IMPACT OF URINE DIVERSION ON DOWNSTREAM WASTEWATER TREATMENT PLANTS

by Umakant Badeti

Thesis submitted in fulfilment of the requirements for the degree of

Doctor of Philosophy

under the supervision of Dr Sherub Phuntsho and Prof Hokyong Shon

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CERTIFICATE OF ORIGINAL AUTHORSHIP

I, **Umakant Badeti**, declare that this thesis is submitted in fulfilment of the requirements for the award of **Doctor of Philosophy, in the School of Civil and Environmental Engineering/ Faculty of Engineering and Information Technology** at the University of Technology Sydney.

This thesis is wholly my own work unless otherwise referenced or acknowledged. In addition, I certify that all information sources and literature used are indicated in the thesis.

This document has not been submitted for qualifications at any other academic institution.

This research is supported by the Australian Government Research Training Program.

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Umakant Badeti

Date: 27/10/2023

I dedicate this thesis to my parents

Vijayasree and ***Anjanee Prasad***

and my sister

Ratna Manjusha

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Writing this doctoral thesis, and particularly the endeavour of gathering and analyzing data for its completion, especially during pandemic has been a paramount academic challenge. Yet, it stands as the most rewarding experience in my lifetime. This journey would not have been possible without the love, unwavering support, patience, and guidance of some remarkable individuals. First and foremost, I extend my heartfelt appreciation to my parents, Vijayasree and Anjanee Prasad, whose sole dedication was to provide my sister and me quality education. To my parents, I dedicate this thesis in recognition of all the sacrifices they made in nurturing and fostering both my sister and me. I also dedicate this thesis to my dear sister, Ratna Manjusha, whose enduring love and encouragement have been a constant source of inspiration. I am deeply grateful for the professional sacrifices she made in supporting our parents while I was committed to my PhD away from home in Australia. In addition, I extend this dedication to my dearest friend Haritha, for her unwavering support and care throughout my PhD and stay in Sydney.

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Part of the work performed during the PhD project resulted in peer-reviewed publications and presentations, which are listed hereafter in the order of acceptance and by category. They are directly or indirectly related to the main topics of this Thesis, and all of them have been accepted and published by the end of that project.

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2. Badeti, U., Pathak, N. K., Volpin, F., Dorji, U., Freguia, S., Shon, H. K., & Phuntsho, S. (2021). Impact of source-separation of urine on effluent quality, energy consumption and greenhouse gas emissions of a decentralized wastewater treatment plant. Process Safety and Environmental Protection, 150, 298-304. (**Impact Factor 8.9, Q1)**
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3. *Umakant Badeti et al*., 2020. Impact of source-separation of urine on effluent quality, energy consumption and greenhouse gas emissions of a decentralized WWTP. Oral presentation at International Conference on the "Challenges in Environmental Science and Engineering" (CESE-2021).
4. *Umakant Badeti et al*., 2021. Impact of urine diversion and conversion to fertiliser products in Sydney Central Park. Oral presentation at Rich Earth Summit, 2021, 1-3 November 2022.
5. *Umakant Badeti et al*., 2022. Process Optimisation of a full-scale MBR converting urine to fertiliser products. Poster presentation at IMSTEC, 2022, 4-8 December 2022.

LIST OF ABBREVIATIONS

AL-DS	**A**ctive **L**ayer facing **D**raw **S**olution
AL-FS	**A**ctive **L**ayer facing **F**eed **S**olution
AEMs	**A**nion **E**xchange **M**embranes
CAPEX	**C**apital **C**osts
COD	**C**hemical **O**xygen **D**emand
CEMs	**C**ation **E**xchange **M**embranes
DCMD	**D**irect **C**ontact **M**embrane **D**istillation
DI	**D**eionised water
DOC	**D**issolved **O**rganic **C**arbon
DS	**D**raw **S**olution
ED	**E**lectrodialysis
EDX	**E**nergy **D**ispersive **X**-ray spectroscopy
EPS	**E**xtracellular **P**olymeric **S**ubstances
ES	**E**lectrolyte **S**olution
FO	**F**orward **O**smosis
FDFO	**F**ertilizer **D**riven **F**orward **O**smosis
FS	**F**eed **S**olution
FU	**F**resh **U**rine (Nitrogen as urea)
HC	**H**igh **C**oncentration
HU	**H**ydrolysed **U**rine (Nitrogen as ammonia
HRT	**H**ydraulic **R**etention **T**ime
IEMs	**I**on **E**xchange **M**embranes
ISS	**I**nternational **S**pace **S**tation
LC	**L**ow **C**oncentration
LC-OCD	**L**iquid **C**hromatography-**O**rganic **C**arbon **D**etection
LMW	**L**ow **M**olecular **W**eight
MBR	**M**embrane **B**ioreactor
MD	**M**embrane **D**istillation
MLSS	**M**ixed **L**iquor **S**uspended **S**olids
NOB	**N**itrite **O**xidizing **B**acteria
NF	**N**ano **F**iltration
OCV	**O**pen **C**ircuit **V**oltage
OPEX	**O**perational **C**osts
PD	**P**ower **D**ensity
RED	**R**everse **E**lectrodialysis
RSF	**R**everse **S**alt **F**lux
RO	**R**everse **O**smosis
SEM	**S**canning **E**lectron **M**icroscopy
SRSF	**S**pecific **R**everse **S**alt **F**lux
TDS	**T**otal **D**issolved **S**olids
TFC	**T**hin-**F**ilm **C**omposite
UF	**U**ltra**F**iltration
UV	**U**ltra **V**iolet
WWTP	**W**astewater **T**reatment **P**lant

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ABSTRACT

Source separation and collection of urine has been proposed as the most suitable approach of recovering of nutrients from wastewater and reusing as renewable fertilizers which inevitably diverts urine and reduces nutrient loads reaching the wastewater treatment plants (WWTP) and ultimately reducing wastewater treatment costs. In this study the impact of urine diversion on the biological treatment processes at a decentralized wastewater treatment plant was investigated using BioWin software for simulations after validating the data from a real WWTP located at the Sydney Central Park building. The simulation study shows that up to 82% N and 30% P load to the WWTP can be reduced by completely diverting urine from the wastewater through source separation and collection. Under the standard WWTP operating conditions, the simulations suggest that 90% urine diversion can help reduce 33% of aeration energy and 25% CO_2 emissions. The impact on the treatment capacity, treatment process, and capital costs were also investigated. The simulation shows that if 75% of the urine is diverted, the treatment capacity of the existing WWTP could be doubled. In fact when the urine diversion exceeds 75%, it was found that the anoxic tank for biological denitrification becomes redundant and the current wastewater treatment process (activated sludge process) could be replaced with a much simpler treatment process with 24% reduction in the capital expenditure (footprint) cost while producing similar effluent quality. Sensitivity analysis suggests that by operating the bioreactor at higher mixed liquor suspended solids concentrations, it could help increase the treatment capacity of the existing plant by about 3.5 times at 75% urine diversion. The model was further applied to investigate the performance of a full-scale urine treatment system by using membrane bioreactor. The simulations showed that at optimised DO set point of 3 mg/L, the energy consumption was 3 kWh/kgN at a volumetric nitrification rate of 310 mgN/L/d which is only about 25-30% of the total energy consumed in the synthesis of a virgin chemical fertilisers. Finally,

a comprehensive techno-economic evaluation was conducted to compare the effects of urine diversion and fertiliser processing systems using biological nitrification and road transportation to agricultural farms at different distances from the decentralized system. If urine is diverted, it can help reduce 10-50% of the cumulative energy demand, 10-50% in the operating costs while the capital costs ranged 10% decrease to 15% increase based on the nitrification rates that can be achieved for urine treatment.

CHAPTER 1: INTRODUCTION

1.1. Research background

Moving from a linear economic model of "take-make-dispose" to a circular one of "take-make-revalorize-reuse" is gaining popularity as an emerging sustainable policy goal due to growing resource prices and climate change (Gregson et al. 2015). This also entails spending money on adaptable, decentralised sewage treatment systems that would enable us to collect and reuse the water, energy, and nutrients in our wastes. The complexity and expense of recovering value farther downstream are, however, significantly increased by the heterogeneous and "dilutive" nature of ordinary wastewater streams. To achieve more robust and energy-efficient sanitation, source-separation of urine, faeces, and greywater sources has been suggested (Figure 1-1) (Larsen & Gujer 1996; Larsen & Gujer 1997). This would allow for the possibility of creating "recycled" fertilisers helping to meet the growing need for nitrogen, phosphorus, and potassium (N-P-K) fertilisers, protecting our cities' capacity to handle an increasingly dense urban population (FAO 2017a).

The dense nutrient content of urine has the undesirable side effect of greatly raising the cost and complexity of treatment when paired with combined wastewater. Additionally, the downstream treatment of residential wastewater is made more challenging by the high content of pharmaceuticals in the urine (i.e., 2-3 orders of magnitudes higher than ordinary sewage water). When urine is handled separately, the cost and complexity of the treatment are both greatly decreased because of its low volume and generally stable composition (Larsen et al. 2009).

Figure 1-4 presents a conceptual model in support of the claim that source-separation and reuse of urine can promote a circular nutrient economy while enhancing the efficiency of our wastewater treatment system. Particularly in the context of urine diversion, this thesis will focus on understanding and quantifying the impact of urine diversion (for conversion to fertiliser products) on an advanced and decentralised wastewater treatment plants. The downstream impacts to the wastewater treatment plants include the wastewater composition, capital cost and operating costs in particular energy consumption, economics, etc.

In Australia, sustainable building practices are gaining prominence, with a focus on features that enhance the sustainability ratings of buildings. One key aspect of these sustainable practices is wastewater reclamation and reuse. It's clear that incorporating urine diversion into decentralized wastewater treatment plants (WWTPs) can significantly reduce nutrient loads to water reclamation facilities. This thesis seeks to explore the implications of implementing urine diversion at a decentralized level within a catchment using a treatment process configuration similar to that of the Sydney Central Park WWTP, which employs advanced configurations and membrane processes for water reuse. To date, there has been no investigation into the specific impacts of urine diversion in such a context. Furthermore, no economic analysis has been conducted to establish the groundwork for a business model for urine diversion in a decentralized wastewater system. Notably, this study marks the first instance in which the effects of urine diversion on a wastewater treatment process have been investigated, utilizing BioWin v.6.2 for this purpose. The case study Sydney Central Park has been detailed in section 1.2 below.

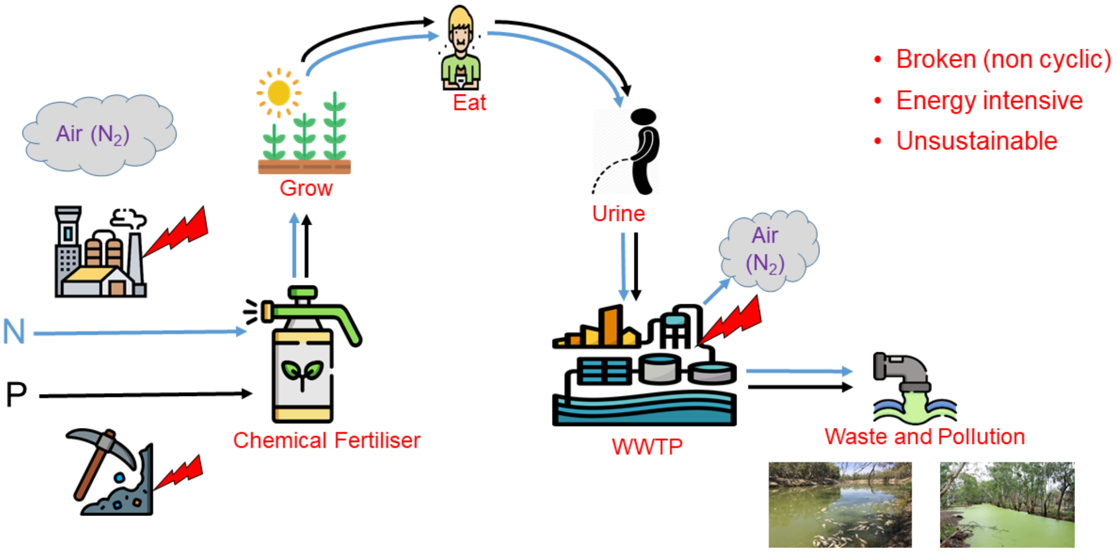


Figure 1-1(a). Conventional nutrient cycle

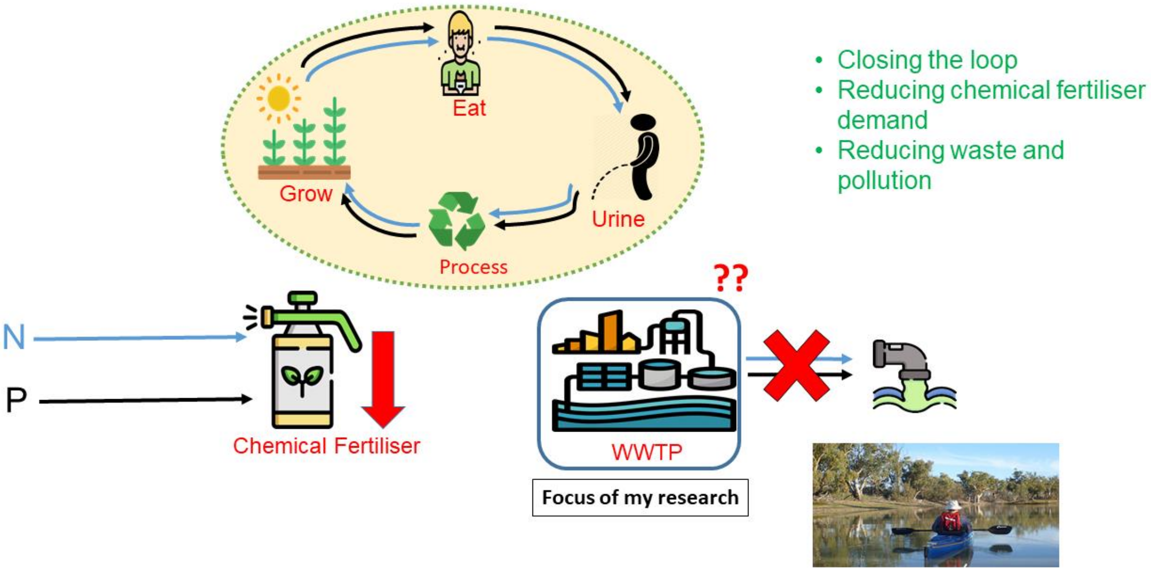


Figure 1-1(b). Nutrient cycle after urine diversion

1.2. Case study: Sydney Central Park WWTP

1.2.1. Central Park precinct

The \$2 billion Central Park development situated on the southern border of the CBD is a primarily residential complex built on the former Carlton and United Breweries site in Chippendale. It is home to over 50 active retail spaces, with more than half of the area allocated to food establishments that vary in size, ranging from a large supermarket to smaller outlets and restaurants. The precinct is equipped with a newly constructed Palace Cinema on Level 3, and food supplied from within One Central Park. The project's unique architectural elements and design are founded on social and environmental initiatives, with the landscaped area measuring approximately 64,000 m^2, including the vertical gardens. Central Park is among Australia’s greenest urban developments, serving the needs of 5,000 residents and 15, 000 daily workers and visitors. Central Park has earned several 5 Star Green Star - Multi Unit Residential v1 Design Ratings and a 5 Star Green Star - Retail Centre v1 Design Rating, alongside their As Built ratings. The project stands out from others due to its numerous sustainability initiatives and accomplishments, including:

- Central Park's sustainability efforts include an on-site trigeneration plant that provides chilled and hot water, as well as green energy to all of the buildings, leading to an 80% decrease in carbon emissions in contrast to electricity.
- Central Park has installed a recycled water treatment plant, also known as a blackwater system, to decrease the demand from the water supply main. To manage wastewater, Central Park has assigned Flow Systems Pty Ltd (now Altogether Group), which is a sustainable multi-utility company that delivers drinking water, recycled water, wastewater services, embedded energy networks, and hot water systems to 13 communities throughout Sydney.

Central Park collects water from several sources on-site, including rainwater from roofs, stormwater from impermeable surfaces and planter box drainage, groundwater from basement drainage systems, sewage from an adjacent public sewer (sewer mining), sewage from all buildings within the Central Park community, backwash from pools and spas in the development, and irrigation water from all green walls.

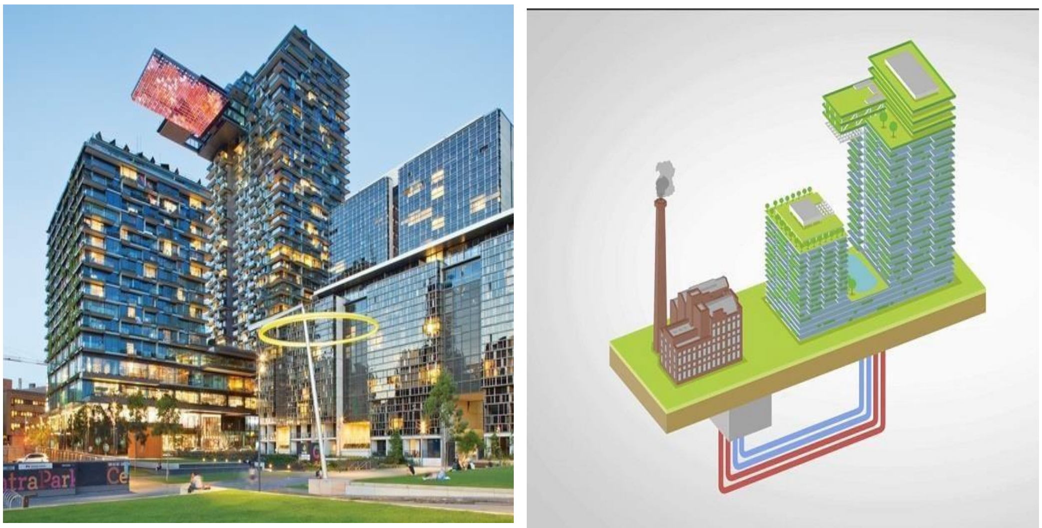


Figure 1-2 (a) Figure 1-2 (b)

Figure 1-2(a). Central Park WWTP mall (b) Central Park wastewater pipeline system

Source (https://watersensitivecities.org.au/solutions/case-studies/central-park2/)

1.2.2. Wastewater management in Central Park

The on-site water recycling plant, resulting in the discharge of approximately 62,050 t/annum (1,190,000 kg/week) to the sewer, does not treat around 25% of the total wastewater generated on Central Park’s site. The Central Park precinct's development was completed at the end of 2018, and the water recycling plant is capable of providing up to 1 ML/day of recycled water to the precinct. Recycled water is exported to the new campus building at UTS, located opposite Central Park, for end-uses like toilet flushing and cooling towers since 2019.

There is significant volume of trade waste generated on-site from the treatment plant. Trade waste is discharged to sewer under a trade waste agreement with Sydney Water. According to estimates by Flow Systems, around 12 kL/day of sludge is discharged to the sewer, which amounts to approximately 4,380 t/annum (84,000 kg/week). At full capacity, this volume is expected to increase. By 2019, when the Central Park precinct development and UTS building are fully operational, the estimated capacity will be approximately 0.7 ML/day, resulting in an estimated 15 kL/day of associated sludge. As a result, it is assumed that the trade waste sludge discharged to the sewer by the end of 2019 will be around 5,475 t/annum (105,000 kg/week). Recycled water is used for all toilet flushing, irrigation, all washing machines and mechanical plant. Thus, Central Park WWTP uses world's best practice technologies to purify recycled water that meets National Standards, is safe to use, crystal clear and odourless.

1.2.3. Treatment processes

Raw sewage first flows to the flow balance tank from where it first passes through fine screens before reaching the biological treatment processes. The bioreactors consist of an anoxic tank, an aerobic tank, and a membrane bioreactor (MBR) tank. Ammonification and denitrification occur within the anoxic tank. In the anoxic tank a low dissolved oxygen level and suspension of the mixed liquor is maintained using a low speed submerged mixer. Nitrification process occurs in the aerobic zone where the ammonia/ammonium is first oxidized into nitrite by the ammonia-oxidizing bacteria and then nitrite into nitrate by the nitrite-oxidizing bacteria during the nitrification process. The MBR tank contains SUEZ's Zee Weed 500 membrane cassettes. Mixed liquor flows from both the membrane tanks to the de-aeration tank (not shown in Figure 1-1), through an overflow weir before flowing back into the anoxic tank through pipework, where the biological treatment cycle continues followed by Ultraviolet (UV) disinfection and Reverse Osmosis (RO). The plant disposes its sludge wastage and RO brine into the Sydney Water sewer line. Sydney Water charges the WWTP operator for the discharge based on nutrient mass loads disposed of by the treatment plant.

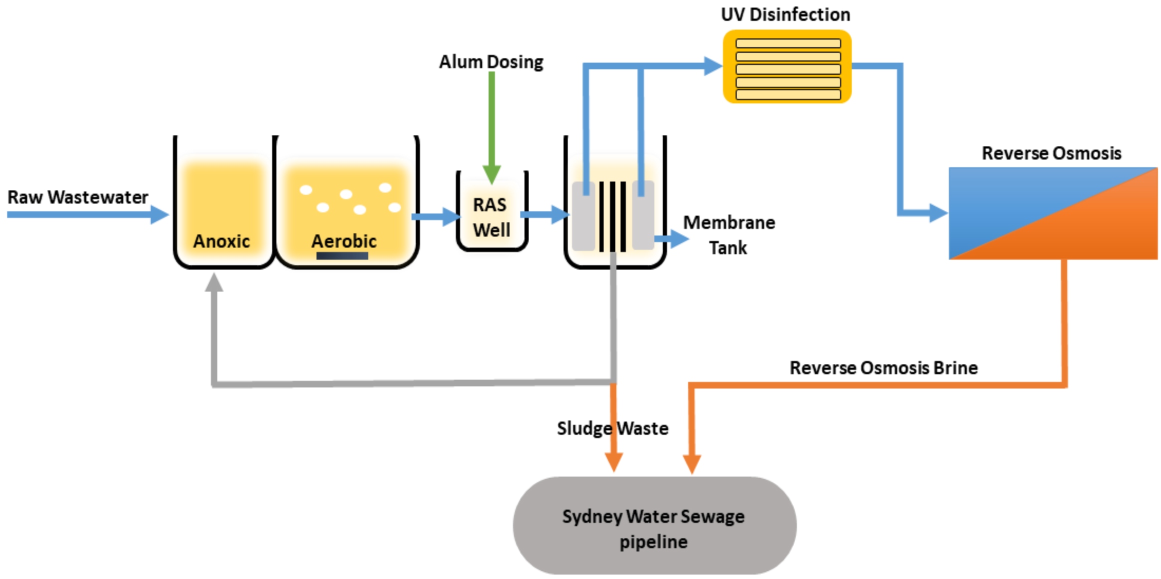


Figure 1-3.Schematic of wastewater treatment processes at Central Park WWTP

1.2.4. Treatment targets and challenges

Recycled water sources in Central Park, such as sewage and greywater, may contain various agents that pose risks to human health, including pathogenic microorganisms and chemicals. Microbial hazards include bacteria, viruses, protozoa, and helminths, while chemical hazards include inorganic and organic chemicals, pesticides, potential endocrine disruptors, pharmaceuticals, and disinfection by-products. While the focus for human health is mainly on microbial hazards, chemicals must also be considered, and long-term exposure to low levels of chemicals raises emerging areas of concern. For the environment, chemical hazards pose a greater risk than microbial hazards, but emerging concerns exist regarding microbial hazards, such as the transmission of antibioticresistant bacteria through waste released into the environment. The primary targets of the plant are to remove the microbial and chemical hazards. Among the various hazards identified, the main purpose of the treatment process that adds complexities to the design of the treatment plant is nitrogen and phosphorus removal. In a biological wastewater treatment processes, nitrification is often the rate-limiting step that determines the space requirements due to its slow rate process and sensitivity to inhibitory factors like temperature, pH and toxic compounds (Henze et al., 2002; Zhou et al., 2020). Inefficient nitrification in activated sludge processes can lead to higher effluent ammonia and total nitrogen concentration due to slow growth rates of nitrifiers. To remove nitrogen from wastewater, denitrification requires additional organic chemicals and significant aeration energy for nitrification, which increases plant footprint and operating expenses (Zhou et al., 2020). Moreover, nitrogen removal requires significant aeration energy for nitrification and high recirculation ratios for denitrification that increases plant footprint. In summary, improving nitrogen removal to the levels that are adequate to decrease eutrophication is a big challenge because of limited land availability, and high capital and operational expenditures. Phosphorus is removed by chemical precipitation; the challenges include high chemical consumption. Chemically precipitated phosphorus is ultimately discharged to Sydney Water sewer line, which is charged at a rate of ~$6.6 per kilogram of phosphorus discharged leading to increase in the operational expenditure.

1.2.5. Relevance of urine diversion in Central Park

In Australia, sustainable buildings are being promoted where wastewater reclamation and reuse is one of the main features of the sustainable star ratings. It is obvious that by implementing urine diversion in such a decentralised WWTP, nutrient loads to water reclamation facilities will be significantly reduced. Reducing nutrient loads could have significant environmental and economic advantages to the treatment plant. Few earlier works have focused on understanding the impact of urine diversion on a centralized WWTP. Hence, in this thesis, we investigated a scenario of a urine diversion on the process performance of a decentralised wastewater treatment plant (WWTP) at the Sydney Central Park multi-complex building water reclamation plant (which involves advanced configurations and membrane processes).

1.3 Objectives and scope of the research aims

This thesis was aimed at understanding and quantifying the impact of urine diversion (upstream) on the wastewater treatment plants (downstream) which could potentially lead towards a more sustainable wastewater management and efficient nutrient recovery in decentralised settings like Central Park. The specific objectives are graphically shown (Figure 1-4) in different segments and are summarised below.

1. The effects of urine diversion on energy consumption, greenhouse gas emissions, effluent quality and operating costs of a typical decentralised wastewater WWTP were quantified.
2. The effects of urine diversion on treatment capacity, space requirement and process configuration and investigate the fate of nitrogen in a decentralised WWTP were quantified.
3. The energy demand and GHG emissions and space requirements for the conversion of source separated urine to fertiliser products via nitrification in a full-scale WWTP were estimated.
4. Full life cycle and techno economic analysis and compare the operations phase of source separation of urine and processing methods with the conventional wastewater recycling method in Central Park WWTP were performed.

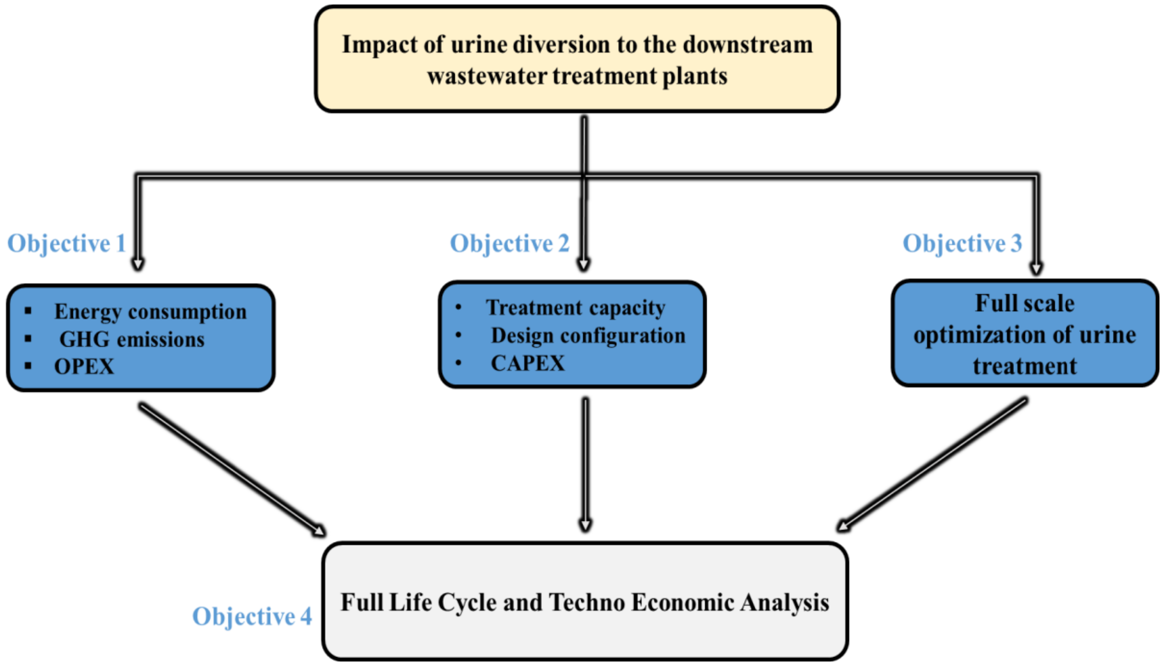


Figure 1-4. Thesis structure on the impact of urine diversion to the downstream wastewater treatment plants

1.4 Thesis Outline

This thesis is organised into seven main chapters as follows:

- Chapter 1 introduces the research background, objectives and scope of the research.
- Chapter 2 provides background information by way of a literature review to highlight the research motivation and gaps in the literature.
- Chapter 3 mainly focuses on the impact of urine diversion on energy consumption, effluent quality and operating costs of Central Park WWTP. The results of this study are published in Process Safety and Environmental Protection.
- Chapter 4 investigates the impact of urine diversion on treatment capacity, capital costs and process design. The results of this study are published in Chemosphere.
- Chapter 5 focused on modelling and investigating the process performance of a full- scale urine nitrification system. The results of this study will be submitted in Process Safety and Environmental Protection.
- Chapter 6 performs a full life cycle and techno economic analysis of urine diversion and conversion to fertiliser products. The results of this study will be submitted in Water Research.

Finally, Chapter 7 concludes the Thesis and provides recommendations for future research.

CHAPTER 2: LITERATURE REVIEW

2.1. Introduction

This chapter begins with a detailed description of the composition of urine (mainly nutrient content, trace organics and changes in composition after hydrolysis) in section 2.2. Following that, it reviews the literature on the following topics: Topic 1: Nutrients in wastewater and challenges. Here the nutrient content in wastewater and the importance and challenges associated with removing nutrients are discussed (section 2.3). Topic 2: Treatment processes for source separated urine Biological, Chemical and Electrochemical processes for stabilisation of urine (section 2.4) Topic 3: Current and past implementation of source separated urine. Here examples of urine recovery and reuse all around the world are presented and analysed (section 2.5). Finally, section 2.6 concludes the review of the literature and highlight its implications to develop a conceptual framework for the study.

2.2. Composition of human urine

Urine is a bodily fluid with a complex and diverse composition, containing over 3000 metabolites identified over the past 30 years. This richness and variability make urine an important source of biomarkers for the diagnosis of various diseases. Urine composition can be influenced by factors such as gender, age, race, diet, medication intake, and physical activity. Additionally, the composition of urine can vary throughout the day in an individual due to the natural metabolic processes required for normal bodily functions. A properly functioning renal system retains proteins like serum albumin (MW: 67,000 Da) while allowing the passage of substances like urea, creatinine, and uric acid (Yang et al., 2007). As a result, urine mainly consists of electrolytes, nitrogenous compounds, vitamins, hormones, and organic molecules (both acids and non-acids). These constituents can accumulate up to a concentration of 35,000 mg/L in typical urine (Putnam, 1971). An individual typically excretes about 1270 g of urine per day. Hence, a significant amount of solids can be recovered from each person (Randall & Naidoo, 2018).

2.2.1 Nutrients in urine

Among the various constituents, nitrogen makes up 14-18% of the solids in dry urine, carbon 13%, phosphorus and potassium about 3.7% (Strauss, 1986). In terms of concentration, urea (or NH_3 after ureolysis) is the predominant compound, typically averaging 6-8 g/L, while K^+ and Na^+ follow with concentrations ranging between 1-2 g/L (Putnam, 1971; Kirchmann & Pettersson, 1994). While urea and ammonia make up the majority of the nitrogen excreted, typically accounting for 75-90%, the remainder includes creatinine (which can reach up to 2 g/L), as well as amino acids and uric acids. Both ppb and ppm quantities of vital trace elements, including B, Cu, Zn, Mo, Fe, Co, and Mn, have been found in human urine (Table 2-1). (McBride & Spiers 2001). Contrarily, urine's heavy metal concentrations are minimal, particularly when measured against that of commercial fertilisers (Jönsson et al., 2005). For instance, cadmium levels in phosphate rock can be several orders of magnitude higher than those in urine (Putnam, 1971; Maurer et al., 2006). Low concentrations of heavy metals but the presence of essential trace elements are positive findings to support the usage of raw material for fertiliser production.

2.2.2 Trace organics (TrOCs) in urine

An average of 64% of the TrOCs (pharmaceuticals and emerging organic contaminants) mass eaten is eliminated through human urine, according to a meta-analysis of 212 medicines conducted by Lienert et al. (2007). This is because they can pass through our kidneys as they are often hydrophilic. According to estimates (Kümmerer, 2009), between 50 and 90% of the antibiotics taken by humans are eliminated via urine as a mixture of metabolites and parent chemicals. For instance, Lienert, Güdel, and Escher's (2007) investigation of the toxicity of 42 drugs in human urine revealed that roughly 67% of them retained at least half of their toxicity. Ibuprofen, acetylsalicylic acid, bezafibrate, carbamazepine, diclofenac, fenofibrate, and paracetamol were discovered to possibly represent an ecotoxicological risk among the examined medications.

Table 2-2 shows the concentration of pharmaceuticals in the urine collected from the UTS urine-diverting toilets and male urinals. This is hardly surprising considering that nearly 50% of all the medicines and hormones in wastewater are found in urine, despite accounting for only 1% of its total volume (Pronk et al. 2006).

Table 2-1 Human urine's organic and inorganic chemical makeup as reported in the literature. The analysis of the urine source separation system at the University of Technology Sydney (UTS) produced the results from the first two rows. The fundamental distinction between urine-diverting toilets and waterless urinals is that the latter dilute urine by two to four times with tap water.

	EC	**COD**	**Urea**	**NH_3-N**	**Creatinine**	**PO_4^{3-}-P**	**K^+**	**Mg^{2+}**	**Na^+**	**Ca^{2+}**	**B**	**Mn**	**Fe**	**Cu**	**Zn**
	[mS/m]	**[mg/L]**	**[g/L]**	**[mg/L]**							**[µg/L]**				
UTS Waterless Urinal (Hydrolysed urine)[1]	40.5 ± 0.7	6174 ± 621	n.d.	6817 ± 145	-	178 ± 15	1604 ± 112	13 ± 21	1971 ± 129	5 ± 31	883 ± 101	<1	19 ± 8	6.7 ± 1.1	200 ± 34
UTS Urine Diverting Toilet (Hydrolysed urine)[1]	25.9 ± 0.4	4873 ± 728	n.d.	3846 ± 121	-	85 ± 5	1387 ± 97	36 ± 25	1204 ± 83	7 ± 20	1170 ± 50	<1	23 ± 11	18 ± 3	137 ± 31
Fresh Urine[2]	15.5 – 19.6	-	9.3 - 23.3	200 - 730	670 - 2150	470 - 1070	750 - 2610	102 - 205	1170 - 4390	30 - 390	435 - 440	0.062	165 - 205	13 10.8	19 - 665

Table 2-2 Pharmaceutical and hormone concentrations in human urine as reported in the literature. The first two rows of results are from an examination of the urine source separation system at the University of Technology Sydney (UTS). The fundamental distinction between waterless urinals and urine-diverting toilets is that the latter dilute urine by two to four times with tap water

	Caffeine	**Carbamazepine**	**Ibuprofen**	**Naproxen**	**Estrone**	**Estriol**	**Fluoxetine**
UTS Waterless Urinals[1] [µg/L]	1475 ± 35	37 ± 3	497 ± 41	<10	<10	6 ± 4	12 ± 2
UTS Urine Diverting Toilet[1] [µg/L]	2165 ± 145	<5	<10	197 ± 195	<10	9 ± 7	<5
	Diclofenac	**Sulfamethoxazole**	**N4 Acetyl-SMX**	**Trimethoprim**	**Hydrochlorothiazide**	**Atenolol acid**	**Ritonavir**
Concentration Range[2] [µg/L]	3.2 - 72	<2 - 6800	<1 - 3500	<2 - 1300	<3 - 134	<1 - 4.6	<1 - 300
Frequency of occurrence[2]	100%	95%	90%	85%	80%	70%	55%

2.2.3 Urine hydrolysis and change in composition

To select the optimal urine treatment approach, it is crucial to have a thorough understanding of how the chemical and biological composition of urine evolves over time. When urine is stored in non-sterile conditions or exposed to high temperatures (> 40℃), the urea in the urine undergoes hydrolysis, producing NH_3/NH_4^+ and CO_2. This process results in a rise in pH, conductivity, and osmotic pressure (Ronteltap et al., 2010; Randall et al., 2016; Larsen et al., 2021b). The differentiation between urine before and after urea hydrolysation resulted from this. The former is commonly known as "fresh urine," whereas the latter is known as "hydrolysed urine". Two moles of NH_3 and one mole of CO_2 are created during the enzymatic hydrolysis of one mole of urea (Equation 1). The enzymatic hydrolysis of one mole of urea produces two moles of NH_3 and one mole of CO2, as shown in Equation 1. Since osmotic pressure is a colligative property of any solution, the accumulation of dissolved species results in an increase in urine osmotic pressure. If urine is stored in non-sterile conditions, this process is unavoidable.

$$NH_2CONH_2 + H_2O + heat \longrightarrow 2\,NH_3 + CO_2 \qquad \text{(Equation 1)}$$

Ureolysis causes the pH to increase to 9-9.5, leading to the precipitation of Mg^{2+} and Ca^{2+} in the form of carbonates and phosphates (Maurer et al., 2006; Randall et al., 2016; Lambert & Randall, 2019). In addition to the potential clogging and scaling of pipes, the precipitation of calcium phosphate and struvite can also result in the sequestration of a portion of the available orthophosphate, the amount of which is dependent on the initial stoichiometric quantity of magnesium and calcium. Following hydrolysis, the rise in pH would cause the equilibrium reaction (2) to shift toward the volatile $NH_3(g)$. This would result in nitrogen losses, as well as the potential for environmental pollution since NH_3 is deemed toxic at high concentrations.

$$NH_3 \overset{pK_a=9.25\ (t=20\ C)}{\Longleftrightarrow} NH_4^+ \qquad \text{(Equation 2)}$$

The high concentration of low molecular weight (LMW) organic acids, including oxalic, uric, citric, hippuric, lactic, and acetic acid in urine, can serve as an electron donor for the growth of anaerobic microorganisms (Udert & Wächter, 2012; Etter et al., 2014). This is coupled with the microbial reduction of sulphate to hydrogen sulphide and the generation of NH_3(g), is responsible for the foul odour of hydrolysed urine that has been stored (Troccaz et al. 2013). Due to these factors, stabilizing and disinfecting urine is often necessary to prevent ureolysis (Randall et al., 2016). In particular, minimizing ammonia volatilization is crucial as it poses both an environmental hazard and results in the loss of valuable fertilizer. Various approaches have been suggested to address these issues, including acidification/alkalization, salinization, electrochemical sanitation, freeze-thaw, biological nitrification, and lactic acid fermentation which have been summarised below in section 2.4.

2.3. Nutrients in wastewater and challenges

2.3.1 Nutrients in wastewater

Nitrogen and phosphorus are common nutrients that occur naturally and are also present in household products such as fertilizers, foods, soaps, and detergents. Human waste is another significant source of these nutrients in wastewater. When nitrogen and phosphorus are easily accessible in water, excessive plant growth can occur, leading to eutrophication. This process involves the proliferation of algae, weeds, and cyanobacteria (blue-green algae), resulting in algal blooms that can deplete oxygen and harm aquatic life. To safeguard freshwater ecosystems, wastewater must be treated effectively to ensure its safe discharge and recycling. With the use of modern flush toilets, wastewater treatment facilities (WWTP) are designed to partially remove nutrients (N, P, and K). As nutrient loads into the ecosystem rise, the steadily declining water quality implies that the status quo scenario is not sustainable. However, this is typically not a technology failure, but rather an inadequacy of treatment brought on by high infrastructure costs and challenges implementing strict P requirements. Domestic wastewater (Badeti et al., 2021), which can be separated into different streams using no-mix toilets or urinals (Randall & Naidoo, 2018). Urine makes up less than 1% of the volume of the domestic wastewater. Wastewater management might be made simpler by using concentrated streams of separated urine (see Figure 2-1). About 80% of the N, 56% of the P, and 63% of the K found in domestic wastewater are present in this urine stream. It was estimated in 2011 that the P content of just human urine and faeces could provide 22% of the world's P needs (Badeti et al., 2021). Additionally, by using source separation, wastewater treatment plants could switch from being net energy consumers to producers (Wilsenach & van Loosdrecht, 2006). This is because biological nutrient removal processes can use less energy. Moreover, in developing nations, centralised wastewater treatment facilities are sometimes overbuilt for a predicted use that may not be achieved (Jimenez et al., 2015).

Currently, 16% of all mined P is released into wastewater treatment plants, discharging about 8% to the environment (Stamm et al., 2022). It is estimated that wastewater treatment facilities annually release 7.4 million tonnes of nitrogen and 1.5 million tonnes of phosphorus into surface waters (see Figure 2-1). If not discharged in accordance with tight rules, this can have a severe impact on nearby ecosystems. For example, improper management of municipal wastewater contributes significantly to eutrophication (Larsen et al., 2021a) and environmental degradation (Larsen, 2020) caused by P. A significant amount of energy is required for the removal of N and P in wastewater treatment facilities. According to estimates, precipitation for P removal needs around 49 MJ kg-1, whereas denitrification for N removal needs about 45 MJ kg-1 (Wilsenach & Van Loosdrecht, 2004). By minimising struvite fouling problems and decreasing sludge generation, reducing nutrient inputs to wastewater treatment plants can also lower operational costs (Etter et al., 2015). Surprisingly, technologies that actually remove the scale as an inorganic fertiliser have been developed in response to the necessity to remove struvite scaling from wastewater treatment plants (Ishii & Boyer, 2015). Effective urine separation may also allow activated sludge operations to run at low sludge ages as a JHB process or a two-stage phoredox process (anaerobic/aerobic with anoxic zone in the sludge return) (e.g. 5 days). By using this design, half may reduce the required reactor size, or an existing reactor could handle twice as much load.

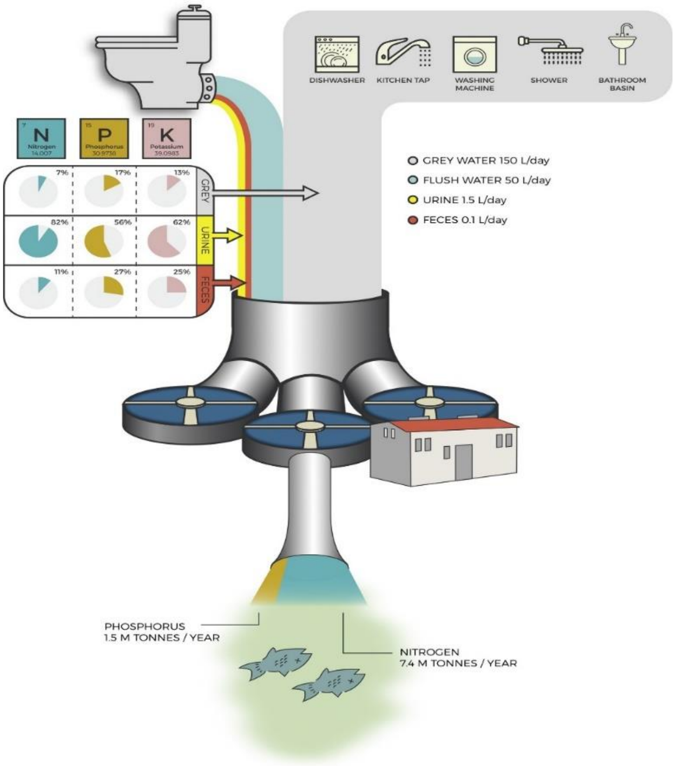


Figure 2-1. Composition of nitrogen (N), phosphorus (P), and potassium (K) in the different streams of domestic wastewater, namely urine, feces, and greywater. The contribution of each nutrient in each stream is expressed as a percentage of the total composition. Taken from (Randall & Naidoo, 2018).

2.3.2 Nutrient removal technologies

As part of the primary treatment process, a majority of wastewater treatment facilities (WWTPs) utilize physical methods such as skimming to remove floatable organic and inorganic materials, and sedimentation to remove settleable organic and inorganic solids. (Rout et al., 2021). The secondary treatment phase ensures that any remaining organics and trace amounts of dissolved nutrients are biologically removed. Therefore, even after conventional secondary treatment, some suspended and dissolved debris as well as a significant part of nutrients may still be present in the effluent of WWTPs (Jasim, 2020).

Secondary treatment alone may not be sufficient to effectively remove certain components of wastewater, including nutrients, refractory organics, heavy metals, and other substances. As a result, the use of a tertiary or advanced treatment system becomes essential. Given the lack of tertiary treatment capabilities at conventional wastewater treatment plants (WWTPs), the nutrient-enriched secondary effluents from these facilities are regarded as the main point source of nutrient contamination. During the early stages of developing nutrient removal technologies, traditional aerobic technologies like the activated sludge process (ASP) were initially used for treating household wastewater, with the expectation of their potential to remove nutrients. The widespread adoption of these technologies, particularly in developing nations, is severely constrained by the high operational and capital costs associated with their deployment. Similar studies have revealed that physical methods are expensive owing to alternative comparator to more passive biological systems like trickling filters, or extensive treatment technologies which limit energy consumption and cost (Le-Clech, 2010). Similarly, it is frequently used to reduce the nutrient concentration of the effluent before it is released into surface waterways by chemical dosing. Therefore, due to the variables mentioned above, most traditional treatment techniques are unable to meet the tight nutrient release criteria and hence more advanced technologies become necessary but they have high capital and operating costs. Hence, biological nutrient removal techniques are generally more preferred because of these reasons (Tchobanoglus et al., 2003).

2.3.3 Biological Nutrient Removal

The term "Biological Nutrient Removal" (BNR) refers to different configurations of activated sludge processes that utilize microorganisms living in aerobic, anaerobic, and anoxic conditions to eliminate nitrogen (TN), phosphorus (TP), and chemical oxygen demand (COD) via metabolic mechanisms such as denitrification, nitrification, and biological phosphorus removal. The type of redox conditions present depends on the electron acceptors available, such as oxygen in the aerobic zone, nitrogen oxides in the anoxic zone, and organic substances that can act as both electron donors and acceptors in the anaerobic zone. For the removal of nitrogen, an aerobic-anoxic condition is beneficial, whereas the alternation of anaerobic and aerobic conditions is necessary for the removal of phosphorus (Ekama et al., 2011; Ogunlaja & Parker, 2015).

Biological Nitrogen removal (BNR) is typically achieved by a series of two-step processes that involve nitrification and denitrification (shown in Figure 2-3). Furthermore, nitrification is a two-step process in which ammonium is first converted to nitrite by AOB (nitritation process), which is then followed by NOB's conversion of nitrite to nitrate (nitratation process). Ammonia and nitrite are used by the AOBs and NOBs as energy sources, CO_2 as a source of carbon, and O_2 as an electron acceptor (Rout et al., 2021). Nitrosomonas is the most prevalent AOB, while Nitrosopira, Nitrosovibrio, Nitrosococcus, etc., are all capable of performing process of nitritation. Similar to this, Nitrobacter is the most prevalent NOB; however, Nitrospira, Nitrococcus, Nitrocystis, and Nitrospina are also known to participate in the nitratation process. While the NOBs are associated within the alpha subdivision of the Proteobacteria, the AOBs are allied to one another in the beta subdivision. The nitrification process has the following stoichiometry:

In the nitrification process, 0.08 g of inorganic carbon is required for the synthesis of 0.16 g of new cells, while 4.57 g of O_2 and 7.14 g of $CaCO_3$ (alkalinity) are used per gram of NH_4^+ oxidised. Similar to nitrification, denitrification is finished by facultative heterotrophic bacteria through a succession of subsequent reduction steps. First, nitrate is converted to nitrite by the enzymatic activities of nitrate, nitrite, nitric oxide, and nitrous oxide reductases, respectively. Next, nitrite is converted to di-nitrogen gas through the intermediates of nitric oxide, nitrous oxide, and nitrogen gas. Nitrate and/or nitrite are used by the bacteria during the denitrification process as electron acceptors and organic matter is used as a source of carbon and energy.

There are two types of denitrification: assimilatory (nitrate is converted to ammonium for use in biosynthesis) and dissimilatory (nitrate is employed as an electron donor). type acceptor for respiration. Most prevalent heterotrophic Alcaligenes, Bacillus, Chromobacterium, and others are genera of denitrifying bacteria. such as Neisseria, Paracoccus, and Pseudomonas. Depending on the type of electron donor, the stoichiometry of the denitrification process changes. According to Aponte-Morales et al. (2013), the overall reaction is as follows when organic carbon serves as an electron donor: 3.57 g of $CaCO_3$ are produced during the denitrification process for every gram of NO_3 decreased, recovering half of the alkalinity that was lost during the nitrification phase. The overall mechanism of biological nitrogen removal and the evolution of wastewater treatment plants over the year to incorporate nitrogen removal process is shown in Figure 2-2.

Alum, also known as aluminum sulfate, is commonly used in wastewater treatment plants (WWTPs) to remove phosphorus through a process called chemical precipitation. Alum contains aluminum ions, specifically aluminum sulfate [$Al_2(SO_4)_3$], which dissociate in water (Metcalf et al., 1979). When alum is added to the wastewater, it reacts with the phosphate ions (PO_4^{3-}) present in the wastewater. This reaction forms insoluble aluminum phosphate compounds ($AlPO_4$), or aluminum hydroxide compounds, depending on the pH conditions(Arcas-Pilz et al., 2021). The formed aluminum phosphate or aluminum hydroxide compounds are solid and do not dissolve in water. They precipitate out of the wastewater as solid particles. These solid particles settle down to the bottom of the wastewater treatment tank due to their increased density.

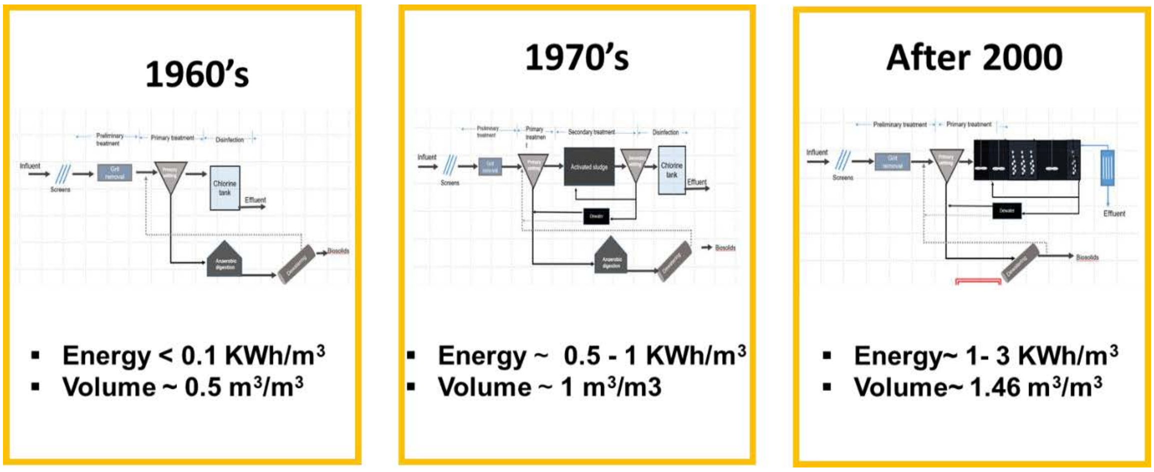


Figure 2-2 - Evolution of wastewater treatment plants and configuration from 1960-present

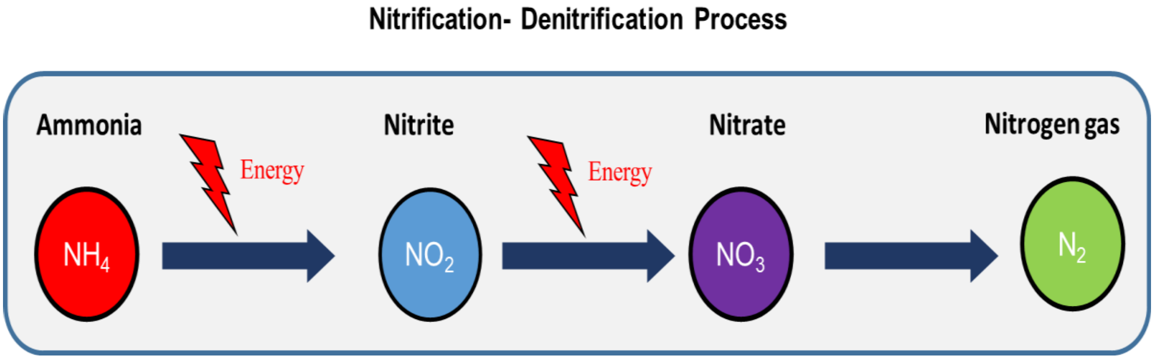


Figure 2-3 - Energy Intensive Biological Nitrogen removal process in WWTP

2.4. Treatment processes of source separated urine

Various urine treatment processes are employed for nutrient recovery, aiming to harness the valuable nutrients present in urine, such as nitrogen and phosphorus, while minimizing their environmental impact (Maurer et al., 2006). One common method is struvite precipitation, where magnesium and phosphate are added to urine, resulting in the formation of struvite crystals that can be harvested and used as a slow-release fertilizer in agriculture (Simbeye et al., 2023). Another approach involves urine stabilization through pH adjustment or storage, which reduces the risk of pathogen transmission and odor issues while preserving nutrient content. More advanced technologies like electrolysis can efficiently separate nitrogen and phosphorus ions, enabling their direct recovery for agricultural or industrial applications (Courtney & Randall, 2021). These innovative approaches not only reduce the need for synthetic fertilizers but also promote sustainable nutrient management, contributing to a more environmentally friendly and resource-efficient future (Larsen et al., 2021b). These have been discussed in detail below:

2.4.1. Precipitation and Crystallisation

A direct method for recovering the P in urine is through precipitation as $MgNH_4PO_3 \cdot 6\ H_2O$, i.e struvite or through $KMgPO_3 \cdot 6\ H_2O$, i.e K-struvite (Krähenbühl et al., 2016) and calcium phosphate. Both struvite and K-struvite are broadly utilised as slow release fertilisers. Struvite precipitation spontaneously happens when the required amount of magnesium is added to hydrolysed urine. Mg^{2+} addition is needed as the molar concentration of Mg^{2+} is lower compared to that of PO_4^{3-}. MgO, $Mg(OH)_2$, and $Mg(Cl)_2$ were found to be appropriate for this process (Kabdaşlı & Tünay, 2018). Even though struvite crystallization begins at pH as low as 7.5, the ideal pH for the process was found to be around 9 (Ronteltap et al., 2010). When ammonium is removed, K-struvite was found to precipitate instead (shown in Figure 2-4). Also, since the molar ratio of NH_4^+:K:PO_4:Mg^{2+} in urine is usually 260:13:6:1, even with the addition of external Mg^{2+} source, only P will be removed with this method. Moreover, urine dilution because of flushing was found to reduce the amount of phosphorus recoverable through this process. The major disadvantage of this process are the minimal N recovery due to undesirable molar ratio of NH_4^+:K:PO_4:Mg^{2+} in urine which necessitates the need for additional Mg. Hence, the process becomes uneconomical due to high input cost of magnesium salts and operational cost. Also, struvite precipitation by adding Mg is not always cheap as the magnesium-based slats are generally more expensive compared to other salts such as calcium and potassium-based salts. Urine piping is required to convey urine from urinal to storage irrespective of whether struvite is extracted or not and so comparing struvite extraction to blocked pipe does not arise.

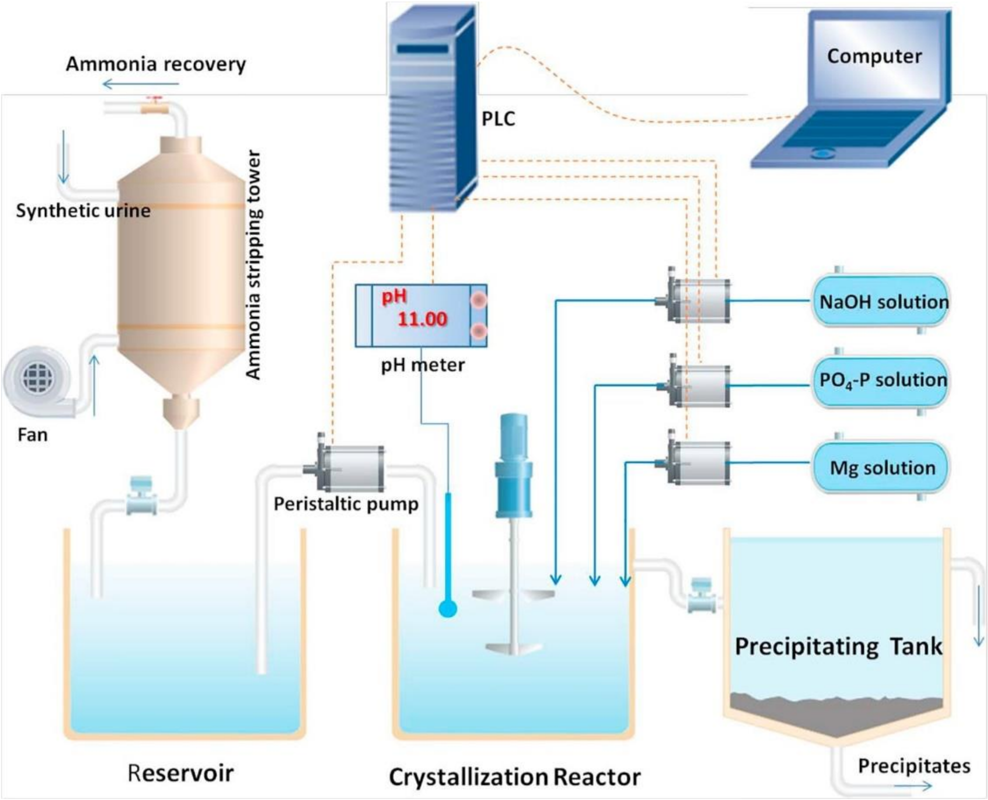


Figure 2-4. Schematic diagram of a struvite and K-struvite precipitation process. Taken from (Gao et al., 2018).

2.4.2. Biological stabilisation by partial and complete nitrification

According to (Maurer et al., 2006), urine stabilization involves a range of processes designed to prevent the degradation of organic matter and associated problems. These processes aim to (i) break down organic matter, which in turn prevents malodors, (ii) impede the volatilization of ammonia, and (iii) prohibit unwanted precipitation that can cause operational difficulties like pipe clogging or membrane fouling.

The primary objective of biological stabilization is to decrease the pH level of urine by means of nitrification. This process accomplishes the objective of eliminating a majority of the organic substances present, which includes compounds that contribute to foul odors (Udert & Wächter, 2012). Urea in fresh or stored urine can be quickly converted to ammonium by a biological action with high enzymic task, regardless of the substrate. While partial nitrification is a commonly used technique for biological stabilization, complete nitrification has been employed by some researchers to produce a stable substrate suitable for supporting algae growth (Coppens et al., 2016). In conventional wastewater treatment, the oxidation of ammonium and nitrite is typically uncomplicated. However, the reaction rates may be hindered in concentrated urine solutions containing high levels of salinity (Moussa et al., 2006), free ammonia, and nitrous acid (Anthonisen et al., 1976), leading to an imbalance between AOB and NOB (Udert & Wächter, 2012).

The alkalinity in urine, which is mostly derived from urea hydrolysis (Equation 3), leads in equal amounts of total ammonia and nitrate during partial nitrification. However, when ammonia is oxidized, the pH of the system drops to about 5.4, which leads to a decrease in the activity of the dominant ammonia-oxidizing bacteria, Nitrosomonas eutropha, due to energy limitation. Therefore, it is crucial to maintain a narrow pH range by governing the inflow rate of urine to ensure consistent nitrate generation, as partial nitrification can result in the formation of ammonium nitrite. Only biofilm reactors have successfully achieved sustained partial nitrification, according to experiments conducted before 2019, with nitrification rates per surface similar to those found in conventional wastewater systems reported that nitrification rates ranged between 0.5 and 2.5 gN/m^2/d (which are at similar nitrification rates per surface as in municipal wastewater systems). Earlier studies on urine nitrification focused primarily on assessing the feasibility and stability of the process, rather than optimization. These studies have demonstrated that the process can be stable for extended periods, with one study achieving stable operation over an year. In a pilot plant study, (Fumasoli et al., 2016) observed an inverse correlation between the nitrification rate and the inlet total ammonia concentration, indicating that inhibition by salinity and volatile ammonia is a relevant factor in nitrification of urine.

$$H_2CONH_2 + 2\,H_2O \longrightarrow NH_3 + NH_4^+ + +HCO_3 \quad \text{(Equation 3)}$$

The necessary reactor volume for achieving partial nitrification is estimated to be around 50 litres per person for heavily concentrated urine, and about 10 litres per person for diluted urine. The volume of the reactor needed for partial nitrification must be balanced against the energy needed for the subsequent evaporation operations. The main hurdle for achieving stable partial nitrification is the prohibition of nitrite-oxidizing bacteria by HNO_2, which is due to the accumulation of nitrite. Another challenge in urine nitrification is the growth of acid-tolerant ammonia-oxidizing bacteria (AOB) when operated at low pH values, which can cause a significant decrease in pH levels and result in the emission of harmful volatile nitrogen compounds, such as NO, N_2O, NO_2, and HNO_2. This can have detrimental impacts on air quality, climate, and loss of acid-sensitive nitrifiers. Although maintaining pH within a narrow range is important, there are not many reliable ways to monitor and regulate nitrite levels, which can lead to inhibition of nitrite-oxidizing bacteria (NOB) by nitrous acid accumulation, further destabilizing the process.

Another research gap is the production of nitrous oxides while operation, which has recently obtained attention in the field of wastewater treatment.

In order to achieve complete nitrification, which involves the oxidation of ammonia to nitrate, it is necessary to supplement more alkalinity. It was demonstrated in both biofilm systems and reactors with suspended biomass, and has been shown to be effective within a pH range of 6.5 to 8. Similar to partial nitrification it has been observed that higher urine dilutions result in increased nitrification rates. Coppens et al. (2016) demonstrated that the use of a halotolerant inoculum reduced start-up time by half compared to an inoculum from a wastewater treatment plant. Despite the positive impact of the halotolerant inoculum on start-up time, it did not contribute to a faster end reaction rate, showing that it is difficult for ammonia oxidizers in particular to adapt over the long run to high saline concentrations. The authors identified ammonia oxidisers as less resistant to high saline concentrations compared to nitrite oxidisers, further highlighting the importance of appropriate inoculum selection and salt management strategies for successful nitrification in urine treatment processes. The finding of salt inhibition in urine nitrification is consistent with the results of Mackey et al. (2014), who reported low process rates when seawater was used to dilute urine in the place of of tap water, as has been performed in similar studies (Jiang et al., 2011). This provides further evidence of the negative impact of salt on urine nitrification and highlights the need for appropriate management of salt concentrations in urine treatment processes.

Dilution with tap water is often not an issue in many real-world applications of full nitrification, such as heterotrophic denitrification in sewers. Hence, compared to partial nitrification, where volume reduction is frequently the primary goal, the appropriate dilution for these reasons is less crucial. In fact, the use of a larger volume of water for urine flushing can often provide needed dilution of both ammonia and salinity, except in cases such as Hong Kong, where seawater is used for flushing. However, the main challenge in implementing complete nitrification is the addition of alkalinity, as highlighted by Oosterhuis and Van Loosdrecht (2009).

2.4.3. Chemical stabilisation using acid and base addition

The enzyme that breaks down urea is prevented from acting by chemical stabilisation, which also stops the pH from increasing. Various ions (F^-, Zn^{2+}, Ag^+)were shown to be not effective by Ray et al. (2018) whereas acid was found to be effective as previously described by Maurer et al. (2006). Alkaline and electrochemical technologies have only recently begun to take off. There is a dearth of recent literature on the treatment of hydrolysed urine with acid and on the use of filtration and precipitation to reduce fouling of membranes (Horn et al., 2023; Simbeye et al., 2023). Further research is needed to investigate the effectiveness of these techniques and their potential for practical implementation in urine treatment processes.

Antonini et al. (2012) demonstrated the effectiveness of stabilizing stored urine using a strong acid, however advised against the adoption of this technique on a small scale in Vietnam due to potential hazards. Jiang et al. (2017) established a mathematical correlation between the pH level and the lost ammonia during the distillation process of acidified stored urine. Their findings revealed that nitrogen recovery was as high as 99.5% at a pH below 4, but decreased to less than 50% at a pH greater than 7. Maurer et al. (2006) discussed the stabilisation of stored urine and noted that a substantial amount of acid is necessary because of the high alkalinity of urine resulting from urea hydrolysis.

Ray et al. (2018) proposed an economically viable solution for stabilizing fresh urine using weak acids to inhibit urea hydrolysis. Similarly, Saetta et al. (2019) conducted modelling studies using real-time conductivity and pH analysis to dose acetic acid and maintain a low pH of about 4, for short-term prohibition of urea hydrolysis in building-scale urine collection systems (Simha et al., 2021b). Minimal mixing and less time for hydrolysis to take place inside the pipes might be caused by less frequent urination and small urine quantities.

Urea's alkaline stabilization is an alternative to acidic neutralization. To achieve the pH >11 required for urease prevention in urine, Randall et al. (2016) compared the effectiveness of $Ca(OH)_2$, $CaCO_3$, and slaked lime. The study found that slaked lime, or Ca(OH)2, was the most suitable option due to its low cost and solubility. This allowed for the appropriate amount of slaked lime to dissolve from a vast quantity poured into a urine contained reactor (Randall et al., 2016).

This stabilisation technique has the added benefit of precipitating phosphorus immediately, which could result in the creation of a distinct P-based fertiliser. Direct alkaline stabilisation in liquid urine can be replaced by a similar procedure in a dry bed. Dutta and Vinnerås (2016) showed that a $Ca(OH)_2$ and wood ash mixture can sustain pH >10, which in this solid mixture was found to retain urea even if some ammonia was lost. Only with wood ash, Senecal and Vinnerås (2017) achieved pH values higher than 10. Simha et al. (2018) showed that the idea of an anion exchanger resin may work by interchanging OH^- ions against Cl^- ions to deliver additional base to fresh urine before these processes. However, no follow-up initiatives have been seen since 2019. The stability of pH has major implications for both liquid and dry alkaline stabilisation. According to Senecal and Vinnerås (2017) CO_2 absorption from the air led to a low pH. According to Simha et al. (2018), pH lowering is caused by buffering of alkaline earth metals and ammonia production from urea decomposition in addition to CO_2 absorption.

Therefore, the main gap of research for alkaline stabilisation is the design of a system that effectively blocks CO_2 uptake from the air, resulting in a crucial positive feedback to increase urea hydrolysis (Vasiljev et al., 2022). Making bricks from sand and stabilising urine that produces bacteria that can resist bases is an alternative to using alkaline urine (Lambert & Randall, 2019). It is possible to generate a low-cost construction material with a compact strength equivalent to traditional bricks by combining calcium with the carbonate formed during urea hydrolysis (Chipako & Randall, 2020b). The resulting calcium carbonate acts as a binding agent for the sand particles, creating a sturdy final product.

2.4.4. Electrochemical processes for stabilisation

Urine has been stabilised using electrochemical techniques that remove organic components, prevent urea breakdown, or kill microbes. Mainly urine stabilisation in electrolytic cells in discussed this section. An extensive amount of research has been focused on the removal of organic contaminants using microbial fuel cells, which also produce energy (Arve & Popat, 2021; Yang et al., 2021).

The majority of urine electrolysis experiments use indirect chlorine oxidation as the main method of removing organic and nitrogen compounds; occasionally NaCl was actually added to speed up the oxidation process (Sophia & Muralidharan, 2015; Chun et al., 2018). Chlorine is a strong oxidant, which is generated by the oxidation of chloride. Additionally, hydroxyl radicals are significant oxidants, mainly for boron-doped diamond electrodes (De Paepe et al., 2020).

Zöllig et al. (2017) investigated the electrolysis of actual stored urine in batches of tests using either a titanium anode with a thermally destroyed IrO_2 sheet or a BDD. As soon as the studies begin, organic molecules are quickly eliminated on both electrodes. Ammonia oxidation on BDD was noticed prior to the COD elimination experiment. In comparison to rates reported in biofilm systems, the COD degradation rates were roughly 10–50 times greater. However, a significant amount of energy is being used (Larsen et al., 2021b).

2.4.5. Nutrient recovery by volume reduction and microalgae

Volume reduction helps in the recovery of numerous nutrients. Evaporation, membrane processes like Electro dialysis and algal nutrient uptake are all of engineering interest. In addition to water and resource reclamation from seawater and RO brines, urine presents an opportunity for generating water and salts through crystallization technologies. A recent review has aptly referred to urine as the "liquid gold" of wastewater due to the valuable resources it contains, which can be recovered. For instance, urine can be utilized to create a wide range of fertilizers, potentially reducing the necessity for resource-intensive mining operations. Furthermore, freeze crystallization offers a method to recover water in the form of ice from these various wastewater sources. Researchers have even explored the application of freeze crystallization processes for seawater treatment. Additionally, a hybrid process combining RO and freeze crystallization holds potential for maximizing water production while reducing overall costs. In general, freeze crystallization stands out as an energy-efficient process because the heat required for freezing is nearly seven times less than the latent heat needed for evaporation.

2.4.5.1 Drying and distillation processes

Both drying and distillation rely on evaporation; however distillation procedures have the ability to recover water and require a significant amount of energy (Udert & Wächter, 2012), this isn’t usually noticed in a drying process. All non-volatile chemicals in urine are primarily concentrated by evaporation, which has the major advantage of keeping micronutrients in the finished fertiliser product (Harder et al., 2019). However there is a potential harm if organic micropollutants are not removed throughout this method. Risk of some ammonia volatilisation persists, but depending upon the effectiveness of the prior stabilisation process and the process design, it can be reduced.

After onsite alkaline stabilisation, drying has generally been recommended for on-site settings in the bathroom. High air temperatures are advantageous for the procedure, although (Randall et al., 2016), found that alkaline stabilised urea loses stability above 40 °C. (Dutta & Vinnerås, 2016) applied a pressurised air stream to a drying bed based on wood ash to evaporate alkaline stabilised urine.(Senecal & Vinnerås, 2017) obtained a solid product equal to commercial NPK fertilisers, at a temperature of 35°C and achieved more than 80% nitrogen recovery. The relevant high loss of ammonia to the atmosphere clearly shows the need for process improvement. According to Simha et al. (2018), higher temperatures caused higher evaporation rates; increased airflow rates were only beneficial up to a point, and a deeper bed was ineffective. The improvement of rates dependent on temperature is required since urea does not exhibit chemical stability over 40°C. Udert and Wächter (2012) used distillation in a laboratory-scale evaporator to remove water from partially nitrified urine, producing either a dry product or a highly concentrated liquid product. In a commercial distillation reactor with the addition of vapour compression and heat recovery, Fumasoli-Hug (2016) reported that the implementation of this technique produced a concentrated nutritional solution with composition (w/w) of roughly 2% K, 0.2% P and 5% N.

According to reports, distillation requires about 110 Wh/L of energy, whereas water evaporation without energy recovery requires about 710 Wh/L. The minimal nitrogen loss is trapped in the distillate and not released into the atmosphere, which is another crucial factor. Although the benefit of drying is relevant on a small scale, there are still some significant research gaps in the areas of energy recovery and reducing ammonia losses. Energy recovery is distillation's undeniable benefit. Distillation is a commonly used industrial process, but if it needs to be scaled down, more innovative approaches such as membrane distillation may be more promising.

2.4.5.2 Membrane Processes

Hydraulic and osmotically driven membrane processes represent versatile technologies applicable in various fields, such as water treatment, desalination, power generation, and solution concentration. High-pressure membrane techniques like nanofiltration (NF), reverse osmosis (RO), and forward osmosis (FO) have demonstrated their effectiveness in dehydrating water and wastewater solutions. Additionally, these processes have been explored to concentrate urine, enhancing its suitability for efficient struvite precipitation or the production of concentrated fertilizers. This is vital because urine typically contains lower nutrient concentrations compared to commercial liquid fertilizers, often being 10-1000 times more diluted, leading to significantly increased transportation costs. Furthermore, ongoing research has delved into the utilization of less restrictive NF membranes to produce pharmaceuticals-free fertilizer solutions, offering an environmentally friendly approach to nutrient recovery and waste management.

Forward osmosis and MD are the main membrane techniques used for volume reduction that are under development. Additionally, MD is a research technology similar to traditional distillation. There are major membrane fouling issues with ultrafiltration; however it has been demonstrated that it is an efficient pre-treatment procedure before the osmotic membrane processes targeted at lowering volume and eliminating 99% of the suspended particles in hydrolysed urine (Ouma et al., 2016). In the treatment of fresh urine, which contains nitrogen primarily in the form of the uncharged urea molecule, and in the treatment of stored urine, where nitrogen is primarily contained in the acid-base pair NH_4^+/NH_3, there are significant differences to consider in membrane processes. Fouling is a common issue for all membrane applications, often resulting in short experiment durations. For on-site membrane treatment applications, where routine membrane cleaning might be challenging, this poses a substantial issue. Therefore, it is crucial to develop effective fouling prevention and cleaning strategies to ensure the practical implementation of membrane processes for urine treatment.

Only volatile substances can travel through a hydrophobic membrane due to vapour pressure, which is created by the temperature differential across the membrane. The technique is suited for stable urine, where the concentration of volatile ammonia is low, because it is expected that in urine, water vapour, NH_3, and volatile organic components will pass through the membrane. MD has more energy requirements than vapour compression distillation with heat recovery. To treat water with a salinity in spiral wound MD modules, Winter et al. (2011) and Fumasoli-Hug (2016) estimated 107 kWh/m^3 for concentrating partially nitrified urine using vapour compression distillation and heat recovery. The main benefit of MD over traditional vapour compression distillation may be a simpler setup process and a potential for using low-grade heat rather than electricity. There are few studies using stabilised fresh urine and MD. DCMD was evaluated on a urea solution that had previously been made from urine using a FO procedure by Ray et al. (2019) Although some COD was shifted, there was only a partial transfer of urea as a result of this process. The amount of urea recovered during the MD process on the urea solution ranged from 72 to 92%, and different stabilisation techniques had no impact on recovery. Despite the lower COD concentration, the membrane experienced substantial fouling, raising concerns about the transferability of MD to fresh urine.

According to Tun et al. (2016) the transfer of nitrogen over the membrane for stored urine is proportional to the amount of free ammonia present in the feed solution and negligible at pH 6 following chemical acidification. For stabilisation, Xu et al. (2019) suggested partial or full stabilisation, which would offer the added benefit of reducing a significant portion of the COD. Only total nitrification produced a high nitrogen recovery of 94%

with an unexpectedly high pH of 8.3 following partial nitrification, but the laboratory studies were too brief to confirm the anti-fouling effects of biological therapy. Lack of long-term experience, particularly with regard to fouling, represents a significant research gap for FO. In addition, only hydrolysed urine with a low pH is suitable as feed and requires a significant amount of acid for neutralisation, comparable to the FO procedures Maurer et al. (2006). Khumalo et al. (2019) employed a new method to address DCMD's poor ammonia rejection at high pH. They altered the membrane properties with nanoparticles to produce more evenly spaced, smaller holes.

Reverse osmosis (RO) has demonstrated potential as a treatment method for concentrating unhydrolyzed urine, as indicated by Courtney and Randall in 2022. Prior research in urine membrane filtration primarily focused on hydrolyzed urine, as urea, being a small and uncharged compound, presented challenges in recovery when using brackish water RO and nanofiltration (NF) membranes, as highlighted by Ray et al. in 2020. Nonetheless, Courtney and Randall (2022) revealed that by employing a high-pressure seawater RO membrane and incorporating a preliminary air bubbling treatment with Ca(OH)2 stabilized urine, it was possible to achieve a concentration factor of at least 2.5, while also reclaiming 86% of the urea present in the brine stream. The urine could have theoretically been further concentrated, but this was constrained by the size of the laboratory equipment. However, there are limitations associated with a process that relies solely on RO for urine concentration.

2.4.5.3 Nutrient uptake by algae

Algae can absorb nutrients, which can be followed by separation and a drying process to reduce their volume biologically (shown in Figure 2-4). Previous studies have focused only on experimental results for the intricate process of algae growth on urine and have not investigated the processes of separation and drying.

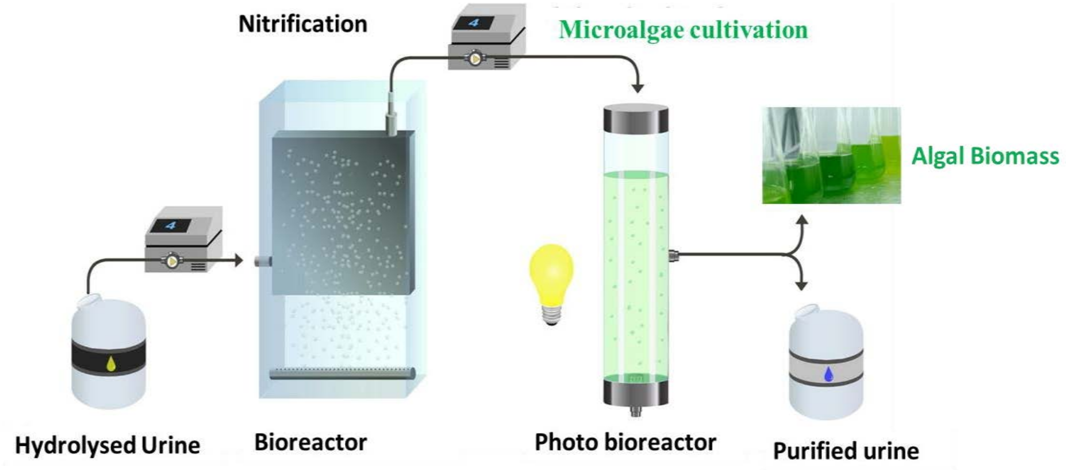


Figure 2-5. Schematic overview of the urine nitrification bioreactor set-up and algae production (Adapted from (Coppens et al., 2016))

Many studies have used urine as a source of nutrients to cultivate algae for various purposes such as energy production, slow-release fertilization, and the extraction of chemical byproducts from algae. However, despite the significant variation in algal species, reactor layout, and urine dilution in the literature, the use of a culture medium in addition to urine is essential. This is due to the inadequate supply of micronutrients in urine and the fact that fresh urine has a high N:P ratio (around 28:1) in comparison to the generic needs of algae (16:1), which makes it incapable of supporting algal growth on its own. Therefore, the addition of phosphorus is necessary for quantitative nitrogen removal (Tuantet, 2015). Additionally, phosphorus crystallization during storage can further increase the N:P ratio, while essential elements like Ca and Mg are left out.

Salt and nitrogen compounds, particularly free ammonia above 140 gN.m-3 (Tuantet et al., 2014) and high nitrate concentrations exceeding 1000gN/m^3 (Coppens et al., 2016), inhibit algae growth, similar to biological stabilization. Therefore, full strength urine is not a suitable substrate. Optimal long-term results were achieved by Tuantet et al. (2014), who optimized a photobioreactor for approximately eight months at a pH of 7, with a dilution factor of 2 and with the addition of missing nutrients. The maximum nitrogen uptake rate was 1300gN/m^3/d, and the highest phosphorus uptake rate was 150 g/m^3/d. The configuration of a bioreactor is challenging because of the limited penetration of light, which is further complicated by the dark colour of urine, as well as the inadequate nutrient content of urine for algae growth (Coppens et al., 2016). To overcome inhibition and improve light penetration, bioreactors are usually designed for diluted urine.

Tuantet et al. (2019) demonstrated the possibility of producing microalgae continuously in a photobioreactor with a 5mm light path. However, a trade-off arises when trying to achieve efficient nitrogen removal and photosynthetic efficiency simultaneously while treating urine at reasonable nitrogen concentrations. To avoid inhibiting algae growth with free ammonia, the hydraulic retention time (HRT) must be extended, resulting in higher biomass concentration and nutrient removal. As the biomass concentration rises, however, photosynthetic efficiency decreases due to the high biomass concentration, necessitating even shorter light paths than those evaluated in the study. This presents a significant hurdle for the recommended open raceway ponds employed for algae production, as demonstrated in a pilot operation of a 0.5m deep raceway pond by (Chatterjee et al., 2019), where heavy dilution was necessary to achieve only 50% nitrogen recovery. Such a system would require a footprint of 3m^2/p, depending on daily nitrogen output from urine. Thus, in addition to the post-processing difficulties, creating an efficient photobioreactor is a critical research focus for reducing urine volume through algae growth.

2.5. Current and past implementation of source separation of urine

It is essential for the adoption of source separating technologies to examine similar real-world events and identify the triumphs and failures. Three studies were conducted: two in academic settings, one with a South African municipality, and one with a German office building in the Hesse region. The case studies instead concentrate on real-world implementations of source separation systems with an emphasis on potential problems rather than the technology.

2.5.1 eTekhwini municipality

The South African city of Durban is home to the eTekhwini municipality. The municipality built an estimated 75,000 urine diversion toilets in 2001 (Roma et al., 2013). After three years of operation, a survey of user satisfaction was carried out in 2004. More than 78% of respondents said they were happy with the restroom system. After ten years of operation, a comparable survey was carried out. On the other hand, more than 70% of respondents said they were dissatisfied with the current system. This shift in view could be attributed to improper maintenance, which led to unpleasant odours and operating issues. Pipes had detached from the pedestal as a result of poor installation workmanship and construction materials. In addition, misuse and vandalism caused toilet seats to break (Roma et al., 2013). This demonstrates how crucial proper installation and ongoing maintenance are necessary for the success of such devices. As a result, the project did not incorporate any urine or feces recycling measures, and urine was instead directed to flow into a soak pit situated adjacent to the toilets (Roma et al., 2013). The inability to use this urine in a nutrient recovery process is indicative of a massive loss of potentially recoverable phosphorous. This indicated a need for further education in terms of sanitation as a necessary preface to the implementation of any resource recovery orientated system at this scale. This demonstrates how crucial proper installation and ongoing maintenance are to the success of such systems. Additionally, it was shown that residents of the municipality view toilets that don't flush as inferior. This could lead to a greater conviction in source-separating measures and encourage adequate maintenance by highlighting the economic and environmental benefits (Roma et al., 2013). A substantial number of pathogens were found in the faeces produced by the community, according to early testing of local faecal and urine samples in the community (Bischel et al., 2015). Pathogens such as protozoa and helminths were among them (Bischel et al., 2015). As a result, the project did not involve the recycling of faeces or urine. Additionally, a soak pit next to the toilets was authorised to receive urine (Bischel et al., 2015). The inability to recover nutrients from this urine indicates a significant loss of possibly recoverable phosphorous. This suggested that additional training in sanitation was required before any system focused on resource recovery at this level could be put into place.

2.5.2 University of Technology Sydney

The University of Technology, Sydney (UTS) Institute for Sustainable Futures (ISF) research on sustainable sanitation examined the challenges in replacing current sanitation systems with more environmentally friendly urine diversion (UD) systems in 2013 (Mitchell et al., 2013). The project included tanks for storing and transporting urine, waterless urinals, and piping systems for urine samples. Regulations and institutions, user habits, operation and execution, 83 agriculture trials, market and socio-cultural acceptance were all considered in the analysis of the project's performance. It was discovered that (i) a lack of policies to support or promote urine diversion and reuse, (ii) a lack of regulations for source-separated urine practices, and (iii) a small market share could all hinder the growth of urine diversion systems.

2.5.3 University of Florida

At the University of Florida, a study was carried out with a primary focus on the social acceptance of urine separation facilities. There was no system in place for this study. Instead, about 8800 students responded to an online questionnaire (Ishii & Boyer, 2016).

It was discovered that after being informed of the advantages of a waterless system, almost 84% of individuals who took part supported the suggested solution. The likelihood of saving water was shown to be the primary driver of perceived support (Ishii & Boyer, 2016). The opinions of one's peers about the suggested systems also had an impact on one's opinion of them. Peers are more inclined to support alternative garbage collection methods if they see their peers supporting them (Ishii & Boyer, 2016). It was advised that in order for these technologies to be successful, local government officials must raise awareness of them and publicise their advantages for the environment (Ishii & Boyer, 2016). Additionally, it was shown that when participants were asked if they would be ready to pay to install the technology, support for the suggested system drastically declined. When considering implementing this system in the majority of public situations, this ostensibly might serve as a barrier (Ishii & Boyer, 2016).

2.5.4. GTZ building

In order to integrate cutting-edge sanitation concepts in a public workplace, the German Federal Ministry of Education and Research supported a two-phase collaborative initiative. The German Agency for Technical Cooperation's (GTZ) main building's urine separation system was designed and built as part of the first stage programme, which ran from 2005 to 2006. After that, the Maßalsky GmbH institution's SANIRESCH programme was carried out to investigate the viability of adopting an ecological sanitation (ecosan) concept for the separate treatment and recycling of urine, brown-, and greywater from an urban office building (Winker & Saadoun, 2011). Health and hygienic issues, environmental effects, the availability of local resources, operation and implementation difficulty, financial and economic viability, and social-cultural acceptance were all taken into consideration while analysing the system's performance. Following the project's conclusion, it was determined that: (i) the system produced an end product that was hygienic and safe to use for agricultural purposes; (ii) both farmers and consumers demonstrated a willingness to use urine as a liquid fertiliser; and (iii) the project was deemed economically feasible under favourable circumstances. However, more research must be done on technologies that can improve the transportability and application effectiveness of the urine product.

2.6. Source separation of urine and its impact on the downstream WWT infrastructure

This chapter provides a comprehensive overview of urine composition and transformation under storage, and highlights the numerous benefits of source-separating and reusing human urine. Furthermore, it was emphasized that the recovery of nutrients from urine could significantly reduce our reliance on energy-intensive fertilizer production processes and finite resources. Through a review of the state-of-the-art processes for the treatment of human urine, several research gaps were identified. In particular, the understanding on impact of urine diversion in decentralised settings (with advanced wastewater treatment configurations) is still limited. As a result, the specific gaps and objectives addressed in the thesis are described in the introduction of each technical chapter.

Previous studies covered both source separation and nutrient recovery from urine for both centralised and decentralised settings (Wilsenach & Van Loosdrecht, 2004; Wilsenach et al., 2005). However, these studies focused only on the effect of urine diversion on centralised WWTPs that do not target wastewater reuse applications. In Australia, sustainable buildings are now being promoted where wastewater reclamation and reuse is one of the main features of the building’s sustainable star ratings. It is obvious that by implementing urine diversion in such a decentralised WWTP, nutrient loads to water reclamation facilities will be significantly reduced (Kummu et al., 2012; Matar et al., 2022). This study therefore, examines the impacts of implementing urine diversion at a decentralized level in the catchment of a treatment process with a similar configuration to that of the Sydney Central Park WWTP (which involves advanced configurations and membrane processes for water reuse) has never been investigated before.

For the first time we have investigated the effect of urine diversion on a wastewater treatment process with the help of BioWin v.6.2 (EnviroSim Associates Ltd., Canada). There several examples where BioWin software has been be used for such simulations (Rieger et al., 2012; Wang et al., 2012). BioWin 6.2 uses the integrated activated sludge (AS) model which is a blend of the worldwide models ASM1, ASM2d and ASM3 proposed by the IWA. The BioWin integrated AS model consists of 50 state variables and 60 process expressions. These expressions are used to describe the biological processes occurring in activated sludge systems, several chemical precipitation reactions, and gas–liquid mass transfer for six gases. In order to describe these processes 78 kinetic parameters and 54 stoichiometric coefficients are included in the BioWin AS model. It has been found to be most accurate and intensively researched biological model which reduces required calibration effort (Liwarska-Bizukojc & Biernacki, 2010; Holloway et al., 2019).

CHAPTER 3: IMPACT OF SOURCE-SEPARATION OF URINE ON EFFLUENT QUALITY, ENERGY CONSUMPTION AND GREENHOUSE GAS EMISSIONS OF A DECENTRALIZED WWTP

Summary

The impact of urine diversion on the biological treatment processes at a decentralized wastewater treatment plant (WWTP) was investigated. BioWin software was used for the simulations, and the model was firstly validated with data from a real WWTP. The simulations showed that upto 82% N, 30% P, 6% chemical oxygen demand (COD) load to the WWTP can be reduced by complete urine diversion but effluent N reduction was notable up to 75% urine diversion. Under the current WWTP operating conditions, the simulations suggest that 33% of aeration energy can be saved by 90% urine diversion. Direct N_2O and CO_2 emissions in the treatment processes can also be reduced by 98% and 25% respectively. Indirect green house gas emissions can also be reduced by 20%. Overall, the reduction in the discharge of nutrients and in the operation of blowers was found to contribute to a 22% reduction in the operating costs (on energy consumption and nutrient discharge).

3.1. Introduction

Currently, numerous wastewater treatment facilities are under pressure to enhance their effluent quality, especially associated with nutrients. Due to rapid population growth in cities like Sydney, wastewater loads to the wastewater treatment plants (WWTP) have been increasing, making nutrient removal processes harder . Hence, WWTPs are continuously adjusted and improved to meet the permissible standards. Activated sludge processes are the most widely used treatment technology all around the world, although the process is energy-intensive, as it is designed to remove organics, nitrogen, and phosphorus loads from wastewater. Municipal wastewater comprises a combination of wastes from different origins, including yellow (urine), brown (feces), and grey (kitchen and bathroom) waters (Larsen & Gujer, 1996; Wilsenach & Loosdrecht, 2002). Loads of nutrients in municipal wastewater are generally much more than that required for the growth of bacterial biomass in the activated sludge process. Therefore, biological nitrification and denitrification processes are employed to remove excess nitrogen from the wastewater (Metcalf et al., 1979). Although effective, both nitrification and denitrification processes produce N_2O (a greenhouse gas) (Winkler & Straka, 2019). Moreover, nitrification requires energy intensive aeration (BNR increases the energy utilization of the plant by 30–50%) (Winkler & Straka, 2019). Denitrification requires organic carbon and often for wastewaters with a low COD:N ratio addition of external carbon source is often needed which is expensive (A$1.14 per kg leftover nitrate eliminated utilizing methanol). Phosphorus removal from wastewater can be performed using physico-chemical methods like precipitation, biological treatment or combinations of both. (Sedlak, 1991).

The growing importance of lowering the effluent concentrations to meet the discharge standards and the increasing value given to sustainability has stimulated efforts to recover valuable resources from wastewater, including water, nitrogen, and phosphorus (Collivignarelli et al.; Guest et al., 2009; Volpin et al., 2020a). In conventional activated sludge processes most of the nutrients are stocked in biological sewage sludge and many efforts have been made to recover these nutrients (Collivignarelli et al.). However, most of these nutrient recovery processes processes are unsustainable. This has driven the recent rise of alternative sanitation concepts, like urine diversion as a decentralized strategy to manage nutrients in water treatment facilities. The majority of the nutrients in the municipal wastewater arise from urine. Several articles have been published on the composition of urine and its contribution to wastewater treatment facilities. It has been reported that urine contributes about 70-80% of the nitrogen, 45-50% of the phosphorus in the domestic wastewater (Larsen & Gujer, 1996; Larsen & Udert, 1999; Maurer et al., 2006; Lienert et al., 2007) although it makes up only 1% of the overall wastewater volume. Therefore, urine diversion and its treatment have gained popularity in the past few years and is seen as a possible option for sustainable wastewater management.

Larsen and Gujer (2001); Wilsenach and Van Loosdrecht (2003) and Wilsenach and van Loosdrecht (2006) , to mention a few, have shown through modeling that urine diversion from the wastewater can improve effluent quality, increase treatment capacity, and reduce energy consumption at an existing water treatment facility. More recently, the work of (Hilton et al., 2020) revealed that urine diversion can also reduce global warming potential. Some studies have also been aimed to produce non-odorous, highly concentrated liquid fertilizers from diverted urine and have observed a similar performance to commercial fertilizers (Udert & Wächter, 2012; Freguia et al., 2019; Volpin et al., 2020b). This makes the diverted urine a valuable resource, in addition to the environmental and economic benefits it offers to the wastewater treatment processes. Therefore, urine diversion and its treatment are gaining popularity in both academic research and applied research as a possible option for sustainable wastewater management although large-scale application is yet to pick-up. The specific objectives of this study were to investigate the impact of various percentages of urine diversion on (1) influent wastewater characteristics; (2) effluent water quality; (3) energy consumption; (4) overall cost savings; and (5) greenhouse gas emissions.

3.2. Methods

3.2.1. Description of Central Park WWTP

The WWTP investigated in this study is located in the basement of Central Park Mall, Sydney. The plant is operated and maintained by Flow Systems, Australia (a licensed water utility company). The plant receives domestic wastewater and trade waste from commercial centers (shops and offices). Raw sewage first flows to the flow balance tank from where it first passes through fine screens before reaching the biological treatment processes. The bioreactors consist of an anoxic tank, an aerobic tank, and a membrane bioreactor (MBR) tank. Ammonification and denitrification occur within the anoxic tank. In the anoxic tank a low dissolved oxygen level and suspension of the mixed liquor is maintained using a low speed submerged mixer.

Nitrification process occurs in the aerobic zone where the ammonia/ammonium is first oxidized into nitrite by the ammonia-oxidizing bacteria and then nitrite into nitrate by the nitrite-oxidizing bacteria during the nitrification process. The MBR tank contains SUEZ's Zee Weed 500 membrane cassettes. Mixed liquor flows from both the membrane tanks to the de-aeration tank (not shown in Figure 3-1), through an overflow weir before flowing back into the anoxic tank through pipework, where the biological treatment cycle continues followed by Ultraviolet (UV) disinfection and Reverse Osmosis (RO). The plant disposes its sludge wastage and RO brine into the Sydney Water sewer line. Sydney Water charges the WWTP operator for the discharge based on nutrient mass loads disposed of by the treatment plant. This treatment process train (excluding UV and RO processes) has been configured on BioWin modeling software and then calibrated under the current operating conditions of the treatment plant. It includes the following processes: (1) growth and decay of Ordinary Heterotrophic Organisms (OHOs), (2) growth and decay of methylotrophs, (3) hydrolysis, adsorption, ammonification and assimilative denitrification, (4) growth and decay of Ammonia Oxidising Biomass

(AOB), (5) growth and decay of Nitrite Oxidising Biomass (NOB). In order to describe these processes 78 kinetic parameters and 54 stoichiometric coefficients are included in the BioWin AS model (Liwarska-Bizukojc & Biernacki, 2010).

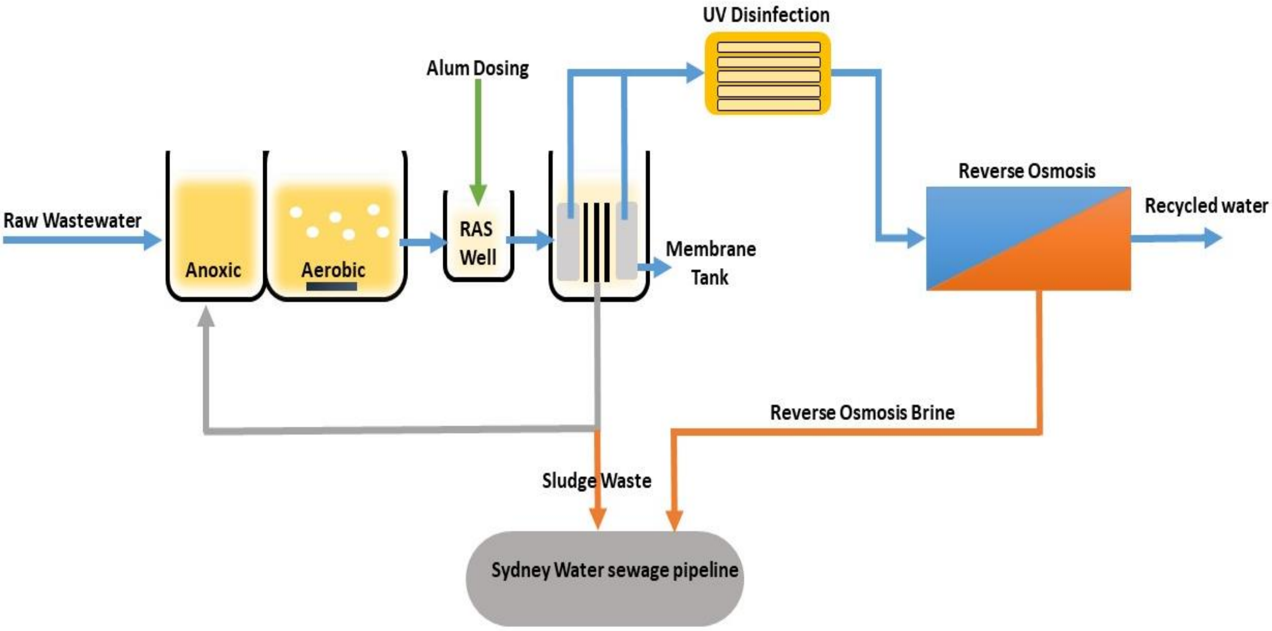


Figure 3-1. Schematic of wastewater treatment processes at Central Park WWTP.

3.2.2. Model Calibration and Validation

BioWin modeling platform was used in this investigation (Ltd., 2021). BioWin is commonly used in wastewater businesses as a process examination tool and to plan or advance WWTPs. It actualizes cutting edge models of biological and physical treatment units. It has been used widely to predict the effluent quality, energy consumption, and greenhouse gas emissions of a treatment plant (Puchongkawarin et al., 2015; Alvarez-Gaitan et al., 2016; Elawwad et al., 2017). The treatment process at Central Park was configured on BioWin according to the current operating conditions of the plant. This model was validated and compared with the results obtained from historical data from Central Park's data management system (Tables 3-1 and 3-2).

3.2.3. Model parameters.

The total suspended solids concentration was maintained around 5000 mg/L by adjusting the sludge wastage. The dimensions and operating conditions of the existing bioreactor and membrane tank were used in the model. An average temperature of 20 °C was considered. All simulations were done for steady-state without variation in influent flow rate or concentrations. A population equivalent of 2695 was assumed based on the influent COD data (using 161 gCOD/EP as per (Wilsenach & Van Loosdrecht, 2004)). Various other important paramters of the simulation are given in supplementary information.

3.2.4. Influent wastewater characteristics after urine diversion.

Different scenarios of urine diversion percentages from 0% to 100% via urine separation at the source were investigated. Wilsenach and Van Loosdrecht (2003) reported an equation based on the population equivalent, chemical oxygen demand, and nutrient loads in urine to predict the composition of wastewater after urine diversion and the same equation has been used in this study. The average volume of urine produced (including flush water) was assumed to be 2 L per population equivalent per day including flushwater (Wilsenach & van Loosdrecht, 2006). Nitrogen in fresh urine is mainly in the form of urea which hydrolyses to ammonium and bicarbonates in wastewater. It was assumed that nitrogen in urine would reach the treatment plant as ammonium. The weight fractions of wastewater assumed for this study are shown in Table 3-3. It was assumed that these weight fractions would remain the same with urine diversion. The composition of urine including flush water (calculated based on nutrient mass loads as per (Wilsenach & Van Loosdrecht, 2004)) are as follows: TN 6000 mg/L; TP 500 mg/L; COD 6000 mg/L.

3.2.5. Effluent Quality and Cost savings

The effluent quality of wastewater after the secondary treatment process (MBR effluent) and waste sludge and RO brine in the sewer line and associated costs with nitrogen and phosphorus discharge have been quantified. The percentage of nitrogen removed as biomass and denitrified with urine diversion have also been investigated. Based on the current Independent Pricing and Regulatory Tribunal (IPART) (https://www.ipart.nsw.gov.au) determination, the spendings on nutrient trade waste discharge to Sydney Water sewer pipeline were given at 1.792$ per kilogram of total Kjeldahl nitrogen and 6.792$ per kilogram of phosphorus.

3.2.6. Energy consumption

Energy consumption analysis in the secondary and MBR treatment processes at Central Park WWTP included aeration, pumping, mixing. The power requirement of mixing in the anoxic tank was based on an average value of 10 W/m^3 mixed volume (Grady et al., 1999). The energy required for pumping (including recirculation) was based on average daily flow rates, the density of the liquid, pump efficiency, and total head. The power requirement for pumping was calculated on BioWin using Eq. (1), (2) and (3).

$$P = (Q * H_{total} * g * \rho / \eta / 1000) \quad (1)$$

$$\rho = \rho_{H2O} + A * MLSS \quad (2)$$

$$H_{Total} = H_s + H_d \quad (3)$$

where, ρ = density of liquid; Q = flow rate through pipe (m^3/s); g = gravity constant (m/s^2) ; H_{total} = total system head (m); H_s = static head (m); H_d = dynamic head (m); A = constant (m^3/g) ; MLSS = mixed liquor suspended solids concentration (g/m^3) ; η = Pump efficiency. The default values of these parameters in the software were used in the analysis.

The aeration energy includes meeting the oxygen demand for the activated sludge process and membrane scouring to reduce membrane fouling. The treatment plant has employed fine diffusers for bioreactor aeration and coarse bubble diffusers for scouring, the same was used in the model as well. The cost of electricity was taken at 0.29$/kWh (Kim et al., 2019).

3.2.7. GHG emissions

Direct and indirect GHG emissions were quantified. BioWin predicts the off-gas N_2O and CO_2 emissions from nitrification, denitrification, and carbonaceous oxidation processes occurring in the plant. Indirect emissions from electricity consumption were calculated based on emission factors reported in the literature.

Table 3-1: *Actual Influent and Effluent concentrations at Central Park WWTP, Compared to Simulated Effluent Concentrations*

Parameter	**Unit**	***Actual Influent (average)**	***Actual Effluent (average)**	**Simulated Effluent (BioWin)**
Flow rate	ML/d	0.434	0.39	0.4
COD	g/m^3	1000 ± 220	37	51.15
TN	g/m^3	90 ± 15	10.7	7.83
NH_4-N	g/m^3	72 ± 14	0.21	0.22
NO_3-N	g/m^3	NA	6	5.3
TP	g/m^3	20 ± 4	3	3.4

*Source: Central Park internal database

***Table 3-2:** Actual Energy consumption of primary treatment processes (Bioreactor and MBR) compared to simulated data*

Power Categories	BioWin value (kWh/m^3)	*Actual value (kWh/m^3)
Blowers (aeration and scouring)	0.63	NA
Mixing	0.09	NA
Pumping	0.02	NA
Total	0.75	0.83

*Source- Central Park internal database

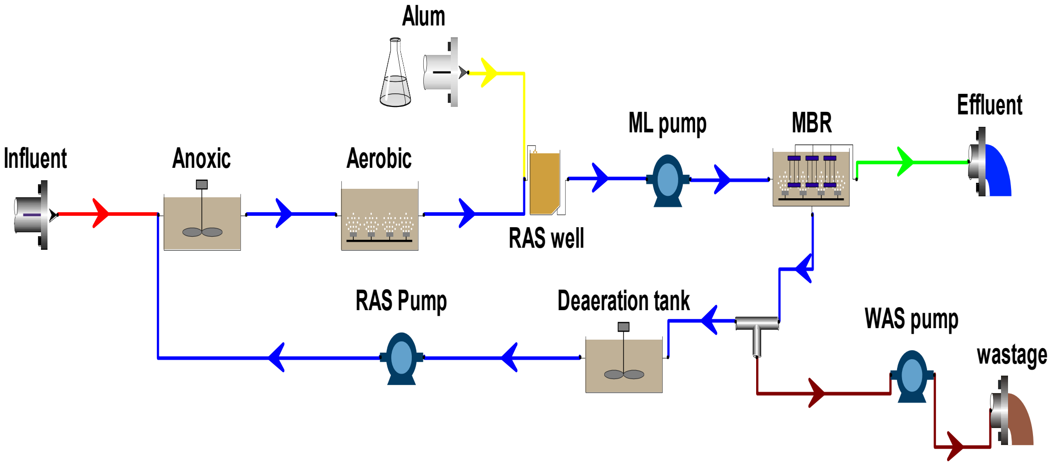


Figure 3-2. Configuration of Central Park WWTP used to simulate on BioWin (Simulation conditions: 1) Flow rate – 0.434 MLD 2) Temperature 20’C 3) MLSS – 5000 mg/L)

3.3. Results and Discussions

3.3.1. Influent wastewater characteristics

The change in wastewater characteristics with an increase in the urine diversion rate from 0 to 100% was investigated in this section. The analysis reveals that 82% N, 30% P, 6% COD, and 1.2% volumetric flow rate can be reduced in wastewater after 100% urine diversion. Table 3-3 shows a comparison of predicted changes in wastewater characteristics in this work and the figures reported in the previous studies. The predicted changes in the urine composition are in good agreement with those reported in the literature (Larsen & Gujer, 1996; Jimenez et al., 2015). However, a slightly lower reduction in phosphorus load is estimated in this study compared to the literature.

Table 3-3 - Percentage reduction in influent loads to the WWTP after complete urine diversion

Parameter	**This work**	**Jimenez et al. (2015)**	**Fewless et al. (2011)**	**Wilsenach and Van Loosdrecht (2004)**	**Larsen and Gujer (1996)**
Nitrogen	82%	80%	80%	80%	87%
Phosphorus	30%	50%	45%	50%	57%
COD	6%	0	8.6%	7.4%	12.5%
Volumetric flow rate	1.2%	not reported	not reported	12%	1.4%
Type of wastewater	High strength (municipal)	Medium strength (municipal)	Household	Medium strength (municipal)	Low strength DWW

3.3.2. Effluent Nitrogen and COD

The reference process at 0% urine diversion had an effluent concentration of 0.22 mg NH_4-N/L, 5 mg NO_3-N/L, at a sludge age of approximately 15 days. Simulation results revealed that urine diversion did not have a significant impact on effluent ammonium concentration whereas effluent nitrate concentration was decreasing with increasing urine diversion. As shown in Figure 3-3, the influent COD:N ratio increases with increasing urine diversion, and therefore the denitrification capacity in the anoxic tank increases as it results in lower effluent nitrate concentration. Ammonium concentration in the effluent was found to slightly increase at higher urine diversion which is contrary to the expectation that lower ammonium influent loads would result in correspondingly lower effluent ammonium concentrations. As previously suggested (Wilsenach & Van Loosdrecht, 2004), this may be caused by decreasing concentrations of nitrifying biomass as influent ammonium loads decrease. These results are in good agreement with effluent ammonium and nitrogen results reported previously by Wilsenach and Van Loosdrecht (2004) and Jimenez et al. (2015).

Approximately 19% of the total influent nitrogen load is removed as sludge wastage at 0% urine diversion. This removal becomes more significant at higher urine diversion rates. Figure 3-3 shows the reducing dependency on denitrification for nitrogen removal with urine diversion. At 90% separation, most of the influent N is removed by biomass growth (ending up as sludge wastage) with denitrification contributing only a small percentage of N removal. Figure 3-4 shows the effluent quality for nitrogen after urine separation efficiencies between 0% and 90%. At urine diversion above 90%, nitrogen limiting conditions occur which means that too little nitrogen is available for biomass formation. This leads to a sudden increase in effluent COD and complete process failure (as shown in Figure 3-5). This indicates that less nitrate is produced due to low ammonium oxidation although the capacity to reduce nitrate by denitrification increases because of higher COD:N ratio (Carrera et al., 2004; Wilsenach & Van Loosdrecht, 2004).

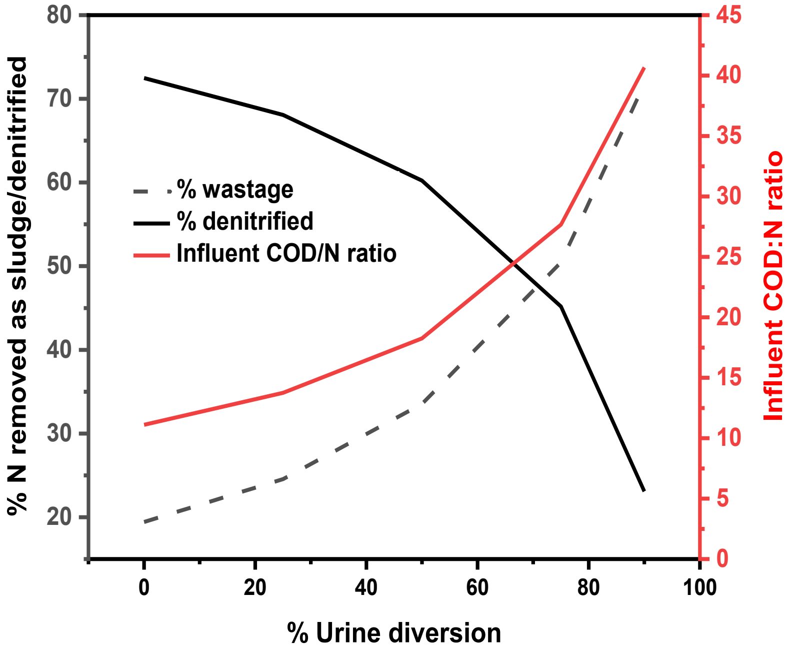


Figure 3-3 Variations in the total N removal from the system through waste sludge biomass and denitrification process at different urine diversion rates.

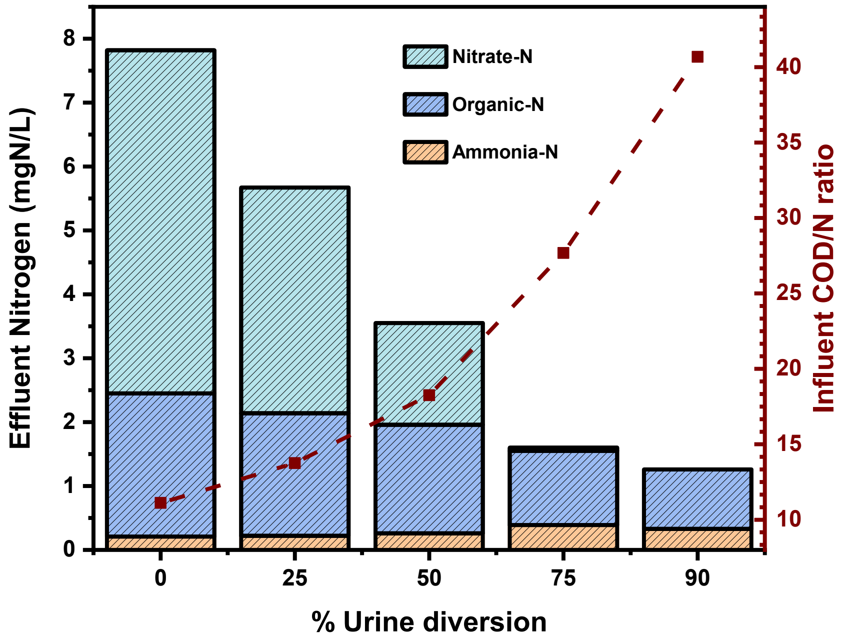


Figure 3-4 Impact of urine diversion on the effluent nitrogen concentrations.

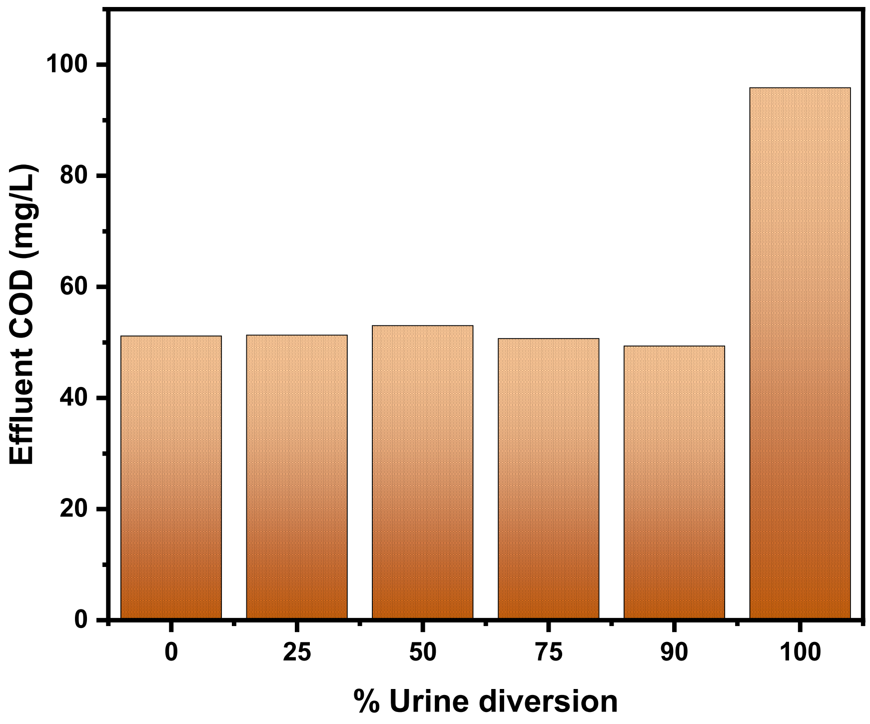


Figure 3-5 Effluent COD concentration with urine diversion.

3.3.3. Energy Consumption

Figure 3-6 shows the impact of urine diversion on the total energy consumption of the WWTP at different urine diversion rates. Results reveal that 22% of the total energy consumption of the WWTP can be saved with 90% urine diversion. Since the operating conditions are kept constant for all the simulations, there is no change in energy consumption with mixing, pumping, and membrane scouring. Energy consumption with pumping can be reduced with urine diversion by operating at aerobic-anoxic recycle rate to the minimum required for nitrate removal (Metcalf et al., 1979; Vance et al., 2015). In this study pumping accounts to only 2.5% of the overall energy consumption (Figure 3-6). Hence, the operating condition is kept constant and changes in pumping energy is neglected. As such, if only aeration is considered, this represents about 33% of savings in the energy required for the aeration for biological nitrification, which is quite significant. The aeration requirement is the major energy component impacting the energy balance of the WWTP (Drewnowski et al., 2019). As urine diversion increases, the energy required for aeration (biological nitrification) decreases because of reduced oxygen uptake rate for nitrification with urine diversion. However, the oxygen uptake rate for carbonaceous oxidation is slightly increased due to a higher COD/N ratio with urine diversion. Part of the COD is oxidized by using nitrate as an electron donor. Oxidation via nitrate is intrinsically less efficient, as 8 electrons are involved in the ammonia-to-nitrate oxidation by O_2, but only 5 of these are available for carbonaceous oxidation as nitrate is reduced to N_2. Previously, it was reported that there is no further reduction in aeration energy requirement after more than 40% urine diversion, as most of the aeration energy requirements in that study were driven by consumption of carbonaceous material (Jimenez et al., 2015). However, it is expected that as long as denitrification occurs there would be an effect on oxygen uptake rate. Hence, in this study, we find that even for high percentages of urine diversion, aeration energy is reduced.

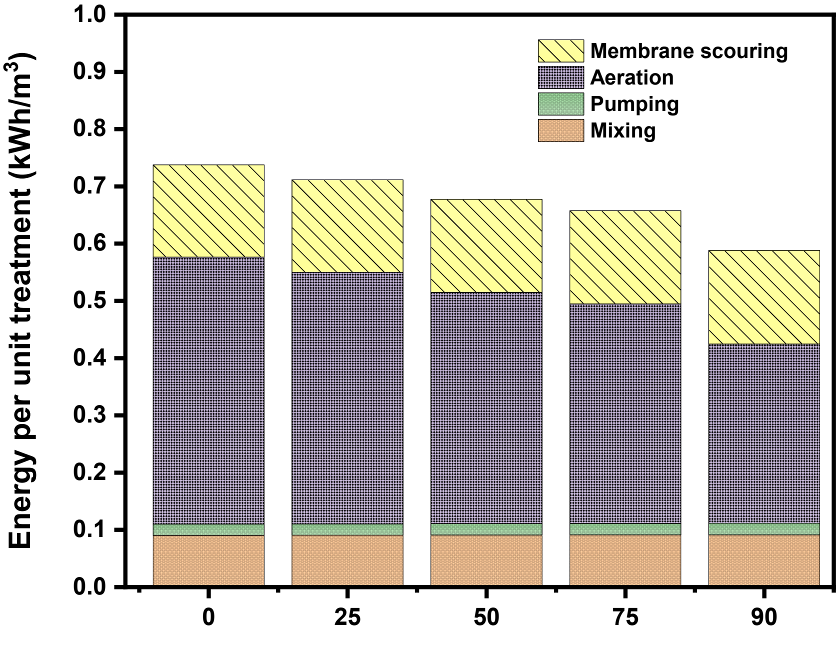


Figure 3-6 Impact of urine diversion on the total energy consumption of the WWTP as simulated for the Sydney Central park WWTP with a capacity of 0.434 MLD.

3.3.4. Overall cost savings

The effect of urine diversion on the operating expenditure associated with nutrient discharge to Sydney Water sewage pipeline and energy consumption are presented in Figure 3-7. The cost analysis revealed that 90% of urine diversion could reduce about 22% of the operational expenditure cost with energy consumption alone contributing up to 53% of this total cost savings (Silva et al., 2016; Jin et al., 2018). All of the influent phosphorus is found as biomass or chemical precipitate or reverse osmosis brine in the Sydney Water sewer pipeline. Therefore, the WWTP benefits from the phosphorus which has been diverted with urine (as the phosphorus discharged into sewer pipeline is reduced). Apart from the nitrogen released via denitrification, the rest of the total N is discharged in the sewage pipeline. However, Sydney Water does not charge for nitrate discharge to its sewer line and hence, urine diversion does not affect the expenditure of the plant on nitrogen much.

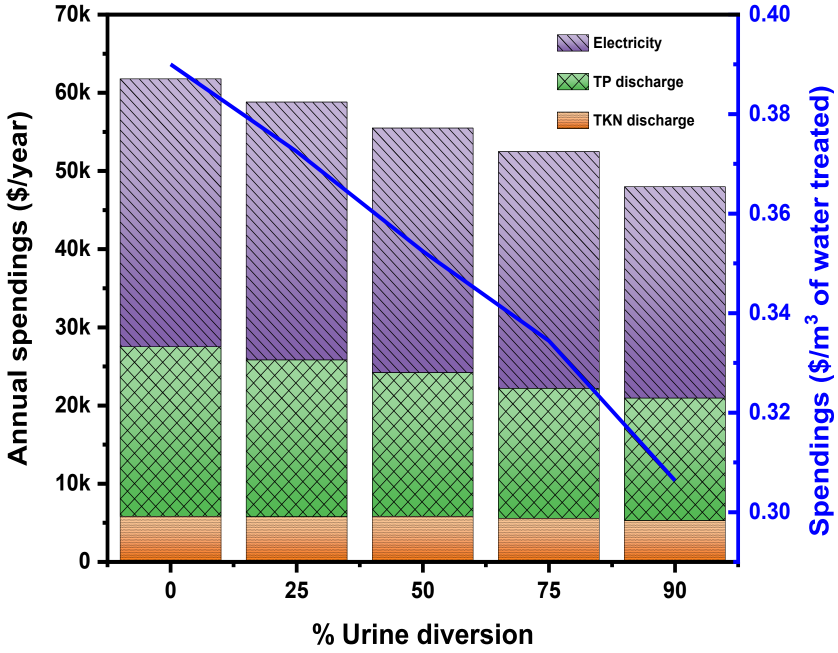


Figure 3-7 Impact of urine diversion on cost savings (TKN, TP discharge, and electricity consumption).

3.3.5. Greenhouse gas emissions

Figure 3-8 represents the impact of urine diversion on direct and indirect GHG emissions from a WWTP. At the reference scenario (0% urine diversion), the direct N_2O and CO_2 emissions from the plant have been estimated at 46.4 $kgCO_{2e}/d$ and 464 $kgCO_2/d$ respectively while the indirect emissions through energy consumption are 268 kg CO_{2e} / d. Simulation results revealed that with urine diversion, direct emissions (produced from nitrification-denitrification and COD oxidation processes) significantly reduced, while the indirect emissions due to energy consumption reduced only slightly. This significant reduction in the direct emissions from the plantover is due to reduced influent nitrogen and a slight reduction in COD load coming to the WWTP. With reduced nitrogen loads, the mass of nitrogen undergoing nitrification and denitrification reduces which in turn reduces the N_2O emissions from the WWTP (Vance et al., 2015; Igos et al., 2017; Yoon et al., 2017; Cui et al., 2019; Wang et al., 2019). A similar effect on the N_2O emissions was reported earlier under various nitrogen loading (Yang et al., 2009; Massara et al., 2017). With 90% urine diversion, it is observed that 98% of N_2O, 25% CO_2, 20% indirect emissions from the plant are reduced.

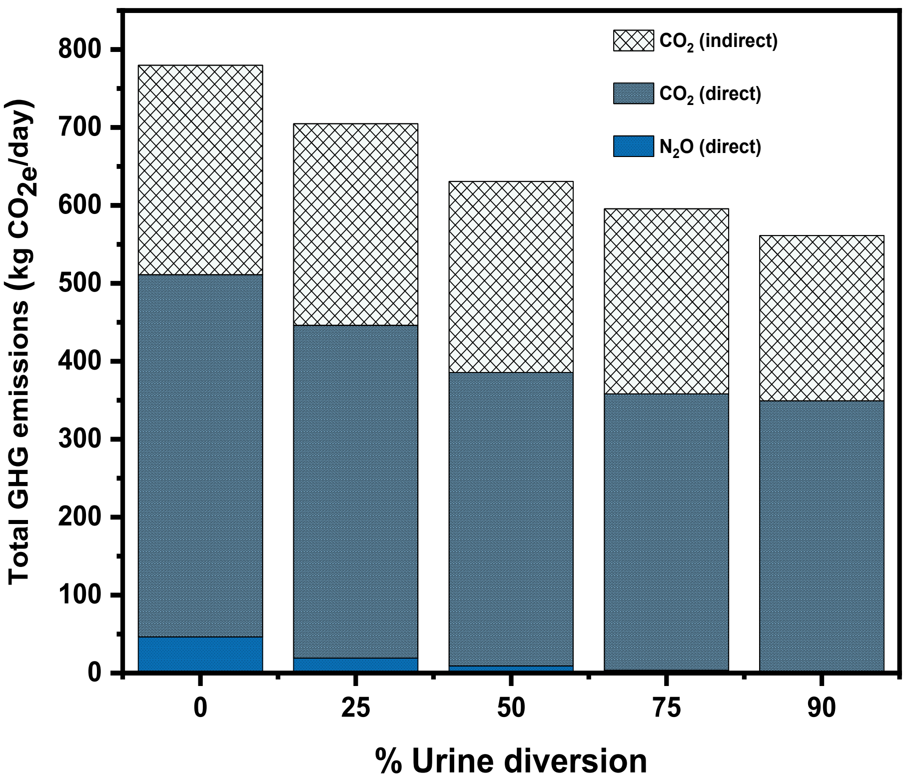


Figure 3-8 Impact of urine diversion on greenhouse gas emissions.

3.4. Conclusions

A decentralized wastewater treatment plant like Sydney Central Park would experience a clear environmental and economic advantage if urine is collected and treated separately. Simulation shows that urine diversion can reduce up to 82% of nitrogen, 30% of phosphorus loads reaching the WWTP although it reduces the COD by only 6%, and the volumetric flow by 1.2%. At urine diversion higher than 50%, the influent COD/N ratio significantly increases to the point where the influent nitrogen is adequate only for biomass growth and hence avoiding the necessity of denitrification processes. Although the urine diversion had little effect on effluent ammonium concentration, the nitrate concentration decreased with increasing urine diversion. At urine diversion above 75%, the effluent total nitrogen concentration did not show any further significant improvements. The decrease in effluent loads would be more significant where wastewater with a low COD:N ratio has to be treated, which is less favorable for N removal. 90% of urine diversion reduces the energy required for aeration by up to 33% and even higher energy savings are expected after optimizing the operating conditions. Urine diversion can significantly reduce direct N_2O and CO_2 emissions by 98% and 25% respectively while indirect emissions for energy consumption are reduced by 20%. Finally, urine diversion can save about 22% of the WWTP operating costs and about 52% of this savings is due to reduced energy consumption for aeration during biological nitrification. These results provide a pathway towards more sustainable wastewater management which is one of the main criteria to obtain high star ratings for commercial buildings in Australia. This would encourage developers to promote urine diversion especially in decentralised settings. This study did not consider the treatment of diverted urine. The technologies to recover nutrients from urine are still being optimized and their costs for the purpose of producing renewable fertilizers are not available in the literature. Similarly, there is no agreement on the likely price range of urine-derived fertilizers. Therefore, a full economic analysis for urine diversion is not possible until urine processing technologies are further developed and optimized. Hence, future research maybe needed in this direction to understand the full economic analysis for urine diversion.

CHAPTER 4: IMPACT OF SOURCE-SEPARATION OF URINE ON TREATMENT CAPACITY, TREATMENT PROCESS, AND CAPITAL EXPENDITURE OF A DECENTRALIZED WWTP

Summary

In this study, the impact of urine diversion on treatment capacity, treatment process, and capital costs of a decentralized wastewater treatment plant (WWTP) was investigated using BioWin software for simulations. The simulations showed that with 75% urine diversion, the treatment capacity of the WWTP can be almost doubled compared to 0% urine diversion although increase was not very significant for urine diversion above 40%. With 75% or more urine diversion, it was found that the current complex treatment process can be replaced with a simple aerobic membrane bioreactor (MBR) to produce the same effluent quality, with significant reduction in the plant footprint. Anaerobic membrane bioreactor followed by adsorption process can also be an alternative process, although further investigations are needed to understand the feasibility of this approach. Replacing the treatment process with a simple aerobic membrane bioreactor can save up to 24% capital costs mostly by reducing the space requirement. Sensitivity analysis revealed that by operating the bioreactor at higher MLSS concentrations (9 g/L instead of 5 g/L) could help increase the WWTP treatment capacity by about 3.5 times at 75% urine diversion. Hence, urine diversion (until nitrogen limiting conditions occur) can increase the treatment capacity of an existing wastewater treatment plant and reduce the capital expenses on space requirement of the treatment plant.

4.1. Introduction

Water is an essential natural resource, which is crucial to life, work, food security, and sustainable economic advancements (Barnett & Morse, 2013; Dorji et al., 2022). However, this resource has come under stress due to growing human population, urbanization and climate change. This demands improved management to safeguard water security and to protect receiving water bodies from environmental pollution (Metcalf et al., 1979). Water security serves different purposes, including protection of environmental flows and consequent support of associated ecosystems, provision of water for drinking and irrigation, and uptake of wastes through abiotic/biotic cycling (Carey & Migliaccio, 2009). Appraisals demonstrate that 60% of the world will live in urban areas by 2030 (Division, 2008). With a growing population size, there is a more prominent freshwater demand, and consequently, increasing volumes of wastewater are produced, particularly in urban areas (Sherbinin et al., 2007). The sustainability of basic water resources, especially in urban areas, relies on efficient wastewater management.

Sydney has been experiencing a rapid population growth rate. The Metro Strategy (2005) anticipates that Sydney's population will continue to grow at an average annual change of 1.85%. The population is about 5.3 million in 2020 and is predicted to reach 7.7 million by 2050. This is expected to increase the volumes of wastewater proportionally. Hence, in the future, it is expected that higher volumes of wastewater will need to be treated by the wastewater treatment plants (Carey & Migliaccio, 2009; Teklehaimanot et al., 2015). Commissioning additional wastewater treatment plants would increase space and volume requirement per unit treatment of wastewater. Availability of space especially in urban areas is limited, and land is expensive and would substantially contribute to capital expenditure (Nabarro & Smart, 1978; James, 2016; Gurran et al., 2020). Hence, alternative ways to treat wastewater that can increase the capacity of the existing wastewater treatment plants without the need of additional land or space are essential.

In a biological wastewater treatment processes, nitrification is often the rate-limiting step that determines the space necessities due to its slow rate and sensitivity to inhibitory factors like temperature, pH and toxic compounds (Henze et al., 2002; Zhou et al., 2020). Inefficient nitrification causes process failure in activated sludge processes by producing higher effluent ammonia and total nitrogen concentration due to the slow growth rates of nitrifiers (Zhou et al., 2020). Besides, low carbon availability in domestic wastewater demands the addition of organic chemicals, and thus increases the operating expenses to enable complete nitrogen removal from wastewater (Sun et al., 2010). Moreover, nitrogen removal requires high energy inputs in the form of aeration (for nitrification) and high recirculation ratios (to enable denitrification). Denitrification step also increases the space and volume requirement of the treatment process. In summary, improving nitrogen removal to levels that are adequate to decrease eutrophication is challenging because of limited land availability, and high capital and operational expenditure. Alternative approaches for nitrogen removal like anammox and ion-exchange to remove nitrogen from side streams have been developed (Ma et al., 2016; Wang et al., 2017). Nitrogen removal by Anammox process is at present applied in more than a hundred full-scale installations for treatment of anaerobic digestion reject water (Van Hulle et al., 2010; Lackner et al., 2014; Lotti et al., 2015). However, mainstream applications of the anammox process face obstacles due to slow growth rate and high sensitivity to low temperature conditions and necessity of a certain influent NO_2^--N/NH_4^+-N ratio (Morales et al., 2015; Tomaszewski et al., 2017). Ion exchange processes to remove nitrogen from wastewater are technically and economically unfeasible (Zhou et al., 2020). Hence, there is a convincing need to develop new technologies for stable and effective nitrogen removal that require less space than the current removal processes.

It has been reported widely that urine contributes approximately 70-80% of the nitrogen, 45-50% of the phosphorus in domestic wastewater, although it makes up only 1% of the overall wastewater volume (Wilsenach & Van Loosdrecht, 2004; Maurer et al., 2006; Randall & Naidoo, 2018; Badeti et al., 2021). Various source separation methodologies (like waterless urinals or no-mix toilets) have been tested which offer a new and simple technique for collecting urine within office blocks and other commercial buildings (Gundlach et al., 2021). Also, reusing of nutrients at the source offers a more economical and natural strategy for fertiliser production since minimal energy is required and waste streams are converted to valuable products (Flanagan & Randall, 2018). Hence, several approaches have been proposed over the years to efficiently recover nutrients from source separated urine (Maurer et al., 2006; Udert & Wächter, 2012; Freguia et al., 2019; Volpin et al., 2020a; Volpin et al., 2020b). In addition, waterless urinals offer an excellent way to save substantial amount of water which is generally consumed for flushing. Simultaneously, the nutrient loads to wastewater treatment plants are significantly reduced by implementing source separation of urine (Kvarnström et al., 2006; Vinnerås & Jönsson, 2013; Almuntashiri et al., 2021). Therefore, urine diversion and its treatment have gained popularity in the past few years and are seen as a possible option towards efficient nutrient recovery and sustainable wastewater management. Previous studies taken by Wilsenach and Van Loosdrecht (2004), Wilsenach and van Loosdrecht (2006), Chipako and Randall (2020a), Badeti et al. (2021) and Hilton et al. (2020) to mention few, have shown that urine diversion from wastewater can unlock treatment capacity, reduce effluent nitrogen concentration, and reduce the energy consumption of centralised wastewater treatment plants. In our previous work (Badeti et al., 2021), we have investigated the effect of urine diversion on effluent nitrogen, energy consumption, and greenhouse gas emissions of Sydney Central Park WWTP (a decentralised wastewater treatment plant that involves membrane treatment processes for water reuse). Our findings in the previous work (Badeti et al., 2021) have revealed that 33% of the aeration energy and 25% of the total greenhouse gas emissions could be reduced by diverting about 90% of the urine.

In this work, we have extended our investigations to study the effect of urine diversion on the treatment capacity, treatment process and capital costs of an existing wastewater treatment plant using BioWin modelling software. Recent efforts to change existing wastewater treatment plants to be energy neutral or positive has resulted in the development of anaerobic MBR (usually known as AnMBR). Since aeration is not required, AnMBR consumes less energy than aerobic processes (Liu et al., 2020). Additionally biogas is produced which can be converted to electricity. However, poor nitrogen removal is one of the disadvantages of this process which can add complexities to the post treatment (Dvořák et al., 2016). However, with urine diversion a major portion of nitrogen is diverted at source which avoids the need for nitrogen removal (Badeti et al., 2021). The feasibility of incorporating aerobic MBR and AnMBR processes with urine diversion has been investigated in this study. Most of the treatment plants employ secondary clarifiers or membrane processes to remove MLSS from the biological treatment processes (Metcalf et al., 1979; Henze et al., 2002). MLSS concentration in the bioreactor directly determine the solids loading rate to the secondary clarifier (James et al., 2015) or the membrane fouling propensity in the case of a membrane bioreactor (MBR) (Schwarz et al., 2006; Kawasaki et al., 2011). Hence, operating at a constant MLSS concentration is generally a standard practice for wastewater treatment plants to keep the clarifier in safe operation or reducing MBR membrane fouling propensity. However, another study (Wilsenach & Van Loosdrecht, 2004) showed that operating at a constant MLSS concentrations limits the potential to increase the capacity of a wastewater treatment plant when the urine diversion is increased beyond 40% due to insufficient growth of nitrifying bacteria. MLSS concentration cannot be increased too much to provide sufficient bacterial growth in the bioreactor when clarifiers are used to separate suspended solids. However, in the case of MBR processes, the treatment capacity of the bioreactor can be enhanced by increasing the total membrane area to operate at lower hydraulic retention time (HRT) and increase MLSS concentrations but maintaining similar membrane flux or even at lower flux so that membrane fouling is controlled. Hence, a sensitivity analysis was performed to investigate the treatment capacity when operated at higher MLSS concentrations. Finally, a comparative economic assessment was conducted for these three different configurations.

4.2. Methodology

4.2.1 Description of the wastewater treatment plant

The WWTP investigated in this study is located in the basement of the Sydney Central Park located at Ultimo, in front of the University of Technology Sydney Ultimo campus. The plant receives mixture of domestic wastewater (from apartments and public toilets) and trade waste from commercial centres (shops and offices). Wastewater produced is treated through biological and membrane treatment processes (including MBR, UV, and RO). A complete description of the treatment processes at Central Park WWTP has been described in our previous submission (Badeti et al., 2021). In our previous study, we have configured these processes (excluding UV and RO) on BioWin modelling platform, calibrated and validated with real data (effluent water quality and energy consumption) of Central Park WWTP. The same model has been used to extend our investigation on the following parameters in this study.

4.2.2. Treatment Capacity

The potential increases in the treatment capacity of the existing decentralised WWTP (i.e, increased number of people connected from the current capacity of 1.0 MLD) with urine diversion were investigated. The target effluent concentrations include ammonium nitrogen NH_4-N = 1 mgN/L and total nitrogen TN = 7.8 mgN/L from their influent concentrations of NH_4-N = 72 mgN/L and total nitrogen TN = 90 mgN/L (Badeti et al., 2021). The increase in flow rate was determined through iteration. Mixed liquor suspended solids (MLSS) in the reactor was kept constant, and the reference effluent concentration was not exceeded. With increasing influent flows, the SRT was decreased to maintain constant MLSS concentration in the bioreactor. The increase in treatment capacity was compared to the capacity of the reference scenario. The number of people connected was calculated based on the inflow rate (0.15 m^3/d per EP).

4.2.3. Treatment Processes

Two different treatment processes are proposed in this study with a lesser footprint compared to the reference scenario. These treatment configurations were modelled on BioWin to determine the effluent nitrogen concentrations produced under different urine diversion. The inflow rate (0.434 MLD) and temperature (20°C) were the same for all three scenarios. The brief description and total volume of the treatment process are described in Table 4-1. The MLSS were maintained at 5000 mg/L in configurations A and B. Since, configuration C was an anaerobic system, the total suspended solids were increased to 10,000 mg/L for configuration C. The fate of nitrogen and effluent nitrogen concentration in both configurations were monitored for various percentages of urine separation.

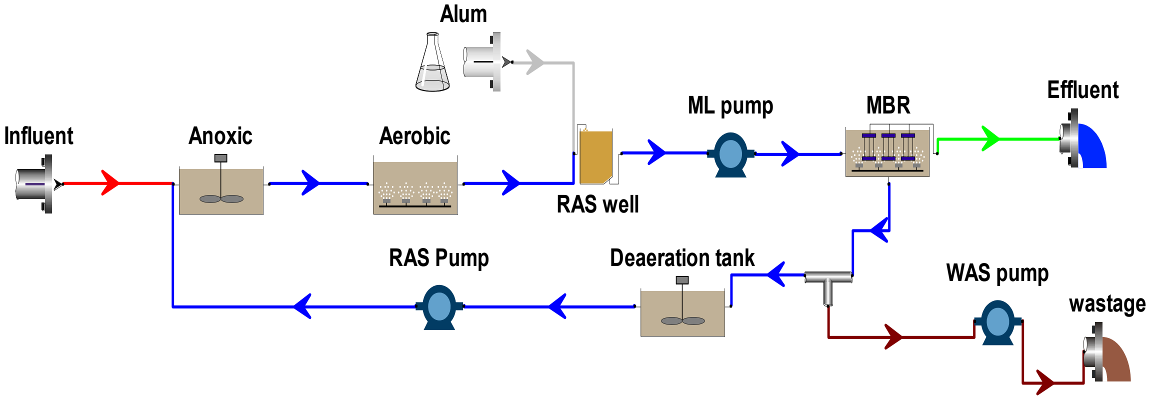


Figure 4-1(a)

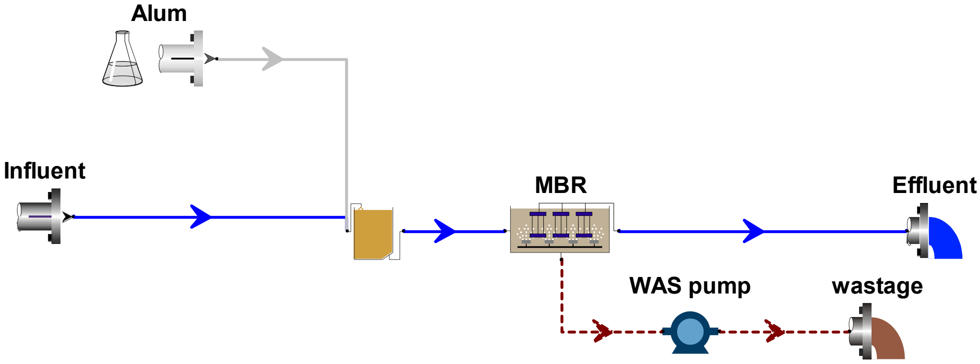


Figure 4-1(b)

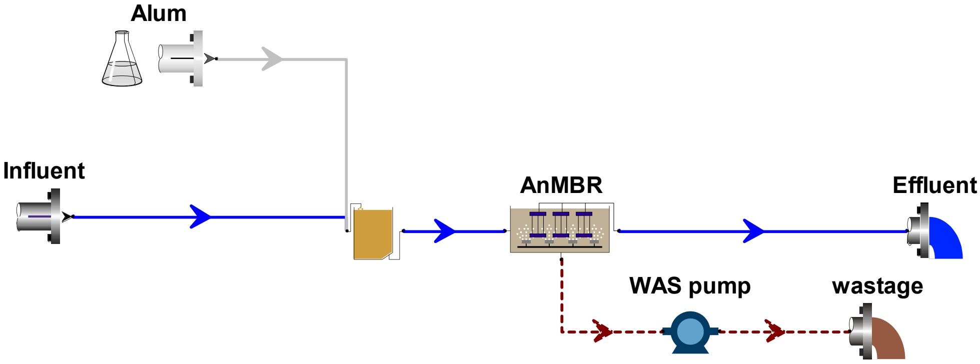


Figure 4-1(c)

Figure 4-1 (a) Schematic of current treatment process at Central Park (b) Schematic diagram of Alum-MBR (c) Alum-AnMBR proposed in this study. These processes are generated by BioWin.

Table 4-1 Different treatment processes compared

Configuration	Description	Total Volume (m^3)
A	Anoxic - Aerobic - Alum - Membrane tank (See Figure 4-1a)	660
B	Alum - Membrane bioreactor(See Figure 4-1b)	500
C	Alum - Anaerobic Membrane bioreactor (See Figure 4-1c)	500

*The effect of urine diversion on configuration A has been investigated in our previous study (Badeti et al., 2021). In this study we have investigated configuration B and C.

4.2.4. CAPEX estimations

The CAPEX estimation includes the cost of space and civil works. The cost of commercial space in Central Park was assumed at AU\$1000/m^2/year as provided by the Central Park's asset management office and this value has been used for space cost. The cost of civil work are based on values reported in (Fernández-Álvarez et al., 2014; East, 2018) and it includes costs of process units such as reactors, pumps and membranes. Other components such as the cost of the membrane module, valves, and pipeline have not been included in this study. More details on the cost analysis has been provided in supplementary information (Fig S1). The annualized CAPEX cost (A\$/yr, CAPEXa) was determined at an interest rate of 6% and plant availability of 0.95 for a 20-year plant lifetime (i.e. n). The CAPEXa cost in A\$/yr is therefore calculated based on the following equation:

$$\text{CAPEXa} = ((\text{Total CAPEX cost}) * i * (i + 1)^n) / ((1 + i)^n - 1) \qquad (1)$$

4.2.5. Sensitivity Analysis

The potential variations in the treatment capacity of the existing decentralised WWTP (i.e, increased number of people connected) with urine diversion, when operated at various MLSS concentrations were investigated. These effluent concentrations were taken as target values (NH_4 = 1mgN/L and TN = 7.8 mgN/L). The increase in flow rate was determined through iteration. Total suspended solids in the reactor were maintained at 7.5 g/L and 9 g/L, and the reference effluent concentration was not exceeded. With increasing influent flows, the SRT was decreased to maintain constant MLSS. The increase in treatment capacity was compared to the capacity of the reference scenario.

4.3. Results and Discussions

In this section, the effect of increasing urine diversion rates on the following parameters has been discussed in detail.

4.3.1. Treatment capacity

Figure 4-2 shows the potential increase in treatment capacity with increasing urine diversion at source up to 75%. The treatment capacity for the reference scenario at 20°C was 1MLD (i.e. 6666 population equivalent). For 10% urine diversion the aerobic zone was increased by 100 m^3 (with urine diversion the required anoxic tank volume was reduced and aerobic tank was increased respectively so that total volume (anoxic + aerobic) remained constant), which made a capacity increase of 40% possible. For 45% urine separation and higher, the complete anoxic zone became an aerobic zone. The maximum increase in influent capacity relative to the reference flow rate, with urine separation and increasing the aerobic zone at the expense of the anoxic zone, was 92%. At higher urine separation percentages, the rate of capacity increase with urine separation was relatively small. From 0 to 15% urine separation the treatment capacity was limited by effluent nitrogen concentration. For more than 15% urine separation effluent ammonia was the limiting factor. The decrease in nitrifiers made the maximum allowable ammonium concentration limiting, although the total nitrogen effluent concentration still decreased.

Previously, Wilsenach and Van Loosdrecht (2004) evaluated the effect of urine separation on the treatment capacity of a centralised BCFS (Biological–chemical phosphorus and nitrogen removal) treatment process operated under Dutch climate and working conditions. The authors observed a similar effect on treatment capacity with urine separation. However, the authors observed no further increase in the treatment capacity for urine diversion above 40% due to insufficient growth of nitrifiers which led to higher effluent ammonia concentrations.

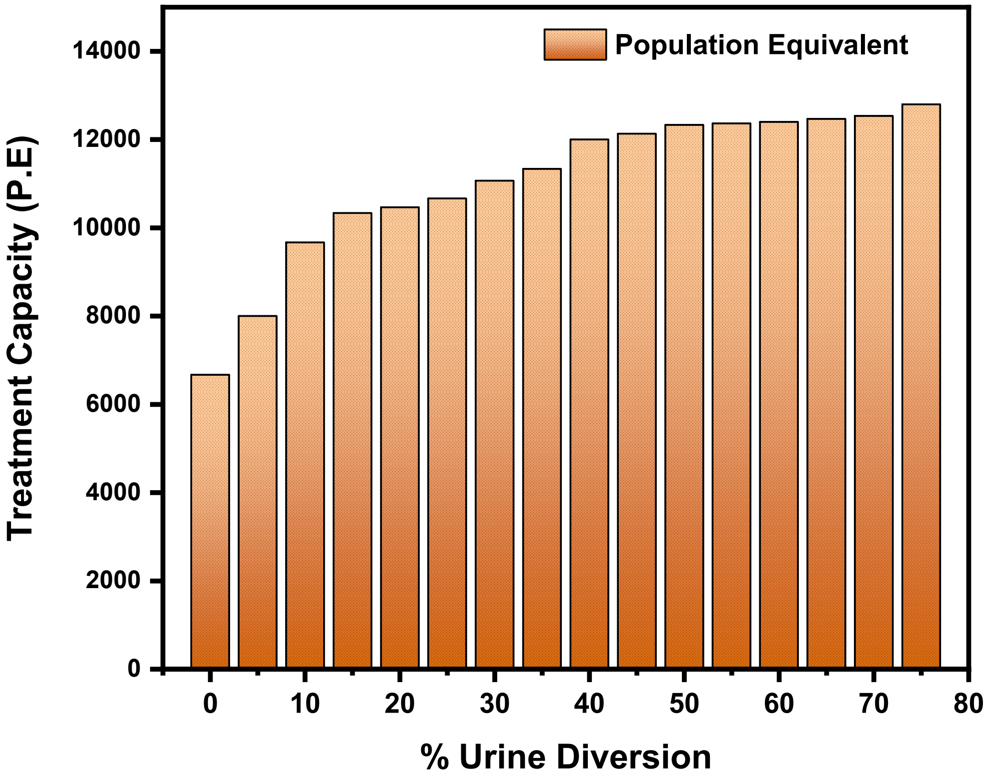


Figure 4-2. Effect of urine diversion on treatment capacity of Central Park WWTP (~6666 EP). MLSS = 5000 mg/L, effluent total nitrogen and ammonia targets of 7.8 mgN/L and 1mgN/L respectively.

4.3.2. Process Configuration

In this section, the possibility of modifying the reference or existing configuration of the Central Park WWTP to a simpler process has been evaluated. The effect of urine diversion on effluent nitrogen concentration in process A has been reported in our previous study

(Badeti et al., 2021) while processes options B and C have been investigated in this study. Figure 4-3 shows the effluent nitrogen concentrations when no urine is diverted from the wastewater streams. The effluent nitrogen loads are significantly low (< 7.8 mgN/L) in configuration A since most of the nitrogen is removed via denitrification. But in configuration B and C the concentrations of effluent nitrogen are 58 mgN/L and 78 mgN/L respectively which are much higher than the permissible limit of 7.8 mgN/L without any urine diversion.

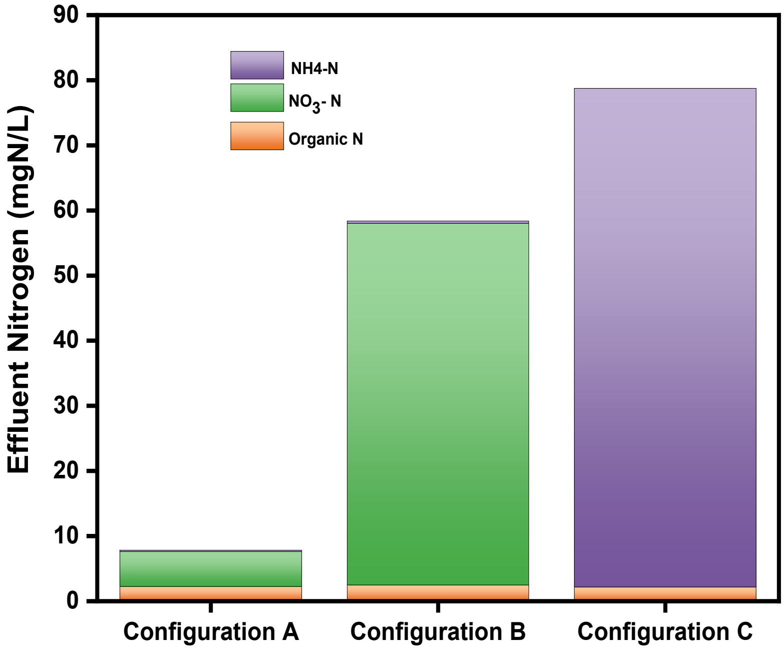


Figure 4-3. Comparison of the effluent nitrogen concentration in the effluent in the three different processes without any urine diversion. The influent concentration of the raw wastewater assumed are NH_4 = 72 mgN/L, NO_3 < 0.1 mgN/L, organic-N = 18 mgN/L

4.3.2.1. Process Configuration B

Figure 4-4 and Figure 4-5 show the effluent nitrogen concentration and mass balance of nitrogen (diverted, denitrified, removed as sludge wastage, effluent) in Configuration B. As shown in Figure 3 and 4, at 0% urine separation, about 70% of the influent nitrogen is found as nitrate in the effluent and this is due to the absence of an anoxic tank in configuration B unlike in A where most of the nitrogen is denitrified in Configuration A. By increasing urine diversion, the influent nitrogen load is reduced. The influent wastewater COD/N ratio however increases with urine diversion reaching a point where most of the low influent nitrogen is mostly removed as sludge biomass and hence the necessity of an anoxic tank to perform denitrification process is not needed because nitrogen concentration below 7.8 mg/L can be easily achieved. Previously, various studies have observed similar results treating wastewater with a high COD/N ratio (Holakoo et al., 2007; Khan et al., 2011; Rezakazemi et al., 2018). Hence, for more than 75% urine diversion, process B was sufficient to meet the effluent nitrogen concentration targets.

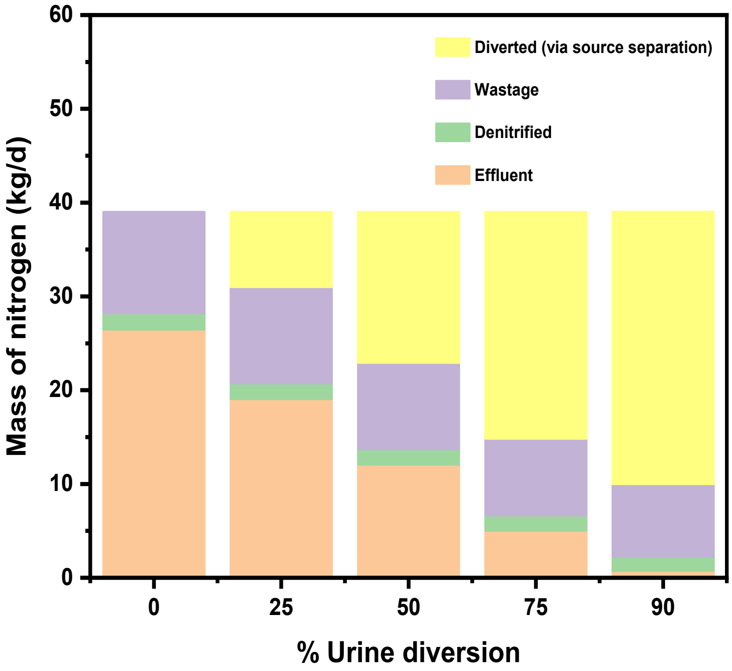


Figure 4-4. Fate of nitrogen in Scenario B with urine diversion. WWTP capacity of 1 MLD or 6666 PE. The influent concentration of the raw wastewater assumed are NH_4 = 72 mgN/L, NO_3^- < 0.1 mgN/L, organic-N = 18 mgN/L

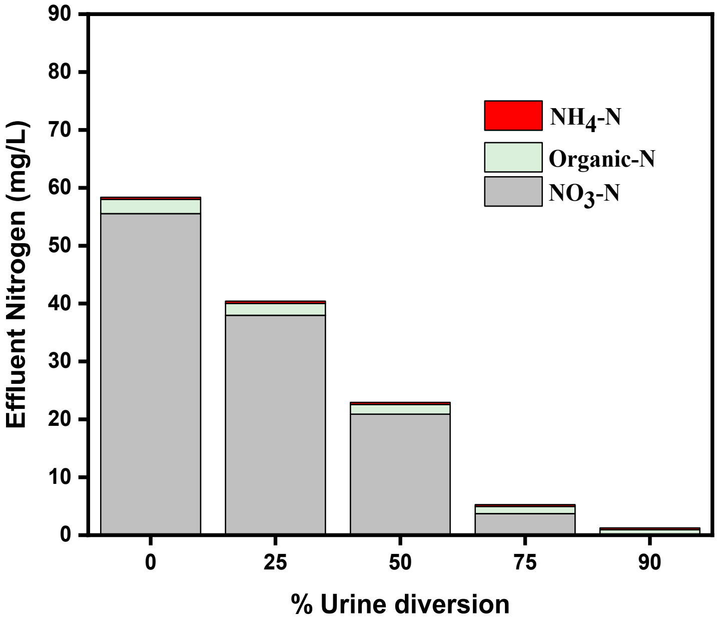


Figure 4-5. Effluent nitrogen concentration in Scenario B with urine diversion. The influent concentration of the raw wastewater assumed are NH_4 = 72 mgN/L, NO_3^-< 0.1mgN/L, organic-N = 18 mgN/L

4.3.2.2. Process Configuration C

Figure 4-6 and Figure 4-7 show the effluent quality and mass balance of nitrogen (recovered, sludge wastage, effluent) with urine diversion in Configuration C. As shown in Figure 4-6 and 4-7 at 0% urine separation, about more than 80% of the influent nitrogen was released in the effluent as mostly ammonium. Only a minute fraction of the influent nitrogen is removed through biomass synthesis. It has been reported in the literature that while treating wastewater, the minimum proportion of COD:N:P in the wastewater to be treated should be around 250:5:1 for anaerobic treatment (Ammary, 2004). For anaerobic treatment, the necessary nitrogen and phosphorous concentrations are lower than the case for aerobic treatment because the anaerobic treatment produces less sludge biomass generally compared to the aerobic treatment process (Chan et al., 2009). Hence, only a minute fraction of influent nitrogen is removed as sludge wastage in an anaerobic system. Most of the influent nitrogen is therefore stays in the effluent.

Although with urine diversion, the influent nitrogen loads and hence the effluent nitrogen concentrations can be significantly reduced in configuration C. However, only at 100% urine the total nitrogen is within the acceptable limit of 7.8 mg/L although this is mostly in the form of NH_4-N. The disadvantage of this configuration is that it produces higher effluent ammonium and COD concentrations compared to the reference scenario which can be an issue for the reverse osmosis membrane process for advanced water treatment for water reuse. Higher concentration of organics and nutrients in the effluent stream which needs to be treated using RO operation increases the fouling potential of RO membranes A zeolite adsorbent to remove excess ammonia and organics could be added to the reference scenario as an alternative option. Previous studies have reported that adsorption treatment of AnMBR effluent by zeolite can produce similar effluent quality to that of aerobic treatment (Gu et al., 2019; Li et al., 2020). However, these processes require significant use of adsorbents including process complexities in the reuse of adsorbed ammonia. In our case however, the dependence on adsorption is reduced by diverting urine at source and makes the nutrient recovery process simpler. The proposed configuration C is one potential option that can be considered and explored. However, further research is needed to understand the feasibility of these novel integrated processes.

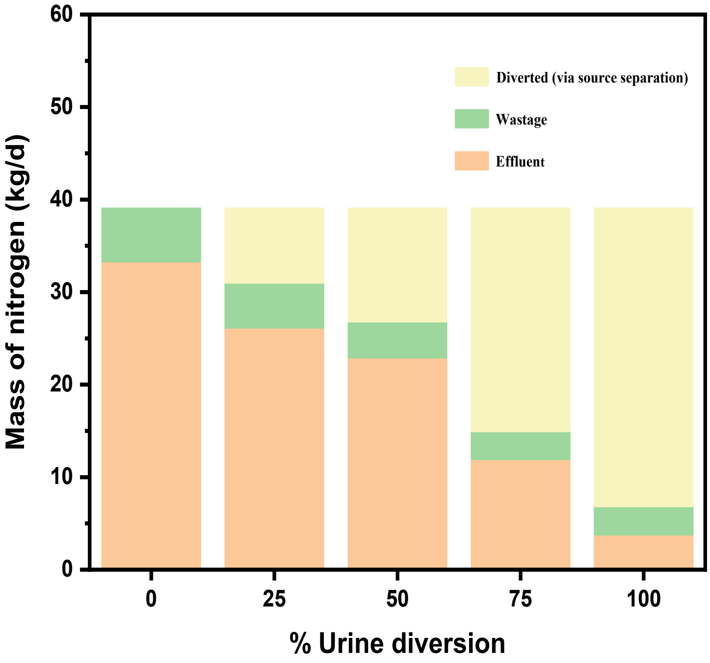


Figure 4-6. Fate of nitrogen in Scenario C with urine diversion

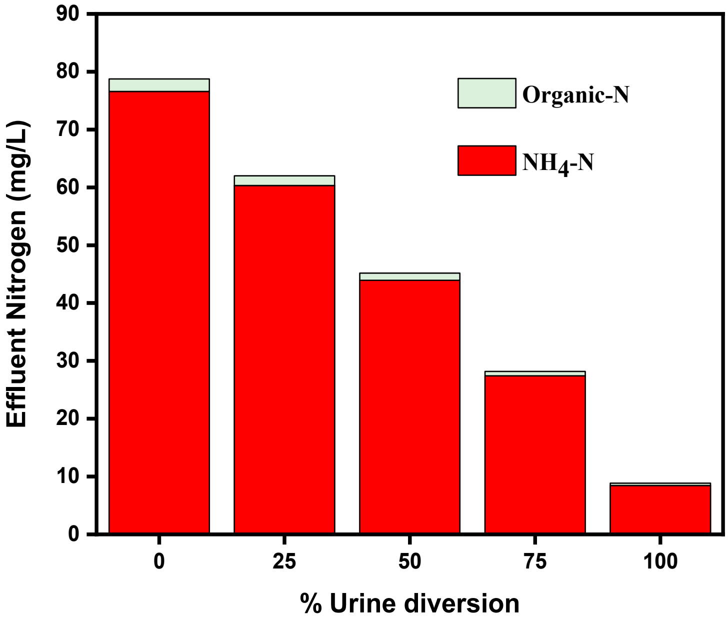


Figure 4-7. Effluent nitrogen concentration in Scenario C with urine diversion

4.3.3. Capital Costs

In the previous section it was observed that the proposed configuration B was able to produce effluent nitrogen (< 7.8 mgN/L) and ammonia (< 1mg/L) concentrations within permissible limits after 75% urine diversion without the need of anoxic tank. In this section, we compared the CAPEX costs of process A and B. Figure 4-8 shows the capital expenditure per unit treatment on space requirement and civil work for process A and B. Reference scenario (Process A) had a capital cost of 0.78 $\$/m^3$. Results reveal that 24% of the capital expenditure on space required and civil work could be saved in process B. As expected, the major factors responsible for the CAPEX of the plant are space utilisation cost. In process B, the anoxic tank and recirculation of nitrified mixed liquor are not required as most of the nitrogen is recovered with urine diversion which reduces the associated space and civil work requirements. In this particular case study, we find that most of the capital costs are with space occupancy and the contribution of civil work costs are comparatively less. However, this may not be true in the case of wastewater treatment plants which are located in areas with low land occupancy rates. Hence, these findings may not be applicable to the general wastewater community as space in the heart of Sydney are not representative of most locations.

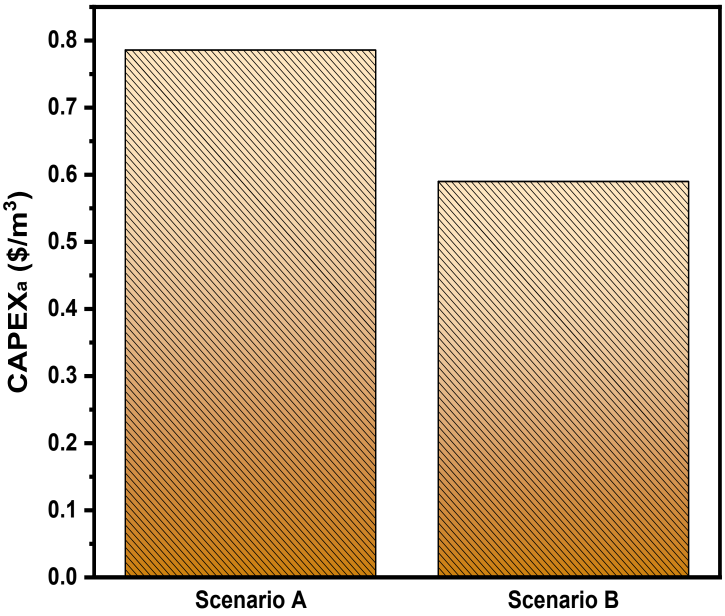


Figure 4-8. Capital expenditure of Configuration A and Configuration B (With urine diversion Configuration A can be replaced by configuration B which reduces the capital costs of the treatment process.)

4.3.4. Sensitivity analysis: Effect of sludge age on treatment capacity

The influence of MLSS concentrations on the capacity of the treatment plant has been investigated in this section under different urine diversion scenario. Earlier in Figure 4-2, it was shown that the plant capacity does not increase significantly for urine diversion beyond 40%. Figure 4-9 shows the simulated maximum treatment capacity of the plant for various percentages of urine diversion. These results show that the treatment capacity of the WWTP at higher urine diversion can be significantly enhanced by operating the aerobic bioreactor at a higher MLSS concentrations (simulated for 5, 7.5 and 9 g/L MLSS concentrations). For urine diversion over 20%, there is a significant increase in the treatment capacity when the bioreactor is operated at 7.5 g/L and 9 g/L increased from 5 g/L as simulated earlier in Figure 4-2. Operating at higher MLSS concentration increases the SRT and provides sufficient time for the nitrifiers to oxidise ammonia and produce effluent ammonia < 1 mg/L (Kos, 1998; Henze et al., 2002; Wilsenach & Van Loosdrecht, 2004). For example, at 5 g/L MLSS concentration, the WWTP capacity increases only by 85% when urine diversion is increased from 0% to 50%. However, if the bioreactor is operated at a higher MLSS concentrations of 7.5 and 9.0 g/L, the treatment capacity increases by 168% and 220%, respectively when urine diversion is increased from 0 to 50%. This treatment capacity further increases by 180% and 250% when the urine diversion is further increased from 0% to 75%, which is a very significant increase. The sensitivity analysis therefore indicates that by operating the bioreactor at higher MLSS concentration of 9.0 g/L (instead of 5 g/L) and at 75% urine diversion, the treatment capacity of the WWTP can be increased by about 3.5 times compared to the reference scenario (no urine diversion).

However, MLSS concentration did not significantly impact the treatment capacity for urine diversion below 20% and this is likely because effluent nitrogen was the limiting parameter for treatment capacity below 20%. However, it has been well reported that increasing the MLSS concentrations would decrease the critical permeate flux and lower oxygen transfer rate (Schwarz et al., 2006). It is expected that the membrane surface area and scouring air flow would be increased to obtain critical flux when operated at higher MLSS concentrations. This may increase the capital and operational expenditure of the plant. Further investigations are needed to understand the effect of operating at higher MLSS concentrations on the capital and operational expenditure of the treatment plant.

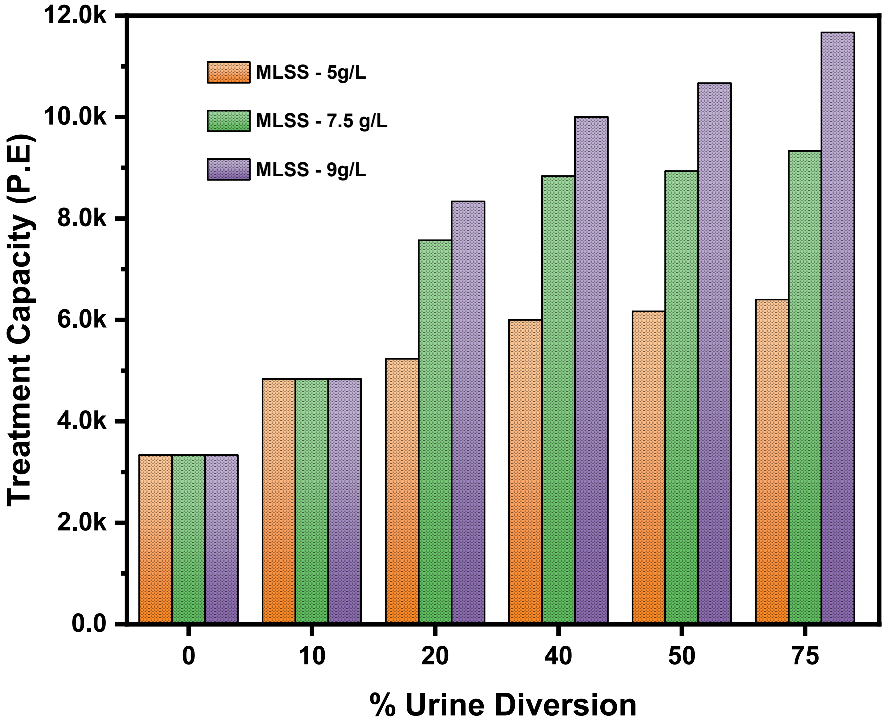


Figure 4-9. Effect of operating MLSS concentration on treatment capacity of Central Park WWTP

4.4. Conclusions

This simulation study demonstrated that urine diversion allows for higher COD loading to the existing WWTP without the need for further process modification but still meets the required effluent nutrient concentrations. Diverting urine up to 75% can help double the treatment capacity of the WWTP, although the rate of enhanced capacity becomes less significant beyond 40% urine diversion. However, the sensitivity analysis indicated that, operating the bioreactor at higher MLSS concentration can significantly help increase treatment capacity even at higher urine diversion rates. At a higher urine diversion, above 75%, it was found that the current complex treatment configuration can be replaced with a simple aerobic membrane bioreactor to produce the same effluent quality significantly reducing plant footprint. Replacing the current process with a simple membrane bioreactor would reduce about 25% of the capital costs mostly due to a reduction in space requirements. The findings of this particular study may not be applicable to the general wastewater community as space in the heart of Sydney are not representative of most locations and hence specific case studies are needed.

CHAPTER 5: ESTIMATION OF ENERGY CONSUMPTION, GREENHOUSE GAS EMISSIONS AND PROCESS DESIGN FOR TREATMENT OF SOURCE SEPARATED URINE AT A DECENTRALISED WWTP

Summary

In this study, we have calibrated the BioWin model to the high ammonia containing sidestream urine treatment conditions to estimate the energy consumption, greenhouse gas emissions and volumetric nitrification rate for source separated urine at a decentralised wastewater treatment plant. Firstly, we calibrated and validated modelling output from Biowin simulation software with lab scale urine MBR treatment process. The Biowin model was further upscaled and used to investigate the performance of a full scale urine treatment system. The simulations showed that at optimised DO set point of 3 mg/L, the energy consumption was found to be 3 kWh/kgN and greenhouse emissions of 25.6 $kgCO2_e/m^3$ at a volumetric nitrification rate of 310 mgN/L/d, which is only about 25 - 30% of the energy consumed in the synthesis of a virgin fertiliser. This study shows that the total combined energy of a MBR wastewater treatment plant (urine diverted) and urine treatment separately in a urine MBR for fertiliser recovery, is similar to the total energy demand of a conventional MBR wastewater treatment plant without nutrient recovery. However, diverting urine reduces GHG emissions and footprint of a wastewater treatment plant in comparison to the conventional process. Besides, nutrients are fully recovered through source separation of urine and treatment thereby contributing to both fertiliser economy and circular economy.

5.1. Introduction

A major part of the world food supply relies on the chemical fertilisers synthesised from non-renewable resources including energy sourced from fossil fuels (Arcas-Pilz et al., 2021). Phosphorus fertilisers are derived from phosphorus rocks and the phosphorous mining reserves are limited to only few regions of the world (Geissler et al., 2018). The resource base's extent is under question, and although demand has increased annually due to population growth and the production of biofuels, supply remains limited. Furthermore, only a few nations dominate the available supplies (Dixon, 2018). Production of nitrogen fertiliser relies on an energy-intensive Haber-Bosch process which contributes to greenhouse gas emissions (Wang et al., 2021). In recent years, fluctuations of up to 800% in the prices fertilizer products have resulted in food shortages and civil unrest in numerous developing nations (Tulsidas et al., 2019).

Due of the negative consequences and resource constraints of conventional fertilisers, viable substitutes must be needed (Maurer et al., 2006). Food intake by people is the main source of these nutrients in wastewater, and advanced nutrient removal technologies are needed at the wastewater treatment plants to remove and protect the aquatic environment from the effluent discharge. The source separation of urine and use it as a fertiliser after adequate treatment is seen as one of the options to recycle nutrients to achieve circular economy rather than the current practice of linear economy of removing nitrogen from the wastewater (Udert & Wächter, 2012). Although it contributes to only1- 2 % of the wastewater volume, urine contributes about 45-50% phosphorus, 80% of the nitrogen and 50-80% of potassium in domestic wastewater (Badeti et al., 2021; Badeti et al., 2022). Source separation of urine diversion has the potential to enhance wastewater management, recover nutrients, and minimise energy-intensive aeration at the wastewater treatment plant before being removed from the system as utilities place an increasing emphasis on sustainability(Hilton et al., 2020).

Since fertilisers have a market value, source separation and treatment of urine to convert into safe and valuable fertiliser has the potential to develop new business industry and business opportunities while at the same time contributing nutrient recycling and circular economy (Larsen et al., 2021a). The most straightforward strategy to increase urine value as a fertiliser is to create an odourless, safe fertiliser that is more widely accepted by society (Simha et al., 2021a). Direct utilisation of urine as a fertiliser has been normal in the past and also currently practised in many rural parts of the world (Larsen, 2020) however, this approach is unsuitable for urban communities. The high water content in urine (up to 96%) makes the transport to long distant agricultural fields expensive, ammonia volatilisation from urine gives unpleasant odour and also causes high nitrogen losses, and microorganisms and micropollutants present in urine potentially pose a health risk to the consumers of the agricultural products produced using urine fertiliser (Smith et al., 2020). Hence, several technologies have been explored to produce sanitised, non-odorous, highly concentrated fertiliser (reduced water volume) from source separated urine (Volpin et al., 2020a; Larsen et al., 2021b).

Udert and Wächter (2012), suggested combining thermal distillation with biological urine oxidation to create a stable and highly concentrated fertiliser solution while also removing organics and lowering pH. A sequencing batch reactor was employed in the process' first step to partially convert NH_4^+ to NO_3^- and remove foul-smelling organics. The alkalinity in the urine was also consumed during nitrification, resulting in a drop in pH from 9.2 (the usual pH of hydrolysed urine) to 5.5 - 6.5. By adjusting the HRT, the pH of the reactor was kept constant because feeding raw, hydrolysed, alkaline urine increased pH while nitrification decreased it. The second procedure was thermally evaporating the stabilised and acidified urine to eliminate water and raise the concentration of dissolved solids. Membrane treatment processes like reverse osmosis and freezing have also been used to concentrate urine and produce a liquid fertilizer. Following this, Fumasoli-Hug (2016) looked into how the pH set-point affected the system's operational costs, pathogen inactivation effectiveness, and biomass selection (Bischel et al., 2015). Similar studies have been carried out by various other researchers in different reactor configurations (Coppens et al., 2016; De Paepe et al., 2018; Volpin et al., 2020b; Jiang et al., 2021). The expenses of these systems for producing renewable fertilisers are currently being studied and optimised, hence they are not yet included in the literature. Similar to this, there is no consensus on the expected price range of fertilisers made from urine. Therefore, until urine processing technologies are further developed and perfected, a complete economic analysis for urine diversion is not possible.

Our previous studies investigated the effect of urine diversion on energy consumption, GHG emissions, footprint of Sydney Central Park WWTP and have shown that aeration energy, plant footprint can be significantly reduced (Badeti et al., 2021; Badeti et al., 2022). However, these studies did not include the energy, process design requirements of the source separated urine. Hence, it does not provide a complete evaluation of the process performance of the treatment plant when source separation is implemented. Most of the previous studies on nitrification of urine have been aiming to demonstrate conceptual feasibility either at a lab or pilot scale.

To our knowledge, there is no study in literature that modelled full scale urine nitrification systems using membrane bioreactor. Hence, for the first time in this study we have performed modelling studies on a full scale MBR system for urine treatment. While other urine treatment technologies are being studied and developed however, this study focussed on the MBR treatment as this approach was widely studied and investigated by our group at UTS. This approach may result in two MBR system running in parallel: one for WW treatment and another and much smaller one for the urine treatment to basically achieve saintisation and biological nitrification. We have also quantified the direct N_2O, CO_2 and indirect CO_2 emissions from urine treatment. Our model predicts a lesser energy consumption in comparison to some of the pilot and lab scale studies reported in literature. Studies like this aimed at full-scale optimisation are needed and important to form the basis of a business model for urine diversion. The specific objectives of this study were to 1) develop and validate a mathematical model for urine pre-treatment, 2) quantify energy and nitrate production rate for pre-treatment of urine for a full scale system, 3) quantify direct and indirect GHG emissions of a full scale system, and 4) Compare process performance of conventional and modified treatment process (which includes urine diversion).

5.2. Methods

5.2.1. Model development and theory

To treat hydrolysed source separated urine, a lab scale UF-MBRs with a total effective working capacity of 4.5 L was used. The activated sludge was taken from the Central Park Wastewater Treatment Plants, Sydney, Australia used for seeding nitrification in the urine MBR. This system was operated for 140 days, which included start-up phase and stable operation phase. More details and operation of the MBR have been described in (Jiang et al., 2021).

BioWin software and BW controller (Ltd., 2010), based on ASM2d, were used as the simulation platform in this study. BW Controller is a separate Windows application which links to BioWin via the COM interface in Windows (Ltd., 2021). A biological tank with extra membrane rejection criteria was defined for the MBR unit. In this investigation, the default rejection rates for particles and colloids were set at 100% and 95%, respectively. The simulation module may provide simulations of both steady states and dynamic states. Different control methods, such as on/off, step, and PI/PID, might be used in conjunction with the process simulation. Modeling of nitrification as a two-step process that took into account AOB and NOB populations. Since enhanced nitrifying systems have been proven to have a rather high content of heterotrophic bacteria, they were also included to be able to represent the overall biomass concentration. The default model description for growth and decay of AOB and NOB is described below in detail.

5.2.2. Growth and Decay of Ammonia and Nitrite Oxidising Biomass

By using the energy from the conversion of ammonia to nitrite or nitrous oxide organic material is created from inorganic carbon which helps in the growth of AOB biomass (fixing CO2). NOB biomass growth occurs by converting nitrite to nitrate and utilising the energy that results to create organic material from inorganic carbon. The base rate expressions for the AOB and NOB growth process is generated by combining the ammonia and nitrite Monod expression, the ammonia and nitrite oxidising biomass concentration and the maximum specific growth rates specifically. The growth rate expressions are modified to account for the effects of the environment (off at low dissolved oxygen), nutrient concentrations and pH inhibition.

The modifications done to the default growth kinetic equations of AOB and NOB to account for the high ammonia concentration and high background salinity of urine has been explained in section 5.3.1.

5.2.3. Lab and Full-scale Model

Energy analysis for biological nitrification of hydrolysed urine includes aeration and pumping. The energy required for pumping (including recirculation) was based on average daily flow rates, the density of the liquid, pump efficiency, and total head. The default values of these parameters in the software were used in the analysis. The aeration energy includes meeting the oxygen demand for the growth of AOB and NOB process and membrane scouring to reduce membrane fouling. We have assumed that fine bubble diffusers are used for aeration and coarse bubble diffusers are used for membrane scouring in the model. The BioWin AS model prediction accuracy was checked by the calculation of the average relative deviation (ARD) according to the following equation (Liwarska-Bizukojc et al., 2011):

$$ARD = \frac{1}{N} \times \sum_{i=1}^{N} \frac{|(mi - pi)|}{mi} \times 100\% \qquad \text{(Equation 4)}$$

where N is the number of observations, pi is the output variable's projected value, and mi is the output variable's measured value. The ARD values were determined for the dynamic simulation results in relation to four major output variables: NH_4-N, NO_3-N, pH, Inflow rate.

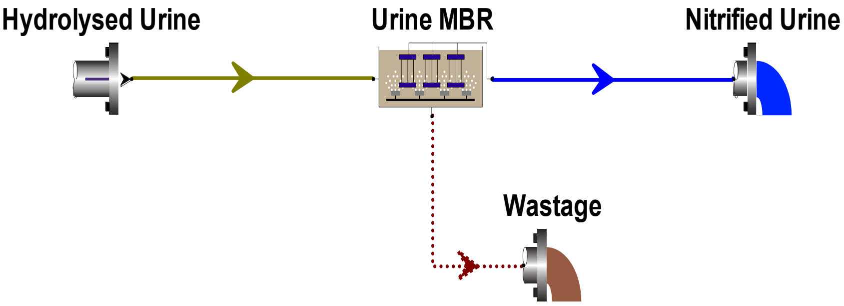


Figure. 5-1. Configuration of Urine MBR system used to simulate on BioWin

5.2.4. Process design configurations and greenhouse gas (GHG) emissions

Direct and indirect GHG emissions were quantified. BioWin predicts the off-gas N_2O and CO_2 emissions from nitrification, denitrification (if any), and carbonaceous oxidation processes occurring in the plant (de Haas, 2018). Indirect emissions from electricity consumption were calculated based on emission factors reported in the literature.

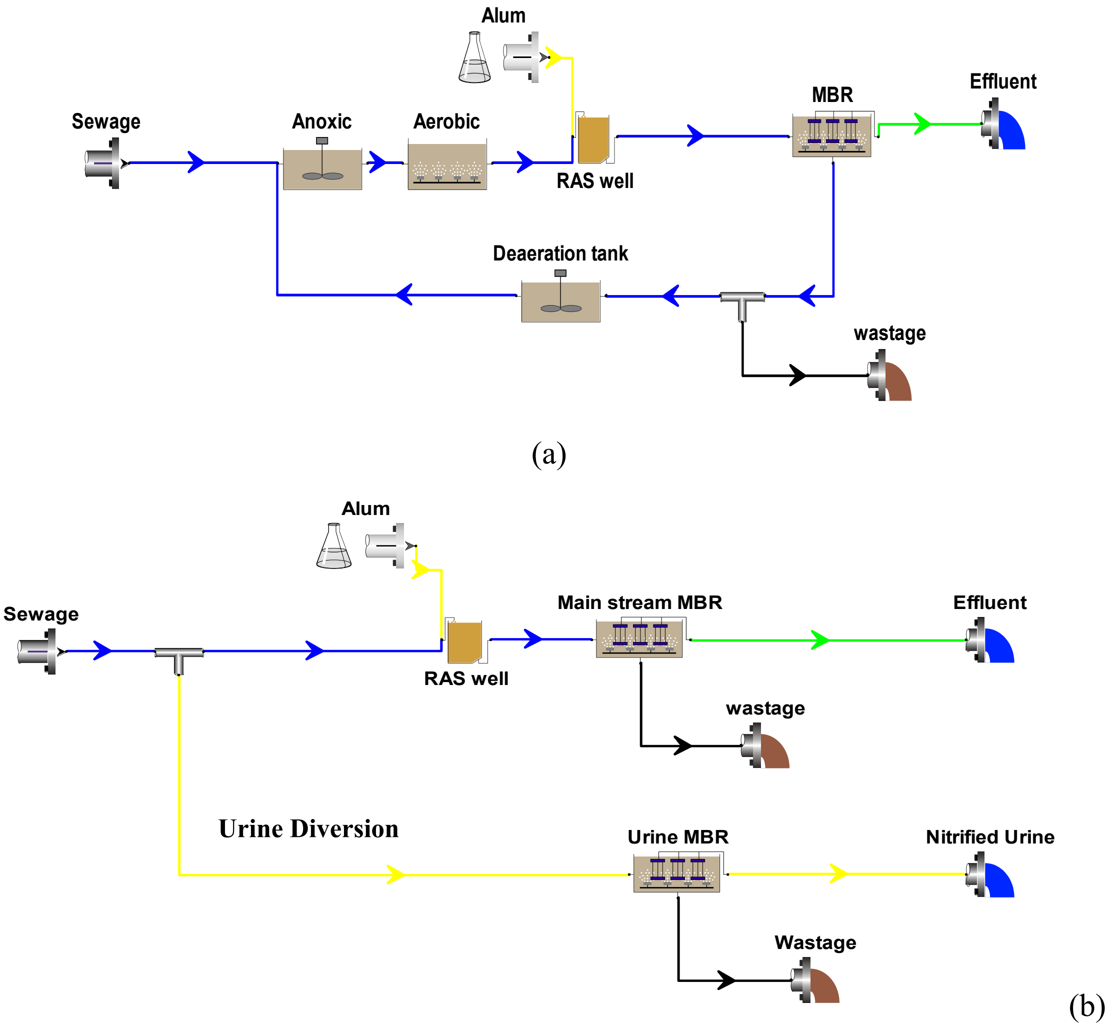


Figure 5-2 (a) Schematic of current main stream wastewater treatment process at Sydney Central Park and (b) the modified main stream with a side stream MBR urine treatment process.

5.3. Results and Discussions

5.3.1. Model Development and Calibration

The lab scale MBR (Jiang et al., 2021) was operated for 140 days which involved two stages: start-up (0 – 90 days) and stable operation stage (90 – 140 days). In the start-up stage the sludge was gradually acclimatised to the new substrate. As the salinity, C/N ratio and pH of urine differed significantly from municipal wastewater, urine was initially diluted 100 times. Following the increase in the abundance and activity of ammonia and nitrite oxidising bacteria, urine dilution was gradually decreased until a stable operation stage was reached using the full strength urine (undiluted) with a pH controller. To replicate the start-up process on BioWin, a steady state simulation was performed by manipulating the kinetic constants and variables. Following a steady state simulation, a dynamic simulation with a pH controller using BW controller module was performed. A detailed description of the model and assumptions have been given below.

All the influent urine characteristics needed for the model have been reported in our earlier study (Jiang et al., 2021) and they were used directly. Influent alkalinity was theoretically estimated from the reported influent ammonia values based on the urea hydrolysis reaction (Ray et al., 2018). Influent COD fractionation values were taken from Jönsson et al. (2005) and used in this study. The default model is not capable to represent the effluent nitrogen with the measured data because the system is quite different from the typical system - when BioWin solves the equation for steady state it needs to provide a reasonable seed for the numerical calculation. In the default model the seed is not reasonably close to the stabilised state of the real scenario, hence we get a washout solution. So, we have calibrated the model by increasing the seed organisms in the reactor. This provides more organisms to improve the seeding in the reactor. Also, the system operates at much higher concentrations than typical, so BioWin is not familiar with other factors working here as they do not play a significant role in more typical systems at lower concentrations (e.g. product substrate inhibition constants, etc.). Hence, we have modified these constants based on previous values reported in literature (Table S1, S2, and S3).

The high salinity of urine inhibits the growth of AOB and NOB, so in order to accommodate for the high background salinity we reduce the maximum specific growth constants. These values were chosen based on previous studies in literature which performed nitrification for high saline wastewater in the similar range of urine (Lu et al., 2009; Wang et al., 2009). We modified the maximum growth specific growth rate for nitrifiers accordingly to account for the high background salinity. The activity coefficients used in BioWin are calculated using the Davies approximation (limited to I.S. of 0.6) which means that if the ionic strength is higher than about 0.6 the activity coefficients used in the water chemistry will not be as accurate. The ionic strength of urine used for our laboratory study is <0.6 hence no further calibration was needed to the Davies approximation used in water chemistry equations.

The affinity and inhibition constants for NH_3 and HNO_2 were taken from the literature (Van Hulle et al., 2007; Fumasoli et al., 2015), because the biomass in our experiments was exposed to similar reactor conditions. All other growth parameters were taken from other study as is (Jubany et al., 2008). The calibrated model was dynamically validated using the measured data of the lab scale MBR. The daily values of influent conditions, DO operation were entered into the software. Figure 5-3 shows the comparison between the measured values reported in our earlier study (Jiang et al., 2021) and simulated values (from this study) of ammonia and nitrate. The measured and simulated values of effluent characteristics were compared and the average relative deviation (ARD) was calculated using the simulated and the measured data as a simple check for the accuracy of the model as described by Liwarska-Bizukojc et al. (2011).

The model showed a good representation of the measured data for ammonia and nitrate. The average relative deviation between the simulated data and the measured data was calculated where ARD values were less than 20%. ARD values less than 20% are considered acceptable according to Liwarska-Bizukojc et al. (2011). We used this calibrated model to upscale and estimate energy consumption, volumetric nitrate production rate and greenhouse emission for a full-scale urine treatment system which has been discussed in the following sections.

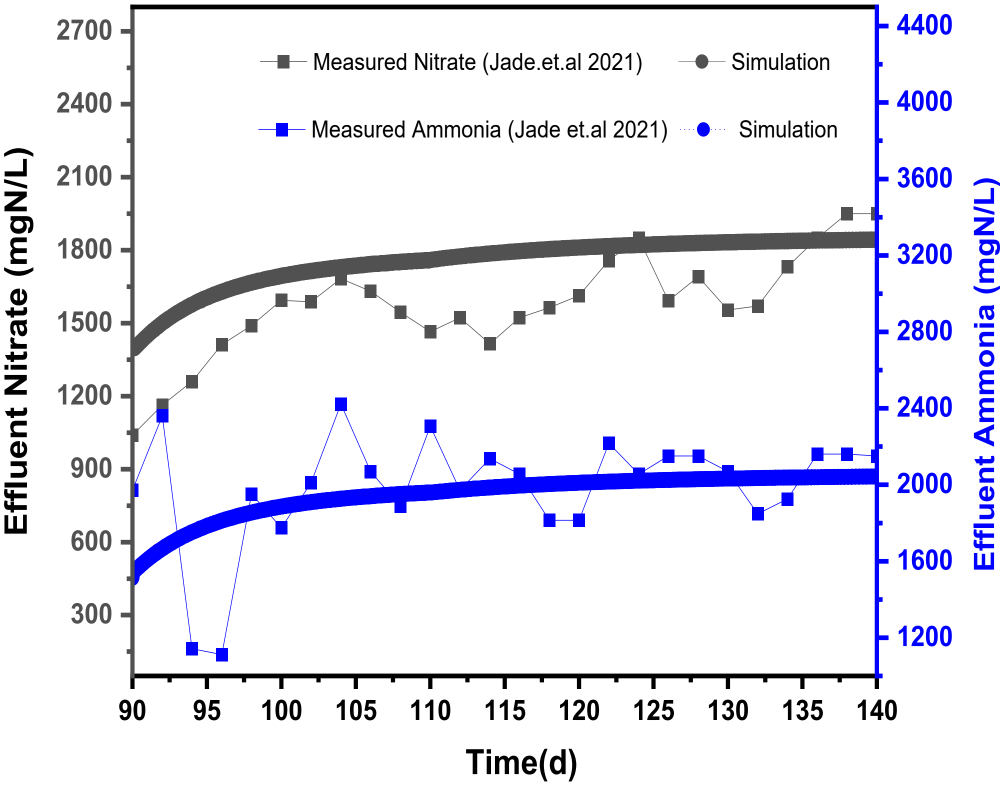


Figure 5-3. Comparison of measured and simulated effluent ammonia and nitrate concentrations

5.3.2. Energy and Nitrate Production Rate from a urine treatment

The calibrated model established in section 3.1 was up scaled and used to simulate a full-scale urine MBR treatment process for source separated urine. The effect of various DO set points on the energy consumption, nitrate production rate is presented in Figure 5-4. The results showed that at a DO set point of 6 mg/L, the electricity demand for nitrification is 6 kWh/kgN. A decrease of the DO set-point from 6 to 3 mg/L was predicted to decrease the aeration energy demand to just around 3 kWh/kgN (decrease of about 50%) with minimal impact on the nitrate production rate (<1%). However, when operated at DO< 3mg/L nitrate production rate was reduced and DO became the limiting factor for the growth of AOB and NOB. Besides, we noted that operating at low DO levels may have adverse effects on membrane fouling and mixing of the reactor (Gil et al., 2011). However, in a full scale urine treatment system, this can be mitigated by employing coarse bubble diffusers and mixers targeting the membrane zone only (Pearce, 2019). By operating at a an optimum DO set point, the air flow rate can be reduced which in turn reduces the blower energy consumption as reported in the literature (Le-Clech et al., 2005).

Previous laboratory and pilot scale studies (Etter et al., 2014; Volpin et al., 2020b; Ren et al., 2021) operated using DO > 6mg/L for full reactor mixing and membrane air scrubbing. However, in a full scale system, stirrer can be used instead of aeration to provide sufficient mixing. The intermittent coarse bubble diffusers beneath the membrane can also be provided for membrane scouring, thus limits deposition on the membrane surface (Judd, 2005; Le-Clech, 2010). These fouling-controlling strategies will avoid the need for operation of MBR at a high DO set point which in turn will reduce the energy demand. Our results show that the total energy demand can be reduced by operating at DO of 3 mg/L without effecting the nitrification rate. These results are much lower than the energy demand reported by others (Udert & Wächter, 2012; Etter et al., 2014; Etter et al., 2015; Fumasoli-Hug, 2016; Fumasoli et al., 2016) in their studies on a pilot scale system (shown in table 2).

The significant difference in the results from literature is possibly due to the difference in the scale of system, reactor configuration and aeration efficiency. Coarse bubble diffusers were used for aeration in this study (Etter et al., 2015) which have relatively low oxygen transfer efficiency and increased energy demand (Ashley et al., 1992). This energy consumption found in this study 3kWh/kgN for urine treatment which produces ammonium nitrate is significantly lower in comparison to for synthetic nitrogen fertiliser production through energy intensive Haber Bosch process (~10-14 kWh/kgN). The limitation of this study is that cost of concentration or sodium removal hasn't been considered in the calculation.

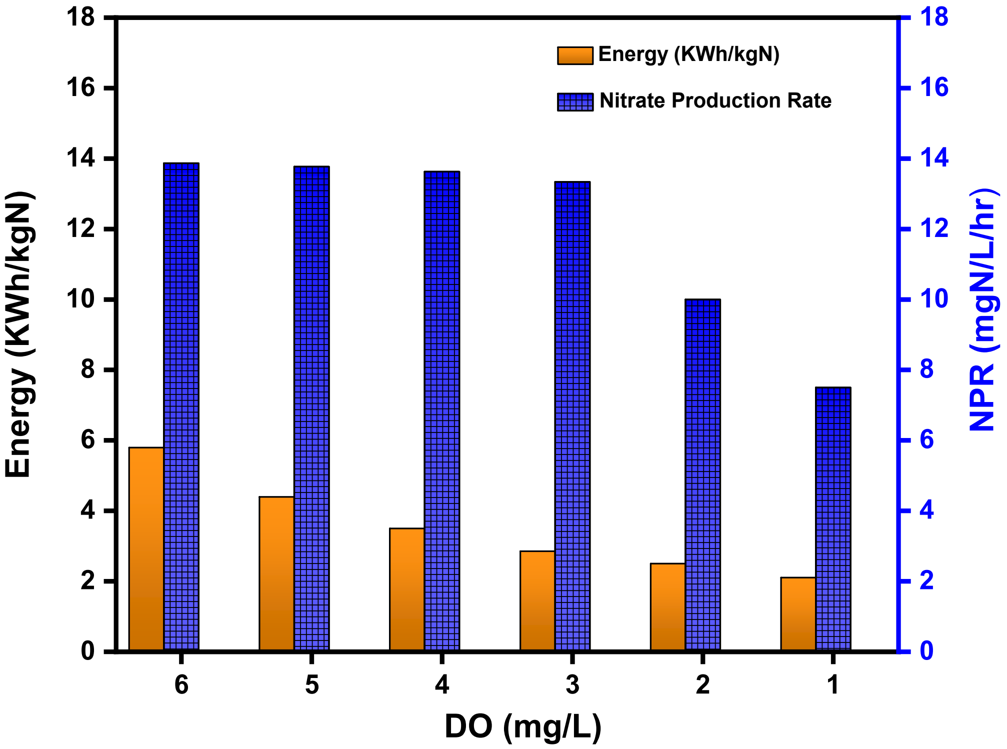


Figure 5-4. Effect of DO set point on energy consumption and nitrate production rate simulated for a full scale urine MBR system

5.3.3. GHG emissions from a urine MBR treatment system

Figure 5-5 shows the overall GHG emissions from pre-treatment of urine at various DO set points. At DO set point of 6 mg/L, the direct N_2O and CO_2 emissions from the plant has been estimated at 5.5 $kgCO_{2e}/m^3$ and 10.37 $kgCO_{2e}/m^3$ respectively while the indirect emissions through energy consumption were 14.25 $kgCO_{2e}/m^3$. At a lower DO set point of 3mg/L, the direct N_2O emission was increased to 9 $kgCO_{2e}/m^3$ but the indirect emissions were decreased to 7 kg CO_{2e}/m^3. This simulation results revealed that by lowering the DO set point, the direct N_2O emissions increased by 63%, however indirect CO_2 or GHG emissions reduced by 50% due to lower energy demand. The significant increase in N_2O emissions at lower DO set point could be due to increase in hydroxylamine formation. These results show that the emissions are quite sensitive to the change in DO set point. Previous studies reported a similar trend on the effect of DO set point on the N_2O emissions in nitrification process (Kampschreur et al., 2009; Massara et al., 2017; Massara et al., 2018). In oxygen limiting conditions, AOB use nitrite as the terminal electron acceptor to save oxygen for the oxygenation reaction of ammonia to hydroxylamine (Kuypers et al., 2018). Despite the higher emissions of N_2O at lower DO set point however, the overall GHG emission is still lower due to reduced CO_2 contribution. Our modelled N_2O emissions are between 0.006 and 0.01 kg N_2O per kg N influent which is within the medium range compared to other full-scale WWTPs typically between 0.001 and 0.25 kg N_2O/kg N, (Law et al., 2012; Bollon et al., 2016). Overall, decreasing the DO set point could lead to a reduction in off-site CO_2 emissions and an increase in onsite N_2O emissions. To our knowledge, this is the first such modelling or measurement results of GHG emissions from urine treatment.

Table 5-2 - Comparison of values predicted in this study and various other studies in literature

Reference	**Capacity (L)**	**Reactor type**	**Influent (mgN/L)**	**Energy (KWh/kgN)**	**Vol nitrification rate (mgN/L/d)**	**GHG ($kgCO_2/m^3$)**
This study	25000	MBR	3000	3	310	25.6
(Fumasoli et al., 2016)	120	MBBR	1790	11	640	-
(Fumasoli et al., 2016)	120	MBBR	4140	-	120	-
(Udert & Wächter, 2012)	2.6	MABR	2390	-		-
(Jiang et al., 2021)	4.5	MBR	4006	-	447	-
(Ren et al., 2021) Ren et al., 2021)	80	MBR	2950	-	95 - 365	-

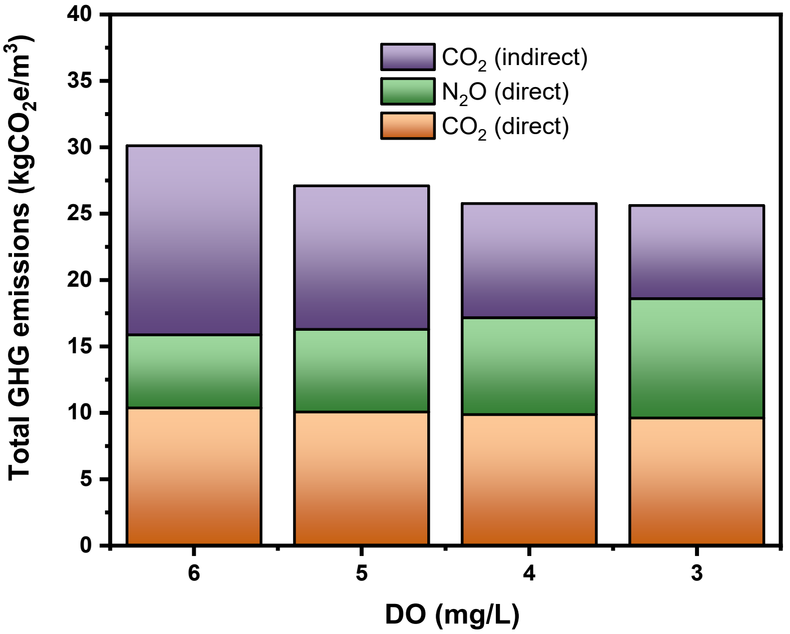
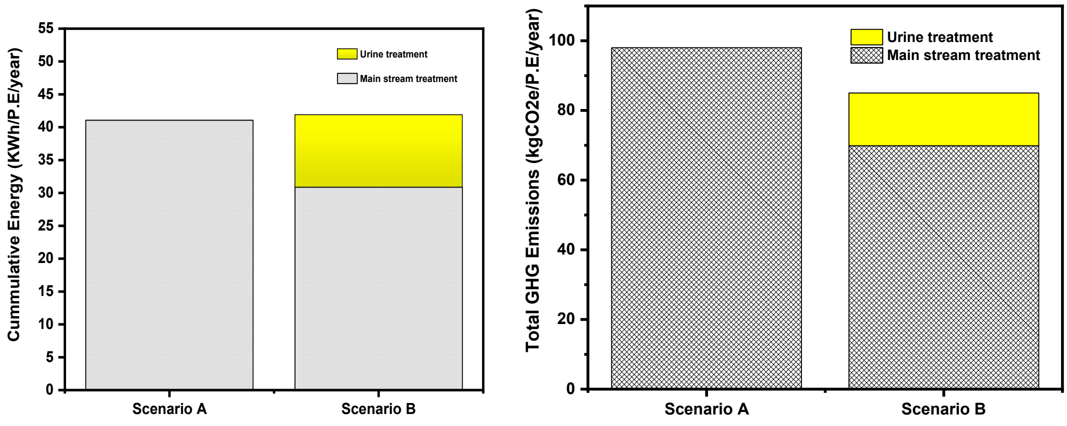


Figure 5-5. Effect of DO set point on total GHG emissions

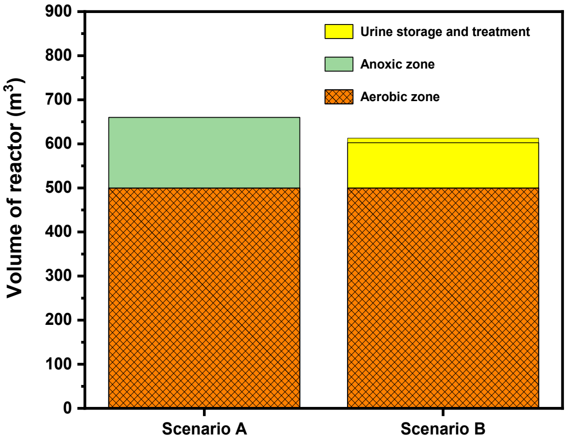
5.3.4. Cumulative Energy, GHG Emissions and Plant Footprint

In this section, we have made a comparison between two scenarios A and B: Scenario A - conventional MBR wastewater treatment with no urine diversion and Scenario B a modified wastewater treatment with 90% urine diversion. Scenario B consists of two processes: B1 (modified mainstream wastewater treatment system) and B2 (side stream urine treatment system). Figure.6 (a), (b), (c) show the cumulative energy, total GHG emissions and volume requirements for these two scenarios respectively. Our results reveal that cumulative energy consumption is similar for both the scenarios A and B however; the total GHG emissions and total bioreactor volume were respectively reduced by 13 and 10% in scenario B. We have performed a simple mass balance of nitrogen and theoretically estimated the oxygen requirements for both these scenarios to support our findings below.



(a) (b)

Figure 5-6 (a) – Cumulative Energy demand in Scenario A and B. (b) - Total GHG emissions in Scenario A and B



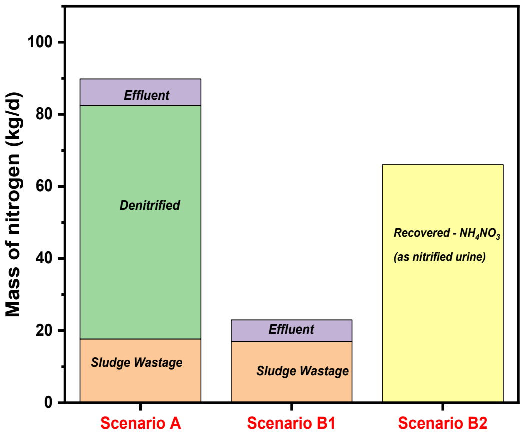


Figure 5-6 (c) – Total volume requirement in Scenario A and B

Figure 5-6(d) – Fate of nitrogen in scenario A and B1 & B2

5.3.5. Mass balance of nitrogen and oxygen consumption

a) Scenario A: conventional MBR wastewater treatment system (Figure. 5-2a)

In scenario A (when no urine is diverted), most of the influent nitrogen in the wastewater influent is nitrified and then denitrified in the anoxic tank to meet the effluent nitrogen standards (<7.8mgN/L). About 90 kg of nitrogen per day enters the treatment plant in the form of TKN. In the aerobic zone ammonia/ammonium is converted to nitrate by energy intensive aeration and about 18 kgN leaves the system via sludge wastage. From the aerobic zone, about 60-80% of the sludge is recirculated to anoxic tank for denitrification. In this process nitrogen in the form of ammonia (NH_3/NH_4^+) undergoes oxidation states from -3 to +5 losing 8 electrons to convert into nitrate and further 5 electrons are gained to be further denitrified. In the denitrification process, theoretically about 60% of the oxygen consumed for nitrification is conserved. This treatment design also needs recirculation which consume pumping energy.

b) Scenario B: Conventional MBR treatment system (without denitrification) and urine diversion

In process B1, 23 kgN/d of nitrogen enters the plant and about 17 kgN/d this is disposed via sludge wastage. About 67 kgN/d is diverted from process B1 through source separation of urine which then enters process B2. At optimum conditions, a maximum of 50% of the influent ammonia nitrogen in the urine is converted into nitrate limited by the alkalinity to nitrogen ratio in the urine (Udert & Wächter, 2012)). So, oxygen is consumed to oxidise only 33 kg of nitrogen in the urine. Since nitrogen is diverted from the wastewater through source separation of urine for a separate treatment (in B2), the denitrification process in the Scenario B1 becomes unnecessary. Hence, the pumping energy for the recirculation of activated sludge can be avoided in this scenario. However, due to the absence of denitrification there is no oxygen recovery.

Figure 5-6(d) compares the mass balance of nitrogen in both the scenarios. Although the effluent nitrogen concentrations in both the scenarios A and B are similar however, the results show that about 60-80% of nitrogen in the wastewater is lost from the system through denitrification in scenario A, an opportunity lost for recycling as a valuable nutrient. On the contrary, all the nitrogen is fully recovered for recycle and reuse in the Scenario B. We have theoretically quantified the oxygen requirements in both these scenarios (see the supplementary information, Table S1, S2, S3 and S4). Based on our estimations, scenario B (B1 and B2) requires about 5-10% more oxygen compared to Scenario B however, it saves pumping energy for recirculation of sludge as anoxic tank is not required anymore. Although the cumulative energy consumption between Scenario B are not different however, scenario B is highly advantageous because it helps in the recovery of all the valuable nutrients (N, P, K, etc.) found in the wastewater for various application in food crops, urban farming, public parks, etc. thereby contributing the nutrient circular economy. Other studies have compared treatment processes with and without urine diversion and have reported similar conclusions (Wilsenach & Van Loosdrecht, 2003; Wilsenach & van Loosdrecht, 2006).

5.4. Conclusions

In this work, we calibrated the BioWin model for side stream urine conditions from source separated urine via nitrification process. With calibration, we were able to validate our simulation results with the measured data from well-controlled laboratory operation. We also compared the conventional wastewater management design with a modified treatment process after urine diversion. The results have shown that at an optimum aeration rate (DO 3 mg/L), the energy consumption for urine treatment via MBR is about 3 kWh/kgN and greenhouse emissions was at 25.6 $kgCO_{2e}/m^3$. We further modelled and compared the process performance of two treatment process design scenarios: conventional MBR wastewater treatment system without urine diversion and with urine diversion but without denitrification. Results revealed that the cumulative energy demand for both the scenarios would be similar however; the GHG emissions and plant footprint can be slightly reduced if urine diversion and treatment is implemented. In the conventional wastewater treatment processes, most of the influent nitrogen is denitrified to meet the effluent nitrogen standards and hence no onsite nitrogen recovery is possible. However, urine diversion from the wastewater provides an opportunity to efficiently recover nitrogen and used for various applications such as vertical garden, urban farming, lawn and public parks, etc. For existing buildings, retrofitting a separate plumbing system for urine diversion can be a significant cost, but this approach will be more suitable for a new building projects. In a future where sustainable star-rated buildings are becoming the standard, decentralised wastewater recycling and urine source separation are poised to play a pivotal role. This strategy promises not only to reduce wastewater treatment costs but, more importantly, to contribute significantly to the preservation of vital nutrients within a circular economy. It's worth noting that P nutrient in particular is a finite resource on our planet with no substitutes and conserving them is of pivotal importance.

For existing buildings, the consideration of retrofitting a separate plumbing system for urine diversion may seem like a substantial cost and hence decision may rely on the evaluation within the context of holistic environmental conservation. While options like afforestation or the implementation of abatement technology for N2O emissions are valid approaches to offset emission however, they don't address the critical issue of nutrient loss, especially P in wastewater. Therefore, decision to embrace urine source separation should be driven by the urgency to preserve valuable nutrients rather than to solely reduce emissions.

CHAPTER 6: TECHNO ECONOMIC ASSESSMENT OF URINE DIVERSION AND CONVERSION TO FERTILISER PRODUCTS AT SYDNEY CENTRAL PARK

Summary

Urine diversion has been proposed as an approach for producing fertilisers while reducing nutrient loads on wastewater treatment plants (WWTP). Techno economic assessment was performed to compare the impacts on the operations phase of nutrient management at the WWTP and fertiliser production at a decentralised scale with and without urine diversion. Scenario modelling include distance from the location of the urine generation along with sensitivity analysis to understand the importance of key parameters, such as the electricity and fertiliser price, labour cost, flush volume of the urine. The urine diversion scenario performed better than the traditional system, with the potential to cut the total energy consumption by 10–50%, the operational expenses by 10–50%, and the capital expenditures by 10%–15%. Sensitivity analysis revealed that fertiliser price and flush volume were found to be the most sensitive parameters that determine the net savings from urine diversion. According to this study, urine diversion might be widely used as a method of nutrient recovery and sustainable wastewater management.

6.1. Introduction

Synthetic fertilisers made from non-renewable resources are used to produce about 50% of the world's food requirements (Kummu et al., 2012). Although the size of the resource base is arguable, supplies are dominated by a small number of countries, are limited, and the demand for them has increased in part due to rising meat consumption and the creation of biofuels (Cordell et al., 2009). The production of nitrogen fertiliser is a natural gas-dependent process that accounts for 1.2% of global energy consumption and the resulting greenhouse gases (Walling & Vaneeckhaute, 2021). Food riots have broken out in numerous nations because of price increase, of up to 800%, for phosphate rock and other fertiliser products in recent years (Nayak-Luke et al., 2022). Renewable and dependable alternatives are required since traditional fertilisers have negative effects and are resource constrained.

The main source of the essential nutrients in domestic wastewater is human food intake. Heavy investments are needed by the water utilities to remove them in order to protect the aquatic environment. According to a recent study by Siddik et al. (2021), the U.S. uses 2% of its total electricity for water and wastewater treatment systems (Walling & Vaneeckhaute, 2021). Urine contributes to about 50% of the phosphorus and 80% of the nitrogen in wastewater, despite making up less than 1% of the total volume of wastewater (Badeti et al., 2021). Full-scale source separation of urine has the potential to reduce nutrient loading in effluents, enhance wastewater management, recover valuable resources, and save some of the energy used in aeration as utilities place an increased emphasis on sustainability (Maurer et al., 2006). Fertilisers produced from human urine recover significant nutrients, have the potential to promote plant development, and contain lower amounts of heavy metals (Maurer et al., 2006). However, making fertilisers from urine will have an impact on the environment (Ishii & Boyer, 2015). New infrastructure technologies, like pressurised pipe networks or truck collection, will be needed for urine collection and transportation (Saetta et al., 2019).

Cleaning strategies using chemicals would be needed to stop the uncontrolled production of volatile ammonia and the formation of precipitates that obstruct pipe infrastructure (Saetta et al., 2019). It may be necessary to concentrate urine using techniques like reverse osmosis (RO), freeze-thaw, or distillation to increase nutrient concentrations, which are significantly lower in urine than in synthetic fertilisers (Courtney & Randall, 2021; Larsen et al., 2021b; Moharramzadeh et al., 2022). Alternative methods of concentrating nutrients include struvite precipitation, ammonia capture through ion exchange, and urea absorption (Larsen et al., 2021b). There may also be a need for additional treatment to eliminate pathogens and remove micropollutants found in urine (Almuntashiri et al., 2022). The selection of process for nutrient recovery is crucial. For example, if niche liquid fertilizers are produced that can be sold for a premium, then the process economics change significantly.

Techno-economic assessment (TEA) is a suitable method for comparing the economic viability of urine diversion systems with conventional systems, identifying areas of high impact, and revealing potential trade-offs and opportunities for system enhancement. (Garrido-Baserba et al., 2018). Most of the techno economic studies on urine diversion in literature have been limited to centralised wastewater treatment plants or hypothetical communities or communities in Sweden, Germany and Luxembourg (Lamichhane & Babcock, 2012; Remy & Jekel, 2012). These studies also lack several of the factors required for realising the maximum life cycle, and the environmental and economic benefits of urine separating systems. Previous studies by Ishii and Boyer (2015), and Landry and Boyer (2016) performed TEA and reported that urine diversion has no significant monetary benefits than the conventional process. However, these studies included struvite precipitation processes in their investigations, which required the addition of expensive chemicals like MgO and recovered only partial amount of nutrients from urine (Kabdaşlı & Tünay, 2018; Yan et al., 2021). Also, the type of product produced from source separated urine is important. An alternative method for stabilizing and recovering nitrogen in urine is biological nitrification. Biological processes like nitrification have been well studied for urine stabilisation and complete nutrient recovery due to their little demand for chemicals and energy (Udert & Wächter, 2012; Larsen et al., 2021b). Nitrogen loss occurs through urea hydrolysis, however acid or base addition to stabilise can slowdown urea hydrolysis and prevent loss of ammonia via volatilisation. This will prevent from losing ammonia before biological nitrification. However this is challenging as well especially in high rise buildings where such approach cannot be adopted. A few biological reactors for urine stabilisation have been piloted and are moving towards industrialisation (Fumasoli et al., 2016). Yet, they are more resilient at the size of bigger buildings or building complexes like Central Park due to potential inhibitory effects by salinity or volatile ammonia (Larsen et al., 2021a; Wald, 2022). In addition, there has been no economic analysis done so far to form the basis of a business model for urine diversion on a decentralised wastewater system, which separates urine and performs biological stabilisation. In Australia, no work has been published on a case study involving a decentralised water reclamation process. This work, therefore, aims to understand the economic impacts of urine diversion for the particular case of the decentralised WWTP at Sydney Central Park.

In this study, the economic implications of urine diversion and its conversion to fertiliser products were investigated in comparison to conventional methods at a building scale within decentralised environments such as Central Park. Our previous research provided a more comprehensive analysis of the effects on wastewater treatment (Badeti et al., 2021; Badeti et al., 2022). The conventional approach for managing urine involved the use of a wastewater system, which produced and distributed synthetic fertilisers containing equivalent amounts of nutrients. By quantifying the relative differences between these two approaches, the study aimed to evaluate the advantages and drawbacks of urine diversion and fertiliser production. We have also evaluated an independent scenario of realistic UTS building producing when urine fertiliser which can be sold at Aurin price. Aurin is the world's first fully approved urine-based fertilizer. It is a universal fertilizer created from human urine and has received approval for unrestricted use in the fertilization of all types of plants in Switzerland, Liechtenstein, and Austria. This liquid fertilizer is rich in essential nutrients required for plant growth, including nitrogen, phosphorous, and potassium, as well as a wide array of trace elements like iron, zinc, and boron. Major cities in Switzerland have successfully integrated Aurin into their fertilizer programs, demonstrating its efficiency. Different scenarios have been considered which required the transportation of urine derived fertiliser to distinct locations to explore how the distance of urine derived fertiliser transport influenced the overall economic performance. Sensitivity analysis was also performed to further investigate the influence of variable parameters like cost of electricity, petrol, labour, alum, flush volume and the uncertainty of others. The specific purpose of this study was to quantify the cumulative energy demand and economic impacts (capital and operational) of managing N and P in urine source separation and biological nitrification as compared with the conventional wastewater treatment. The results of this study are expected to highlight the potential benefits and costs of source separation in a decentralised setting with respect to nutrient recovery, fertiliser offsets, thus also identifying major areas of the system requiring a deeper investigation.

6.2. Methods

6.2.1. Description of urine processing methods and scenarios evaluated

To capture system-wide differences, the study's scope encompassed the production and distribution of flush water, collection of wastewater, production and transportation of fertilizer, and wastewater treatment, as demonstrated in Figure 6-1. Further details on these alternatives can be found in Section 3. It was assumed that source separated urine will be stabilised by biological nitrification (nitrified urine was not concentrated) and then transported if needed to the farms/irrigation sites. The processing costs and energy demands of Urine derived fertiliser products were compared to conventional fertilisers. The study assumed that 90% of urine in each of the examined scenarios would be used for fertiliser production. However, processes that were not affected by urine diversion, such as primary sludge treatment and hauling screening to landfills, were excluded from the analysis. To ensure the equivalence of nitrogen, phosphorus, and potassium mass in the fertilisers, ammonium nitrate, potash, and phosphorus fertilisers were used across all alternatives. These conventional fertilisers were applied to both the conventional and diversion options to give a similar quantity of nitrogen and phosphorus, despite differing nutrient recovery ratios. The study did not include an analysis of farm application of the fertilisers, as previous research indicated that there were no significant differences in plant ingest and runoff between urine-originated fertilisers and synthetic fertilisers.

6.2.2. Techno Economic Assessment

For the cumulative energy demand analysis, the functional unit of one kilogram of nitrogen equivalent was used, while the cost analysis was based on a functional unit of one-person equivalent for a year. The treatment of wastewater, which included urine where applicable, was considered in the analysis, as urine diversion can result in a considerable decrease in the amount of nitrogen and phosphorus that arrives at the treatment plant, which can have a significant impact on treatment. All alternatives provided equal masses of nitrogen and phosphorus in the fertilisers.

Australian Life cycles Inventory was used for most of the economic analysis which included the wastewater treatment costs, nutrient discharge costs, synthetic fertiliser cost, urine treatment and transportation costs. It was assumed to have built out urine separation systems so the cost of dual piping and urine separation toilets were not included in this study.

6.2.3. Sensitivity Analysis

A sensitivity analysis was performed to assess the reliability of the results, expand the testing of urine separation to a wider scope of scenarios, and clarify the ways in which model parameters and significant assumptions impacted the economic viability of urine diversion. A total of four separate simulation scenarios were created and six parameters were evaluated. These values were collected from available literature and various sources. The variation in cost savings were compared to the reference scenario.

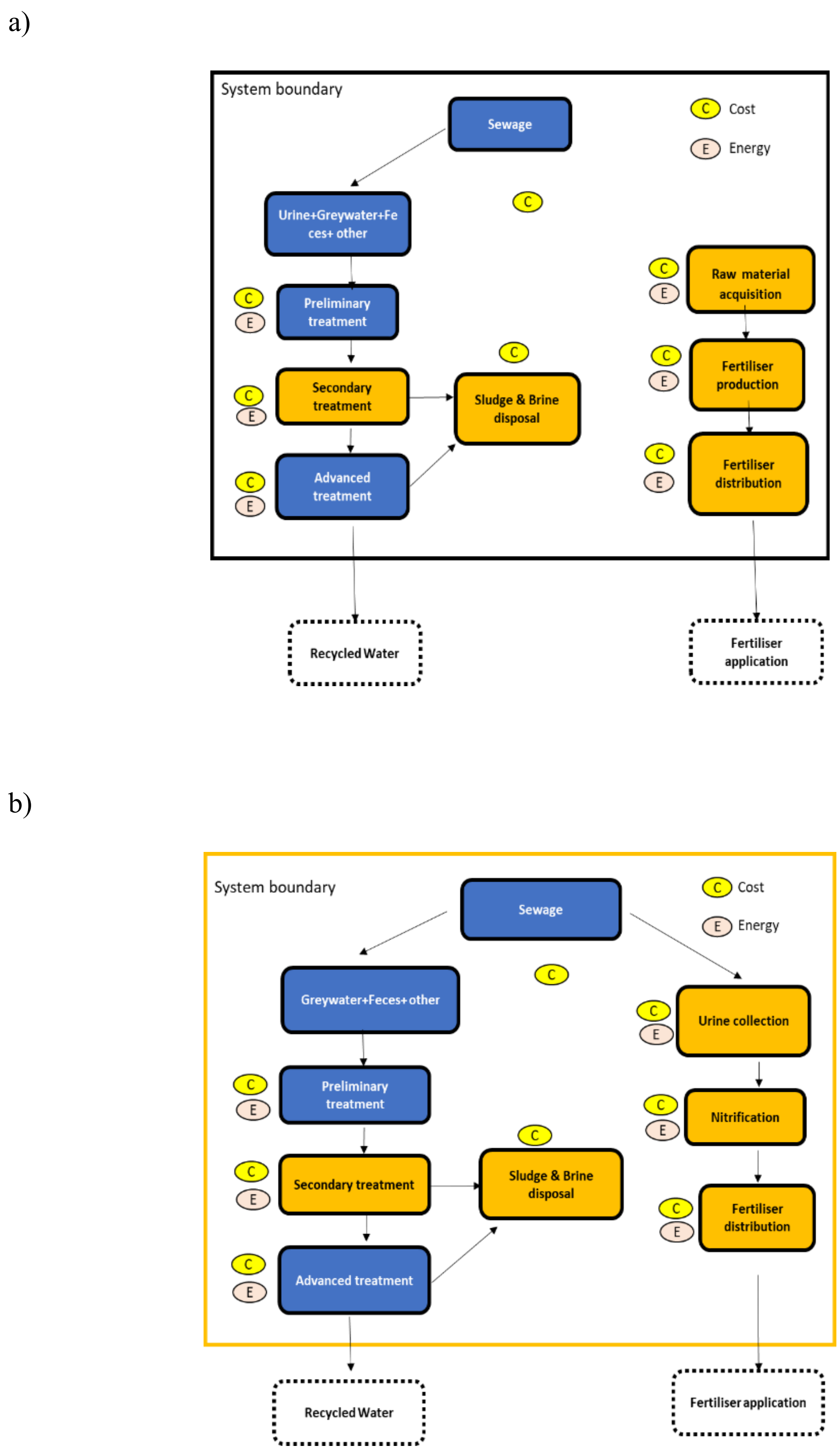


Figure 6-1. (a-b) System diagram for both scenarios. (a) Scenario A : Conventional (b) Scenario B: Urine nitrification alternative. The yellow boxes in the analysis were used to show that a process was either unique to a specific substitute or that urine diversion had a significant impact on its environmental impact.

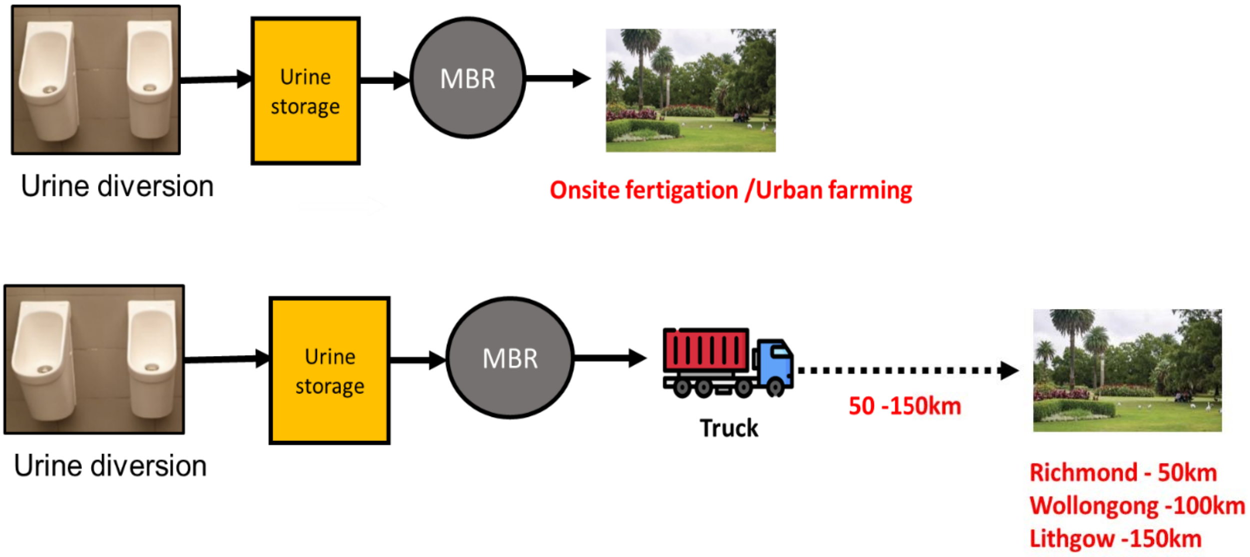
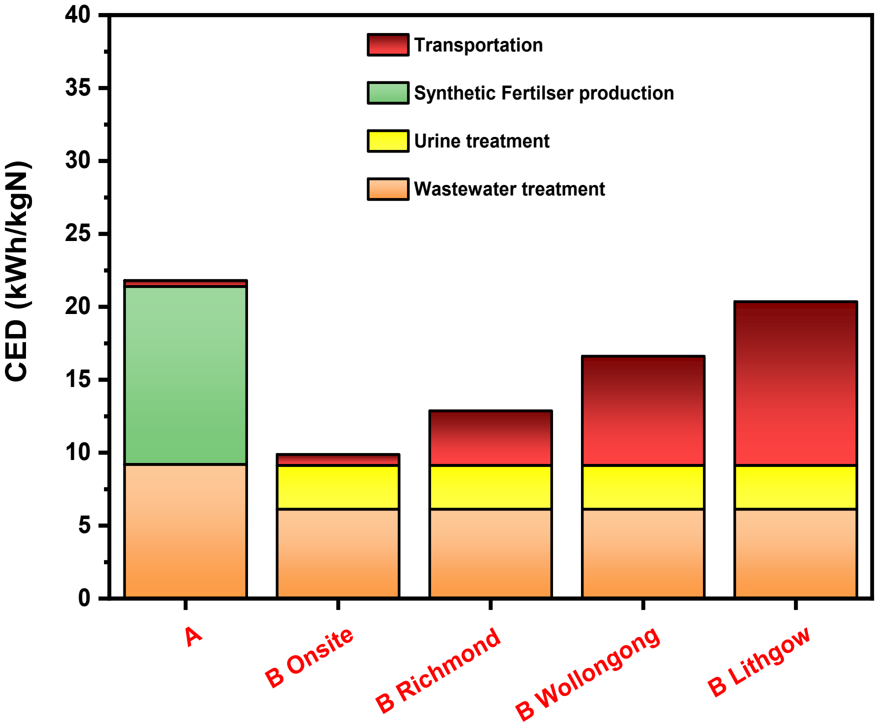


Figure 6-2. Urine transport options in scenario B

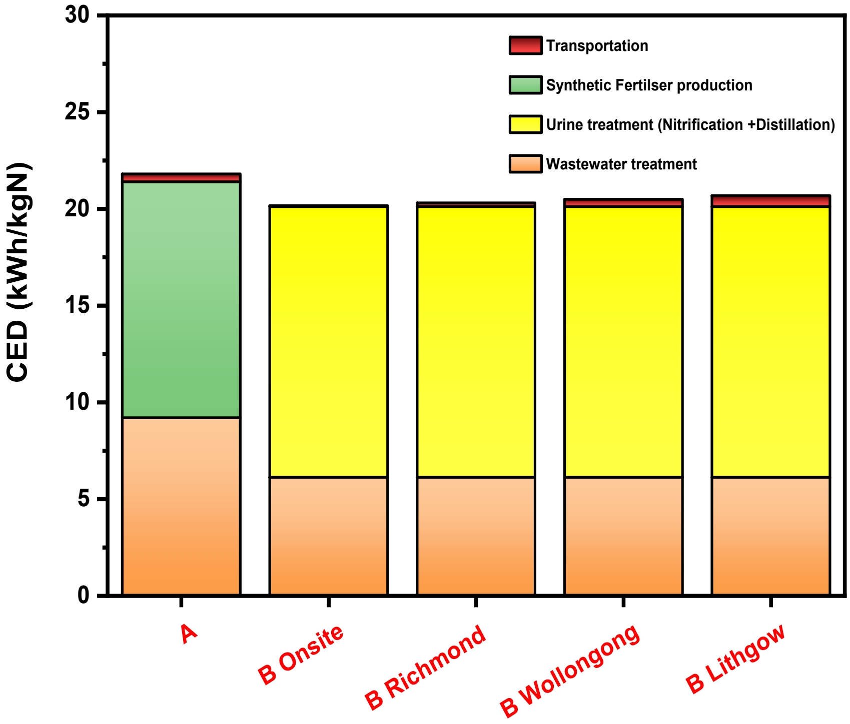
6.3. Results

6.3.1. Cumulative Energy Demand

Figure 6-2 shows the cumulative energy demand (CED) of scenario A and scenario B. In Scenario A, the CED accounts for about 23 kWh/kgN. The transportation cost in scenario A was marginal as the fertiliser has about 25-40% N content which is about 40-60 times more concentrated than the urine derived fertiliser (nitrified urine). In scenario B, onsite, the CED was significantly reduced by 60% to 10 kWh/kgN. This was due to the reduction in aeration energy and the avoidance of the energy required to process commercial nitrogen fertiliser via Haber-Bosch process (Mudahar & Hignett, 1985). In scenario B "Richmond", the urine derived fertiliser at Central Park was transported about 50 km to the agricultural farms in Richmond. The need for transportation counterbalanced the net energy gain that was saved in aeration and synthetic fertiliser production. Similarly, the CED increased to 16.6 kWh/kgN and 20.3 kWh/kgN in scenario B "Wollongong" and "Lithgow" respectively. Figure 6-3 shows the effect of performing distillation post nitrification in these scenarios; by performing distillation the fertiliser was concentrated (4 - 6% N), however, there was no significant energy gain due to the high energy consumption of vacuum distillation process (80-100 Wh/L (Etter et al., 2015)). Previously, Hilton et al. (2020) studied the effect of urine diversion on a city scale and evaluated the CED for which urine was acid stabilised and concentrated about 5 times using a reverse osmosis process. Table 6-1 shows a comparison of the net energy consumption between this study and a previous study (Hilton et al., 2020). The net energy gain was higher in this process due to the assumption for low energy requirements for reverse osmosis process (0.009 kWh/L (Hilton et al., 2020)). Hence, results found in this study were lower.



(a)



(b)

Figure 6-3 (a) Cumulative Energy demand for each scenario per kilogram nitrogen management (Urine treatment included only nitrification). (b) Cumulative Energy demand for each scenario per kilogram nitrogen management (Urine treatment includes nitrification and distillation to concentrate).

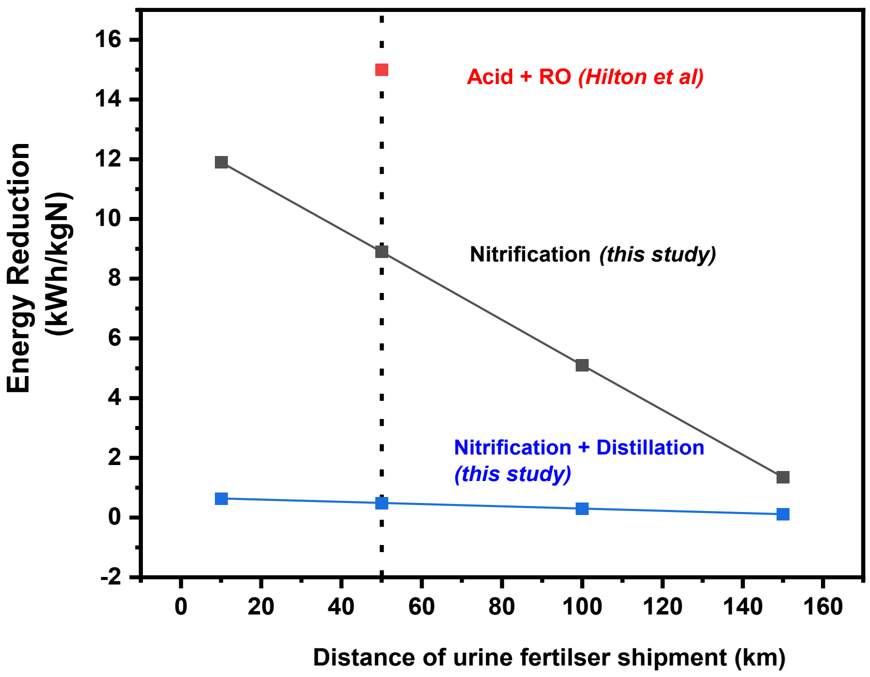


Figure 6-4. Relation between net energy savings and distance of fertiliser shipment

6.3.2. Operating Cost Analysis

In section 3.1scenario B, it was observed that the CED increased proportionally with the distance between the urine derived fertiliser production site and the farms. In this section, we compared the operating expenditure for both scenarios associated with nutrient discharge to Sydney Water sewage pipeline, energy consumption for wastewater, source separated urine treatment, chemical cost for phosphorus removal, and transportation costs. The analysis revealed that the operating expenses in scenario A accounted for about $440,000.00/yr (shown in Figure 6-4). The analysis revealed that in scenario B onsite, about 36% of the operating costs could be saved. The cost of electricity consumption was reduced due to the reduction in aeration energy, the quantity of alum required to remove phosphorus was also reduced as with urine diversion, about 30% of the phosphorus was recovered as a fertiliser and hence it did not need to be removed from the wastewater treatment process. Therefore, the cost required to purchase alum to maintain the effluent P limit was significantly reduced.

The nutrient discharge cost, that Sydney Water charges the operator, was also decreased because phosphorus and nitrogen discharge was cut down significantly. In scenario B the operator need not account for the purchase of commercial synthetic fertiliser as it could be produced at the treatment plant by source separation and hence it could be a significant advantage. However, in Scenario B "Richmond", B "Wollongong", B "Lithgow" these savings were counterbalanced by the cost requirement to transport the urine derived fertiliser to Richmond, Wollongong and Lithgow which reduced the savings to 23, 14, 3% respectively. Previous studies on the techno economic assessment of urine diversion with only partial nutrient recovery through struvite precipitation have concluded that no significant monetary benefits are found (Ishii and Boyer (2015); Landry and Boyer (2016)). In this study, we found that significant cost savings could be possible depending on the travel distance of the fertiliser shipment. The difference in the findings in this study could be due to the choice of urine processing technology and urine derived fertiliser produced.

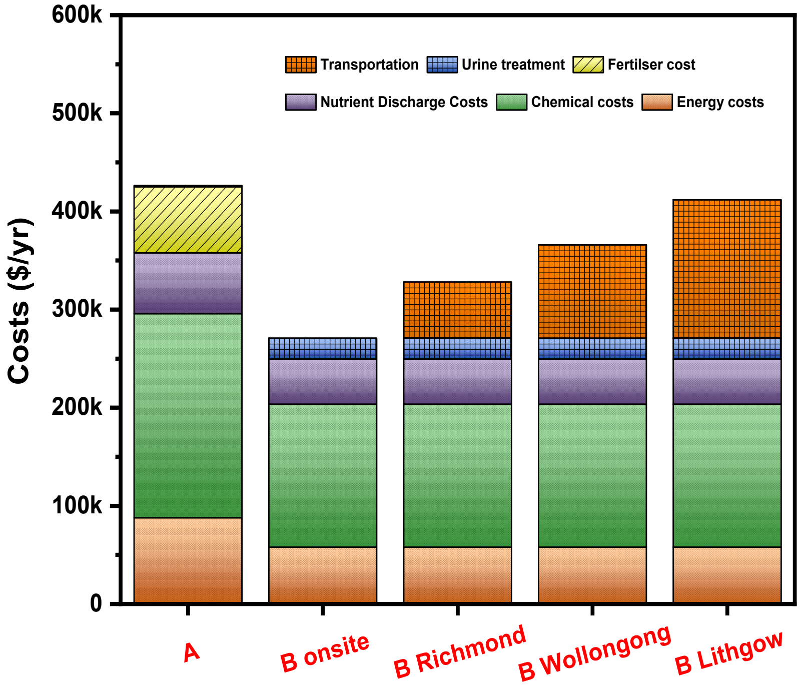


Figure 6-5. Operating cost for each scenario per year

6.3.3. Capital Cost

In our previous studies Badeti et al. (2021); Badeti et al. (2022), we observed that the anoxic tank could be avoided, which compacted the plant footprint and significantly reduced the capex (capital expenditure) of the treatment plant. However, this did not consider the space and capital expenditure needed for urine treatment and transportation. In this section, we compared the capital costs of scenario A and scenario B. In scenario B three sub scenarios have been evaluated (B_{120}, B_{310}, B_{640} – representing urine nitrification performed at 120, 310 and 640 mgN/L/d nitrification rates). The basis of selecting these scenarios have been explained below.

Figure 6-5 shows the capital expenditure that includes space requirements and process design cost for scenario A and B. The reference scenario revealed a capex cost of about $9/kgN of which 24% of the capex was for the space requirement and when this space was saved in configuration B the capex was reduced to $6.5/kgN. In process configuration

B, the anoxic tank and recirculation of nitrified mixed liquor were not required as most of the nitrogen is already removed. However, it would need to accommodate capital costs for nitrification of urine, which would need space and infrastructure. Our estimation showed capital costs of $1.5 per unit of influent nitrogen. The capital costs associated with nitrification of urine were dependent on the nitrification rates that can be achieved as it determined the space requirement for urine treatment plant after considering the nature of biological nitrification process especially in high strength ammonia solutions.

In the initial studies on urine nitrification, the primary focus was not on optimising the process, but rather on determining its viability and assessing the stability of the procedure, which demonstrated stability over a period of 12 months (Udert & Wächter, 2012). Fumasoli et al. (2016), in a pilot plant experiment, discovered an inverse relationship between the nitrification rate and the influent nitrogen concentration (shown in table 5-2), highlighting the significance of salt and/or free ammonia inhibition and operating conditions in the context of urine nitrification. Previous studies achieved a minimum nitrification rate of 120 mgN/L/d and revealed that further optimising the reactor configuration and operating conditions can increase the nitrification rate to 640 mgN/L/d (as ammonium-nitrogen transformed into nitrate NO_3-N per litre of reactor volume per day). Based on our full-scale modelling study in chapter 5, the results revealed a nitrification rate of 310 mgN/L/d for a full scale system. Capital expenditures for all these scenarios (B_{310}, B_{120}, B_{640} representing 310, 120 and 640 mgN/L/d) have been estimated and are shown in Figure 6-5. If the urine treatment process was designed for the worst-case scenario (B_{120}); the capital costs were estimated to be about 10% higher than scenario A. But if the urine treatment can be performed at optimised conditions (B_{640}) the capital cost costs are expected to reduce by 15%.

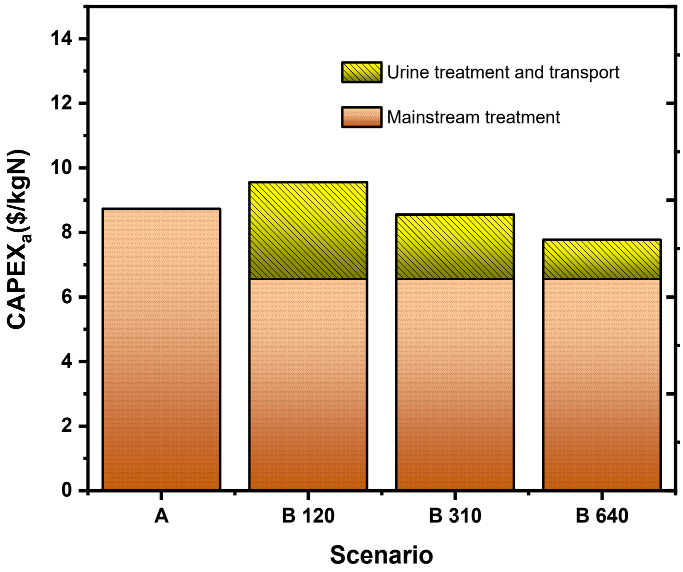
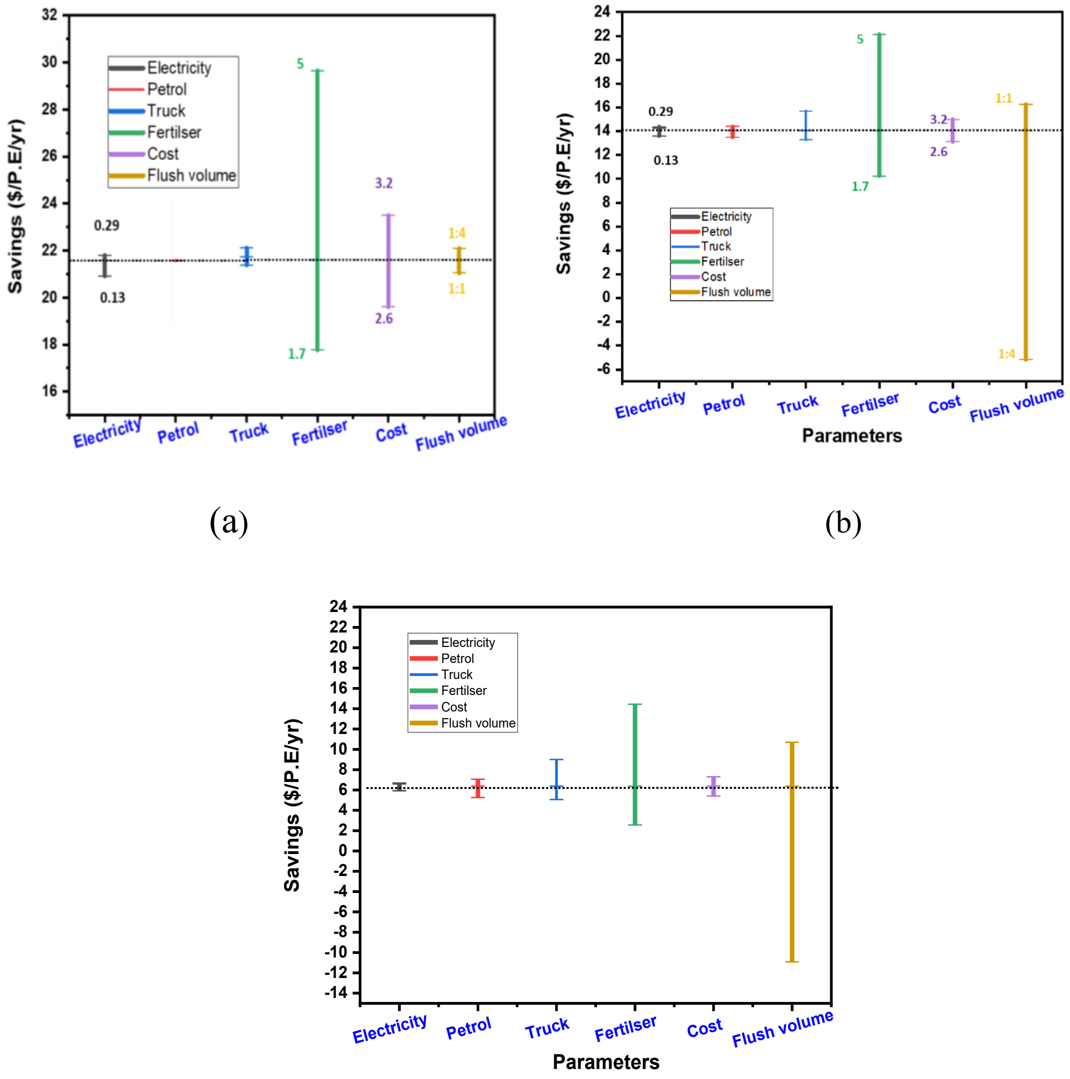


Figure 6-5. Annualised capital costs for each scenario per kilogram nitrogen management

6.3.4. Sensitivity Analysis

The sensitivity analysis conducted for the operating costs (as detailed in Section 6.3.2) revealed that the trade price for the fertiliser cost and the flush volume both affected the net savings to the greatest extent in most of the studied scenarios. Only in scenario B onsite, the flush volume did not affect the net savings but fertiliser and chemical (alum) costs effected the net savings. The influence of various parameters like electricity, petrol, and fertiliser cost on the overall operating costs of both scenarios has been investigated under different urine diversion scenarios.

Earlier in Figure 6-2, it was observed that the savings were possible with urine diversion. Figure 6-6 (a) (b) (c) showed the influence of parameters on cost savings. These results showed that the cost-benefit of urine diversion was robust. Our results suggested that the viability of the urine diversion process largely depended on the market and expected revenue for the generated nutrient product. The increasing prices of nitrogen fertilizers, with the expectation of further price hikes in the future, indicated potential promise for the adoption of our technology. However, the reliance on the value of the nutrient product made the technology vulnerable to market fluctuations, with little influence over the retail price of nitrogen. From the perspective of technology optimization, the findings clearly indicated a need to reduce the energy consumption of urine transport or concentration, particularly for transportation distances exceeding 50 km, to make it a feasible alternative for nitrogen recovery.



(a)

(b)

(c)

Figure. 6-6 (a) (b) (c). Sensitivity analysis results for the scenarios with urine diversion (a)- Scenario B onsite (b)- Scenario B Richmond (c) Scenario B Wollongong. The dashed horizontal lines represent the median value cases.

6.3.5. Realistic UTS Scenario (without WWTP)

The utilization of urine diversion has been suggested as a potential solution for generating sustainable fertilizers and decreasing nutrient loads to WWTPs. Full cost analysis of a realistic urine diversion scenario in a UTS building was performed. The total production cost for producing a liquid concentrated fertiliser by nitrification and distillation processes was estimated. Sensitivity analysis revealed that the market price of the fertiliser was found to be the most sensitive parameter to determine the net savings from urine diversion. Based on the estimations the cost of producing concentrated liquid fertiliser (~ 3-5 % N) is about \$4.3/L. Results show that about \$80-86k/year net revenue is possible if the fertiliser can be sold at the price of Aurin fertiliser (shown in Fig 6-7), which has been commercialised.

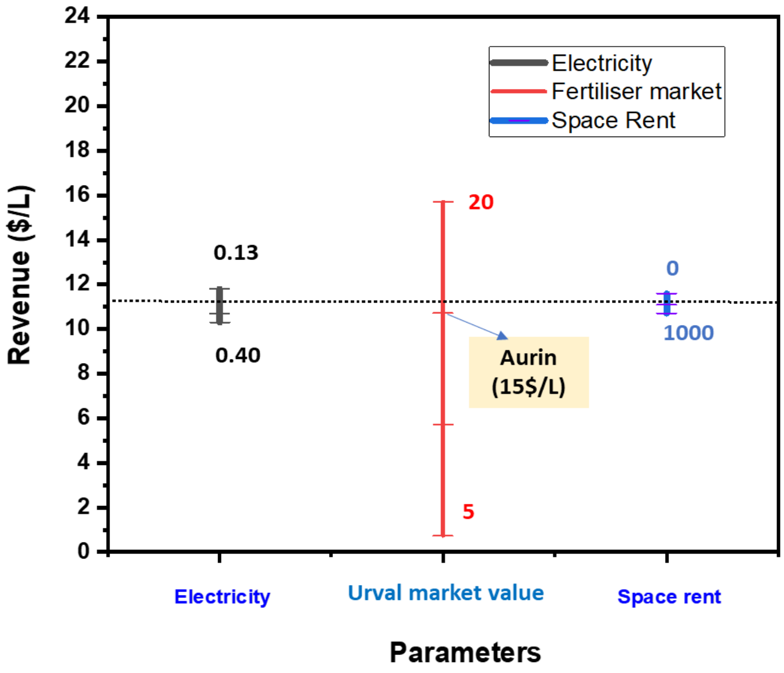


Figure. 6-7. Net revenue generation per litre of urine fertiliser bottle sold (UTS scenario –without WWTP)

6.4. Conclusions

In this work, we performed a full techno assessment on the impact of urine diversion and conversion to fertiliser products in Sydney Central Park. The results have shown that about 10-50% of the cumulative energy demand can be saved depending on the distance that urine derived fertiliser would need to be transported. The capital expenses were expected to be slightly more than the conventional scenario. Sensitivity analysis revealed that flush volume and fertiliser cost were critical parameters in determining the net possible saving from urine diversion. These results showed that the utilities would need to carefully evaluate application options associated with the urine derived fertiliser options for the process to show significant benefits. As buildings with sustainable star ratings become a norm in the future, decentralised wastewater recycling and reuse with source separation of urine will be seen as a promising strategy to not only reduce wastewater treatment costs but also to contribute to nutrient circular economy. For existing buildings, retrofitting a separate plumbing system for urine diversion can incur a significant cost; therefore, this approach will be more suitable for a new building project.

CHAPTER 7: CONCLUSION AND RECOMMENDATIONS

This thesis is aimed at understanding and quantifying the impact of source separation of urine (for conversion to fertiliser products) to the downstream wastewater treatment facilities and lead towards a sustainable wastewater management while helping efficient recovery of nutrients for reuse. Concluding remarks and recommendation for each study are described below.

7.1. Impact on energy consumption, GHG emissions and operating costs

Simulation shows that urine diversion can reduce up to 82% of nitrogen, 30% of phosphorus loads reaching the WWTP although it reduces the COD by only 6%, and the volumetric flow by 1.2 %. At urine diversion higher than 50 %, the influent COD/N ratio significantly increases to the point where the influent nitrogen is adequate only for biomass growth and hence avoiding the necessity of denitrification processes. About 90% of urine diversion reduces the energy required for aeration by up to 33% and even higher energy savings are expected after optimizing the operating conditions. Urine diversion can significantly reduce direct N_2O and CO_2 emissions by 98% and 25% respectively while indirect emissions for energy consumption are reduced by 20%. Finally, urine diversion can save about 22% of the WWTP operating costs and about 52% of this savings is due to reduced energy consumption for aeration during biological nitrification.

7.2. Impact on treatment capacity, process design and capital expenditure

Here, we modelled the impact of urine diversion on the treatment capacity, process design and capital costs of Sydney Central Park WWTP. The results have shown urine diversion at source for nutrient recovery can significantly help reduce the influent nitrogen concentration making nitrogen removal process a lesser priority for a WWTP. A 75% urine diversion can help double the treatment capacity of the existing WWTP without the need for a bioreactor upgrade, although above 40% urine diversion, the impact is less significant. However, sensitivity analysis indicated that, operating the bioreactor at higher MLSS concentration could further help increase the treatment capacity although further studies are needed to measure the exact effect of operating at high MLSS concentration.

At a urine diversion above 75%, anoxic tank for biological nitrification becomes redundant and hence a simple aerobic MBR is able to produce the similar effluent quality thereby helping reduce the plant footprint and capital expenditure cost under similar operating conditions or significantly help increase the capacity of the existing treatment plant. These findings may be more relevant to large cities such as Sydney where space can be limited and its space value is very significant. Urine diversion can also be a smart strategy where wastewater treatment infrastructure is already running at near capacity and will require upgrades in the near future requiring additional footprint.

7.3. Full scale conversion of urine to fertiliser products

In this work, we have calibrated the BioWin model to the high ammonia containing side stream urine treatment conditions to estimate the energy consumption, greenhouse gas emissions and volumetric nitrification rate for source separated urine at a decentralised wastewater treatment plant. Firstly, we calibrated and validated modelling output from Biowin simulation software with lab scale urine MBR treatment process. The Biowin model was further upscaled and used to investigate the performance of a full-scale urine treatment system. The simulations showed that at optimised DO set point of 3 mg/L, the energy consumption was found to be 3kWh/kgN and greenhouse emissions of 25.6

$kgCO_{2e}/m^3$ at a volumetric nitrification rate of 310 mgN/L/d which is only about 25-30% of the energy consumed in the synthesis of a virgin fertiliser. This study shows that the total combined energy of a MBR wastewater treatment plant (urine diverted) and urine treatment separately in a urine MBR for fertiliser recovery, is similar to the total energy demand of a conventional MBR wastewater treatment plant without nutrient recovery. However, diverting urine reduces GHG emissions and footprint of a wastewater treatment plant in comparison to the conventional process. Besides, nutrients are fully recovered through source separation of urine and treatment thereby contributing to both fertiliser economy and circular economy.

7.4. Full life cycle and techno economic study

Urine diversion has been proposed as an approach for producing fertilisers while reducing nutrient loads on wastewater treatment plants (WWTP). Techno economic assessment was performed to compare the impacts on the operations phase of nutrient management at the WWTP and fertiliser production at a decentralised scale with and without urine diversion. Scenario modelling include distance from the location of the urine generation along with sensitivity analysis to understand the importance of key parameters, such as the electricity and fertiliser price, labour cost, flush volume of the urine. The urine diversion scenario performed better than the traditional system, with the potential to cut the total energy consumption by 10–50%, the operational expenses by 10–50%, and the capital expenditures by 10%–15%. Sensitivity analysis revealed that fertiliser price and flush volume were found to be the most sensitive parameters that determine the net savings from urine diversion. According to this study, urine diversion might be widely used as a method of nutrient recovery and sustainable wastewater management. Potential income of about \$80-86k/year net revenue can be generated from sales of a concentrated liquid fertiliser in high rise buildings like UTS.

7.5. Recommendations for future work

The following research questions may be of interest for further investigations:

Application of urine diversion in the existing buildings through retrofitting: One of the drawbacks of this case study with the Sydney Central Park is that it lacks urine diversion and treatment facilities although it has a water recycling plant. Implementing urine diversion at the Sydney Central Park will incur retrofitting costs which needs to be accounted and the same applies if urine diversion is to be universally applied in the whole City of Sydney. A detailed economic study including business models for urine derived fertiliser on the potential of implementing urine diversion in a city-wide coverage is one area of research that needs consideration in the future.

Urine treatment technology options: Currently, the urine treatment technologies are yet to be fully commercialised and tested for larger-scale applications and hence there is a still a lot of research works that needs to be done to develop a technology that is energy efficient, less operational complexity and affordable that can make an economic sense for the urine-derived fertiliser. A range of urine treatment technology options should be developed targeting different fertiliser reuse scenarios and technology that ensures safety of the users from the pathogens, micropollutants such as pharmaceuticals and other harmful salts. There is a limited understanding of what happens to these substances when treated urine is utilized as a fertilizer in soils. If subsequent research indicates a possible hazard to human and environmental well-being resulting from this recycling process, it will become necessary to incorporate an additional treatment step for the removal of micropollutants. Such an inclusion would inevitably influence the outcomes of lifecycle analysis.

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APPENDIX

CHAPTER 3: Supplementary Information

Table S1**-** Summary of wastewater fractions used for model calibration

Name	**Value**
Readily biodegradable (including Acetate) [gCOD/g of total COD]	0.16
Acetate [gCOD/g of readily biodegradable COD]	0.15
Non-colloidal slowly biodegradable [gCOD/g of slowly degradable COD]	0.75
Unbiodegradable soluble [gCOD/g of total COD]	0.05
Unbiodegradable particulate [gCOD/g of total COD]	0.13
Cellulose fraction of unbiodegradable particulate [gCOD/gCOD]	0.5
Ammonia [gNH_3-N/gTKN]	0.8
Particulate organic nitrogen [gN/g Organic N]	0.5
Soluble unbiodegradable TKN [gN/gTKN]	0.02
N:COD ratio for unbiodegradable part. COD [gN/gCOD]	0.035
Phosphate [gPO_4-P/gTP]	0.5
P:COD ratio for unbiodegradable part. COD [gP/gCOD]	0.011
Reduced sulfur [H_2S] [gS/gS]	0.15
Ordinary heterotrophic COD fraction [gCOD/g of total COD]	0.02
Methylotrophic COD fraction [gCOD/g of total COD]	1.00E-04

Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04
Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04
Anaerobic ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04
Phosphorus accumulating COD fraction [gCOD/g of total COD]	1.00E-04
Propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04
Acetoclastic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04
Hydrogenotrophic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04
Sulfur oxidizing COD fraction [gCOD/g of total COD]	1.00E-04
Sulfur reducing propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04
Sulfur reducing acetotrophic COD fraction [gCOD/g of total COD]	1.00E-04
Sulfur reducing hydrogenotrophic COD fraction [gCOD/g of total COD]	1.00E-04
Endogenous products COD fraction [gCOD/g of total COD]	0

Table S2- Kinetic parameters for bioreactions in the Biowin model.

Heterotrophic	**Value**	**Arrhenius**
Max. spec. growth rate [1/d]	3.2	1.029

Substrate half sat. [mgCOD/L]	5	1
Anoxic growth factor [-]	0.5	1
Denite N2 producers (NO3 or NO2) [-]	0.5	1
Aerobic decay rate [1/d]	0.62	1.029
Anoxic decay rate [1/d]	0.233	1.029
Anaerobic decay rate [1/d]	0.131	1.029
Fermentation rate [1/d]	1.6	1.029
Fermentation half sat. [mgCOD/L]	5	1
Fermentation growth factor (AS) [-]	0.25	1
Free nitrous acid inhibition [mol/L]	1.00E-07	1
Ammonia Oxidising	Value	Arrhenius
Max. spec. growth rate [1/d]	0.9	1.072
Substrate (NH_4) half sat. [mgN/L]	0.7	1
Byproduct NH_4 logistic slope [-]	50	1
Byproduct NH_4 inflection point [mgN/L]	1.4	1
Denite DO half sat. [mg/L]	0.1	1
Denite HNO_2 half sat. [mgN/L]	5.00E-06	1

Aerobic decay rate [1/d]	0.17	1.029
Anoxic/anaerobic decay rate [1/d]	0.08	1.029
KiHNO$_2$ [mmol/L]	5.00E-03	1
Nitrite Oxidising	Value	Arrhenius
Max. spec. growth rate [1/d]	0.7	1.06
Substrate (NO$_2$) half sat. [mgN/L]	0.1	1
Aerobic decay rate [1/d]	0.17	1.029
Anoxic/anaerobic decay rate [1/d]	0.08	1.029
KiNH$_3$ [mmol/L]	0.075	1

CHAPTER 4: Supplementary Information

Table S1: Design and operating parameters and effluent nitrogen concentrations for Central Park WWTP, for urine diverted fractions of 0–75%.

% UD	Units	0%	10%	20%	40%	50%	75%
HRT	H	15.6	10.7	9.9	8.85	8.34	7.66
SRT	D	6.2	4.94	4.49	3.8	3.8	3.52
Effluent TN	mgN/L	7.42	7	6.07	6.16	5.26	2.01

Effluent NH_4-N	mgN/L	0.91	0.93	0.98	1.05	1.02	0.98
Effluent NO_3-N	mgN/L	3.28	3.02	2.17	2.45	1.98	0
Anoxic tank	ML	0.163	0.068	0.048	0.01	0	0
Aerobic + MBR tank	ML	0.425	0.52	0.54	0.578	0.588	0.588

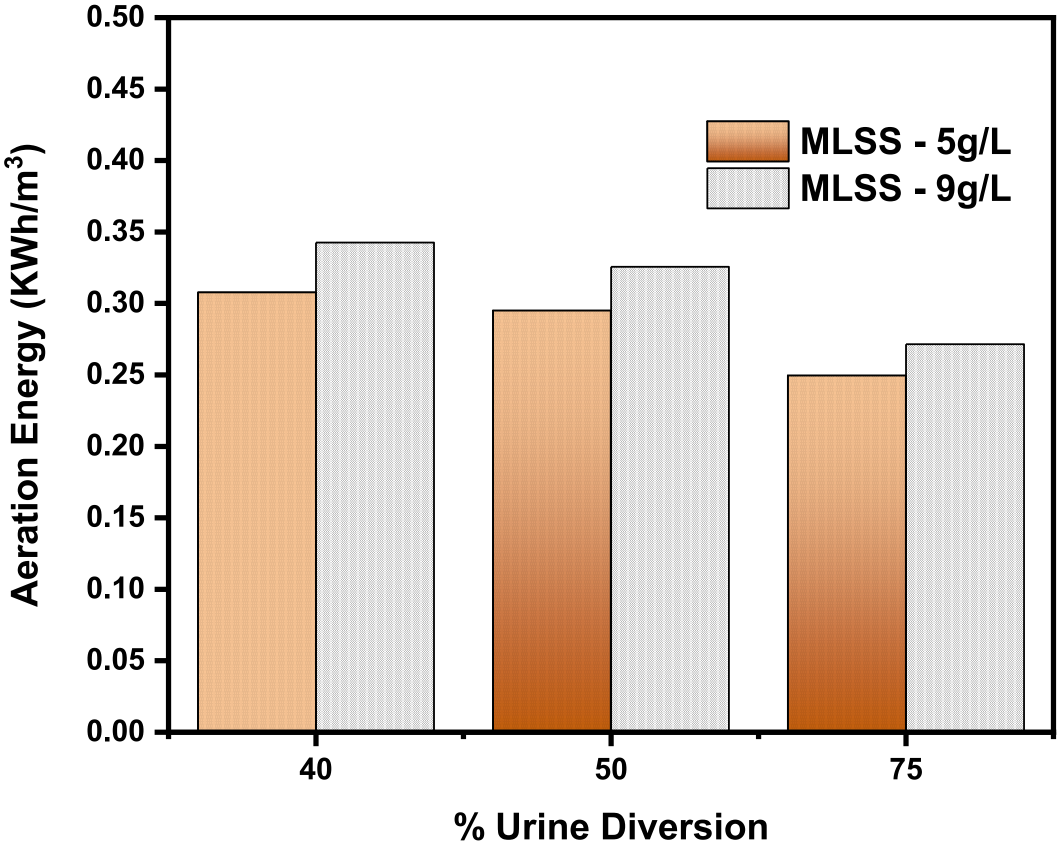

Fig. S1: Effect of operating MLSS concentration on aeration energy of Central Park WWTP

CHAPTER 5: Supplementary Information

Table S1- Kinetic parameters for AOB in the BioWin model.

Name	Default	Value	Arrhenius
Max. spec. growth rate [1/d]	0.9	0.65	1.072
Substrate (NH_4) half sat. [mgN/L]	0.7	0.7	1
Byproduct NH_4 logistic slope [-]	50	50	1
Byproduct NH_4 inflection point [mgN/L]	1.4	1.4	1
Denite DO half sat. [mg/L]	0.1	0.1	1
Denite HNO2 half sat. [mgN/L]	5.00E-06	5.00E-06	1
Aerobic decay rate [1/d]	0.17	0.17	1.029
Anoxic/anaerobic decay rate [1/d]	0.08	0.08	1.029
$KiHNO_2$ [mmol/L]	5.00E-03	1.46E-04	1

Table S2- Kinetic parameters for NOB in the BioWin model.

Name	Default	Value	Arrhenius
Max. spec. growth rate [1/d]	0.7	0.5	1.06
Substrate (NO_2) half sat. [mgN/L]	0.1	0.1	1
Aerobic decay rate [1/d]	0.17	0.17	1.029

Anoxic/anaerobic decay rate [1/d]	0.08	0.08	1.029
KiNH$_3$ [mmol/L]	0.075	0.0175	1

Table S3. Wastewater fractions used in the BioWin simulation.

Name	Default	Value
Fbs - Readily biodegradable (including Acetate) [gCOD/g of total COD]	0.16	0.25
Fac - Acetate [gCOD/g of readily biodegradable COD]	0.15	0.15
Fxsp - Non-colloidal slowly biodegradable [gCOD/g of slowly degradable COD]	0.75	0.75
Fus - Unbiodegradable soluble [gCOD/g of total COD]	0.05	7.00E-03
Fup - Unbiodegradable particulate [gCOD/g of total COD]	0.13	1.60E-03
Fcel - Cellulose fraction of unbiodegradable particulate [gCOD/gCOD]	0.5	0.5
Fna - Ammonia [gNH$_3$-N/gTKN]	0.66	0.9
Fnox - Particulate organic nitrogen [gN/g Organic N]	0.5	0.5
Fnus - Soluble unbiodegradable TKN [gN/gTKN]	0.02	0.02
FupN - N:COD ratio for unbiodegradable part. COD [gN/gCOD]	0.07	0.07
Fpo4 - Phosphate [gPO$_4$-P/gTP]	0.5	0.5
FupP - P:COD ratio for unbiodegradable part. COD [gP/gCOD]	0.022	0.022

Fsr - Reduced sulfur [H_2S] [gS/gS]	0.15	0.15
FZbh - Ordinary heterotrophic COD fraction [gCOD/g of total COD]	0.02	0.02
FZbm - Methylotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZao - Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	0.01
FZno - Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	0.01
FZaao - Anaerobic ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZppa - Phosphorus accumulating COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZpa - Propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZam - Acetoclastic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZhm - Hydrogenotrophic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZso - Sulfur oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZsrpa - Sulfur reducing propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZsra - Sulfur reducing acetotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04

FZsrh - Sulfur reducing hydrogenotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZe - Endogenous products COD fraction [gCOD/g of total COD]	0	0

Constants used in steady state simulation

FZao - Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	0.01
FZno - Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	0.01

Constants used in dynamic simulation

FZao - Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZno - Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04

Table S4. Theoretical estimation of oxygen consumption in scenario A and B

	Scenario A	Scenario B
Mass of N oxidised (kgN)	72	32
Mass of N denitrified (kgN)	65	0
Mass of oxygen consumed (kgO_2)	309.6	137.6

Mass of oxygen recovered (kgO_2)	179.568	0
Net oxygen (kgO_2)	130.032	141.9
Recirculation	Yes	No
Out put	Recycled water	Recycled water + Nitrified Urine