INVESTIGATING THE POTENTIAL OF DEEP ROW ENTRENCHMENT OF PIT LATRINE AND WASTE WATER SLUDGES FOR FORESTRY AND LAND REHABILITATION PURPOSES

by

David Still[1], Bobbie Louton[1] , Babatunde Bakare[2], Craig Taylor[2], Kitty Foxon[2] and Simon Lorentz[2]

[1]Partners in Development
[2]University of KwaZulu Natal

ISBN

August 2012

Obtainable from:

Water Research Commission
Private Bag X03
Gezina
0031

The publication of this report emanates from a project titled: Investigating the potential of deep row entrenchment of pit latrine and waste water treatment works sludge for forestry and land rehabilitation purposes (WRC Project No. K5/1829).

EXECUTIVE SUMMARY

Introduction

While South Africa struggles to meet its sanitation backlogs, a substantial amount of existing basic sanitation infrastructure, which includes both conventional pit latrines and ventilated improved pit latrines (VIPs), has reached or is reaching the end of its design life. Urgent interventions are required to deal with the escalating accumulation of sludge in these basic units. The options for disposal of this sludge are few. Because of the concentrated nature of pit sludge, only a limited quantity can be processed at a wastewater treatment works (WWTW) before COD and solids overload of the system causes process failure. The 1.5 cubic metres of sludge removed from a typical pit is the equivalent of 560 kℓ of typical sewage in terms of nitrogen (measured as TKN) and 1064 kℓ of sewage wastewater in terms of total suspended solids (TSS).

Not only is the build-up of sludge in on-site sanitation systems reaching a critical stage, but the accumulation of wastewater sludge at wastewater treatment works (WWTW) is now becoming a major problem in some metropolitan municipalities and many of the district municipalities of South Africa. The number of landfills prepared to accept sludge is diminishing and disposal costs at landfills are exorbitant. Furthermore, the coastal WWTWs that currently discharge sludge to sea outfalls are anticipating curtailment of this option by the Department of Water Affairs in accordance with changing international standards.

South Africa's new *Guidelines for the Utilisation and Disposal of Wastewater Sludge* encourage sludge management options that include recovering energy, recycling the nutrients or synthesising commercial products from the sludge. Disposal without beneficiation is to be considered the last resort: when the sludge quality is of a high enough standard to be used beneficially, the sludge producer must prove to the authorities which beneficial use options were investigated and why they were not feasible, before disposal will be allowed.

The potential benefits of the nutrients (nitrogen, phosphorus and potassium as well as micro nutrients) and the high organic carbon content of sludge have been well demonstrated. Sludge is utilised for agricultural practices in many countries, and the burial of faeces and other household waste as a soil management system appears to have been used as early as the period 5 000 BC - 1450 AD by inhabitants of the Amazon Basin, where the technique produced a deep nutrient and organism rich soil which continues to have the capacity to support intensive agriculture and a high population density. This is in contrast to the poor, leached soils adjacent to the area which are typical of the equatorial regions of the world. The South African climate causes rapid decomposition of organic matter which has contributed to the deterioration of the physical properties of cultivated soil. Improving these properties through the introduction of organic carbon present in sludge would yield a number of benefits in the South African context, including reducing reliance on expensive fertilisers derived from scare phosphorus reserves and increasing food security when applied at the subsistence level.

The utilisation of sludge must, however, be managed using stringent safety measures in order to prevent the uptake of harmful chemicals by humans or animals, contamination of the environment or the potential of exposure of humans to the pathogens found in sludge. Surface application of sludge on land for agriculture therefore requires that the sludge is first treated to destroy pathogens and achieve stability. Other options currently in use are composting the sludge with other organic material or mixing or covering the sludge with soil in natural veld or tree plantations. In the Washington, D.C./Baltimore, Maryland region of the USA, a method of entrenching sludge has been used since the 1980s which overcomes the need for treatment. Sludge is buried in deep rows, covered with soil and planted with trees. This method provides adequate management of odour and the risk of disease transmission and can be used to benefit forestry, wildlife habitats or assist in the reclamation of land.

Project scope and methodology

This study investigated the application of the deep row entrenchment method under South African conditions with the aim of establishing whether the use of pit latrine and wastewater sludges as a fertilizer substitute for the agro-forestry sector can be recommended and thereby also establishing a safe and beneficial technique for the disposal of faecal and wastewater sludges.

The development of the deep row entrenchment method in the 1970s and subsequent studies were reviewed as were the South African guidelines for the utilisation of wastewater sludge. No guidelines currently exist for the classification or utilisation of pit latrine sludge. Pit latrine sludge was buried at different loading rates on a sandy site at a disused oxidation pond in Umlazi, south of Durban, and wastewater sludge from the Howick Wastewater Treatment Works was entrenched at a Sappi research site near Howick. The sites were characterised and monitored over time in order to document the movement of nutrients out of the trenches and changes in groundwater over time. The HYDRUS-2D was used to model the movement of soil-water, phosphorus and nitrate. The fate of pathogens over time was also investigated. Tree growth was monitored in order to document differences in growth rates between trees provided with different loading rates of sludge and control groups. Two smaller studies were also conducted to investigate the impact of sludge on tree growth under controlled conditions and the application of deep row entrenchment for on-site sludge disposal.

Key findings

In all trials undertaken in this study, trees grown on sludge showed significantly greater growth than trees grown on native soil, with the exception of wattle trees in one study (where the difference was not significant), which may be attributable to the ability of wattle on poor soil to fix nitrogen from the atmosphere. Eucalyptus grown on pit latrine sludge and wastewater treatment sludge showed similar benefit over control trees early in the study (75% increase in diameter on pit latrine sludge and 73% on wastewater treatment sludge); however, by the end of the study the margins had decreased significantly. In addition while trees planted on trenches containing more wastewater treatment sludge initially showed faster growth than trees grown on trenches containing less sludge, this distinction later disappeared. It is unknown whether both of these trends would continue throughout the remainder of the growth cycles of the trees or whether further changes in the entrenched sludge would alter these trends. However total biomass is proportional to the cube of diameter and after 25 months of growth the trees grown over the sludge were still more than 60% larger than the control trees.

In a controlled 26 week study using constructed tree towers, eucalyptus trees grown over a core of pit latrine sludge showed nearly 4 times greater above ground biomass than the control group (which were given regular doses of liquid fertilizer), with mean leaf area 6.5 times greater, foliar nitrogen concentrations nearly 3 times greater and foliar phosphorus double that of the control group. In the large scale trials nutrient concentrations in leaves and wood were not significantly enhanced in trees grown on wastewater sludge at approximately two years, however.

Nitrate, phosphorus and pH fluctuations in the downstream monitoring boreholes remained within acceptable ranges throughout the study, despite the volumes of sludge buried being significantly in excess of agronomic rates.

The HYDRUS-2D model was found to provide meaningful predictions of soil-water, phosphorus and nitrate movement at the study sites.

Values for volatile solids, COD and moisture decreased rapidly during the first year of entrenchment for pit sludge, with slower degradation indicated thereafter. All analytes achieved lower values in the trench than were measured at the bottom (most degraded) layer of sludge in a pit, indicating that more complete degradation of sludge can be achieved under entrenchment over three years than can be achieved in pits even over a period of many years, possibly due to the action of soil fungi.

While a significant number of helminth ova were found in freshly exhumed pit latrine sludge, after 2.8 years of entrenchment less than 0.1% were found to be potentially infective (containing underdeveloped or non-motile larvae) and none of the eggs contained motile larvae. If sludge is entrenched without contamination of the surface soil occurring, it provides a safe means to contain pathogens and thus represents a utilisation option that is appropriate even for untreated sludges or VIP sludge.

Recommendations

For municipalities in South Africa, the deep row entrenchment method opens up a range of possibilities for the disposal of both wastewater and pit latrine sludge, overcoming the problems associated with the stabilisation of sludges, while providing benefits to non-edible crops and to soil. Potential risks to the environment or public health can be managed effectively with periodic monitoring of groundwater and soil. Partnerships between municipalities and forestry could provide mutual benefit to both, with sludge handled, applied and monitored by forestry companies on their own land or with sludge entrenched and monitored by municipalities on municipal land with a forestry company contracted to manage a timber crop on the entrenchment site. On site entrenchment of pit latrine sludge at the household level accompanied by planting of trees where space is available overcomes the difficulties and costs associated with the transport, treatment and disposal of sludge. Entrenchment on small decentralised plots is another option which may prove useful to municipalities. A guideline prepared in consultation with local experts is provided in Appendix A to facilitate implementation of the deep row method; this is based on South African legislation and regulations, the recommendations of studies conducted in the US over the past 40 years and the knowledge gained from this research.

Further study of the impact of deep row entrenchment of sludge would allow the technique to be optimized in terms of benefits to a fruit or timber crop grown on the sludge and improving the cost-effectiveness of entrenchment techniques. In addition, a better understanding of the changes in and movement of sludge would enable minimum criteria for monitoring to be developed to ensure that risks are responsibly managed without incurring undue costs. As further study provides a more complete understanding of these variables over time, the guidelines provided in this document can be developed into an authoritative regulation adopted at the national level to facilitate broader application of this technique. To achieve this, the following areas warrant further study:

- The impact of different sludge loading rates over a full growth cycle on crop growth and nutrient concentrations

- The fate of nitrogen at an entrenchment site due to a better understanding of the processes of leaching and denitrification
- The movement and concentration of nutrients through the soil profile and groundwater over time and with consideration for variable soils, slopes and other hydro geological conditions
- The impact of sludge on soil characteristics in terms of moisture retention, soil conditioning and the long-term rehabilitation of poor soils
- The role of soil fungi in degradation of sludge
- The impact of entrenchment on pathogens and the timeframe in which total deactivation can be assumed

In addition, further work is needed to develop models for municipalities to implement this method cost effectively across a range of conditions.

ACKNOWLEDGEMENTS

The project team thanks the following organizations and individuals for their assistance and collaboration through the course of this project.

eThekwini Metropolitan Municipality
Sappi (Pty) Ltd
Umgeni Water
Hans Salisbury, University of KwaZulu Natal
Babatunde Bakare, University of KwaZulu Natal
Professor Norman Pammenter, Biological Sciences, University of KwaZulu Natal
Dr Nicola Rodda, Biological Sciences, University of KwaZulu Natal

Members of the project reference group:

Jay Bhagwan	Water Research Commission (Chairman)
Jeff Hughes	University of KwaZulu-Natal
Gordon Borrain	Umgeni Water
John Annandale	University of Pretoria
Eyob Tesfamariam	University of Pretoria
John Harrison	eThekwini Water and Sanitation
Cyprian Mazubane	Department of Human Settlements
Iris Lebepe	Department of Human Settlements
Ian Pearson	WATSUP Development
Richard Holden	TCTA
Speedy Moodliar	eThekwini Water and Sanitation

TABLE OF CONTENTS

FIGURES

TABLES

1. INTRODUCTION

The land application of waste water treatment works (WWTW) sludge is a potential alternative to landfills which have increasingly high costs of construction and operation. The method has been applied by ERCO, a company in Maryland in the USA (in conjunction with the University of Maryland) that has functioned under a research permit from the Maryland Department of Environment since 1983 to use the deep row entrenchment technique (Kays et al., 2000).

Deep row applications of sludge for forestry production has been reported to have the potential to solve many of the problems associated with agricultural land application and other land disposal methods. Surface sludge application has been shown to provide plant nutrients, enhance soil productivity, and improve soil properties (WRC, 1997); however, offensive odours and perceptions of health and environmental problems may result in application restrictions. Deep row entrenchment involves placement of sludge into trenches that are immediately covered with soil, eliminating odour problems and safely containing human pathogens in the sludge. The area can then be planted with trees that have high nutrient consumption rates (e.g. eucalyptus). The major environmental concern with deep row entrenchment is the potential for contamination of groundwater by nitrate (NO_3) and phosphorus (P) leached from sludge.

In addition to the management of risks, this study explored the potential for pit latrine and wastewater sludges as sources of fertilizing nutrients which would become available at slow rates as the sludge degrades.

Building on the Maryland studies, two primary studies were conducted to test the entrenchment of pit latrine and wastewater sludge under the geophysical conditions found in South Africa:

- Approximately 1200m^3 of VIP sludge was entrenched at a site owned by the eThekwini Metro Municipality and planted with approximately 1400 trees.
- 360m^3 of wastewater sludge was entrenched at a site owned by Sappi in Howick and planted with approximately 3000 trees

Characterisations of the sites and collection of baseline data, installation of monitoring equipment and regular monitoring and sampling were conducted in order to study four key variables:

- Impact of entrenched sludge on the environment (primarily leaching of pollutants)
- Impact of entrenched sludge on the timber crop
- Impact of entrenchment on sludge (degradation processes)
- Impact of entrenchment on the survival pathogens in the sludge

Parameters and conditions taken from the sites were used to generate simulations which were compared with observed data in order to model and predict trends over time.

Two smaller trials also investigated aspects of the entrenchment of VIP sludge. Twelve trees were grown in containers with a sludge core, while another twelve trees were grown with no sludge, to observe the impact of sludge on growth and nutrient levels. At two households, pit sludge was entrenched on site and planted with fruit trees, to explore the potential of on-site entrenchment to enhance food security.

Existing guidelines deal only with the treatment, use and disposal of WWTW sludge and do not address the deep row entrenchment technique or the management of pit latrine sludge. While existing legislation and guidelines must guide the application of this technique in South Africa, the goal of this research is to contribute to the further development of guidelines and protocols for the management of both pit latrine and WWTW sludge in South Africa, specifically in terms of deep row entrenchment. Accordingly, guidelines specific to the entrenchment of pit latrine and WWTW sludge were prepared to aid municipalities and industry in the implementation of this method with proper management of risks to the environment and public health.

2. EXISTING FRAMEWORK FOR THE LAND APPLICATION OF SLUDGE

2.1 Legislation and regulations

The disposal of WWTP sludge is regulated and controlled by the Department of Water Affairs in terms of the National Water Act (Act 36 of 1998) and the Environment Conservation Act (Act 73 of 1989). Other legislation which influences the use and disposal of sludge includes:

- The Fertilisers, Farm Feeds, Agricultural Remedies and Stock Remedies Act (Act 36 of 1947)
- The Conservation of Agricultural Resources Act (Act 43 of 1983) (CARA)
- The National Health Act (Act 61 of 2003) (HA)
- The Water Services Act (Act 108 of 1997) (WSA)
- The National Environmental Management Act (Act 107 of 1998) (NEMA)

(Snyman and Herselman, 2006a)

In 1998, the Department of Water Affairs and the Department of Environmental Affairs and Tourism developed guidelines and regulations to control the disposal of waste:

- Minimum Requirements for the Handling, Classification and Disposal of Hazardous Waste
- Minimum Requirements for Waste Disposal by Landfill
- Minimum Requirements for the Monitoring of Water Quality at Waste Management Facilities

In line with the resolution of the World Summit on Sustainable Development held in South Africa in 2003, a series of *Guidelines for the Utilisation and Disposal of Wastewater Sludge* was drafted during 2006-2009 to support the appropriate and sustainable use of wastewater sludge. The guidelines stress that by utilising the energy or nutrients available in sludge, sludge management may contribute to sustainable development:

- Volume 1: Selection of management options
- Volume 2: Requirements for the agricultural use of sludge
- Volume 3: Requirements for the on-site and off-site disposal of sludge
- Volume 4: Requirements for the beneficial use of sludge at high loading rates
- Volume 5: Requirements for thermal sludge management practices and for commercial products containing sludge

These guidelines do not apply to pit latrine sludge or other untreated sludge from dry on-site sanitation systems, which differs in a number of characteristics from secondary sludge which has been treated at a wastewater treatment works. A classification system, as well as guidelines for usage and disposal, is urgently needed for pit latrine sludge as the Ventilated Improved Pit (VIP) latrine is the standard for basic sanitation that has been delivered in South Africa. In the absence of guidelines specifically dealing with pit sludge, the classification system provided for wastewater sludge provides a useful frame of reference as research on the management of pit sludge goes forward.

2.1.1. Classification of waste water sludge

Waste water treatment works sludge is classified according to three parameters: microbiological quality (presence of pathogens), stability and pollutant content.

Microbiological class	A: Unrestricted use	B: General use	C: Limited use
Stability class	1: Stable	2: Partially stabilised	3: Unstable
Pollutant class	a: Minimal restriction	b: Moderate restriction	c: High restriction

Sludge with an ***A1a*** classification will have the least restrictions applied to its usage. A sludge heavily contaminated with pathogens, with no stabilisation or vector attraction reduction and heavily contaminated with pollutants will be classified as ***C3c***. Sludge may be reclassified after it has been treated to stabilise the solid content or destroy pathogens, or if action is taken to reduce the attraction of vectors. Vectors are animals or insects (e.g. rates, flies, mosquitoes) which are attracted by the odour of unstable sludge and may then spread disease by carrying pathogens away from the sludge with them.

2.1.1.1 Microbiological class

In order to determine the microbiological class of sludge, three independent samples must be subjected to tests of pathogen or pathogen indicator content. The level of restriction of the use of the sludge will depend on its classification.

Microbiological class	A: Unrestricted use	B: General use	C: Limited use
Criteria	All three samples comply with the following standard	Two of the three samples comply with the following standard	The sample that failed does not exceed the following standard
Faecal coliforms (CFU/g dry)	< 1 000	< 1 x 10^6	1 x 10^7
Helminth ova (Total viable ova/g dry)	< 0.25 (or one viable ova/4g)	< 1	4

Pathogen levels in pit latrine sludge or untreated sludge can be expected to be too high for the sludge to qualify even for a classification of ***C***. *Ascaris lumbricoides* (large roundworm) is typically used as a marker for parasites because its eggs are very hardy: if treatment leaves no viable Ascaris eggs it can be assumed that all other parasites have been eliminated as well. *Escherichia coli* (E. coli) and salmonella are used as a marker for viruses and bacteria.

A number of thermal and other treatment processes can destroy pathogens adequately to produce a sludge which can be used without restriction from a microbiological point of view (Romdhana et al, 2009). These include:

- **Pasteurisation.** Sludge is heated by direct steam injection or with a heat exchanger and kept at a temperature of at least 63°C for at least 30 minutes (or at 91°C for 10 minutes). Tyndallisation is a process which involves repeated pasteurization at 30–40°C at 12 to 24 hour intervals.

- **Ionizing radiation.** Sludge is exposed to a source of radiant energy with high penetration power.

- **Chemical stabilisation.** Lime, chlorine, quaternary ammonium, ozone and other chemicals can be used to disinfect sludge but may not be effective for all parasites. This can be combined with raising the pH of the sludge above 11 or reducing it below 3, which has a toxic effect on most microorganisms. High amounts of lime, or low amounts of lime combined with anoxic storage, raises the pH and temperature of the sludge sufficiently to destroy pathogens.

- **Heat conditioning–dewatering.** Temperatures of 140–240°C are maintained for a period of 15–60 minutes while pressuring sludge at 17–27 bars.

- **Low frequency ultrasound** can be combined with heat to deactivate pathogens; the addition of anti-microbial agents may be required to achieve a Class A product.

- **Extreme high temperatures.** Heating processes such as incineration, vapour-gasification and pyrolysis raise the sludge temperature to above 500°C, structurally destroying pathogens.

- **Heat drying.** The application of direct or indirect heat kills pathogens and can produce a very dry material, eliminating the need for further vector attraction reduction before application as a fertilizer or soil conditioner. Thermal drying processes include fry drying, solar drying, drum drying and agitated conductive drying.

- **Digestion.** In autothermal thermophilic aerobic digestion (ATAD) systems, the sludge reaches temperatures of 55 – 65°C while it is aerated over a period time, effectively killing pathogens (Piterina, Bartlett and Pembroke, 2010). Composting occurs through essentially the same process; however as faecal sludge only has a carbon:nitrogen ratio of approximately 6:1, it is necessary to add carbonaceous material, such as domestic or garden waste, to achieve optimal conditions for composting -- approximately a 30:1 ratio. Composting produces a nutrient-rich humus which can be used beneficially as a fertiliser and soil conditioner. Mesophilic anaerobic digestion, where temperatures of approximately 35◦C are reached under conditions where oxygen is not present (usually wet conditions), also reduces pathogens significantly but to a lesser extent. (Berg and Berman, 1980)

Management options and restrictions

The following options and restrictions apply to class ***C*** wastewater sludge:

- Agricultural use at agronomic[1] rates is permitted if stability class 1 or 2 is achieved. Vegetables shall not be grown which are consumed raw, that touch or are below the soil/sludge mixture or have harvested parts below the soil surface. Crops whose edible parts do not touch the soil/sludge mixture, shall not be harvested until 90 days after the last sludge application. Animals shall not be grazed on land until 90 days after the last sludge application. Notably, there are no restrictions on the production of industrial crops (eg. timber).

- For beneficial use other than agricultural use at agronomic rates, care should be taken not to expose the public and workers to pathogens. Access to land with low public exposure (private farmland) is restricted for 90 days after the last sludge application.

- Saleable products such as fertiliser may not be produced because the risk of infection is unacceptable.

- If for some reason no beneficial use of sludge is possible, it can be co-composted with general waste at a landfill following the protocols outlined in Volume 3 of the *Guidelines for the Utilisation and Disposal of Wastewater Sludge* (Herselman and Snyman, 2009).

2.1.1.2 Stability class

Stability refers to the fraction of sludge which is still degradable. Unstable sludge, which still has a large component of organic matter, attracts organisms such as rodents, flies and mosquitoes. These organisms operate as vectors for disease transmission as they are attracted to unstable sludge and then potentially carry away with them pathogens present in the sludge. Unstable sludge also has a strong and unpleasant odour relative to more stable sludge. Odour is a critical factor in terms of public acceptance of sludge application within a community, which is a further reason why restrictions must be placed on the use of unstable sludge. Unstable sludge can be reclassified more favourably if one of the options for reducing the attraction of vectors can be applied.

[1] The *agronomic* rate is the rate at which the nutrients in the sludge are used by the crop on an annualised basis.

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

Measuring the chemical oxygen demand (COD) provides an indication of how well stabilised the sludge is. COD indicates the amount of oxygen required to oxidize all the organic matter into carbon dioxide (CO_2). In contrast to wastewater sludge, which is relatively uniform in terms of degradation, sludge removed from a pit latrine that has recently been in use can be expected to contain sludge at various stages of stabilization that are mixed together. Variables such as moisture, temperature, characteristics of the surrounding soil and the presence of rubbish in the pit affect the rate of degradation.

Management options and restrictions

- For wastewater sludge, stability class ***1*** sludge may be used for agriculture at agronomic rates. Other beneficial options may be used with the application of one vector reduction option and saleable products are permitted if the long term stability of the sludge is ensured.

- Partially stabilised class ***2*** sludge may be applied for agricultural or other beneficial uses with vector reduction and careful management of odour or other characteristics which may create a nuisance. Saleable products are permitted if the sludge will stabilise further over time.

- Unstable class ***3*** sludge may not be used for agriculture or saleable products. At least one vector reduction application must be applied before consideration of any land application. High rate application of this sludge could cause long-term effects and source control should be implemented.

2.1.1.3 Pollutant class

Pollution class is determined from a full total elemental analysis of the sludge. Elements typically of concern for their potential toxicity are listed below; however, this list is not exhaustive.

Pollutant class mg/kg	a	b	c
Arsenic (As)	• <40	• 40 - 75	• >75
Cadmium (Cd)	<40	40 - 85	>85
Chromium (Cr)	<1200	1200 - 3000	>3000
Copper (Cu)	<1500	1500 - 4300	>4300
Lead (Pb)	<300	300 - 840	>840
Mercury (Hg)	<15	15 - 55	>55
Nickel (Ni)	<420	420	>420
Zinc (Zn)	• <2800	• 2800 - 7500	• >7500

Sludge which comes exclusively from domestic sources, such as pit latrines, is not considered a risk for pollutants as is sludge which may include industrial waste. Sludge which has been treated at a wastewater treatment works must undergo a pollutant analysis to determine its classification.

Management options and restrictions

- Class ***a*** sludge may be applied to land as long as the agronomic rate is not exceeded. Some restrictions apply to producing edible goods. Because the sludge is of a high quality which can be used beneficially, non-beneficial disposal options are not permitted without demonstrating to the authorities why the sludge could not be used beneficially.

- Class ***b*** sludge can be applied to land if the combined metal content of the sludge and the receiving soil fall within permitted ranges. Details for determining maximum thresholds are provided in Volume 2 of the Sludge Management Guidelines. Some saleable products are restricted. It is recommended that class ***b*** sludge be limed at 25kg lime/tonne wet sludge to immobilise metals, after which the sludge should be retested.

- Class ***c*** sludge may not be used in agriculture. Stricter control should be implemented at the origin of the pollutants in order to improve the classification of sludge in the future.

2.2 Management options and limitations for pit sludge and waste water sludge

The threat of disease transmission is the most significant restriction on the use of untreated sludges and consequently they should not be used to produce saleable products without undergoing treatment to destroy pathogens. They could, however, be applied for agricultural or other beneficial purposes if one vector reduction option is applied (such as entrenchment) and if management practices ensure that requirements are met regarding odour control and that workers and the public are protected from contact with pathogens. In addition, soil analysis would be required to assess whether the receiving soil could accommodate the pollutant load of class ***b*** wastewater sludge, and class ***c*** wastewater sludge would be disqualified from land application.

While no system is in place for classifying pit sludge, it can be expected to contain infectious pathogens and be at least partially unstable but not pose a significant threat in terms of pollutants. Even if it was stabilized and sterilised, pit latrine sludge would typically not be appropriate for surface or shallow incorporation methods for agriculture due to the rubbish content. Entrenchment, however, contains pathogens and odours and is feasible even when sludge contains rubbish.

2.2.1 Agriculture and other beneficial options

According to the Fertilisers, Farm Feeds, Agricultural Remedies and Stock Remedies Act (Act 36 of 1947) a fertiliser can only be classified as an organic fertiliser if it contains less than 20% ash and 40% water. Sludge does not comply with these criteria and therefore cannot formally be classified as an organic fertiliser. Sludge does, however, offer a number of significant benefits to agriculture as it:

- contains major plant nutrients (calcium, magnesium, potassium, phosphorus, nitrogen)
- contains some essential micronutrients (zinc, copper, molybdenum and manganese)
- improves physical properties of soil (better soil structure, increased water retention capacity and improved soil water transmission)

Table 2.1: Typical nutrient content of dry wastewater sludge in South Africa (Snyman & Herselman, 2006b)

Nutrient	Range
Total N	3.2 - 4.5 %
Total P	1.5 - 1.7 %
Total K	0.2 - 0.3 %
Organic content	40 - 70 %

If wastewater sludge has been treated to eliminate pathogens, a wide range of agricultural applications are possible. In Washington State in the USA, for example, King County's Biosolids Program has been active since the early 1970s developing a range of products from sludge for surface application; for some of its programmes demand for sludge even exceeds supply (King County, 2009). Composted sludge mixed with sawdust is sold commercially as compost; sludge treated with heat digestion is applied to wheat and other crops in partnership with local landowners; other programmes involve the cultivation of fruit trees, enhancing pasture land and wildlife habitat, and improving forestry products. Durban has developed a pelletizing plant which pasteurises and pelletises the sludge from its VIP toilets and is currently exploring the commercial potential of this sludge as a plant fertiliser. Pelletising is an option which is being implemented increasingly by cities around the world. One example is the city of Boston in the United States, where centuries of discharge of raw sewage has damaged the harbour ecosystem and alternatives such as deep sea dumping, incineration and land filling have not been acceptable (DeCocq, Gray and Churchill, 1998).

In South Africa, a number of requirements apply to wastewater sludge utilised for agriculture:

- Sludge must be stored in a properly designed facility that minimises the impact on the environment and must be applied as soon as possible or, alternatively, adequate precautions must be taken not to generate odours or attract vectors.
- Sludge should be applied at rates that do not exceed the plant nutrient requirements (agronomic rates). An application rate of 120 tonnes dry mass per hectare per year may not be exceeded for industrial crops and 60 tonnes for crops to be used for animal feed. If this is not sufficient for a specific crop's requirements, a motivation can be submitted to the Department of Water Affairs for consideration on the basis that it can be accommodated with existing pollutant levels in the receiving soil.
- The sludge application method must not contribute to soil erosion.
- To protect groundwater and surface water from pollution, buffer zones of at least 5m should be maintained from an aquifer and 200m from surface water boreholes. This requirement may be relaxed on condition that proof is provided that the groundwater and surface water is adequately protected.
- Sludge should not be applied to land within 500 m from dwellings.
- The quality of sludge and soil in some cases must be monitored over time.
- The sludge producer and sludge user are required to keep records.

Other beneficial options for wastewater sludge presented in Volume 4 of the *Guidelines* include (Herselman & Moodley, 2009):

- Using sludge as a landfill cover, with layers of municipal waste alternated with dewatered sludge/compost
- Rehabilitation of disturbed/degraded soils (nutrient depletion, erosion, acidity and salinity, poor physical properties, reduced biological activity) after mining activities, intensive farming and industrial activities
- Rehabilitation of mining waste deposits
- Remediation of contaminated soil
- Establishment of golf courses, race courses, vineyards, road embankments, public parks, natural forests and plantations
- Use of sludge as growth medium for plants, flowers and seedlings
- Instant lawn cultivation

The *Guidelines* also highlight the value of sludge as an energy source. One option for this is the production of biogas from sludge in order to power wastewater treatment works. eThekwini Municipality is an example of a South African Municipality exploring this route. Another energy option that is being explored internationally is processing sludge into "e-fuel" for co-combustion with coal (replacing 10-15% of crushed coal) in order to reduce industrial consumption of fossil fuels.

2.2.2 Sludge application methods

Wastewater sludge can be added to soil as a once-off application in order to improve its capacity to sustain vegetation. If crops require more than three applications over a five year period at rates higher than agronomic rates, this will be classified as continuous sludge application.

2.2.2.1 Surface application

When applied to the soil surface, sludge can be sprayed on as a liquid or applied in a dewatered form. In Sweden, willows (*Salix spp*) grown for biofuel were irrigated with the wastewater extracted during the dewatering of sewage sludge, while 10% of the dewatered sludge was applied to the plantations (Melin, Aronsson and Hasselgren, 2004). In France, the ERESFOR project found significant increases in the diameter of poplar and pine with the application of liquid sludge while forest floor biodiversity was improved by composted sludge and limed sludge (Thomas-Chery et al, 2007). However, Marx, Berry and Kormanik (1995) found that liquid application in plantations of trees less than 8 years old stimulated competing vegetation and decreased tree growth. Lazdiņa, et al (2007) reported similar problems using surface applications of sewage sludge in willow plantations. In King County in Washington State in the USA, surface application of sludge which has been treated by heated digestion to kill pathogens has been used since 1973 for a range of beneficial uses and a number of agricultural applications have been in use in partnership with local farmers (King County, 2009). Marx, Berry and Kormanik (1995) found that liquid sludge sprayed on the surface did improve tree growth at applications of approximately 400 kg nitrogen per hectare but that higher application rates were not more effective. In a research study investigating the usefulness of lime-stabilised secondary sludge for the production of forest seedlings, Lambert et al (1985) found that surface application rates over 45 000 kg of sludge per hectare were detrimental to the plants.

In South Africa, the University of Pretoria has researched the application of wastewater sludge to soil for turfgrass production (Tesfamariam, Annandale, Steyn and Stirzaker, 2009). Turfgrass was grown on an 8 month rotation and when harvested the sludge was removed with the turf and new sludge was applied for the next growth cycle. This allowed a large quantity of sludge to be exported through the production of turfgrass, providing a means of consumption of municipal sludge, reducing the vulnerability of turf during harvest, transfer and planting and continuing to provide a slow release source of nutrients to the crop after replanting (Muse et al 1991 in Tesfamariam, Annandale, Steyn and Stirzaker, 2009). For the turfgrass study, application rates of 8, 33, 67, and 100 megagrams (Mg, equal to metric tonnes) of sludge per hectare were trialled using class ***A1a*** wastewater sludge (see Section 2.1.1; Tesfamariam, Annandale, Steyn and Stirzaker, 2009). At rates up to 67 Mg/ha the crop showed improved establishment rates and colour. The cohesiveness of sod during handling and transport improved at rates of 8 Mg/ha and 33 Mg/ha but deteriorated at higher rates. By applying sludge at a rate of 100 Mg/ha the removal of native soil at harvest could be avoided, but unacceptably high levels of nitrogen appeared in the leachate at this application rate. Over the two year period of the trial, the mass balance of nitrogen was preserved in the top meter at a rate of 67 Mg/ha, with total nitrogen decreasing over time at lower rates. Significant leaching to the lower 450mm to 750mm occurred at the 67 Mg/ha rate, however, while lower application rates produced leachate concentration below the South African drinking water standard limits. Mass balance of phosphorus was preserved at application rates of 33 Mg/ha, with total phosphorus decreasing at lower rates and accumulating – mainly in the top 150 mm – for higher application rates. Researchers therefore recommended an optimal application rate of 33 Mg/ha, with monitoring of leachate by wetting front detectors to ensure that it did not exceed 300 mS/m during the initial 2 months after planting.

2.2.2.2 Shallow incorporation

In Australia, surface application of sludge achieved a 30% increase in the growth rates of existing pine plantations, while incorporation of sludge into the soil surface prior to planting improved tree height by up to 50% after 5 years. Tree diameter was increased by 85% and the density of the wood produced was not affected (Polglase and Crowe, 1999). In a study in the United States in 1970s investigating the reclamation of “borrow pits” (areas where the A and B soil horizons have been removed to provide earth fill for the construction of dams, highways or buildings) sludge or fertiliser was disked into the soil surface at a rate of 34 kg/ha dry mass and planted with fescue and loblolly pine (Berry and Marx, cited in Marx et al, 1995). After 3 years, seedling volumes (diameter^2 x height) were 28 times greater on sludge plots than nonsludge plots; at age 18 years the sludge appeared to continue to contribute to enhanced growth, with volume of trees on sludge plots 80 fold that of trees on control plots (Marx et al, 1995). These authors also demonstrated successful rehabilitation of eroded lands in the Tennessee Copper Basin, but found that public fear of sludge and high transportation costs limited the application of sludge.

2.2.2.3 Injection

Another method of sludge application which has not yet been used widely is the injection of sludge beneath the soil surface for agriculture. In 2009, students at the University of Kwa-Zulu Natal, with support from the Water Research Commission, designed a prototype for an apparatus to inject sewage sludge beneath the soil surface in order to enhance sugar cane cultivation (Still and Foxon, 2012). The sludge was liquefied and pumped through a discharge pipe into a channel at least 300 mm below the soil surface, ensuring an adequate barrier between pathogens and pollutants in the sludge and the soil surface and surface water.

The cost of sludge application by this method was estimated at R2480 per hectare per annum (or R124/tonne), while the cost of using inorganic fertilizer in current practice is R2 400/ha. The Waste Water Treatment Works which participated in this study currently disposes of sludge (50% moisture) at a landfill site at a cost of R300/tonne including labour, operations, transport and land. Disposing of sludge through injection would therefore realise significant savings over current practice. Utilisation of sludge by the sugar cane industry could therefore be cost effective for both parties if the treatment works offset the cost to the sugarcane industry.

2.2.2.4 **Deep row entrenchment**

The deep row entrenchment technique pioneered in the United States in the 1970s overcomes many of the limitations encountered with surface application or shallow incorporation. By burying sludge in deep trenches covered with soil and planted with trees, odour is controlled, increasing the ranges of sites which are appropriate for application. If stringent protocols are followed during application of sludge to the site to prevent surface contamination, pathogens are contained at a safe depth below the surface and do not present a risk to humans or animals. This represents a vector reduction, making it allowable for raw sludge that has neither been stabilised nor sterilised to be applied directly to land, which is not allowed with surface application or shallow incorporation. In addition, sludge can be applied at a loading rate high enough to ensure sufficient nutrients for crops for an entire 6-7 year growing cycle, dramatically reducing the costs and risks associated with the more frequent applications required for surface application where high application rates are not possible. The risk of contamination of local streams is reduced as sludge is not present in surface runoff; also because nutrients mineralise slowly under the anaerobic conditions in the trench. Potential contaminants are freed from the sludge at a slow rate, with the surrounding tree roots acting as a nutrient sink - taking up nutrients and reducing the risk of leaching.

The deep row entrenchment technique involves preparing trenches manually or with a backhoe with dimensions typically about 200m long, 600mm wide and 1.2 – 1.5 m deep, with rows spaced 2.4 – 3 m between centres (Kays, Felton, and Flamino, 1999). The depth of the trench is determined by the volume of sludge to be applied. The trench is filled with sludge to within 300 mm of the surface and then backfilled with the overburden heaped. Trees or other vegetation are planted on or between trenches. Variables to consider are trench dimensions, spacing, method of filling (layered with soil or co-composted with vegetable matter), species, composition and density of vegetation and end purpose (Kays et al, 1999).

3. PREVIOUS TRIALS OF DEEP ROW ENTRENCHMENT

3.1 Pioneering of method in Maryland, USA, 1972

In 1971, the Blue Plains wastewater facility which served two million people in the Metropolitan Washington, DC area was treating approximately one million cubic meters of wastewater per day and generating 200-300 tonnes of digested sludge per day (representing 40 to 60 tonnes of solids) (Walker, 1975). With the passage of the Clean Water Act in 1972, the facility was required to reduce the solids in the effluent it discharged. In response, the US Department of Agriculture, in collaboration with the Environmental Protection Agency, the Maryland Environmental Service and a number of other agencies, launched a cooperative effort to find an environmentally acceptable option for sewage sludge disposal that would also benefit soil and crops. Walker (1975) provides comprehensive documentation of the methodology used and the detailed analyses conducted on all aspects of the study.

Sludge was hauled to the site and dumped in trenches dug along the contour using dump trucks and front end loaders. In addition to sludge leaking from the trucks during transit, depositing of sludge at the site resulted in considerable surface contamination and contamination of the wheels and mud flaps of the trucks, which then tracked sludge back onto the roads. Temporary storage pits were dug on site to attempt to alleviate this problem but this gave rise to a new set of problems. The researchers recommended using a cement mixer or pump in order to pipe the sludge directly into the trenches to avoid the risk of contact with the surface.

Three different sludges were used and applied to trenches as follows:

- Dewatered digested sludge (20-25% solids) was applied at 800 dry metric tonnes/hectare to trenches 600 mm wide x 600 mm deep with a 600 mm space between trenches; 1150 dry metric tonnes/hectare were applied to trenches 600 mm wide x 1.2 m deep with a 1.2 m space between trenches.
- Undigested (raw limed) dewatered sludge (18-25% solids) was applied at the same application rates: 800 dry metric tonnes/hectare to the 600 mm deep trenches and 1150 dry metric tonnes/hectare to the 1.2 m trenches.
- Undigested (raw limed) liquid sludge (5-8% solids) was discharged pneumatically from a tanker truck into trenches 600 mm wide x 1.2 m deep with a spacing of 2. 4m–3.2 m between trenches. A wider distance between trenches was used for this application because of the tendency of the sandy soils to collapse into the trenches. In addition, it was found that trenches could only be filled halfway with liquid sludge or it would overflow when the trenches were backfilled; this restricted the application rate to 125 dry metric tonnes/hectare.

Trenches were typically ten percent overfilled after which they were backfilled as the trencher dug the next trench. A variety of surface treatments were trialled: plots were either left in ridges, levelled and disked or levelled, disked and cross-ripped. Levelling left a 280-350 mm soil covering above the sludge and ground surface. This soil covering was limed and either fertilised or treated with digested sludge at 35-60 dry metric tonnes/hectare in order to establish a crop. It was found that while leaving rows ridged until planting reduced erosion, it was difficult to establish ground cover on ridges. Cross ripping at right angles to the trenches did not appear to aid plant growth.

Fescue grass, maize, soybeans and alfalfa, as well as fruit and shade trees, were grown on the site. It was difficult to plant crops over the liquid sludge application because conditions in the rows remained wet and unstable. Crops performed best where sludge was shallower and trenches were closest together, with the poorest growth on the trenches filled with liquid sludge with trenches spaced furthest apart. Initially the growth of maize appeared to be inhibited by insufficient oxygen and excessive ammonia, as well as volatile compounds. This did not impact the fescue grass, however. Irrigation of the site was necessary during the course of the two year trial.

Drain lines and diversion ditches were installed to carry surface and groundwater to a 0.4 hectare drainage pond for monitoring. Forty test wells were drilled to monitor groundwater at and adjacent to the site. Chloride and nitrate levels increased in groundwater, with chloride peaking at 18 months and nitrate a year later; both decreased after peaking but at 4 years neither had returned to background levels (Sikora, 1978). While the sandy soil of the site provided conditions of extreme porosity for testing the movement of the pollutants through the soil, a clay layer beneath the sandy soil may have created a barrier which slowed movement. Movement of nitrate-nitrogen was detected from the 1.2 m raw liquid sludge trench, where close contact with the soil created unfavourable conditions for denitrification and promoted rapid conversion of nitrogen to mobile nitrate (Walker, 1975). In contrast, little nitrogen movement was found from the 1.2 m trench filled with digested sludge, which at 18 months had dewatered less than the shallower trenches or trenches filled with raw sludge. Denitrification occurred most significantly under the 600 mm deep raw limed dewatered sludge, which contained a high percentage of organic material.

Raising the pH of the sludge to over 11.5 by liming dramatically decreased the numbers of salmonella and faecal coliforms present in the sludge (Walker, 1975). However, several months after entrenchment the sludge pH dropped and numbers increased again temporarily, eventually decreasing over time. Salmonella persisted for less than 10 months. Low levels of total and faecal coliforms were still found in the soil more than two years after burial. Movement of faecal coliform or salmonella out of the sludge into the surrounding soil or groundwater was found only a few centimetres from the entrenched sludge, however. The sludge was not monitored for viruses as none had been detected in the sludge during testing prior to burial.

No downward movement of heavy metals was detected. Low concentrations of metals were taken up by plants during the study, although it appeared that these may become increasingly available to plants over time. It appeared that the higher pH in the entrenched sludge may have rendered metals insoluble, while phosphates and organic matter in the sludge may have sequestered metals, limiting movement of metals out of the trench.

3.2 Trial of deep row entrenchment for forestry, Maryland USA, 1983

In the early 1980s ERCO, Inc., a sand and gravel mining company in Maryland, pioneered the use of deep row entrenchment as a means of beneficiation for the forestry industry and for wildlife habitat creation. Hybrid poplar, which has a high nitrogen demand, was planted on land which had been mined for gravel (Kays et al, 1997). Rows 760 mm deep and approximately 1 m wide were prepared approximately 2.5 m apart from their centres. The sludge averaged about 20 percent solids, had a pH of 7.0 – 8.0 and a nitrogen content averaging 3.32 percent before application. In the later part of the study (from 1992) sludge was required by law to be stabilised with lime before application, which reduced the nitrogen to 1.5 – 2.5 percent. Sludge was applied at 460 mm depth to the rows for an application rate of 382 dry tonnes per hectare and a special demonstration plot was treated with 560 mm sludge for a rate of 660 dry tonnes per hectare. The remaining 200-300 mm was filled with soil that had been removed from the trench and treated to maintain a pH of 6.2. The trenches were levelled and disked in preparation for planting.

Stem cuttings from hybrid poplar trees were planted densely over approximately 40 hectares. High density planting was chosen in order to accommodate tree mortality and quickly utilize the sludge. Tree growth remained steady but slow throughout the experiment, rather than dropping significantly after a few years as expected, and diameters were too small for the trees to be of commercial value, possibly because planting density was too high. The reduction of nitrogen levels to 1.5-2.5 percent through mandatory lime stabilization later in the study may have resulted in inadequate nitrogen levels, requiring higher application rates.

Nitrogen mineralization was assessed annually using composite samples from three depths and the utilization and fate of phosphorus was studied (Kays et al, 1997). According to regulations trees could only be harvested once foliar nitrogen dropped below 3.5 percent and total nitrogen mineralization reached approximately 70 percent. When the trees were actively growing, foliar nitrogen measured over 3.5 percent; 6 to 9 years after planting nitrogen levels in the sludge had reduced to 2.3 percent, indicating that much of the sludge had mineralised. Nitrogen levels averaged 0.02 percent only 15 cm either side of the entrenched sludge and 0.04 percent below it. Visual assessments of root growth patterns indicated that at three years after planting roots surrounded the entrenched sludge but did not penetrate it; after six years roots were found throughout the buried sludge with a large root mass around the entrenched sludge appearing to act as a sink, taking up nitrogen as soon as it was mineralized and minimising leaching.

After harvest, the trees were wet chipped. At the beginning of the study chips were used as mulch on site and were mixed into the soil during the next sludge application. Because of the high carbon/nitrogen ratio, woodchips could serve as a sink for any available nitrogen, reducing the risk of nitrogen leaching from the site. (Gouin in Kays, 1999).

While the site had been massively eroded at the beginning of the study, after repeated rotations of deep row entrenchment it was restored to a stable forested habitat with abundant wildlife.

Over 13 years, groundwater sampled from 7 monitoring wells showed no elevation in levels of nutrients, pH, metals, or pathogens. The clay soil beneath the sludge may have assisted in preventing movement of potential pollutants through the soil.

From 2001, the University of Maryland and the Washington Suburban Sanitary Commission partnered with ERCO to investigate in greater depth the effect of the planting density and sludge application rates on water quality, plant growth and economic viability described by Kays et al (2007). In 2002, a new study was commenced on a 1.3 ha plot at the ERCO test site. The plot was partitioned into three blocks, each of which contained various application rate/tree density combinations. Thirty treatments were designed combining three different tree density rates (0, 717, 1063 trees/ha) with three different sludge application rates (11900, 23800, 35800 kg N/ha); three replicates and three control treatments (no sludge, no trees) were also included in the design. The outer two rows of trees around the perimeter of each treatment were used as buffers and samples were collected only from the innermost 16 trees in each treatment in order to reduce the possibility of edge effects. The three sludge application rates were assigned randomly within each block, while tree densities were not randomised because of logistical considerations.

Groundwater was monitored with a zero-tension pan lysimeter placed 305 mm directly below the bottom of a deep row at the centre of each treatment, suction lysimeters placed at 150 mm, 300 mm, and 610 mm below the sludge and at 150 mm and 300 mm from the sides of sludge, and seven groundwater monitoring wells installed at depths ranging from 7.6 m to 39 m. Entrenched sludge did not impact on the groundwater at two years into the study. In fact, no migration of nitrate to groundwater was found at the site throughout the course of the entrenchment studies conducted at the site from 1983 to 2005. Kays et al (2007) found that at these application rates nitrate was not released from the sludge to the environment during the first two years, and while ammonium did leach from the trench, it quickly became bound in the soil and in this study did not travel further than 600mm. After two years it appeared that the trees were just beginning to access the sludge, and foliar nitrogen and phosphorous levels were similar to fertilized trees. Subsoiling prior to planting appeared to provide a number of benefits: shorter crop rotation length might be possible due to saplings establishing themselves more quickly in less compacted soil, lower mortality and higher growth despite more severe browsing by deer. A study of the economic potential of sludge application indicated little gain for the 23,800 kg N/ha rate, but substantial economic gains at higher application rates despite greater costs for application.

3.3 Deep row entrenchment in Philadelphia, US, 2005

Since the late 1970s, the Philadelphia Water Department has used more than 1 million tonnes of sludge for the reclamation of approximately 2 000 hectares of mine lands. The conventional method of application over these three decades for mine reclamation has been the surface application of dewatered sludge at rates of up to 134 dry tonnes/ha using manure spreading equipment followed by plowing to attempt to incorporate the sludge into the top 150-300 mm of soil then planting with eucalyptus and grasses.

In 2005, deep row entrenchment was tested for the reclamation of a coal mine (Toffey, Flamino, Pepperman et al, 2007). An area of 5,6 ha was prepared with two control plots on which no sludge was entrenched and three treatments were tested of 134, 160 and 224 dry tonnes per hectare. Lime was added at approximately 11 tonnes /ha, and the area was planted with hybrid poplars.

By the end of the second growing season, trees planted on the 224 t/ha treatment were on average 2 to 3 times higher and nearly 10 times heavier than the control group. Planting on or between rows yielded the same growth rates. Trees in the 224 t/ha treatment had more extensive root systems, with the largest roots growing in the direction of the sludge. Foliar sampling showed higher nutrient levels with the capacity to utilise additional nitrogen. Monitoring of pathogens showed that die off continued and there was no evidence of regrowth a year after entrenchment. Nitrogen mobilisation out of the trench was slowed by the high demand of the trees and the anaerobic conditions and cooler temperatures which slowed microbial activity in the trench. Researchers calculated that, with generation of $C0_2$ by equipment used in entrenchment considered, approximately one tonne of CO_2 equivalents was sequestered per wet tonne of sludge through the growth of the trees and increase of carbon in the soil.

3.4 Discussion

Utilisation of sludge through surface application, shallow incorporation and deep row entrenchment all demonstrate benefits to vegetation and improvement of soil. Some studies suggest that shallow incorporation yields better results than surface application and deep row entrenchment yields better results than shallow incorporation. Entrenchment has a number of advantages over other methods:

- Far greater quantities of sludge can be applied, reducing the frequency of application and thereby reducing costs and risks of contamination.

- Entrenchment improves the structure and organic content of soil more intensively and, when used to rehabilitate soils, shows the potential to restore the capacity of land to support vegetation and wildlife.

- With surface application, aerobic soil microbes rapidly convert organic nitrogen to ammonia and nitrate nitrogen, releasing most of the nitrogen present in sludge over the first three years. With entrenchment, sludge is held in anaerobic conditions in the trench until tree roots slowly begin to penetrate the sludge and introduce oxygen. Cooler temperatures and lack of oxygen in the trenches slow microbial activity. This results in nutrients being released from the sludge at a slow rate, which both sustains the provision of nutrients to the tree for the duration of a 6-8 year rotation and slows the rate at which nitrates leach into the soil, reducing the risk of surface or groundwater contamination (Toffey 2007), and saving costs in terms of frequency of application.

- As sludge is not available on the surface, it will not promote forest floor vegetation which could inhibit the growth of young trees.

- In terms of safety, while sludge applied by other methods will be restricted based on the stability and microbiological risk of the sludge, deep row entrenchment is applicable for pit sludge and wastewater sludge unless it falls into Pollutant class ***c***.

- In terms of stability, burial of sludge represents a vector reduction placing the sludge out of reach of vectors and eliminating the issues of odours (with the exception of the period during the entrenchment process itself), making deep row entrenchment an option for all stability classes. This can also allow for smaller buffer zones between the area of application and human settlements.

- In terms of microbiological risks, burial also dramatically reduces the risk of contact with pathogens, making deep row entrenchment an option for all microbiological classes. If protocols are followed carefully to prevent contamination of the environs during entrenchment, pathogens which pose a risk to humans and animals will not be present on the surface. No restrictions on animal grazing are therefore needed. Findings of research into pathogen survival after entrenchment at 2,5 years indicate significant die off, suggesting that when workers disturb the site at harvest at 7 years, there will not be a risk of infection.

- In terms of pollutant classes, deep row entrenchment is acceptable for all household sludge and for class ***a*** and ***b*** wastewater sludge if, in combination with the pollutant content of the receiving soil, it does not exceed the maximum burden allowed. The slow release of pollutants held in the sludge when entrenched also assists with controlling the risk of surface or groundwater contamination.

A final consideration is the presence of detritus which is typical of pit sludge in South Africa and has proven highly problematic in both the removal of sludge from pits and its disposal. While the presence of rubbish in sludge would obviously limit its potential for surface application or shallow incorporation, with deep row entrenchment it can simply be buried without being extracted from the sludge with no harm to crops.

Although the research that has been conducted to date on deep row entrenchment has shown positive results, there is need to investigate how the sludge entrenchment technique may be applied under South African conditions and what safe working procedures (handling and transport, maximum application rates, etc.) should be developed to protect the health of workers, local communities and the environment. While other trials have focussed exclusively on the entrenchment of treated sludges from waste water treatment works, in the South African context where the VIP pit latrine is the recommended minimum norm for basic sanitation provision it was vital to investigate deep row entrenchment as an option for the beneficial use of pit latrine sludge as well.

4. THEORETICAL BASIS FOR DEEP ROW ENTRENCHMENT TRIALS IN SOUTH AFRICA

The trials conducted in this study aimed to investigate the applicability of deep row entrenchment in the South African context in terms of the following variables:

- Impact of nutrients supplied by sludge on biomass of trees which may increase efficiency or profits in the timber industry
- Impact of entrenchment on degradation of sludge over time
- Fate of pathogens in entrenched sludge over time
- Impact of nutrients supplied by sludge on biomass of trees which may increase efficiency or profits in the timber industry

The following four studies were conducted in order to assess the effect of entrenched sludge on trees and the environment in the South African context and to explore the feasibility of deep row entrenchment as a beneficial disposal option:

- Entrenchment of VIP pit sludge at Umlazi E ponds
- Entrenchment of wastewater sludge at Sappi (Howick)
- Two smaller trails
  - Tree tower trial using pit sludge
  - On-site burial of pit sludge

Both pit latrine sludge and treated sewage sludge were tested under varying conditions. The treatment works sludge used for the Sappi trial was well stabilised, while the pit latrine sludge used for the Umlazi trials, tree tower trials and on-site burial comprised a mixture of older, more stabilised sludges and fresher, unstable sludges from different levels of the pit.

4.1 Effect of entrenchment on sludge

This project investigated whether the entrenchment of unstable VIP latrine sludge in soil might result in a greater degree of stabilization than can be achieved in the pit latrine. Buckley *et al* (2008) classified the sludge found in a full pit latrine into four layers. Layer (i) is the top layer which contains recently deposited faeces with components which can be rapidly degraded through aerobic processes. This layer is negligibly small and is not measurable in practice. Layer (ii) is made up of the top aerobic section of the pit. In this layer, aerobic degradation of hydrolysable organic material occurs at a rate limited by the aerobic hydrolysis of complex organic molecules to simpler compounds. Layer (iii) is covered by layers (i) and (ii) and, as a result, oxygen is not available for aerobic processes and anaerobic processes dominate. Anaerobic digestion proceeds at a significantly slower rate than in the layer above, and is controlled by the rate of anaerobic hydrolysis of complex organic molecules to simpler molecules. In the lowest layer (iv), no further stabilisation of organic material occurs within the remaining life of the pit.

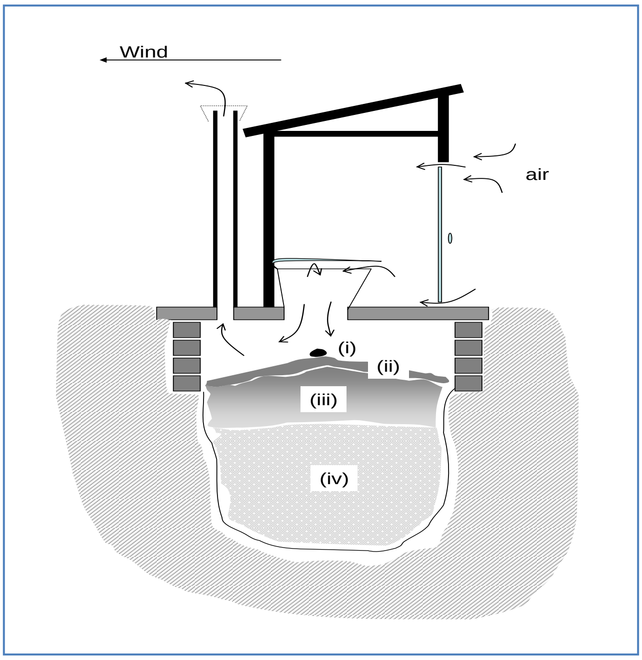


Figure 4.1 Diagram showing the different theoretical layers within a pit latrine. (Buckley et al, 2008)

This theory applies when the contents of the pit are not disturbed (for example, by liquid saturating the pit contents) so that the depth of material correlates to the length of time that it has been in the pit. In this case, the amount of biodegradable solids as a fraction of total solids should decrease with increasing depth for samples collected from the surface layer (i) through to layer (iii) and should remain constant in layer (iv). This would be observed as decreases in chemical oxygen demand (COD), volatile solids (VS) and biodegradability of pit latrine sludge content as a function of total solids as one digs from the surface layer down to the bottom layer of the pit. It should also be noted that depending on the household habits and local environmental conditions, and the history of these factors, there will be considerable variation in the moisture content, organic content, non-biodegradable content and microbial population of material with time as it is added to the pit, and therefore variations will occur within a single pit and between different pits.

During the processes of being dug out of the pit and reburied in trenches the sludge from these four layers will be mixed together to some extent. The sludge will contain a mixture of old, relatively well-degraded sludge, and fresh, unstable sludge. Thereafter the fresher portions will undergo similar processes as occurred in the pit latrine as the less stable components continue to stabilize.

In addition, some aerobic and possibly fungally mediated degradation processes may become possible under the conditions of entrenchment that did not occur in the pit. Field studies of full pit latrines which are no longer in use (ie. do not contain fresh, undegraded material) have indicated that further stabilization occurs from the soil/sludge interface inwards (Morgan, 2004), possibly due to the action of soil fungi. While in wet environments bacteria are primarily responsible for degradation (De Boer et al 2005; Del Giorgio and Cole 1998) in soils, soil fungi contribute significantly to the biodegradation of organic material. Various studies have demonstrated that the presence of organic matter in soils or organic fertilization of soil has a positive influence on the soil fungi population (Abbott and Murphy, 2003). This is because bacteria cannot bridge air-filled voids found in soil due to their unicellular body form (De Boer et al, 2005) while fungi, which have a hyphal/mycelial growth form, can, allowing them greater motility through the soil (Griffin, 1985; de Boer et al 2005). While certain soil bacteria (eg. Actinomycetes) have developed a hyphal growth form, fungi are responsible for most of the heterotrophic processes and degradation of recalcitrant organic compounds taking place in the soil (de Boer et al, 2005; Griffin, 1985). In addition, fungi hyphae have a greater ability than bacteria to translocate nutrients within the soil (Jennings, 1987; de Boer et al 2005) and are largely responsible for the formation of mycorrhiza and the decomposition of lignocelluloses, processes upon which terrestrial ecosystems rely. As soil fungi grow best in moist but well aerated soil conditions with pH near neutral they do not find the conditions favourable in a pit which is still in use, where conditions are largely aerobic with no air-filled voids and where chemical agents may be present. Without the presence of certain soil fungi cellulosic cell components in the pit cannot be broken down, limiting the extent of degradation that occurs. After a pit is no longer in use, or if sludge is transferred to a burial pit or trench, conditions become favourable for soil fungi and, as a result, further degradation of the sludge may then become possible.

4.2 Impact on tree growth

This project investigated the potential for sludge to be used in association with agroforestry to both enhance timber production and provide a sustainable means of disposing of sludge. The nutrients present in sludge (especially nitrogen, phosphorus and potassium) may be released slowly as the sludge is degraded, resulting in a sustained nutrient source for the timber crop. According to Jönsson et al. (2004), burial of sludge increases the organic content of the soil which enhances the moisture retention characteristics, ion-buffering capacity and general fertility of the soil. It has been documented that in plantations trees draw the water available within the surroundings into the plantation area (Dons, 1987, Duncan, 1993). The planting of trees near the buried sludge would therefore have the added advantage of a net movement of water into the burial site to supply the water requirements of the trees. Toffey et al (2007) found that trees grown on sludge showed improved growth rates from the first year, particularly when the receiving soil was poor. In a deep row entrenchment study involving poplars, Kays et al (1999) found that the tree roots only penetrates the entrenched sludge 3-6 years after planting. Tree roots developed in a large mass around the trenches which appeared to take up the nutrients as they were mineralised.

4.3 Effect of sludge on the environment

For entrenchment of sludge to become a viable option, the movement of nutrients and contaminants present in the sludge through the soil and the potential risk of contamination of the groundwater must be understood.

When sludge is fresh, most of the nutrients it contains are bound in organic compounds and cannot be accessed by plants. The most significant elements making up the organic compounds in sewage are hydrogen (H), oxygen (O), carbon (C), nitrogen (N), phosphorus (P) and sulphur (S) (Marais and Ekama, 1984). As bacteria break these compounds down, inorganic compounds and energy are released and the sludge becomes more stable. If oxygen is present, aerobic bacteria multiply rapidly and decompose the material.

In deep row entrenchment, the conditions within the trench are initially anaerobic (Kays et al 1999). As roots begin to penetrate the sludge oxygen is introduced into the sludge gradually, enabling aerobic digestion which slowly releases nutrients from organic compounds into an inorganic form which the tree can utilise. Thus the buried sludge provides a nutrient source which releases nutrients to the tree over its lifetime.

If nitrogen leaches through the soil as nitrate, harmful concentrations in surface and ground waters can result. Kays et al (1999) found that entrenched sludge averaged a nitrogen content of 3.3% at the time of application which reduced to 1.2% by the end of the growth cycle. (Kays, unknown date) suggested that while much of the nitrogen had been mineralised, as much as 40% may have been converted to nitrogen gas (N_2) by denitrification due to the wet anaerobic conditions in the trench.

Phosphorus is an essential nutrient for plants and animals and its limited availability in most fresh waters limits the propagation of some plants. If phosphorus levels in streams and surface waters rise as a result of leaching from sludge, accelerated plant growth and algae blooms can result which then rapidly deplete the oxygen available in the water. The end product is water with low dissolved oxygen which cannot support aquatic life including certain fish, invertebrates, and other aquatic animals.

In this study, the possible nutrient impact from deep trench disposal of sewage sludge and pit latrine waste on water resources was investigated in four phases:

- Firstly, the site was characterised so that the dominant flow pathways could be described. To do this the surface topography, subsurface geophysics and subsurface hydraulic properties were surveyed. Using Electrical Resistivity Tomography a 2-dimensional distribution of the electrical resistivity of the subsurface along selected transects was derived, indicating soil and geologic layering as well as the water distribution in the subsurface. Interpretation of this geophysics survey provided a first indication of the likely flowpaths from the entrenchment site and allowed boundary conditions to be defined for later simulation of subsurface water and nutrient movement. ERT surveys later in the study were used to highlight the effects of water and salinity movement from the trenches. The measurement of soil hydraulic characteristics (water retention and hydraulic conductivity) was used in interpreting observed water and nutrient movement as well as providing the hydraulic parameters for simulation of water and nutrient fluxes.

- Secondly, monitoring of the water and nutrient dynamics in the subsurface was aimed at quantifying the concentration, rates and prevalence of water and nutrients in the soil and groundwater. The monitoring also provided observations of soil water and nutrient status which could be compared to the simulated status and so allow for calibration of the simulation, so that accurate simulated discharges of water and nutrients (which could not be directly observed) could be reported.

- A third phase of the estimation required an assessment of the various observations and surveys to deduce flow pathways, duration of discharges and concentration of nutrients in the subsurface flow.

- The fourth and final phase required simulation of water and nutrients, including a water balance of rainfall, evapotranspiration and subsurface fluxes and storage. From these results the discharge of subsurface water and nutrients to either groundwater or stream could be estimated. The HYDRUS-2D model was used to simulate water and nutrient movement through the trenches and the soil at the Umlazi site. While the interactions between topography, vegetation, climate, surface water, groundwater and soil chemical processes at a given location are too complex to be explicitly modelled, these processes can be approximated with conceptual simplifications of the inputs to the model based on field evidence, soil survey data, and norms provided in the literature.

4.4 Fate of pathogens

Faecal sludges contain excreted pathogens (bacteria, viruses, protozoa and helminths) which may represent a health risk if they reach the groundwater. While secondary sludge which has been treated at a waste water treatment plant will have reduced levels of viable pathogens, it is still potentially infective. Pit latrine sludge has been found to remain highly pathogenic even after remaining for years outside of a host in the pit environment.

In a study of the movement of bacteria through the soil around low water use toilets built on fairly permeable soil, it was found that contamination levels dropped rapidly with distance from the soakaway: at a distance of 3m horizontally from the soakpit bacteria measurements had returned to background levels (van Ryneveld, Fourie, and Palmer, 2012).

The fate of pathogens after sludge has been entrenched was unknown. For entrenchment to be employed as a viable method, it is important to understand for what period the entrenched sludge remains pathogenic, even if there is no movement of pathogens out of the trench. This study investigated the fate of helminth eggs over time, using the hypothesis that the number and viability of *Ascaris* ova per mass of sample would decrease with increasing amount of time spent buried in the trench. The effect of the following variables on helminth egg load was explored:

- age of sludge (or depth in a pit latrine)
- exhumation and subsequent deep row entrenchment of the sludge
- time spent in the trench environment

Ascaris ova can be categorised as potentially viable or non-infective (i.e. containing a necrotic larva or unfertilized egg). An ovum was recorded as being potentially viable if

- It was undeveloped (might at some time in the future develop a larva)
- Had a fully developed and motile larva (this is the only category that may be regarded as infective with certainty)
- Had an immotile larva that was not obviously dead

In the case of an immotile larva, the larva may be alive but dormant, or may have recently died but not yet become necrotic. If it has recently died, this means that it was recently infective. It is not apparent how long a larva will remain intact after dying before it is obviously necrotic. However, the assumption was made that a dead larva will become necrotic within a few months. Thus, any ovum which contained an intact larva must have been alive and therefore infective a relatively short time prior to being sampled.

This hypothesis is important for interpreting the *Ascaris* ova data; while only the presence of motile larvae in ova confirms that the sludge tested is definitely infective, the presence of immotile larvae indicates that until a short time before sampling (or even after sampling, but before examination under the microscope) the sludge was definitely infective.

In terms of health risk assessment, a sludge may be allocated a binary classification as infective or not-infective. Thus, if there is a detectable number of undeveloped ova, ova with motile larvae or ova with immotile larvae per gram of sludge, it may be concluded that the sludge should be treated as infective.

5. ENTRENCHMENT OF PIT LATRINE SLUDGE

As over a million VIP toilets have been provided by South African municipalities to meet the need for basic sanitation, the disposal – or preferably, beneficial use – of pit latrine sludge is a critical issue. Burial of pit sludge on the same premises as the pit from which it is removed is ideal if there is space. For this study the burial of pit sludge in combination with the planting of trees to both limit movement of pollutants out of the sludge and enhance the nutrient values in the fruit crop of the trees was tested on a small scale with two homes. Thereafter a large scale study was conducted in which approximately 1200 m^3 of pit latrine sludge was entrenched on a research site which was then planted with approximately 1400 trees. A smaller, controlled study was conducted in which 24 trees were grown in containers, half of which were enhanced with a core of sludge within the soil.

5.1 Entrenchment of sludge at Umlazi E ponds

The eThekwini Metropolitan Municipality has taken a proactive approach to the challenges of emptying pits and finding appropriate options for the disposal and utilisation of sludge. Having recently completed a cycle of emptying the 35 000 VIPs within the metro, the city had a considerable volume of pit sludge to dispose of. One option eThekwini had developed to utilise its pit latrine sludge was a heat treatment system which removes rubbish from sludge and processes it into pathogen-free pellets which can be used by the municipality as a fertiliser and potentially sold as well. In the interest of exploring new options for the beneficial use of sludge, eThekwini offered three sites for potential use for a deep row entrenchment study. The site of a disused wastewater works in Umlazi was selected.

5.1.1 Site assessment and selection

The Umlazi site had been used as wastewater treatment works by eThekwini Municipality until heavy flooding damaged the oxidation ponds in 1987. The land was valueless because it was below the 1:50 year flood line. The ponds were situated on gently sloping ground at the foot of a steep, densely populated hill side in the Umlazi residential zone E section. The lower edge of the site was approximately 80 m from the Umlazi River. The site was 20km south west of the city centre with an average annual rainfall of 1000 mm.

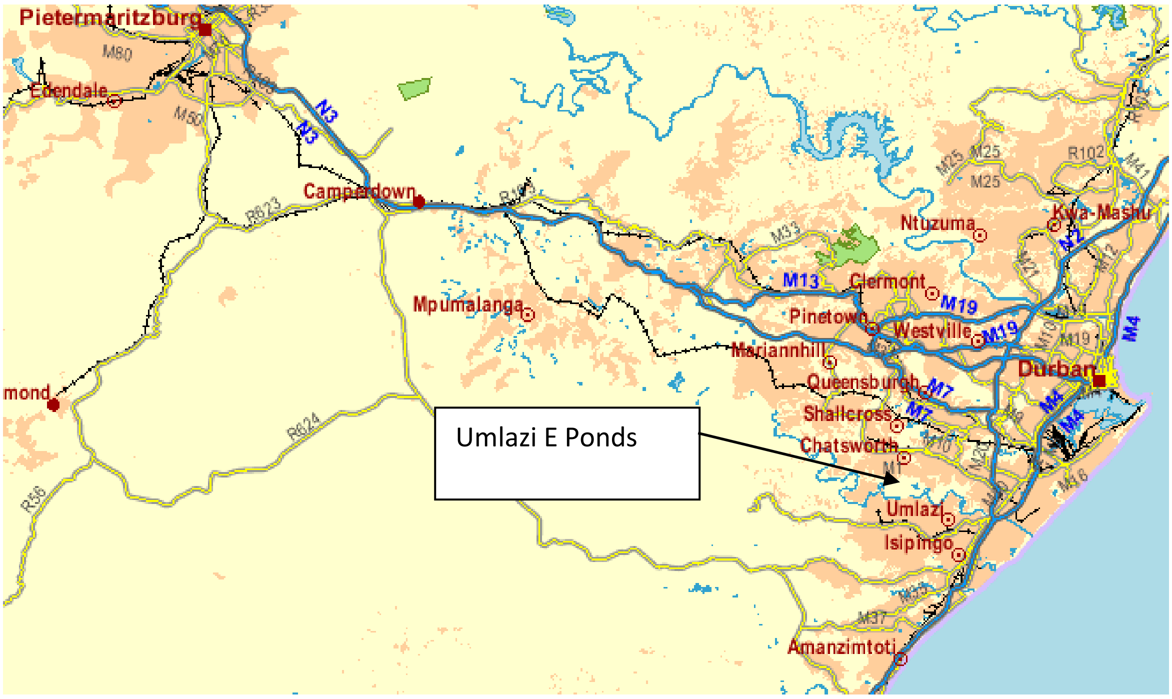


Figure 5.1. Location of Umlazi pit latrine sludge burial site

Of the three ex-ponds the northernmost had been used for sand winning operations and the southern, and lowest, pond was essentially a wetland. The central pond was selected for this trial. It was approximately 5000m^2 in size and had fine, silty river sand with very little nutrient content. Trial pits were dug to a depth of two metres without encountering any change in the soil make up and without reaching the water table.

The selection of this site provided a number of advantages:

- Municipal land was ideal as the municipality was effectively a partner in this research project.
- There was a precedent of use of the site for the treatment of sewage.
- It was situated below the 1:50 year flood line so the land had no value for other purposes.
- It was close to the pits which provided sludge for the study.
- The only apparent disadvantage was that the soil was of a very poor quality, with almost no agricultural value. This did, however, provide an opportunity to demonstrate whether the entrenchment of sludge could reclaim an infertile site.

Because the trial site bordered residential areas of Umlazi (E section), the local councillor was consulted and appraised of developments on the site. Several community meetings were held to discuss the study and to obtain permission to proceed. Permission was also obtained from the Department of Water Affairs to conduct the study at this site.

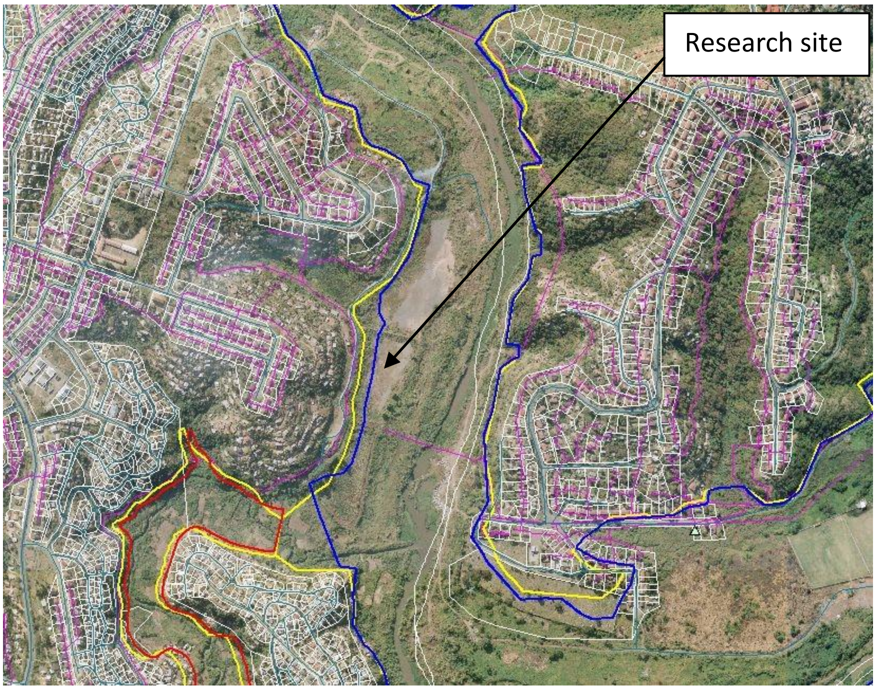


Figure 5.2 Umlazi site: 1:50 year flood line shown in blue and 1:100 year flood line in yellow

Figure 5.3 Research site at the beginning of the study, cleared of vegetation

5.1.2 Geophysical characterisation of the site

A range of geophysical surveys were conducted to provide a baseline characterisation of the research site and to provide insight into the dominant controls of the subsurface water fluxes.

5.1.2.1 Electrical resistivity characterisation

The electrical resistivity of the ground is determined by a number of geological parameters, such as mineral and fluid content, porosity, degree of water saturation in the rock and dissolved ions in the water. Resistivity measurements allow the various subsurface soils and geological and hydrogeological features, as well as aquifer parameters such as transmissivity and hydraulic conductivity, to be characterised (ABEM, 2005). The electrical resistivity of rocks and soils in a survey area can vary by several orders of magnitude. The presence of clay minerals strongly affects the resistivity of sediments and weathered rock (Dahlin and Loke, 1998) because clay minerals act as electrically conductive particles, absorbing and releasing ions and water molecules on their surface through an ion exchange process. Metals, on the other hand, have extremely low resistivity values. Igneous and metamorphic rocks typically have high resistivity values, while sedimentary rocks, which usually are more porous and have higher water content, normally have lower resistivity values. Wet soils and fresh groundwater have even lower resistivity values. Clayey soil normally has a lower resistivity value than sandy soil. Because resistivity values have a much larger range, compared to other physical quantities mapped by other geophysical methods, resistivity and other electrical or electromagnetic based methods are very versatile geophysical techniques.

Electrical Resistivity Tomography (ERT) technology allows resistivity to be measured without damaging the soil surface. Two dimensional electrical imaging is generated using a large number of electrodes connected to multi-core cable. In order to obtain a 2D electrical image, horizontal and vertical data is collected using automatic sequential measurements of current and potential locations.

Figure 5.4. ERT survey at Umlazi to determine subsurface layering and soil water profile

An ABEM Lund Imaging System, consisting of a basic charging unit, Electrode SelectorES10-64, four Lund spread cables, a quantity of cable joints, cable jumpers and electrodes together with a Terrameter SAS 1000 was used on site for data acquisition. For each transect, a Wenner array, which has the advantage of accurately resolving more horizontal layers than vertical layers, was used with an electrode spacing of 0.5 and 1m. The DGPS was used to record the electrode geographical coordinates.

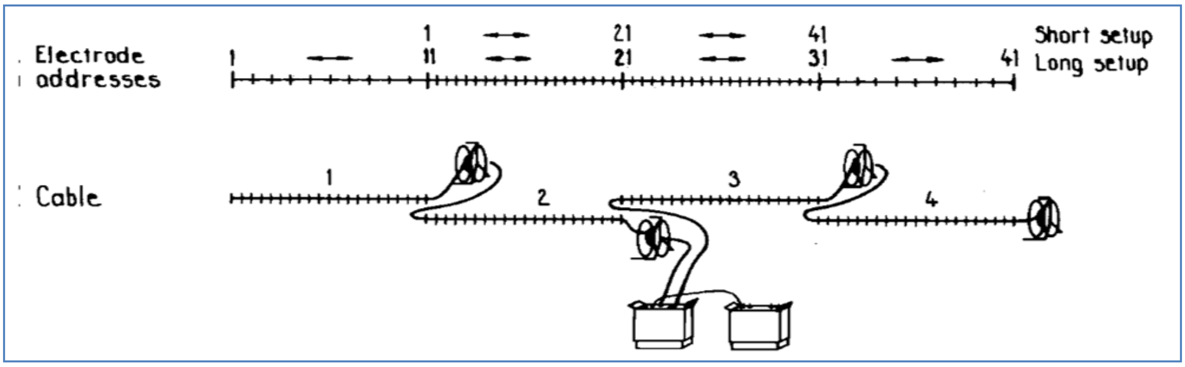


Figure 5.5. Schematics of the ERT instrument and typical cable layout as used in the geophysical surveys.

Measurements were taken by injecting current into the ground through two current electrodes (see Figure 5.5) and measuring the resulting voltage difference at the two potential electrodes (P). The instrument automatically selects different current and potential electrode pairs so that the resistivity can be estimated along the length and depth of the transect as shown in Figure 5.6.

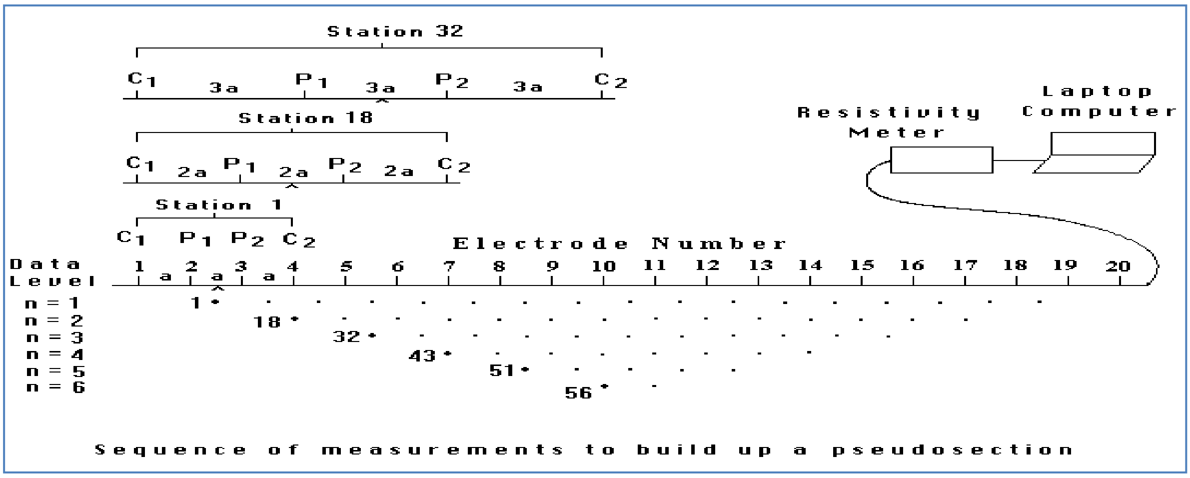


Figure 5.6. Sequence of measurement for a typical Electrical Resistivity Tomography survey (Griffiths and Barker, 1996)

From the current (I) and voltage (V) values, an apparent resistivity (pa) value is calculated:

pa= k V / I

where k = geometric factor, which depends on the arrangement of the four electrodes, used at any particular time in the sequence, and making up the voltage excitation and resistance measurement pairs.

This calculation gives the resistivity of the ground as if it were homogeneous. The relationship between this "apparent" resistivity and the "true" resistivity of the ground (which is not homogeneous) is a complex relationship, used in the inversion of the "apparent" resistivity to yield a 2-dimensional spatial distribution of the actual subsurface resistivities. To generate a geological simulation of the area, resistivity measurements are interpreted on the basis of typical resistivity values for different types of subsurface materials. The resistivity of the rocks is also affected by the amount and quality of water in pore spaces and fractures and the degree of connection between the cavities (Table 5.1). The resistivity of groundwater varies from 10 to 100 ohm meters, depending on the concentration of dissolved salts.

Table 5.1 Typical electric resistivity values for common rocks, minerals (Kollert, 1969) and natural waters (Delta Mine Training Centre, 2005) used to interpret resistivity data

Common rocks/materials	**Resistivity (ohm meters)**	**Common rocks/materials**	**Resistivity (ohm meters)**
Clay	1 – 100	Granite	200 – 100,000
Graphitic schist	10 – 500	Basalt	200 – 100,000
Topsoil	50 – 100	Limestone	500 – 10,000
Gravel	100 – 600	Slate	500 – 500,000
Weathered Bedrock	100 – 1000	Quartzite	500 – 800,000
Gabbro	100 – 500,000	Greenstone	500 – 200,000
Sandstone	200 – 8,000		
Ore minerals	Resistivity (ohm meters)	Ore minerals	Resistivity (ohm meters)
Pyrrhotite	0.001 – 0.01	Pyrite	0.01 – 100
Galena	0.001 – 100	Magnetite	0.01 – 1,000
Cassiterite	0.001 – 10,000	Hematite	0.01 – 1,000,000
Chalcopyrite	0.005 – 0.1	Sphalerite	1000 – 1,000,000
Water Type			Resistivity (ohm meters)
Precipitation or rainfall			30-1000
Surface water, in areas of igneous rock			30-500
Surface water, in areas of sedimentary rock			10-100
Groundwater, in areas of igneous rock			30-150
Groundwater, in areas of sedimentary rock			>1
Sea water			Approx. 0.2
Drinking water (max. salt content of 0.25%)			>1.8
Water for irrigation/stock (max. salt of content 0.25%)			>0.65

Two transects were surveyed initially: along the long axis of the entrenchment site and at right angles between the site and the river (Figure 5.7). A Wenner Long protocol with 2 m electrode spacing was used for Transect 1 on the long axis (approximately 120 m) of the deposition area and both Wenner Long and Short with electrode spacing of 1.5 m was used for Transect 2 between the disposal site and the river, a distance of approximately 60 m. The survey indicated a perched water table at about 2 m below surface, probably fed by the local hillslope, with a shallow high resistivity zone at the northern end (Transect 1). Electrical conductivity was found to be relatively low, indicating that salinity or EC could be used as a tracer after application of the sludge, considering the porous nature of the soil.

Figure 5.7 Transects used for ERT survey in September 2008

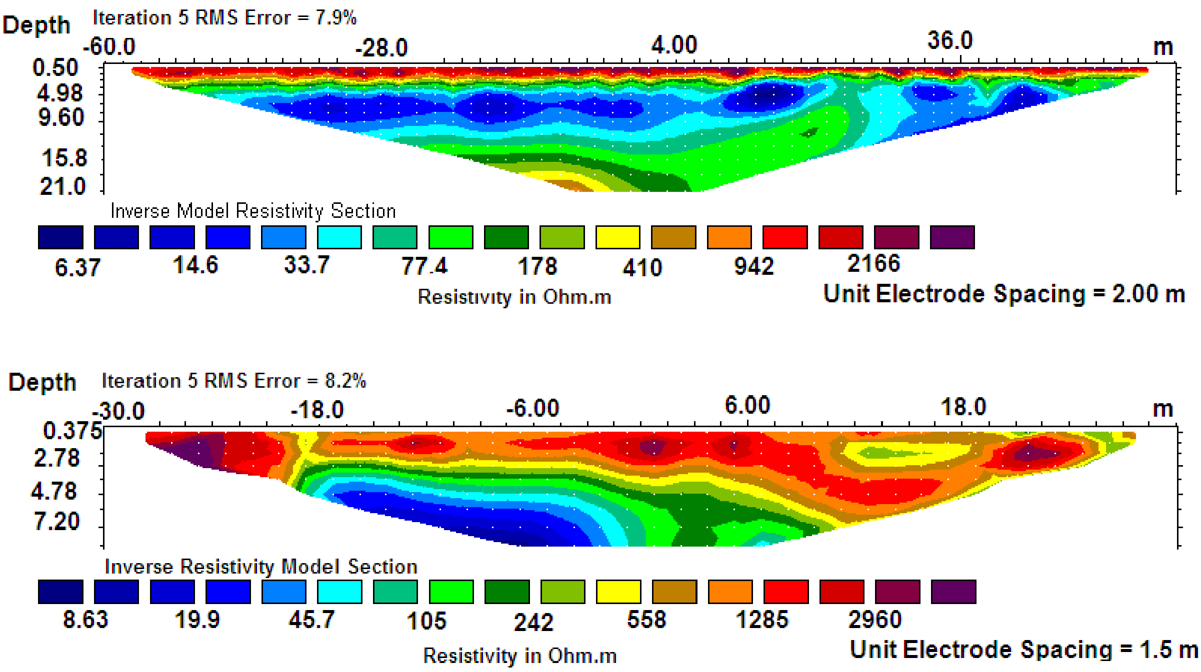


Figure 5.8 Transects 1 north-left and south-right (top) and 2, upslope-left and stream-right (bottom)

5.1.2.2 Soil characterisation

Soil samples were were collected from the prepared pits in April 2008 and analysed for Electrical Conductivity (EC) and acidity as well as texture and hydraulic characteristics (water holding capacity and hydraulic conductivity).

Soils from all sampling points and depths fell under the USDA classification of sands with textures similar to one another, indicating high infiltration rates and low water holding capacities. The soils were found to be acidic, indicating that the pH of the sludge would be critical with respect to the microbial population and trees grown on the site.

Table 5.2 Analysis of soil samples collected from pits in April 2008

Sample name	Particle size analysis				pH	EC
	Sand %	Silt %	Clay %	Texture class		
Southeast (1.3m)	93.2	2.9	3.8	Sand	5.1	0.079
Southwest (2m)	94.3	2.6	3.1	Sand	5.9	0.061
Northeast (1.2m)	97.1	0.7	2.1	Sand	5.3	0.033
Northwest (2m)	97.2	0.7	2.1	Sand	5.0	0.06

The hydraulic properties of the soil were characterised in order to understand the influx, uptake and distribution of water in the entrenchment zone which could impact on the movement of pollutants out of the trenches. This was done by estimating the retention of water in the soil and the hydraulic conductivity in saturated and unsaturated soil on the basis of a wide range of measurements.

In-situ measurements were made at 5 surface points in the survey area (yellow dots 1-5 in Figure 5.9), 5 points on top of the filled trenches (yellow dots 6-10) and 2 locations for measurements at different depths (yellow dots 11-12). At these two locations measurements were taken at depths of 100 mm, 200 mm, 400 mm, 600 mm, 800 mm, 1 m and 1.5 m. While GPS coordinates were taken at each point, the accuracy was not sufficient to pinpoint the locations on site for follow-up measurements.

For each measurement, a standard procedure was followed (Lorentz, 2001). Tension infiltrometer measurements were made in duplicate, followed by duplicate double ring infiltrometer measurements on the same spots. Thereafter, the rings were removed and undisturbed soil samples were taken of the material beneath the measurement locations for laboratory bulk density, water retention and textural analyses. In addition, at each duplicate location, one disturbed sample was taken of the soil condition before and one after the tension infiltrometer measurement. These two samples provided initial and final water content data for the unsaturated hydraulic conductivity analysis. The undisturbed sample was used to determine the water retention characteristic and bulk density while part of the disturbed sample was used for a particle size distribution analysis. The data were then used to derive parameters for input to the Hydrus-2D model to analyze the movement of water and dissolved compounds in the soil.

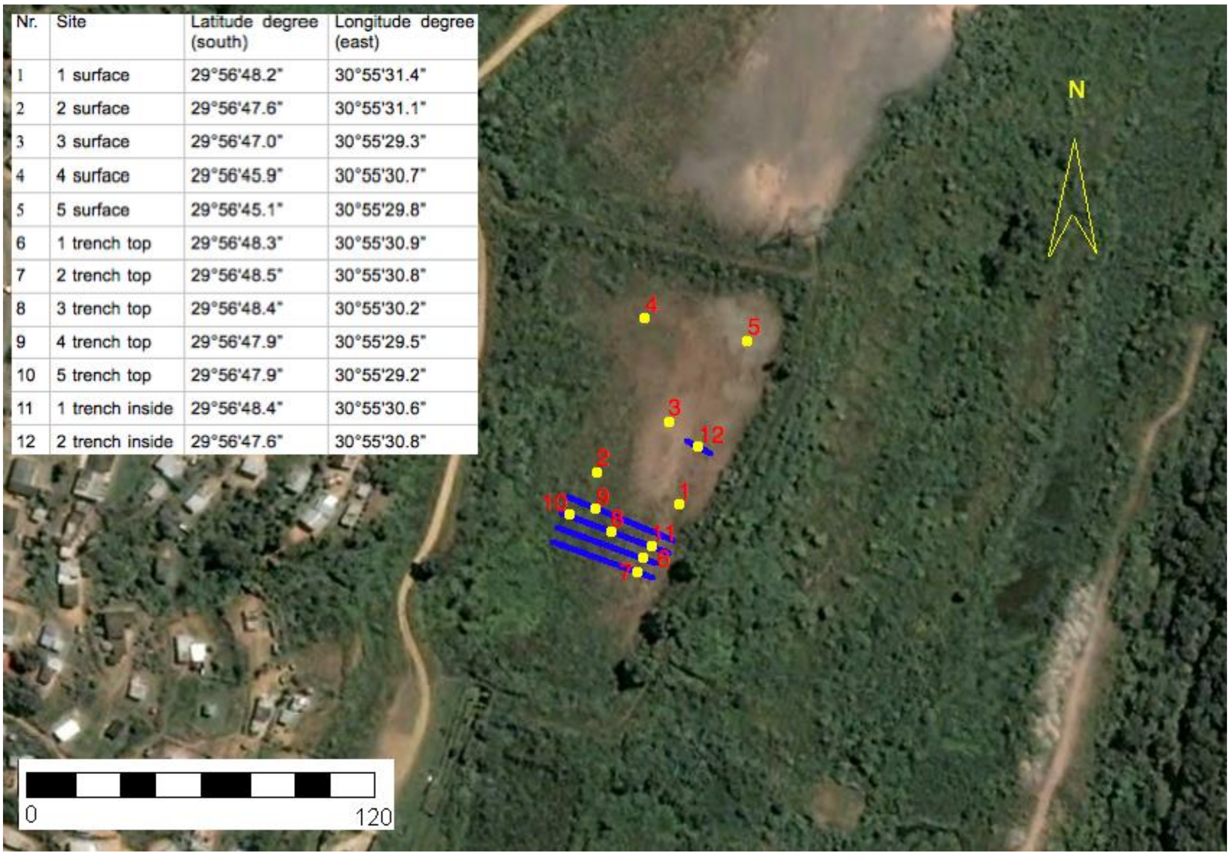


Figure 5.9 The Umlazi entrenchment site indicating positions of trenches (blue line) and measurement points (yellow dot)

All of the soil samples tested fell into the sand category, with clay percentages varying between 1 and 7%, except for the Surface 5 position which classified as a sandy loam with a clay content of 20%.

Table 5.3. Summary of particle size fractions

Site	Depth (mm)	Sand (%)	Silt (%)	Clay (%)
Surface position 1	0	91	5	4
Surface position 3	0	85	8	7
Surface position 4	0	94	3	3
Surface position 5	0	82	18	20
Trench top position 1	0	96	2	2
Trench1 inside	200	94	4	2
Trench1 inside	600	97	2	1

Typical hydraulic characteristics are shown for the water retention at trench site 2a in Figure 5.10 and for hydraulic conductivity at Site 2 in Figure 5.11. For clarity and completeness, the saturated hydraulic conductivity is shown on the plot at a tension value of 1 mm. It is worth noting that many of the characteristics showed an order of magnitude difference between the saturated hydraulic conductivity of the ponded double ring infiltrometer test and the unsaturated hydraulic conductivity at a small tension of 5 mm (Figure 5.11).

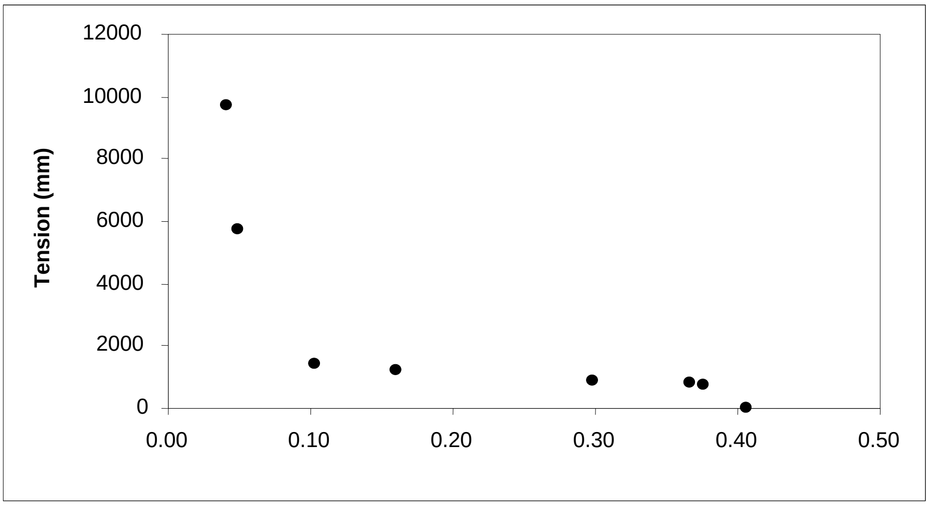


Figure 5.10 Typical water retention characteristic at Trench 2 at 200mm

There was no observable trend in Ksat with depth in the two trenches measured. However, the Ksat determinations on the undisturbed surface and those at 100mm below surface in the trenches were generally lower than the deeper Ksat values determined in the trenches. This could reflect a layer of fines deposited on the surface or residue from the previous use of these ponds. The 10 measurements performed on undisturbed surface material were generally lower than those completed on the surface of the disturbed trench fill (Table 5.6).

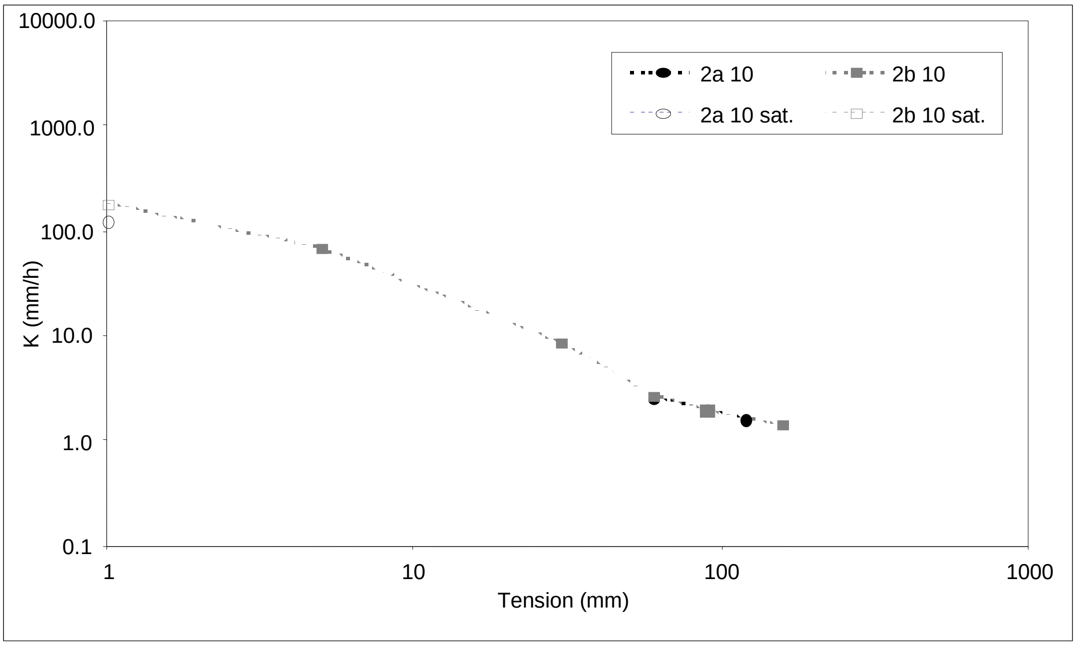


Figure 5.11. Hydraulic conductivity characteristic at Trench 2 100mm

Table 5.4 Summary of the saturated hydraulic conductivities at surface and trenched sites (The number in brackets represents the site number shown in Figure 5.9)

SURFACE				SUBSURFACE			
Surface		Surface on trench		Trench 1 (11)		Trench 2 (12)	
Position	Ksat (mm/h)	Position	Ksat (mm/h)	Depth (mm)	Ksat (mm/h)	Depth (mm)	Ksat (mm/h)
1a (1)	89	1a (6)	404	100	30	100	123
1b (1)	98	1c (6)	385	100	21	100	178
2a (2)	96	2a (7)	713	200	327	200	382
2b (2)	36	2b (7)	936	400	312	200	366
3a (3)	82	3a (8)	827	400	369	400	346
3b (3)	150	3b (8)	885	600	866	400	172
4a (4)	183	4a (9)	1020	600	822	600	553
4b (4)	132	4b (9)	531	800	1226	600	312
5a (5)	571	5a (10)	249	800	930	800	366
5b (5)	375	5b (10)	251	1000	888	800	290
				1000	911	1000	596
				1500	1699	1000	229
				1500	3109		

5.1.3 Preparation of trenches

Parallel trenches spaced 3 metres from their centres were excavated mechanically using a backhoe.

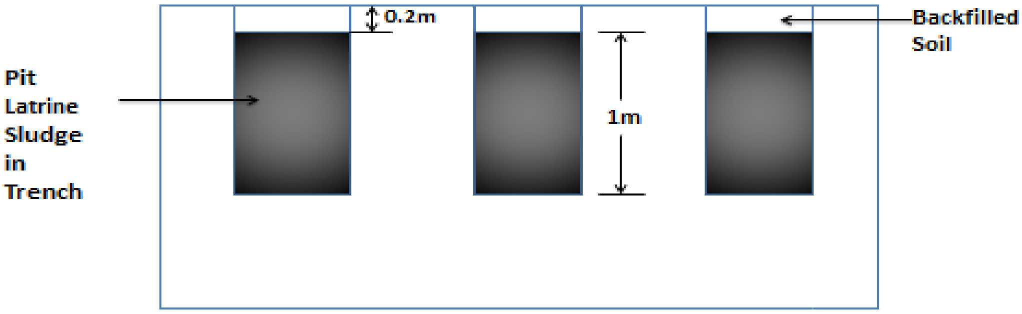


Figure 5.12. Trench design at Umlazi sludge burial site

While the planned width was 1 m and depth 2 m, the action of digging the loose, sandy soil caused the sides of the trenches to collapse. As a result, the final dimensions were variable, with trenches typically wider (1.2 m to 1.5 m wide) and shallower (1.3 m to 2 m deep) than planned. In general it was not possible to dig 2 m deep trenches with vertical sides, so sides sloped outwards towards the surface. Moreover, when the soil was particularly wet after rain it was not possible to dig deeper than approximately 1.3m and at these times the trenches were wider. Weather slowed progress at times and the site was periodically flooded towards its lower (southern) end. The site drainage was eventually improved by the construction of a cut off drain and pipe through the southern berm wall, allowing the remainder of trenches to be dug.

Figure 5.13 Variable conditions resulting in variable pit dimensions

5.1.4 Sludge application

Between September 2008 and January 2010 approximately 1200 m^3 of VIP pit sludge was buried in 900m of trench. Trenches were filled to within 500 mm of the surface and the soil was backfilled, with the overburden heaped along the top of the trench to a height of approximately 400 to 600 mm. Community representatives insisted that the sludge be covered with a layer of sand between deliveries to reduce odour problems. As a result the sludge was not applied in a single continuous layer.

5.1.5 Tree planting

The site was planted with over 1400 seedlings of *Eucalyptus grandis (*commonly known as flooded Eucalyptus or rose Eucalyptus), *Acacia mearnsii*, banana and *Cryptocarya Latifolia.* Rows were planted as follows.

- Main section (22 rows): 8 rows of Eucalyptus and 14 rows of wattle trees were planted.
- Experimental section (9 rows): Rows E1, E2, E8 and E9 were planted with Eucalyptus only while rows E3, E4, E5, E6 and E7 were planted with a combination of Eucalyptus and wattle. The third row in this zone was the only row that was to contain any sludge while the seventh row contained a commercial fertilizer. Sludge was inadvertently also added to one end of row E2 however.
- Section C (9 rows): While the other sections had been planted exclusively with Eucalyptus and wattle, which are low-maintenance, fast-growing trees commonly grown commercially for the timber industry, the opportunity was taken to plant 3 of the rows in this section with banana and 2 with *Cryptocarya Latifolia*, an indigenous species. Some of the rows which were planted were not trenched.

5.1.6 Monitoring

Monitoring procedures for tree growth, changes in the sludge, soil and groundwater and the fate of pathogens are described below.

5.1.6.1 Tree growth

Trees were measured every three months. Both height and stem diameter measurements were taken until height exceeded 5 m. For trees that were taller than 5 m, a stem diameter measurement was taken at a height of 1.37 m from the base of the tree. According to standard forestry practice the stem diameter at 1.37 m from the base of trees taller than 4 m enables the estimation of the total tree biomass. Visual assessments of the growth and health of trees were also conducted when it was possible.

5.1.6.2 Groundwater

The following instrumentation was installed at the entrenchment site (Figures 5.14 and 5.15):

- piezometers to record groundwater levels and for drawing samples (numbered UEP1, UEP2, UEP3, UEP4 and UEP5)
- 2 watermark sensors to record soil water tension at various depths (numbered Umlazi_WM_01 and Umlazi_WM_02)
- 4 wetting front detectors (WFDs) to indicate percolation depth and sample the percolating liquid at various depths (numbered UEP3-200 mm, UEP3-300 mm, UEP3- 500 mm and UEP3-800 mm)
- 5 deep boreholes, established before commencement of the trial, between the trenching site and the river (numbered BH1, BH2, BH3, BH4 and BH5).

Figure 5.14 Location of borehole sites at Umlazi

Figure 5.15 Location of piezometer/WaterMark/Wetting Front Detector (WFD) sites

Figure 5.16 Installation of wetting front monitors (left) and monitoring of piezometers (right)

Groundwater samples from the five background monitoring wells were taken every 1-2 months. The depth to the water table at each well was recorded and twice the volume of the standing water within the well was purged before a water sample was taken to ensure that the sample was representative of the groundwater around the well rather than of the water standing in the well. Samples were analysed for chloride, COD, conductivity, sodium, ammonia, nitrate and nitrite, dissolved oxygen, pH and orthophosphate as well as *T. coli*, *E. coli* and total organisms. A Hach Model DR/2010 Spectrophotometer was used to analyse the nitrate and phosphorus concentration in the water samples from the piezometers and WFDs as well as the borehole samples of 20 July 2011.

Unfortunately the WaterMarks and wetting front detectors were vandalised in the early stages of the project and so, due to an inundation of leaked water from a local sewer main and the fact that the wetting front detectors required burial simultaneously with the sludge disposal, these instruments were not replaced. Water samples were collected from the piezometers on one occasion and thereafter were found dry. The nearby Umlazi river was also sampled during the monitoring period. Rainfall, temperature, relative humidity and solar radiation measurements were obtained from the nearest weather station (Data for station [0240808A2]-Durban South AWS -29.9650 30.9460) from the South Africa Weather Services.

5.1.6.3 Characteristics of sludge

Samples were taken of sludge before burial to provide a baseline analysis of its composition. Thirty samples were collected over a period of six weeks in order to assess variability.

After the sludge was entrenched, samples were extracted with a soil auger for analysis over a period of 29 months in order to establish whether significant change occurred in the characteristics of the sludge after entrenchment and especially whether the sludge stabilised to a greater degree in the trenches than was possible in the bottom layer of latrine pits.

A number of biological, physical and chemical analyses were conducted on the sludge samples to determine moisture content, solids (total and volatile solids), chemical oxygen demand (COD), aerobic biodegradability, Total Kjeldahl Nitrogen (TKN) and Free and Saline Ammonia (FSA). Standard Methods (APHA, 1998) were used to analyse the collected sludge samples where applicable and where no appropriate method was published, adaptations of existing methods were used or entirely new methods were developed.

ERT technology was also used to determine the spatial distribution of moisture in the entrenched sludge and study the changes occurring in the sludge and its effect of the surrounding soil.

5.1.6.4 Pathogens

Over a period of 17 months (3 March 2010 to 28 July 2011), 115 points along the trenches were sampled. Where a thick layer of sludge was found buried in the trench, two samples were taken, an '***a***' sample just below the upper sludge-sand interface and a '***b***' sample some 20 cm further into the sludge. At the time of sampling, the amount of time that each sample had spent in the trench was calculated from the date at which the trench was filled. Samples were between 6 and 34 months old. A total of 220 samples were analysed. Samples were analysed using the modified AMBIC protocol (Hawksworth et al., 2005): the sample size was increased from 1 g (5 replicates) to 5 g or 10 g (only one sample analysed). It was found that due to the nature and content of the mixed soil/compost/faecal samples that were exhumed from the trenches, it was easier to be less selective of smaller or larger particles when weighing out the 10 g sample than when struggling to accurately weigh out 1 g. Also, it was found that analysing one much larger sample was less time consuming and gave a better estimate of the average egg load in a sample than analysing a number of 1 g samples. This implies that a higher degree of variance in data from samples taken directly from pit latrines might be attributable to the less precise measurement technique.

In addition, 10 samples were incubated at 30ºC in a humid environment for 14 days to see whether undeveloped ova would develop.

5.1.6.5 Challenges

A number of difficulties arose during monitoring. When the young saplings were damaged by goats, each tree had to be individually fenced, which proved time consuming and costly at R31-R52/tree (it was felt that a longer fence around the entire site would have been re-appropriated for other purposes within days and was therefore not contemplated). Vandalism of groundwater monitoring equipment interfered with data collection and for a time problems with flooding rendered data meaningless until a berm wall was built and the area was drained. Piezometers continued to be monitored, but, since the area was inundated by a local mains leakage, the wetting front detectors and watermark sensors were not replaced. After the mains leak had been repaired and the drainage of the site remediated, core samples were taken in the trenched area to determine subsurface nutrient distribution.

5.1.7 Results

Results for the Umlazi entrenchment cover tree growth, changes in the sludge, soil and groundwater and the fate of pathogens.

5.1.7.1 Tree growth

Tree growth was measured periodically over the course of the trial for wattle and eucalyptus trees in the experimental block (Rows 1- 9). For purposes of comparison, tree measurements from the following rows were used because they were all planted in October 2009.

Row 3: Sludge entrenchment
Row 4: Buffer (no treatment - roots possibly accessing sludge in Row 3)
Row 5: Control (no treatment)
Row 7: Fertiliser treatment

Row 4 was measured to monitor whether there was any effect that might suggest that the tree roots could be accessing nutrients from sludge entrenched in Row 3. Measurements gave no indication of enhanced growth in Row 4 over Row 5 (see Figures 5.18 and 5.19). From January 2011 it was no longer possible to measure tree height with any accuracy, so measurements of tree diameter taken at 1.37 m from this point onwards were used for comparison. Due to unauthorised felling of trees for firewood by individuals in the community, as well as mortality, the number of trees for each species dwindled to as few as 6 in some rows by the end of the study, with the result that comparisons were not statistically significant. The trends that were observed were informative, however. Mean values for measurements taken from 17 January 2011 (1.25 years after planting) periodically until 29 March 2012 (2.5 years after planting) are presented below for both species.

Figure 5.17 Eucalyptus trees at planting in February 2009 (top) and in January 2012 (bottom)

For eucalyptus, there was consistent growth for all four rows, with trees planted on entrenched sludge (Row 3) showing consistently higher growth than the buffer (Row 4), control (Row 5) or fertilizer treatment (Row 7) rows.

The advantage experienced by trees with access to sludge appeared to peak at 1.5 years after planting (April 2011) with 75% greater mean stem diameter than control trees and 30% greater stem diameter than fertiliser treatment trees. The advantage over control trees then declined to 46% greater stem diameter at approximately 2 years after planting (August 2011) and 13% at 2.5 years after planting (March 2012).

Note that biomass is proportional to the *cube* of the diameter, being a product of area (which is proportional to the square of diameter) and height (which is linearly correlated with diameter – see Figure 6.51). A difference of 13% in diameter is therefore equivalent to a difference of 45% in biomass.

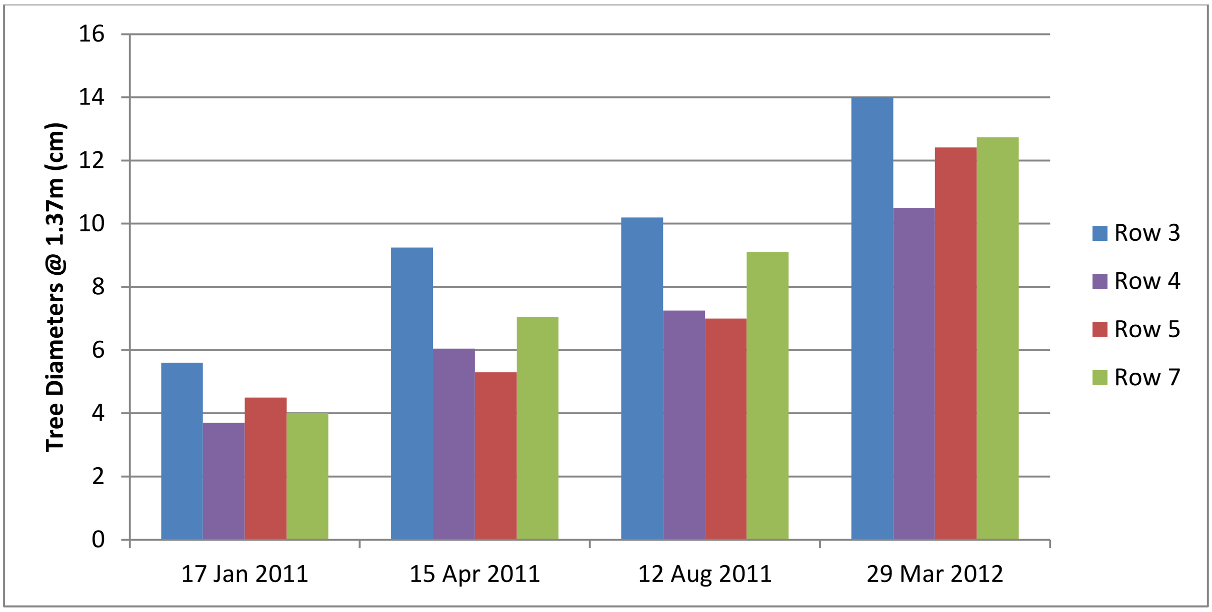


Figure 5.18 Changes in stem diameter (measured at 1.37m) for experimental eucalyptus rows over time (mean values)

Table 5.5 Mean values for stem diameters and relative gains in growth for different treatments for eucalyptus trees

Growth of eucalyptus					
	Diameter at 1.37mm			Percentage greater diameter	
Month	Row 3: sludge	Row 5: control	Row 7: fertiliser	Row 3 over Row 5	Row 3 over Row 7
January 2011	56 mm	45 mm	40 mm	24%	40%
April 2011	93 mm	53 mm	71 mm	75%	30%
August 2011	102 mm	70 mm	91 mm	46%	12%
March 2012	140 mm	124 mm	127 mm	13%	10%

For wattle, the effect of sludge on growth was more limited. While Row 3 (sludge treatment) showed enhanced growth over other treatments on 17 January 2011, the next measurement (15 April 2011) showed an apparent decrease in mean diameter for Row 3. It is possible this could be attributed to the loss of some of the largest trees, but no unauthorised felling of trees was known to happen during this period and it would be more likely that smaller, weaker trees would have been among natural mortalities than the largest trees. It is most likely that the apparent decrease in size is a data collection error, and while this could have occurred in either January 2011 or April 2011, the April data is in line with the trend of subsequent data, while the January data is not. If the trend seen in data from April onwards was extrapolated to January, it would show mean diameter for Row 3 to be less than that for Row 7. Assuming that the 15 April 2011 data is correct, growth in both Row 5 (control) and Row 7 (fertiliser) was consistently greater than growth in Row 3 (sludge) from 15 April 2011 onward. This may have been due, in part, to the fact that wattle has the ability to fix nitrogen from the atmosphere, with the result that a lack of nitrogen in native soil would not represent a limitation to growth for this species to the same extent that it would for eucalyptus.

It was expected, however, that apart from nutrient benefits the sludge would have improved the soil structure over the poor native soil and enhanced conditions for growth on that basis alone. It is unclear why, instead, growth appeared actually more limited in the sludge row than the control and fertilizer rows, with the mean diameter in the control row 18%, 38%, and then 16% greater than mean diameter in the sludge row over successive measurements. In addition, growth in the fertilizer row was inferior to the control row, with trees in native soil performing the best over three sets of measurements. This may be due to some extent to variation in the native soil from one row to the next. It must be stressed, however, that the numbers of trees involved in this particular trial is too small to draw any significant conclusions.

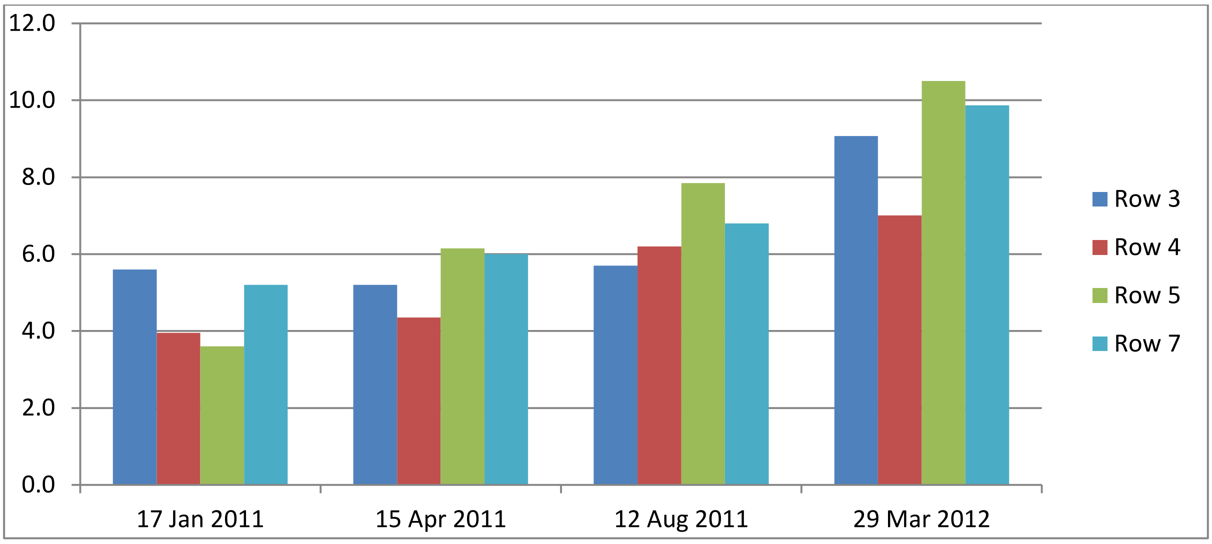


Figure 5.19 Changes in stem diameter at 1.37 m (in cm) for experimental wattle rows over time (mean values)

Table 5.6 Mean values for stem diameters and relative gains in growth for different treatments for wattle trees

Growth of wattle					
	Diameter at 1.37mm			Percentage greater diameter	
Month	Row 3: sludge	Row 5: control	Row 7: fertiliser	Row 3 over Row 5	Row 3 over Row 7
January 2011	56 mm	36 mm	52 mm	56%	8%
				Row 5 over Row 3	Row 7 over Row 3
April 2011	52 mm	62 mm	60 mm	18%	15%
August 2011	57 mm	79 mm	68 mm	38%	19%
March 2012	91 mm	105 mm	99 mm	16%	10%

5.1.7.2 Groundwater

The surface topography of two transects were surveyed on 29 June 2011 in order to determine the water table gradients (Figure 5.20). The first transect ran from the trench site to the stream and the second along the borehole line from BH2 to BH5. The water table lay at a depth of approximately 2m below the entrenchment site and was typically 5m below surface at the boreholes between the site and the river. The flow from along the hill slope was channelled downward to the river as indicated with the expected pathway of water and nutrients in Figure 5.21.

Figure 5.20 Layout of surveyed transects from trench zone to stream and along the borehole line (29 July 2011).

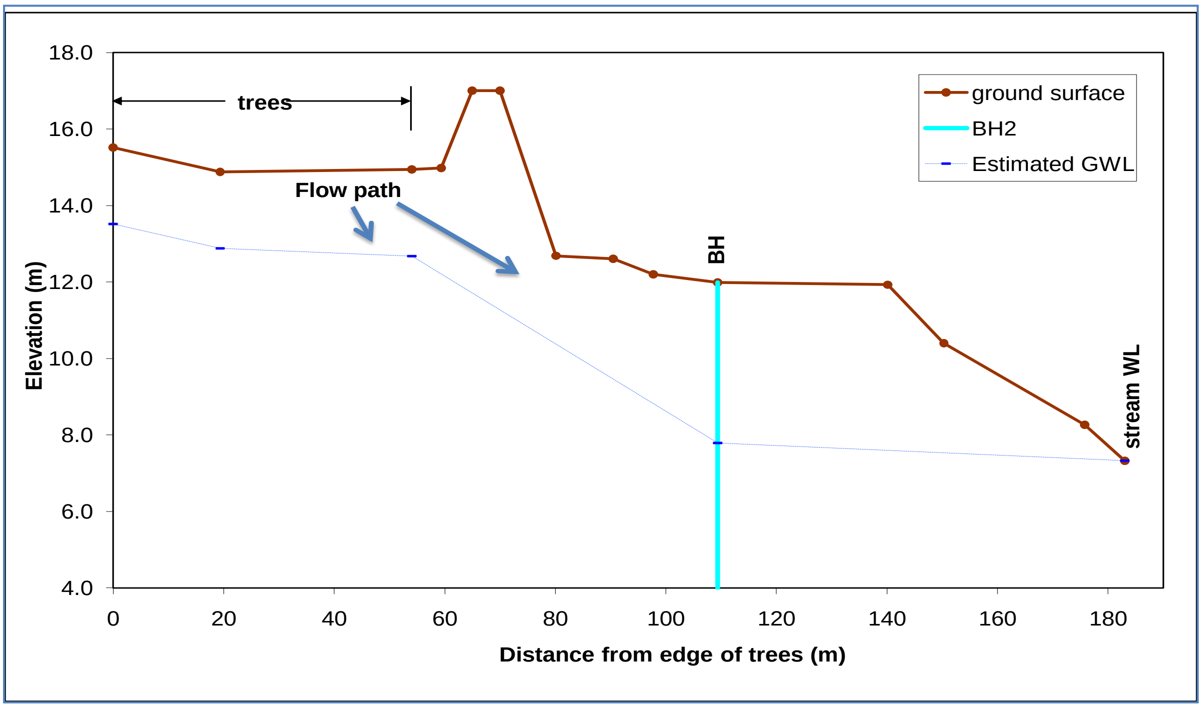


Figure 5.21 Surface topography, expected flow path and groundwater levels at 01 July 2009 at the Umlazi site.

Throughout the monitoring period, December 2008 to June 2011, the water table depth in the monitoring boreholes varied between 3 m and 6 m below surface, being lowest in the dry season of 2010 (Figure 5.22).

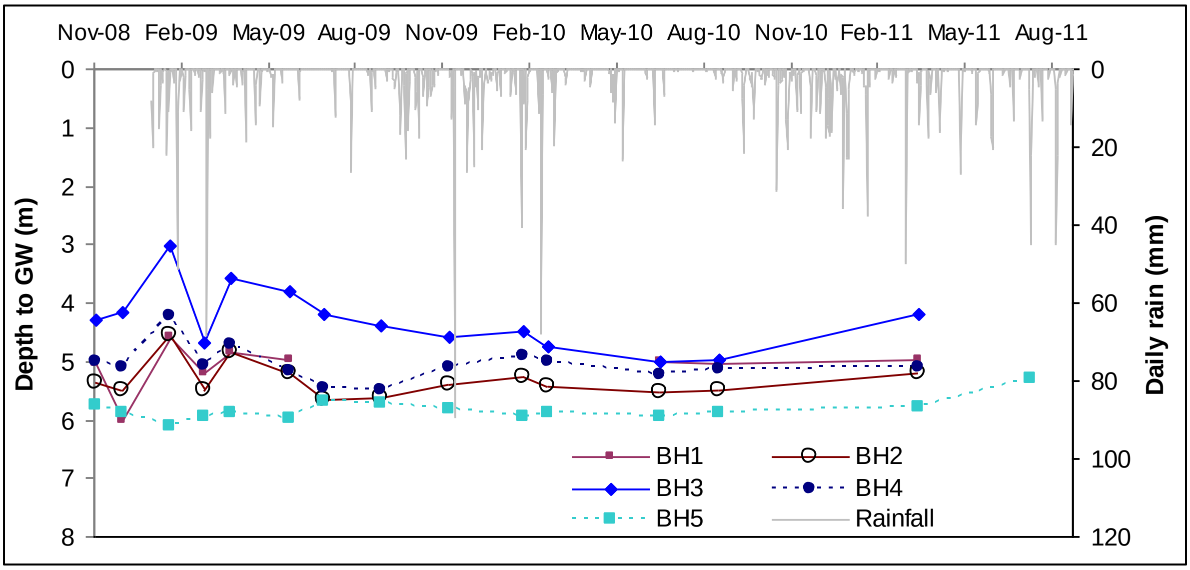


Figure 5.22 Time series of monitoring borehole water levels at Umlazi

❖ *Nitrate*

The nitrate concentration in water samples collected from the piezometers and wetting front detectors, before they were destroyed, ranged from approximately 50 mg/ℓ to as high at 600 mg/ ℓ (Figure 5.23). This can be attributed to their proximity to the entrenched sludge. Nitrate fluctuations in the boreholes, (Figure 5.24), remained within acceptable ranges (less than 44 mg/ ℓ measured as nitrate, EPA drinking water standard) throughout the period of the study, exhibiting only 2 outliers. The first outlier (30 mg/ℓ) was measured in borehole BH5 on 25 February 2009 and was associated with an elevation of NO3 concentrations in the remaining boreholes, but these were all below 5 mg/ℓ on that day. These increases in concentration could be due to a series of dry days during the wet season, resulting in concentrated recharge from the unsaturated soils. The second outlier (36 mg/ℓ) occurred in BH3 on 28 February 2010. None of the other boreholes reflected this increase and thus this value in BH3 should be ignored. Besides similar periods of slightly elevated concentrations, no seasonal pattern could be discerned in the nitrate record.

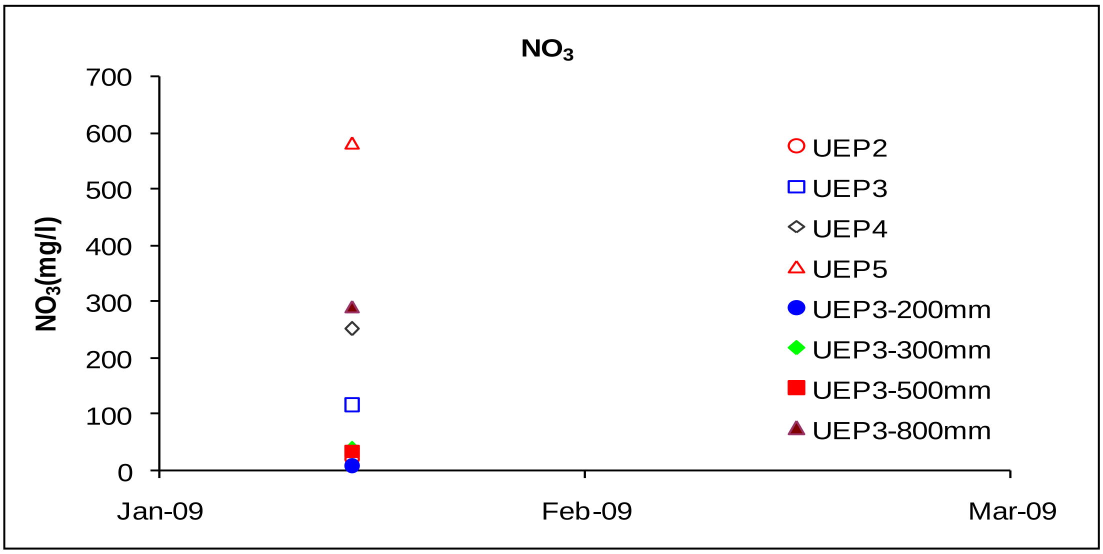


Figure 5.23 Nitrate concentration in piezometers and wetting front detectors before they were destroyed

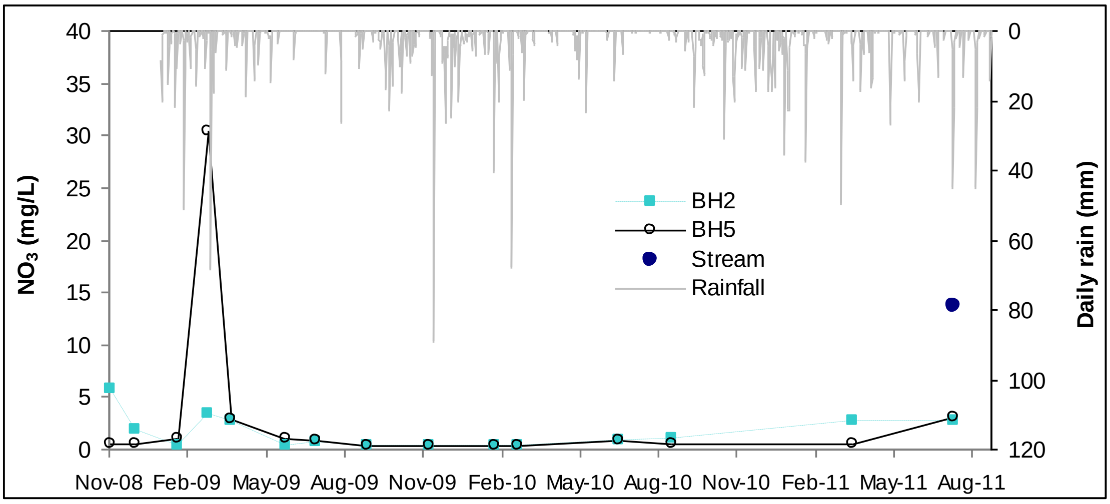


Figure 5.24 Nitrate concentration in boreholes 2 and 5 and the stream

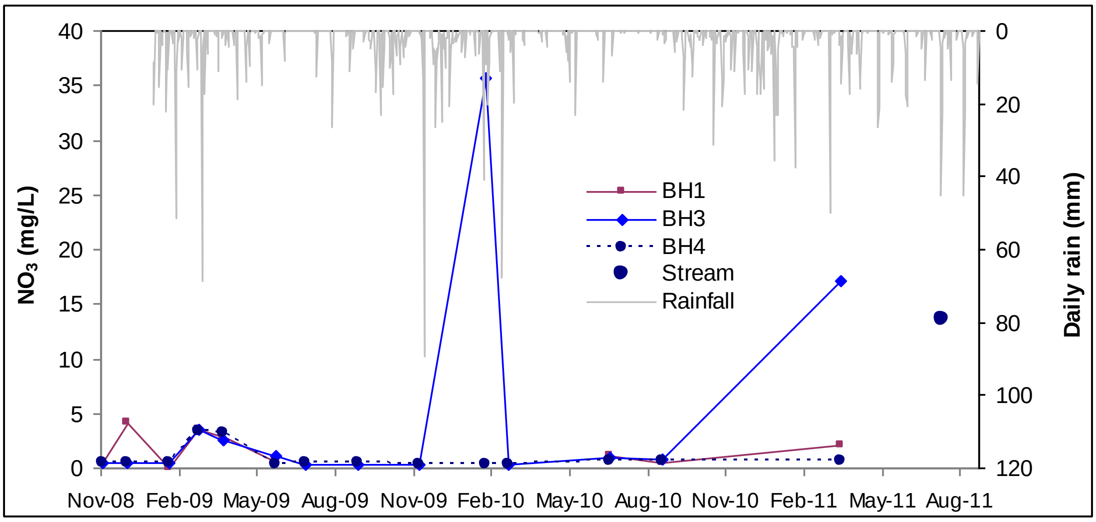


Figure 5.25 Nitrate concentration in boreholes 1, 3 and 4 and the stream

- ***Phosphorus (P)***

Water samples collected from the piezometers and wetting front detectors before they were destroyed were also analysed for phosphorus concentration. Water samples from the wetting front detector at a depth of 800mm (UEP3-800 mm) analysed at the beginning of monitoring indicated the highest phosphorus concentration of 225 mg/ℓ. This wetting front detector was located at the bottom of the trenched VIP sludge and hence accumulated the highest concentration of leachates generated from the sludge. The other wetting front detectors and piezometers indicated phosphorus concentrations ranging between 0.26 and 9.26 mg/ℓ at the initial stages of monitoring (Figure 5.26). The water quality in the boreholes (Figure 5.27) frequently exceeded the eutrophic critical value for orthophosphate, P of 0.05 mg/ℓ, recommended by the EPA and also exceeded the South African Water Quality Guideline for eutrophication (0.025-0.25 mg/ℓ) during the monitoring period. The phosphorus concentration in the stream was 0.17 mg/ℓ, (19 July 2011), and consistent elevated P concentrations in the stream was evidenced by the fact that algae and water hyacinth covered most of the waterway. These were all considerably higher than the recommended eutrophication threshold value (0.05 mg/ℓ) although, in a highly impacted catchment, (Roodeplaat), the DWA has recommended a treatment works effluent orthophosphate concentration of 0.5 mg/ℓ to reduce eutrophication. Borehole phosphorus concentrations were also higher during the dry periods, apparently due to lower dilution and declined during the rainy period as infiltrated precipitation caused dilution.

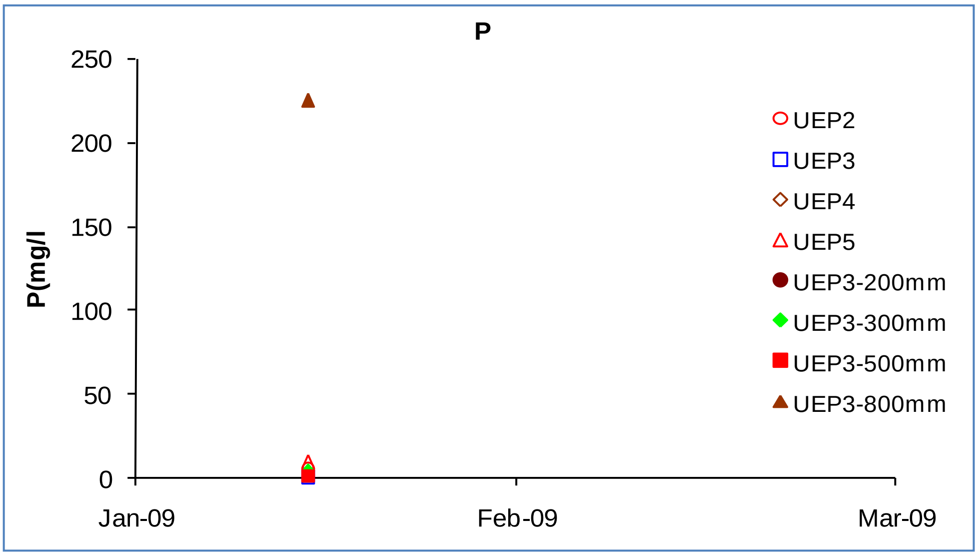


Figure 5.26 Dissolved phosphorus concentration in piezometers and wetting front detectors

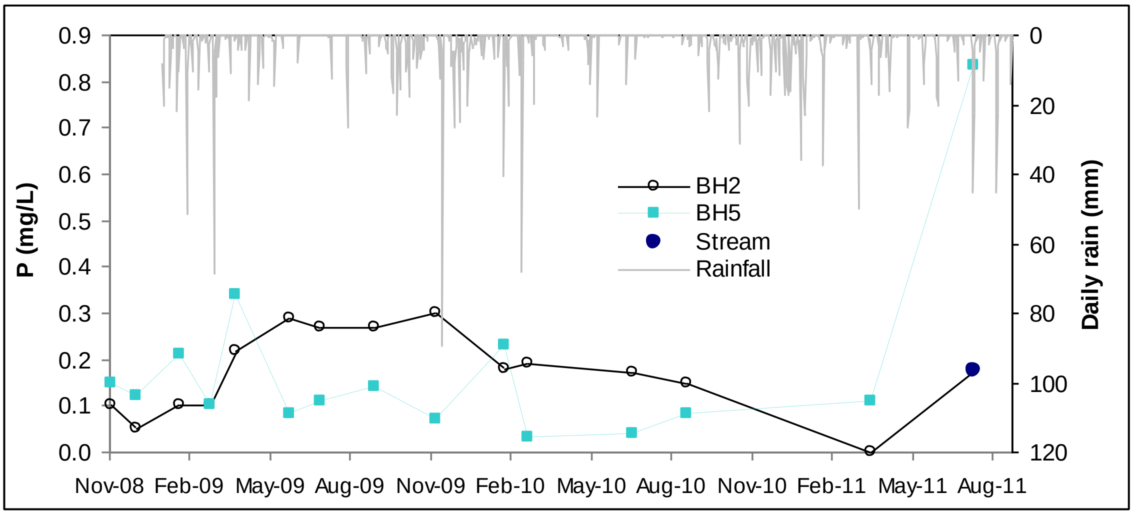


Figure 5.27 Phosphorus concentration in boreholes 2 and 5 and stream

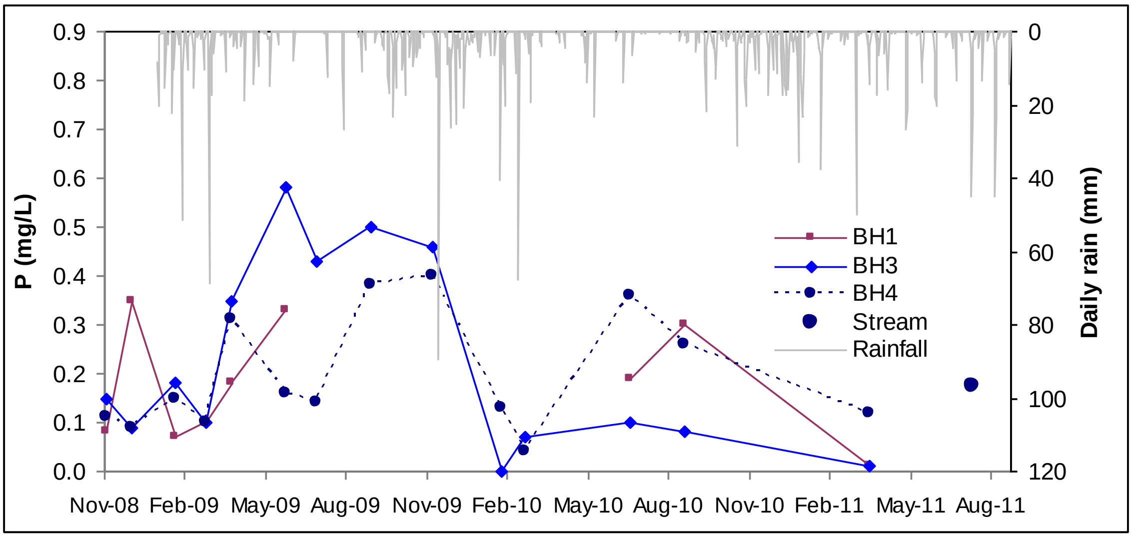


Figure 5.28 Phosphorus concentration in boreholes 1, 3 and 4

Elevated phosphorus levels in the buried sludge indicated leaching of phosphorus from the sludge either by adsorption to and transport with colloids or as organic phosphorus. Nevertheless, these mechanisms did not appear to extend beyond the trenched area as the orthophosphate concentrations in groundwater were considerably lower than those measured in the trenched zone. This was possibly a result of the high phosphorus binding capacity of the materials in the unsaturated zone and the groundwater aquifer between the trench site and the groundwater monitoring location. The entrenchment of sludge cannot be assumed to be related to the level of phosphorus recorded in stream water as refuse and other waste material were observed to be dumped into the stream by nearby residents. The phosphorus concentrations in the groundwater peaked during the dry season of 2009, but only once did they exceed 0.5 mg/ℓ, (BH3 0.58 mg/ℓ on 27 May 2009). However, the dry season concentrations frequently exceeded the maximum concentration for eutrophication of 0.25 mg/ℓ of the South African Water Quality Guidelines in the 2009 and subsequent winters in all boreholes, confirming that elevated P concentrations result from the lack of dilution of percolating water during the dry periods, possibly combined with high P uptake during the wet seasons. However, these data do not indicate an increasing trend such as may result from consistent elevated loading from the trenched area.

❖ ***pH***

The pH is a measure of the acidity of groundwater: the lower the pH, the more acidic the water. At the typical temperature of groundwater, a pH of 7 is considered neutral. Therefore, a pH less than 7 is acidic and a pH greater than 7 means the water is alkaline. The pH state of surface water is especially important since aquatic organisms are able to tolerate only a very narrow pH range. A pH value higher than 8 or lower than 6 for stream water can threaten the survival of aquatic organisms and compromise the diversity of the ecosystem in the stream. High pH levels can occur when algae and aquatic vegetation use CO_2 for photosynthesis. Low pH can also be caused by the respiration of aquatic vegetation or from bacterial decay of organic matter in the water producing high levels of CO_2. Low pH in the water can also cause toxic chemicals to become mobile. Measured pH in boreholes ranged from 6.3 to 7.5 and can therefore be said to have been within acceptable levels. pH levels appear to drop slightly on occasion, possibly in response to recharge rain events (Figure 5.29).

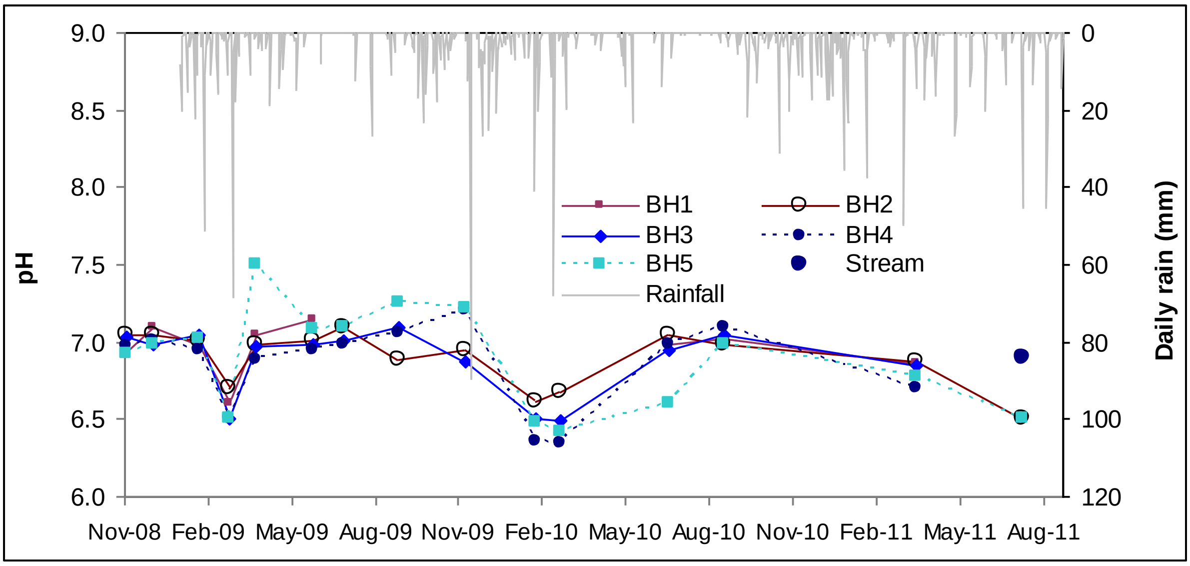


Figure 5.29 pH of borehole water samples and stream

❖ ***Electrical conductivity***

Electrical conductivity (EC) values reflect the concentration of total ions and ranged between 40 and 95mS/m (Figure 5.30). On only two occasions (2 December 2008: 95mS/m in BH1 and 31 August 2009: 85 mS/m in BH2) did the EC exceed a value recommended for drinking and irrigation, 75 mS/m (EPS standard converted from 500mg/ℓ). EC variations did not correlate with nitrate or phosphorus variations and generally fluctuated around the EC of the stream, measured in July 2011.

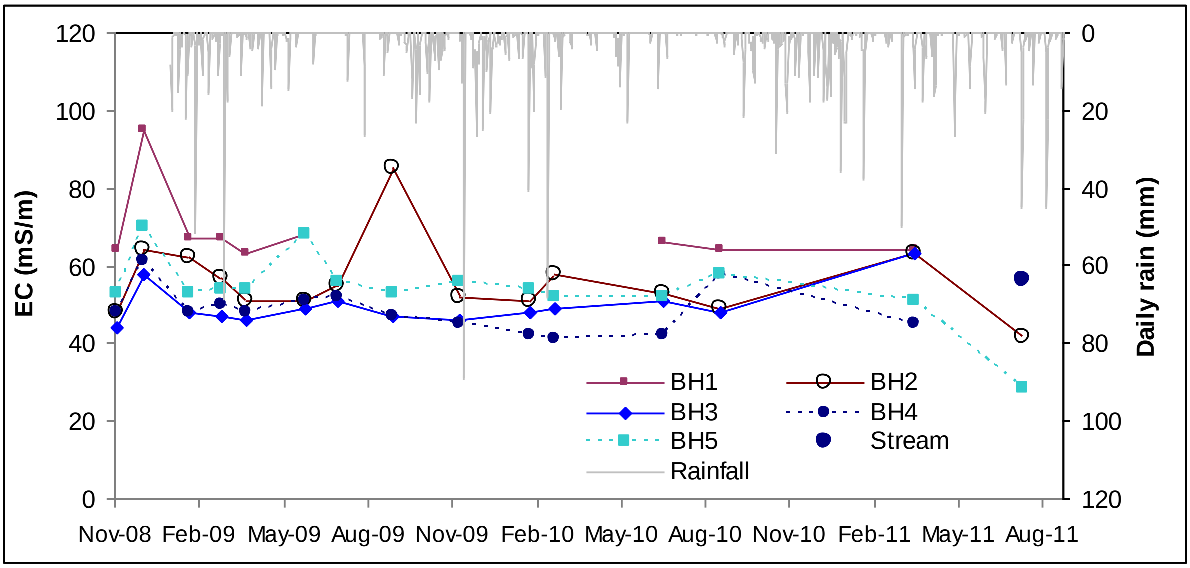


Figure 5.30 EC of borehole water samples and stream

❖ *ERT Monitoring*

Additional Geophysical Electrical Resistivity Tomography (ERT) surveys were conducted one and three years after trenching. In August, 2009, a survey was completed on the same transects as the initial survey.

Electrical resistivity distribution indicated a resistive layer of dry sand above a conductive saturated layer (aquifer) and a resistive bed rock both before and after waste burial. However the saturated layer was less conductive before entrenchment than after (Figure 5.31). The presence of the sludge entrenched above the saturated layer may have contributed to the change in resistivity distribution. The difference in the shapes of the layers was due to noise effect caused by the presence of wires covering planted trees after entrenchment.

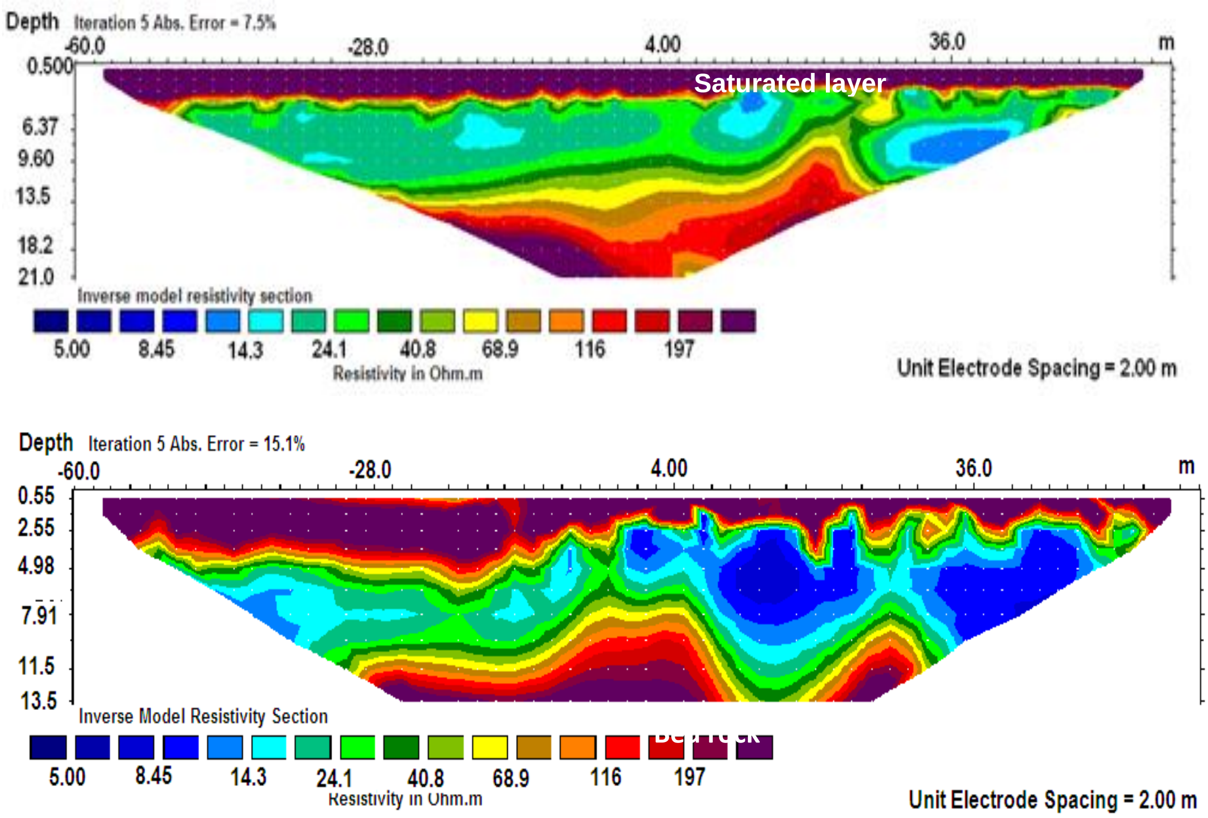


Figure 5.31 Transect 1 inverse model resistivity section before entrenchment in September 2008 (top) and a year later in August 2009 (bottom)

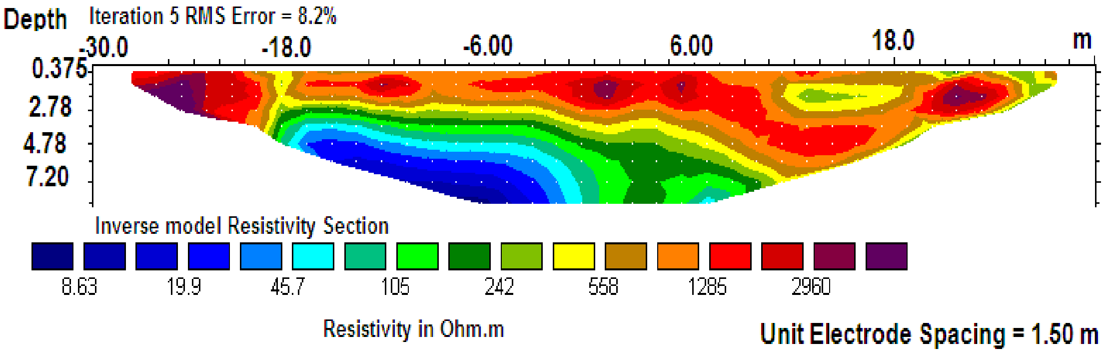


Figure 5.32. Umlazi Transect 2 resistivity before entrenchment in September 2008

Another ERT survey was conducted in July, 2011 on four new transects. During this survey, water was occasionally poured on the dry, sandy soil to improve contact with the electrodes. The survey indicated a resistive 2m layer of dry sand (>2500 Ω.m) lying above a conductive saturated layer (aquifer) (<300 Ω.m).

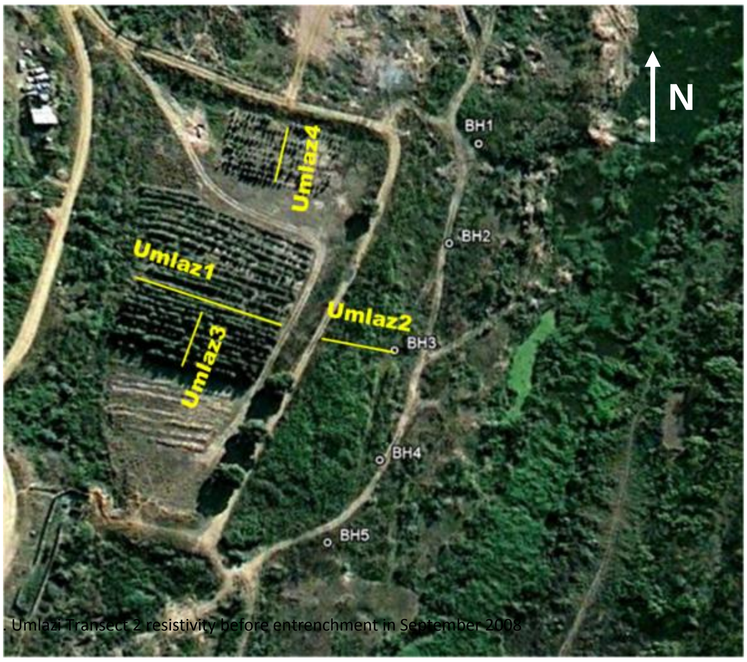


Figure 5.33 Transects Umlaz1-4 used for ERT study in July 2011

Transect Umlaz1 (5.33) showed lateral variations of resistivity in the top 2m layer which may have resulted from the presence of the trees and root activity (Figure 5.34). Transect Umlaz2 indicated a relatively moist sandy soil (1500 Ω.m) in the top meter (Figure 5.34). In Transect Umlaz3, the material in the trenches appeared to have only marginally lower resistivity than the topsoil at the same depth, which may have been caused by the presence of the trees or the sludge within the trenches; high reeds at this location were also indicative of the presence of water close to the surface. Umlaz4 indicated a resistive layer of dry sand and rock (>2500 Ω.m) over 3m depth and at about 5m depth shows very low conductivity, suggesting the presence of water at the south side of this transect (Figure 5.34). This coincides with the water levels recorded in the boreholes in this area.

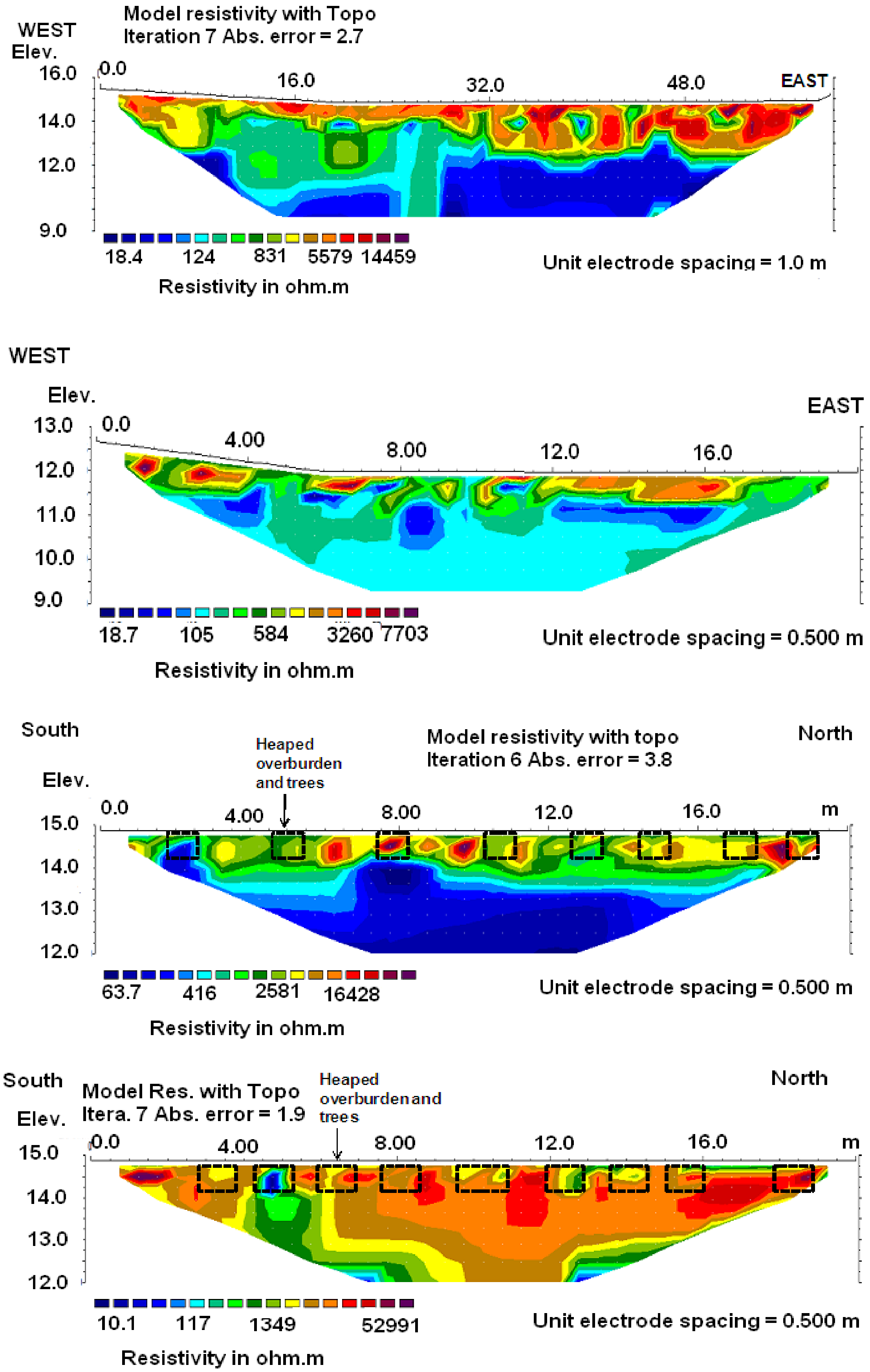


Figure 5.34 ERT inverse model resistivity section Umlazi 1-4 from top to bottom (July, 2011)

5.1.7.5 Modelling to predict trends over time

A two-dimensional finite element grid was generated for a subsurface cross-section between the trench site and the stream using the HYDRUS-2D model (Šimůnek *et al*., 1999). ERT profiles were applied to define the spatial dimensions of the outer boundary as well as the distribution of different soil materials and the geological layout of the subsurface. A schematic cross section of the subsurface, showing the trenches, stream and boundary condition is shown in Figure 5.35. Crucial for the model application is the setting of reasonable boundary conditions. The distribution of material types is shown in Figure 5.36. Three groups of input variables were used to model soil water and nutrient fluxes:

- Initial conditions for topography, hydrologic characteristics of the soil materials, distribution of hydraulic head, and location and concentration of solutes.
- Boundary conditions describing the rules that govern the movement of water into and out of the perimeter of the modelled area.
- Time dependent conditions including changing rainfall, vegetation water-use rates model characteristics that influence the time-step selection and the iterative process used to evaluate groundwater and solute movement over the duration of the model run time.

Saturated water content, residual water content, water retention parameters and saturated hydraulic conductivity, values in the cross section were assigned based on clay, silt and sand percentages derived from subsurface sampling. In the near surface trenched area, measured hydraulic characteristics were used.

Table 5.7 Fitted hydraulic characteristics (van Genuchten and Mualem) of different soil layers and PLW sludge used in the initial simulation.

Material	Class	θ_r cm^3/cm^3	θ_s cm^3/cm^3	a (cm^{-1})	n(-)	K_S $cmday^{-1}$	L(-)
1	Loamy sand	0.057	0.41	0.124	2.28	350.2	0.5
2	PLW	0.0999	0.4972	0.01	1.2685	22.1	0.5
3	Clay loam	0.095	0.41	0.019	1.31	6.24	0.5

θr = residual volumetric water content (cm3/cm3),
θs = volumetric water content at saturation (cm3/cm3),
Ks = Hydraulic conductivity at saturation (cmday-1),
α = approximation for the inverse of the air entry pressure (cm^{-1}),
n = pore size distribution index and
L = pore connectivity parameter.

Water flow and solute transport were modelled within the flow domain comprising the 180 m between the sludge burial site and the nearby stream, and a profile depth of 14 m.

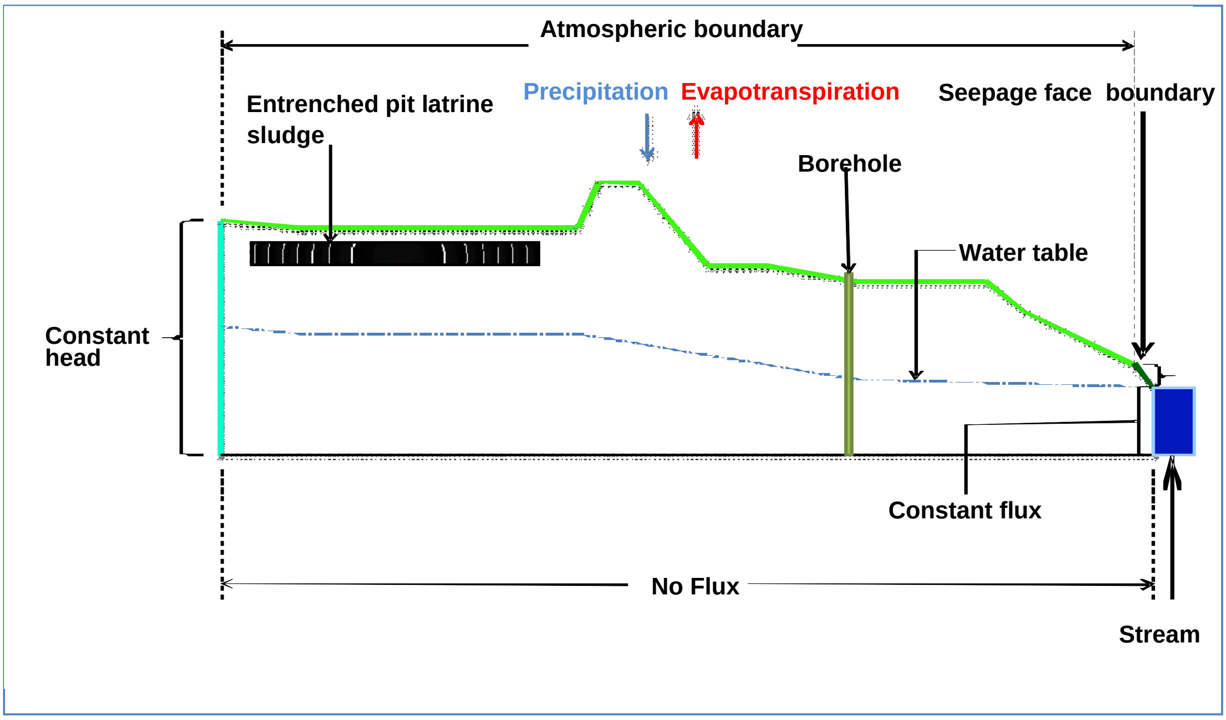


Figure 5.35 Conceptual model and boundary conditions

Three materials or layers were distinguished, pit latrine sludge, loamy sand and clay loam and are distributed as shown in Figure 5.36.

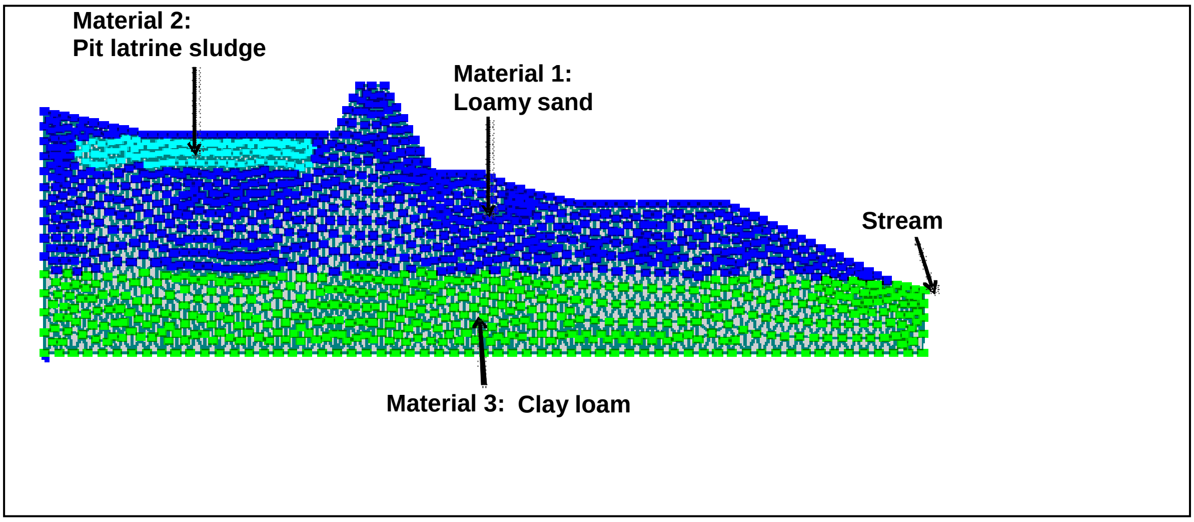


Figure 5.36 Material distributions in profile at Umlazi site

Daily potential evapotranspiration was calculated based on temperature and humidity data from the nearby weather station using the ACRU model. Initial concentrations of solutes (nitrate and phosphorus) were set at zero in the whole domain and only the sludge application area was set at a uniform value of 580.89 mg/ℓ nitrate and 225.25 mg/ℓ phosphorus, based on observations in the trenches. The minimum permissible soil potential at the surface, input as a constant value to restrict soil water evaporation when the soil surface becomes dry, was set at 15000 cm. Every triangle (or element) in the finite element space is formed by three nodes. On the boundary, these nodes are set with specific water flux and solute transport conditions (Figure 5.37). The light green boundary indicates an atmospheric boundary condition with precipitation as influx and actual evapotranspiration as out flux. The dark green boundary at the stream is a seepage face that allows water to leave the profile when positive water pressures develop at this boundary. Because these boundary conditions assume unhindered outflow, it could not be assigned to the entire vertical side, or the profile would have drained completely. In reality, water backs up on the fractured crystalline bedrock. To imitate this process a constant flux boundary condition is applied that enables outflow at the bottom. Because crystalline bedrock has small porosity (between 0-10 % according to Ward & Robinson, 2000) and small coefficients of hydraulic conductivity between 10^{-5}-10^{-10} m/s the applied fluxes were set at 0.9 cm/d. In order to observe the simulated pressure heads, water content and nutrient concentrations, 3 observation points were positioned in the grid at selected positions in the profile (Figure 5.37).

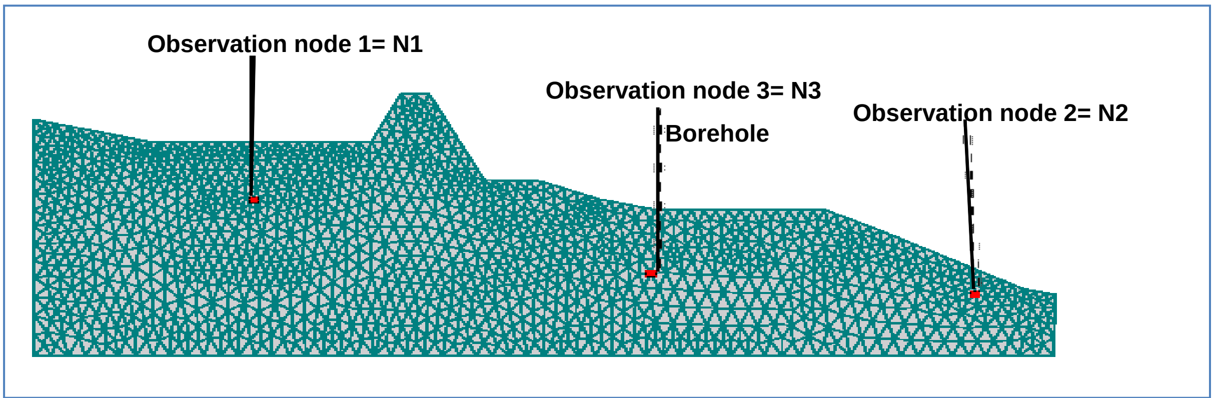


Figure 5.37. HYDRUS-2D finite element grid of soil profile of the Umlazi site with observation nodes

Specific physical and chemical parameters (phosphorus adsorption isotherms, denitrification rate constant, and dispersion coefficient) were used to simulate the movement of nitrate and phosphorus through the soil profile. Concentrations of nitrate (580.89 mg/ℓ) and phosphorus (225.25 mg/ℓ) measured prior to the simulation (1 January 2009) were assumed to be constant across the pit latrine sludge in the trench and were used as the initial condition for solute transport. The conversion of nitrate to N_2 and N_2O (denitrification) was determined as a first-order reaction (Bohn *et al*., 1985) using the denitrification rate constant based on travel time and nitrate adsorption by soil (Reneau *et al*., 2005). The analysis considered the Umlazi soil profile to 14 m below surface. Root water uptake was not simulated initially, since the tree water uptake was considered low over the first year of simulation. This also comprised a worst case scenario for migration of nutrients towards the stream. Longitudinal dispersivity and transverse dispersivity values, adapted from Reneau *et al*, 2005, were selected for the simulation. These parameters determine the rate of spreading, (and thus dilution) of the nutrient solute plume, in the direction of flow (longitudinal) as well as orthogonal to the flow path (transverse), as it moves in the subsurface.

Table.5.8. Solute transport and reaction parameters for materials outside trench, (adapted from Reneau *et al*., 2005)

Material	Bulk density	NO_3 - N				PO_4^{-3} - P
		SinkL1' (d^{-1})	L (cm)	T (cm)	D(cm^2d^{-1})	K_d ($mg^{-1}m\ell^3$)
1	1.6	0.005364	20	4	0.84	29.70
2	1.21	0.005364	10	2	0.84	34.29
3	1.27	0.005364	10	2	0.84	34.29

SinkL1': first-order degradation rate constant for dissolved phase in the decay chain reaction (day^{-1});
L: longitudinal dispersivity (cm);
T: transverse dispersivity (cm);
D: molecular diffusion coefficient in free water (cm^2day^{-1});
K_d: Langmuir PO_4^{-3} – P adsorption isotherm coefficient ($mg^{-1}.ml^3$).

Simulation results

The simulations results can be taken as illustrative of the likely solute plume migration and dispersion. Wetting and drying due to rainfall and soil water evaporation resulted in the formation of wet and dry zones in the near surface of the soil profile. However, the flow regime quickly responded to the constant head boundary condition on the hillslope side of the trench area. Within the first 100 days of simulation all the observation nodes are at or near saturation.

The dewatering of the sludge in the simulations was confounded by the movement of the water table from an initial condition to a constant head boundary condition which resulted in a steady state groundwater level some 2 m below the surface. The simulated sludge material subsequently wetted up due to the proximity of the water table as shown in the series of water content distributions in Figure 5.38. This was similar to conditions on site where elevated water tables were caused by a leaky sewer pipe upslope of the trenched site and by poor surface water drainage in the first period of monitoring. Nevertheless, fluxes of water through the buried waste were evident in the migration of the nitrate from the trenches into the unsaturated zone and groundwater as shown in Figure 5.38. With the growth of the trees and subsequent wetting and drying cycles, the trenched material has been observed, through auguring, to be drying and weathering. This follows observations from trenched sludge experiments elsewhere. The first entrenchment trials conducted in the 1970s showed that gravimetric flow prevailed and sludge rows dewatered from the top down. In fact, 19 months after the entrenchment of raw, limed sludge in sandy soil the top 20% had weathered to a peat like consistency (Walker, 1974). Another trenching study conducted between 1977-1980 (Sikora, *et al*., 1982) in well-drained silt loam soils similarly found the largest amount of sludge dewatering to have occurred in the first 20 months, but overall, the amount of dewatering that occurred was less than that observed in sandy soils due to the slower percolation through silty soils. Regardless, the sludge pack dewatered from the top down. Varying soil conditions impact the transport of rainfall and, by association, the leaching of soluble components out of the trench; this effect was expected to be immediate and pronounced in the sandy soils characterised by high hydraulic conductivities at the Umlazi site. Hence the characterisation of the buried pit latrine waste as initially containing dissolved nutrients of the concentrations measured in the first few months and thereafter being subject to rain water leaching and soils water evaporation, without further generation of nutrients from the buried waste, is justified.

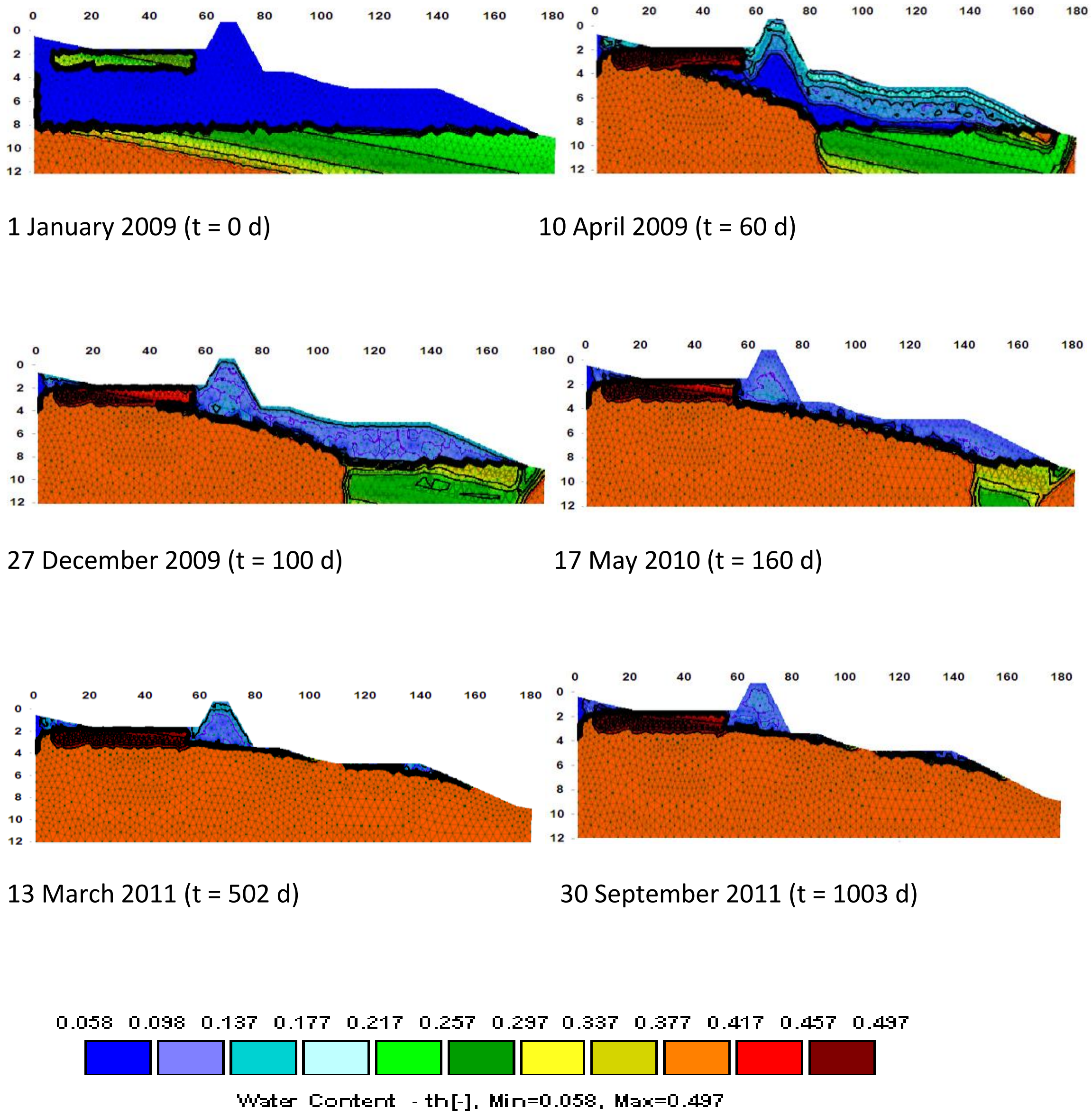


Figure 5.38 Simulated water content distributions in the flow domain for 6 characteristic times during the period from 1 January 2009 to 30 September 2011 at Umlazi (x-axis represents depth of profile and y-axis represents distance to stream in m).

The progress of the nitrate plume is clearly illustrated in the simulated profiles shown in Figure 5.40. Although the simulation was started only in January 2009, the nitrate plume had migrated outside of the burial site by the winter of 2009 (20 June 2009 in Figure 5.40). The dispersion of the original concentrations was also evident as dilution due to mixing, adsorption and chemical modification occur in the migration pathway. By 17 May 2010, the simulation revealed plume concentrations to be similar to background sources.

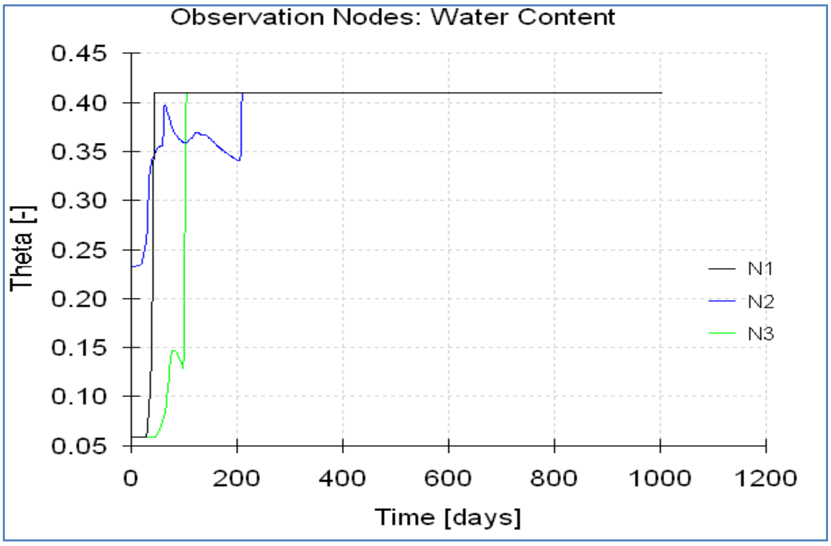


Figure 5.39 Simulated water content at observation nodes

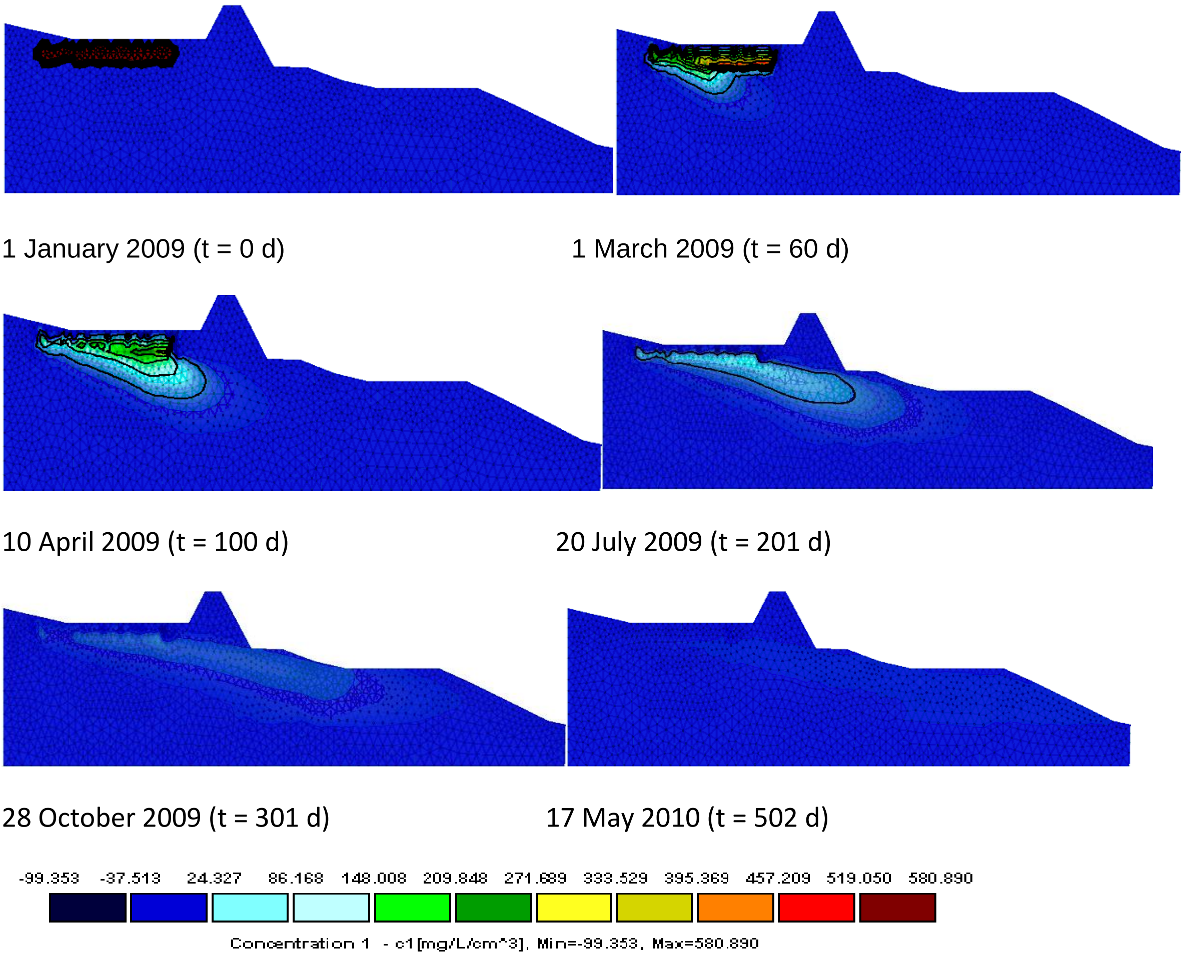


Figure 5.40 Simulated NO3 concentration distributions in the flow domain for eight characteristic times during the period from 1 January 2009 to 30 September 2011 at Umlazi

The arrival of the plume and the dispersion and dilution process was also evident in the time history of nitrate concentrations at the observation node points. At observation node 1, below the waste trenches, (Figure 5.37), the nitrate concentration peaks within the first 100 days at 100 mg/ℓ (Figure 5.41). However, by the time the peak of the plume reached the borehole observation site, (node 3), 200 days later, the maximum concentration had been reduced to 7 mg/ℓ. Nitrate concentrations peaked at the near-stream observation node (node 2) a further 200 days later and peak nitrate concentrations were below 1 mg/ℓ.

Comparing the simulated nitrate concentrations at the boreholes with the observed series (Figure 5.42) reveals similar concentrations between simulated and observed time series, so the dilution, dispersion and change processes appear to have been simulated accurately. The simulated peak concentrations seem to have a delayed response compared to the measured data. This could be due to the simulations starting in January 2009, while the wastes were buried in late 2008. Nevertheless, the simulations confirm that the nitrate plume from the trenched site does not elevate nitrate concentrations in the groundwater above natural levels.

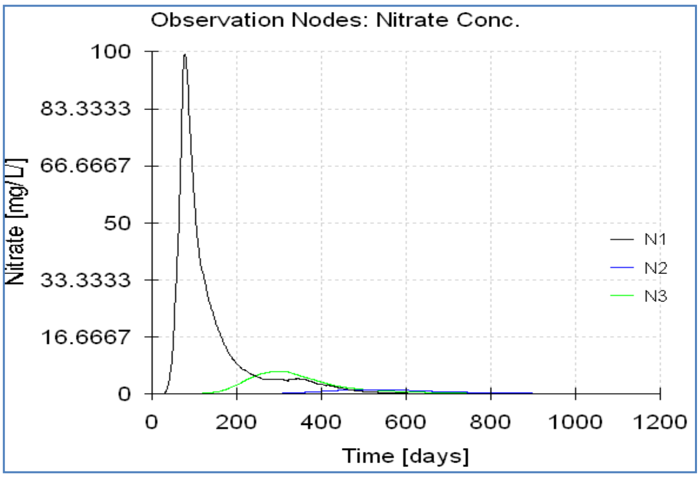


Figure 5.41 Simulated nitrate concentration at nodal points

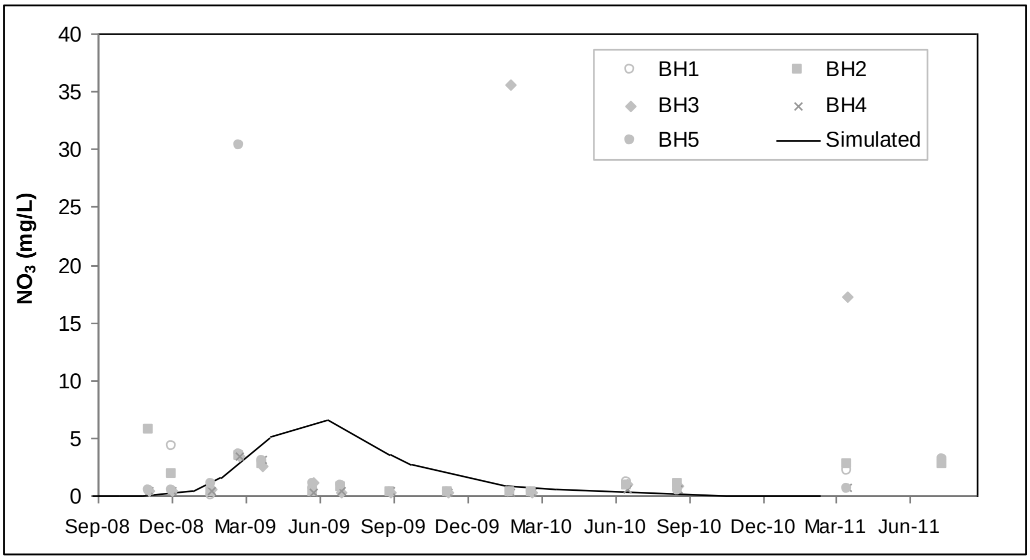


Figure 5.42 Observed (all boreholes) and simulated (node 3) nitrate concentration

Similar observations can be made for the phosphorus plume, shown in Figure 5.34. Due to the strong adsorption of P to the soils, concentrations of the dissolved phase continue throughout the simulation period. However, simulated concentrations were systematically reduced along the flow pathway (Figure 5.44) and were well below those observed at the borehole sites (Figure 5.45).

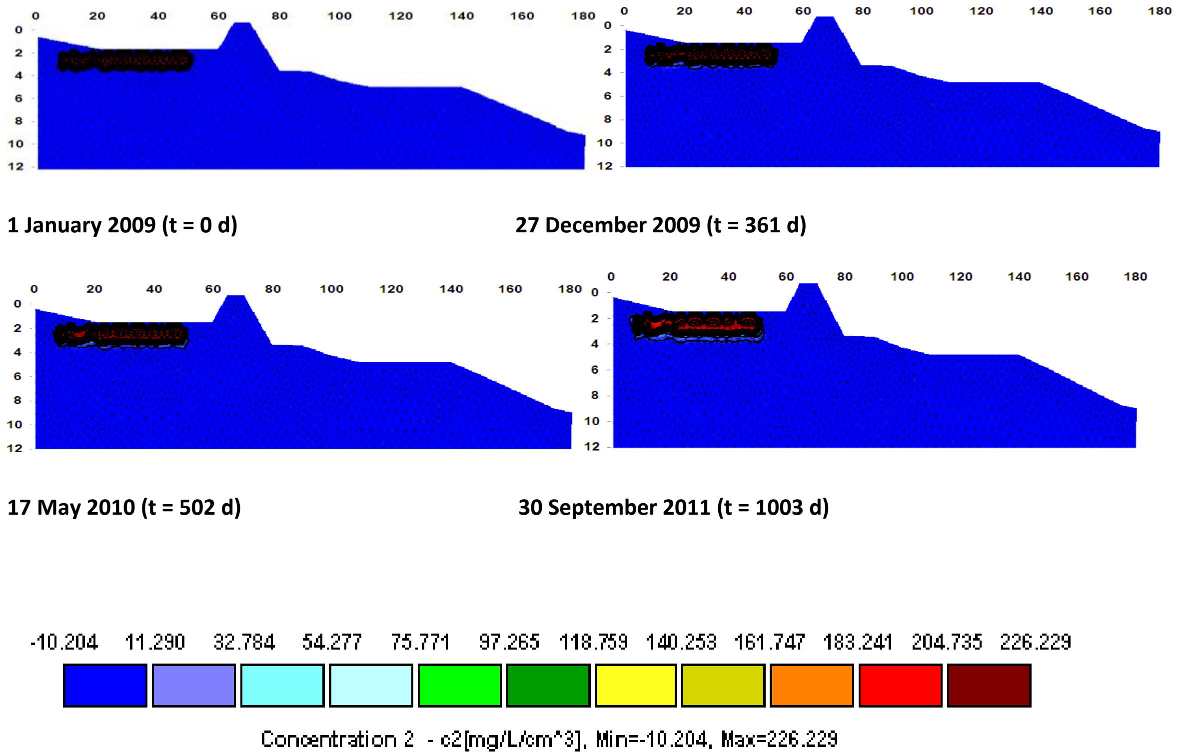


Figure 5.43 Simulated phosphorus concentration distributions in the flow domain for four characteristic times during the period from 1 January 2009 to 20 September 2011 at Umlazi. (x-axis represents depth of profile and y-axis represents distance to stream in m).

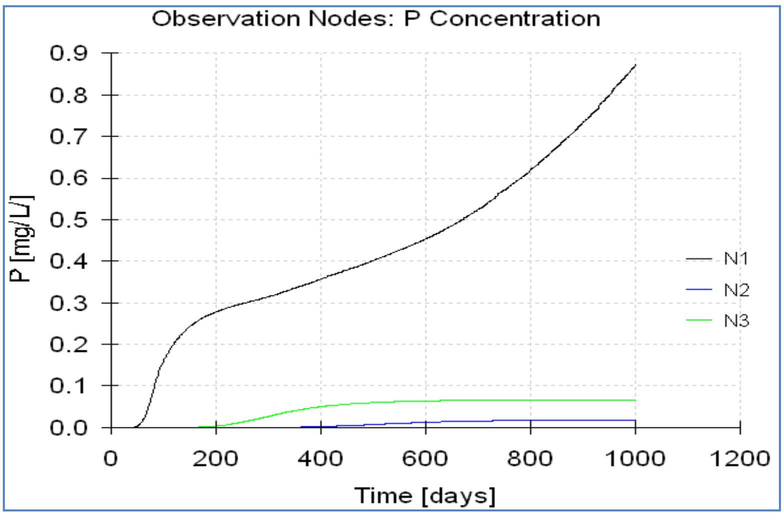


Figure 5.44 Simulated phosphorus concentration at nodal points

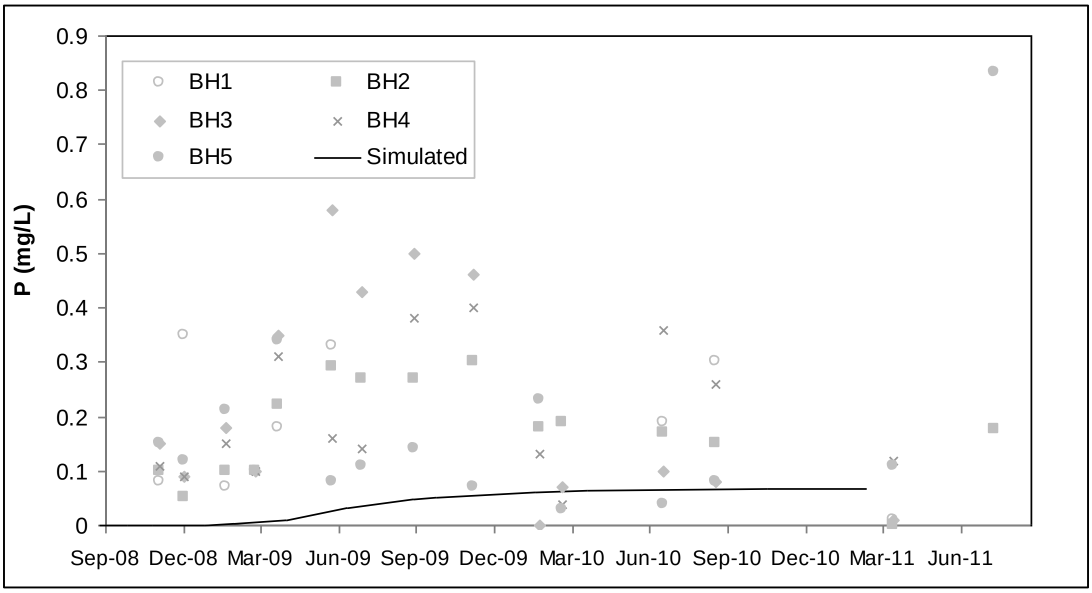


Figure 5.45 Observed (all boreholes) and simulated phosphorus concentration

5.1.7.3 Sludge/soil characteristics

Table 5.9 presents a summary of the measured characteristics of pit latrine material delivered to the Umlazi E entrenchment site (reported as average pit contents), and of material that had been entrenched for 12 months and 28 months. These values are compared to similar measurements performed on samples taken from the bottom of 16 pit latrines after they had been emptied for comparative purposes.

Table 5.9 Summary of VIP sludge contents at different layer within the pit. Data are presented as mean value ± 95% conf. interval, (n = number of observations) [min, max]

Parameter	Units	Average pit contents	Sludge entrenched for 12 months	Sludge entrenched for 28 months	Sludge taken from the bottom of pit latrines
Moisture	g H_2O/g sample %	75 ± 1 (n=90[2]) [63, 89]	54 ± 3 (n=75) [25, 66]	36 ± 17 (n=10) [6, 56]	67 ± 3 (n=48) [35 , 87]
VS	gVS/g dry solids %	59 ± 3 (n=90) [18, 86]	29 ± 2 (n=75) [12, 51]	23 ± 13 (n=10) [0.4, 49]	37 ± 4 (n=48) [4, 74]
COD	mg COD/g dry solids	0.25 ± 0.03 (n=60) [0.10, 0.53]	0.14 ± 0.01 (n=60) [0.10, 0.23]	0.12 ± 0.05 (n=10) [0.05, 0.22]	0.24±0.03 (n=48) [0.09,0.49]
Biodegrad-ability	g biodeg. COD/ g total COD%	27 ± 4 (n=7[3]) [22, 32]	15 ± 5 (n=5) [12, 19]	not measured	17 ± 6 (n=8) [8, 35]

[2] Replicate measurements of moisture, VS and COD for independent samples are reported as though they were different observations because of the heterogeneity of the material.
[3] Only one observation is recorded for each sample tested since only one biodegradability test was performed on each sample, although replicate COD values were obtained from each test for the calculation of biodegradability.

COD and volatile solids are both used as markers for biological degradation processes, since the concentration of COD, and the relative fraction of volatile solids (those solids that ignite and burn off a sample held at 550ºC) will both decrease with time as a result of biodegradation. Analysis of total COD, volatile solids content and moisture content showed significant scatter in all data sets, which is consistent with the highly heterogeneous nature of pit latrine contents. However, the general trend for all three data sets is an initial high value of analyte, followed by a period of rapid decrease in the first 6-12 months, and thereafter followed by a much slower, (possibly not significant) decline in concentration (Figure 5.46). In each case, the variance for the measurement was very high (Figures 5.46, 5.47, 5.48). It was found that biodegradation does result in a decrease in COD concentration and the fraction of dry solids that are classified as ignitable and therefore possibly biodegradable. Results are presented on a dry mass basis (i.e. mass of COD/mass of dry sample); this is to allow direct comparison of values from different samples by eliminating the effect of dilution with moisture.

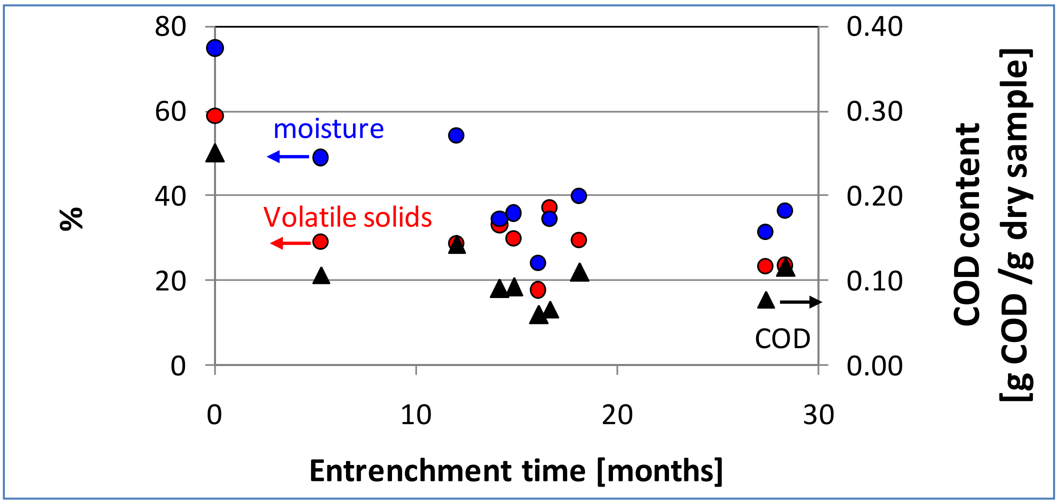


Figure 5.46 Total COD (black triangle), volatile (ignitable) solids (red circle) and moisture content (blue circle) of VIP sludge after a period between 0 and 30 months of entrenchment.

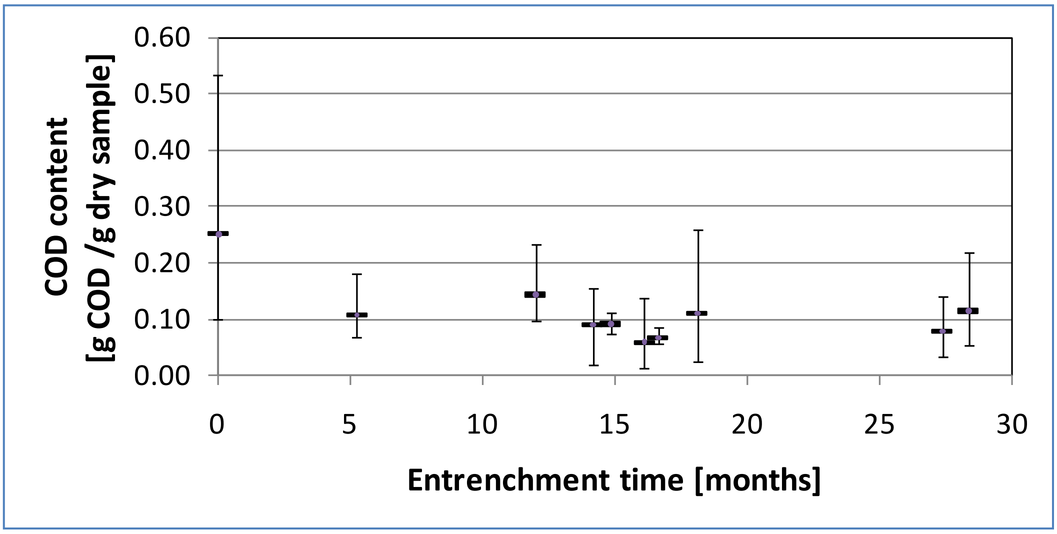


Figure 5.47 Total COD concentration in sludge samples as a function of period entrenched. Error bars indicate maximum and minimum values measured.

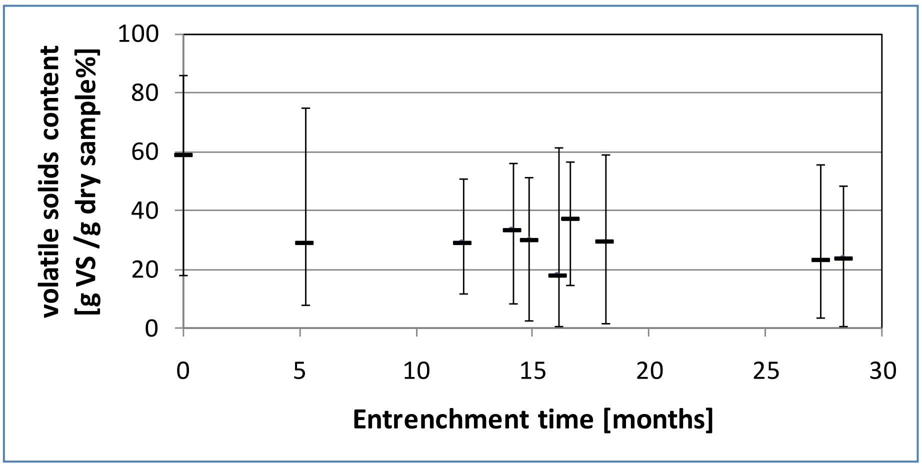


Figure 5.48 Fraction of total solids that are volatile in sludge samples as a function of period entrenched. Error bars indicate maximum and minimum values measured.

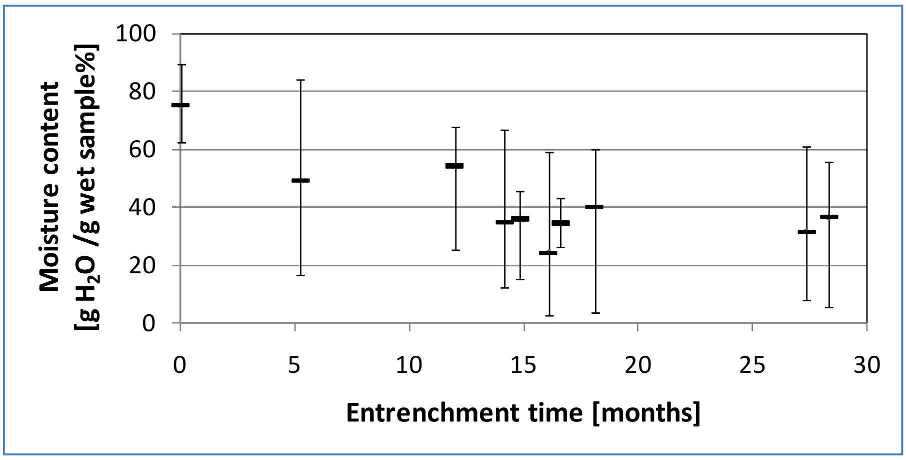


Figure 5.49 Moisture content in sludge samples as a function of period entrenched. Error bars indicate maximum and minimum values measured.

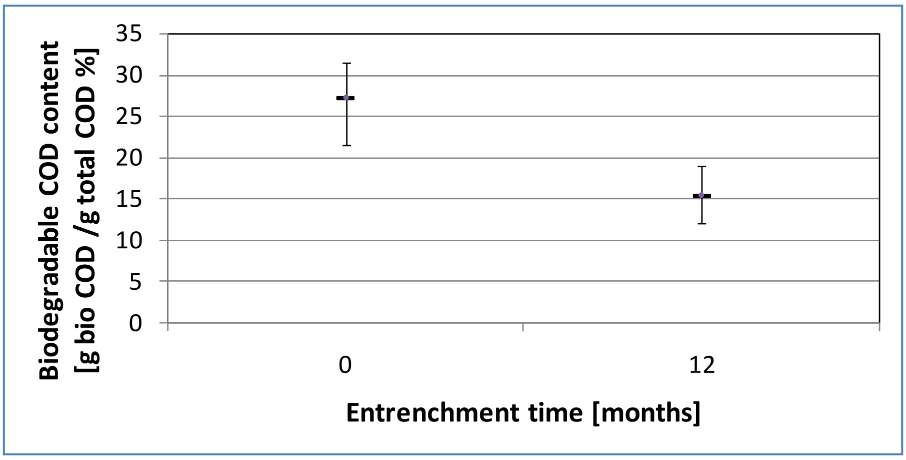


Figure 5.50 Aerobic biodegradability of pit latrine samples before and after 12 months of entrenchment. Error bars indicate maximum and minimum values measured.

Figure 5.46 shows that the fraction of COD that can be aerobically biodegraded in a batch test decreases from an average value of 27% biodegradable COD in the fresh pit sludge to 15% after being buried for 12 months. This confirms that a significant amount of biodegradation occurs during entrenchment.

Theoretically, moisture content can leach into or out of pit latrine contents as a result of rain or groundwater ingress; thus it is conceivable that the moisture content in the entrenched sludge could show significant fluctuations due to seasonal changes. In fact, the moisture content of the entrenched sludge seems to track the general decrease in COD concentration. The soil at the burial site had good drainage properties, and the water table was found to be below the level at which sludge was buried. It is therefore proposed that moisture loss may have accompanied biological degradation and that the rate of reduction in moisture content is a function of biodegradation rate (contrary to the situation within pit latrines).

For all of the analytes presented, the lowest values reported in this study are lower than the equivalent concentrations measured in samples taken from the bottom of a pit latrine. This means that either (i) a greater degree of stabilisation occurs in pit sludge that is buried in an entrenchment for 3 or less years, than occurs in the pit in 14 years of operation; or (ii), the amount of sand that is entrapped in samples taken from entrenchments dilutes the measured concentration of solids. The reduction in biodegradability relative to that measured in the bottom of a pit latrine could not be accounted for by dilution with sand, since addition of sand would dilute both total and biodegradable COD. However, the variance in the method for measurement of biodegradability in pit latrine samples is inherently large, and the measured value (16%) is not much larger than the corresponding value of entrenched sludge after 12 months (15%). Differences for COD, volatile solids fraction and moisture content are significantly lower than the equivalent bottom-of-pit samples but these may be influenced by mixing with sand. Therefore, these results indicate that it is possible that the action of soil fungi can break down pit latrine content further than is achievable in a pit latrine, but the data is not sufficiently precise to prove the hypothesis.

Nitrogen is an essential nutrient for plant growth; the potential value of entrenching sludge with agroforestry is that nitrogen and possibly phosphorus present in the sludge may be a slow-release fertiliser for plant growth. If significant further biodegradation of sludge occurs in the entrenchment, it is conceivable that nitrogen and phosphorus will be released into the soil and be available for plant uptake. Nitrogen content in the sludge before burial and after significant periods of entrenchment were measured. Nitrogen was measured as Total Kjeldahl Nitrogen (TKN) and as Free and Saline Ammonia (FSA – NH_3/NH_4^+) according to Standard Methods (APHA, 1998).

It was found that an order of magnitude difference existed between the amount of TKN in the sludge before entrenchment and after a period of 29 months. This amount of nitrogen may have been released into the soil and become available for nitrogen uptake in plants. There is no similar decrease in FSA concentration. This is because FSA is water-soluble and therefore will drain out of the pit with any moisture flow. Thus FSA may be regarded as an intermediate that is controlled by the rate of biodegradation of organically bound nitrogen (reporting as TKN) and the rate at which it is drained away. Therefore, FSA concentrations may fluctuate with time and location.

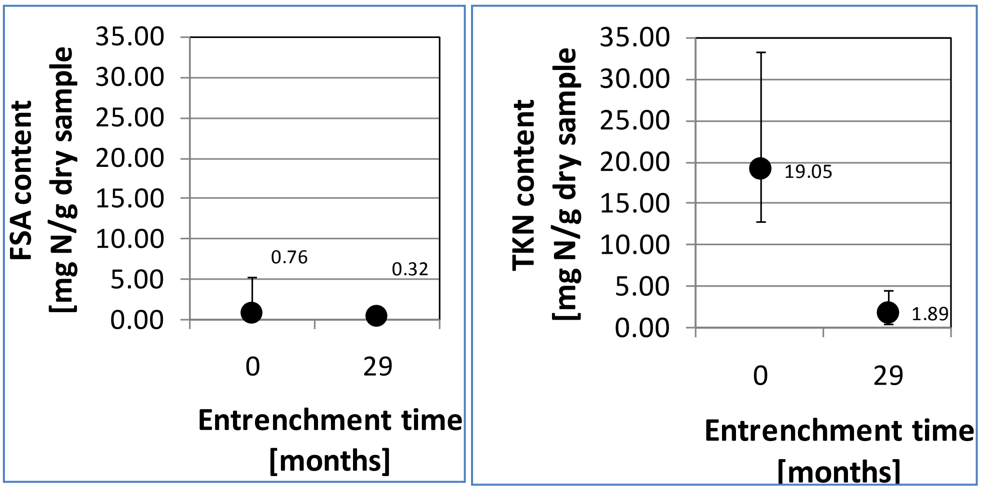


Figure 5.51 Total Kjeldahl Nitrogen (TKN) and free and saline ammonia (FSA) concentration in pit latrine sludge before and after entrenchment for 29 months.

The amount of TKN released by the sludge is calculated from the difference in TKN on a dry basis between the two measurements, i.e. 17.2 mg N/g dry sample are lost over a period of 29 months. This corresponds to approximately 4.3 kg N/tonne wet sludge. Since only one measurement interval is available, the rate of release cannot be accurately calculated, but must be equal to or greater than 4.3 kg N/tonne wet sludge per 29 months or approximately 1.8 kg N/tonne wet sludge/year. Given that most degradation seems to occur in the first 12 months, the rate may be as high as 4 kg N/tonne wet sludge/year.

5.1.7.4 Fate of pathogens

In order to obtain representative data for pit latrine sludge at the time of entrenchment, samples were obtained from 3 heights (near the top, near the middle, near the bottom) from 10 pit latrines in eThekwini Municipality (Hawksworth et al., 2005). It was assumed that the results from the 10 pit latrines could be regarded as an indication of the load of ova in the exhumed sludge that was delivered to the entrenchment site and therefore provide a representative starting value for ova load during entrenchment. It should be noted, however, that these pit latrines may have belonged to a community that did not exhibit the same average ova load as the global average.

It was hypothesised that since different depths in the pit sludge have different ages since excretion (i.e. the age of the sludge increases as you go deeper into the pit), and that helminth ova deactivate over a period of many years, that there would be a decrease in the number and viability of *Ascaris* ova with increasing pit depth.

Each sample was divided into five 1 g replicates and the AMBIC protocol (Hawksworth et al., 2005) was used to enumerate ova. *Ascaris* ova were found in all samples. The lowest measured *Ascaris* ovum count was 142 per gram of sample (wet weight) (142 ova/g w.w.) and the largest was 3937 ova/g w.w. with most of the samples analysed showing a total ovum count of between 200 and 1000 ova/g w.w. Of these ova, the average viability per sample varied between 20 and 40%. The high variance of the measurements (a common characteristic of this kind of measurement) resulted in large confidence intervals. Although the mean values of ova count decrease with increasing depth in

6 of the 10 pits, the change is not statistically significant indicating that there is no basis to conclude that there is a relationship between number of ova or ovum viability with pit depth. These results imply that significant deactivation of *Ascaris* ova does not occur during residence in the pit. Nevertheless, it is possible that cross-contamination of samples occurred during the pit emptying process.

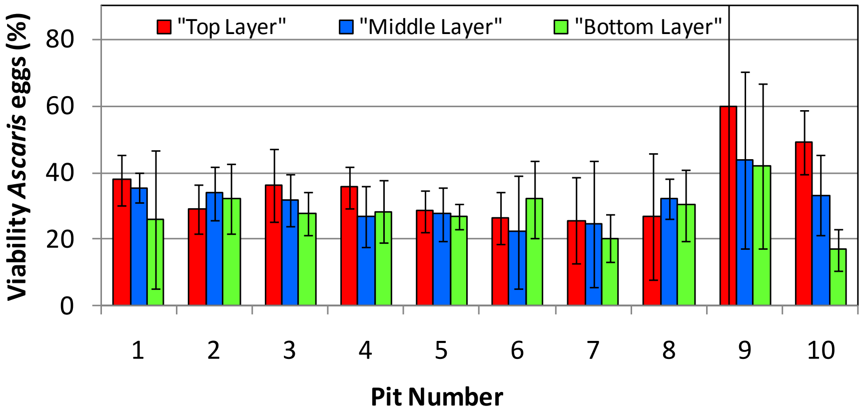


Figure 5.52 *Ascaris* ovum viability for top (a), middle (b) and bottom (c) samples from 10 VIP latrines. Error bars indicate 95% confidence on the mean.

In Figure 5.53, the average eggs per gram and the average viability across all samples within a pit are presented for each pit. Ovum viability was around 30%, although significantly higher values were observed for pit 9. Ova counts were generally below 1000 ova/g w.w., but all samples from all pits had significant numbers of ova.

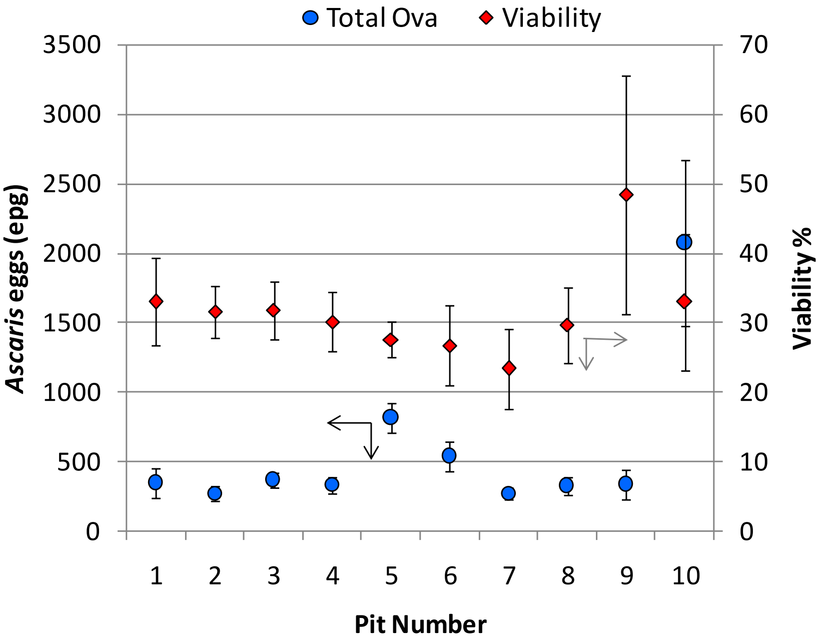
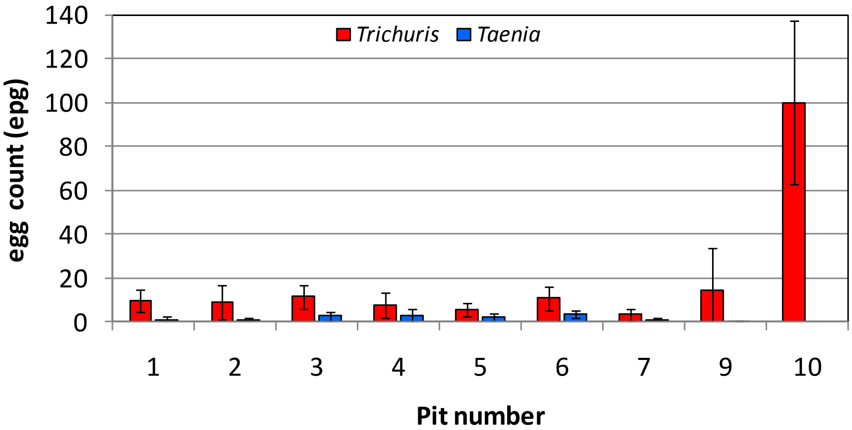


Figure 5.53 Ascaris total ova counts (all categories of ova) and viability (potentially viable/total ova) for samples collected from 10 pit latrines. Error bars indicate 95% confidence on the mean.

In addition to *Ascaris* ova, *Taenia* and *Trichuris* ova were found in almost every sample analysed. Once again, there did not appear to be a difference in prevalence with pit depth for these species. Ova of *Toxocara* worms (canine or feline roundworm) and *Enterobius vermicularis* (pinworm) were found in samples taken from pit latrine 9.

Figure 5.54 Taenia and Trichuris prevalence for VIP latrines 1-7 and 9-10. Error bars indicate 95% confidence on the mean.



These data were compared with the samples taken from the entrenched sludge. Calculations for correlation between calculated properties were performed in two ways: firstly all pit sample and trench sample data was considered as a single data set with pit samples assigned a time value of 0 (i.e. condition of sludge at the day of burial); Secondly, the pit data was excluded from the set and only the trench sample data subjected to correlation analysis. The correlation analysis was performed in both ways to understand the contribution of the pit data to the analysis. If both methods of analysis gave similar results, it would indicate that the pit samples were a reasonable substitute for day 0 samples, i.e. sludge buried and dug up on the same day). A difference in results would suggest that there was some fundamental difference between the two data sets; for example that a degree of mixing with sand in the trench interfered with egg recovery in the modified AMBIC procedure. However, both sets of analyses led to the same conclusions.

In Figure 5.55, the total number of eggs counted and potentially viable eggs counted in each sample are plotted against the period spent by the sample in the trench. Egg counts as high as 4000 eggs/g were observed, with an average value of 428 over all the entrenched samples, and 564 eggs/g over all the samples analysed before entrenchment (time 0).

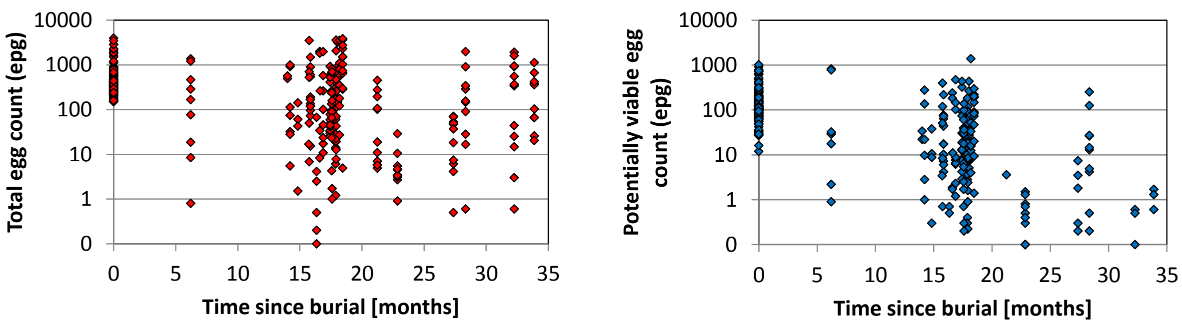
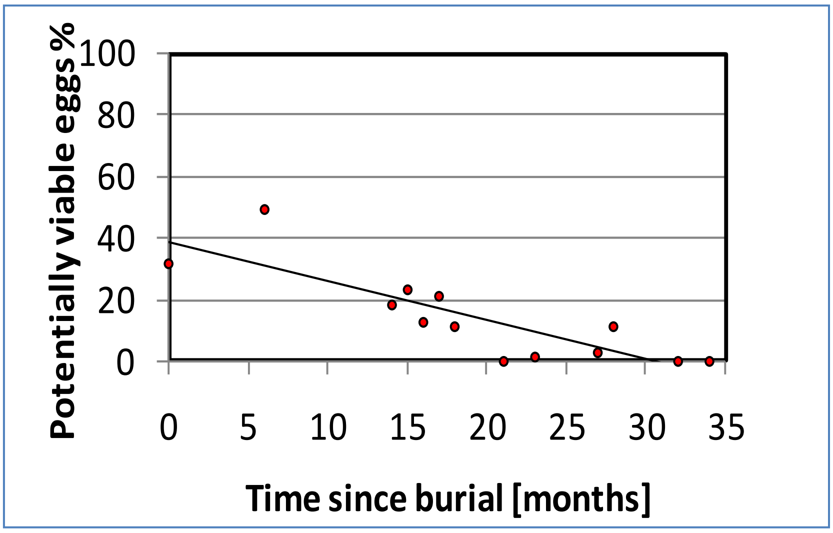


Figure 5.55 Total and potentially infective *Ascaris* ova counts on samples taken from pit latrine sludge before burial (Time 0) and samples entrenched at the Umlazi site.

Given the extremely heterogeneous nature of the material being studied, and the variability in the starting load of helminth ova in the sludge, it is difficult to calculate a meaningful deactivation rate. A significant number of helminth ova may still be found in the sludge nearly 3 years after being buried in trenches. However, the fraction of eggs that were definitely infective, or potentially infective, decreased significantly over a period of 2.8 years: In 10 samples taken after 1 030 days or 2.8 years spent in the trench, a total of 37 904 ova were counted, but only 36 of these (<0.1%) were potentially infective (non-motile larvae, or undeveloped) and none contained motile larvae.

These data were analysed on a month-by-month basis: the age of samples (in terms of how long they were in the trench) was ranked and data samples of a similar age were considered together. All the ova and all potentially viable ova from all samples of a similar age were counted and the overall fraction f_t was evaluated for that age as follows:

$$f_t = \frac{\text{Total number of potentially viable eggs in all samples of age t}}{\text{Total number of not infertile eggs in all samples of age t}}$$



This analysis reduces the noise of the data and allows a better visual picture of the results to emerge. These data clearly indicate a reduction in potentially infective ova with time, demonstrating that significant deactivation of helminths occurred in the trenches with time and that a period of 30 to 36 months (2 ½ to 3 years) was sufficient to reduce potentially viable ova to very low concentrations.

Figure 5.56 Potentially infective *Ascaris* ova (fraction of *not infertile ova*) on a month-by-month basis

In Figure 5.57 the relative fraction of different larvae conditions are plotted against time. The purpose of this comparison is to see whether there is a net change from motile to non-motile to necrotic larvae, indicating a deactivation of ova with time.

These plots indicate that except in just-exhumed pit sludge, there are very few ova with motile larvae (definitely infective). The data indicate that all samples have non-motile larvae visible in *Ascaris* ova, but that the relative fraction of these decreases significantly with time. In contrast, the relative fraction of necrotic larvae increases with time. This suggests that as time progresses, the potentially infective ova in a sample of buried pit latrine sludge is deactivated through death and decay of the larvae. It is not possible to say at which point a larva dies, but clearly this occurs during the stage when the larva is non-motile, and has occurred sometime before the larva became necrotic.

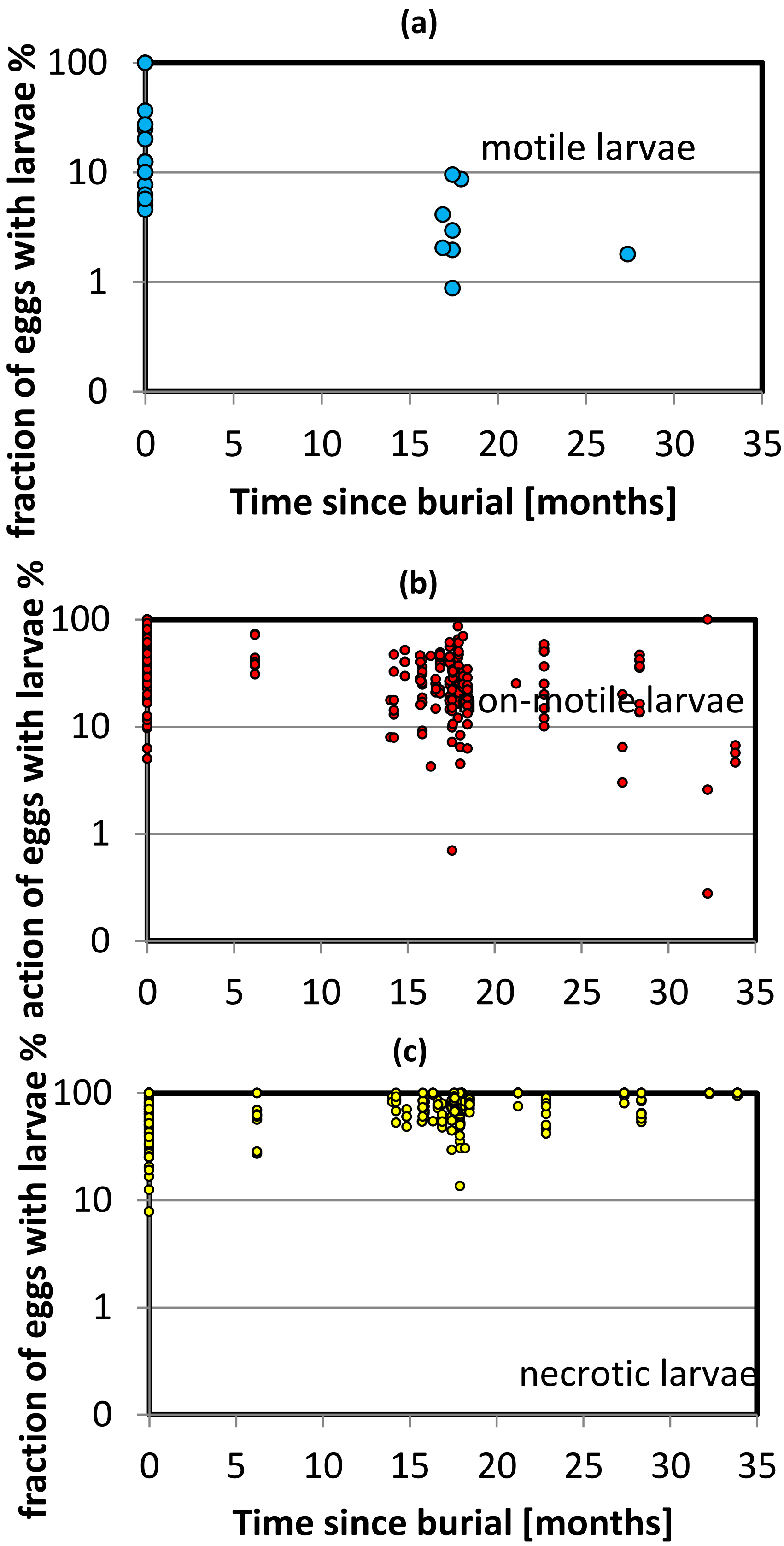


Figure 5.57 Change in condition of visible larvae with time for (a) motile, (b) immotile, (c) necrotic larvae

If the best fit straight line for Figure 5.57 (b or c) are extrapolated to the horizontal axis, they predict an average time for converting non-motile larvae to necrotic larvae (i.e. definitely dead) of 3.6 years (using all data) or 3.2 years (using trench data only). This is more conservative than the previous estimate of 2.7 years and furthermore only represents an average value (i.e. while most larvae visible should be necrotic, natural variance between samples indicates that a portion may not yet be necrotic).

Incubation experiments

Fifteen samples were tested from trenches where sludge had been buried for different ages between 1 year 9 months and 2 years 9 months according to Table 4.2. These samples were divided into two portions, and the first portion was subjected to the AMBIC protocol within a week of being sampled. The second portion was incubated at 30ºC in a humid environment for 14 days before being subjected to the AMBIC protocol.

Table 5.10: Age of samples subjected to incubation

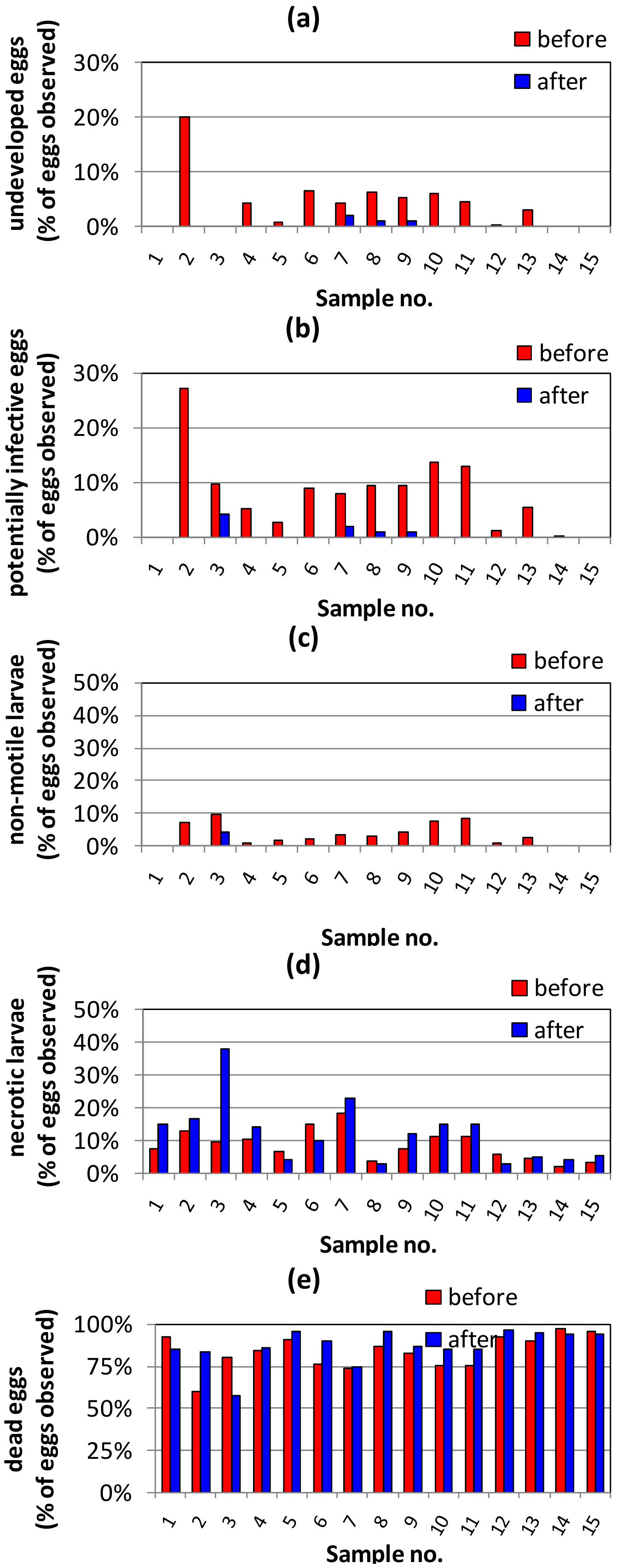
Samples	Age
1	1y 9m
2-3	1y 10m
4-11	2y 3m
12-13	2y 4m
14-15	2y 9m

Figure 5.58 shows that the fraction of undeveloped ova decreased as a result of incubation. However, the decrease was not matched by an increase in potentially infective ova (Figure 5.58 b). In fact this category shows a substantial decrease where only 4 of the 15 samples had any potentially infective ova, and most of the ova in this category were undeveloped. If 14 days of incubation did not result in development of these ova, it is unlikely that they would ever develop and therefore can be considered non- infective.

Looking at the observable larvae, Figure 5.58 (c) indicates that except for sample 3, after incubation there were no non-motile larvae. This means that virtually all non-motile larvae that existed before incubation must have reported as necrotic larvae or dead eggs after 14 days of incubation. This is confirmed by Figure 5.58 (d) which shows that 11 of the 15 samples show an increase in the necrotic larvae fraction.

The fraction of ova that were dead increased significantly after incubation from an average value of 80% dead to an average of 84% dead (Figure 5.58 e).

These results suggest that for samples buried for more than 2 years, while ova counts may indicate potentially infective eggs, it is probable that these eggs are not in fact infective. However, this conclusion hinges on the assumption that incubation for 14 days at 30ºC in a humid environment will cause undeveloped ova to develop viable larvae and that the incubation conditions themselves do not in fact result in the deactivation of ova/larvae.



5.2 Tree tower trials

Figure 5.50 Ova categories as a % of total ova observed before and after incubation for 5 categories of data (a) undeveloped ova, (b) potentially infective ova, (c) non-motile larvae and (d) necrotic larvae, (e) dead

In order to investigate the impact of pit latrine sludge on tree growth under more controlled conditions, a study was conducted in which *acacia mearnsii* (wattle) and *eucalyptus grandis* were grown in pots containing a core of faecal sludge surrounded by sand. The study explored the ability of sludge to aid plant growth in comparison with fertiliser for both species and assessed the impact of sludge on photosynthetic parameters and spatial root distribution.

5.2.1 Preparation of the site

Twenty-four planting containers were constructed by stacking four concrete manhole rings on a concrete base, creating a 1m high structure with an internal height and diameter of approximately 250 mm and 750 mm. A layer of PVC sheeting was partially embedded in the base of each column to prevent seepage into the base, with a channel exiting the column to allow drainage. The joints between the manhole rings were sealed.

Figure 5.59 Experimental site at University of Kwa-Zulu Natal (Howard campus)

5.2.2 Sludge application

A 250 mm base layer of river sand was placed in each of the 24 columns. Faecal sludge was removed from full pit latrines, transported in 100 litre bins and placed in the columns the same day. For the 12 experimental columns, a 450 mm diameter polycarbonate cylinder was placed in the centre of each base layer of sand and filled with sludge to an approximate height of 500 mm above the base layer. This created a core of sludge faecal sludge 500 mm in height and 450 mm in diameter, with a volume of 0.22 m^3. The area around the cylinder was filled with sand, creating an annulus approximately 150 mm in width. The core of sludge was covered with a 250 mm layer of river sand.

Figure 5.60 A polycarbonate cylinder (left) was used to fill the tower with a core of sludge ringed by river sand (right)

The remaining 12 columns were not treated with any faecal sludge but were filled with river sand to the same height as columns in the treatment group (approximately 1 m). Throughout the experiment these columns were treated with fertiliser so as to serve as a positive control in the experiment.

5.2.3 Tree planting and maintenance

Trees were planted in the columns the day after the columns were prepared. Eucalyptus (*eucalyptus grandis*) and wattle (*acacia mearnsi*) were selected because they are fast-growing and are commercially grown as a source of timber. *Acacia mearnsii* has been found to show a significant growth response to applications of phosphorus and potassium, but less response to nitrogen applications because of its ability to fix atmospheric nitrogen (du Toit, 2002). Eucalyptus responds well to fertiliser, growing as high as 20 m in 3 years under favourable conditions (Campinhos, 1980). In the 12 experimental columns containing a core of faecal sludge, 6 seedlings of each species were planted. In the 12 control columns, 6 seedlings of each species were planted. These were bedded in 300 mℓ of compost to improve water retention and provide nutrients for early seedling establishment, which may have proven problematic in sand.

Plants in the experimental group were irrigated immediately after planting with 1 ℓ of water and were watered as needed thereafter. Plants in the control group received 1ℓ of a 1 g/ℓ aqueous fertilizer after planting and biweekly thereafter, alternating with a 2.5 mℓ/ℓ aqueous trace element solution applied alternate weeks. Plants in the control group were irrigated with the same quantities of water and at the same times as plants in the experimental group throughout the experiment. No herbicides or other chemical treatments were used on the columns during the experiment. Aphids were removed by spraying with water and the columns were weeded weekly.

At approximately two months, all wattle seedlings were staked with bamboo to prevent lodging. Trees were harvested at approximately 6 months (26 weeks) after planting.

5.2.4 Analysis and monitoring

Analyses were completed of sand, sludge and plant tissue samples. Plants were monitored for growth, chemical and physical change and gas exchange. Plant height was measured after planting and thereafter every second week. Stem diameter was measured every month. Photosynthetic measurements were taken on expanding young leaves of both species when plants were approximately six months old. Measurements of assimilation, intercellular CO_2, stomatal conductance and transpiration were taken. The response of assimilation to light (light response curves) was measured in the morning from 8:00 to 13:00 on alternating control and treatment plants. CO_2 response curves (A-ci curves) were taken from 8:00 to 13:00 under saturating light conditions. Transpirational water use efficiency (WUE) was calculated using data for transpiration and assimilation. Soil and leachate quality were monitored.

At harvest, tree trunks were cut as close to the base of the stems as possible. Twigs were cut from the trees and leaves were separated from the twigs. Dry biomass was determined using a 200 g sample of each component (leaf, twig or trunk) which was then air dried to constant weight after which a 100 g sample of each was oven dried. Total leaf area was estimated by determining the ratio of leaf area per unit leaf mass of a 50 g sample for each replicate. Root distribution was analysed after the towers were dismantled.

5.2.5 Results

5.2.5.1 Growth

Table 5.11 shows the measured growth data for the 24 trees over the six month trial period.

Height

For eucalyptus, difference in height between the experimental and control groups became apparent by the ninth week. By the end of the study at 26 weeks, the mean height of the experimental group was approximately 2.3 times greater than that of the control group (244 cm and 108 cm respectively). While tree height levelled off soon after the midpoint of the study, mean root collar diameter increased throughout the period of the study. At 26 weeks, mean root collar diameter measured at 40 mm for the experimental group and 34 mm for the control group -- a relatively small difference in comparison with the considerable differences observed in tree height, indicating that biomass was preferentially partitioned into stem diameter rather than stem length in the control group.

For wattle, however, differences in mean height between the experimental group (270 cm) and control group (232 cm) were small and differences between stem diameter at harvest were insignificant (45 mm and 39 mm).

Table 5.11 Growth data for experimental and control groups of both species

Parameter	Eucalyptus		Wattle	
	Experimental	Control	Experimental	Control
Height	244 cm	108 cm	270 cm	232 cm
Root collar diameter	40 mm	34 mm	45 mm	39 mm
Mean leaf area	11.97 m2.	1.83 m2.	3.73 m2.	1.95 m2.
Specific leaf area	18.9 m2.kg-1	10.6 m2.kg-1	5.9 m2.kg-1	4.7 m2.kg-1
Total aboveground biomass	1517 g	362g	1558g	1038 g
Leaf biomass	42.1%	48.4%	40.7%	40.1%
Trunk biomass	22.1%	23.6%	29.4%	30.2%
Twig biomass	35.8%	28.0%	29.8%	29.7%

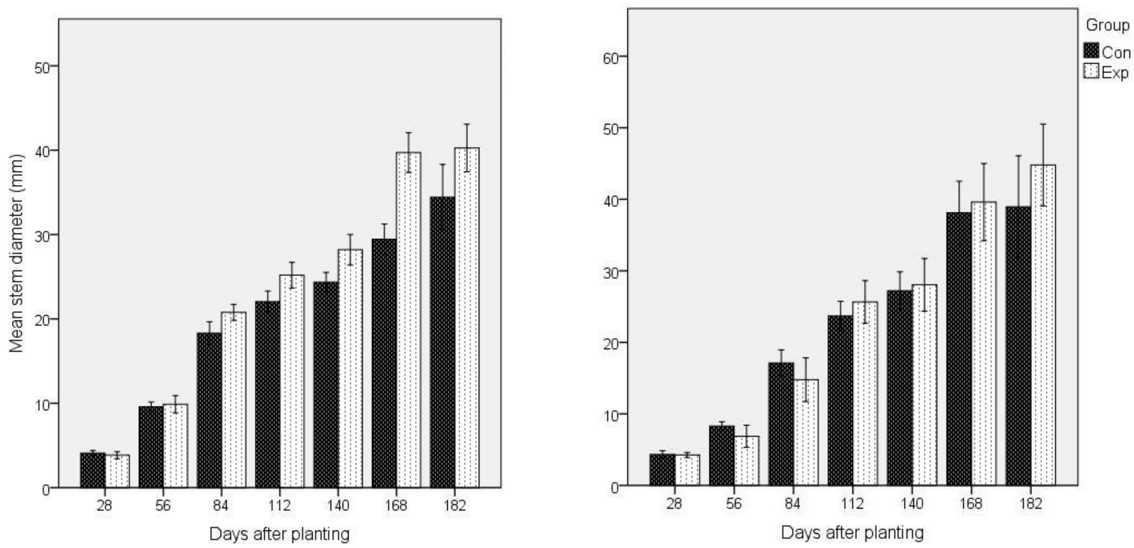


Figure 5.61 Mean height of eucalyptus (left) and wattle (right) over 26 weeks

Mean leaf area

For eucalyptus, mean leaf area was 6.5 times greater than in the control group (11.97 m^2 and 1.83 m^2), indicating that foliage production was severely restricted in the control group where leaves were visibly fewer in number and smaller than those of the experimental trees. Leaf area did not appear to increase visibly during the final two months of the study, suggesting growth in terms of both height and foliage production had reached a ceiling. Leaf loss, indicating reduced leaf lifespan, was evident in the control group where branches were bare of leaves closer to the trunk, probably due to nutrient stress.

For wattle, mean leaf area for the experimental group was twice that of the control group (3.73 m^2 and 1.95 m^2), with vigorous lateral growth at the base of the trees, while leaf production for trees in the control group appeared to halt approximately one month before harvest, rather than experiencing leaf loss.

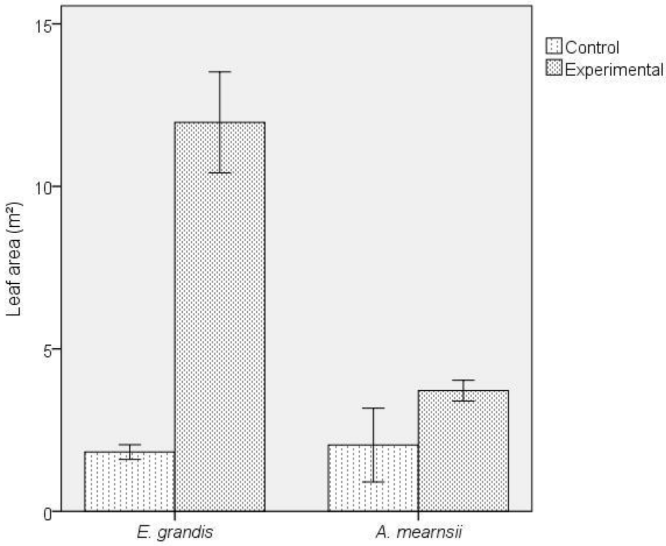


Figure 5.62 Mean leaf area of eucalyptus (left) and wattle (right) after harvest

Specific leaf area

Specific leaf area indicates a compromise between maximum light capture for photosynthesis and the functional constraints which require carbon. Eucalyptus typically has thick leaves with low specific leaf area in native habitats but in an environment of abundant nutrient and water supply specific leaf area can increase dramatically (Kirschbaum and Tompkins, 1990; Kirschbaum et al., 1990). Specific leaf area was significantly greater in experimental trees for eucalyptus (18.9 and 10.6 m^2.kg-1) but the difference was insignificant for wattle (5.9 and 4.7 m^2.kg-1).

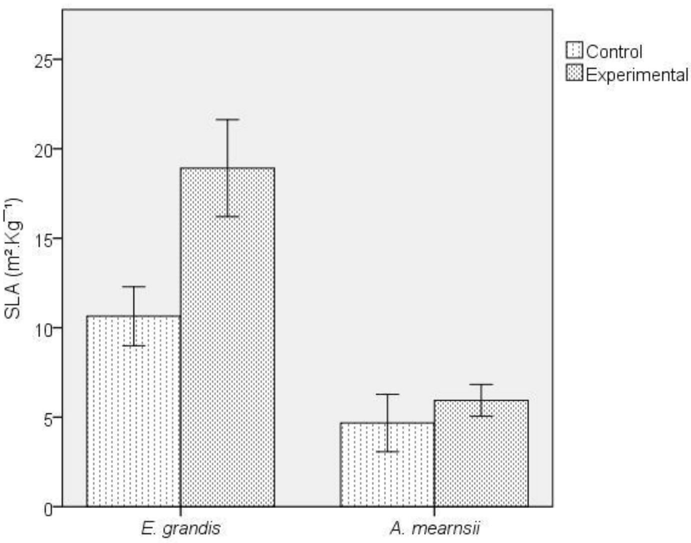


Figure 5.63 Specific leaf area for eucalyptus (left) and wattle (right) after harvest

Dry biomass partitioning

For eucalyptus, total aboveground biomass was almost four times greater in the experimental group for each of the measured components (trunk, twigs and leaves) than in the control group (1517 g and 362 g). Leaves represented the greatest resource sink for both experimental and control groups (42.1% and 48.4%), with trunks accounting for approximately one fifth of total dry biomass (22.1% and 23.6%) and twigs roughly a third of total dry biomass (28.0% and 35.8%).

Differences between experimental and control groups were less marked for wattle (1558g and 1038g for total aboveground biomass), with nearly identical partitioning for leaves (40.7% and 40.1%), trunk (29.4% and 30.2%) and twigs (29.8% and 29.7%).

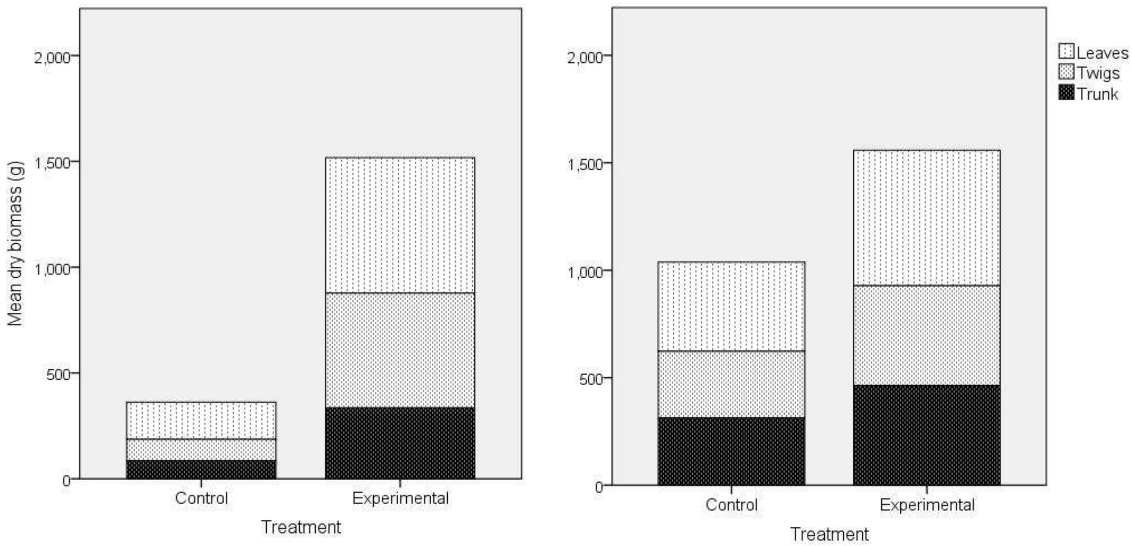


Figure 5.64 Mean dry biomass partitioning of leaves, twigs and trunk for eucalyptus (left) and wattle (right).

It is clear that the growth of both species was enhanced by the application of faecal sludge, with greater benefit to eucalyptus which does not have the nitrogen-fixing properties that support the ability of wattle to thrive in poor soil. While the control trees had received fertilizer, their growth was poorer than expected, which may be due to the porosity of the sand resulting in excessive leaching of nutrients away from tree roots. The enhanced growth in the presence of sludge observed in this study was consistent with growth observed during a 12 month study in which eucalyptus treated with sludge showed 86% greater biomass over control trees (da Silva et al, 2011).

5.2.5.2 Gas exchange measurements

The experimental trees of both species showed an enhanced capacity for photosynthesis resulting from increased foliar macronutrient concentrations due to the higher nutrient availability in sludge. Constructed A-ci curves, showing the net CO_2 assimilation rate (*A*) versus the calculated substomatal CO_2 concentration (C_i), indicate that RuBP, an organic substance required for photosynthesis, was more limited in control trees of both species. RuBisCO, an enzyme which catalyzes the reaction between RUBP and carbon dioxide in order to make carbon from the atmosphere available to the plant, was limited for wattle, resulting in decreased carboxylation efficiency.

Maximum CO_2 assimilation (Amax) and maximum electron transport (Jmax) rates were positively affected by faecal sludge application in both species. For eucalyptus there was a positive correlation between the concentrations of foliar nitrogen and phosphorus and increased light-saturated photosynthesis (Amax), maximum carboxylation (Vcmax) and electron transport rate (Jmax) which suggests that these macronutrients had a major impact on photosynthetic capacity. In wattle, however, there was not a strong correlation, suggesting that nitrogen and phosphorus were not acutely limiting, although the correlation between foliar nitrogen and Jmax indicates that foliar nitrogen concentration had an effect at the electron transport level, with phosphorus playing a less critical role. While the maximum assimilation rate under saturating light (Amax) was enhanced in the experimental group of both species, there was a more significant difference in eucalyptus (17.3 and 12.7 µmol.m-2.s-1) than wattle (16.4 and 11.7 µmol.m-2.s-1). There were no significant differences in photochemical efficiency (apparent quantum efficiency) in either species.

For eucalyptus, water use efficiency (WUE) under saturating light was almost doubled in experimental trees, possibly due to the combined effect of increased A and decreased transpiration (E) as well greater nutrient availability relative to control trees. For wattle, however, WUE was slightly higher for control trees.

At low photosynthetic active radiation (PAR) (>500 µmol.m-2.s-1), photosynthetic rates were similar between control and experimental groups of both species indicating a fairly equal effect of light limitation irrespective of treatment. However, at higher PAR the curves of both species diverged and at light saturation A was considerably suppressed in control groups of both species relative to that of experimental groups. Assimilation of control groups levelled at lower PAR compared with experimental groups, indicating that these trees were unable to utilise increased PAR due to underlying biochemical limitations to photosynthesis.

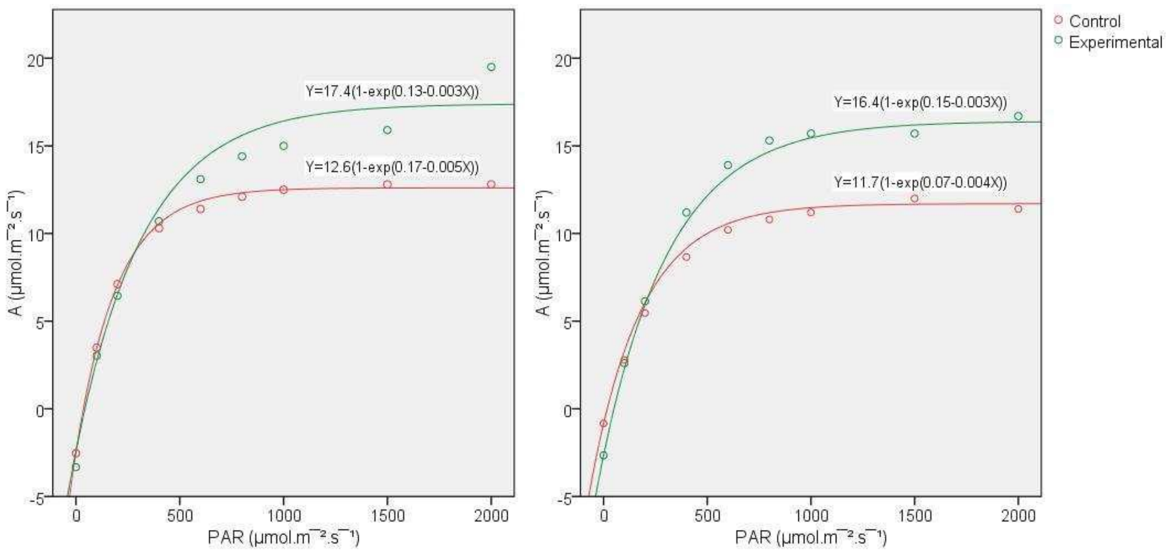


Figure 5.65 Representative light response curves of eucalyptus (left) and wattle (right)

Light compensation points were slightly, but not significantly, decreased in the control groups of both species, with a high degree of variation around the mean. The mean light compensation point for eucalyptus was 39.4 µmol.m-2.s-1 for trees in the experimental group and 33.0 µmol.m-2.s-1 for trees in the control group. In wattle, the mean light compensation point was 46.7 mol.m-2.s-1 for experimental trees and 36.6 µmol.m-2.s-1 for the controls.

Table 5.12 Gas exchange data for experimental and control groups of both species

Parameter	Eucalyptus		Wattle	
	Experimental	**Control**	**Experimental**	**Control**
Maximum assimilation rate (Amax)	17.3 µmol.m-2.s-1	12.7 µmol.m-2.s-1	16.4 µmol.m-2.s-1	11.7 µmol.m-2.s-1
Light compensation points	39.4 µmol.m-2.s-1	33.0 µmol.m-2.s-1	46.7 µmol.m-2.s-1	36.6 µmol.m-2.s-1
Dark respiration rates	1.9 µmol.m-2.s-1	2.5 µmol.m-2.s-1	1.7 µmol.m-2.s-1	1.2 µmol.m-2.s-1
Maximum electron transport (Jmax)	41.2 µmol.m-2.s-1	18.2 µmol.m-2.s-1	43.1 µmol.m-2.s-1	29.2 µmol.m-2.s-1
Carboxylation efficiency	0.15 mol.m-2.s-1	0.11 mol.m-2.s-1	0.16 mol.m-2.s-1	0.12 mol.m-2.s-1
CO_2 compensation point	127.7 µmol	103.5 µmol	108.0 µmol	106.0 µmol
Photorespiratory loss	15.7 µmol.m-2.s-1	9.1 µmol.m-2.s-1	14.8 µmol.m-2.s-1	10.7 µmol.m-2.s-1
A under saturating light	17.2 µmol.m-2.s-1	12.6 µmol.m-2.s-1	16.2 µmol.m-2.s-1	11.6 µmol.m-2.s-1
Leaf transpiration under saturating light	4.1 mmol.m-2.s-1	3.2 mmol.m-2.s-1	2.7 mmol.m-2.s-1	4.2 mmol.m-2.s-1
Water usage effectiveness (WUE)	6.2 µmol.mmol-1	3.2 µmol.mmol-1	4.0 µmol.mmol-1	5.4 µmol.mmol-1

5.2.5.3 Foliar nutrient concentrations

For eucalyptus, all primary and secondary macronutrients occurred in greater concentrations in the foliage of the experimental trees with the exception of calcium, indicating improved nutrition. Nitrogen (N) concentration in the foliage of experimental trees (329 mg.kg-1) was nearly three times greater than in control trees and phosphorus (P) (31 mg.kg-1)was approximately double. Differences in potassium (K) were less significant (105 and 72 mg.kg-1).

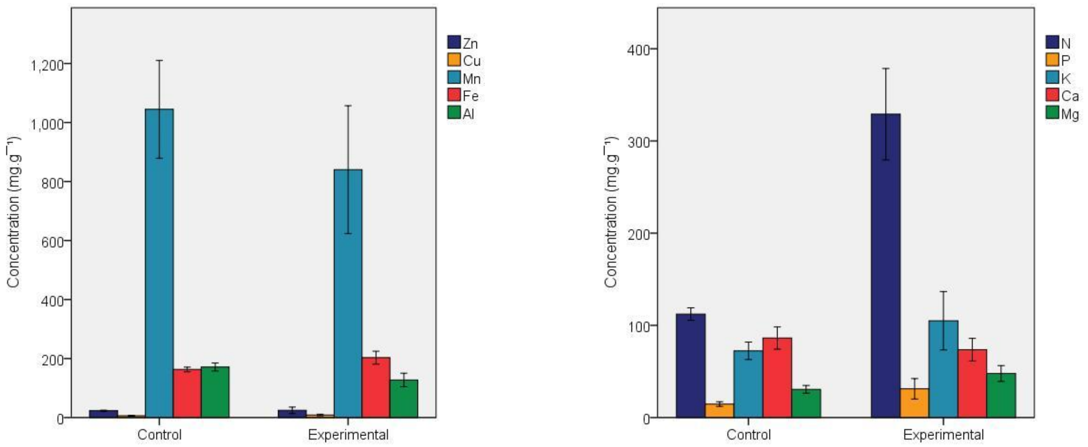


Figure 5.66 Foliar micronutrients (left) and macronutrients (right) in eucalyptus

Deficiencies of these nutrients, in particular nitrogen, likely had the greatest limiting effect on tree growth in the control group, which exhibited lighter green foliage and purple colouration on growing tips in the latter part of the growing phase and had a lower N:P ratio (7.9) than the experimental group (11.1) For micronutrients, iron (Fe) was significantly greater in experimental trees (203 mg.kg-1 and 163 mg.kg-1) and aluminium (Al) was greater in control trees.

Figure 5.67 Growing tips of *E. grandis* from trees of experimental (left) and control (right) groups at approximately 5.5 months after planting

Wattle showed similar trends, but with statistically greater concentrations in control trees than with eucalyptus. Calcium again appeared in greater concentration in control trees. Nitrogen was higher in foliage of experimental trees (319 mg.kg-1 and 233 mg.kg-1; p=0.006). The concentration of foliar phosphorus (P) in the leaves of experimental trees (45 mg.kg-1) was approximately 4.5 times greater than in control trees and potassium (K) in the experimental group (126 mg.kg-1) was approximately twice that of control trees (p<0.0005). Nitrogen (N) 319 mg.kg-1 was similar to levels in eucalyptus. A higher N:P ratio for the foliage of control trees indicates that phosphorus was more deficient than nitrogen and no visible symptoms of K deficiency were evident in control trees. Magnesium (Mg) was also higher for experimental trees (27 mg.kg-1 compared with 18 mg.kg-1). For micronutrients, Zn was statistically greater in experimental trees than control trees (18 and 26 mg.kg-1), while levels in control trees were higher for Cu, (9 and 4 mg.kg-1), iron (Fe) (258 and 179 mg.kg-1) and aluminium (Al) (169 and120 mg.kg-1).

Table 5.13 Adequate and actual macronutrient ratios for wattle

Macronutrient	N	P	K	Ca	Mg
Adequate	100	4.4	11.7	9.1	2.8
Experimental	100	4.4	26.8	17	7.8
Control	100	14.1	39.4	11.4	8.5

Table 5.14 Adequate and actual macronutrient ratios for eucalyptus

Macronutrient	N	P	K	Ca	Mg

Optimal	100	8	35	2.5	4
Experimental	100	9.3	31	22.7	14.6
Control	100	13.1	64.6	77.2	27.3

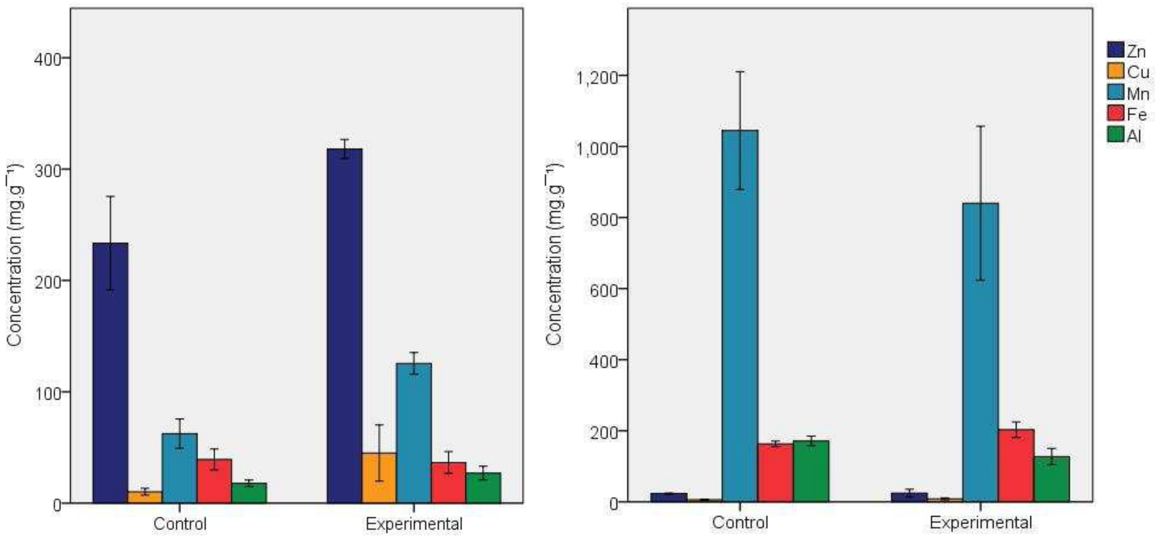


Figure 5.69 Foliar macronutrients (left) and micronutrients (right) for wattle

In contrast to the eucalyptus, wattle showed little difference between control and experimental groups in terms of foliar nitrogen. High nitrate levels present in the sludge would inhibit the fixing of N_2 into usable ammonium in the root nodules of the tree. Because the control group used sand which was low in nitrogen, N_2-fixing activity was probably high which offset the limiting effect of the poor soil. Comparing the control groups of the two species, wattle showed statistically greater leaf, twig, trunk and total dry biomass than eucalyptus, while there were not significant differences in total dry biomass between the experimental groups of both species.

Spatial root distribution

At the end of the study, the tree towers were dismantled and root development in the sand and at the sand/sludge interface was examined. The sludge cores for experimental towers were too dense to permit the extraction of roots from the core itself for analysis of patterns and biomass, however it was evident that root distribution was not impeded by the presence of sludge.

Figure 5.7068 Root development of eucalyptus around sludge core

Fine roots were matted densely just beneath the surface for both species and also on the periphery of the sludge core in response to the higher nutrient levels and possibly also high moisture levels in the sludge. Fine roots decreased sharply with depth, which is usual in forest species. Medium and coarse roots of both species were fairly evenly distributed throughout the soil, reaching to the bottom of the towers. Medium and coarse root intersections were counted on three horizontal planes for experimental trees: above the sludge core (125 mm), through the sludge core (500 mm), and below the sludge core (625 mm). Root intersects were generally greater in the outer ring of sand for both experimental and control groups, growing laterally out to the column walls and then turning downward, with only a few coarse and medium roots below the stump, penetrating the sludge core in the case of the experimental group. In an earlier study using eucalyptus, however, Bouillet et al (2002) documented medium root density to be greatest below the stump, suggesting that the lateral rooting in this study may have resulted from early containerised growth. This may have also prevented the development of a normal tap root system, which in both species would have promoted stronger penetration of roots into the sludge core.

A greater proportion of root intersections in both species were found to intersect the faecal sludge core at a depth of 500 mm compared to the control trees, particularly in the wattle, where 31.3% of roots at that depth had intersected the faecal sludge core while only 7.9 % of roots in the control were found in the equivalent area. This indicates that although the sludge core was likely too wet initially to have promoted root penetration, over time faecal sludge was able to support root growth and did not inhibit normal root distribution. In eucalyptus, a mean of 88 root intersects occurred in the experimental eucalyptus trees while 155 root intersects occurred in the control trees, suggesting that control trees had produced a greater root biomass in order to forage over a greater volume of soil in the nutrient poor soil. In wattle, however, the opposite occurred: a mean of 46 root intersects occurred in the control trees while 164 root intersects - representing a 357% increase -- occurred in the experimental trees. The control group did, however, contain more fine roots, which assist in acquiring nutrients and water.

5.3 On site burial of pit latrine sludge

Preceding these more detailed investigations, a small trial was conducted in August 2008 to investigate the possibility of using on-site burial of sludge to provide nutrients to fruit trees on the same plot where the sludge was removed. Two homeowners willing to participate in the study were identified in the Inadi area northwest of Pietermaritzburg. Each had a pit which had filled and was no longer in use.

The two pits were emptied manually. As the pits had been constructed without a lining of concrete blocks or some other material, the sides sloped inward towards the bottom of the pit and the length and width of the pit at the surface was not a reliable indicator of the actual volume of the pit. The sloping sides of the pit at the first household reduced the length and width from 1.05 m x 0.75 m at the surface to 0.7 m x 0.7 m at the bottom of the pit. The expected volume of 2.008 m^3 based on a 2.55 m depth was instead closer to 1.629 m^3 (or approximately 20% less).

Both pits were reported to have been full when they were closed and homeowners reported that only a 300-400 mm covering of earth had been required to fill in the pit to ground level. As the volume of sludge had decreased as a result of degradation over time, the first household had added more soil to fill the depression, while the second household had added garden waste. When the pits were emptied, faecal sludge was only encountered at a depth of 1.65 m (first household) and 1.15 m (second household). The volume of sludge recovered from the pits that were closed when full was considerably less than expected; the height of the sludge contents had decreased from approximately 2.25 m to 0.9 m (60% drop) for one pit and from 1.9 m to 1.05 m (45% drop) for the second.

The depth at which the sludge was first encountered was recorded. The sludge comprised two distinct components. One was a dryer, peaty, black, odourless fraction that appeared to be well stabilised, the second was a wetter, more "fresh" looking fraction with a strong unpleasant smell. The peaty fraction was found in contact with soil whereas the malodorous fraction was found in clumps of non biodegradable material, particularly plastic bags. Initially, the pit sludge seemed to contain a considerable quantity of non-biodegradable matter (plastic bags, cloth, bottles, etc.). However, the volume of this material amounted to only 0.08 m^3 (8%) for the pit at the first household and 0.18 m^3 (14%) for the pit at the second household. Non-biodegradable material was removed manually and placed in a separate pile.

Holes for the fruit trees were dug approximately 900 mm square and deep. Trees serving as a control were planted in only the soil that had come from the hole, others were planted with a handful (approx. 40 g) of superphosphate fertilizer scattered around the base of each tree and worked in superficially with a fork, and others were planted with pit sludge added in 90 ℓ batches to layers of soil returned to the holes. The sludge was mixed into the soil layers with a garden fork. It was ensured that no sludge was incorporated into the top 300 mm of soil around each tree. The owners were asked to subsequently treat all trees identically with regard to watering and weeding.

Figure 5.691 Manual excavation of a pit. Start of the sludge layer clearly visible 1.65m below the surface.

Accurate measurement of sludge volumes removed from the pit was difficult as soil removed from the sides of the pit during excavation was added to the sludge; sludge removed from the pit at the second household appeared to include a substantial amount of soil. The possibility exists that soil was used to cover faeces when the pit was in use.

5.3.1 Sludge characterisation

Sludge samples taken from four depths in the pit were analysed for moisture, volatile solids (TS - Ash) and COD. Moisture content was lower than is typical of pits in use, suggesting that some dehydration of pit contents had occurred during the three or four years during which the pit contents had been buried. There were no significant differences in volatile solids between any of the samples tested, irrespective of the height or appearance of the samples. A comparison of mean values of volatile solids as a fraction of the total solids, eliminating the diluting effect of moisture, revealed no significant differences in volatile solids between any of the samples tested, irrespective of the height or appearance of the samples.

5.3.2 Tree planting and sludge application

Holes were prepared for the fruit trees approximately 900 mm square and deep. The two homeowners were offered any three fruit tree varieties of their choice. The first homeowner selected three orange, three mandarin and three lemon trees. The trees were planted with each of the three trees of the same species receiving one of three treatments:

1. 315 ℓ sludge layered in 90 ℓ batches with soil and mixed with a garden fork.
2. A handful (approximately 40 g) of superphosphate scattered around the base of each tree and worked in superficially with a fork.
3. Tree planted in only the soil that had come from the hole (control)

It was ensured that no sludge was incorporated into the top 300mm of soil around each tree. The owners were asked to subsequently treat all trees identically with regard to watering and weeding.

The second household selected two peach and four orange trees which received one of the following treatments:

1. A handful (approximately 40g) of superphosphate scattered around the base of each tree and worked in superficially with a fork.
2. 240ℓ sludge layered with soil and mixed with a garden fork.
3. 320ℓ sludge layered with soil and mixed with a garden fork.

Figure 5.70 Orange trees planted from top left in, respectively, soil mixed with sludge, soil with 40g super phosphate and soil only.

Until mid-2011, the trees grew well, with only one mortality of the 15 planted. Table 5-15 shows the record of the growth of the trees from planting in August 2007 until August 2011. While the number of trees in the study was too small for data to be statistically significant, the percentages of height increase from height at planting did reflect the trend of trees showing the least increase with no treatment and increasing benefit with 40 g super phosphate, 240 ℓ sludge and 315-320 ℓ sludge, with the exception of mandarin, which showed a far higher increase in the growth of the tree which received no treatment. The reason for this is not known, but a possible explanation for a high value for this “no treatment” tree is the presence of a specific high nutrient source close to the tree (eg. carcass or manure disposal at an earlier time). Because the growth of the trees treated with superphosphate and sludge appeared dramatically lower than that of other species, however, it may be more likely that the two treatment trees experienced a set back which compromised their growth, such as disease or pests.

Table 5.15 Comparison of increase in height for Inadi fruit trees planted on sludge enhanced soil vs those planted on ordinary soil over 4 years

Variety	Treatment		Tree height (in metres)			
	Sludge	S Phosphate	23/8/07	25/8/11	Growth	(% increase)
Orange Z	-	-	0.96	1.35	0.39	41
Orange Z	-	40g	0.85	dead	dead	
Orange K	-	40g	0.9	2.45	1.55	172
Orange K	240ℓ	-	0.95	2.7	1.75	184
Orange Z	315ℓ	-	0.95	2.6	1.65	173
Orange K	320ℓ	-	0.88	2.8	1.92	218
Orange K	320ℓ	-	0.6	1.97	1.37	232
Mandarin K	-	-	0.76	1.95	1.19	156
Mandarin K	-	40g	0.69	1.2	0.51	74
Mandarin K	315ℓ	-	0.75	1.3	0.55	73
Peach K	-	40g	1.27	4.5	3.23	254
Peach K	240ℓ	-	1.3	4.9	3.6	276
Lemon Z	-	-	0.76	2.5	1.77	229
Lemon Z		40g	0.76	2.6	1.84	242
Lemon Z	315ℓ		0.69	2.86	2.17	314

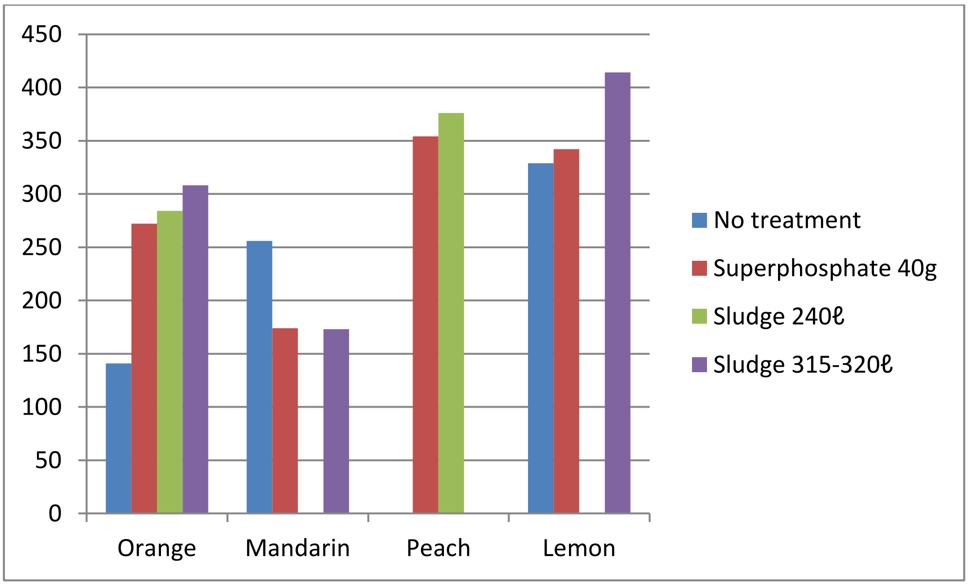


Figure 5.71 Increase in growth (%) of fruit trees with different treatments from time of planting to 4 years after planting

A foliar nutrient analysis conducted on trees at approximately 3.3 years after planting did not show consistent trends for trees with no sludge, super phosphate or sludge. For orange, the tree with no treatment had higher levels of N, K, Zn and Cu than the tree treated with sludge, although the tree treated with sludge showed 132% higher growth over original height. For mandarin, the tree which had received no treatment but showed increased growth approximately double that of the mandarin trees which received sludge or super phosphate showed elevated levels of calcium and K over the treatment trees, while the tree treated with sludge showed dramatically high levels of sodium (1003 mg/Kg over 442 and 421). Trees which had received sludge treatments showed the lowest levels of K for mandarin and lemon.

Table 5.16 Nutrient analysis of fruit trees with different treatments at 3.3 years, with highest value for species shown in light blue

Tree	Treatment	N	Ca	Mg	K	Na	Zn	Cu	Mn	Fe	P	Al
		%	%	%	%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	%	mg/kg
Orange 3	None	3.8	1.2	0.3	2.0	565	28	10.3	24	137	0.3	107
Orange 1	315 ℓ sludge	3.4	2.5	0.4	1.5	486	24	8.1	26	231	0.3	154
Mandarin 3	None	1.9	2.8	0.2	1.4	421	8	5.4	24	140	0.1	142
Mandarin 2	S.phosphate	1.8	2.5	0.2	1.2	442	10	5.4	24	155	0.1	128
Mandarin 1	315 ℓ sludge	2.2	1.4	0.3	0.9	1003	6	2.8	20	120	0.2	104
Lemon 3	None	1.6	3.4	0.3	1.1	321	12	8.2	28	189	0.1	169
Lemon 2	S.phosphate	1.7	4.2	0.3	0.9	161	12	7.6	24	179	0.1	147
Lemon 1	315L sludge	1.6	4.1	0.4	0.6	180	16	6.0	28	178	0.2	136

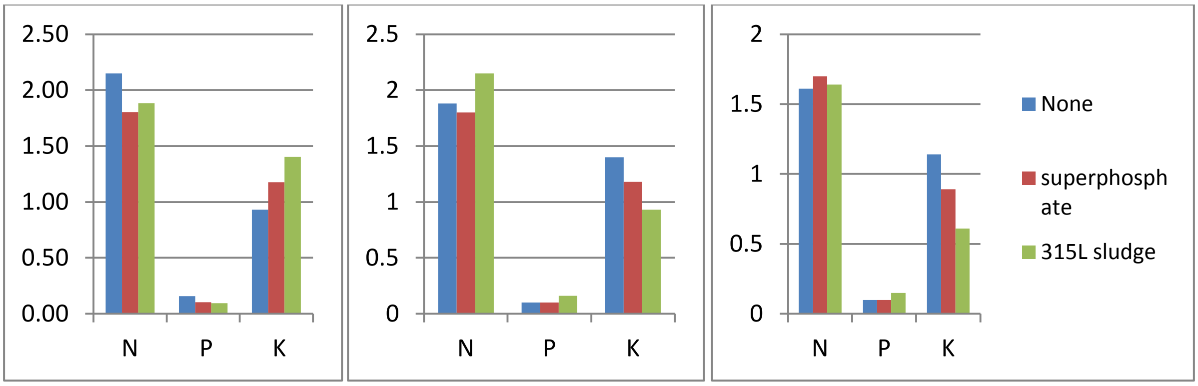


Figure 5.724 Foliar concentrations of nitrogen (N), phosphorus (P) and potassium (K) for orange (left), mandarin (centre) and lemon (right) trees for different treatments at 3.3 years after planting (% dry matter)

Homeowners reported that the trees produced excellent fruit and that neighbours had inquired about using the same methods. During the second half of 2011 however several of the citrus trees were diagnosed with greening disease (a serious disease endemic to citrus in this region) and all symptomatic branches on all trees were removed. With up to 50% of branches removed, it was not possible to gather further data comparing the growth and nutrients levels of trees from this study which would be meaningful.

Because the number of trees in this study was too small for data to be statistically significant or for observations or possible effects to be substantiated, it would be useful for a larger, more controlled study to be conducted to investigate more thoroughly the benefit of sludge to fruit trees which could represent a significant contribution to food security for families with onsite sanitation.

5.4 Conclusions

- **Tree growth**

In the Umlazi study, eucalyptus trees grown on sludge showed enhanced growth over trees grown with fertiliser or on native soil. Enhanced growth appeared to peak at 75% greater stem diameter for trees grown on sludge over control trees at 1.5 years after planting, but with a decreasing margin thereafter. It remains to be seen whether eucalyptus grown on entrenched sludge will show lasting benefits over the remainder of the growth cycle or whether the growth curves will eventually converge. With wattle, however, trees in both control and fertiliser rows demonstrated superior growth over trees grown on entrenched sludge, with the mean diameter of trees grown on native soil greater than that of trees planted with fertiliser. As the number of trees in the group that was measured declined over the course of the study, the numbers at the conclusion were not statistically significant.

Trees grown under controlled conditions in containers filled with sludge cores showed very markedly improved growth and nutrient levels over trees grown with no sludge. The improvement was greater for eucalyptus than for wattle, which as a nitrogen-fixing species may be less affected by nitrogen limits in unamended soil.

The small study in which sludge was buried on site and planted with fruit trees was inconclusive in terms of demonstrating the impact of sludge on nutrient levels and tree growth because of the small number of trees in the study and that fact that the trees in the study were affected by greening disease. It stands to reason, however, that the nutrients in sludge would benefit fruit trees both in terms of growth and nutrient content. This warrants further investigation in a larger and more controlled study. What was perhaps more significant was that the homeowners willingly participated in the study and did not report any reluctance to eat the fruit from the trees grown on the sludge.

- **Changes in sludge**

While data showed high variance, it was clear that biodegradation and dewatering occurred in pit latrine sludge after entrenchment at the Umlazi site. It appeared that an initial rapid degradation and moisture loss occurred, probably as a result of the most recently deposited and therefore unstabilised pit latrine contents degrading. Thereafter, a slow decrease in volatile solids, COD and moisture was observed, until final values were reached that appeared to be lower than the lowest values obtained in pit latrine samples. This decrease below the concentrations measured in the samples taken from the bottom of full pit latrines, where the sludge was apparently fully stabilised, could be explained by the action of fungi that result in cellulase activity that reduces the remaining potentially degradable material. Visually, degradation of sludge was apparent in both the Umlazi trial and the Inadi fruit tree trial where sludge which was excavated after two or more years of having been covered with soil was distinguishable from the surrounding soil only by the presence of rubbish. Sludge which was exhumed from pits which had been covered with soil for at least four years was found to have dewatered significantly and showed no difference in volatile solids between samples of any depth (age) or appearance.

❖ **Impact on groundwater**

Although the monitoring at the Umlazi site was hampered by drainage and instrumentation problems, sufficient observations have been made to conclude that:

- No excessive loading of nitrate or phosphorus has moved beyond the burial site;
- Simulations indicate a response time of 100 days between burial and observation of a nitrate plume at the borehole. However, the peak plume concentrations do not exceed background seasonal variations of nitrate;
- Occasional elevated concentrations of nitrate and phosphorus were observed in the borehole record, but the long term mass loading from the buried waste site seems unlikely to exacerbate eutrophic conditions in the nearby stream which is already affected by high levels of pollution in the catchment;
- The modelling of phosphorus and nitrogen are further enhanced by the inclusion of relevant parameters and processes and the extended observations.

❖ **Fate of pathogens**

While pit latrine sludge remained highly pathogenic regardless of the amount of time it had been in the pit latrine environment, a clear reduction in potentially infective ova was observed with time after burial in soil. It was found that entrenchment over a period of 30 to 36 months (2 ½ to 3 years) was sufficient to reduce potentially viable ova to negligible concentrations.

If sludge is disposed of at the same household site where the pit was emptied, as was done in the small Inadi study, risks of contamination of vehicles or roads from pathogens during transport to a final disposal or treatment site are eliminated. If trees are planted over the sludge, the risks of pathogenic sludge being disturbed by planting or building, which is a potential risk to new householders if sludge is buried in a location of which they are unaware, is reduced.

6. ENTRENCHMENT OF WASTEWATER SLUDGE (SAPPI, HOWICK)

The Howick Waste Water Treatment Works (WWTW) currently disposes of approximately 700 tonnes of sludge (measured in dry weight) every year at the Curry's Post landfill site. The capacity of the WWTW to dry the sludge to the 40-60% moisture content required for landfill acceptance is limited, particularly during the wet season. In addition, this option fails to exploit the nutrients or energy contained in sludge.

In February 2009, Sappi Forests, a forestry/timber plantation company, was asked to participate in a deep row entrenchment trial by providing a site for the experimental burial of sewage sludge from the Howick WWTW. After several rounds of discussions a protocol was approved for burial of sludge at SAPPI's Shafton Karkloof plantation, located 10 km east of Howick on the Karkloof road. A formal letter from DWAF/WRC stating that they supported the trial was provided before the trial commenced.

Figure 6.1 Commencement of sludge entrenchment at Sappi's Shafton plantation in the Karkloof area

6.1 Preparation of site

The experimental site, which encompasses 2 hectares on gently sloping ground (11%), was covered with large amounts of timber trash and in line with Sappi's standard practice it was necessary to first burn this before work could begin. In November 2009, the site was pegged out into 30 blocks each

20 m x 20 m in size, with a 10 m buffer zone between blocks. Six trenches oriented downslope and spaced 3m apart centre to centre were dug in each block using a TLB (tractor-loader-backhoe) initially and then later using an excavator. Each trench was prepared to dimensions of 20 m in length, 0.6 m width and 1.5 m depth.

An excavator was required to dig the trenches in ten of the plots because of large stumps. This also provided tracks into the site for the vehicles transporting sludge. Once the trenches were dug, sludge was carted to the site using tip trucks and the required number of loads was dumped next to each block (Figure 6.2).

Figure 6.2 Prepared trench (left) and delivery of sludge from WWTW (right)

6.2 Sludge application

The Howick Waste Water Treatment Works analysed the sludge designated for burial and provided Sappi with the sampling results indicating a sludge classification of C1a (Untreated for pathogens, stable, low pollutant content). Sludge samples were also analysed for water holding capacity, hydraulic conductivity and nutrient content. Moisture levels were found to be higher (60%) and nitrogen levels lower than expected. This may be because the sludge had been stockpiled for up to 6 months prior to the commencement of the study, which would have resulted in a reduction in nitrogen content. The entrenchment moreover took place during a period of wet weather, which would have increased the moisture content.

Five different treatments were allocated randomly to the 30 blocks (Figure 6.2), assuming a density of 1200 kg/m^3 and a 1% nitrogen content in the wet sludge. The trenches were backfilled to allow the required amount of sludge to be filled to 300 mm below the surface. Thereafter each treatment was topped with 300 mm of the excavated topsoil. The five treatments comprised:

- Treatment 1 (T1): 250 mm sludge (120 dry tons/ha), providing 2 250 kg nitrogen/hectare
- Treatment 2 (T2): 500 mm sludge (240 dry tons/ha), providing 4 500 kg nitrogen/hectare
- Treatment 3 (T3): 750 mm sludge (360 dry tons/ha), providing 6 750 kg nitrogen/hectare

- Treatment 4 (T4): Control: trench dug and refilled with no sludge application
- Treatment 5 (T5): Control: No trench, no sludge. Standard planting in small holes.

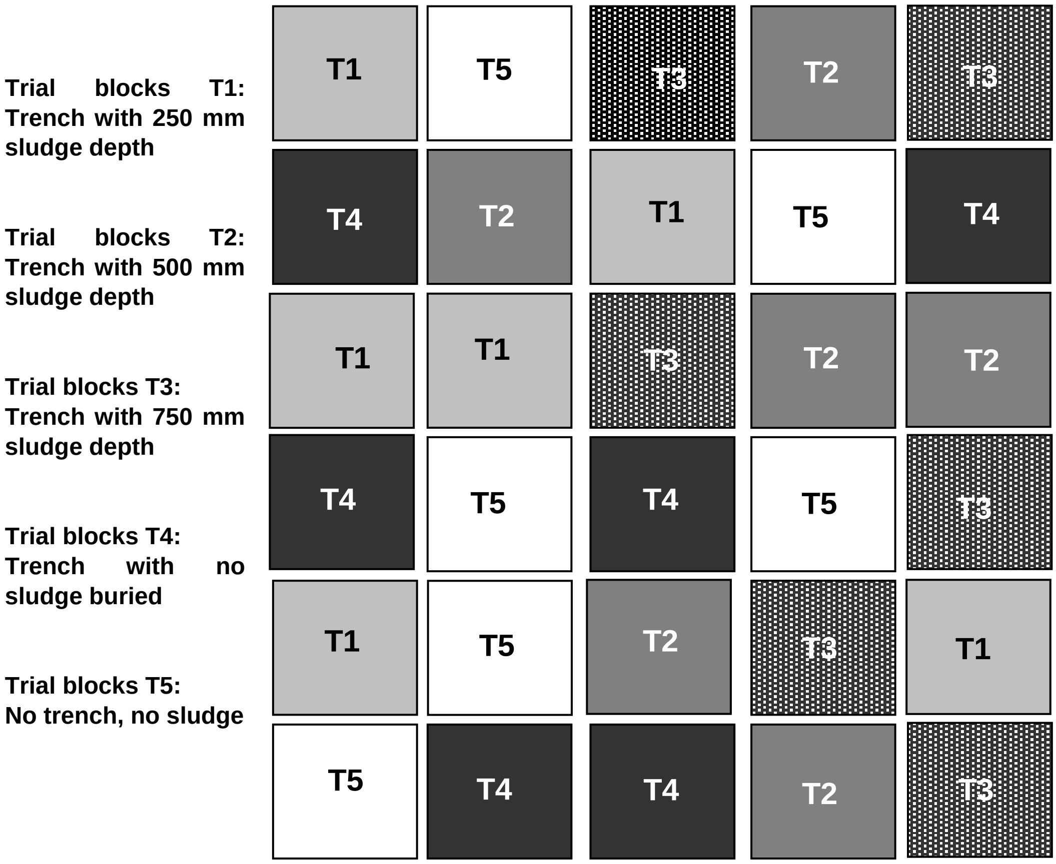


Figure 6.3 Layout of treatment blocks at Sappi site

6.3 Characterisation and monitoring

A number of studies were conducted to characterise the site. These included a topographic survey, geophysics surveys of nine transects on the site using Electrical Resistivity Tomography (ERT) technology and characterization of the soil and waste material including particle size, hydraulic conductivity and water retention characteristics.

Monitoring comprised local (within each treatment) and hillslope scale observations of soil water and groundwater dynamics and nutrient concentrations. The tree growth was also monitored periodically.

6.3.1 Geophysical characterisation

Geophysical characterization of the subsurface using ERT was introduced at two scales. The first was conducted to define the nature and extent of the geological formations and distribution of deep water bodies surrounding the trial site and the second, a series of much shallower and shorter surveys, to determine the water distribution in and adjacent to the trenches as well as to define subsurface, preferential flow pathways which may intersect the trenched material and move nutrients rapidly down the hillslope.

The same ERT technique was used as was described for the Umlazi study (Section 5.1.2.1).The positions and extent of the surveys are show in Figure 6.4.

ERT surveys were conducted in eight transects using the Wenner Long and Wenner Short protocols with electrode spacing of 0.5 m or 1 m. The deep surveys included a long transect along the entire length of the site (Transect A-B) as well as transects adjacent to each of the two boreholes (BH1 and BH2). Shallower surveys were conducted at the two piezometers on the lower slope (SP6 and SP7) and along short transects intersecting the trenches for each of the four trenched treatments (T1, T2/T3 combined and T4). The topsoil at the site was generally moist and offered good contact with the electrodes. Occasionally, water was poured to improve electrode contact with the soil. Due to a cable malfunction, electrode positions had to be excluded from measuring on several stations during the survey.

Transect AB, completed early in the trial (December 2009), revealed a top layer of 1-2 m with low resistivity values ranging from 300 to 800 Ωm, suggesting a relatively moist soil (Figure 6.5). The layer below this had high resistivity values of up to 3000 Ωm which varied laterally, suggesting weathered and fractured igneous or metamorphic bed rock. Below 10 m the resistivity dropped, suggesting the presence of a fractured aquifer. GPS readings were also taken every 10 m along this transect using the Trimble differential GPS to allow for topographical rectification of the ERT data.

Figure 6.4 Map of ERT transects, boreholes, piezometers and soil water sensing nests at the SAPPI Shafton sludge burial site (Google Earth image

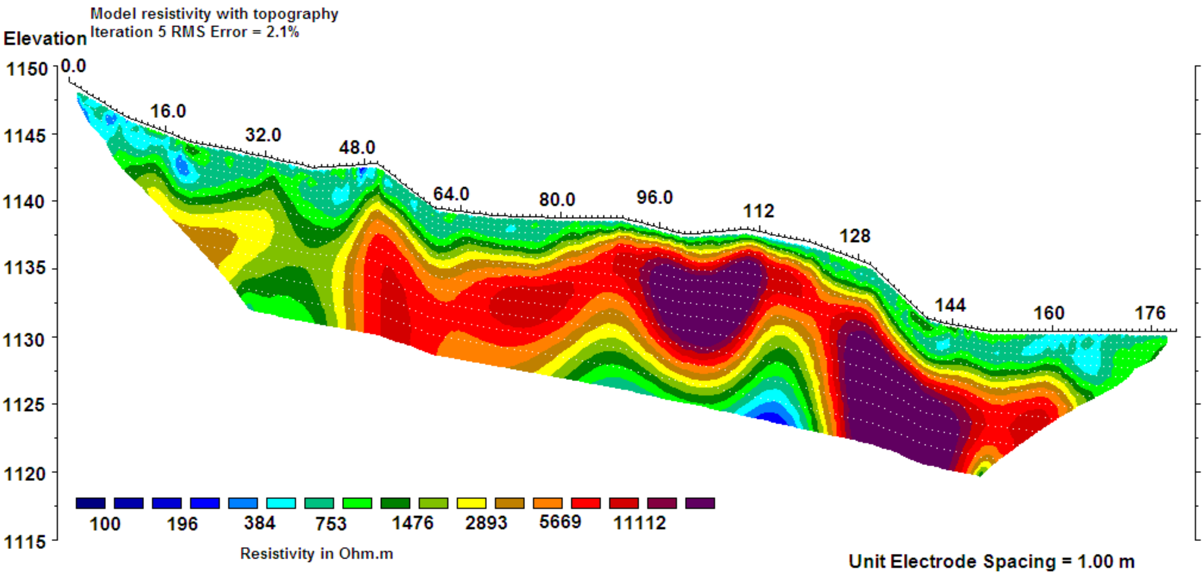


Figure 6.5. Resistivity for Transect A–B (7 December 2009)

Later in the trial, surveys were concluded on the lower end of the trial area to confirm the across-slope connectivity of groundwater sources. The study of a 60 m transect 5 m north of borehole BH1 indicated an igneous or metamorphic bed rock on the eastern side (resistivity values > 5000 Ω.m) with lower resistivity values on the western side (<1000 Ω.m) suggesting relatively shallow, fractured and weathered bed rock containing water (Figure 6.6). The drilling report indicated that borehole BH1 had 4 m of shale and loam, then 26 m of dolerite, showing more evidence of weathering than the transect near BH2. Observed groundwater depths in BH1 varied between 2 to 3.5 m, suggesting a shallow supply from the weathered shale, overlying the fractured dolerite in the west.

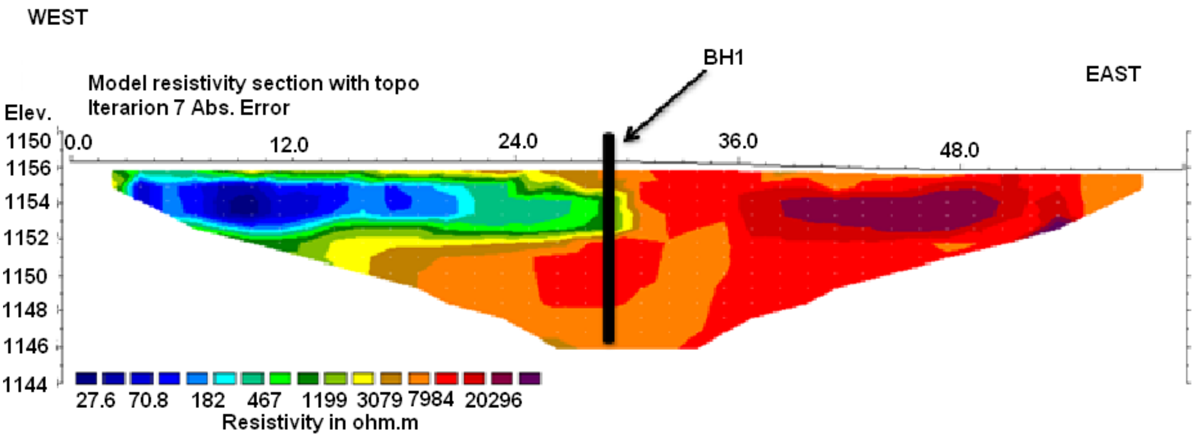


Figure 6.6. Resistivity for transect near Borehole BH1 (12-18 July 2011)

Near borehole BH2, the survey of a 60 m transect indicated that a layer of topsoil of 1 m to 4 m in thickness (resistivity values ranging from 1000 Ω.m to 5000 Ω.m) lay above a layer of igneous or metamorphic bed rock (resistivity values > 5000 Ω.m). At a depth of 10 m resistivity decreased, suggesting the presence of an aquifer located in weathered bedrock. The drilling report indicated borehole BH2 had 2 meters of loam and thereafter 28 meters of dolerite. The average water depth in borehole BH2 was measured at 10 m, confirming a source in the fractured dolerite.

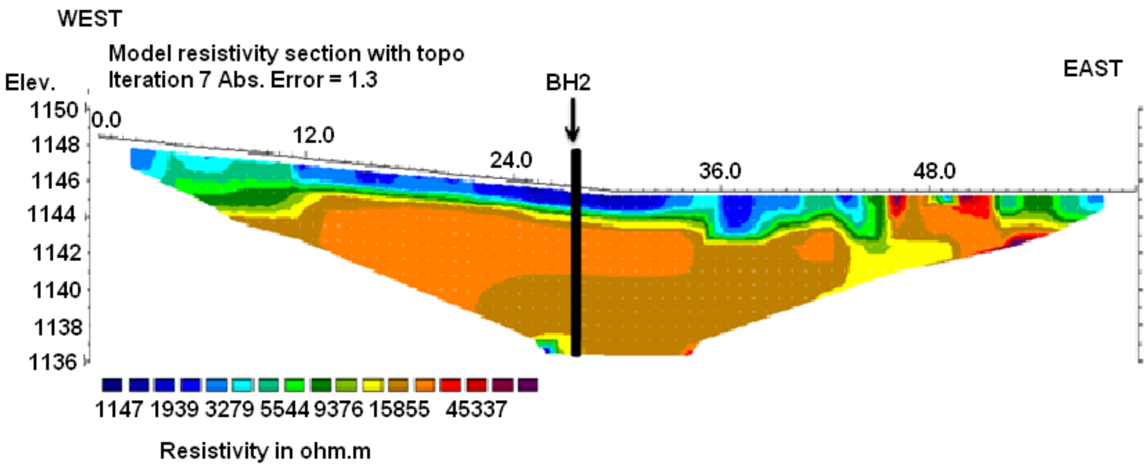


Figure 6.7. Resistivity for transect near Borehole BH2 (12-18 July 2011)

The first of the shallower surveys (June 2011) comprised a 40 m transect, surveyed on the northern side of the road at the downslope edge of the trial, near piezometer SP6, (Figure 6.7). This indicated a top layer of approximately 1 m to 2 m of topsoil with resistivity values ranging from 1900 Ω.m to 5000 Ω.m above a layer 4 m in depth with lower resistivity (<300 Ω.m), indicating a water source in weathered material. Depths of water during the summer period at SP6 ranged between 1.3 and 2.8 m below surface, confirming this as a zone of near-surface preferential flow.

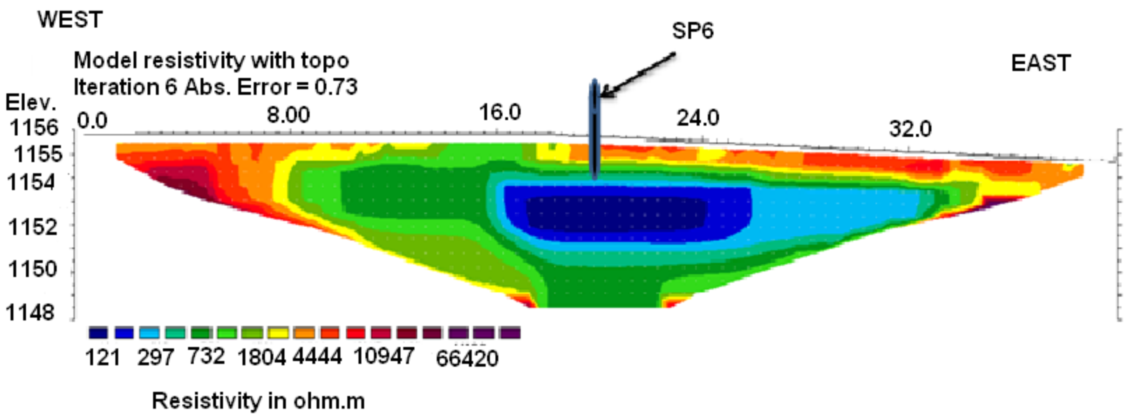


Figure 6.8. Resistivity for transect near piezometer SP6 (12-18 July 2011)

A transect on the northern side of the road, 5 m south of piezometer SP7, was also surveyed in June 2011 (Figure 6.9). The top 4m showed resistivity values ranging between 2000 Ω.m and 5000 Ω.m corresponding to wet topsoil with a deeper layer showing high resistivity values (> 5000 Ω.m), suggesting igneous or metamorphic bed rock. Variations in resistivity in the near surface in this transect also suggests the existence of a preferential pathway which could conduct near-surface lateral flows rapidly during high rainfall events. Water levels observed during the summer at this location varied between 2.3 and 2.8m.

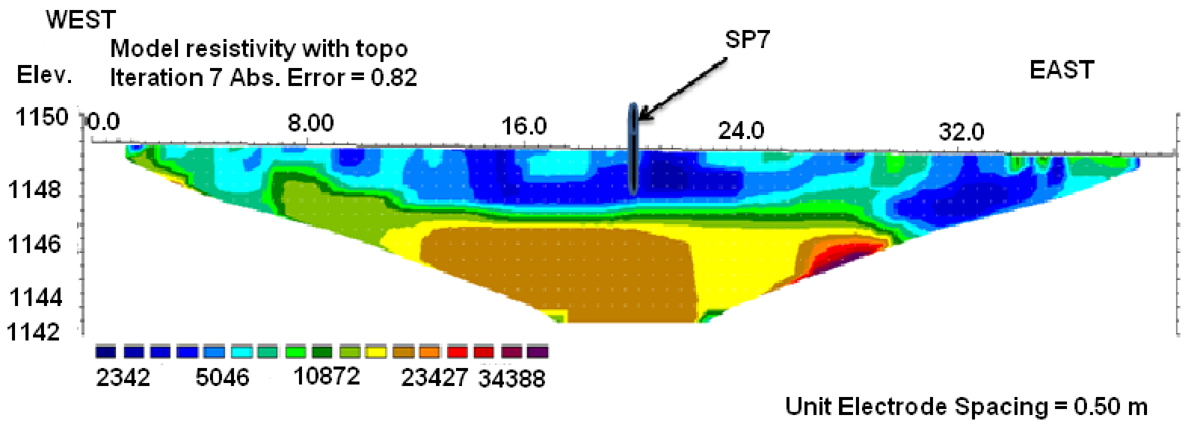


Figure 6.9 Resistivity for transect near piezometer SP7 (12-18 July 2011)

Surveys within the trial area were conducted to assess the water distribution within and between the trenches. A 20 m transect surveyed in the Treatment 1 (T1) plot revealed high resistivity values (>2500 Ω.m) in the trenches contrasting with the more conductive layer of unsaturated soil (<1000 Ω.m) adjacent to it (Figure 6.10). This could be attributed to the different bulk densities and water holding capacities of the sludge and the native soil.

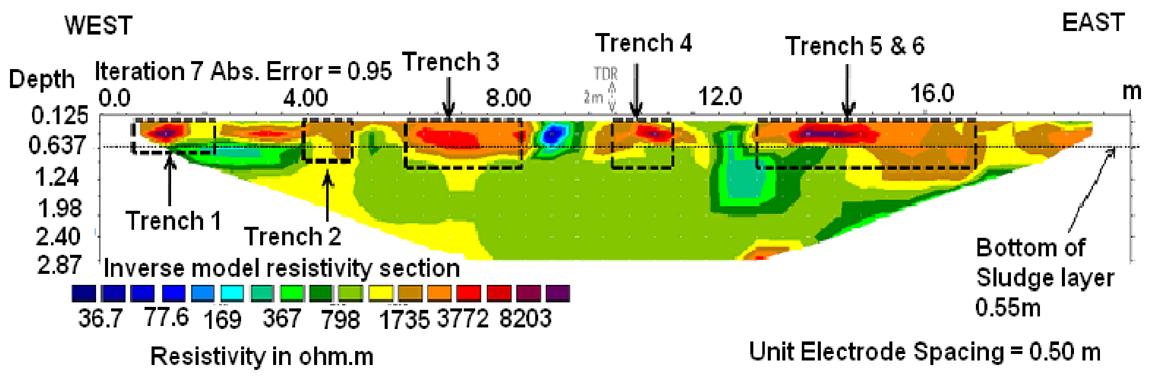


Figure 6.10. Resistivity for transect of T1 plot (12-18 July 2011)

A 55 m section transecting adjacent T2 and T3 plots (and including the 10 m buffer between the plots) was surveyed. On the surface, the trenches had high resistivity values (>2500 Ω.m) in contrast with the more conductive layer of the unsaturated soil between the trenches (<1000 Ω.m) (Figure 6.11). However, the bottom of the trenches (1.2 m) was not as clearly defined in this survey as in the other treatment plots that were surveyed. This could have been caused by the presence of water in a layer between 0.7 m and 2 m. Between the treatments and below 2 m, the soil had a high resistivity of <3000 Ω.m, suggesting dry conditions from surface to a lower bedrock. The data indicated that, although the trenches were dry at the time of this winter survey, sufficient moisture had been supplied to adjacent areas, possibly due to ingress through the trenched material in the summer.

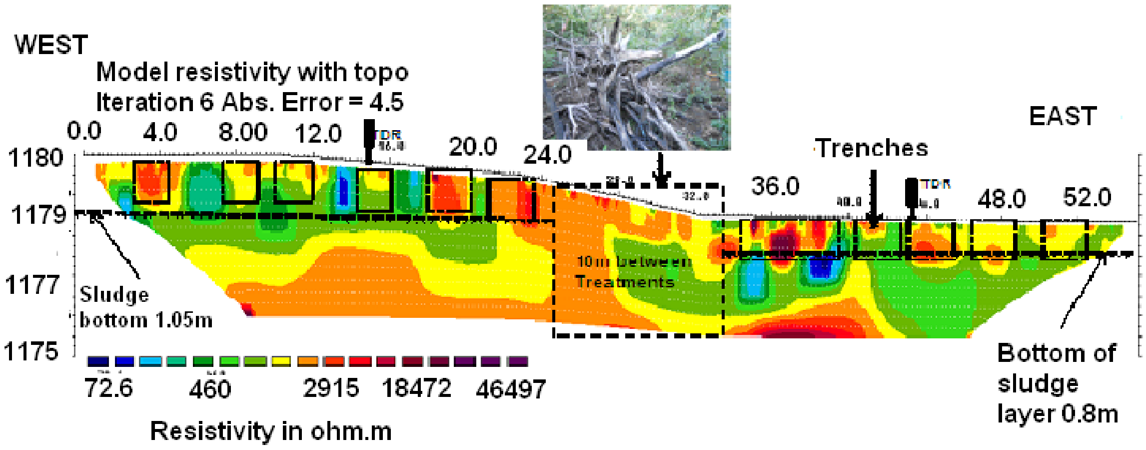


Figure 6.11. Resistivity for transect across T2 and T3 treatments (12-18 July 2011)

An ERT survey was conducted on a 20 m transect in a Treatment 4 (T4 plot – trenches backfilled with no sludge). High resistivity values in the trenches (>2500 Ω.m) contrasted with the more conductive layer of unsaturated soil (<1000 Ω.m) between the trenches (Figure 6.12). Lower resistivity at a depth of 0.7 m to 1.7 m may have been caused by a hole 3 m north of the section that had been dug for soil sampling.

Comparing the results shown in Figures 6.10 to 6.12 does indicate that some of the variation in resistivity may be due to the disturbance of the soil resulting from the trenching and backfilling process, and is not solely attributable to the presence or absence of sludge.

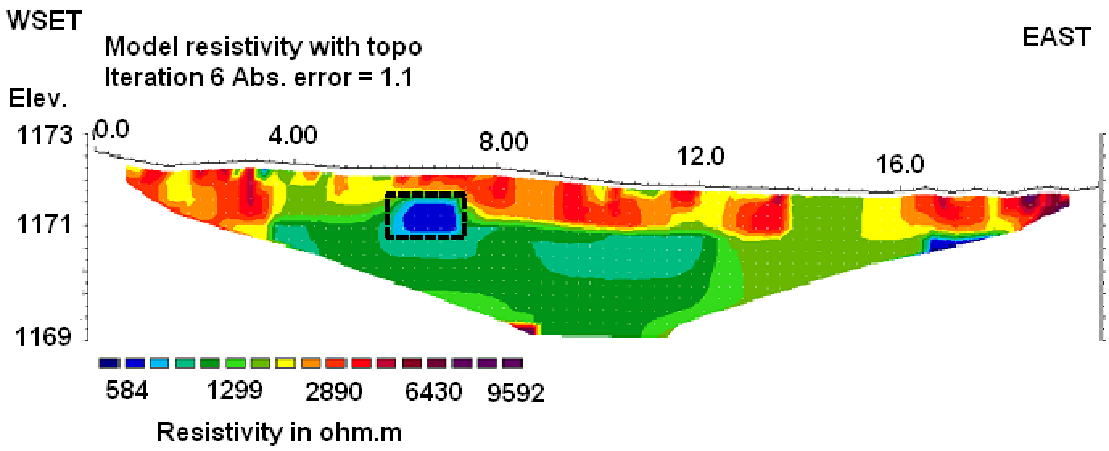


Figure 6.12. Resistivity for transect of T4 plot (12-18 July 2011)

6.3.2 Soil characterization

Tests were conducted to establish the particle size distribution, hydraulic conductivity and water retention characteristics of soils beside the trenches of all four treatments. These results were used to model water movement, uptake and distribution as well as the movement of contaminants from the sludge through the soil.

Saturated hydraulic conductivity was measured using a double ring infiltrometer (near surface) and the Guelph permeameter (to 1.5 m depth) methods. In each trial plot, four undisturbed samples were collected in 5 cm diameter steel coring rings at depths of 0–30 cm, 30–60 cm, 60-90 cm, and 90–150 cm, in order to determine soil water retention. Another set of undisturbed soil samples and sludge was collected and were taken the laboratory where standard sieve and hydrometer methods (Gee and Baunder, 1986) were used to determine sand, silt and clay fractions in the profile. The four undisturbed soil samples from each treatment (T1, T2, T3 and T4) were used to determine the soil water retention characteristics at 2, 5, 10, 20, 50, 100, 200, 500, 1000, 1500 kPa pressure using a controlled outflow cell and pressure plate extractor.

6.3.3 Instrumentation

Instruments installed at the site for monitoring purposes included, Time domain Reflectometry (TDR) probes (volumetric water content), WaterMark sensors (soil water suction), wetting front detectors (WFD) (for indicating arrival of a wetting front and for collecting samples for analysis), boreholes, piezometers (Figure 6.4) and a meteorological station.

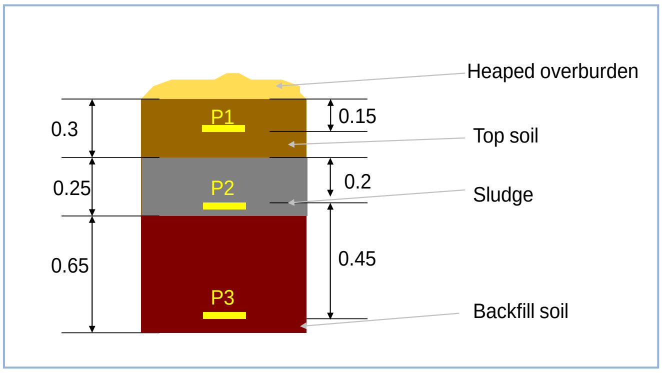


Figure 6.13 Positioning of TDR probes in TI block

During the burial of the sludge, Time Domain Reflectometry (TDR) probes were installed in one selected trench of each of treatments 1-4 to measure volumetric water contents and electrical conductivity in the soil and sludge. In each selected trench, probes were installed at the following depths (Figure 6.13):

- P1: in the 300mm soil covering above the sludge (150 mm)
- P2: in the sludge (0.5 m)
- P3: below the sludge (1.2 m)

Figure 6.14 Installation of a pair of wetting front detectors at depths of 400mm depth (left) and 1m (right)

Wetting front detectors were installed in the trenches at depths of 400 mm and 1 m below surface (Figure 6.14). Samples of the drainage liquid were collected for analysis of the migration dynamics of sludge effluent. A weather station was established at the site to measure temperature, barometric pressure, humidity, wind speed, wind direction, and precipitation. Seven piezometers were installed, five at the lower (south east) corner of each treatment (SP1-SP5) and two between the stream and the experimental plot (SP6, SP7). The depths of the installed piezometers ranged from 2.5 to 3 m. Figure 6.15 shows a schematic layout of the instrumentation installed in the trial plots. The two 60 m deep boreholes were positioned in the valley line up-gradient (BH1) and down gradient (BH2) from the site, between the experimental plot and the stream running on the southern boundary (Figure 6.4).

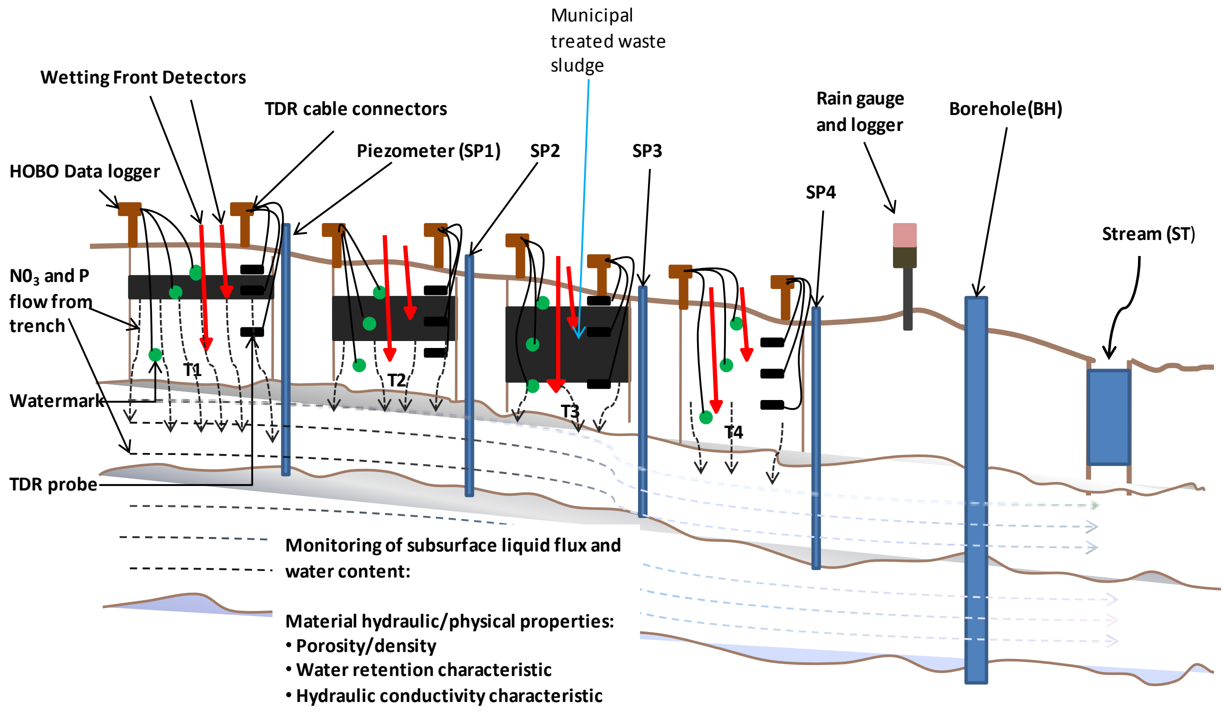


Figure 6.15: Schematic of instrumentation for monitoring of subsurface water at the Sappi Shafton site

6.3.4 Groundwater

Samples were collected from the wetting front detectors, piezometers and boreholes to monitor the impact of the entrenched sludge on ground water.

The seven piezometers were monitored at regular intervals. When free water was found in the base, the water depth was measured and a sample was collected for analysis. Piezometers SP1 to SP5 occasionally revealed evidence of lateral water seepage, but only through the existence of moisture at the soil/bedrock interface, with insufficient accumulation to collect a sample during the monitoring period. However, the lack of observed water in these stations does not preclude the rapid accumulation and subsequent depletion of perched water from rainfall events occurring between visits to the site. Piezometers SP6 and SP7, however, located downslope of the treatment area, had standing water in them for most of the rainy season. The water table fluctuation was recorded and samples taken over 18 months (22 Dec 2010 – 27 May 2011).

The volumetric water contents (TDR probes) within the monitored trench of each treatment were measured manually by means of PCTDR software at least once a month during the dry season, and once a week during the wet season.

Groundwater depths in the boreholes (BH1 and BH2) were monitored regularly and water samples collected for analysis (June 2010 – July 2011). For this, the depth to the water table at each well was recorded and twice the volume of the standing water within the well was purged before a water sample was taken to ensure that the samples were representative of the groundwater around the well rather than of the water standing in the well. The stream was also sampled at regular intervals during the monitoring period (October 2010 – July 2011).

The water samples collected from piezometers, boreholes, WFD and the stream were analysed for ^{2}H and ^{18}O Isotopes, nitrates, phosphates, pH, electrical conductivity, hydraulic conductivity and Oxidation Reduction Potential (ORP). The **s**table water isotopes are used to identify likely sources of water, flow paths and evaporation processes. Electrical conductivity (EC) was also measured in order to estimate the amount of total dissolved salts (TDS), or the total amount of dissolved ions in the water. The EC data can also be used to supplement the isotope data in characterising the sources and flow pathways of water. Oxidation Reduction Potential (ORP) was measured in order to characterize the oxidation-reduction state of the water on a scale from approximately -300 mV (strongly reducing) up to +500 mV (strongly oxidizing). Samples were analysed for *E. coli*, heterotrophic plate count, nitrate, ammonium, chloride, sodium, conductivity and COD.

Rainfall, temperature, relative humidity and solar radiation measurements for the Sappi site were downloaded from the weather station located at the Sappi airstrip, 800 m from the site.

6.3.5 Tree growth

Measurements were taken every 3 months to monitor tree growth. Initially 12 trees were measured (4 trees in each of three rows), but on Sappi's recommendation this was increased to 32 trees in the centre of each block (8 trees in each of 4 rows). Dead or damaged trees were also included in the measurement. Initially, measurements were taken using a 5m survey staff. Once tree height exceeded 5m stem diameter was measured at a height of 1.37 m (chest height) from the base of the tree to estimate total tree biomass. Visual assessments of the growth and health of trees were also conducted when it was possible.

6.4 Results

6.4.1 Impact on groundwater

Subsurface water and nutrient dynamics were monitored at three different scales. The first was at the local scale, within the trenches, the second monitored near surface, preferential pathways on the hillslope scale and the third monitored deep groundwater aquifers at the downslope end of the trial site.

6.4.1.1 Local scale monitoring

Soil Water Content

All profiles where data were collected from TDRs[4] were wet during late summer of 2010 (water content >0.30) and became dry during the winter (<0.20), (Figures 6.16 - 6.19). Measurements from probes located within the sludge (0.5 m depth) displayed the highest amplitude in wetting and drying, indicating the relatively high capacity of the sludge to store and release water. For the T3 trench (highest sludge loading), water content at this depth remained high throughout the dry season.

None of the profiles responded to the extreme event of March 2010 (140 mm), possibly due to its short duration (the trees at that stage still being too small to have any effect on soil moisture). The levels of drying and subsequent wetting in the profiles (except T3 for the first season) reflect the wetting and drying dynamics in the trench without sludge T4 (Figure 6.22), indicating that the sludge, apart from an initial slow release of moisture, behaves similarly to the disturbed parent soil filled into the trench at T4.

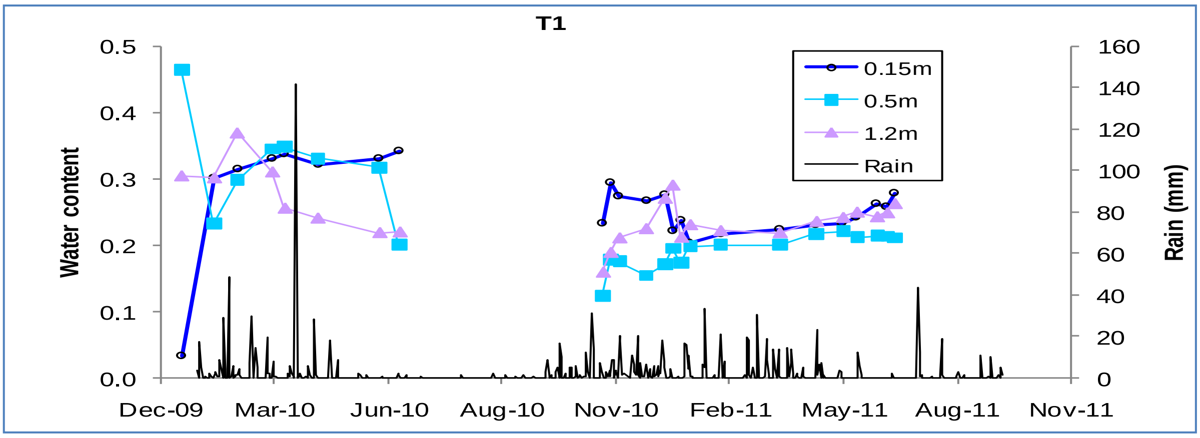
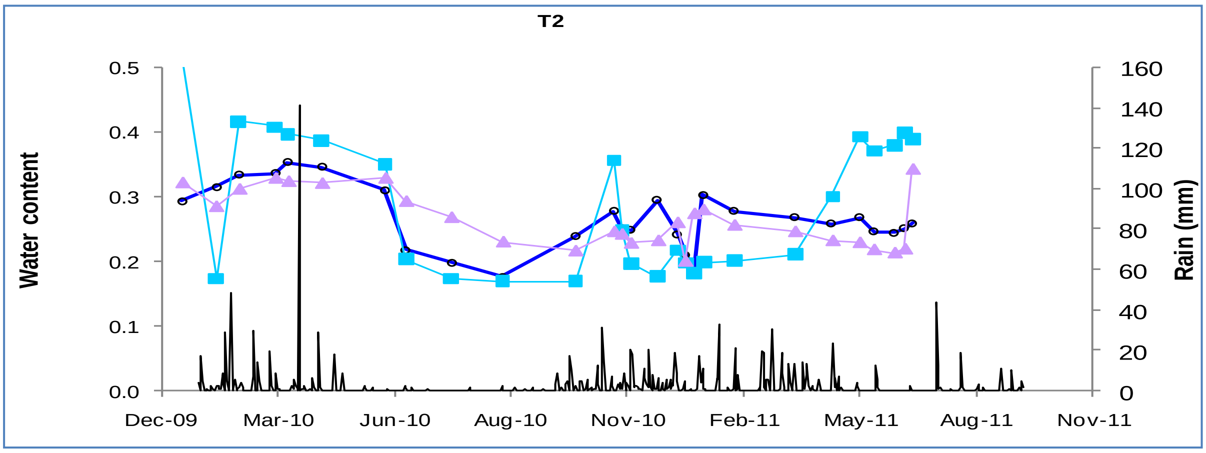


Figure 6.16. Dynamics of rainfall and soil water content for T1 block



[4] The missing data for Treatments 1 and 4 occurred due to TDR cable damage at the site which was later repaired.

Figure 6.17. Dynamics of rainfall and soil water content for T2 block

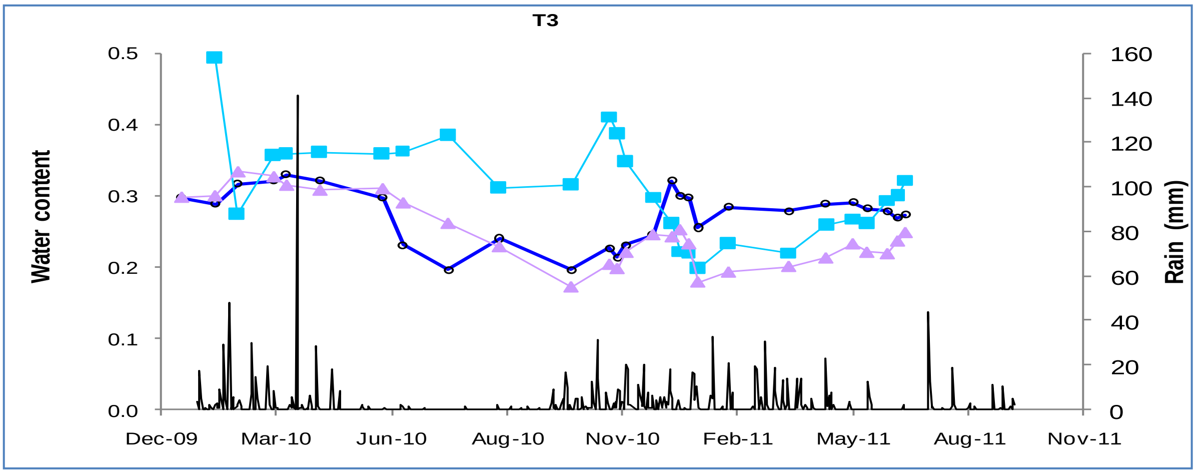


Figure 6.18. Dynamics of rainfall and soil water content for T3 block

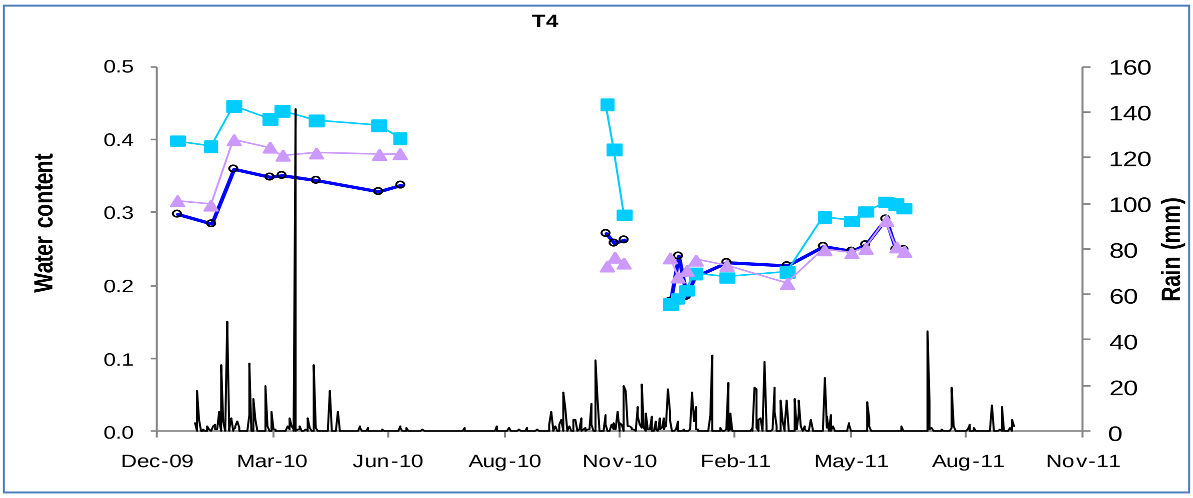


Figure 6.19. Dynamics of rainfall and soil water content for T4 block

Wetting front detectors

Samples collected from wetting front detectors were analysed for nitrate, phosphorus, pH and Oxidation-Reduction Potential (ORP). Nitrate concentrations collected from all three sludge treatments at both depths (0.5 m and 1.2 m) decreased gradually after initial installation and then increased with the onset of rain in October 2010, peaking in December 2010. After the initial flushing of available nitrate by rain, concentrations built up in the profile again over a period of three months. For the T3 trench (highest sludge loading), the deep concentrations did not begin this increase in October 2010, but continued decreasing until December 2010. However, the concentrations at 1.2 m in T3 were the highest throughout the early period of observation. In general, the observed concentrations were high (100 – 500 mg/ℓ), even for an agricultural soil environment, warranting continued monitoring.

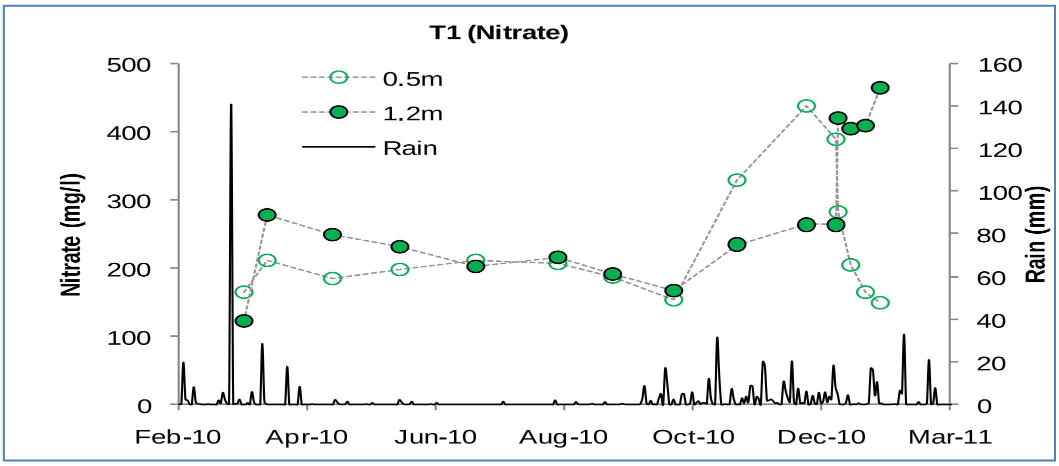


Figure 6.20. Observed nitrate concentrations in T1

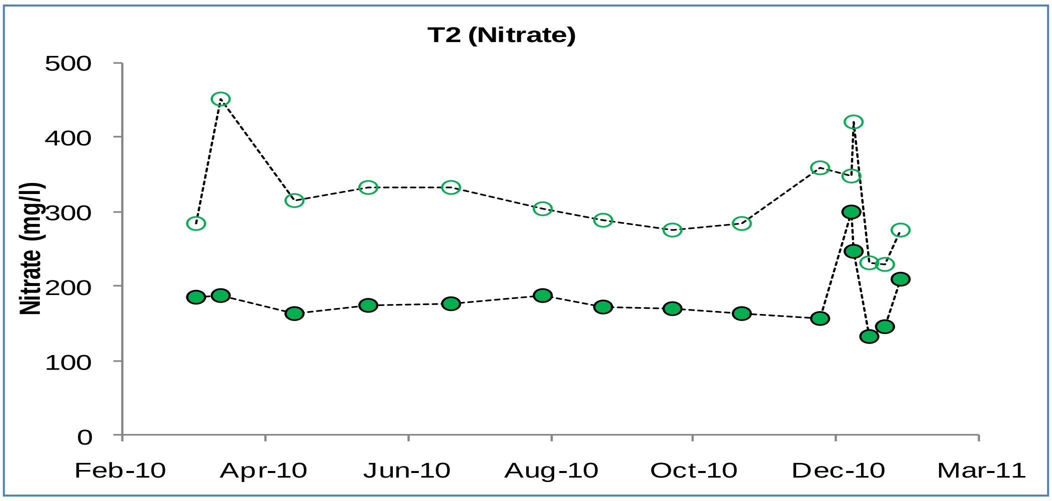


Figure 6.21. Observed nitrate concentrations in T2

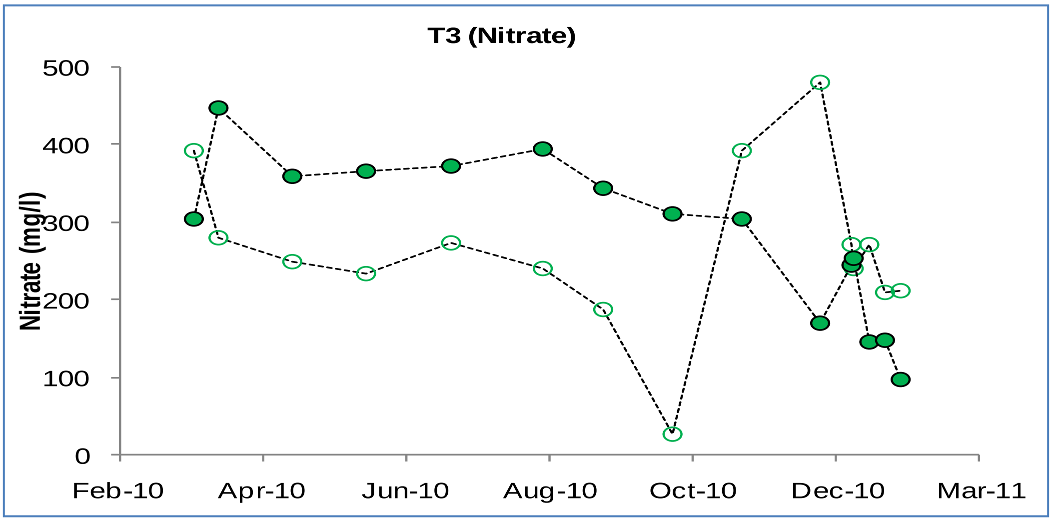


Figure 6.22. Observed nitrate concentrations in T3

The phosphorus concentrations in the middle of the T1 treatment sludge layer were consistently high (20 mg/ℓ) which appeared to be an anomaly, as concentrations at all other sites and depths were generally below 3 mg/ℓ with the exception of the 0.5 m level in T3, where phosphorus levels reached a peak of approximately 18mg/ℓ in October 2010. These results reflected the spatially varied source of phosphorus in the sludge and also the temporal variations in phosphorus movement.

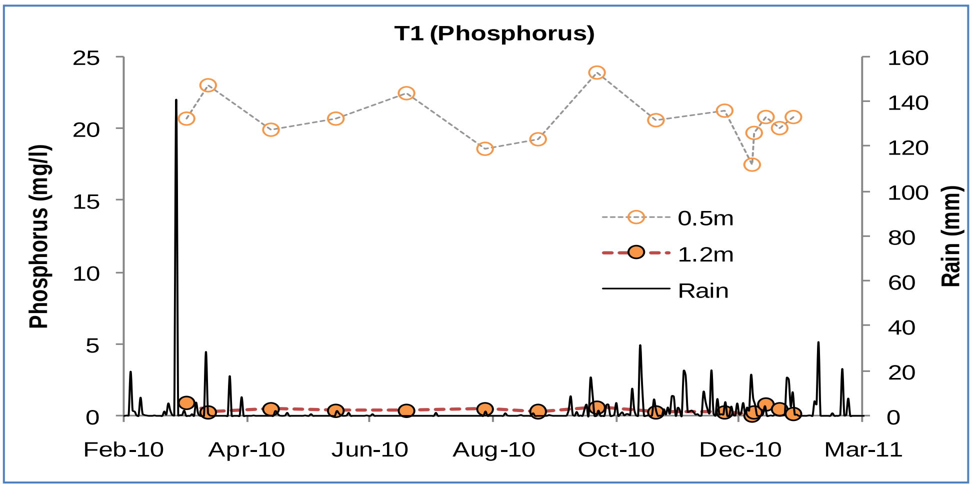


Figure 6.23. Observed phosphorus concentrations in T1

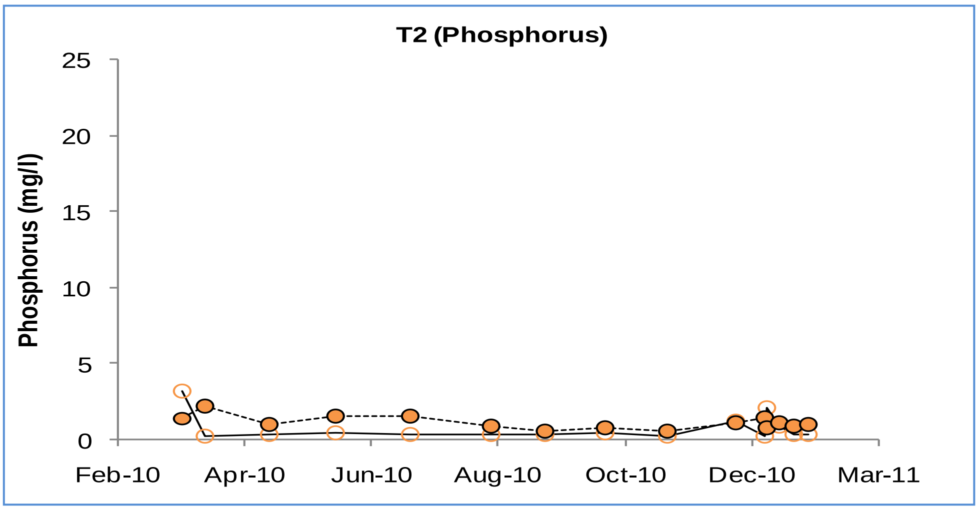


Figure 6.24. Observed phosphorus concentrations in T2

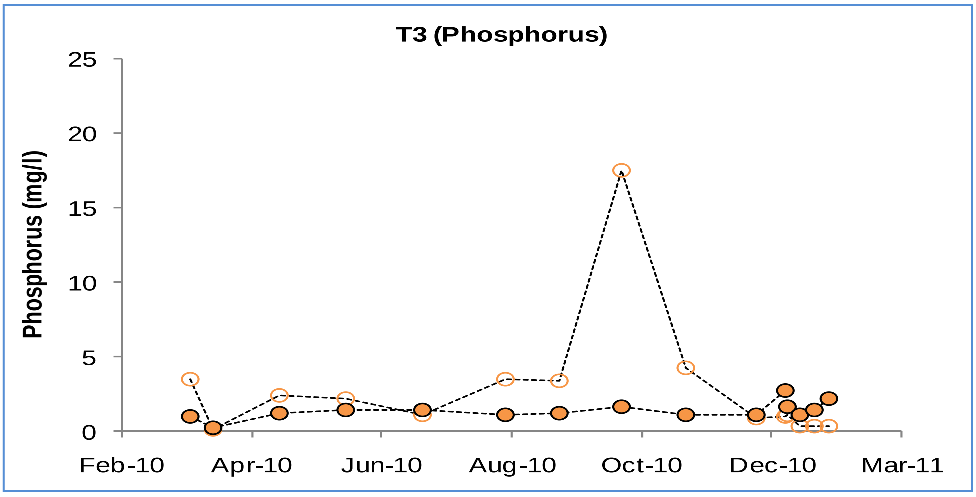


Figure 6.25. Observed phosphorus concentrations in T3

6.4.1.2 Piezometers, boreholes and stream

The borehole water levels reflected the intersection of two different sources, as suggested by the geophysics. The upslope borehole, BH1 had phreatic water at an average of 2.1 m below surface throughout the observation period, while the downslope borehole, BH2, with a lower ground surface elevation, had an average phreatic surface at 11.3 m below ground. The isotopic values also indicated that the sources of the water in these boreholes were different.

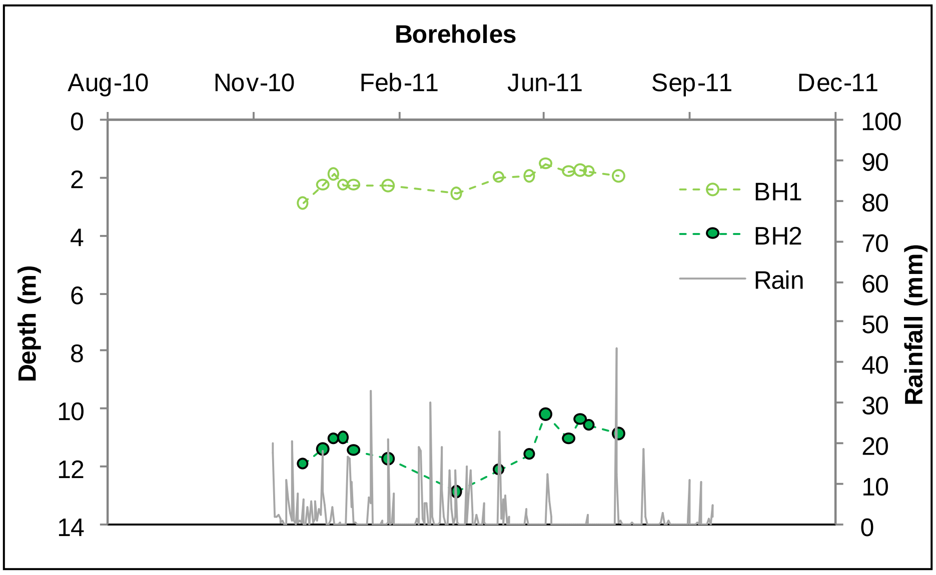


Figure 6.26 Groundwater depths in boreholes BH1 and BH2

The only two piezometers which reflected near surface water during the monitoring period were SP6 and SP7, located on the downslope edge of the trial area. SP6 was located in a possible drainage channel, while SP7 was on higher ground. Hence the longevity of the perched water in SP6 (is greater than in SP7 for the wet season of 2010/2011.

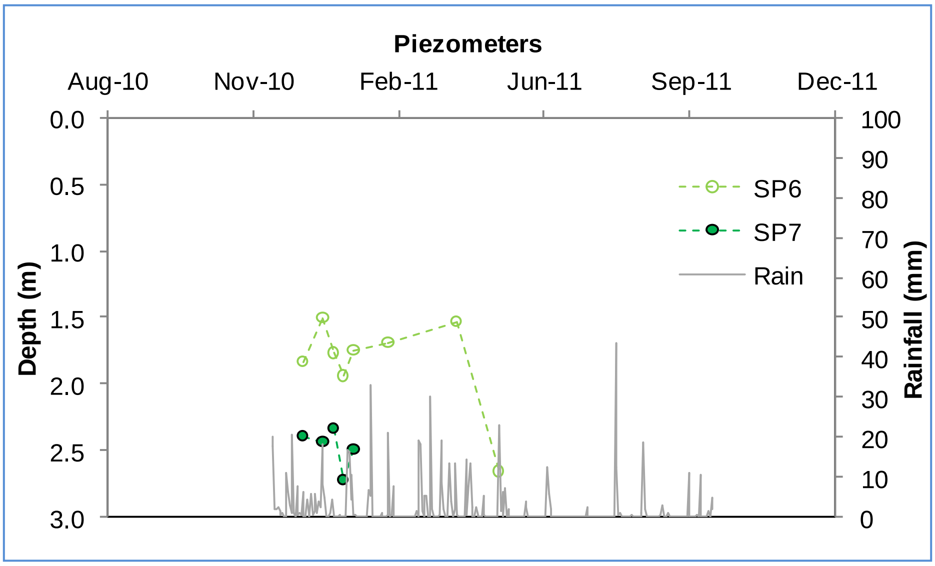


Figure 6.27 Perched groundwater depths in piezometers SP6 and SP7

Nitrate concentrations in the boreholes increased with the onset of the wet season in October 2010, peaked in April 2011 and then subsided. These concentrations were not nearly as high as those found in the infiltrating water and may reflect natural seasonal variations in nitrate concentrations. The values were well below the drinking water standard (45 mg/ℓ as NO_3) and the range of concentrations in the boreholes was similar to that in the stream.

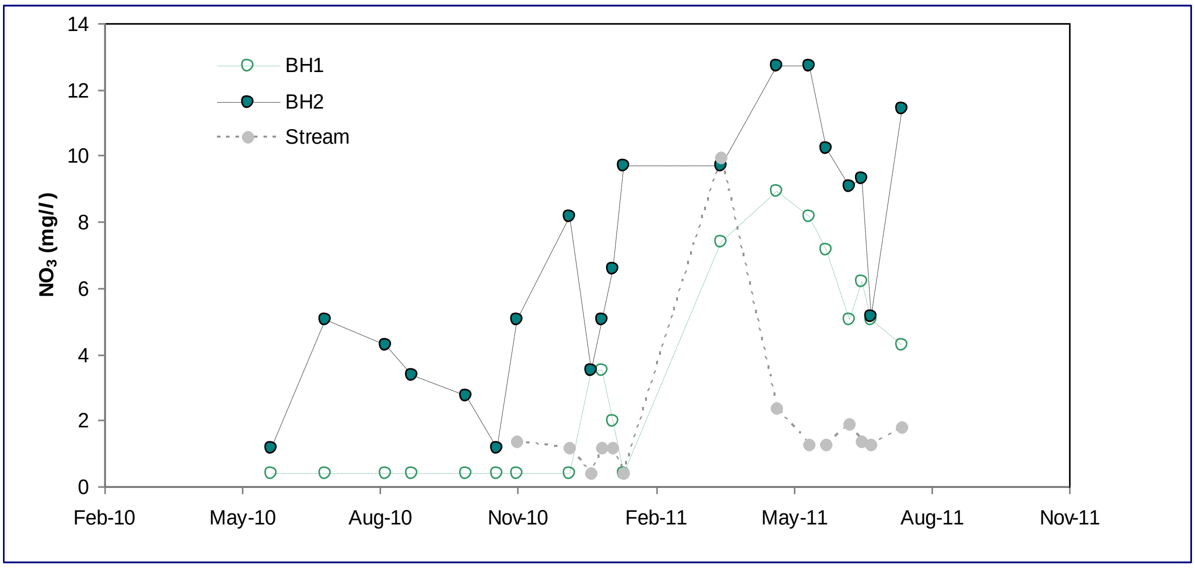


Figure 6.28. Observed nitrate concentration in boreholes and stream

Nitrate concentrations in piezometers SP6 and SP7 varied widely: concentrations in the shallower piezometer, SP6, ranged from 10 to 200 mg/ℓ while concentrations in the deeper piezometer, SP7, ranged from 100 to 700 mg/ℓ. These variations reflected accumulation along preferential flow paths during the wet season rather than a direct response to rainfall events. SP6, located in a subsurface waterway, had a deeper flow regime than that of SP7 and also continued for much longer than in SP7. Concentrations in SP6 were similar to those intercepted by the wetting front detectors in the trench profiles, suggesting a connection between the infiltrated water and the accumulated subsurface, free water discharge at the soil/bedrock interface. This mechanism of the interception of buried waste by rapid, near surface lateral flow is often overlooked and requires monitoring and assessment of the total load from these pathways.

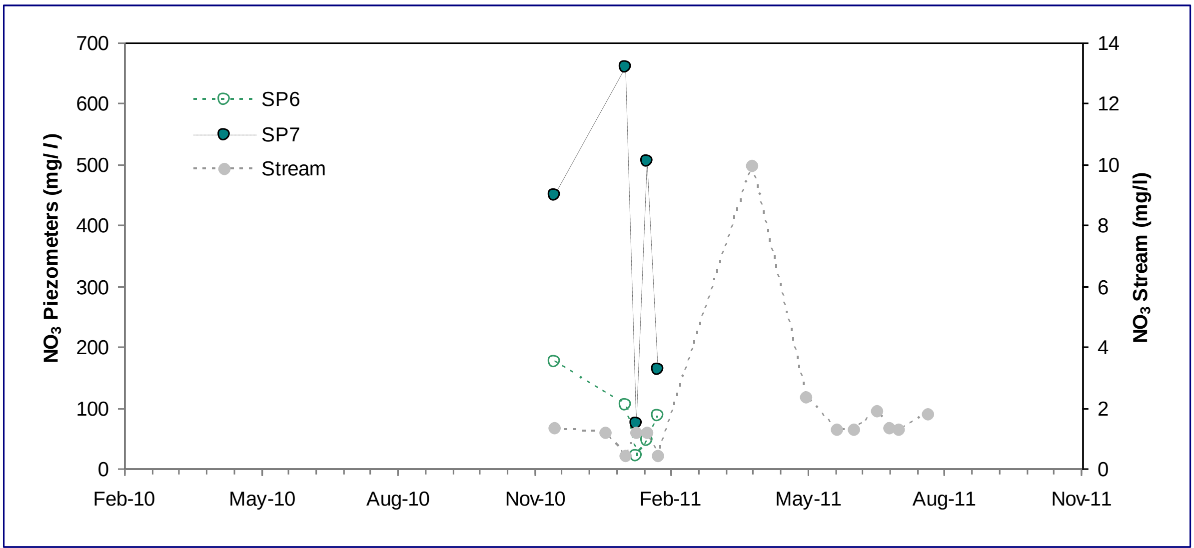


Figure 6.29. Observed nitrate concentration in piezometers (SP6 and SP7) and stream

Phosphorus (P) concentrations in the boreholes and stream varied throughout the monitoring period, but appeared to increase from May 2011 onwards. The range of P concentrations in the boreholes was equal to that in the stream and was assumed to be similar to subsurface contributions upstream of the trial site.

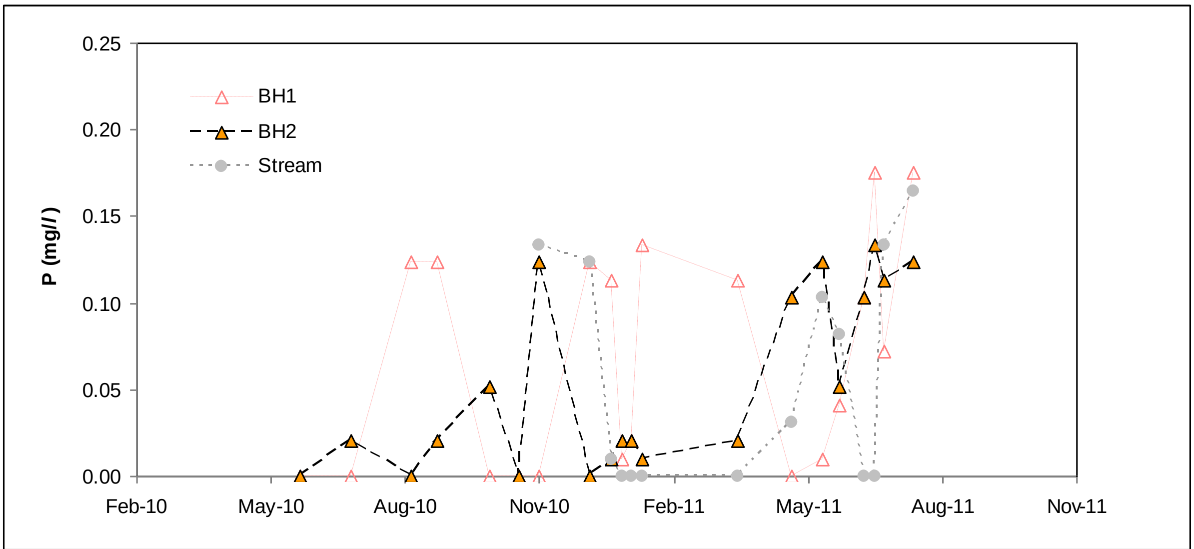


Figure 6.30 Observed phosphorus concentrations in boreholes and stream

Phosphorus concentrations ranged widely (0.1 – 23 mg/ℓ) in the piezometers SP6 and SP7 but were similar to infiltrated water detected in the wetting front detectors. The phosphorus concentrations reflected the nitrate concentrations in these piezometers, with SP7 having higher phosphorus concentrations than SP6.

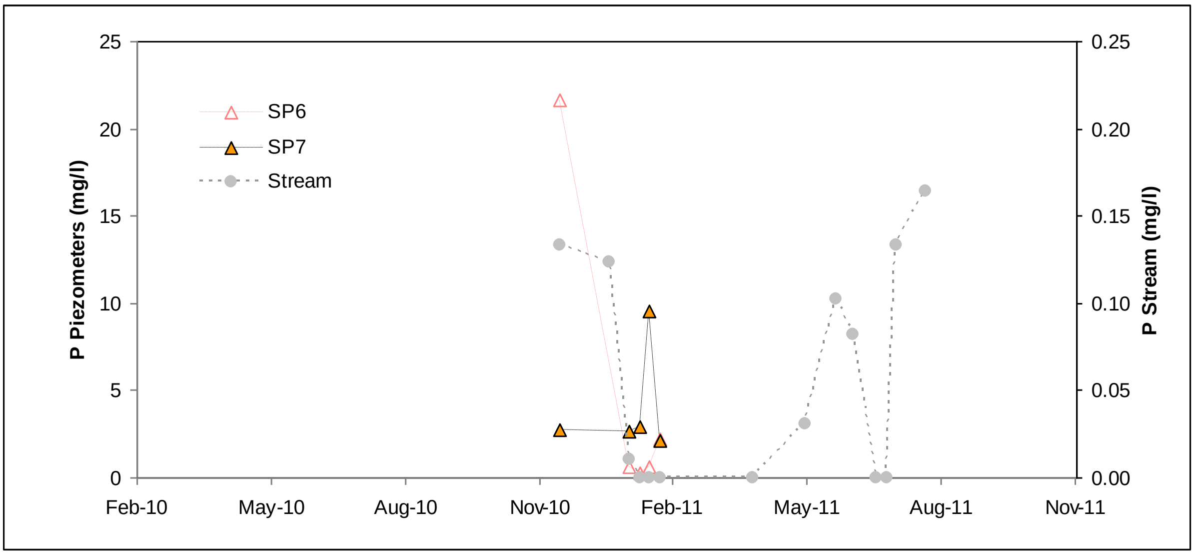


Figure 6.31 Observed phosphorus concentrations in piezometers and stream

Stable isotopes

Hydrogen and oxygen isotopes of water vary in time and space due to their fractionation in the natural environment. Isotope fractionation is accompanied by a number of processes, including phase change, transportation, diffusion, reduction, oxidation, chemical reaction, and biological metamorphism. Data from samples drawn from the boreholes, piezometers and the stream (ST), (Figure 6.32) clustered to the left of the global meteoric water line (GMWL) (Figure 6.33). The range indicated that they were predominantly of meteoric origin.

The borehole isotopes clustered in two intersecting zones, with isotopes from BH1 being generally more depleted than those of BH2 indicating different source and pathways of the groundwater in these two boreholes (Figure 6.33). The piezometer and stream water isotope values also clustered in the region of BH1, indicating a probable connection of these shallower sources with the stream. This highlights the importance of the near surface lateral flows in contributing solutes to the stream.

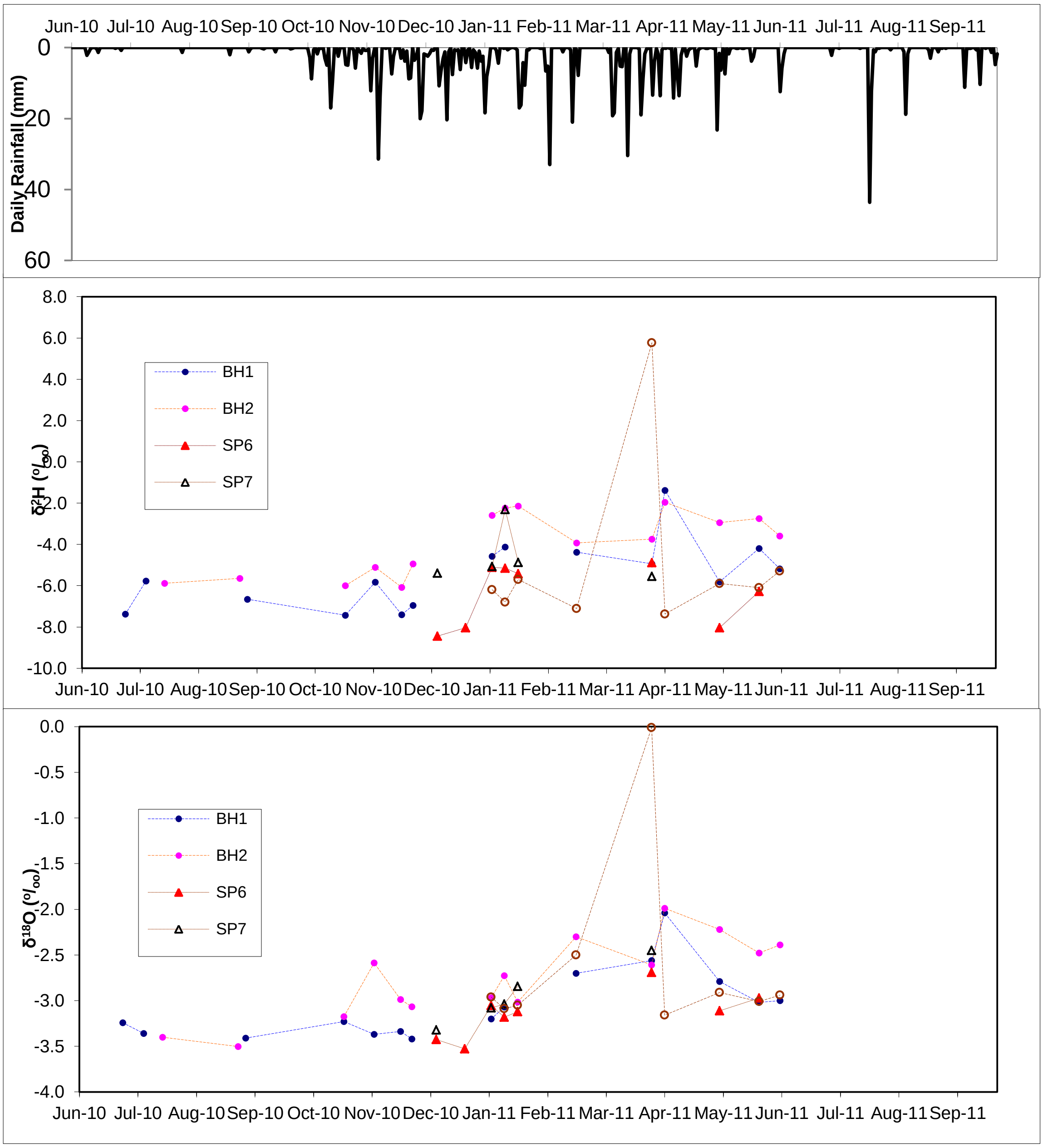


Figure 6.32. Stable isotope responses in the boreholes, piezometers and stream

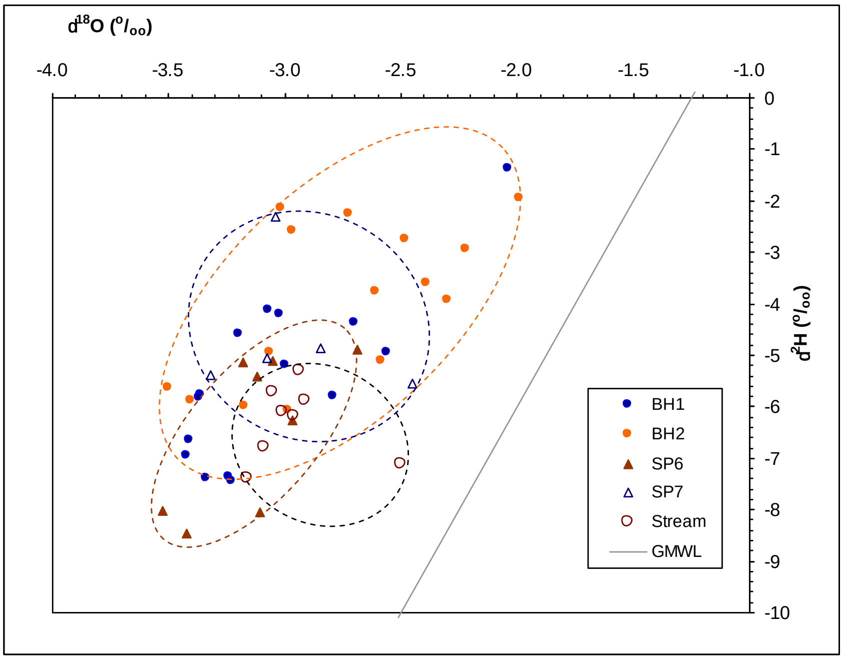


Figure 6.33. Stable isotope responses in the boreholes, piezometers and stream

6.4.2 Modelling

Simulation modelling of water and solute movement was performed at two scales. At the local scale the water balance of the trenches was simulated using the HYDRUS-2D model and at the hillslope scale, the mass loading from the near surface was estimated using Darcy's law for saturated flow.

6.4.2.1 Local scale modelling

Local scale observations at each trench type comprised soil water content dynamics; collection of infiltrated water and subsequent analysis for nitrates and phosphorus, as well as observations of accumulation of water and solutes at the soil/bedrock interface. The water content dynamic record was used to inform the water balance simulations, while the nitrate and phosphorus concentrations of water collected within the profiles was used to inform the simulation of nutrient fluxes.

As with the Umlazi site, the model HYDRUS-2D was used to simulate water flow and solute transport. A two dimensional simulation was done of the responses in each of the trenched treatments.

The material characteristics and the soil water and solute dynamics time series were analysed to develop parameters for simulation and calibration of the soil water and solutes. The soil hydraulic characteristics measurements were used to predict the van Genuchten parameters (α, n, L), using the Rosetta program. These predictions were modified during the Hydrus-2D modelling to accurately predict the variability of observed water content and nitrate and phosphorus concentrations in the soil. From these simulations the soil water fluxes of evaporation, transpiration, drainage and the solute migration fluxes were quantified for each site. A historic time series of met data was used to extend the simulations beyond the monitoring period in order to assess the impact of wet and dry periods as well as extreme events.

To run the simulation in HYDRUS 2-D, observation points were placed at points 0.15 m, 0.5 m and 1.2 m. These were the same positions where TDR probes have been placed for volumetric water content reading in a trench in each treatment. Wetting front detectors located at 0.5 m and 1.2 m also collected samples for nitrate and phosphorus analysis. The depths of soil layers in the trenches of each treatment were 0-30, 30-60, 60-90, 90-150 cm; but with variation inside the trenches of each treatment with the introduction of WWTP sludge of different depths. The boundary conditions used for each cross-section were surface (atmospheric), bottom (free drainage), right sides (seepage face) and left side (constant head) (Figure 6.34). The initial condition of nitrate and phosphorus transport at the beginning of the simulation (1 January 2010) for each cross-section were assigned using measured nitrate (357 mg/ℓ) and phosphorus (23 mg/ℓ) concentrations from samples collected from wetting front detectors installed in the sludge. The constructed finite element mesh with the selected boundary and initial conditions were used to simulate water flow, nitrate and phosphorus transport. In addition to precipitation, the potential evaporation and the potential transpiration were required for model application. Daily potential evapotranspiration rates were calculated with the ACRU model, using data from the nearby weather station.

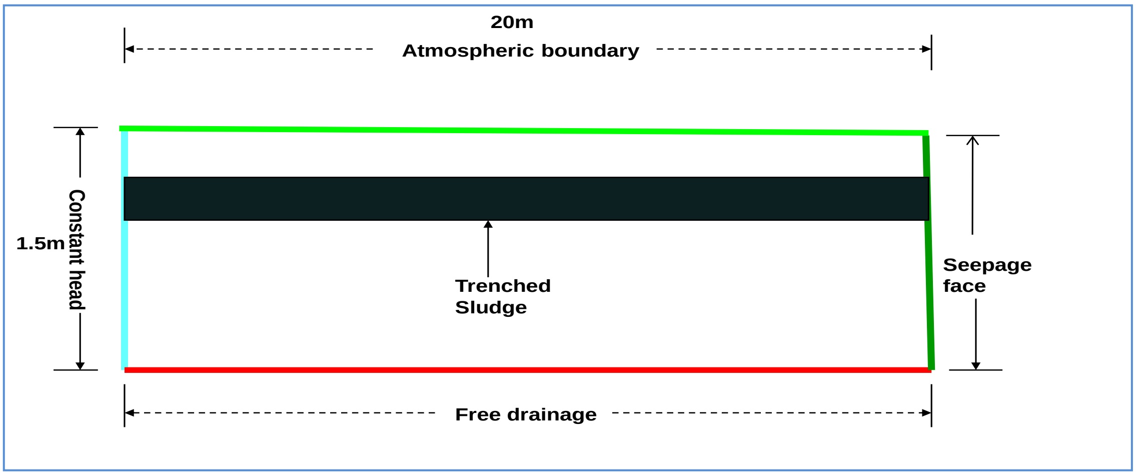


Figure 6.34 Conceptual model and boundary conditions

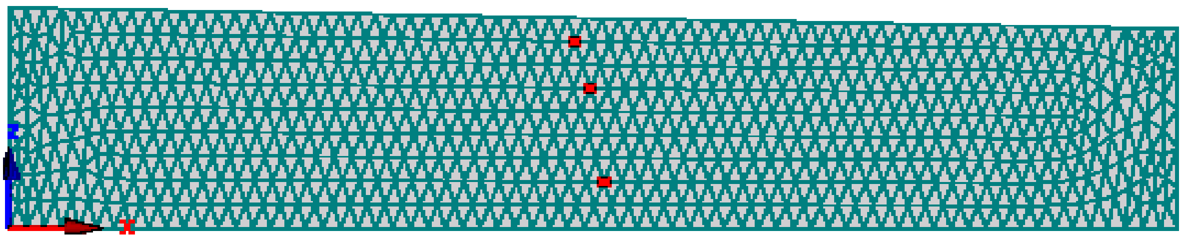


Figure 6.35 Locations of observation nodes, showing the finite element mesh

Volumetric water content readings from the TDRs were used as observed data (Figure 6.36). The different patterns of water content distributions between the treatments are thought to be mainly due to the amount of sewage sludge in the trench. Soil physical properties in the four treatments did not show significant variation. The results showed that simulated and observed water contents followed a similar trend. The correlation coefficient between observed and simulated water contents in all treatments (T) at the 3 depths (P1=0.15 m, P2=0.5 m and P3=1.2 m) varies from 0.0642 to 0.785 with the exception of T3P2 with a value of -0.793. Root mean square error (RMSE) between simulated and observed values was also estimated to examine the predictability of the model. RMSE values varied from 0.019 to 0.059. This indicates that HYDRUS-2D can be used to simulate the water distribution with satisfactory accuracy.

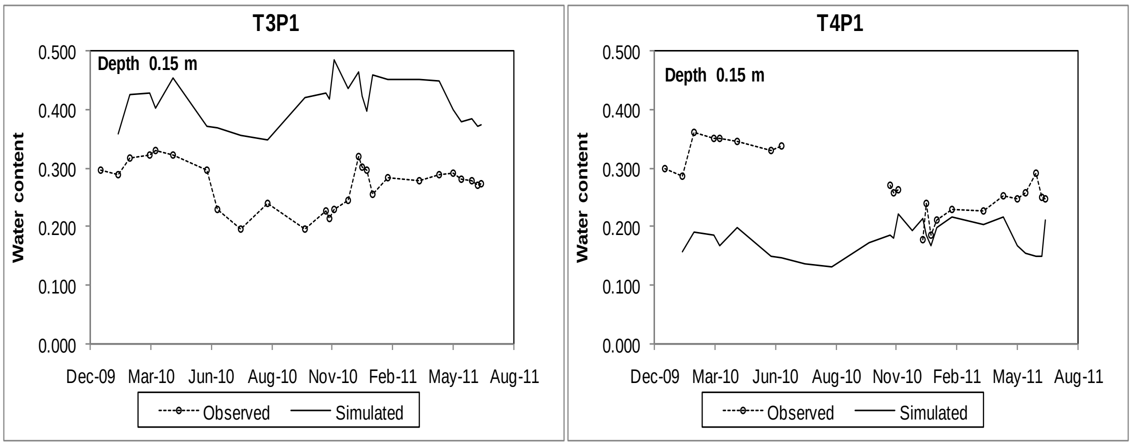


Figure 6.36 Observed and simulated water contents at 0.15 m for Treatment 3 (highest sludge loading, left) and Treatment 4 (no sludge, right)

Using dispersivity values adapted from Reneau *et al*., 2005, observed and simulated nitrate concentrations were compared for treatments 1, 2 and 3. To examine the predictability of the model, the simulation was carried out to predict the nitrate distribution using a simulation period of 638 days. Simulated and observed values of nitrate showed some variations in trend (Figures 6.37-6.42). The correlation coefficient between simulated and observed nitrate concentration varied from -0.602 to 0.959. The RMSE between simulated and observed varied from 3.84 to 74.19. The disparity might have occurred through biological or chemical changes following collection affecting the observed data. The collected samples were kept in cold storage until analysis, but further precautions may have been necessary. For example the samples could have been treated with mercuric chloride (HgCl2) to inhibit bacteria activity, applicable for nitrogen and phosphorus forms. In addition, the parameter selection and inherent model processes may require refinement.

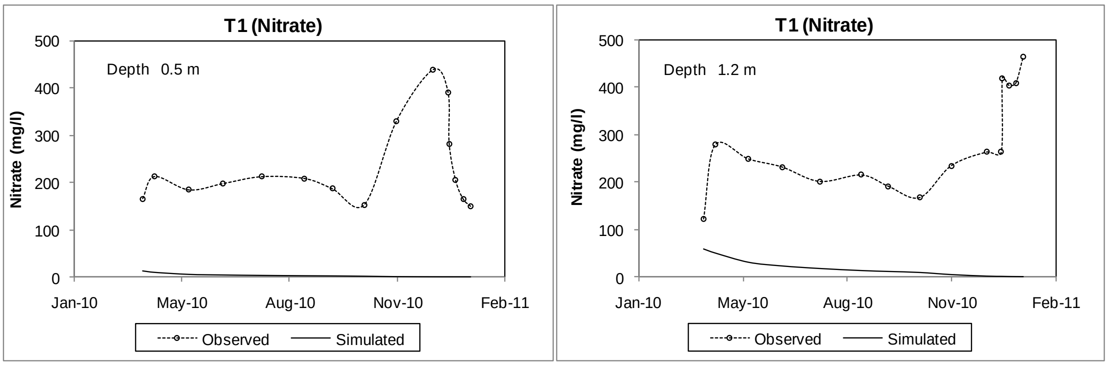


Figure 6.37 Observed and simulated nitrate in T1 at depths of 0.5 and 1.2 m

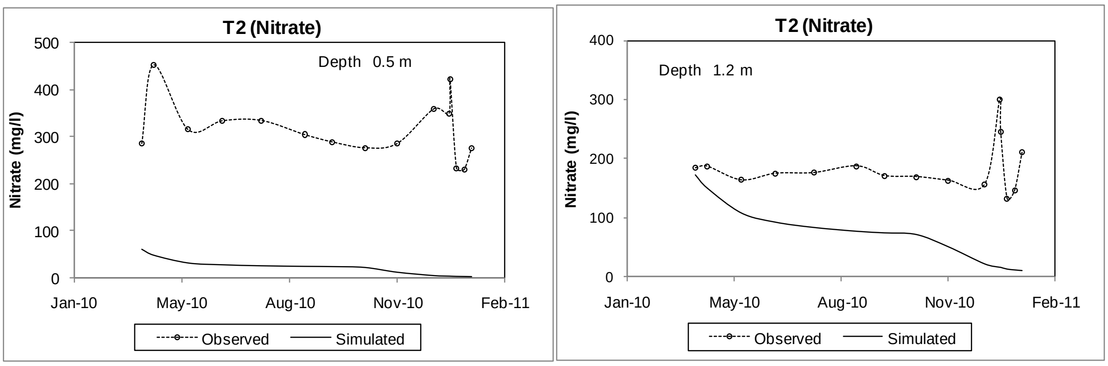


Figure 6.38 Observed and simulated nitrate in Treatment 2(T2) at depth 0.5 and 1.2 m

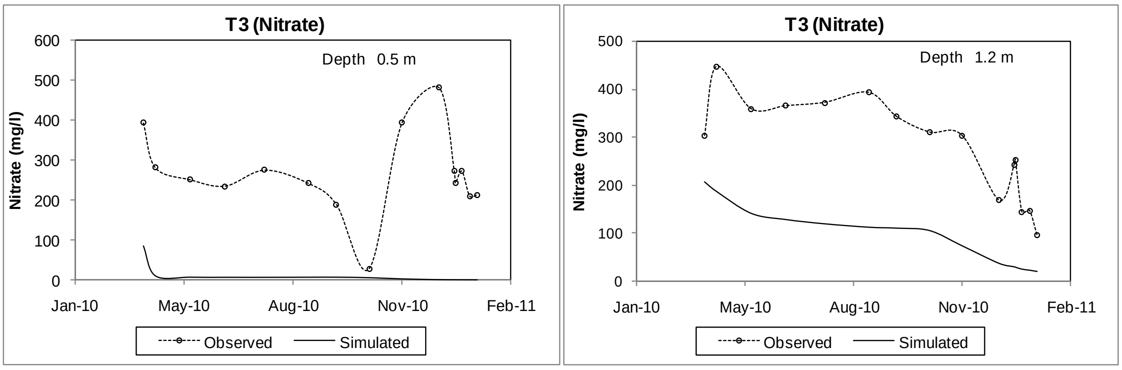


Figure 6.39 Observed and simulated nitrate in T3 at depths of 0.5 and 1.2 m

In all figures it is clear that the output of the HYDRUS model are close in some experimental results while some show significant differences. The differences between model and experiments may be as a result of the rapid movement of water through the soil without sufficient time for reaction between the soil and solute (Hillel, 1998). These differences may also occur as a result of the increase in error with depth due to preferential flow paths and/or the choice of incorrect adsorption isotherms.

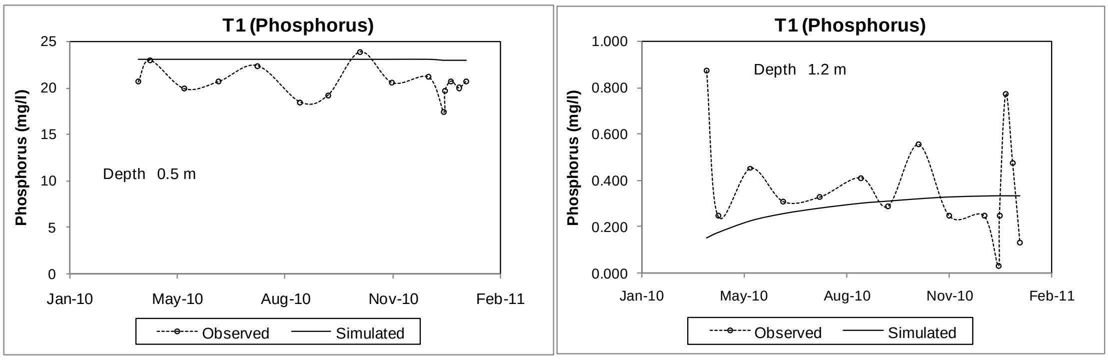


Figure 6.40 Observed and simulated phosphorus in T1 at depths of 0.5 and 1.2 m

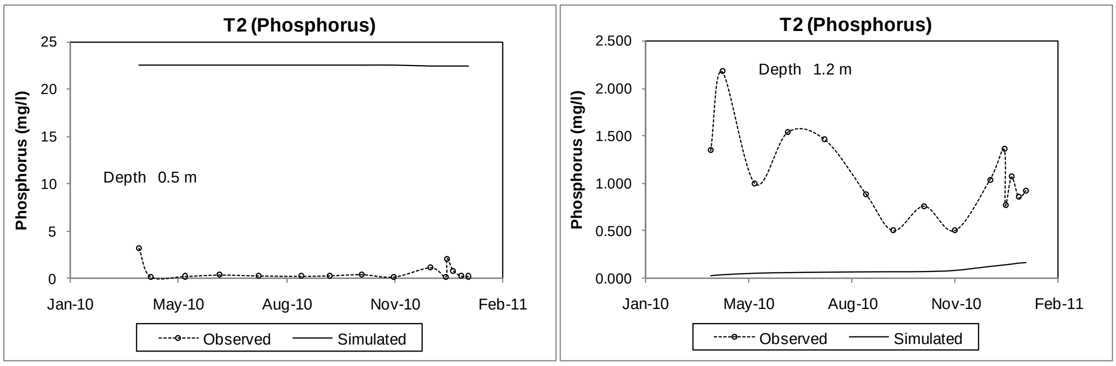


Figure 6.41 Observed and simulated phosphorus in T2 at depths of 0.5 and 1.2 m

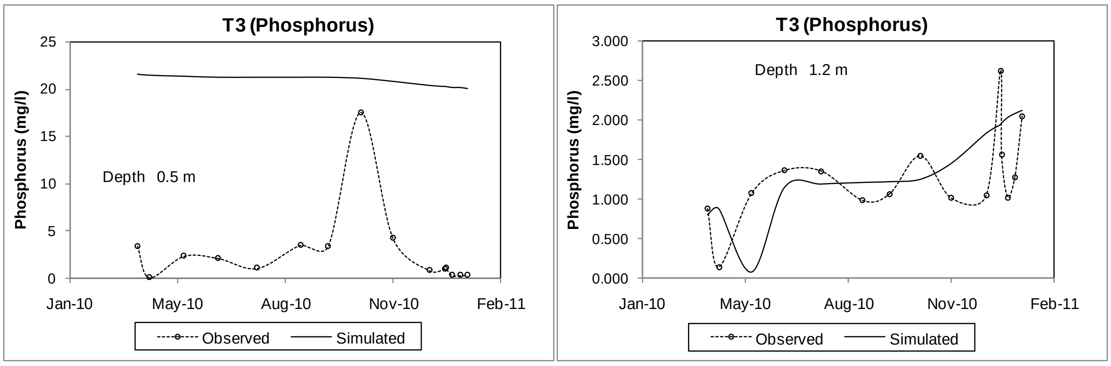


Figure 6.42 Observed and simulated phosphorus in T2 at depths of 0.5 and 1.2 m

6.4.2.2 Hillslope modelling

Hillslope scale observations included the accumulated lateral flow observed in piezometers at the downslope edge of the trial area. Borehole levels and concentrations were also used to interpret the likely pathways and contribution of nutrient to the fractured rock aquifer from the buried sludge. Nutrient concentrations measured in the stream were used as a baseline in assessing fractured rock groundwater impact.

These observations were used collectively to derive a sketch of the likely flowpaths and storage elements in the trial site (Figure 6.43). Water infiltrating into the profile was subject to evapotranspiration. During periods of high rainfall, subsurface water accumulated on the soil/bedrock interface, where it probably flowed laterally. These responses were rapid in most of the hillslope, but extended periods of perched water were observed at the lower end of the trial site. While some seepage water may have percolated into the fractured bedrock, no evidence of this was found in the borehole monitoring.

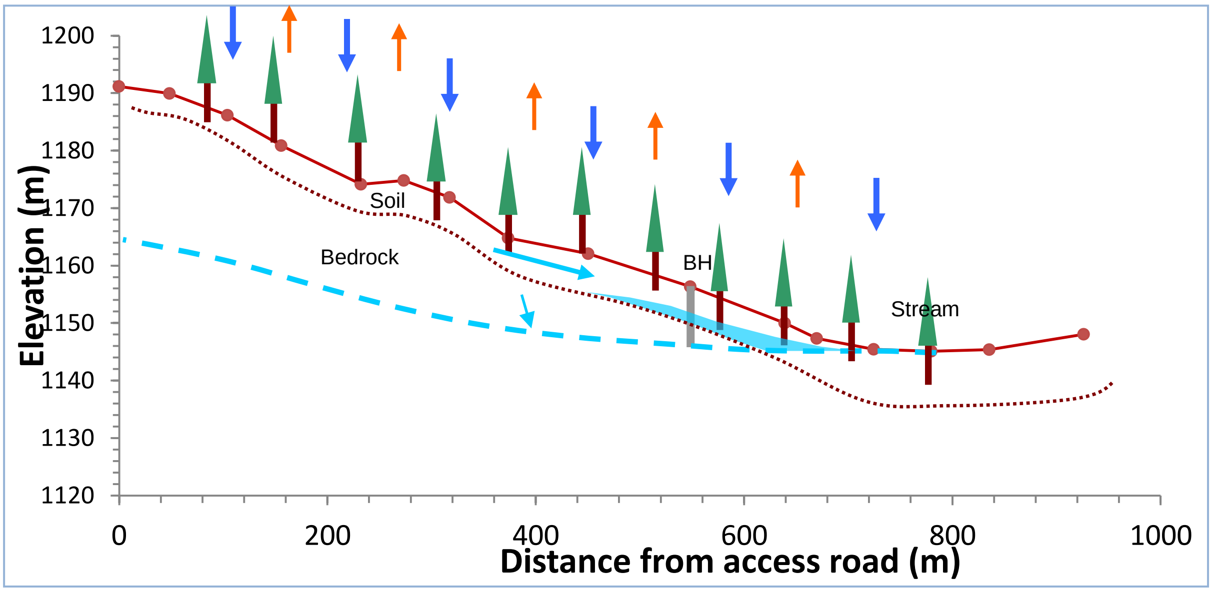


Figure 6.43 Sappi trial site schematic flow path, showing rainfall input (dark blue), evapotranspiration (orange), subsurface pathways and accumulation (light blue).

An estimate of the likely load discharging from the near-surface, lateral flow source has been estimated. The geophysics sections of the SP6 and SP7 piezometer locations were used, together with observed depths in the piezometers, to calculate a cross sectional area, Ai, of down slope discharge (Figure 6.43) for the prevailing depth of flow. The hydraulic gradient, ΔH/ΔL, of this near-surface discharge was taken as the surface slope, surveyed along the western edge of the trial (Figure 6.4) and the hydraulic conductivity of the material, Ksat, was measured using a Guelph permeameter. The discharge, Qi, for each time interval, Δti, between sampling events, was then calculated using Darcy's law as:

$$Qi = Ai.Ksat.\Delta H/\Delta L$$

The mass load, M, of NO_3 and P was determined for each sampling interval, i, by multiplying the respective concentration, Ci, with the discharge, Qi and the time interval. These were summed to derive the cumulative mass load during the sampling period as:

$$M = \sum_{i-1}^{n} Q_i C_i \Delta t_i$$

The mass loading in the stream was also estimated for the monitored period as a comparison of the mass contributed from the trial hillslope. The catchment area of the stream (Figure 6.45) was multiplied by the average monthly runoff factor and measured rainfall for the monitored months to determine an average discharge in the stream. This was multiplied by the measured concentrations of NO_3 or P and the time interval to yield a mass load. These loads were cumulated and are shown in Figure 6.46.

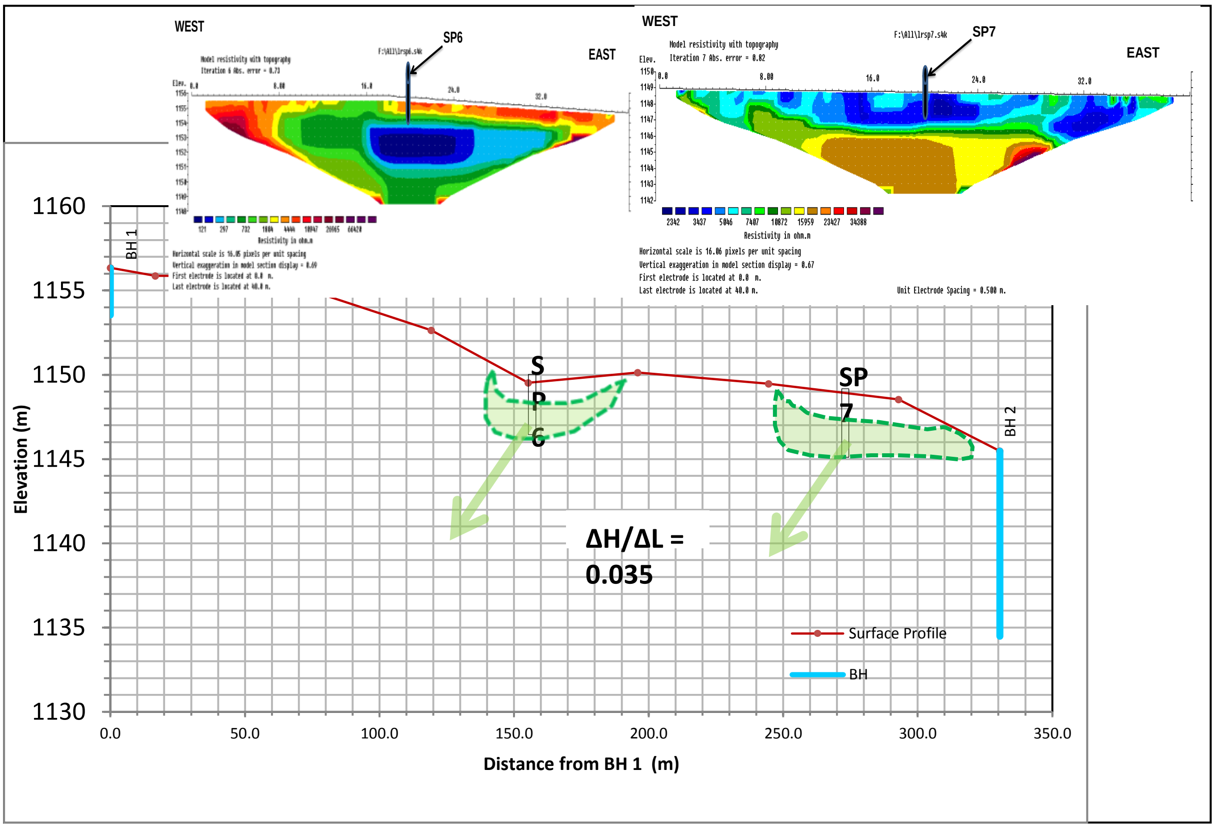


Figure 6.44 Illustration of the near surface preferential flowpath cross sectional areas at piezometers SP6 and SP7

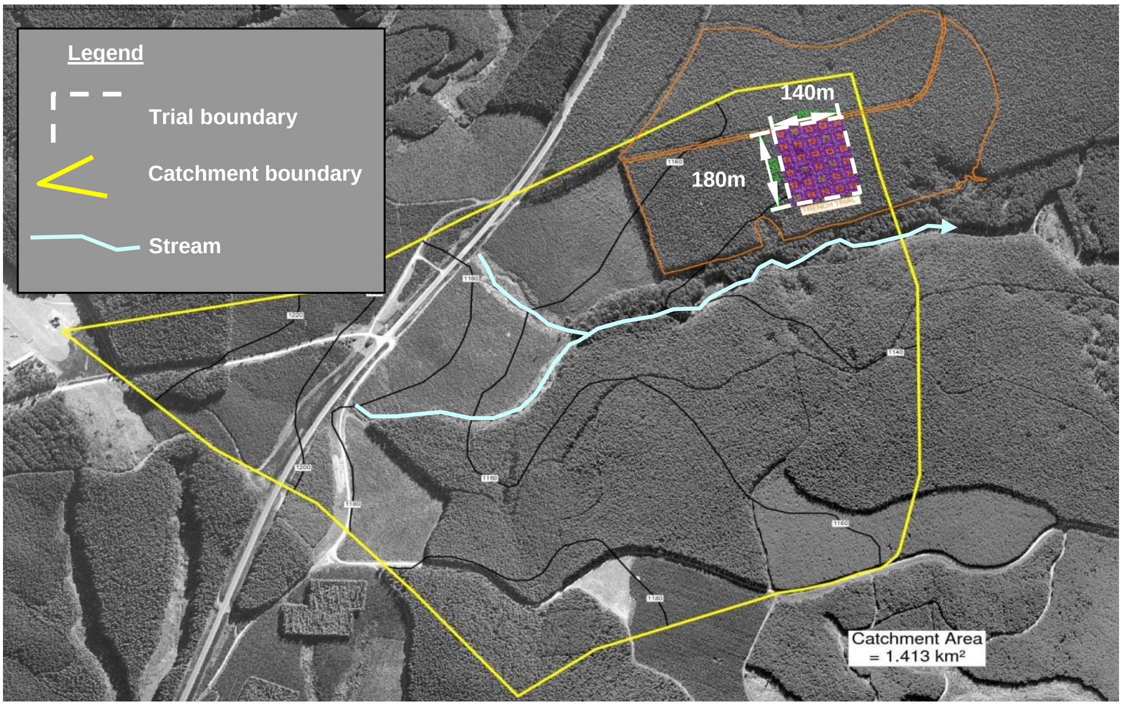


Figure 6.45 The catchment area of the stream on the southern edge of the trial.

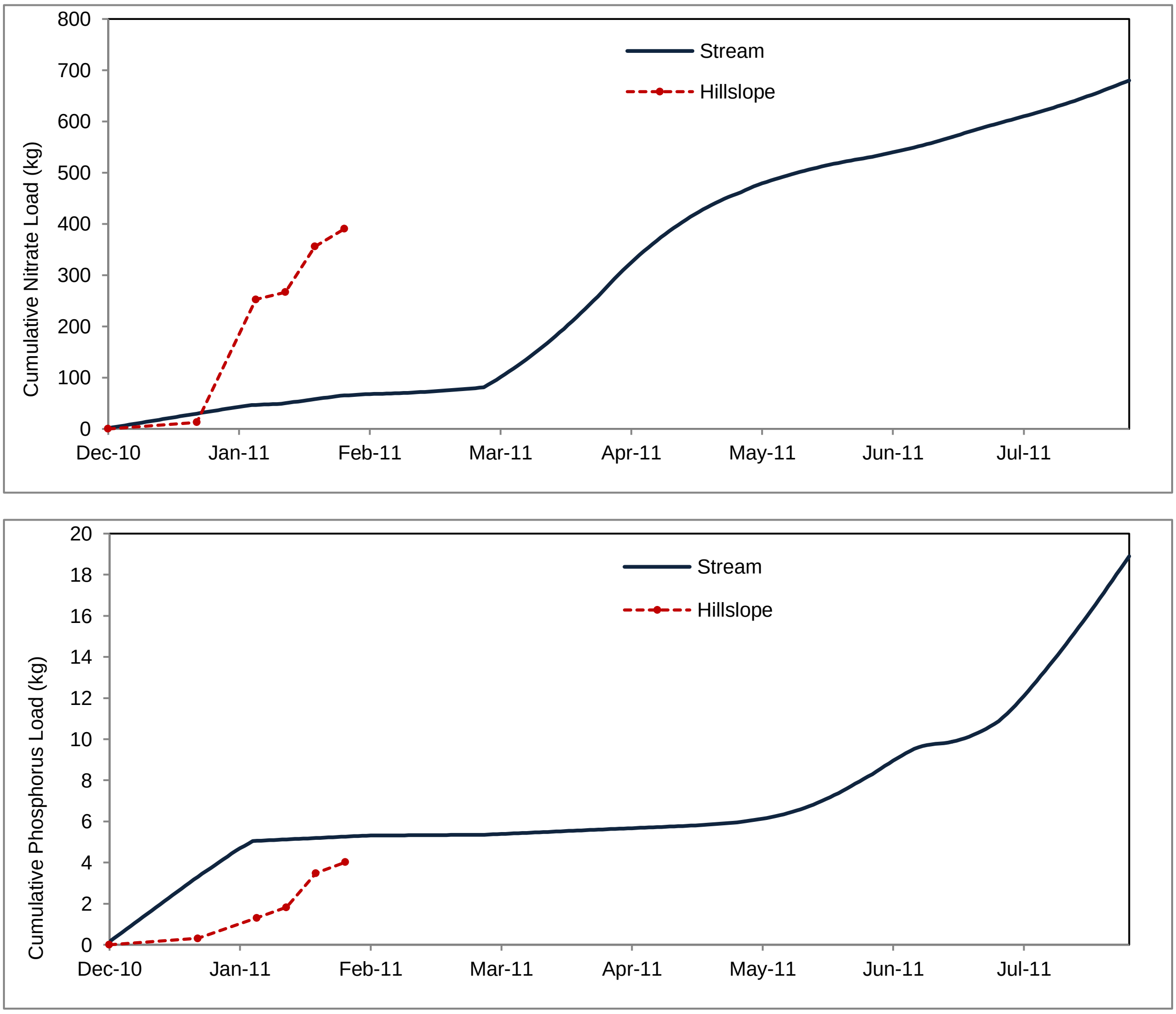


Figure 6.46 Cumulative loads of NO_3 (above) and P (below) from the near surface source and in the stream

The modelled mass loading of NO_3 from the hillslope exceeded that in the stream in the wet season of 2010/2011 (Figure 6.44). Modelled mass loading of P from the hillslope was similar to that in the stream for the monitored period. It was not possible to determine whether the mass loads in the stream were derived as a direct consequence of the discharge from the trial hillslope since the stream sampling did not include upstream (background) sampling. However, the potential loading from the hillslope was high, with a potential load of 400 kg NO_3 and 4 kg P estimated to discharge from the trial site. There was a buffer zone of open grassland between the trial site and the stream, and this could possibly take up a significant portion of the nutrient load from the trial.

TREATMENTS	
Treatment 1	250mm sludge
Treatment 2	500mm sludge
Treatment 3	750mm sludge
Treatment 4	Trench, no sludge
Treatment 5	No trench, no sludge

6.4.3 Tree growth and nutrients

The trees which had been treated with sludge were found to have, on average, a greater circumference than those which had not. At one year after entrenchment increased sludge loading showed a correlation with increased growth, with median stem diameter at 1.37 m for Treatment 3 15% greater than Treatment 2 and 26% greater than Treatment 1. Average median diameter for combined sludge treatments was 73% greater than for combined control treatments.

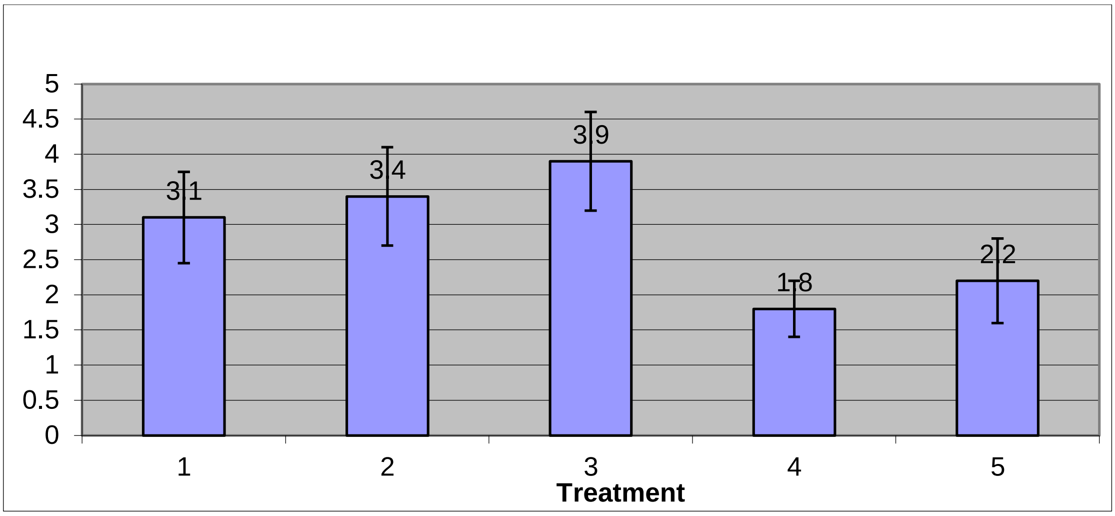


Figure 6.47 Median of stem diameters at 1.37 m (cm) – 7 January 2011 (1 year after planting)

By 1.25 years, however, trees planted on sludge (Treatments 1,2,3) continued to show increased growth over trees planted on no sludge (Treatments 4,5) but there was no longer a correlation between sludge loading and growth and the margin of benefit for trees planted on sludge had narrowed. Median stem diameter for trees in the combined sludge treatments was 51% greater than for trees in the combined control groups, but growth for Treatment 2 had overtaken Treatment 3 by 2%.

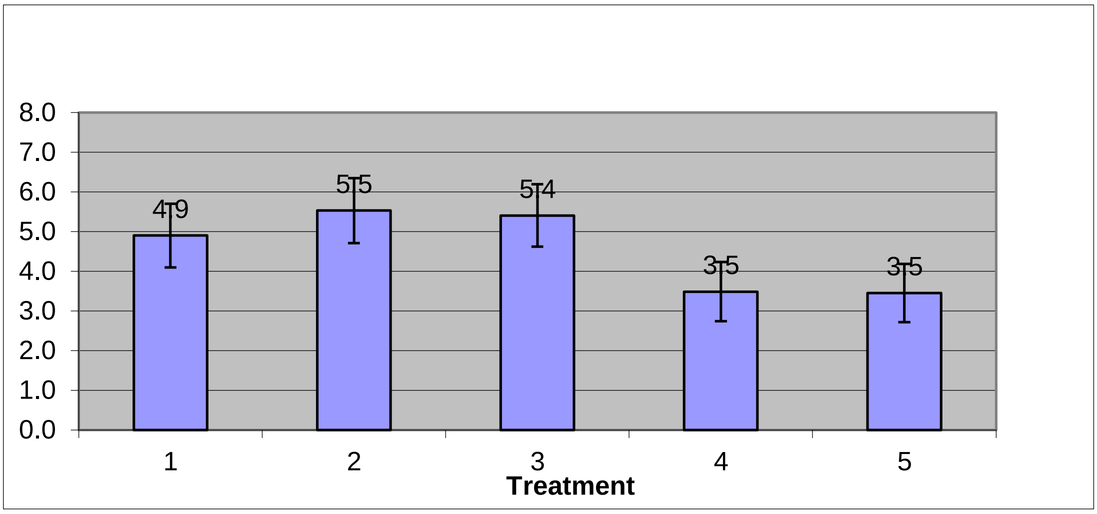


Figure 6.48 Median of stem diameters at 1.37m above base - 28 April 2011 (1.25 years after planting)

Final measurements taken for this study at 2.25 years after entrenchment showed continuing trends of greater growth for sludge treatments over no sludge treatments but no apparent benefit from higher sludge loadings. The average of median areas for sludge treatments (1, 2 and 3) had narrowed further to 39% greater than the average of median area for no sludge treatments (4 and 5). Among sludge treatments, however, the median area for Treatments 1 and 2 (lower sludge loadings) was 7% greater than that of Treatment 3 (highest sludge loading).

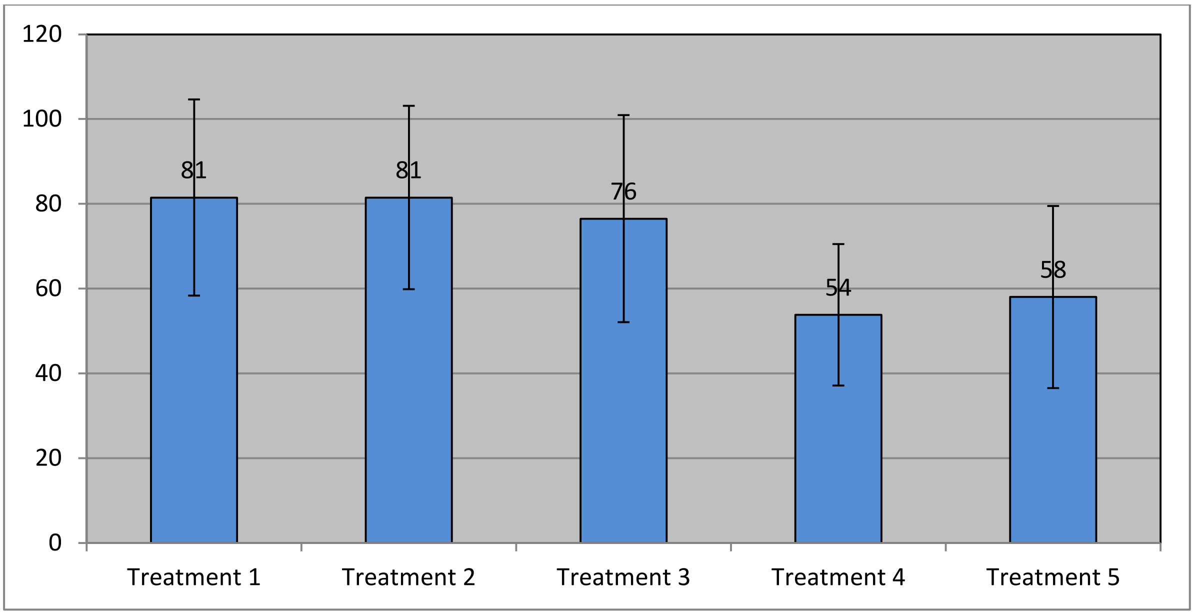


Figure 6.49 - Median tree areas for different treatment methods in cm at 20 March 2012 (cm) (2.25 years after planting)

In Figure 6.50 a comparison of median areas over time for all treatments shows the growth of trees in control treatments (T4 and T5) clearly lagging behind that of sludge treatments.

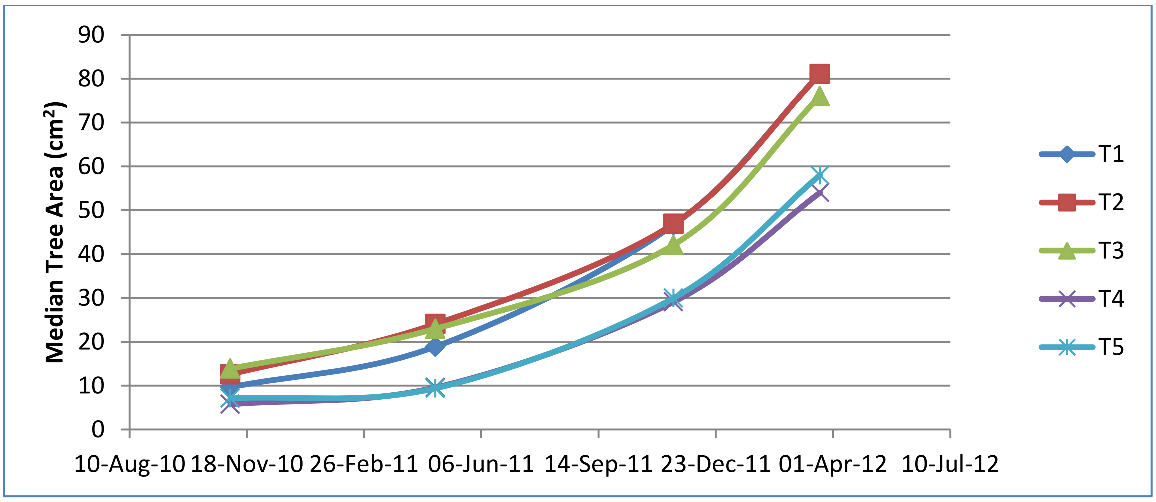


Figure 6.50 - Change in tree area with time

The heights of thirty trees from different treatment plots were measured. The relationship between tree height and circumference was found to be fairly linear (Figure 6.51).

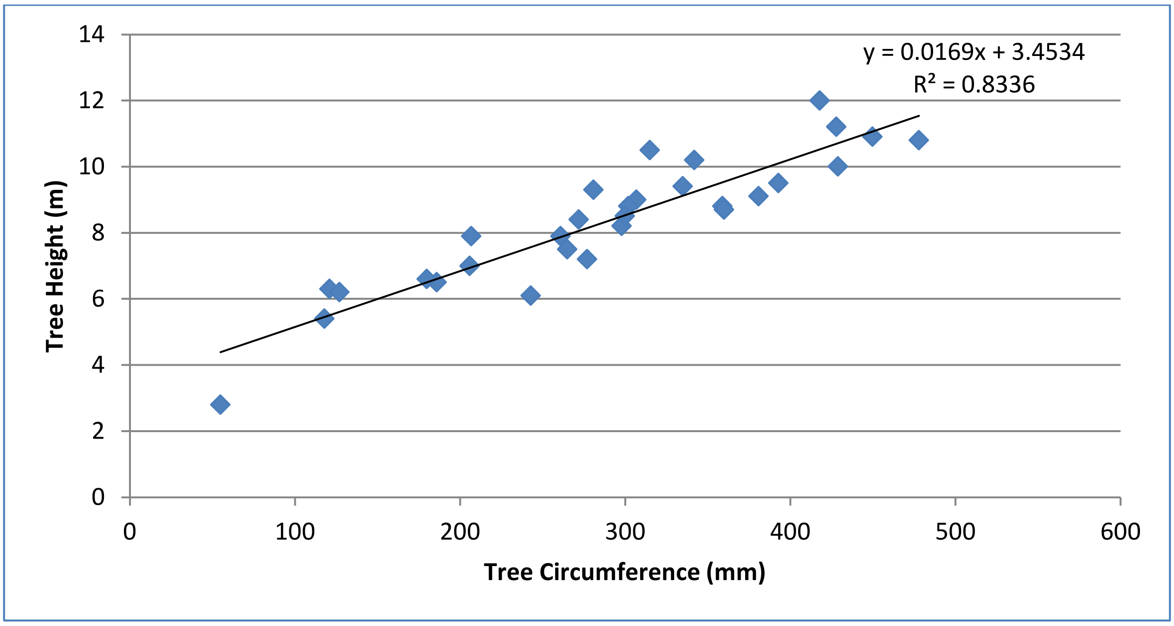


Figure 6.51 - Tree circumference vs height at 2.25 years

Foliar samples were taken and analysed for nutrient content at approximately two years after entrenchment. Median values for trees in blocks with sludge treatments (T1-T3) were significantly higher for zinc (Zn) and manganese (Mn) and marginally higher for nitrogen (N), potassium (K), copper (Cu) and phosphorus (P), with equal values for magnesium (Mg). Median values for trees in blocks with no sludge were higher for calcium (Ca), sodium (Na) and iron (Fe). Both showed high levels of aluminium (Al). Differences in moisture content between treatment blocks were insignificant.

Wood core samples were also taken and analysed for nutrient content at approximately two years after entrenchment. Samples were taken from all T3 blocks (highest sludge loading) and all T5 blocks (no sludge). Median values for trees in T3 blocks were significantly higher for zinc (Zn) and manganese (Mn), marginally higher for potassium (K) and equal for nitrogen (N). T5 blocks showed significantly higher values for trees grown on native soil for sodium (Na) and iron (Fe) with aluminium (Al) concentrations approximately double that of samples from trees grown on sludge. T5 treatments showed marginally higher values for calcium (Ca), magnesium (Mg), copper (Cu) and phosphorus (P).

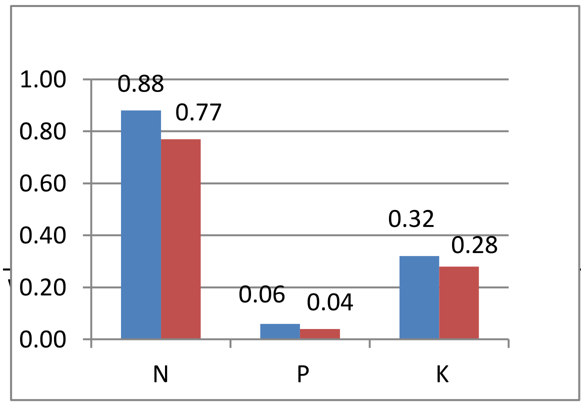
Table 6.1 Nutrient analysis for leaf samples from Treatment (T) blocks (B) with median values for sludge and no sludge treatments

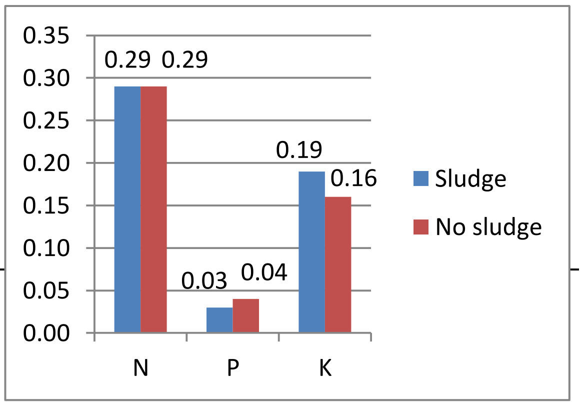
Sample	N	Ca	Mg	K	Na	Zn	Cu	Mn	Fe	P	Al	Moist
	%	%	%	%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	%	mg/kg	%
T1 – B1	0.96	0.28	0.10	0.31	671	7	2.4	391	55	0.06	66	67
T1 – B8	0.88	0.40	0.09	0.29	460	14	2.3	1056	69	0.05	110	63
T1 – B11	1.06	0.34	0.09	0.32	768	9	2.5	362	56	0.06	70	66
T2 – B15	0.88	0.46	0.11	0.26	660	10	2.4	711	65	0.05	103	64
T3 – B13	0.88	0.33	0.08	0.32	658	13	2.4	1041	55	0.07	96	63
T3 - B20	0.78	0.37	0.09	0.34	845	12	2.4	858	61	0.05	98	65
MED T1-T3	0.88	0.36	0.09	0.32	674	11	2.41	797	59	0.06	96	64
T4	0.68	0.52	0.08	0.25	657	4	1.7	528	53	0.04	94	60
T5	0.91	0.48	0.10	0.33	884	6	2.5	493	96	0.05	90	65
T5	0.80	0.51	0.08	0.30	714	6	2.3	443	59	0.04	80	63
T5	0.73	0.51	0.11	0.27	662	5	2.1	354	76	0.04	124	65
MED T4/T5	0.77	0.51	0.09	0.28	688	6	2.21	468	67	0.04	92	64

Table 6.2 Nutrient analysis for wood samples from Treatment (T) blocks (B) 3 and 5

Sample	N	Ca	Mg	K	Na	Zn	Cu	Mn	Fe	P	Al
	%	%	%	%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	%	mg/kg
T3 –B3	0.32	0.14	0.13	0.23	107.3	21	4.9	298	77	0.05	26
T3 –B5	0.31	0.10	0.11	0.19	276.6	17	3.2	132	98	0.03	23
T3 –B13	0.21	0.08	0.12	0.19	190.8	6	2.5	100	78	0.02	21
T3 –B20	0.31	0.10	0.13	0.22	169.2	13	3.0	192	49	0.04	25
T3 – B24	0.26	0.08	0.12	0.15	256.3	9	3.0	137	70	0.03	28
T3 – B30	0.28	0.10	0.12	0.15	236.3	11	2.6	84	43	0.02	21
MEDT3	0.29	0.10	0.12	0.19	213.5	11.71	2.98	134.3	73.86	0.03	24.39
T5 – B2	0.47	0.11	0.15	0.21	318.1	19	5.7	108	170	0.09	62
T5 – B9	0.28	0.15	0.16	0.14	189.9	8	3.4	87	101	0.04	46
T5 – B17	0.22	0.10	0.13	0.16	315.7	8	2.5	114	59	0.03	21
T5- B22	0.22	0.10	0.14	0.15	422.8	8	3.2	49	106	0.03	55
T5 – B26	0.30	0.12	0.12	0.16	206.2	8	2.9	58	87	0.04	45
T5 – B19	1.07	0.16	0.22	0.35	256.0	41	10.0	105	245	0.24	100
MED T5	0.29	0.12	0.14	0.16	285.8	8.45	3.27	95.51	103.8	0.04	50.69

The fact that the differences in foliar and wood concentrations of the key nutrients nitrogen (N),





phosphorus (P) and potassium (K) were insignificant suggests that at two years after entrenchment trees grown on sludge are utilizing key nutrients for enhanced growth rather than accumulating higher concentrations of these in plant tissues.

Figure 6.52 Median values for nitrogen, phosphorus and potassium in leaf samples from treatment blocks with sludge (T1-T3) and without sludge (T4-T5) (left) and from wood samples from treatment blocks T3 and T5 (right) (% dry matter basis)

6.4.4 Fate of pathogens

Limited investigation of the presence and fate of pathogens in the entrenched wastewater sludge was conducted in this study for the purpose of establishing values for comparison with the untreated pit latrine sludge entrenched for the Umlazi study. Further investigation of the fate of pathogens in entrenched wastewater sludge at the Sappi site is planned for a subsequent study designed to monitor the effects of entrenchment over the remainder of the growing cycle of the timber crop planted for this study.

At two years after entrenchment, three samples of wastewater sludge were collected from the wastewater treatment works which supplied the sludge for entrenchment to provide a representative baseline for sludge at the time of delivery to the trenches. Six samples were also collected from Treatment 3 blocks – three from Block 3 and three from B lock 20. In each case a single sample of 20g was subjected to the AMBIC protocol and recovered ova were enumerated and categorised. Potentially infective (undeveloped or immotile larvae) Ascaris eggs were found in each of the three samples of sludge which had not been entrenched, with numbers ranging from 8 to 39 per 20 gram sample. No helminth eggs were found in the 3 samples from Block 20, while potentially infective eggs were found in each of the three samples from Block 3 with numbers ranging from 1 to 20 eggs per 20 gram sample. While the lower numbers found in the entrenched sludge may correspond with the deactivation of eggs in trenches over time, the number of samples was too small to be significant.

Tables 6.3 and 6.4 show the helminth data per gram (Table 6.3) and per 20 gram sample (Table 6.4).

Table 6.3: Ranges of total number of Ascaris ova and numbers of potentially infective ova found in sludge samples of wastewater sludge (newly dried) and sludge that had been entrenched for two years (per gram)

Sample	Total no. of eggs /g	No. of potentially infective eggs/g
WWTP sludge	0.95-3.1	0.4-1.95
Treatment 3 block 3	0.05-3.85	0.05-1.45
Treatment 3 block 20	0	0

Taenia eggs were found in two of the three samples of unentrenched sludge and two of the three samples of entrenched sludge. As it is not possible to verify in the laboratory if the eggs are viable or not, they are considered potentially infective. Trichuris eggs were found in one of the three newly dried samples and two of the three entrenched samples.

One year later another set of three samples of dried sludge from the Howick WWTP was again analysed for Ascaris. This time the range for the samples was from 0.15 to 7 potentially infective eggs/g i.e. somewhat higher, but still in the same general ballpark.

Table 6.4 Numbers of helminth ova found in sludge samples of wastewater sludge (newly dried) and sludge that had been entrenched for two years

Identification Details	Sample Mass	Taenia	Trichuris	*Ascaris*-un-developed egg	*Ascaris*-immotile larva in egg	*Ascaris*-necrotic larva in egg	*Ascaris*-dead egg
WWTW 1	20 gm.	0	7	8	0	0	11
WWTW 2	20 gm.	5	0	19	0	0	18
WWTW 3	20 gm.	23	0	39	0	8	15
T3 Block 3 1 – Soil surface	20 gm.	0	0	0	1	0	0
T3 Block 3 2 – Soil/sludge interface	20 gm.	1	2	1	0	0	0
T3 Block 3 3 – 300 mm into sludge	20 gm.	26	16	9	20	31	17
T3 Block 20 1 soil surface	20 gm.	0	0	0	0	0	0
T3 Block 20 2 Soil/sludge interface	20 gm.	0	0	0	0	0	0
T3 Block 20 3 300 mm into sludge	20 gm.	0	0	0	0	0	0

6.5 Conclusions

After two years Eucalyptus trees grown with access to wastewater treatment works sludge showed significantly enhanced growth over trees grown without access to wastewater treatment sludge. The margin of benefit was narrowing with time, however, and higher sludge loading rates did not show significantly greater benefit to trees than lower sludge loading rates. The difference in nutrient levels in leaves and wood for trees treated with sludge and trees not treated with sludge was insignificant.

Table 6.5: Comparative increase in diameter and biomass over time at Sappi Shafton trial

Time after planting	Increase in diameter for trees planted over sludge vs controls	Increase in biomass (proportional to diameter^3) for trees planted over sludge vs controls
12 months	• 73%	• 420%
15 months	51%	• 240%
25 months	18%	64%

High concentrations of nutrients were evident in the water infiltrating into and through the sludge at all loading rates. The cumulative potential for mass loading of nutrients in the down slope pathway during the wet season and for extreme events has been estimated to be approximately 400 kg NO_3 and 4 kg P. The effects of vertical seepage of nutrients into the deep aquifer in fractured rock was not observed in the deep borehole, but the potential for seasonal fluctuations and sustained increases has not been determined.

Lower numbers of helminth eggs were found in sludge after two years of entrenchment than were found in sludge taken directly from the waste water treatment works. This may indicate deactivation of eggs over time under conditions of entrenchment.

7 CONCLUSIONS

❖ Tree growth

In all trials trees grown on or above VIP or wastewater sludge showed improved growth characteristics compared to a negative control, with the exception of wattle trees grown in the Umlazi study.

Eucalyptus grown on pit latrine sludge in the **Umlazi study** showed benefits over trees in control groups. The margin of difference appeared to peak at 1.5 years after planting with trees grown on sludge but then narrowed to 46% greater stem diameter at approximately 2 years after planting and 13% at 2.5 years. For wattle trees in this study, however, trees in both control and fertiliser rows demonstrated superior growth over trees grown on entrenched sludge, with the mean diameter of trees grown on native soil greater than that of trees planted with fertiliser. Because of the small sample size results were not statistically significant, however. In a controlled **tree tower study** in which wattle trees were grown on a core of pit latrine sludge, enhanced growth was noticed over trees grown without sludge, but the margin was significantly narrower than for eucalyptus trees, suggesting that control trees may have fixed nitrogen from the air.

For eucalyptus trees grown on wastewater treatment works sludge in the **Sappi trial**, trees planted on sludge showed a mean increase in diameter of 73% over trees grown without sludge at 1 year after planting. The difference appeared to narrow thereafter, with trees grown on sludge showing 51% greater diameter compared with trees not grown on sludge after 15 months. At the final measurement at 2.25 years after planting the difference in diameter between experimental trees and controls had reduced to 18%. As biomass is proportional to the cube of diameter, however, this still translated into a difference of 64% in total biomass. In addition, while trees grown on different sludge loadings at the Sappi trial initially showed greater growth for higher loadings, this effect also diminished rapidly and at the final measurement trees grown on the two lower loadings showed 7% greater growth than trees grown on the highest loading. While growth of eucalyptus was found to be enhanced significantly by access to sludge during the first 2-3 years after planting, it is unclear whether these trends will continue for the remainder of the growth cycle of these trees.

Eucalyptus grown on a core of pit latrine sludge in a controlled tree tower showed nearly three times greater nitrogen concentration and approximately two times phosphorus concentration in foliage over trees in the control group (which were given regular does of fertilizer), while eucalyptus grown on entrenched wastewater sludge showed marginally higher foliar concentrations of nitrogen and phosphorus over control trees.

❖ Changes in sludge

While data showed high variance, it was clear that rapid biodegradation and dewatering occurred in pit latrine sludge during the first year of entrenchment, after which the rate of degradation slowed. This may be due to the presence of fresher sludge at the top of the pit which had not yet degraded in the pit. Finals values reached under conditions of entrenchment were, however, lower than values achieved in the oldest sludge samples from pits, suggesting that soil fungi are able to degrade sludge more completely than is possible under conditions in the pit.

- **Impact on groundwater**

In addition to the differences relating to the characteristics of pit latrine sludge and wastewater treatment works sludge, the sites used for studies of pit latrine sludge (Umlazi) and wastewater treatment works sludge (Sappi) differed in terms of subsurface properties. The Umlazi trial was sited on a flat, alluvial, sandy deposit, with a shallow water table (2 m to 5 m), while the Sappi site was constructed in shales, overlying a dolerite sill on a relatively steep slope, with opportunities for near-surface, preferential flow which could intersect the waste trenches and carry nutrients downslope. Despite these differences, NO_3 and P concentrations generated within the trenches in the early phase of the study were found to be similar for both trials (Table 7.1). While pit latrine waste yielded higher P concentrations than WWTW sludge, concentrations observed in piezometers downslope of the deposits at both locations were similar. In addition, concentrations observed in receiving streams and nearby boreholes outside of the study sites were similar and no impact was found from the trenched waste over the period that was monitored. Water in the receiving streams do not appear to have been affected in excess of their seasonal concentration variations.

For entrenched pit latrine sludge, no excessive loading of nitrate or phosphorus was found beyond the entrenchment site. Simulations indicate a response time of 100 days between burial and observation of a nitrate plume at the borehole. However, the peak plume concentrations do not exceed background seasonal variations of nitrate; Occasional elevated concentrations of nitrate and phosphorus were observed in the borehole record, but the long term mass loading from the buried waste site seems unlikely to exacerbate eutrophic conditions.

For entrenched wastewater sludge, high concentrations of nutrients were evident in the water nutrients in the down slope pathway during the wet season and for extreme events was estimated to be approximately 400 kg NO_3 and 4 kg P. Nutrients were not found to have reached the deep aquifer, although the potential for this developing over time or occurring during seasonal fluctuations was not determined.

Table 7.1 Comparison of the nutrient concentrations between the Umlazi and Sappi site

Location	UMLAZI Pit Latrine Waste in Alluvium						KARKLOOF WWTW Sludge in Shale/Dolerite Hillslope					
	NO_3			P			NO_3			P		
	Min	Max	Med	Min	Max	Med	Min	Max	Med	Min	Max	Med
In trench	8	290	36	0.6	225.3	3.5	26	480	234	0.03	23.9	1
In piezometer	20	581	184	0.3	9.3	2.5	20	658	134	0.3	21.6	2.4
In borehole	0.04	35.6	0.6	0	0.83	0.15	0.4	13	5	0	0.17	0.03
In stream			3.1			0.17	0.4	10	1.3	0	0.16	0.02

- **Fate of pathogens**

Pathogen counts were significantly different between the WWTW sludge used at the Sappi trial and the pit latrine sludge used at the Umlazi trial. The average helminth egg load was approximately

1000 times greater in the pit latrine sludge samples than in the WWTW sludge. This difference may be ascribed to the different history of the sludges.

Sludge sampled from pit latrines contained high loads of infective *Ascaris* ova – (142 - 3937 eggs per gram wet weight with an average viability between 20 and 40%) - as well as quantities of *Taenia* and *Trichuris* ova. There was no indication that over time significant deactivation of ova took place in the latrine pits. Pit sludge, regardless of the length of time that it has been in a pit, should therefore be regarded as hazardous to the health of those working with the sludge.

Once pit sludge was buried in trenches associated with agro-forestry, however, a significant deactivation of helminth ova occurred. By 2 ½ to 3 years after entrenchment deactivation was sufficient to reduce potentially viable ova to very low concentrations. Of 145 907 ova counted which had been entrenched for more than two years, only 3 contained motile larvae, and these three were all in the same sample.

This indicates that the process of exhuming and burying pit latrine sludge in trenches, and then leaving it undisturbed for a period exceeding 2 years is sufficient to result in the deactivation of a significant proportion of helminth eggs. The exact length of time required for this to occur has not yet been established. However, from a microbial health risk assessment point of view, it is evident that the application of simple barriers, such as the use of gloves, hand washing and regular deworming would be adequate to mitigate any risk to workers exposed to sludge entrenched for more than 2 years. It is not yet clear what risks might arise as a result of accidental contact with entrenched sludge if it was dug up for any reason, although the data indicate that these risks would not be great.

8 RECOMMENDATIONS

Implement deep row entrenchment at the municipal level

Entrenchment of sludge is a relatively simple solution for sludge disposal which enables nutrients to retained and accessed over time. Municipalities can implement this method in a variety of different arrangements in order to achieve both disposal and beneficiation in a cost effective manner. Some scenarios are mapped out in the guidelines provided in Appendix A, along with recommendations for protecting public health and the environment when entrenchment is used.

Further research

While the impact of entrenchment on the environment is well enough understood to enable the method to be used safely and responsibly with appropriate monitoring, there are many aspects of the processes which occur during entrenchment which could be better understood with further study. This can happen both as municipalities implement this method and gather experience and information with time, as well as with continued research. It is recommended that the Umlazi and Sappi studies reported in this document are continued for the remainder of the growing cycles. In addition, lessons learned in this study could be incorporated into future research studies. These include:

- Plan the study to continue for the entire length of the growth cycle in order to observe trends over time. A number of factors could impact the release of nutrients or pollutants from the sludge and it cannot be assumed that benefits or negative impacts can be projected into the future on a linear path.
- Collect baseline data for native soil, groundwater and sludge before entrenchment in order to accurately track changes in soil, water, nutrients, pollutants and pathogens over time.
- Ensure that the transport and disposal of sludge is undertaken with careful planning to limit contact between workers and sludge and to ensure that sludge is deposited directly into trenches with no contamination of the surrounding area.
- Plant and measure a large enough sample of trees in each treatment to accommodate mortality over the course of the study due to disease or damage by humans or animals.
- Avoid planting species which fix nitrogen, in order to isolate benefits of nitrogen taken up exclusively from the soil.
- Ensure careful monitoring and documentation during the application of sludge or other treatments to ensure that trees and treatments documented on paper reflect the reality on the ground
- Number trees to be measured to ensure that individual trees can be tracked over time. This can allow errors to be picked up (eg. trees appearing to grow smaller) and phenomena to be investigated (eg. if a particular group of control trees are showing exceptional growth, the soil in that particular area can be examined).

Further work on the Deep Row Entrenchment of wastewater treatment sludge and on-site faecal sludges should address the following questions:

❖ **Tree growth**

- What comparative difference on tree growth, biomass and timber quality can be achieved using deep row entrenchment over the full growth cycle (8 to 10 years) ?

❖ **Changes in soil/sludge**

- Work out the long term fate of all nutrients added to the soil with the sludge
- Collect and analyse samples of entrenched sludge from different depths to determine decomposition rates over time
- The role of fungi in stabilisation of sludge in entrenchments could be further investigated through biochemical and microscopic techniques.

❖ **Fate of pathogens**

- Sampling for pathogens should include soil coring and piezometer water sampling.
- The HYDRUS model could be tested for simulation of pathogen migration and evaluated against migration in unsaturated and saturated soils in earlier studies.

❖ **Impact on groundwater**

- Continued monitoring of the deep groundwater through the boreholes and the near-surface lateral flows through the piezometers is recommended for at least two further seasons. This should be conducted in conjunction with upstream (background) and downstream sampling in the stream.
- Periodic gravimetric sampling of the profiles within and between the trenches to track the level of nutrients near the source.
- Continue to observe the rapid and highly concentrated mass loading from the near-surface.
- Investigate the remediation potential of planted trees by uptake of solutes, to minimize groundwater pollution. Three dimensional modelling could improve the simulation of preferential lateral discharges.
- Simulate plant water and nutrient uptake for understanding the total water and nutrient fluxes. With the potential for the use of a soil-water and nitrate movement model being

demonstrated, further research into the calibration and validation of the model has potential to improve the model performance.

❖ **Economics and application models**

- Explore and develop a range of models for cooperation between municipalities and the timber industry which can provide mutual benefit.

❖ **Safe practices**

- Understanding that sludge contains viable pathogens, develop procedures and protocols for entrenchment that ensure protection of humans and the environment during transport (eg. required signage for vehicles and protocols for spills) and entrenchment (storage of sludge on site, protective wear for workers, protocols to prevent spills and in the case of spills).

9 REFERENCES AND RESOURCES

ABEM (2010). Instruction Manual, ABEM Instrument AB. ABEM printed matter 93109, Hamngatan 27, S-17266 Sandbyberg, Sweden

Abbott L. and Murphy D (2007). Soil biological fertility: A key to sustainable land use in agriculture. Springer (Netherlands).

APHA (1998). Standard methods for the examination of water and wastewater. *20th Edition.*

Berg, G. and Berman, D. (1980). Destruction by anaerobic mesophilic and thermophilic digestion of viruses and indicator bacteria indigenous to domestic sludges. *Applied and Environmental Microbiology, 39(2), 361-368*.

Berry, C.R. and Marx, D.H. (1980) Significance of various soil amendments to borrow pit reclamation with loblolly pine and fescue. *Reclamamation and Revegetation Research 3:87-94*

Berry, C.R. (1986) *Use of municipal sewage sludge for improvement of forest sites in the Southeast* USDA [Online] Available: http://www.treesearch.fs.fed.us/pubs/1884

Bouillet, J.P., Laclau, J., Arnaud, M., M'Bou, A.T., Saint-Andre, L. and Jourdan, C. (2002). Changes with age in the spatial distribution of roots of *Eucalyptus* clone in Congo impact on water and nutrient uptake. *Forest Ecology and Management, 171, 43–57.*

Buckley C.A., Foxon, K.M., Brouckaert, C.J., Rodda, N., Nwaneri, C., Balboni, E., Couderc, A. and Magagna, D. (2008). Scientific Support for the Design and Operation of Ventilated Improved Pit Latrines (VIPs) and the Efficacy of Pit Latrine Additives. Water Research Commission. Report No. TT357/08.

Buswell, C. (2006) Fate And Transport of Nitrogen at a Deep Row Biosolids Application Hybrid Poplar Tree Farm. Masters Thesis, University of Maryland.

Campinhos, E. (1980). More wood of better quality intensive silviculture with rapid-growth improved Eucalyptus spp. for pulpwood. In *Proceedings, Tappi Annual Meeting Atlanta, GA. p. 351-357.*

da Silva, P.H.M., Poggiani, F. and Laclau, J.P. (2011). Applying sewage sludge to eucalyptus grandis plantations: effects on biomass production and nutrient cycling through litterfall. Applied and Environmental Soil Science, Hindawi Publishing Corporation.

Dahlin, T. and M.H. Loke, 1998. Resolution of 2D Wenner resistivity imaging as assessed by numerical modelling. *J. Applied Geophys*., 38: 237-249

De Boer, W., Folman, L.B., Summerbell, R.C. and Boddy, L. (2005). Living in a fungal world: impact of fungi on soil bacterial niche development. *Federation of European Microbiological Societies, 29:795-811.*

DeCocq, J., Gray, K. and Churchill, R. (1998) *Sewage sludge pelletization in Bostonne: Moving up the pollution prevention hierarchy.* National Pollution Prevention Center for Higher Education, Ann Arbor (USA).

Del Giorgio P.A. and Cole J.J (1998). Bacteria growth efficiency in natural aquatic systems. *Annual Review of Ecological system, 29:503-541.*

Department of Water Affairs and Forestry (1996). *National Sanitation Policy – White Paper* Available: www.dwaf.gov.za/Documents/

Department of Water Affairs and Forestry (2000). *Model Water Services Bylaws* Available: www.dwaf.gov.za/Documents/

Department of Water Affairs and Forestry (2001a). Draft guidelines for the removal and disposal of faecal waste from pit latrines and septic tanks.

Department of Water Affairs and Forestry (2001b). Study report on management of faecal waste from on-site sanitation systems in South Africa. DWAF Report no. C356

Department of Water Affairs and Forestry (2003). *Strategic Framework for Water Services* Available: www.dwaf.gov.za/Documents/

Departments of Agriculture, Health, and Water Affairs (1997). *Permissible Utilisation and Disposal of Sewage Sludge. Edition 1.* WRC Report No. TT85/97.

du Toit, B. (2002). Wattle Nutrition. *In: Dunlop, RW and MacLennan, (L) Black Wattle: The South African Research Experience, Chapter 6: p 47-64*. Institute for Commercial Forestry Research, Pietermaritzburg, South Africa.

Duncker, L.C., Matsebe, G.N. and Moilwa, N. (2007) The social/cultural acceptability of using human excreta (faeces and urine) for food production in rural settlements in South Africa. WRC Report No TT 310/07

Dons A. (1987). Hydrology and sediment regime of a pasture, native forest, and pine forest catchment in the central North Island, New Zealand. NZ *Journal of Forestry Science, 17,*(2/3): 161-78.

Duncan M.J. (1993). *Does planting pines reduce recharge?* New Zealand hydrological annual symposium. NIWA Freshwater, Christchurch (New Zealand).

Elliot, H.A. and O'Connor, G.A. (2007) Phosphorus management for sustainable biosolids recycling in the United States. *Soil Biology & Biochemistry 39: 1318–1327.* [Online] Available: www.elsevier.com/locate/soilbio

Foxon, K. (2007). Progress report (Jan 2006 to Jan 2007). WRC Project No. 1629.

Gee, G.W., and J.W. Baunder. 1979. Particle size analysis by hydrometer: a simplified method for textural analysis and sensitivity test of measured parameters. Soil Sci Soc. Am. J. 43: 1004-1007.

Gilmour, J.T., Cogger, C.G., Jacobs, L.W., Evanylo, G.K. and Sullivan, D.M. (2003). Decomposition and Plant-Available Nitrogen in Biosolids: Laboratory Studies, Field Studies, and Computer Simulation. *Journal of Environmental Quality 32:1498–1507*.

Glaser B. (2007). Prehistorically modified soils of central Amazonia: a model for sustainable agriculture in the twenty-first century. *Philosophical Transactions of the Royal Society B: Biological Sciences, 362(1478): 187-196.*

Griffin D.M. (1985). A comparison of the roles of bacterial and fungi. In: Bacteria in Nature (Leadbetter E.R. and Poindexerter J.S Eds), Bacterial Activities in Perspective, vol. 1, pp.221-255. Academic Press London.

Griffiths D.H. and Barker R.D.,1993. Two-dimensional resistivity imaging and modelling in areas of complex geology. Journal of Applied Geophysics, 29, 211-226.

Hawksworth, D.J., Archer, C., Appletonne, C.C., Rodda, N. and Smith, M. (2005). *The development of a method to recover Ascaris ova from urine diversion toilets*. Pollution Research Group. School of Biological and Conservation Sciences.

University of KwaZulu-Natal (Howard College Campus), Durban. WISA (2006). Biennial conference, Durban, South Africa.

Herselman J. E., Burger L. W. and Moodley P. (2009). Guidelines for the Utilisation and Disposal of Wastewater Sludge: Volume 5 of 5: Requirements for Thermal Sludge Management Practices and for Commercial Products Containing Sludge. Water Research Commission TT 351/09, Pretoria, South Africa.

Herselman J. E. (2009a). Technical Support Document to the Development of the South African Sludge Guidelines: Volume 4: Requirements for the Beneficial Use of Sludge at High Loading Rates. Water Research Commission K5/1622/2/09, Pretoria, South Africa.

Herselman J. E. (2009b). Technical Support Document to the Development of the South African Sludge Guidelines: Volume 5: Requirements for Thermal Sludge Management Practices and for Commercial Products Containing Sludge. Water Research Commission K5/1622/3/09, Pretoria, South Africa.

Herselman J. E. and Snyman H. G. (2009a). Guidelines for the Utilisation and Disposal of Wastewater Sludge: Volume 3 of 5: Requirements for the On-site and Off-site Disposal of Sludge. Water Research Commission TT 349/09, Pretoria, South Africa.

Herselman J. E. and Moodley, P. (2009b). Guidelines for the Utilisation and Disposal of Wastewater Sludge: Volume 4 of 5: Requirements for the Beneficial Use of Sludge at High Loading Rates. Water Research Commission TT 350/09, Pretoria, South Africa.

Herselman J. E. and Snyman H. G. (2009c). Technical Support Document to the Development of the South African Sludge Guidelines: Volume 3: Requirements for the On-site and Off-site Disposal of Sludge. Water Research Commission K5/1622/1/09, Pretoria, South Africa.

Hillel, D., 1988. Environmental Soil Physics. Academic Press, San Diego, CA, pp: 771.

Jennings, D.H. (1987). Translocation of solutes in fungi. *Biological Review, 62:215-243.*

Jimenez, B., Maya, C., Sanchez, E., Romero, A., Lira, L. and Barrios, J.A. 2002. Comparison of the Quantity and Quality of the Microbiological Content of Sludge in Countries with Low and High Content of Pathogens. Water Science and Technology 46(10):17-24

Jönsson, H., Richert, S.A., Vineras, B. And Salomon, E. (2004). *Guidelines on the use of Urine and Faeces in Crop Production*. EcosanRes Publication Series. Stockholm Environment Institute. Stockholm.

Kostyanovskiy, K.I., Lasley, K.G., Evanylo, G.K., Sukkariyah, B.F. Shang, C and Zheng, H. (2008). Paper presented at the 2008 National Meeting of the American Society of Mining and Reclamation, Richmond, VA..

Kays, J.S. (unknown date) *Use of deep-row biosolid applications to grow forest trees* University of Maryland Cooperative Extension [Online] Available: http://www.naturalresources.umd.edu.

Kays, J.S and Hammond, E. (unknown date) *Site preparatonne and the effect of subsoiling on survival and growth of hybrid .poplar.* University of Maryland Cooperative Extension [Online] Available: http://www.naturalresources.umd.edu.

Kays, J.S., Flamino, E., Feltonne, G. and Flamino, P.D. (1997). Use of deep-row biosolid applications togrow forest trees: a case study. In: The Forest Alternative: Principles and Practices of R Residual Use. *Preliminary Proceedings*. C.L. Henry (ed.). University of Washingtonne, Seattle, WA.

Kays, J.S., Feltonne, G. and Flamino, E. (1999) Deep-Row Application of Biosolids to Grow Forest Crops on Mine Spoils: Potential Utilization for the Baltimore, MD - Washingtonne, D.C. Metro Area. University of Maryland Cooperative Extension [Online] Available: http://www.naturalresources.umd.edu.

Kays, J. S., E. J. Flamino, G. Feltonne, & P. D. Flamino. (2000). Use of deep-row biosolids applications to grow forest trees: a case study. In C.L. Henry, R.B. Harrison, and R.K. Bastian (Eds.), The Forest Alternative: Principles and Practice of Residuals Use. (pp. 105-110). Seattle, WA: University of Washingtonne College of Forest Resources

Kays, J. S.; Feltonne, G. K.; Buswell, C. U.; Flamino, E. J. (2007) Deep Ros Incorporation of Biosolids to Grow Hybrid Poplar Trees on Sand and Gravel Mine Spoils in Southern Maryland. Water Practice, 1(1) 1-15.

Kelly, G. (2006) *Recycled Organics in Mine Site Rehabilitation: A review of the scientific literature* Department of Environment and Conservation NSW [Online] www.environment.nsw.gov.au

King County (2009). *Alternative uses and market opportunities for biosolids.* Report by King County, Washingtonne, USA.

Kirschbaum, M.U.F. and Tompkins, D. (1990). Photosynthetic responses to phosphorus nutrition in Eucalyptus grandis seedlings. *Australian Journal of Plant Physiology, 17, 527–535.*

Kirschbaum, M.U.F.; Bellingham, D.W. and Cromer, R.W. (1992). Growth analysis of the effects of phosphorus nutrition on seedlings of Eucalyptus grandis. *Australian Journal of Plant Physiology, 19, 55–66.*

Kollert, R (1969). Groundwater exploration by the electrical resistivity method. Geophysical memorandum ABEM printed matters (90019). Sweden.

Korentajer L. (1991). A review of the agricultural use of sewage sludge: benefits and potential hazards. Water SA, 17(3), 189-196.

Lambert, D.H., Weidensaul, T.C., Borger, D.C and Rhodes, L.H. (1985) *Use of Sewage Sludge for Forest-Tree Seedling Production.* EPA/600/S2-85/095.

Lazdiņa, D., Lazdiņš, A., Kariņš, Z. and Kāposts, V. (2007) Effect of sewage sludge fertilization in short-rotation willow plantations. Journal of Environmental Engineering And Landscape Management . 15: 2, 105–111.

Lorentz, S. 2001. Hydrological Systems Modelling Research Programme: Hydrological Processes. Phase I: Processes Definition and Database. Report to the Water Research Commission on the project: *Hydrological Systems Modelling Research Programme: Processes Research*. WRC Report 637/1/01.

Marais, G. v R. and Ekama, G.A. (1984). Fundamentals of Biological Behaviour. In *Theory, Design and Operation of Nutrient Removal Activated Sludge Processes*. Water Research Commission.

Marx, D.H., Berry, C.R. and Kormanik, P.P. (1995) *Application of Municipal Sewage Sludge to Forest and Degraded Land.* USDA-Forest Service Institute of Tree Root Biology Athens, Georgia [Online] Available: http://www.srs.fs.usda.gov/pubs/ja/ja_kormanik015.pdf.

Melin, G., Aronsson, P. and Hasselgren,K. (unknown date) *Recycling of wastewater and sludge in Salix plantations* [Online] Available: http://www.fao.org/forestry/media/10445/1/0.

Morgan, P. (2005) *Ecological Sanitation in Southern Africa: Many approaches to a varied need.* Proceedings of the 3rd International Ecological Sanitation Conference.

Murphy, K.O'H. (1997) Permissible utilisation and disposal of sewage sludge. WRC Report No. TT 85/97

Muse, J.K., Mitchell, C.C Jnr. and Mullins, G.L. (1991). Land Application of Sludge. Community Resource Development Publications, Auburn University, Auburn.

Nwaneri, C., Mkhize, S., Buckley, C.A. and Foxon, K.M. (2007) Research into the processes occurring inside the pit of Ventilated Improved Pit Latrines In Progress report of WRC Project K5/1630, Scientific support for VIP design and operation

Piterina, A.V., Bartlett, J. and Pembroke, TJ. (2010) Evaluation of the removal of indicator bacteria from domestic sludge processed by autothermal thermophilic aerobic digestion (ATAD). *International Journal of Environmental Research and Public Health*, 7, 3422-3441.

Polglase, P. and Crowe, M. (1999) *Sustainable use of biosolids (sewage sludge) in plantation forests* [Online] Available: http://www.csiro.au/science/PlantationBiosolidUse.html.

Reneau Jr, B.B., G. Hassan, C. Hagedorn, and M. Saluta. 2005. *On-site Waste Treatment and Disposal by Sequencing Batch Reactor: Drip Irrigation.* Crop and Soil Environmental Sciences Dept., Virginia Polytechnic Institute and State University, Blacksburg, VA, P 72

Rijnsburger, J., 1998, *Scale of wastewater treatment* In: A.. Balkema, H. Aalbers, E. Heijndermans (Eds.), *Workshop proceedings Sustainable municipal waste water treatment systems,* ETC/WASTE, Working Document, edition January 1998, WASTE, the Netherlands

Riley, J.P., Grasshoff, K. and Vipio, A. 1972. Nutrient Chemicals. In: Goldberg, E.D. (ed). A Guide to Marine Pollution. Gordon and Beach, New York. pp 81-110.

Robinson, M.B., Polglase, P.J. and Westonne, C.J. (2002) *Loss of mass and nitrogen from biosolids applied to a pine plantation* Australian Journal of Soil Research 40:6 1027-1039.

Romdhana, M.H. , Lecomte, D., Ladevie, D, and Sablayrolles, C. (2009). *Monitoring of pathogenic microorganisms contamination during heat drying process of sewage sludge*. Process Safety and Environmental Protection, 87, 377–386.

Roos, A. and Rosenqvist, H. (2000) A survey of short rotation willow growing in Sweden. *Proceedings of the International Forestry Conference, 8-9 August, 2000. Malaysia, p. 11.*

Sikora, L.J. (1978). Water quality at a sludge entrenchment site. *Groundwater 16(2): 96-104.*

Sikora, L. J., Burge, W. D. and Jones, J. E. (1982) Monitoring of a municipal sludge entrenchment site. *J Environ Qual 11:321-326.*

Šimůnek, M., Šejna, and M. Th. Van Genuchten. 1999. The HYDRUS-2D software package for simulating two-dimensional movement of water, heat, and multiple solutes in variably saturated media. Version 2.0.

Smernik, R.J., Oliver, I.W. and McLaughlin, M.J. (2004) Changes in the nature of sewage sludge organic matter during a twenty-one-month incubation *J. Environ. Qual. 33:1924–1929.*

Smith S.R. (1996). *Agricultural recycling of sewage sludge and the environment* [Review of international scientific literature on the potential impacts of recycling sewage sludge on farmland with almost 1000 references.]

Snyman, H.G. and Herselman, J.E. (2006a). Guidelines for the utilisation and disposal of wastewater sludge Volume 1: Selection of management options WRC Report No. TT 261/06

Snyman, H.G. and Herselman, J.E. (2006b) Guidelines for the utilisation and disposal of wastewater sludge Volume 2: Requirements for the agricultural use of wastewater sludge. WRC Report No. TT 262/06

Taylor, C (2011). An Investigation into the Potential of Faecal sludge for Plant Production, submitted in fulfilment of the academic requirements for the degree of Master of Science in the School of Biological Conservation Sciences, University of KwaZulu-Natal Durban.

Tesfamariam, E.H., J.G. Annandale, J.M. Steyn, and R. J. Stirzaker. (2009) Exporting Large Volumes of Municipal Sewage Sludge through Turfgrass Sod Production. *Journal of Environmental Quality* (38):1320-1328.

Thomas-Chery, A.L., Carnus, J.M., Denais, L. and Benbrahim, M. (2007) *The experimental network ERESFOR: Sewage sludge applications on forest lands* INRA, France [Online] Available: www.bordeaux.inra.fr/eresfor

Toffey, W., Flamino, E., Pepperman, R., Grous, A., Drumheller, A. and Garvey, D.(2007) *Demonstrating Deep Row placement of biosolids in coal mine land reclamation* [Online] Available: http://www.clu-in.org/conf/tio/ecoresources3/resources/Toffey-2007-Hybrid-Poplar-Results.pdf.

van Lier, J.B., Zeeman, G., van Buuren, J.C.L. and Lettinga, G. (1999) Decentralised sanitation concepts: perspectives for reclamation of domestic sewage for agricultural production by using low-cost treatment technologies. EP & RC Foundation, Wageningen, The Netherlands.

van Ryneveld, M.B. & Fourie, A.B. (1997). A strategy for evaluating the environmental impact of on-site sanitation systems. *Water SA, 23(4).*

Walker, J.M. (1975). *Trench incorporation of sewage sludge.* From: Proceedings of the National Conference on Municipal Sludge Management, Allegheny County, PA. Information Trans., Inc. Washingtonne, D.C.

WRC, (1997). Permissible Utilization and Disposal of Sewage Sludge. WRC report TT 85/97. Water Research Commission, Pretoria.

APPENDIX A

GUIDELINES FOR MUNICIPAL ENTRENCHMENT OF PIT LATRINE AND WASTEWATER SLUDGE

While further research of the benefits and risks associated with deep row entrenchment is warranted to better understand the interaction of entrenched sludge with the environment and with the species of tree or other crop it has been entrenched to benefit, municipalities can begin to implement this method with proper management and monitoring.

The South African *Guidelines for the Utilisation and Disposal of Wastewater Sludge* provide a matrix for the development of new options for wastewater sludge management; however, deep row entrenchment is not addressed specifically . In addition, the existing guidelines have been developed exclusively for sludge which has been treated at a waste water treatment works (WWTW) and the implications for VIP sludge are not addressed. While two of the applications addressed in these guidelines can be used for either VIP sludge or WWTW sludge, it is important to keep in mind that separate analyses of these different sludges should be made before application as they differ in several key respects:

- While VIP sludge can be expected to have a low pollutant content because it originates exclusively from domestic sources,[1] it is less stable than WWTW sludge and water soluble elements have not yet been removed, with the result that greater movement of pollutants through the soil and water table might be seen[2].
- Because WWTW sludge originates from multiple domestic, commercial and industrial sources where pollutant disposal may fluctuate widely over time and from one treatment works to another, the pollutant level of WWTW sludge can never be assumed and must always be assessed, along with the receiving soil, before application, to ensure that the combined loading of pollutants at the entrenchment site will not pose a threat to the environment.
- While processes at the WWTW may have reduced the presence of pathogens in WWTW sludge significantly, both WWTW sludge and VIP sludge can be assumed to have viable pathogens and should be treated as hazardous materials. However, because the odour of WWTW sludge has been reduced significantly through stabilisation, it will not attract vectors to the extent that VIP sludge would. In addition, as the smell of WWTW sludge is less offensive it can be applied at closer proximity to settlements than VIP sludge without creating a nuisance. Sludge from pits which have not been disturbed for some time may also have stabilised to the point that it has no odour.

[1] This assumption could, however be questioned on the basis that the pit often has to accommodate the solid waste of the household if solid waste collection is not provided, and waste associated with home businesses, could potentially contain significant pollutants.
[2] Heidi Snyman, pers. comm.

Volume 4 of the *Guidelines* addresses two types of applications of wastewater sludge at higher than agronomic rates: once-off application of less than three times in a five year period, or continuous sludge application, a method which requires less stringent planning and monitoring, which is more than three times in a five year period. With deep row entrenchment, sludge is normally applied on a crop planting/harvest cycle which is typically longer than five years, and thus could be categorised as "once-off" application. However, entrenching sludge in deep rows allows sludge to be applied at much higher rates, making it necessary to consider some of the implications addressed in the guidelines for "continuous sludge application."

The guidelines presented in this chapter are intended to provide a practical framework for entrenchment of both wastewater and pit latrine sludge which adequately protects the environment and public health while avoiding excessive measures which could prove prohibitive to the implementation of this method both in terms of costs and time. Volume 4 of the *Guidelines* forms the basis of many of the recommendations, in addition to the experience gained with entrenchment in the trials conducted in this study as well as earlier trials conducted abroad. While these recommendations may prove over time to be overly conservative, they are intended to ensure that all variables are considered. As more experience is gained with this disposal method in practice, it is expected that these guidelines will continue to be revised until they reflect a solid understanding of the interactions between entrenched sludge and the surrounding environment.

The following guidelines provide options for municipal application of the deep row entrenchment method for both VIP sludge and treatment works sludge:

❖ *Commercial entrenchment (pit latrine sludge or treatment works sludge)*

This would involve a larger scale partnership between a municipal waste water treatment works and a forestry company in which the sludge generated within a municipality is transferred to a forestry company through some mutually beneficial arrangement. Alternatively, the municipality might contract a forestry company to manage a timber crop on entrenched sludge on municipal land.

❖ *Municipal decentralised entrenchment (pit latrine sludge or treatment works sludge)*

This option involves entrenchment of sludge on relatively small tracts of municipal land, particularly in rural areas where utilising sludge close to its point of origin on smaller plots represents a significant savings over transporting sludge long distances for more centralised use/disposal. Trees or other non-edible crops would be grown on the sludge.

❖ *Municipal burial sludge on household premises (pit latrine sludge)*

This option provides recommendations for situations where there is space at the site where a pit is emptied for the sludge to be entrenched on site. The entrenchment of the sludge provides a protective barrier over the pathogens, and the planting of trees provides a means to derive some benefit from the nutrients in the sludge.

1. Commercial deep row entrenchment for forestry (PIT LATRINE / WASTEWATER SLUDGE)

Both VIP and WWTW secondary sludge can be entrenched within an existing forestry site, reducing the costs and problems associated with other disposal options and yielding an economic benefit in terms of greater biomass of timber produced. While fruit trees and food crops with edible parts grown above the soil could benefit from the nutrient supply provided by entrenched sludge, the possibility of the covering layer of soil being disturbed by heavy rains or activities and thereby exposing pathogens in the sludge makes it an inappropriate option for commercial production of food. Class ***A*** sludge which has been treated to destroy pathogens may however be used to grow food commercially either through surface application, shallow incorporation or deep row application.

1.1 Legal requirements

The sludge producer and sludge user must have a legal agreement/contract if the sludge is utilised by a third party. [3] Formal application must be made to the competent authority for authorisation of the proposed activity at the proposed site. Details for this process can be found in the Environmental Impact Assessment Regulations (GN No. R. 660 of 30 July 2010). The applicant must appoint an Environmental Assessment Practitioner (EAP) to prepare the necessary basic assessment, scoping report or EIA, including handling the public participation process required for obtaining authorisation. A rejected application can be appealed; in addition, it is possible to apply for exemption from the process.

The costs and benefits to stakeholders in the local community must be assessed prior to commencing the activity, in addition to needs and rights. If an environmental impact assessment is required a public participation process must be undertaken.

Once the applicable permits and authorisations have been granted, management and monitoring of the entrenchment site becomes self-regulatory. DWA will provide a list of the records that must be kept by the sludge producer and disposal site owner/operator.

1.2 Financial viability

In order to determine whether deep row entrenchment of sludge for commercial forestry is a viable option for a municipality, a cost benefit analysis should be conducted to determine:

[3] The conditions which must be included are provided in the Guidelines for the utilisation and disposal of wastewater sludge, Volume 4, Appendix 3.

- The current expenditure by the municipality sludge disposal versus anticipated expenditure for disposal of sludge by entrenchment. Can expenses be reduced by simplifying or bypassing some stages of treatment at the works which are not essential for sludge destined for entrenchment? The costs of testing sludge for pollutant levels before disposal must be added in.
- The current expenditure on transport to the disposal site versus anticipated expenditure for transport to a deep row entrenchment site (including worker and public safety measures and measures to address contamination of vehicles if sludge is currently not transported off site)
- Income, if any, generated by existing disposal alternatives versus anticipated income generated by deep row entrenchment
- Period for which this cost benefit analysis can be expected to apply: Can the designated forestry site accommodate all the sludge generated at the works or will it reach capacity after a certain number of years? How often can sludge be entrenched or will it be stockpiled at the works during the growing cycle?

Similarly, a cost benefit analysis should be conducted by the forestry company to determine:

- Increased / decreased costs of obtaining, developing and operating a site for deep row entrenchment due to more stringent environmental impacts (limitations on site selection, preparation of trenches, entrenchment equipment, monitoring, savings on fertiliser, protective equipment and training for personnel)
- Cost of desired quantity of sludge purchased from municipality
- Anticipated increased savings/profits resulting from entrenchment (increased biomass production)

If the sludge is to be used for the rehabilitation of mine spoils or restoration of habitat, a similar cost benefit analysis can be conducted to determine the viability of deep row entrenchment in contrast with alternatives.

1.3 Site characterisation and selection

A site with any of the following characteristics should not be considered for deep row entrenchment unless mitigation measures can be taken to alleviate any potentially negative impacts:

- Within the 1 in 100 year flood line (wetlands, vleis, pans and flood plains) due to risk of water pollution.
- Unstable areas (fault zones, seismic zones and dolomitic or karst areas)
- Steep gradients(greater than 15 degree) due to potential for instability and erosion, and greater movement of pollutants
- Distance to fissured rock below surface less than 3m
- Areas of groundwater recharges
- Highly permeable soils
- Areas immediately upwind of a settled area

- Natural habitat of endangered species which could be disturbed by entrenchment activity

If the site is to be surveyed for the first time, the following characterisations should be completed in order to assess whether the site is suitable and, if it is, to establish baseline data to be used to assess the impact of entrenchment on the environment over time. Characterisations should be submitted as expert reports with the EIA application:

- **Topography and hydrogeology:** Location and depth of hard, impervious layers and permanent and perched water tables should be characterised. The direction and flow of underground waters and potential for underwater drainage should be assessed.

- **Soil:** The surface and subsoil soils should be assessed in terms of structure, permeability and cation exchange capacity (CEC) to indicate the extent to which the soil will retain contaminants and minimise leaching into the surrounding area. Soils should have a clay content of at least 20% and a pH of above 6.5 to limit mobility of metals. Lime can be added to the soil to raise the pH. A baseline analysis of the nutrients, trace elements and metals present in the soil should be conducted in order to assess the impact of sludge on native soil over time.

The Department of Water Affairs provide two systems of assessing the metal content in the receiving soil depending on the use of sludge. In cases where sites with limited public access are used to cultivate industrial crops (e.g. timber), maximum permissible levels (MPL) are provided for metal concentrations for the receiving soil. For more details refer to Section 9.1.7.1. This system is designed for wastewater sludge. As VIP sludge comes from domestic sources it is not expected to contain harmful levels of metals.

- **Surface water:** Possible surface water resources should be identified and sampled to determine baseline values which can be used for comparative purposes should surface contamination occur during sludge entrenchment.
- **Groundwater:** The aquifer must be classified in terms of yield, depth, and strategic value. A buffer of 200m should be maintained from surface water, boreholes and the recharge zone of major aquifers, sole-source aquifers or other strategic aquifers. The hydraulic gradient should be determined to assess the position of the monitoring boreholes. Groundwater quality (up gradient and down gradient) should be assessed to establish baseline values for monitoring.
- **Site stability assessment:** An engineering assessment must be completed in order to inform the spacing and orientation of trenches. As a precautionary principle the shear strength of sludge should be assumed to be zero. Trenches should be dug parallel to the ground contours rather than parallel to the slope direction.

1.4 Characterisation of sludge

An initial characterisation of both the soil and sludge can provide information to optimise application rates and economic gain. These include:

- Physical characteristics: pH, total solids (TS), volatile suspended solids (VSS), volatile fatty acids (VFA)
- Nutrients: Total Kjeldahl Nitrogen (TKN), Total Phosphorus (TP) and potassium
- Metals and micro-elements
- Organic pollutants

Testing of sludge to determine pollutant loads must be done any time there is reason to believe there has been a change in the sludge stream at the waste water treatment works or if there is reason to believe pollutants have been disposed of in a pit. Unless sludge has undergone special treatment to kill pathogens, it can be assumed that it poses a risk for disease transmission and safety measures should be followed for transport, handling and burial.

While there are some restrictions on the use of microbiological class[4] ***B*** or ***C*** sludge or pollutant class ***b*** or ***c*** sludge for the growing of crops to be eaten by humans or livestock, there are no restrictions on any of the categories for industrial crops with the exception of restricting public access where sludge with microbiological class ***B*** or ***C*** is applied to the surface. As proper burial overcomes the risk of human contact, deep row entrenchment can be considered for all sludge classes.

1.5 Designing and preparing the site

The site should be designed and prepared to allow adequate access for heavy trucks and field equipment in all weather conditions.

Trenches should be dug parallel to the contours of the slope. Trench spacing and dimensions will depend to some degree on the results of the soil stability assessment and depth to the water table. Enough space must be left to accommodate the vehicle which is delivering the sludge and the backfill heaped beside the trench without the trench collapsing. It is advisable to prepare test trenches to ensure that the desired dimensions and spacing of trenches are feasible for the specific site conditions.

In terms of the impact on degradation of entrenched sludge over time, optimal dimensions for trenches are 800mm deep and 600mm wide. The spacing between rows should be determined by standard practice for the intended tree species but should also take into consideration soil stability, with adequate space to accommodate the vehicles digging the trenches and delivering the sludge. Assuming a row spacing of 3m between the centres of trenches 800mm deep and 600mm wide, this

[4] See Introduction for sludge classification system

will allow for an application rate of 990m^3 of sludge per hectare of plantation, allowing a covering of 300mm of soil over the entrenched sludge.

The backfill soil should be heaped over the trench to allow settling and to prevent erosion. Monitoring boreholes should be located to intersect groundwater moving away from a disposal site.

1.6 Handling of sludge

Due to the likelihood of viable pathogens in sludge it should be handled as a hazardous waste (containing infectious substances) during transportation. Transporters must be informed of the nature and risks of the load and carry accurate documentation. Hazchem placards should be fitted to the vehicles. All road accidents must be reported to the Department of Transport Local Authorities, the Competent Authority and the Department of Water Affairs and steps must be taken to minimize the impact of any contamination on public health and the environment.

1.6.1 Storage

Storage of sludge at the disposal site is not recommended because of the issues of odour, vector attraction and potential contamination of the site surface or leachate. Sludge should ideally be delivered directly to the trench and covered with soil the same day. If storage of WWTW sludge on site cannot be avoided, it should be stored in a specially prepared pit to avoid surface contamination and covered securely so that it cannot be accessed by animals or other vectors. If sludge is stored in an area for more than 90 days then the area is considered a disposal area and a disposal permit is required. VIP sludge should not be stored on site because of its higher vector attraction factor.

1.6.2 Preventing transmission of pathogens

It is recommended that municipalities provide regular (3 monthly) deworming treatment to workers working with sludge and provide a full orientation to educate workers about pathogens, routes of transmission and procedures to protect their health before they begin work. Workers must wear protective gear while handling sludge to prevent infection by bacteria, viruses or intestinal parasites in the sludge. Eggs can become airborne when sludge is handled and so it is important that workers wear masks, gloves, boots and overalls when handling sludge. It is also critical that rigorous protocol be established to avoid transfer of pathogens to surfaces by wearing contaminated gloves, boots or clothing out of the site or when driving vehicles and to ensure that vehicle wheels do not become contaminated, carrying pathogens with them when they leave the site. It is also important that workers are provided with the means to disinfect their clothing so that contaminated clothing is not taken to their homes, increasing the likelihood of infection of their family members. Facilities should be provided at the site for workers to shower and disinfect their hands as needed during and at the end of their work day.

1.7 Application of sludge

1.7.1 Application rate

While the Department of Water Affairs recommends an application rate limit of 120 tonnes dry sludge/ha/year for surface and shallow incorporation of sludge, it also provides a formula for calculating a permissible application rate (PAR) which can be used to demonstrate that higher application rates can be safe for a particular soil without negatively impacting the environment. The authorising authority may approve a higher application rate based on the particular characteristics of the site in terms of soil properties, depth to aquifer, type of aquifer, distance from surface water resource, and other characteristics. Entrenching sludge for forestry allows for far higher loading rates than would be possible on the surface and also moderates the impact of metals in a number of ways different to other utilisation options:

- While high loading rates for surface application of sludge for agriculture would dramatically increase the presence of pollutants in run-off, entrenched sludge at any loading rate should have no contact with surface water if application was completed without contaminating the soil surface.
- While surface application of sludge results in highly aerobic conditions, under which nitrification occurs rapidly and metals mobilise more quickly through the soil profile, entrenchment results in anaerobic conditions, resulting in slow release of pollutants from the sludge as tree roots introduce oxygen into the sludge over the course of several years.
- While landfill disposal creates conditions where sludge is entrenched and nutrients and pollutants have no means to move except through the soil profile, deep row entrenchment involves the planting of trees which take up nutrients, leachate and metals, minimising the movement of these out of the trench. While uptake of high levels of metals in edible crops could represent a health risk, uptake by industrial crops represents no health risks.

While surface application is usually done yearly or more frequently, once-off application of high loading rates can be used with entrenchment to match the growing cycle of the crop. This also minimises human contact with sludge as application is both far less frequent and, once applied, the covering soil provides a barrier between the sludge and workers.

Data from sites used for entrenchment in the US for nearly 30 years and sites currently in use in South Africa over 3 years indicate that application rates of 360 to 800 tonnes dry sludge/ha/year provide maximum benefit to trees and have not negatively impacted the environment. However, ideal loading rates and risk to the environment will vary depending on tree species, pollutant levels in receiving soil and sludge, and site factors such as soil permeability.

The permissible application rate (PAR) can be calculated as follows:[5]

$$\mathrm{PAR} = \frac{\mathrm{MPL} - \mathrm{Soil}_{\mathrm{conc}}}{\mathrm{Sludge}_{\mathrm{conc}}} * 3900$$

[5] Guidelines Vol. 4 p. 22

Where:

PAR	= permissible application rate (tonne/ha)
MPL	= maximum permissible level (mg/kg) (See Table 1)
$Soil_{conc}$	= the actual metal content of the soil (mg/kg)
$Sludge_{conc}$	= metal concentration in the sludge that will be applied (mg/kg)
3900	= conversion factor to account for soil density (1.3 t/m^3) and sludge incorporation depth of 300 mm

Table 1 Maximum levels of elements allowed in sludge to be applied at high loading rates

Elements	Maximum Permissible Level (MPL) mg/kg
Arsenic (As)	20
Cadmium (Cd)	5
Chromium (Cr)	450
Copper (Cu)	375
Lead (Pb)	150
Mercury (Hg)	9
Nickel (Ni)	200
Zinc (Zn)	700

As discussed previously, these guidelines were put in place for management of wastewater sludge and are not designed for pit sludge, which may be less stable and more variable in terms of potentially harmful components. However, as pit sludge originates from domestic sources, it is not expected to contain high levels of pollutants. In the absence of parameters set specifically for pit sludge, the wastewater guidelines can provide a measure of safety in the entrenchment of pit sludge.

1.7.2 Application method

Sludge should be delivered directly to trenches to prevent surface contamination and contamination of vehicles which may drive over sludge deposited at the side of trenches. Trenches should be backfilled the same day with a minimum of 300 mm native soil cover. This will result in an overburden which will subside over time.

1.7.3 Planting and care of trees

Trenches can be levelled at right angles at the time of planting. Trees can be planted using standard forestry practices. This study has found that trees planted on or between rows of entrenched sludge grow equally well, as sludge can be accessed laterally by roots. There is no need to plough the soil prior to sludge application.

1.8 Monitoring

It is the duty of the sludge user to ensure that both during and after the period that sludge is being actively utilised the safety of groundwater is not compromised and pathogenic material is safely contained. Baseline values of groundwater and soil should be established before sludge is entrenched at the site in order to provide a basis for assessing the impact of the entrenched sludge on the site over time. It should be kept in mind that unanticipated factors may arise which could alter the environmental impact of entrenchment during or after the period of use. For example:

- Heavy rains or flooding may cause pollutants and contaminants to rise to the surface or move further, more quickly or in higher concentrations through groundwater
- Encroachment of human settlement may reduce buffers, result in use of water near the entrenchment site for drinking which was not the case at the time of the site characterisation and increase the risk of human contact with contaminated areas
- Informal use of the site which may disturb sand covering which contains pathogens: e.g. sand winning, building, grazing

Should the sludge disposal site be affected by any of the above, sludge entrenchment should be suspended until such time as the conditions once again are conducive to continuation, if in fact this should occur.

1.8.1 Surface and ground water

Monitoring groundwater on the site should include both microbiology (faecal coliforms and E. Coli) and chemistry: alkalinity, organic nitrogen, nitrate-nitrogen, ammonium-nitrogen, chlorides, pH, COD, zinc, cadmium, copper and specific conductivity. Nitrate-nitrogen ($N0_3$) should not exceed 10 mg/ℓ in groundwater if the water will be used for drinking. In the event that the depth of the water table is less than 5m, monitoring should be done 3 monthly for dry sludge and monthly for liquid sludge and during the rainy season. Composite samples should be collected and analysed 20-50m upstream and downstream from the site and from any boreholes located within 1km of the site. Less frequent monitoring may be adequate where the soil clay content is greater than 35%, where dewatered sludge is entrenched above a water table deeper than 10m, or where liquid sludge is entrenched above a water table deeper than 20m. In some cases monitoring of groundwater may not be necessary due to the depth of the water table or other factors. This must be demonstrated through a study conducted by a qualified person. Detailed procedures and methodology for sampling and testing surface and groundwater are provided in the *Guidelines for the Utilisation and Disposal of Wastewater Sludge* (Volume 3, Appendices 1.3, 3 and 4.)

1.8.2 Soil

Monitoring of soil allows the movement of pollutants to be detected before they reach the groundwater, providing an early warning system. If sludge of pollutant class ***b*** or ***c*** is being used, soil must be monitored to ensure that metal content does not exceed the Maximum Permissible Level. (See Section 9.1.7.1). Increased concentrations of chloride provide a first indicator of the movement of elements and the possibility of contamination. The required frequency of monitoring will be determined by the clay content and pH of the soil and the water content of the sludge. If the site contains different soil types, different monitoring schedules may be necessary. Soils should be sampled at 100 mm intervals to a depth of at least 500mm below the bottom of the trench. Crops can be monitored for uptake of metals throughout the growing cycle by comparing foliage samples with samples from crops in similar soils.

1.9 Managing risks to environment and public health

If sludge has been deposited directly from the transport vehicle into trenches without spilling and covered with 300 mm soil, there is no need to restrict access to the site for the public who may use the forest recreationally. However, if any surface area has been contaminated by contact with sludge during application, those areas should be covered with an additional 300 mm of soil or fenced off for 3 years with signage clearly indicating a biohazard. If general contamination of the site has occurred, the entire site should be restricted for a period of 3 years with all workers entering the site wearing appropriate protective equipment and trained in procedures to protect themselves from infection with pathogens. Run-off from a site with surface contamination should be prevented from leaving the site by constructing cut-off trenches or bund walls down-gradient of the entrenchment site to intercept run-off. The bund wall must be high enough to intercept the surface run off with 0.5m to spare. The water must be recycled on site or treated before it is discharged. Similarly, if groundwater monitoring indicates elevated levels of pollutants in the leachate, leachate should be collected in a drainage pond and recycled or treated. Planting of vegetation can also be used to take up ground and surface water and slow movement from the site.

1.10 Closure of site

If entrenchment is to be discontinued at the site, an aftercare plan should be developed to manage potential ongoing risks to public health or the environment dependent on the planned future use of the site by the forestry company. Soil samples should be assessed for pathogen viability and levels of metals and nutrients which could migrate to the groundwater over time. If there does seem to be a risk, the plan should include ongoing monitoring until the risk is resolved and should address issues of ongoing restriction of the site to people, animals or agricultural use if needed to protect public health. If the final rotation of trees has been harvested, a final planting of trees or other plants can be used to assist with the management of both issues, preventing contact with pathogens by humans or animals through digging or farming and providing a sink for nutrients and limiting the movement of soil water which may contain pollutants.

2. Decentralised entrenchment (PIT LATRINE AND WASTEWATER SLUDGES)

Entrenching VIP sludge, septic tank sludge, raw sewage or treated sewage at small, decentralised sites near the source of the sludge has a number of advantages over transport to and treatment at municipal waste water treatment works (WWTW):

- Operations can be handled by a single small entrepreneur
- Lower transport costs
- Minimal overhead and infrastructure required
- Minimal skills required for daily operation (no complex processes or machinery to manage)
- Timber or fruit can be grown to benefit the local community
- Easier for small or rural municipalities to set up than hazardous waste sites (less red tape)

Deep row entrenchment of VIP sludge planted with trees can provide effective management of risks that would be involved with landfill disposal of VIP sludge:

- Direct entrenchment of sludge and burial the same day prevent contamination of surfaces or equipment with pathogens
- Small, dedicated sites with restricted access provide the control of a hazardous waste landfill without the red tape
- Solid waste disposed of in VIPs is automatically co-disposed of with sludge, with the potential for disposal of other solid waste with the sludge if needed
- Planting of trees creates a nutrient sink, reducing movement of potentially harmful nutrients from the trench
- Utilization of sludge by trees allows for repeated disposal of sludge at the same site over cycles of entrenchment and harvest, unlike landfills where re-use of site can only be done by adding additional layers

2.1 Site characterisation and selection

Requirements for site selection and characterisation in terms of environmental and public safety are the same as for deep row entrenchment for forestry and can be found in Section 1.3 of this document.

The following factors should be considered to assess the long term viability of the operation:

- Is the site large enough to accommodate the volume of sludge it will receive over the growth cycle (trench size/volume/size of community serviced/planting cycle – i.e. trenches that are planted are tied up for 6-9 years, can stockpiling be avoided?)
- Distance to source of sludge and means of transport (are VIPs that will need emptying in future years located further from the burial site?)
- Access to site for vehicles transporting sludge

2.2 Characterisation of sludge

While the characteristics of VIP sludge, in contrast to WWTW sludge, may be highly variable from pit to pit, the following assumptions can be made:

- Microbiological content: Pit sludge can be expected to contain high levels of viable pathogens and therefore can be assumed to have a microbiological classification of c.
- Stability: Unless a particular pit has been left unused for more than a year, VIP sludge contains both fresh sludge and sludge which may be several years old, representing all stability classes. Because it does contain some fresh, unstable sludge, it must be classified as Stability class c, requiring vector reduction which is provided by covering it with earth during entrenchment.
- Pollutant content: VIP sludge can be assumed to contain low levels of pollutants because it originates from domestic sources.

2.3 Designing and preparing the site

Trenches should be dug parallel to the ground contours. Trench spacing and dimensions will depend on the spacing of trees. It is advisable to prepare test trenches to ensure that the desired dimensions and spacing of trenches are feasible for the specific site conditions.

In terms of the impact on the degradation of the entrenched sludge over time, optimal dimensions for trenches are 800 mm deep and 600 mm wide, spaced 2.4 m apart edge to edge. This will allow for an application rate of 990 m^3per hectare with 300 mm backfill covering the sludge. The backfill over the trenches should be left heaped or ridged until planting to allow the backfill to settle and prevent erosion.

Monitoring boreholes should be located to intersect groundwater moving away from a disposal site.

Provision should be made at the site for cleaning, disinfection and storage of equipment and protective gear which may have come into contact with pathogens in the sludge. Facilities should be provided at the site for workers to shower and disinfect their hands as needed during and at the end of their work day. Workers should not be required to take work overalls or boots home for cleaning due to the risk of transmitting diseases from contaminated clothing.

2.4 Handling of sludge

Guidelines for safe transport and preventing the transmission of pathogens during the handling of sludge are provided in Section 1.6 of this document. VIP sludge should not be stored at the disposal site due to its vector attraction factor.

2.5 Application of sludge

Sludge should be delivered directly to trenches to prevent surface contamination and contamination of vehicles which may drive over sludge deposited at the side of trenches. The trenches are then backfilled with a minimum of 300mm native soil. Sludge should be covered the same day it is delivered to the trenches to prevent odour and attraction of vectors.

2.6 Planting of trees

Trees may be grown to produce timber for building, fencing, fuel or for the paper and pulp industry. Species which have a high demand for water will assist in absorbing the leachate produced by sludge, reducing the risks of groundwater contamination. Trees with some nitrogen-fixing capacity, such as wattle (*acacia mearnsii*) will benefit less from increased nitrogen loading through sludge. While fruit can be grown safely on trees planted on sludge buried on site at households, fruit should not be grown commercially for sale unless adequate measures are taken to disinfect the fruit from possible surface contamination.

2.7 Monitoring

It is the duty of the sludge user to ensure that both during and after the period that sludge is being actively utilised the safety of groundwater is not compromised and pathogenic material is safely contained. Baseline values of groundwater and soil should be established before sludge is entrenched at the site in order to provide a basis for assessing the impact of the entrenched sludge on the site over time. It should be kept in mind that unanticipated factors may arise which could alter the environmental impact of entrenchment during or after the period of use. For example:

- Heavy rains or flooding may cause pollutants and contaminants to rise to the surface or move further, more quickly or in higher concentrations through groundwater
- Encroachment of human settlement may reduce buffers, result in use of water near the entrenchment site for drinking which was not the case at the time of the site characterisation

2.8 Surface and ground water

Monitoring of water on the site should include both microbiology (faecal coliforms and E Coli) and chemistry: alkalinity, organic nitrogen, nitrate-nitrogen, ammonium-nitrogen, chlorides, pH, COD, zinc, cadmium, copper and specific conductivity. Nitrate-nitrogen ($N0_3$) should not exceed 10 mg/ℓ if the water will be used for drinking. In the event that the depth of the water table is less than 5m, monitoring should be done 3 monthly for dry sludge and monthly for liquid sludge and during the rainy season. Composite samples should be collected and analysed 20-50 m upstream and downstream from the site and from neighbouring residential wells. Less frequent monitoring may be adequate where the soil clay content is greater than 35%, where dewatered sludge is entrenched above a water table deeper than 10m, or where liquid sludge is entrenched above a water table deeper than 20 m. In some cases monitoring of groundwater may not be necessary due to the depth of the water table or other factors. This must be demonstrated through a study by a qualified person. Detailed procedures and methodology for sampling and testing surface and groundwater are provided by DWA (Herselman & Snyman, 2009).

2.8.1 Soil

Monitoring of soil allows the movement of pollutants to be detected before they reach the groundwater, providing an early warning system. The required frequency of monitoring will be determined by the clay content and pH of the soil and the water content of the sludge. If the site contains different soil types, different monitoring schedules may be necessary. Soils should be sampled at intervals to a depth of at least 500 mm below the bottom of the trench.

2.9 Protecting public health and the environment

Animals and the public must be restricted from sites which are in use for burial of VIP sludge in order to prevent sludge being disturbed and pathogens spread. The site should be fenced securely with a gate that is locked when workers are not on the property. Stringent protocols must be followed by workers to prevent the spread of pathogens via vehicle wheels, tools or protective wear as they enter and exit the site.

2.10 Closure of site

If entrenchment is to be discontinued at the site, soil samples should be assessed for pathogens and metals, and viability and levels of metals and nutrients which could migrate to the groundwater over time. If there is a risk of ongoing contamination, an aftercare plan should be developed to manage potential threats to public health or the environment dependent on the planned future use of the site by the municipality or the company which owns the site until year soil sampling indicates that the levels of pathogens and metals have reduced to within acceptable limits. A final planting of trees or other plants on the site can assist with preventing contact with pathogens by humans or animals through digging or farming.

3. On-site burial of sludge (VIP)

Depending on the status of public health in the area, the sludge found in VIP latrines may contain significant levels of pathogenic organisms. This sludge is therefore regarded in South Africa as hazardous waste and it therefore becomes problematic to transport and dispose if it is concentrated in large volumes (as borne out by the preceding section of these guidelines). By far the least cost lowest risk disposal option is to bury the pit sludge close to the VIP where it was abstracted. This will not have any more impact on the groundwater than the VIP from which it was abstracted.

In future regulations may be promulgated to allow the concentration of pit sludge such that, for example, the waste from an entire small village could be entrenched on one site. However, South Africa's Waste Management Act and Environmental Management Act currently preclude this option.

Sludge that is removed from a VIP pit can be buried on site if the home owner agrees and if an appropriate burial site can be found. On site burial of sludge reduces the risks (in terms of disease transmission) and costs of transporting sludge to a disposal or treatment site and eliminates the costs and difficulties involved with treatment or other disposal options. In order to utilise the sludge beneficially, trees which yield fruit, building material, shade or fuel can be planted above or alongside the buried sludge, with the following benefits:

- Nutrients which could potentially contaminate the groundwater are taken up by the trees
- Nutrients available in sludge improve the quality of fruit or wood, enhancing food security or economic security
- Householders are unlikely to expose the pathogens buried in sludge through digging, planting, or play, creating a health risk, if trees are planted over the burial site(s).

3.1 Site selection and preparation

No permit or notification of authorities is required for on-site burial of sludge from a VIP. However, as burial of sludge on site essentially involves digging small pits, the same variables should be considered for site selection as those for VIPs. In particular, the following factors should be considered in siting burial locations for sludge on site:

- If groundwater is present in the VIP pit, sludge should not be buried at locations lower on the site and should be placed in shallower holes.
- On-site burial should not be used in very sandy or gravelly soils because of the potential of groundwater contamination. (See Section 2.3)
- Burial holes should not be located on eroded banks or cutaways where activity could erode the vertical face into the sludge itself, exposing pathogens.
- Burial holes should not be located within 15m of a stream.

Holes or trenches should be prepared of adequate proportions to contain the sludge while also allowing for 300 mm of soil backfill on top of the sludge surface. This cover is partly to keep animals and people away from the sludge, and partly to provide space for trees to be planted above the sludge.

The volume of sludge removed from a full VIP pit will depend on the dimensions of the pit. Typically the volume will range between 1.5 m^3 and 2.5 m^3. Assuming a volume of 2.0 m^3, the recommended dimensions for a disposal pit are 2 m long by 1m wide by 1.3 m deep. Alternately, a pit which is 1.4 m by 1.4 m dug to the same depth will have the same capacity. Depending on soil conditions and the desired number and location of trees to be planted, sludge can also be buried in a trench or divided between several different planting holes. If a single disposal trench is used rather than a pit, the recommended dimensions would be 8m long by 0.5 m wide by 0.8 m deep.

3.2 Transfer of sludge from VIP pit to burial holes

Sludge may contain a variety of harmful pathogens. These can become airborne when the sludge is disturbed, meaning that they can settle on surfaces or enter workers' lungs. It is essential that workers removing sludge follow careful protocols to protect themselves as well as household surfaces. Contamination can occur by workers placing contaminated tools or gloves on household surfaces (walking around in contaminated boots, laying tools on the ground, benches, etc), touching taps or door handles with contaminated gloves or hands, washing hands or equipment under household taps or borrowing householder tools or spilling sludge on the ground.

Basic principles to be kept in mind are:

- Pit emptiers must wear protective clothing (masks, overalls, gloves and boots)
- Work areas (lip of pit, area where tools and equipment are placed) must be protected with tarpaulins to prevent surface contamination
- Workers must not use tools belonging to the household for pit emptying
- Workers must not wash contaminated tools, clothing or hands at the household tap
- Any contamination which occurs must be remedied and the householder must be alerted

Pit emptiers should be provided with long handled tools, protective equipment for themselves and the site, bins for transporting the sludge and the necessary equipment to deal with contamination of surfaces should it happen (lime, clean shovel, stakes, tape).

After tarpaulins are placed on the lip of the pit, a disposal bin is placed on the tarpaulin. Workers remove the sludge from the pit into the bins using long handled shovels and rakes. Tools are placed on the tarpaulin and the bin is carried to the tarpaulin placed on the lip of the first burial hole.

3.3 Planting and care of trees

After sludge is placed in the hole, the top 300mm of the hole is backfilled with soil using a clean spade and the remainder of the soil that has been removed is heaped over the burial site. After trees have been planted over the holes, householders should be provided with information on how to care for their trees (pruning, diseases, etc). The period of time that the buried sludge will provide an adequate nutrient supplement will depend on the tree species, nutrient values in the native soil and the quantity of sludge buried. Typically the tree will not require any supplementary fertilization for at least five years. After that, if necessary, householders can work manure and composted organic waste into the soil around the tree to provide additional nutrients.

3.4 Protecting public health and improving sanitation

Once sludge has been buried on site, clear instructions should be given to the householders that the sites should not be disturbed through digging or planting for a minimum of three years to prevent contact with buried pathogens which may still be viable.

If any contamination of surfaces has occurred during transfer of sludge from the pit to burial holes, these areas should be treated with lime, covered with 100mm soil and restricted with danger tape and stakes. The household should be told that the danger tape should be left for one month to prevent contact with concentrated lime.

Diarrhoeal diseases are a significant cause of the high rate of death among children under five years of age in South Africa. These diseases are spread through contact with faeces which happens when a person does not wash his or her hands with soap after using the toilet or when open defecation is practiced. In addition, it is very common for residents of communities with a history of poor sanitation to be infected with intestinal parasites which are also spread by contact with faeces. Both diarrhoeal diseases and intestinal parasites can pose a serious threat not only to young children but to the elderly and to individuals who are malnourished as a result of poverty or who have weakened immune systems as a result of HIV, TB or other illnesses. It is therefore recommended that municipalities take advantage of pit emptying as a natural point in the cycle of on-site sanitation provision to provide hygiene education and deworming medication to all members of a household. The household should be provided with a dose to be taken immediately and a follow up dose to be taken after six months.