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 Mg ha^{-1} yr^{-1} which was based on studies conducted in other countries. It was recently updated to 10 Mg ha^{-1} 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 Mg ha^{-1} 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 Mg ha^{-1} 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 Mg ha^{-1} 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 Mg ha^{-1} yr^{-1} sludge limit.
3. Under high hay yield productin 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).

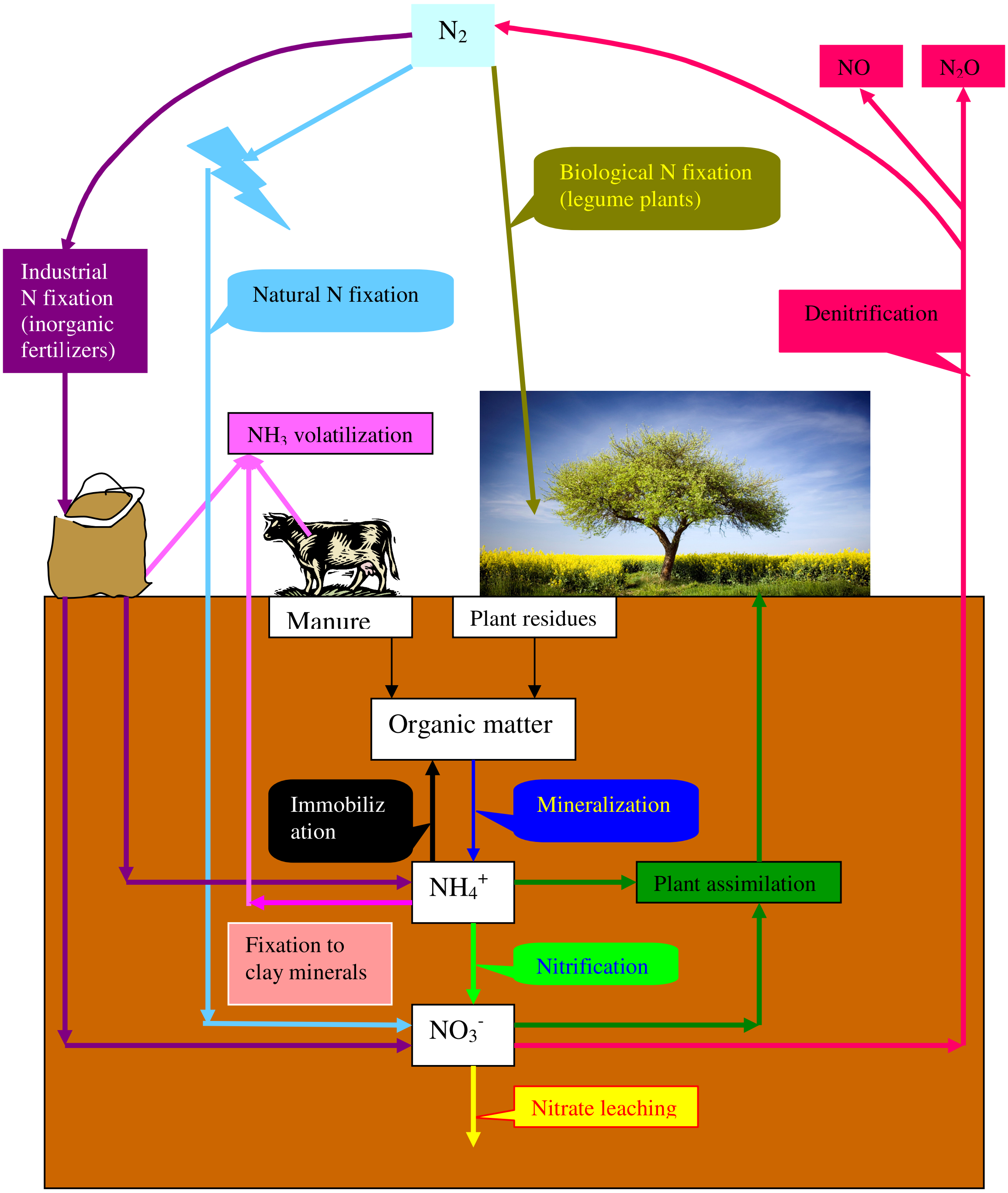

Figure 2.1 Simplified nitrogen cycle in terrestrial plant-soil system.

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).

<table>
<tr><td colspan="2">Treatment method</td><td>Mineralization (% of initial organic N)</td></tr>
<tr><td rowspan="3">Anaerobically digested</td><td>Liquid</td><td>20 – 40</td></tr>
<tr><td>Dewatered</td><td>25 – 45</td></tr>
<tr><td>Heat-dried</td><td>25 – 45</td></tr>
<tr><td colspan="2">Aerobically digested</td><td>30 – 50</td></tr>
<tr><td colspan="2">Lagooned</td><td>10 – 30</td></tr>
<tr><td colspan="2">Lime-stabilized</td><td>30 – 60</td></tr>
<tr><td colspan="2">Composted</td><td>0 – 30</td></tr>
<tr><td colspan="2">Drying bed</td><td>15 – 40</td></tr>
<tr><td colspan="2">Oxidation ditch</td><td>30 - 50</td></tr>
</table>

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).

<table>
<tr><th colspan="4">Treatment method</th><th>Volatilization rate (% of initial ammonia lost)</th></tr>
<tr><td rowspan="6">Anaerobically digested</td><td colspan="3">Liquid (incorporated)</td><td>20 – 40</td></tr>
<tr><td rowspan="5">Dewatered</td><td rowspan="4">Incorporated</td><td>Agronomic rate</td><td>14 – 50</td></tr>
<tr><td>Double agronomic rate</td><td>25 – 49</td></tr>
<tr><td>Lime amended</td><td>45 – 134</td></tr>
<tr><td>Reduced pH</td><td>12 – 28</td></tr>
<tr><td colspan="2">Surface applied</td><td>51 - 127</td></tr>
<tr><td colspan="4">Aerobically digested (incorporated)</td><td>6 – 12</td></tr>
<tr><td colspan="4">Lagooned (incorporated)</td><td>4 – 20</td></tr>
<tr><td colspan="4">Lime-stabilized (incorporated)</td><td>14 – 22</td></tr>
<tr><td colspan="4">Drying bed (incorporated)</td><td>2 – 5</td></tr>
<tr><td colspan="4">Oxidation ditch (incorporated)</td><td>9 - 23</td></tr>
</table>

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 denitrificaction 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).

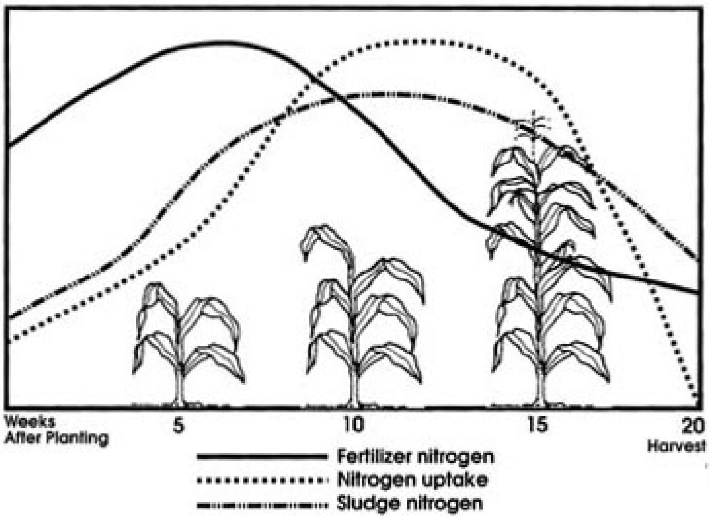

Figure 2.2 Nitrogen requirement of maize during the growing season and nitrogen availability from fertilizer compared with sludge (adapted from Muse et al., 1991).

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 methemeglobinemia 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).

	South African microbiological class				US microbiological class	
	A	B		C	A	B
Organism	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
Salmonela 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^{o}C (average temperature 45^{o}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 Sulivan et al. (1997), prairie grass (*Bromus uniloides (Willd)*) dry matter yield increased as sludge application rate increased to 26.9 Mg ha^{-1} (1064 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 Mg ha^{-1} yr^{-1} (>800 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 20-50 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 kg N ha^{-1} increased the concentration of NO_3-N in the grass to hazardous levels for livestock consumption (>3 mg g^{-1}). Regarding the health 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). In 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.

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

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).

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

Where

CN ratio_{decomp} – is CN ratio of the decomposing pool

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

CNratio_{pool} – is the CN ration of the receiving pool.

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

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.

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

Where

T - is real time soil temperature

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

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

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

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

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.

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

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

$$\text{If } 0.7 \leq WFP \leq 1 \quad WF = WF_{sat} + (1 - WF_{sat}) * ((1 - WFP) / (1 - WFP_{high}))\ \hat{}\ 2 \tag{7}$$

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, where as 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.

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

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.

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

Where

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

NH_4 – is the mass of ammonium N

NO_3 – is the mass of nitrate N

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

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

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

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

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

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.

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

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

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

Where

WFP – water filled porosity

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

DWF – denitrification soil water function

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

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

Where

$N_{denitrified}$ – denitrified N mass

k – potential denitrification constant (0.000032)

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

RRF – respiratory response function

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

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

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^{o}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 kg NH_3-N ha^{-1} 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 (NH_4-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 NH_4-AV is greater than zero. If the NH_4-AV is greater than zero, Cropsyst estimates ammonia volatilization using Eq. 16.

$$NH_{3\text{-volatilized}} = ((K_H * NH_{3\text{-conc}})/p_{atm}) * TTC \qquad (16)$$

Where

$NH_{3\text{-volatilized}}$ – is volatilized ammonia mass

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

$NH_{3\text{-conc}}$ – 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 Mg ha^{-1} yr^{-1} and was recently increased to an upper limit of 10 Mg ha^{-1} 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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