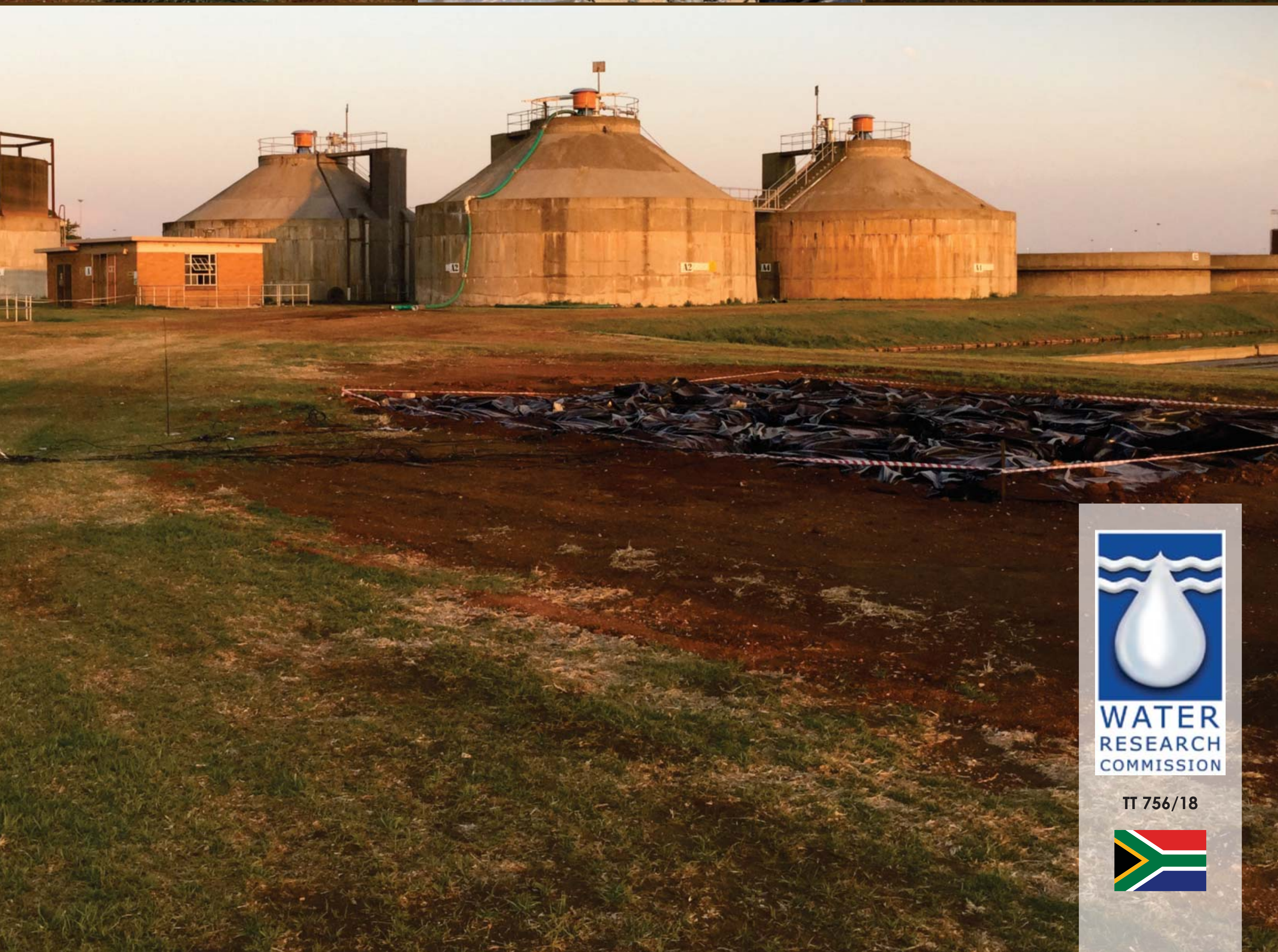
CHARACTERISING MUNICIPAL WASTEWATER SLUDGE FOR SUSTAINABLE BENEFICIAL AGRICULTURAL USE

EH Tesfamariam, T Badza, T Demana, J Rapaledi and JG Annandale



WATER RESEARCH COMMISSION

TT 756/18

CHARACTERISING MUNICIPAL WASTEWATER SLUDGE FOR SUSTAINABLE BENEFICIAL AGRICULTURAL USE

Report to the

Water Research Commission

by

EH Tesfamariam, T Badza, T Demana, J Rapaledi and JG Annandale

Department of Plant and Soil Sciences, University of Pretoria, Pretoria 0002, South Africa

WRC Report No. TT 756/18

July 2018

Obtainable from:

Water Research Commission

Private box X03

Gezina, 0031

orders@wrc.org.za or download from www.wrc.org.za

The publication of this report emanates from a project entitled 'Characterizing municipal wastewater sludge for sustainable beneficial agricultural use'. (WRC Project No. K5/2477)

ISBN 978-1-4312-0997-2

Printed in the Republic of South Africa

EXECUTIVE SUMMARY

World human population growth in general is expected to place significant new demands on already strained renewable and non-renewable natural resources, thereby causing environmental degradation, while reinforcing social inequity and poverty. Hence, the United Nations (UN) developed an agenda of 'Sustainable Development Goals' to assist with attending to these challenges, of which waste recycling forms the core to achieving more than half of them. Recycling of wastewater sludge in agricultural lands serves as a source of water (liquid sludge) and crop nutrients (both solid and liquid sludge), thus playing a crucial role in reducing the pressure on fresh water bodies (liquid sludge) and lowering greenhouse gas emissions from fertilizer production. Wastewater sludge is a mixture of materials of biological and non-biological origin with varying levels of toxic effects on the environment. Therefore, for sustainable use of these resources, suitability characterization should be set for their use in agricultural lands, and for the rate at which sludge should be applied. This process should take into account the following: crop nutrient requirements adjusted to the given agro-ecological zone, farming system and soil type, nitrogen fertilizer value of sludge as affected by treatment processes, dewatering techniques, and sludge drying depths. The existing South African Sludge Guidelines 'Guidelines for the Utilisation and Disposal of Wastewater Sludge, volume 2, 2006' lacks such detailed instructions for sludge users. This study was therefore geared to provide the necessary guidelines. The overarching aim of the study was to establish quantitative relationships between sludge fertilizer value (nitrogen release rate) and wastewater sludge treatment processes, drying techniques, and sludge nitrogen (N) composition, and to use these to refine the current South African sludge guidelines. This aim was addressed through the following detailed objectives:

- Characterize selected representative liquid wastewater sludges
- Determine the effect of sludge treatment methods, sludge drying time and drying depth (thickness) above drying beds, on sludge N and organic matter content
- Establish a quantitative relationship between sludge N fertilizer value (N release rate) and sludge treatment processes, post treatment dewatering techniques, and sludge N composition
- Refine the current South African sludge guidelines.

To achieve these objectives, four activities were undertaken: a literature study, liquid municipal wastewater sludge characterization, laboratory incubation studies, field experiments and model scenario simulations. Following a thorough literature review and synthesis of existing knowledge, a quantitative relationship was established between sludge N mineralization on the one hand and wastewater sludge treatment processes, sludge organic-N content, and sludge dewatering techniques on the other. Liquid wastewater sludge characterization was conducted on samples (activated and digested sludges) collected from 18 wastewater treatment plants (WWTPs) in Gauteng, South Africa. Laboratory incubation studies were conducted to investigate the effects of wastewater sludge treatment processes (anaerobic versus aerobic digestion) and post-treatment dewatering and drying techniques (polymerised and belt-pressed versus paddy dried) on sludge N and organic matter contents, sludge organic matter composition, potential N mineralization and organic matter decomposition. Field trials were conducted at Vlakplaats and Waterval plants of the East Rand Water Care Company (ERWAT), Gauteng, South Africa, to quantify the effect of wastewater sludge drying depth above drying beds (5 cm, 10 cm, 15 cm, 20 cm, and 25 cm) on the total N, inorganic-N, total carbon (C) and N fertilizer value. Finally, a calibrated and validated Soil Water Balance (Swb-Sci) model was used to determine cumulative N mineralization from sludges of varying N contents of the five South African agro-ecological zones under both rain-fed and irrigated farming systems.

The pH of all wastewater sludges originating from 18 WWTPs was within acceptable ranges for use in agricultural lands ($pH_{(H2O)}$ of 5.5 – 9.5). Similarly, the trace metal contents of sludges from most of the candidate WWTPs were below the threshold values for use in agricultural lands. The only exceptions were anaerobically digested sludges from two plants (pollutant class A) and two activated sludge plants (pollutant class B). All of the anaerobically digested sludges and the combined anaerobically-aerobically digested sludges had exceptionally high salinity (EC), exceeding the agricultural threshold value of 200 mS m^{-1}, making them unfeasible for use in agricultural lands. Besides, these sludges had Total Suspended Solids (TSS) values exceeding the threshold value of 25 mg L^{-1}. In contrast, sludges from waste-activated plant had EC values of less than 200 mS m^{-1} and TSS values of less than 25 mg L^{-1}. The problems associated with high TSS on soil hydraulic conductivity, however, could be minimized by scheduled intervention through cultivation. From the preliminary investigation it could be deduced that the application of wastewater sludge according to crop water requirements would lead to excess nutrient application above crop nutrient demands, which would compromise sustainability. *Henceforth, wastewater liquid sludge should be applied according to crop nutrient requirements.*

Wastewater sludge treatment and dewatering techniques affect the N content of sludge, thus influencing the fertilizer value. *Sludge dried in thin layers of 10 to 15 cm thickness generated higher total N and N fertilizer value than sludge dried in thick layers of 20 cm and higher. The final decision on the choice of sludge drying thickness on drying beds, however, depends on other factors, including the availability of land for drying and the ease of sludge collection from drying beds.*

Inter-comparison of a six-year cumulative N mineralization rate (%) among years in which a single 10 t ha^{-1} sludge was applied during the lowest rainfall year, an average rainfall year, and the highest rainfall year showed little variation. Similarly, cumulative N mineralization rate (%) from a single sludge application rate of 10 t ha^{-1} for sludges with a range of N contents (2% to 6%) showed significant variation during the first and second year after application. However, it stabilized afterwards, showing little variation among sludges of similar origin, but varying N contents. Cumulative N mineralization at a specific site, however, was generally higher under irrigation compared with dryland farming. *From this it can be deduced that a single N release rate per farming system (dryland vs irrigated) could be used at a specific site for agricultural lands that use sludge in the long term.* This should be synchronized with crop nutrient requirements because crop nutrient requirements also vary, depending on the availability of water. *Hence, sludge guidelines for beneficial land application ought to be synchronized with the South African fertilizer guidelines as presented in the Fertilizer Handbook (FSSA, 2007).* This higher level of integration would contribute towards achieving the UN and South African Sustainable Development Goals.

To continue building on the body of knowledge of the fertilizer value of sludge that has been developed to date, the following further studies are recommended:

- Field assessment of the current findings through model simulations
- Investigation of the dynamics of contaminants of emerging concern in the sludge – soil (solid and solution phase) – plant – animal – human food chain

ACKNOWLEDGEMENTS

The publication of this report emanates from a project titled 'Characterizing municipal wastewater sludge for sustainable beneficial agricultural use' **(WRC Project No. K5/2477).**

The project team gratefully acknowledges and thanks Dr John Zvimba (project manager) and Mr Bennie Mokogonyana (project co-ordinator) and the following institutions for providing funds:

- Water Research Commission
- East Rand Water Care Works (ERWAT)

The project team is grateful for the additional assistance in kind from ERWAT for the past 12 years by providing land, electricity and water for our long-term field experiment studies at Hartbeesfontein, sludge drying depth studies at Vlakplaats plant and Waterval plant, and provision of sludge and information from all of their plants.

Members of the Water Research Commission Reference Group Committee for this project were:

Dr J N Zvimba	:	Water Research Commission (**chairperson**)
Mr J Visagie	:	University of Pretoria
Dr H Snyman	:	Golder Associates
Dr E Herselman	:	Golder Associates
Mr J Topkin	:	ERWAT
Mr D Wilson	:	eThekwini Municipality
Prof T de Jager	:	University of Pretoria

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

Act_FertN-req	Actual nitrogen fertilizer requirement of the crop
ADF	Acid Detergent Fibre
ADL	Acid Detergent Lignin
AeD	Aerobic digestion
AnBp	Anaerobic digestion
CEC	Cation Exchange Capacity
COD	Chemical Oxygen Demand
EC	Electrical Conductivity
ERWAT	East Rand Water Care Company
Fer_Req	Optimum fertilizer requirement
Fer_NReq	Optimum nitrogen fertilizer requirement of the crop
Inorg_N_Mass	Mass of inorganic nitrogen
Inorg_N_Conc	Concentration of inorganic nitrogen
NDF	Neutral Detergent Fibre
Org_N	Organic Nitrogen
N_{min}	Mass of Organic nitrogen mineralized
SARA	Sludge Application Rate Advisor
SD	Standard Deviation
Soil_N	Mass of soil inorganic nitrogen in the top 60 cm soil layer
TN	Total Nitrogen
TS	Total Solids
TSS	Total Suspended Solids
UN	United Nations

WRC	Water Research Commission
WWTP	Wastewater Treatment Plant

CHAPTER 1 - INTRODUCTION

1.1 Background

The ever-increasing human population in the world and consequent increase in food and water demand has constrained already stretched limited natural resources. The global population, which is currently at 7.6 billion, is expected to increase to 8.6 billion by 2030, to 9.8 billion by 2050, and to 11.2 billion by 2100 (United Nations, 2017). Natural resources will not be able to satisfy the expected rise in human food and water demands in a decade's time under current 'business as usual' use of resources. The United Nations has developed an agenda of Sustainable Development Goals to assist in attending to these challenges (United Nations, 2017) (Figure 1.1). Recycling of wastes in general and recycling of wastewater sludge in particular form the core of the solution to more than of half of the 17 goals.

Figure 1.1 United Nations' Sustainable Development Goals

Source:https://www.un.org/sustainabledevelopment/sustainable-development-goals/

1.2 Motivation

Recycling of wastewater sludge in agricultural lands as a source of water for irrigation can reduce the pressure on freshwater resources, because agriculture uses 70% of the fresh water globally. In addition, wastewater sludge serves as a source of crop nutrients, which are critically needed by crops for their growth and development, because it is rich in mineral nutrients and organic matter. However, wastewater sludge is a cocktail of wastes originating from various sources. Therefore, proper characterization of this resource for its suitability for use in agricultural lands is crucial to achieving the relevant Sustainable Development Goals.

Similarly, dewatered wastewater sludge, with acceptable quality for agricultural use, is used as a source of essential crop nutrients and as a soil conditioner. Usually sludge is applied to agricultural lands according to crop nitrogen (N) requirements. A large proportion (>70%) of the N in sludge, however, is in organic forms and has to be mineralized before crops can utilize it. The rate at which the organic-N is mineralized and the total N content of a sludge, however, are influenced by various factors. Inherent factors that can affect the total N content of a sludge and its mineralization rate include the origin of the sludge, wastewater treatment processes, dewatering techniques, and sludge drying depth on drying beds. Despite this, the relationships between sludge N content and N mineralization rate on the one hand and sludge treatment processes, drying techniques, and sludge drying depth on drying beds on the other have not yet been established. Setting quantitative relationships between these factors is essential, because sludge recommendation rates rely on sludge N content and release rate. Therefore, proper characterization of the effects of wastewater treatment methods, dewatering techniques, and drying depth on drying beds is essential towards sustainability.

The final aim of characterizing wastewater sludge should be to synchronize crop water and nutrient requirements with sludge water content, N content and sludge N release rate. This enables the reuse of resources (nutrients and water) to enhance productivity (reduce hunger and poverty and enhance good health and wellbeing), reduce ground and surface water contamination (clean water and sanitation, reduce negative impacts on life below water, and life on land), reduce contribution to climate change (by minimizing the use of commercial inorganic fertilizer), encourage the responsible use of resources (responsible consumption and production), in due course enhancing sustainability (sustainable cities and communities), thus striving towards achieving the Sustainable Development Goals.

1.3 Aim of the study

The main objectives of the study were:

1. To characterize selected representative liquid wastewater sludges and assess:
    a. Their feasibility for land application
    b. The criteria for decision on the rate of land application (according to crop water requirement or crop nutrient requirements)
2. To determine the effect of sludge treatment methods, sludge drying time and drying depth (thickness) on drying beds on:
    a. The nitrogen content of sludge
    b. The organic matter content and composition of sludge
3. To establish a quantitative relationship between sludge N fertilizer value (N release rate) and sludge treatment processes, post-treatment dewatering techniques, and sludge N composition.
4. To refine (as an addendum) the current South African Sludge Guidelines, 'Guidelines for the Utilisation and Disposal of Wastewater Sludge, Volume 2 (2006)'

1.4 Approach to the study

To achieve the stated objectives, the following activities as presented in the diagram were undertaken:

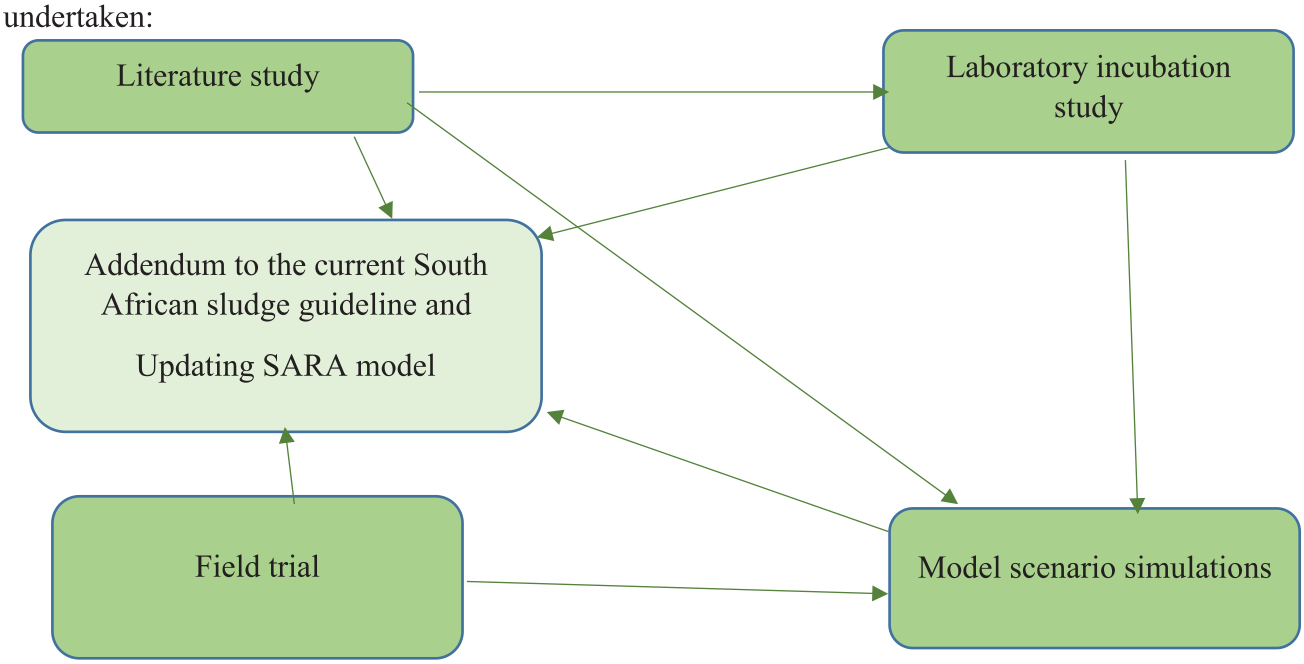


Figure 1.2 Flow diagram of activities undertaken to update existing South African sludge guidelines and Sludge Application Rate Advisor (SARA) model

CHAPTER 2 - LITERATURE REVIEW

2.1 Introduction

Recycling treated wastewater sludge in agricultural lands could play a significant role in achieving several of the UN Sustainable Development Goals. Goals that can be achieved through recycling wastewater sludge in agricultural lands include; poverty alleviation, responsible consumption and production, sustainable cities and communities, clean water and sanitation, good health and wellbeing, climate action, life below water, life on land, and zero hunger. Recycling treated wastewater sludge in agriculture serves as a source of water (liquid sludge) and nutrients (liquid and dewatered sludge). Insufficiency of water and nutrients are the most critical limiting factors for agricultural production in sub-Saharan African countries, where 50% of the population lives below the poverty line. The use of treated wastewater sludge in agricultural lands can relieve the pressure on freshwater resources, while boosting agricultural productivity because of the nutrients added with the sludge. In this case, the need for commercial inorganic fertilizers is decreased, thus minimizing greenhouse gas emissions during fertilizer manufacturing. In addition, the use of wastewater sludge in agricultural lands under controlled irrigation management practices safeguards ground and surface water bodies from contamination, leading to clean water and sanitation while protecting life below water. Consequently, the use of wastewater sludge in agricultural lands is receiving great attention worldwide. However, this is due not only to the unprecedented increase in water demand or wastewater disposal challenges, but to the shortage of fresh water, especially in arid and semi-arid regions, increases in fertilizer costs, greenhouse gas emissions during fertilizer production, and the ever-increasing world population (Hanjra et al., 2012).

2.2 Wastewater treatment processes and their associated effects on sludge nutrient availability for crops

The physical, chemical, and biological properties of wastewater sludge vary depending on their origin (domestic or industrial) (Feigin et al., 2012), treatment processes and treatment stage (Bresters et al., 1997; Kunhikrishnan et al., 2012). The main processes or operation units involved in the wastewater treatment include preliminary, primary, secondary and tertiary stages (Tchobanoglous & Burton, 1991b; Ramalho, 2012 [Table 2.1]).

Table 2.1 Conventional classification for wastewater treatment processes

Primary	Secondary	Tertiary (advanced treatment)
Screening	Activated sludge process	Micro-screening
Sedimentation	Extended aeration (total oxidation process)	Precipitation & coagulation
Flotation	Contact stabilization	Adsorption (activated carbon)
Oil separation	Other modifications of conventional activated sludge process (tapered aeration, step aeration, complete mix activated sludge process)	Ion-exchange
Equalization	Aerated lagoons	Reverse osmosis
Neutralization	Wastewater stabilization ponds	Electrodialysis
	Trickling filter	Nutrient removal processes
	Anaerobic treatment	Chlorination & ozonation
		Sonozone process

Source: Adapted from Ramalho (2012)

2.2.1 Primary sludge

Primary sludge is produced as a result of the sedimentation process of the primary stage when all solids are settled at the bottom of settling tanks or clarifiers (Spinosa & Vesilind, 2001; Kadlec & Wallace, 2008), while some are floating or suspended. The main aim of primary sedimentation is to reduce the concentration of TSS in the raw wastewater (Kadlec & Wallace, 2008). Pescod (1992) indicated that approximately 25–50% of the biochemical oxygen demand, 50–70% of the TSS, and 65% of oil and grease are removed during primary sedimentation. This sludge is characterized by high populations of pathogens, general grit, organic and inorganic substances. Generally primary sludge contains 3–7% total solids (TS), of which about 60–70% is organic matter on a dry mass basis (Girovich, 1996). In some countries, primary sludge is disposed of on landfills, while in other countries it is not permitted because it is highly putrescible and generates unpleasant odours (Turovskiy & Mathai, 2006).

2.2.2 Secondary sludge

Secondary sludge, also known as biological sludge, is generated from secondary treatment processes. There are three basic biological treatment methods: the trickling filter, the activated sludge process, and the oxidation pond. There is also a fourth, less common method, called a rotating biological contactor. This process uses bacteria and fungi to remove additional wastewater solids and organic matter (Kadlec & Wallace, 2008). It is used mainly for the removal of large quantities of suspended solids, and hazardous and non-hazardous organic and soluble materials. At the secondary stage, TSS reductions of about 85% are expected (Kadlec & Wallace, 2008). The sludge from the secondary treatment stage has about 0.5–2% TS, of which 60–70% is organic matter (Girovich, 1996).

2.2.3 Advanced sludge

Tertiary treatment produces an advanced treated sludge referred to as tertiary sewage sludge. Processes involved in this treatment stage are costly, and are applied only when deemed necessary (Ramalho, 2012). It is used in particular for the removal of nutrients and other residual pollutants and metals that could not be eliminated in the previous treatment stages. Processes included at this stage are oxidation/reduction, chemical precipitation constituted with the use of lime (Bishop, 1995), the addition of iron and aluminium, chemical additions such as metal salts and polymers, and filtration before disposition (US EPA, 1995). Generally, advanced sludge contains 0.2–1.5% TS of which 35–50% is organic matter (Girovich, 1996).

2.2.4 Sludge stabilization

The residues from primary and secondary treatment processes are further stabilized with the objective of reducing pathogens, eliminating odour, and reducing organic matter. The most commonly utilized processes to stabilize solids generated from WWTPs are anaerobic digestion, aerobic digestion, lime/alkaline stabilization, composting, long-term storage in lagoons or reed beds, and thermal processes.

Anaerobic digestion of sludge is the decomposition of organic material in the absence of oxygen. It is a biological and complex process, which requires strong anaerobic conditions to work successfully. The principal steps of this process are hydrolysis, acidogenesis (use of acidogenic/fermentative bacteria in the presence of ammonia (NH_3), carbon dioxide (CO_2) and hydrogen sulphide (H_2S)), acetogenesis (action of acetogenic bacteria) and methanogenesis (process of methanogenic bacteria). This last stage of the process produces high energy methane (CH_4) gas. According to Li et al. (2011) and Nkoa (2014), anaerobic digestion has gained attention because of its capacity to generate energy and to produce nutrient-rich sludge for agricultural use. The composition and quality of the sludge produced, however, depends on the origin of the sludge material, feedstock (wastewater material) properties and digestion operating conditions such as temperature and retention time (Sheets et al., 2015). During the sludge digestion phase, various chemical elements and physical property changes take place. Although there is limited N assimilation by the digestion microorganisms, organic-N was seen to be degraded during aerobic digestion (Li et al., 2011). Owing to minimal N assimilation, anaerobic digested sludge has high levels of total N (TN) of approximately 157 g TN kg^{-1} dry matter (DM), and 50–59 % total ammonia N, which is a combination of NH_4 and free NH_3 (Fouda et al., 2013). However, the ratio of NH_4 to NH_3 varies depending on temperatures and pH levels (Sheets et al., 2015). High amounts of inorganic-N content in anaerobic digested sludge were observed in studies by Yoshida et al. (2015). They indicated that because of high levels of inorganic-N, there was an immediate availability of N for plants after anaerobic digested sludge was applied.

Aerobic digestion of sludge takes place under the supply of oxygen for proper performance of aerobic and facultative microorganisms during the digestion process. This operation is governed by raw sewage characteristics, pH, mixing, oxygen requirement, temperature and sludge retention time, of which the last two are considered the key parameters (Chang et al., 2011). Aerobic digestion is a two-stage process consisting of direct oxidation of biodegradable matter and endogenous respiration, in which cellular material is oxidized. The latter stage is the dominant phase of the aerobic process and leads to the production of sludge material, CO_2, H_2O, and NO_3 (Zupančič & Roš, 2008). Under aerobic digestion, sludge biomass is oxidized to CO_2 and H_2O, while organic-N is transformed to NO_3 through mineralization (Al-Ghusain et al., 2002) and N is released in this process as ammonium (NH_4) (Zupančič & Roš, 2008). When conditions are conducive, that is, when there is sufficient dissolved oxygen and a suitable alkalinity pH level, NH_4 is further oxidized via nitrification process to form NO_3, leading to acidification of the environment (low pH) (Hamoda & Ganczarczyk, 1980). In the process, TN reduction was observed to be 33.7 % (Al-Ghusain et al., 2002).

Lagoon stabilization is a biological system that is a more natural sludge treatment process. Lagoons are artificially or naturally aerated as a way of promoting biological oxidation. Sludge quality with respect to its microbiological pathogen load and reduction of chemical elements is enhanced through the use of lagoons. Nitrogen is among the various elements reduced by their use. According to Middlebrooks and Pano (1983), the greater percentage of N in lagoons is in the form of ammonia N and exists in aqueous solution as ammonia or ammonium ion. Although a number of factors govern the rate of gaseous N losses in ammoniacal form from lagoon ponds, pH is reported to be the major controlling factor. The pH levels of lagoon ponds are usually about 8.0 or less. Middlebrooks and Pano (1983) indicated that alkaline pH values shift the equilibrium equation $NH_3 + H_2O <-> NH_4 + OH^-$ towards gaseous ammonium production. The NH_3-N is easily lost into the atmosphere through volatilization, hence reducing the total N content in lagooned sludge. According to Middlebrooks and Pano (1983), there is a positive linear relationship between hydraulic retention time and gaseous NH_3-N losses in lagoons. Besides gaseous losses from lagoon ponds in ammonia form, N is lost through denitrification and sedimentation of insoluble organic-N. Nonetheless, volatilization and sedimentation of organic-N through biological uptake are the main pathways by which N is removed in lagooned sludge (Maynard et al., 1999).

Lime stabilization is a crucial process in wastewater management if the environment is to stay free of contaminants post sludge application. The prime objective of this process is to stabilize sludge by reducing potential substrate producing odour, nutrient elements, and microbiological pathogens to acceptable levels (Stoll, 1996; Schneiter et al., 1982). Lime stabilization enhances N removal through nitrification by increasing alkalinity while adding C, which speeds up denitrification (Ren et al., 2015). The use of alkaline agents results in enhanced thickening and dewatering of sludge due to high pH and Ca^{2+}. Usually lime stabilization is favoured over other stabilization processes such as composting, lagoon, drying, and chemicals owing to its cost effective nature (Ren et al., 2015).

2.3 Wastewater sludge dewatering and drying techniques and their effects on sludge nutrient availability for crops

Dewatering is a post sludge-treatment technology that is employed to reduce sludge volume by reducing its water content and increasing solid concentration, hence enhancing its handling and transportation during disposal. Dewatering is governed by several factors, which include the origin of the sludge, the physical and chemical properties of the sludge, such as floc structure, bound water content, hydrophobicity and surface charges (Jin et al., 2004, Mikkelsen & Keiding, 2002). Wang et al., (2014) indicated that digested sludge has a large percentage of water content (> 75 %). This makes it difficult to dewater. Therefore, this requires chemical treatment such as the addition of coagulants, which reduce water-to-solids adsorption and facilitate dewatering. Although mechanical treatments can be used, these are hardly effective (Wang et al., 2014). The incorporation of chemicals in sludge is known as chemical conditioning. Chemical conditioning has been used widely to facilitate dewatering and sludge compressibility relative to mechanical treatment (Wang et al., 2014). It effectively destroys the polymeric matrix and enhances the release of extracellular polymeric substances (EPS) and bound water in sludge flocs (Vaxelaire & Cézac, 2004, Wu et al., 1998, Sheng et al., 2010). Chemical additives such as aluminium sulphate, ferric chloride, enzymes and surfactants are used (Neyens et al., 2004, Ruiz-Hernando et al., 2013). Surfactants are reported to reduce the surface tension between liquid-to-liquid and liquid-to-solid interphases (Rosen, 2004).

As a means of sludge stabilization, dewatering is performed in various ways, including belt-pressing, sand drying beds, reed beds, thermal drying, and centrifuge use. Depending on the type of dewatering and pre-treatments that are applied, the physical and chemical properties of the resultant sludge can be different. Dewatering techniques therefore affect the chemical composition of the sludge, including N. In addition, since a larger fraction of the N in sludge is in organic form, its availability for crop uptake is influenced by the dewatering techniques used (Table 2.2). In an evaluation of the effect of sludge stabilization on the fertilizer value of sludge, Yoshida et al., (2015) reported a reduction in the total N content of sludge as a result of dewatering. This reduction was evident in the organic and inorganic fractions of N.

Table 2.2 Properties of primary, digested, dewatered digested, and dried digested sludge in the incubation study

Sludge type	NO_3-N	NH_4-N	Total N	Inorganic-N	C : N ratio
		mg g^{-1} d.w		(% of total N)	
Primary sludge	0.3 (0.1)	1.8 (0.5)	26.9 (4.9)	8	15
Digested sludge[1]	1.1 (0.4)	31.4 (6.62)	70.2 (8.2)	46.3	4.2
Dewatered digested sludge[2]	0.1 (0.1)	4.9 (1.5)	43.8 (6.3)	11.5	6.3
Dried digested sludge[3]	0.1 (0.1)	6.7 (1.9)	44.8 (5.9)	15.1	6

Source: Adapted from Yoshida et al. (2015)
Note: Standard deviation in parenthesis (*n*=3)
[1] Generated by mesophilic digestion of primary and secondary sludge
[2] Digested sludge, dewatered using a centrifuge after addition of polymer coagulant
[3] Dewatered digested sludge dried using heat recovered from a sludge incinerator

Heat or thermal drying is one of the stabilization processes towards producing an environmentally friendly sludge for use in agriculture. According to Reimann (1989), heating is done in two phases, at 30–75 ºC and 75–190 ºC at a pressure of 26 bars for a very short period such as four hours. The resultant product after heating is a flocculated sludge with reduced water affinity due to the broken gel structure. Basically, heat breaks down chemical bonds of various elements and likewise N properties in sludge. According to Smith and Durham (2002), inorganic-N was lost from the digested biosolids during high-temperature drying by ammonia volatilization. These authors reported that thermal conditioning significantly increased the mineralizable organic-N content in digested biosolids.

Drying beds are have long been used worldwide as a means of sludge dewatering. There are various types of drying beds. The most common ones are sand drying beds, paddies, and concrete lined beds (Figure 2.1). Dewatering and drying via drying beds was reported to reduce total N in the form of NH_4-N by almost 50% (from 25–60% to as low as 10–30%) (Sullivan et al., 2007).

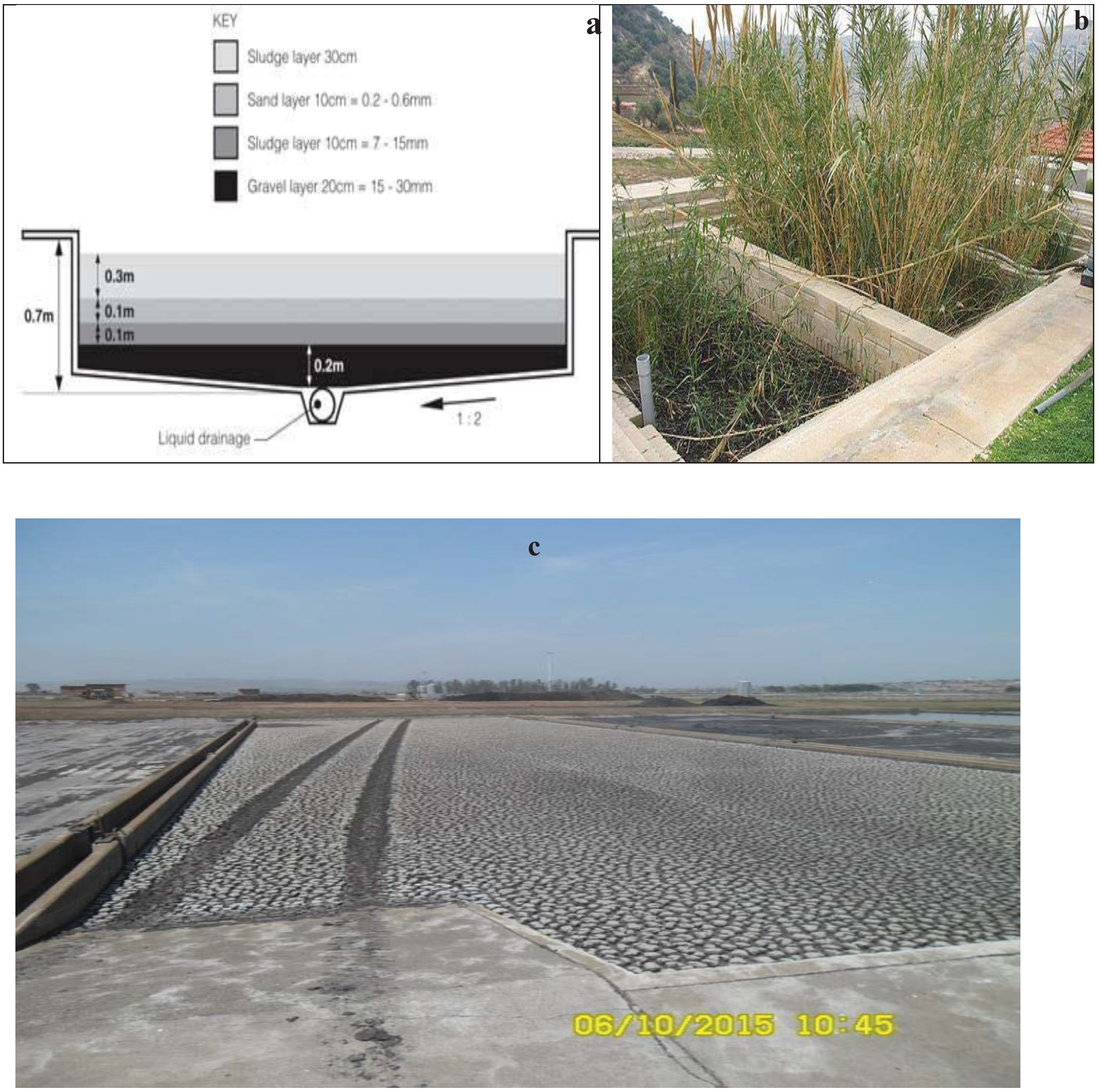


Figure 2.1 Drying beds of various forms: cross-section of a sand bed construction (a), planted sand bed (b) and concrete bed (c)

Source: Wang et al. 2007

CHAPTER 3 - CHARACTERIZING LIQUID MUNICIPAL WASTEWATER SLUDGE FOR ITS SUITABILITY TO USE IN AGRICULTURAL LANDS

3.1 Introduction

Wastewater sludge generated from municipal WWTPs is constituted of various characteristics, including organic and inorganic compounds (Feigin et al., 2012). These are governed by factors such as the source of the sludge, and the type and degree of wastewater treatment process to which it has been subjected. The suitability of wastewater sludge for beneficial agricultural use is determined by these characteristics. The major factors that limit the use of wastewater sludge in agricultural land include pollutants (organic and inorganic), pathogens, and stability (odour) (Snyman & Herselman, 2006), salinity (EC), pH, and Chemical Oxygen Demand (COD) (South African National Water Act 36 of 1998, 2013).

The objectives of this section of the report are:

a. To characterize treated liquid wastewater sludge from 18 representative wastewater treatment plants around Gauteng for their suitability for use in agricultural lands
b. To assess the macro nutrient composition (N, P, and K) of sludge from these representative 18 wastewater treatment plants
c. To determine the criteria according to which liquid sludge should be applied in agricultural lands (according to crop nutrient or water requirements)

NB. The names of the WWTPs used for this study are withheld for privacy reasons and are represented by letters of the alphabet.

3.2 Materials and Methods

3.2.1 Sludge sampling and preparation

Sludge samples were collected from eighteen WWTPs in Gauteng, South Africa. The sampling period spanned from mid-September to the first week of November 2015. The sludge samples included both activated and digested sludges. Twenty litres of liquid activated sludge were collected at sampling points just after final clarification. Similar amounts of digested sludge were collected at digester outlets. The samples were kept in a cold room below 4 ^{o}C until analysed.

Sludge subsamples of 500 ml were collected from the 20 L sample to determine TS. The samples were weighed, followed by liquid and solid separation. The wet solids were oven dried at 50 ^oC to determine dry mass. The decanted liquid was filtered twice. The first filtration was done by a 2 μm filter. The filters average weight was determined as an average of three separately weighed filters. After filtration, the filters were allowed to air dry and were re-weighed individually. The difference between the dried filter plus retained solids and the average clean filter weight was the mass of the retained solids. This was added to the dried solids to give TS. The second filtration was done using a 1.1 μm filter to determine the TSS of the decanted liquid. The filtered liquid was then used for anion analysis through ion chromatography and NH_4 determination by the colorimetric method.

3.2.2 Chemical analysis

Wastewater sludge pH and EC were determined on the decanted liquid after solid separation using benchtop pH and EC meters. NO_2, NO_3, P_2O_4 and sulphate were analysed by ion chromatography. NH_4 was analysed by the colorimetric method using an automated flow system (autoanalyser). Trace metals such as Fe, Zn, Mn, Cu, and Al were analysed using an ICP-OES after digestion with 7ml HNO_3 (conc. nitric acid) and 3ml $HClO_4$ (perchloric acid) at temperatures up to 170 ^oC in a 100 ml vol. flask. Total C and N was analysed using a Carlo Erba NA 1500 C/N/S analyser.

3.3 Results and Discussion

3.3.1 Physical and chemical characterization of wastewater liquid sludge

The results for the selected physical and chemical characterization of the liquid wastewater sludge collected from 18WWTPs around Gauteng are presented below.

3.3.2 Physical characterization of wastewater sludge

The physical properties measured included TS and TSS. Generally, solids determine sludge viscosity. Total solid percentage determines the volume of sludge to be handled and whether sludge should be classified or behaves as a liquid effluent, sludge or bio-solids (Al-Malack et al., 2002). Each of these physical parameters is discussed below.

Total solids

The TS content of the sludges under investigation ranged from 0.16% to 16% (Figure 3.1). Most of the candidate WWTPs, 12 of the 18 (67%), had TS contents of less than 1%, while 4 had between 1% and 3% solid contents.

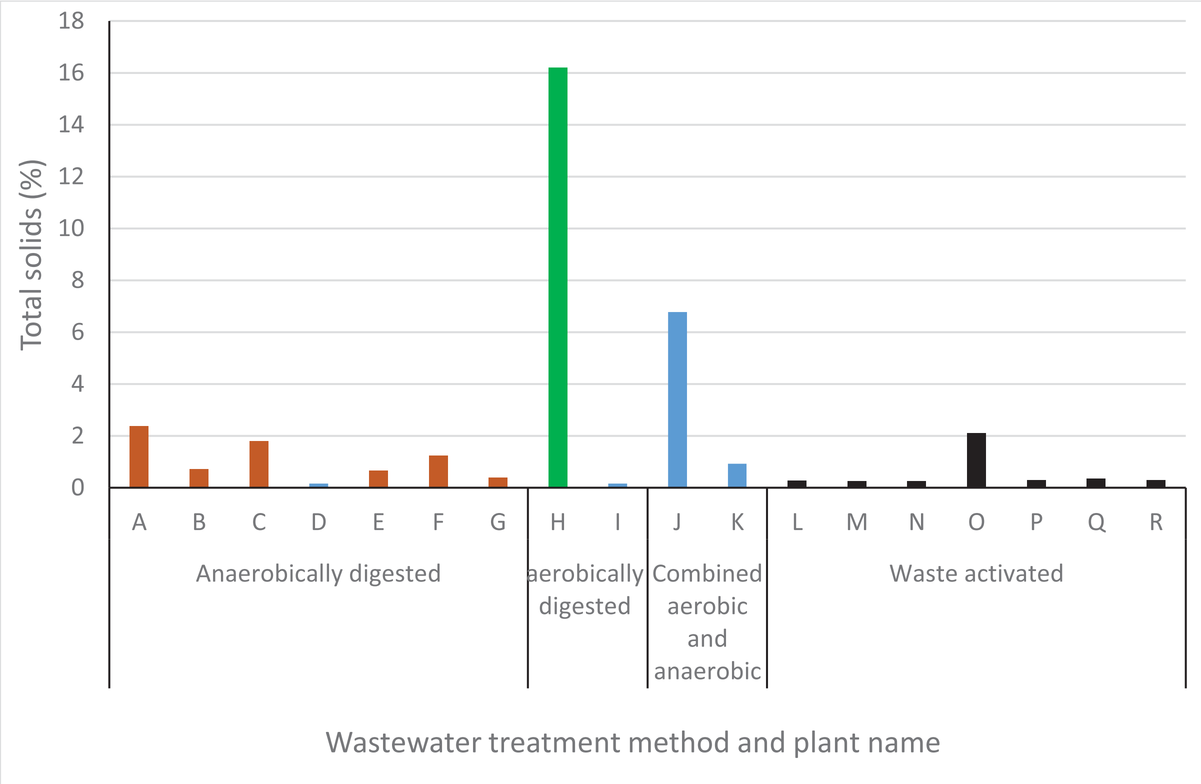


Figure 3.1 Total solid content (%) of 18 wastewater treatment plants around Gauteng, South Africa

Total suspended solids

The total suspended solids of the candidate WWTPs ranged from 0 to 558 mg L^{-1} (Figure 3.2). About half of the WWTPs had TSS concentrations below the recommended limits of 25 mg L^{-1} (National Water Act, No 36 of 1998), while the rest exceeded the recommended limits for use as a source of irrigation in agricultural lands.

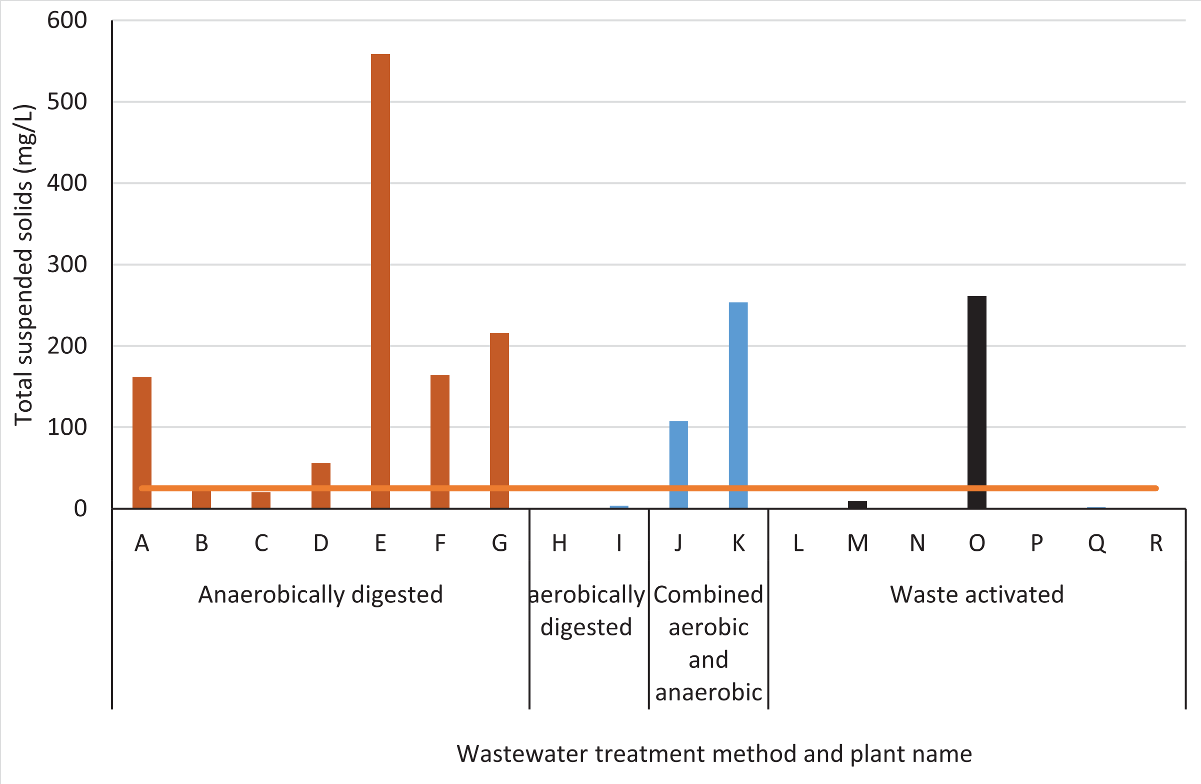


Figure 3.2 Total suspended solids of 18 wastewater treatment plants around Gauteng, South Africa

Previous studies have shown that high concentrations of suspended solids added with wastewater sludge to soils negatively affected soil hydraulic conductivity (Clanton & Slack, 1987; Vinten et al., 1983a, 1983b). This reduction is attributed mainly to the physical process of clogging the soil by the coarse fraction of suspended solids (Vinten et al., 1983b). The concentration of suspended solids in almost all of the sludges originating from anaerobically digested and combined aerobic and anaerobic digestion exceeded the allowable threshold levels, while sludge that had been aerobically digested and waste activated remained well below the threshold level, indicating its suitability for use as a source of water or nutrients through irrigation. However, the problems associated with surface sealing could be mitigated through regular cultivation practices.

3.3.3 Chemical characterization of liquid wastewater sludge

The chemical properties measured for chemical characterization of the candidate sludges including pH, and EC are presented below.

Electrical conductivity

The EC of the candidate sludges ranged from 44 mS m^{-1} to 821 mS m^{-1} (Figure 3.3). Most of the wastewater sludge from the candidate WWTPs, namely 11 of the 18 plants (61%), had EC values exceeding the recommended limit value of 200 mS m^{-1}. All of the anaerobically digested sludges, the combined aerobic and anaerobically digested sludges, and one of the aerobically digested sludge had EC values that exceeded the 200 mS m^{-1} threshold value. Six of the seven waste-activated sludges, however, had EC values lower than the threshold level.

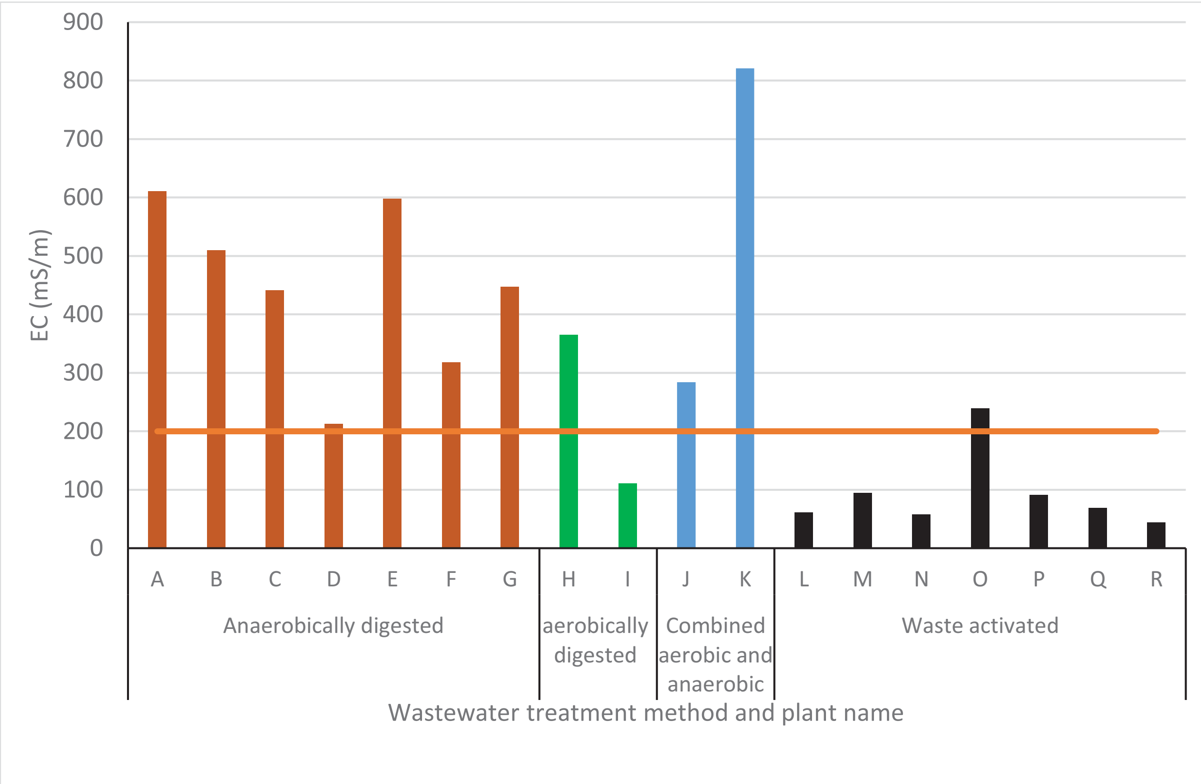


Figure 3.3 Electrical conductivity values of 18 wastewater treatment plants around Gauteng, South Africa

The EC of a solution indicates the amount of salts dissolved in it. An increase in the concentration of salts in the soil solution negatively affects crop access to soil water by increasing the osmotic strength of the soil solution. Such a decrease in the access to water by crops results in a decline in stomatal opening and gas exchange by crops, leading to a reduction in crop growth and final harvestable dry matter (Abdul Qados, 2011). Generally, the level and types of negative impacts from high soluble salts added to the soil through irrigation depend on the species of the dissolved salts. For instance, long-term application of irrigation water that is rich in sodium will cause soil sodicity, besides having a direct osmotic effect on the crop. Soil sodicity causes detrimental effects to the soil structure, stability, and the soil hydraulic properties (Levy et al., 2014). Such negative effects on the physical and chemical properties of soil would ultimately lead to a decline in agricultural productivity.

Most of the wastewater sludges investigated in this study appeared to have high salinity, which must be considered in agricultural land application. The only exceptions were sludges of activated waste origin. Unless there is enough water to conduct leaching practices, using such sludges in agricultural lands could compromise the sustainable use of the receiving soils.

Wastewater sludge pH

The pH of all the sludges under investigation was within the ranges prescribed for use in agricultural lands (pH of 5.5 To 9.5) (Figure 3.4).

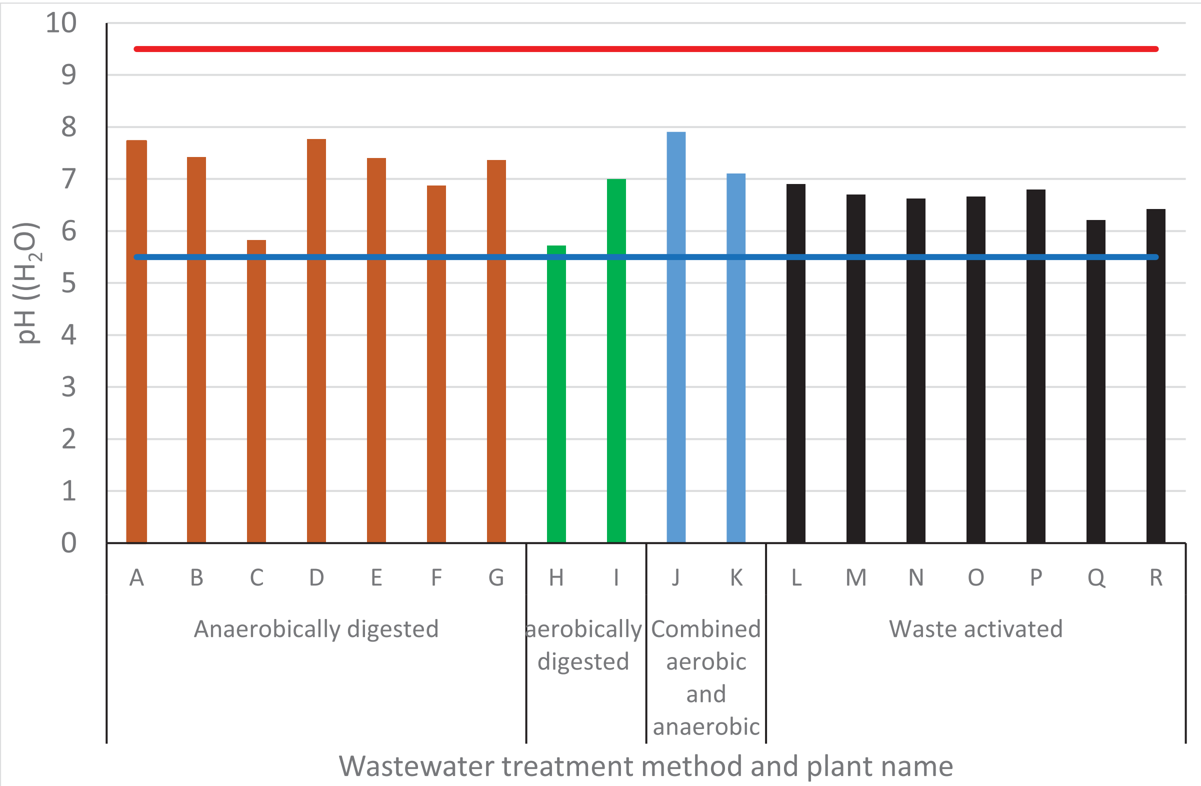


Figure 3.4 pH (H_2O) of wastewater sludge from 18 wastewater treatment plants around Gauteng, South Africa

The pH of all sludges of activated waste origin showed little variation among them (6.21–6.9) with an average value of 6.62, while sludges from other wastewater treatment origin showed relatively higher variation.

The pH of irrigation water is crucial as it affects the pH of the soil in the long term. High and low pH both cause nutritional disorders in crops because the availability of nutrients in the soil for crop uptake is influenced by soil pH. Generally, soil pH ranges of 6.0–7.5 are considered acceptable for most crops because most crop nutrients are available within these pH ranges. Under acidic soil conditions of pH(H_2O) less than 5, a deficiency of major nutrients such as P, K, Ca, Mg, and N can be experienced by plants, while at high pH(H_2O) of 8 and above, crops can experience trace element deficiency such as Mn, B and Fe. The pH of all the sludges investigated, however, was within the optimal range. In this case, pH may not be a limiting factor for agricultural use of these sludges.

Wastewater sludge trace metal concentration

The National Water Act 36 of 1998 does not provide the trace metal concentration limits of wastewater sludge used in agricultural lands. Therefore, in this study the authors used the South African Sludge Guidelines for the Utilisation and Disposal of Wastewater Sludge (Snyman & Herselman, 2006) to classify the wastewater sludges. Analytical results showed that except wastewater from plant D (high in lead (Pb): pollutant class C), wastewater from plant F (high in cadmium (Cd): pollutant class C), and wastewater from plants M and O (high in Pb: pollutant class B), all others fell in the category of pollutant class A (Table 3.1). Sludges of pollutant class A can be applied according to crop nutrient requirements without any limitation. Therefore most of the sludges were feasible for use in agricultural lands in terms of their trace metal concentration.

Table 3.1 Trace metal concentration (dry mass basis) of wastewater sludge collected from 18 wastewater treatment plants around Gauteng, South Africa

Wastewater treatment plant name (code)	Pb	Cr	Zn	Cd	As	Hg	Cu	Ni
	mg kg^{-1}							
Anaerobically Digested								
A	240.2	731.7	4.52	14.52	21.36	1.84	0.74	350.56
B	260.7	480.9	1.57	3.58	5.51	0.84	0.3	50.86
C	117.3	122.3	1.91	3.81	35.39	1.01	0.46	104.4
D	1371.3	1092	1.61	2.65	8.26	0.86	2.08	138.53
E	69.3	132.7	1.51	4.33	6.11	1.93	0.46	43.03
F	234.1	133.4	1.92	167.18	2.92	0.98	0.27	112.46
G	140.8	836.8	10.45	5.41	13.71	0.81	0.29	76.54
Aerobically Digested								
H	11.4	30.2	0.31	0.29	1.47	0.32	0.12	11.99
I	64.4	87.9	0.25	3.47	1.62	0.31	0.12	18.46
Combined aerobically and anaerobically digested								
J	38.4	419.7	1.46	7.77	3.21	0.59	0.31	232.25
K	117.4	296.3	4.28	5.27	7.77	0.75	0.31	110.72
Waste activated sludge								
L	34.4	56.2	1.37	0.55	5.07	1.74	0.14	34.07
M	806.9	505.8	0.59	0.68	8.08	4.61	0.25	38.51
N	21.7	32.6	0.68	0.56	3.5	0.37	0.09	26.14
O	634.2	709.9	0.71	0.85	24.48	2.85	0.31	162.05
P	98.6	38.8	0.56	1.02	1.92	0.49	0.1	28.42
Q	25.6	549.7	2.82	0.96	2.86	0.35	0.17	74.74
R	28.8	29.1	0.64	0.68	4.41	1.49	0.2	26.16
South African Sludge Guidelines pollutant class upper limits								
Pollutant class A	<300.0	<1200.0	<2800	<40.0	<40.0	<15.0	<1500	<420.0
Pollutant class B	300-840	1200-3000	2800-7500	40-85	40-75	15-55	1500-4300	420
Pollutant class C	>840	>3000	>7500	>85	>75	>55	>4300	>420

NB. Red highlights indicate the concentration of the given element is very high (class C) and sludge of this quality may not be used in agricultural practices

Yellow highlights indicate that the concentration of the given element is moderately high (class B) and the sludge can only be used in agriculture if the soil analysis is favourable

3.3.4 Plant macro nutrients and organic matter composition of wastewater sludge

In this section the macro nutrients that are critical to crops and plants in general for their growth and development are discussed. These elements include N, P, and K. Similarly, organic matter is included as it plays a crucial role in facilitating soil health by improving the physical, chemical and biological properties of a soil as a crop growth medium.

3.3.5 Organic matter composition of wastewater sludge

The organic matter fraction of the total solids ranged from 33% to 79% (Figure 3.5). Most of the WWTPs (17 of 18) had organic matter contents exceeding 50% of TS. The organic matter content of sludges of waste-activated origin had generally higher organic matter (>66%) than sludges that had undergone aerobic, anaerobic treatment methods and a combination of the two.

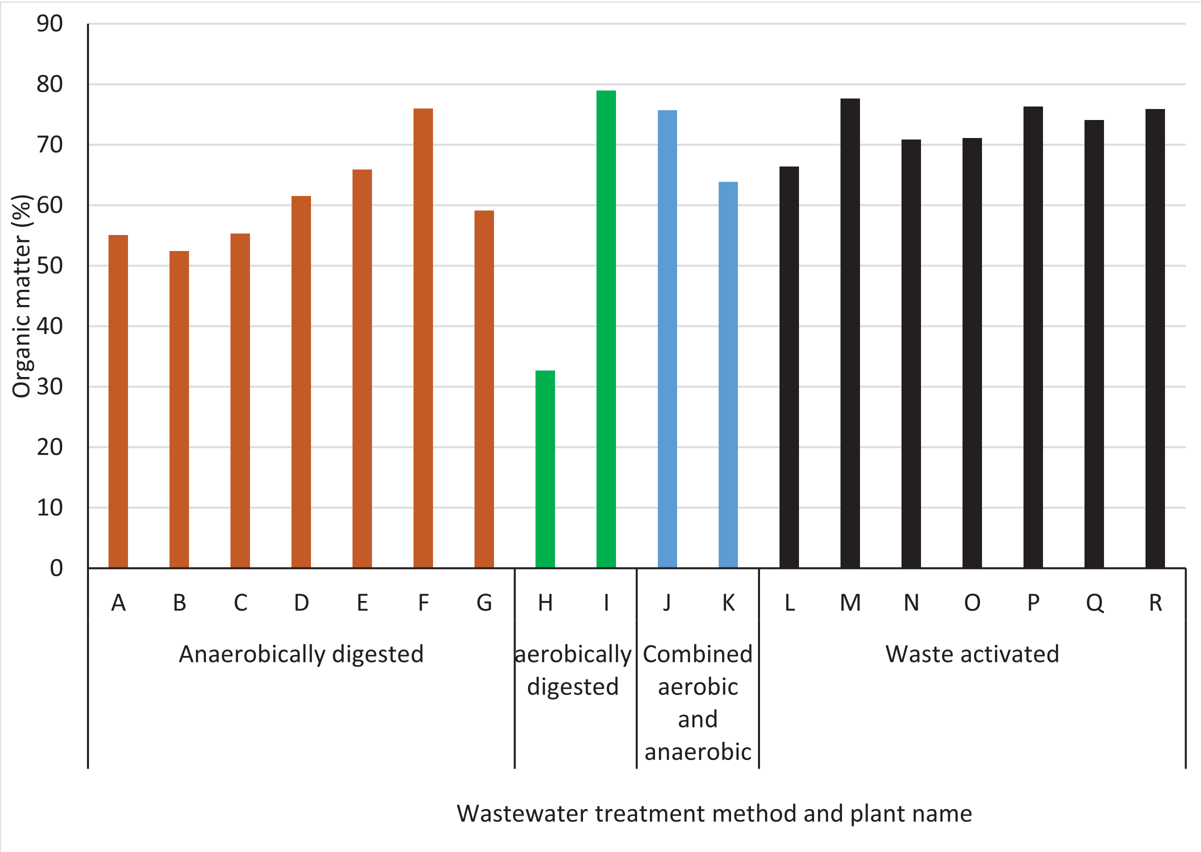


Figure 3.5 Organic matter composition of sludge from 18 wastewater treatment plants around Gauteng, South Africa

Sludge as a source of organic matter (Xu et al., 2010) is generally considered a good soil conditioner (Bell et al., 2004). The sludge from almost all WWTPs had above 50% organic matter, indicating that they could serve as good soil ameliorants. The only exception was the sludge from wastewater treatment H, which is mainly of industrial origin, and is very rich in indecomposable inorganic minerals. This was proved after oxidation at high temperature in furnace (550 ºC) ashing, which yielded an ash content of 67%.

3.3.6 Plant macro nutrients

The macro nutrients that are discussed in this section are N, K, and P. The nitrogen content of the wastewater sludge ranged from 1.5% for wastewater treatment H to 6.8% for sludge from wastewater treatment I (Figure 3.6). The N content of sludges of waste-activated origin showed little variation (SD = 0.52) among the WWTPs, with an average value of 5.87%. Similarly, sludges of combined aerobic and anaerobically digested origins had very little variation (SD = 0.14) between the treatment plants with average N contents of 4.1%.

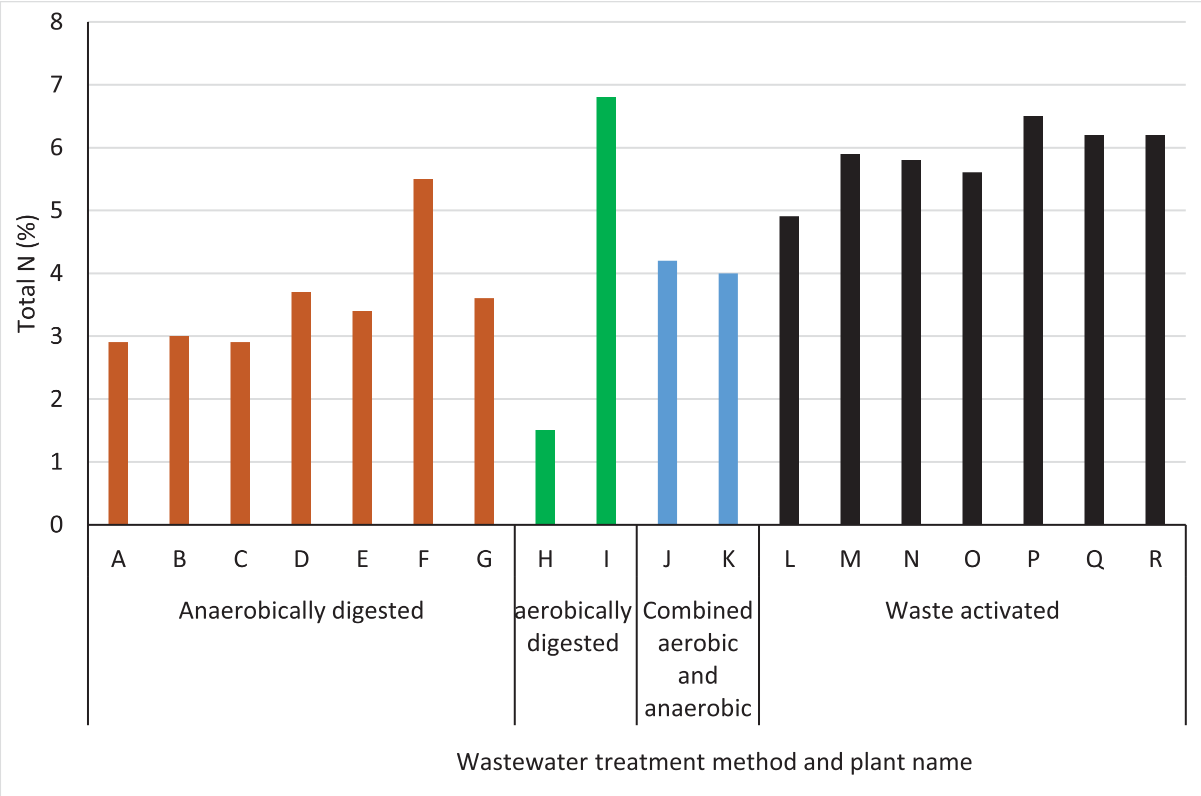


Figure 3.6 Total nitrogen content of sludge (%) from 18 wastewater treatment plants around Gauteng, South Africa

The P fraction of the solids of the wastewater sludges varied from 1.26% to 3% (Figure 3.7). Most of the WWTPs had P contents between 1.5% and 2.5%. The N : P ratio of crops is (8 : 1 to 10 : 1), so the N : P ratio of sludges is high (1 : 1 to 4 : 1). This implies that if wastewater is applied based on crop N requirements, there will be excess P accumulation in the soil in the long run.

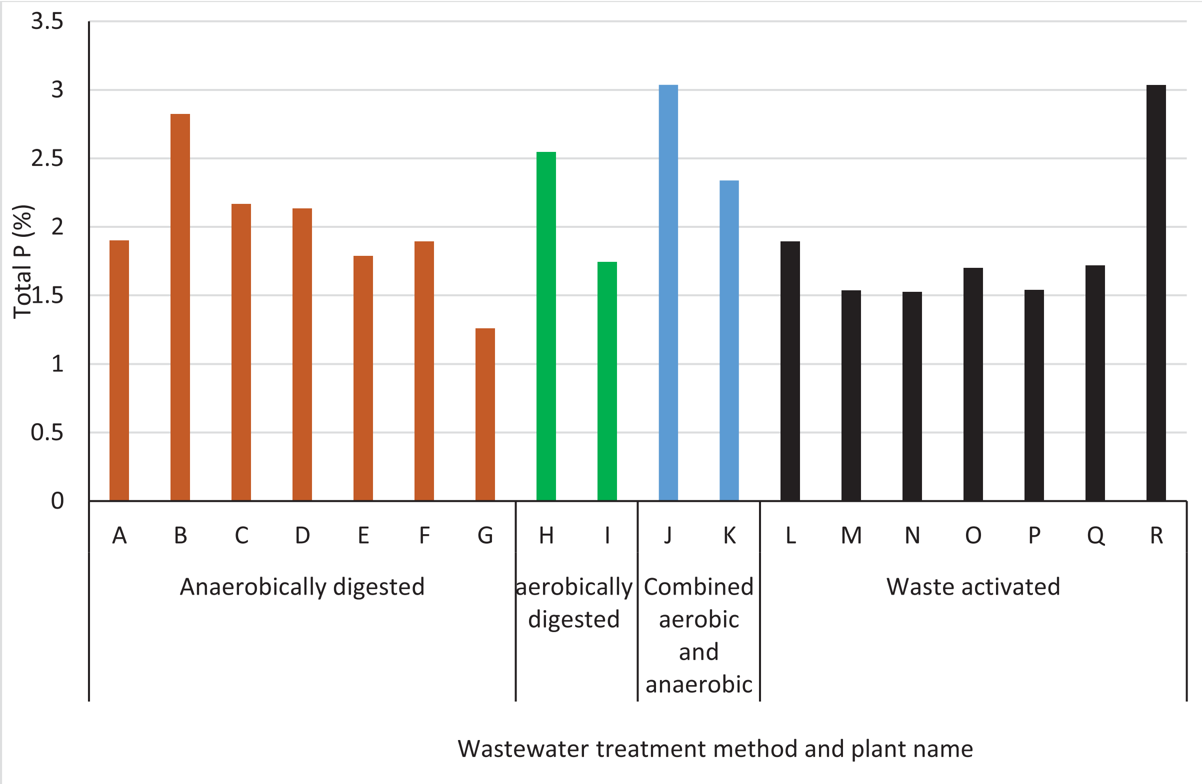


Figure 3.7 Total phosphorus content of sludge (%) from 18 wastewater treatment plants around Gauteng, South Africa

Generally the K content of the candidate wastewater sludges was very low (0.15% to 1.5%) (Figure 3.8) compared with both N and P. This is despite high crop K requirements, which are similar to N. Sludges of anaerobically digested origin had relatively lower K content. In contrast, sludge of waste-activated origin had relatively higher K than others.

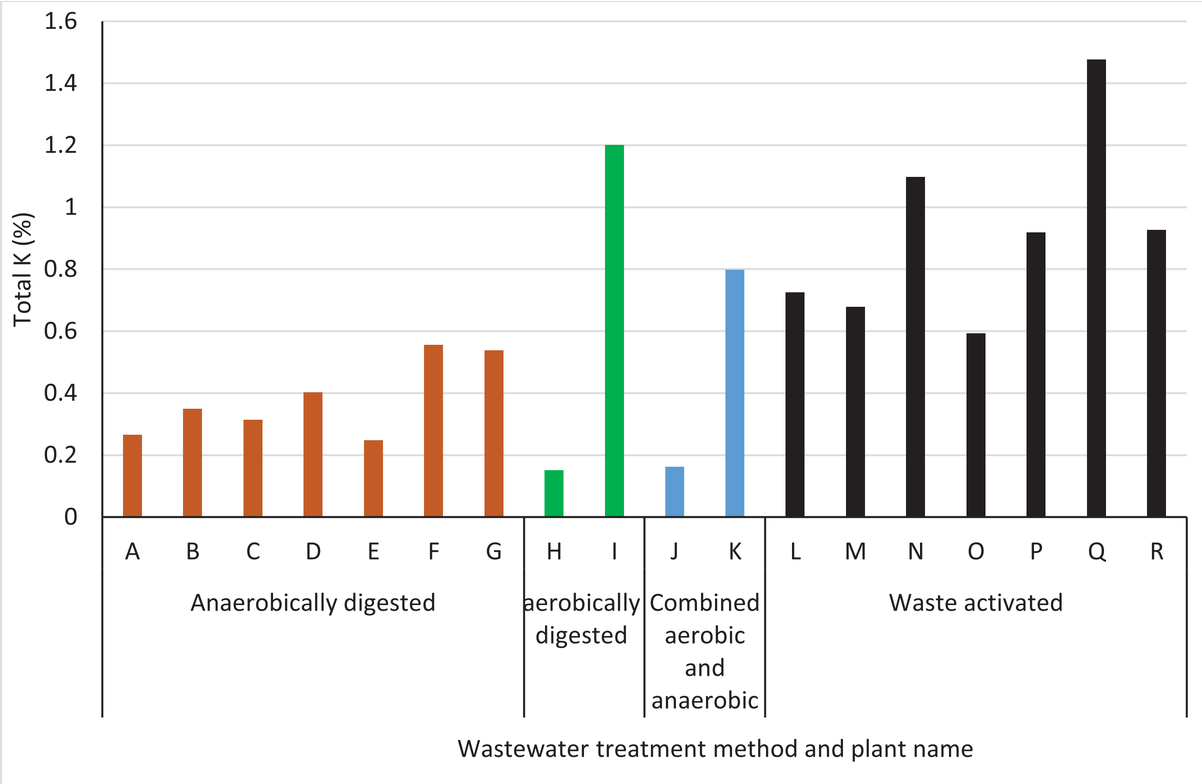


Figure 3.8 Total potassium content of sludge (%) from 18 wastewater treatment plants around Gauteng, South Africa

The critical factor that must be considered when using wastewater sludge for irrigation is the amount of nutrients added with the water. If the amount of nutrient added with the sludge exceeds the crop nutrient requirements, it would compromise environmental health and sustainability because of excess nutrient accumulation, leaching, and losses to surface water bodies with runoff. To get a broader picture of the environmental implications of applying wastewater sludge according to crop water requirements, previous estimates of supplemental irrigation water requirements for all South African provinces are used from the WRC report by Schulze (1997) (Table 3.2). Assuming that crops are planted throughout the year (summer (maize) and winter (barley)), the total nutrient requirements for both crops under irrigation would be about 500 kg ha^{-1} N, 50 kg P, and 450 kg ha^{-1} K. If the sludge is applied according to crop water requirements, the amount of N, P, and K added is 23, 126, and four times higher than the crop requirements.

Table 3.2 Provincial gross mean annual irrigation requirements (mm) and nitrogen, phosphorus, potassium addition when wastewater sludge is applied according to crop water requirements

Province	Gross mean annual irrigation requirement (mm)	Gross addition of nutrients per hectare land (kg)		
		N	P	K
Limpopo	1616	14835	6561	2036
Mpumalanga	1187	10897	4819	1496
North West	1947	17873	7905	2453
Northern Cape	2225	20426	9034	2804
Gauteng	1418	13017	5757	1787
Free State	1565	14367	6354	1972
KwaZulu-Natal	1008	9253	4092	1270
Eastern Cape	1337	12274	5428	1685
Western Cape	1711	15707	6947	2156

NB. Computation of N, P, and K added with irrigation assumed 2% solid content, N sludge content of 4.59%, P sludge content of 2.03%, and sludge K content of 0.63%

Source: Schulze (1997)

From this, we can deduce that wastewater sludge should not be applied according to crop water requirements, but according to crop nutrient requirements. This computation assumes the mean N value of 4.59%, P value of 2.03%, and K values of 0.63%, which is the mean value of all WWTPs under investigation. Similarly, the computation assumes the mean wastewater sludge solid content of 2%.

3.4 Conclusion

The pH of all wastewater sludges was within the acceptable ranges for use in agricultural lands ($pH_{(H2O)}$ of 5.5–9.5). Similarly, the trace metal contents of most sludges were below the threshold values for use in agricultural lands. The only exceptions were two anaerobically digested (pollutant class C) and another two activated sludges (pollutant class B). All of the anaerobically digested sludges and the combined anaerobically-aerobically digested sludges had exceptionally high salinity (EC), exceeding the threshold value of 200 mS m^{-1}, making them unfeasible for use in agricultural lands. Besides, these sludges had TSS values exceeding the threshold value of 25 mg L^{-1}. In contrast, almost all of the waste-activated sludges had EC and TSS levels of less than the threshold values. The problems associated with the effects of high TSS on soil hydraulic conductivity, however, could be minimized by scheduled intervention through cultivation. From the preliminary investigation it has been clearly indicated that wastewater sludge application according to crop water requirements could lead to excess nutrient application above crop nutrient demands, which would compromise sustainability. Henceforth, wastewater liquid sludge should be applied according to crop nutrient requirements.

CHAPTER 4 - THE EFFECT OF WASTEWATER SLUDGE TREATMENT PROCESSES AND DEWATERING ON THE FERTILIZER VALUE OF SLUDGE

4.1 Introduction

Sludge composition is influenced mainly by the source. Nevertheless, wastewater treatment and post treatment dewatering/drying techniques can significantly alter organic matter composition (Banegas et al., 2007; Fernandez et al., 2007) and C and N mineralization of sludge (González-Ubierna et al., 2013; Mattana et al., 2014), thus influencing the fertilizer value of sludge. Despite this, there is little quantitative information available on the effects of wastewater treatment methods, sludge drying time and depth on drying beds on the fertilizer value and organic matter content and composition of sludge.

The objectives of this section of the report are:

a. To determine the effect of wastewater treatment (aerobic vs anaerobic digestion) on:
    i. Sludge organic matter content and composition
    ii. Sludge N content and composition
b. To quantify the effect of sludge drying time and depth (thickness layer) on drying beds on:
    i. Sludge organic matter content and composition
    ii. Sludge nitrogen content and composition
c. To quantify the effects of wastewater treatment methods and drying techniques on the N fertilizer value of sludge

4.2 Materials and Methods

4.2.1 The effect of wastewater treatment processes on the nitrogen content, organic matter content and composition of sludge

Sludge sampling and preparation

Two wastewater sludges of the same origin, but from different wastewater treatment methods (aerobic digestion (AeD) vs. anaerobic digestion (AnBp)) were collected from a plant in the Johannesburg area. The anaerobically digested sludge was produced through the conventional mesophilic anaerobic process. The AnBp sludge was further treated with polymer to improve compressibility and was dewatered using a belt press. The samples for the study were collected when they were ready, after the belt press. The AeD sludge was collected from the clarifiers as liquid sludge with 97% moisture content and was allowed to dry on drying beds, from which the samples were collected for the study. Both sludges were allowed to dry at 40 ºC over 24 hours to bring them to similar moisture content before application to the soil and analysis thereafter.

The fertilizer value for the candidate sludge used in this study was estimated based on a laboratory incubation experiment conducted at the University of Pretoria over 65 days. The incubation was done using sandy loam Hutton soil, which was collected from ERWAT, near Johannesburg. A parallel study on carbon dioxide evolution from similarly incubated soils was conducted to determine the carbon decomposition rate of the sludge.

Soil-sludge mix incubation for N mineralization and P release

The incubation study was conducted in 1163 cm^3 volume airtight containers at a constant temperature room at 25 ± 1 ºC. The amount of 100 g soil was mixed with the candidate sludge to meet the current South African maximum sludge application limit of 10 t ha^{-1} (Snyman & Herselman, 2006a). The soil moisture was maintained at field capacity by adding distilled water based on mass difference.

The carbon decomposition study was conducted in airtight 2000 cm^3 volume desiccators, where 100 g soil was also mixed with the candidate sludge to meet the current South African upper sludge application limit of 10 t ha^{-1}. A zero control treatment, which did not receive sludge, was included during the incubation study. The soil moisture was maintained at field capacity by adding distilled water based on mass difference.

The sludge and soil were mixed thoroughly at the commencement of the study. Both CO_2 evolution and inorganic-N release analyses were conducted at similar time intervals during the incubation study. The analyses intervals were set at days (d) 0, 1, 3, 7, 15, 30, 45, and 65. Both N mineralization and CO_2 evolution experiments were aerated during sampling intervals for the first seven days of incubation. After day 7, the N-mineralization experiment was opened and weighed once a week and at sampling time to replenish water to field capacity, while the CO_2 evolution set up was opened only at sampling time.

The cumulative amount of net N mineralized from sludge was calculated using Equation 1.

$$N_{\text{min(net)}} = N_{\text{(sludge amended soil)}} - [N_{\text{(control)}} - N_{\text{(initial in sludge)}}] \qquad \text{Equation 1}$$

Where $N_{\text{min(net)}}$ is net N mineralization at each sampling time.

The net-N mineralized was expressed in terms of net-N mineralized per unit organic C applied (g kg^{-1}) and net-N mineralized per unit organic-N applied (g kg^{-1}) using Equations 2 and 3, respectively.

$$N_{\min(net)}\ (per\ organic\ C\ applied) = \frac{Net\ N\ mineralized(g) per\ kg\ soil}{Organic\ C\ added\ from\ sludge\ per\ kg\ soil}$$

Equation 2

$$N_{\min(net)}\ (per\ organic\ N\ applied) = \frac{Net\ N\ mineralized\ (g) per\ kg\ soil}{Organic\ N\ added\ from\ sludge\ per\ kg\ soil}$$

Equation 3

4.2.2 Effect of sludge drying depth on drying beds on the nitrogen content, organic matter content and composition of wastewater sludge

Field experiment

A field trial was conducted at Vlakplaats plant, ERWAT, Gauteng to determine the effect of sludge drying depth above drying beds on sludge N content, C content and organic matter composition. Five drying depth treatments (5, 10, 15, 20, and 25 cm) with three replications were laid out on the field in a completely randomized design. The experiment was conducted for two seasons (autumn and winter) to represent two extreme climatic conditions (warm and cold). The drying period per season was determined by the maximum number of days (drying time) required to reach 90–95% solid. The sludge was poured to the desired drying depth into plastic containers (0.62 m long * 0.62 m wide * 0.38 m deep), which were dug into the soil to mimic the concrete drying beds (Figure 4.1). The bases of the containers were perforated to allow drainage of supernatant. The bottom of the containers were lined with geo-fabric to cover the perforations to allow leaching of solution and retaining solids in the container. The leached supernatant was captured by another container buried below the drying beds for analysis.

Figure 4.1 Plastic containers used to mimic concrete drying beds

Selected chemical properties of sludge used

The wastewater sewage sludge used for this study was obtained from Vlakplaats plant, ERWAT. The sludge was anaerobically digested and can be categorized as class A1a. Selected chemical properties of this sludge are presented in Table 4.1.

Table 4.1 Selected properties of the anaerobically digested liquid sludge used for this study during the autumn and winter seasons

Season	pH	EC	Water content	Total N	Total C	NO_3	NH_4
	H_2O	dS m^{-1}		%		mg kg^{-1}	
Autumn	6.76	3.31	96.84	0.13	57.63	1065.00	605.00
Winter	6.83	3.33	96.40	0.14	57.90	524.00	342.00

Sampling during the drying period

During the sludge drying period, samples were collected regularly from the 'drying beds' for targeted physical and chemical properties analysis. Samples were collected weekly for the first three weeks, followed by every second week for the rest of the study period, until the percentage solid concentration reached 90–95%. The samples were immediately placed in a plastic container, sealed, placed in a mobile refrigerator, and transported to the University of Pretoria. The temperature inside the mobile refrigerator was maintained at 4 ºC. The samples were transferred from the mobile refrigerator to a cold room and remained there until the samples were analysed.

The moisture content of samples at every sampling interval (%) was determined immediately on arrival at the university. Samples of 20 g mass from each treatment and replication were weighed immediately on arrival (wet mass). The 20 g samples were then dried in an oven at a temperature of 40 °C to a constant mass (dry mass). The moisture content of the sludge for each sampling time was then estimated using Equation 4.

$$\text{Moisture (\%)} = \frac{[weight\ undried\ sample + container] - [weight\ dried\ sample + container]}{[weight\ undried\ sample + container] - weight\ empty\ container}\ x\ 100$$

Equation 4

4.2.3 Analytical methods

General analytical methods

Sludge samples were ground to pass through a 150 μm screen and analysed for total C and N using a Carlo Erba NA1500 C/N analyser (Carlo Erba Strumentazione, Milan, Italy). Sludge samples were extracted in 1:5 ratio (sludge: solution) using 1 M KCl and were tested for ammonium and nitrate with the Lachat autoanalyser (Lachat Quick Chem Systems, Milwaukee, WI, USA).

Both soil and sludge samples were analysed for pH and CEC following the standard procedures presented in Non-affiliated Soil Analyses Work Committee (1990). Total P, Ca, Mg, P, K, Na, Fe, Zn, Mn, Cu, Al, and B 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 analysed using the Bray-1 extraction method.

Organic matter fractionation

Sludge organic matter fractionation studies were conducted using the Van Soest method as follows. Freeze-dried samples were extracted with petroleum ether using the Soxtec^{TM} 2043 lipid extraction system to determine the fraction of lipids. The lipids extraction was followed by estimation of the neutral detergent fibre (NDF) as described by Goering and Van Soest (1970). The NDF solution was made up of sodium lauryl sulphate, disodium ethylenediaminetetraacetate (EDTA) dehydrate, sodium borate decahydrate, disodium hydrogen phosphate anhydrous, 2-ethoxyethanol (ethylene glycol monoethyl ether) purified grade and distilled water. A freeze-dried sample (1 g) was placed in a 50 ml crucible and boiled for 1 hour with 100 ml NDF solution at 100 ^{o}C. After boiling, the solvent was filtered through the crucible and the residues that remained in it were oven dried at 60 ^{o}C and not ashed. Extraction of NDF was conducted using the Fibertec^{TM} 2010 Auto Fiber Analysis System.

The NDF extraction was followed by acid detergent fibre (ADF) extraction as described by Goering and Van Soest (1970). The ADF solution was made up of cetyltrimethyl ammonium bromide (CTAB) and 1 N H_2SO_4. One gram of freeze-dried sample, milled to pass a 1-mm sieve, was placed in 50 ml crucible and boiled for 1 hour with 100 ml ADF solution at 100 ^{o}C. After boiling, the solvent was filtered through the crucible and the residues were oven dried at 60 ^{o}C and not ashed. Extraction of ADF was conducted using the Fibertec^{TM} 2010 Auto Fiber Analysis System.

Finally, acid detergent lignin (ADL) was determined as reported by Goering and Van Soest (1970). This procedure is a two-step process. Initially the dried residue from the ADF extraction was treated with 72% sulphuric acid for 3 hours at room temperature. The 72% sulphuric acid digested sample was filtered through the crucible, and the residues that remained in the crucible were oven dried at 60 ^{o}C and weighed before ashing. This is followed by ashing of the dried residue. Similar to the NDF and ADF, extraction of ADL was conducted using the Fibertec^{TM} 2010 Auto Fiber Analysis System.

Fractions of the neutral detergent soluble fraction (Equation 5), hemicellulose (Equation 6), cellulose (Equation 7) and lignified fraction (Equation 8) were calculated as follows:

Soluble compounds fraction (SOL) = original mass – NDF- lipids fraction Equation 5

Hemicellulose (Hem) = NDF - ADF Equation 6

Cellulose (Cel) = ADF - 72 % H_2SO_4 Equation 7

Lignified fraction (Lign) = 72 % H_2SO_4 - Ash Equation 8

4.3 Results and Discussion

4.3.1 Effect of wastewater treatment and dewatering techniques on the nitrogen, phosphorus, organic matter content and composition of sludge

Despite having the same wastewater source, the aerobically digested sludge dried in beds to a moisture content of 20% had 14% higher total N than the anaerobically digested belt-pressed sludge (Figure 4.1). The inorganic-N fraction accounted for less than 1% of the total N.

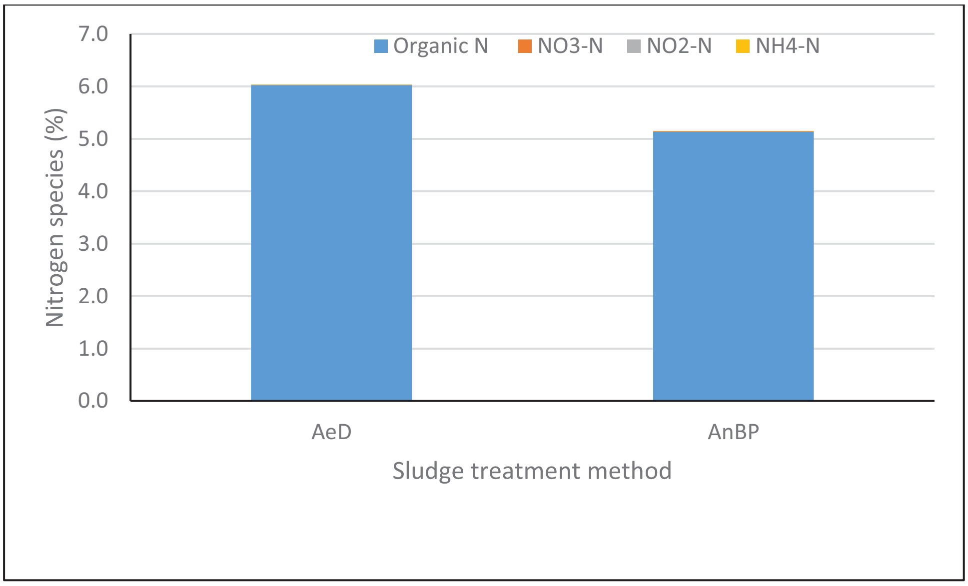


Figure 4.2 Total nitrogen and inorganic-N (NH_4, NO_3-N, and NO_2-N) of wastewater sludge of the same origin but treated and dried by either anaerobic digested polymerized and belt-press dried (AnBp) and aerobically digested paddy dried (AeD)

Generally, the N content of the aerobically digested sludge (6.04%) from this study was within the ranges reported by Rigby et al. (2016) for sludges that underwent similar treatment processes (4.8–8.5% liquid; 4.0–6.2% for dewatered and dried in beds). Similarly, the N content of the anaerobically digested belt-pressed sludge (5.15%) was within the ranges reported for sludges that underwent similar processes (4.3–15% liquid; and 2.2–6.3% for dewatered, belt-pressed and drying beds), as reported by Rigby et al. (2016). The inorganic-N fractions of both aerobically digested and anaerobically digested belt-pressed municipal sludges (<0.02% of the total N) were very low compared with the values reported for municipal sludges that underwent similar sludge treatment processes, as reported by Rigby et al. (2016) (>0.4%). Thus a larger proportion (>99%) by mass of the total N was organic and has to be mineralized before it can be used by plants. The rate at which this organic-N is released per kg of sludge or organic-N applied, however, is influenced by the organic matter composition of the sludge.

Similar to total N, the organic matter content of the aerobically digested sludge (50%) was slightly higher than that of the anaerobically digested, belt-pressed sludge (46%) (Figure 4.2a). A larger proportion of the organic matter was dominated by soluble compounds followed by hemicellulose (Figure 4.2b), while the lipids accounted for about 1% of the total organic matter for both sludge types.

The soluble fraction accounted for more than 70% of the total sludge organic matter for both sludge types, which is in agreement with previous studies (Parnaudeau et al., 2004; Zhao et al., 2011; Tesfamariam et al. (in press)). The fraction of soluble compounds for anaerobically digested belt-pressed sludge (74.5%) was similar to that of aerobically digested sludge of the same origin (72.3%). Hence, the treatment and dewatering techniques had little effect on the organic matter composition of the sludge. Nonetheless, it had a slight effect on the total organic matter content of the sludge.

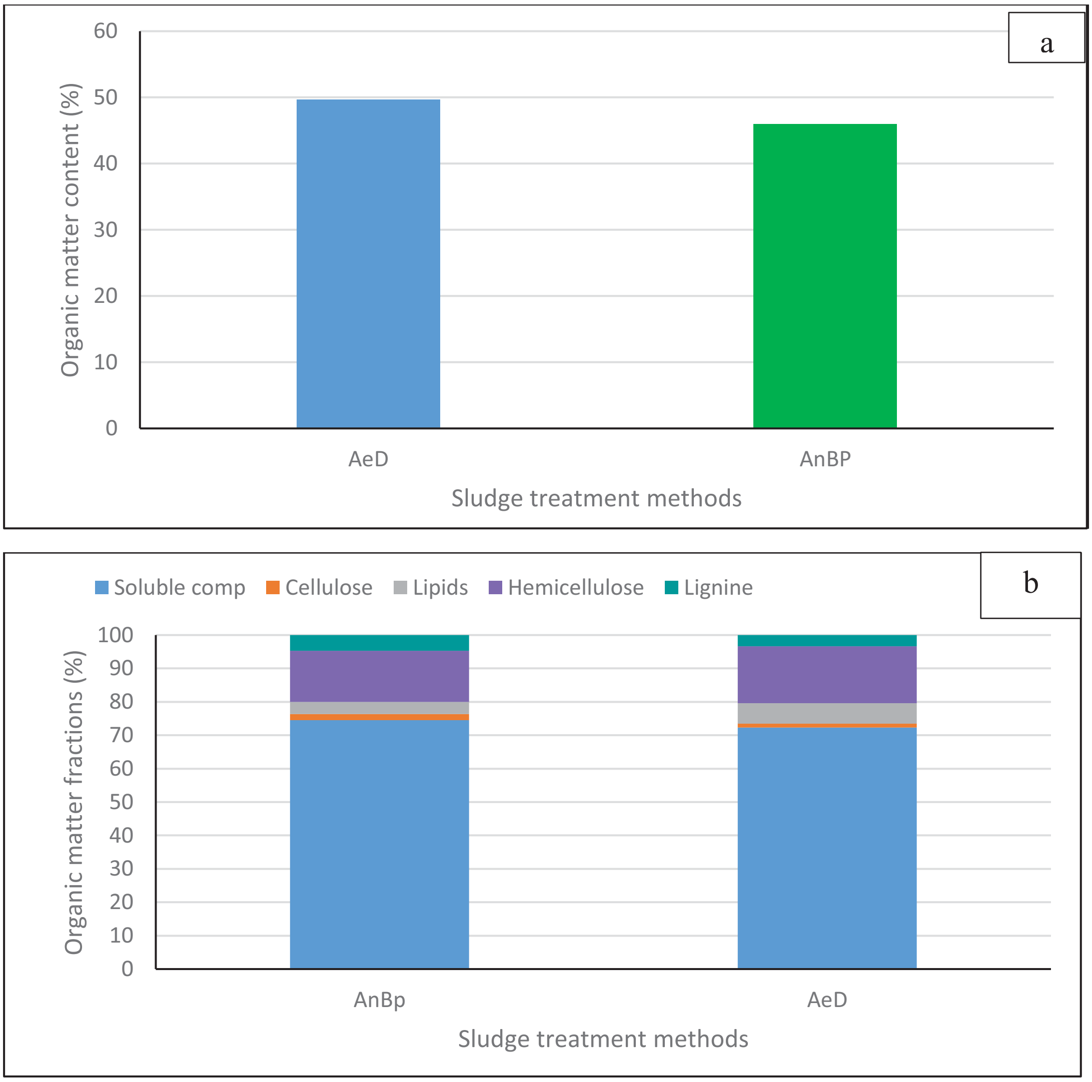


Figure 4.3 Organic matter content (a) and organic matter composition fractions (b) of wastewater sludge of the same origin but treated and dried by either anaerobic digested polymerized and belt-press dried (AnBp) and aerobically digested paddy dried (AeD)

Unlike the total N and organic matter content, the total P (Figure 4.4a) and the plant available P (Bray-1P) (Figure 4.4b) of the aerobically digested sludge were lower than those of the anaerobically digested belt-pressed sludge. The plant available fraction (Bray-1P) accounted for less than 1% of the total P.

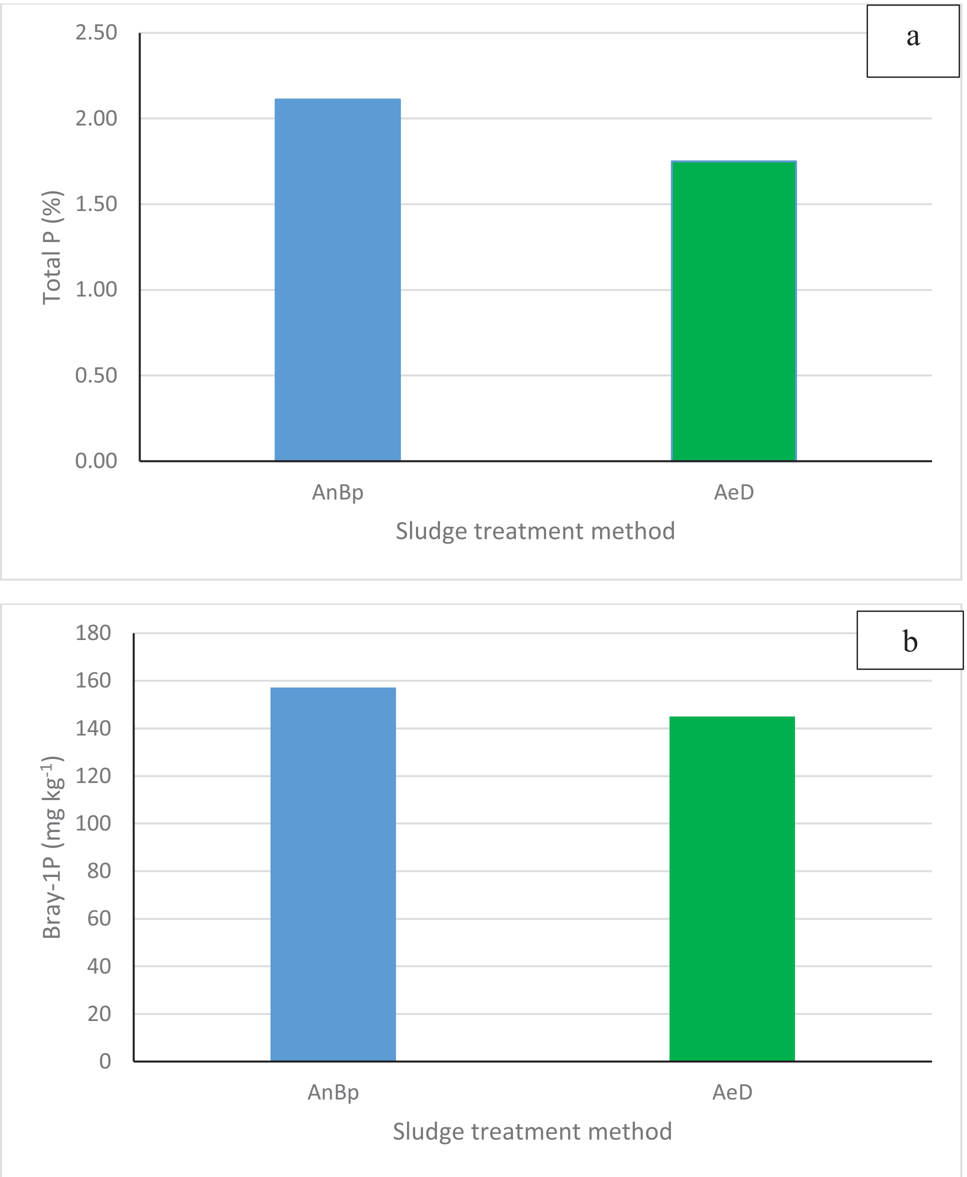


Figure 4.4 Total phosphorus content (%) (a) and plant available (Bray-1P) (mg kg^{-1}) (b) of wastewater sludge of the same origin but different treatment and dewatering techniques (anaerobic digested polymerized and belt-press dried (AnBp) versus aerobically digested paddy dried (AeD))

Generally, the P content of the candidate sludges was within the ranges of 1.5% to 3.5% (dry weight basis) reported by Sullivan et al. (2007). Despite having the same origin, the AnBp had slightly higher total and plant available P. This is probably attributed to the adsorption of some of the P ions by the polymers added to the AnBp, compared with the AeD sludge, which might have leached some of the ions with the water during the drying process. Such leaching losses would have environmental implications if continued.

4.3.2 Effect of sludge drying depth on drying beds on the nitrogen content, organic matter content and composition of wastewater sludge

Nitrogen content of wastewater sludge as affected by sludge drying depth on drying beds

The total N content of wastewater sludge was highest for the 10-cm drying depth both during autumn (3.53%) and winter (3.72%) seasons (Figure 4.5). In contrast, the lowest total N content was recorded for the sludge drying depth of 25 cm during both autumn (2.62%) and winter (2.65%). The inorganic fraction of the total N accounted for 8% to 10% of the total N.

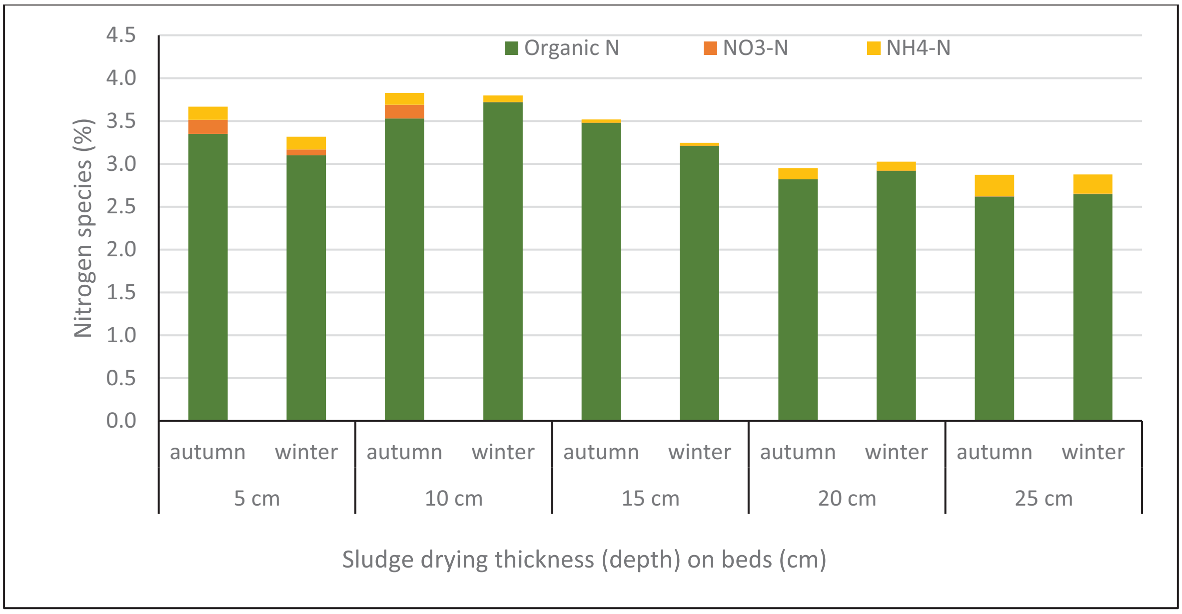
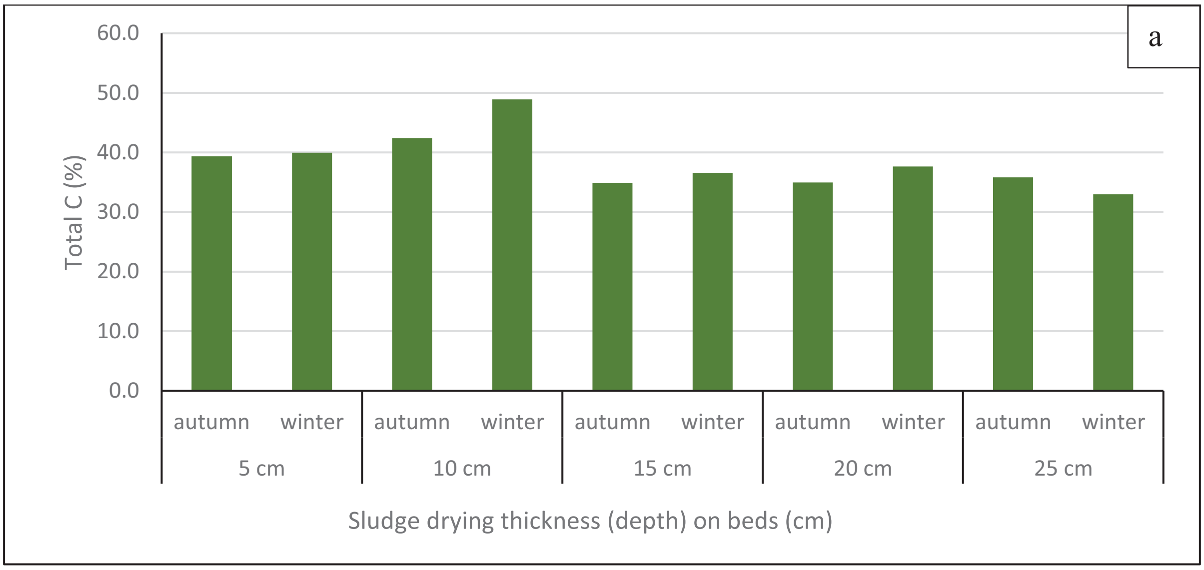


Figure 4.5 Nitrogen content of anaerobically digested wastewater sludge as affected by sludge drying depth (thickness) on drying beds

Total N was highest for the 10 cm drying depth, followed by 15 cm and 5 cm drying depths, while thicker drying depths of 20 and 25 cm resulted in lower total N content by the end of the drying period. These findings agree with those of Malobane (2014), who reported higher total N content from shallow drying depths. The lower total N content recorded at 20 and 25 cm depths probably has two reasons: a) enhanced NH_3 volatilization; and b) denitrification losses from the middle to lower sections of the drying beds, which remained quite wet for longer during the drying process.

Organic matter content and composition of wastewater sludge as affected by sludge drying depth on drying beds

At 10% moisture level, sludge dried at 10 cm drying depth had the highest total C (42% in autumn and 49% in winter) (Figure 4.6a), while the lowest total C was recorded for the 15, 20 and 25 cm drying depths (35%) in autumn and for the 25 cm drying depth in winter (33%). A larger proportion of the sludge organic matter was in the form of soluble organic fraction (75%-80%), followed by lipids (10%-15%), and lignin (6%-8%) (Figure 4.6b). Hemicellulose and cellulose together contributed to less than 4% of the total organic matter composition at all drying depths.



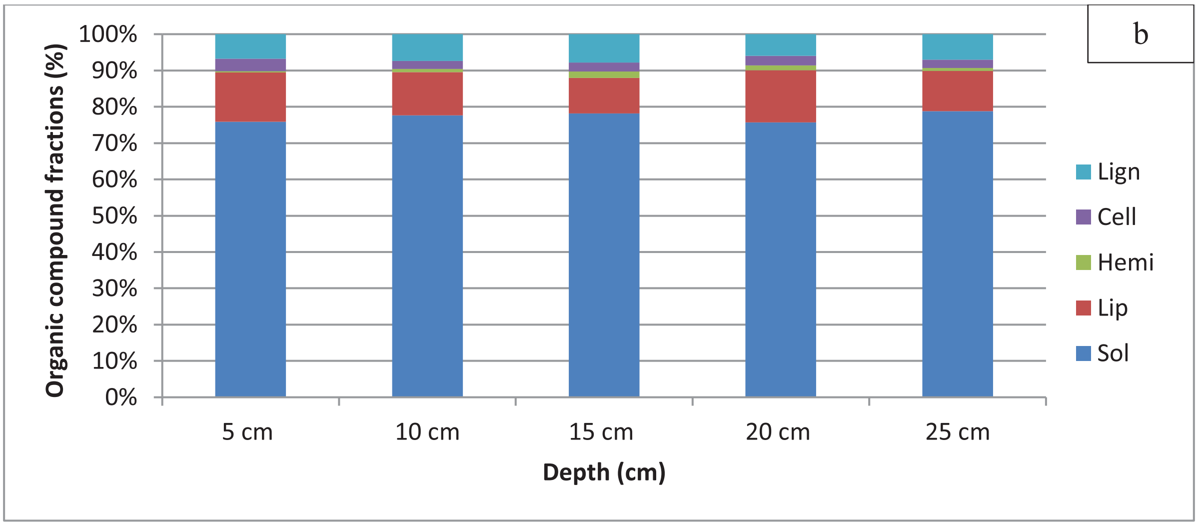


Figure 4.6 Total carbon content (a) and organic matter composition (b) of anaerobically digested wastewater sludge as affected by sludge drying depth on drying beds

Sol: soluble organic compounds; Lip: lipids; Hemi: hemicellulose; Cell: cellulose; Lign: lignified fraction

Generally the total carbon (oven dry mass basis) of the dried sludge was 50% lower than the initial liquid sludge. This was attributed mainly to the loss of carbon in the form of CO_2 through organic matter decomposition during the drying process. It was also apparent that sludge total C content at 10% moisture level was significantly higher for the 5 and 10 cm drying depths than the 15, 20, and 25 cm drying depths. This is in agreement with previous studies that reported a decrease in C content as time progressed during sludge composting (Nayak et al., 2013). Microbes use C as the source of energy during organic matter breakdown (Leslie & Lim, 1980), thus leading to the decline of C from the substrate. In general, the organic matter composition of sludge did not show noticeable differences between drying depths, though there were some slight differences. Soluble organic compounds, which were dominant, play a key role in N mineralization and C decomposition of organic matter (Alexandra & José, 2005). The reported higher proportion of soluble compounds is in agreement with previous findings by Parnaudeau et al. (2004); Mottet et al. (2010); Zhoa et al. (2011); and Malobane (2014).

4.3.3 Effect of wastewater treatment and drying techniques on the nitrogen fertilizer value of sludge

There was huge variation in the organic-N content of sludges from 1.93% for anaerobically digested sludge (dried in concrete beds, collected from hips) to 4.93% for activated sludge dried in concrete beds in thicknesses of less than 10 cm (Table 4.2). Similarly the N mineralization varied between 22% for anaerobically digested sludge and 59% for aerobically digested sludge. Generally, the data show that N mineralization increased with an increase in total organic-N. Such a general trend of a rise in N mineralization with an increase in the N content of a sludge implies an improvement in the fertilizer value of a sludge. Therefore, WWTPs and dewatering techniques that enhance the N content of a sludge improve the N fertilizer value of the sludge.

Table 4.2 Nitrogen mineralization studies of anaerobically digested sludge under controlled temperature and moisture conditions

Sludge origin	Study site	Treatment process	Drying technique	Additional treatment	Solids (%)	Incubation period (days)	Organic C (%)	Organic N (%)	Mineralization (%)	Reference
Vlakplaats, Gauteng, SA	University of Pretoria	Anaerobic digestion (probably collected from hips)	Drying bed		50	56	11.6	1.93	26.2	Nobela (2011)
Vlakplaats, Gauteng, SA	University of Pretoria	Anaerobic digestion	Drying bed		95	100	24.8	2.78	31.3	Malobane (2014)
Fort Worth, TX, USA	Arkansas, USA	Anaerobic digestion	Belt filter press	Lime stabilized	24	75	23	2.88	28	Gilmour et al. (2003)
Springdale, AR, USA	Arkansas, USA	Anaerobic digestion	Belt filter press		12	75	32	5.36	39	Gilmour et al. (2003)
Trinity River Authority, TX, USA	Arkansas, USA	Primary-secondary anaerobic digestion	Belt filter press	Lime stabilized	45	75	27.1	2.4	22	Gilmour et al. (2003)
Baltimore, MD, USA	Arkansas, USA	Anaerobic digestion	Heat dried		97	75	30.5	4.24	30	Gilmour et al. (2003)
Dewitt, MI, USA	Michigan, USA	Rotating biological, anaerobic digestion			23	75	29	3.72	34	Gilmour et al. (2003)
Baltimore, MD, USA	Michigan, USA	Anaerobic digestion	Heat dried		97	75	30.5	4.24	22	Gilmour et al. (2003)
Mason, MI, USA	Michigan, USA	Primary-secondary anaerobic digestion			45	75	27.6	3.58	28	Gilmour et al. (2003)
Tacoma, WA, USA	Tacoma, WA, USA	Aerobic-anaerobic thermophilic digestion	Belt filter press		25	75	32.5	3.73	33	Gilmour et al. (2003)
Chelan, WA, USA	Tacoma, WA, USA	Anaerobic digestion	Drying bed		96	75	31.1	4.5	31	Gilmour et al. (2003)
Ellensburg, WA, USA	Tacoma, WA, USA	Anaerobic digestion	Drying bed		94	75	29	3.84	45	Gilmour et al. (2003)
Rock Creek, OR, USA	Tacoma, WA, USA	Anaerobic digestion	Belt filter press		17	75	27.1	3.82	40	Gilmour et al. (2003)
Stayton, OR, USA	Tacoma, WA, USA	Aerobic digestion	Belt filter press	Lime stabilized	42	75	25.7	3.88	59	Gilmour et al. (2003)
Ekurhuleni, Gauteng, SA	University of Pretoria	Activated sludge	Drying bed (<10 cm drying depth)		95	100	29.7	4.93	47.5	Malobane (2014)
UK	University of Pretoria	Thermally hydrolysed sludge	Belt filter press		50	100	18.1	2.83	36.6	Malobane (2014)

4.4 Conclusion

Wastewater sludge treatment and dewatering techniques affect the nitrogen content of sludge, thus influencing its fertilizer value. Sludge dried in thin layers of 10 to 15 cm thickness have higher total N and N fertilizer value than sludge dried in thick layers of 20 cm and higher. The final decision on the choice of sludge drying thickness on drying beds, however, depends on other factors, including the availability of land for drying and ease of sludge collection from drying beds.

CHAPTER 5 - NITROGEN FERTILIZER VALUE OF SLUDGE

5.1 Introduction

The amount of organic-N that can be mineralized and become readily available for plant uptake is dictated by the type of sludge, sludge treatment method, type and level of drying (Rigby et al. 2016) and weather (mainly rainfall and temperature) (Ogbazgi et al., 2016). Therefore, quantitative understanding of the N dynamics under this matrix is crucial for sustainable use of wastewater sludge in agricultural lands. Besides, crops have varying nutrient requirements. Generally the application rate of commercial inorganic fertilizers is determined by the crop nutrient requirements (crop specific) and the prevailing climate, mainly rainfall, and the availability of supplemental irrigation. The Fertilizer Society of South Africa developed a handbook to guide fertilizer advisors and farmers (FSSA, 2007). It is therefore of utmost importance to synchronize the South African Sludge Guidelines for Beneficial Land Application (2006) with the South African Fertilizer Handbook (2007) to enhance agricultural productivity, while minimizing negative environmental impacts from nitrate leaching and phosphorus accumulation.

The objective of this section of the report is to develop an Addendum to refine the current South African Sludge Guidelines Volume 2 of 5 (2006).

To achieve the stated objective, the following methodology was used.

5.2 Materials and Methods

This addendum was developed based on model simulations using a calibrated computer (SWB-Sci) model. The model was calibrated and validated with field experimental studies (Tesfamariam et al., 2015) and used to test various hypotheses on sludge application across South African agro-ecological zones (Ogbazghi et al., 2016). To generate the data needed to develop this addendum, controlled incubation studies from South Africa and the USA, which are presented in Table 4.2, were used to calibrate the model. The calibrated model was used to run scenario simulations across South African agro-ecological zones to assess N mineralization rates during the first five to six years following a once-off sludge land application (10 t ha^{-1}). Such simulations were conducted for each representative site using sludges of varying N contents. From these simulations, the effect of initial sludge N content on the six-year cumulative N mineralization rate was evaluated.

Once the effect of sludge N content on N mineralization rate had been established, scenario simulations on N mineralization were conducted for five of the six South African agro-ecological zones (arid, semi-arid, sub-humid, humid, and super-humid) under both rain-fed and irrigated farming systems. Finally, the N mineralization rates established from the scenario simulations were used to establish the sludge application rates according to crop N requirements for the main crops grown in various regions of South Africa as stipulated in the South African Fertilizer Handbook (FSSA, 2007).

5.3 Results and Discussion

5.3.1 Nitrogen mineralization

Sludge N release rates varied among sludge treatment methods and dewatering techniques. Besides this, sludge stockpiling (storage) time plays significant role in determining its N fertilizer value. Although many sludge mineralization studies have been conducted under controlled temperature and soil water conditions around the globe, most lack detailed information about the sludge treatment processes and sludge moisture content at time of incubation, while others do not present detailed information on the methodology. In this report, only sludge mineralization studies with detailed information about the sludge treatment and dewatering techniques and the methodologies for the incubation studies (Table 4.2) are considered. It was apparent that N mineralization from anaerobically digested municipal sludge under controlled temperature (25 ºC) and moisture conditions varied between 22% and 59% (Table 4.2). This variation seemed to have a very strong positive correlation ($r^2 = 0.92$) with the solid content of the sludge (Figure 5.1). This correlation, however, did not include data from two sites (Tacoma, WA, USA and Steyton, OR, USA), which drastically lowered the coefficient of correlation. The best fit correlation equation, which was able to predict N mineralization based on the solid percentage, was Harris model Equation 5.1

$$N_{min} = \frac{1}{(-1.03 + 1.05(Solid_{\%})^{5.49})} \qquad \text{Equation 5.1}$$

Another correlation was established between organic-N and N mineralization. The relationship between organic-N and N mineralization was promising, with good correlation coefficient ($r^2 = 0.85$) (Figure 5.2). This relationship was best expressed with an exponential association equation (Equation 5.2).

$$N_{min} = 37.72(1 - e^{-0.529 \times Org\_N\%}) \qquad \text{Equation 5.2}$$

Where N_{min} is nitrogen mineralization; Org_N% is organic-N content of sludge in percentage

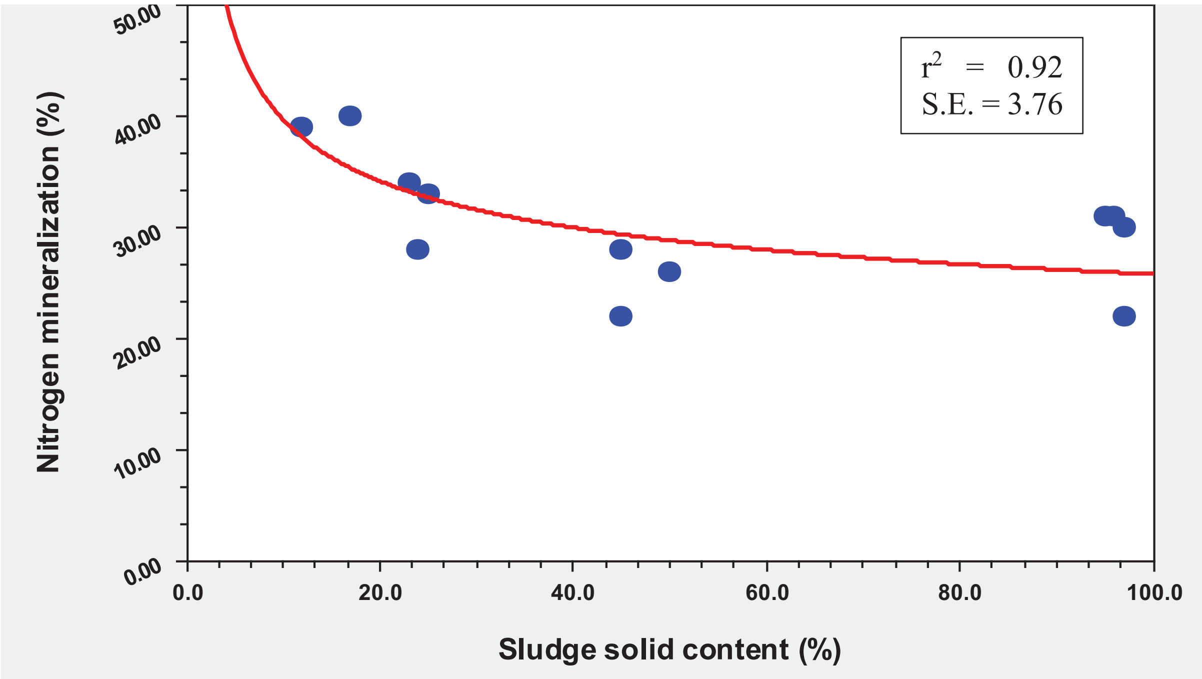


Figure 5.1 Nitrogen mineralization as a function of sludge solid content

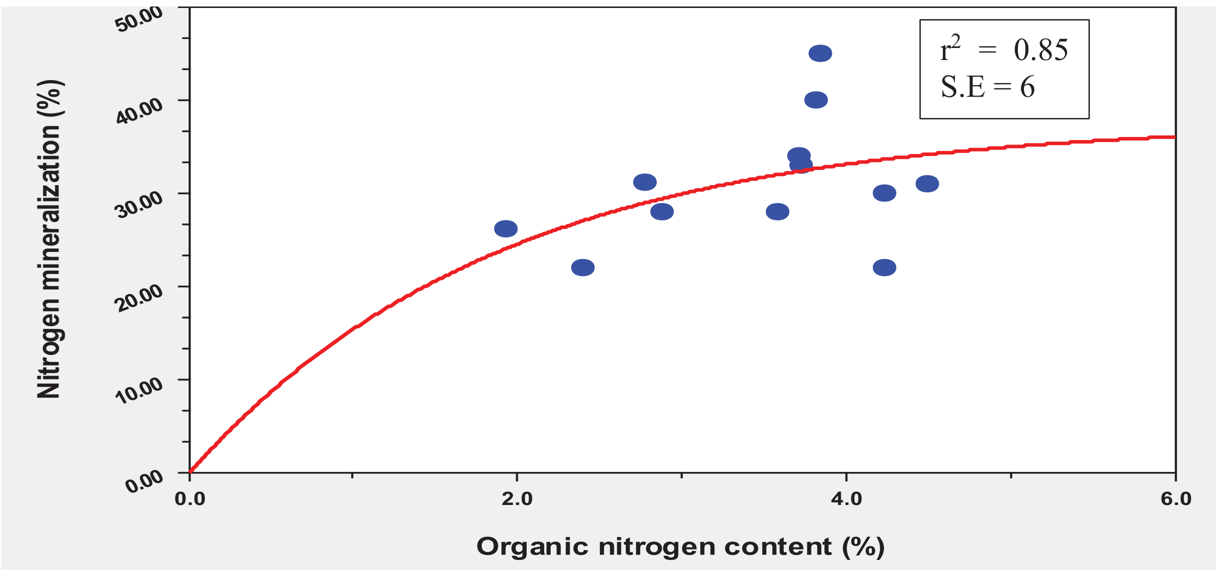


Figure 5.2 Nitrogen mineralization as a function of sludge organic nitrogen content

For this report, the correlation between organic-N and N mineralization was preferred to the solid percentage versus mineralization function, mainly because the latter is influenced by post-treatment sludge water content changes because of rain, which is often the case.

5.3.2 Sludge nitrogen mineralization across South African agro-ecological zones

The effect of rainfall on annual and cumulative nitrogen mineralization

Previous model simulations on nitrogen mineralization across South African agro-ecological zones have shown that besides sludge properties (source, treatment processes and drying techniques), N mineralization is influenced by climatic factors (Ogbazgi et al., 2016). In this report further simulation was conducted to investigate the amount of N mineralized per year during the next five consecutive years from a single sludge application. This was tested for representative sites from each of the five South African agro-ecological zones during low, average, and high rainfall years. The model simulation results show that the overall (cumulative of six years) N mineralization in arid areas remained similar (40% of organic-N mineralized in 6 years) regardless of the amount of rainfall at the time of sludge application (Figure 5.3).

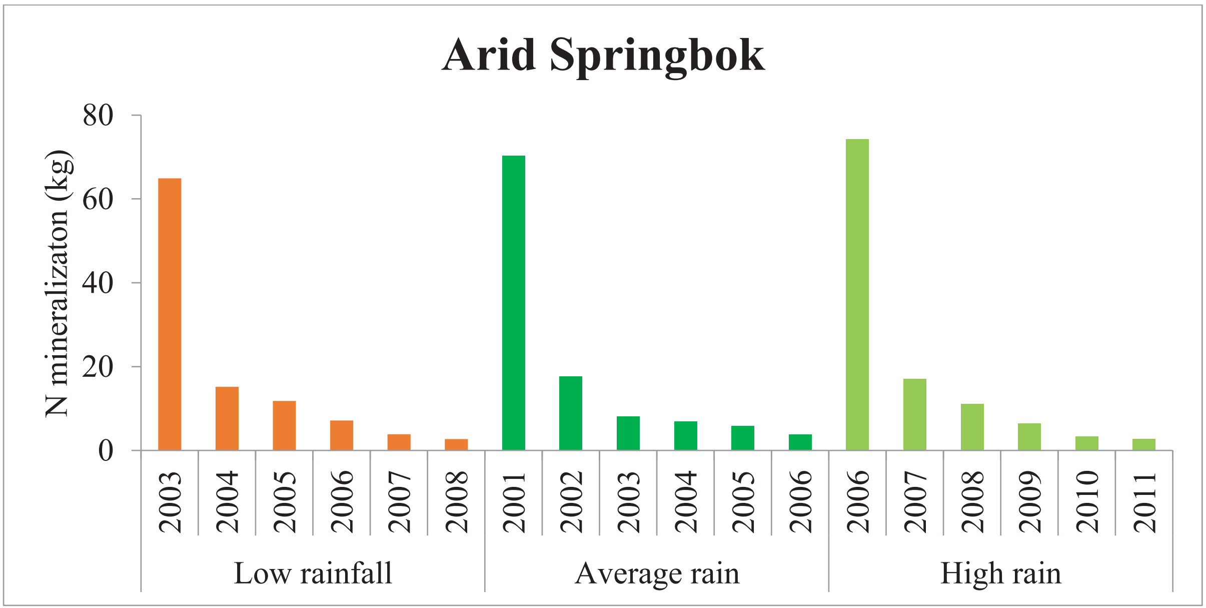


Figure 5.3 Nitrogen mineralization from sludge-amended sandy clay loam soil applied at a rate of 10 t ha^{-1} (organic-N content of sludge = 2.78%) in the arid agro-ecological zone of Springbok

This was mainly because the N that was not mineralized during the low rainfall year (residual organic-N) at the time of application was mineralized in the years that followed. The only difference observed was during the first year of application, when the N mineralization in a relatively higher rainfall season was about 4% higher than in the low rainfall season. The percentages of N mineralization during the first year of sludge application were 23%, 25%, and 27% of the organic-N applied during the low (131 mm), average (225 mm), and high (304 mm) rainfall years, respectively. Mineralization from residual organic-N in the second year after sludge application decreased to 7% (low rainfall), 8% (average rainfall), and 8% (high rainfall). On the third year, nitrogen mineralization from residual organic-N was 6% (low rainfall), 4% (average rainfall), and 6% (high rainfall). In the fourth, fifth, and sixth years, nitrogen mineralization from the residual organic-N decreased to 4% and below for all.

In the semi-arid area of Polokwane, the cumulative N mineralization during the first six consecutive years, after a single 10 t ha^{-1} sludge application, was 42% during the low rainfall year (223 mm), 45% during the average rainfall year (388 mm), and 47% during the high rainfall year (513 mm) (Figure 5.4). The main difference between the years occurred in the first year of application (26% low rainfall, 29% average rainfall and 31% high rainfall).

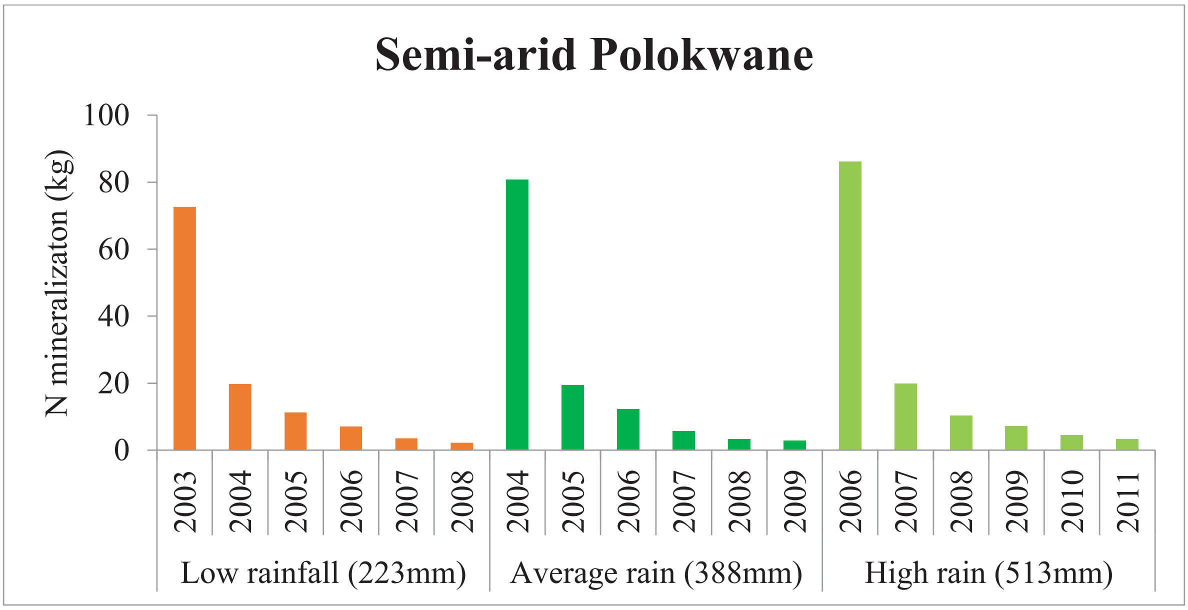


Figure 5.4 Nitrogen mineralization from sludge amended sandy clay loam soil applied at a rate of 10 t ha^{-1} (organic-N content of sludge = 2.78%) in the semi-arid agro-ecological zone of Polokwane

In the sub-humid area of Johannesburg, the cumulative N mineralization within six years after a single sludge application of 10 t ha^{-1} was 43% (low rainfall), 52% (average rainfall), and 62% (high rainfall) of the organic-N applied (Figure 5.5). The major difference in N mineralization between the years occurred during the first year of application (28% low rainfall; 32% average rainfall; 36% high rainfall) and second year of application (9% low rainfall; 13% average rainfall; 14% high rainfall).

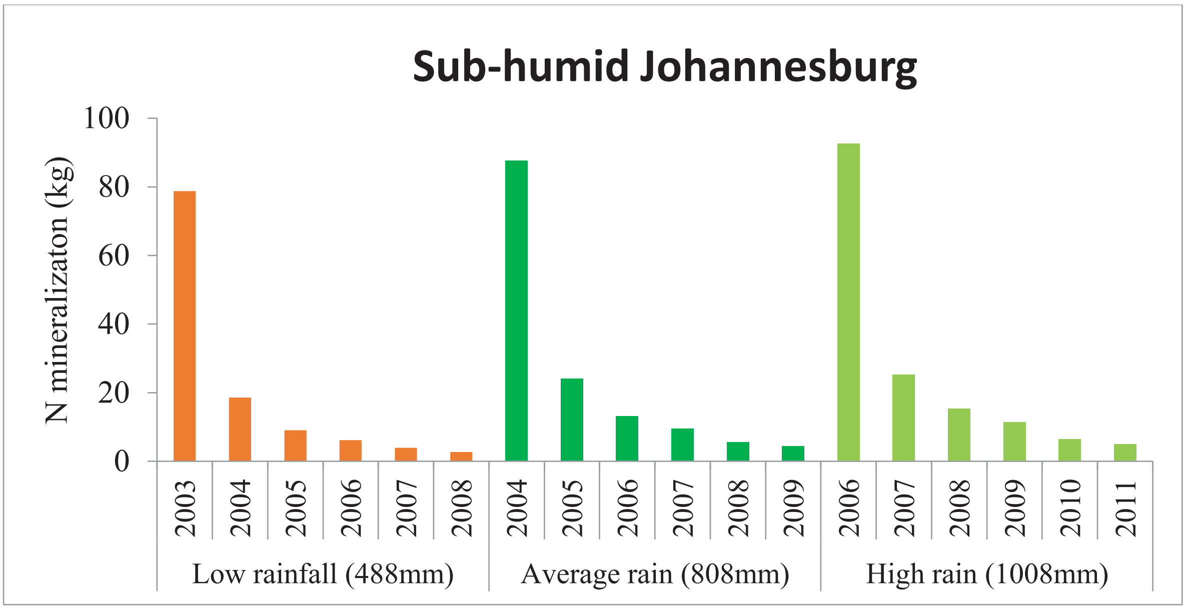


Figure 5.5 Nitrogen mineralization from anaerobically digested sludge-amended sandy clay loam soil applied at a rate of 10 t ha^{-1} (organic-N content of sludge = 2.78%) in the sub-humid agro-ecological zone of Johannesburg

In the humid zone of Durban, cumulative N mineralization within six consecutive years of a single sludge application rate of 10 t ha^{-1} was 51%, 57%, and 64% when rainfall during the sludge application year was low, average, and high, respectively (Figure 5.6). Nitrogen mineralization during the first year of application was 36% (low and average rainfall years) and 39% (high rainfall year). Mineralization from residual organic nitrogen in the second and third years after sludge application was 11% and 6% (low rainfall), 17% and 9% (average rainfall), and 23% and 10% (high rainfall), respectively.

Cumulative nitrogen mineralization from six consecutive years following a single 10 t ha^{-1} sludge application on a sandy clay loam soil in the super-humid area of Nelspruit was 55%, 68%, and 73%, when sludge was applied during low, average, and high rainfall seasons, respectively (Figure 5.7). More than 40% of the organic-N was mineralized in the first year of application when sludge was applied during the average and high rainfall seasons.

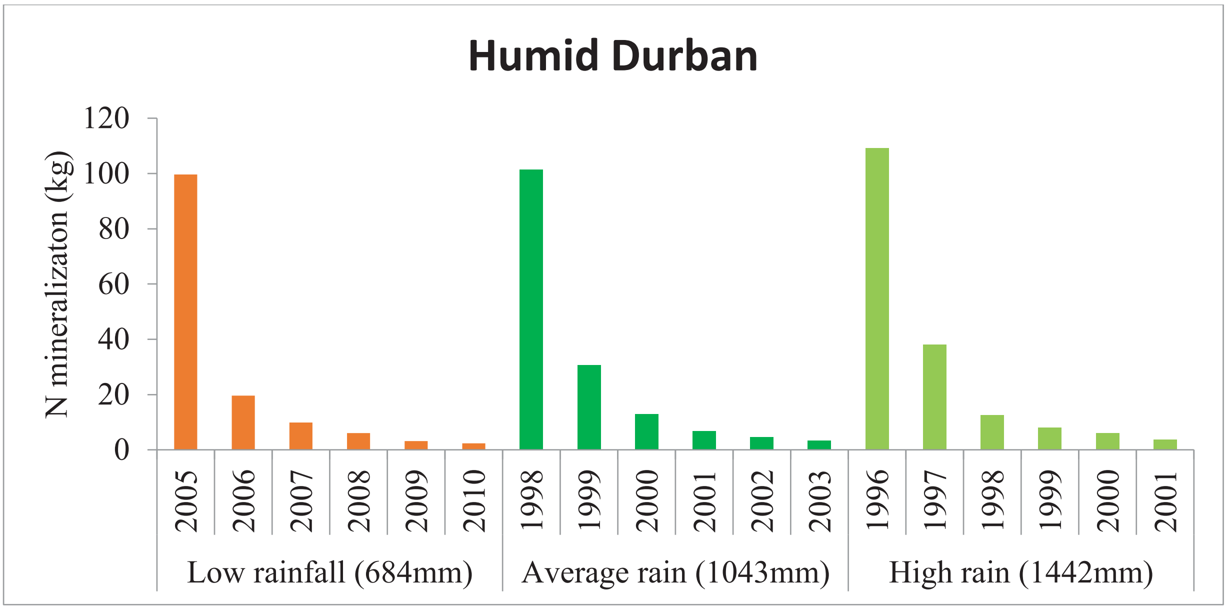


Figure 5.6 Nitrogen mineralization from anaerobically digested sludge-amended sandy clay loam soil applied at a rate of 10 t ha^{-1} (organic-N content of sludge = 2.78%) in the humid agro-ecological zone of Durban

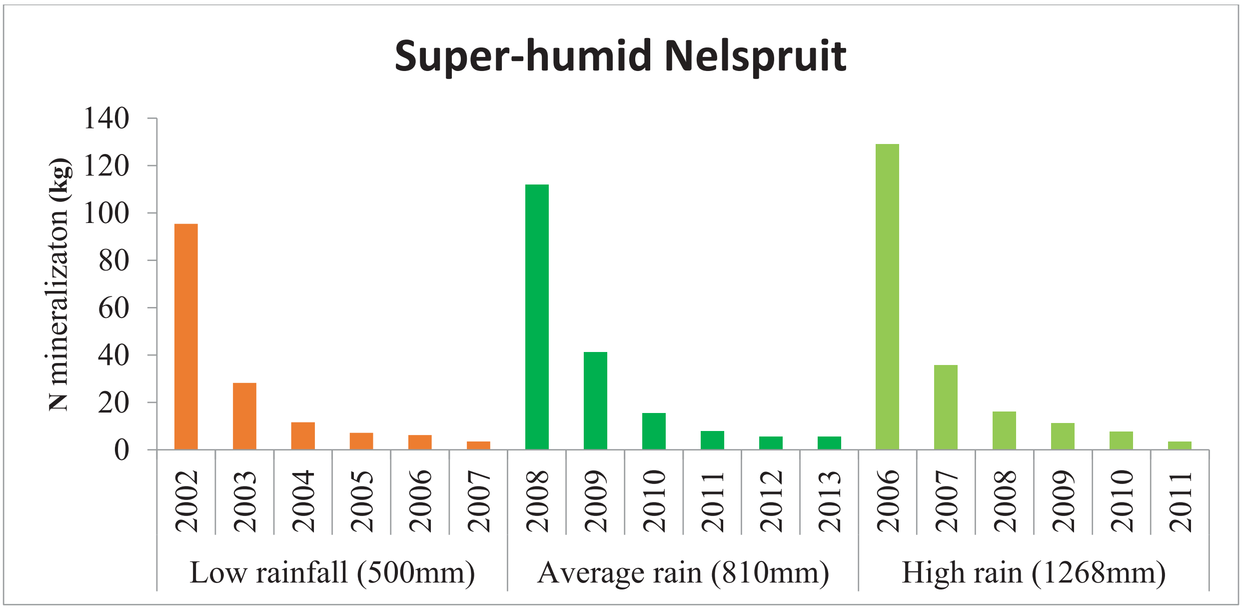


Figure 5.7 Nitrogen mineralization from anaerobically digested sludge amended sandy clay loam soil applied at a rate of 10 t ha^{-1} (organic-N content of sludge = 2.78%) in the super-humid agro-ecological zone of Nelspruit

Mean cumulative N mineralization across agro-ecological zones varied linearly with the increment in rainfall from 40% in the arid areas to 65% in the super-humid zone of Nelspruit. Generally N mineralization from the average rainfall year was similar to the average of the low and high rainfall years in most cases. This is in agreement with the findings of Ogbazghi (2016), who reported that, except for the exceptionally low rainfall anomalous year, N mineralization remained similar to the average N mineralization for all study years. Therefore, the average rainfall year is used in the next section to evaluate the effect of sludge organic-N content on annual N mineralization across representative cities in five of the six South African agro-ecological zones.

5.3.3 The effect of sludge organic nitrogen content on nitrogen mineralization rate across South African agro-ecological zones

Scenario simulations were run using the SWB-Sci model to investigate the effect of sludge organic-N content on the N mineralization of sludges for representative sites in five of the six South African agro-ecological zones. The five agro-ecological zones are arid, semi-arid, sub-humid, humid and super-humid.

The effect of sludge organic-N content on N mineralization rate in the arid agro-ecological zone of South Africa

Under rain-fed conditions, N mineralization during the first year of application in the arid zone (Springbok) was lowest (20%) for the sludge with 2% organic-N. As the sludge organic-N content increased to 4%, nitrogen mineralization also increased gradually to 30% (Figure 5.8). Further increase in sludge organic-N content above 4% fluctuated between 30% and 33% mineralization, indicating that it had reached the maximum possible mineralization rate under the prevailing climatic conditions. The trend in N mineralization observed during the first year of sludge application was better expressed using an exponential association equation. The six-year cumulative N mineralization from a single 10 t ha^{-1} sludge application, however, remained similar among the sludges of varying organic-N content. The only exception was the mineralization rate from the sludge with 2% organic-N content, which had slightly lower mineralization (39.6%) than the rest (41–44%). From this, it can be deduced that in the arid zone of South Africa, a single cumulative N mineralization rate of 42% could be used to determine the amount of N released from anaerobically digested sludge under rain-fed farming in a land that has received sludge for more than 5 to 6 years.

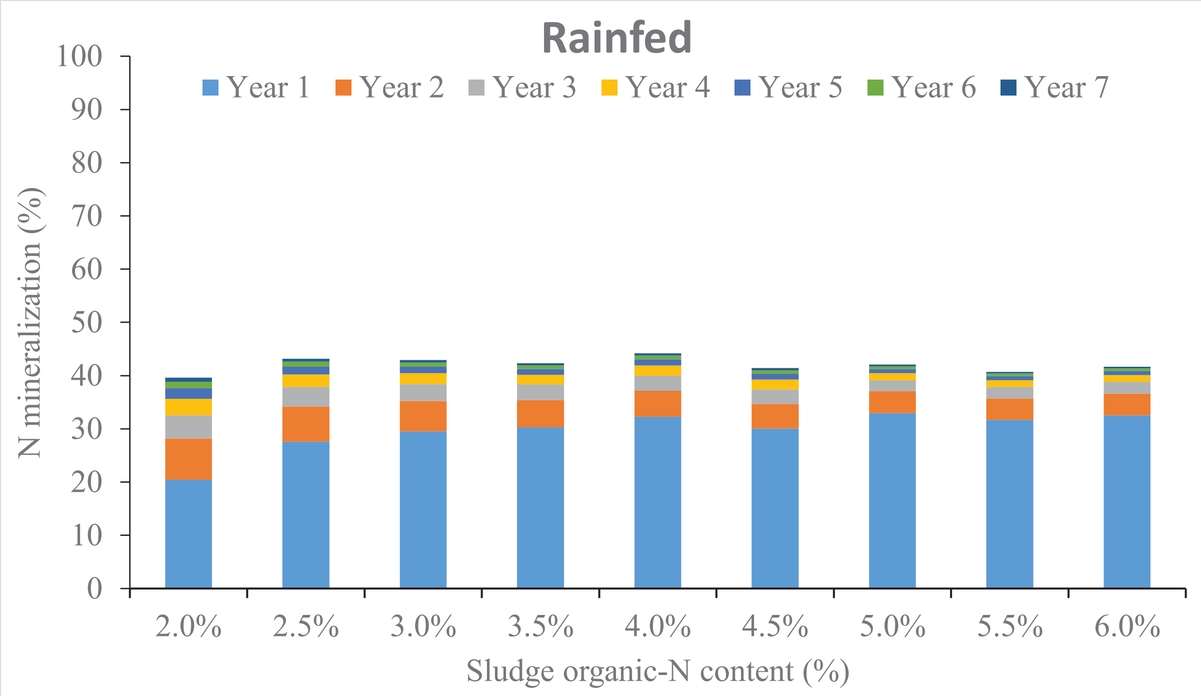


Figure 5.8 Six year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the arid region of Springbok under rain-fed farming

*Sludge with varying organic-N contents

Similar to the rain-fed system, N mineralization during the first year of application under irrigation showed a gradual increment as the sludge organic-N increased from 2% (23% mineralization) to 4% (36% mineralization) (Figure 5.9). The trend in N mineralization from sludges with organic-N content exceeding 4%, however, remained almost constant. The mean cumulative N mineralization under irrigated conditions was 47.6%. Cumulative nitrogen mineralization was generally higher under irrigated conditions (45% to 50%) compared with a rain-fed (39.6% to 44%) system. This is mainly due to the enhanced availability of water, which created an environment that was conducive to microbial activity to decompose more organic matter per unit time. Similar to the rain-fed system, it can be deduced that in the arid zone of South Africa, a single cumulative N mineralization rate of 48% could be used to determine the amount of N released from anaerobically digested sludge under irrigated farming in a land that has received sludge for more than 5 to 6 years.

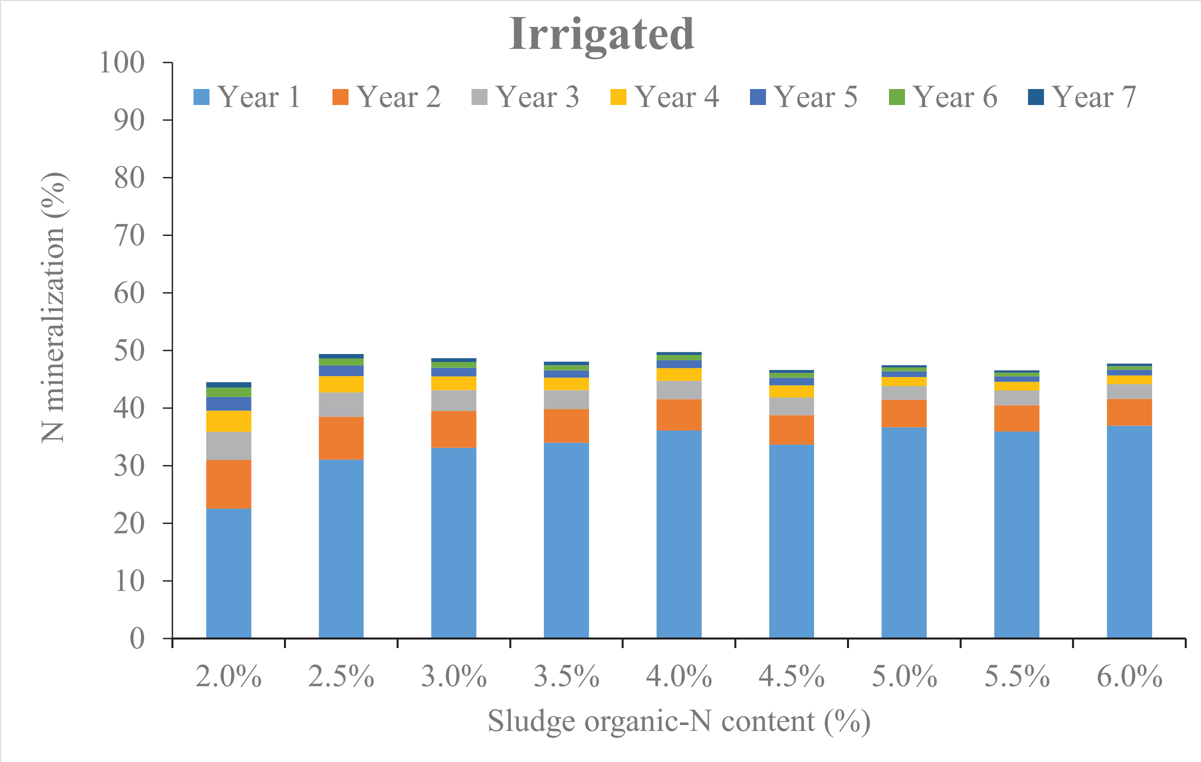


Figure 5.9 Six year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the arid region of Springbok under irrigated farming

*Sludge with varying organic-N contents

The effect of sludge organic-N content on N mineralization rate in the semi-arid agro-ecological zone of South Africa

For this study, the city of Polokwane was selected to represent the semi-arid agro-ecological zone mainly because of the availability of long-term weather data for the city. Nitrogen mineralization during the first year of application gradually increased from 23% for sludge organic-N content of 2% to 34% for sludge organic-N content of 4% (Figure 5.10). Further increase in sludge organic-N content above 4%, however, did not show an increment in N mineralization rates. The seven-year cumulative N mineralization rate following a single 10 t ha^{-1} sludge application varied from 43% for the sludge with 2% organic-N to 50% for the sludge with 4% organic-N. Except for the sludge with organic-N content of 2%, N mineralization from sludges with higher organic-N contents varied between 46% for sludge with 5.5% organic-N and 50% for sludge with organic-N content of 4%.

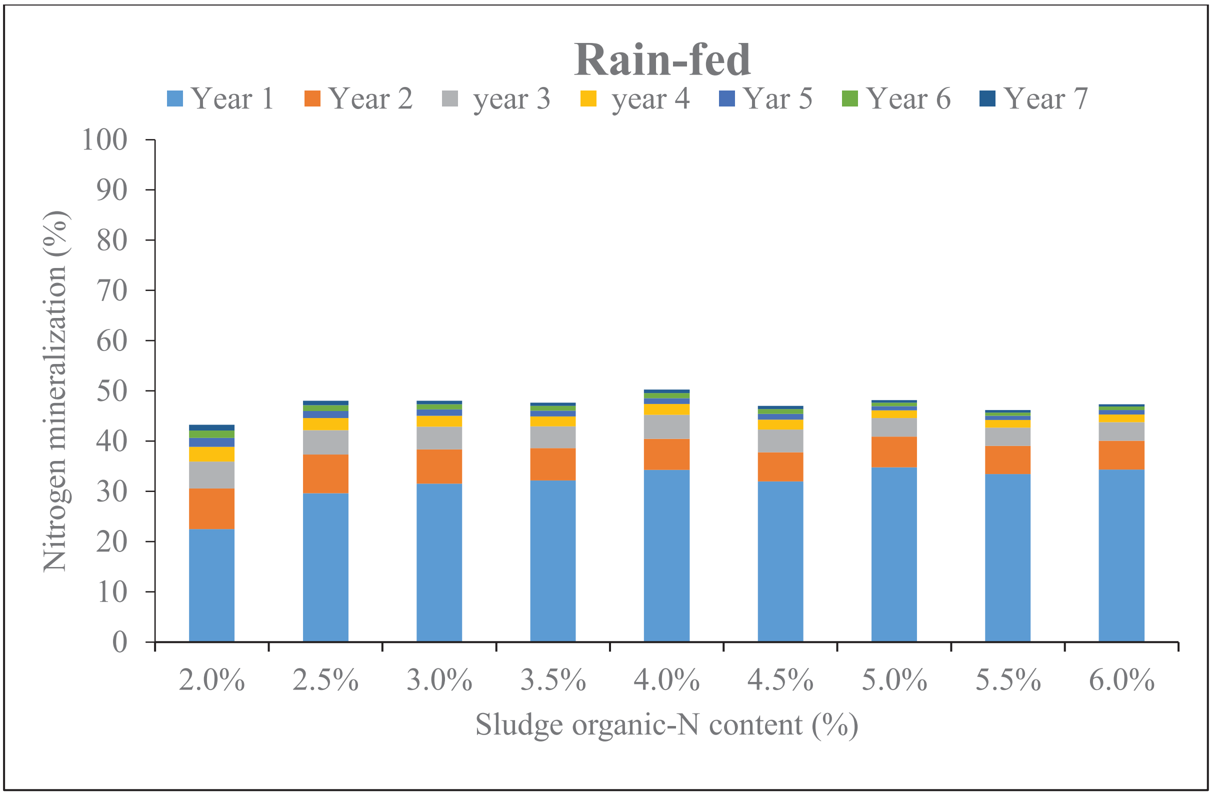


Figure 5.10 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the semi-arid region of Polokwane under a rain-fed farming

*Sludge with varying organic-N contents

Similar to the rain-fed system, N mineralization in the first year of sludge application gradually increased from 25% for the sludge with 2% organic-N content to 39% for the sludge with 4% organic-N (Figure 5.11). The cumulative N mineralization at the end of the seventh year varied between 55% for the sludge with organic-N content of 2% and 63% for the sludge with organic-N content of 4%. The mean cumulative N mineralization was 47% under rain-fed and 59% under irrigation.

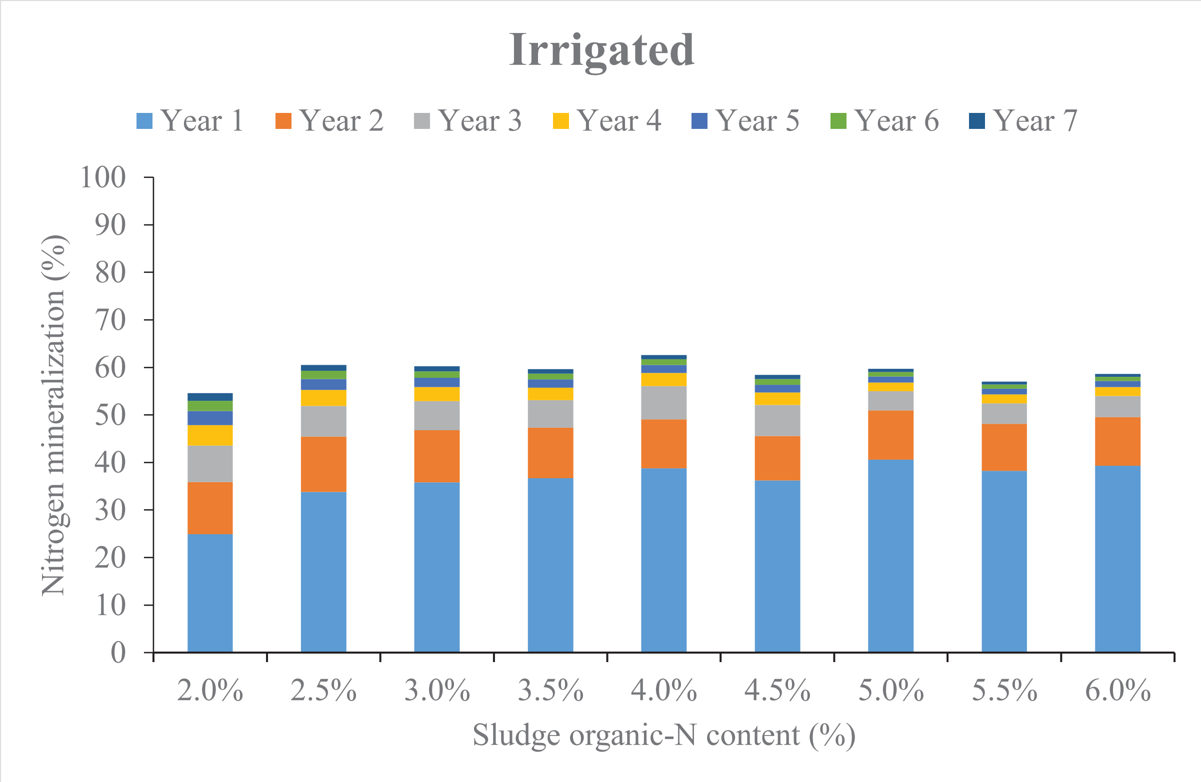


Figure 5.11 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the semi-arid region of Polokwane under irrigated farming.

*Sludge with varying organic-N contents

The effect of sludge organic-N content on N mineralization rate in the sub-humid agro-ecological zone of South Africa

Seven years cumulative nitrogen mineralization under rain-fed conditions in the sub-humid zone of Johannesburg area ranged from 49% for a sludge with 2% organic-N to 58% for a sludge with organic-N content of 4% (Figure 5.12). The mean cumulative N mineralization under a rain-fed system was 55%. Nitrogen mineralization during the first year of application followed similar trends to that of the arid and semi-arid areas.

Cumulative N mineralization (7 years) under irrigation (Figure 5.13) was generally higher than rain-fed systems (60% for a sludge with 2% organic-N to 70% for a sludge with organic-N content of 4%) (Figure 5.12). The mean cumulative N mineralization rate under irrigated systems in the sub-humid zone of Johannesburg was 67%.

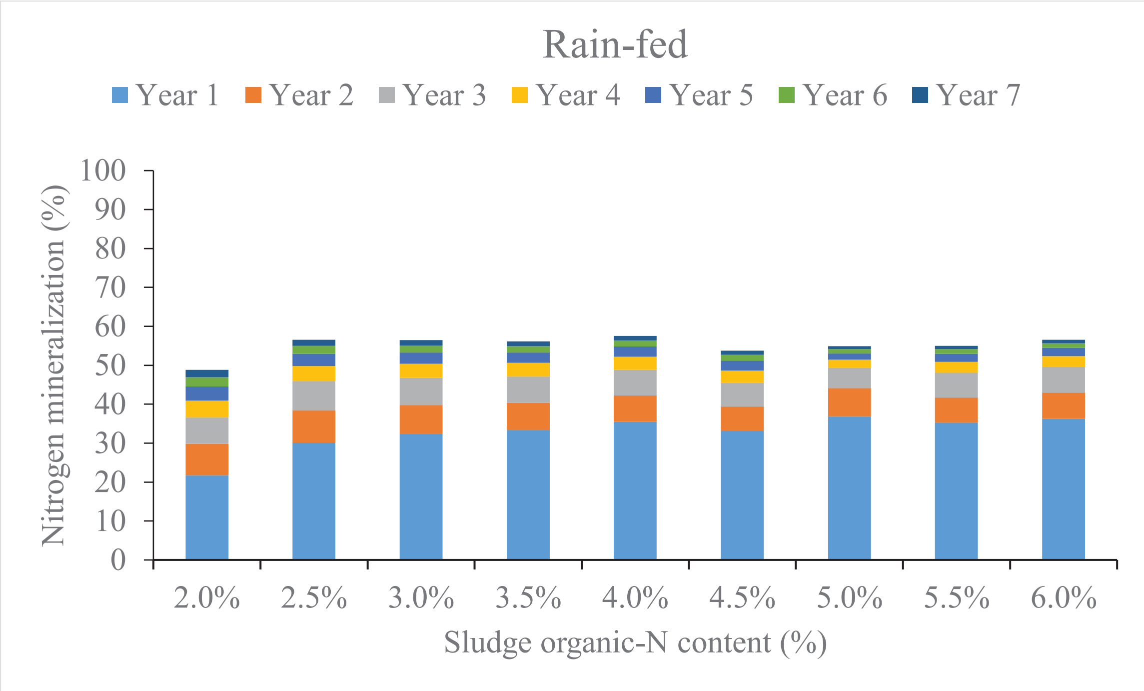


Figure 5.12 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the sub-humid region of Johannesburg under rain-fed farming

*Sludge with varying organic-N contents

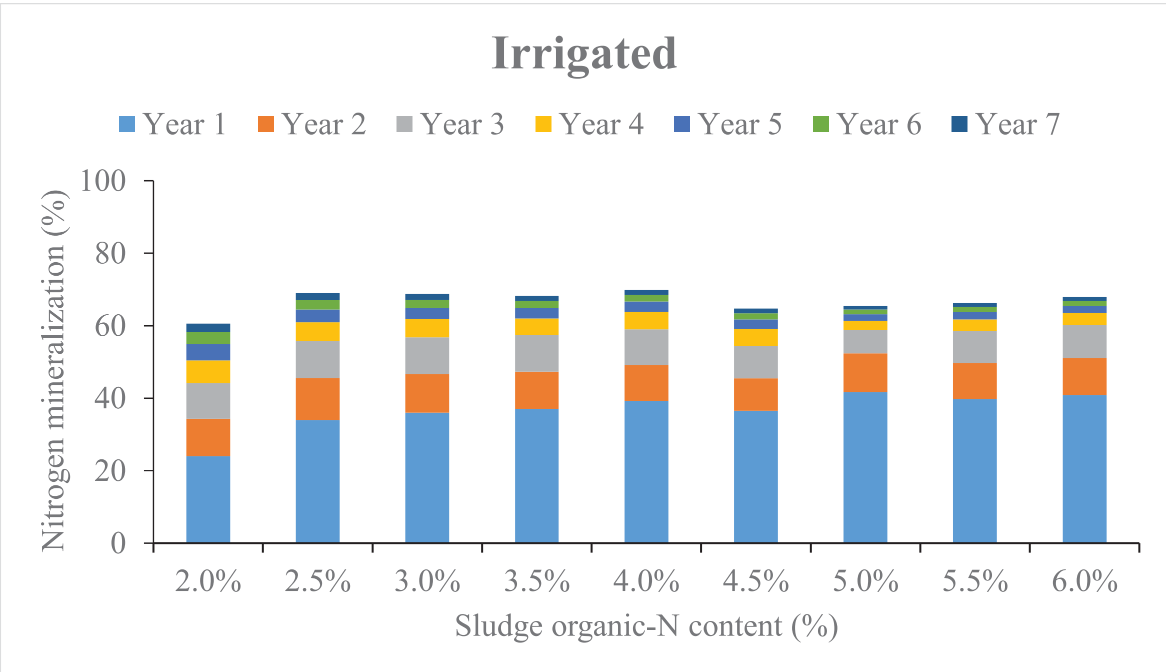


Figure 5.13 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the sub-humid region of Johannesburg under irrigated farming

*Sludge with varying organic-N contents

The effect of sludge organic-N content on N mineralization from sludge amended soils in the humid agro-ecological zone of South Africa

Cumulative (7 years) N mineralization under rain-fed conditions in the sub-humid zone of Durban area varied between 54% for a sludge with an organic-N content of 2% to 60% for a sludge with organic-N content of 4% (Figure 5.14). The mean cumulative N mineralization rate under rain-fed systems in this agro-ecological zone was 58%. Similar to the other agro-ecological zones, cumulative N mineralization rate during the first year of sludge application in the humid zone of Durban increased from 26% for a sludge with organic-N content of 2% to 40% for a sludge with organic-N content of 4%.

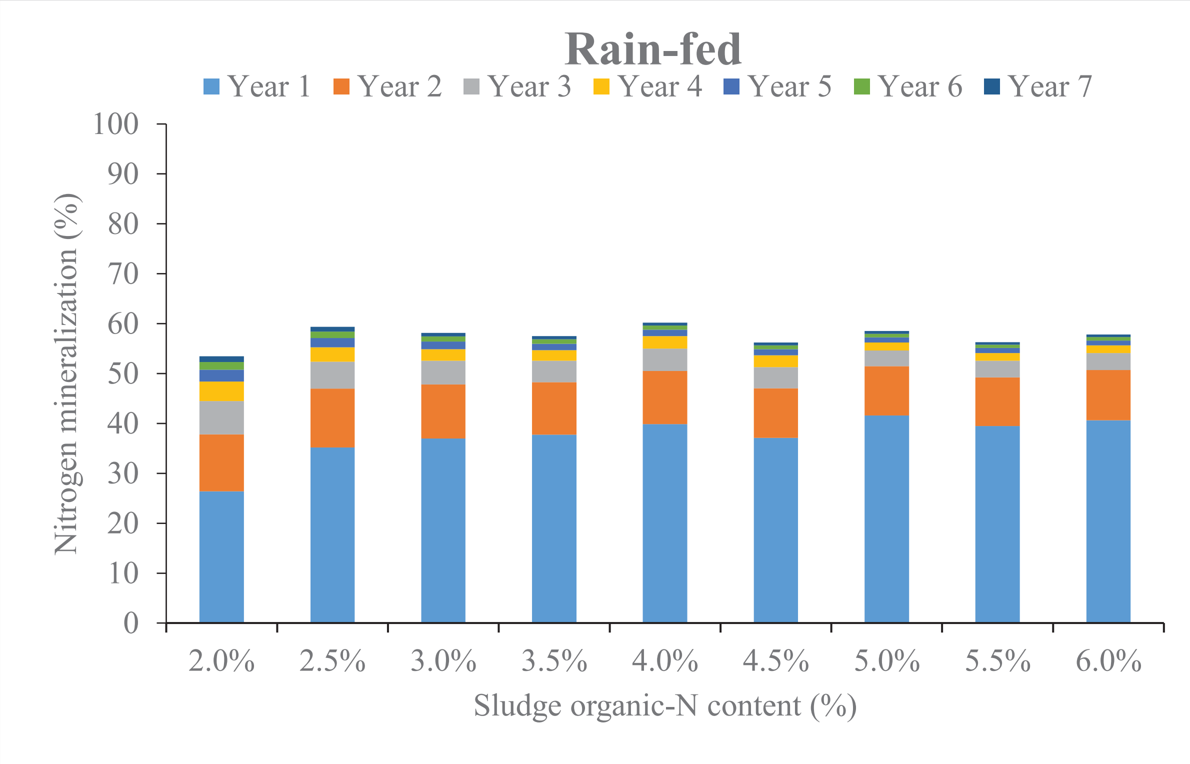


Figure 5.14 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the humid zone of Durban under rain-fed farming

*Sludge with varying organic-N contents

Cumulative (7 years) N mineralization from single 10 t ha^{-1} sludge application under irrigation varied between 68% for sludge with organic-N content of 2% to 75% for a sludge with organic-N content of 4% (Figure 5.15). The mean cumulative N mineralization rate under irrigation in the humid zone of Durban was 73%.

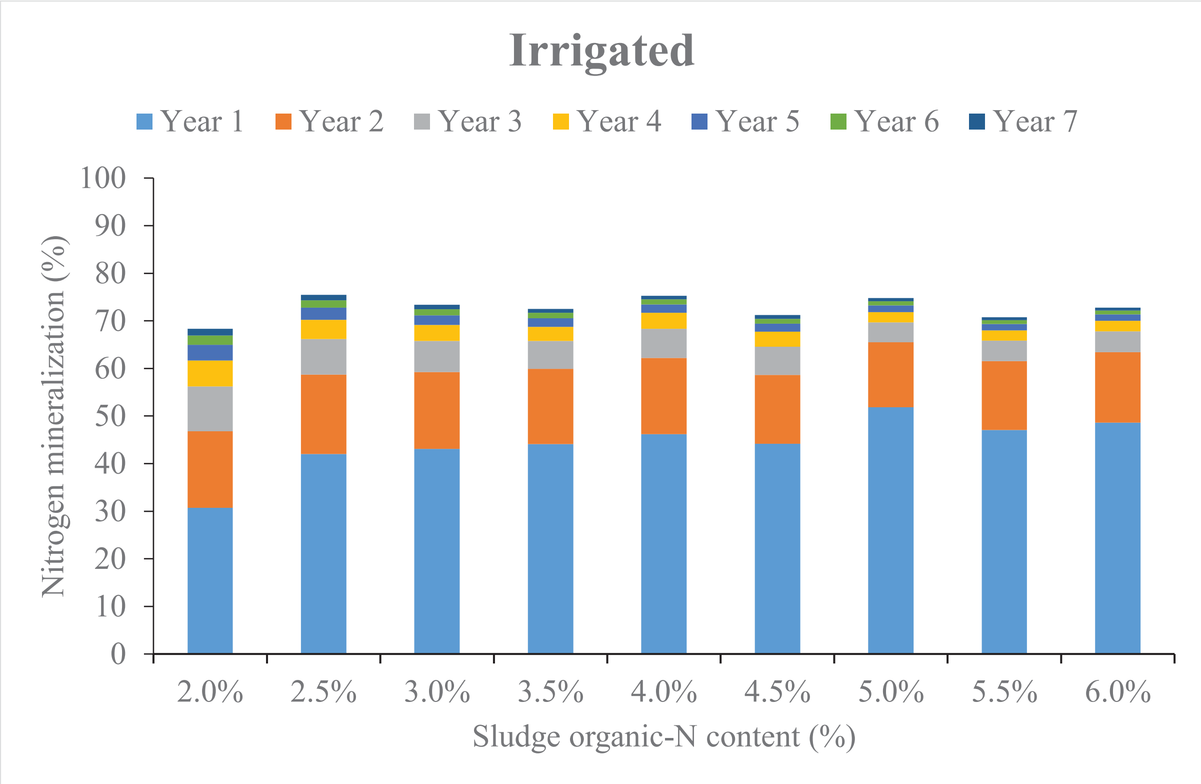


Figure 5.15 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the humid zone of Durban under irrigated farming

*Sludge with varying organic-N contents

The effect of sludge organic-N content on N mineralization from sludge-amended soils in the super-humid agro-ecological zone of South Africa

Cumulative (7 years) N mineralization from a single 10 t ha^{-1} sludge application under rain-fed farming in the super-humid agro-ecological zone of Nelspruit ranged from 64% for a sludge with 2% organic-N content to 72% for a sludge with an organic-N content of 4% (Figure 5.16). Similar to other agro-ecological zones, N mineralization during the first year of sludge application showed a gradual increment from 30% to 47% as the sludge organic-N content increased from 2% to 4%.

Cumulative N mineralization under irrigation in the super-humid zone of Nelspruit varied between 83% for a sludge with organic-N content of 2% and 92% for a sludge with organic-N content of 4% (Figure 5.17). The mean cumulative N mineralization in this agro-ecosystem under irrigation was 88%.

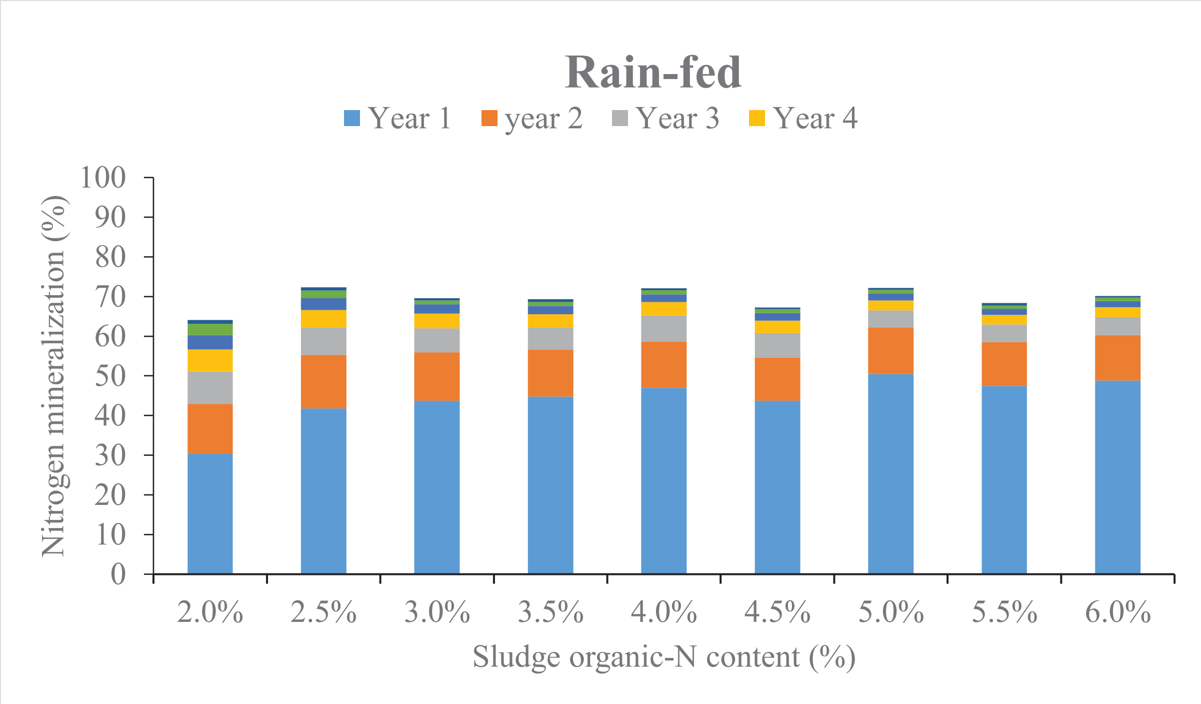


Figure 5.16 Seven year cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the super-humid zone of Nelspruit under rain-fed farming

*Sludge with varying organic-N contents

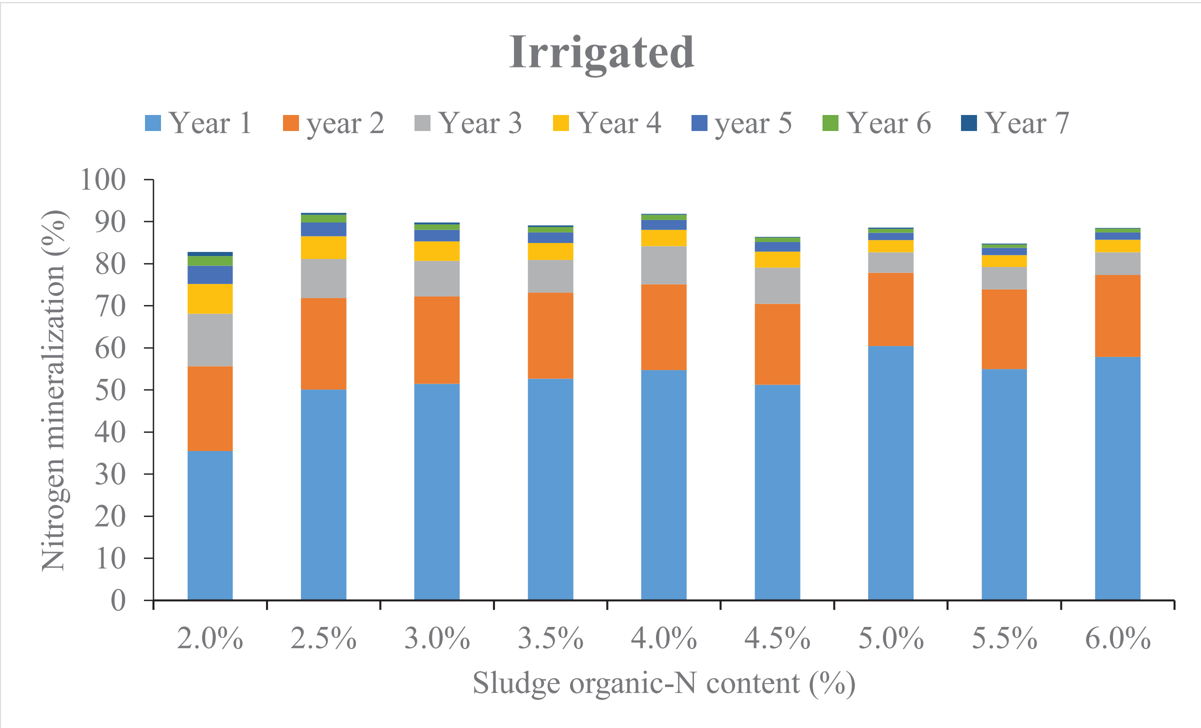


Figure 5.17 Seven years of cumulative nitrogen mineralization from a single 10 t ha^{-1} anaerobically digested sludge application in the super-humid zone of Nelspruit under irrigated farming

*Sludge with varying organic-N contents

5.3.4 Summary of nitrogen mineralization from sludge amended soils

Model simulation results of N mineralization from anaerobically digested sludge for five of the six South African agro-ecological zones are summarized in Table 5.1. This table presents expected cumulative N mineralization from anaerobically digested sludge applied to a land for more than five years. For lands receiving sludge for the first, second and third time, the expected mineralization rates are presented in Table 5.2.

Table 5.1 Annual nitrogen mineralization of anaerobically digested sludge from lands that received sludge for more than six years

Agro-ecological zone	Representative site	Farming system	N mineralization (%)	N release (kg N per ton sludge)
		Rain-fed	42	$Y = 4.19 * N_O$
Arid	Springbok	Irrigated	48	$Y = 4.76 * N_O$
		Rain-fed	47	$Y = 4.75 * N_O$
Semi-arid	Polokwane	Irrigated	59	$Y = 5.90 * N_O$
		Rain-fed	55	$Y = 5.55 * N_O$
Sub-humid	Johannesburg	Irrigated	67	$Y = 6.69 * N_O$
		Rain-fed	58	$Y = 5.76 * N_O$
Humid	Durban	Irrigated	73	$Y = 7.28 * N_O$
		Rain-fed	70	$Y = 6.82 * N_O$
Super-humid	Nelspruit	Irrigated	88	$Y = 8.67 * N_O$

'Y' in the linear equation presented on the fifth column of Table 5.1 represents the amount of N (kg) mineralized from a ton of sludge

- 'N_O' in the linear equation presented on the fifth column is the organic-N content of sludge used in percentage

Table 5.2 Annual nitrogen mineralization from soils amended with sludge for their first, second, and/or third year of application

Agro-ecological zone	Representative site	Farming system	N mineralization per year (%)			N release (kg N per ton sludge)		
			Year 1	Year 2	Year 3	Year 1	Year 2	Year 3
Arid	Springbok	Rain-fed	30	5	3	$Y = (3.72 * N_O) – 2.61$	$Y = (0.21 * N_O) + 1.12$	$Y = (0.104 * N_O) + 0.66$
		Irrigated	33	6	3	$Y = (4.22 * N_O) – 3.11$	$Y = (0.26 * N_O) + 1.17$	$Y = (0.12 * N_O) + 0.74$
Semi-arid	Polokwane	Rain-fed	32	7	4	$Y = (3.88 * N_O) – 2.50$	$Y = (0.44 * N_O) + 0.74$	$Y = (0.28 * N_O) + 0.56$
		Irrigated	36	10	6	$Y = (4.49 * N_O) – 3.09$	$Y = (0.93 * N_O) + 0.43$	$Y = (0.27 * N_O) + 1.13$
Sub-humid	Johannesburg	Rain-fed	33	7	7	$Y = (4.20 * N_O) – 3.23$	$Y = (0.58 * N_O) + 0.47$	$Y = (0.56 * N_O) + 0.35$
		Irrigated	37	10	9	$Y = (4.75 * N_O) – 3.83$	$Y = (0.96 * N_O) + 0.24$	$Y = (0.72 * N_O) + 0.72$
Humid	Durban	Rain-fed	37	11	4	$Y = (4.60 * N_O) – 3.08$	$Y = (0.88 * N_O) + 0.60$	$Y = (0.16 * N_O) + 0.99$
		Irrigated	44	15	6	$Y = (5.62 * N_O) – 4.24$	$Y = (1.30 * N_O) + 0.85$	$Y = (0.18 * N_O) + 1.50$
Super-humid	Nelspruit	Rain-fed	44	12	6	$Y = (5.60 * N_O) – 4.14$	$Y = (1.03 * N_O) + 0.56$	$Y = (0.26 * N_O) + 1.16$
		Irrigated	52	20	8	$Y = (6.59 * N_O) – 4.86$	$Y = (1.74 * N_O) + 0.89$	$Y = (0.19 * N_O) + 2.13$

Y in the linear equation is the amount of N released from a ton of sludge
N_O is organic-N content of sludge used in percentage

5.3.5 Implications of the findings for existing South African sludge guidelines

The reported variation on N mineralization between farming systems (rain-fed as opposed to irrigated) within an agro-ecological zone and the variations among agro-ecological zones indicate that flexible guidelines are needed that take these factors into account. Hence, sludge guidelines and sludge upper limits ought to be synchronized with the South African fertilizer guidelines as presented in the fertilizer handbook (FSSA, 2007). This handbook provides a list of the most commonly grown crops in South Africa and their fertilizer requirements.

5.3.6 Crop fertilizer requirement estimation

The first phase of fertilizer application recommendations for a given crop growing in a given soil under given weather conditions is to determine the optimum fertilizer requirement (Fer_Req) for maximum productivity in that area. The South African Fertilizer Handbook (FSSA 2007) has established the relationship between target yield and the nutrient requirements to achieve the target yield for a range of most commonly grown crops in South Africa. To synchronize South African Sludge Guidelines (volume 2) with the fertilizer handbook, target yield data of selected crops and their corresponding fertilizer requirements have been used from the fertilizer handbook to develop regression equations as presented on Table 5.3. Similar regression equations can be developed for other crops.

Table 5.3 Regression equation for the prediction of optimal N level required to achieve target yields for maize, wheat and sunflower (FSSA, 2007)

Crop	Region	Regression equation	Comment
Maize	All maize growing areas	Fer_NReq = (25 * target yield) - 30	
	Southern and western Free State	Fer_NReq = 7.403 * $(\text{target yield})^{2.06}$	
	Central Free State	Fer_NReq = (20 * target yield) – 7.5	
	Eastern Free State	Fer_NReq = 15.26 * $(\text{target yield})^{1.34}$	Mean target yield and fertilizer requirement was used to develop regression equation
	North West	Fer_NReq = (20 * target yield) – 15	
	Mpumalanga	Fer_NReq = (20 * target yield) – 10	
Wheat	Limpopo (Springbok Flats and Dwaalboom)	Fer_NReq = (15 * target yield) – 14.17	
	Eastern Cape coastal area	Fer_NReq = (17.57 * target yield) – 4.35	
	Swartland	Fer_NReq = (40.30 * target yield) – 13.80	
	Southern Cape	Fer_NReq = (32.15 * target yield) – 29.65	
	Irrigated land	Fer_NReq = (25.74 * target yield) – 2.87	
Sunflower	All sunflower growing areas	Fer_NReq = (65 * target yield) – 75.67	Mean values used to generate equation

Fer_NReq: optimum fertilizer requirement for a given target yield

Once the fertilizer requirement has been established, the next step is to collect soil samples from representative sites in the field for analyses to determine the fertility status of the soil. Soil samples for testing soil nutrient status need to be taken before fertilizer application to the next crop. In areas where the soil properties differ within a field, the field should be divided into management areas with similar characteristics, and fertilizer recommendations should be made accordingly. The actual fertilizer requirement of the crop, in this case N (Act_FerN_req), should then be computed by subtracting the amount of plant-available nutrients in the top 60 cm soil layer from the optimum fertilizer requirement (Fer_Req) as presented in Equation 5.1.

$$Act\_FerN\_req = Fer\_NReq - Soil\_N$$

Equation 5.1

Where Act_FerN_req is actual N fertilizer requirement of the crop; Fer_NReq is optimum fertilizer requirement of crop for maximum productivity under the given soil and climatic conditions; and Soil_N is the amount (mass) of inorganic-N (nitrate + ammonium + nitrite) in the top 60 cm soil layer.

The mass of inorganic-N (Inorg_N_Mass) in the top 60 cm soil layer is estimated from measured bulk density and inorganic-N concentration (Inorg_N_Conc) using Equation 5.2.

$$Inorg\_N\_Mass = Inorg\_N\_Conc * bulk\ density * Area * Depth$$

Equation 5.2

Where Inorg_N_Mass is mass of inorganic-N in the top 60 cm soil layer; Inorg_N_Conc is the concentration of inorganic-N in the top 60 cm layer; bulk density is the bulk density of the top 60 cm soil layer; Area is the area of the land to be fertilized; and Depth is the thickness of the soil layer where the inorganic-N concentration measurement was conducted (0.6 m).

The actual N fertilizer requirement (Act_FerN_req) of a crop that should be applied from a sludge is computed using Equation 5.3.

$$Act\_FerN\_req = NH_4\text{-}N + NO_3\text{-}N + NO_2\text{-}N + N_m$$

Equation5.3

Where NH_4-N is ammonium-N; NO_3-N is nitrate-N; NO_2-N is nitrite-N; and N_m, mineralizable N (the fraction of organic-N that is converted into inorganic-N in the soil, agro-ecological zone dependent). This is computed using equations presented in Tables 5.1 and 5.2. This information is incorporated in the SARA model.

The variation in crop target yield and associated nutrient requirements between rain-fed and irrigated farming systems and between agro-ecological zones indicates the need for sludge upper limits for specific farming systems and agro-ecological zones. These guidelines would promote optimal crop production while protecting the environment. Since the amount of sludge that should be applied to satisfy crop N requirement is influenced largely by the sludge organic-N content, the upper sludge limit should be adjusted accordingly. For instance, the upper sludge limit could be set at the rate that satisfies the optimum fertilizer requirement of a crop. In this case, the contribution from the soil (Soil_N) is excluded from Equation 5.1.

5.4 Conclusion

The synchronization of sludge application rates with farming system crop nutrient requirements and agro-ecological zones should be the way forward for resilient beneficial agricultural use of treated wastewater sludge. In all South African agro-ecological zones, irrigated systems had higher crop nutrient requirements than rain-fed. Under rain-fed cropping systems, areas that receive higher rainfall have higher crop nutrient requirements. Similarly, sludge decomposition rate is influenced by the availability of soil water and the level of soil temperature. Sludge decomposition rate is slowed down under low soil moisture conditions, low soil temperature, very high soil temperature, and when soil pores are saturated with water. Henceforth, sludge recommendation rates should be able to integrate both the crop nutrient requirements and sludge nutrient release rate. In this report, computer model assisted sludge nitrogen release rates for five of the six South African agro-ecological zones were established to fine-tune South African Sludge Guidelines, volume 2 (2006). It is of utmost importance to conduct field validation of these findings on selected sites across South African agro-ecological zones.

REFERENCES

ABDUL QADOS AMS (2011) Effect of salt stress on plant growth and metabolism of bean plant *Vicia faba* (L.) *Journal of the Saudi Society of Agricultural Science.* **10(1)** 7-15.

ALEXANDRA B & JOSÉ B (2005) Organic matter decomposition and the soil food web. In: The importance of soil organic matter. *FAO Soils Bulletin*. **80**. 5-8, FAO, Rome, Italy.

AL-GHUSAIN I, HAMODA MF & EL-GHANY MA (2002) Nitrogen transformations during aerobic/anoxic sludge digestion. *Bioresource Technology.* **85(2)** 147-154.

AL-MALACK MH, ABUZAID NS, BUKHARI AA & ESSA MH (2002) Characterization, utilization, and disposal of municipal sludge: The state of-the-art. *Arabian Journal for Science and Engineering.* **27(1)** 3-27.

BALATSKY AV, BALATSKY GI & BORYSOV SS (2015) Resource demand growth and sustainability due to increased world consumption. *Sustainability* **7(3)** 3430-3440.

BANEGAS V, MORENO JL, MORENO JI, GARCÍA C, LEÓN G & HERNÁNDEZ T (2007) Composting anaerobic and aerobic sewage sludge using two proportions of sawdust. *Waste Management.* **27(10)** 1317-1327.

BELL M, BARRY G & PU G (2004) Mineralization of N from biosolids and the adequacy of the assumptions in the current NLBAR calculations. *Biosolids Specialty II Conference*, Sydney 2-3 June 2004. CDROM Conference Proceedings ISBN 0-908255-62-4, 2004.

BISHOP, P (1995) *Municipal sewage sludge: management, processing and disposal*. CRC Press, USA.

BRESTERS AR, COULOMB I, DEAK B, MATTER B, SAABYE A, SIPNOSA L, UTVIK, ÅØ, UHRE L & MEOZZ, P (1997) *Sludge treatment and disposal: Management approaches and experiences.* Environmental Issues Series No. 7. European Environment Agency, Copenhagen, Denmark.

CHANG TC, YOU SJ, DAMODAR RA & CHEN YY (2011) Ultrasound pre-treatment step for performance enhancement in an aerobic sludge digestion process. *Journal of the Taiwan Institute of Chemical Engineers.* **42(5)** 801-808.

CLANTON CJ & SLACK DC (1987) Hydraulic properties of soils as affected by surface application of wastewater. *Transactions of the ASAE* **30(3)**, 683-587. American Society of Agricultural and Biological Engineers, St. Joseph, MI, USA.

FEIGIN, A, RAVINA, I & SHALHEVET, J (2012) *Irrigation with treated sewage effluent: Management for environmental protection.* Springer Science & Business Media. Berlin.

FERNÁNDEZ JM, PLAZA C, HERNÁNDEZ D & POLO A (2007) Carbon mineralization in an arid soil amended with thermally-dried and composted sewage sludges. *Geoderma.* **137(3)** 4497-503.

FOUDA S, VON TUCHER S, LICHTI F & SCHMIDHALTER U (2013) Nitrogen availability of various biogas residues applied to ryegrass. *Journal of Plant Nutrition and Soil Science.* **176(4)** 572-584.

FSSA (2007) *Fertilizer handbook*. The Fertilizer Society of South Africa. 6th edition. Beria Printers, South Africa, pp. 99-123.

GILMOURJT, COGGERCG, JACOBSLW, EVANYLOGK & SULLIVANDM (2003) Decomposition and plant available nitrogen in biosolids: Laboratory studies, field studies, and computer simulation. *Journal of Environmental Quality*. **32(4)**1498-1507.

GIROVICH MJ (1996) Biosolids treatment and management: Processes for beneficial use. CRC Press, USA.

GOERING, HK & VAN SOEST, PJ (1970) Forage fibre analysis: Apparatus, reagents, procedures and some applications. *Agricultural Handbook,* **379**, USDA Agriculture Research Service, Washington, DC, USA.

GONZÁLEZ-UBIERNA S, JORGE-MARDOMINGO I, CARAVACA MT, VALVERDE I & CASERMEIRO MA (2013) Sewage sludge application in Mediterranean agricultural soils: Effects of dose on the soil carbon cycle. International Journal of Environmental Research. **7(4)** 945-956.

HAMODA M & GANCZARCZYK J (1980) Control of aerobic digestion by substrate concentration. *Canadian Journal of Civil Engineering.* **7(3)** 456-465.

HANJRA, MA, BLACKWELL, J, CARR, G, ZHANG, F & JACKSON, TM (2012) Wastewater irrigation and environmental health: Implications for water governance and public policy. *International Journal of Hygiene and Environmental Health* **215(3),** 255-269.

JIN B, WILÉN BM & LANT P (2004) Impacts of morphological, physical and chemical properties of sludge flocs on dewaterability of activated sludge. *Chemical Engineering Journal,* **98** 115-126.

KADLEC RH, & WALLACE S (2008) *Treatment wetlands* (2^{nd} ed.), CRC Press, USA.

KUNHIKRISHNAN A, BOLAN NS, MÜLLER K, LAURENSON S, NAIDU R & KIM WI (2012) The influence of wastewater irrigation on the transformation and bioavailability of heavy metal(loid)s in soil. *Advances in Agronomy,* **115,** 215.

LEVY GJ, FINE P, GOLDSTEIN D, AZENKOT A, ZILBERMAN A, CHAZAN A & GRINHUT T (2014) Long term irrigation with treated wastewater (TWW) and soil sodification. *Biosystems Engineering,* **128,** 4-10.

LI Y, PARK SY, and ZHU J (2011) Solid-state anaerobic digestion for methane production from organic waste. *Renewable and Sustainable Energy Reviews,* **15** 821-826.

MALOBANE ME (2014) Using the organic carbon fractions of the Van Soest method to determine compounds responsible for C and N mineralization from sludge-amended soils. MSc Dissertation, University of Pretoria, South Africa, pp. 56–64.

MATTANA S, PETROVICOVA B, LANDI L, GELSEMMINO A, CORTES P, ORTIZ O & RENELLA G (2014) Sewage sludge processing determines its impact on soil microbial community structure and function. *Applied Soil Ecology*. **75** 150-161.

MAYNARD H, OUKI S & WILLIAMS S (1999) Tertiary lagoons: A review of removal mechanisms and performance. *Water Research,* **33** 1-13

MIDDLEBROOKS EJ & PANO A (1983) Nitrogen removal in aerated lagoons. *Water Research,* **17** 1369-1378.

MIKKELSEN LH & KEIDING K (2002) Physico-chemical characteristics of full scale sewage sludges with implications to dewatering. *Water Research,* **36** 2451-2462.

MOTTET A, FRANCOIS E, LATRILLE E, STEYER JP, DÉLÉRIS S, VEDRENNE F & CARRÈRE H (2010) Estimating anaerobic biodegradability indicators for waste activated sludge. *Chemical Engineering Journal.***160(2)** 488-496.

NEYENS E, BAEYENS J & DEWIL R (2004) Advanced sludge treatment affects extracellular polymeric substances to improve activated sludge dewatering. *Journal of Hazardous Materials* **106** 83-92.

NKOA R (2014) Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. *Agronomyfor Sustainable Development.* **34(2)** 473-492.

NOBELA M (2011) Influence of biosolid stability, temperature and water potential on nitrogen mineralisation in biosolid amended soils. University of Pretoria, South Africa, pp. 153-159 (MSc. dissertation).

OGBAZGHI ZM, TESFAMARIAM EH & ANNANDALE JG (2016) Modelling N mineralisation from sludge-amended soils across agro-ecological zones: A case study from South Africa. *Ecological Modelling*. **322** 19-30.

PARNAUDEAU V, NICOLARDOT B, & PAGÈS J (2004) Relevance of organic matter fractions as predictors of wastewater sludge mineralization in soil. *Journal of Environmental Quality*. **33** 1885-1894.

RAMALHO, R. (2013) *Introduction to wastewater treatment processes*.(2nd ed.) Elsevier, USA.

REIMANN D (1989) Thermische schlammkonditionierung [Thermal conditioning of sludge]. *Klärschlammentsorgung*. Düsseldorf: VDI-Bildungswerk**,** 12-26.

REN W, ZHOU Z, JIANG LM, HU D, QIU Z, WEI H & WANG L (2015) A cost-effective method for the treatment of reject water from sludge dewatering process using supernatant from sludge lime stabilization. *Separation and Purification Technology,* **142** 123-128.

RIGBY H, CLARKE BO, PRITCHARD D, MEEHAN B, BESHAH F, SMITH SR & PORTER NA (2016) A critical review of nitrogen mineralization in biosolids-amended soil, the associated fertilizer value for crop production and potential for emissions to the environment. *Science of the Total Environment*. **541** 1310-1338.

ROSEN MJ (2004) *Surfactants and interfacial phenomena*. John Wiley & Sons, New York, NY, USA.

RUIZ-HERNANDO M, MARTINEZ-ELORZA G, LABANDA J & LLORENS J (2013) Dewaterability of sewage sludge by ultrasonic, thermal and chemical treatments. *Chemical Engineering Journal,* **230,** 102-110.

SCHNEITER RW, MIDDLEBROOKS EJ & SLETTEN RS (1982) Lime stabilization and land disposal of cold region wastewater lagoon sludge. *Environment International,* **7,** 207-213.

SCHULZE RE (1997) *South African atlas of agrohydrology and climatology*. WRC Report TT82/96 Water Research Commission, Pretoria, South Africa.

SHEETS JP, YANG L, GE X, WANG Z and LI Y (2015) Beyond land application: Emerging technologies for the treatment and reuse of anaerobically digested agricultural and food waste. *Waste Management.* **44** 94-115.

SHENG GP, YU HQ and LI XY (2010) Extracellular polymeric substances (EPS) of microbial aggregates in biological wastewater treatment systems: a review. *Biotechnology Advances.* **28** 882-894

SMITH S & DURHAM E (2002) Nitrogen release and fertiliser value of thermally-dried biosolids. *Water and Environment Journal.* **16,** 121-126.

SNYMAN H & HERSELMAN J 2006. Guidelines for the utilization and disposal of wastewater sludge, Volume 2: Requirements for the agricultural use of wastewater sludge. WRC Report TT 262/06. Water Research Commission, Pretoria, South Africa.

SPINOSA L & VESILIND PA (2001) *Sludge into biosolids: processing, disposal, utilization.* IWA Publishing, London, UK.

STOLL U (1996) Liquid effluent treatment, sewage sludge management and industrial effluent standards. *Resources, conservation and recycling.* **16** 113-133.

SULLIVAN DM, COGGER CG and BARY AI (2007) *Fertilizing with biosolids*. Oregon State University Extension Service. Corvallis, OR, USA.

TCHOBANOGLOUS G. & BURTON FL (1991) *Wastewater engineering:Treatment, disposal, reuse*. Metcalf and Eddy, McGraw-Hill, New York, NY, USA..Chap. 1-4

TESFAMARIAM EH, ANNANDALE JG, STEYN JM, STIRZAKER RJ & MBAKWE I (2015) Use of the SWB-Sci model for nitrogen management in sludge-amended land. *Agricultural. Water Management.* **152** 262-276.

TUROVSKIY IS & MATHAI P (2006) *Wastewater sludge processing*. John Wiley & Sons, Inc. New York, USA.

UNITED NATIONS POPULATION INFORMATION NETWORK (1994) Population and sustainable development. *International Conference on Population Development*, Cairo, Egypt, 5-13 September 1994. Accessed 13 November 2017. http://www.un.org/popin/icpd/infokit/infokit.eng/6interre.html.

US EPA (1995) *Process design manual: Land application of sewage sludge and domestic septage.* EPA/625/R-95/001. EPA, Washington, DC.

VAXELAIRE J & CÉZAC P (2004) Moisture distribution in activated sludges: A review. *Water Research,* **38(9),** 2215-2230.

VINTEN AJA, MINGELGRIN U & YARON B (1983a) The effect of suspended solids in wastewater on soil hydraulic conductivity. I. Suspended solids labelling method. *Soil Science Society of America Journal.* **47(3)**, 402-407.

VINTEN AJA, MINGELGRIN U & YARON B (1983b) The effect of suspended solids in wastewater on soil hydraulic conductivity. II. Vertical distribution of suspended solids. *Soil Science Society of America Journal.* **47(3)**, 408- 412.

WANG LF, HE DQ, TONG ZH, LI WW & YU HQ (2014) Characterization of dewatering process of activated sludge assisted by cationic surfactants. *Biochemical Engineering Journal,* **91,** 174-178.

WANG LK, LI Y, SHAMMAS NK & SAKELLAROPOULOS GP eds. (2007) Biosolids treatment processes, in *Drying beds: Handbook of environmental engineering*. Volume 6, 403-429, Humana Press, Totowa, NJ, USA.

WU CC HUANG C & LEE D (1998) Bound water content and water binding strength on sludge flocs. *Water Research,* **32** 900-904

YOSHIDA H, NIELSEN MP, SCHEUTZ C, JENSEN LS, CHRISTENSEN TH, NIELSEN S & BRUUN S (2015) Effects of sewage sludge stabilization on fertilizer value and greenhouse gas emissions after soil application. *Acta Agriculturae Scandinavica. Section B—Soil & Plant Science* **65** 506-516.

ZHAO L, WANG XY, GU WM, SHAO LM & HE PJ (2011) Distribution of C and N in soluble fractionations for characterizing the respective biodegradation of sludge and bulking agents. *Bioresource Technology.* **102(22)** 10745-10749.

ZUPANČIČ GD & ROŠ M (2008) Aerobic and two-stage anaerobic–aerobic sludge digestion with pure oxygen and air aeration. *Bioresource Technology.* **99** 100-109.

9781431209972