessarily in total N balance. In this study, the 16 Mg ha⁻¹ yr⁻¹ sludge treatment significantly improved weeping lovegrass hay yield but was not in total N mass balance. The same treatment accumulated an extra 649 kg N ha⁻¹

(supply less uptake) or 330 kg N ha^{-1} (final less initial soil N storage change) following four consecutive years of sludge application and hay harvest events. Despite this net positive total N mass balance, there was neither residual nitrate nor ammonium accumulation in the soil profile above the initial values. In addition, the few soil solution samples collected from 0.3 m deep WFDs presented low nitrate concentrations. The implications are that sludge applications of up to $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ should have minimal environmental impacts through nitrate leaching in the medium-term (four years) for weeping lovegrass hay production under the prevailing climatic and edaphic conditions of the study site.

Therefore, the following hypothesis was accepted:

- Sludge applications of up to $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ that produced the highest hay yield under the prevailing climatic and edaphic conditions were not prone to excessive nitrate leaching in the medium-term (four years).

The following hypothesis was rejected:

- Under high hay yield production conditions, N supply from double the norm can be fully utilized.

Nevertheless, long term model simulations should be conducted because the mineralization of N from sludge could take several years before reaching steady-state conditions.

5.4 Total P mass balance and residual Bray-1P

5.4.1 Total P mass balance

Sludge applications of all rates resulted in a net positive total P mass balance (CUM-PS less CUM-PU) following four years of sludge applications and weeping lovegrass hay harvest events (Table 5.11, columns 4 and 5). Based on the mass balance calculation of total P imported with sludge less exported with weeping lovegrass hay yield, sludge applications of $4 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ supplied a cumulative excess of 279 kg P ha^{-1} during the four year study.

Table 5.11 Cumulative total P supply (CUM-PS), uptake (CUM-PU), and mass balance of a weeping lovegrass treated with three sludge rates, inorganic fertilizer, and a control

Treatment	Four years cumulative total P mass balance				
	CUM-PS	CUM-PU	CUM-PS less CUM-PU	Change in soil P storage	Mass balance difference
	kg ha ⁻¹				
Control†	0	59	-59	-50	-9
4 Mg ha ⁻¹ yr ⁻¹	352	73	279	302	-23
8 Mg ha ⁻¹ yr ⁻¹	705	93	612	660	-48
16 Mg ha ⁻¹ yr ⁻¹	1409	102	1307	1316	-9
Inorganic fertilizer‡	160	93	67	96	-29

† Zero sludge and inorganic fertilizer

‡ 40 kg P ha⁻¹ per year applied

Soil sampling to a depth of 0.8 m (Change in soil P storage) also supported after trends for the total P supply less uptake mass balance (Table 5.11). Most of the

excess P added with the sludge was located in the 0-0.1 m soil layer with a slight increase in the deeper 0.1-0.3 m soil layer (Fig. 5.6). The slight total P increment in this soil layer was most probably due to the physical migration of colloidal sludge particles between cracks formed during dry periods of the year, or from preferential flow of particulate P (Jensen et al., 2000; Brock et al., 2007). The grave concern with P surface accumulation is the potential threat to surface water bodies through transport by runoff, enhancing the rate of eutrophication in fresh water bodies (Carpenter et al., 1998).

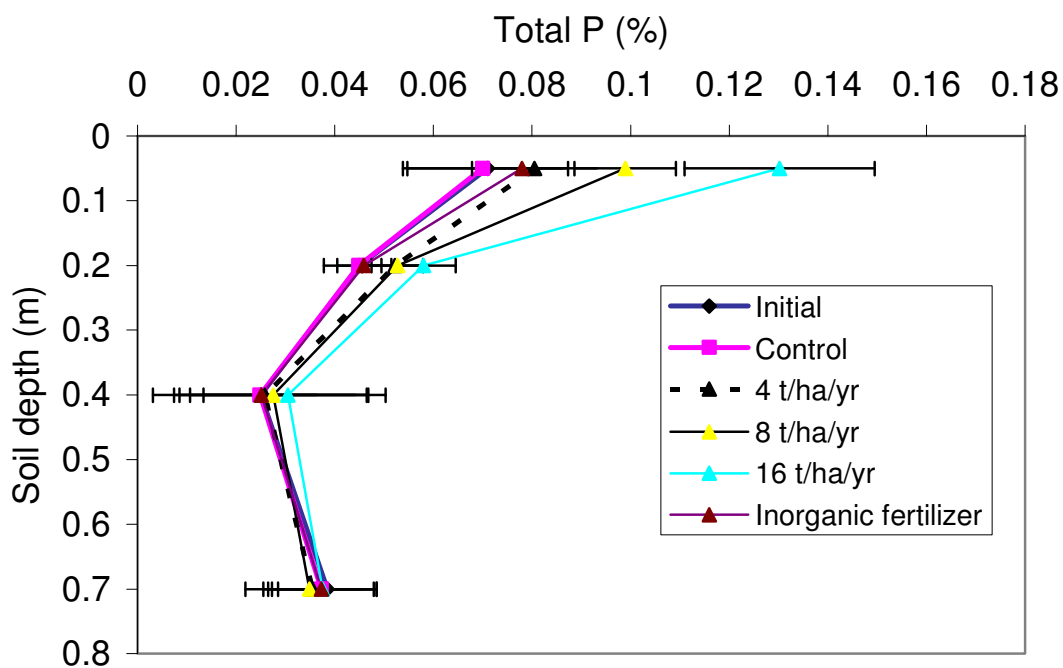


Figure 5.6 Initial soil profile total P and after four consecutive years of treatment applications in a weeping lovegrass hay production trial.

There was a mass balance difference between the supply less uptake mass balance (Table 5. 11, column 4) and the final less initial soil profile total P mass balance (Table 5. 11 column 6). The possible sources of difference are sampling errors, P content variation within the sludge matrix and soil heterogeneity.

The main reason for the accumulation of P in the soil profile was the low sludge N:P ratio of the sludge compared with that of crops. Therefore, P accumulation is inevitable if sludge is applied according to crop N demand (Shober and Sims, 2003).

5.4.2 Soil profile residual Bray-1 extractable P

Bray-1P decreased as the sludge application rate increased to 8 Mg ha⁻¹ yr⁻¹, but increased at higher rates (Table 5.12). Four years of sludge application according to the limit reduced residual Bray-1P by 42%. This is in contrast to the net positive total P mass balance reported for the same treatment on Table 5.11. On the other hand, sludge applied at double the limit increased residual Bray-1P in the top 0.5 m by 33 and 58%, compared with the control and 8 Mg ha⁻¹ sludge treatments, respectively.

Generally the mean background Bray-1P concentration of the soil in the 0-0.3 m soil stratum (65 mg kg⁻¹) was higher than the optimum concentration required for most crops (25-30 mg kg⁻¹). It was also higher than the concentration for an

optimum soil quality (50 mg kg^{-1}) above which the risk for surface water body pollution increases as reported (Sims and Pierzynski, 2000).

The decline in Bray-1P reported for the 4 and 8 Mg ha^{-1} sludge treatments was mainly due to the FeCl_3 added to the sludge at the waste treatment plants, which reduced the solubility of P (Huang et al., 2007; O'Connor et al., 2004; Huang and Shenker, 2004; Elliott et al., 2002; Maguire et al., 2000). The build up of Bray-1P reported from doubling of the sludge upper limit is most probably because the P supply from the sludge exceeded the buffer capacity of the soil and sludge (Elliott and O'Connor, 2007) or a decline in the sludge's P-sorbing capacity over time (Lu and O'Connor, 2001). It is also possible that the Fe-P minerals may have released P by dissolution as reported by Huang et al. (2007). Therefore, it is apparent from this study, that although, sludge treated with FeCl_3 reduced P plant availability; it is likely that P availability could also increase at higher rates over time.

Despite the high background concentration and additional P added with the sludge, there were no visible phosphorus toxicity symptoms during the study period. This is most probably due to the high Zn added with the sludge, because previous studies conducted by Loneragan et al. (1979), Safaya (1976), Parker (1997), Webb and Loneragan (1988), and Silber et al. (2002), all show that plant P toxicity is enhanced under Zn deficient conditions.

Table 5.12 Residual Bray-1P in the top 0.5 m soil stratum after the second hay cut of dryland pasture (weeping lovegrass) during the 2004/05 to 2007/08 growing seasons.

Treatment	Initial	After 2 nd hay cut in 2005/06	After 2 nd hay cut in 2007/08
		kg ha ⁻¹	
Control†	274	282	254
4 Mg ha ⁻¹ yr ⁻¹	274	253	230
8 Mg ha ⁻¹ yr ⁻¹	274	150	159
16 Mg ha ⁻¹ yr ⁻¹	274	367	378
Inorganic fertilizer‡	274	276	295

† Zero sludge and inorganic fertilizer applied

‡ 200 kg N and 40 kg P per hectare per annum applied

In summary, doubling of the annual sludge upper limit to satisfy crop N demand enhanced the build up of total P. Furthermore, this study showed that plant availability of P (Bray-1P) from dryland pasture soils fertilized with surface applied sludge, treated with Fe-Cl₃, is affected by the rate of sludge application. Consequently, Bray-1P decreased as the sludge application rate increased to 8 Mg ha⁻¹, but increased at higher rates over time.

Therefore, the following hypotheses were accepted:

- Sludge application according to weeping lovegrass N demand results in the accumulation of total and Bray-1P in the soil profile.

5.5 Conclusions

Weeping lovegrass hay yield, crude protein content, and rainfall use efficiency increased with doubling of the annual upper limit sludge application norm for

sludge with 2.56% mean N. Highest yield was harvested from years which experienced high rainfall compared with similar treatments experiencing low rainfall for all sludge treatments. Weeping lovegrass annual N uptake increased significantly as the sludge application rate was doubled. Sludge applied according to the norm of $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ was not sufficient to satisfy weeping lovegrass N demand. Doubling the upper sludge limit norm did not cause the accumulation of nitrate and ammonium in the profile. Sludge applied at double the norm, however, increased both total and Bray-1P with time.

Therefore, the following hypotheses were accepted under the prevailing climatic conditions:

1. Sludge application above the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ norm could significantly increase dryland pasture (weeping lovegrass) hay yield, crude protein content, and water use efficiency.
2. The ideal sludge application rate to satisfy dryland pasture (Weeping lovegrass) N demand is dynamic and could exceed the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge norm.
3. Sludge applications of up to $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ that produced the highest hay yield under the prevailing climatic and edaphic conditions are not prone to excessive nitrate leaching in the medium term (four years).
4. Sludge application according to weeping lovegrass N demand results in the accumulation of total P.

5. Sludge application according to weeping lovegrass N demand could result in the accumulation of Bray-1P at higher rates.

The following hypothesis was rejected under the prevailing low rainfall conditions of the study site:

- Under high hay yield production conditions, N supply from double the norm can be fully utilized.

Nevertheless, long-term model simulations should be conducted under varying rainfall, sludge N concentration, sludge type, and soil types in order to get a site specific real-time ideal sludge loading rate because N mineralization from sludge could take several years before reaching steady-state conditions.

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CHAPTER 6

TURFGRASS

The following hypotheses are tested in this chapter.

High sludge surface loading rates well above recommendations based on crop removal:

1. Are possible without reducing turf growth and quality.
2. Do not cause an accumulation of N and P below the active root zone.
3. Can minimize soil loss through sod harvesting, and
4. Do not cause unacceptably high nitrate and salt leaching.

6.1 Turfgrass growth and quality

6.1.1 Establishment rate

Establishment rate was estimated from mean percent basal cover. Sludge application rates above the 1997 South African upper limit norm (8 Mg ha⁻¹) significantly improved turf establishment rates (Table 6.1). 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.

6.1.2 Turfgrass colour

Turfgrass colour rating significantly improved with increase in sludge application rate (Table 6.1). Turfgrass which received sludge rates higher than the former South African upper limit norm exceeded the 'acceptable' colour ratings (7 - 9). The highest colour rating was scored by the 67 and 100 Mg 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. 6.1) also clearly illustrated that there was an increase in the availability of N with increase in sludge

Table 6.1 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 rate	Days after sludge application				Seasonal mean visual colour rating†		Sod integrity (percent harvestable sod per unit area)	
	2005		2006		2005	2006	2005	2006
	34	65	34	65				
	Mean vegetative cover, %				Colour ratings		%	
Control	34	58	36	68	4.0	3.8	72	71
8 Mg ha ⁻¹ per cut	41	74	43	75	4.8	5.3	72	73
33 Mg ha ⁻¹ per cut	56	88	59	87	7.0	7.3	93	96
67 Mg ha ⁻¹ per cut	63	95	68	95	8.5	8.5	90	88
100 Mg 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

application rate during most of the growing season.

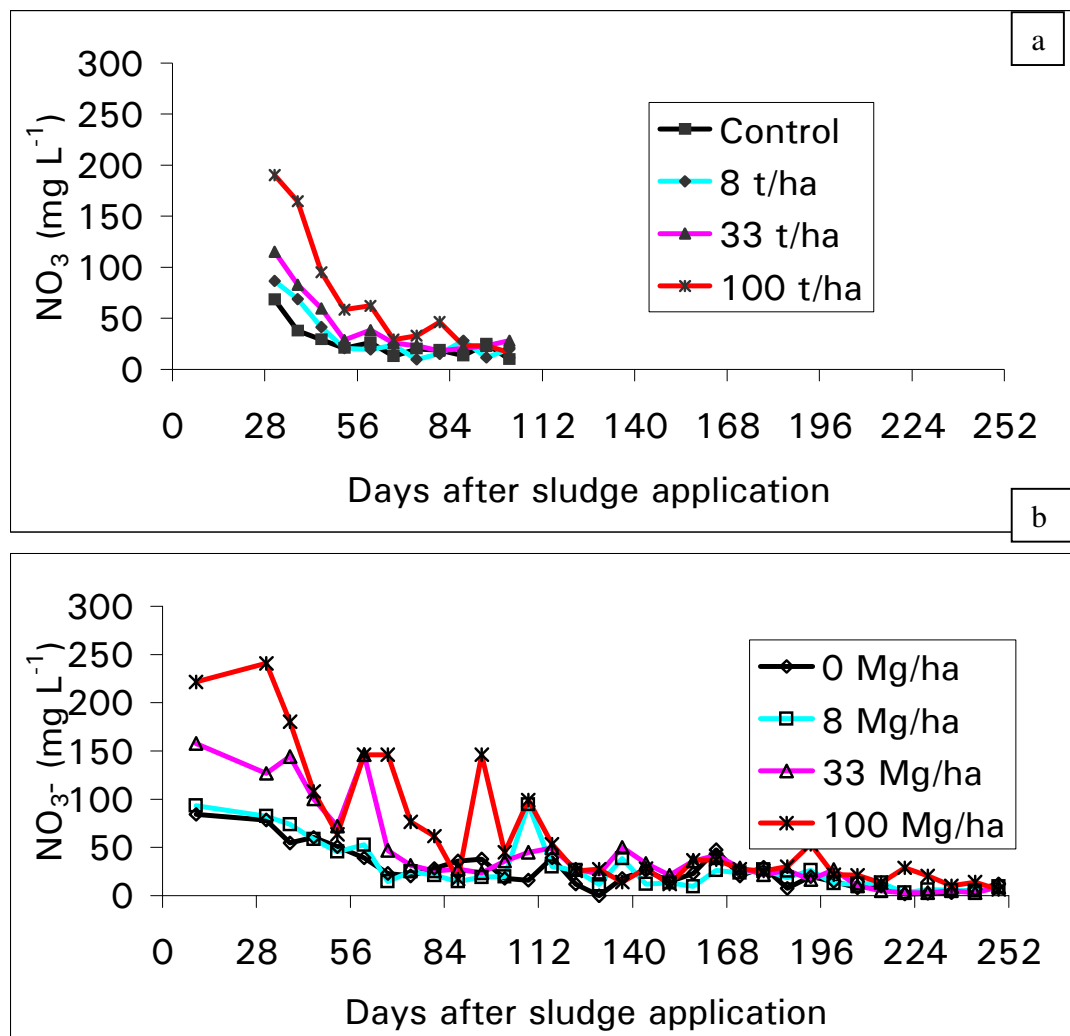


Figure 6.1 Concentration of nitrate in soil solution samples collected from wetting front detectors installed at 0.30 m of a turfgrass sod (*Pennisetum clandestinum*) trial for four sludge application rates (0 Mg ha^{-1} , 8 Mg ha^{-1} , 33 Mg ha^{-1} , and 100 Mg ha^{-1}) during (a) year 2005 and (b) 2006.

6.1.3 Sod integrity

The percentage of sods that did not break during handling increased as the sludge application rate increased to 33 Mg ha⁻¹, but decreased at higher rates (Table 6.1). The 0 Mg ha⁻¹ sludge treatment produced heavy sods up to 54 Mg ha⁻¹ heavier than the treatment receiving 100 Mg ha⁻¹. An increase in the sludge application rate to 33 Mg 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 Mg ha⁻¹, the sod became weaker and a greater proportion fell apart during handling and loading. Nevertheless, sod integrity was still good at 67 Mg ha⁻¹, and was not significantly different from the 33 Mg 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 Mg ha⁻¹ treatment, and the lowest from the 100 Mg 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 Mg 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 Mg 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 Mg ha⁻¹, well above the maximum recommended limit, but that sod integrity decreased after 33 Mg ha⁻¹.

6.2 Accumulation of N and P in soil below active root zone

6.2.1 Nitrogen

The mass of N exported with the sod increased significantly with increase in sludge application rate (Table 6.2), 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 meter of soil declined over the two year period in all treatments receiving less than 100 Mg ha⁻¹ sludge (Table 6.3). Based on soil sampling over this depth (change in soil storage), sludge applications of around 67 Mg ha⁻¹ were approximately in balance for nitrogen (Table 6.3). 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. 6.2a). After two years, the total N content of the top 0.15 m layer was less than the initial total N content 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 Mg ha⁻¹ and the 0.45-0.75 m layers of the 67 Mg ha⁻¹ treatments (Fig. 6.2a), 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 Mg ha⁻¹ showed a marked depletion in nitrate levels by the end of the study (Fig. 6.2b). Rates of 33 Mg 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 6.2. Total N imported with sludge, vs. exported with sods and clippings during the 2004/05 and 2005/06 growing seasons at East Rand Water Care Works, Johannesburg, South Africa.

Sludge application rate	N imported with sludge vs. exported with sod and clippings									
	Imported		Exported							
			2005				2006			
			Sod		Clippings	Total	Sod		Clippings	Total
	2005	2006	Soil	Plant			Soil	Plant		
Mg 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 6.3 Total nitrogen and total phosphorus mass balances after two years of sludge application and sod harvest events for five sludge application rates during the 2005 and 2006 growing seasons

Sludge application rate	Nitrogen				Phosphorus			
			Change in	Mass			Change in	Mass
	Imports	Exports	soil storage†	balance difference‡	Imports	Exports	soil storage†	balance difference‡
	kg ha ⁻¹							
Mg 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).

The concentration of ammonium was elevated at applications greater than 33 Mg ha⁻¹ (Fig. 6.2c). 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.

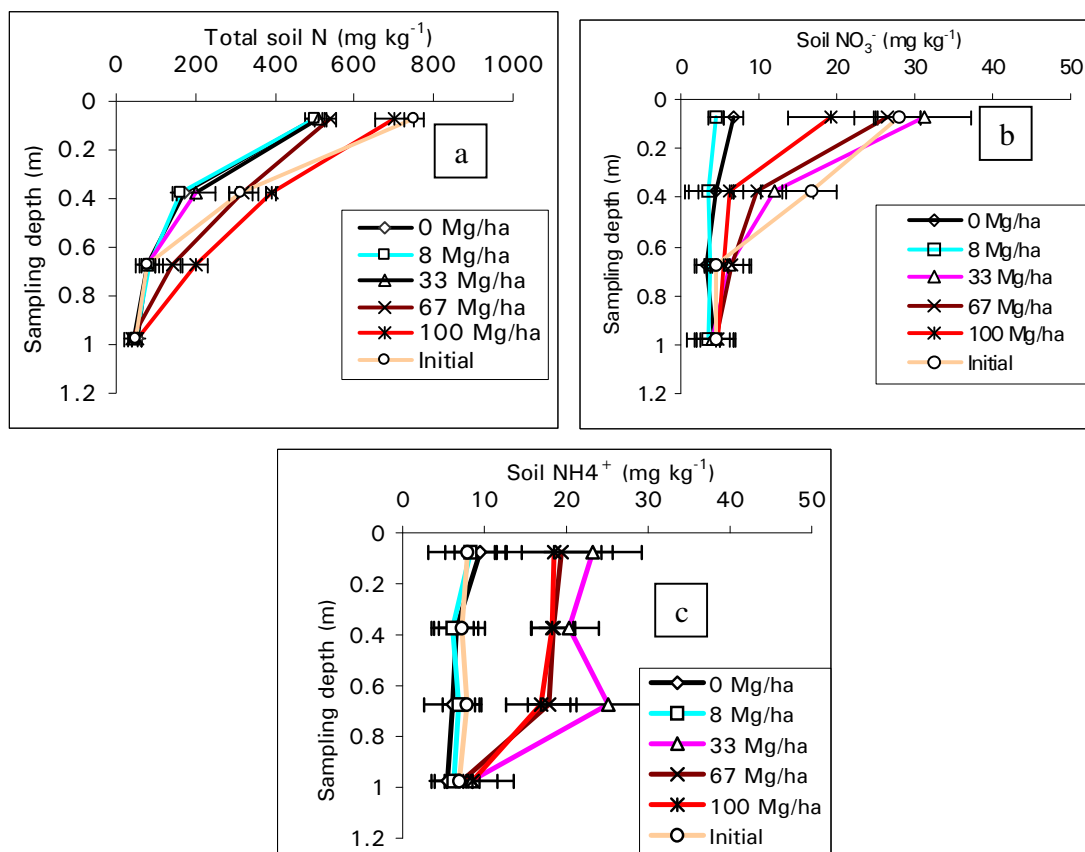


Figure 6.2 Soil profile (a) total N (b) nitrate (c) ammonium (d) total P (e) Bray-1 extractable P, and (f) electrical conductivity (ECe) as affected by two consecutive years of sludge application at five rates (0, 8, 33, 67, and 100 Mg ha⁻¹) in a turfgrass sod (*Pennisetum clandestinum*) field trial, sampled before treatment application in 2005 (initial) and after two sod harvests in 2006.

6.2.2 Phosphorus

The mass of P exported with the sod increased significantly as the sludge application rate increased (Table 6.4), 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 Mg ha⁻¹ of sludge, based on soil sampling over the 1.0 m depth (Table 6.3). Based on this sampling depth (change in soil storage), sludge application rates of 33 to 67 Mg ha⁻¹ were approximately in balance for phosphorus. However, the 67 Mg 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. 6.3a and 6.3b). Sludge application rates higher than 33 Mg 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. 6.3b). The build up of total P in the top 0.15 m soil layer of the 67 and 100 Mg 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).

- 1 Table 6.4 Total phosphorus imported with sludge, vs. exported with sods and clippings during the 2005 and 2006 growing
- 2 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			
	2005	2006	Sod		Clippings	Total	Sod		Clippings	Total
			Soil	Plant			Soil	Plant		
Mg 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 _t , %			6	5	4	4	6	5	6	5

- 3 † Coefficient of variation

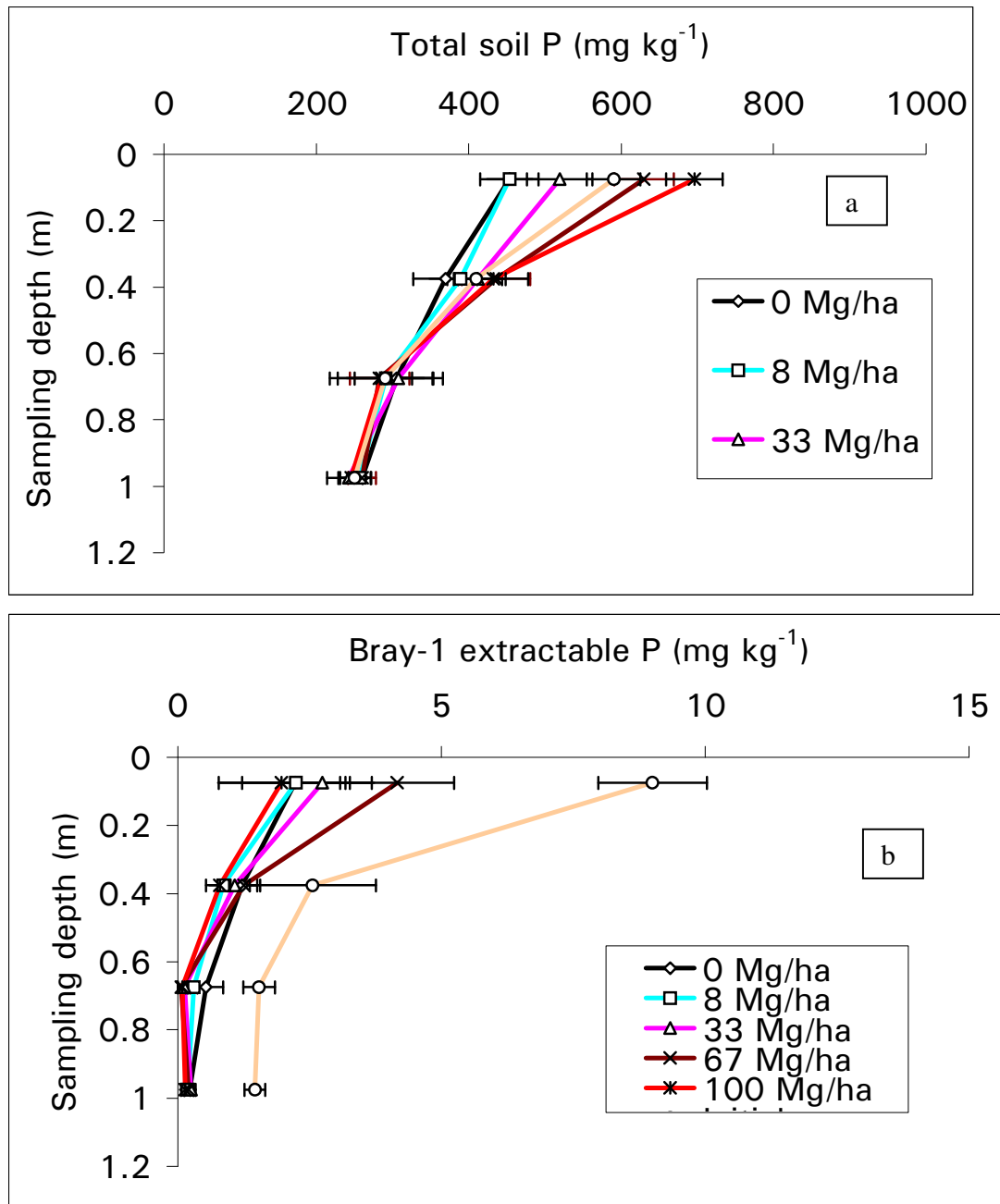


Figure 6.3 Soil profile (a) total P (b) Bray-1 extractable P as affected by two consecutive years of sludge application at five rates (0, 8, 33, 67, and 100 Mg ha⁻¹) in a turfgrass sod (*Pennisetum clandestinum*) field trial, sampled before treatment application in 2005 (initial) and after two sod harvests in 2006.

The 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 loadings of less than 67 Mg ha⁻¹ could most probably be due to plant uptake. For treatments receiving sludge rates higher than 67 Mg 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 meter of soil decreased over the duration of the trial for rates not exceeding 67 Mg ha⁻¹. Total P content within the same depth also declined for rates not exceeding 33 Mg 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 Mg 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 loading rates of 33 Mg ha⁻¹. Total P accumulation was evident mainly in the top 0.15 m of treatments which received sludge loading rates higher than 33 Mg ha⁻¹ under the prevailing management practices and climate.

6.3 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 6.5). The depth of soil exported from the site decreased significantly with increasing sludge application rates above 8 Mg ha⁻¹. The application of 33, 67, and 100 Mg ha⁻¹ saved 120, 217, and 290 Mg 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 Mg ha⁻¹ treatments saved soil from being exported which could have taken 86, 155, and 207 years to form, respectively.

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 6.5). This also has a direct financial implication for sod transportation. Thus sludge applied above agronomic limits, will also reduce sod transportation costs.

In summary, the application of sludge did help to reduce the rate of soil loss through sod harvesting and should reduce transportation cost through lowering sod mass. Rates as high as 100 Mg ha⁻¹ were required to completely avoid soil loss. However, such high loading rates deleteriously affected sod integrity.

Table 6.5. 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	
	Mg 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

6.4 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 Mg ha⁻¹ treatment did not have WFDs, this treatment is excluded from the discussion that follows.

6.4.1 Nitrate leaching

In the beginning of the season, soil solution nitrate concentration increased with sludge application rate (Fig. 6.1). 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. 6.1), presumably because the greater demand from the turf matched the mineralization rate from the sludge. Similar results and trends were recorded both during the 2004/05 (Fig. 6.1a) and 2005/06 (Fig. 6.1b) 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 ($0\text{--}255\text{ mg L}^{-1}\text{ NO}_3^-$). It was also less than the maximum nitrate concentration in leachate from a simulated golf green ($376\text{ mg L}^{-1}\text{ NO}_3^-$) reported by Shuman (2001).

The concentration of nitrate leachate from all treatments remained higher than South African drinking water standards ($44\text{ mg NO}_3^- \text{ L}^{-1}$) (Korentajer, 1991) for the first two to three months in the 100 Mg ha^{-1} sludge treatment (Fig. 6.1). The wetting front detectors installed at 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 Mg ha⁻¹ sludge treatments (26, 29, and 43 mg NO₃⁻ 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 NO₃⁻ L⁻¹) (Vlassak and Agenbag, 1999). The seasonal average nitrate concentration for the 100 Mg ha⁻¹ treatment (63 mg NO₃⁻ L⁻¹), however, exceeded both limits. Compared with the zero sludge treatment, the addition of 8, 33, and 100 Mg ha⁻¹ sludge increased the average seasonal nitrate concentration of the leachate by 3, 17 and 36 mg NO₃⁻ L⁻¹ respectively.

6.4.2. Salt leaching

In the beginning of the season soil solution EC increased with sludge application rate (Fig. 6.4). The EC of soil solution samples from the 100 Mg ha⁻¹ treatment collected 10 days after sludge application were only slightly higher than the threshold value with 10% yield reduction of 300 mS m⁻¹ for kikuyu (Yiasoumi et al., 2005). Nevertheless, highest mean percent vegetative cover was recorded for this treatment (Table 6.1). Treatments receiving less than 100 Mg ha⁻¹ sludge, however, had lower soil solution EC values than this threshold (Figs. 6.4a and b). For soil solution samples from the 33 and 100 Mg ha⁻¹ treatments, the 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. 6.5) 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 Mg ha^{-1} significantly increased soil salinity, but the levels were still acceptably low (Fig 6.5).

In summary, the requirement to leach salts is the most difficult aspect of managing large volume sludge applications. Nitrate leaching cannot 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 6.2), it would have been significant. Based on the EC of soil water and the growth studies, the leaching fraction could have been reduced. It may also be necessary to apply the sludge in two applications, and delay the second application until low nitrate was measured in the wetting front detectors. This high leaching fraction, however, could simulate worst case scenarios.

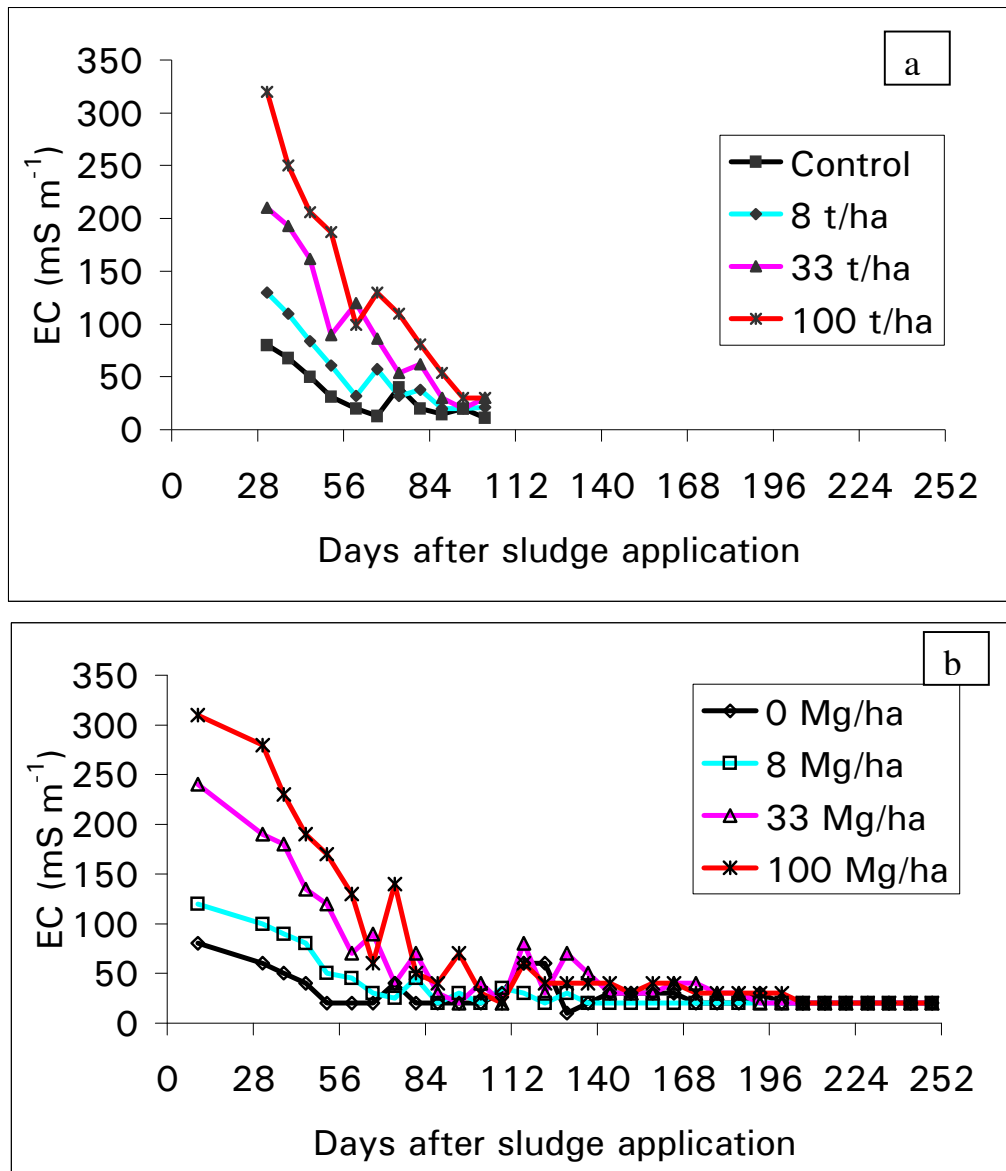


Figure 6.4 Electrical conductivity of soil solution samples collected from wetting front detectors installed at 0.30 m of a turfgrass sod (*Pennisetum clandestinum*) trial for four sludge application rates (0 Mg ha⁻¹, 8 Mg ha⁻¹, 33 Mg ha⁻¹, and 100 Mg ha⁻¹) during (a) year 2004/05 and (b) 2005/06 growing seasons.

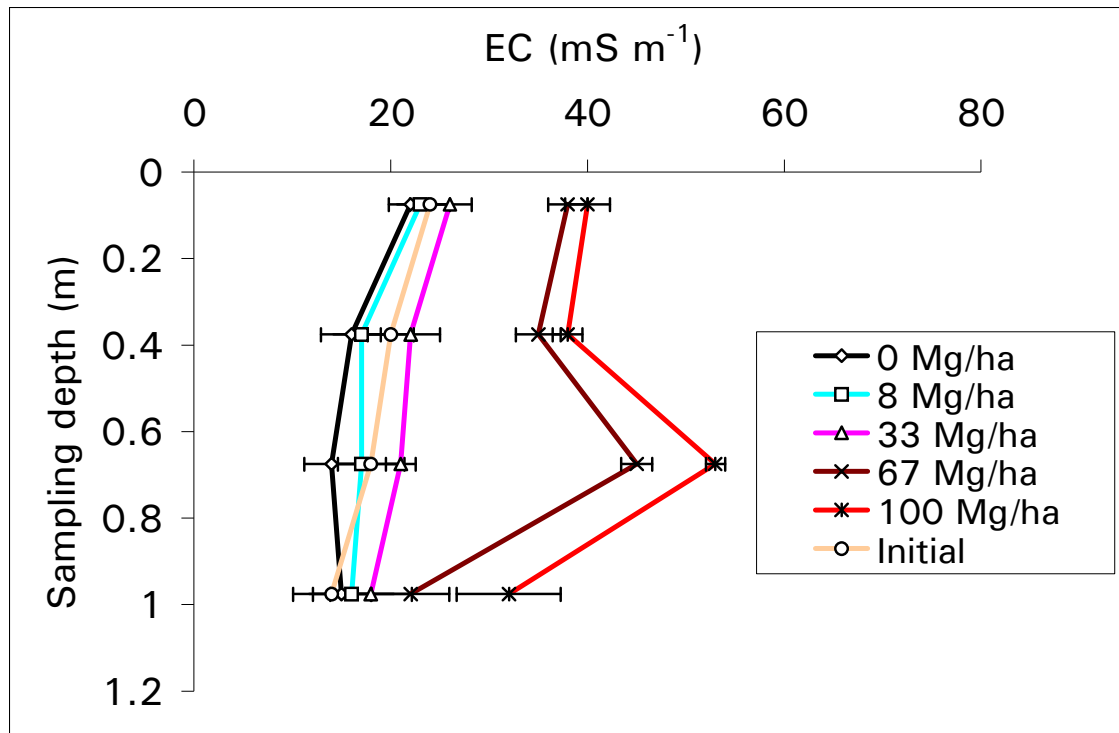


Figure 6.5 Soil profile electrical conductivity as affected by two consecutive years of sludge application at five rates (0, 8, 33, 67, and 100 Mg ha⁻¹) in a turfgrass sod (*Pennisetum clandestinum*) field trial, sampled before treatment application in 2004/05 (initial) and after two sod harvests in 2005/06.

6.5 Conclusions

The study demonstrated several advantages to large volume sludge loading of turf for sod production. Establishment rate and turf colour improved up to 67 Mg ha⁻¹. Very high loadings of 100 Mg ha⁻¹ had no deleterious effect on the turf growth, despite the salt content of the sludge. High loadings 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 Mg ha⁻¹ sludge was applied. At the end of the study the 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 loading are apparent when comparing the N and P mass balance 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 Mg 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 Mg ha⁻¹, four times higher than the 1997 South African upper recommended 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 Mg ha⁻¹ application treatments. Great care would be required to minimize nitrate leaching during the first two months. 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 Mg ha^{-1} could be possible if the sludge applications were split by delaying the second application until low nitrate was measured in the wetting front detectors, as long as turf quality was not negatively affected.

Therefore, all hypotheses were accepted for application rates not exceeding 33 Mg 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. Further research is needed for other soil types and possible runoff losses from fields with some gradient.

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CHAPTER 7

NITROGEN MODELLING

The chapter will focus on N modelling as N is the most likely limiting factor for crop production with sludge and most sludge guidelines are based on N.

Although P accumulation in N based agricultural sludge application is causing concerns, the complex nature of P in Fe and Al salts 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.

7.1 Model calibration

Field data collected during the 2004/05 growing season was used to determine specific crop parameters of maize, oat and weeping lovegrass in order to calibrate the SWB 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, harvestable dry matter, and maximum grain N concentration were determined from field data. The remaining parameters were obtained from literature.

Selected measured variables, namely total aboveground biomass, grain yield, leaf area index, and aboveground biomass and grain N uptake (Fig. 7. 1) were used to test the success of model calibration. Accuracy of model simulations were assessed based on statistical parameters proposed by De Jager (1994) as presented in Table 7.1.

Table 7.1 Model evaluation statistical parameters with their reliability criteria (after De Jager, 1994)

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

Generally the model was calibrated successfully for maize, oats, and weeping lovegrass. This is because most of the simulated values for the selected variables agreed closely to measured values (Fig. 7.1) and all the statistical parameters were within the ranges prescribed by De Jager et al. (1994) (Table 7.2). Maize aboveground biomass was underestimated towards the end of the season. Nevertheless, the difference was not significant and all the statistical parameters

were within the ranges prescribed by De Jager (1994) (Table 7.2). 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 ($52.74 \text{ kg N ha}^{-1}$).

Model simulation results for oats LAI, aboveground biomass, grain yield, aboveground biomass N uptake and grain N uptake were estimated with high accuracy. Similarly, simulated weeping lovegrass hay yield was close to the measured values for the first cut of each year. It was, however, underestimated during the second cut causing the underestimation of hay N uptake during this harvest cycle. Nevertheless, all the variables were estimated within the acceptable ranges of statistical accuracy.

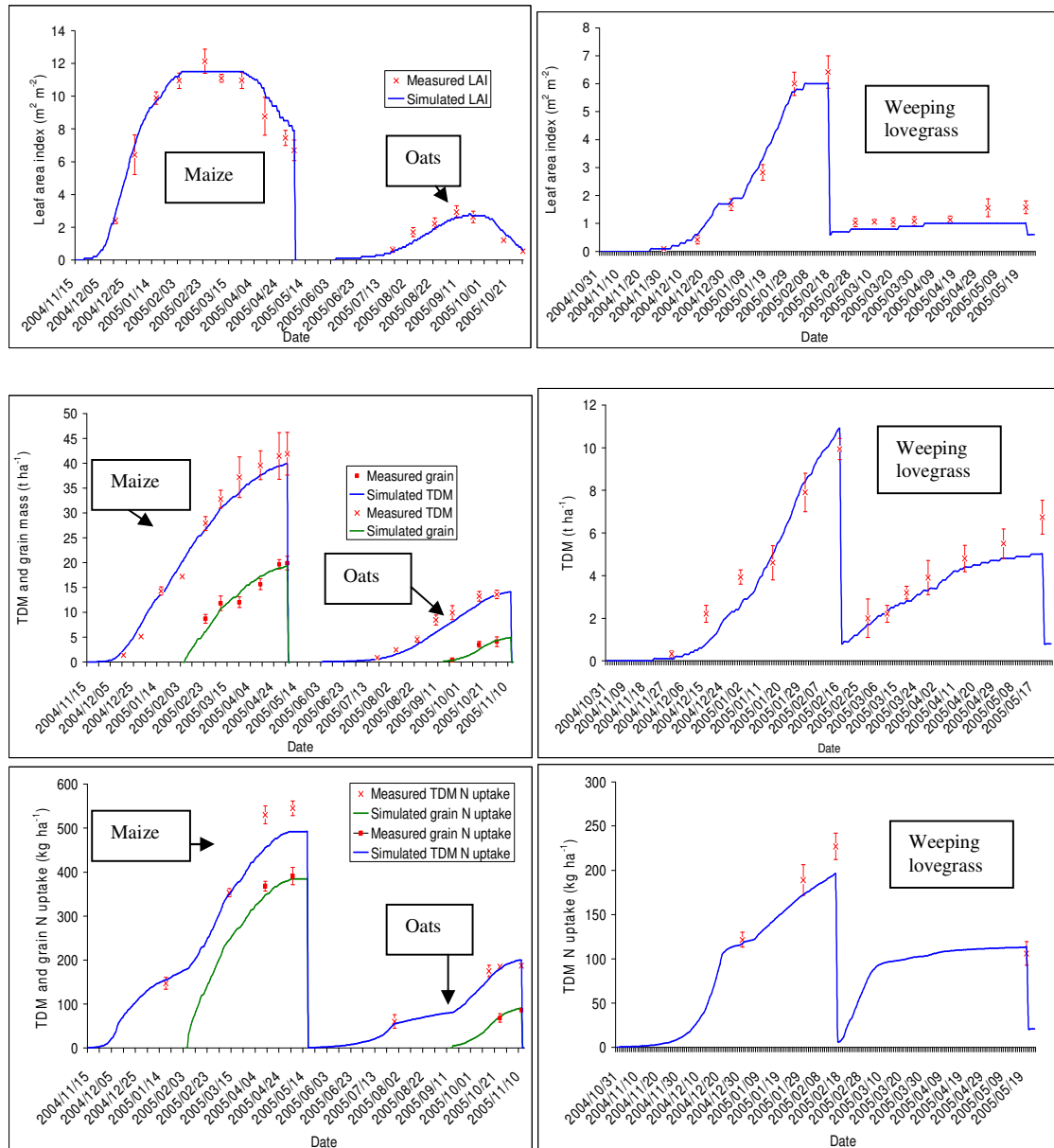


Figure 7.1 Simulated (solid lines) and measure values (symbols with standard deviation) from top to bottom of leaf area index, aboveground biomass (TDM), and aboveground biomass N uptake for the 16 Mg ha^{-1} sludge treatment.

Table 7.2 Statistical parameters of the SWB model calibration simulations for maize, oats, and weeping lovegrass 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	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	3	0.90	0.77	17.39	0.95
Weeping lovegrass					
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.

7.2 Model corroboration

Model corroboration was conducted using variables collected from independent data sets during the 2004/05 to 2007/08 growing seasons for maize, oats and weeping lovegrass. The variables used to evaluate the accuracy of the model

were LAI, aboveground biomass, grain (agronomic crops), and aboveground biomass and grain N uptake.

7.2.1 Agronomic crops

Generally evaluation of the model against three year combined independent data sets for dryland maize and irrigated maize-oat rotation proved to be very successful. All indicators were simulated for selected treatments, as all the statistical parameters were within the accuracy limits prescribed by De Jager (1994) (Table 7.3).

Simulations for selected treatments and their statistical analyses are presented in Figs. 7.2 to 7.4 and Table 7.3. Although, the model underestimated in some instances and overestimated in others, the statistical parameters for both dryland maize and irrigated maize-oat rotation were within the accuracy limits. There were, however, a few exceptions, such as for the leaf area index (LAI), aboveground biomass (TDM), and grain yield (HDM) of the 8 Mg ha⁻¹ yr⁻¹ sludge treated dryland maize. The model significantly underestimated LAI and TDM during 2004/05 and 2007/08 growing seasons and HDM during the 2004/05 and 2005/06. This was indicated by a slightly higher MAE (%) value of 29, for the LAI and 20 for both TDM and HDM, compared with the prescribed range of < 20%. Generally the model predicted the indicators more accurately under irrigated maize-oat rotation than dryland maize cropping.

Table 7.3 Statistical parameters of the SWB model corroboration for maize, oats, and weeping lovegrass using combined data collected during the 2004/05 to 2007/08 growing seasons.

Variables	n	D	RMSE	MAE (%)	R ²
Dryland maize 8 Mg ha⁻¹ per annum sludge treatment					
LAI	11	0.80	0.38	29	0.98
Aboveground biomass	16	0.87	2.37	20	0.99
Aboveground biomass N uptake	9	0.85	31.00	17	0.93
Grain	10	0.87	1.10	20	0.91
Grain N uptake	6	0.88	17.00	14	0.99
Irrigated maize-oat rotation 8 Mg ha⁻¹ per annum sludge treatment					
LAI	25	0.82	1.00	12.22	0.98
Aboveground biomass	26	0.97	2.55	10.68	0.99
Aboveground biomass N uptake	17	0.92	44.01	14.00	0.96
Grain	17	0.95	1.17	7.00	0.98
Grain N uptake	10	0.89	41.26	14.00	0.95
Irrigated maize-oat rotation 16 Mg ha⁻¹ per annum sludge treatment					
LAI	23	0.94	0.71	18	0.96
Aboveground biomass	29	0.94	2.69	15	0.97
Aboveground biomass N uptake	15	0.89	43.00	15	0.97
Grain	18	0.92	1.59	16	0.97
Grain N uptake	10	0.90	31.71	15	0.97

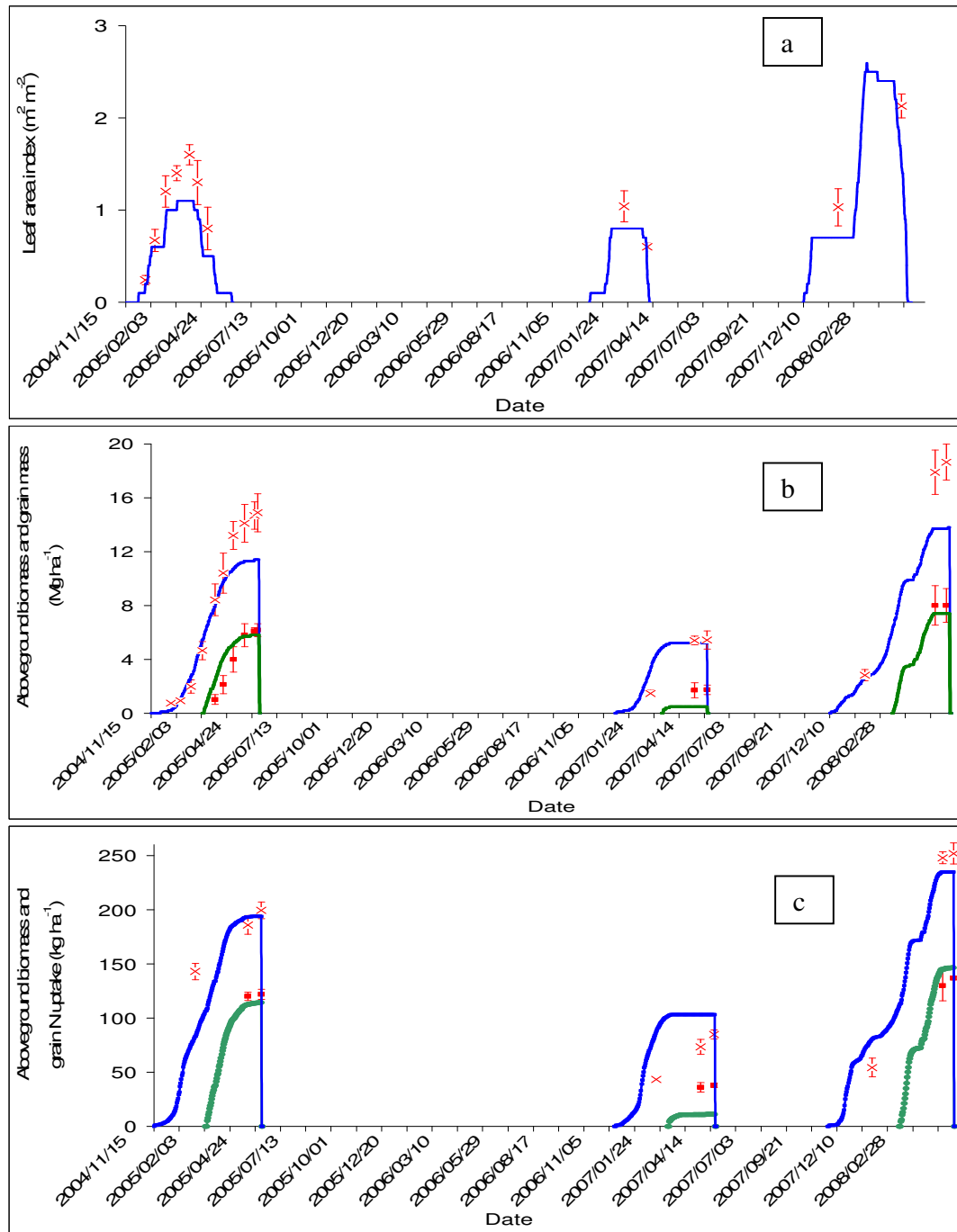


Figure 7.2 Simulated (solid lines) and measure values (symbols) of leaf area index (a), aboveground biomass (TDM), and aboveground biomass N uptake (c) for the 8 Mg ha⁻¹ per annum sludge treated dryland maize during the 2004/05 to 2007/08 study period.

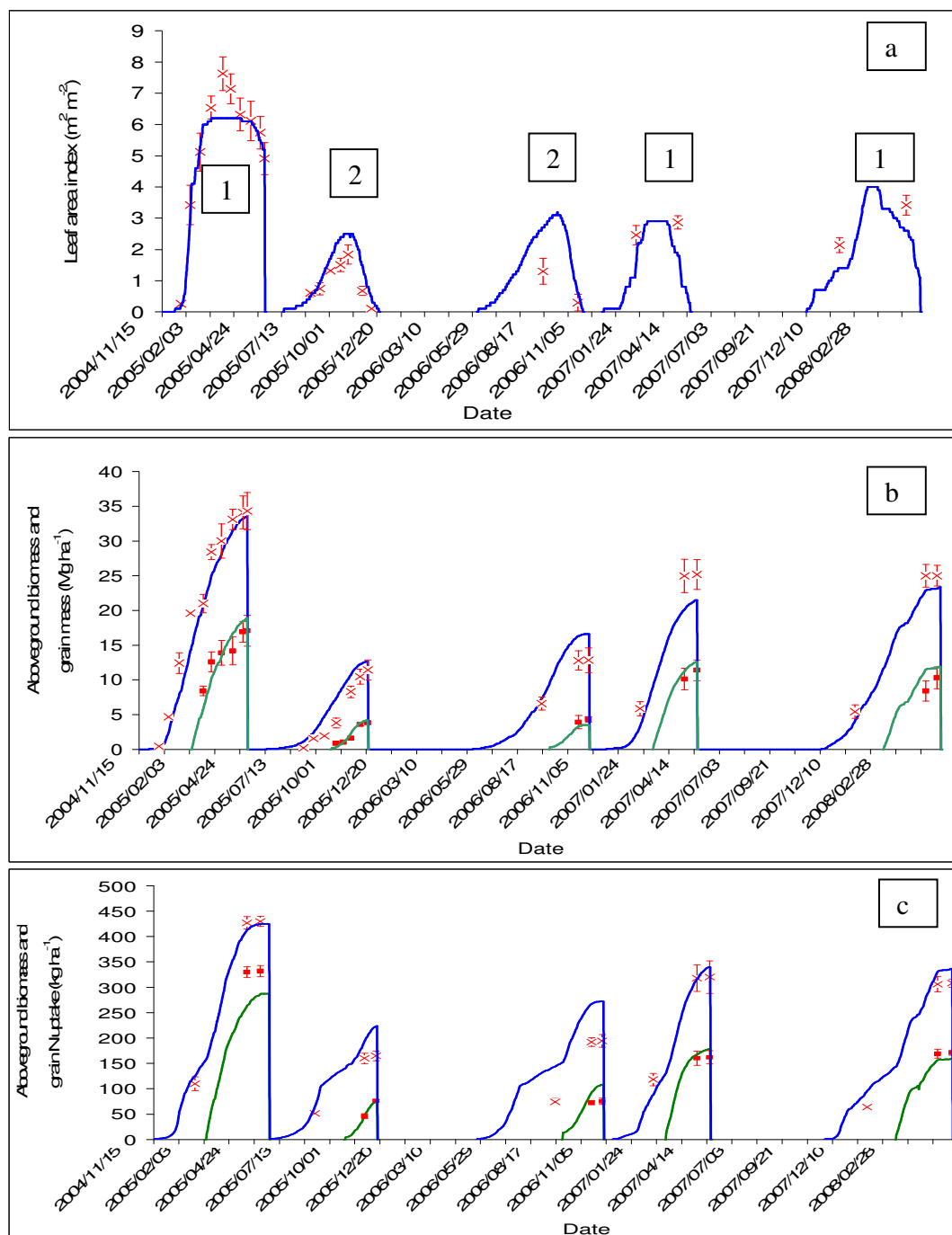


Figure 7.3 Simulated (solid lines) and measure values (symbols) of leaf area index (a), aboveground biomass (TDM), and aboveground biomass N uptake (c) for the 8 Mg ha⁻¹ per annum sludge treated irrigated maize(1)-oat(2) rotation during the 2004/05 to 2007/08 study period.

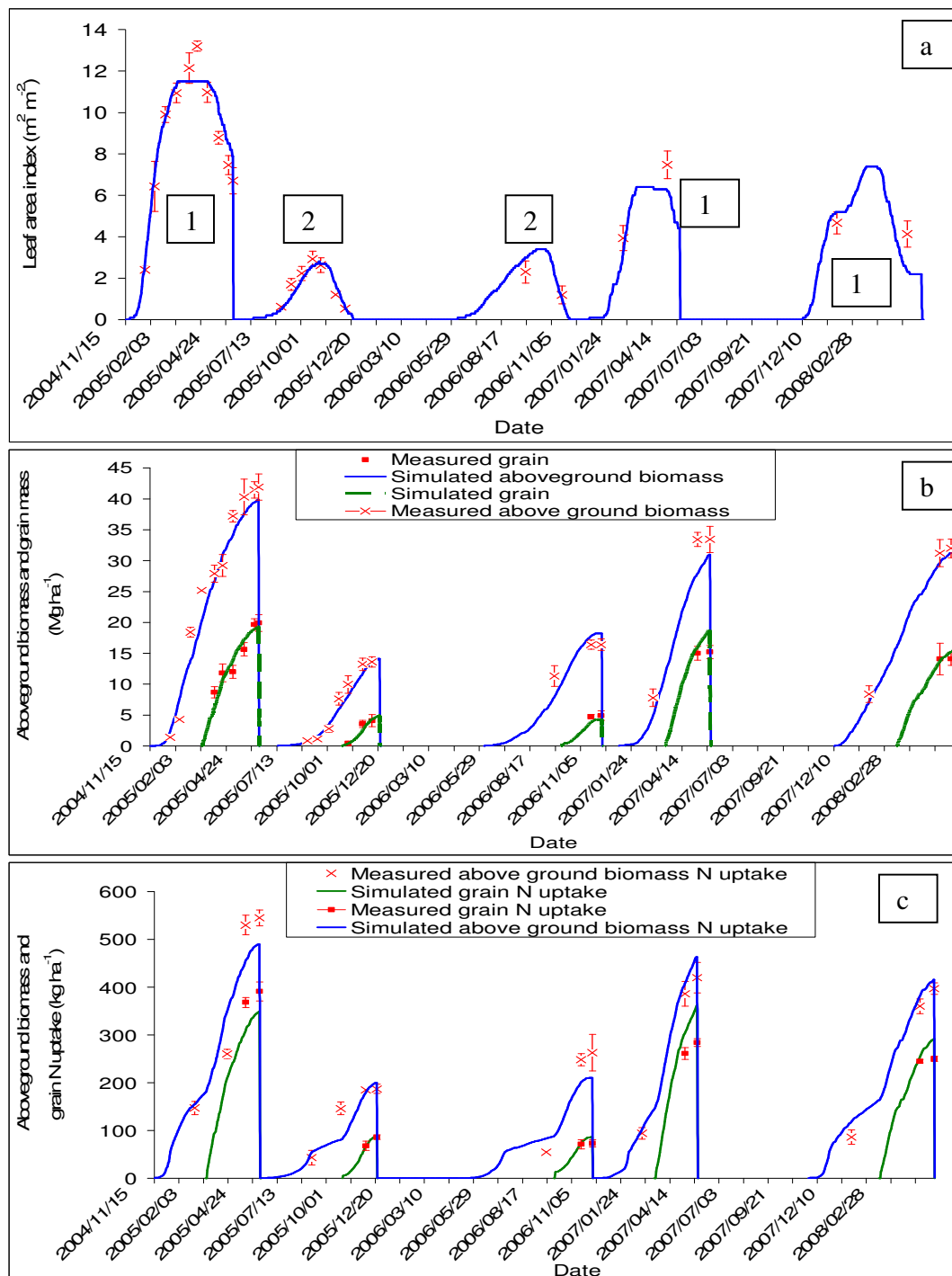


Figure 7.4 Simulated (solid lines) and measure values (symbols) of leaf area index (a), aboveground biomass (TDM), and aboveground biomass N uptake (c) for the 16 Mg ha⁻¹ per annum sludge treated irrigated maize(1)-oat(2) rotation during the 2004/05 to 2007/08 study period.

7.2.2 Weeping lovegrass

During model corroboration of weeping lovegrass using four year combined data of all treatments, prediction of LAI, aboveground biomass, and aboveground biomass N uptake had poor agreement with the measured values. All the statistical parameters for the above mentioned variables were not within the prescribed ranges. Updating soil water content after every hay cut using field measurements, however, improved model performance (Table 7.4).

Table 7.4 Statistical parameters of the SWB model corroboration for weeping lovegrass without and with updating soil water content after every hay cut using combined data collected during the 2004/05 to 2007/08 growing seasons.

Variables	n	D	RMSE	MAE (%)	R ²
Weeping lovegrass					
Without updating soil water content					
LAI	102	0.74	0.83	38	0.85
Aboveground biomass	102	0.51	1.97	37	0.74
Aboveground biomass N uptake	30	0.36	35.00	24	0.64
Soil water content updated after every hay cut					
LAI	102	0.92	0.43	22	0.95
Aboveground biomass	102	0.84	1.13	22	0.91
Aboveground biomass N uptake	30	0.49	32.00	21	0.71

Individual simulations, without updating soil water content, (Figs. 7.5 to 7.7) and after updating soil water content (Figs. 7.8 to 7.10) were also conducted for selected treatments, namely for the control, 8, and 16 Mg ha⁻¹ sludge treatments. The statistical analyses for these simulations are presented in Appendix (Table A3). For scenarios, where the soil water content was updated after every hay cut, the model simulated the correct trend and magnitude for LAI, thereby showing good predictability under various N supply conditions varying from zero added N to far more than the crop demand. Although the model underestimated LAI in most cases, the statistical parameters were within the prescribed limits. The simulated LAI for all the treatments was predicted with a RMSE of 0.44-0.47 m² leaf area m⁻² ground area.

During model corroboration runs of scenarios where the soil water content was updated after every hay cut, predictions of weeping lovegrass aboveground biomass for the control, 8, and 16 Mg ha⁻¹ sludge treatments were underestimated in most cases during the study period. The statistical parameters for the 8 and 16 Mg ha⁻¹ sludge treatments were within the prescribed ranges. The control treatment, however, had lower coefficient of determination (0.71) than the prescribed (>0.8).

Aboveground biomass N uptake model prediction during model corroboration was poor for all treatments, because the statistical parameters were not within the prescribed ranges (Table 7.4). Generally, the poor model performance for

perennial dryland pasture (weeping lovegrass) was because the model does not simulate long-term perennial grass growth mechanistically. The model does not have a mechanistic way of simulating the dormancy during winter and re-growth in spring for perennial grass. Therefore, the model needs to be improved.

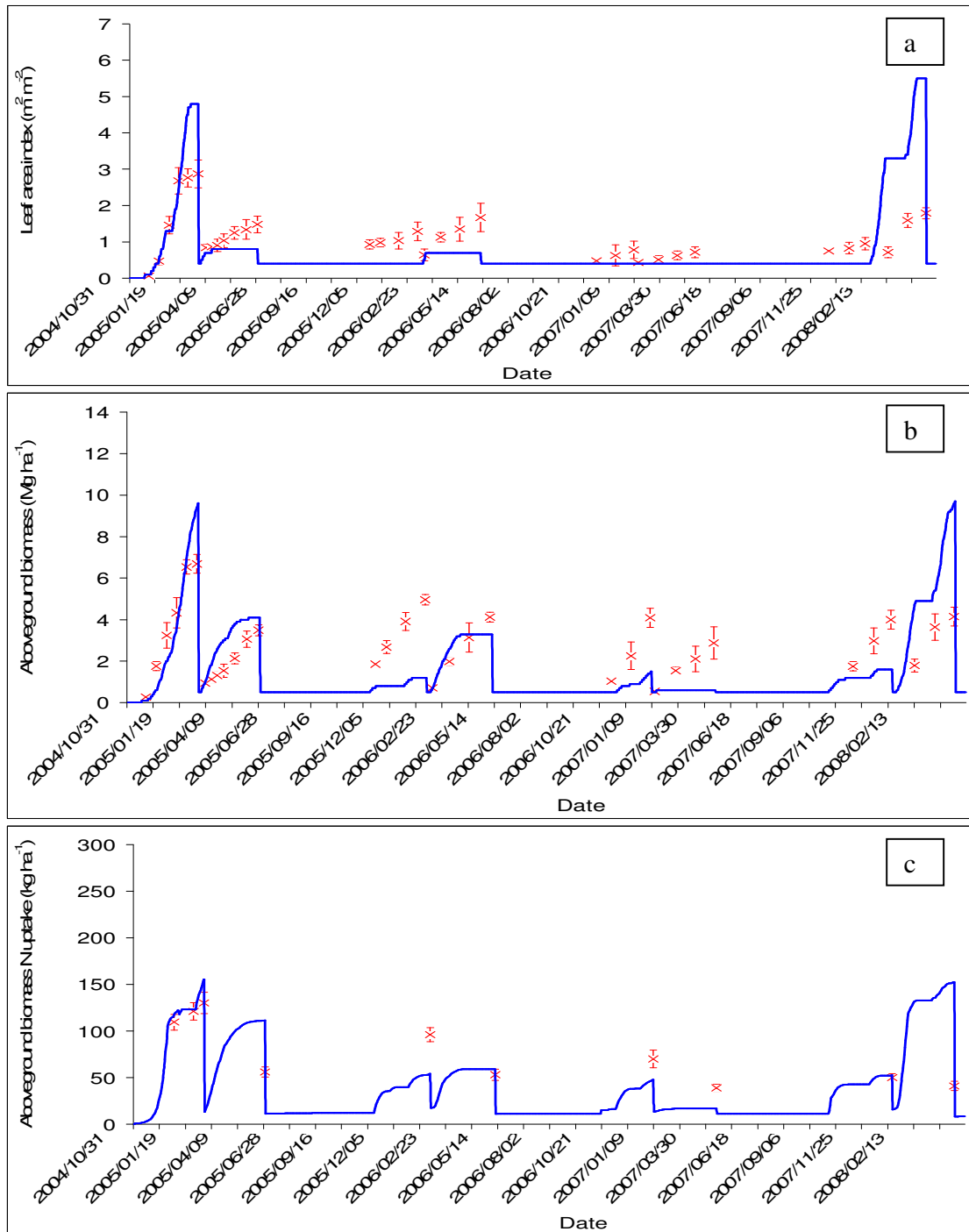


Figure 7.5 Simulated (solid lines) and measured values (symbols with standard deviation) of weeping lovegrass leaf area index (a), aboveground biomass (b), and aboveground biomass N uptake (c) for the control treatment (0 nutrients applied) during the 2004/05 to 2007/08 study period (without updating soil water).

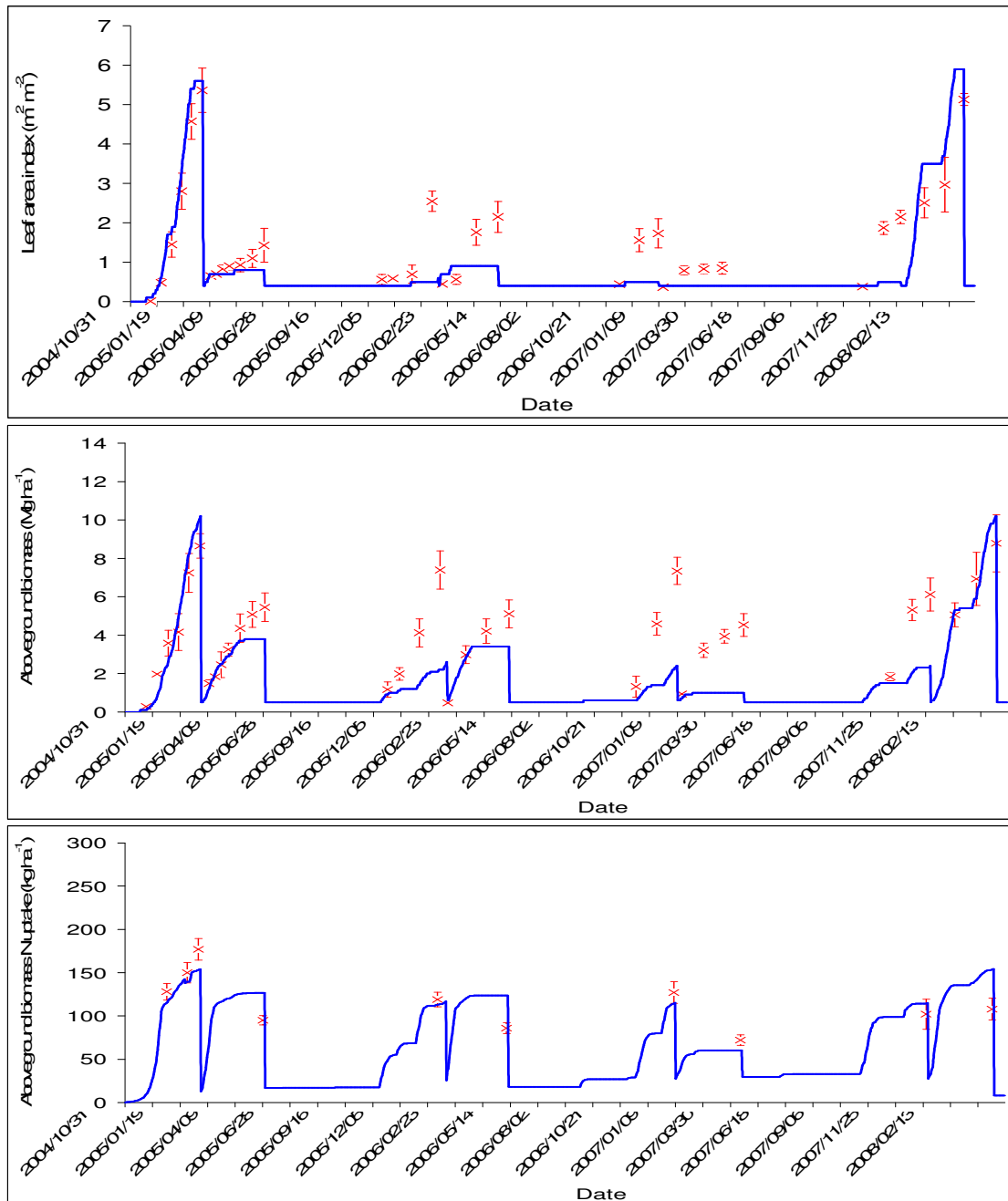


Figure 7.6 Simulated (solid lines) and measured values (symbols with standard deviation) of weeping lovegrass leaf area index (a), aboveground biomass (b), and aboveground biomass N uptake (c) for the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatment during the 2004/05 to 2007/08 study period (without updating soil water).

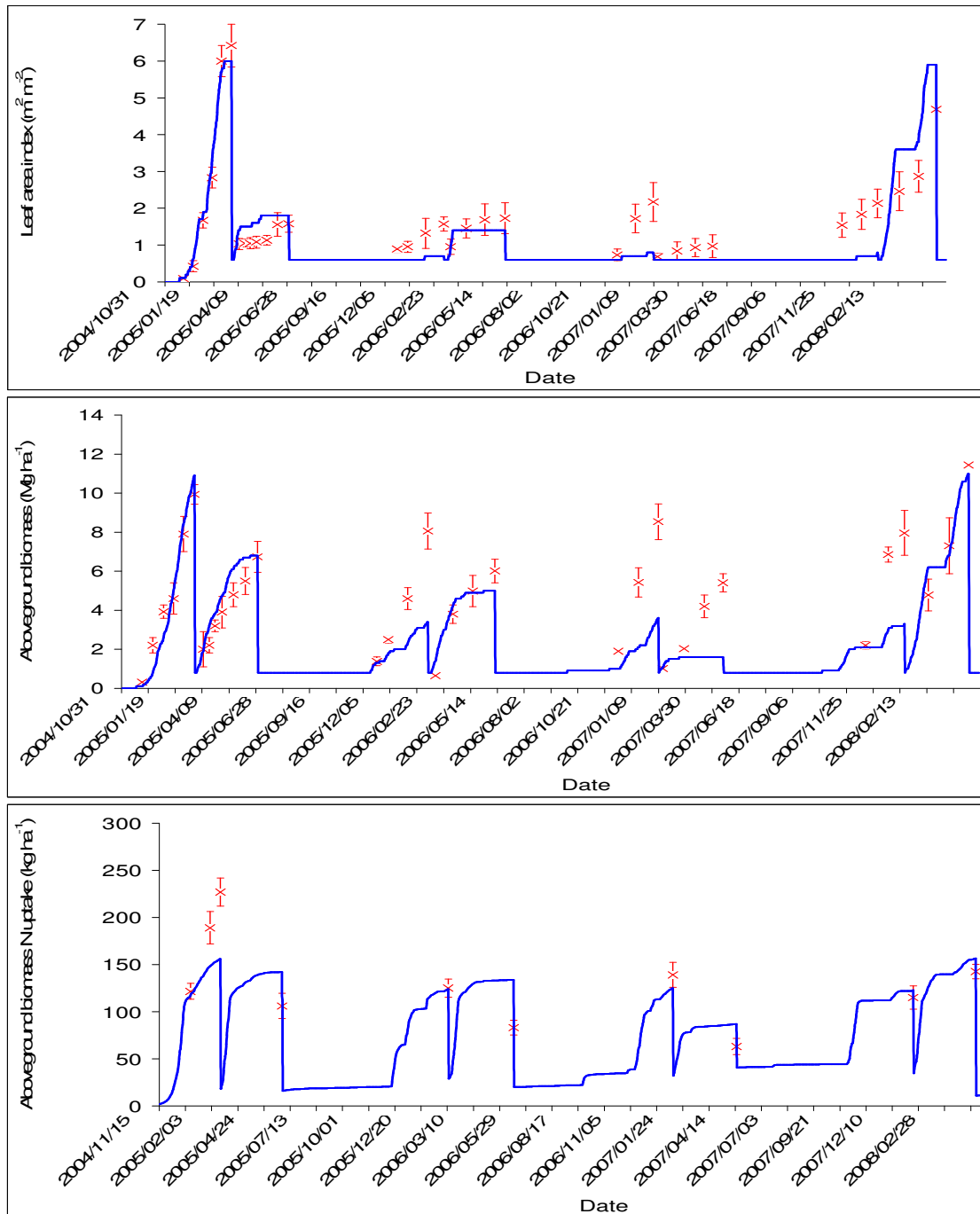


Figure 7.7 Simulated (solid lines) and measured values (symbols with standard deviation) of weeping lovegrass leaf area index (a), aboveground biomass (b), and aboveground biomass N uptake (c) for the $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatment during the 2004/05 to 2007/08 study period (without updating soil water).

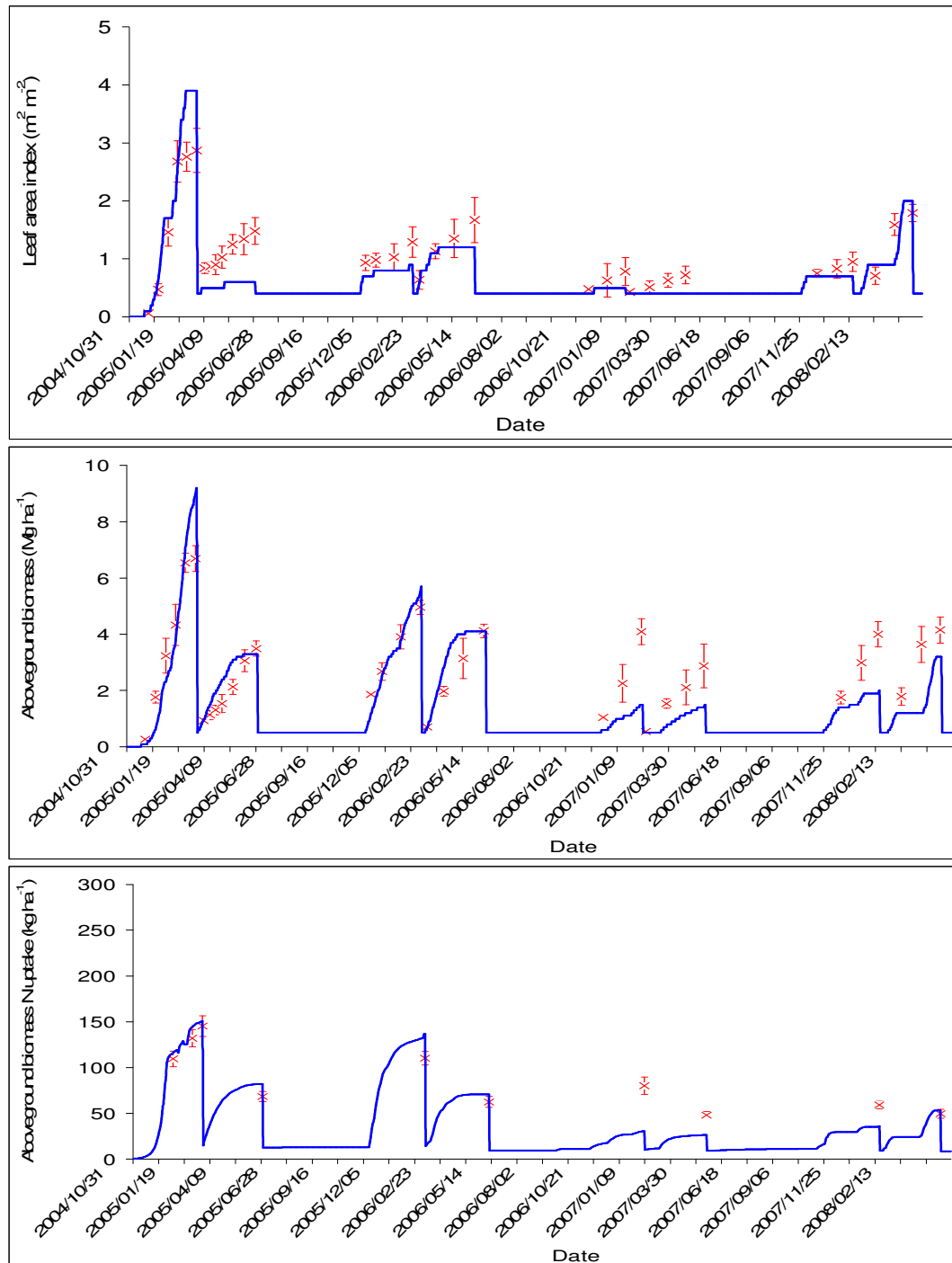


Figure 7.8 Simulated (solid lines) and measured values (symbols with standard deviation) of weeping lovegrass leaf area index (a), aboveground biomass (b), and aboveground biomass N uptake (c) for the control treatment (0 nutrients applied) during the 2004/05 to 2007/08 study period (without updating soil water).

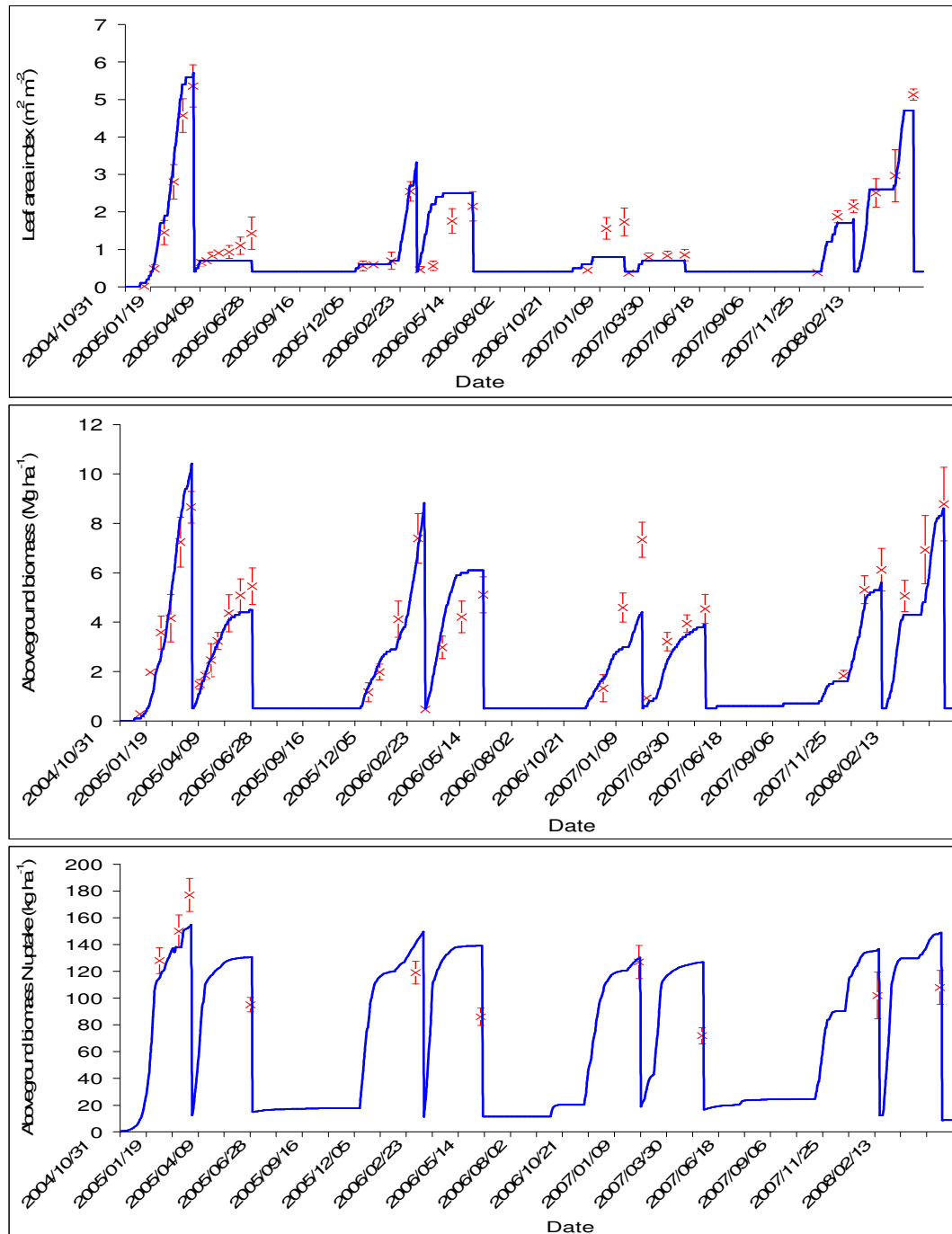


Figure 7.9 Simulated (solid lines) and measured values (symbols with standard deviation) of weeping lovegrass leaf area index (a), aboveground biomass (b), and aboveground biomass N uptake (c) for the $8 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatment during the 2004/05 to 2007/08 study period (without updating soil water).

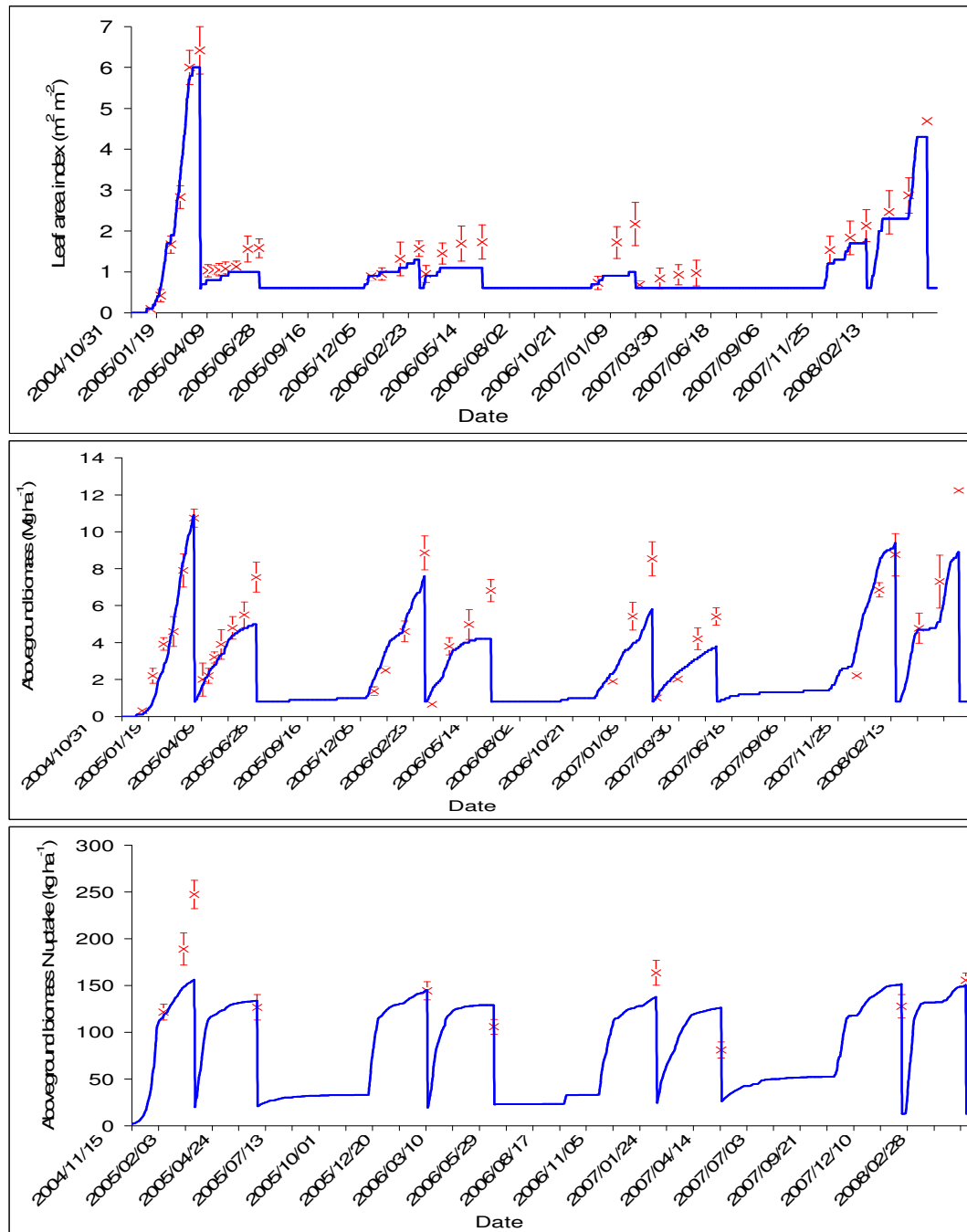


Figure 7.10 Simulated (solid lines) and measured values (symbols with standard deviation) of weeping lovegrass leaf area index (a), aboveground biomass (b), and aboveground biomass N uptake (c) for the $16 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ sludge treatment during the 2004/05 to 2007/08 study period (without updating soil water).

7.3 Conclusions

The model was successfully calibrated for maize, oats, and weeping lovegrass. The model simulated both LAI, aboveground biomass, grain yield, aboveground biomass N uptake, and grain N uptake with acceptable accuracy. Model corroboration conducted using three year independent sets of data also proved accuracy of the model in simulating the above mentioned variables for dryland maize and irrigated maize-oat rotation. The model shows promise as a decision support tool. For dryland pasture, the model predicted similar variables of interest with lower accuracy for long-term simulations. The simulation was improved with updating profile water content after every hay cut. This was mainly because the model does not simulate long-term perennial grass growth mechanistically, and the model needs to be improved. Model accuracy in predicting nitrate leaching from agricultural lands treated with sludge should, however, be tested using data collected from controlled systems such as drainage lysimeters.

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CHAPTER 8

CONCLUSIONS

Sludge applied at double the old 8 Mg ha⁻¹ limit significantly increased grain and forage yield in two of the three years under dryland maize and in all of the three years studies for the irrigated maize-oat rotation. Grain and forage yield of irrigated maize exhibited a decreasing trend over time, due to the greater crop N uptake than that supplied with sludge. In contrast, under dryland maize cropping, a net positive total N accumulation was apparent for sludge rates of 8 Mg ha⁻¹ and higher. Nitrogen exported in the forage was about twice that of the grain, under both dryland and irrigated conditions. Generally, N exported in the forage from the irrigated rotation was at least three times higher than similar treatments under dryland maize forage production. Soil solution samples collected from wetting front detectors at 0.3 and 0.6 m depths indicated higher nitrate leaching risks from inorganic fertilizer treatments than from the 16 Mg ha⁻¹ sludge treatment, usually shortly after fertilizer applications. Residual nitrate after crop harvest in the soil profile of the 16 Mg ha⁻¹ dryland maize treatment was higher than the irrigated maize-oat rotation treatment having the same sludge application. Nitrogen mineralization from sludge could, however, take several years before reaching steady-state conditions. Therefore, long-term model simulations across a range of soil types, sludge N concentrations, and sludge qualities should be conducted. Sludge applications at all rates studied resulted in the accumulation of total P and Bray-1P, both for dryland and irrigated cropping

systems. The only exception was Bray-1P in the 8 Mg ha⁻¹ irrigated rotation which remained similar throughout the study.

Doubling of the 8 Mg ha⁻¹ sludge limit increased weeping lovegrass hay yield, crude protein content, and water use efficiency. Sludge applied at the 8 Mg ha⁻¹ limit was not sufficient to satisfy the demand of weeping lovegrass, resulting in crop N uptake from soil reserve and a decline in total profile N. Sludge applied at 16 Mg ha⁻¹, however, exceeded crop N demand resulting in the accumulation of total N, but not residual nitrate and ammonium. The low residual nitrate and ammonium together with the low nitrate concentration in soil solution collected from the 0.3 m deep wetting front detectors of the 16 Mg ha⁻¹ treatment indicated low risk of ground water pollution through nitrate leaching. Total P accumulation was evident for all sludge rates. Bray-1P decreased as sludge rate increased up to 8 Mg ha⁻¹. This is in contrast to similar treatment under dryland maize cropping. Sludge applied at 16 Mg ha⁻¹, however, increased Bray-1P significantly across years.

Increasing sludge application rates up to 67 Mg 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 Mg ha⁻¹, but deteriorated at higher rates. High loading rates of 100 Mg ha⁻¹ minimized soil loss from the site but caused unacceptably high leaching losses.

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 steady state conditions. 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, 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 low accuracy due to the non mechanistic approach followed by the model when simulating perennial grasses.

Municipal sludge could be applied above the 8 Mg ha^{-1} limit to satisfy the high N demand of intensive cropping systems such as the irrigated maize-oat rotation and dryland pasture with minimal environmental impacts through nitrate leaching. Soil P accumulation and its availability should, however, be monitored for overall environmental sustainability. Ideally the upper sludge limit to satisfy crop N demand should be dynamic and could exceed the old 8 Mg ha^{-1} upper limit. 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.

According to this study, anaerobically digested paddy dried sludge could be applied as high as 33 Mg ha⁻¹ for turfgrass sod production - four times higher than the 8 Mg ha⁻¹ limit. This rate improved turf growth and quality with minimal environmental impact through nitrate leaching.

This study will help to improve the local knowledge of the benefits and disadvantages of sludge application in agricultural lands. Moreover, results from this study will contribute to the plan to update the current guideline on agronomic crops (Volume 2 (Snyman and Herselman, 2006)) and the development of Volume 4 on beneficial use at high loading rates, which is still under development.

Results from this study confirm previous studies showing total P accumulation in N based agricultural sludge management practices. Results will also contribute to the few field scale long term P availability studies on Fe salt treated sludges applied to agricultural land at various rates, different cropping systems, and management practices. The N model looks promising especially for dryland maize and irrigated maize-oat rotation to be used as decision supporting tool for sustainable sludge use in agricultural lands.

Recommendations for future research include:

- A monitoring protocol to complement the decision support system, to ensure that leaching losses were kept within reasonable limits.
- The maximum number of years which sludge could be left without incorporation under perennial pasture grasses with minimal environmental impacts 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 model to simulate perennial grasses mechanistically.

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APPENDIX

Table A1 Selected macro nutrients and heavy metals supplied from three sludge rates to dryland maize, irrigated maize-oat rotation and dryland pasture during the 2004/05 to 2007/08 growing seasons.

Selected macro nutrients supplied from three sludge rates									
Year	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹
	K			Ca			Mg		
	kg ha ⁻¹								
2004/05	15.0	30.0	61.0	100.5	201	402	21.0	43.0	86.0
2005/06	3.0	6.0	12.0	52.0	104	208	2.4	4.7	9.5
2006/07	2.8	5.5	11.0	70.0	140	280	3.3	6.6	13.3
2007/08	5.4	10.8	21.6	40.0	80	160	4.6	9.0	18.0
Cumulative supply	26.2	52.3	105.6	262.5	525	1050	31.3	63.3	126.8

Year	Selected heavy metals supplied from three sludge rates												
	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹				
	Cd			Hg			Cr						
	kg ha ⁻¹									As			
	2004/05	0.0065	0.013	0.026	0.0068	0.014	0.027	0.21	0.42	0.83	0.028	0.057	0.11
	2005/06	0.00028	0.00056	0.0012	0.00008	0.00016	0.00032	0.006	0.012	0.024	0.00072	0.00144	0.00288
	2006/07	0.0006	0.0012	0.0024	0.00012	0.00024	0.00048	0.012	0.024	0.048	0.00092	0.00184	0.00368
	2007/08	0.076	0.15	0.30	0.007	0.014	0.28	2	4	8	0.072	0.14	0.28
Cum. supply	0.08338	0.16476	0.3296	0.014	0.0284	0.3078	2.228	4.456	8.902	0.10164	0.20028	0.39656	

Year	Selected heavy metals supplied from three sludge rates												
	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹	4 t ha ⁻¹	8 t ha ⁻¹	16 t ha ⁻¹				
	Pb			Zn			Ni						
	kg ha ⁻¹									Cu			
	2004/05	0.22	0.44	0.87	1.84	3.68	7.36	0.095	0.19	0.38	0.39	0.78	1.56
	2005/06	0.038	0.075	0.15	0.017	0.035	0.069	0.0055	0.011	0.022	0.013	0.026	0.051
	2006/07	0.00548	0.01096	0.022	0.0834	0.17	0.33	0.0039	0.0078	0.016	0.18	0.36	0.72
	2007/08	0.4	0.8	1.6	9.3	18.7	37.2	0.58	1.16	2.3	2.11	4.21	8.42
Cum. supply	0.66348	1.32596	2.642	11.2404	22.585	44.959	0.6844	1.3688	2.718	2.693	5.376	10.751	

Table A2 Selected macro nutrients and heavy metals supplied from three sludge rates to turfgrass sod production during the 2004/05 to 2005/06 growing seasons.

Selected macro nutrients supplied from three sludge rates									
Year	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹
	K			Ca			Mg		
	kg ha ⁻¹								
2004/05	126.00	255.00	380.00	828.83	1682.77	2511.60	176.81	358.99	535.80
2005/06	23.43	47.57	71.00	431.05	875.15	1306.20	19.50	39.60	59.10
Cumulative supply	149.43	302.57	451.00	1259.88	2557.92	3817.80	196.31	398.59	594.90

Selected heavy metals supplied from three sludge rates												
Year	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹
	Cd			Hg			Cr			As		
	kg ha ⁻¹											
2004/05	0.0538	0.1092	0.163	0.0561	0.1139	0.17	1.71	3.48	5.19	0.23	0.47	0.71
2005/06	0.0023	0.0047	0.007	0.0007	0.0014	0.002	0.05	0.10	0.15	0.01	0.01	0.02
Cum. supply	0.0561	0.1139	0.170	0.0568	0.1153	0.172	1.76	3.58	5.34	0.24	0.48	0.73

Selected heavy metals supplied from three sludge rates												
Year	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹	33 t ha ⁻¹	67 t ha ⁻¹	100 t ha ⁻¹
	Pb			Zn			Ni			Cu		
	kg ha ⁻¹											
2004/05	1.80	3.65	5.45	15.18	30.81	45.99	0.79	1.60	2.38	3.21	6.51	9.72
2005/06	0.31	0.63	0.94	0.14	0.29	0.43	0.05	0.09	0.14	0.11	0.22	0.32
Cum. supply	2.11	4.28	6.39	15.32	31.1	46.42	0.84	1.69	2.52	3.32	6.73	10.04

Table A3 Statistical parameters of the SWB model corroboration for weeping lovegrass (Soil water content updated after every hay cut) using combined data collected during the 2004/05 to 2007/08 growing seasons.

Variables	n	D	RMSE	MAE (%)	R ²
Weeping lovegrass control treatment					
LAI	34	0.83	0.44	29	0.90
Aboveground biomass	34	0.71	1.03	27	0.85
Aboveground biomass N uptake	10	0.60	22.8	20	0.99
Weeping lovegrass 8 Mg ha⁻¹ per annum sludge treatment					
LAI	34	0.93	0.47	20	0.95
Aboveground biomass	34	0.87	0.94	17	0.92
Aboveground biomass N uptake (all data)	10	-0.27	35.00	25	0.37
Weeping lovegrass 16 Mg ha⁻¹ per annum sludge treatment					
LAI	34	0.95	0.40	18	0.98
Aboveground biomass	34	0.84	1.28	19	0.91
Aboveground biomass N uptake	10	0.30	39.32	18	0.72