Separate Collection and Treatment of Urine and the Impacts on Urban Wastewater Treatment in South Africa

JA Wilsenach, M McMillan, A Mbaya, KG Motlomelo, C Anderson, A Pascal, S Brown, J Germanis, MR Mashego, JA du Plessis & GA Ekama

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SEPARATE COLLECTION AND TREATMENT OF URINE AND THE IMPACTS ON URBAN WASTEWATER TREATMENT IN SOUTH AFRICA

Report to the

WATER RESEARCH COMMISSION

by

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EXECUTIVE SUMMARY

Figure 1 Earth rise[1], as seen from the moon. Spaceship earth is a finite system that has to supply the demand of an increasing number of passengers

RATIONALE

Environmental impacts of industrialisation and population growth are vast. The growing population's standard of living has improved in many respects and that is fuelled by the consumption of the earth's resources. Different impacts can be divided into many broad categories, but a few key factors have captured our attention. Firstly, food security relies on productive arable land, nutrients, water and energy. Figure 1 shows the finite nature of planet earth and its resources, against the desolate landscape of the moon. Secondly, while energy is based on fossil fuels, all other resources should be considered as finite too. The Haber-Bosch process burns and reduces natural gas to produce urea for fertiliser. Phosphate is mined from concentrated rock ore for fertiliser production. Potassium is minded from concentrated salt deposits for fertiliser production. Following their consumptive use,

[1] Photograph used by permission of NASA

both phosphate and potassium become dispersed in the environment and recovery from such low concentrations would be enormously energy intensive. Surface water is being polluted by nitrogen and phosphate from sewerage (in its raw or treated form) as well as from agricultural runoff. Excess nutrient concentrations lead to eutrophication and deterioration of water quality. Water supply to cities has led to large water transfer schemes, disrupting the flow patterns of natural rivers. In South Africa, roughly two thirds of all water is used for the irrigation of crops.

In the International Space Station, all water, including urine, is recycled and re-used. The view of planet earth as a space station requires a radical re-think of our resource management.

BACKGROUND

Separate collection and treatment of urine is a relatively recent topic in urban wastewater management. Many research institutions and technology developers have since the 1990s, within the framework of sustainable development, introduced programmes to manage the different waste streams from households separately. A crucial development was the modern no-mix flush toilets from Northern Europe that collect urine and faecal matter separately.

Figure 2 Colour coded wastewater, including grey, brown, and three samples of yellow water, which all display vastly different characteristics

In domestic wastewater, high loads of organic matter (particulate Chemical Oxygen Demand), some nutrients and the vast majority of pathogens originate from flushed faeces, which is also termed “brown water”, as illustrated in Figure 2. High loads of organic matter (soluble Chemical Oxygen Demand) originate from kitchen sinks, washing machines, dish

washers baths, showers and hand basins, termed “grey water”. The majority of the nutrients found in wastewater (nitrogen, phosphate and potassium) and a relatively small amount of Chemical Oxygen Demand originate from flushed urine, termed “yellow water”. Furthermore, although not the focus of this project, urine contains a large fraction of the pharmaceutical residue, hormones and other endocrine disrupting chemicals found in domestic wastewater.
If different domestic waste streams could be managed separately, it could open up ways in which to recover and recycle nutrients, while at the same time to improve the treatment of municipal wastewater at lower energy demand.

OBJECTIVES AND AIMS

The objectives of this work were to install a pilot scale urine separation system within a sewer catchment for the first time in South Africa, and thereby demonstrate the concepts, create public awareness, and to do in-depth investigations on the composition and treatment of urine, grey water and brown water. Specific objectives were to:

- Reconfiguring existing urinals and install no-mix toilets to collect urine and faecal water separately.
- Analyse the composition of urine, faecal flush water and grey water, and quantify the daily loads from a small office block.
- Treat undiluted urine from men’s urinals in a sequencing batch reactor designed for biological nitrification and denitrification.
- Treat diluted urine from no-mix toilets with waste activated sludge in an anoxic-aerobic digester.
- Treat grey water and brown water, without urine, in a biological nutrient removal activated sludge process, at different sludge ages, in order to investigate the impact of urine separation on the design and effluent quality of wastewater treatment works.

This project aimed at a comparison of the results from experimental bio-reactors with existing mathematical activated sludge models, and where appropriate, to adapt these for future use. Ultimately, the project aimed to contribute to new ways of thinking that, when implemented, would improve wastewater treatment, water recycle and nutrient recovery.

METHODOLOGY

Two top of the range no-mix toilets were imported from Sweden and installed to replace conventional toilets at the CSIR in Stellenbosch. One no-mix toilet was installed in the ladies’ room and one in the men’s room. The water supply to three existing urinals was closed off, and odour traps were retrofitted to these urinals. Wastewater and urine from the toilets was piped to a collection courtyard.

Over time, samples of the separately collected urine, yellow water, brown water and grey water were analysed. Samples were analysed at CSIR’s water laboratory in Stellenbosch according to standard analytical methods.

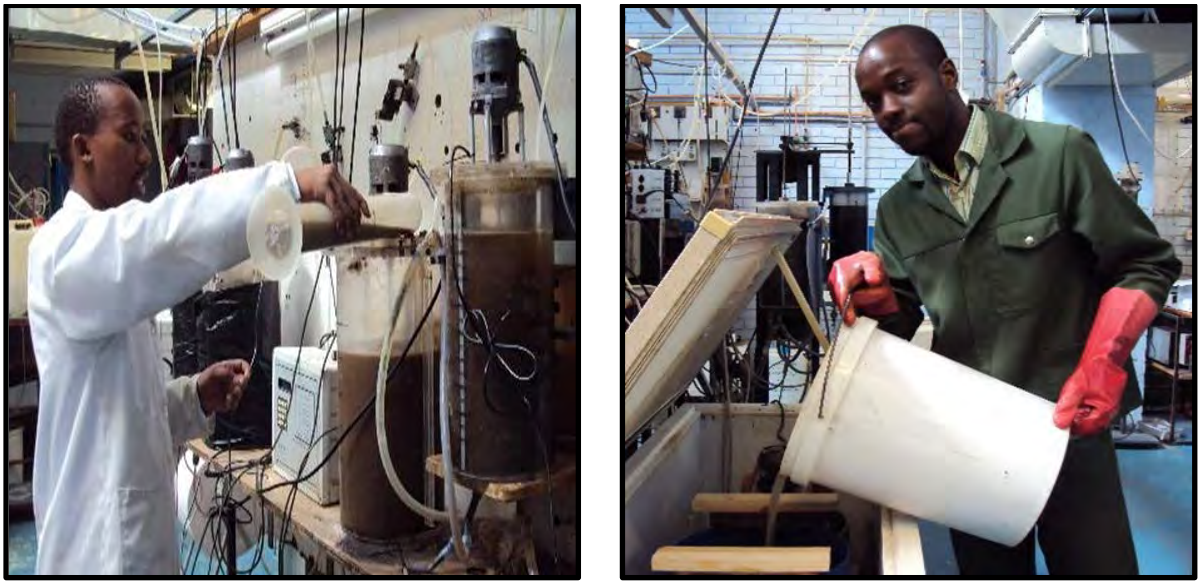

Figure 3 Experimental set-up for anoxic-aerobic sludge digestion with urine (left) and making up a 50:50 mixture of grey:brown water with separately collected waste streams (right)

Three bench scale bioreactor systems were investigated (two at University of Cape Town and one at Stellenbosch University). The reactors were fed with the separately collected waste streams.

RESULTS AND DISCUSSION

The composition and loads from different waste streams

Total Kjeldahl Nitrogen (TKN) concentrations ion undiluted urine were analysed to be between 6,000 and 7,000 mg/l (maximum 10,450 mg/l), while Chemical Oxygen Demand concentrations (COD) were even higher, at 7,000-8,000 mg/l (maximum 14,606 mg/l). The average concentration of phosphate in urine was 262 mg P/l. It is interesting that the first sample analysed had the maximum concentration out of 50 samples, at 734 mg P/l. This sample was taken before the build-up of biofilm in the pipe that accelerates hydrolysis of urea, which raised the pH to 9 and led to the natural precipitation of phosphate salts (Figure 4).

Urine contains high concentrations of inorganic salts. The average total dissolved solids concentration from undiluted was 14,583 mg TDS/l with relative standard deviation (RDS) of 31%, over n = 50 samples. The average electrical conductivity was 3,260 mS/m (RDS = 39%). Flush water clearly had a great influence on the composition of yellow water, and more so in the case of urine from the ladies' toilet. Total dissolved solids in yellow water from men's and ladies' no-mix toilets respectively had average concentrations of 3,865 mg/l (RDS = 149%, n = 8), with EC = 1,274 mS/m (RDS = 133%), and 2,837 mg/l (RDS = 190%, n = 8), with EC = and 903 mS/m (RDS = 172%). By contrast, the total dissolved solids concentration in brown water was only 651 mg/l, with EC = 70 mS/m. The average COD of

brown water was 3,566 mg/l, with TKN only 140 mg/l and phosphate of 16 mg P/l. Grey water had an average TDS of 722 mg/l, with EC = 24 mS/m. The average COD concentration of grey water was 2,208 mg/l, with TKN of 47 mg/l and phosphate only 3 mg P/l.

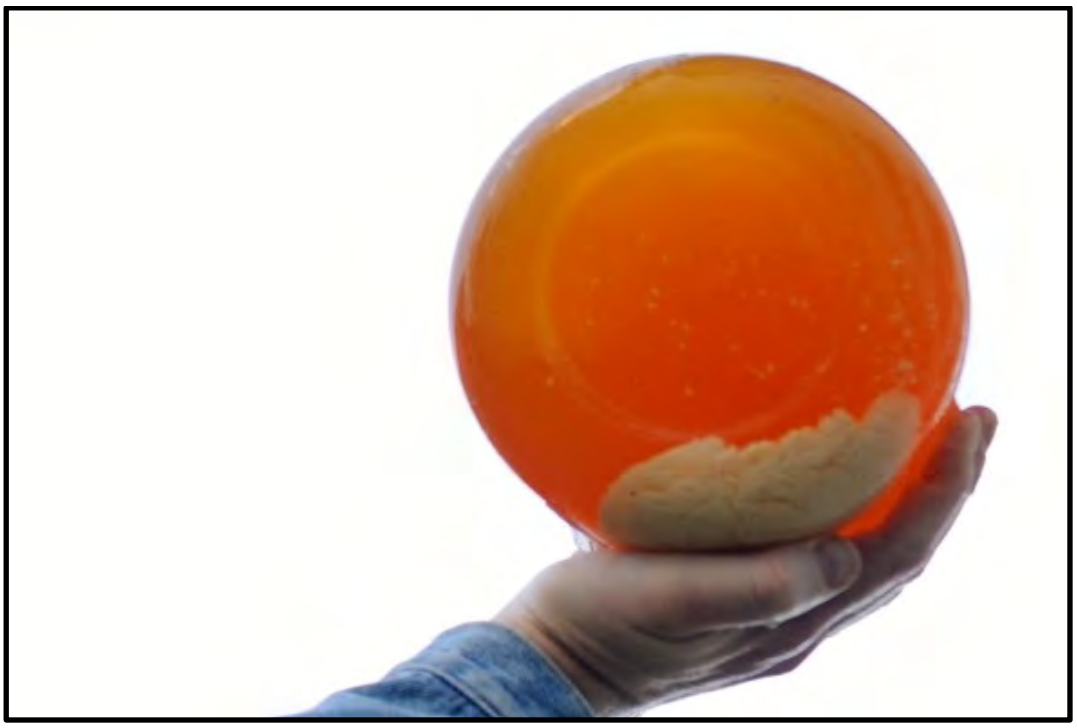

Figure 4 Dissolved salts contained in urine precipitate naturally at high pH

The analysis of undiluted urine, yellow water, grey water and brown water has shown that the majority of nutrients in wastewater originate from urine. At the particular CSIR office block, at least 66% of the nitrogen load and 43% of the phosphate load originated from urine produced over one 24 hour period.

Nitrification and denitrification of source separated urine in a sequencing batch reactor

An experimental study was conducted in which a 20 l sequencing batch reactor was used for the nitrification-denitrification of undiluted urine. A stable system was achieved with a sludge age of 20 days, in which one litre of urine was exchanged daily for one litre of reactor content. In this mode, the reactor converted 60-70% of the approximately 6,000 mg TKN/l in urine to nitrite mostly, and relatively small amounts of nitrate. Furthermore, through the oxidation of organic matter in urine, under anoxic conditions, it was possible to remove an average of 33% of the total nitrogen load in urine to N_2 gas. The final effluent consisted of a consistent ammonium-nitrite mixture. The available alkalinity in urine limited the nitrification of all ammonium to nitrite. It was found that salinity, as well as high nitrogen concentrations, inhibited the growth and activity of nitrite oxidisers to a much greater extent than ammonium oxidisers. The process is more effective in removing nitrogen and in oxidising ammonium, if

the oxidised nitrogen product is limited to nitrite instead of nitrate. It is therefore important to prevent or minimise the dilution of urine with flush water during separation and collection.

Figure 5 Two samples of urine; on the left is undiluted stored urine from men's urinals while on the right is urine treated in the sequencing batch reactor

Process automation would be possible simply by on-line measurement of pH and ROP. These two measures are tell-tale indicators of the process sequence and performance. The sequencing batch reactor was successfully modelled using a spread sheet based iterative approach. The stoichiometry of the process could be described mathematically from first principles (i.e. known microbiological processes). The kinetic of the system was based on the actual process, which was governed, and limited compared to process rates in conventional activated sludge systems, by high ammonium concentrations, high salinity and the effect of extreme high and low pH values. The model simulation was calibrated with the measured oxygen uptake rate, and a good fit was obtained. The results of the sequencing batch reactor and the modelling approach is discussed in great detail in the M.Sc. (Eng.) thesis of Morgan McMillan (Stellenbosch University).

Nitrification and denitrification of urine with waste activated sludge in an anoxic/aerobic reactor

Management and further treatment of waste activated sludge is often the bugbear of wastewater treatment works. A relatively simple and straightforward method of sludge digestion is through extended aeration, despite the perceived high energy cost. An improvement on the extended aeration, or aerobic digestion, is anoxic-aerobic sludge digestion. This system allows for nitrification of the ammonium released during sludge decay and lysis, as well as denitrification of the formed nitrate with COD that becomes available from the same decay. Urine collected from no-mix toilets (which diluted urine by at least

50%) was treated in an anoxic-aerobic digester, with a 3 hour air-on and 3 hour air-off sequence in a 6 hour cycle, which also stabilized concentrated waste sludge from a biological nitrogen and phosphate removal activated sludge system. A total daily influent flow that consisted of 600 ml urine (roughly 1,200 mg COD/l, 1,500 mg TKN/l, and 150 mg P/l)) and 600 ml concentrated WAS (roughly 20,000 mg COD/l, 1,500 mg TKN/l, 1,500 mg P/l) were dosed to a 12 litre anoxic-aerobic digester operated at 20 d retention time. Proportionally the nitrogen load on the system was equal from the waste activated sludge and the urine. In the case of phosphate, 90% of the P load was from the WAS and only 10% from the urine. From operating the system for a period of 1 year with and without Ca or Mg hydroxide dosing, it was found that (1) nitrification was complete (effluent < 2 mg FSA-N/l) with negligible nitrite (<1 ngNO2-N/l), (2) denitrification removed 60-70% of the nitrate generated, and (3) about 1/3 of the phosphate released from the WAS was precipitated with Mg co-released from the phosphorus accumulating organisms (PAO) polyphosphate and (4) Ca and Mg hydroxide dosing increased the phosphate precipitation approximately stoichiometrically and partially restored the alkalinity lost by precipitation. An improvement in the denitrification efficiency was achieved by dosing urine during the anoxic phase of the batch process. Even through urine contributed very little COD to the overall influent load, the COD in urine was much more bio-degradable than that of the waste activated sludge, in which the COD is made up of inert particulate material (for a large part), as well as activated sludge biomass, which had to decay and hydrolyse first before it could become available as substrate for denitrification. The results of this work is further discussed in the B.Sc. (Eng.) thesis of Koali Motlomelo (University of Cape Town).

Impacts of separate collection of urine on biological nutrient removal activated sludge processes

Conventional wastewater treatment, with activated sludge processes, requires sludge ages of around 20 days to ensure good nitrification. The required sludge age results in large bioreactors and to some extent, large clarifiers too. Separated collection of urine at source could be a way to change the conventional approach, and decrease the required sludge age considerably, and even to simplify biological nutrient removal activated sludge (BNRAS) wastewater treatment plants (WWTP). Although urine makes up less than one percent of the volume of wastewater, it contains around 50% of the total phosphate load and up to 80% of the nitrogen load in wastewater. The impact of urine diversion on BNRAS processes was investigated in a laboratory scale reactor with a University of Cape Town (UCT) BNR system configuration, receiving mixes of grey and brown wastewater collected separately at the CSIR in Stellenbosch. Interestingly, the grey water and brown water collected for experiments had lower overall concentrations than that measured initially at CSIR. Out of 13 batches, brown water concentrations were 2,238 mg COD/l, 166 mg TKN/l and 50 mg P/l. Grey water concentrations were 1,434 mg COD/l, 65 mg TKN/l and 27 mg P/l. After reaching steady state, the laboratory-scale UCT system was operated for 90 days at a sludge age of 20 days and a temperature of 20ºC. Effluent ammonia and nitrate concentrations (both <1 mg N/l) and aerobic batch test on sludge harvested from the aerobic reactor revealed the nonexistence of nitrifers in the system treating a 50:50 (by volume) grey and brown water mix. A series of batch tests were done, which pointed to 6 factors to prove the non-existence of nitrifiers in the system, which included (i) no nitrite or nitrate in the effluent (ii) no nitrate

generation when fed with excess ammonium in batch tests (iii) no decrease in alkalinity (iv) phosphate release in the anoxic compartment (v) measure oxygen uptake rate yielded a good COD balance and (vi) a low nitrogen fraction of the sludge produced. This meant that nearly all N was used for biological growth and no nitrate was produced, and the P removal proceeded via the normal biological excess P removal (BEPR). In fact, the batch tests indicated that the system might have been nitrogen deficient. The removal efficiencies of the system were 86%; 90% and 93% for Organics (COD), Total Kjeldahl Nitrogen (TKN) and Total Phosphorus (TP) respectively and the effluent concentrations were 85.3 mg COD/l, 2.9 mg TKN/l, 0 mg NO_3-N/l and 1.3 mg TP/l. These removal performances were all better than those achieved in a control system, with exact design parameters and an equivalent COD feed, from a real domestic wastewater treatment works. A second system was operated according to the Johannesburg (JHB) process configuration, which also received a 50:50 mixture of grey and brown water. The JHB process is similar to the UCT process, but includes an anoxic zone in the return activated sludge stream, to remove any nitrate before entering the anaerobic zone.

Based on the results of the UCT process, the JHB process was operated at a sludge age of only 5 days, in order to test the hypothesis that urine separation could lead to smaller bioreactors. Surprisingly however, the JHB process struggled with COD removal. A large fraction of bio-degradable particle organics was not removed. Therefore, the fraction of ordinary heterotrophic organisms was relatively lower. In turn, this resulted in less of the bio-degradable soluble organics that could be converted to volatile fatty acids for consumption by the phosphate accumulating organisms. Finally, phosphate removal was poor in the JHB system. This poor performance is perhaps more indicative of the fluctuation in the specific substances from the CSIR wastewater, which was not “buffered” against variation in the same way that a municipal wastewater stream does. It is possible that the composition of wastewater changed faster than the ability of bacteria to adapt[2]. The mass balances showed that if all the degradable COD was utilized, then the very low effluent standards of 0.1 mg P/l could be achieved in both UCT and JHB process configurations. Furthermore, based on the mass of COD load on a wastewater treatment works, the reactor can be reduced by 50% if urine was collected separately. From the perspective of an existing system, an activated sludge reactor basin could treat double the design load if urine was collected separately at source! Results of this work on biological nutrient removal activated sludge systems are discussed at length in the M.Sc. (Eng.) thesis of Andre Mbaya (University of Cape Town).

CONCLUSIONS

The concept of urine separation is still in its early stages, but the prospects of achieving more sustainable urban water management are very good. Urine separation could lead to better wastewater treatment, e.g. increased plant capacity, improved effluent quality and reduction in energy consumption. With complete urine separation, the need for nitrification

[2] Adaptation or acclimatization of bacteria to substrate may be a clue for understanding the often inconsistent performance of small and package plants.

and denitrification falls away at activated sludge plants, which would be operated at short sludge ages (5 days) with anaerobic phosphate release and aerobic phosphate uptake.

Treatment of undiluted source separated urine in a sequencing batch reactor could be described in terms of the normal activated sludge model stoichiometry. The specific process rates were however less than normal activated sludge kinetics, due to salinity of urine, as well as extreme high free ammonia and nitrous acid concentrations. Still, due to the small volume of urine (by comparison to the overall domestic wastewater production), the lower rate is not of concern. At least one third of the nitrogen in urine can be removed over nitrite by utilizing the COD in urine. The resulting ammonium-nitrite mixture is the ideal feed for an Anammox process that removes nitrogen without the need for organic carbon. An alternative systems, which consists of treating yellow water from no-mix toilets with waste activated sludge, was investigated in an anoxic-aerobic digester. This is a fairly simple system, which could augment existing overloaded treatment works immediately, or eventually be developed to remove all nitrogen in a side stream process.

However, in order to advance beyond experimental work, some risks need to be taken (capital risks); Also, strong political will, significant financial investment and large scale public education are required to embark on the source separation of urine route in urban sanitation, because any deviation from the current water borne sanitation will be less convenient for users but better for the environment. Adoption of urine separation sanitation by middle- and high-income urban groups will change the perception that urine separation is an inferior system only for low-income communities.

RECOMMENDATIONS FOR FUTURE RESEARCH

Further improved performance of the UCT and JHB systems, where urine is collected separately, could be possible if the following is considered:

- A larger no-mix installation, with urine separation, to generate grey and brown water, the characteristics of which must better represent the overall population. Real domestic or municipal wastewater must be collected from a larger and socio-economically more diverse group of people. This may require the installation of no-mix toilets in places such as shopping malls or airports.
- A larger BNR system set up. A lab- or possibly pilot-scale JHB system (e.g. 2000 m^3 in volume) can be operated at the same SRT of 5 days which will treat grey and brown water collected from a large and more diverse population. *Note:* Further precautions must be considered when operating the new system such as toilet paper which is biodegradable COD must be part of the treated WW or the use of excessive detergents in cleaning no-mix toilets must be avoided because they can be detrimental to the microorganisms in the AS.
- An extensive analysis of the results. Measurements on the JHB system must be evaluated by comparison with predictions of the models for BNRAS systems. Also, the separated and brown water samples must be analysed for micropollutants such as hormones, environmental estrogens and pharmaceutical residues which was not done in this investigation.

Some ideas are presented in the main text for the application of urine separation technology, but there is much room for further optimisation operating regimes of bioreactors and hybrid system configurations that combine side stream processes for treatment of source separated urine with waste activated sludge or sludge reject water. There should be a step-wise introduction of ideas, at the appropriate time and place, which will gradually change the way we think and work with waste streams. Ultimately, the design and management of sustainable earth systems require radical change.

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The staff members in D-Block at the Council for Scientific and Industrial Research's Stellenbosch offices are thanked for their indispensable daily contributions in using the no-mix toilets!

SKILLS DEVELOPMENT AND KNOWLEDGE TRANSFER

Six graduate and post graduate students worked on this project. Their final reports are included on a Compact Disc with this report, or obtainable from the Water Research Commission:

Student	University	Degree	Thesis title
Morgan McMillan	SUN	M.Sc. (Eng.)	Nitrification and denitrification of source separated urine in a sequencing batch reactor.
Andre Mbaya	UCT	M.Sc. (Eng.)	The impact of source separation of urine on biological nutrient removal activated sludge plants.
Koali Motlomelo	UCT	B.Sc. (Eng.)	Treatment of source separated urine with concentrated WAS from a BEPR reactor in an anoxic-aerobic digester.
Charney Anderson	CPUT	B.Tech. (Chem.)	Investigation of urea hydrolysis of source separated urine.
Aluwani Maumela	UCT	B.Sc. (Eng.)	Evaporation of aerobically pre-treated urine.
Chakala Mphafi	UCT	B.Sc. (Eng.)	Redesign of toilets, drains and sewers for existing building to allow source separation of urine (practical feasibility study).

Two official workshops were organised and well attended as part of this project. The workshop programmes are included In Addendum of this report:

Date	Place	Number of presentations	Number of attendees
28 July 2009	CSIR, Stellenbosch	7	66
15 February 2011	Paradise Valley, eThekwini	9	49

Four oral presentations were presented as full papers at conferences after peer review. The articles are available online via the eWISA knowledge base:

- Motlomelo, K.G., Ekama, G.A. and Wilsenach, J.A. (2012) Co-stabilization of source separated urine and concentrated waste sludge from a biological nutrient activated sludge system. WISA Biennial Conference, Cape Town.
- Mc Millan, M., Wilsenach, J.A., Germanis, J. and du Plessis, J.A. (2012) Biological nitrification and denitrification of source-separated urine in a sequencing batch reactor. WISA Biennial Conference, Cape Town.
- Ekama, G.A., Wilsenach, J.A. and Chen, G.H. (2012) Urine separation and saline sewage treatment for more sustainable urban wastewater management. WISA Biennial Conference, Cape Town.
- Mbaya, A, Germanis, J, Wilsenach, J.A. and Ekama, G.A. (2010) The impact of urine diversion on biological nutrient removal activated sludge systems. WISA Biennial Conference, Durban.

One internationally peer reviewed article resulted from this project. The article is available from IWA online publishing:

- Ekama, G.A., Wilsenach, J.A. and Chen, G.H. (2011) Saline sewage treatment and source separation of urine for more sustainable urban water management. Water Science and Technology 64(6) 1307-1316.

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LIST OF ABBREVIATIONS

AE	Aerobic reactor
AN	Anaerobic reactor
AS	Activated Sludge
AX	Anoxic reactor
BEPR	Biological Excess Phosphorus Removal
BNRAS	Biological Nutrient Removal Activated Sludge
COD	Chemical Oxygen Demand
d	day
˚C	Degrees Celsius
DO	Dissolved Oxygen (mg O/ℓ)
DSVI	Diluted Sludge Volume Index
f_{CV}	COD/VSS ratio (mg COD/mg VSS) of the mixed liquor
f_i	VSS/TSS ratio (mg VSS/mg TSS) of the mixed liquor
f_N	Nitrogen to VSS ratio of the mixed liquor (mg N/mg VSS)
$f_{S'bs}$	Fraction of the total COD which is readily biodegradable (soluble)
$f_{S'up}$	Unbiodegradable particulate COD *fraction* of the total influent COD in wastewater
$f_{S'us}$	Unbiodegradable soluble COD *fraction* of the total influent COD in wastewater
F- (prefix)	Denotes Flux
FSA	Free and Saline Ammonia
JHB	Johannesburg
K	Potassium
ℓ	Litres
MBR	Membrane bioreactor
Mg	Magnesium
MLTSS	Mixed Liquor Total Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
N	Nitrogen
ND	Nitrification-denitrification
NO_2^-	Nitrite
NO_3^-	Nitrate
NO_x	Nitrite and nitrate
OUR	Oxygen Utilization Rate
p	person

P	Phosphorus
PAO	Polyphosphate Accumulating Organism
PST	Primary Settling Tank
Q_i	Influent flow rate
RBCOD	Readily Biodegradable COD
rpm	revolution per minute
S_{ti}	Total influent COD wastewater concentration (mg COD/l)
S_{use}	Unbiodegradable soluble COD concentration of the system effluent
S_{usi}	Unbiodegradable soluble COD concentration of the system influent
SRT	Solids Retention Time
SS	Suspended Solids
SST	Secondary Settling Tank
St_{effl}	Total effluent COD concentration
St_{ws}	Total COD concentration of the aerobic mixed liquor
TKN	Total Kjeldahl Nitrogen
TP	Total Phosphorus
TSS	Total settleable solids (mg TSS/l)
VSS	Volatile settleable solids (mg VSS/l)
WWTPs	Wastewater Treatment Plants

CPUT	Cape Peninsula University of Technology
CSIR	Council for Scientific and Industrial Research
SUN	Stellenbosch University
UCT	University of Cape Town
WRC	Water Research Commission

1 INTRODUCTION

1.1 PRIMARY DRIVERS FOR CHANGE IN WASTEWATER TREATMENT

1.1.1 A brief history of major change drivers in wastewater treatment

Water borne sanitation and municipal wastewater treatment have seen many changes over the past 150 years. The primary driver that changed the bucket systems of England and Western Europe in the late 19th century was the discovery of microbial pathogens and the revelation that these organisms travel easily from human waste to drinking water. Following the introduction of centralised sewer systems, which drastically improved the health of the urban communities they served, rudimentary treatment plants were developed to protect receiving water against oxygen depletion. Trickling filters, such as the Daspoort Eastern Works, built in Pretoria in 1910, effectively removed the chemical oxygen demand in wastewater. The space required for expansion of trickling filter plants, in order to maintain treatment capacity for growing urban populations, became so large that suspended activated sludge systems were introduced. In many cases, activated sludge plants did not replace trickling filters, but augmented the treatment capacity of old trickling filters. Later still, in order to address the problem of eutrophication, wastewater treatment plants had to change in order to remove nutrients. After the research done at University of Cape Town (), biological nutrient removal activated sludge processes became the norm for large new treatment plants. What is driving change these days? The cost of urban land and space limitation has further led to development and implementation of membrane bioreactors. With these systems, clarifiers are no longer required while reactors are more compact with higher sludge concentrations, although at much higher operation and maintenance cost. Still, what else is, or should be, driving future changes?

No-Mix technology was re-introduced into the scientific research community of modern Europe mostly by Larsen and Gujer (1996) and by Otterpohl (2002), who argue that the city of the future could be without sewers. In this kind of city, rain water would be infiltrated, grey water would be treated and recycled on site for household purposes other than drinking water. Brown water (toilet water without urine) can be converted to biogas, while yellow water (flushed urine) can be converted to fertilizer. Vinneras and Jonsson (2) claim that if urine and faeces were collected separately, up to 91% of N, 83% of P and 59% of K, respectively, can be recovered and recycled from domestic wastewater. Maurer et al. (2006) have worked out a number of options for collection, storage and treatment of urine, separately from the mixed wastewater stream.

1.1.2 Environmental impacts related to nitrogen

On a global scale, the natural nitrogen cycle is controlled mainly by the interaction of five biological processes. These processes are (i) fixation of atmospheric nitrogen, (ii) uptake of nitrogen as part of growth (plant and animal anabolism), (iii) mineralisation (decay), (iv) nitrification and (v) nitrate reduction or denitrification. Atmospheric nitrogen is unavailable for use by organisms. In order for living organisms to use nitrogen, nitrogen gas must first be converted to ammonium, nitrate, or organic nitrogen like urea. Nitrogen is fixed by bacteria in the soil and plant roots through metabolic pathways. Other less important nitrogen conversion processes include lightning, forest fires and lava flows. In wastewater treatment, the three main processes that involve nitrogen are growth, nitrification and denitrification.

- Bacteria found in biofilm or suspended solids, which are employed to treat domestic wastewater, take up nitrogen in order to build cell mass. However, the ratio of organic substrate to nitrogen encountered in wastewater is skewed from the perspective of organic growth, so that only a fraction of the total nitrogen is removed in this way.
- Ammonium at different concentrations is toxic to different fresh water animals. Furthermore, ammonium would ultimately consume oxygen in a natural nitrification process and deplete surface water from oxygen. Therefore, advanced wastewater treatment processes are designed for nitrification, which is the oxidation of ammonium through a series of processes, ultimately to nitrate.
- Nitrate in fresh water encourages growth of algae and eutrophication. Nitrate is also a toxic substance in drinking water. Therefore, advanced wastewater treatment processes are also designed for denitrification, which is the reduction of nitrate (or nitrite) to elemental nitrogen gas that is released into the atmosphere.

Wilsenach and van Loosdrecht (2003, 2004, and 2006) have argued that nitrogen removal is the single most important process that determines the scope, size and efficiency of advanced biological nutrient removal processes. Modern versions of the Modified UCT process, which includes denitrification and biologically enhanced excess phosphate removal, seem to be close to the pinnacle of this technology. Further improvement of advanced wastewater treatment, to deal with emerging pollutants for instance, cannot be of an incremental nature and requires a step function to an alternative system.

As opposed to the natural nitrogen cycle, the industrial production of ammonia and urea has revolutionised modern agriculture. While the rate of food production was historically limited in part by the rate of natural nitrogen fixation, the commercialisation of the Haber-Bosch process and industrial nitrogen based fertilisers has led to more than a doubling of the reactive nitrogen available since 1950, from 150 million tonnes to more than 350 million tonnes (International Fertilizer Association). Most of this nitrogen terminates in surface water, as either end-of-pipe discharges at wastewater treatment works, or as diffuse pollution.

A critical assessment of the anthropogenic nitrogen cycle's impacts on the environment should be an important driver for change. Human intervention has all but erased the boundaries between the natural and the industrial nitrogen cycle, which means that changes now require the re-design of earth systems, in which nature and industry interacts through complex feedback loops.

1.1.3 Environmental impacts related to phosphate

Eutrophication is a serious problem in surface water that stems from high nutrient concentrations and manifests in excessive growth of algae and cyanobacteria. Secondary problems that result from eutrophication in surface water include poor penetration of sunlight, release of toxins, and ordinary decay of algal biomass that in turn leads to depletion of dissolved oxygen. However, even if all nitrogen were removed through wastewater treatment, surface water is always in contact with atmosphere, through which nitrogen could be re-introduced through fixation. Phosphate, on the other hand, is the one element in nature, without which no life is possible. Removal of phosphate is therefore paramount in surface water quality control. Phosphate was first removed through chemical precipitation. However, the chemical dosing process (e.g. via ferric chloride) is expensive and leads to an increase in salinity. Enhanced biological excess phosphate removal activated sludge processes were developed to prevent eutrophication in a more cost effective way. Nevertheless, without

exceeding the effluent standards (some as low as P < 1 mg/l), large wastewater treatment works still discharge tonnes of phosphate per annum into sensitive waters. For example, if a wastewater treatment works discharges 400 Mℓ/d upstream of the Hartbeespoortdam at a very low effluent concentration of 0.5 mg P_{tot}/l, it amounts to 73 tonnes of P per annum. Yet, the target mean annual phosphate load to turn around the hypertrophic condition of Hartbeespoortdam is only around 50 tonnes of P per annum (Harding, 2008). At this point, not the effluent concentration, but the total mass is of concern.

Figure 6 Two images of eutrophication in the Hartbeespoortdam, with satellite image overview from 2010 (left) and detail air photograph of water condition at the wall (right)

Phosphate is a finite resource, which is mined extensively for the production of mineral fertiliser. Foskor (Pty) Ltd in South Africa is an international player with its Phalaborwa mine and phosphoric acid plant in Richards Bay. The total resource estimate at Pahlaborwa mine is 2000 Mt (Foskor, 2007). Phosphate rock extraction has varied between 2600 kt in 2006 and 2200 kt in 2010. This resulted in phosphoric acid production of between 626 and 622 kt P_2O_5 in 2006 and 2010, respectively. By comparison, if one assumes a domestic wastewater contribution of 2 gP/person.day, and 50 million people living in South Africa, then the maximum recovery of phosphate would be around 75 kt P_2O_5 per annum. Clearly, with Foskor's current emphasis on phosphate export from mining, large scale recovery of phosphate from waste streams, such as domestic wastewater, will not be immediately economically attractive. However, ultimately the price of phosphoric acid is expected to increase with the depletion of high quality phosphate rock which is easily mined. Currently the economics of resources are driven by complex markets. For instance, the price of rock phosphate was R1,700/ton in 2009, but dropped to R900/ton in 2010. At the same time, the price of sulphuric acid, used for production of phosphoric acid, was R4,400/ton in 2009, but down to R600/ton in 2010 (Foskor, 2013). It seems that "market sentiments" such as the commodity price hike of 2009, have a much greater influence on price than long term fundamentals.

Sub-Saharan Africa experiences chronic food shortages, which is at least in part, related to relative nutrient deficiency in soils. For example, Sub-Saharan Africa uses only around 8 kg/ha, whereas Africa as a whole uses around 20 kg/ha, South Africa 60 kg/ha and the world average is around 93 kg/ha. Countries that export food use considerably more fertilizer, e.g. North America (98), Asia as a whole (146), Western Europe (175) and East Asia (202 kg/ha). The "Green revolution" of Asia has seen dramatic increases in harvest yields through this increased fertilizer use. African Union states

set a target of increasing fertilizer use to 50kg/ha in 2015 (Pitse, 2007). It is interesting that China, who has vast phosphate reserves, continued to increase their import of phosphate. Clearly, food security is based partly on long term phosphate availability. In future, recovery and re-use of phosphate, instead of the current linear flux, would drive changes in the global nutrient system.

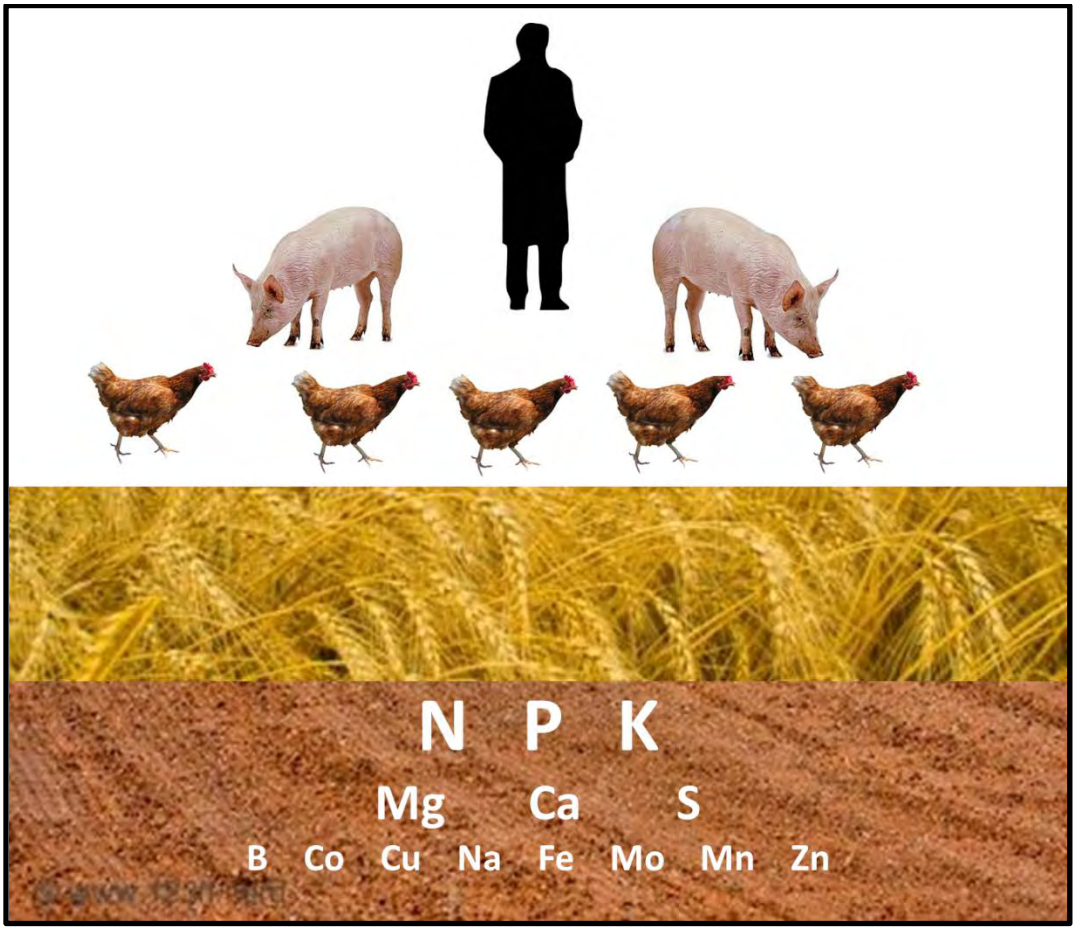


Figure 7 The modern food pyramid, including major nutrients and micro-nutrients in the soil system, industrial grain production, animal husbandry, all engineered for consumption by man

The African Union is set on increasing fertilizer dosing in the near future, but some European Union member states have already introduced policies to promote recovery of finite elements, especially phosphate. Sweden for example has set a goal that by 2015 at least 60% of all phosphate present in wastewater should be recovered for use in productive land (Swedish EPA, 2002). One way of phosphate recovery is through precipitation of struvite ($MgNH_4PO_4.6H_2O$). Struvite precipitation under special conditions, where ammonium has been removed, also allows for the recovery of potassium, which is another finite element like phosphate used predominantly in industrial fertilizer, as $KMgPO_4.6H_2O$ (Wilsenach et al., 2007).

If a phosphate recovery is not introduced on a national scale, as part of a macro-economic policy, recovery and re-use of phosphate will become attractive on a local scale. As the price of fertilizer and transport is expected to increase, the economic viability of phosphate recovery and local re-use (e.g. as struvite) may increase for medium enterprises.

1.1.4 Energy demand versus water quality control

Improved effluent quality normally implies a higher energy demand, due to aeration, pumping, mixing, filtration, etc. The electricity that drives modern wastewater treatment works can be viewed from two problematic angles. Firstly, energy that is derived from finite resources, such as fossil fuels must be conserved. Secondly, the coal fired power stations that are largely employed for generation of electricity are under scrutiny for their emission of harmful gasses. Most often, CO_2 is cited for its contribution to climate change. However, many more harmful gasses are emitted which have direct negative impacts. Such gasses include sulphurous and nitrogenous compounds that lead to air pollution (smog) and acidification.

In March 2013, ESKOM had 44 GW installed capacity, of which 38 GW was supplied by coal fired power stations (ESKOM 2012 Annual Report, Statistical tables, www.eskom.co.za). Much is made of the "water-energy nexus", but just how much electricity is required for wastewater management? Swartz et al. (2013) show that wastewater treatment and sludge management require up to 80% of the electricity of wastewater management, with the remaining 20% required for sewerage collection and transport through pumping. The primary energy requirements for advanced wastewater treatment and sludge management were worked out by Wilsenach and van Loosdrecht (2006) as 0.92 kWh/m^3, which is equivalent of 0.27 kWh/m^3 as electricity. Assuming that all 50 million South Africans were connected to water borne sanitation and activated sludge treatment plants, which is not yet the case, and that a constant average 10 watt per person of electricity is required for the wastewater collection and treatment infrastructure, then 500 MW of electricity would be required. Wastewater transport and treatment therefore require only 1.25% of the national electricity demand. From the perspective of the global energy situation, wastewater management should hardly be considered any further. This argument is developed in greater detail still by Svardal and Kroiss (2009) who reason that the long term value of the fresh water is much higher for sustainability than the energy and carbon footprint generated in maintaining its quality. For South Africa, which is faced by an increasing demand for fresh water from a fixed resource, compounded by problems of poor effluent quality discharge and the problems arising from de-facto indirect wastewater reuse, the energy cost and/or carbon footprint of proper wastewater treatment is of negligible national concern.

From the perspective of water services authorities, who pay for electricity to run wastewater treatment plants, the situation is somewhat different. Here it would be prudent to adopt energy efficient design and minimisation of energy losses, optimisation of aeration, power factor corrections, etc. Still, these considerations are subject to maintaining the right effluent quality. As such, the energy demand of existing systems should not drive changes towards alternative systems.

If water services authorities resort to desalination, for fresh water supply, the consideration of the overall energy demand changes yet again. Fenu et al. (2010) show that membrane bioreactors (0.64 kWh/m^3) typically require double the electrical energy as compared to conventional activated sludge plants (0.3 kWh/m^3). For the reverse osmosis of sea water, the electricity requirement is an order of magnitude higher, up to 10 kWh/m^3. Energy demand should be an important driver for change when water recycle and ultimately water desalination are being considered as part of strategic planning.

1.1.5 Emerging pollutants

The endocrine system of vertebrates consists of the glands that secrete hormones directly into the bloodstream. Hormones carry information that regulates body functions, such as metabolism, tissue

growth, sleep and reproduction. Endocrine disrupting chemicals (EDC) is an umbrella term that refers to external substances that enter and alter the endocrine system. EDCs mimic hormones and can therefore interact with the hormones' receptors. A large number of different types and increasing quantities of EDCs have been released into the environment since the 1950s, coupled with repaid industrialization and the proliferation of new chemical compounds. A common effect of endocrine disruption, through widespread distribution of estrogenic compounds, has been described as the "feminization of nature" (Cadbury, 1999). An emerging theme is "developmental feminization", which considers disruption and change at a structural or functional level that is caused by exposure to estrogenic compounds during embryonic or foetal life (McLachlan, 2001). Furthermore, the effects of EDCs during development can be transferred over generations, from mother to child, due to the persistence of EDCs in body fat which is mobilised during pregnancy and lactation (Colborn et al., 1993). Many EDCs enter the wastewater stream, through human excretion (especially rest products of oral contraceptives) and medicine rests. Activated sludge treatment processes are not designed specifically for removal or conversion of EDCs, but it has been found that normal operating condition could remove 85% of certain EDCs, such as estradiol, estriol and ethinylestradiol (Johnson and Sumpter, 2001). Other EDCs may be more persistent and not removed as well, or at all. Even so, the concentrations at which EDCs pose a risk are disputed. Classical risk assessment methodology relies on linear extrapolation that uses high dose testing to predict responses to doses at environmental concentrations. However, EDCs with estrogenic activity can be active at extremely low background concentrations (Welshons et al., 2003).

Other emerging pollutants include engineered nano-materials (e.g. Farre et al., 2009), heavy metals (also known as the "toxic" metals, such as mercury, lead, zinc, copper, chromium and arsenic) and persistent organic pollutants, referred to as POPs (such as chlorinated pesticides aldrin, dieldrin, dichlorodiphenyltrichloroethane (DDT), endrin, heptachlor, hexachlorobenzene, mirex and toxaphene) produced by synthetic chemical manufacturing or as by-products of industrial processes. These chemicals can act as endocrine disruptors, cause harm to the reproductive system and impair the immune system. Widespread pollution over the past decades has resulted in elevated concentrations in marine environments, with harmful effects being seen, for instance, on marine mammals (Mouton and Botha, 2012). This array of emerging pollutants, irrespective of their toxicity, ubiquity or impact, don not originate from human excretion, and is therefore not topical within the context of this report and not considered further in any detail.

Greater demand for fresh water will increase pressure to recycle domestic wastewater, which means that pollutants are re-introduced and caught up in the recycle stream. With greater exposure to direct recycle, consumers should drive change towards systems that deal more effectively with emerging pollutants, especially organic endocrine disrupting chemicals.

1.1.6 Urbanisation

Global population growth is forecast to take place almost exclusively in cities of the developing world. Many such cities do not have proper sanitation systems yet and struggle to manage the growth of its established population, compounded by an inbound migration of people. Almost half the world's urban population live in slums (UN Habitat). In total more than 1 billion people the world over live in slums.

Much of the American and European cities of the 19th century looked like the slums of the modern day mega-cities in the developing world, with their notorious un-sanitary conditions. For example, in Liverpool during the 1830s one third of the population lived in cellars with earth floors (Ashton, 2006),

but slum clearance programmes of the 1960s and 1970s have all but demolished these areas. Vale (2007) sketches a similar picture of New York city in the 1930s. In Paris, slums existed throughout the city until it was demolished and cleared in the 1970s. Asian cities like Singapore still had 130,000 people living in squalid and un-sanitary *attap kampungs* (slums) during the 1950s. The public housing authority of Singapore constructed new buildings, along a master plan for a new city development, to which slum dwellers gradually moved. Today, Western Europe is not associated with slums and informal settlements, while Singapore has become a model of urban water management and water sensitive design.

The clearing of slum areas and improvement of living conditions seem to have always had two indispensable ingredients: economic growth and proper planning. However, while there is economic growth in the developing world, it's not immediately clear what kind of planning must be adopted. The problems associated with urbanisation and slums are of a much bigger scale than before. An important question, that no-doubt occupies the minds of city planners, is thus: how should we plan, for when, and what should that future look like. Based on the changes that occurred in the past decades, current planning cannot mimic that of Western Europe, America or isolated parts of Asia. Indeed, even the developed countries are asking how they should rehabilitate ageing and compromised infrastructure. More and more there is a sense that at the end of its life, existing water and wastewater systems could be replaced by alternative systems.

1.1.7 Economic considerations

The economy can be a driver for change, but could also be the result and reflection of system changes driven for social and political reasons. Consider the example of phosphate recovery from wastewater in a town, or community. If 100,000 people are using a system that recovers nutrients directly (e.g. source separation of urine and faeces) which yields 2 gram of phosphorus per person per day, and if this is converted into a phosphate based fertilizer, at average prices (R6,000 per ton P_2O_5), then a maximum sales value of R75,000 per month may be possible. Considering the effort and cost of collection, transport, storage, processing and marketing, the potential income does not promise a booming business. The problem with this analysis is the assumption that the free market correctly determines the price of finite resources through levels of supply and demand. However, there is a lack of complete information. Questions regarding the true value of nutrients, of energy, of water, of fertile land, or even: of pristine land … are mostly answered by speculation. Governments try to regulate the economy to limit inflation and prevent un-sustainable growth due to speculation. An equally important function of regulation is stimulation of other sectors of the economy, which includes political motives (such as protection of labour, distribution of wealth, etc.). The simple example above illustrates that the free market may not yet be driving change for phosphate recovery (because it's far more "economical" to mine rock phosphate on large scale). However, a longer term vision should drive policy change, not only to preserve finite resources, but to stimulate new sectors within the economy. South African examples of this kind of stimulation are the *Working for wetlands* and *Working for water* programmes, which employs unskilled and unemployed youth to restore and preserve freshwater ecosystems, which improves normal river runoff. In a sense then, economic considerations are not inherent drivers for change. Rather, system changes can, and should, drive an improved overall economic system.

Life cycle costing determines the true value of a system, which is far beyond capital cost. A methodology for determining the net present value and the internal rate of return for water systems were documented by van Vuuren and van Dijk (2006). In this methodology, life cycle costing of an

infrastructure system can be determined, where operation and maintenance are important cost contributors. For pumping systems, Anderson (2012) shows that 85% of the life cycle cost can be incurred during operation. However, very little generic or actual overview data is available on Life Cycle Costing of South African wastewater infrastructure. Still, if a 100 Mℓ/d treatment plant, which typically cost R1,000 million to construct, runs at R1.50/m^3 (ERWAT, 2010, escalated) for up to 30 years, then around 63% of the life cycle cost is incurred during the operational phase. The cost of planning and design of wastewater infrastructure is less than 5% of the life cycle cost.

Two important conclusions should be drawn here. Firstly, the importance of proper planning and design cannot be overemphasised. Small improvements have effects year in and year out during operation, and more of the life cycle cost should be invested in this phase. Especially since step changes in a system can result in massive economic effects (either positive, or negative). Fear of error may therefore prevent serious consideration of step changes, and result in the selection of business as usual. Secondly, if the cost of operation already contributes the bulk of life cycle costing, then in future, with increasing costs of resources and energy, the operational cost can only increase relative to existing systems. That is, unless the system is changed in fundamental ways through new concepts, developments and designs.

1.1.8 Socio-political response

For systems changes to occur, administrators and politicians need a strong mandate, which can only be issued by the communities they serve. This mandate can be formulated by the governing bodies that rule society, often because complex issues are at play that require translation of highly technical information into policy briefs, but this must still be endorsed by the population. As such, no changes can come about without political will. Furthermore, administrative ownership is required to ensure the effective implementation of policies.

Sanitation is not only of communal concern, but it can be a private and sensitive issue. Hence, the motto: “Water is Life – Sanitation is Dignity”. In South Africa, communities of informal settlements often aspire to flush toilets and water borne sewers. Changes to a perceived high level sanitation system can therefore easily be perceived as a lower level of service. Moreover, when this perceived lower level system is presented with urban renewal, or formalisation of informal settlements, the stigma of “second rate services for second rate citizens” is easily evoked. Presented correctly however, alternative sanitation systems are acceptable, which was the case in eThekwini Metro, where in 2011 just over 50% of 912,000 households in the greater Durban area had access to waterborne sanitation. At the same time, 92,000 households had been supplied with dry urine diversion toilets within only six years, as part of a planned programme that included extensive facilitation and training. The installation of urine diversion toilets will continue in eThekwini to eradicate the service backlog. Similarly, in Pook se Bos (City of Cape Town) a mobile sanitation system was introduced in 2007, which eventually that led to introduction of other dry no-mix toilets at Du Noon and Klipheuwel.

Despite the apparent success of these alternative systems that were eventually met with social acceptance, there may be a case for city wide no-mix systems. In other words, alternative sanitation techniques are not only for the informal settlements, the underdeveloped and the poor, but also for the redevelopment of existing neighbourhoods and town centres, for the sophisticated and the rich.

1.2 NO-MIX TECHNOLOGY AS PART OF WATER BORNE SANITATION

Sustainability considerations apply to the water cycle and demand that the nitrogen, phosphate and micro-pollutant loads to surface water be reduced to improve and conserve surface water quality. In view of the increasing water demand for food production and urban use, together with the aspirations of many to increase their quality of life, will increase the burden on water resources. To sustain this growth will require greater recycle and re-use of available water, which in turn demands better effluent quality of used waster. Could this be achieved through separation of urine at source?

Sanitation in the urban centres of South Africa involves a mixture of first world technology and a first class service, with squalor and complete lack of services. In most cases, the eradication of the service backlog is implemented as part of water borne sewer systems. By contrast, the eThekwini Local Government's response was to provide dry sanitation with urine diversion to semi-rural and per-urban areas. The sole aim of these urine diversion toilets is faecal desiccation for ease of operation during emptying. Although this is a great improvement on the old pit latrines, the diverted urine is not (yet) managed in this system and soaks into soil. Furthermore, irrespective of improved toilet facilities, untreated grey water discharge poses threats to health and surface water quality. The water related health or illness in communities is determined not only by the *quality* of bulk drinking water, but also by the *quantity* of water available. In areas with little access to drinking water, the same water is cascaded from one household purpose to the next, which results in poor overall water quality. Sufficient water supply inevitably results in high grey water production, which necessitates some kind of collection, removal and treatment system. With household water connections, flush toilets result almost automatically.

Sewers have been and will probably remain the most effective transport mechanism for household liquid waste. Sewers concentrate faecal matter to a single point source, where adequate treatment is not complicated. However, sewers dilute urine and to a large extent disperse nutrients and micro-pollutants in the receiving surface water. Is it then conceivable that not all liquid wastes be transported via sewers? The first principle of cleaner production and resource recovery applied in industrial waste management would be to prevent dilution of concentrated streams. Source separation of urine not only offers improved conservation of water quality but also water quantity. Source separation of urine with separate treatment and nutrient recovery could bring significant surface water benefits (quality improvement and quantity savings) and unlock greater wastewater treatment capacity at existing wastewater treatment plants, all aspects aligned with greater sustainability of urban water systems.

Where and how would urine separation systems be implemented? Radical thinking requires gradual change. A long term view therefore includes hybrids of existing and ideal systems. Introduction of urine separation systems could be viable, when considering the economies of scale:

- Public buildings (hospitals, libraries, prisons, markets, etc.), transport terminals (airports, train stations, bus and taxi ranks) and office blocks and commercial centres house many people where the toilet facilities are shared and used at greater frequency than household toilets.
- The cost of installation (or retrofitting) of no-mix toilets, fittings and pipes at these places could be insignificant in relation to total development cost, which is not the case with households.
- Volumes generated could be sufficiently large to justify collection by tanker truck, or treatment on site.

Again we emphasise that separate collection of urine, as part of a sewer catchment, is not aimed at the low cost housing developments, but should involve the well-to-do, the rich and the famous.

1.3 AIMS OF THE PROJECT

The project aims and objectives are listed below:

No.	Aim
1	Reconfiguring toilets and urinals to allow (partial) urine separation on pilot scale.
2	Determining the composition of urine from a random selection of professional people in South Africa (scientific staff at CSIR), as well as the composition of the remainder of wastewater, generated mostly during office hours.
4	Demonstrating the effectiveness of treating wastewater with less urine than normal in varying quantities to achieve very low nutrient effluent concentrations (DWA special authorisation), as well as relatively low salt effluent concentrations.
5	Treating un-diluted urine biologically in an aerobic/anoxic reactor, to stabilise liquid and nitrify ammonium to nitrite or nitrate, remove nitrogen and maintain a balanced pH.
6	Treating urine biologically with waste activated sludge, to nitrify ammonium to nitrate and denitrify the nitrate with the organic material that is released from the decay of biomass in waste activated sludge, while at the same time precipitating phosphate
7	Assessing operational issues, such as struvite and other forms of scaling in urine drains, odours, etc.
8	Understanding the additional processes coming into play other than those prevalent in conventional centralised activated sludge to the level of incorporation into mathematical modelling.
9	Creating awareness for the potential positive impacts of urine separation and the feasibility of its implementation.
10	Ultimately improve effluent water quality in southern Africa to improve the practices of direct and indirect water recycle.

2 LITERATURE AND TECHNOLOGY OVERVIEWS

2.1 COMPOSITION OF URINE

The composition of urine is reflective of human dietary consumption and therefore the main constituents of urine are urea, P, K, S, Ca, Cl and Mg as well as trace elements such as B, Cu, Zn, Mo, Fe, Co and Mn (Kirchmann and Pettersson, 1995; Udert et al., 2006; Maurer et al., 2006; Feng et al., 2007). Though the heavy metal content of human urine is relatively low, the concentration of elements such as copper, mercury, nickel and zinc are still 10-500 times higher than in surface waters (Kirchmann and Pettersson, 1995). Urine also contains high loads of pharmaceuticals and hormones excreted by the human body (Maurer et al., 2006).

Nitrogen

Nitrogen is the most abundant of all the elements in urine and can be found in different forms. Before urea hydrolysis, 85% of total nitrogen is present in as urea and 5% as total ammonia. After the process of hydrolysis, 90% of nitrogen is present as total ammonia (Udert et al., 2006). Stored urine may contain very small fractions of nitrite/nitrate-N.

Phosphate

Dissolved phosphate makes up for 95 -100% of the total phosphorous in urine however during storage of urine this figure is influenced by precipitation as large amounts of phosphorous can be fixed in precipitates (Ciba-Geigy, 1977; Udert et al., 2003c).

Potassium

Potassium is the third major nutrient in urine of which the concentration is not really affected by chemical alteration processes of urine. Potassium is known to be included in certain precipitates that form in stored urine, but this is not always common or only consumes a small fraction of potassium when it occurs (Udert et al., 2006)

Magnesium and calcium

Magnesium and calcium is present at a relatively low concentrations in fresh urine (Table 1) however, the dilution of urine with water can add additional amounts of these two elements (Maurer et al., 2006).In stored urine the majority of magnesium and calcium can be present in the form of precipitates (Udert et al., 2006).

Sulphur

Sulphate contains 90% of the total sulphur content of urine with compounds such as sulphuric acid esters and neutral sulphur compounds accounting for the rest of the sulphur content (Udert et al., 2006; Ciba-Geigy, 1977). In stored urine, where there is an absence of oxygen, nitrate or nitrite, sulphate is prone to reduction by microbial activity as it becomes the most favourable electron acceptor (Udert et al., 2006).

Table 1 Composition of urine from different sources of collection

Source		Fresh Urine	School		House-hold	Workplace		
Reference Literature		1	2	2	3	4	5	6
Dilution	(-)	0	0.33	0.33	0.75	0.26	#	1
pH	(-)	6.2	8.9	9	9.1	9	9	9.1
N_{tot} (TKN)	(g/m^3)	8830	2610	1795	3631	1793	-	9200
$NH_4^+ + NH_3$	(g/m^3)	463	2499	1691	3676	1720	4347	8100
$NO_3^- + NO_2^-$	(g/m^3)	-	0.07	0.06	<0.1	-	-	0
P_{tot}	(g/m^3)	800-2000	200	210	313	76	154	540
COD	(g/m^3)	2737	-	-	-	1650	6000	10000
K	(g/m^3)	1315	1150	875	1000	770	3284	2200
S	(g/m^3)	3450	175	225	331	98	273[b]	505[b]
Na	(g/m^3)	4970	938	982	1210	837	1495	2600
Cl	(g/m^3)	233	2235	2500	1768	1400	2112	3800
Ca	(g/m^3)	119	13.34	15.75	18	28	-	0
Mg	(g/m^3)	0.019	1.5	1.63	11.1	1	-	0
Mn	(g/m^3)	0.97	0	0	0.037	-	-	-
B	(g/m^3)		0.44	0.435	-	-	-	-

1: Ciba-Geigy (1977)

2: Kirchmann & Petterson (1995)

3: Jonsson et al. (1997)

4: Udert et al.(2003b)

5: Ronteltap et al. (2003)

6: Udert et al. (2006)

[a]Dilution factor as obtained from related referenceliterature by Maurer et al. (2006)

Dilution factor defined as $V_{urine}/[V_{urine} + V_{water}]$

[b]Sulfate-S(SO_4^{2-}S) value only.

Microbial urea hydrolysis

The nature of the urine collection system influences the composition of urine as the ratios of the constituents varies according to user group and can differ over time (Maurer et al., 2006). It is therefore difficult to establish exact formula for the composition of urine. The composition of fresh urine (i.e. urine immediately after leaving the body) also differs from that of stored urine in collection systems as certain chemical alterations occur due bacteria and dilution with water.

When urine is exposed to a non-sterile environment, the enzyme urease from ubiquitous bacteria catalyse hydrolyses of urea in urine to bicarbonate and ammonia (Udert et al., 2003; Maurer et al., 2006):

$$NH_2(CO)NH_2 + 3H_2O + CO_2 \rightarrow 2NH_4^+ + 2HCO_3 \quad \text{(Eq. 2.1)}$$

This reaction leads to an increase in the pH of the urine solution from 6 to a level of approximately 9. Udert et al. (2003b) found that urease-active bacteria, responsible for urea hydrolysis, grow mainly in the pipes of a collection system and get washed into the storage containers where the urea is then completely hydrolysed within only a few days. Udert also estimates that complete urea hydrolysis can occur in the pipes of a collection system under full flow conditions, thus resulting in a hydrolysis time of only a few minutes.

Mineral precipitation

The hydrolysis of urea is associated with a sharp increase in pH and together with the formation of ammonia and bicarbonate triggers the precipitation of mineral compounds such as struvite (MgNH4PO4•6H2O), hydroxyapatite ($Ca(PO4)_6(OH)_2$) and, if urine is diluted with water, calcite ($CaCO_3$) (Udert et al., 2003b/c). Struvite and hydroxyapatite are both phosphate minerals and after precipitation has commenced it can be found that 30% of phosphate in undiluted urine is in the form of precipitates (Udert et al., 2006). Only small amounts of urea (11-24%) have to be hydrolysed in order for a significant quantity of precipitation (87-97%) to occur. Calcium and magnesium addition to urine can enhance precipitation processes as well (Udert et al., 2006).

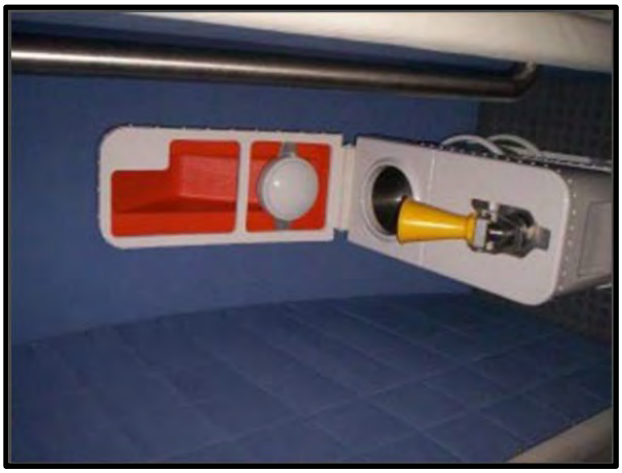
Ammonia volatilisation

After hydrolysis occurs, stored urine contains high concentrations of ammonia (Table 1). Ammonia is a volatile substance which results in urine in collection systems being an unstable solution of which any agitation can lead to the volatilisation of ammonia (Udert et al., 2006; Hellstrom and Johansson, 1999).

Despite the fact that NH_3 is highly soluble in water, minor losses of ammonia-N can occur in urine storage systems that are not closed or properly sealed. This can lead to odour problems and negatively affect human health and the environment (Galloway and Cowling, 2002; Udert et al., 2006). Up to 33% of the of ammonia in urine is volatile ammonia and volatilisation does become a bigger problem during transport or handling of urine as much greater ammonia losses and odour problems can occur than during storage (Udert et al., 2006). Ammonia volatilisation might be a possible characteristic to take into consideration during treatment of urine such as when the urine solution is aeration or stirred.

2.2 SEPARATE COLLECTION OF URINE

Table 2 Examples of no-mix toilets for every possible application



No-mix toilet designed for zero gravity

Once potty-trained, earthlings hardly ever contemplates using a toilet (… provided it's clean). Living in space changes this attitude fundamentally – a vacuum assisted toilet system collects faecal matter and urine, through separate receptacles, requiring some degree of concentration. The International Space Station is a confined space with limited resources, but necessity is the mother of invention: Faecal matter is wrapped, and returned to earth, and urine is recycled for the value of the water!

Wostman's original urine separation toilet

During the 1990s, groups in Sweden were driven by a national debate to recycle phosphate and prevent eutrophication of the Baltic sea, to start experimenting with separate collection of urine. This rudimentary design was used by EAWAG in Zurich as part of the Novaquatis project. An important design flaw is the low position of the urine basin, which intercepts high volumes of flush water that dilutes the separated urine. Cost around 600 Euro.
http://www.wost-man-ecology.se/

Dubletten urine separation toilet

Another Swedish no-mix toilet, with dual flush mechanism (1 l/fluh for urine and 5 l/flush for faeces). Leakage in the urine pipe creates bad odours, but leakage can be avoided by making the connections correctly. This toilet has a slightly longer shape and comes with a special child seat, positioning small children to the back of the toilet and preventing defecation in the front (urine) compartment. Cost around 700 Euro.
http://www.dubbletten.nu/

Gustavsberg urine separation toilet

Top of the range, high quality porcelain no-mix toilet, manufactured by a subsidiary of Villeroy & Boch. From the outside, this wall mounted unit looks identical to conventional flush toilets. Also equipped with a dual flush mechanism. Two of these toilets were imported from Sweden and installed at CSIR, Stellenbosch. Cost around 700 Euro.
http://www.gustavsberg.com/

Roediger low flush vacuum toilet with urine separation

Ultra-modern vacuum assisted low flush no-mix toilet. Vacuum toilets have been in use on boats, trains and aeroplanes for decades. The use of vacuum reduces water consumption drastically, while operating with smaller diameter pipes. Good for preventing dilution of separately collected urine, but the vacuum system is high technology, and is appropriate in limited cases only. Most expensive no-mix toilet at 1,080 Euro.
http://www.roevac.com/

Waterless urinals

Water conservation and demand management has seen the introduction of waterless urinals, especially in the Western Cape. This is an example of waterless urinals at a wine farm's tasting and sales room.

Traditional Chinese Urine Diversion Toilet

Squatting toilet for separate collection of urine and faeces. A ping-pong ball sits on the urine outlet and acts as an odour trap, while the faecal receptacle is closed by a lid. Ash is added to faecal matter (note bucket with small spade in background).

Modern Chinese Urine Diversion Toilet

Modern waterless no-mix toilet produced in China after Swedish-Chinese design collaboration, for use in multi-storey buildings and allows dry faeces collection and separate urine collection, as part of the Erdos-Eco Town Project. It is equipped with a flushing device for ash and a mecanism to seal off the drop shaft. Cost around 100 Euro.
http://www.meilongco.com/

Dry Urine Diversion Toilets – eThekwini

Dry urine diversion toilets, made of plastic, installed at more than 90,000 households by eThekwini Water and Sanitation. (Durban, South Africa). Based on a structured questionnaire using mobile phone technology, low levels of satisfaction were reported, with the facilities as well as perceived smell in the toilets and malfunctioning of the pedestal, and low use of these toilets when a pit latrine is present in the dwelling perimeter. Educational and promotional activities must stress the economic return derived from reusing urine and excreta in urban agriculture (Roma et al., 2013).

2.3 PROCESS TECHNOLOGY FOR TREATMENT OF SOURCE SEPARATED URINE

Different process technologies have been proposed for treatment of source separated urine. These technologies range from physical, chemical to more-or-less conventional microbiological processes, and the choice of technology derives from the rationale for treatment:

- Recovery of fresh water from urine is the prime driver for urine process technology in the International Space Station. Every litre of fresh water ferried by a rocket is one extra kilogram, which also has to be brought back to limit orbital debris. This makes water more expensive than the solar energy produced on-board. The “package plant” shown in Figure 8 treats urine in a series of steps, including precipitation of salts, and a centrifuge-distiller. Without gravity, normal distillation would not function as on earth, which is why the distiller is spun to produce an artificial gravity field. The distillate is further filtered, and 93% of water is recovered for drinking. (http://www.nasa.gov/mission_pages/station/behindscenes/waterrecycler.html)
- Distillation of urine on earth would drive off (evaporate) ammonia gas, which could be re-dissolved in concentrated sulphuric acid, for further use in ammonium-sulphate fertiliser production. This technology assumes an industrial process further down the value chain that can take on the ammonium-sulphate and beneficiate this further.
- Ammonia can be recovered through adsorption on natural zeolites, such as clinoptilolite. Ammonia may also be recovered through ion exchange mechanisms.
- Freeze drying concentrates the nutrients in urine in a smaller volume still, which reduces transport costs.

- Urine can be sprayed directly onto land as a fertilizer, although the high salt concentration of urine has come under scrutiny for salinization of soil.
- Phosphate recovery through precipitation of struvite ($MgNH_4PO_4.6H_2O$), or calcium apatite ($Ca_3(PO_4)_2$) has been widely established as possible techniques.

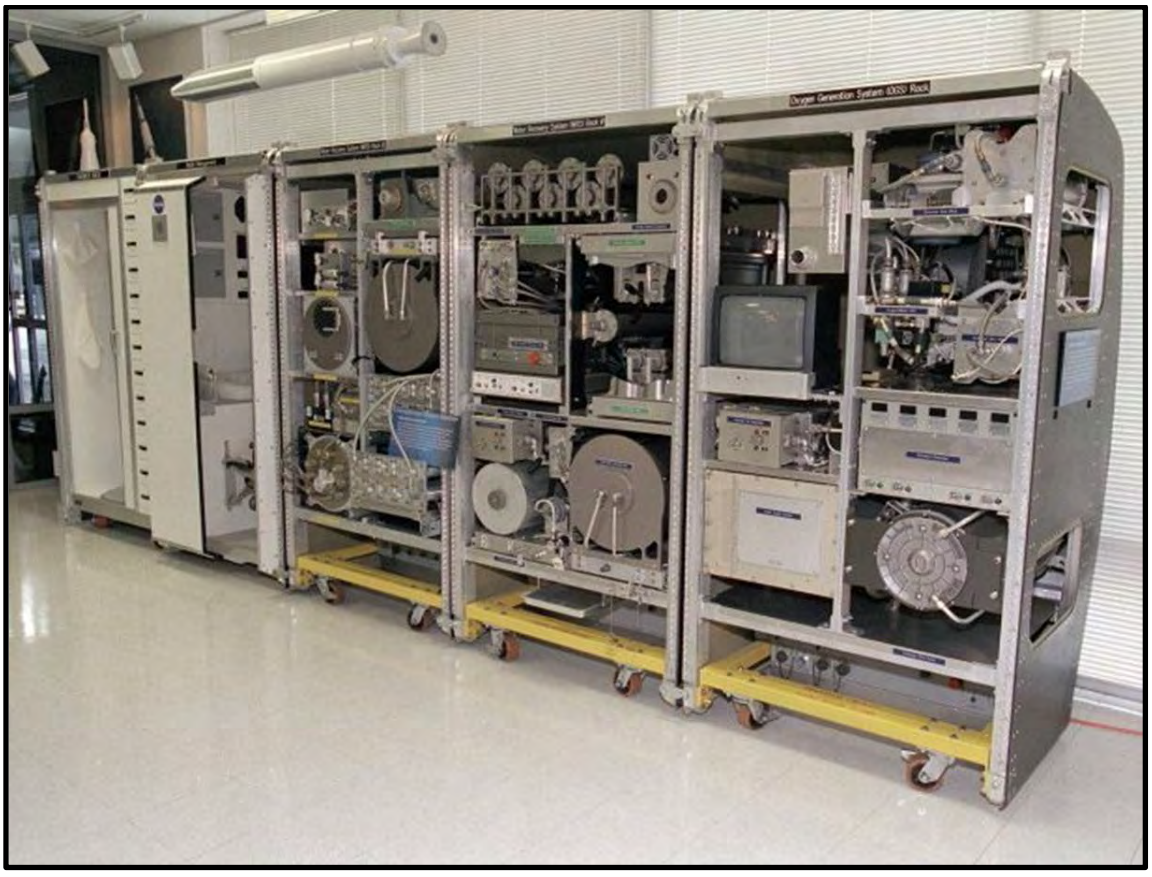


Figure 8 Treatment plant installed in the International Space Station for the complete recovery of fresh water from urine[3]

This project focussed on the potential impact that source separation can have on wastewater treatment. The primary driver is thus the improvement of surface water quality, with the ultimate aim of recycle. Therefore, the recovery of nutrients from urine at this stage is of secondary importance. We would here argue that a simple system that allows better mainstream wastewater treatment is more important than a complicated system for nutrient recovery, even though this may ultimately be a secondary aim. Therefore, amongst the many possible treatment processes, this project would focus on biological treatment of urine, which would be fairly simple to implement, and which could be easily integrated with existing biological treatment works. In a radical programme, all challenges should rather not be dealt with at the same time. The challenge would remain effective separate collection in the bathroom, storage and transport. Some of the novel biological processes that allow gradual integration of wastewater treatment with treatment of source separated urine, based on the nitrogen removal pathways shown in Figure 9, are discussed below:

- Bio-Augmentation Batch Enhancement (BABE) is a fairly simple process unit that grows nitrifying organisms in a side stream reactor, typically in (part of) the return activated sludge stream, with a highly concentrated nitrogen load. Without much competition, and with longer

[3] Photograph used by permission of NASA

residence time, nitrifiers grow outside of the main process parameters, after which they are introduced into the main process to “mop up” remaining ammonium (Salem et al., 2003).

- The Single reactor High activity Ammonium Removal over Nitrite (SHARON) has been implemented successfully on full scale. This process unit works for concentrated waste streams at temperatures above 30^{o}C, where the specific growth rate of ammonium oxidising bacteria is markedly higher than that of nitrite oxidising bacteria. Furthermore, the sludge age is limited (typically one day) so hat ammonium oxidisers can grow, but nitrite oxidisers are effectively washed out. The absence of nitrite oxidisers prevents production of nitrate, and therefore a “short-cut” for biological nitrogen removal. An external carbon source is then dosed into the batch to reduce nitrite to nitrogen gas, through which alakilinity consumed in the nitrification process is restored. Overall, less oxygen is consumed in the aerobic nitrification process, and less COD is consumed in the anoxic denitrification process, compared to conventional nitrification and denitrification.

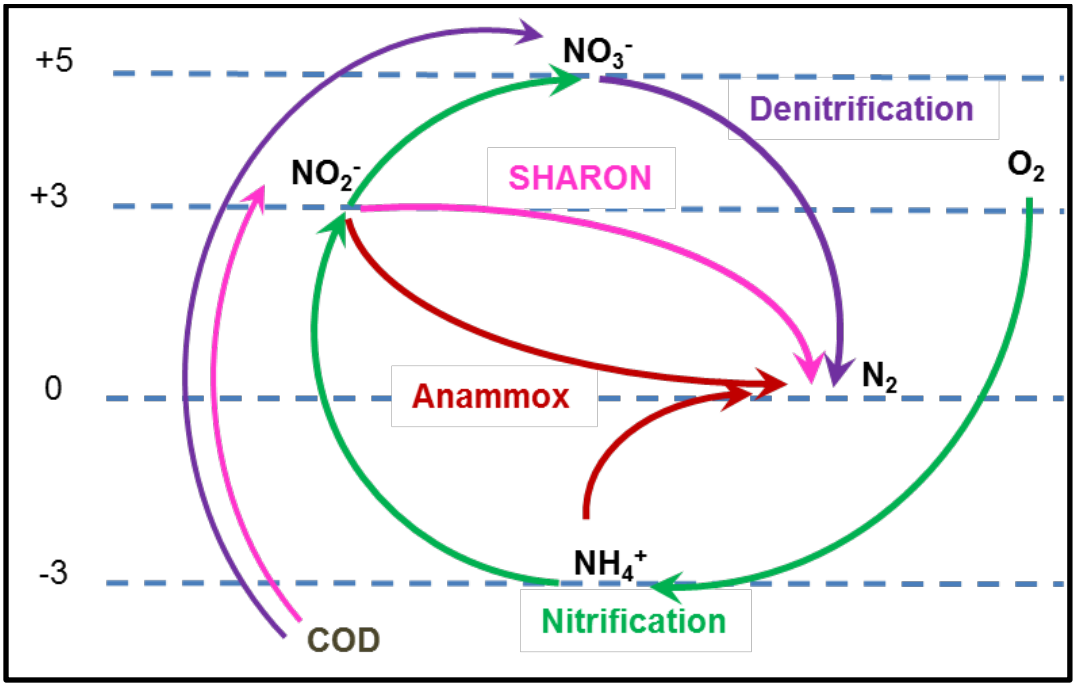


Figure 9 Nitrogen removal pathways; Conventional nitrification and denitrification, SHARON, and combined SHARON-Anammox, in relation to redox levels.

- Anaerobic Ammonium Oxidation was discovered in the 1990s and has since been commercialised in full scale granular sludge reactors. The anammox bacteria are completely autotrophic (no COD is required for nitrogen removal) and obtain their energy from ammonium, like normal ammonium oxidising bacteria, but with nitrite as the final electron acceptor. While conventional nitrification and denitrification require 13 moll electrons per moll nitrogen removed, the anmmox process requires only 3 moll electrons per moll of nitrogen removed. This efficiency of the biochemical anammox pathway translates directly into full scale process efficiency. Anammox bacteria were recently found to play a significant role in the nitrogen removal process within the old trickling filters at Daspoort wastewater treatment works (Wilsenach et al., 2013). The combined SHARON-Anammox process requires nitrification of 50% of ammonium to nitrite only, and then routing the ammonium-nitrite stream through an Anammox reactor. Wilsenach (2006) has shown that this process could be employed to remove nitrogen from source separated urine.

3 RETROFITTING NO-MIX TOILETS AT CSIR, STELLENBOSCH

3.1 NO-MIX TOILETS

The male and female restrooms at office block D at CSIR in Stellenbosch were fitted with source separating (no-mix) toilets. The waste streams were diverted to separate onsite storage containers, one for brown water from ladies' and gents' toilets combined, one for male urine and one for female urine. Figure 10 shows one of the two no-mix toilets, ordered from Villeroy & Boch (Gustavsberg, Sweden), after installation at CSIR. The old floor mounted toilets were broken out and discarded, and floor tiles were found to match the existing. Urine is separated in the front compartment of the toilet and flows down the metal pipe (Figure 11). The urine collected from the no-mix toilets contains flush water also, and its chemical concentrations are therefore diluted and its mineral composition altered by the chemicals contained in potable water. Figure 12 shows the cast iron pipes from the old toilets and the new PVC pipes after installing the no-mix toilets.

Figure 10 No-Mix toilets installed in the men's and ladies bathrooms on the 1st floor, with an electronic counter installed to determine the number of toilet users for per person load calculations.

Figure 11 Stainless steel urine drain from men's no mix toilet installed at CSIR. Notice the slight bend in the pipe which acts as an odour trap.

Figure 12 Retrofitting pipe work sections to accommodate collection of brown water and yellow water. Note old cast iron drains and new PVC pipe.

Figure 13 Overhead view of the No-Mix toilet showing a front compartment for urine separation and the back compartment for faecal waste as for standard toilets.

3.2 WATERLESS URINALS

The male urinals were also retrofitted so that urine was diverted to a fourth onsite container. The urine from the three containers were analysed individually. Figure 14 shows waterless urinals in the bathrooms CSIR which have been retrofitted. Urinals were retrofitted with one-way plastic valves and operated without flush water. They were also cleaned without the use of detergents to allow accurate chemical analysis of undiluted urine. No urine traps are needed since the valves close immediately after the urine passes through, thus preventing any bad odours.

3.3 GREY WATER COLLECTION

The effluent pipe from the kitchen sink was also retrofitted to a 50 L collection tank (Figure 15) so that composite samples were taken and sent for chemical analysis. The grey water volume was also recorded twice daily, and from this information hydraulic loads of grey water were calculated. Collected grey water was decanted into 150 L drums and transported together with brown water and urine to UCT Water Laboratory, to make up feed solutions for the mixed wastewater reactors.

Figure 14 **Urinals retrofitted with one-way waste valves which allow the urine to pass and prevent odours from escaping.**

Figure 15 **Grey water collection tank and pipe layout showing the inlet pipe, discharge line, overflow, and sampling point**

3.4 COLLECTION AND STORAGE

The urine from both no-mix toilets and from three gent urinals were re-directed to collection tanks. This outside area was well ventilated, and also enclosed by a low brick wall to allow wash-down water to flow into a drain connected to the sewer system. This area was purposely built to isolate it from the adjacent storm water drain to avoid wastewater contaminating the storm water.

Figure 16 Waste stream collection and storage area in the courtyard outside the CSIR office block, with brown water collection tank and blue macerator pump line seen top left (left) and four urine collection tanks 40 l ladies' no-mix urine, 25 l men's urinals and 25 l men's no-mix urine, plus overflow to sewer (right)

The collection site (Figure 16) was the most suited because of its availability and close proximity to the office toilets. Although all the collection tanks were sealed tightly, the overflow connections were not well sealed because the urine tanks had to be lifted and emptied weekly. This meant that some ammonia vapours were escaping through the overflow connection points in both the gent urinal and gents no-mix urine tanks. This design error was compounded by hot summer temperatures (40°C ±) which aggravated complaints about bad urine/ammonia smells coming from the collection site.

It is obvious that a tiled floor is easier to clean than a cement floor. Therefore, an addition building feature for the collection site should include a ceramic tiled floor. This would be far easier to clean and remove bad odours caused by spillages of urine or brown water during sampling.

3.5 COST OF MATERIAL AND INSTALLATION

The capital and labour costs for purchasing and installation of no-mix toilets, waterless urinals, pipes and fittings, collection tanks and re-routing the kitchen grey water was in the order of R45,000. The labour work included minor construction of a waste collection site and installation (including fitting, tiling, welding tank stands, and electrical connection for the grinder pump). The major cost was for purchasing and transporting the two no-mix toilets and amounted to R16,479, of which the toilets cost around R3,800 each. At the time, a good quality local toilet would cost around R1,500. The cost of three 1-way valves (for retrofit in urinals) was only R1,094. The plumbing work entailed replacing the old toilets with the new no-mix toilets and laying new waste pipes to direct the separated waste streams to outside collection tanks, which amounted to R8,832. Much of the material needed for the collection tanks were found on site, and the cost is not included. The 3-phase grinder pump, float switch and electronic counting devices expenditure was R7,467, R266, R655 respectively. A 20 m 3-phase extension and control box for the grinder pump cost R 8,000, and the welding work necessary to reinforce the metal stand of the brown water tank came to R1,926. The time cost associated with the planning, design and construction supervision of the system is more difficult to quantify, but would have been in the order of R75,000. Motivating the project internally and obtaining all the relevant permissions has been the most significant time cost.

3.6 PERCEPTION SURVEY

The aim of the perception survey or questionnaire was to see how the users' perceptions changed over the duration of this project, from the time they first used the toilet and again later after they were more accustomed to the no-mix toilets. This would give one some expectation of people's reaction using these toilets for the first time in public places, should this technology ever reach the implementation and commercialisation stages. There were mainly young professional researchers and analytical staff in CSIR's employment using these toilets. Therefore, this survey was slightly biased – the users were scientifically-minded people who were aware of the environmental impacts of pollution caused by poorly treated wastewater. If this study was conducted in a public place, then it can be fair to assume that their answers would be very different with regard to the technical questions. However their answers would be similar to the more general questions about the actual use of the toilet, and comparison of the new toilet to a normal toilet.

The questionnaire was designed to collect anonymous information to add an element of privacy for the participants, and also to facilitate an open channel for the users to express themselves without feeling embarrassed. However, strangely enough all participants used their real name instead of their pseudonym. This indicated that they were not shy to express themselves and that they clearly understood the reasons and benefits driving this technology. The cleaners were also asked some questions to establish if they encountered any problems with cleaning the no-mix toilets.

Two sets of questionnaires were distributed. The first questionnaire was distributed and completed in January 2009 while the toilets were being installed. And the second questionnaire was completed in October 2009 after the no-mix toilets had been used for several months. For complete results, please refer Addendum A. Most participants were indifferent or nervous at first because this was the first time they had ever seen or used a no-mix toilet. However, after their initial fears were overcome they reported that these toilets were no different to the standard toilet and were just as comfortable. They also stated that this technology was a step in the right direction, and that it had great potential for improving effluent quality through the source separation and treatment of urine. The results from the

second surveys indicate how users' perceptions change with time, towards a more accepting attitude of this new and innovative technology. Provided it has been proven safe, many participants were comfortable about the idea of using nutrients recovered from urine as a raw fertiliser material for growing agricultural or industrial crops.

Figure 17 Toilets complaints and complements box situated outside the gents and ladies toilets where the new No-Mix toilets have been installed.

The conclusions from these questionnaires are summarised below. It should be noted that these are the views of researchers and that the public might have different opinions. However, most of the challenges of implementing new technology and sustained behavioural change may be overcome.

- More than half the participants answered that no-mix technology improves wastewater treatment by separating urine and hence removing the nutrients.
- All participants were aware of additional benefits such as decreased treatment cost for wastewater, and a cleaner environment.
- The majority of participants (80%) were happy with the no-mix toilets.
- Most of the participants (73%) acknowledged that they were indifferent, curious, or had no expectations of the no-mix toilet.
- Only the women experienced problems as a result of toilet paper blocking the metal urine pipe after flushing. It was suggested that the toilet design could be improved by adding a cover or sieve to prevent this from occurring.
- Two thirds of the participants answered that a high percentage of urine was successfully separated. The remaining one third said they were not sure and would have to wait and see the laboratory results. Initial results suggest that these toilets do separate urine reasonably well, but there is still room for improvement.
- The majority of participants said that educating, informing and explaining to new users about the no-mix toilet is essential for them to understand the logic and reasons behind urine separation

technology. This will assist them in overcoming their initial fear and nervousness which might make them otherwise reluctant to use the no-mix toilets.

- All participants agreed that it was a good idea to install the no-mix toilets in high traffic buildings. They also said that they would move into an apartment with no-mix toilets installed.
- When asked whether source-separated urine could be used as a fertilizer, there was no decisive response. Only 30% answered NO correctly, because of the high salt content in urine. But there are other concerns too, such as pharmaceuticals, hormones, and PCP products.
- The recovery of nutrients from source-separated urine will need to go through the proper stages of testing to deem such waste based fertilisers safe for use in the agricultural industry. But it is encouraging to see that at this preliminary stage in the study, its use and potential is largely accepted.

No samples were taken from individuals and therefore ethical clearance was sought for this study.

There was also only one written complaint captured from a no-mix toilet user who wrote: “it’s like peeing in a test tube”. This complaint came about at the time when access to the second (standard) toilet in each bathroom was denied to encourage the CSIR staff to use the no-mix toilet only, therefore “restricting their ability to choose” which toilet to use. This was only done on specific days when the COD, N and P loads were being determined. From the questionnaires, another valid comment was that “everyone seemed interested and eager to try it (the no-mix toilet), but won’t be prepared to pay much extra to have a no-mix system installed at our home.”

4 COMPOSITION OF URINE AND OTHER STREAMS

4.1 WASTEWATER ANALYSIS

Wastewaters from different no-mix toilets, waterless urinals, and reconfigured grey water systems were collected (as shown in Chapter 3) and analyzed for the list of chemical components normally found in wastewater. The different streams consisted (as shown in Figure 17) brown water, which is toilet paper and faecal matter from no-mix toilets, grey water from the office kitchenette, yellow water from the ladies' no-mix toilet, yellow water from the men's no-mix toilet and undiluted men's urine from waterless urinals.

Figure 18 Typical samples of separately collected wastewater streams

Samples were analysed at CSIR's Stellenbosch water laboratory (Figure 18).

Free and Saline Ammonia (FSA) – SMWW Method 4500-NH3 C

FSA concentrations were determined using a titrimetric method. The method involves converting all saline ammonia (NH_4^+) in a sample to free ammonia (NH3) then steam distilling it into a indicating boric acid (H_3BO_3) solution which turns from purple to green as the ammonia is absorbed. The boric acid solution is then titrated with a sulphuric acid solution. The boric acid solution turns back to purple and the endpoint of the titration is reached when the purple solution stops gaining intensity in colour. The concentration of ammonia-N in the solution can then be calculated from the volume of sulphuric acid titrated.

Total Kjeldahl Nitrogen (TKN) – SMWW Method 4500-Norg B

Amino Nitrogen of organic materials is converted to ammonium in the presence of sulphuric acid, potassium sulphate and copper sulphate catalyst. The free ammonia is also converted to ammonium. After base addition the ammonia is distilled from the alkaline medium and absorbed in boric / sulphuric acid. The ammonia is then determined by titration with a standard mineral acid.

Nitrite and Nitrate – SMWW Method 4500-NO3 H

Nitrite and Nitrate concentrations were determined by means of an automated colorimetric method facilitated by an auto analyser. A sample is inserted into the instrument wherein all nitrate-Nis reduced to nitrite-N by hydrazine. The nitrite then reacts with an added colouring reagent to form a coloured solution. The colour intensity of the solution is determined by the concentration of nitrite present. The solution is passed through a spectrophotometer which measures the intensity of the coloured solution.

The result is compared to pre-established nitrite-N standards from which the concentration of nitrite-N in the sample is determined. The auto analyser also conducts a parallel operation on the same sample during which the reduction step is omitted. This only measures the original amount of nitrite-N present in the sample and no nitrate-N reduced to nitrite-N. The concentration of nitrate-N in the sample can be determined by calculating the difference in obtained nitrite-N values.

Chemical Oxygen Demand (COD) – SMWW Method 5220 C

The COD concentrations were determined using a closed reflux, titrimetric method. The method involves the digestion of a sample after adding K2Cr2O7 and H2SO4 and then titrating with Fe(SO4)2(NH4). The endpoint of the titration is reached when the solution changes colour from bright yellow to dark brown. The endpoint is sudden and clear which gives a close to exact result. Errors in the result may be contributed to the step in which H2SO4 is added to a sample as a rapid reaction can lead to the sudden formation of vapours which may escape from the flask. Precaution is taken by adding the acid slowly so as to ensure minimum vapour losses.

Ortho-Phosphate (Ortho-P) – SMWW Method 4500-P C

Ortho-P concentrations were determined by means of a colometric method. A colouring reagent is added to the sample which reacts with ortho-P to develop a coloured solution. The intensity of the colour is determined by the ortho-P concentration in the sample. The sample is inserted in to a spectrophotometer which determines the absorbance of the coloured solution. The absorbance value of the sample is compared to pre-established standards from which the concentration ortho-P in the sample is be calculated.

Figure 19 Analytical methods for measuring chemical oxygen demand via acid fermentation (left) and ICP optical emission spectrometer for Ca, Mg, K and Na (right)

Alkalinity

The alkalinity was determined with a 5 point titration method (Moosbrugger et al., 1992). The sample is titrated with acid to several end points. The volume titrated at each end point is used to calculated the true alkalinity of a sample. This method differs from a standard titration in that it has various endpoints with which the alkalinity is calculated.

Total Dissolved Salts (TDS)

Total dissolved solids were determined gravimetrically when a well-mixed sample is filtered into a pre-weighed dish. The filtrate is evaporated and dried to constant weight at 180ºC (Standard Method).The measurement of pH and Electrical Conductivity are determined electrochemically using standard methods (Standard Method). Alkalinity of water is its acid-neutralizing capacity. A known sample

volume is titrated against a standard acid to a predefined pH endpoint (4.5) and the resultant volume used to calculate the sum of all titrable bases (Standard Method).

Calcium, Sodium, Magnesium & Potassium were analysed using the Inductively Coupled Plasma Optical Emission Spectroscopy (ICP OES)

This spectroscopic method employs an Argon plasma gas generated by radio frequency and magnetic flux producing extremely high temperatures which atomizes the sample completely and its constituent atoms are detected by photomultipliers. The analysis consists of running a set of calibration standards with suitable QC solutions followed by the samples. These concentrations will be expressed in ppm (mg/L.)

Phosphate were determined using the Flow Injection Analyzer (FIA)

This colorimetric technique is based on the principle of the addition of reagents specific to the analyte which results in colour formation. The absorbance measured at the specific wavelength for the particular colour is thus proportional to the analyte concentration. The analysis consists of running a set of calibration standards with suitable QC solutions followed by the samples. These concentrations will be expressed in ppm (mg/L.)

Chloride and Sulphate was determined using an Auto Analyser

This is a colorimetric technique and is based on the same principle as Phosphate. For Sulphate determination the analytical method used was an Auto analyser (turbidimetric technique). This is based on the addition of a chemical reagent which reacts with the SO43- and forms a precipitate, and the measure of turbidity is thus proportional to the concentration of SO43- . The analysis consists of running a set of calibration standards with suitable QC solutions followed by the samples. These concentrations will be expressed in ppm (mg/L.)

4.2 ANALYSIS OF URINE

Average values for chemicals in urine, collected from three different streams at CSIR, are shown in Table 3. The data from urine collected from waterless urinals, on a continuous basis, is divided further into three sets. The first set comprises of all 50 samples, the second set contains the first four samples (n=4) only which represent samples of urine collected and analyzed while the piping and tank systems were relatively clean. The third set of data (n=46) contains all samples after these initial four samples, after which time it is evident that chemical changes were taking place in the stored urine. This last set only is shown in Table 3. On average there were 10 men using these urinals at CSIR, and about of 4.5 ℓ/d of urine was collected daily (0.45 ℓ/ p.d). By comparison, the average amount of urine produced is around 1.2 ℓ/d per person, which of course also includes urine passed at households and public toilets. The results are compared with data from Ciba Geigy and NASA in Table 4.

The chemical changes occurring in stored urine were also investigated in a separate study, which lead to data supporting the occurrence of urea hydrolysis and struvite precipitation (see Charney Anderson thesis). These bio-chemical processes occurred spontaneously only after one to two weeks from the time of commissioning the urine diversion and collection system, which provided ample time for biofilms to develop inside the newly fitted pipes and collection tanks.

Potassium and sodium

CSIR's urine data in general displays a very wide standard deviation. However, overall K and Na show the lowest deviation relative their average values (2xSTDEV/average) with values of 30% and 26%. This observation suggests that because K and Na salt concentrations are the most consistent, also verified by data from the township, they are good proxies to use when analyzing urine data for trend determinations. Also, there was hardly any change in the K and Na concentrations when comparing the first and last sample batches of urine from gents urinals (n=4 to n=46), which indicates that K and Na salt concentrations are quite constant in fresh and stored urine. After K and Na, parameters with the next lowest deviations are Cl, TDS and EC with values of 30%, 34% and 37% respectively. This is not surprising since the total dissolved solids and electric conductivity are both directly proportional to concentrations of salts and dissolved particles.

Table 3 Average composition of urine collected from office block at CSIR

Source		**Male No-Mix Toilets**	**Female No-Mix Toilets**	**Urinals**
Samples	**(-)**	**8**	**8**	**46**
K	(mg/l)	545.6	431.9	1866.8
Na	(mg/l)	619	524.4	2141.7
Ca	(mg/l)	22.6	20.4	30.3
Mg	(mg/l)	0.8	1.9	3
NH_4^+	(mg/l)	1402.5	978.8	3705.5
Ortho phosphate	(mg/l)	93	62.6	249
Total P	(mg/l)	121.71	79.31	-
SO_4^-	(mg/l)	575.8	336.5	1284.2
Cl	(mg/l)	1111.5	704.4	4098
COD	(mg/l)	1890.6	1271.5	7719.9
TKN	(mg/l)	2113.2	1273.8	6603.5
TDS	(mg/l)	3865	2837.1	14161.5
Alkalinity	(mg $CaCO_3$/l)	5375.3	3688.3	18172.6
EC	(mS/m)	1273.8	903.1	3414.4
pH	(-)	9.1	8.9	9.1
DOC	(mg/l)	459	240.1	2587.9
DOC	(mgCOD/l)	1224	715.8	6901.1
[1] Only 6 samples tested				

Calcium, magnesium and urea hydrolysis

The progression from freshly sampled urine to that of stored urine can be seen in the decreasing trend of Mg and Ca concentrations, starting from NASA's data, then in CSIR first 4 samples, followed by CSIR's last 46 samples. For Ca the concentration trends are 186, 100, and 30 mg Ca/L and for Mg the concentrations are 104, 55, and 3 mg Mg/L respectively. Conversely, the decreasing Ca and Mg

concentrations are mirrored by increasing pH values of 6.2, 7.4 and 9.1. The question arises: what is the meaning of the decreasing Ca and Mg trends and the increasing pH values?

As fresh urine undergoes urea hydrolysis, urea is converted into ammonium and bicarbonate, the pH rises as additional alkalinity is produced. Ammonium produced is in equilibrium with aqueous ammonia, which in turn is in equilibrium with ammonia gas. The amount of ammonia released in the gas phase depends on a number of factors like the quantity of water the urine is diluted with, temperature and partial gas pressure.

$$NH_2(CO)NH_2 + 3H_2O \longrightarrow 2NH_4^+ + HCO_3^- + OH^- \quad \text{(Eq. 3.1)}$$

$$NH_4^+ + OH^- \longleftrightarrow NH_3\,(aq) + H_2O \quad \text{(Eq. 3.2)}$$

$$NH_3\,(aq) \longleftrightarrow NH_3\,(g) \quad \text{(Eq. 3.3)}$$

The increase in pH catalyzes the precipitation of Ca and Mg salts, hence their diminishing presence in the urine's liquid phase (Ca and Mg change from dissolved salts to precipitates).

Another interesting point to note in NASA's data is the low ammonium (NH_4^+) concentration of 581 mg NH_4^+-N/L. This verifies that most of the urea has not yet hydrolysed, since it is about 1/6 of the ammonium concentration in CSIR's stored urine.

Figure 20 Schott bottle filled with stored urine and precipitated salts on the bottom.

Phosphate and struvite

Phosphate concentrations in urine also decrease as Ca and Mg precipitate. NASA and CSIR n=50 data shows phosphate concentrations of 1120 and 255 mg PO_4-P/L respectively. This dbig difference most likely occurs as a result of struvite ($MgNH_4PO_4.6H_2O$) and hydroxyapatite ($Ca_5(PO_4)_3OH$) crystal formations when pH of stored urine exceeds 8.0. The most common form of struvite in stored urine is

magnesium ammonium phosphate (MAP), although struvite crystals in the form of potassium magnesium phosphate (KMP) are also possible but they generally form after all the ammonium is depleted.

4.3 FLUSHED URINE (MEN'S AND WOMEN'S NO-MIX TOILETS)

All elements and parameters measured in the gent's urine were significantly higher than that of the ladies no-mix urine. When comparing Na, TDS and EC, the relative increase was approximately 50%. It is doubtful that the higher gent's urine concentrations are diet related. This was not expected because both toilets are identical. A more plausible reason would be that more flush water is mixed with the women's urine, thus causing this observed dilution. This could be due to differences in toilet behavior between men and women, but it cannot be ruled out that the ladies' toilet just used more flush water (due to specific connection and installation). Volumes of flush water were measured during the commissioning phase and it was discovered that the no-mix toilet in the ladies bathroom discharged 7% and 22% more water into the urine collection tank for small and big flushes respectively (which were specified to be between 3 and 5 ℓ of water into the brown water tank, and 200-300 mℓ of water down the no-mix urine pipe together with the urine). These water volumes were measured before the toilets were actually used.

The Na and TDS concentrations in urinal urine and gent's no-mix urine were compared and the average dilution factor of 3.5 was calculated. Assuming the Na and TDS concentrations in undiluted women's urine is the same as men's urine, then the ladies urine is diluted by a factor of about 5.3 by flush water. Very low Mg concentrations also indicate that struvite is forming in both the ladies and gent's no-mix urine collection tanks, even though urine is diluted 3-5 times with flush water. Struvite precipitation occurs here due to the combined pre-condition of high Mg, phosphate, ammonium concentrations and high pH values of 8.8-9.0.

4.4 BOTTLE EXPERIMENTS FOR COMPLETE UREALYSIS

Comparisons of CSIR data sampled from continuously supplied urine collection tanks against samples from isolated urine stored in schott bottles showed that greater correlations were noticed between the substances in the schott bottle experiments. This observation was evident after examining the Pearson's tables generated from the data sets. The correlation outputs vary from 1 to -1. A correlation value of 1 indicates that the two components are directly proportional (or perfectly positively correlated). For example, the correlation value for magnesium versus calcium is 0.92 for stored urine (Table 4), which means that a decrease in magnesium concentration is associated with a linear decrease in the calcium concentration. Conversely, a correlation of -1 implies that the measured components exhibit an inverse relationship, e.g. with an increase in the calcium concentrations, there is a linear decrease in the chloride concentration (correlation value of -0.98, Table 4). When the Pearson's correlation approximates zero it means that the two components are not correlated, e.g. the Pearson value for TDS and magnesium is -0.06. This value indicates that these two substances are independent.

Table 4 Person's matrix for concentrations of chemical components of separated urine in 3 schott bottles analysed within 24 hours.

	Ca	K	Mg	Na	SO4	Cl	PO4	TKN	NH4	COD	Cond	TDS	DOC	Alk	pH
Ca	1.00	-0.95	0.92	-0.94	-1.00	-0.98	-0.44	-1.00	-0.96	-0.85	-1.00	-0.45	-0.51	-0.62	-0.61
K		1.00	-1.00	0.79	0.97	0.87	0.70	0.97	0.82	0.98	0.96	0.14	0.76	0.34	0.33
Mg			1.00	-0.73	-0.95	-0.82	-0.76	-0.94	-0.77	-0.99	-0.93	-0.06	-0.81	-0.25	-0.25
Na				1.00	0.91	0.99	0.11	0.92	1.00	0.63	0.93	0.72	0.20	0.84	0.84
SO4					1.00	0.96	0.52	1.00	0.93	0.90	1.00	0.37	0.59	0.54	0.54
Cl						1.00	0.25	0.96	1.00	0.74	0.97	0.62	0.34	0.76	0.76
PO4							1.00	0.50	0.17	0.84	0.48	-0.60	1.00	-0.44	-0.44
TKN								1.00	0.94	0.89	1.00	0.38	0.58	0.56	0.55
NH4			Positive		Negative				1.00	0.68	0.95	0.68	0.26	0.81	0.81
COD			1.0		-1.0					1.00	0.88	-0.08	0.88	0.12	0.11
Cond		high	>0.75		<-0.75						1.00	0.41	0.55	0.58	0.58
TDS		med	0.35<0.75		-0.35<-0.75							1.00	-0.53	0.98	0.98
DOC		low	<0.35		>-0.35								1.00	-0.36	-0.36
Alk				.-+0.0										1.00	1.00
pH															1.00

Three 5 L samples of urine were stored in Schott bottles and sub-samples were taken within the first day and again after 3 months, and the analysis is shown in Table 4 and Table 5 respectively. In Table 4 potassium has 10 high value correlations above 0.75 (red cells). It is the parameter with the most frequent (positive) high correlation, followed by sodium, ammonium and chloride all with seven high values.

Table 5 Person's matrix for concentrations of chemical components of separated urine in 3 schott bottles analysed after 3 months of storage.

	Ca	K	Mg	Na	SO4	Cl	PO4	TKN	NH4	COD	Cond	TDS	DOC	Alk	pH
Ca	1.00	-0.42	0.54	-1.00	-0.88	-0.91	0.48	-0.94	-0.99	-0.46	-0.91	-0.97	-0.41	-1.00	-1.00
K		1.00	1.00	0.50	0.81	0.77	0.59	0.71	0.53	1.00	0.77	0.62	1.00	0.44	0.34
Mg			1.00	-0.61	-0.87	-0.84	-0.49	-0.80	-0.63	-1.00	-0.84	-0.71	-0.55	-0.46	-0.99
Na				1.00	0.92	0.94	-0.40	0.96	1.00	0.54	0.94	0.99	0.49	1.00	0.98
SO4					1.00	1.00	0.00	0.99	0.93	0.83	1.00	0.96	0.80	0.88	0.83
Cl						1.00	-0.06	1.00	0.95	0.79	1.00	0.98	0.76	0.91	0.87
PO4							1.00	-0.14	-0.37	0.56	-0.06	-0.27	0.60	-0.47	-0.55
TKN								1.00	0.97	0.74	1.00	0.99	0.70	0.94	0.90
NH4			Positive		Negative				1.00	0.56	0.95	0.65	0.52	0.99	0.98
COD			1.0		-1.0					1.00	0.79	0.98	1.00	0.47	0.38
EC		high	>0.75		<-0.75						1.00	1.00	0.76	0.91	0.87
TDS		med	0.35<0.75		-0.35<-0.75							1.00	0.61	0.98	0.95
DOC		low	<0.35		>-0.35								1.00	0.43	0.33
Alk				.-+0.0										1.00	0.99
pH															1.00

Table 5 shows clearly that after 3 months storage, some of the chemicals correlate better, for instance TDS and electrical conductivity. Other chemicals don't correlate at all after storage, such as phosphate, which precipitates as salts at high pH. Both potassium and sodium are key components (proxies) to monitor because they have good correlations with most of the other components. The sample data in Table 4, although not fully hydrolysed, is already affected by the urine deposits in the urinals and collection pipe work. Therefore it is not the closest data set to fresh urine. However, data in Table 6 is the most indicative of fresh urine because its samples were the first samples collected after the urine diversion work was completed.

Table 6 Person's matrix for concentrations of chemical components of urine from the first four samples (n=4) taken from a continuous collection and storage system

	Ca	K	Mg	Na	SO4	Cl	PO4	TKN	NH4	COD	EC	TDS	Alk	pH
Ca	1	0.83	0.38	0.38	0.86	0.97	0.97	0.98	-0.62	0.78	0.09	0.84	-0.63	0.42
K		1	0.47	0.18	0.46	0.83	0.90	0.87	-0.08	0.89	0.56	0.48	-0.09	0.56
Mg			1	1.00	0.20	0.13	0.20	0.49	-0.26	0.61	0.28	0.05	-0.35	-0.67
Na				1	0.22	0.14	0.20	0.49	-0.29	0.59	0.25	0.07	-0.37	-0.67
SO4					1	0.87	0.80	0.76	-0.92	0.36	-0.43	0.99	-0.91	0.38
Cl						1	0.99	0.92	-0.60	0.66	0.01	0.89	-0.59	0.62
PO4		Positive	Negative				1	0.95	-0.50	0.76	0.16	0.82	-0.50	0.59
TKN		1.0	-1.0					1	-0.51	0.88	0.25	0.73	-0.54	0.32
NH4		high	>0.75	>-0.75					1	-0.08	0.69	-0.87	1.00	-0.09
COD		med	0.35<0.75	-0.35<-0.75						1	0.67	0.32	-0.13	0.13
EC		low	<0.35	<-0.35							1	-0.44	0.65	-0.02
TDS			.-+0.0									1	-0.85	0.52
Alk													1	-0.01
pH														1

Additional chemical results from separated urine that was stored and collected twice per week, is shown in Table 6 and Table 7. The tank from which these samples were collected had a noticeable scale and some biofilm on the inside. It is expected that the presence of this biofilm may have caused differences in the chemical analysis. Table 7 clearly represents a different environment from that shown in Table 7. This is seen by comparing the number and degree of correlations between these two tables, which is for all values combined.

Data in Table 6 shows the results from the initial first four samples, which show a close approximation with fresh urine because the urine from these samples have been stored in a relatively sterile environment, with minimal urea hydrolysis taking place (i.e. low ammonia concentrations). A perfect correlation (1.00) between alkalinity and ammonium is evident, which indicates the trend shown in equation 3.1 and 3.2 caused by urea hydrolysis – simultaneous increase of ammonia and bicarbonate. In Table 6, there are very good (positive) correlations between PO_4^{3-}: Cl^- and PO_4^{3-}:SO_4^{2-}, because phosphate is still an anion with fresh urine and behaves like other anions. However, in Table 5 (and Table 7) there are no correlations because after three months the phosphate has precipitated out of solution. Pearson's correlations were used when analysing the chemical data from urine collected from the gent's urinals, to see if any trends could be identified between the variables. Pearson's correlation can be described as a measure of the similarity or discrepancy of data between two components or variables, and reflect the degree of linear relationship.

Table 7 Person's matrix for concentrations of chemical components of urine from 24 samples taken, after the first four samples, from the same collection tank.

	Ca	K	Mg	Na	SO4	Cl	PO4	TKN	NH4	COD	EC	TDS	Alk	pH
Ca	1	0.14	0.90	-0.10	-0.05	-0.15	0.36	0.02	-0.25	-0.41	0.15	-0.06	-0.12	-0.17
K		1	0.33	0.61	0.56	0.39	-0.12	0.54	0.00	0.47	0.39	0.12	0.15	-0.50
Mg			1.00	0.01	0.12	-0.01	0.50	0.29	-0.07	-0.20	0.24	-0.13	0.07	-0.23
Na				1.00	0.53	0.40	-0.42	0.31	0.20	0.45	0.32	0.30	0.18	-0.23
SO4					1.00	0.65	-0.04	0.52	0.17	0.37	0.07	0.24	0.22	-0.19
Cl						1.00	0.09	0.20	0.26	0.74	0.30	-0.02	0.21	-0.25
PO4			Positive	Negative			1.00	-0.02	0.32	-0.15	0.03	-0.08	0.29	0.19
TKN			1.0	-1.0				1.00	0.36	0.24	-0.03	-0.11	0.38	-0.08
NH4		high	>0.75	>-0.75					1.00	0.32	-0.02	0.13	0.70	0.50
COD		med	0.35<0.75	-0.35<-0.75						1.00	0.29	-0.14	0.19	-0.22
EC		low	<0.35	<-0.35							1.00	0.04	-0.11	-0.34
TDS			.-+0.0									1.00	-0.07	-0.03
Alk													1.00	0.30
pH														1.00

4.5 URINE FROM INFORMAL SETTLEMENTS

Urine samples from Pooke se Bos Township were taken from a dry urine diversion containerised facility called MobiSan, and are operated by locally trained caretakers (Figure 21). The idea for this pilot project started from a Dutch consortium (Vitens-Evides, Landustrie and LeAF). MobiSan is a demonstration unit with the intention of offering proper basic sanitation and high hygiene levels to informal settlements that lack sanitation infrastructure. The general operating procedure is for urine to be collected, stored and then transported to a nearby wastewater treatment works, while treatment of faecal waste involves air-drying and mixing before re-using in compost.

The relative standard deviation of chemical parameters from urine samples taken from MobiSan are all below 15% (except for COD 49% and DOC 55%), which is significantly more consistent than those values obtained from CSIR's gent's urinal urine data. Two reasons are offered for this:

- Roughly 500 people contributed to those urine samples, whereas at CSIR there are at most 10 men using the urinals and therefore higher deviations are expected.
- The urine tank was half full, and the urine was a true composite sample over a long period.

The largest distinctions between the average values of CSIR gent's urinal urine and urine from MobiSan are in K, TDS and DOC concentrations. The K concentration in CSIR's urine is 1,848 mg/L and in samples taken from MobiSan 781 mg/L. It is generally accepted that diet plays an important role in urine composition, but it's not certain which specific foods results in the large salt variations. The huge difference in TDS values of 14,583 mg/L and 5,476 mg/L (difference of 9,107 mg/L TDS) from CSIR and MobiSan urine is difficult to account for. By adding the differences in all the salts (K, Na, Ca, Mg, NH4, PO4, SO4, Cl), only 4,611 mg/L of the total difference of 9,107 mg/L TDS is accounted for.

Figure 21 Dry urine diversion toilets (MobiSan prototype unit) at Pooke se Bos Informal Settlement in Cape Town.

The COD concentrations of 7,771 mg COD/L (CSIR) and 6,944 mg COD/L (Pook se Bos) are more or less similar. COD can be traced back to foods high in carbohydrates, which form a substantial part of the diet of low income people living in the informal settlement. TKN in urine results from the breakdown of proteins, and since the CSIR staff can afford more luxury groceries like meat and fish it is not surprising that the CSIR urine TKN value is almost double that of urine sampled at Pook se Bos, 6,551 mg TKN/L and 3,496 mg TKN/L respectively.

4.6 GREY WATER

shows the complete chemical compositions of grab samples, composite samples, and a grey water sample mixed with large volumes of spent wine, and a summary of the results relevant to grey water only are shown in **Table 10**. There is not much difference in concentrations resulting from the different sampling techniques used, except maybe for TKN and DOC. More interesting to note, the sample intoxicated with spent wine indicates a two-fold increase in TKN and COD, compared to grab sampling results. The wine sample also showed higher potassium, sodium and calcium concentrations. It was initially thought that grab samples taken during the morning or late afternoon reflect low nutrient and mineral concentrations, as they miss out on the peak nutrient loads once dishes are washed after lunch. However, it was later discovered that spent wine entered the grey water collection tank, after the wine tasting on the previous day, and this caused the concentration increases of the variables mentioned above.

4.7 BROWN WATER

Brown water (faecal water) is also collected and macerated in a 200 L tank, before being sampled and tested for its chemical composition. This tank firstly acted as balancing tank for highly irregular flow on such small scale, and secondly contributed to the hydrolysis reactions that normally occur in sewers. The brown water was transferred into 20 L buckets and then transported to UCT for processing. Excess brown water was discharged to the sewer via the underflow pipe (see **Figure 19**).

Table 8 CSIR separated wastewater composition table showing concentrations of all chemical compositions which are compared to NASA and local WWTPs.

	REFERENCE	COD mg/L	DOC mg/L	TKN mgN/L	NH_4^+ mgN/l	P_{tot} mgP/L	PO_4^{3-} mgP/l	K_{tot} mg/l	K^+ mg/l	Na^+ mg/l	Ca^{2+} mg/l	Mg^{2+} mg/l	Cl^- mg/L	SO_4^{2-} mgSO_4^{2-}/L	TDS mgTDS/L	Alk mg$CaCO_3$/l	EC mS/m	pH
Undiluted urine	Ciba Geigy, 1977, average	9600		9200	581		1120		2160	4120	186	104	3800	1059				6.2
	NASA, 1971, average	8205		6300	465		770		1680	2780	210	112	5135	2944				
	CSIR, 2008, office average	7771	2604	6551	3363		262		1848	2147	36	7	3958	1307	14583	16596	3260	8.9
	(RSD, n=50)	54%	52%	46%	68%		99%		29%	25%	112%	368%	34%	59%	31%	65%	39%	9%
	CSIR, 2008, first samples	6562		5495	491		296		1933	2295	100	55	3457	1967	16083	1343	1875	7
	(RSD, n=4)	41%		7%	52%		45%		18%	17%	30%	16%	84%	98%	23%	232%	18%	6%
	CSIR, 2008, following samples	7720	2588	6603	3705		249		1841	2142	30	3	4098	1284	14161	18173	3414	9
	(RSD, n=46)	55%	52%	46%	48%		106%		30%	26%	79%	215%	29%	46%	31%	42%	32%	3%
	CSIR, 2008, office minimum	5430	1000	4497	379		69		1179	1262	10	0	1556	789	10219	80	1650	7
	CSIR, 2008, office maximum	14606	3625	10450	5200		734		2615	3158	119	61	5833	2880	24570	28830	6400	9
	NASA, 1971, minimum	5810		4700	200		470		750	1170	30	20	1870	489				
	NASA, 1971, maximum	10600		7900	730		1070		2610	4390	390	205	8400	5400				
	CSIR, 2009, informal settlement	6944	912	3496	2759	182	166		781	1448	25	1	2433	704	5476	11040	2800	0
	(RSD, n=3)	49%	55%	4%	12%	7%	6%		3%	3%	15%	13%	8%	11%	3%	3%	8%	0%
Diluted uirne	CSIR, 2008, Men's no-mix toilets	1891	459	2113	1403	122	93	499	546	619	23	1	1112	576	3865	5375	1274	9
	(RSD, n=8)	171%	261%	171%	147%	116%	143%	68%	135%	126%	112%	346%	165%	216%	149%	75%	133%	6%
	CSIR, 2008, Ladies' no-mix toilets	1272	240	1274	979	79	63	447	432	524	20	2	704	337	2837	3688	903	9
	(RSD, n =8)	209%	224%	170%	316%	251%	244%	226%	177%	275%	241%	505%	253%	332%	190%	197%	172%	13%
Brown water	CSIR, 2008, Men and ladies' no-m	3566	91	140	41	44	16	42	31	25	19	10	129	553	651	355	70	7
	(RSD, n=8)	128%	68%	69%	119%	38%	58%	31%	97%	93%	80%	94%	364%	348%	60%	160%	90%	10%
Grey water from kitchen	CSIR, 2008, Grab samples	2380	163	25	0		2		11	15	7	2	127	86	570	9	40	5
	(RSD, n=6)	136%	146%	60%	141%		191%		105%	41%	40%	74%	428%	74%	90%	179%	266%	12%
	CSIR, 2008, Composite samples	2208	822	47	0		3		17	20	14	3	21	61	722	24	24	5
	(RSD, n =14)	146%	229%	207%	360%		227%		135%	69%	144%	76%	95%	104%	143%	155%	83%	24%
Domestic municipal wastewater	UCT, Westfleur raw	810		88	65	13	10		18	100	46	9	129			288	117	7.4
	UCT, Simonstown raw	584		51	35	10	7						130			208	98	7.5
	UCT, Mitchell's Plain raw	1274		101	70	17	13		22	89	30	4	102			341	114	7.4

A manually operated submersible grinder pump installed in the collection tank homogenises the brown water, so that samples taken are the most representative of the actual wastewater composition. In addition, the installed grinder pump also simulates maceration taking place at pump stations. During pump operation, the brown water is circulated through the recycle pipe (left hand side) and up several meters high so that the required head is reached to prevent cavitation. Pumping through a small diameter hose further breaks up and homogenises solids. In the event of the brown water tank reaching maximum capacity, an overflow pipe was included to allow wastewater to flow into the sewer.

4.8 TYPICAL COMPOSITION OF RAW DOMESTIC WASTEWATER IN RELATION TO URINE, BROWN WATER AND GREY WATER

Westfleur and Simonstown wastewater treatment works in Cape Town are entirely domestic treatment works, while Mitchell's Plain wastewater may contain some industrial wastewater. The majority is still domestic wastewater. The average COD to nutrient ratio of these wastewaters compares closely to the ratios of wastewater at Hardenberg and the Dutch average. This allows comparison of this project with that of Wilsenach and van Loosdrecht (2003, 2004, 2006). The separated wastewater streams were analysed at CSIR and compared to mixed raw wastewater data obtained from local WWTPs Westfleur, Simonstown and Mitchell's Plain. Comparisons are made easier after converting the concentrations into COD to nitrogen to phosphorus ratios, with COD normalised at 1000 mg/L, as shown in Table 9.

Table 9 Raw COD:N:P wastewater data from WWTPs in South Africa and Europe, as well as the COD:N:P ratios of brown and grey wastewater separated and measured at CSIR.

	COD	**N**	**P**
Westfleur	1000	108	17
Simonstown	1000	87	16
Mitchell's Plain	1000	97	13
Average	**1000**	**91.7**	**15.3**
Hardenberg (Meijer, 2001)	1000	96.0	15.2
Dutch average (STOWA)	1000	93.1	14.9
Brown (CSIR, mean ratio)	1000	47.3	19.7
Grey (CSIR, mean ratio)	1000	19.0	1.1
Brown: grey (COD 50:50)	**1000**	**30.3**	**6.8**

During one day experiments at CSIR, all the wastewater leaving a small office block of about 15 people, was captured. The motivation for this experiment was to determine where the majority of the nutrient load originated from. This shows again the relative contribution of urine to the nutrient load in domestic wastewater. It is interesting to note the significant load of COD in urine, which is a potential carbon source for the treatment of source separated urine. Again, these results illustrate that although urine accounts for the smallest flow it has the largest nitrogen load. The hydraulic load of urine in the top left pie chart shows that it accounts for about 7% of the daily wastewater volume. This is contrary to literature which states that urine should only make up 1% of the volume. This is because the wastewater captured during this experiment was for only part of the day, from 08:00 to 16:30 and thus

excludes wastewater resulting from showing, bathing, washing clothes, etc., which would reduce the contribution that urine is currently showing in Figure 20.

By separating urine at source, and analyzing the COD, N, and P in brown and grey water, a 50:50 volume mix of brown and grey water reveals a much higher COD to nutrient ratio of 1000 : 30.3 : 6.8 in contrast to the local average of raw mixed wastewater which includes urine of 1000 : 91.7 : 15.3.

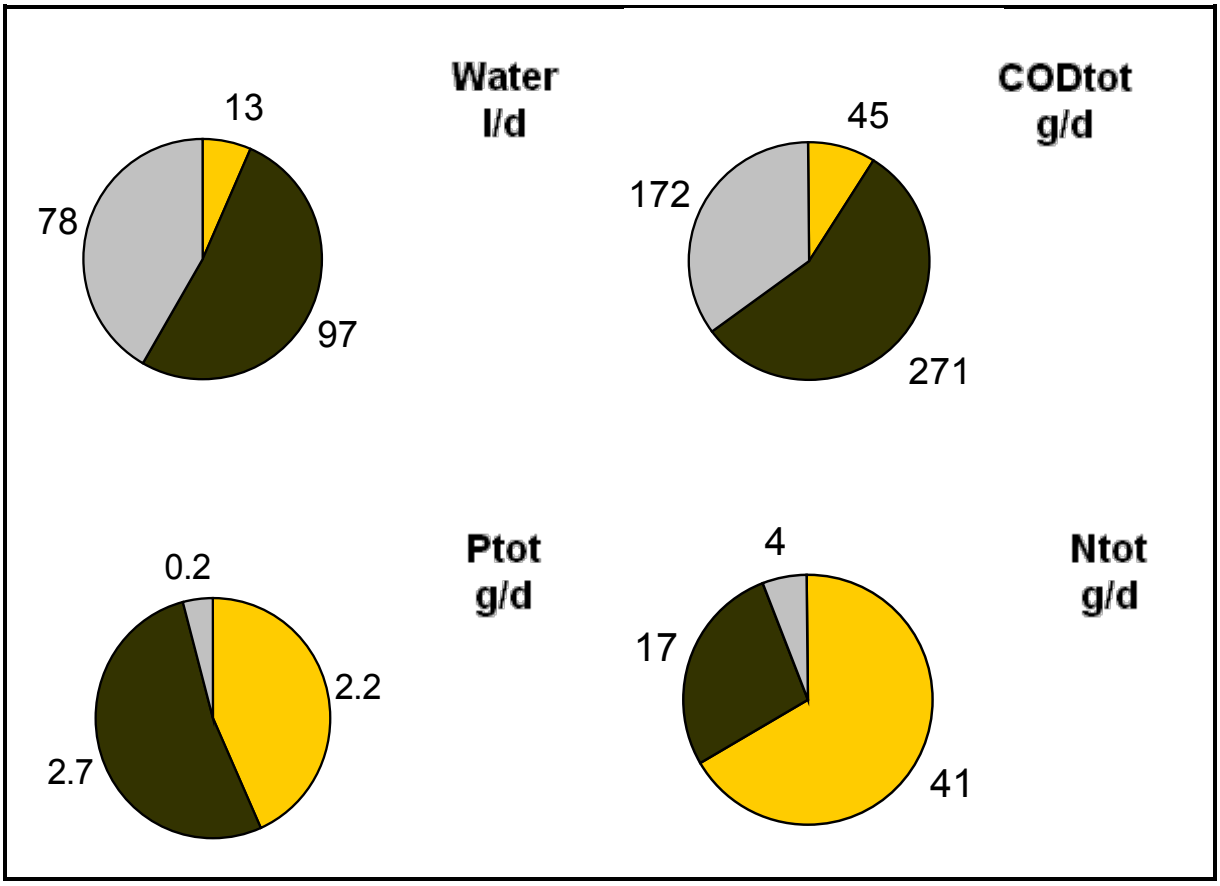


Figure 22 Pie charts with results from 24 hour experiment that captured and analysed all wastewater from the office block (Yellow water = urine, grey water = kitchen water, brown water = faecal water).

This clearly shows that a reduction of 67% nitrogen (from 91.7 to 30.3) and a 56% reduction in phosphorus (from 15.3 to 6.8) can be achieved if complete urine separation is implemented within an entire catchment of wastewater. This reduction in nutrients closely verifies the contribution that urine makes to domestic wastewater reported by Larsen *et al*. (1996), who stated that urine makes up 80% N and 50% P in raw wastewater.

5 NITRIFICATION AND DENITRIFICATION OF SOURCE SEPARATED URINE IN A SEQUENCING BATCH REACTOR

5.1 INTRODUCTION AND AIMS

In Chapter 4 it was shown that the majority of nutrients originate from urine, but that urine makes up less than 1% of the volume of municipal wastewater. This insight underlies all the drivers for change (Chapter 1.1) that may lead to separate collection and treatment of urine. However, if urine was collected separately, how would it be managed and treated? To answer this question, a simple biological system was proposed that nitrifies urine and produces ammonium-nitrite, for which there are two possible applications:

1. A decentralised reactor, for example at a large office block, commercial centre or airport, treats urine and discharges the effluent into the normal sewer system, where in-sewer denitrification takes place, thus further removing nitrogen and at the same time preventing biological sulphate reduction, or
2. A side stream reactor, at a centralised wastewater treatment works, produces an ammonium-nitrite mixture (similar to the SHARON process) for further treatment and complete autotrophic nitrogen removal in an Anammox reactor.

This part of the project focused on the treatment of undiluted source separated urine in a 20 L sequencing batch reactor (SBR) for the biological removal of nitrogen, using activated sludge. The operation of the reactor included an aerobic and anoxic phase for nitrification and de-nitrification.

The early stages of the study mainly involved attaining a stable biological system in which bacteria adapted to high free ammonia concentrations. The nitrifying organisms were of greater concern than the denitrifying organisms as they proved to be more inhibited than initially expected. The following stage in the study entailed establishing the maximum daily feed volume, the required sludge age and the optimal duration of anoxic/aerobic phases.

Some limitations for the biological treatment of urine were quantified. Firstly, the alkalinity in urine is present as bicarbonate, following urea hydrolysis. The alkalinity is however insufficient to counter the acid formation during nitrification. Secondly, the organic carbon is too low to achieve complete denitrificaiton, even if the more effective route over nitrite is followed. Furthermore, the pH > 9 of stored urine moves the equilibrium from ammonium towards free ammonia, which is toxic, even to ammonia nitrifying organisms at high concentration.

5.2 MATERIALS AND METHOD

The experimental setup used for this study consisted of a lab scale sequencing batch reactor (SBR). A set of smaller batch reactors were implemented, in order to run side experiments for shorter periods without affecting the main reactor. The reactor vessel consisted of a vertical Perspex tube with 153 mm inside diameter, 1020 mm high and 20 litres operating volume. The reactor height was later extended by 300 mm allowing for headroom and preventing spillage during operation. The reactor was fitted with three inlet/outlet valves (one at the top, one at the base and one in the middle). Two probe ports were inserted at either sides of the mid-level valve. The top and bottom valves of the reactor were connected to an 8 mm recirculation tube rigged through a 0.75 kW peristaltic variable speed drive bi-directional pump (Figure 21). An air nozzle was fixed into the base of the reactor. Inside the reactor, a fine bubble disperser was connected, and the outside end of the nozzle was

connected to a compressed air line (100 kPa), fitted with a humidifier to reduce excessive evaporation from the reactor. The conditions inside the reactor were monitored by means of pH and reduction-oxidation-potential (ROP) probes installed at mid-level ports. The probes were linked to an online data logger of which the logging interval could be set as required. Temperatures were measured manually with a mercury thermometer.

Figure 23 Sequencing batch reactor from top, with beige recirculation line, two blue Watson Marlow pumps, and plastic air lines (pressurised air left, and vent line right)

The sequence of the different stages of the reactor (Figure 22) was manually controlled, withdrawing effluent from the mid-level valve and adding new feed into the top of the reactor. The volumes of effluent and feed were measured in volumetric flasks to determine and replace the evaporative volume losses from the reactor. The duration of cycle phases were maintained by manually closing and opening the air valve. The air supply during aerobic phases was automated by means of a solenoid valve controlled by the OUR meter. No additional alkalinity was added. The data logger unit monitored and stored values for pH, Reduction oxidation potential (ROP), Temperature (T), Specific conductance (SC) and Dissolved Oxygen (DO).

The oxygen utilisation rate (OUR) was measured by a control/logging processor, linked to a separate DO probe in the reactor and a solenoid valve installed on the air line. The processor maintained the DO within a set DO concentration interval. This was facilitated by the solenoid valve that was controlled by the processor. The processor was programmed to close the solenoid at the pre-set DO high point concentration. The DO concentration would then decrease down to a pre-set DO low point concentration before the solenoid valve would open. The OUR was calculated by the device based on the time between shutting and opening the valve and the set point interval.

The experimental system was inoculated with sludge from a lab scale reactor at the University of Cape Town wastewater lab. The UCT reactor treated domestic wastewater with different characteristics compared to undiluted urine. It was thus foreseen that organisms in the sludge would require time to adapt to urine feed. Upon inoculation on 25/5/2009, 2 litres of urine was added and the

system was aerated. On 26/05/2009 two litres more urine was added however not enough time was allowed for the first batch of ammonia to be oxidised. In retrospect, the excess feed increased pH above 9, and raised free ammonia concentrations to a toxic level. The reactor was operated in this state, without further addition of urine, to see if biomass would adjust. After three weeks operation, without notable changes in pH and ROP, it became clear that the nitrification process failed. This was rectified by decanting 10 l of reactor content and replacing the discharged volume with fresh sludge from UCT (15/06/2009). Only small amounts of urine were then added (500 ml), based on the pH and ROP responses, from which it was evident that the system had recovered (similar, but not as pronounced as Figure 23).

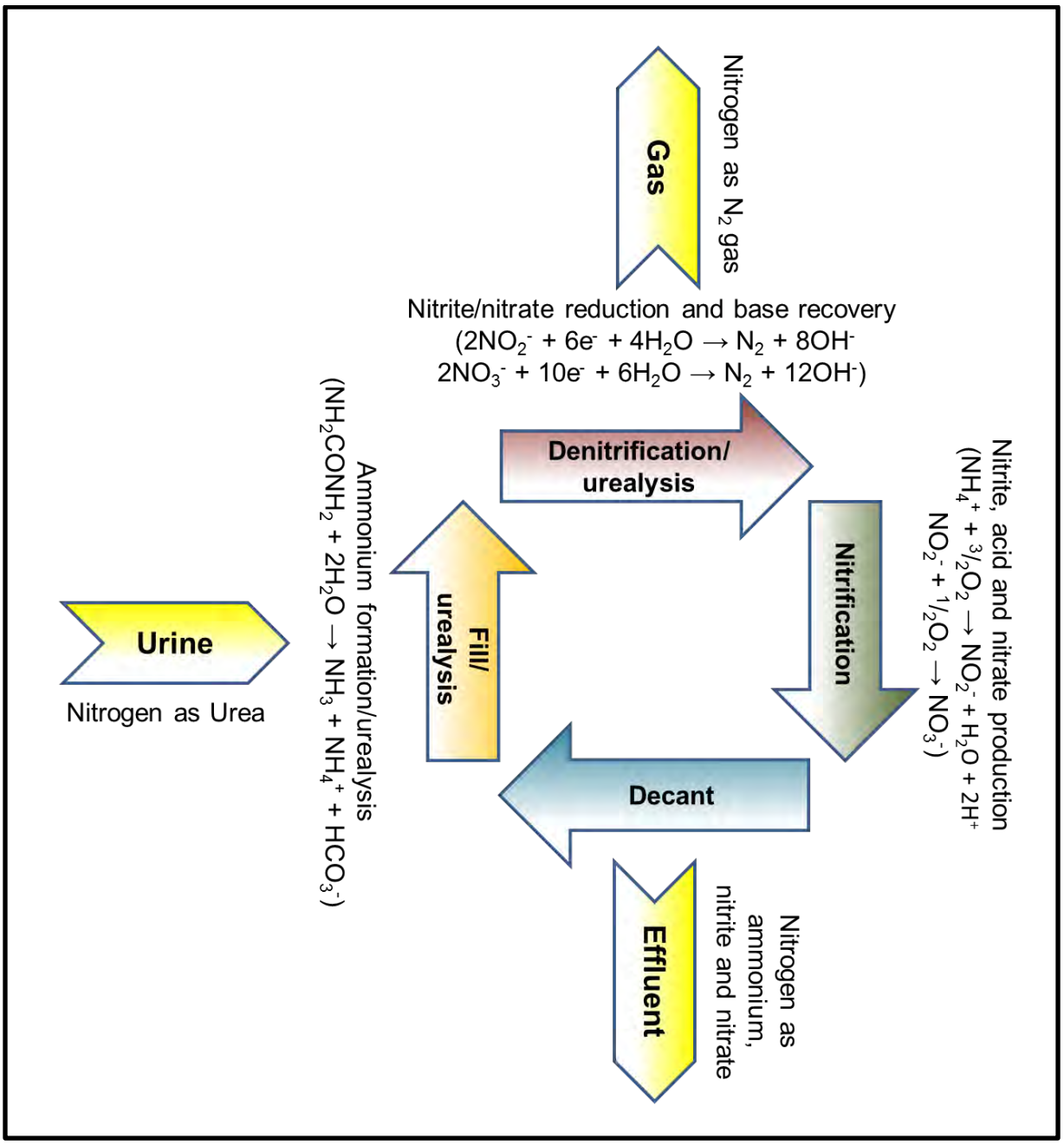


Figure 24 Process stages of the sequencing batch reactor for nitrification and denitrification of urine

The chemical analysis were done according to the methods described in Chapter 4. The changes in concentrations over a cycle were however small relative to the base concentrations (background)For example, assume the TKN concentration at the end of a cycle was 2,000 mg N/L. During the volume exchange 1 L of effluent is withdrawn from the 20 L system and 1 L of urine is added again with a TKN concentration of 6000 mg N/L. Therefore the TKN in the in the system at the start of the cycle is:

(19 L x 2000 mg N/L + 1 L x 6000 mg N/L)/20 L= 2200 mg/L.

Therefore the change in the TKN concentration during the cycle is 200 mg N/L. If the chemical analysis of TKN has an error of only 5%, then this relates to 100 mg N/L, which is half of the change in TKN concentration over a cycle. Dilutions were made stepwise using volumetric flasks and 5 ml to 10 ml pipettes to ensure accuracy of the large dilutions required for some of the methods. Samples were diluted for standard chemical analysis, according to the following dilution factors:

- Nitrite and nitrate x 5,000 (x10x10x10x5)
- TKN and FSA x 100 (x10x10)
- COD x20 (x10x2)

Slight errors could therefore have huge impacts on the analytical values.

5.3 RESULTS

From experimental results, the pH and redox oxidation potential (ROP) data gave good indications on the progress of nitrification and de-nitrification, showing an operating pH range from 6.0 to 8.5 and ROP from 250 mV to -50 mV (Figure 23). The decrease in pH indicates the nitrification stage where alkalinity is consumed. The sudden increase of pH to above 8, indicates feeding an untreated batch of urine. The SBR's temperature was controlled using a heating element inserted into the reactor. Temperature control for this reactor was added after winter temperatures dropped to 10ºC, and the colder temperatures noticeably slowed down the reactor performance.

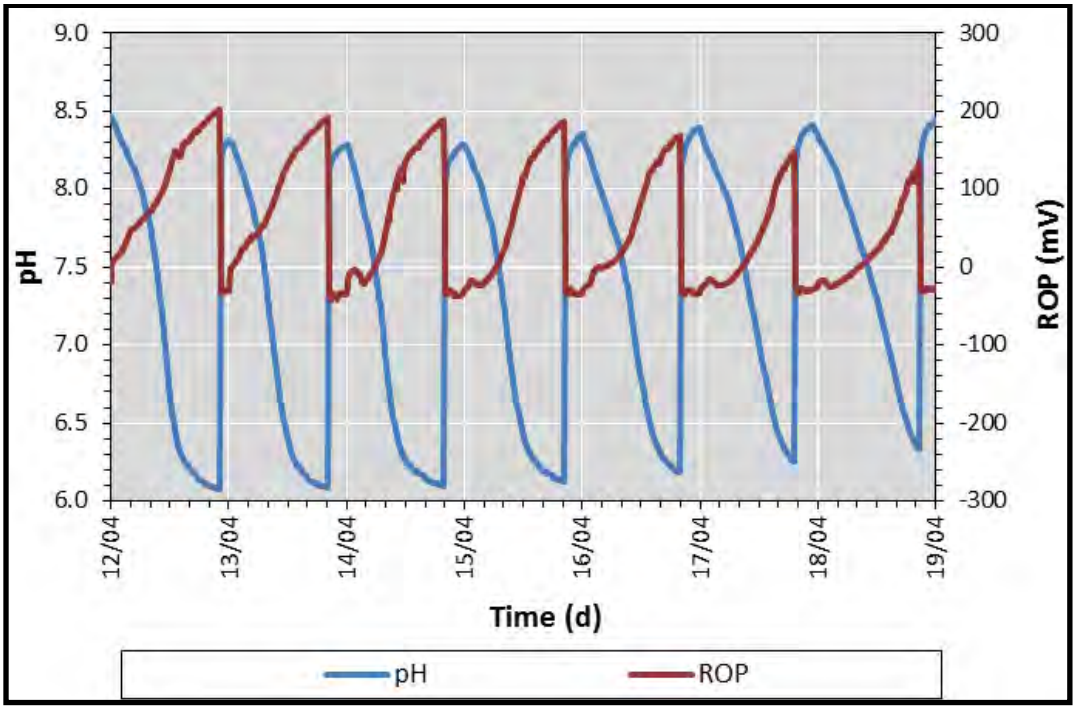


Figure 25 Periodic fluctuating of pH (blue) and Reduction/Oxidation Potential (red), treating 1.25 L of urine per day in the sequencing batch reactor

At a closer glance of the anoxic phase (Figure 24), the pH rate of change and variation thereof, over the extent of the phase, is evident. The intial rapid pH increase, which ensues for approximately 10 minutes (0.7% of cycle time), is associted with the feed procedure that adds alkalinity and causes the pH to increase by almost two pH units, from 6.1 to 7.9 through rapid mixing. Thereafter, the pH increases slowly to around 8.25 by the end of the anoxic phase. The rate of this secondary pH increase is significantly slower than the rate of the initial increase and reduces as pH rises further. The reason for this increase is the production of alkalinity during the denitrification of nitrite (or nitrate) at the same time oxidising organic material (COD) under anoxic conditions. As the organic material available for denitrification decreases, the rate of the reaction slows down as can be seen by the decreasing slope of the pH curve.

The reduction-oxidation potential (ROP, also called redox potential) is measured in mV and gives an indication of the ratio between chemicals in a system with electrons to donate, relative to chemicals that can accept electrons. In a reduced system, more chemicals are present that can donate electrons compared to chemicals that can accept the electrons, which gives a negative redox value. Vice versa, in an oxidised system, with a positive value, more chemicals will be present that can accept electrons that that on offer.

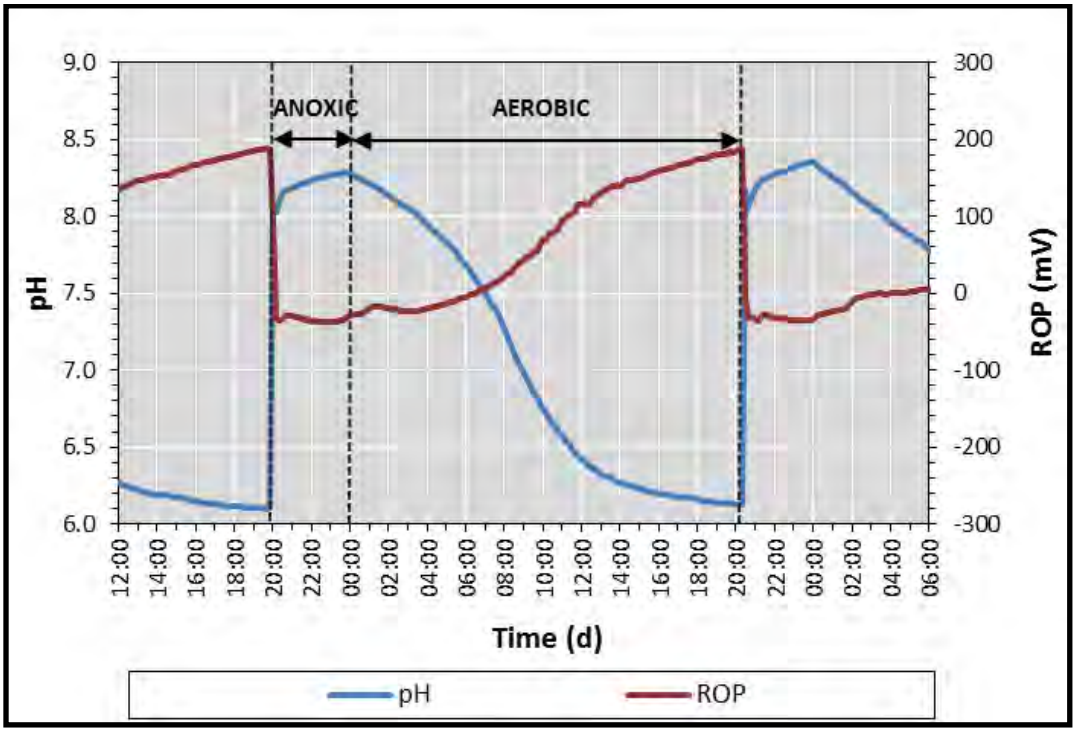


Figure 26 Changes in pH and Reduction/Oxidation Potential during processing of one batch of urine, where urine was fed at the onset of the anoxic phase

During the fill phase of the SBR, ammonium and dissolved organic carbon compounds are added, both of which are electron donors. An immediate drop in ROP is seen. During the anoxic phase, the dissolved organic matter is oxidised to biomass and CO_2, with nitrite or nitrate, which in turn is reduced to nitrogen gas that escapes to atmosphere. Both CO2 and N2 have no redox potential and the ROP stays more or less constant during the anoxic phase, and into the aerobic phase. As soon as ammonium is nitrified, the pH starts to drop as the bicarbonate buffer is consumed. However, the redox potential does not immediately increase and can lag the change in pH profile considerably. Two possible explanations are offered:

- An amount of slowly degradable COD, either from slowly degradable components in urine or by-products from endogenous respiration is still present while being slowly oxidised, thereby maintaining a low redox potential.
- As soon as redox potential reaches zero, the rate changes increases, which may indicate that the organic material has been oxidised and that redox potential is determined as ammonium (donating 3 electrons) is converted by oxygen that accepts the electrons and remains as nitrite (ready to accept 3 more electrons). As soon as the pH drops 6.5, the rate of pH change slows down, which indicates the decrease in the nitrification rate. This is also reflected in the redox potential where the rate of change decreases over the same period.

The substrate and product concentrations measure over one batch is shown in Figure 25. Virtually no nitrate appears in the system. During the anoxic phase a decrease in nitrite can be seen in accordance with denitrification. Nitrite then increases slightly during the aerobic phase, and faster in the first six hours. However, during the same time, COD seems to increase, which cannot be explained for the 10 hour period. Considering the mix pump rate of 4 l/min, the reactor content replacement rate is 5 minutes. Both ammonium and TKN decreases during the aerobic phase, in accordance with nitrification, but the values during the anoxic phase are somewhat erratic. The analysed concentrations are prone to some level of error, due to small expected variations against a high background concentration, as explained in the Methodology. It is therefore more useful to consider the concentrations over many periods.

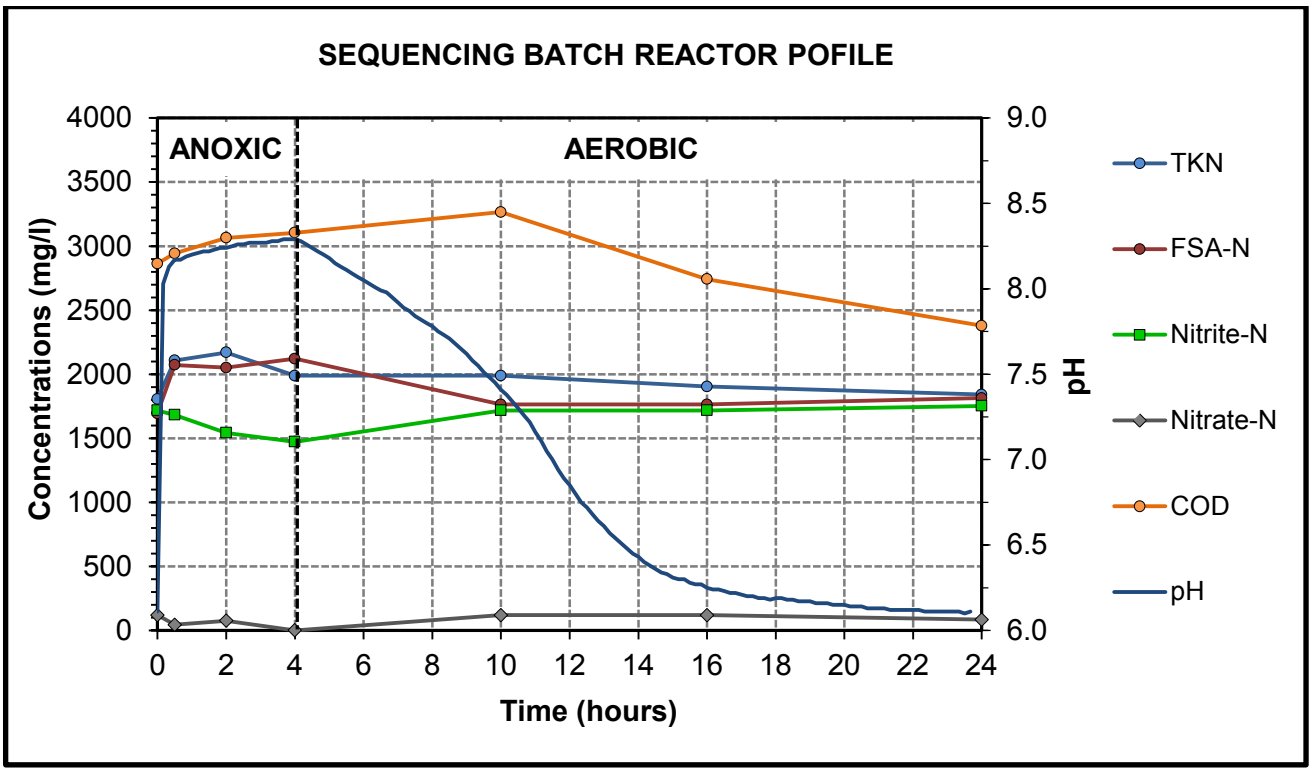


Figure 27 Typical profile of substrates and products in the sequencing batch reactor, over one 24 hour processing period (2010.04.13)

The anoxic phase was found the near optimum at 4 hours. One series of experiments, with anoxic periods of 8 hours, showed that there was no marked improvement on denitrification. On the contrary, if there was any effect it seems that the longer anoxic period, at the cost of a shorter aerobic period,

led to deterioration in nitrification. Possibly, the decay of nitrifiers, which continues during the anoxic phase, started to impact the observable growth rate and overall nitrification reaction.

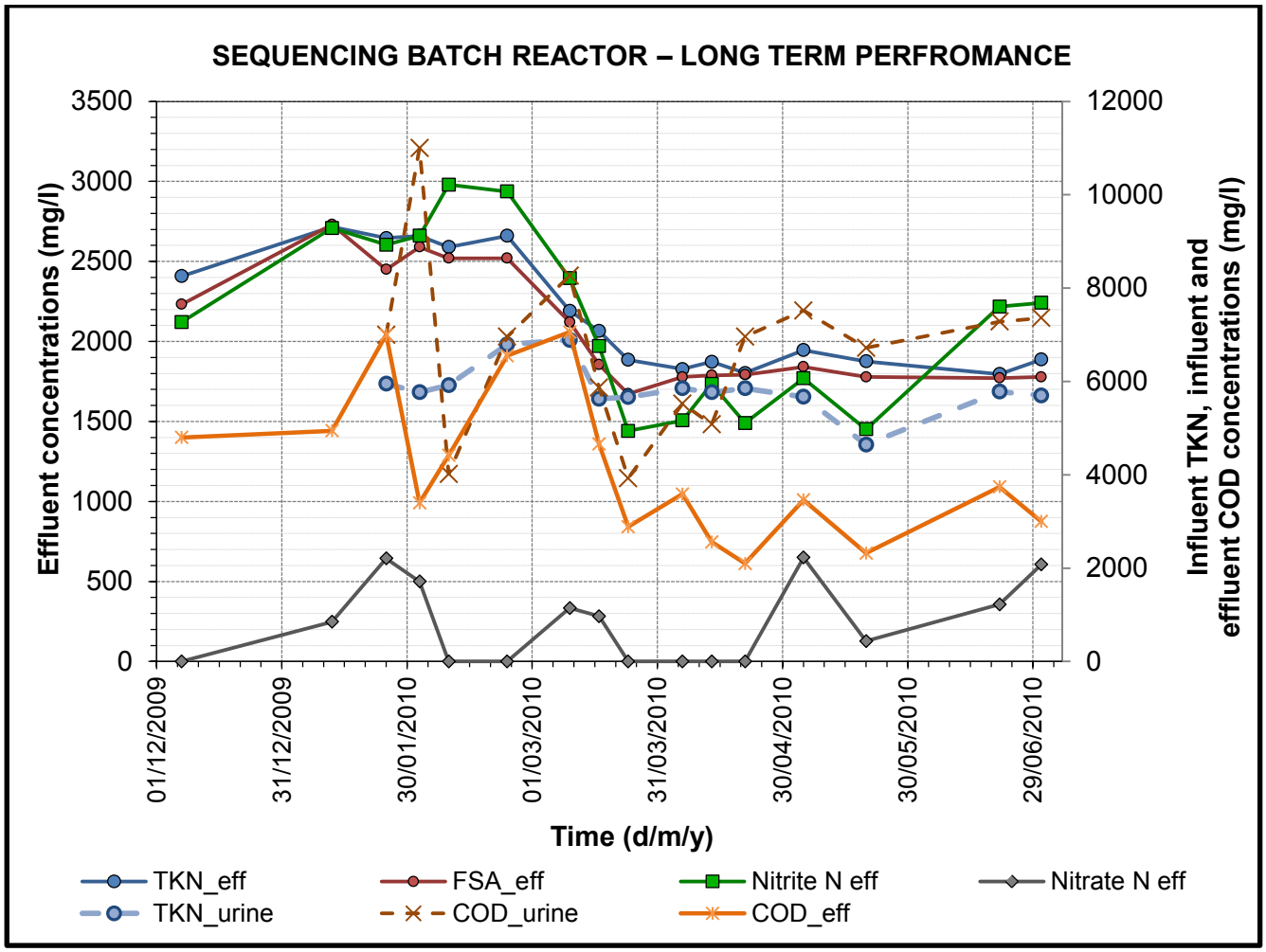


Figure 28 Long term profile of sequencing batch reactor for nitrification and denitrification of urine

The system reached steady state during operation (including practical aspects of consistent sequencing, sludge withdrawal, etc) at around March 2010, as seen in the long term profile of Figure 28. Although TKN in the influent urine had a fairly constant concentration, “steady state” is somewhat of a misnomer in this regard, considering the variation of COD in the influent urine.

The concomitant decrease in TKN, ammonium and nitrite for the period from mid-February towards the end of March, could be related to changes in the processing of COD. Over the first part of the experiment, the COD in the effluent was not much less than the COD in urine. Till the beginning of March, nitrite concentrations were also higher than ammonium. From March onward, the effluent COD concentrations were clearly much less than the COD in urine, which was brought about by more effective denitrification. However, effluent COD concentrations remained between 2,500 and 3,500 mg/l during the steady state operation.

From Figure 26 one tends to see some apparent correlations between COD in the influent, the effluent COD and the effluent nitrite concentrations. However, this is not backed up by the data, or any kind of mass balance. It is more likely the effect of dilution errors than lead to these variations.

Total Suspended Solids (TSS), Inert Suspended Solids (ISS) and Volatile Suspended Solids (VSS) concentrations varied between 1700 mg/l and 3300 mg/l before steady state was reached, with an initial high inert solids contribution. This indicates that a high degree of endogenous respiration which produces endogenous residue that manifests as ISS. This is possibly due to the adjustment of the organisms to system conditions during which organism decay is high relative to organism growth. During steady state the total solids concentration ranged between 1100 mg/l and 2000 mg/l. The VSS fraction averaged at more than 80% of the TSS which indicates that majority of sludge consisted of biomass.

From the equation for denitrification in Figure 22, the minimum amount of COD required for the denitrification of 1 Moll nitrite is equivalent to 3 electrons, or 1.71 gCOD/gN. From the values in Table 11, on average 3,176 mg COD/l was removed for 1,627 mg N/l, or 1.95 g COD/gN. This low value seems plausible considering that bacterial growth is low, due to inhibiting factors. Much of the biomass decay products therefore becomes available again as substrate. Furthermore, it is not clear how much COD is generated autotrophically through growth and decay of nitrifiers.

Table 10 Average concentrations of components in urine and effluent during steady state operation

	FSA mg N/l	**TKN** mg N/l	**Nitrate** mg N/l	**Nitrite** mg N/l	**COD** mg COD/l
Urine[1]	5396	5849	0	0	6496
Urine[2]	5198	5618	0	0	6249
Effluent[2]	1747	1857	367	1767	3073

1 Average of 14 analyses of urine over experimental period, for comparison with results in Chapter 4
2 Average of the final 9 analyses, during which the reactor was at steady state conditions, for comparison with effluent results, over the same period

Finally, the effects of high nitrogen concentrations, which forms free ammonia (NH_3) and free nitrous acid (HNO_2) in solution, as well as the effects of salinity, were tested in 3 litre batch experiments, with sludge from the SBR. The batches were run for 12 hours, with an initial high ammonium concentration of 2,400 mg N/l and an initial low ammonium concentration of 800 mg/l. High salt concentrations were made up with sodium chloride to 2,500 mS/m, and low salt concentrations were made up with sodium chloride to 500 mS/m. The final nitrite-N and nitrate-N concentrations of the batch experiments are compared in Table 12. The sums of nitrite-N and nitrate-N concentrations at the end of the batch experiments are also revealing.

Table 11 Summary of results from four batch tests with different FSA and salt concentrations

Batch and condition	**Nitrite (mg N/l)**	**Nitrate (mg N/l)**	**Nitrite + nitrate (mg N/l)**
Batch 1: High FSA, non-saline	48.3	23.1	71.2
Batch 2: Low FSA, non-saline	46.8	39.1	85.9
Batch 3: High FSA, saline	45.8	20.3	66.1
Batch 4: Low FSA, saline	51.9	27.2	79.1

The results are not too surprising. Over the same period the batch with low ammonia and low salt concentrations had the highest conversion (fastest rate) considering both formation of nitrite and

nitrate. This was followed by the low ammonium high saline batch. Clearly, the high ammonia concentration inhibits the process more than salts in this range. This could probably be attributed to the significantly lower FSA levels under which these experiments were conducted. Considering only nitrite conversion, the differences are not so big. The major dissimilarity between experiments with high and low ammonia, is seen in the difference in final nitrate-N concentration. The nitrite oxidisers are more prone to inhibition by high ammonia concentrations than the ammonia oxidisers. These experiments mirror the work of Anthonisen et al., 1976, who showed that nitrite oxidisers are inhibited by both free ammonia and nitrous acid. Furthermore, inhibition of nitrite oxidisers occurs at relatively low concentrations, while ammonia oxidisers are only inhibited at extreme ammonia concentrations, like undiluted urine. The appearance of nitrate in the effluent () from April onwards, could be explained by the operating conditions, during which the pH never dropped to below 7 in an attempt to improve the nitrification rate. This would have prevented the formation of free nitrous acid and thus eliminated one of the growth inhibitors.

5.4 DISCUSSION AND CONCLUSION

Process reactions involved in the nitrification and denitrification of urine are relatively simple and follow the known stoichiometry. Based on the composition of urine, the final products of a denitrification-nitrification sequencing batch reactor could already be predicted. From the hydrolysis of urea (Figure 22) and the ammonium-bicarbonate ratio of 1, one would expect only 50% conversion of ammonium to nitrite due to the formation of acid. However, a consistent ammonium oxidation of 66% was achieved. The additional ammonium oxidation was made possible by the recovery of alkalinity during denitrification in the first phase of each sequence. Roughly half of all produced nitrite and nitrate was reduced, presumably to nitrogen gas. Generally speaking, for every three parts of Total Kjeldahl Nitrogen in urine, two parts would be nitrified, of which one part would be denitrified and released as gas, leaving one part ammonium and one part nitrite in the effluent. The maximum duration of the anoxic phase was established at 4 hours to compromise for the longer aerobic phase required by nitrifying organisms.

From a process point of view, denitrification over nitrite is preferred because of the better utilisation of organic substrate and therefore greater N removal potential. It was found that the growth and activity of nitrite oxidising bacteria is inhibited by the high salinity of urine, more so than the inhibition of ammonium oxidising bacteria. Furthermore, growth of nitrite oxidising bacteria was also inhibited by the high free ammonia concentration as well as the high nitrous acid concentration that formed in the sequencing batch reactor when treating undiluted urine, exacerbated by extreme high pH 8.5 and low pH 6.

Process kinetics are clearly pH dependant. For complete inhibition of nitrite oxidisers and production of nitrate, this is an advantage. The process could be integrated as part of a larger system of waste treatment, management and even beneficiation. As a pre-treatment process unit, the effluent produced during this study is nearly identical to that required by an Anammox process.

If nitrite production is not specifically required, then addition of alkalinity and running a continuous flow system, without the pH extremes of the batch process, should be considered. Such a process would be ideal for pre-treatment of urine before discharge of the nitrate effluent into an existing sewer, for prevention of sulphate reduction, and for in-sewer denitrification.

Process automation could be possible simply by on-line measurement of pH and ROP. These two measures are tell-tale indicators of the process sequence and performance. The sequencing batch reactor was successfully modelled using a spread sheet based iterative approach (Chapter 6).

6 MATHEMATICAL MODEL – NITRIFICATION/DENITRIFICATION IN URINE

6.1 MODELLING STRUCTURE

This section summarises a model for biological nitrification and denitrification of urine in a sequencing batch reactor (SBR) system. A model was developed in a spread sheet by means of a time step integration method. The model is aimed at describing the given system over the duration of a 24 cycle in incremental time steps of 10 minutes. The techniques and assumption used to develop the model are explained.

The bacteria facilitated biochemical processes which define an activated sludge system are dependent on various factors that can be categorised by the groups of organisms involved. These factors are related to the biological behaviour and requirements of the organisms. Every group of organisms have different requirements with respect to substrate and environment. This essentially defines the functional roles of organism groups. The development of a model is centred around the mathematically description of the biological behaviour of functional organisms and the quantification of influential factors. Organisms may perform differently from one system to the next, but the biological fundamentals of activated sludge processes remain valid and can be used as the platform for developing a model of an unfamiliar system. Data from experimental work serves in verification of parameters and estimation of unknowns which assists in refining and calibrating a model specific to the system. Models are not perfect and may have limitations, but they can be sufficiently accurate and are useful in gaining better insight into the inner workings of a given system.

Activated sludge systems are in reality very intricate and it is not always possible to determine all variables or represent every biological mechanism in detail. Therefore various simplifications have to be made in developing a model. The simplifications are based on reasonable assumptions and principles that make it possible to bridge complexities and create a model that is plausible and representative of the real system.

The development of this model is based on the fundamentals of activated sludge systems. It involves the biological activity of organisms, chemical changes and conversions, and the interactions that occur within the system environment. In order to explain the approach used to develop the model, the system is broken down into the various components which it consists of Biological behaviour, Wastewater characteristics, External variables, System constraints, Growth Kinetics and Stoichiometric Conversions

6.2 PROCESS STOICTRIOMETRY

Bacteria are unicellular prokaryotes that reproduce through binary fission. Binary fission is a replication process whereby two daughter cells are produced from a mother cell (Bitton, 2005). Therefore increase in bacteria biomass occurs mainly through an increase in cell numbers rather than mass of individual cells. Bacteria synthesise biomass by converting substrate into new cell constituents through internal processes. The organisms acquire the necessary substrate for biomass synthesis the environment in which they live. The relative small size and shape of bacteria provide them with a high surface-to-volume ratio which is an advantage in substrate uptake.

Bacterial life and growth is essentially the product of metabolic processes. Metabolism is the sum of all chemical processes in a bacterial cell through which the organism acquires energy, maintains life

functions and synthesises new biomass. These chemical processes are facilitated by bacterial enzymes which serve as catalysts for series chemical reactions that break down organic and inorganic compounds into a form which is utilisable for internal processes:

$$\text{Substrate} \xrightarrow{\text{Enzyme}} \text{Intermediate product} \xrightarrow{\text{Enzyme}} \text{…….....…} \xrightarrow{\text{Enzyme}} \text{Final product}$$

Metabolic processes can be divided into two key types namely catabolism and anabolism.
Energy is derived from catabolic processes which involve chemical oxidation-reduction reactions between electron donating substrates and electron acceptors. This energy in turn is utilised during anabolic processes in combination with carbon and other essential nutrients (inorganic compounds) to maintain cellular activity and synthesise new cell mass. The type of energy and carbon sources utilised by bacteria are related to the kind of organism.

Bacteria require favourable environmental conditions in order to function. These conditions mainly refer to pH, temperature and oxygen levels. Organisms can function at their peak when conditions are optimal, however the deviation from optimal of any one condition reduces organism activity; the more a condition diverges from optimal, the greater the reduction in organism activity. The presence of substances that are toxic to bacteria lead to inhibition and when in combination with sub-optimal environmental conditions, a cumulative inhibition effect can develop to a point where bacterial activity completely ceases.

When organisms reach a stage where growth is no longer possible, cells enter an endogenous mode during which internally stored products and cell constituents are metabolised to produce energy for cell maintenance. Anabolism is no longer relevant but catabolic processes continue. Cell mass reduces until all energy sources are depleted after which the cell expires and breaks open (lysis). The endogenous process is therefore associated with decay in biomass.

There are many different species of bacteria which can make up the microbial population of a biological environment such as an activated sludge system. The characteristics of the organisms, with respect to biological behaviour, requirements and functional roles, differ from one species of bacteria to the next and therefore have to be deliberated in order to develop a plausible model. The identification of every individual species of bacteria is impractical (virtually impossible) but grouping organisms by type and similarity, provides an adequate approach to recognising the significance of the diversity in bacterial population composition. Bacteria are grouped and every group is identified as a separate entity with a specific functional role and set of attributes (concentration, growth, decay…) which collectively describe multiple types of similar organisms as one. Similarities in organisms can be found in their metabolic processes and requirements which in effect determine their role in an activated sludge system. Metabolism therefore serves as a good basis for classifying organisms as explained in the following section.

The biological behaviour of bacteria is related to its metabolism. Energy and carbon, as two major requirements for metabolism, is used in classification of bacteria. Energy is derived from external sources which can be of three types namely organic, inorganic or light (the latter being less relevant in activated sludge systems). The type of energy source that bacteria utilise depends on the type organisms. Organisms, such as the bacteria in activated sludge systems, which obtain energy through the oxidation of chemical compounds, are classified as chemotrophs. The sub division of chemotrophs is established on the nature of the chemical compound and the source of carbon. Chemotrophic organisms that utilise organic compounds as source of energy and carbon classifies as

chemo heterotrophs (chemotrophic heterotrophs). Organisms that utilise inorganic compounds as energy source and carbon dioxide as carbon source are classified as chemoautotrophs (chemotrophic autotrophs)

Functional roles of organism groups

Heterotrophs

This group of organisms are primarily responsible for the removal of organic carbon (measured as COD) from wastewater. The same group of organisms also facilitate the conversion of nitrate/nitrite to nitrogen gas. Under aerobic conditions the organisms use oxygen as terminal electron acceptor.

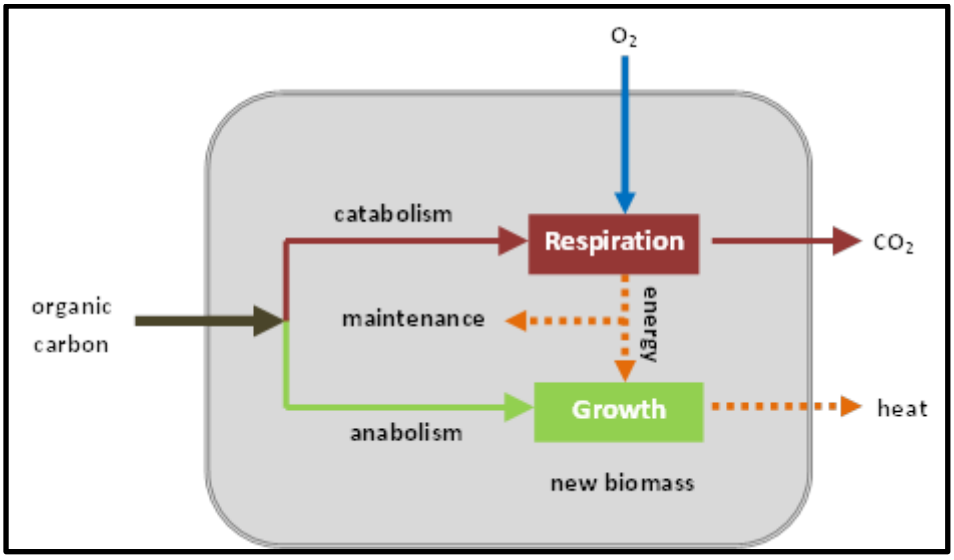


Figure 29 Heterotrophs under aerobic condition

However when oxygen is not available the organisms can revert to available nitrite or nitrate as electron acceptor.

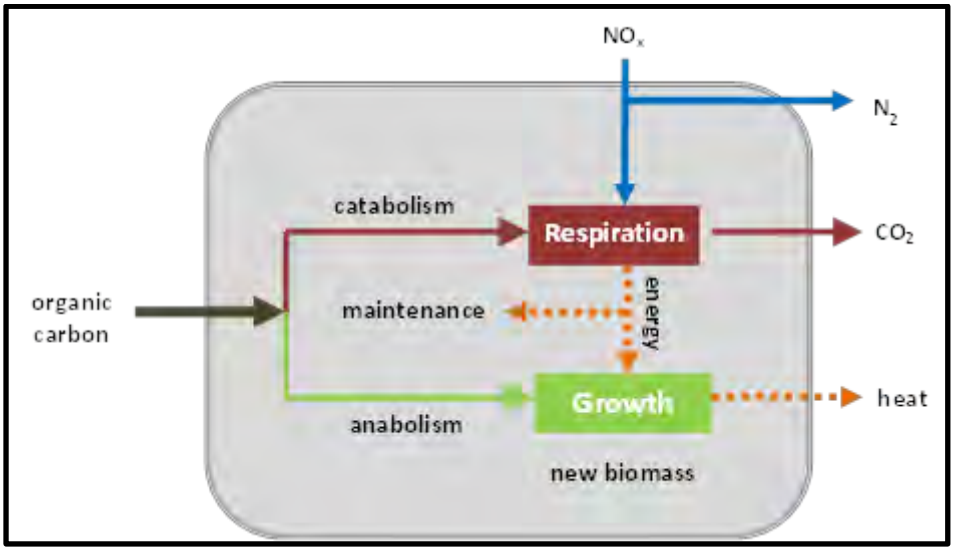


Figure 30 Heterotrophs under anoxic conditions

These organisms are therefore modelled as one entity which has different sets of attributes for aerobic and anoxic conditions.

Autotrophs

Organisms of this group are responsible for nitrification under aerobic conditions. Autotrophic nitrifiers can be subdivided into two types. The first type is the ammonia oxidisers which convert ammonia to nitrite.

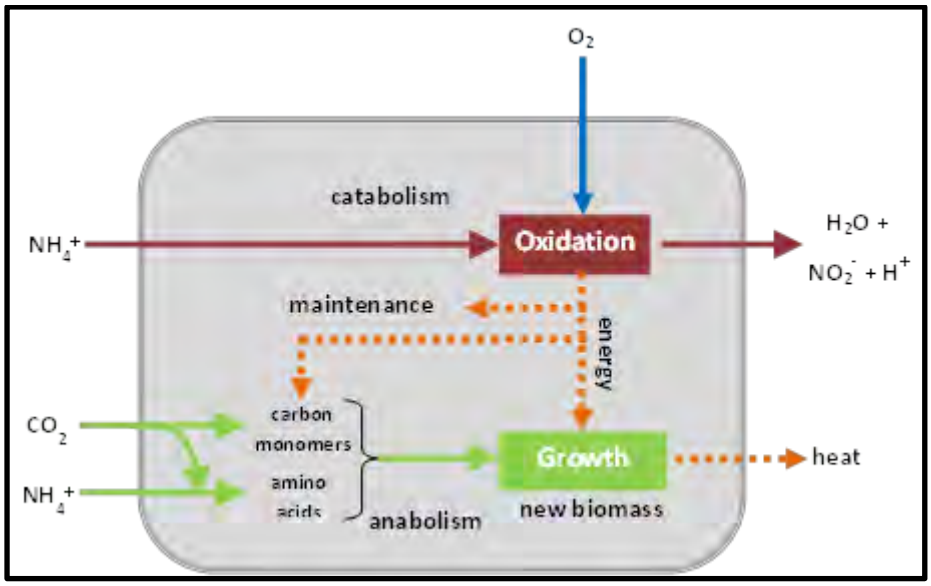


Figure 31 Autotrophic nitrifier: ammonia oxidisers convert ammonia to nitrite

The second type of autotrophic nitrifiers is nitrite oxidisers which convert nitrite to nitrate. These organisms are therefore inherently dependant on ammonia oxidisers if there is no other source of nitrite in the system.

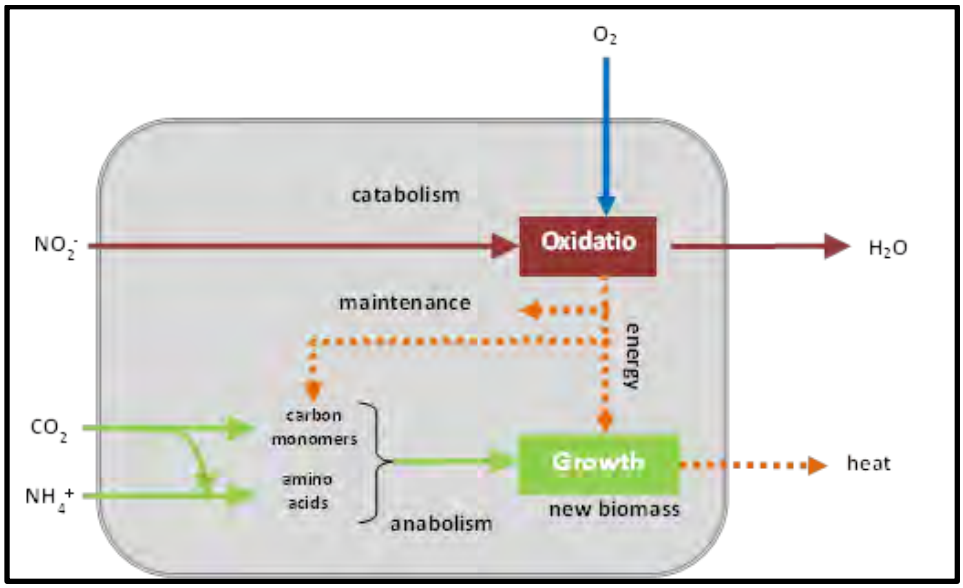


Figure 32 Autotrophic nitrifier: nitrite oxidisers convert nitrite to nitrate.

Nitrification is normally modelled as one step in which the amount of ammonia oxidised is considered proportional to the amount of nitrate formed without any residual intermediate nitrite. However in the urine treatment system it is evident that full nitrification does not always result (Chapter 5), which necessitates the ammonia oxidisers and nitrite oxidisers be considered as two different entities.

6.3 PROCESS KINETICS

Growth Dynamics

A bacteria population consists of multiple active cells which have a collective biomass. In a favourable environment bacterium multiply which results in population growth. Therefore growth of bacteria biomass can be defined as the increase in biomass through an increase in the number of bacteria cells. Accordingly growth rate can be defined as the increase in bacterial biomass per unit time. The growth and growth rate is related to substrate concentration, uptake and conversion thereof. The growth of bacteria biomass under batch conditions is depicted in Figure 32 and is characterised by 4 main phases.

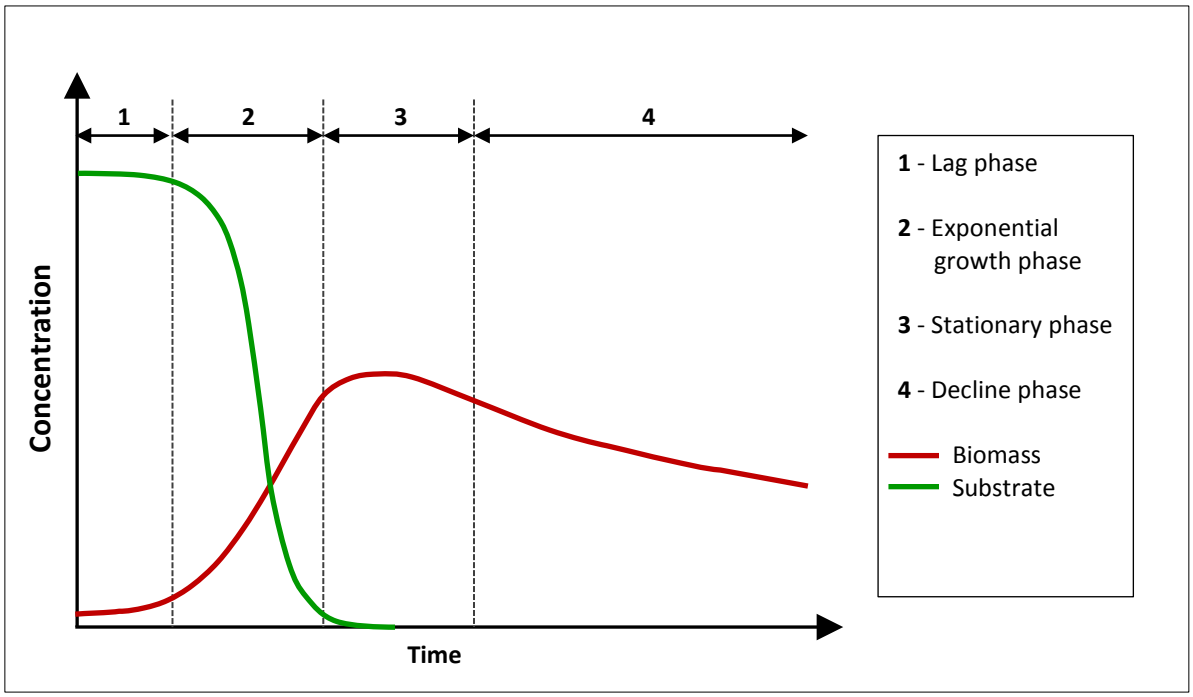


Figure 33 Typical growth curve of bacteria under batch conditions.

The first phase is a period during which bacteria are active but do not reproduce as the organisms are adjusting to the environment (Gerardi, 2006; Bitton, 2005). The amount of substrate utilised during this period is marginal and biomass growth is insignificant. This phase typically occurs under start-up conditions of a new biological system or when a system recovers from toxic conditions.

The transition from the first to second phase is accompanied by substrate uptake as organism begins to synthesise new biomass. During the second phase there is a rapid increase in biomass and a significant decline in substrate. Growth occurs at an exponential rate as substrate is converted to new biomass.

The third phase is characterised by little change in biomass concentration as growth is limited by the low concentrations of substrate. The biomass concentration reaches a peak which reflects the maximum growth capacity for the amount of substrate that was available in the system.

The fourth phase is a period marked by a steady decay in biomass concentration. Substrate is depleted and growth is no longer supported which prompt organisms to enter an endogenous mode.

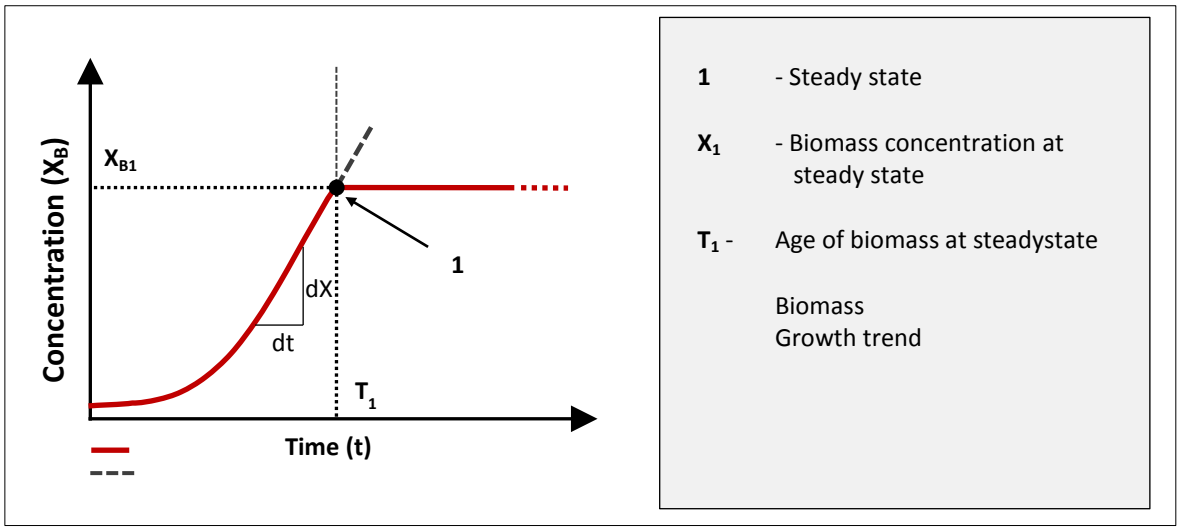


Figure 34 Growth curve steady state reached.

The functioning of activated sludge systems are based on sustaining growing biomass at a regulated concentration with a constant supply of substrate. This is achieved through withdrawal of sludge in amounts that control the limits of biomass concentration whilst keeping a balance between substrate rich influent and treated effluent to maintain a constant system volume. The withdrawal of sludge also regulates the average age of the biomass which is a key element in activated sludge systems as sludge age largely defines the characteristics of a treatment system.

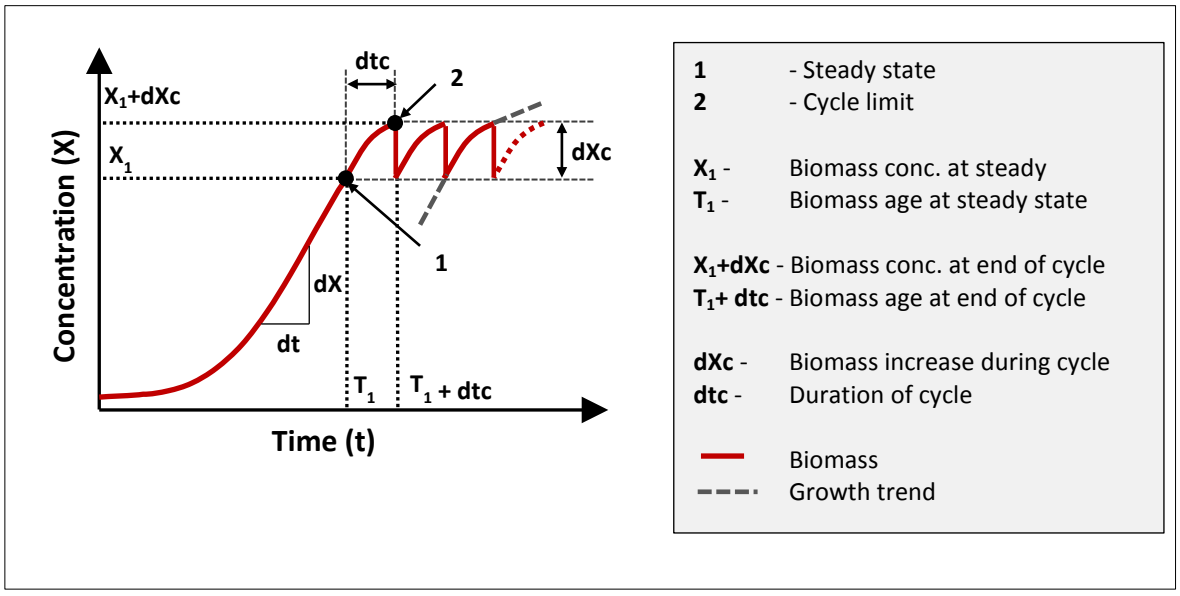


Figure 35 Periodic withdrawal of sludge.

In a continuous flow activated sludge system, biomass is withdrawn on a continuous basis. The biomass concentration will grow in a similar manner as illustrated in the bacterial growth curve until reaching a steady state where the rate of biomass production is equivalent to the rate of biomass withdrawal. Hence there is no significant change in biomass concentration after steady state is reached (Figure 33).

In a batch system biomass is controlled through the periodic withdrawal of sludge. This results in cyclical increases of biomass concentration during the periods between sludge withdrawals (Figure 35).

Kinetic equations

Consider a system with fixed growth conditions containing a biomass concentration with uniform characteristics. Assume that growth in biomass is on-going and that changes in substrate have a negligible effect. The change in concentration over a period of time is:

$$dX = X_1 - X_0 \qquad \text{(Eq. 6.1)}$$

dX = change in biomass concentration (mgX/l)
X_0 = initial biomass concentration (mgX/l)
X_1 = biomass concentration after a period of time (mgX/l)

The rate of change in biomass over the period of time is proportional to the change in biomass and the duration of the growth period:

$$\frac{X_1 - X_0}{\Delta t} = \frac{dX}{dt} \qquad \text{(Eq. 6.2)}$$

Δt = duration of growth period (d)
dX/dt = rate of change in biomass (mgX/ℓ/d)

X_1 can be expressed as a function of X_0 and dX/dt by rearranging Equation 6.2:

$$X_1 = X_0 + \frac{dX}{dt} \cdot \Delta t \qquad \text{(Eq. 6.3)}$$

With a constant rate of change, the biomass concentration at any point in time after the initial concentration can be determined as:

$$X_t = X_0 + (dX/dt)\cdot t \qquad \text{(Eq. 6.4)}$$

The rate of change in biomass concentration (dX/dt) is a function of the microbial growth rate and decay rate. Therefore the rate of change in population can be denoted as:

$$dX/dt = \mu\cdot X_0 - b\cdot X_0 = X_0\cdot(\mu - b) \qquad \text{(Eq. 6.5)}$$

μ = biomass growth rate (mgX/mgX.d = 1/d)
b = biomass decay rate (mgX/mgX.d = 1/d)

Substituting 6.5 into equation 6.4:

$$X_t = X_0 + X_0\cdot(\mu - b)\cdot t \qquad \text{(Eq. 6.6)}$$

Equation 6.3 forms the basis of modelling bacterial growth for the system. The difficulty however is the variation in the system conditions over time. These variations result in continues fluctuation of the kinetic rates μ and b which ultimately means that the rate of change in biomass concentration (dX/dt) also varies over time.

Therefore the change in organism population over long periods (e.g. 24 hours cycles) during which system conditions vary, cannot be determined accurately for a single set of values for μ and b over the period duration.

A solution is found by implementing a method of time step integration in which a modelling period is divided into incremental time intervals. The value of dX/dt is determined for every time increment based on the μ and b for the residing system conditions during the given incremental period. The cumulative X_t is determined by selecting X_t from the preceding increment as X_0 in the following time increment:

$$X_{i\cdot t} = X_{(i-1)\cdot t} + X_{(i-1)\cdot t}\cdot(\mu_i - b_i)\cdot t\cdot i \qquad \text{(Eq. 6.7)}$$

- i = time increment number
- t = time increment duration (d)
- $X_{i\cdot t}$ = mass of bacteria at end of i^{th} time increment (mg/l)
- $X_{(i-1)\cdot t}$ = mass of bacteria at beginning of i^{th} time increment/end of $(i-1)^{th}$ time increment (mg/l)
- μ_i = average growth rate during time increment i
- b_i = average decay rate during time increment i

Growth rate

The growth rate can be sensitive to various environmental conditions. The application of a kinetic model to describe the organism growth in activated sludge must therefore be capable of including the various factors which affect growth rate. There are several models that have been developed to describe suspended cell microbial growth however it is the model proposed by Monod (1949) that has been widely adopted activated sludge models. The original Monod equation relates growth rate to a single growth-controlling (or growth-limiting) substrate through a maximum specific growth rate (μ_{max}) and an affinity constant (Ks) for the specific substrate:

$$\mu = \mu_{max}\cdot\frac{S}{K_s + S} = \mu_{max}\cdot f(S) \qquad \text{(Eq. 6.8)}$$

- μ = specific growth rate (1/d)
- μ_{max} = maximum specific growth rate (1/d)
- S = limiting substrate concentration (mg/l)
- K_s = substrate half-saturation (affinity) constant (mg/l)
- $f(S)$ = saturation function for substrate

The growth of microbes in activated sludge systems can however be affected by numerous growth limiting substances. Therefore the original Monod equation has been refined in order to incorporate constants and terms which describe the possible effect of additional growth limiting nutrients other than a single substrate. The terms have the same form as the Monod term. The effect of dissolved oxygen concentration for example is introduced into the equation:

$$\mu = \mu_{max}\cdot\frac{S}{K_s + S}\cdot\frac{S_O}{K_O + S_O} = \mu_{max}\cdot f(S)\cdot f(O) \qquad \text{(Eq. 6.9)}$$

- S_O = dissolved oxygen concentration
- K_O = dissolved oxygen affinity constant
- $f(O)$ = saturation function for dissolved oxygen

A saturation function has a value between 0 and 1 where a value of zero reflects complete limitation on growth and a value of one signifies no limitation on growth by the specific substrate.

Additionally physical and chemical factors, such as temperature and pH, affect growth rate. The effect of temperature changes can be incorporated by means of an exponential expression:

$$\mu=\mu_{max}(20°C)\cdot e^{(\kappa\cdot(T-20))} \qquad (Eq.\ 6.10)$$

$\mu_{max}(20°C)$ = maximum specific growth rate at 20 degrees Celsius
K = temperature constant

There are many (thousands) chemical reactions involved in catabolism and anabolism but it amounts to three main processes relevant to activated sludge systems:

- Assimilation
- Respiration
- Biomass production

In activated sludge systems wastewater characteristics determines the amount and type of substrate available. In turn the substrate will generally define which type of organisms can grow as well as the required environmental conditions to be maintained for optimal process performance. System configurations and operation strategy also make it possible to accommodate numerous types of organisms in a system so as to facilitate various treatment processes.

6.4 COMPUTATIONAL TOOLS

Figure 35 gives a graphical representation of how the model was implemented in spread sheet format. Based on bacterial populations that change in biomass concentration over time, the changes in substrate concentrations are determine. These changes are brought about by change in growth and decay rate which are as a result of various influencing factors, including pH, alkalinity and temperature and oxygen concentration. The substrate concentrations could in this system mostly be assumed *not to be* the rate limiting factors, as substrate concentrations were always high. To model or predict a function of various factors requires that bacterial growth closely approaches the estimation of organism concentration (X_2) after a given time based on the initial concentration (X_1). The increase in biomass is therefore a function of the initial concentration and a factor which describes the rate of increase. Separate growth rates and growth functions were used in the modelling of heterotrophic organisms, ammonium oxidising organisms and nitrite oxidising organisms. Changes that occur due to growth and substrate utilisation are introduced into the next iteration. In this way, the consumption of alkalinity by growth of ammonium oxidising organisms becomes one of the important rate limiting factors, practically ceasing nitrification at low alkalinity. The spread sheet model is available as an electronic addendum to this report.

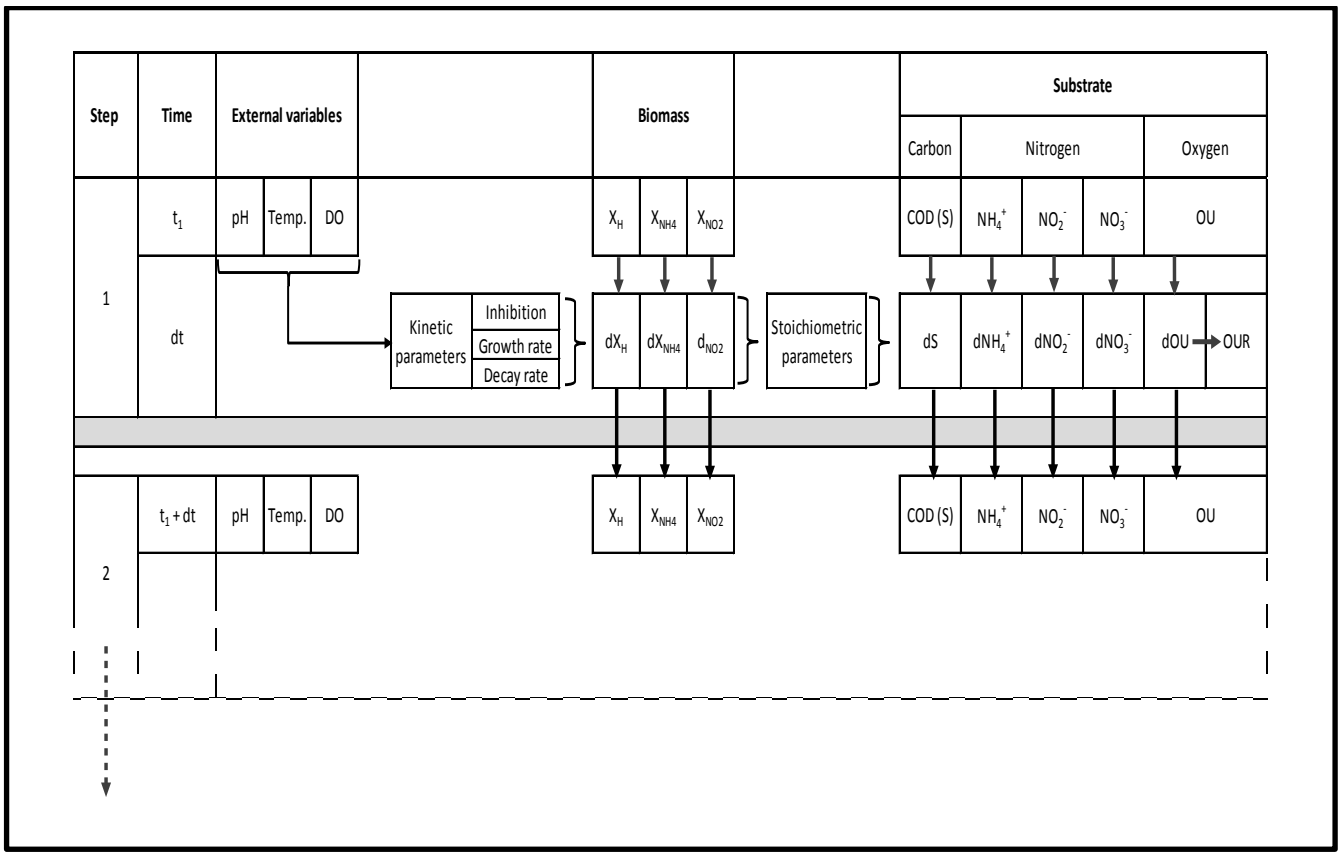


Figure 36 Graphical representation of the iterative process modelling structure

6.5 THE ROLE AND IMPORTANCE OF PH

pH plays an important role because nitrification is directly affected by pH. When pH is out of the range 6.5-7.5, nitrification is inhibited.

Figure 36 shows the measured increase in pH up to 8.5 immediately after the system is fed. This is because of the addition of high concentrations of ammonium (NH_4^+ species) and bicarbonate which are both hydrolysis products of urea. Nitrification slowly takes place resulting in the pH decreases due to the production of H^+ species. When the pH reaches 7.5, the rate of nitrification consequently increases until a pH of ~6.5, whereby the rate of nitrification decreases. The nitrification rate remains constant with constant pH of 6.

The value of pH was not modelled, but to simplify the model, the measured pH values were imported into the model, to govern all growth kinetics and inhibition, as well as ammonium-ammonia and nitrite-nitrous acid speciation. In Figure 36 the other parameters, including ammonium, nitrite, nitrate and COD are all calculated based on the mathematical model for the urine nitrification-denitrification system.

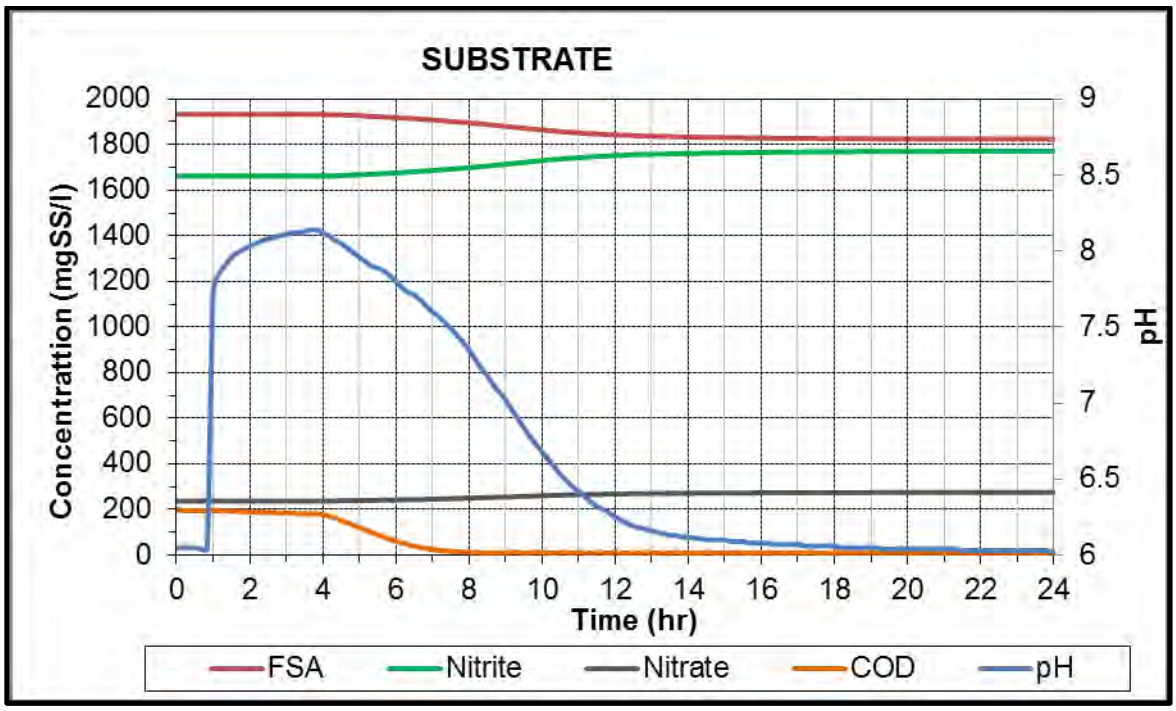


Figure 37 Importance of pH on modeling.

6.6 SIMULATIONS

Further Oxygen Utilization Rate (OUR) model simulations are shown and explained in graphicals below:

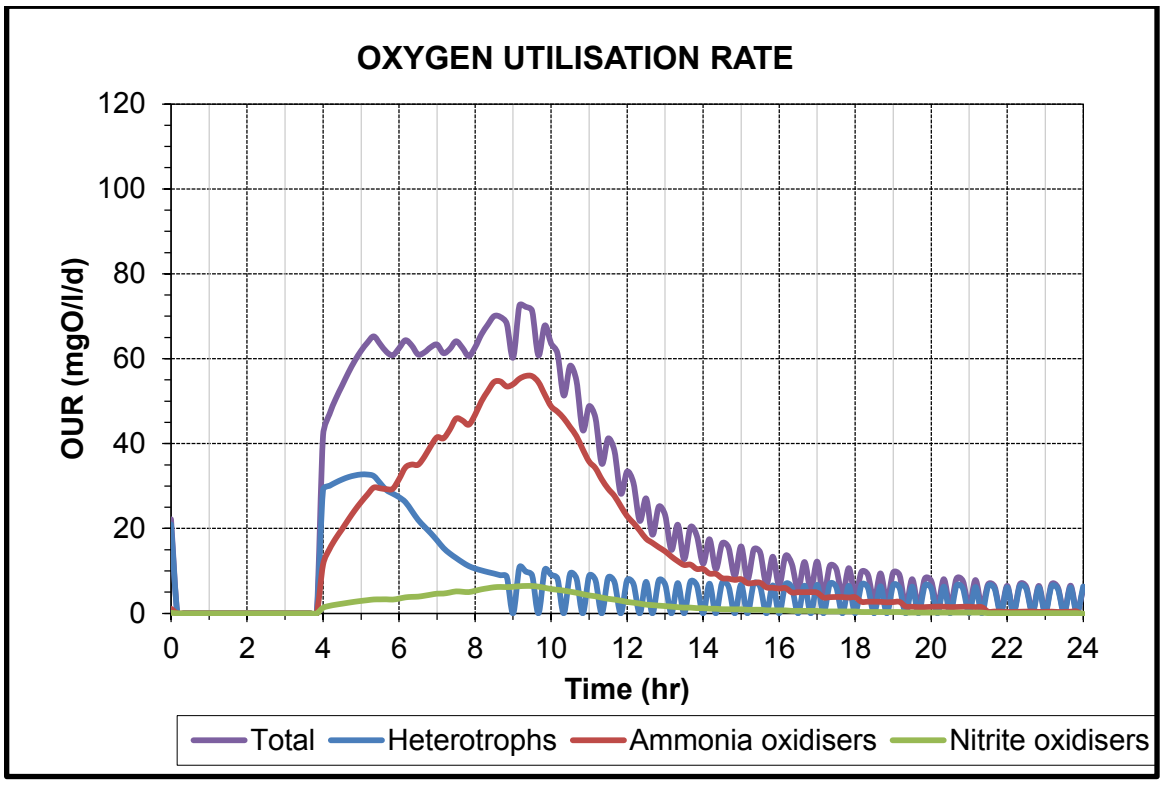


Figure 38 OUR model simulations.

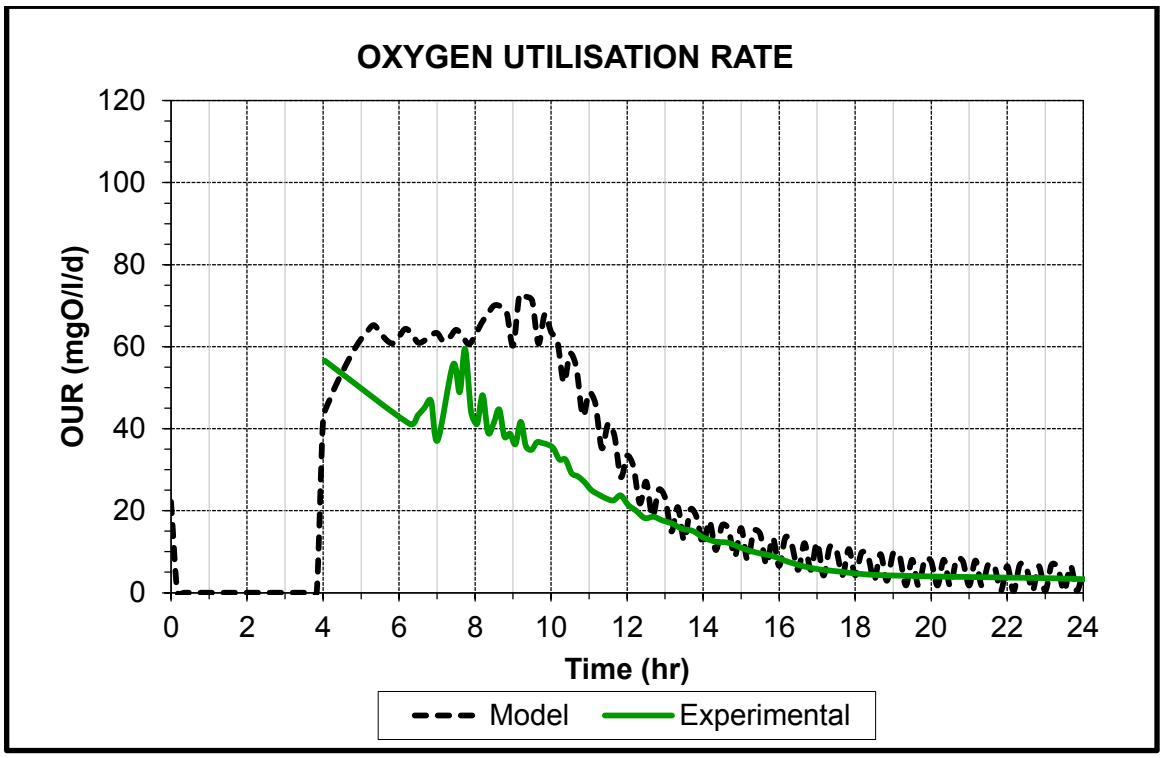


Figure 39 OUR model simulations.

The model predicts that the OUR will remain zero during the anoxic cycle and this is confirmed by the experimental results. The model predicts that at the start of the aerobic phase, the OUR will sharply increase and remain constant for about 6 hours, then decrease exponentially thereafter.

However the experimental results show a non-constant exponential decrease of the OUR, during the initial 6 hours of the aerobic phase, which is slightly different from the model prediction. This difference could be due to the type and degradability of COD. For instance, the model assumes readily available soluble COD and biodegradable particulate COD as substrates. In reality, however, there are many different substrates, from short chain fatty acids to complex molecules such as hormones and medicine rests, which all have different degrees of biodegradability. One should therefore expect that the model and the measured OUR values do not coincide too well. However, after the initial aerobic period, when biodegradable COD left in the system after the anoxic phase had been consumed, the model values and the measures OUR show a good fit.

7 NITRIFICATION/DENITRIFICATION OF URINE WITH WASTE ACTIVATED SLUDGE IN AN ANOXIC/AEROBIC DIGESTOR

7.1 INTRODUCTION AND AIMS

In the context that urine separation and collection is implemented, the collected urine would still have to be treated. Urine can be treated in novel biological systems, such as the combined Sharon-Anammox processes (Wilsenach, 2006). However, these are complex treatment systems and should be implemented with caution in developing countries with low skills levels. A much simpler system may be available, which is investigated in this chapter.

Vogst and Ekama (2012) described the anoxic-aerobic digestion of waste activated sludge. In this process, biomass decays and undergoes hydrolysis that releases nutrients (nitrogen from cell mass, and phosphate from polyphosphate storage products). During the nitrification stage ammonium is oxidised to nitrate, which is denitrified during the anoxic stage, with COD released during sludge hydrolysis. This system is fairly easy to operate and can be incorporated into existing wastewater treatment infrastructure. This chapter further investigates the anoxic-aerobic sludge digestion process at room temperature (20^0C), neutral pH of 7 and a long sludge age (20d) to which urine from no-mix flush toilets were added (~1,500 mg N/l). The system was operated and studied with respect to:

- Stabilization of waste activated sludge (WAS) by breaking down the organic biomass through anoxic-aerobic digestion.
- Oxidation of carbonaceous material measured as Chemical Oxygen Demand (COD) to Carbon Dioxide (CO_2) and Water (H_2O).
- Removal of nitrogen from added urine and that released from sludge digestion, by nitrification of ammonium (NH_4^+) to nitrate (NO_3^-) during the aerobic phases of system operation and de-nitrification of nitrate to nitrogen (N_2) gas during the anoxic phase.
- Precipitation of phosphate from added urine and that released from sludge digestion, as $NH_4MgPO_4.6H_2O$, $Mg_3(PO_4)_2$ and/or $Ca_3(PO_4)$.

7.2 MATERIALS AND METHOD

Experimental set-up

A 12L laboratory scale anoxic-aerobic digester (Figure 39) was operated at a 20d sludge age, 20^0C temperature over 18 months, during which time 15 raw sewage batches from Athlone wastewater treatment works were added to the parent Biological Nutrient Removal (BNR) system. A total of 10 out of the 15 sewages batches represent steady state operation of the AAD.

Control timers were used on the digester to switch air on and off over 3h intervals, so that the total aeration stage was 12h/d. The digester was fed with 06 ℓ/d of urine from CSIR's no-mix toilets and 0.6 ℓ/d of activated sludge (thickened to 2%) from a biologically enhanced phosphate removal UCT process operated in the same laboratory. A waste sludge rate of 0.6 ℓ/d was maintained for 20d sludge age in the anoxic-aerobic digester. Apart from the direct removal of WAS (0.6 ℓ/d) a further 0.6 ℓ/d was removed, which was centrifuged. The solid material was returned to the anoxic-aerobic reactor, and the supernatant was discarded to make room for 0.6 ℓ/d of urine.

A dissolved oxygen (DO) meter was installed and the system DO was controlled between a low set concentration of 2 mg/l and a high set concentration of 5 mg/l. Two solenoid valves controlled

aeration of the digester. The first air solenoid valve opened and closed the air line on 3 hour intervals to control anoxic and aerobic conditions. The second solenoid valve in series with the first was controlled by the OUR recorder. The valve would open at the low DO set-point and close at the high DO set-point.

The digester concentration was maintained constant by mixing with a stirrer driven by an electrical motor. Both the influent (urine and activated sludge) and effluent streams were analysed for chemical oxygen demand (COD), magnesium (Mg), Calcium (Ca), Potassium (K), N, P, alkalinity, Total suspended solids (TSS) for effective digester operation and performance monitoring.

Figure 40 Anoxic-aerobic digester, with stirrer, air pipes and OUR recorder

Feed characterization

Average results of parameters measured on the WAS from the UCT parent process are presented in this chapter. From the raw sewage results measured, the following WAS concentration characteristics were calculated for each of the sewage batches: Polyphosphate Accumulating Organisms (X_{BG}), Heterotrophic Organisms (X_{OHO}), Autotrophic Nitrifying Organisms (X_{ANO}), PAOs endogenous residue (X_{EG}), OHOs endogenous residue (X_{EH}), Un-biodegradable particulate COD (X_{ii}), VSS (X_V), Active biomass fraction (f_{ai}) COD/VSS (f_{cv}) ratio, TKN/VSS (f_n) ratio, TP/VSS (f_P) ratio. The average WAS measurements and calculated parameters are used to characterise the WAS feed. The calculated f_{cv}, f_n and f_p are compared to the accepted experimental values of 1.48 mg COD/mgV SS, 0.1 mg

TKN/mg VSS and 0.025 mgTP/mg VSS within a statistical range of ±10% and used as a basis for validation of the experimental results and calculated parameters.

The high COD, TKN, TP, VSS and TSS values were obtained taking into consideration the centrifugal thickening of the WAS from the parent reactor by a factor of 2. For example, to calculate the TKN concentration, the following formula was used for each daily parent system WAS measurement:

TKN feed = 2x(TKN_unfiltered – TKN_filtered) + TKN_filtered

Measurements on the AAD

Wastewater measurements were conducted on the AAD WAS to determine AAD operational parameter concentrations. The following parameter concentrations were measured: Unfiltered COD, Filtered COD, Unfiltered TKN, Filtered TKN, FSA, Unfiltered TP, Filtered TP, OP, Unfiltered K, Filtered K, Unfiltered Mg, Filtered Mg, unfiltered Ca, filtered Ca, VSS, TSS, pH, alkalinity and Total OUR_t. The average results are presented in this chapter for each of the 10 steady state batches.

First principles steady state calculations

From the parent reactor WAS, collected urine feed characteristics and measured AAD parameters, steady state first principle mass balance calculations were carried out to determine AAD performance with respect to organic matter degradation (Active OHO biomass stabilization), TKN, TP, Potassium (K), Magnesium (Mg) and Calcium (Ca) removal for each of the 10 raw sewage and no-mix urine batches. Mass balances within ±10% statistical residue were accepted as valid.

Organic matter degradation (Active OHO stabilization) performance of the system was evaluated using South African Guidelines for the Utilization and Disposal of Wastewater Sludge: Volume 1 (WRC, 2006). The guidelines are formulated under the following South African Acts: *The National Water Act (Act of 1998), The Water Act (Act 54 of 1956), The environment conservation Act (Act 73 of 1989), The fertilizers, farm feeds, Agricultural remedies and stock remedies Act (Act 36 of 1947), The conservation of Agricultural resource act (Act 43 of 1983), The National Health Act (Act 64 of 2003), The Water services act (Act 108 1997), The national environmental management act (Act 107 of 1998).*The guidelines specify a VSS fraction (f_{sr}) vector attraction reduction of > 38% for both aerobically and anaerobically digested sludge's.

7.3 RESULTS

A range of 90% < mass balance < 101% over the AAD was achieved for COD for all of the 10 steady state WAS and urine feed batches. A range of 87% < N mass balance >104% was obtained for N with only one batch falling out of the acceptable range by ±3%. Good P mass balances were achieved for only 4 batches at an average of 100% the rest of the P mass balance was out of range. The overall batch mass balance ranged 62%< P mass balance<128%. Good balances were achieved for K for 9 batches with only batch 10 falling out of ranged due to missing experimental data. The K mass balance ranged from 85% to 111%. Good Mg mass balances were obtained for only 4 batches and this ranged from 61% to 118%. Relatively poor Ca mass balances were achieved an overall batch mass of 65%. Only batches 1 and 10 achieved good Ca mass balances of 100% and 92% respectively.

7.3.1 UCT BEPR WAS and urine feed characterization

The detailed average feed characterization for each of the 10 feed batches is presented in Addendum C for WAS and no-mix urine.

The mean COD concentration of the 10 batches was ~14,483 mg COD/l. The average highest batch COD value (batch 8) varies with the mean COD by 3% and the lowest value by 2%. This shows that the organic matter characteristics of the raw sewage batches were similar. This is shown by the relatively similar COD/VSS ratio values (Addendum C, Table 29) that range from 1.32 mg COD/mg VSS to 1.43 mg COD/mg VSS. All the f_{cv} values are within 10% of the widely accepted value of 1.48 mg COD/mg VSS.

The average TKN across all batches was ~ 969 mg TKN/l. The variation of the average TKN batches around the mean is larger at about 20% for the maximum and minimum values. However the TKN/VSS ratios of all sewage batches were found to be within 10% of the widely accepted mean value of 0.1 mg TKN/mg VSS. Most values were within 1% of 0.1 TKN/mg VSS. The WAS TP concentration in BEPR systems is highly dependent on the phosphate uptake capacity by the Polyphosphate organisms (PAOs) in the aerobic zone of the parent reactor, which in turn is dependent on concentration of the Volatile Fatty Acids (VFAs) and fermentable Readily Biodegradable Organics (FRBCOD) in the raw sewage (Wentzel et al., 1990). Phosphate uptake can reach a maximum of 0.38 mg PolyP/mg VSS (Wentzel et al., 1990). The TP/VSS ratio of the batches ranged from 0.06 mg TP/mg VSS to 0.20 mg TP/mg VSS. Batches 1 and 2 were found to have the least f_p values (<0.1 mg P/mg VSS) and bathes 3 to 10 obtained f_p values >0.1. This is characteristic of WAS from BEPR systems.

For the yellow water (flushed urine) average unfiltered and filtered COD concentrations of ~ 1,200 mg/l and 1,100 mg/l respectively was measured across all batches. This implies ~ 10% of the COD within the urine batches was solid. This could be due to impurities in the no mix urinals. Averages of ~1,200 mg/l and ~1,000 mg/l of TKN and FSA respectively as N were measured across all batches. Most of the 20% of the particulate TKN could be due to organic impurities, from cross contamination, in the no mix urinals. Some of the N could be from precipitated ammonia (<0.6%) (Udert et al,.2003) as Struvite ($MgNH_4PO_4.6H_2O$).

The urine composition results show evidence of urine precipitation, most probably as Calcite ($CaCO_3$), Hydroxyapatite ($Ca_5 (PO_4)_3 (OH)$ and/or struvite ($MgNH_4PO_4.H_2O$) (Udert et al., 2003). Precipitation of Struvite is most common in no-mix urinals (Udert et al., 2003). This is confirmed by the results as they show about 80% of Magnesium (Mg) in solid form in the measured urine batch samples. About 70% of the Ca was found in solid form. This could indicate the presence of hydroxyapatite. Almost all the phosphate (P) in the urine precipitates either as Struvite or Hydroxyapatite (Gujer et al., 2003). Filtered P samples were not measured, but the measured unfiltered P concentration is assumed to be closely equal to the measured unfiltered sample. Measuring and classification of the P precipitates concentration would require special sampling and measurement procedures and were not part of the objectives of this research study.

7.3.2 Organic matter removal and WAS stabilization

Waste activated sludge (WAS) from biological nutrient removal (BNR) systems contains high concentrations of biodegradable and un-biodegradable organic matter. The un-biodegradable organic

matter comprises of influent particulate COD (S_{upi}) and the endogenous residue (X_{EH}). The S_{upi} accumulates in the BNR system over the sludge age. The X_{EH} is generated from active biomass degradation at the rate 0.048/d and accumulates in the BNR over the sludge age. The un-biodegradable particulate exits the BNR via the waste activated sludge stream. However, WAS from BNR systems also contain biodegradable organic matter incorporated in the active organism mass (MX_{BH}). BNR systems with relatively shorter sludge ages (<5 days) have relatively high X_{BH} fractions (f_{ai}) and need to be stabilized before discharge into the receiving environment. During the aerobic digestion, the biomass organic matter is broken down and the organics are released to the bulk liquid. The UCT parent system was operated at an SRT of ~10 days. 0.048/d of the broken down biomass accumulates as endogenous residue and is enmeshed in the sludge mass with the digester S_{upi}.

Figure 40 shows the AAD average concentration of the mixed liquor and final effluent COD measured per batch of sewage added to the parent BNR system. The mixed liquor COD within the AAD remained in the range 10,000 mg/l-12,000 mg/l, depending on different parent sewage and WAS batches. The final effluent COD concentration ranged at 100-150 mg/l at an average of 141 mg/l over the 10 WAS and urine batches. The average filtered influent COD concentration was ~65 mg/l. This indicates urine contains some soluble un-biodegradable (~8%). The effluent COD concentration does not meet the legislation requirements of 75 mg/l. However the digester effluent would normally be recycled to the inlet works, or if it still contains nitrate, directly to the anoxic zone of the main stream activated sludge process.

To further evaluate the AAD performance, mass balance calculations of experimental results (detail per batch included in Addendum C) over the AAD were conducted. The average FSt_i of the 10 batches was 9,300 mg/d and FS_{te} 6,700 mg/d. The difference between the FS_{ti} and FS_{te} gives the percentage of COD released by the biomass into the bulk liquid (around 28%). It was assumed that urine contains only biodegradable organics. This biodegradable organics are used for new biomass synthesis during the aerobic phase of AAD operation. However, the urine organics were later used for de-nitrification when the urine feed was added during the anoxic phase of the AAD.

Oxygen utilization rate of the digester was measured through a computer program. The program plots a straight line graph of the utilization rate at every aerobic phase of AAD operation.

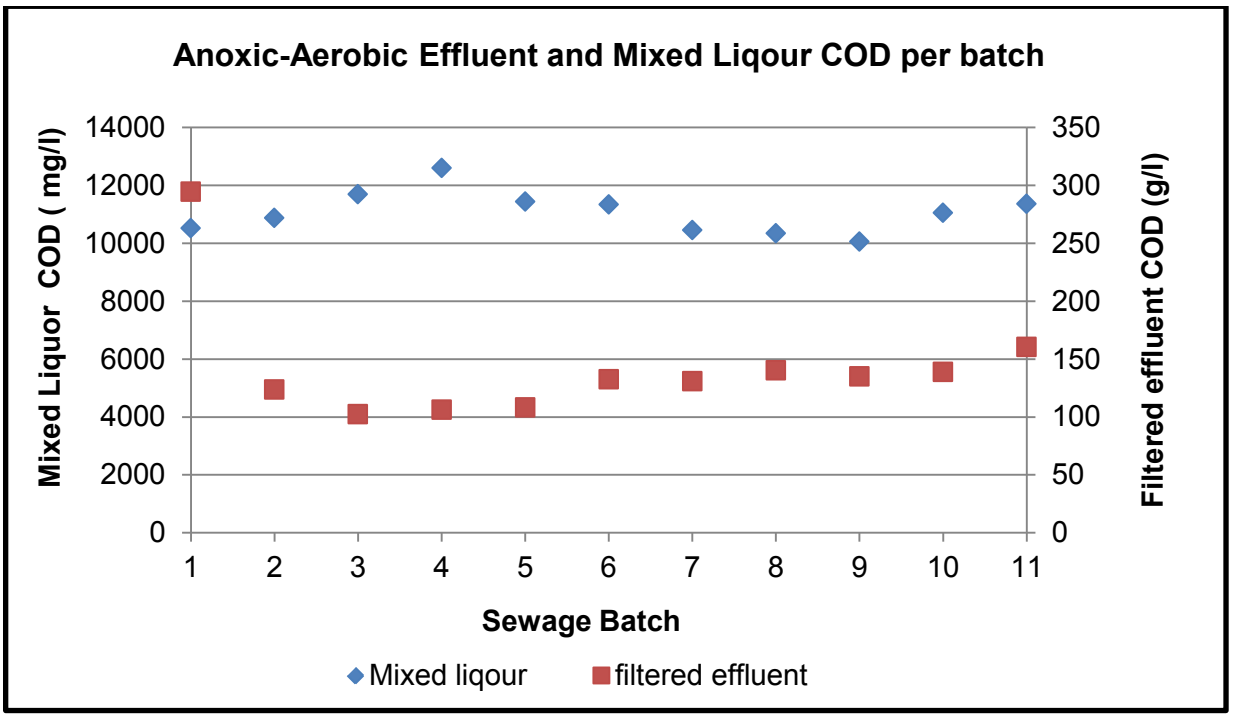


Figure 41 Average AAD effluent and mixed liquor COD concentrations measured

Figure 41 shows a typical OUR graph that was obtained during AAD steady state operation. The area under the graph represents the total mass of oxygen (FO_t) utilized for both nitrification and organic matter degradation. The vertical line that is displayed by the graph at time t_0 most likely represents oxygen utilized for new biomass synthesis from the COD in urine.

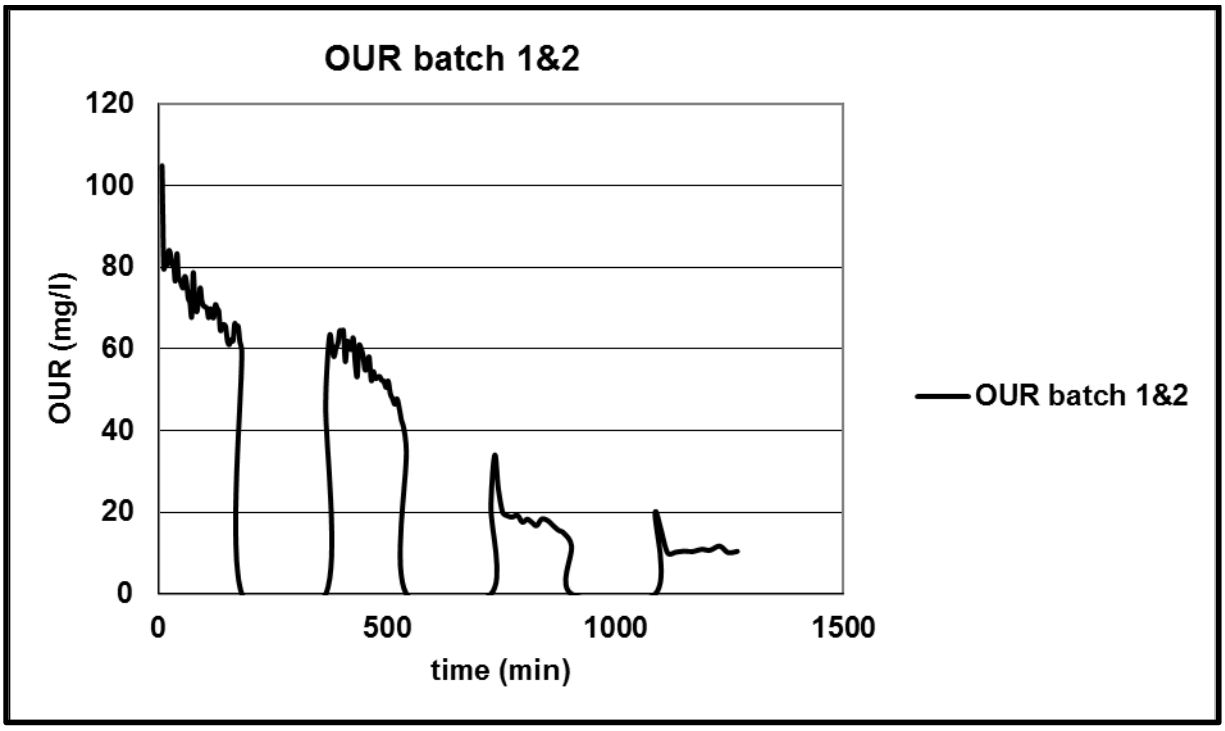


Figure 42 Typical steady state OUR graph

According to the activated sludge growth model the specific growth rate of biomass (cell synthesis) is a function of substrate concentration surrounding the organism with a specific biomass yield

coefficient (~0.45 mg VSS/mg COD). The second phase of the model explains the organism degradation rate which is constant at 0.24/d at room temperature (Ekama et al., 1986). The OUR is modelled as a function of the biomass degradation rate and growth. The higher the measured urine feed COD concentration the higher the OUR and hence $FO_{synthesis.}$ From OUR measurements, the total average FO_n required for biomass endogenous respiration was found to be ~2163 mg O/d. The OUR_n is always constant in the AAD. As shown in Figure 41, this rate is constant at ~14.5 mg/l.h. The expected FO_{Ncalc} was also calculated from the aerobic digestion model (Ekama et al., 1986). The model shows that the FO_n required for aerobic digestion of WAS is a function of influent active biomass (FX_{BHi}). The measured and calculated FO_n averages vary. This is due to the fact that the measured WAS and calculated VSS vary. Measured values depict high VSS. Mass balances were obtained from the $FS_{ti,}$ FS_{te}, $FO_{synthesis}$ (utilizing COD from urine) and measured FO_n (endogenous respiration).

This was used to calculate the OUR for urine organics for new biomass synthesis for each of the 10 urine batches, which yielded an average mass balance of 100.4%, with maximum of 113% and a minimum of 93%. The detail values are included for each batch in Table 32 of Addendum C. The good COD balance indicates the validity of the experimental measurements and hence conclusions drawn from the experimental results.

Table 12 shows calculated steady state operational parameters. The calculated and measured steady state volatile suspended solids varies significantly (>20%) for all batches, except for batches 2, 5, and 9 that vary within 5% of the statistical mean value. The measured values however give result in better COD balances. The COD/VSS ratio calculated for most sludge batches was within 10% accuracy (1.30-1.42 mg COD/mg VSS) of the widely accepted value of 1.48 mg COD/mg VSS. Furthermore, the AAD TKN/VSS ratio ranged from 0.07 mg N/mg VSS to 0.1 mg N/mg VSS, with a mean of 0.08 mg N/VSS for all batches. These indicators validate the accuracy of the experimental results.

Table 12 Calculated AAD average volatile suspended solids concentration

Batch	X_{BH} (mg/l)	X_{EH} (mg/l)	X_{ii} (mg/l)	Calculated X_V (mg/l)	Measured X_v (mg/l)	f_{ae} (%)	f_{sr} (%)
1	805	304	5072	6181	7549	0.107	32
2	713	303	6742	7758	7980	0.089	37
3	754	308	5151	6213	8732	0.086	21
4	776	506	5387	6669	8732	0.089	33
5	791	396	8359	9547	9768	0.081	37
6	557	285	5775	6616	9827	0.057	20
7	520	275	4925	5720	8077	0.064	26
8	694	414	8359	9467	7298	0.095	53
9	628	274	6663	7565	7487	0.084	45
10	539	419	7115	8074	8564	0.063	40

An average of 12,9680 mg VSS/l was measured in the influent WAS. Based on the measured influent WAS VSS and AAD VSS, of each batch, the average removal of the f_{sr} values shown Table 12 was 34%.

The endogenous residue results from broken down WAS X_{BH} and endogenous respiration of the new biomass that grew from urine biodegradable organics. A VSS destruction of 38% conforms to the legislative requirements for digested sludge (WRC, 2006). Moreover, the average active WAS fraction (f_{ae}) was at an average of 8%. This result shows a very stable sludge and is well within maximum legislative requirements (WRC, 2006), hence the AAD performance with regard to WAS stabilization is very good.

7.3.3 Nitrogen removal

The AAD was operated intermittently with alternative 3h aerobic and anoxic periods. The WAS feed contained an average of 3 mg N-FSA/l as soluble and 0.1 mg N-OrgaN/mg VSS stored in the biomass. This N was released to the digester bulk liquid during endogenous respiration and nitrified during the aerobic phase to nitrate.

During the initial setup of the AAD system, the 12L reactor was filled with thickened WAS (2%) form a BNR BERPR system. Urine was added at daily increments of 50 ml to progressively build Autotrophic Nitrifying (ANO) mass. This continued over 30 days whereby 0.6 L of urine was added. The specific growth rate of ANOs is relatively lower (~0.1 mg VSS/mg N) relative to that of OHOs (0.45 mg VSS/mg COD). However with increasing added urine inflow volumes, at 300 ml of urine feed, nitrification slowed down despite maintaining the urine inflow rate. It was realised that with increasing urine volumes, higher hydrogen (H^+) ion concentrations are produced during nitrification resulting in AAD pH decrease to <6. An 80 g/l solution of alkaline sodium bicarbonate was prepared and added to the system to maintain a neutral pH of 7±0.5. An average AAD pH of 7.2 was maintained during the operation period. With digester pH adjustment and control, complete nitrification was achieved for the urine and WAS FSA (1300 mg/l) feed over the steady state operational period.

Through profiling of the system where by 30 minute interval samples were extracted, it was found that at steady state complete nitrification occurs within the first aerobic phase of the AAD operation, after which the FSA concentration remains constant (~3 mg/l). During anoxic phases, FSA was released during endogenous respiration to the AAD bulk liquid. This was visible from the OUR peaks in Figure 42 at the start of every aerobic first phase post the anoxic period. Partial de-nitrification was achieved in the digester with ~40% of the influent N-NO_3 being de-nitrified to N_2 gas. When feeding was switched to the start of the anoxic phase, de-nitrification improved to ~60% of the influent N-NO_3 feed.

The FSA effluent concentration remained in the range 3 mg/l-4.5 mg/l across all batches at steady state operation of the AAD signifying complete nitrification. This was irrespective of the average influent FSA concentration which is line with the activated sludge model for a single reactor (Downing et al., 1964), which states that nitrification is independent of the Influent FSA (N_{ai}). A measured average of 2080 mg O_2/d was used for Nitrification of urine FSA to NO_3^-. The activated sludge model predicts an average of 2863 mg O_2/d. Some FSA was released by the biomass during endogenous respiration. From the OUR graphs, the nitrification of this released FSA is instantaneous as shown by sharp spikes on the OUR graph at the start of each aerobic phase.

Nitrification was further investigated to determine the time frame of this bio-process. Figure 44 shows measurements over 30 minutes intervals during the first aerobic and anoxic phases (3h) for a typical batch.

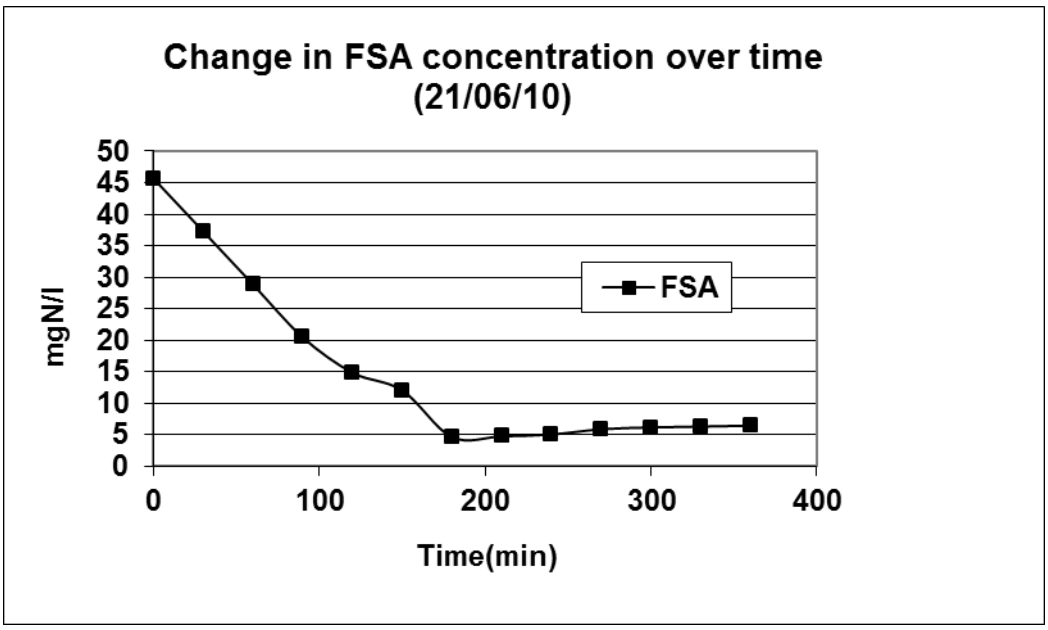


Figure 43 Change in NH_4^+ concentration over time within AAD after feed.

N removal occurred during the four 3h anoxic phases per day (12h) by de-nitrification of the available nitrate (NO_3) with endogenously generated organics. Approximately 0.38/d of the Volatile Settleable Solids (VSS) were degraded in the AAD over the 10 steady state WAS batches and therefore it is expected that the OHO and PAO mass released biodegradable organics at the endogenous respiration rates of 0.24/d (Ekama et al., 1986) and 0.034/d (Wentzel et al., 1990) respectively. To determine the extent of de-nitrification, an N balance was carried out across all batches.

The N balance was calculated as the ratio of the total effluent N ($FTKN_e$) of the total influent N ($FTKN_{in}$) where:

$FTKN_{in} = FTKN_{WAS} + FTKN_{urine}$

$FTKN_e = FTKN_{AAD} + FN_{ne} + FN_{den}$

Hence N balance (%) = $(FTKN_e/FTKN_{in}) \times 100$

The influent N flux (FN_{ai}) which was ~ 830 mg N/d for the 10 WAS and urine batches consisted of the urine FSA flux ($FN_{n\text{-}uri}$) ~ 630 mg/d, and the WAS N ($FN_{a\text{-}rel}$) ~200 mg N/d embodied in the WAS biomass (FX_{BHi}) at an average of ~0.08 mg N/VSS. The effluent N flux (FN_{te}) which was ~ 680 mg N/d consisted of supernatant nitrate flux (FN_{ne}) at ~440 mg N/l.

The difference between the FN_{ai} and the FN_{ne} for each batch was the de-nitrified nitrate flux ($FN_{ndenitrified}$) and this differed across batches. N de-nitrification efficiency of the AAD ranged from 34% to ~69%. Initially it was thought that the N de-nitrification efficiency depended on the influent urine FSA flux ($FN_{n\text{-}uri}$) and the flux of NO_3 generated VSS endogenous respiration (FN_{ti}) since the De-nitrification potential (D_p) would be constant across WAS batches. However as shown in Table 13, higher de-nitrification efficiencies (up to ~60%) can be achieved and lower efficiencies (< 35%) irrespective of the nitrate generated from the urine ($FN_{n\text{-}uri}$).This indicates that D_p of the AAD is characteristic of the influent WAS. The average de-nitrification efficiency of the 10 WAS batches was 50%. This implies 50% of the NO_3 generated within the AAD system will be converted to N_2.

Table 13 N balance carried out across all batches.

Batch	$FN_{a\text{-}rel}$ (mg/d)	$FN_{n\text{-}uri}$ (mg/d)	FN_{WAS} (mg/d)	FN_{ai} (mg/d)	FN_{ne} (mg/d)	$FN_{ndenitrified}$ (mg/d)	$N_{denitrified}$ (mg/l)	% Rem
1	178	848	433	1026	404	622	52	61
2	203	684	429	887	334	553	46	62
3	128	671	436	799	431	368	31	46
4	196	543	469	739	486	253	21	34
5	253	770	451	1023	313	710	59	69
6	108	538	410	645	396	249	21	39
7	114	533	367	647	398	249	21	38
8	258	666	385	924	390	534	45	58
9	287	568	381	855	399	456	38	53
10	280	446	454	726	324	402	34	55

The average N/VSS ratio of the WAS remained constant across the AAD, which indicates good N mass balances. The average mass N balance over 10 batches was 96%

De-nitrification of the AAD was investigated further through 30 min profiles during the aerobic and anoxic phases of the AAD.

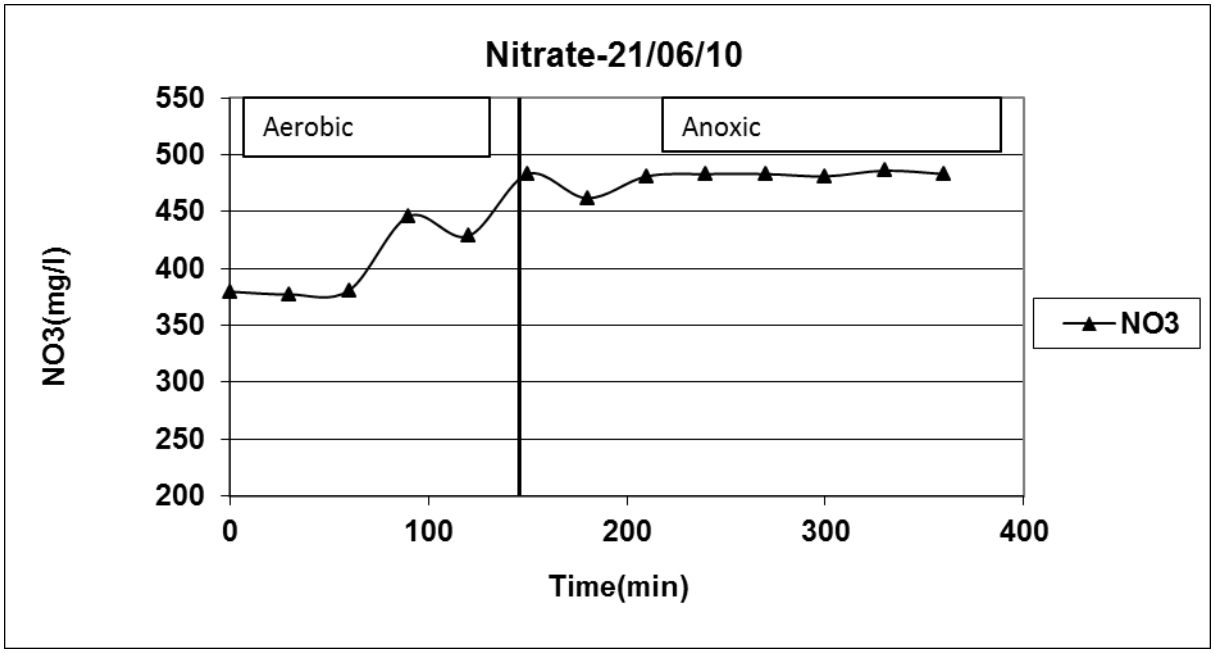


Figure 44 Change in nitrate concentration (N_n) with time during the first aerobic-anoxic phases (0h- 6h) of the 24h period for batch 7.

Figure 44 shows the change in nitrate concentration (N_n) with time during the first aerobic-anoxic phases (0h- 6h) of the 24h period for batch 7. The AAD N_n difference between the start and end of the aerobic phase is 82 mg/l which is an increase in AAD N_n over the aerobic phase as urine N_a nitrifies to

N_n. During the anoxic phase the AAD N_n remains constant till the end of the phase. During the aerobic phase there was a change of 42 mg/l N_n increase in the AAD as the N_a that had been released during endogenous respiration of the previous anoxic phase was nitrified. The N_n remained constant throughout the final anoxic phase of the AAD indicating no de-nitrification.

Feeding of the AAD was switched to the start of the Anoxic phase on the 22-August 2010 to investigate the improvement of de-nitrification due to extra biodegradable urine organics. The switch of the feeding phase was made for urine batch 9 (with N_a ~825 mg/l). As shown in Figure 45 , the AAD N_{ne} decreased by ~15% showing improvement in de-nitrification. However the new batch of urine was added due to depletion of the batch 9 volume. Batch 10 had a relatively higher (1300 mg/l) N_a and this led to the subsequent increase of ~15% of the AAD N_{ne}. However it should be noted that the AAD N_n increased to by 15% equal to the initial decrease despite higher batch N_a being added. This confirms improvement of de-nitrification with extra urine biodegradable organics made available for de-nitrification when the feeding sequence is switched to the start of the anoxic phase.

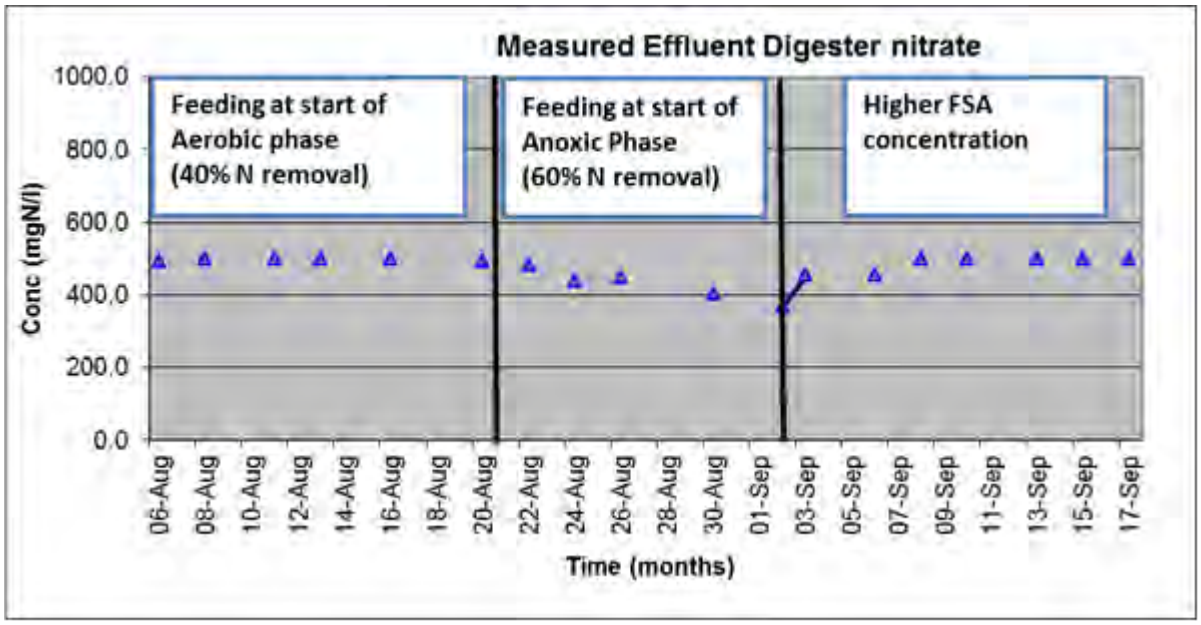


Figure 45 Measured effluent digester nitrate

7.3.4 Precipitation of P, K, Mg and Ca

The WAS from the UCT BEPR system was rich in P (up to 0.2 mg P/mg VSS), which is a characteristic of biological enhanced phosphate removal sludge (Wentzel et al., 1990). Phosphate was released into the bulk liquid with decay of biomass in the ADD, together with potassium and magnesium. This confirms findings about various authors that during biological excess P uptake in the BEPR systems, uptake of K and Mg cat ions also takes place to counter negative charge of phosphate (Wu et al., 2006). K and Mg release investigations have been conducted mostly for anaerobic digestion. It has been found that K and Mg release occur during anaerobic digestion. If the bulk liquid FSA, OP, Mg, K and Ca concentrations create ionic products high enough to exceed the solubility products of the Ca and Mg Phosphate and Carbonate minerals, then mineral precipitation takes place. To investigate the possibility of precipitation, a mass balance of the bulk liquid Mg, K and Ca measured as filtered in the WAS, urine and final effluent was conducted and shown in Table 14 below.

Table 14 Mass balance of the bulk liquid Mg, K and Ca measured as filtered in the WAS, urine and final effluent.

Batch no	$FFCa_i$ mg/d	$FFCa_e$ mg/d	$FFCa_P$ mg/d	$FFMg_i$ mg/d	$FFMg_e$ mg/d	$FFMg_r$ mg/d	FFK_i mg/d	FFK_e mg/d	FFK_r mg/d
1	27	15	13	49	65	16	257	294	37
2	31	14	17	55	68	12	241	311	70
3	25	17	8	43	72	29	267	286	19
4	23	13	10	29	100	71	218	264	46
5	26	10	16	48	86	38	217	306	89
6	12	8	4	29	64	35	222	280	58
7	12	13	0	21	83	62	238	298	60
8	23	14	9	28	106	78	182	392	n.a
9	16	14	1	28	80	52	256	308	52
10	7	13	0	19	96	77	57	360	n.a

From the results shown in Table 14, about 50% of the Ca that was added to the AAD precipitated assuming that Ca release did not occur in the AAD. This was worked through the calculation of the difference between the filtered influent Ca flux ($FFCa_i$) and the filtered effluent Ca flux ($FFCa_e$). Ca normally precipitates as Calcite (CaCO3) and or Hydroxyapatite (Ca5 (PO4)3 (OH). K and Mg release occurred in the AAD. The released K ranged from 37 mg/l to 60 mg/l. Batches 8 and 10 give values out of range. The filtered K samples of this batches spilled during sampling procedures hence results could not be measured.

Table 15 Mass flux of phosphate in the AAD

Batch	FP_{in} mg/d	FP_e mg/d	FP_r mg/d	OP_{VSS} mg/d	P_P mg/d	OP_{End} mg/d
1	85	248	163	173	10	67
2	87	203	116	310	194	65
3	86	232	146	251	105	66
4	75	430	355	363	8	78
5	95	383	288	352	64	78
6	90	346	256	164	-92	81
7	113	345	232	335	103	73
8	79	514	435	504	69	78
9	96	416	320	730	410	90
10	87	386	299	628	329	83

Table 15 shows the Filtered influent P (FP_{in}), Filtered effluent (FP_e), calculated released P (FP_r) which is the difference between FP_{in} and FP_e, OP_{VSS} which is the OP released during biomass degradation, precipitated P (P_p) given by the difference between OP_{VSS} and FP_r and Expected OP release (OP_{End}) at endogenous respiration (0.04/d).

From the average value of VSS broken down per batch, using the measured P/VSS ratio, the average flux of P released per batch as OP_{VSS} was calculated. This was compared with the expected OP release having assumed the P is released at the endogenous respiration rate of the PAOs and assuming PAOs endogenous residue accumulation rate of 0.25/d at 20^0C.The results show that the PAOs release P at a much higher rate than that of their endogenous respiration rate. The difference between OP_{vss} and FP_r was the precipitated P. Precipitation efficiency ranged from 2% to 50% and varied across batches. The overall P precipitation efficiency was ~23%. The purpose of the experiment was not to maximise P precipitation, but simply to observe natural behaviour. Dosing of magnesium oxide, instead of sodium bicarbonate, would restore alkalinity and result in improved phosphate precipitation, as struvite (for instance).

The P release and precipitation trends were further investigated by conducting similar profiles but for the final aerobic-anoxic phases (final 6h of the 24hr period).

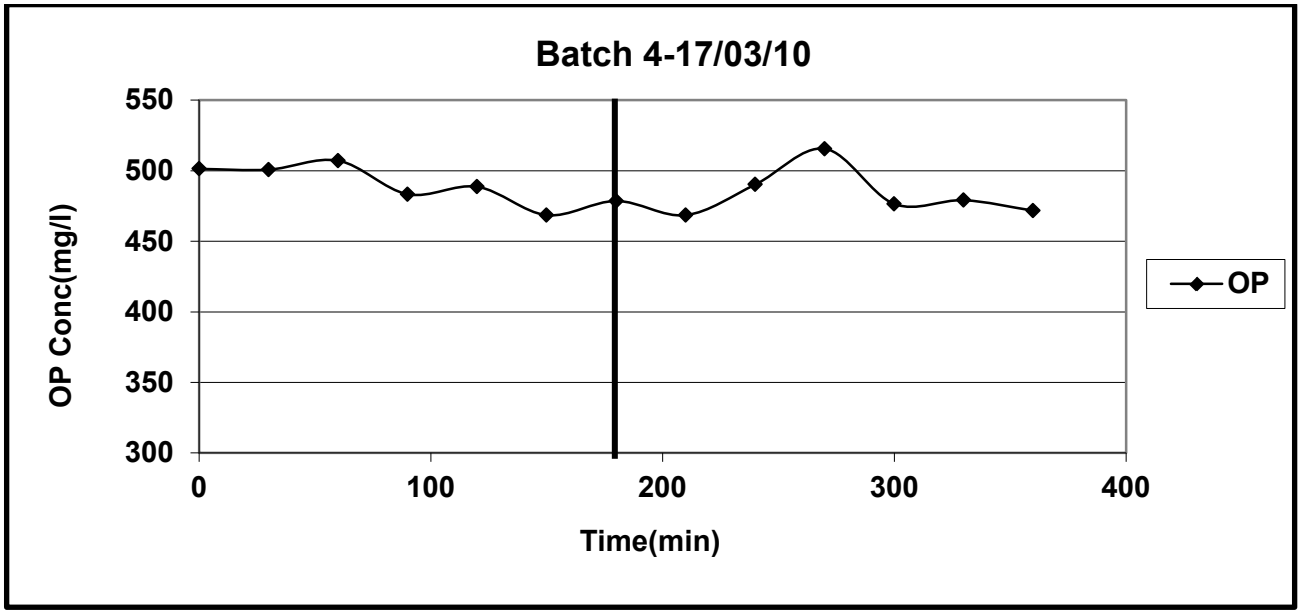


Figure 46: Variation of AAD concentration with time during the final aerobic-anoxic phase for batch 4

Figure 46 shows the variation of AAD concentration with time during the final aerobic-anoxic phase for batch 4. It is important to note that at the start of the final aerobic phase the OP level was 500 mg P-OP/l, which was a net increase of 100 mg P-OP/l (Ju. L.K et al., 2005), from an initial P concentration measured during start of the first aerobic phase of the same bacth Over the 6h period, there was a net AAD P decrease of 30 mg P-OP/l. This indicates P precipitation occurred at a higher rate relative to the P release. P release occurred simultaneously with precipitation as shown by the sinusoidal trend of the profile. The profile suggests lower P release rates during the aerobic phase and increased P release rates during the anoxic phase.

P release in aerobic-anoxic digestion of BEPR sludge can be investigated by categorizing BEPR sludge into three components, namely: Inorganic P precipitates, organically bound P and Polyphosphate accumulated P (PAOs) (Ju L.K et al., 2005). P release of up to 130 mg P/L can occur in the AAD (Ju L.K et al., 2005). Inorganic P precipitation is affected by factors such as solubility products and pH. Organically bound P is depended on the fraction of VSS mass broken down during AAD digestion. It is not conclusive, but is results suggest that PAOs release P at a higher rate than that of their hydrolysis rate. The AAD pH was controlled at 7±0.5, hence mineral precipitation due to redox potential change is ruled out. Precipitation could only have been due to cat ions solubility product exceeding that of inorganic mineral compounds.

Since the P remains in the solid (filterable) phase with the release of OP from the biomass and mineral precipitation, the extent of the mineral precipitation cannot be determined from the measurements alone, because it is not clear how much of their PolyP do the PAOs release under aerobic digestion conditions. Under anaerobic digestion (AD) conditions it has been observed that the PAOs release all their PolyP in a few days of AD at a much faster rate than the release of their OrgP via the hydrolysis/acidogenesis of their biomass organics (Harding et al., 2010), but this may not be the case in AAD with endogenous respiration.

In the case of the AAD, the focus was on whether the PAOs release their PolyP at the endogenous respiration rate (Which is very slow ~0.04/d) or a relatively higher rate. It was observed that the PAOs released their PolyP at a relatively faster rate to that of their endogenous rate. This was further investigated in detail by the modelling of the UCT NDBEPR and AAD systems as a connected unit with the three phase (aqueous-gas-solid) plant wide WWTP model of Ikumi et al. (2001), which include mineral precipitation.

7.4 CONCLUSION

With addition of alkalinity, nitrification was complete and results in low effluent ammonium concentrations from the anoxic-aerobic digester. The effluent quality produced by the AAD with urine treatment and that without urine treatment is almost the same with the exception of the N effluent concentration. However the AAD with urine treatment can be operated to achieve close to zero effluent filtered N ($FSA+NO_3$) by controlling the addition of urine KgN/KgTSS WAS at the same N flux as the WAS. It was observed that the AAD with 50% aeration time (3h air on and 3h air off) removed twice as much N as the N content of the WAS. The N content of main stream wastewater treatment waste activated sludge production is ~ 4 g N/Person.d and the N in the urine is 8 g N/Person.d, for middle and high income people (high meat died). The AAD as operated in this experimental investigation achieved 50% N removal for the received load. Dosing of urine during the anoxic stage was found to improve the denitrification potential of the system.

8 IMPACTS OF URINE SEPARATION ON WASTEWATER TREATMENT IN ACTIVATED SLUDGE UCT AND JOHANNESBURG PROCESSES

8.1 INTRODUCTION AND AIMS

Modern biological nutrient removal activated sludge (BNRAS) wastewater treatment processes remove organics, nitrogen (N) and phosphorus (P) to prevent receiving water de-oxygenation and eutrophication. Under normal dry weather conditions, process plants receive a mixture of brown water (faeces and toilet paper), yellow water (urine) and grey water (kitchen and bathroom). Based on previous work, urine contains about 80% of the N, 65% of endocrine disruptors (Lienert et al., 2007) and 50% of the P in mixed municipal wastewater, but it represents less than 1% of the total wastewater volume (Wilsenach, 2006). Therefore, if urine were collected separately, the nutrient loads on municipal wastewater treatment plants will reduce significantly, but will this automatically relate to a significant difference in effluent quality or process economics? Here the key questions are:

(1) With nitrification being the BNRAS size governing bioprocess via the sludge age, will this still be required with wastewater comprising only brown and grey water?
(2) With a significantly reduced P load, will the effluent P concentration also decrease and to what value?
(3) With eutrophication prevention requiring ever decreasing effluent N and P concentrations, e.g. 0.1 mg P/ℓ, is achieving such limits related to the N and P loads or by other factors that set the limits of the BNRAS technology, e.g. affinity constants in Monod kinetics?
(4) What will be the COD:TKN:TP ratio of urine separated (grey and brown) municipal wastewater?

The minimum N and P requirements for biological (heterotrophic) growth depend on sludge age, the shorter the sludge age the higher the N and P requirements and decrease when settled wastewater is treated. For activated sludge N/VSS (f_n) and P/VSS (f_p) ratios of 0.10 g N/gVSS and 0.025 g P/g VSS respectively, the influent TKN/COD and TP/COD ratios for raw and settled wastewater at 5, 8, 10 and 20d sludge age are given in Table 28.

At these ratios, all of the influent N and P will be used for sludge production so that no nitrification denitrification (ND) and biological excess phosphate removal (BEPR) will be required. From Table 28, for 1000 mg COD/l, and for sludge ages between 5 and 10 days, the influent TKN concentration of mixed grey and brown wastewater would need to be below 31 to 26 mg N/l for raw wastewater and 25 to 20 mg N/l for settled wastewater to eliminate nitrification and denitrification from the BNRAS system. For the same influent COD concentration and sludge ages, the TP concentration should be below 9.3 to 7.9 mg P/l for raw wastewater and 7.6 to 6.1 mg P/l for settled wastewater to eliminate biologically enhanced phosphate removal. To unlock capacity at existing BNRAS WWTPs, it is much more important for nitrification to be eliminated, because sludge ages can then be halved from 15-20d to 8-10d. BEPR can be achieved at short sludge ages (5-10d).

In their model simulation study of the impacts of urine diversion on WWTPs, Wilsenach and van Loosdrecht (2003) utilized a BNR system called the Biologisch/Chemisch Phosphaat and Stikstof (Nitrogen) removal process (BCFS). Their predictions showed a 70% decrease in total effluent nitrogen content with increasing urine separation up to 60% and nearly 100% P removal without urine separation. At greater urine separation, the model showed no improvement in effluent quality due to the magnitude of the Monod affinity constants (do these affinity constants also set the limit of

technology in real WWTPs?). The predicted effluent N and P concentrations only decreased below the affinity constant limit concentrations once nutrient limitation took place.

Table 16 Influent TKN/COD and TP/COD ratios for complete removal of N and P without requiring ND and BEPR.

Sludge age (d)	Raw Wastewater		Settled Wastewater	
	TKN/COD	TP/COD	TKN/COD	TP/COD
5	0.031	0.0093	0.025	0.0076
8	0.028	0.0084	0.022	0.0066
10	0.026	0.0079	0.020	0.0061
20	0.023	0.0068	0.016	0.0049

8.2 MATERIALS AND METHOD

The UCT system was selected because it offers the advantage that the nitrification denitrification (ND) processes can occur independently of the biological excess phosphorus removal (BEPR). This section describes the experimental set-up, operating, monitoring and analytical measurement of the lab-scale UCT type BNRAS at 20 days sludge age and operating temperature of 20˚C.

Experimental Set-up

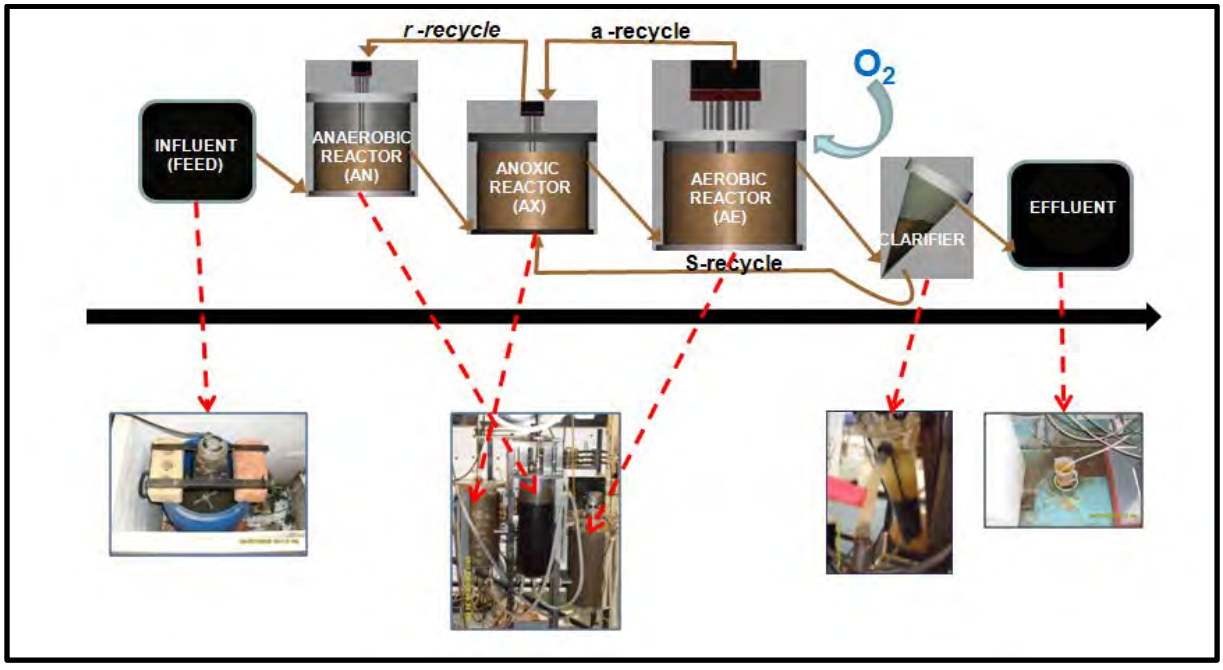


Figure 47 The nitrification/denitrification biological excess phosphorus removal (NDBEPR) system.

The experimental set up consisted of a laboratory scale NDBEPR UCT system with the following components: a 15 ℓ refrigerated (8^{o}C) feed drum, a 7 ℓ anaerobic (AN) reactor, a 10 ℓ Anoxic (AX) reactor; a 15 ℓ aerobic (AE) reactor, a 2 ℓ clarifier and a 20 ℓ bucket for collecting effluent (Figure 52). Three recycle flows, which were pumped by the same influent flow multi-channel peristaltic pump, connected the reactors and clarifier, i.e. *a*-recycle from the aerobic to the anoxic reactor, *r*-recycle

from the anoxic to the anaerobic reactor and the *s*-recycle from the clarifier underflow to anoxic reactor. The system design parameters are listed in Table 29.

Table 17 UCT system's design and operating parameters

System parameters	CAS UCT
Sludge age (day)	20
Anaerobic mass fraction (%), Volume (ℓ)	23.6 [1]/ 7
Anoxic mass fraction (%), Volume (ℓ)	37.6 [1]/ 10
Aerobic mass fraction (%), Volume (ℓ)	38.8 [1]/ 15
a-recycle (AE to AX)	1:4
r-recycle (AX to AN)	1:2
s-sludge Return Recycle (SST to AX)	1:2
Hydraulic retention time (day)	2.13
MLVSS concentration (mg/ℓ)	2500 [2]/3450[3]
MLTSS concentration (mg/ℓ)	3350 [2]/4100[3]
Influent flow (ℓ/d)	15
Feed COD concentration (mg/ℓ)	1000
[1] & [3] for the given a- and r-recycle ratios; [2] predicted using processes and modelling of NDBEPR BNR (Wentzel et *al.*, 1992).	

Wastewater separation, collection, storage and blend

Separately collected and stored grey water and brown water was trucked from CSIR (Stellenbosch) to the Water Research Laboratory at UCT. The grey and brown were separately macerated and stored in 400 ℓ refrigerated stainless steel tanks at 4ºC. Depending on the COD concentration of the grey and brown water, a collected batch lasted up to 3 weeks, after which it was discarded and a new batch collected. Upon delivery to the UCT laboratory, each of grey and brown wastewater was tested separately for Chemical Oxygen Demand (COD), Total Kjeldahl Nitrogen (TKN), Free and Saline Ammonia (FSA) and Total Phosphorus (TP). Out of 13 batches, average brown water concentrations were 2,238 mg COD/l, 166 mg TKN/l and 50 mg P/l. Average grey water concentrations were 1,434 mg COD/l, 65 mg TKN/l and 27 mg P/l.

Feed Preparation

In order to estimate the grey and brown water mix, the changes in TKN/COD and TP/COD ratios were calculated at different mixing proportions. The grey and brown wastewaters had COD, TKN and TP concentrations around 1800, 45 and 5 and 7000; 260 and 90 respectively. From these ratios, the brown water contained about 50% more N and P per mass of COD. Because the per capita generation of grey and brown water varies quite widely, it was decided to mix the grey and brown water with a relatively high proportion of brown water so that the TKN/COD and P/COD ratios of the mixture are relatively high. Grey and brown water were mixed 50/50 and diluted with tap water to the target COD concentration of 1000 mg/ℓ. Daily, after thoroughly mixing the stainless steel tank content, the required volume of grey and brown water was collected through a fine (1 mm) mesh (to prevent

blockages of the UCT system interconnecting tubes), and the tap water added. In this way 15.2 ℓ was prepared, a 200 mℓ sample taken and 15 l transferred to the influent feed drum. The feed drum was gently mixed (1-2 rpm) to minimize settlement of particulates. The 15 ℓ was pumped into the system over 23.5 to 24 hours. At the end of the 24h period, the particulates not pumped into the system were collected in a small volume of effluent (200 mℓ) and added to the anaerobic reactor.

Sampling and Testing

Samples were collected from the influent, anaerobic, anoxic and aerobic reactors and effluent and tested to evaluate the system's performance every second day. The parameters tested are listed in Table 30. The sample volumes collected from the reactors were part of the daily mass of sludge wasted from the system to establish the 20d sludge age (1425 mℓ if no samples were taken). Sampling and sludge wasting were done 2-3 hours after feeding thereby ensuring that the system had adjusted to the new days' feed. All samples taken were immediately filtered through 0.45 µm filter membranes and 2 drops 8.6 g/l HgCl added to prevent further biological activity and stored at 4^{o}C until analysis the next day.

The oxygen utilization rate (OUR) in the aerobic reactor was measured continuously and automatically using a technique detailed by Randall et al. (1991). A Dissolved oxygen (DO) probe – YSL Model 5739 – was suspended in the aerobic reactor and connected to an automated DO meter/OUR data logger (HiTech Microsystems), which controlled the reactor DO between high (5 mg O/l) and low (2 mg O/l) set points. When the DO reached the low set point, a solenoid valve in the air supply line opened and the reactor was aerated until the DO reached the high set point, when the solenoid valve shut off the air supply. During each air-off period, the DO versus time was measured and the OUR calculated by linear regression. The DO time data, OUR, time, correlation coefficient and temperature were stored in the OUR meter. The OUR results for each day's feed were downloaded from the OUR meter to a computer at the end of the feeding period prior to sludge wasting. The data were imported into a spread sheet program where it was plotted and the average OUR calculated for the day. The DO/OUR meter was reset at the start of the next days feed. The DO probe was regularly calibrated in DO saturated tap water (9.1 mg O/l) and sodium thiosulphate solution (0 mg O/l).

Table 18 Sampling position and parameter measurement

	COD [1]	**TKN** [2]	**FSA** [3]	**NO_2** [4]	**NO_3** [5]	**P_{tot}** [6]	**TSS** [7]	**VSS** [8]	**OUR** [9]	**DSVI** [10]	**Ph** [11]
Influent	*U;F*	*U;F*	*F*	*F*	*F*	*U*					*U*
AN[a]				*F*	*F*	*F*	*U*	*U*			
AX[b]				*F*	*F*	*F*	*U*	*U*			
AE[c]	*U*	*U*		*F*	*F*	*U;F*	*U*	*U*	*	*U*	*U*
Effluent	*U;F*	*U;F*	*F*	*F*	*F*	*F*					*U*

[a], [b], [c] Anaerobic, Anoxic & Aerobic reactor respectively; U = *Unfiltered* sample; F = sample *Filtered* through Schleicher & Schull ME 25/21 0.45 µm membrane filter; * = Direct measurement taken; [1] to [11] refer to "*Standard Methods*" for the examination of Water and Wastewater", American Public Health Association, 13th Edition, 1971.

8.3 RESULTS

8.3.1 UCT process configuration

The results presented here are from the last 60 days of UCT system operation, during which time 5 batches of wastewater were fed. For each wastewater batch (which was accepted to represent a steady-state period), the daily results were averaged (after analysis for outliers). These steady-state averages were used to assess the performance of the system and the following process characteristics were calculated: System COD, N and P flux (mass/d) balances; influent unbiodegradable soluble and particulate COD fractions ($f_{S'us}$ and $f_{S'up}$ respectively, Ekama and Wentzel, 1999); influent readily biodegradable RBCOD concentration; mixed liquor VSS/TSS, COD/VSS, TKN/VSS and TP/VSS ratios; nitrate and P flux (mass/d) changes across each reactor.

The N mass balance compares the exiting N via the effluent, waste sludge stream and nitrate denitrified (from a nitrate balance over the anoxic and anaerobic reactors) with the N entering the systems via the influent TKN. Similarly, the COD balance was calculated by comparing the exiting COD via the effluent, waste sludge stream and oxygen utilized in the aerobic reactor (corrected for nitrification) with the COD entering the systems via the influent COD. The mass balances are calculated to determine the accuracy and reliability of the response data. Mass balances in the range 90 to 100% are considered acceptable and indicate reliable data.

UCT system Nitrogen mass balance

The daily flux of nitrogen that enters the system in the form of influent TKN (FNti) exits it as:

i. Flux of N denitrified (FNOxd, mg N/d)
ii. Flux of N in the waste sludge (FNtws)
iii. Flux of N in the effluent, i.e. TKN (FNteffl) plus nitrite and nitrate (FNOxeffl)

The different components (i, ii, iii) of the N balance for the 5 wastewater batches are related in equation 8.1 and shown graphically in Figure 53 and . The unaccounted for N (Uncc) is the N deficit to balance nitrogen at 100%. FNO_{xd} is obtained from a nitrate and nitrite mass balance over the anaerobic, anoxic and clarifier sections of the system (see red dotted line in Figure 54). Because insignificant nitrite was generated (average for all batches was 0.2 mg NO_2-N/ℓ), NO_3 was accepted equal to NO_x (NO_2+NO_3). The aerobic reactor and effluent nitrate concentration was found to be very low (< 1 mg N/l). Because the effluent exits from the aerobic reactor, such low nitrate concentrations are only possible if negligible nitrification takes place in the aerobic reactor. Without nitrification, there cannot be any denitrification in the anoxic (and anaerobic) reactor. The nitrate mass balances over the anoxic (and anaerobic) reactors confirmed this (Figure 53). Less than 7% of N exits the systems via denitrification and this is most like due to low spurious background nitrate concentrations which are probably really zero.

$$\%N\_balance = \frac{100(MNOxd + MNtws + MNteffl + MNOxeffl)}{MNti} \quad (8.1)$$

Because no (negligible) nitrification takes place in the system, the nitrogen balance should close at 100% with nearly all the N exiting the system as N bound in waste activated sludge (WAS). While it is true that nearly all the measured N exiting the system exits as N in WAS (96%), the N balance is not 100%. The N balance over the 5 wastewater batches is only 60 to 75%. Provided it can be confirmed

that nitrification in the system is negligible, this means that the measured N/VSS ratio of the sludge in the aerobic reactor is too low. Reasons for this have not been established, but are being investigated.

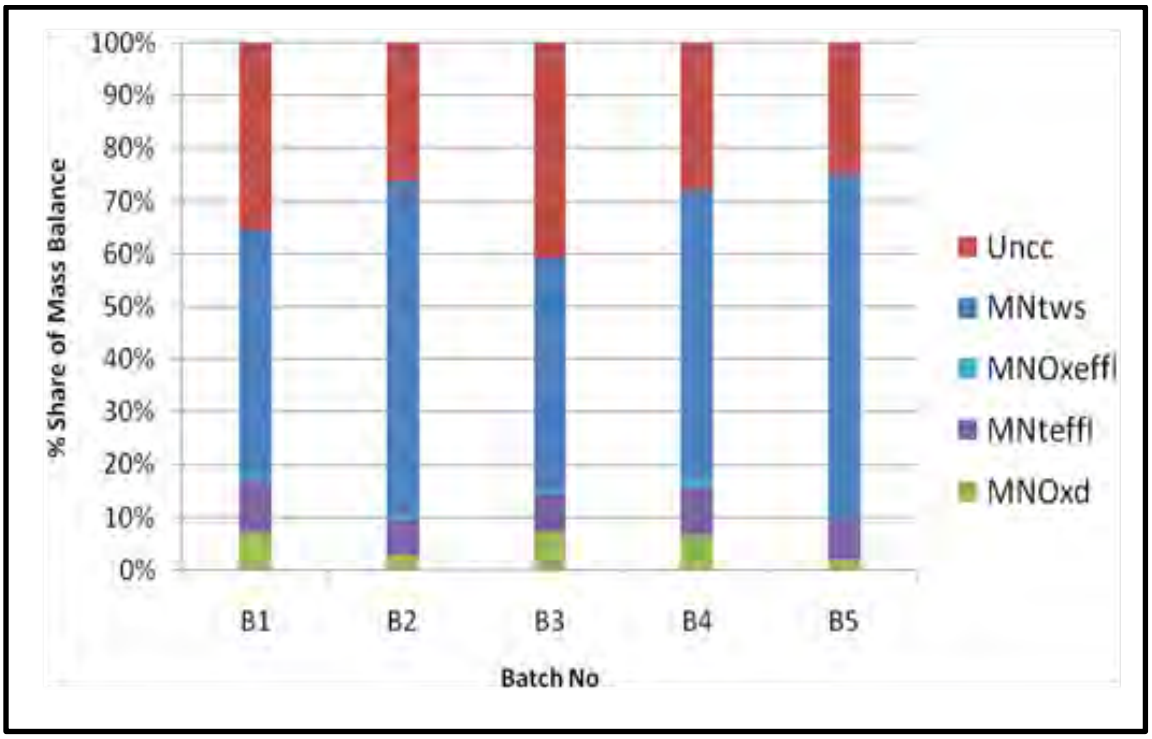


Figure 48 Graphical representation of the percentage Nitrogen mass balance for the sewage wastewater batches – at steady state – for the Conventional UCT system.

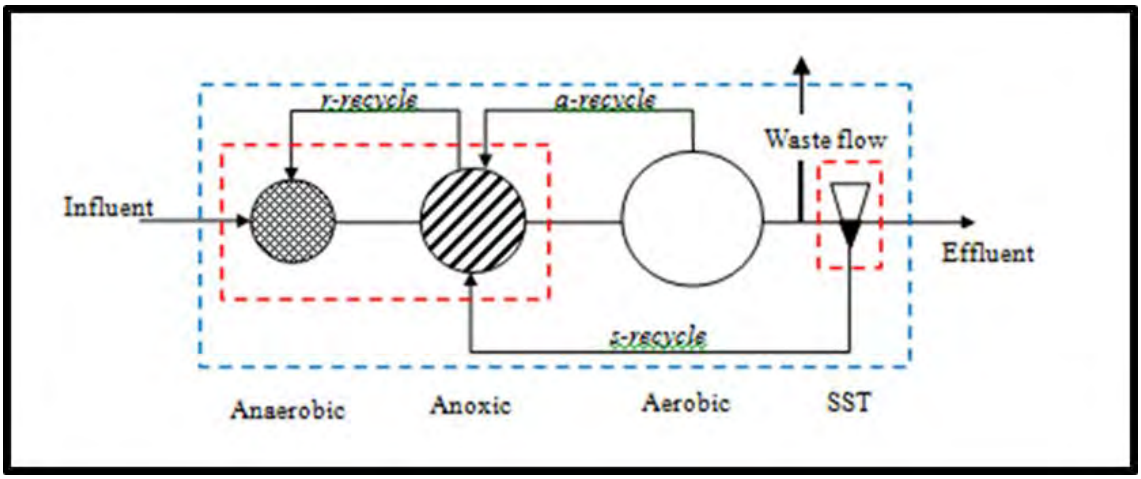


Figure 49 Schematic layout of the UCT system. Red and blue dotted lines indicate the mass balance boundaries around the un-aerated zones and clarifier and overall system respectively.

UCT system COD mass balance

For 100% COD balance, the flux of COD entering the system should be accounted by the:

i. Flux of oxygen demand required per day for degradation of carbonaceous material in the aerobic reactor (FO_c, mg O/d)
ii. Flux of oxygen recovered by denitrification of nitrate and nitrite (FO_d, g O/d)

iii. Flux of COD in the waste sludge (FSt_{ws}, gCOD/d)
iv. Mass of COD in the effluent (MSt_{effl}, gCOD/d)

FO_c is obtained by subtracting the nitrification oxygen demand (FO_n, gO/d) from the measured oxygen demand, i.e. $FO_m = OUR_m * V_{aerobic} * 24$ and adding the oxygen recovered by denitrification (FO_d). FO_n is 4.57 mg O/mg N nitrified times the flux of nitrate generated by nitrification, $FO_n = 4.57*(FNO_{xd} + FNO_{effl})$ where 4.57 represents the oxygen requirement for the nitrification of ammonia to nitrate.

FO_d is 2.86 times the flux of nitrate denitrified, i.e. $FO_d = 2.86*FNO_{xd}$ where 2.86 is oxygen equivalent of nitrate when denitrifying one mg N of nitrate to nitrogen gas.

$$\%COD\_balance = \frac{100(MO_c + MO_d + MSt_{ws} + MSt_{effl})}{MSt_i} \quad (8.2)$$

The measured FO_c is then added to the flux of COD exiting the system via the effluent and waste flows to obtain the total COD exiting the system (Equation 8.2). The COD results are shown in Figure 55.

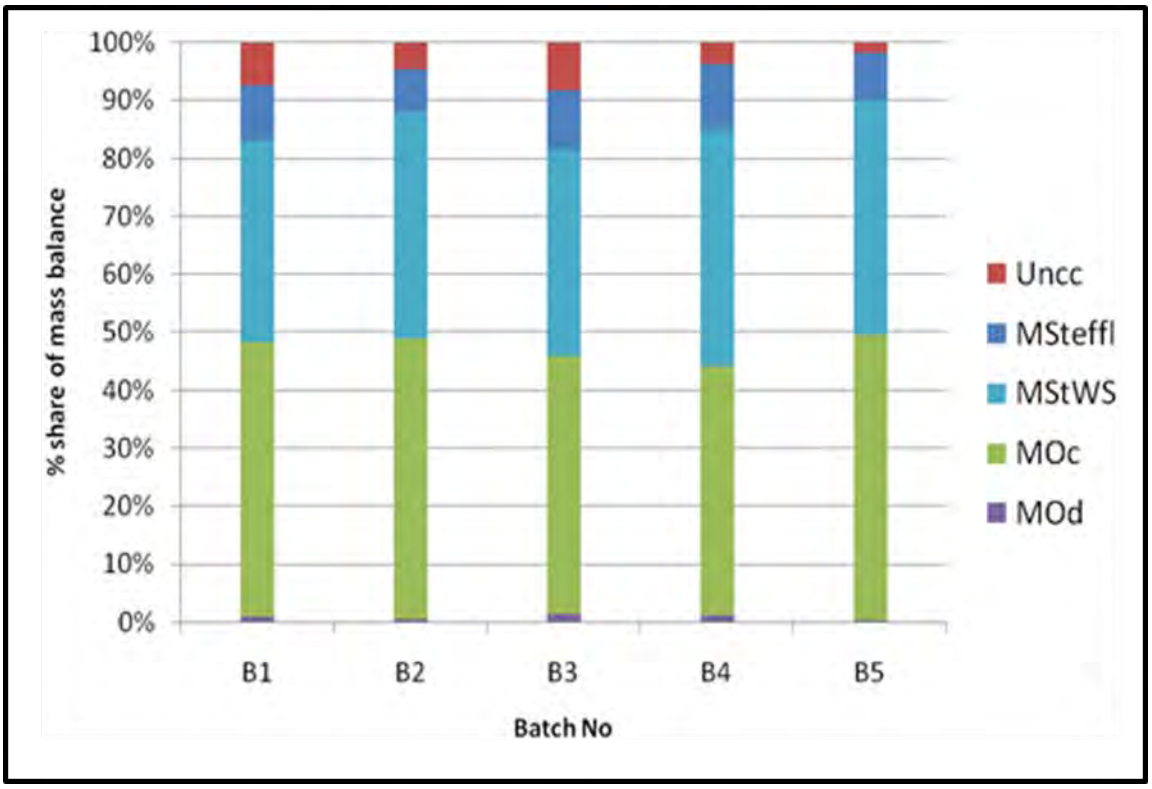


Figure 50 Graphical representation of the percentage COD mass balance for the sewage wastewater batches – at steady state – for the UCT system.

Because no nitrification took place in the system, the oxygen demand of nitrification (FO_n) and oxygen recovered by denitrification (FO_d) are both zero and so the measured oxygen demand (FO_m) could be used directly in the COD balance ($FO_c = FO_m$). Doing this yielded between 92 to 99% COD mass balances over the system. These are among the highest COD balances obtained on a BEPR system – usually COD balances in NDBEPR systems vary between 80 and 95% (Ekama and Wentzel, 1999). The COD mass balance is the first confirmation that no nitrification took place in the system.

UCT system COD, N and P removal performance

The overall average influent and effluent COD, TKN, FSA and TP concentrations and COD, N and P removal efficiency for the UCT system are listed in Table 31. The COD removal was good at 86% and the N and P removals were 90 and 93% respectively. The unbiodegradable soluble COD concentration is higher than the general standard limit of 75 mg COD/ℓ.

Table 19 Overall average influent and effluent COD, TKN, FSA and TP concentrations and COD, N and P removal efficiency for the UCT system at 20d sludge age. ([1]Unfiltered sample; [2] 0.45 filtered sample)

PARAMETER	Influent	Effluent	Removal Efficiency
COD (mg COD/l)	1041.8	108.7[1] (85.3[2])	86%
TKN (mg N/l)	53.2	4.7[1] (2.9[2])	90%
FSA (mg NH_4^+-N /l)	23.6	2.6	96%
TP (mg P/l)	17.3	1.3	93%

Fixing the unbiodegradable soluble COD fraction ($f_{S'us}$) for the UCT system at the measured value (i.e. $f_{S'us}$ = filtered effluent COD/total influent COD), the $f_{S'up}$ fraction was calculated such that the calculated VSS mass in the system matched the measured average mass of VSS in the system (Ekama and Wentzel, 1999). This calculation requires the influent readily biodegradable COD (RBCOD) concentration (S_{bsi}), which was measured as the difference between the membrane filtered influent and effluent COD concentrations, i.e $S_{bsi} = (S_{tsi} - S_{use})$. Because no nitrate was recycled to the anaerobic reactor, it was accepted that all the influent RBCOD was converted to volatile fatty acids (VFA) and taken up by the phosphate accumulating organisms (PAOs) in the anaerobic (AN) reactor. Table 33 lists the influent wastewater and aerobic reactor mixed liquor characteristics. The measured P mass changes in each reactor calculated from a mass P mass balance over each reactor (divided by the influent flow of 15 ℓ/d) are listed in Table 34.

Table 20 Influent and Aerobic Mixed Liquor Characteristics for the 5 batches of sewage over the steady state period for the conventional UCT system at 20d sludge age.

Batch No	Influent Characteristics					Aerobic Mixed Liquor Characteristics			
	TKN/ COD	TP/ COD	$f_{S'us}$	$f_{S'up}$	RB-COD	VSS/ TSS	COD/ VSS	TKN/ VSS (f_N)	TP/ VSS (f_p)
B1	0.047	0.0122	0.07	0.14	0.31	0.82	1.25	0.079	0.029
B2	0.044	0.0172	0.06	0.21	0.27	0.85	1.27	0.089	0.035
B3	0.061	0.0223	0.09	0.21	0.18	0.84	1.24	0.094	0.047
B4	0.058	0.0173	0.11	0.27	0.34	0.84	1.18	0.088	0.030
B5	0.048	0.0142	0.08	0.24	0.32	0.86	1.26	0.093	0.032
AVE	**0.049**	**0.0166**	**0.08**	**0.21**	**0.28**	**0.84**	**1.24**	**0.089**	**0.035**

Table 21 P mass changes – i.e. P_{uptake} & $P_{release}$ – across each reactor for the 5 batches of sewage over the steady state period for the conventional UCT system at 20d sludge age.

Batch No	ANAEROBIC (AN)		ANOXIC (AX)		AEROBIC (AE)	
	P_{uptake} mg P/ℓ influent	$P_{release}$ mg P/ℓ influent	P_{uptake} mg P/ℓ influent	$P_{release}$ mg P/ℓ influent	P_{uptake} mg P/ℓ influent	$P_{release}$ mg P/ℓ influent
B1	0	16.7	0	165.9	194.5	0
B2	0	26.9	0.4	0	41.3	0
B3	0	14.2	0	4.1	35.1	0
B4	0	20.0	0	0.6	33.6	0
B5	0	59.0	0	24.1	95.3	0
AV=	**0**	**27.36**	**0.08**	**38.94**	**79.96**	**0**

8.3.2 Johannesburg process configuration

The brief results presented here are from the 154-day operation of the Johannesburg (JHB) system, operated at 5 day sludge retention time (SRT). The system was operated for 15 days (3 x SRT) before reaching steady state. During the operating period, 8 batches were fed, which were the same influent as the final 8 batches fed to UCT system (batched 8-15). The rationale for operating the JHB system was to achieve (1) better phosphate removal at the short SRT, without interference of nitrification, which was confirmed to not be taking place in the long SRT UCT system, and (2) smaller system volume due to the low SRT.

The nitrogen mass balances were calculated in the same way as for the UCT system. N balances in the range 100-127% were obtained giving an average mass balance of 115% for the eight wastewater batches. This average is 33% higher than that obtained for the UCT system over the same wastewater batches. The N balance over 100% implies that more N was exiting the system than entering it, with the largest share (79%) of the N balance in waste activated sludge. The amount of denitrified nitrogen was insignificant.

Unexpectedly, all effluent concentrations i.e. unbiodegradable soluble COD, TKN, FSA, TP, orthophosphate and total solids averaged at values much higher than obtained for the UCT system The overall total removal efficiencies were low at 80% for COD, 58% for TKN and 53% for P, compared with 88% COD, 85% TKN and 82% P for the UCT system over the same wastewater batches. The average effluent orthophosphate concentration was 8.5 mg P/ℓ. At the shorter sludge age, the JHB system was expected to have higher concentrations of N and P required for sludge production and consequently lower effluent concentrations than the UCT system. Furthermore, aerobic batch tests done on the effluent from the JHB system with waste activated sludge harvested from another BNR reactor system fed normal wastewater from Mitchell's Plain wastewater treatment works, revealed that the 0.45 µm membrane filtered effluent COD of 125.1 contained 70% biodegradable COD (Sbse). The same effluent also contained 95% organic N biodegradable soluble (Nobse). The far greater diversity of ordinary heterotrophic organisms from the normal wastewater

system, were able to utilize all the biodegradable soluble organics exiting the JHB system, which the low diversity and constantly adapting biomass in the JHB system were not able to utilize. The shorter sludge age of the JHB system, with the higher biomass turnover rate than the UCT system, exacerbated this phenomenon, resulting also in reduced utilization of slowly biodegradable particulate organics, and hence lower N and P concentrations for sludge production. Most of the JHB effluent concentrations were high and did not represent what had been expected from a scenario of 100% urine diversion.

8.3.3 Nitrification batch tests

Figure 51 Batch reactors set up for aerobic batch tests.

To confirm whether or not nitrification was taking place in the UCT system, batch tests were conducted on sludge harvested from the aerobic reactor. The aerobic batch test reactors are shown in Figure 56. To start the batch test, the reactor was filled with 3 ℓ mixed liquor. Since endogenous conditions needed to be established, the mixed liquor was aerated for a long period (at least 12 hours), during which the OUR was measured, before a small volume concentrated ammonium chloride solution (5 gNH_4Cl/ℓ) was added. The OUR continued to be measured until endogenous conditions were again re-established. The difference between the total OUR and the OUR related to endogenous respiration was considered the oxygen utilized (OU) due to NH_4-N oxidation, which is related to the ammonia nitrified via OU/4.57. During the test, grab samples were taken, immediately filtered (0.45 µm) and stabilized with 2 drops of 8.6 g/ℓ HgCl. The samples were stored overnight at 4°C and analysed for NO_3^--N, NO_2^--N, and NH_4^+-N.

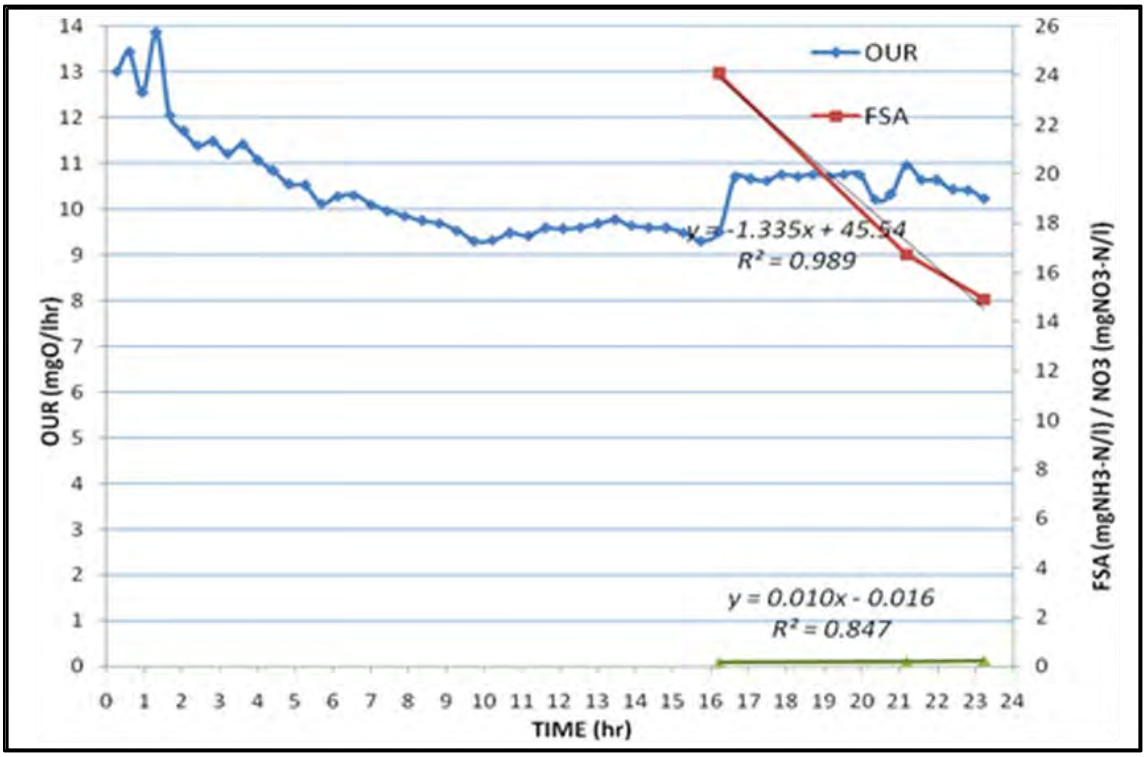


Figure 52 OUR, FSA and NO_3^- results from the first batch test (October 4, 2009)

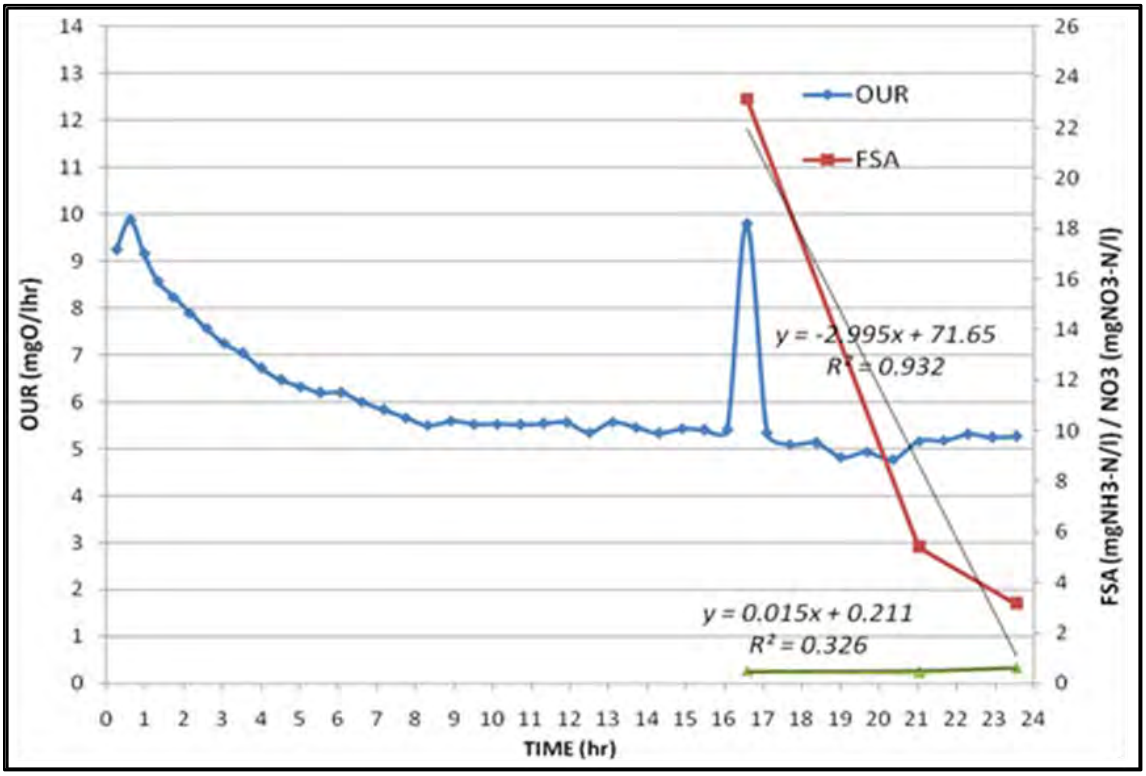


Figure 53 OUR, FSA and NO_3^- results from the second batch test (November 2, 2009)

The OUR, FSA and NO_3^- results of two batch tests, conducted a month apart, are shown in Figure 57 and Figure 58. Although the concentration of ammonia (NH_3) decreases from 24 to 15 mg N/l over 7h,

the nitrate production is negligible, i.e. only 1/150 of the ammonia decrease. The OU was only 15% of that expected from the ammonia decrease. Nitrification is therefore not taking place. The decrease in ammonia and small increase on OUR is probably a response of the ordinary heterotrophic organisms (OHOs), accustomed to ammonia deficient growth in the UCT system, to a sudden ammonia surplus.

8.4 DISCUSSION

8.4.1 Low aerobic reactor and effluent nitrite (NO_2) and nitrate (NO_3)

Provided the clarifier contains little accumulated sludge to minimize denitrification, the concentrations of NO_2 and NO_3 in the aerobic reactor and effluent will be the same. This was the case in this investigation. The recorded NO_2 & NO_3 in the aerobic reactor were *95%* on average of the concentrations in the effluent for all 5 batches. The effluent nitrite and nitrate were on average 0.2 mg NO_2-N/ℓ and 0.4 mg NO_3-N/ℓ respectively. These low NO_2 and NO_3 indicate the absence of nitrification in the activated sludge.

8.4.2 Uncorrected measured OUR yields good COD balance

For BNR systems with nitrification and denitrification, the OUR measured in the aerobic reactor (OUR_m) is the sum of the nitrification OUR (OUR_n) and carbonaceous OUR (OUR_c) minus the equivalent OUR recovered by denitrification (OUR_d). If there is no nitrification in the BNR system, then OUR_n and OUR_d are zero and the measured OUR_m = OUR_c. For each wastewater batch, the measured OUR, without accounting for any nitrification, gives good COD mass balances (>93%, see Figure 55).

8.4.3 Low nitrogen fraction of the sludge f_N (mg N/mg VSS)

From many years experimental work, the N content of activated sludge including OHOs and PAOs (f_n) has been accepted at 0.10 mg N/mg VSS in steady state models (Marais and Ekama, 1976; WRC 1984, Wentzel et al., 1990; Henze et al., 2008) and kinetic simulation models (ASM1, Henze et al., 1987; UCTPHO, Wentzel et al., 1992; ASM2, Henze et al., 1995). In this investigation a somewhat lower value was measured (0.089, Table 31), which may indicate a nitrogen deficiency due to a very low influent TKN concentration.

8.4.4 No decrease in alkalinity

No decrease in average alkalinity was measured between the influent and the aerobic mixed liquor sludge for all the batches. Average influent and effluent alkalinity concentrations remained unchanged at 81 mg $CaCO_3$/ℓ. Given that 7.14 mg/ℓ as $CaCO_3$ is consumed per mg FSA-N nitrified (Henze et al., 2008), the unchanged alkalinity provides further evidence that there was no nitrification in the system.

8.4.5 P release in the anoxic reactor

Both the JHB and the UCT systems indicated consistent phosphate release in the anoxic reactor (between 16-50% of the sum of P-release over the anaerobic and anoxic reactors). This could only happen if no external electron acceptor – nitrate in this case – was present. What has happened in fact was that the anoxic reactor became a second anaerobic reactor, or merely an extension of the first anaerobic reactor.

8.4.6 No nitrate generation in aerobic batch tests

The results of two batch tests on sludge harvested from the aerobic reactor conducted a month apart show that although the concentration of ammonia decreases over time at 1.2 to 2.8 mg FSA-N/(ℓ.h), the nitrate generation is negligible, <0.03 mg NO_3-N/(ℓ.h), only 1/150 of the ammonia decrease, and the increase in OUR is less than 10% of that expected from nitrification of ammonium. This finaly confirmed the absence of nitrifiers in the activated sludge. The observed decrease in FSA and the increase in OUR, are probably responses of the ordinary heterotrophic organisms and phosphate accumulating organisms, from the sudden availability of excess ammonia for growth.

8.5 CONCLUSIONS

Even though persuasive evidence was found to confirm the absence of nitrification within the UCT system, this raises new questions. Measurements throughout the experiments have shown sufficient ammonium in the effluent at between 3 to 5 mg NH_4^+-N/ℓ.

- Why was the nitrogen content of all bacterial biomass (f_n = 0.89) less than the historically measured average of f_n = 0.1 mg N/mg VSS?
- Why did the mass balance calculations point to excess nitrogen content of bacterial biomass (f_n = 0.135) to achieve a 100% biomass.
- Why wasn't it possible for nitrifiers to accumulate in the system with ammonium always present at higher concentrations than the affinity constants in the aerobic reactor?

What we can conclude is that with high efficiency urine separation, nitrification is no longer required in biological nutrient removal activated sludge processes. Six factors pointed to the absence of nitrification in the system,

1. virtually zero nitrate in the aerobic reactor and effluent (<1 mg NO3-N/ℓ);
2. no alkalinity consumption in the aerobic reactor, which would indicate that ammonia was nitrified;
3. the measured OUR, uncorrected for nitrification, yielded a good (> 95%) COD mass balance;
4. low N content of the activated sludge (fn = 0.09 mg N/mg VSS);
5. phosphate release in the anoxic reactor, which indicated that it was a continuation of the anaerobic reactor; and
6. no nitrate generation and associated nitrification OUR could be stimulated in aerobic nitrification batch tests on sludge harvested from the UCT system.

However, the absence of nitrifiers may not be due to ammonia limitation because the effluent FSA concentrations were high (3-5 mg/ℓ). Activated sludge systems with nitrification achieve much lower effluent FSA concentrations than this.

The UCT system's average filtered effluent concentrations were 85 mg COD/ℓ, 32 mg SS/ℓ, 4.4 mg TKN-N/ℓ, 3.5 mg NH3-N/ℓ,and 2.1 mg PO4-P/ℓ and the removal efficiencies (based on unfiltered effluent concentrations) good at 92%, 92%, 86% and 89% for COD, TKN, FSA and TP respectively. However, these effluent concentrations are not yet good enough to limit receiving water de-oxygenation and eutrophication. However, investigation revealed that an average of 60% organic biodegradable soluble COD (Sbse) and 25% organic biodegradable soluble N (Nobse) were contained in the effluent. This cannot be attributed to the process set-up, because similar units were satisfactorily operated in the past. Rather, this points to something strange in the influent wastewater. The inconsistency of the experiment's wastewater (from a small office block) caused acclimatization

problems for microorganisms (OHOs and possibly ANOs), which was shown by further degrading the effluent COD with activated sludge from a conventional system. For the shorter sludge age of the JHB system, the problem of bacterial acclimatization and diversity was worse. Had the collected wastewater been from a large population group, with a more consistent wastewater composition, biomasss would have better adapted and complete utilization of biodegradable organics would have been possible. In that case, it is likely that low effluent FSA concentrations (less than 1 mg N/ℓ) would have been obtained in the lab systems in the absence of nitrification. . It can be reasonably stated P concentrations, e.g. 0.1 mg P/ℓ, can be achieved with the lower N and P loads from urine separation.

The suppression of nitrification and denitrification will unlock great potential at BNRAS processes, because the SRT can be reduced. This implies that the size of the plant can be reduced significantly (by nearly 50% as shown in this experimental investigation) or alternatively more wastewater can be treated for the same overall reactor volume.

An advantage of nitrification is that all nitrogen not consumed during growth of ordinary heterotrophic or phosphate accumulating organisms, would be mopped up by the nitrification process. An important question, in systems designed for urine separation, is the control and management of a more exact waste stream that would always allow stoichiometric removal of nitrogen; to prevent excess effluent ammonium with too much TKN in the influent, or excess effluent COD with too little.

9 SCENARIOS FOR IMPLEMENTATION AND DEVELOPMENT OF NO-MIX TECHNOLOGY

9.1 RADICAL THINKING – GRADUAL CHANGE

In many ways, separate collection and treatment of urine is radically different from the existing wastewater infrastructure system. Radical thinking does not, however, require an all-or-nothing one stop change. After all, it used to be said that it's impossible to teach women how to read, a statement that's proven wrong by many readers of this report today … Based on a future vision, incremental and modest step function changes could all lead in the same direction. This approach is sensitive to the fact that much of existing infrastructure has not reached its maximum service life, and at the same time that people require time to change.

We therefore need to imagine existing infrastructure in the context of future no-mix streams, and how different units could be reconfigured. This chapter explores a few options, and considered the obvious consequences of changes towards a no-mix system.

9.2 COMPOSITION OF URINE WASTEWATER AND RE-USE

9.2.1 Salinity and drinking water recovery

De facto indirect water recycle has been in practice for as long as cities have been sewered. In future, direct water recycle will become the norm rather than the exception. One serious problem, introduced with wastewater treatment, reclamation and water recycle, is the ever increasing salinity. This is already a problem in Windhoek (Namibia) where wastewater effluent is directly recovered and treated for drinking water.

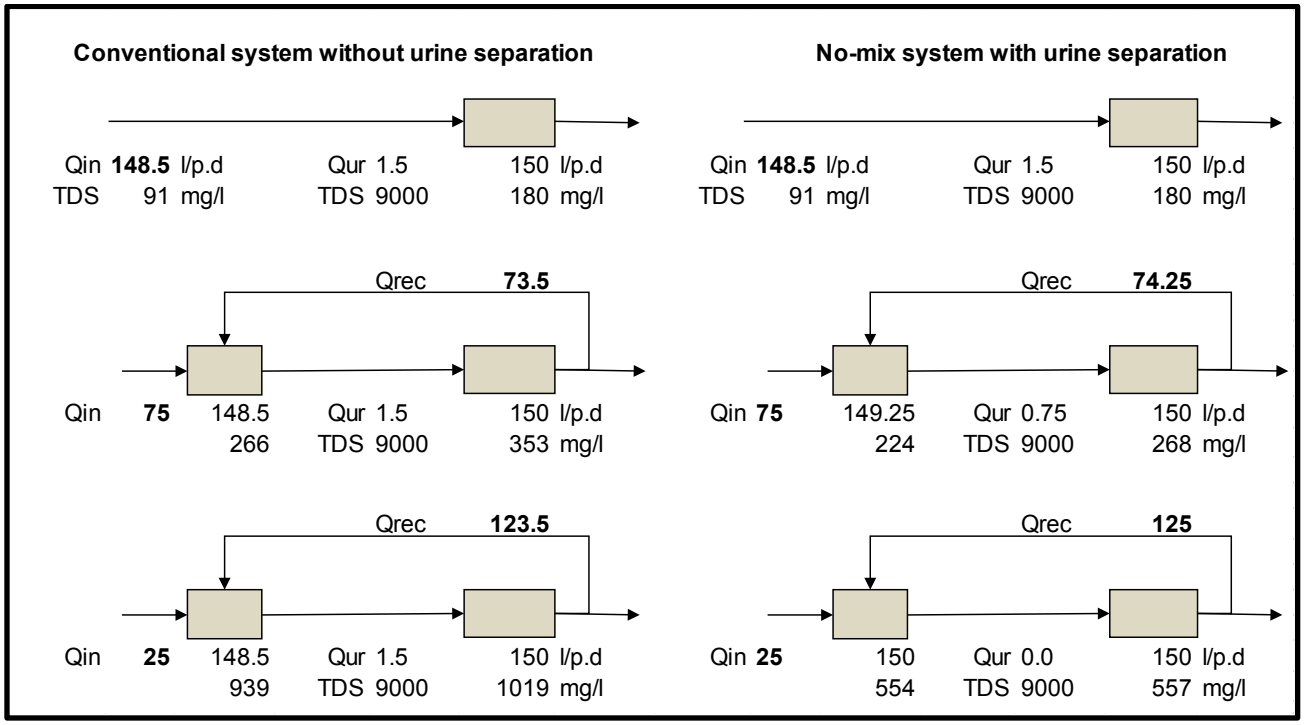


Figure 54 Effect of urine and urine separation on the salinity of recycled water

From the CSIR analysis of the composition of urine (Chapter 4) and based on concentrations of Na^+, K^+, Cl^- and SO_4^{2-}, the total dissolved solids concentration of urine is 9,000 mg TDS/l, or 13.5

gTDS/p.d., for an average urine contribution of 1.5 l/p.d. The average daily salts contribution of urine is roughly half of the daily wastewater load. Figure 60 shows an example of water recycle, and the effect of increased salinity with and without urine separation. If 1 litre of water is recycled for every litre of fresh water, the salinity is expected to double. In severe cases, where five litres is recycled for every one litre of fresh water, the salinity exceeds 1,000 mg/l, which is undesirable. Moreover, this effluent of the same salinity is then discharged into the environment. On the other hand, if urine separation is gradually introduced, then with 1:1 water recycle, the salinity is only around 270 mg TDS/l. With complete urine separation, and with 5:1 water recycle, the salinity would increase to only half (i.e. 560 mg TDS/l) of what would have been the case without urine separation. The latter concentration may be thought of as a more sustainable concentration of salts in drinking water and to discharge into the environment.

Source separated urine is already a concentrated stream, in which natural precipitation of salts was seen to occur. After biological treatment and phosphate removal, the urine stream could be desalinised at much lower cost (due to the much smaller volume and already high concentration) than desalination of the entire wastewater stream.

9.2.2 Micro-pollutants

The occurrence and fate of micro-pollutants (especially endocrine disrupting chemicals) were not studied as part of this project. It is however widely thought that much of the micro-pollutants originate from urine. One pollutant of special interest is the residual compounds or daughter products of synthetic oestrogen, found in oral contraceptives, which is excreted only with urine. In much the same that salts are concentrated in drinking water, with increasing water reclamation and recycle percentages, one can expect resilient micro-pollutants to build up in a water cycle.

Separate collection and treatment of urine allows for targeted removal of micro-pollutants through all the mechanisms that occur within conventional activated sludge systems, as well as with additional mechanisms which become feasible due to the relative small and highly concentrated stream.

9.2.3 Phosphate and potassium recovery

The phosphate concentration in urine is high enough for direct precipitation with additional cations. In biological excess phosphate removal processes like the JHB process, phosphate is concentrated in sludge and can be released at high concentration in an anaerobic environment. In no-mix systems, a concentrated phosphate stream, resulting from biological concentration of brown and grey water must be mixed again with the treated urine stream to recover phosphate. Based on the values in Chapter 4, and comparing the first magnesium concentrations with latter magnesium concentrations, around 60 mg of magnesium is removed through natural precipitation in urine. This is most likely as struvite ($MgNH_4PO_4.6H_2O$) and we can conclude that 75 mg P/l was precipitated at the same time. Calcium was also removed through natural precipitation, but possibly as calcite. The CSIR average measured concentration was 250 mg P/l, which may underestimate the “true phosphate concentration” in urine with 23%. Since prevention of urea hydrolysis in urine collection systems is not feasible, the amount of phosphate precipitated naturally should be accepted as a given. However, if urine contributes 43% of the total phosphate load, then potentially 90% of phosphate may be precipitated as potassium struvite (57% plus 77% of 43%). In stored urine, double the amount of Potassium is available for the maximum potential requirement of potassium struvite precipitation from urine and black and grey water combined.

Ammonium struvite precipitates preferentially over potassium struvite (Wilsenach et al., 2007), so in order to recover potassium as a valuable nutrient too, precipitation of potassium magnesium phosphate must be configured to take place after ammonium removal.

9.3 SALINE WATER TOILET FLUSHING AND URINE SEPARATION

Hong Kong has a dual water distribution system, and toilet flushing is done by filtered and chlorinated seawater. This approach has led to a saving of one third of conventional fresh water consumption, which is imported from main land China. This however also leads to high concentrations of sulphate in sewers (200 mg SO_4^{2-}/l). The sulphate reduces biologically in sewers and forms hydrogen sulphide (H_2S), which escapes into the atmosphere above the wastewater, where it then re-dissolves into the moisture on the sewer walls. Here, the following reaction takes place: $H_2S + 2O_2 \rightarrow H_2SO_4$. Due to the formation of sulphuric acid, sewer crowns corrode and collapse.

In Hong Kong it is believed to be cheaper to repair sewers than to buy extra fresh water. Furthermore, the sulphate can be used as an electron carrier in the SANI process, which is shown in Figure 61. Organics are removed in a upflow anaerobic sludge blanker reactor with biological sulphate reduction. The anaerobic effluent is then contacted with nitrate in an anoxic zone, which is reduced, while the hydrogen sulphide is re-oxidised anoxically to sulphate. In the following aerobic zone, ammonium is nitrified to nitrate, and any remaining sulphide is oxidised aerobically to sulphate. A great advantage of this process is the low sludge production (mostly autotrophic processes) and lower than normal aeration requirement since, COD is oxidised with sulphate as electron acceptor and oxygen is only required for nitrification. Phosphate removal is not accounted for in this scheme, but the search is on for a biological sulphate reducing excess phosphate accumulating organism!

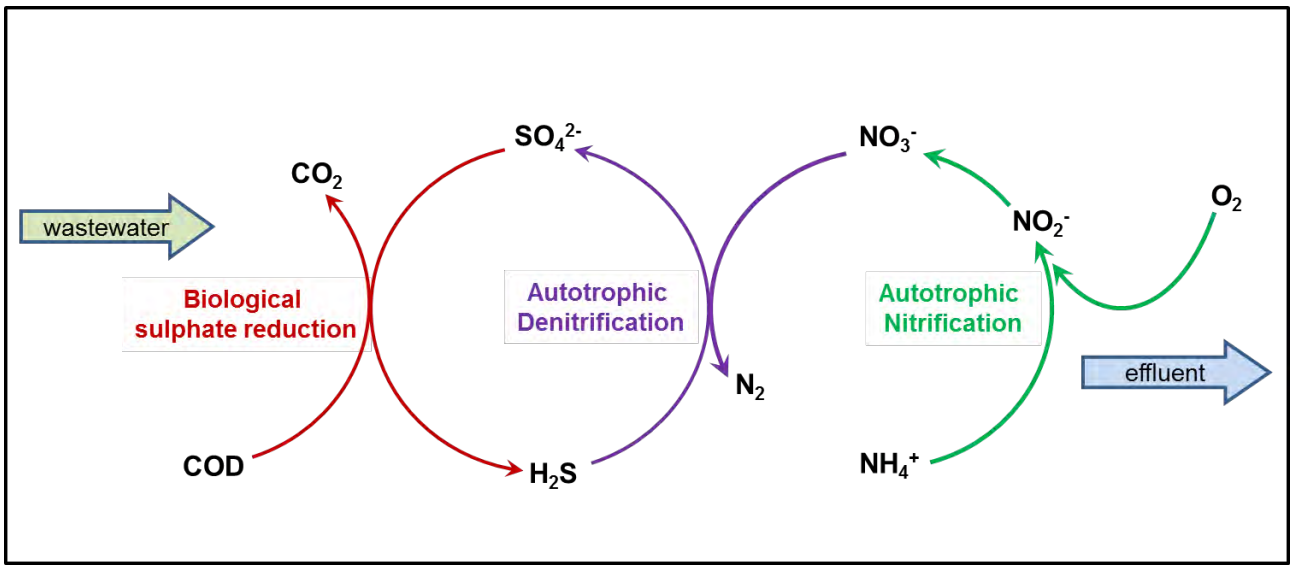


Figure 55 The SANI wastewater treatment process, with the sulphur cycle interposed between the carbon and nitrogen cycle

In mining towns, not situated at the coast, the SANI process could be used in the treatment of Acid Mined Drainage (AMD). Partially treated AMD, where acid has been neutralised with limestone, and gypsum removed, still has a high sulphate concentration of 1,000-1,500 mg SO_4^{2-}/l, similar to seawater. Instead of seawater (like Hong Kong), Johannesburg could use pre-treated AMD for toilet

flushing. Separate collected and treatment of urine offers a mechanism of protection against in-sewer sulphate reduction and crown collapse, which is described in the next section.

9.4 DECENTRAL NITRIFICATION OF URINE AND IN-SEWER DENITRIFICATION

Chapter 5 of this report described the performance of a sequencing batch reactor for treatment of undiluted urine. The natural composition of urine (i.e. COD:TKN ratio) dictates that around one third of the nitrogen can be removed to nitrogen gas when denitrification is over nitrite. Furthermore, the TKN:alkalinity ratio of urine dictates that the production of acid during nitrification can be neutralised only for half the ammonium. The natural effluent therefore consists roughly of a 1:1 ammonium nitrite or ammonium nitrate mixture. If additional alkalinity were added, then all ammonium would be nitrified. The inhibition of nitrite oxidisers due to high free ammonia and nitrous acid concentrations would no longer apply, and all the nitrogen could be converted to nitrate. The use of milk of magnesia (MgO), instead of lime as alkalinity, would at the same time provide the magnesium required for struvite precipitation. All phosphate in urine could be recovered in this way. A sequencing batch reactor would still be desired, in which a high pH can be achieved, required for struvite precipitation. Once precipitated, struvite will not readily dissolve at neutral pH.

Consider installation sequencing batch reactors, at strategic decentral locations (in the basement of large buildings, in industrial areas, etc.) and that the concentrated nitrate could be discharge from these reactors into normal water borne sewers where toilets are flushed with saline water (sea water or pre-treated and neutralised acid mine drainage). The saline water introduces sulphate to the sewer system, but now, ordinary heterotrophic organisms will outperform the biological sulphate reducing organisms, and the former will form biofilm on the sewer walls. Now, instead of corrosive H_2S, the denitrification of nitrate in sewers will only produce nitrogen gas. Furthermore, the sewer becomes part of the wastewater treatment process so that nitrogen and much of the COD would have been removed by the time it reaches the treatment works. The sewer would most likely, due to increased biomass concentration also improve the breakdown/fermentation of slowly degradable and perhaps the more resilient organic compounds. Decentral removal of phosphate (say up to 50%) makes the removal of phosphate in the SANI process also less of a problem. The SANI process is then employed mostly for the treatment of brown and grey water.

9.5 RECONFIGURED WASTEWATER TREATMENT PLANTS

The basic configuration of a wastewater treatment works that treats domestic wastewater and no-mix urine separately, but in an integrated way, is shown in Figure 62. The three important premises of this configuration are highlighted:

1. The emphasis of treatment shifts from main stream treatment to sludge and supernatant treatment, i.e. the highest organic and nutrient load is dealt with outside of the high volumetric flow. This approach allows not only for better natural effluent quality, but also for optimal consumption and recovery of natural resources: a maximum of COD can be converted to biogas, nitrogen can be removed in complete autotrophic processes, phosphate can be recovered for re-use and energy (aeration) is minimised. “Wastewater treatment plants” then become “resource recovery ”
2. Nitrogen removal in the main stream is less important in terms of overall load, and nitrogen removal takes place predominantly in side stream processes that targets the combination of sludge supernatant and source separated urine. This means that eventually much less nitrifiers grow in the main stream process and that very low effluent concentrations of

ammonium may not be achieved. Therefore, a maturation and polishing system is required, which could be a series of ponds and/or constructed wetlands. At the same time, natural disinfection can take place, as well as re-oxygenation before final effluent discharge.

3. Phosphate has to be concentrated in the main stream, removed with waste sludge and then precipitated as a recoverable product as part of the urine/supernatant treatment. Phosphate can be concentrated simply through primary settling, or even enhanced primary settling with poly-electrolyte dosing. In activated sludge processes, such as the Johannesburg process, phosphate accumulating organisms store poly-phosphate, which can be released again, at higher liquid concentration, in an anaerobic environment.

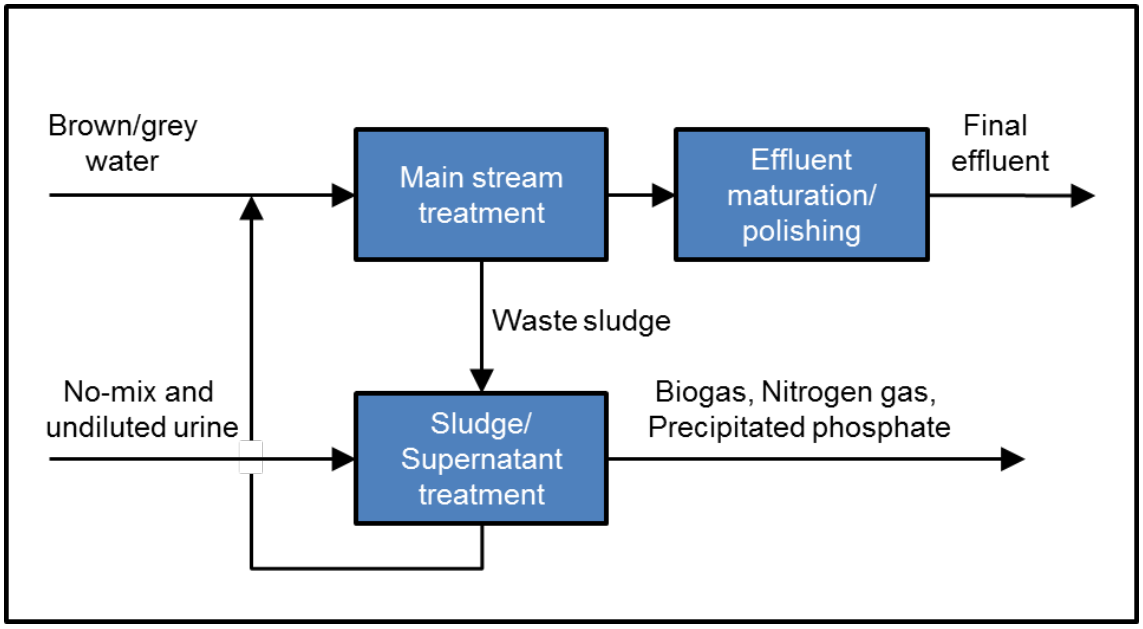


Figure 56 Process configuration for treatment of domestic wastewater and source separated urine

The sections below deal with various combinations of main stream processes and sludge/supernatant handling. Main stream processes could be as simple as trickling filters, extended aeration, or ideally a two stage Phoredox (anaerobic/aerobic) or JHB processes (anaerobic/aerobic/anoxic RAS) with short sludge age of 5 days that maximise sludge production and nutrient uptake from the main stream. Depending on the type of waste sludge produced, sludge handling could be anaerobic digestion, anoxi/aerobic digestion or even direct dewatering. However, from all these processes a supernatant of filtrate stream is produced with high nutrient concentrations that should be mixed with source separated urine.

9.5.1 Trickling filter performance improvement (main stream treatment)

South Africa has over 150 wastewater treatment plants that employ filters in one way or another. Low loaded trickling filters are good at nitrification, and achieve some level of denitrification. Wilsenach et al. (2013) found that anammox bacteria play an important role in the overall removal of nitrogen from the low loaded trickling filters at Daspoort's Eastern Works. These trickling filters are downstream of primary settling tanks and treatment the main wastewater stream very well, to nitrogen effluent concentrations as low as 10 mg N/l. Not nearly all trickling filters perform this well, and some form of augmentation would be welcome at many plants.

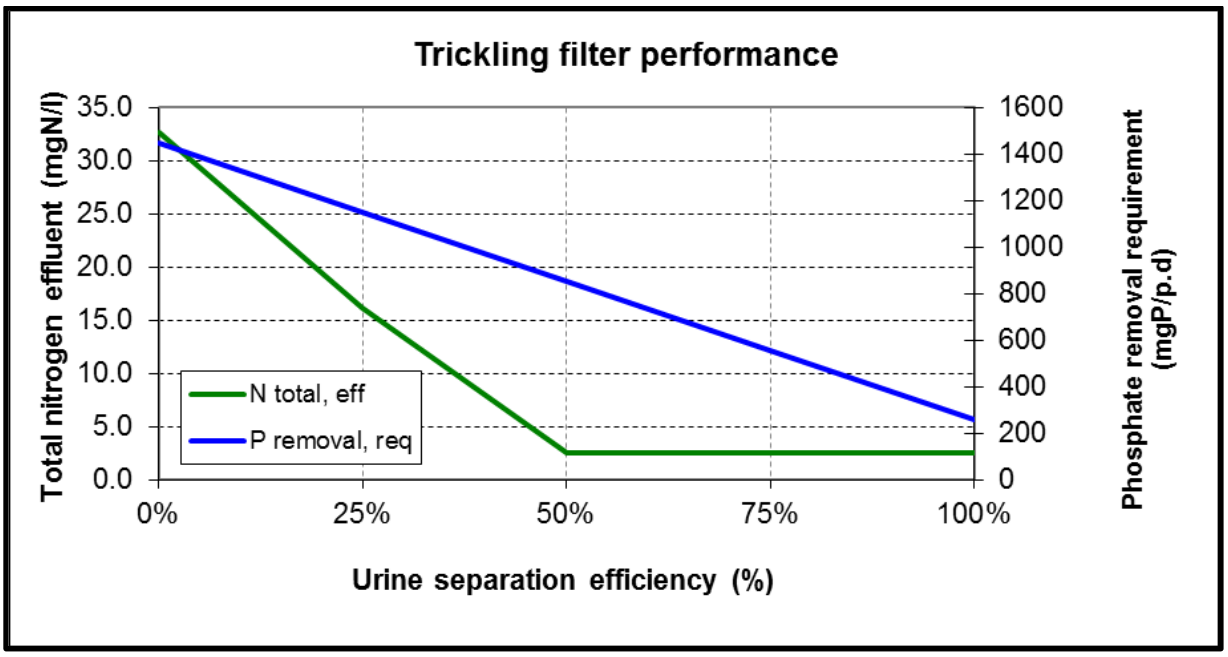


Figure 57 Improved effluent quality and reduced chemical precipitation with urine separation

The Nitrogen removal capacity of trickling filter depends on factors such as filter height, media, hydraulic and organic loading rate, and could be seen as a fixed ability. A fixed amount (load) of nitrogen is removed per day, while a higher load will result in increased effluent concentrations, and a lower load in decreased effluent concentrations. Therefore, with increasing urine separation efficiency, a linear improvement of effluent quality from existing trickling filters can be expected. Figure 63 shows an example where normal domestic wastewater, of the same composition as the average wastewater shown in Table 9 is treated in a trickling filter. Assuming that 30% of the organic, nitrogen and phosphate load is removed with primary settling, and that the trickling filter is of such design that only 50% of the settled sewage nitrogen is removed biologically, then 32 mg N/l would be found in the effluent, mostly as nitrate. With urine separation, there is a linear improvement in effluent quality with increasing urine separation efficiency, as each mg N less in the raw water, is a mg NO_3^- less in the effluent.

Trickling filters do not remove phosphate biologically and trickling filter plants rely on ferric chloride (or some other chemical) dosing to precipitate phosphate. With increasing urine separation, the amount of ferric chloride is reduced linearly with increasing urine separation efficiency. The phosphate removed with raw sludge, together with the phosphate from the separated urine could be precipitated at much higher concentration and recovered as calcium or magnesium phosphate.

9.5.2 Anoxic/aerobic sludge digester for N-removal and P-recovery

The work presented in chapters 7 and 8 can be integrated as shown in Figure 64. This is an ideal system for a gradual shift in wastewater collection and transport towards no-mix technology. It is also a fairly simple system, without the need for intricate automation and control that would be required in the more technologically advanced process shown in Figure 65. Considering the Johannesburg process in combination with the anoxic/aerobic digestion of waste activated sludge, the focus of nutrient removal would shift towards sludge handling with increasing urine separation efficiency.

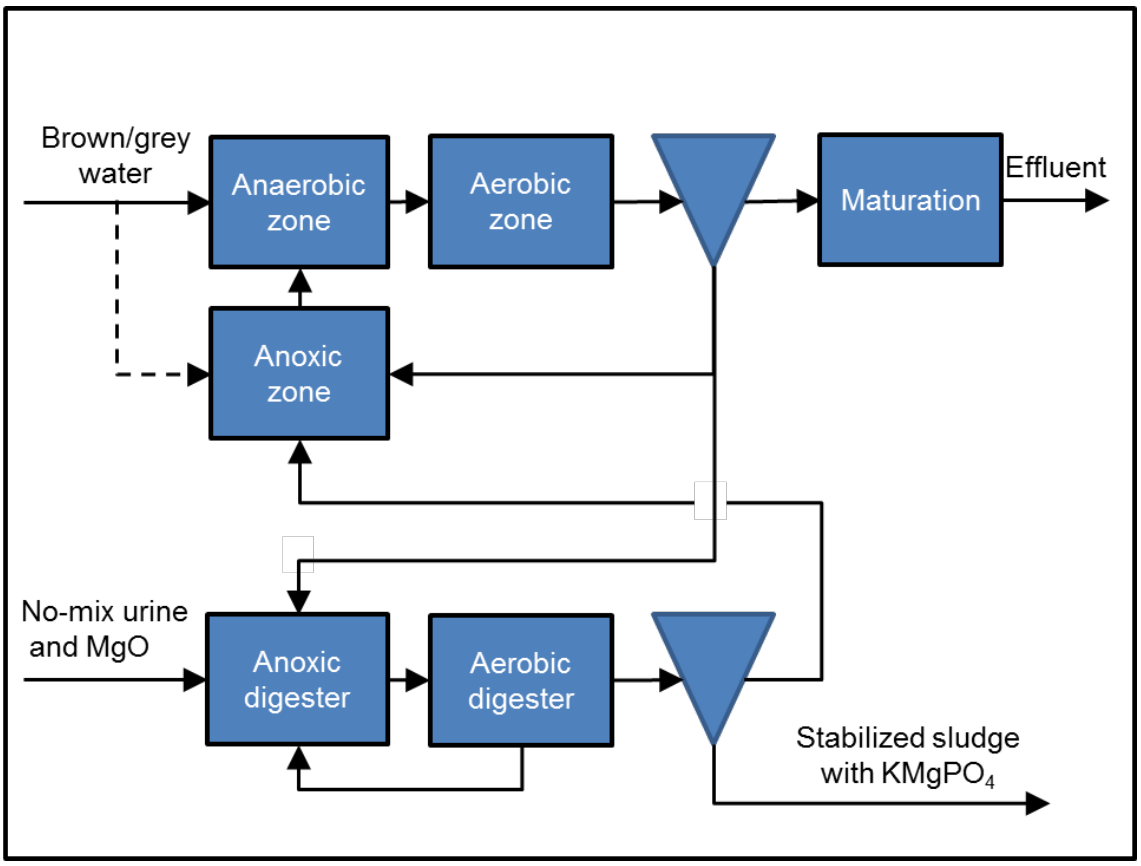


Figure 58 Combined treatment of domestic wastewater (grey and brown water) and source separated urine integrated with anoxic/aerobic digestion of waste activated sludge.

Ultimately, the main stream JHB process is focussed only on the removal of phosphate from the grey and brown water, in the anaerobic/aerobic configuration. The anoxic zone in the RAS removes any returned nitrate to ensure that the main stream reactor remains truly anaerobic for P-release and fatty acid uptake. The waste activated sludge from the JHB process is then treated with urine in an anoxic/aerobic digester. Here COD from urine is used for denitrification, together with some decay products from the waste activated sludge, and ammonium from urine, as well as from sludge hydrolysis is nitrified in the aerobic digester. Addition of MgO in the aerobic zone, where the ammonium concentration must be near zero, precipitates potassium phosphate and adds alkalinity for further nitrification.

The more effective urine separation within the catchment becomes, the shorter the sludge age in the JHB process would need to be, allowing additional treatment capacity for increasing loads.

9.5.3 Combined SHARON/Anammox process and phosphate crystallisation (supernatant treatment)

The ultimate combination of biological nitrogen removal and resource recovery is shown in Figure 65. For this system, the main stream treatment could be as simple as primary settling tanks with trickling filters. In this configuration, raw sludge or very unstable waste activated sludge is preferred to maximise the production of biogas and potential energy. Organic material broken down in sludge digestion would release both N and P in high concentration. At this point, source separated urine is introduced. Alternatively, urine can be introduced with WAS into the digester to buffer the digestion

with bicarbonate and convert COD in urine to biogas too, but the danger is ammonia toxicity of methanogens.

In the SHARON process, 50% of the ammonium is converted into nitrite, either through inhibition of nitrite oxidisers or maintaining low sludge ages at high temperature (T = 35°C) that effectively wash out nitrite oxidisers. The ammonium nitrite is then converted to nitrogen gas in the complete autotrophic Anammox process. After this step, ammonium is removed, and magnesium can be added for the recovery of potassium struvite. The treated supernatant is returned to the main stream.

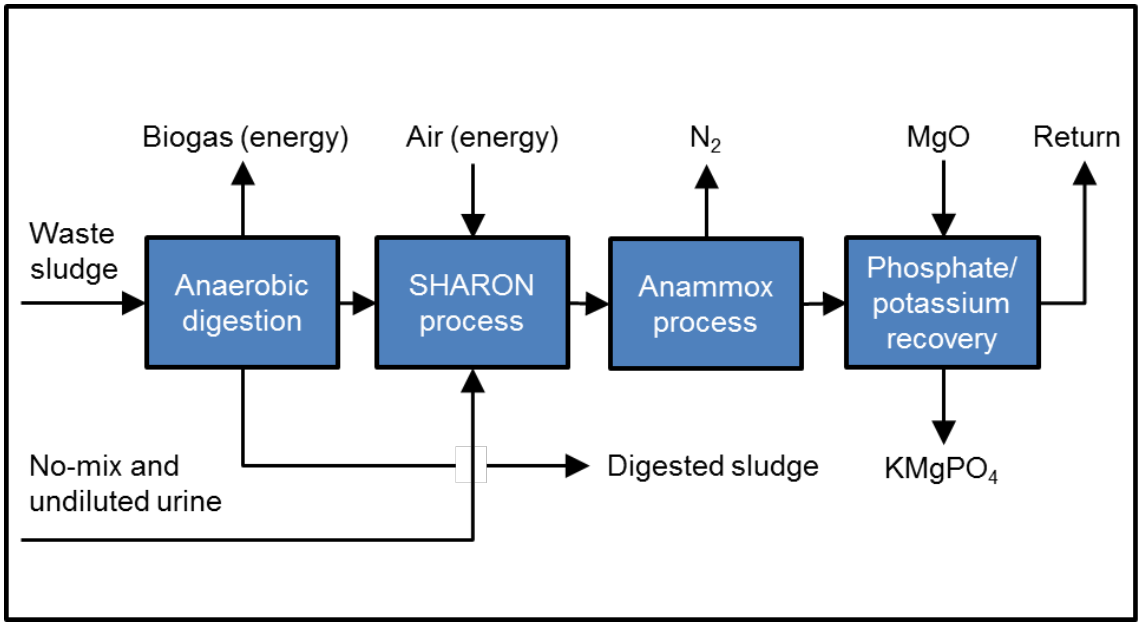


Figure 59 Combined SHARON-Anammox and phosphate crystallisation process

Currently, the cost of fertilizer, and the inherent energy cost, may not yet be high enough to warrant the recovery of ammonium from domestic waste streams. Furthermore, while SHARON/Anammox and other similar variants are still novel processes, these will grow in popularity with water utilities as the cost of energy increases relative to other goods and services. If the cost of energy continues to increase in real terms, at a higher rate than other costs, one may expect the recovery of ammonium, to replace production in the Haber-Bosh process, to become an interesting proposition. Ultimately, the energy efficient and low sludge production combined SHARON/Anammox process may be replaced at the end of its lifetime by an ammonium recovery process. However, at that time, logistical arrangements would have been streamlined for the separate collection, transport and handling of source separated urine and other no-mix streams. A further requirement at that time would be the return of recovered ammonium to agriculture, which adds another layer of complexity. Either way, the combined SHARON/Anammox process would have been more than just a useful technology bridge: it would already have contributed to a much more resource sensitive urban wastewater management system.

10 CONCLUSIONS

10.1 COMPOSITION OF URINE

Urine is a waste stream with extremely high concentrations of nitrogen (TKN = 6,600 mg N/l), COD (7,800 mg/l), phosphate (300 mg/l) and salts in general (14,400 mg TDS/l and 3,400 mS/m). Fresh urine changes composition completely as soon as it enters the non-sterile collection system. Urea hydrolysis results in formation of ammonium and bicarbonate, which increases the pH and leads to natural precipitation of minerals, including struvite, apatite and calcite. The measured concentrations of phosphate, calcium and magnesium must therefore have been considerably less than in fresh urine.

The high concentration of salts in urine contributes to the problem of steadily increasing salinity experienced in cities where wastewater effluent is further treated and recycled as drinking water.

When urine was stored and samples were taken over time, it was found that the TKN, ammonium and alkalinity increased with each subsequent measurement. During storage, urea and other organic nitrogen compounds are hydrolysed to ammonium and bicarbonate, seemingly more effective than with the chemical reagents of the analytical methods. The standard methods of analysis do not account for 100% of these concentrations in urine.

At least two thirds of the total nitrogen and almost half of the phosphate produced from sanitary waste originated from urine. Almost 10% of the COD originated from urine. Between 50% and 55% of the COD and phosphate originated from brown water. Grey water contributed 35-40% of the COD, and very little nutrients.

10.2 TREATMENT OF URINE

Urine from waterless urinals can be treated biologically in its undiluted form in a sequencing batch reactor (SBR). The high concentration of nitrogen, combined with the pH extremes of the SBR lead to high concentrations of free ammonia (at pH 8.5) and free nitrous acid (at pH 6.0-6.5). These compounds seem to inhibit the growth of nitrite oxidising bacteria, more than that of ammonium oxidising bacteria. This is seen in the high concentrations of nitrite that were produced, with very little nitrate. Denitrification over nitrite has the advantage that the limited COD in urine is used more effectively, i.e. more nitrogen is removed per unit of COD. The denitrification process also restores alkalinity consumed during nitrification. In this way, with an average influent concentration of 5.8 gTKN/l, nitrogen was converted to effluent concentrations of around 1.9 gTKN/l, 1.8 gNO_2^- and 0.3 gNO_3^-/l. Therefore, around 30% of the influent nitrogen was removed. The remaining ammonium nitrite stream is ideal for final treatment in an anammox process for autotrophic nitrogen removal. The nitrogen in urine could therefore be removed biologically to nitrogen gas, without the addition of any external COD.

Diluted urine from no-mix toilets was treated with waste activated sludge in an anoxic-aerobic digester. The waste activated sludge from the parent reactor contained a high fraction of phosphate accumulating organisms, amongst ordinary heterotrophic organisms and nitrifiers. The loads of nitrogen from WAS and from urine into the anoxic/aerobic digester were equal. Ultimately, around 30% of the volatile suspended solids were broken down, and the nitrogen added with the urine were

denitrified completely. Generally speaking, for every ten grams of nitrogen, five were denitrified, three remained part of the biomass and 2 were returned as nitrate.

10.3 IMPACT ON WASTEWATER TREATMENT

Operation of a UCT laboratory process with a 50:50 mixture of brown and grey water, at a sludge age of 20 days, resulted in a system in which no nitrifying organisms were present. Six independent factors pointed to the complete absence of nitrifiers. One of the findings was a lower than normal nitrogen content of the activated sludge, i.e. 0.89 mg N/mg VSS, which could point to a nitrogen deficient environment, where all the influent nitrogen was already consumed in bacterial cell synthesis. However, effluent ammonium concentrations were never very low (2.5 mg N/l), in contrast with conventional nitrifying systems that have much lower effluent concentrations (< 0.5 mg N/l). Herewith comes an unexpected conclusion: it is clearly not the affinity of nitrifiers for ammonium that limits or prevents their growth in the no-mix wastewater, but the low influent load. There is simply not enough ammonium (as energy source) to build and maintain a viable autotrophic biomass (i.e. the rate of decay and predation exceeds growth rate). This finding was not predicted by activated sludge models (Wilsenach and van Loosdrecht, 2004). Still, the overall removal of nitrogen was very good, at 90%, with zero nitrite/nitrate in the final effluent. Removal of phosphate was also very good at 93%. However, the removal of COD at sludge age of 20 days was less than expected: only 86% removal efficiency, with final filtered COD concentrations of 85 mg COD/l.

Operation of a JHB laboratory process gave poor but interesting results at 5 days sludge age. COD removal was even less effective than in the UCT process, but interestingly, the effluent COD was “mopped up” completely by biomass from another activated sludge reactor from the same laboratory, which was fed on normal municipal wastewater. We conclude that the fluctuation of the brown and grey water’s specific characterisation, as a result of a small population’s contribution, is much greater than that of an entire city where such variations are buffered out. Compounded with the short sludge age of 5 days, the biomass must struggle to adapt to the changing substrate. In the parallel system that “mopped up” the COD, there must be a wider variety of bacteria that’s seen all sorts of substrates before. It would therefore neither be the selection of process, nor the sludge age, that led to the systems poor performance, but the experimental set-up in wastewater collection and composition. This finding may explain some of the problems commonly experienced with package plants.

Ultimately, there is all evidence to believe that with effective urine separation, activated sludge processes could be operated as a two-stage phoredox process (anaerobic/aerobic) or JHB process (anaerobic/aerobic with anoxic zone in the sludge return) at low sludge age (e.g. 5 days). This configuration would reduce the required reactor size by half, or alternatively, treat double the load in an existing reactor.

10.4 PRACTICAL LESSONS

No-mix toilet design

The ladies no mix toilet urine pipe sometimes blocked with toilet paper. In the event that toilet paper landed inside the front compartment there was a good chance some of it would get stuck in the urine pipe after flushing the Gustavsberg no-mix toilet. In that case, urine would be flushed away with brown water after the next toilet use. The blocked urine pipe also led to odours due to stale urine in the toilet paper plug. It was also found that when releasing a full bladder, some urine would overflow

from the urine compartment into the brown water compartment. Both these problems must be addressed in no-mix toilet design. Firstly, ineffective urine separation defeats the purpose of the installation. Secondly, no-mix toilets and therefore no-mix technology would only be acceptable as long as these are as simple to use and maintain as conventional toilets.

Other manufacturers of no-mix flush toilets have devised mechanisms that open and close the urine outlet via a spring release. When a toilet user sits down, the mechanism opens and urine can flow through. As soon as the user stands up, the mechanism closes and no flush water or foreign material can enter the urine drain pipe. However, this mechanism comes with an additional purchase cost, and would no doubt result in more maintenance work due the number of moving parts. It may also be prone to problems due to build-up of precipitants like struvite or calcite.

It was also found that the further into the project, the more dilute the yellow water became (from initial concentrations of 1.5-2.1 g TKN/l down to below 1 g TKN/l. This is most likely due to wear on the flush mechanisms, and these not closing properly in time.

Design criteria to revisit are:
1) Minimise the amount of flush water, in order to prevent dilution of urine.
2) Prevent any urine from ending the brown water compartment.
3) Prevent foreign material (anything other than urine) from entering the urine drain pipe.

Perhaps all the above could be achieved by moving the urine drain to the front of the yellow water compartment, instead of at the divider (Figure 13). A different position of the urine drain and a different slope of the basin could be further investigated. If the design criteria are conflicting, then effective urine separation is much more important than minimising flush water interception. The no-mix toilet design could be reconsidered completely, with a simple device similar to that used as odour traps in the waterless urinals, and with all flush water directed only into the brown water compartment.

Retrofitting and wastewater collection design

To prevent odours from the collection tank, the brown water pipes were routed directly into the collection tank, without additional air vents. The collection tank was open to atmosphere, except for a loosely fitted lid. Still, air entrapment inside the collection tank interfered with the functioning of the no-mix toilets. With the collection tank near full volume, the brown water drain backed-up and the no-mix toilets did not flush properly, possibly because of an air-lock in the piping system. This is not thought to be a design error of the toilet, but rather of the specific installation where the system was installed not to vent to atmosphere. Furthermore, the grinder pump installed inside the brown water collection tank was used to pump the wastewater through a recycle pipe (blue flexi-hose) to effectively macerate and homogenise the waste before shipment to UCT. As soon as the pump was turned on, odours occurred in the toilet rooms. The agitation of the collection tank and closed air space must have volatilised odours that leaked through the toilet's odour trap. These problems would have been avoided by an air outlet line connected via T-piece at each of the toilets' brown water outlets, immediately outside the building, vented to above the building's roof.

Counting toilet uses

Electronic counting devices (based on obstructed light beams, as used at shop entrances) did not provide reliable numbers of no-mix toilet users. The signal created by some-one crossing the beam

was registered in a counter with electronic display, in order to quantify toilet uses and relate this to the volume of waste produced. However, it was quickly discovered that the “number” of no-mix toilet users was far too high for the amounts of wastewater produced. It became evident that the cleaners were impacting on the counting system and that their presence inside the toilets was providing false indication of unrealistic high numbers of toilet users.

11 RECOMMENDATIONS

11.1 A STRONG VISION

Separate collection, transport and treatment of urine is a novel concept. However, it has all the prospects of leading to more sustainable urban water management, with increased wastewater treatment capacity, reduction in energy consumption and improved effluent quality and recycled water quality. In order to advance quickly, some risks need to be taken, especially since we concluded that a scaled-up collection and treatment system is needed to quantify and model the system in detail.

Deviation from the current water borne sanitation will be less convenient for users, but will improve the surface water environment (almost immediately) and enhance sustainable development. Adoption of urine separation sanitation by middle- and high-income urban groups will change the perception that urine separation is an inferior system for low-income communities. Therefore, strong political will, administrative leadership, significant financial investment and large scale public education would be vital to embark on a no-mix wastewater pilot system. Education and awareness could take the form of catchy campaigns, with slogans like “Fix no-mix”, or “P from pee.” Therefore:

- The Department of Water Affairs should appoint a “no-mix champion”.
- The Water Research Commission should continue the support for this work, through one of the thrusts (e.g. “Sustainable municipal wastewater and sanitation”).
- Institutions mandated with the incubation and transfer of technology, such as the Council for Scientific and Industrial Research and the Technology Innovation Agency, should support the intellectual and financial funding of a pilot project.
- Water utilities, such as ERWAT, Johannesburg Water, City of Tshwane and City of Cape Town, should follow the example of eThekwini Water and Sanitation, but also in the context of human settlements with sewers.

Town planners have the opportunity to change conventions based on linear consumption with an implicit assumption of infinite resources. The drivers for change that motivate no-mix wastewater technology are equally important for changes in urban design. Politicians, planners, and engineers should convene around the theme of no-mix wastewater technology in a transition towards water sensitive cities. At the same time, no-mix wastewater technology would promote the transition towards “nutrient sensitive cities” within “energy and transport sensitive cities”.

11.2 PILOT PROJECTS

Results from this project, combined with earlier modelling work and international experience show that wastewater and urine treatment technology could be based on activated sludge and water chemistry first principles. Imperfections and non-optimal configurations would no doubt be further perfected. Learning is now needed from large scale complex operations that would ensure the separation, collection transport and management of no-mix streams as smoothly as conventional sewers. No amount of theorizing can offer this learning, which is therefore the motivation for pilot projects at urban places like:

- Sport stadiums (soccer, rugby, cricket)
- Large air ports (OR Tambo, King Shaka, Cape Town)
- State institutions, such as prisons, hospitals, schools, universities, etc.
- Government and parastatal buildings and office blocks
- New multi-storey housing developments, or inner city rejuvenation projects

Apart from these places, remote locations offer scope for pilot projects:

- Eco-estates, wildlife-estates, etc., bordering cities.
- Game Lodges, National – and Provincial Parks
- Freeway service stations as well as Border Control Ports of Entry, where the normal wastewater consist mostly of urine already
- The South African National Antarctic Expedition base (SANAE IV)

Although there are many benefits to no-mix wastewater technology, which includes process customization for waste stream composition, process stabilisation, improved resource efficiency, mineral recovery, better effluent quality, water recycle, etc. the objective of implementation and practical implications are unique for each location. Learning is required on aspects such as human behaviour in public toilets, prevention of and dealing with odours, preventative and reactive maintenance of no-mix systems, storage design and optimization, transportation design and optimization, manual handling and other human interfaces.

11.3 WATER RESEARCH QUESTIONS

Interesting and challenging scientific questions stem from this project, some of which lies at the fringe of existing knowledge or models. The topics listed below could be single questions or more comprehensive research programmes:

1. Optimization of no-mix toilets

For no-mix technology to be successful, it is important that all no-mix toilets are near 100% effective in separating urine and faeces within the toilet. This criterion demands that all urine is collected in a frontal basin and does not overflow into the faecal compartment of the toilet. At the same time, flush water in the urine collection system should be limited to prevent dilution and transport cost. This could be achieved with valves and closing mechanisms, but a free flow system is more elegant and requires less maintenance. The questions are therefore: i) what is the statistical description of direction, pattern, duration and flow rate of urine when discharged from the human body in a seated position (male and female), ii) how can such flood events be modelled or described and iii) how can these flood events be routed hydraulically without overflow, using the minimum of barriers?

2. Activated sludge models and the limits of technology

All the activated sludge models published by the International Water Association (IWA) assume domestic wastewater with nutrients in excess of growth requirements. Based on these models Monod kinetics and affinity constants determine the limits of technology, so that requirements for ultra-low concentrations point towards very long process reaction time with large reactor volumes. Therefore, where the technology is defined by the effluent quality, the daily load is of lesser concern. With no-mix toilets, the wastewater load is radically altered, but this does not necessarily and automatically result in better effluent quality for all parameters. Activated sludge models would predict a nitrifying biomass to develop even with very low ammonium loads, but experimental work shows this is not the case, i.e. no nitrification occurs at low ammonium load, while ammonium remains in the effluent. Therefore: what happens in activated sludge systems, especially with regard to nitrifiers, at low nitrogen load, where no nitrification takes place, and can phosphate be removed to extreme low concentrations in the absence of nitrate? Can this knowledge be used to improve the IWA family of activated sludge models to the fringes of possibility?

3. Cell synthesis under nutrient limitation

If no-mix technology could be introduced, with the aim of improving wastewater treatment through activated sludge without the need for nitrification, then all nitrogen has to be consumed in conventional activated sludge biomass through cell synthesis. One of the implications of such a system would be a perfectly balance waste stream, in which the COD:TKN ration leads to consumption of energy, carbon source and nitrogen in the exact ratio so that nothing is left. With too little nitrogen in the system, the COD may not all be converted by growth, and too much nitrogen would go through the system untouched. However, is there a window of opportunity for sludge to grow, even with a ratio COD:TKN ratio well above normal measurements? Does nitrogen limitation impact on the growth of phosphate accumulating organisms, or would these or similar organisms eventually convert excess COD into storage products? However, would there also be organisms with sufficient affinity for nitrogen to ensure low effluent concentrations, and can these organisms co-exist?

4. Plant wide modelling of integrated no-mix streams

In order to optimise a process configuration where no-mix brown water, grey water and urine are all collected separately, but treated in an integrated plant, the resource value of each stream must be considered. Development of ideas for such an optimisation requires a mass balance approach. Plant wide modelling allows balancing components (nitrogen, phosphate, alkalinity and different fractions of COD) from different influent streams, with the effluent streams. Inversely, can plant wide measurements be used to further quantify and calibrate the composition of the different no-mix streams?

5. Nitrification growth rate under extreme conditions of undiluted urine

Although urine can be effectively treated with activated sludge, the rate is relatively slow due to extreme conditions of high pH and low pH in a sequencing batch reactor, and respectively high concentrations of free ammonia and nitrous acid, as well as salinity. Due to this variety of extremes, the inhibition of nitrite oxidising organisms is generally greater than that of ammonium oxidising organisms, which is a very useful fact. What then are the operating conditions that maximise the growth rate of ammonium oxidising bacteria, while at the same time inhibiting the growth rate of nitrite oxidising bacteria?

6. Urine treatment optimization and automation

Experimental results showed an inverse relation between pH and redox (ORP) levels during the filling, denitrification and nitrification in a sequencing batch reactor treating urine. The pH profile shows a sharp increase due to ammonium bicarbonate addition during filling, a gradual increase due to denitrification, followed by a decrease during nitrification, while the redox level more or less mirrors this profile. Ammonium and COD are both in reduced form and results in a negative OPR, while higher nitrite concentrations result in a positive ORP. From the perspective of plant automation, the anoxic phase must come to an end when the readily available COD has been oxidised through denitrification of nitrite. Stopping the anoxic phase too soon would result in COD being converted aerobically, and maintaining the phase for too long would slow down the overall process rate. Is it possible to calibrate the pH and redox profile to the approximate concentrations of readily available COD, with electrons to offer, amongst the electrons on offer from ammonium, so that the anoxic phase can be stopped at the right time? Furthermore, at the end of the aerobic phase, the pH drops

and the nitrification rate slows down. Can the rate of change in pH and redox profile be calibrated to approximate ammonium and nitrite concentrations so that the process stops at a point where the nitrification rate decreases below a viable value, but where at the same time, a sufficient ammonium load has been converted? Can this knowledge be further refined to maximise ammonium growth rate while effectively inhibiting nitrite oxidation (refer question 5)?

7. Granulation of nitrification/denitrification organisms

Anammox bacteria in an internal recirculation reactor are known to form activated sludge granules, instead of flocs. These slow growing organisms can therefore be maintained at a high density within the reactor, while at the same time reducing the settling time before decanting. Granulation has also been established as a novel process for conventional activated sludge processes treating normal domestic wastewater. Is it also possible to form granulating activated sludge biomass, consisting of ammonium oxidising bacteria and ordinary heterotrophic organisms specialised in denitrification, in a sequencing batch reactor treating urine? Furthermore, is it then still possible to inhibit nitrite oxidising bacteria, within the more protective layers of the granules, and favour denitrification over nitrite?

8. Origin and fate of micro-pollutants in no-mix streams

The present study did not investigate the presence of micro-pollutants, including pharmaceutical residues and endocrine disrupting chemicals (EDC) such as synthetic hormones. Many of these chemicals are found in urine, and it is thought that if urine were collected separately, the micro-pollutants can be removed more effectively in side stream processes. How much and which type of micro-pollutants originate from urine, and how much originate from faeces? Does a sequencing batch reactor for nitrification and denitrification (at say 20 day sludge age) remove micro-pollutants more effectively than a conventional main stream activated sludge plant? With no-mix technology, where grey and brown water can be treated in a JHB process at low sludge age (say 5 days) and low hydraulic residence time, would the remaining micro-pollutants from faeces still be removed effectively?

9. Recovery of phosphate and potassium from urine, yellow water and brown water

Phosphate precipitates naturally in stored urine, as apatite or struvite in the presence of excess ammonium ions. In the absence of ammonium, phosphate is known to precipitate as potassium struvite ($KMgPO_4.6H_2O$). Roughly one half the phosphate in wastewater originate from urine. The other half originate from faces and this phosphate can be removed through phosphate accumulating organisms. How can the concentrated phosphate released from the anaerobic zone of a JHB process be combined (configured) with the phosphate from treated urine, in order to maximise recovery of potassium struvite? What kind of reactor and reactor conditions are required to optimize the recovery of potassium struvite?

10. Validity of analytical methods for determining concentrations of chemicals in urine

When urine is stored, with samples taken weekly for analysis, it seems as if the concentration of total kjeldahl nitrogen and the alkalinity increases with time. However, this cannot be possible, since the total nitrogen and alkalinity cannot be changed by storage only. Clearly some of the urea (or perhaps other nitrogen compounds?) are not seen by the analysis. However, over time, as natural hydrolysis takes place, these compounds are released into solution and react with analytical reagents. What are

the additional steps (e.g. longer acid digestion) to be introduced into the standard operating analytical procedures in order to measure concentrations of nitrogen and alkalinity in undiluted urine accurately?

12 LIST OF REFERENCES

Andersen, L.B. (2012), Lifecycle cost saving benefits. World Pumps, 2012 (12) 24-26

Anthonisen, A. C., Loehr R. C., Prakasam, R. B. S., Srinath, E. G. (1976). Inhibition of nitrification by ammonia and nitrous acid.*Water Pollution Control Federation*, 48(5),835-852.

Ashton, J.R. (2006), Back to back housing, courts, and privies: the slums of 19th century England. Journal of Epidemiology and Community Health, 60 (8):654

The JECH gallery

Bitton, G. (2005). Wastewater Microbiology, 3rd ed., John Wiley and Sons Inc., Hoboken, New Jersey, USA.

Cadbury, D. (1997). *The feminization of nature: our future at risk.* Hamish Hamilton Ltd., London.

Ciba Geigy. (1977). *Wissenschaftliche Tabellen Geigy, 8th Edition, Körperflüssigkeiten.* Basel.

Clayton, J.A., Ekama, G.A. & Wentzel, M.C. (1991). Denitrification kinetics in biological nitrogen and phosphorus removal activated sludge systems treating municipal wastewaters. *Water science and technology.*23 (4-6):1025-1035.

Downing, A.L. and Hopwood, A.P. (1964). Some observations on the kinetics of nitrifying activated-sludge plants. *Aquatic Sciences*, 26(2):271-288.

Ekama, G.A., Marais, G.v.R., Siebritz, I.P., Pitman, A.R., Keay, G.F.P., Buchan, L., Gerber, A. & Smollen, M. (1984). Theory, Design and Operation of Nutrient Removal Activated Sludge Processes: A Collaborative Information Document*. Water Research Commission.*

Ekama, G.A. & Wentzel, M.C. (1999). Denitrification kinetics in biological N and P removal activated sludge systems treating municipal wastewaters. *Water science and technology.*39 (6):69-77.

Feng, D., Wu, Z., Xu, S. (2007). Nitrification of human urine for its stabilisation and nutrient recycling. *Bioresource Technology*, 99:6299-6304.

Fenu, A., Roels J., Wambecq, T., de Gussem, K., Thoeye, C., de Gueldre, G. and van De Steene, B. (2010) Energy audit of a full scale MBR system. Desalination 262 (1-3) 121-128.

Galloway, J.N., Cowling, E. B. (2002). Reactive nitrogen and the world: 200 years of change. *Ambio*, 31(2):64-71.

Gerardi, M. H. (2006) Nitrifying Bacteria, in *Wastewater Bacteria*, John Wiley & Sons, Inc., Hoboken, NJ, USA

Harding, W.R. (2008) Determination of Annual Phosphorus Loading Limits for South African Dams WRC Report No 1687/1/08

Hellström, D. and Johansson, E. (1999). Swedish experience with urine separating systems. *Wasser und Boden*, 51(12):26-29.

Henze, M., Van Loosdrecht, M.C.M., Ekama, G.A. & Brdjanovic, D. (2008). *Biological wastewater treatment: principles, modelling and design.* London: IWA Publishing.

Hu, Z., Wentzel, M.C. and Ekama, G.A. (2001). External nitrification in biological nutrient removal activated sludge systems. *Water science and technology,* 43(1):251-260.

Jönsson, H., Stenström, T.A., Svensson, J., and Sundin, A. (1997). Source separated urine-nutrient and heavy metal content, water savings and faecal contamination. *Water Science and Technology*, 35(9):142-152.

Ju, L.K., Hemant, K.S. and Porteous, J. (2005). Phosphorus Release in Aerobic Sludge Digestion, Water Environment Research, 77(5):553-559.

Kirchmann, H., Pettersson, S. (1995). Human urine – Chemical composition and fertilizer use efficiency. *Fertilizer Research*, 40:149-154.

Larsen, T. A. and Gujer, W. (1996). Separate management of anthropogenic nutrient solutions (human urine). *Water Science and Technology*, 34 (3-4).

Lienert, J., Larsen, T.A., Maurer, M. and Udert, K.M. (2007). Nutrient Cycles and Resource Management: Implications for the choice of Wastewater Treatment Technology. *Water Science and Technology*.

Marais, G.R. & Ekama, G.A. (1976). The activated sludge process.Part 1-Steady state behaviour. *Water SA,* 2(4):164-200.

Maurer, M., Schwegler, P., Larsen, T.A. (2003). Nutrients in urine: energetic aspects of removal and recovery. *Water Science and Technology*, 48(1):37-46.

Maurer, M., Pronk, W., Larsen, T.A. (2006). Treatment processes for source separated urine. *Water Research*, 40:3151-3166.

Monod, J. (1949). The growth of bacterial cultures. *Annual Review of Microbiology*, 3:371-394.

Mouton, M. and Botha. A. (2012) Cutaneous Lesions in Cetaceans: An Indicator of Ecosystem Status? InTech – Creative Commons Attribution License. http://dx.doi.org/10.5772/54432

Otterpohl, R. (2002) Options for alternative types of sewerage and treatment systems directed to improvement of the overall performance. Water Science and Technology 45 (3) 149-158.

Randall, E.W., Wilkinson, A. & Ekama, G.A. (1991). An instrument for the direct determination of oxygen utilisation rate. *Water SA* 17(1):11-18.

Roma, E., Philp, K., Buckley, C., Xulu, S. and Scott D. (2013) User perceptions of urine diversion dehydration toilets: Experiences from a cross-sectional study in eThekwini Municipality. *Water SA,* 39 (2): 305-312

Ronteltap, M., Biebow, M., Maurer, M., and Gujer, W. (2003). Thermodynamics of struvite precipitation in source separated urine. In *Proceedings from the 2nd International Symposium on Ecological Sanitation*, Lübeck, Germany.

Svardel, K. and Kroiss, H. (2009). Energy Requirement for Wastewater Treatment. *Water Science and Technology*, 64(6): 1355-1361.

Udert, K. M., Fux, C., Munster, M., Larsen, T.A., Siegrist, H., Gujer, W. (2003a). Nitrification an autotrophic denitrification of source separated urine. *Water Science and Technology*, 48(1):119-130.

Udert, K. M., Larsen, T.A., Biebow, M., Gujer, W. (2003b).Urea Hydrolysis and precipitation dynamics in a urine collecting system. *Water Research*, 37:2571-2582.

Udert, K. M., Larsen, T.A., Gujer, W. (2003c). Estimating the precipitation potential in urine collecting systems. *Water Research*, 37:2667-2677.

Udert, K.M., Larsen T.A., Gujer, W. (2003d). Biologically induced precipitation in urine-collecting systems. *Water Science and Technolgy*, 3(3):71-78.

Udert, K. M., Larsen, T.A., Gujer, W. (2006). Fate of major compounds in source separated urine. W*ater Science and Technology, 54(11/12)*:413-420.

Vale, L. (2007). From the Puritans to the Projects: Public Housing and Public Neighbors, Harvard University Press

Van Vuuren, S.J. and van Dijk, M. (2006) Life cycle costing analyses for pipeline design and supporting software. WRC Report no. TT 278/06

Vinnerås, B. and Jönsson, H. (2002) The performance and potential of faecal separation and urine diversion to recycle plant nutrients in household wastewater. Bioresource Technology. 84 (3) 275-282

Vogts, M., and Ekama, G.A. (2012) Anoxic-aerobic digestion of waste activated sludge from biological nitrogen and phosphorus removal systems. WISA Biennial Conference, Cape Town.

Wentzel, M.C., Ekama, G.A., Dold, P.L. & Marais, G.R. (1990). Biological excess phosphorus removal. Steady state process design. *Water S.A.* 16(1):29-48.

Wilsenach, J.A. (2005). Options for separate treatment of urine. STOWA 2005-11, ISBN: 90.5773.300.5

Wilsenach, J.A. & van Loosdrecht M.CM. (2002). Separate urine collection and treatment. Options for sustainable wastewater systems and mineral recovery. STOWA 2001-39, ISBN: 90.5773.197.5

Wilsenach, J.A. (2006). *Treatment of source separated urine and its effects on wastewater treatment systems.* Ph.D. Thesis. Delft University of Technology, Delft, Netherlands.

Wilsenach, J.A. and Loosdrecht. (2003). Impact of separate urine collection on wastewater systems. *Water Science and Technology*. 48(1):103-110.

Wilsenach, J.A. and van Loosdrecht, M.C.M. (2006) Integration of processes to treat wastewater and source separation urine. Journal of Environmental Engineering. 132(3) 331-341.

Wilsenach, J.A., Schuurbiers, C.A.H. and van Loosdrecht, M.C.M. (2007). Phosphate and potassium recovery from source separated urine through struvite precipitation. Water Research 41(2) 458-466.

APPENDIX A: PERCEPTION SURVEY QUESTIONNAIRES (CHAPTER 3)

1st questionnaire: January 2009

A total of 18 questionnaires were distributed, out of which 10 were submitted. This section deals with those answers:

Question 1: So, why is wastewater treatment important?

This was answered well, as expected from a professional group of water and health researchers. Most participants said that it was important to "ensure satisfactory human health, protect the environment, and to protect water resources". Other answers pointed out that "water is a limited resource, and wastewater can be re-used".

***Question 2: What are important chemical components in urine? Nutrients: Salts: Organic components*?**

Eight respondents answered this question correctly stating that the nutrients were nitrogen and phosphorous based compounds. Salts included sodium, potassium, magnesium, chloride, and the organics consisted of hormones, metabolites and urea. Two people simply answered that nutrients, salts and organics were all important but failed to give examples of each.

Question 3: What is a no-mix toilet?

Nine people responded correctly, saying that it is a toilet that separates urine from solid waste (faeces), and one person was not sure.

Question 4: Are you a little nervous about the first time you will use a no-mix toilet, or can't you wait, or are you totally indifferent, or something else?

One person did not reply, one person was interested, and one person nervous but still curious. Four people were indifferent, or not bothered, or thought that it would be like any other toilet. Two people were more enthusiastic, and their comments were: "What can I say, it's about as good as Christmas" while another male person, stated "Can't wait, I have been having aiming practise lessons".

Question 5: What should you do with used toilet paper after giving no-mixing your best shot?

Six people indicated that it should be disposed of as usual, at the back of the toilet bowl. Three people stated that the used toilet paper should be thrown in a paper bin, while one person was again not sure.

Question 6: How can a no-mix toilet contribute to improved wastewater treatment?

Most replies confirmed that once urine is diverted from wastewater at source, it can be treated separately, and it will decrease the nutrient load going to the treatment works, thereby improving the effluent quality.

Question 7: How much potable water is used by the average middle income person per day (guess if you don't know!)?

Most participants answered correctly saying that the amount of water used per day was between 120 -300 litre per person per day, depending of socio-economic class. This includes drinking water, washing water, bath/shower water, and toilet water. Two participants gave 25 l/p.d as the water use, perhaps only thinking of the water they consumed or used at work.

Question 8: How much urine is produced by the average person per day (guess if you don't know!)?

Most men answered that they produce about 0.5-1.0 l of urine per day (although one man did answer 2.5 l). The women answered 0.5-2.0 l of urine per day.

Question 9: What are typically good effluent concentrations for advanced wastewater treatment plants? ___ mg COD/l _____mg N/l _____mg P/l

Three of the respondents (specifically the laboratory staff) answered correctly for all three categories, namely, 75 mg COD/l, 10 mg N/l, 1 mg P/l. Most of the other participant did not know and guessed.

Question 10: Where would you imagine the separately collected urine to go to, and what happens there?

Seven people said that it is temporarily stored then treated in a special reactor. Three people said that it can be treated and recycled as a fertilizer for agricultural purposes.

Question 11: What (if any) health risks are associated with separately collected urine?

Three people said that there was no health risks, two people said that there were none if the urine was handled correctly, but there was potentially a risk of contagion if some-one using the system had a bladder infection.

The other half of participants mentioned the presence of hormones and pharmaceutical residues in urine, and there might be some pathogens, viruses and bacteria in the separated urine (maybe as a result of cross contamination). One person also answered that urine contact may cause skin irritation.

An additional health risk (for someone working directly with urine) not mentioned by the participants is the potential hazard of inhaling high concentrations of ammonia gas. This is possible if one works with urine collected in the 25 l collection containers.

Question 12: Would you say you're at least somewhat environmentally conscious?

Nine out of ten people answered YES, and one person said "A little bit". This is overall not surprising since everyone surveyed worked at CSIR's Natural Resource and Environment operating unit.

Question 13: Is enough being done in South Africa in terms of environmental innovation?

Nine out of ten people indicated that there was NOT enough being done in South Africa. One person said “There’s so much scope for more (environmental innovation)!”

Question 14: Are you very happy with the installation of a no-mix toilet at CSIR (and if you say no, why are you not very happy!?)

Four people said YES, another four replied positively with comments such as “it shows innovation, it’s a step in the right direction, it’s a forward thinking approach, and it’s an innovative idea”. Two people were not sure because they had not used one yet. Overall, at that initial stage, it seemed that no-one was really against the introduction of the experimental no-mix toilet installations.

Question 15: So when are you going to ‘fork out’ extra money to install one at your home, huh?

A variety of answers came from this question:

1. “As soon as dual effluent collection becomes available”.
2. “When we have the infrastructure (for urine treatment) in place”.
3. “Make it commercially viable at an affordable price”.

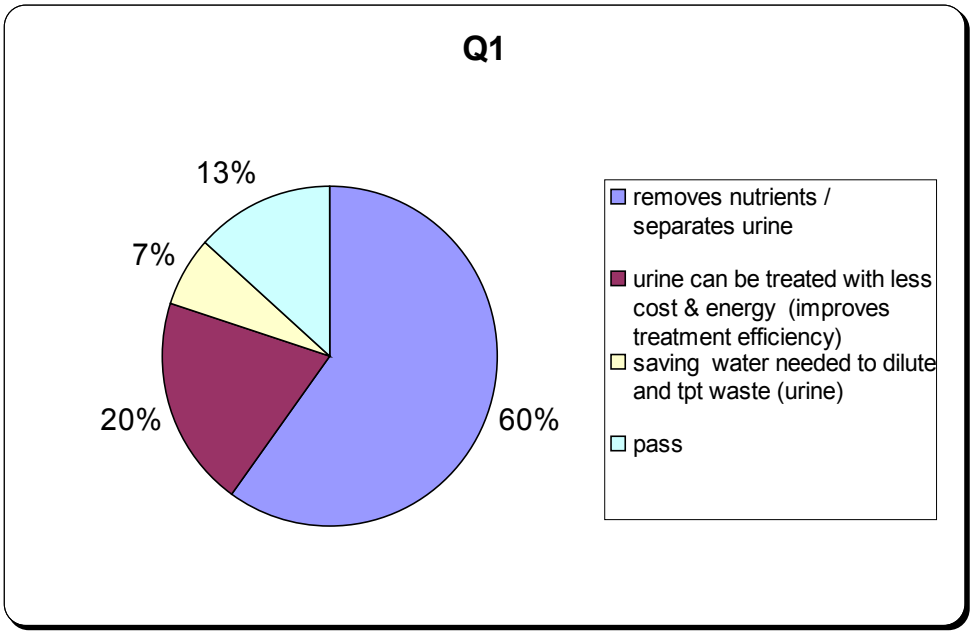
And the rest replied that money was an issue, i.e. the installations were too expensive. This is a fair statement because the imported toilets cost €344/each (approximately R3,830 at the time) which was far more than the price of a standard locally-made porcelain toilet.

2nd questionnaire: October 2009

A total of 15 questionnaires (7 from men, 8 from women) were completed out of 18 handed out.

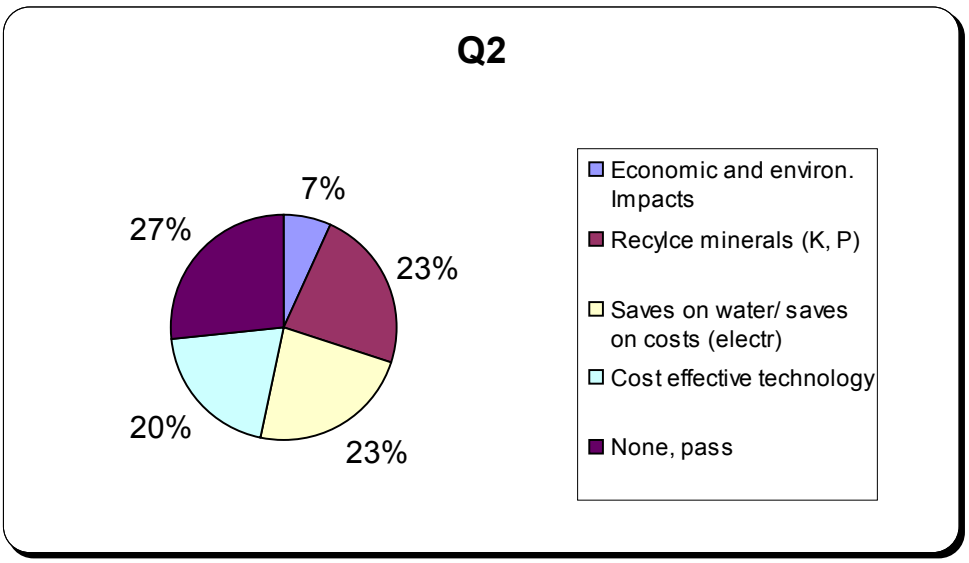
Question 1: Again, a familiar question, but still: How can a no-mix toilet contribute to improved wastewater treatment, or perhaps the entire water cycle?

Nine people answered that no-mix technology improves wastewater treatment by separating urine and hence removing the nutrients. The remaining people said that urine can be treated at a lower cost but this point still needs to be determined by the project team. One person mentioned that "less water will be needed to dilute and transport the waste", which is correct but rather a spin off from this project and not a key focus point.



Question 2: Are there other benefits that you can think of, have heard of, etc.?

As an additional benefit, 23% of respondents said that minerals can be recycled from source separated urine. An additional 23% said that added benefits included water savings, treatment cost savings, and it promotes "green thinking". A total of 20% of respondents said that it would lead to cost effective treatment technology and 27% skipped this question, unable to suggest other benefits.

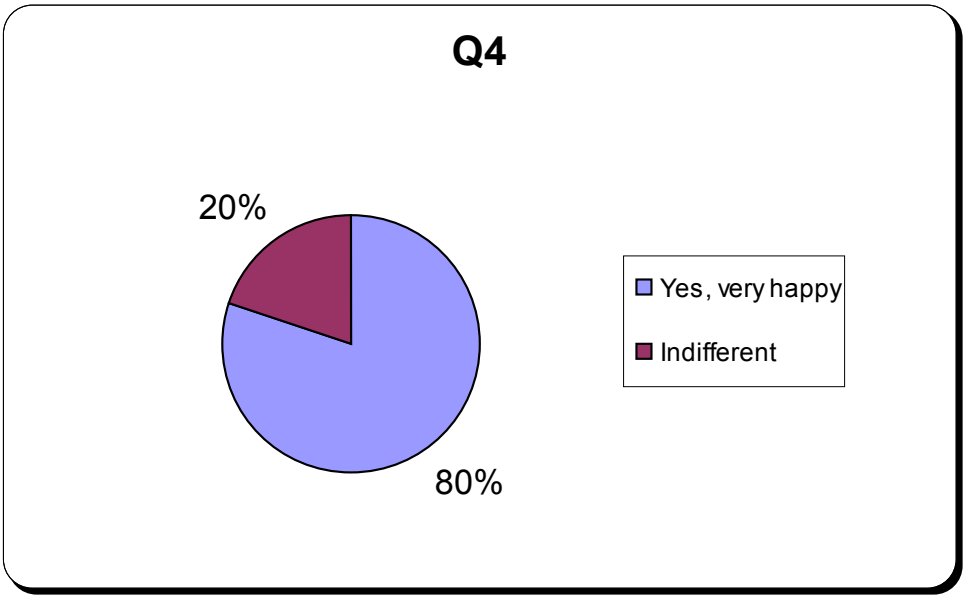


Question 4: Are you very happy with the installation of a no-mix toilet at CSIR (and if you say no, why are you not very happy!?)

The majority of participants (80%) were happy with the no-mix toilets. Examples of some of their comments were as follows:

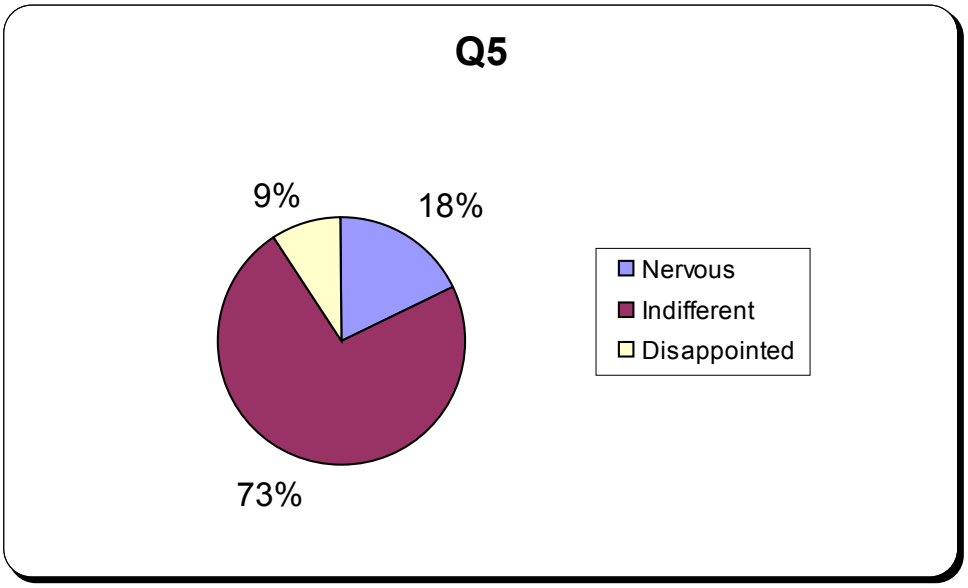
1. "Yes, it's innovative and will contribute to the advancement in science for the CSIR. Practising what we preach in a sense".
2. "Yes, the study may prove to be beneficial to the wastewater treatment industry and may also be a water-saving technology".
3. "Yes, it is innovative and CSIR is setting a good example".

The remaining 20% participants were indifferent.



Question 5: Did the toilets meet your expectations (Were you nervous at first? Were you indifferent? Has it been an anti-climax?)

Most of the participants (73%) acknowledged that they were indifferent, curious, or had no expectation of the no-mix toilet. About 18% said that they were nervous at first. One person replied saying that he was disappointed with the system, because there was too much water mixing in the urine compartment, which would decrease the urine separation efficiency of the system.



Question 6: Have you noticed any operational problems in using these no-mix toilets? If so what can be done to improve the design??

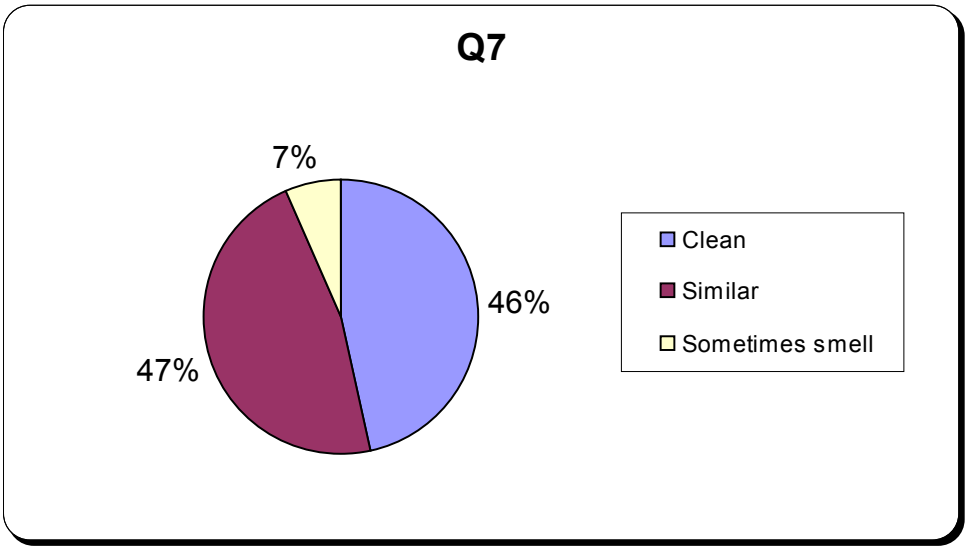
All the men answered that they had not experienced any problems with the no-mix toilets. This may be since men may have used the no-mix toilet less frequently to urinate, since urinals are available in the gents' toilets. Women were possibly more likely to make use of the no-mix toilet, where half of the women had experienced some problems, while the other half stated that no problems were noticed.

The problems experienced by half the women occurred as a result of:

- Toilet paper blocking the narrow metal urine pipe after flushing. This metal pipe had no cover or sieve to prevent the toilet paper causing this blockage.
- With the collection tank near full volume, the brown water drain backed-up and the no-mix toilets did not flush properly, possibly because of an air-lock in the piping system.

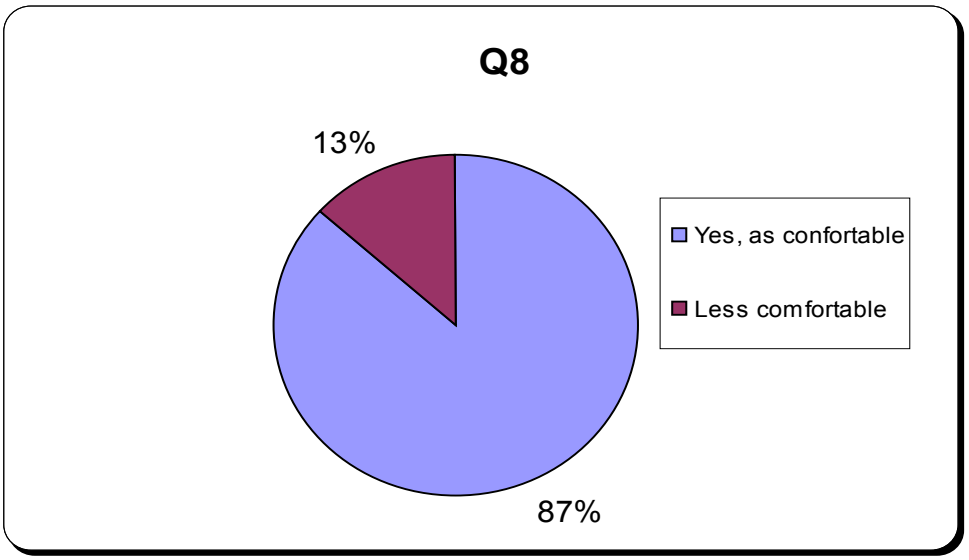
Question 7: Are the no-mix toilets clean, do they smell (more than an ordinary flush toilet)?

The majority reported that there were no bad smells, where half the respondents said the no-mix toilets were clean and half said that they were the same. Only one person reported that the toilets smelled sometimes and this may be a result of the blockage described in question 6 above.



Question 8: Are they comfortable (more or less than an ordinary flush toilet)?

Most of the people said that these toilets were just as comfortable (87%). One person commented that the toilet seats were slightly smaller than the previous (locally-made) toilet seats. Another comment was that one had to adjust their position (which made the toilet less comfortable) because of the way the toilet was designed to separate urine.

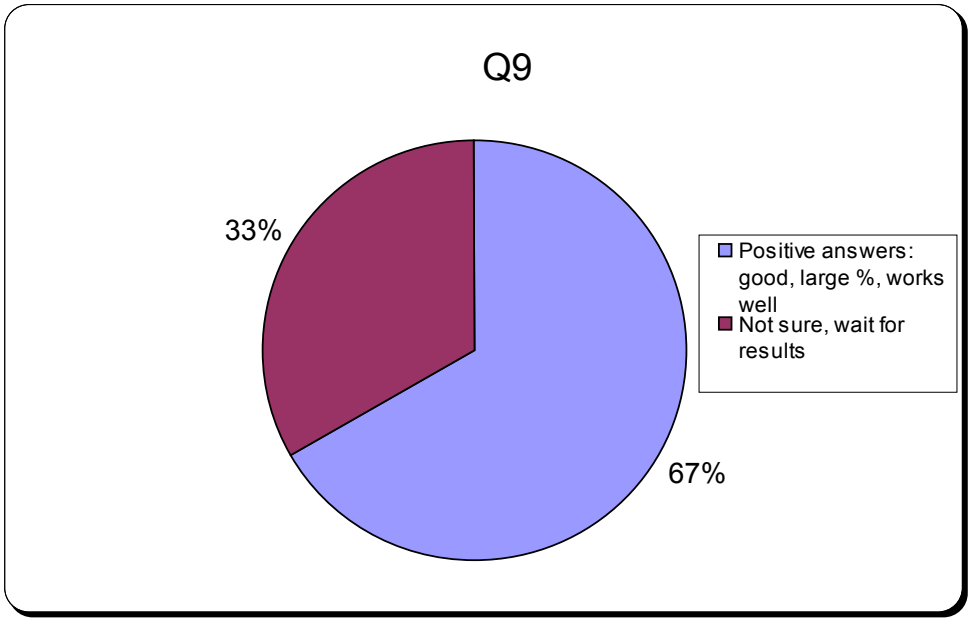


Question 9: How well do you think they work, in terms of urine separation efficiency?

Two thirds of the male and female participants answered that a large volume (high percentage) of urine was successfully separated. The remaining one third said they were not sure and would have to wait and see the laboratory results.

Perceptions of urine separation efficiency

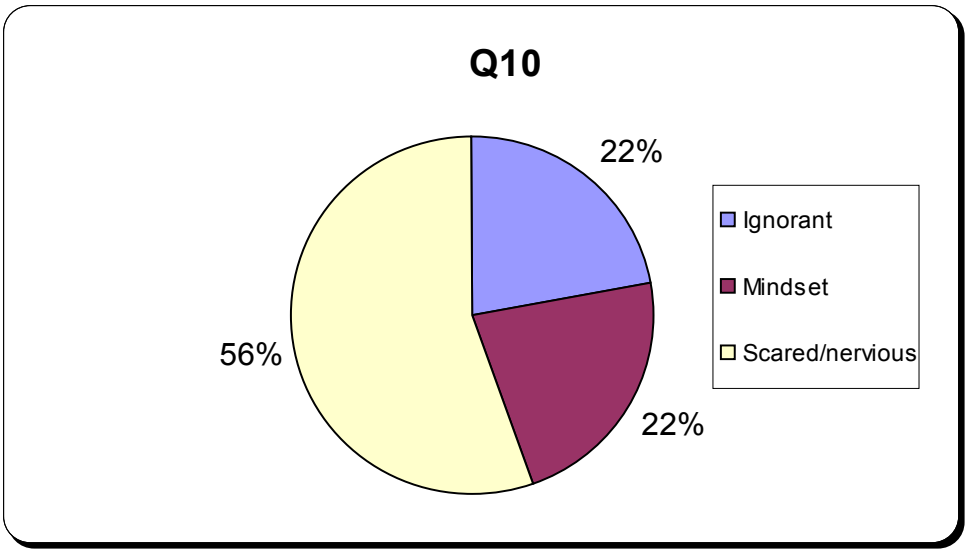
Ladies' responses	Men's responses
Very well	Reasonably well
Good	Pretty well
90%	Seems to work well
Capture large percentage	75%
They work but there is still a transfer (of urine)	Difficult to say
Not sure, will wait to see results	I'm not sure



Question 10: Do you have any ideas why people would feel reluctant to use these no-mix toilets if installed in a public building?

56% of the respondents said that they might be scared or have initial fear, or be nervous and even confused at first, by the new no-mix toilet design. Of the respondents, 22% said people would feel ignorant and therefore reluctant to make use of the no-mix toilet because they are uninformed. And other 22% said people have fixed mind-sets, and they don't want to change their toilet behaviour.

Most of these problems are addressed by answers to question 13 below.



Question 11: Would you move into an apartment/house with a no-mix toilet already installed in it?

All the men and most of the women answered positively, thus supporting this technology. Typical answers were: "Yes", "sure", and "why not".

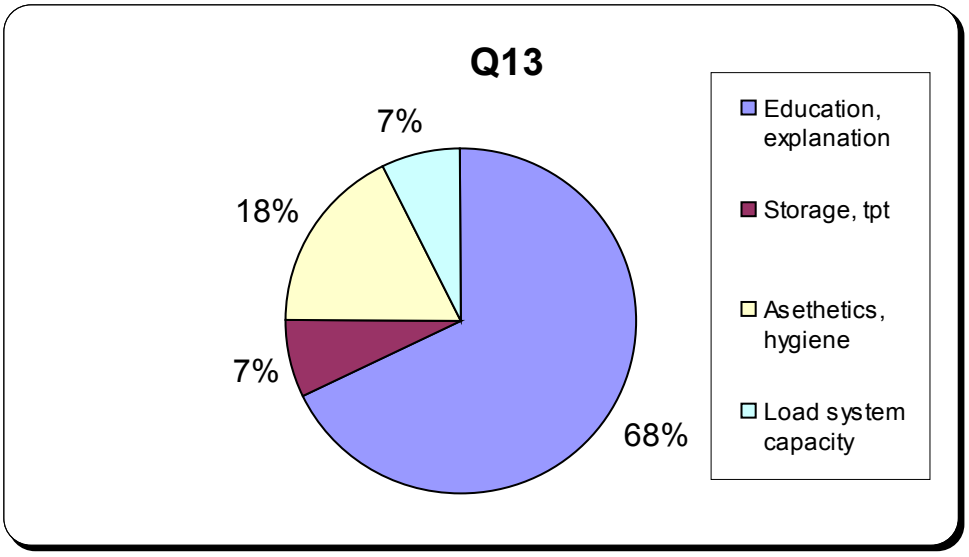
One of the women said that it is a bad idea because toilet paper would get stuck down the urine pipe often. However, there are other no-mix toilet designs commercially available that have overcome this problem, where such designs include urine drain pipes with valves that close when you stand up after using the toilet.

Question 12: Do you think installation of no-mix toilets at places with a high potential "hit rate" such as airports, office blocks, hospitals, schools, etc., would be a good idea?

All the men and women answered positively, saying that it was definitely a good idea and that it had "great potential". From an economical and investment point of view, it makes sense to install these expensive no-mix toilets in buildings with high concentrations of people, all needing to make use of a toilet at some stage.

Question 13: What should be taken into careful consideration from a human perspective before introducing no-mix toilets at such places?

About two thirds of the participants said that education and making information available with regard to no-mix toilets, is essential for the introduction of the system. Users would have to be informed about and coached in order to grasp the logic and reasons that drive urine separation technology. Aesthetics and hygiene were the second biggest concern, followed by storage and transport of the separated urine, and then the engineering design of "the load and system capacity".



Question 14 (for women): If you had to share the toilet with men, would you be worried about them sitting down or not when using the no-mix toilet for urinating only? Is this more of a concern than sharing toilets with men in general?!

Generally, women were not too concerned about sharing a toilet with men, as evident in table below.

Women's perceptions about men using no-mix toilets.

Question 14a **Men sitting down to urinate**	**Number of responses**	**Question 14b** **Sharing with men in general**	**Number of responses**
Not bothered (so long as no mess)	1	Skip	
No, not worried Wouldn't concern me	2	I wouldn't want to share with them in general	1
Yes they mess on the seat	2		
Skip	2		
No I don't like to share with men	1		

From answers to 14a, three women indicated that they don't mind sharing with men, two answered that men mess urine on the seat, and one woman said she does not like sharing with men. For question 14b, one woman said that "I wouldn't want to share with them in general".

Question 14 (for men): Did you have any problems in sitting down when using the no-mix toilet for urinating only? Do you believe your aim is accurate when using the no-mix toilets to urinate in standing position?

Only four men attempted to answer this question and their answers were generally less descriptive than the women's answers. Three of the men had no problem with having to sit and urinate. This shows that the men are open to changing their toilet behaviour to achieve better urine separation. With regard to standing and urinating, one respondent said his aim was not accurate enough, one said "yes it was", and one said only that "aim is important".

General perceptions for using no-mix toilets.

Question 14a Problems sitting to urinate	**Number of responses**	**Question 14b Problems to aim**	**Number of responses**
No	3	No	1
Yes	1	Yes	1
Skip	3	Skip	4
		Aim is important	1

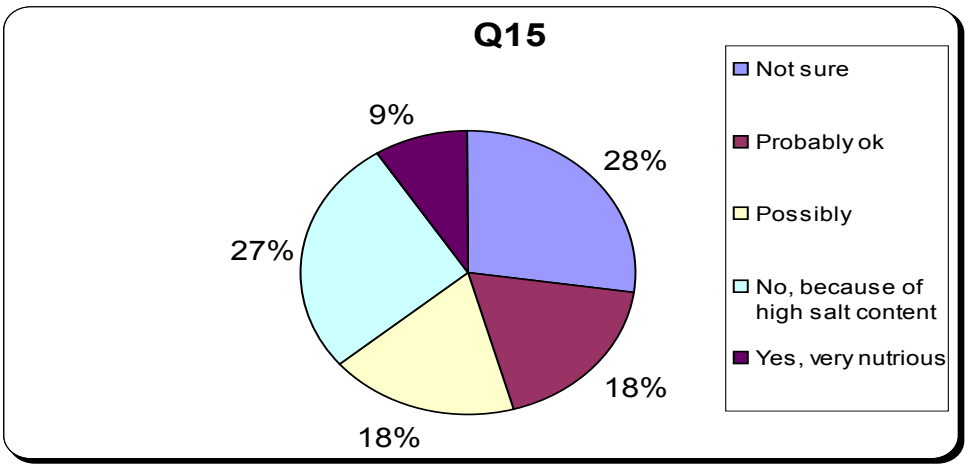
Overall, this question was not well answered. It was possibly an embarrassing question and the participants might have felt awkward in expressing themselves. It was also difficult to interpret if they were answering part a or part b (a questionnaire design error). Those respondents that did answer this question did not write a lot so it was also difficult to gather useful information from these answers.

Question 15: Would it be OK to use source separated urine as an alternative fertilizer? What concerns are there to be addressed (when would it be safe?)

There was a variety of answers to this question. For example:

1. "Not sure, would need to do a risk assessment for EDCs and pharmaceutical (by) products".
2. "Probably would be ok, provided all health risks are eliminated".
3. Possibly, after "addressing the presence of pathogens and pharmaceuticals and breakdown products". Also a concern was the presence of hormones and antibiotics.
4. No, because of the high salt content.
5. Yes, because it is very nutritious.

These answers illustrate that there was no consensus, which is very typical of a relatively new idea that is in its experimental phase. One would have expected more people to have answer 'no', because of the high salinity content in urine, but only 27% gave this as their answer.

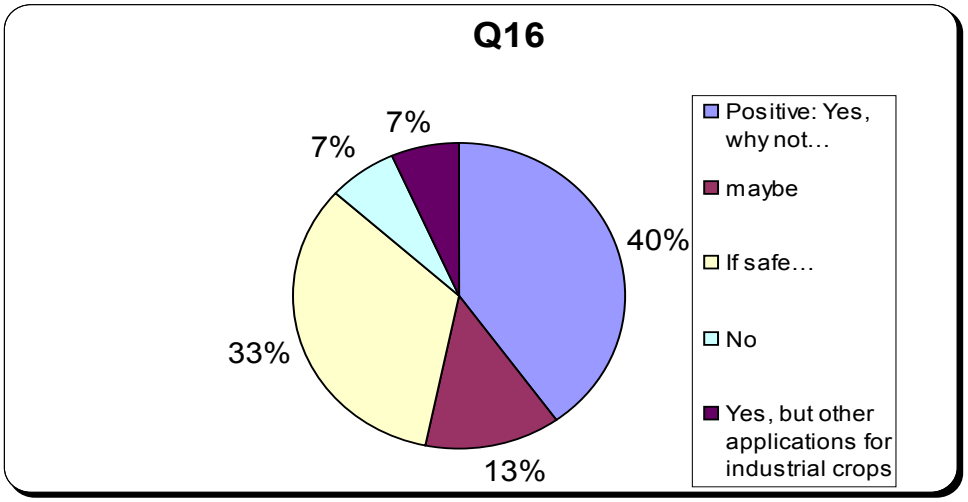


Question 16: If the nutrients in urine were recovered in an engineered process, would you buy food grown from such a waste-based fertilizer?

A total of 40% of respondents answered positively to this question. These answers ranged from saying:

1. "Yes, why not?"
2. " We are already consuming such foods grown from wastewater sludge"
3. "A lot of crops grown locally (e.g. in Kraaifontein) are already using secondary treated effluent for irrigation."

Then, 33% of respondents said yes, if it was proven a safe technology and 13% answered maybe. Lastly, one person expressed concern for human health and he was not sure it was a good idea for food crops but rather for use in industrial crops, ornamentals etc. These results show that most participants were in favour of a waste-based fertilizer, and when proven a safe technology, the majority of the participants would support this technology.



Complaints

One written complaint was received on the 30 June 2009.

"Stop closing off all the toilets please and forcing one to sh%$t in a glorified test tube … I feel like my privacy is being invaded, to say the least. Thank you"

The reason why the toilet doors to the old conventional toilets were closed and only the no-mix toilets made available, was to determine the nutrient and volume load per person in one 24 hour period (Refer chapter 4). For this to be achieved, we needed to ensure that the only toilet uses would be with the no-mix system. The office block was separated from the main building by a large hydraulic laboratory, which meant that one could reasonably argue that the large majority of toilet uses would be captured. Still, whoever refused to use the facility could still walk the distance to the main office block.

One oral complaint was made on 23 November 2009, when a bad smell was noticed in the bathrooms. But this was not from the no-mix toilets but from outside, perhaps from the collection area. Nevertheless, the bad smell created a negative impression and the female participants were reluctant to use the no-mix-toilet that week resulting in a decreased volume of wastewater collected.

Feedback from the cleaners

The cleaners were asked if the no-mix toilets were more difficult to clean when compared to the average toilets. Their reply was YES because we had given them specific cleaning instructions for the no-mix toilets (use no cleaning chemicals in the toilets, don't mop the floor near the electronic counters, etc…). But on a practical level, the no-mix toilets were just as simple to clean compared to the average toilet.

A positive finding was that the cleaners said that it was better for them not to use cleaning chemicals which sometimes irritated their eyes and skin (note: cleaning chemicals were not used for the no-mix toilets because they would have interfered with the chemical analysis of the urine and brown water samples).

There was an ordinary toilet and a no-mix toilet in both the ladies' and gents' bathrooms. When asked in general, which toilet people used the most, they said that they did not know. Despite their answers, the project team suspected that sensitive users avoided the no-mix toilets because they felt uncomfortable knowing that the wastewater that they generated was being tested and used in this study.

APPENDIX B: DETERMINING THE QUANTITIES OF FLUSH WATER FROM THE NO-MIX TOILETS (CHAPTER 4)

Gents

Silver button

No 1 (urine tank)	No 2 (ww tank)
mL	**L**
300	5
280	5
290	5
290	5
300	5
290	5
Average = 291.67	5.0

Gents

Black button

No 1 (urine tank)	No 2 (ww tank)
mL	**L**
180	3.0
170	3.0
140	2.5
280	3.5
160	2.5
230	3.0
Average = 193.33	2.92

Ladies

Silver button

No 1 (urine tank)	No 2 (ww tank)
mL	**L**
225	3.0
345	3.8
335	4.5
325	4.3
320	4.3
Average = 310	3.98

Ladies

Black button

No 1 (urine tank)	No 2 (ww tank)
mL	**L**
230	3.0
240	3.0
215	3.0
230	3.0
260	3.0
Average = 235	3.0

ADDENDUM C – CHAPTER 7 BATCH MEASUREMENTS

Table 22: Unfiltered WAS feed measured parameters from parent reactor

Batch no	COD (mg/l)	TKN (mg/l)	TP (mg/l)	Mg (mg/l)	K (mg/l)	Ca (mg/l)	TSS (mg/l)	VSS (mg/l)	ISS (mg/l)
1	14399	926	902	236	153	96	14653	11112	3540
2	14413	926	728	270	201	149	19282	12569	3357
3	15479	1015	1984	384	192	135	17421	11064	6357
4	15325	991	1834	356	242	121	19152	13032	6120
5	14929	1145	1592	279	214	148	21275	15466	5809
6	14682	911	1389	279	184	117	17636	12235	5401
7	12252	743	2177	289	226	90	16221	10859	5362
8	14929	813	1592	279	214	147	21275	15467	5808
9	14227	1061	2700	397	205	75	21174	13626	7548
10	14192	1154	2590	361	177	74	22916	14368	8548

Table 23: Filtered WAS feed measured parameters from parent reactor

Batch number	COD (mg/l)	TKN (mg/l)	FSA mg/l	NO_3 Mg/l	TP (mg/l)	OP mg/l	Mg (mg/l)	K (mg/l)	Ca (mg/l)
1	28.68	3.24	2.31	1.90	31.24	23.14	14.68	95.93	15.63
2	83.8	9.92	2.85	5.58	31.9	23.52	10.12	92.43	16.21
3	49.80	12.83	2.80	7.17	43.98	27.68	12.83	99.78	14.64
4	59.6	3.08	2.53	10.53	26.5	23.09	11.95	91.12	14.93
5	74.9	4.87	3.19	6.66	26.9	25.97	12.7	89	17.48
6	76.1	6.2	4.64	8.85	28.9	25.67	11.73	85.93	11.56
7	86	4.45	3.16	8.03	26.2	25.55	11	79.33	13.73
8	74.9	4.63	3.96	6.66	26.9	25.97	12.7	89	17.48
9	64.5	5.7	3.26	10	15	18.64	12.84	76.7	12.5
10	47.7	4.27	3.92	7.9	52.2	18.24	32.15	95.7	12.17

Table 24: Waste activated sludge characteristics

Batch number	X_{OHO} (mg/l)	X_{ii} (mg/l)	X_{BG} mg/l	f_{cv}	f_n	f_p	f_{ai}
1	2296	5072		1.39	0.10	0.08	0.207
2	2206	6742		1.32	0.09	0.06	0.176
3	2266	5151		1.43	0.09	0.18	0.205
4	3290	5387		1.25	0.09	0.14	0.252
5	2751	8359		1.31	0.08	0.10	0.178
6	1964	5775		1.42	0.09	0.11	0.161
7	1883	4925		1.43	0.09	0.20	0.173
8	2751	8359		1.31	0.06	0.10	0.178
9	1976	6663		1.33	0.08	0.20	0.145
10	2628	7115		1.19	0.08	0.18	0.183

Table 25: Unfiltered Urine feed measured parameters

Batch number	COD (mg/l)	TKN (mg/l)	FSA mg/l	TP (mg/l)	Mg (mg/l)	K (mg/l)	Ca (mg/l)
1	1546	1623	1413	130	550	441	117
2	1260	1347	1140	133	483	413	156
3	1374	1360	1118	130	186	432	180
4	790	944	905	114		327	155
5	1200	1456	1283	145	300	353	149
6	825	1058	896	137	390	353	102
7	738	1075	888	176	110	212	52
8	831	1268	1110	118	101	269	36
9	1087	1096	946	150	76		46
10	326	825	744	136	n.a	n.a	n.a

Table 26: Filtered urine feed measured parameters

Batch number	COD (mg/l)	TKN (mg/l)	Mg (mg/l)	K (mg/l)	Ca (mg/l)
1	1233	1505	75	380	38
2	847	1236	87	356	44
3	860	1225	66	395	34

4	424	970	43	297	31
5	1009	1378	73	317	34
6	693	978	42	327	18
7	470	870	29	357	21
8	702	1171	40	258	29
9	967	1047	41	388	20
10	536	824	n.a	n.a	n.a

Table 27: Mass balance calculations of experimental results over the AAD.

Batch No	FS_{ti} (mg/d)	FS_{te} (mg/d)	FO_{urine} (mg/d)	FO_{Ccalc} (mg/d)	FO_{Cmeas} (mg/d)	% balance
1	9567	6493	674	1272	2163	98
2	9404	6604	549	1223	2163	99
3	10112	7076	599	1256	2163	97
4	9669	7628	344	1823	2163	105
5	9677	6932	523	1524	2163	99
6	9304	6888	360	1088	2163	101
7	7794	6357	322	1043	2163	113
8	9456	6289	362	1524	2163	93
9	9188	6115	474	1095	2163	95
10	8711	6716	142	1456	2163	104

Table 28: Total N balances over all ten batches.

Batch	$FTKN_{in}$ mg/d	$FTKN_{AAD}$ mg/d	FN_{ne} mg/d	FN_{den} mg/d	Balance %
1	1529	433	404	622	95.4
2	1363	429	334	553	96.5
3	1425	436	431	368	86.7
4	1161	469	486	253	104.0
5	1561	451	313	710	94.4
6	1182	410	396	249	89.3
7	1091	367	398	249	92.9
8	1249	385	390	534	104.8
9	1294	381	399	456	95.4
10	1187	454	324	402	99.4

ADDENDUM D

WORKSHOP PROGRAMMES

Invitation to Workshop

"Effects of urine separation and treatment on wastewater effluent quality"

28 July 2009 | **08h30** | **CSIR, Stellenbosch**

9h00 - Presentations and discussion
The composition of colour coded wastewater, Chemistry of separately collected and stored urine, Nitrification and denitrification of undiluted urine, Treatment of grey water from an office kitchen.

11h00 - Site and laboratory visit
No-mix toilets, collection pipework and storage tanks, bioreactors

11h30 - Presentations and discussion
Effects of urine separation on the effluent quality of BNR processes, The future of source control and side stream wastewater treatment, Infrastructure and management options for source control.

13h00 - Lunch

14h00 - WRC K5/1824 Reference Group and Project Team progress meeting

RSVP; 10 July 2009 JWilsenach@csir.co.za +27 (0)21 888 2533

The composition of colour coded wastewater: Investigating the origins of various substances in mixed wastewater

(Jac Wilsenach; CSIR)

How much of the nutrients (N P K) in mixed domestic wastewater originates from urine and how do we determine this? Concentrations of TKN, COD, DOC, K+, Na+, Mg2+, Ca2+, P, SO42-, Cl-, Alkalinity and conductivity were measured in numerous collections of undiluted urine, flushed urine from men's no-mix toilets, flushed urine from ladies no-mix toilets, brown water (i.e. faecal wastewater with traces of urine) and an office kitchen's effluent. These concentrations are translated into per capita loads. By comparison with well known mixed wastewater compositions, urine is the dominant source of nutrients and salts, while brown water is very rich in organic matter with some nutrients and very little salt. Kitchen effluent is still rich in organic matter, but contains very little nutrients.

Composition and chemistry of stored urine

(Charney Anderson; CSIR/CPUT)

Urine is a sterile liquid inside the human body, but a vile waste in collection systems. What happens to the urine, and what is the fate of the chemicals in urine?. We traced the hydrolyses of urea in separately collected and stored urine. This was done for the newly installed and the "contaminated" system. The changes in chemistry of first flush urine, caught in a separate vessel, over time not only reveals some of the chemistry changes, but pose questions for the analytical methods of this complex liquid. It is clear that salt concentrations in urine is more consistent than organic components that vary significantly from day to day.

Treatment of grey water from an office kitchen in a fixed bed batch reactor

(Jason Germanis; CSIR)

Grey water from an office kitchen has more than twice the COD concentration of conventional raw wastewater, but much less nitrogen and phosphate. A fixed bed bio reactor was filled and aerated in batch mode to remove the COD. Hollow plastic packing material provides a large surface area for biofilm accumulation. Redox and pH was measured online and effluent samples were taken to determine COD removal efficiency.

Nitrification and denitrification of undiluted urine in a sequencing batch reactor

(Morgan McMillan; CSIR/SUN)

Undiluted urine is filled into a 20 l sequencing batch reactor, followed by an anoxic phase for denitrification to remove COD in urine, remove some nitrogen gas and restore alkalinity. The anoxic phase is followed by an aerobic phase with air up flow. A recirculation pump provides additional mixing. Nitrification in traced by a clear pH profile from pH > 8.5 after denitrification, to pH < 7 where the nitrification rate slows down. The goal is an on-site reactor that discharges a liquid with high nitrate concentration into existing sewers, to achieve biological nitrate instead of sulphate reduction.

Effects of urine separation on the effluent quality of a biological nutrient removal process

(Andre Mbaya; UCT)

The *sustainability* of wastewater treatment related to the effluent quality and overall resource utilisation. *Biological Nutrient Removal Activated Sludge* processes can achieve good effluent concentrations for nitrogen & phosphate, and possibly recover phosphate. However, this works better in theory than in practice, and separate collection and treatment of urine could greatly improve design and operation of wastewater processes. The experimental set-up consist a UCT process (anaerobic-anoxic-aerobic) operated with mixed municipal wastewater as well as wastewater from CSIR brown and grey water, with different volumes of urine, or without urine at all.

Future of source control in domestic wastewater treatment

(George Ekama; UCT)

In industry it is not strange to treat separate waste streams separately. The good process engineer considers the composition and volume of each stream to achieve the optimal in treatment efficiency. The end-of-pipe approach of municipal engineers have served us well for many years, but in moving towards better effluent quality and increased greater resource recovery (water, nutrients, biogas), source separation and side stream treatment must be part of the future municipal wastewater treatment strategy. Some new ideas are presented.

Logistics and management of source control on building and city level

(Kobus du Plessis; SUN)

Source control is a wonderful philosophy, but what does it hold for developers, planners, municipal managers, operators, maintenance teams, building owners and occupants? If operation and maintenance of existing municipal services is a major problem, should we not reduce rather than add to the complexity of tasks and duties screaming for attention? Conversely, hidden costs and adverse effects of existing systems – even when they are well maintained - are often not accounted for. The benefits of source control may outweigh the added complexity, if evaluated in a proper framework. A suggestion for such a framework is made along the theme of the waste discharge charge system.

15th February 2011, Paradise Valley, Durban.

TIME	PRESENTATION	SPEAKER
09:00	Registration **TEA/COFFEE**	
THE COMPOSITION OF DOMESTIC WASTEWATERS AND CHARATERISATION OF URINE		
10:00	Introduction: The broader picture of water, wastewater and the loss of nutrients.	Jac Wilsenach Virtual Consul
11:00	The composition of colour coded wastewater: Investigating the origins of various substances in mixed wastewater.	Jason Germanis, CSIR
12:00	Investigation of urea hydrolysis in source-separated urine.	Mlawule Mashego, CSIR
12:30	Envirosan: Sanitation Product Options	Jacques Rust
13:00	**LUNCH**	
THE SEPARATE TREATMENT OF URINE FROM MODERN NO-MIX TOILETS		
14:00	Nitrification and denitrification of undiluted urine in a sequencing batch reactor.	Morgan McMillan, SUN
15:00	Urine treatment in anoxic aerobic digestion of high P content BNR system waste activated sludge.	Koali Motlomelo, UCT
15:45	END	
16:30	**Drinks & snacks @ Buckley's residence:** **49 Essex Grove, Westridge, Durban.**	

16th February 2011, Paradise Valley, Durban.

TIME	PRESENTATION	SPEAKER
08:00	**TEA/COFFEE**	
	NEW AND ALTERNATIVE WAYS OF TREATING WASTEWATER	
08:30	Effects of urine separation on the effluent quality of a biological nutrient removal process.	Andre Mbaya, UCT
09:20	Urine separation and sea water toilet flushing for a more sustainable urban water management.	George Ekama, UCT
10:15	**TEA/COFFEE**	
	COLLABORATIVE RESEACH AND VARIOUS WASTEWATER TREATMENT OPTIONS	
10:45	Process options for different wastewater treatment scenarios.	Jac Wilsenach Virtual Consulting
11:35	Overview of the Gates Urine Project.	Chris Buckley, UKZN
12:15	*Interactive Session: Open forum, discussions & brainstorming*	Chair: Jac Wilsenach
13:30	**LUNCH**	

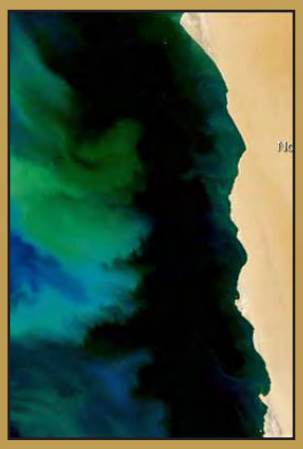
Commercial agriculture relies on nitrogen, phosphate and potassium, applied to vast areas of land as industrial fertilizer. This linear flux of nutrients is not sustainable, because it depends on finite mineral and energy resources.

Due to an excess of nutrients from wastewater treatment works, eutrophication disturbs aquatic eco-systems in fresh water and in the oceans, and leas to overall deterioration of water quality.

Around 75% of the total mass of nitrogen and potassium in municipal wastewater, and almost 50% of the total mass of phosphate, originates from urine, which is only a small fraction (less than 1%) of the volume.

This pilot project demonstrated that separate collection of urine can be practical. It was shown that urinals can be retrofitted to operate without flush water, while modern no-mix toilets give users the same comfort as normal water-borne toilets.

Processes for the treatment of separately collected urine were investigated and are described in this report. Full scale separation of urine could ultimately double the treatment capacity of an existing activated sludge reactor.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