

# **Integrated Validation Plant: Modelling and optimisation of the biological treatment process**

**Puay Ni Qing**

for the degree of:

***Master of Science in Civil Engineering***

**Thesis report**

**June 2015**

## **Committee:**

Prof. Dr. ir. J.B. van Lier

Delft University of Technology,  
Sanitary Engineering Section

Assoc. Prof. Ng How Yong

National University of Singapore,  
Faculty of Civil and Environmental Engineering

Dr. ir. M. de Kreuk

Delft University of Technology,  
Sanitary Engineering Section

Dr. S. Pande

Delft University of Technology,  
Water Resources Section

Sanitary Engineering Section, Department of Water Resources  
Faculty of Civil Engineering and Geosciences  
Delft University of Technology

Department of Civil & Environmental Engineering  
Faculty of Engineering  
National University of Singapore

## Summary

The wastewater infrastructure in Singapore is undergoing a major revamp, which includes the construction of a new water reclamation plant (WRP) in Tuas. This plant aims to maximise the energy recovered from wastewater, with the ultimate aim of achieving energy self-sufficiency. The Integrated Validation Plant (IVP) is a 1000 m<sup>3</sup>/day pilot plant equipped with an AB configuration. The A-stage biosorption process increases the amount of COD recovered in the primary sludge to increase biogas production, whereas the B-stage membrane bioreactor (MBR) treats the wastewater using activated sludge. The effluent then undergoes reverse osmosis (RO) to produce potable water. This report investigates the biological treatment process in the IVP and aims to answer the following question: Given that most WRP in Singapore uses the Modified Ludzack-Ettinger configuration (MLE; also known as the pre-denitrification configuration) for the B-stage, will a change in configuration reduce net energy consumption while achieving good effluent quality?

The research is divided into three main parts with the following objectives:

1. To create and calibrate a Biowin model to simulate the biological treatment process in the IVP
2. To optimise the MBR B-stage by finding the configuration and operating conditions that requires the least influent COD to achieve acceptable effluent quality (in terms of COD, nitrogen and phosphorus), and to determine the appropriate changes in the A-stage (if any) that is required to achieve this effluent quality
3. To explore issues facing the IVP and future developments of wastewater treatment in Singapore – effect of an unaerated deoxygenation tank, feasibility of mainstream Anammox, and modification of the treatment process to reduce chemical dosage in the downstream RO process

The Biowin model for the MBR was first calibrated using jar tests and oxygen uptake tests, then validated with profile studies that are being carried out in the MBR regularly. It was found that Biowin is able to model the biological treatment process relatively well, with the major discrepancy being that Biowin is not able to model ammonia adsorption onto activated sludge in the anoxic zone. With the calibrated model, various configurations and operating conditions were simulated to optimise the MBR. It was found that the optimal configuration is the Anoxic-Oxic-Anoxic-Oxic (AOAO) configuration with a step feed of 50% of the influent entering the first anoxic zone and 50% entering the second anoxic zone. The return activated sludge (RAS) flow should be 73% of the influent flow and the DO in the aerated tanks should be maintained at 1 mg/l.

As the biosorption tank at the IVP is not functioning well, data was obtained from another biosorption pilot plant in Singapore to calibrate the A-stage. Using this calibrated biosorption model, the optimal MBR configuration, and the typical raw influent characteristics for the IVP, it was determined that a functioning biosorption stage will result in insufficient COD entering the MBR for sufficient phosphorus removal. To achieve the required effluent quality, the best option during steady state operations would be to reduce the sludge retention time (SRT) of the biosorption process from 0.5 days to 0.32 days by increasing sludge wastage as this leads to the highest overall COD recovery. On the other hand, the best method to cope with sudden increases in the nitrogen and phosphorus load in the raw influent would be to partially bypass the biosorption

tank and divert some of the raw influent to the primary clarifier as this option results in the fastest response in the effluent quality.

As the membrane tanks in the MBR are aerated to scour the membranes and minimise fouling, the mixed liquor in the tanks have a high concentration of dissolved oxygen (DO). A small deoxygenation tank is installed after the membrane tanks to allow DO to deplete before the mixed liquor is recycled back to the anoxic zone at the start of the MBR. However, the deoxygenation tank is not functioning well due to the presence of a cascade overflow from the membrane tanks. Simulations show that the presence of residual DO in the recycle stream leads to a large deterioration the biological phosphorus removal performance of the MBR, mainly due to reduced COD availability for the polyphosphate-accumulating organisms (PAOs). Modifications to reduce the impact of the cascade include the installation of baffle walls, slopes, or overflow pipes.

One of the technologies for B-stage that can allow even higher COD recovery is mainstream Anammox for nitrogen removal, which consumes less COD than conventional nitrogen-removal processes. As it works best at temperatures of about 30°C, it may be suitable for implementation in Singapore where the average wastewater temperature is about 31°C. Biowin simulations revealed that mainstream Anammox can occur under the local conditions, with CANON and DEMON configurations being more suitable than the SHARON-Anammox configuration. However, a very high solids removal efficiency in the primary clarifier is required to remove sufficient COD in the A-stage for Anammox to develop in the B-stage. Further studies would be required to determine if this is achievable under local conditions.

Lastly, the MBR effluent is sent to a RO process to be reclaimed as potable water. Ammonia and sodium hypochlorite is first added to produce chloramine to prevent biofouling and provide the necessary residual disinfection. It was found that the ammonia concentration in the MBR effluent can be increased sufficiently to remove the need for ammonia dosage. However, the reduced nitrification also resulted in a higher nitrite concentration in the effluent, increasing the amount of sodium hypochlorite needed to produce the required chloramine as sodium hypochlorite also oxidises nitrite into nitrate. In addition, the dosage of anti-scalants can be reduced with better phosphorus removal as the lower effluent phosphorus leads to less scaling potential in the RO process by calcium phosphate.

## Acknowledgements

This thesis report is the result of many months of experiments, modelling, and learning. It would not have been possible without the support of many people and I would like to express my heartfelt gratitude to them:

My thesis committee from TUD – **Prof Jules van Lier**, **Dr Merle de Kreuk**, and **Dr Saket Pande**, for the guidance rendered from more than 10,000 km away. Skype meetings are not the most conducive for thesis discussion and sometimes fraught with technical difficulties, but the assistance given during the limited time that we have were invaluable.

My supervisor from NUS – **A/P Ng How Yong**, for spending time to go through the results with me and giving me numerous suggestions to improve my project and report.

**Dr Tao Guihe** and **Daniel** from Public Utilities Board (PUB), for the discussions about the operations of the IVP and other WRP around Singapore. These gave me useful insights about the issues facing the IVP and ideas regarding potential topics which I could explore for the thesis.

**Dr Henri Spanjers**, for the help and advice on Biowin simulations.

**KK**, **Xiaodi**, and **Fu Chen**, for all the help rendered to me in the NUS lab, especially in teaching me the protocols for analysing samples and operating equipment.

**Agilan**, **Sok Hian**, **Benson**, **Benedict**, and the other PUB interns for showing me the way around the IVP as well as collecting and sampling data, some of which is used in the calibration and verification of the Biowin model.

And last but not least, my family and friends for their endless support during this period.

# Table of Contents

Summary.....	i
Acknowledgements .....	iii
Table of Contents .....	iv
List of figures .....	vi
List of tables.....	vii
Nomenclature.....	viii
1 Introduction.....	1
1.1 Background of wastewater treatment in Singapore .....	1
1.1.1 Deep Tunnel Sewerage Scheme (DTSS) .....	1
1.1.2 Integrated Validation Plant (IVP) .....	2
1.2 Background of topics for investigation .....	4
1.2.1 Nutrient removal .....	4
1.2.2 Biowin simulations.....	4
1.3 Research questions .....	5
1.4 Research methodology .....	5
1.5 Motivation of study.....	6
2 Literature review .....	7
2.1 Biosorption process.....	7
2.2 Biological Excess Phosphorus removal (Bio-P) .....	7
2.2.1 Classical theory of Bio-P removal .....	7
2.2.2 Alternative forms of Bio-P removal .....	8
2.2.3 Glycogen accumulating organisms (GAOs) .....	8
2.2.4 Factors affecting Bio-P removal.....	9
2.3 Nitrogen removal .....	10
2.3.1 Conventional nitrogen removal process.....	10
2.3.2 Alternative nitrogen removal processes.....	11
2.3.3 Nitrous oxide emission .....	13
2.3.4 Denitrifying PAO (DPAO).....	13
2.4 Computer modelling .....	14
3 Methods and materials.....	15
3.1 MBR operation .....	15
3.2 Model calibration and verification .....	15
3.2.1 Calibration and verification process .....	15
3.2.2 Biosorption .....	16
3.2.3 Nitrifier Oxygen Uptake Rate (OUR) test .....	16
3.2.4 Jar test.....	17
3.2.5 Profile studies .....	17
3.2.6 Laboratory analysis .....	18
3.3 Optimisation process .....	19
4 Results and discussion .....	20
4.1 Biowin calibration .....	20

4.1.1	Influent characteristics .....	20
4.1.2	Biosorption .....	20
4.1.3	Nitrifier OUR test .....	21
4.1.4	Jar test.....	22
4.1.5	Calibrated parameters .....	24
4.1.6	Validation of model .....	25
4.2	Optimal MBR configuration .....	28
4.2.1	Optimal configuration for existing MBR .....	28
4.2.2	Optimal configuration not limited to existing MBR pipe layout .....	31
4.2.3	Further optimisation .....	33
4.2.4	Sensitivity of effluent quality to MBR influent COD .....	38
4.3	Effect of biosorption A-stage .....	39
4.3.1	Effect of designed biosorption conditions .....	39
4.3.2	Operating changes to meet effluent limits .....	40
4.3.3	Sensitivity of MBR influent quality to biosorption process changes .....	42
4.4	Discussion of optimisation results .....	43
5	Further discussion.....	44
5.1	Identification of issues in the IVP .....	44
5.1.1	Bio-P removal.....	44
5.1.2	DO concentration in deoxygenation tank.....	45
5.2	Mainstream Anammox.....	51
5.2.1	SHARON-Anammox.....	51
5.2.2	CANON .....	52
5.2.3	DEMON .....	53
5.2.4	Results.....	53
5.3	Downstream RO process.....	55
5.3.1	Introduction .....	55
5.3.2	Combining secondary clarifier and ultrafiltration.....	55
5.3.3	Reduction of ammonia usage .....	55
5.3.4	Reduction of anti-scalant usage.....	56
5.3.5	Conclusion.....	56
6	Further studies.....	58
7	Conclusion .....	61
8	References .....	63
	Appendix A: Membrane fouling.....	66
	Appendix B: Influent sampling data.....	72
	Appendix C: Specifications for Biowin simulations.....	73
	Appendix D: Biosorption sensitivity analysis .....	75

## List of figures

Figure 1: Sewer tunnels and WRP under DTSS .....	1
Figure 2: Process flow of IVP .....	2
Figure 3: MBR schematic with location of online probes .....	3
Figure 4: Process of Bio-P removal .....	8
Figure 5: Main transformation reactions between different forms of nitrogen in wastewater .....	10
Figure 6: Biowin setup for biosorption process .....	16
Figure 7: Sampling locations for profile studies .....	18
Figure 8: Biowin setup for profile studies .....	18
Figure 9: Modelled and measured biosorption data for 2 Dec 2014 .....	21
Figure 10: Modelled and measured biosorption data for 7 Dec 2014 .....	21
Figure 11: Jar test with low effluent $\text{PO}_4\text{-P}$ concentration .....	22
Figure 13: Jar test with high effluent $\text{PO}_4\text{-P}$ concentration .....	22
Figure 12: Jar test at $\text{DO} = 2.5 \text{ mg/l}$ .....	23
Figure 14: 23 Sep 2014 profile study (MLE, 3d SRT) .....	25
Figure 15: 10 Jun 2014 profile study (MLE, 5d SRT) .....	25
Figure 16: 24 Jun 2014 profile study (MLE, 5d SRT) .....	26
Figure 17: 23 March 2015 profile study (AOAO, 5d SRT) .....	26
Figure 18: Design of existing MBR .....	28
Figure 19: Effluent $\text{PO}_4\text{-P}$ concentration for the AAO configuration .....	31
Figure 20: UCT configuration for MBR .....	32
Figure 21: MJHB configuration for MBR .....	32
Figure 22: Variation of required MBR influent COD and effluent TN with RAS in AOA1 configuration .....	34
Figure 23: Sensitivity analysis of influent COD on effluent $\text{PO}_4\text{-P}$ .....	38
Figure 24: Effect of bypass ratio on sCOD, rbCOD and VFA concentrations in MBR influent .....	40
Figure 25: Sensitivity of sCOD, TKN, and TP in the MBR influent to bypass ratio .....	42
Figure 26: Bio-P removal in May 2014 .....	44
Figure 27: Bio-P removal in Sep 2014 .....	45
Figure 28: Typical DO profile in the deoxygenation tank .....	46
Figure 29: Cascades in the deoxygenation tank .....	46
Figure 30: Effect of DO in deoxygenation tank on $\text{PO}_4$ concentrations in the MBR .....	48
Figure 31: Effect of DO in deoxygenation tank on $\text{NO}_3$ concentrations in the MBR .....	48
Figure 32: Effect of DO in deoxygenation tank on $\text{NH}_4$ concentrations in the MBR .....	48
Figure 33: Effect of DO in deoxygenation tank on COD concentrations in MBR .....	48
Figure 34: Installation of baffle walls in the deoxygenation tank to minimise cascade .....	50
Figure 35: Installation of slope in the deoxygenation tank to minimise cascade .....	50
Figure 36: Overflow pipes at beginning of Anoxic 1 tank .....	50
Figure 37: Biowin setup for the SHARON-Anammox configuration .....	52
Figure 38: Biowin setup for the CANON configuration .....	52
Figure 39: Biowin setup for the DEMON configuration .....	53
Figure 40: Biowin setup to produce $2 \text{ mg/l NH}_4\text{-N}$ in effluent .....	56
Figure 41: Biowin setup for the Anammox - MBR process .....	59

## List of tables

Table 1: MBR operating conditions at the IVP.....	15
Table 2: Biosorption parameters .....	16
Table 3: VFA concentrations in the influent samples .....	20
Table 4: Cation concentrations in the influent samples .....	20
Table 5: Maximum growth rate of nitrifiers .....	21
Table 6: Calibrated parameters .....	24
Table 7: Configurations considered in MBR optimisation .....	29
Table 8: Optimisation results for existing MBR layout .....	30
Table 9: Optimisation results not limited to existing MBR layout .....	33
Table 10: Optimisation results of MLE2 and AOA at various RAS flow .....	34
Table 11: Optimisation results for the AOA3 configuration at various DO concentrations.....	35
Table 12: Optimisation results at different influent step feed .....	37
Table 13: MBR influent and effluent quality with BPT .....	39
Table 14: Biowin results for the different alternatives.....	41
Table 15: Comparison of reference and optimised cases.....	43
Table 16: Bacterial population in WAS .....	48
Table 17: Modified kinetic parameters for Anammox in CANON configuration .....	52
Table 18: Comparison of Anammox alternatives .....	54
Table 19: Effluent quality of setup to produce 2 mg/l $\text{NH}_4\text{-N}$ in effluent .....	56



## Nomenclature

AAO	Anaerobic Ammonia Oxidisers (Anammox Bacteria)
Anammox	Anaerobic Ammonia Oxidation
AOAO	MBR configuration with Anoxic-Oxic-Anoxic-Oxic zones
AOB	Ammonia-Oxidising Bacteria
APAO	Actinobacteria PAO
Bio-P	Biological Excess Phosphorus Removal
BPT	Biosorption Tank
CANON	Completely Autotrophic Nitrogen-removal Over Nitrite
COD	Chemical Oxidation Demand
DEMON	De-ammonification
DO	Dissolved Oxygen
DPAO	Denitrifying PAO
DTSS	Deep Tunnel Sewerage Scheme
EBPR	Enhanced Biological Phosphorus Removal
EPS	Extracellular Polymeric Substances
GAO	Glycogen Accumulating Organisms
GLUE	General Likelihood Uncertainty Estimation
HRT	Hydraulic Retention Time
F/M ratio	Food/Mass Ratio
IVP	Integrated Validation Plant
KWRP	Kranji Wastewater Reclamation Plant
LCOCD-OND-UV	Liquid Chromatography coupled with Organic Carbon Detector – Organic Nitrogen Detector – Ultraviolet Detector
MBR	Membrane Bioreactor
MBT	Membrane Tanks
MJHB	Modified Johannesburg
MLE	Modified Ludzack-Ettinger
MLSS	Mixed Liquor Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
NOB	Nitrite-Oxidising Bacteria
OHO	Ordinary Heterotrophic Organisms
OUR	Oxygen Uptake Rate
PAO	Phosphorus Accumulating Organisms
PHA / PHB	Polyhydroxyalkanoate / Polyhydroxybutyrate
RAS	Return Activated Sludge
rbCOD	readily biodegradable COD
RO	Reverse Osmosis
RPAO	Rhodocyclus-related PAO
sCOD	Soluble COD
SMP	Soluble Microbial Products
SHARON	Single reactor system for High activity Ammonium Removal Over Nitrite
SRT	Sludge Retention Time
tCOD	Total COD
TKN	Total Kjeldahl Nitrogen
TMP	Transmembrane Pressure
TN	Total Nitrogen
TOC	Total Organic Carbon

TP	Total Phosphate
TWRP	Tuas WRP
TSS	Total Suspended Solids
UCT	University of Cape Town
UPWRP	Ulu Pandan Wastewater Reclamation Plant
VFA	Volatile Fatty Acids
WAS	Waste Activated Sludge
WRP	Water Reclamation Plant

Explanation of terms:

Raw influent	Influent entering the A-stage from the equalisation tank of UPWRP
MBR influent	Influent entering the B-stage after the primary clarifier

# 1 Introduction

## 1.1 Background of wastewater treatment in Singapore

### 1.1.1 Deep Tunnel Sewerage Scheme (DTSS)

The DTSS is an overhaul of the used water infrastructure in Singapore in order to meet its used water needs in an economical and sustainable manner. The main concept of the DTSS is to build deep sewers to allow used water to flow to the water reclamation plants (WRP) solely under gravity, eliminating the need for intermediate pumping stations. In addition, the existing WRPs will be consolidated, reducing the total number from 6 to 3. As a result, the land space required by the used water infrastructure will be halved to about 150 hectares, freeing up land for higher-value development.

The first phase of DTSS was started in 2001 and completed in 2008. It consists mainly of 48 km of deep sewer tunnels and the Changi WRP at the eastern end of the island. The second phase of DTSS includes an extension of the deep sewer tunnels and a new Tuas WRP (TWRP) at the western end of the island. TWRP has a designed capacity of 800,000 m<sup>3</sup>/day (600,000 m<sup>3</sup>/day domestic wastewater and 200,000 m<sup>3</sup>/day industrial wastewater) and is expected to be completed in 2022.

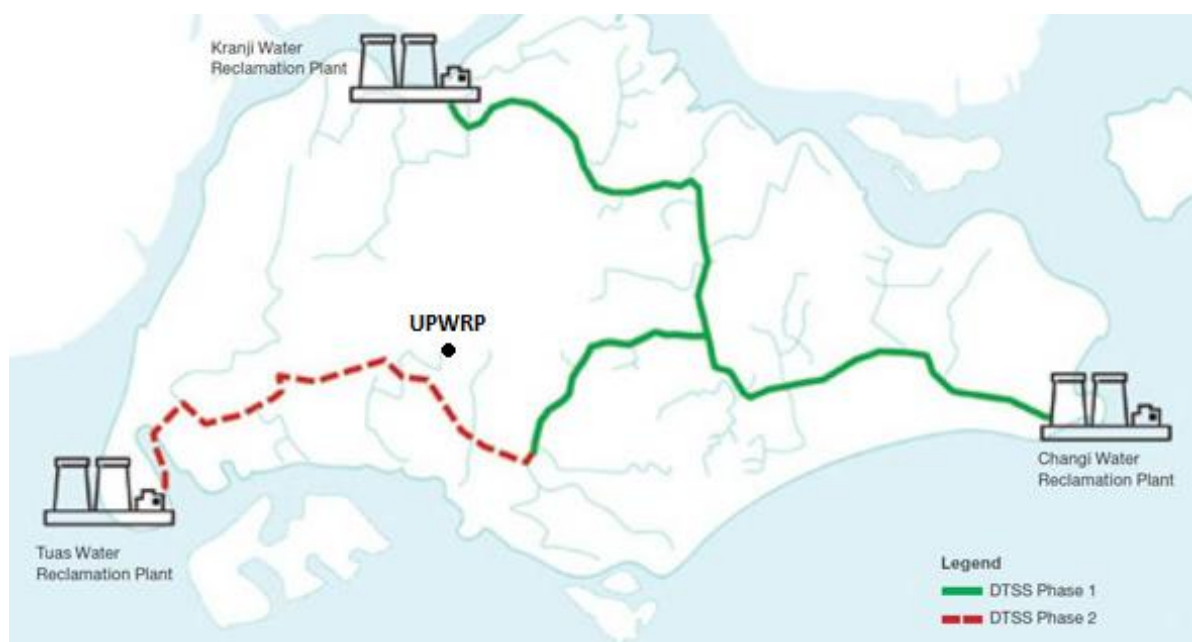


Figure 1: Sewer tunnels and WRP under DTSS (<http://www.pub.gov.sg/DTSS>)

One of the aims of DTSS Phase 2 is to incorporate technologies that are more energy-efficient and compact than the existing WRP, reducing both the energy and land footprints. These technologies are currently being tested at various WRP in Singapore. One of these projects is the Integrated Validation Plant (IVP) located at the Ulu Pandan WRP (UPWRP). 90% of the wastewater originates from domestic sources, while the rest comes from a mix of commercial and industrial sources. UPWRP will be decommissioned after TWRP opens. The results from the IVP will be used to design the domestic wastewater treatment process at TWRP, which is expected to receive similar domestic sewage composition as UPWRP.

### 1.1.2 Integrated Validation Plant (IVP)

The IVP is a pilot plant with a capacity of 1000 m<sup>3</sup>/day and takes influent from the main equalization tank of UPWRP. It operates at a constant influent flow rate, except when the influent pump is stopped due to operational controls. Figure 2 shows the main processes within the IVP, which works in an AB configuration. In the A-stage, the influent undergoes biosorption before undergoing primary clarification. It is then sent to the B-stage consisting of a membrane bioreactor (MBR). The effluent from the MBR is sent to a reverse osmosis (RO) unit to reclaim the effluent as potable water.

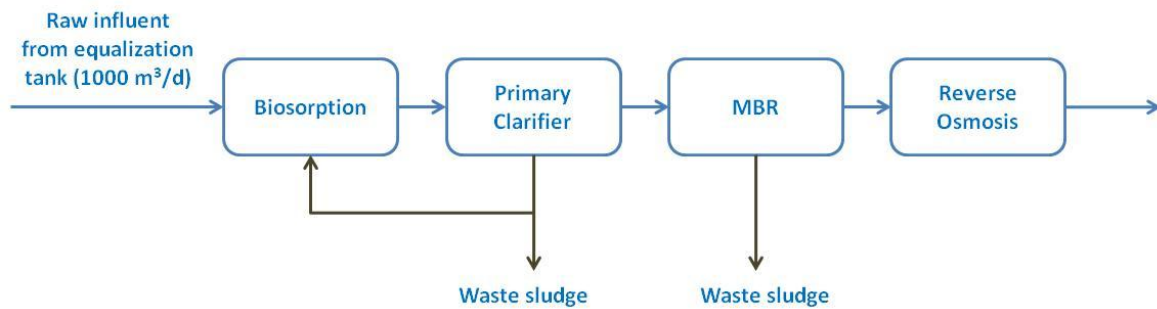


Figure 2: Process flow of IVP

#### *A-stage biosorption*

The A-stage being tested in the IVP aims to maximise the amount of COD in the waste sludge in order to maximise energy recovery through biogas production during sludge digestion. This is achieved through a biosorption process, where biomass growth is used to consume soluble COD (sCOD) and capture colloidal particles to increase the COD removal from the influent. This is achieved through rapid growth of biomass that secretes extracellular polymeric substances (EPS) that binds colloidal particles in the wastewater in order to remove them in the primary clarifier. The biosorption tank has a hydraulic retention time (HRT) of about 30 min, a sludge retention time (SRT) of about 12 hours, and is also kept at a low dissolved oxygen (DO) concentration of about 0.3 mg/l. These values agree well with the typical design parameters for biosorption (Fei, 2012).

However, the biosorption process in the IVP has been largely unsuccessful, showing no significant difference in performance compared to primary settling. This is likely to be due to the total suspended solids (TSS) concentration in the influent. Because of the diurnal pattern in wastewater flow, the sediments at the bottom of the equalisation tank are stirred up by the increased flow every morning and are then pumped into the IVP. This causes daily shock loadings of up to 7000 mg-TSS/l in the influent, compared to an average of about 200 mg-TSS/l. This causes a large portion of biomass in the biosorption tank to be flushed out every morning. As the aerators in the biosorption tank are able to maintain a DO of at least 0.2 mg/l, it is unlikely that the high organic load in the influent caused a low DO concentration in the biosorption process that resulted in a lack of COD uptake by the biomass.

### B-stage membrane bioreactor (MBR)

The IVP uses a submerged MBR with hollow-fibre membranes and operates in a plug-flow manner. Figure 3 shows the layout of the MBR, including the location of online probes and the measured parameters. The volume and HRT of the individual tanks are given at the bottom of the figure. The anoxic and aerobic tanks can only operated in the unaerated and aerated modes respectively. The swing tanks can operate in both aerated and unaerated modes as they are equipped with both diffusers and stirrers. A small deoxygenation tank is installed after the membrane tanks (MBT) for the DO to be depleted before the return activated sludge (RAS) is pumped back to the anoxic zone. Sludge is wasted from the RAS pipe after the deoxygenation tank.

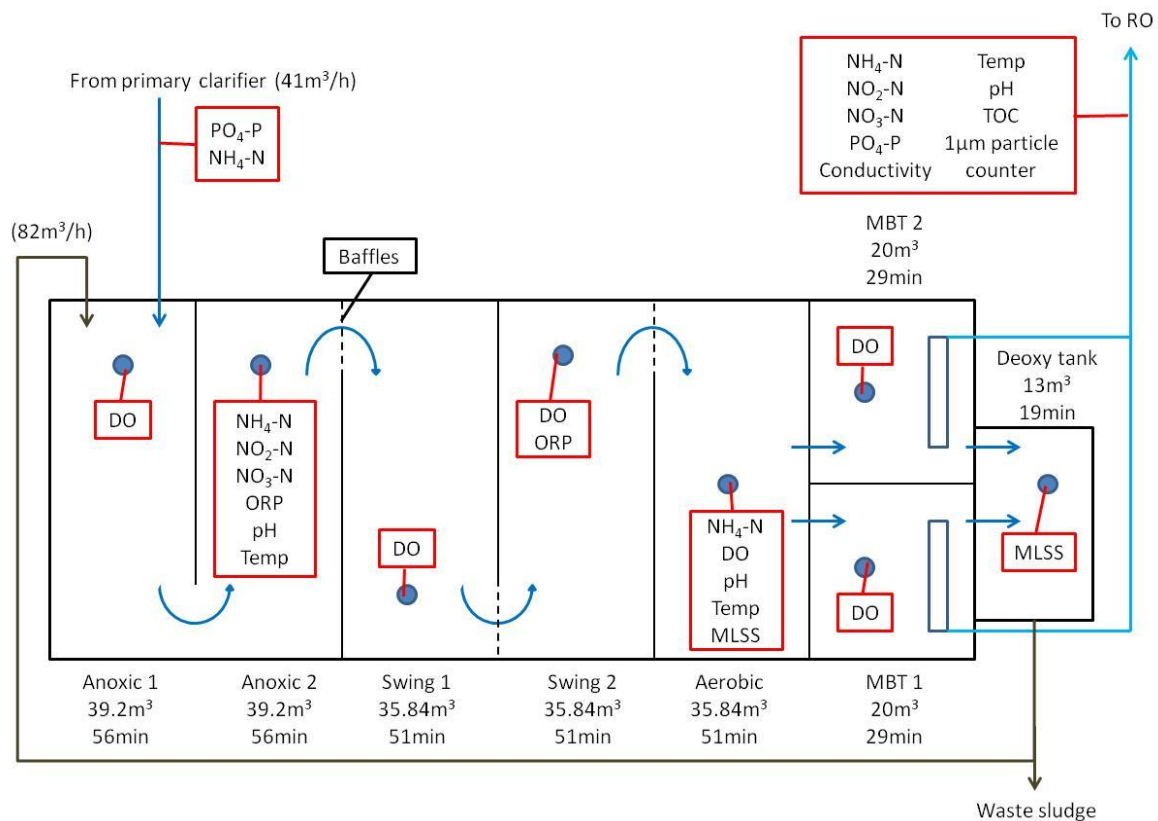


Figure 3: MBR schematic with location of online probes

### MBR Membranes

The MBR membranes are subjected to cycles of filtration and relaxation to reduce fouling. They are also backwashed with permeate during the twice-weekly maintenance clean. The transmembrane pressure (TMP) across the hollow fibre membranes is continuously monitored to identify any increase in fouling. Under normal conditions, the membranes operate with a flux of 25 l/(m²h) and a TMP of about 10 kPa, and a chemical recovery clean is performed whenever the TMP exceeds 20 kPa.

## 1.2 Background of topics for investigation

### 1.2.1 Nutrient removal

Phosphorus and nitrogen are nutrients that are present in small quantities in natural habitats, supporting the growth of aquatic plants that provide food for aquatic animals. However, they are detrimental to the environment when present in large quantities, causing algae blooms and eutrophication. This results in decreased food and oxygen levels in rivers that can lead to the death of aquatic organisms. The water quality may also be negatively affected, making it unfit for human consumption. Hence, care must be taken that large quantities of nutrients are not discharged into the natural waters, especially from wastewater treatment plants that discharges large quantities of effluent relative to the base flow.

Phosphorus is present in wastewater mainly in the form of ortho-phosphate ( $\text{PO}_4$ ), which mostly originates from washing powders based on polyphosphates and from proteins due to human consumption of meat and dairy products (van Haandel & van der Lubbe, 2012). The phosphorus concentration is between 5 and 10 mg/l in the typical municipal wastewater. This can be removed from wastewater via two methods – chemical treatment with dosage of metal salts, or biological treatment using activated sludge. The former option is more commonly used currently, and involves chemical precipitation using alum or iron salts. The latter option involves the incorporation of phosphorus into activated sludge and phosphorus is then removed from wastewater through sludge wastage. This process has gained in popularity since the first successful implementations in the early 1980s due to the reduced chemical costs and lower overall sludge production (Tchobanoglous, Burton, & Stensel, 2003).

On the other hand, nitrogen is present in wastewater mainly in the form of ammonia ( $\text{NH}_4^+$ ) and organic nitrogen such as urea and amino acids (van Haandel & van der Lubbe, 2012). In the conventional process, organic nitrogen is first broken down into ammonia, and then removed from wastewater through nitrification and denitrification. Nitrification oxidises ammonia to nitrite ( $\text{NO}_2^-$ ) and nitrate ( $\text{NO}_3^-$ ) in the presence of oxygen, whereas denitrification then reduces both to nitrogen gas ( $\text{N}_2$ ) in the presence of rbCOD (Tchobanoglous et al., 2003).

### 1.2.2 Biowin simulations

Biowin is a computer simulation package that is able to dynamically simulate the wastewater treatment process. It combines various published models, such as the activated sludge models, anaerobic digestion model, and solids settling model, into a single simulation platform (EnviroSim Associates Ltd). It is a flexible platform that allows investigations into changes in configurations or operations without having to alter conditions in the actual treatment plant.

### 1.3 Research questions

The main research goal of the thesis is to optimise the biological treatment process in the IVP with the aim of maximising the amount of COD that is recovered for sludge digestion. The report seeks to answer the following question: Given that most WRP in Singapore uses the MLE configuration for secondary wastewater treatment, will a change in configuration reduce net energy consumption while achieving good effluent quality?

The sub-questions that will be answered in the report are:

1. Given the high wastewater temperature in Singapore, is Biowin able to model the IVP for use in investigating the effect of changes in configuration?
2. With the existing MBR at the IVP, can changes in configuration and operating conditions reduce the net energy consumption while maintaining effluent quality in terms of COD, nitrogen and phosphorus?
3. Which configuration for mainstream Anammox would be the most suitable for Singapore, considering the limited land space and aim of reducing net energy consumption?
4. How can the secondary wastewater treatment process be modified to reduce chemical dosage for the downstream RO process?

### 1.4 Research methodology

This research is split into three parts – model calibration, plant optimisation, and further exploration of issues facing the IVP and wastewater treatment in Singapore. To calibrate biosorption, data was obtained from the 'Energy+' project at Kranji WRP, which has a stable biosorption pilot plant. On the other hand, nitrifier oxygen uptake rate (OUR) tests and jar tests were conducted on the activated sludge from the IVP to calibrate the MBR B-stage. In addition, profile studies of the MBR were carried out regularly, measuring the sCOD,  $\text{NH}_4$ ,  $\text{NO}_2$ ,  $\text{NO}_3$ , and  $\text{PO}_4$  concentrations at various points in the MBR. Together with the data from the online monitoring program SCADA, the profile studies were used to validate the Biowin model. Additional analysis of both the raw influent from the equalisation tank and the MBR influent from the biosorption process were also carried out to determine the influent characteristics in more detail.

After calibration and validation, the Biowin model was used to optimise the MBR. As the IVP aims to maximise energy recovery from wastewater, the optimal configuration is defined as the one that requires the lowest COD in MBR influent for nutrient removal, hence allowing the biosorption tank to recover the maximum amount of COD. This was identified by simulating various configurations and operating conditions. Finally, the A-stage was added to the Biowin model to investigate how a functioning biosorption stage would affect nutrient removal in the MBR, and if any changes to the biosorption process are required to achieve good effluent quality. The optimal operation of the entire IVP for maximum COD recovery in the waste sludge would also be investigated.

Finally, additional issues facing the IVP and wastewater treatment in Singapore will be explored under the section 'further discussion'. As the deoxygenation tank in the IVP is not functioning well, the Biowin model was

used to investigate the effect of a functioning biosorption tank on the biological treatment process in the MBR. A comparison of various mainstream Anammox configurations was also carried out as Singapore is looking towards this technology to reduce the energy footprint of the WRP. In addition, studies were performed to determine if operational changes in the IVP can benefit the downstream RO water reclamation process through a reduction in chemical dosage.

## 1.5 Motivation of study

The high wastewater temperature in Singapore provides a different set of challenges in wastewater treatment from previous research, most of which are conducted at lower temperatures. In particular, biological nutrient removal from wastewater in warm climates is very different from that in cooler climates. For example, full nitrification can be observed at an SRT of less than 3 days due to a much higher nitrifier activity. Hence, it is interesting to create a model to investigate the impact of various operating conditions and MBR configurations on the biological treatment performance under local conditions.

The available chemical energy content of the raw influent is conservatively estimated to be around 1.7 kWh/m<sup>3</sup> (Lee et al., 2013). This is several times more than the energy needed to treat the wastewater itself. However, almost all wastewater treatment plants (WWTP) are net consumers of energy as the commonly used processes are energy intensive and are not very efficient at recovering the chemical energy available in the influent. The Public Utilities Board (PUB; waterboard in Singapore) is currently investigating technologies that can minimise the net energy footprint of the WRPs, with the ultimate aim of achieving energy self-sufficiency like the Strauss WWTP in Austria. The results from this report will go towards identifying treatment processes and operating conditions that can contribute towards this goal.



## 2 Literature review

### 2.1 Biosorption process

The biosorption process is first invented and patented by Böhnke in the 1970s, and is currently used in many wastewater treatment plants in Germany. It is more commonly known as the A-stage in the AB process, where A refers to Adsorption and B refers to Belebung (=aeration). The feed to the A-stage is high in COD and the rapidly growing organisms form a net to filter particles out of the wastewater (Verdenius & Broeze, 1999). The B-stage operates with a high sludge age for nitrification and removal of slowly biodegradable COD.

The AB process has an advantage in that most of the organic load is removed in the A-stage together with many of the harmful substances. However, the disadvantage is that biological nitrogen and phosphorus removal in the B-stage may be negatively affected due to a lack of COD (Jördening & Winter, 2005).

### 2.2 Biological Excess Phosphorus removal (Bio-P)

#### 2.2.1 Classical theory of Bio-P removal

Normal biomass in wastewater sludge contains about 2.5% phosphorus. Hence, there is a certain amount of phosphorus removed from the wastewater through normal sludge wastage. Under ideal conditions, this usually accounts for between 2 mg/l and 7 mg/l of phosphorus removed (van Haandel & van der Lubbe, 2012). This is inadequate to meet effluent guidelines, which typically ranges from 0.1 mg-P/l to 2.0 mg-P/l (Tchobanoglous et al., 2003). However, studies have revealed the presence of organisms that can uptake more phosphorus than usual and can thus remove sufficient amounts of phosphorus from the wastewater.

Biological excess phosphorus removal (Bio-P removal) is the process where phosphorus is removed through the uptake of phosphorus in excess of the usual 2.5%, also known as luxury phosphorus uptake. These organisms are known as polyphosphate accumulating organisms (PAOs), and can have phosphorus mass fractions of up to 38%. (van Haandel & van der Lubbe, 2012)

Figure 4 shows the process of Bio-P removal. Under anaerobic conditions, the PAOs uptake VFA and stores energy within the cell as polyhydroxybutyrate (PHB), which is a type of polyhydroxyalkanoate (PHA). The energy required is supplied by the degradation of glycogen and breaking of polyphosphate (poly-P) bonds, releasing phosphate into the water (Zeng, Saunders, Yuan, Blackall, & Keller, 2003). Under the subsequent aerobic (or anoxic) conditions, PHAs are metabolised and the energy is used in the presence of electron acceptors for cell metabolism and biomass growth, taking up phosphate to form poly-P and generate glycogen in the process (Oehmen, Carvalho, Lopez-Vazquez, van Loosdrecht, & Reis, 2010). The net growth of biomass and wastage of phosphate-rich sludge result in a net removal of phosphate from the water (Tchobanoglous et al., 2003).

An anaerobic zone is essential for the growth of PAOs. Firstly, it allows for the fermentation of a portion of biodegradable COD into VFA for assimilation by PAOs (Tchobanoglous et al., 2003). Next, heterotrophs

consume COD much faster than PAOs in the presence of electron acceptors such as oxygen and nitrate, and an anaerobic zone is essential to allow PAOs to uptake VFA without competition from heterotrophs. This is often achieved by feeding influent into an anaerobic zone at the beginning of the secondary treatment (van Haandel & van der Lubbe, 2012). However, high phosphorus removal has been observed in various WRPs in Singapore in the absence of such an anaerobic zone. The removal has been found to be due to Bio-P, as jar tests have revealed that the sludge exhibit the typical phosphorus release and uptake cycle of PAOs.

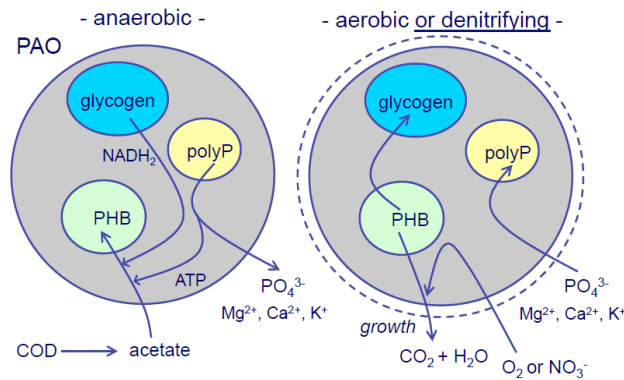


Figure 4: Process of Bio-P removal

### 2.2.2 Alternative forms of Bio-P removal

Recent studies into the various strains of PAOs have revealed that not all strains follow the classical theory as described in Section 2.2.1. Kong, Nielsen, and Nielsen (2005) have identified two major strains of PAOs - Rhodocyclus-related PAO (RPAO) and Actinobacteria PAO (APAO). RPAOs follow the classical model, taking up VFAs under anaerobic conditions and storing the energy as PHAs. However, APAOs are able to utilise more complex substrates such as certain amino acids and do not generate PHAs. The actual storage compound is still unknown, although glycogen has been suggested as a possible storage media (Kristiansen et al., 2013). More studies are required to determine the metabolic pathway for the APAOs.

The classical theory was developed mainly with batch tests or sequencing batch reactors (SBR), often with acetate or other VFAs as the carbon source for the PAOs. As a result, this selects for the RPAOs instead of APAOs. Indeed, Zengin, Artan, Orhon, Satoh, and Mino (2011) found that using glutamate (an amino acid) as the carbon source selects for APAOs and lower concentrations of PHA were observed. Studies have found that the APAO population can be quite significant in actual WWTPs with complex influent feed, and sometimes even dominate over the RPAOs in the activated sludge (Beer, Stratton, Griffiths, & Seviour, 2006; Kong, Xia, Nielsen, & Nielsen, 2006).

### 2.2.3 Glycogen accumulating organisms (GAOs)

Besides PAOs, there are other many other organisms involved in biological wastewater treatment. One particular group of interest are the glycogen accumulating organisms (GAOs), which thrive in similar anaerobic conditions that are conducive for the growth of PAOs. Unlike PAOs, GAOs utilise glycogen as the sole energy source to assimilate VFA during anaerobic conditions. Due to the absence of poly-P, they do not uptake phosphorus during aerobic (or anoxic) conditions. As a result, they do not perform Bio-P removal and is

sometimes the reason behind the failure of Enhanced Biological Phosphorus Removal (EBPR) processes (Oehmen, Saunders, Vives, Yuan, & Keller, 2006). Hence, GAOs are undesirable in (BPR systems as they compete directly with PAOs for VFAs during the anaerobic phase. Kong et al. (2006) have found that GAOs and PAOs constitute up to 6% and 27% of biomass respectively in EBPR plants.

#### **2.2.4 Factors affecting Bio-P removal**

Extensive research has been done on Bio-P removal in the last two decades, including factors that play a part in influencing the phosphorus removal process. Most of the research focuses on the competition between PAOs and GAOs in the biological treatment system.

One major factor affecting Bio-P removal is the pH of the water. It was found that PAOs tend to dominate over GAOs in wastewater with a pH above 7.25 in the anaerobic zone (van Haandel & van der Lubbe, 2012). One explanation for this is that GAOs has an optimal pH of 6.5 and consume more energy to assimilate VFAs at higher pH, causing them to be less efficient at VFA assimilation (Filipe, Daigger, & Grady, 2001). On the other hand, PAOs are able to regulate the internal cellular pH through phosphate release, and this can compensate for the increased electric potential gradient across the cell membrane results at high external pH (Oehmen et al., 2010).

Another significant factor is the temperature of the water. Studies have observed higher phosphorus removal rates at lower temperatures due to it favouring PAOs over GAOs, with PAOs dominating over GAOs at temperatures below 20°C (Oehmen et al., 2007; van Haandel & van der Lubbe, 2012). This has been found to be potentially due to the metabolism pathway of the GAOs for glycolysis, which are able to switch from the Entner-Doudoroff (ED) pathway to the Embden-Meyerhof-Parnas (EMP) pathway at higher temperatures. Glycolysis plays only a small role in PAOs as their metabolism proceeds mainly through the hydrolysis of poly-P, but research has found that they can only utilise the ED pathway for glycolysis (Oehmen et al., 2010).

Other factors include the SRT, VFA composition, influent COD, and cation concentrations. For instance, the optimal SRT was found to be 10 days, with incomplete phosphorus uptake occurring at an SRT of 5 days and reduced fraction of PAOs in sludge at SRTs longer than 10 days. The Bio-P removal process also reacts differently to different VFAs, with the highest rate of phosphorus release observed with acetate and propionate (Mulkerrins, Dobson, & Colleran, 2004). Between these two VFA, Oehmen et al. (2006) found that acetate favours the growth of GAOs whereas propionate favours the growth of PAOs. In addition, while COD is necessary for Bio-P to occur, influent with very high COD values have been found to be detrimental to the process as it favours GAOs over PAOs (López-Vázquez, Hooijmans, Brdjanovic, Gijzen, & van Loosdrecht, 2008; Mulkerrins et al., 2004). Lastly, the uptake and release of phosphate requires cations such as potassium and magnesium (Figure 4), and Mulkerrins et al. (2004) found that their availability affects the efficiency of the Bio-P removal process.



nitrate reducing enzymes and inhibits denitrification (Kampschreur, Temmink, Kleerebezem, Jetten, & van Loosdrecht, 2009). In addition, denitrification requires the presence of electron donors, which are mainly rbCOD in the wastewater (van Haandel & van der Lubbe, 2012).

### 2.3.2 Alternative nitrogen removal processes

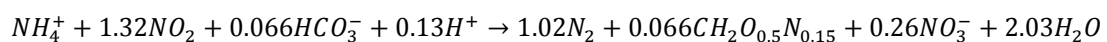
The conventional method of nitrogen removal is a relatively simple and robust system. However, it has high aeration requirements for nitrification and high COD requirement for denitrification (Mota, Head, Ridenoure, Cheng, & Francis, 2005). This increases the energy required for aeration and reduces the energy that can be recovered as COD in the waste sludge. Furthermore, external COD dosage may also be needed if the post-denitrification configuration is implemented (van Haandel & van der Lubbe, 2012). Hence, research is now focused on more energy-efficient means of nitrogen removal.

#### *Nitrite shunt (Nitrification-denitrification)*

The first alternative is to remove nitrogen through nitrification and denitrification, also known as nitrite-shunt. Similar to the conventional nitrification-denitrification process, this process involves oxidation of ammonia to nitrite using AOB, but without further oxidation to nitrate using NOB. The formed nitrite is then denitrified directly to nitrogen gas by regular denitrifiers. Henze, van Loosdrecht, Ekama, and Brdjanovic (2008) reported that this process requires 25% less oxygen and 40% less COD than the conventional nitrification-denitrification process, resulting in significant energy savings. The reduction of nitrite-oxidation can be achieved by a combination of a low DO concentrations in the aerated zones and rapid denitrification of produced nitrite, for instance by using an intermittently aerated reactor (Mota et al., 2005). However, this process may also lead to higher production of nitrous oxide (see section 2.3.3). Nitrite-shunt is currently being used at Changi WRP in Singapore.

#### *Anaerobic Ammonium Oxidation (Anammox)*

Another alternative technology for nitrogen removal is through Anaerobic Ammonium Oxidation (Anammox). This process uses ammonia as the electron donor and nitrite as the electron acceptor in the following reaction (Suneethi, Sri Shalini, & Joseph, 2014):



Anammox bacteria (Anaerobic Ammonia Oxidisers, AAO) operate at an optimal temperature of between 30°C and 37°C and an optimal pH of between 7 and 8.5. Compared to the conventional nitrification-denitrification process, the Anammox process consumes 60% less oxygen and 89% less organic matter, resulting in even higher savings compared to nitrite shunt. (van Haandel & van der Lubbe, 2012)

This process is currently widely applied in treating sludge dewatering centrate as it has high temperature, low COD, and the proper  $NH_4$ /alkalinity ratio (van Dongen, Jetten, & van Loosdrecht, 2001). It is also sometimes used in the treatment of animal waste due to the high nitrogen concentrations in the wastewater (Waki, Tokutomi, Yokoyama, & Tanaka, 2007). Recently, there have been numerous research on implementing

Anammox on mainstream wastewater treatment, and trials are currently undergoing at several plants in Europe such as the Dokhaven WWTP in the Netherlands (Fei, 2012), Strauss WWTP in Austria, and Glarnerland WWTP in Switzerland (Wett, Nyhuis, Takács, & Murthy, 2010). However, one challenge that is faced there is the low wastewater temperature, which can go below 11°C in winter in the Netherlands (Fei, 2012). This leads to very slow growth of AAO and hence poor nitrogen removal through Anammox.

#### Reactor configurations for Anammox

Based on the above equation, the ratio of ammonia to nitrite should ideally be 1:1.32. This ratio is almost never encountered in wastewater as nitrite is usually not present in significant amounts. Hence, Anammox is often coupled with partial nitrification of ammonia in order to generate the required ratio. Several reactor configurations have been designed to achieve this.

One popular two-stage design is the SHARON-Anammox process. The Single reactor system for High activity Ammonium Removal Over Nitrite (SHARON) process was originally developed for nitrogen removal through nitrification-denitrification. The SHARON reactor is operated without sludge retention and with a HRT of about 1 day to wash out NOB and prevent nitrite oxidation (van Kempen, Mulder, Uijterlinde, & van Loosdrecht, 2001). In addition, the DO concentration is also kept at about 1 mg/l (L. Zhang, Zheng, Tang, & Ren-cun, 2008). The SHARON effluent with a nitrite/ammonia ratio of about 1.32 then enters the unaerated Anammox reactor, where they are converted to nitrogen gas through Anammox. On the other hand, heterotrophs would utilise the available COD to denitrify any produced nitrate (van Haandel & van der Lubbe, 2012). This configuration has the advantage in that the partial nitrification and Anammox reactions are now uncoupled, making it more effective than the other configurations if complete nitrogen removal is required (Hao, Heijnen, & van Loosdrecht, 2002).

A common one-stage design is Completely Autotrophic Nitrogen-removal Over Nitrite (CANON), which integrates partial nitrification and Anammox into a single reactor. In this configuration, a DO profile is generated in the biomass using biofilm or granular sludge. This allows for nitrification by AOB in the aerobic layer and Anammox by AAO in the anoxic layer within the same aerobic reactor. However, a fine balance needs to be established among the AOB, NOB, and AAO in the reactor by manipulating the DO and ammonia in the influent. When the ammonia is too low or DO is too high, the influent ammonia is exhausted too quickly, favouring nitrite consumption by NOB instead of AAO and resulting in nitrification instead of Anammox (L. Zhang et al., 2008).

De-ammonification (DEMON) is another one-stage design that commonly uses a SBR with intermittent aeration to achieve cycles of aerobic and anoxic conditions, allowing successive occurrence of partial nitrification and Anammox. This system is simpler to build and operate than CANON, and the aeration is often controlled by online measurements of pH and DO (van Haandel & van der Lubbe, 2012). An alternative configuration of the DEMON process is to use consecutive aerated and unaerated tanks instead of an SBR with intermittent aeration, and this has been successfully implemented on a demonstration scale at the Glarnerland WWTP in Switzerland (Wett et al., 2010).

A comparison of various lab-scale studies shows that the CANON and DEMON configurations have maximum nitrogen removal rates of 1.5 and 0.77 kg-N/(m<sup>3</sup>-reactor.day) respectively (H. Zhang, Hashmi, Zhang, & Zhang, 2014). The lower efficiency of DEMON could be attributed to the fact that AOB and AAO are only active in the aerated and unaerated conditions respectively in the DEMON process, whereas they are active all the time in the CANON process. However, full-scale implementations of the CANON, DEMON, and SHARON-Anammox configurations shows no significant differences in performance, with all plants achieving nitrogen removal rates of between 0.4 and 0.6 kg-N/(m<sup>3</sup>-reactor.day) (Jaroszynski & Oleszkiewicz, 2011).

### 2.3.3 Nitrous oxide emission

One notable by-product of the conventional nitrification-denitrification process is nitrous oxide (NO + N<sub>2</sub>O), which is an undesirable greenhouse gas as it is about 300 times stronger than carbon dioxide (Solomon, 2007). Nitrous oxides can be produced by AOB through nitrifier denitrification, where nitrite is reduced to N<sub>2</sub>O (Kampschreur et al., 2009). It is also produced during denitrification as it is an intermediate product before nitrogen gas is produced.

Kampschreur et al. (2009) also reported on various factors affecting nitrous oxide production during the wastewater treatment process. They identified that the three main operation parameters that affects nitrous oxide production are DO in the aerated tanks, nitrite concentrations in the mixed liquor, and the influent COD/N ratio. A low DO leads to increased nitrous oxide production due to nitrifier denitrification in the oxygen limited conditions, with a DO of below 1 mg/l found to cause 10% of the nitrogen load being released as nitrous oxide. Similarly, a high nitrite concentration in the mixed liquor also leads to higher nitrifier denitrification in the aerobic zone. On the other hand, a low influent COD/N ratio increases nitrous oxide production due to incomplete denitrification, but this factor is not likely to be an issue in the IVP due to a high COD/N ratio of about 10 in the MBR influent. Generally, full-scale treatment plants that are designed for low effluent TN (< 10 mg-N/l) generates lower amounts of nitrous oxide (Foley, De Haas, Yuan, & Lant, 2010).

### 2.3.4 Denitrifying PAO (DPAO)

Even though nitrogen and phosphorus are removed from wastewater via different processes, they are not mutually exclusive due to the existence of denitrifying PAOs (DPAOs), which is a subset of PAOs. In addition to using oxygen as an electron acceptor under aerobic conditions, DPAOs are also able to utilise nitrate during anoxic conditions, hence performing denitrification (Zeng et al., 2003). These organisms are desirable in the IVP as they can denitrify and uptake phosphorus simultaneously, hence reducing the COD required for nutrient removal. However, the phosphorus uptake rate is slower under anoxic conditions compared to aerobic conditions, with the Activated Sludge Model No. 2d (ASM2d) recommending a rate reduction factor of 0.6 for anoxic conditions (Henze et al., 1999).

DPAOs grows much slower than heterotrophs, with a Biowin default maximum growth rate of 0.95 d<sup>-1</sup> at 20°C compared to a heterotrophic maximum growth rate of 3.2 d<sup>-1</sup> (EnviroSim Associates Ltd). As a result, they can only perform significant amounts of denitrification under conditions of low DO, low rbCOD and high nitrate

concentrations where there is a lack of substrate for heterotrophic denitrifiers. As this condition is only present in the small deoxygenation tank in the MBR, it is unlikely that DPAOs contribute significantly to denitrification in the plant.

## 2.4 Computer modelling

Since the 1970s, numerous attempts had been made to model the complex bacterial interactions in the biological wastewater treatment process. In 1987, the International Association on Water Quality introduced a basic model to describe COD removal, oxygen uptake, and cell metabolism of biomass in activated sludge systems. This model is known as Activated Sludge Model No. 1 (ASM1), and also includes nitrifying and denitrifying processes (Henze, Grady, Gujer, Marais, & Matsuo, 1987).

This model served as the basic framework for further improvements to the model. ASM2 introduces the classical Bio-P removal mechanism by PAOs into the model, whereas ASM2d includes the relationship between nitrogen and phosphorus removal in the activated sludge system with the addition of DPAOs (Gujer et al., 1995; Henze et al., 1999). Finally, ASM3 was introduced in 1999 to fix some of the defects and limitations of ASM1 (Gujer, Henze, Mino, & van Loosdrecht, 1999). Besides activated sludge, an anaerobic digestion model ADM1 was introduced in 2002 to simulate the anaerobic digestion of sludge and treatment of wastewater (Batstone et al., 2002).

Biowin (EnviroSim, Canada) is a commercial program that allows users to simulate biological treatment in a WWTP. By default, it uses a general Activated Sludge / Anaerobic Digestion model which is referred to as the Biowin ADSM model. This model combines the above ASM and ADM models into a complete, integrated model to reduce the complexity involved in having to map one model's output to another model's input. In addition to biological treatment, Biowin can also simulate several other aspects of a WWTP as it includes other models such as mass transfer and sludge settling models. (EnviroSim Associates Ltd)





The main differences were found to be the rate of COD consumption in the A-stage and the rates of nitrification and phosphorus uptake in the B-stage. Hence, the maximum growth rates of heterotrophs, nitrifiers, and PAOs were calibrated such that the modelled values agree well with the experimental values. The calibration of the growth rates could be done by modifying the maximum growth rate at 20°C or the corresponding Arrhenius factor. As temperature correction does not matter in Singapore due to the constant wastewater temperature, both alternatives would have the same effect and the Arrhenius factor was altered in this study.

After calibration, profile studies of the MBR under various configurations and operating conditions were used to verify the calibrated parameters. The measured influent quality and MBR operating conditions were simulated in Biowin and the modelled data was compared to the measured data from the profile studies for verification.

### 3.2.2 Biosorption

The biosorption A-stage at the IVP is not functioning well due to daily shock TSS loads upsetting the process. Hence, data was obtained from the biosorption process of the 'Energy+' project at Kranji WRP to calibrate the Biowin model. It operates in the same configuration as that in the IVP, having a biosorption tank (BPT) operating at a low DO, a primary clarifier, and a RAS return flow from the primary clarifier to the BPT. Figure 6 shows the Biowin model used for biosorption calibration and Table 2 summarises the values used in the model.

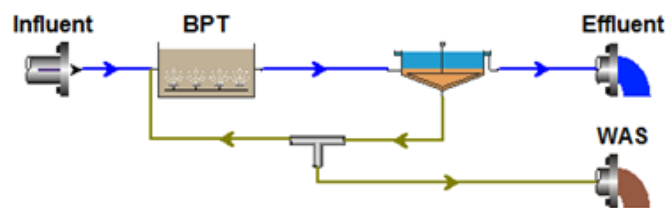


Figure 6: Biowin setup for biosorption process

Table 2: Biosorption parameters

Parameter	Value	Units
Influent flow	2	m <sup>3</sup> /h
Clarifier underflow	0.4	m <sup>3</sup> /h
WAS flow	0.05	m <sup>3</sup> /h
Effluent flow	1.95	m <sup>3</sup> /h
BPT volume	1	m <sup>3</sup>
BPT DO	0.3	mg/l
Clarifier volume	3.42	m <sup>3</sup>
Clarifier depth	1.5	m

### 3.2.3 Nitrifier Oxygen Uptake Rate (OUR) test

Samples taken from the IVP, including the profile studies, indicated very rapid nitrification in the system, as evidenced by the high rate of ammonia depletion in the aerobic zones. This is expected as the high wastewater temperature would lead to very fast growth of nitrifiers. Indeed, the maximum growth rate of nitrifiers is expected to increase by up to 2.67 times for every 10°C increase in temperature (van Haandel & van der Lubbe,

2012). Hence, a method similar to that outlined by the same authors was used to determine the maximum growth rate of nitrifiers at the IVP for use in the Biowin model.

Firstly, 3L of mixed liquor from the aeration tank was aerated in a well-mixed reactor and maintained at 30°C. The Oxygen Uptake Rate (OUR) was measured by repeatedly saturating the mixed liquor with DO and measuring the drop in DO from 5 mg/l to 4 mg/l using a portable DO meter (YSI Incorporated 550A, USA). A plot of DO against time would give the base OUR, indicating the endogenous respiration rate ( $OUR_{en}$ ). This includes the heterotrophic OUR for COD consumption. Next, ammonium chloride was dosed into the reactor for an initial ammonia concentration of about 20 mg-N/l. The OUR was then measured again using the same procedure to get the maximum OUR ( $OUR_m$ ).

The following formulas were then used to calculate the maximum growth rate of nitrifiers:

$$X_n = \frac{Y_n \times R_s \times N_c}{R_h \times (1 + b_n \times R_s)}$$

$$r_n = \frac{OUR_m - OUR_{en}}{4.57}$$

$$\mu_m = \frac{Y_n \times r_n}{X_n}$$

where:

$b_n$  = decay rate ( $0.04 \times 1.03^{(T-20)} = 0.054$  /day)

$N_c$  = nitrified ammonium concentration (7.8 mg-N/l)

$R_h$  = HRT (0.29 d)

$R_s$  = SRT (5 d)

$r_n$  = maximum nitrification rate (mg-N/(l.d))

$X_n$  = nitrifier concentration (mg/l)

$Y_n$  = nitrifier yield (0.1 mg-VSS/mg-N)

$\mu_m$  = maximum growth rate of nitrifiers (1/d)

### 3.2.4 Jar test

Jar tests were also performed to calibrate the Biowin model, focusing on the kinetic parameters in particular. During the jar tests, 2L of RAS was added to 1L of MBR influent in a well-mixed reactor, similar to the RAS/influent ratio in the MBR. The reactor was first operated unaerated for 90 minutes, and then maintained at a DO concentration of 1.5 mg/l or 2.5 mg/l for a further 180 minutes. Samples were taken at regular intervals for laboratory analysis of sCOD,  $NH_4$ ,  $NO_2$ ,  $NO_3$  and  $PO_4$ . The jar tests were then dynamically simulated in Biowin with the SBR module.

### 3.2.5 Profile studies

Profile studies have been carried out in the MBR on an ad-hoc basis when it has stabilised from operational changes. Samples from various locations in the MBR as indicated by yellow crosses in Figure 7 were taken and analysed for sCOD,  $NH_4$ ,  $NO_2$ ,  $NO_3$  and  $PO_4$  using the methods outlined in section 3.2.6. Together with online

operational data recorded, these laboratory results were used for the verification of the calibrated Biowin model using a steady state simulation.

Figure 8 shows the Biowin setup used for verification using the profile studies. As the MBR operates in a plug flow manner whereas Biowin simulates well-mixed reactors, two reactors were used to simulate the Anoxic 1, Anoxic 2, Swing 2, and Aerobic tanks each. The Swing 1 tank was simulated with four reactors instead as it provided a better fit of results. The membrane tanks and deoxygenation tank are relatively well-mixed, and are simulated with only one tank each. Investigations revealed that the deoxygenation tank is not functioning well and has an average DO of about 2 mg/l (see section 5.1.2). Hence, it is simulated with a constant DO of 2 mg/l in the Biowin model.

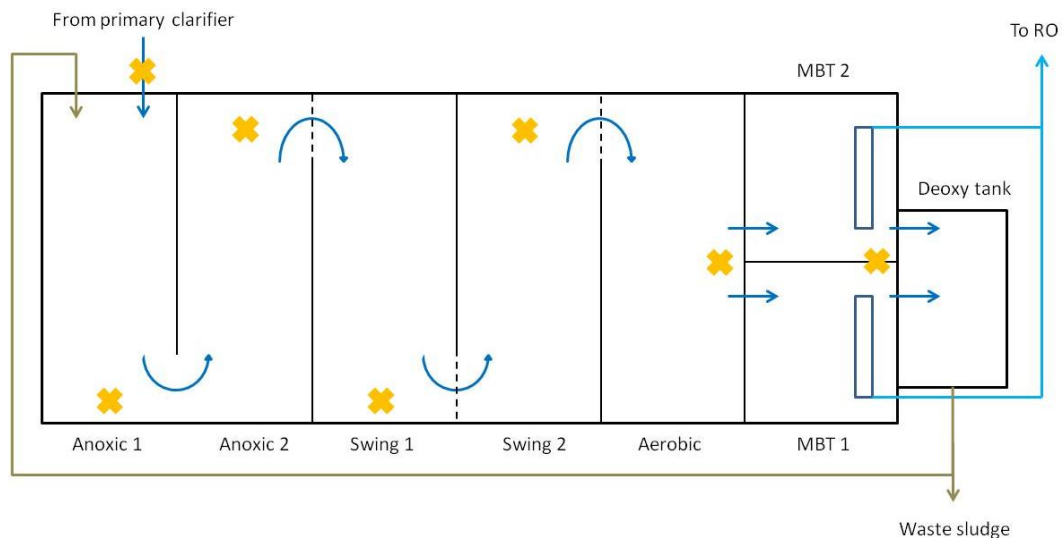


Figure 7: Sampling locations for profile studies

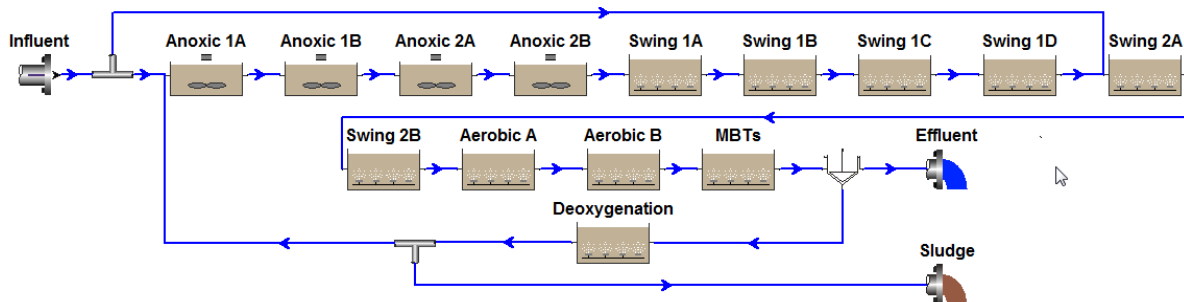


Figure 8: Biowin setup for profile studies

### 3.2.6 Laboratory analysis

The samples from the jar tests and profile studies were analysed using the following HACH kits and read using a spectrometer (HACH DR2800, USA):

- COD: HACH method 8000
- $\text{NH}_4\text{-N}$ : HACH method 8038
- $\text{NO}_2\text{-N}$ : HACH method 8507
- $\text{NO}_3\text{-N}$ : HACH method 10020
- $\text{PO}_4\text{-P}$ : HACH method 8048
- TP: HACH method 8190

In addition, the MLSS and MLVSS concentrations were analysed according to the standard methods (APHA, AWWA, & WEF, 1999). To characterise the MBR influent for Biowin simulations, the VFA concentrations were measured using gas chromatography (Shimadzu GC-2010, Japan) and cation concentrations were measured using ion chromatography (Diomex DX-500, USA).

### 3.3 Optimisation process

After the Biowin model has been calibrated and verified, it is first used to optimise the B-stage MBR in the IVP. As one of the aims of the IVP is to maximise the amount of COD recovered from wastewater, the criterion to identify the optimal MBR configuration is defined as the configuration that requires the least influent COD for good nutrient removal in the MBR. This would maximise the amount of COD that can be recovered from the A-stage while maintaining the effluent quality. Besides the COD value, all other parameters for the MBR influent quality are fixed.

Various configurations based on the existing MBR were first simulated. Next, two popular configurations used for Bio-P removal were simulated assuming that changes could be made to the pipe layout of the MBR. These are the University of Cape Town (UCT) configuration and a modified form of the Johannesburg configuration. Lastly, different operating conditions were also simulated to determine the optimal RAS flow and DO in the aerated tanks. The optimal proportion of influent to be added to the first tank in a step feed was also investigated. A sensitivity analysis was then performed to determine how the effluent quality changes with influent COD.

With the optimal MBR configuration, the calibrated biosorption A-stage is added into the Biowin model to investigate its effect on the treatment performance. As it results in unsatisfactory effluent quality, the operating conditions of the biosorption process were then modified to determine possible changes to the A-stage that can result in good effluent quality. The different modifications were then compared to determine the one that maximises COD recovery while achieving the required effluent quality. The speed of response of the effluent quality to changes in the operating conditions of the A-stage was also examined.

## 4 Results and discussion

### 4.1 Biowin calibration

#### 4.1.1 Influent characteristics

Samples of raw influent (entering the biosorption tank) and the MBR influent (entering the MBR from the biosorption process) were analysed to determine the approximate range of VFA and cation concentrations in the influent as these can affect the biological treatment process. The results are summarised in Tables 3 and 4 below, where n denotes the total number of samples. The raw data can be found in Appendix B.

**Table 3: VFA concentrations in the influent samples**

	Raw influent (n = 7)			MBR influent (n = 9)		
	Avg (mg/l)	Std Dev	Range (mg/l)	Avg (mg/l)	Std Dev	Range (mg/l)
Acetate (C2)	36.62	24.36	11.13 - 83.99	16.27	16.31	1.81 - 48.02
Propionate (C3)	2.64	1.98	0.25 - 5.99	1.14	1.88	0 - 5.95
Butyrate (C4)	0.74	0.50	0.18 - 1.69	0.42	0.38	0 - 1.12
Valerate (C5)	1.06	0.55	0.45 - 1.90	0.66	0.43	0.17 - 1.49
<b>Total</b>	<b>41.06</b>	<b>27.10</b>	<b>12.01 - 93.57</b>	<b>18.49</b>	<b>18.46</b>	<b>2.08 - 56.58</b>

**Table 4: Cation concentrations in the influent samples**

	Raw influent (n = 5)			MBR influent (n = 10)		
	Avg (mg/l)	Std Dev	Range (mg/l)	Avg (mg/l)	Std Dev	Range (mg/l)
K <sup>+</sup>	11.78	3.78	7.50 - 14.88	12.95	3.96	7.34 - 18.65
Mg <sup>2+</sup>	4.30	1.84	2.12 - 5.70	4.23	2.13	0.96 - 6.32
Ca <sup>2+</sup>	22.75	2.16	19.99 - 25.09	25.11	3.12	19.72 - 29.14

The VFA concentrations in the MBR influent has been found to vary widely, and part of this could be attributed to fluctuations in the raw influent. In addition, this difference also indicates the instability of the biosorption process, which can remove most of the influent VFA at certain times but not at others. The cation concentrations are observed to be much lower than the Biowin default concentrations of Mg<sup>2+</sup> (15 mg/l) and Ca<sup>2+</sup> (80 mg/l) (EnviroSim Associates Ltd). However, subsequent simulations revealed that there is no cation limitation, indicating that the influent cation concentration is sufficient for anabolic cell growth and metabolism.

#### 4.1.2 Biosorption

Several sets of 24-hour composite data for the influent and effluent quality in December 2014 were obtained from the 'Energy+' project at Kranji WRP. These data were then used to calibrate the parameters in Biowin. Figures 9 and 10 compare two of the data sets to the modelled values, and the model was found to be able to simulate the biosorption process relatively well.

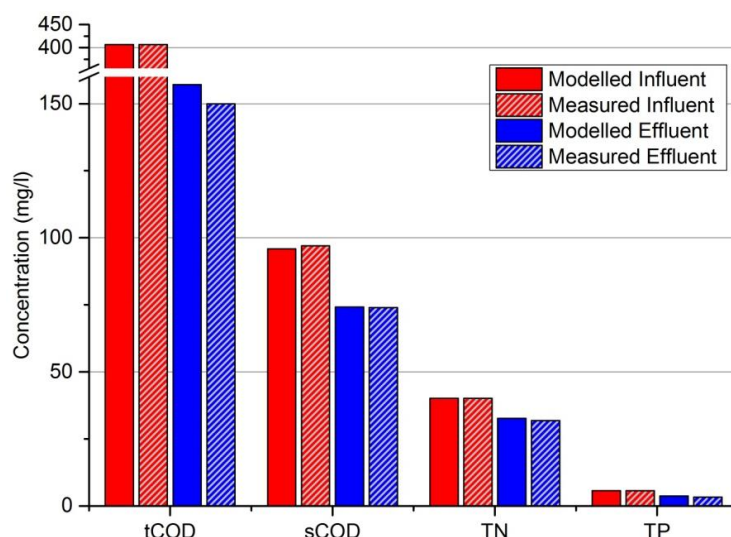


Figure 9: Modelled and measured biosorption data for 2 Dec 2014

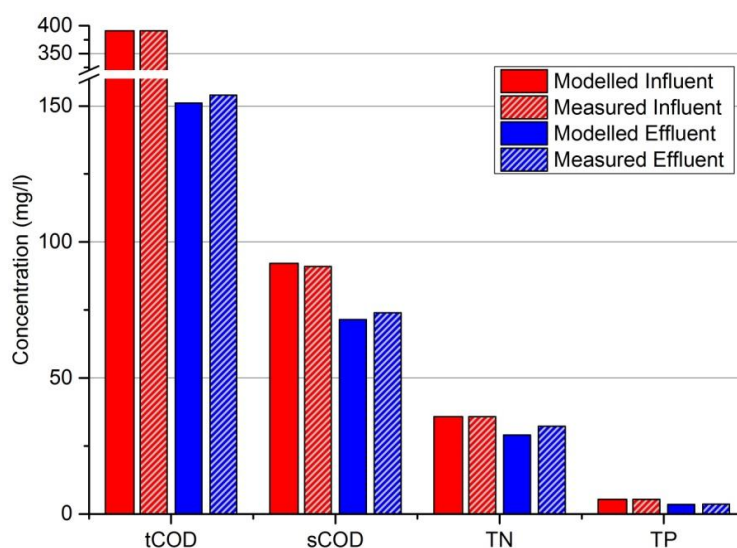


Figure 10: Modelled and measured biosorption data for 7 Dec 2014

#### 4.1.3 Nitrifier OUR test

The nitrifier OUR test was performed 4 times to determine the maximum growth rate of nitrifiers in the IVP. Table 5 summarises the results of the OUR test, where the maximum growth rate of nitrifiers was determined to be  $1.993 \pm 0.083 \text{ d}^{-1}$  at  $30^\circ\text{C}$ . As a start, this is taken to be the maximum growth rate of both AOB and NOB as this measures the rate-limiting nitrification step. They are found to be about 10% and 60% higher than the temperature-corrected default AOB and NOB growth rates of  $1.804 \text{ d}^{-1}$  and  $1.254 \text{ d}^{-1}$  respectively in Biowin (EnviroSim Associates Ltd).

Table 5: Maximum growth rate of nitrifiers

Base OUR ( $\text{OUR}_{\text{en}}$ ) ( $\text{mg-O}_2/(\text{l.hr})$ )	Max OUR ( $\text{OUR}_{\text{m}}$ ) ( $\text{mg-O}_2/(\text{l.hr})$ )	Nitrifier OUR ( $\text{OUR}_{\text{n}}$ ) ( $\text{mg-O}_2/(\text{l.hr})$ )	Max nitrification rate ( $\text{mg-N}/(\text{l.day})$ )	Max growth rate ( $\mu_{\text{m}}$ ) ( $\text{d}^{-1}$ )
22.6	61.1	38.5	202	1.918
22.6	64.9	42.4	223	2.112
23.8	63.5	39.7	209	1.979
23.4	62.8	39.4	207	1.963
Average				1.993

#### 4.1.4 Jar test

A total of 3 jar tests were performed to calibrate the Biowin model. Figures 11 to 13 show the modelled results of the jar tests, consisting of sCOD,  $\text{NH}_4$ ,  $\text{NO}_2$ ,  $\text{NO}_3$ , and  $\text{PO}_4$ . The measured data from the jar tests are also plotted on the same axis for comparison. The calibrated values are summarised in Table 6 in Section 4.1.5.

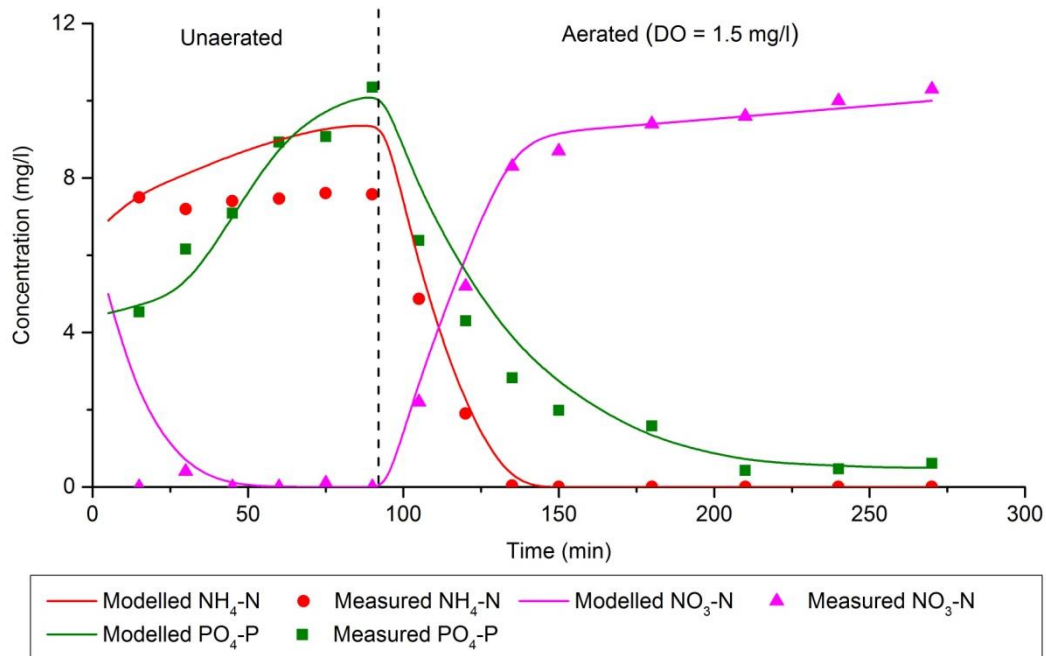


Figure 11: Jar test with low effluent  $\text{PO}_4$ -P concentration, error in sCOD measurement

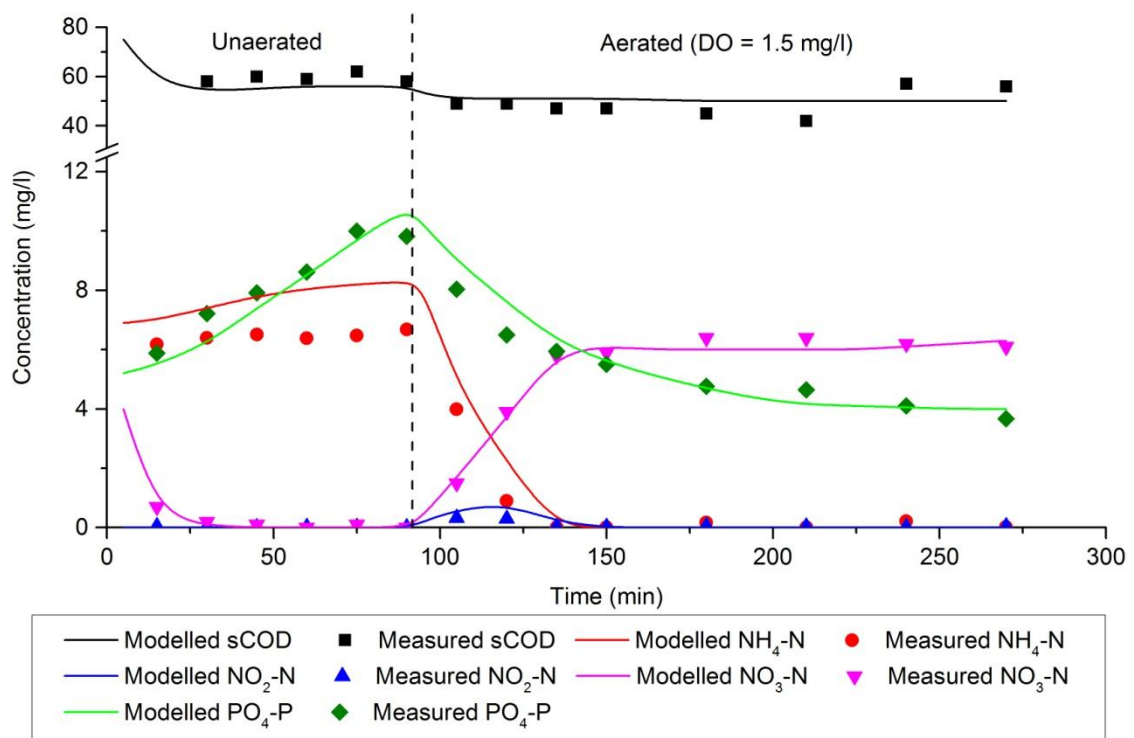


Figure 12: Jar test with high effluent  $\text{PO}_4$ -P concentration



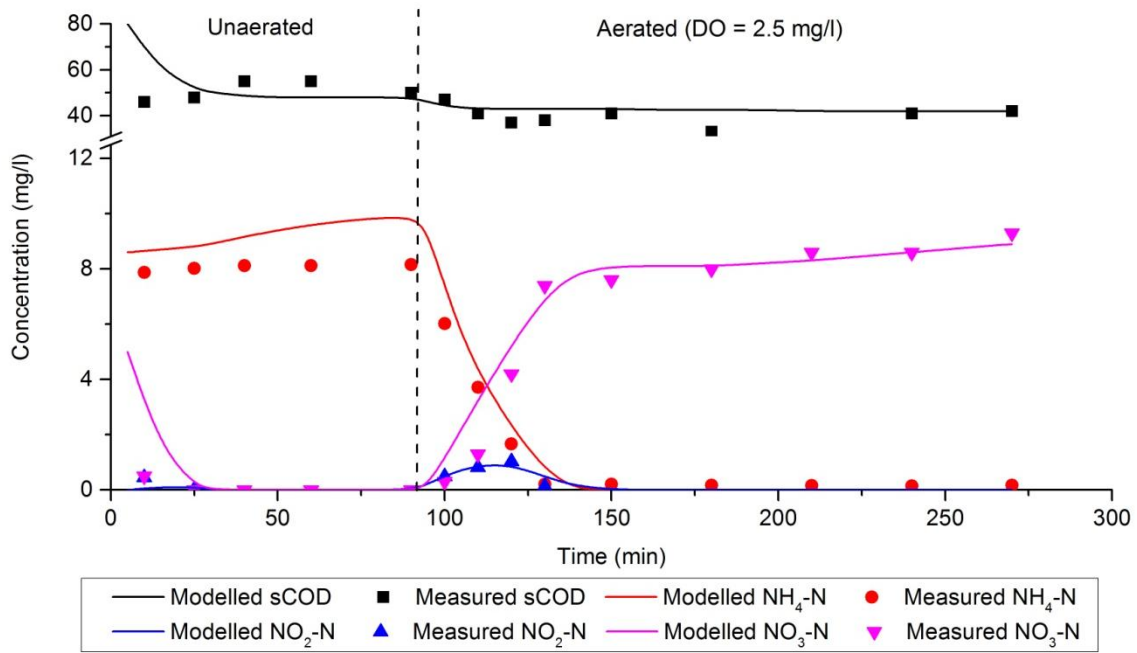


Figure 13: Jar test at DO = 2.5 mg/l, error in PO<sub>4</sub>-P measurement

The modelled data agreed reasonably well with the measured data. The agreement in modelled and measured nitrite values validates the assumption in section 4.1.3 that the measured maximum growth rate of nitrifiers can be applied to both AOB and NOB. The only major discrepancy is the ammonia concentration during the unaerated phase, where the modelled concentration is up to 1.7 mg-N/l higher than the measured values. This accounts for about 20% of the ammonia in the reactor and is likely to be due to adsorption of ammonia onto the activated sludge, which is not modelled in Biowin. With a TSS value of about 3 g/l, the adsorption rate was found to be about 0.56 mg-N/g-TSS. These values agree reasonably well with the findings of Nielsen (1996) that up to 30% of ammonia in the mixed liquor can be adsorbed to activated sludge and that the maximum adsorption rate of ammonia is about 0.5 mg-N/g-TSS.

Besides ammonia, the jar tests also show some discrepancy in the COD and nitrate measurements in the beginning of the simulations. The difference in COD is most likely due to a modelling artefact. Instead of an immediate addition and mixing of the influent and RAS in the jar tests, the modelled SBR has an addition over a period of one minute. On the other hand, the discrepancy in nitrate is most likely due to denitrification occurring in the RAS during the time it takes to transport the RAS to the lab and set up the jar tests.

In addition, the modelled phosphate concentrations also deviate slightly from the measured values. For example, the initial observed rates of phosphate release and phosphate uptake seems to be faster than what Biowin predicts. One reason for this discrepancy may be the presence of APAOs which are not modelled in Biowin. In addition, Barnard, Houweling, and Steichen (2011) also reported that Biowin assumes a lysis mechanism for biodegradation of PAOs when a maintenance approach may be more appropriate, and is hence not able to model PAOs accurately.

#### 4.1.5 Calibrated parameters

Table 6 summarises the calibrated parameters that are found to be different from the Biowin defaults. For the biosorption process, the Arrhenius value for Ordinary Heterotrophic Organisms (OHO) was determined from data obtained from the 'Energy+' project, and corresponds to a maximum growth rate of  $8.3 \text{ d}^{-1}$  at  $31^\circ\text{C}$ . This agrees well with values of  $7 \text{ d}^{-1}$  ( $22^\circ\text{C}$ ) and  $8 \text{ d}^{-1}$  ( $24^\circ\text{C}$ ) determined at treatment plants in Switzerland at warmer temperatures (Kappeler & Gujer, 1992). A value of  $12 \text{ d}^{-1}$  was found for WRPs in Singapore, although this was based on model calibration and may not be very accurate (Cao et al., 2008). The primary clarifier removal efficiency of 96% was also calibrated using the results from 'Energy+'. A primary clarifier without biosorption typically removes 90-95% of settleable solids and 40-60% of suspended solids (Spellman, 2009), indicating that the biosorption process is indeed performing better than a primary clarifier. However, there is still room for improvement as the biosorption A-stage at the Dokhaven WWTP in the Netherlands is able to achieve removal efficiencies of 98-99% (Fei, 2012).

For the MBR, the anaerobic hydrolysis factor was calibrated to be 0.2 using sCOD values in the jar tests and this value agrees well with studies that estimate the anaerobic hydrolysis factor to be in the same order of magnitude as 0.1 (Gujer et al., 1995) and 0.4 (Henze et al., 1999). The Arrhenius values for AOB and NOB were adjusted mainly based on the results of the nitrifier OUR test. The value for PAO was adjusted based on the jar tests, and is slightly higher than the value of  $1.065^1$  found by De Kreuk, Picoreanu, Hosseini, Xavier, and van Loosdrecht (2007).

One interesting observation is that the calibrated Arrhenius factors are all higher than the respective default factors in Biowin. One likely reason is that the default factors were determined at lower temperatures and it is uncommon to have such high wastewater temperatures found in Singapore. Indeed, a review of existing work by van Haandel and van der Lubbe (2012) revealed that most of the nitrification experiments were carried out at temperatures between  $15^\circ\text{C}$  and  $23^\circ\text{C}$ . Similarly, Kappeler and Gujer (1992) investigated the kinetic parameters of heterotrophs at various WWTP at temperatures ranging from  $13^\circ\text{C}$  to  $24^\circ\text{C}$ .

Table 6: Calibrated parameters

Parameter	Default	Calibrated	Remarks
<u>For MBR:</u>			
Anaerobic hydrolysis factor (activated sludge)	0.04	0.20	From sCOD values in jar test
AOB $\mu_{\max}$ Arrhenius factor	1.072	1.083	From nitrifier OUR test and jar test
NOB $\mu_{\max}$ Arrhenius factor	1.06	1.11	From nitrifier OUR test and jar test
PAO $\mu_{\max}$ Arrhenius factor	1	1.09	From jar test
<u>For biosorption:</u>			
OHO $\mu_{\max}$ Arrhenius factor	1.029	1.09	From biosorption results
Primary clarifier removal efficiency	-	96%	From biosorption results

<sup>1</sup> A value of  $\theta = 0.063$  in the form  $\mu_{\max,T} = \mu_{\max,20} e^{\theta(T-20)}$  was found by the authors, where T denotes the temperature. This was converted to the form used by Biowin:  $\mu_{\max,T} = \mu_{\max,20} \theta^{(T-20)}$  at  $30^\circ\text{C}$ .

#### 4.1.6 Validation of model

After calibration, the MBR model was validated using a total of 4 profile studies of the MBR. These studies were conducted when the MBR operation was relatively stable at 3 conditions – MLE configuration (3d SRT), MLE configuration (5d SRT), and AOA configuration (5d SRT) with a 75-25 step feed (75% of influent enters the first anoxic zone and 25% enters the second anoxic zone). The operating details are summarised in Appendix C and a comparison of the modelled and measured values are shown in Figures 14 to 17 below.

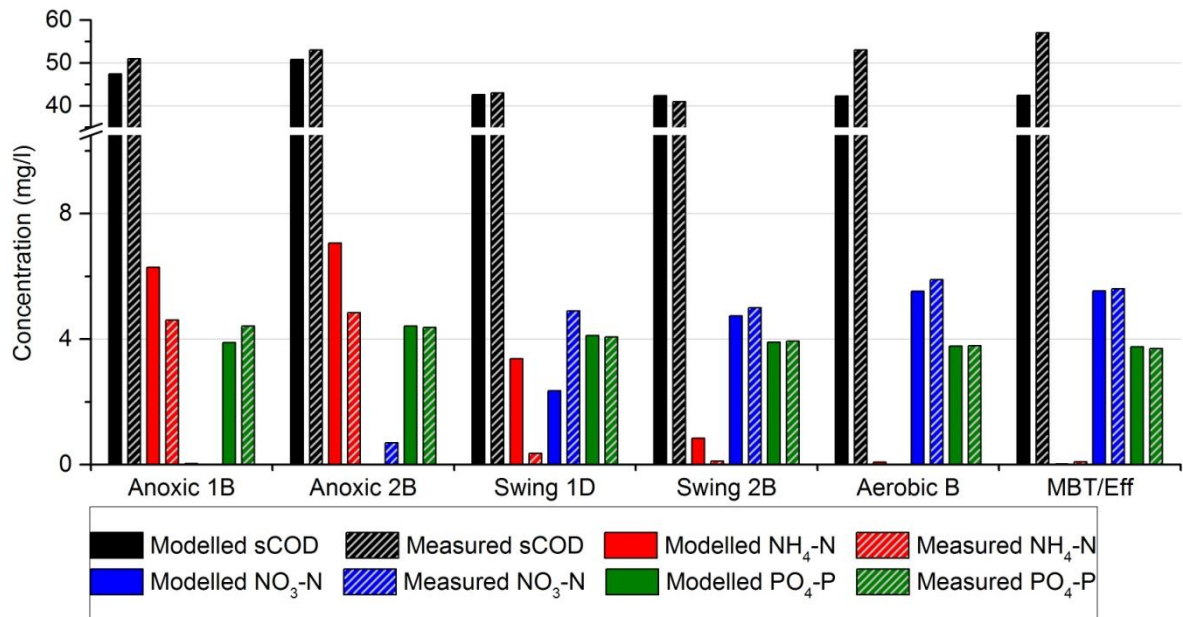


Figure 14: 23 Sep 2014 profile study (MLE, 3d SRT)

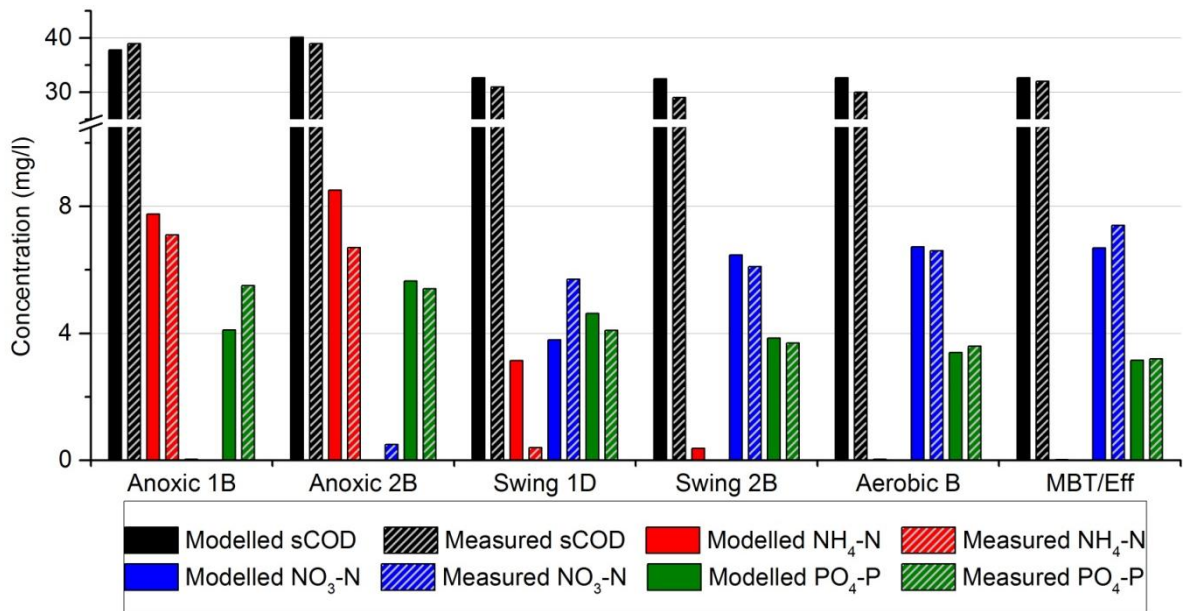


Figure 15: 10 Jun 2014 profile study (MLE, 5d SRT)

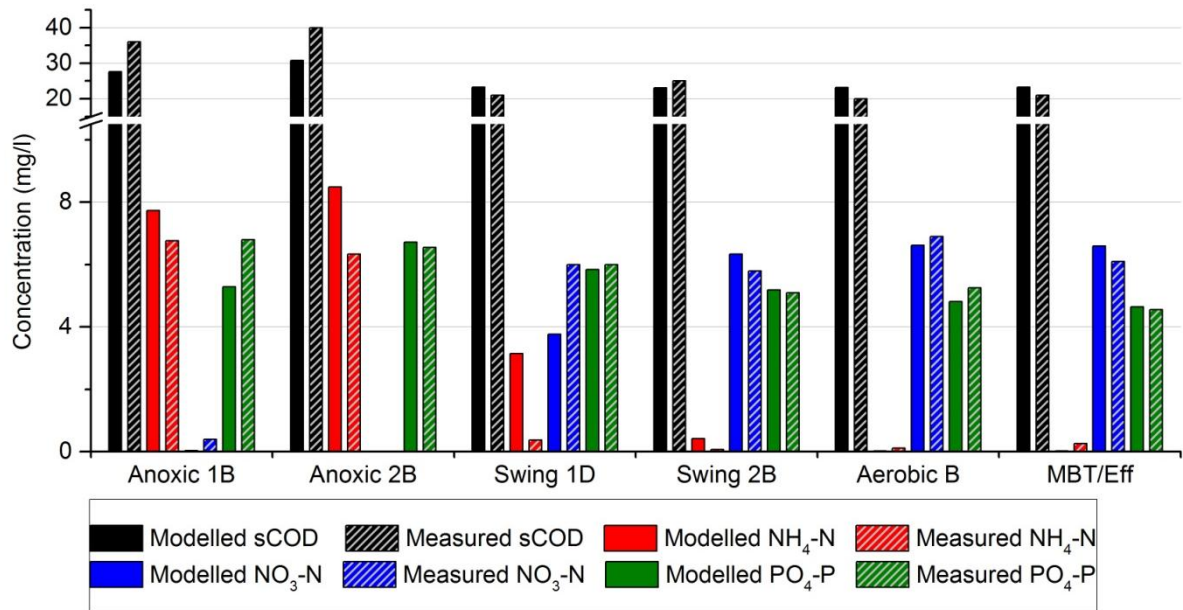


Figure 16: 24 Jun 2014 profile study (MLE, 5d SRT)

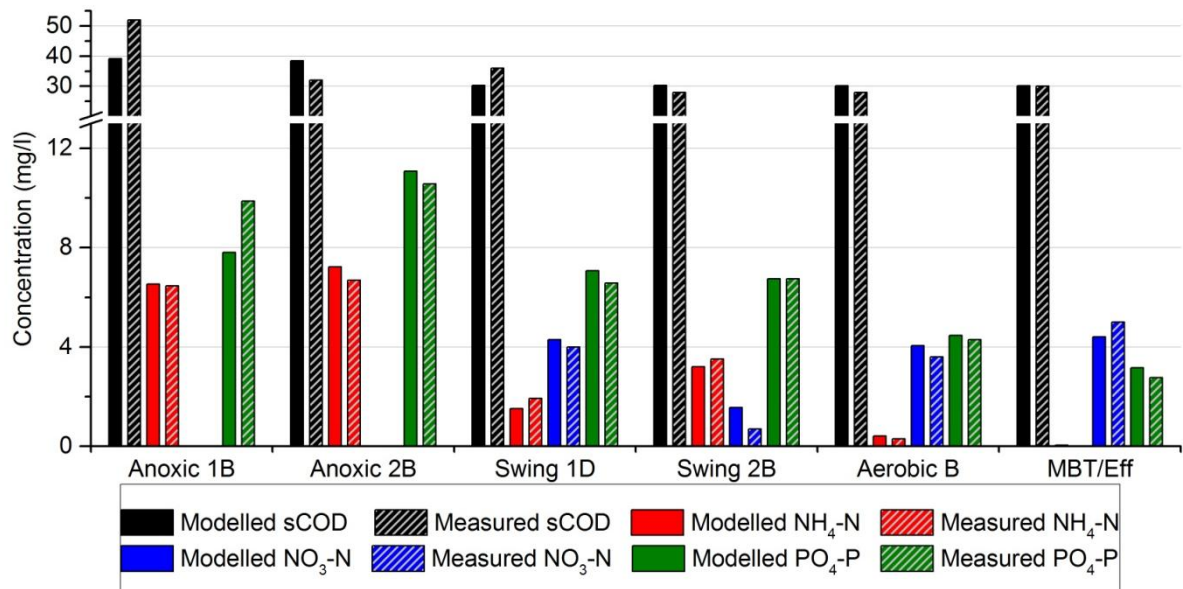


Figure 17: 23 March 2015 profile study (AOAO, 5d SRT)

Similar to the observations in the jar tests, the modelled ammonia concentration in the anoxic tanks are higher than the measured values, with a value of up to 2.2 mg/l higher in the Anoxic 2B tank (end of Anoxic 2 tank). This could be partly explained by the adsorption of ammonia onto the activated sludge. Another possible contributing factor for the lower measured ammonia concentration is that mixed liquor can be observed to flow backwards from the Swing 1 tank to the Anoxic 2 tank when aeration starts in the former. This backflow would introduce DO and partially nitrified water into the Anoxic 2 tank. The presence of a backflow is further confirmed by the occasional detection of nitrate at the end of the Anoxic 2 tank, even though denitrification completes in the Anoxic 1 tank. Hence, both ammonia adsorption onto activated sludge and backflow of mixed liquor would contribute toward a lower measured ammonia concentration in the anoxic tanks.

In addition, the measured ammonia and nitrate concentrations in the Swing 1D tank does not agree well with the modelled values for the first three profile studies performed in 2014 (Figures 14 to 16). The measured data suggest that the actual nitrifying activity in the MBR is much higher than in the model. However, the nitrifying OUR tests in Section 4.1.3 did not reveal the unusually high nitrifying activity required to obtain the measured ammonia and nitrate values in those profile studies. This discrepancy might have been due to the large amounts of water that have been observed to flow backwards from the Swing 2 tank to the Swing 1 tank when aeration starts in the former. The backflow would have contaminated the collected samples with more nitrified water. Baffle walls were installed in January 2015 to minimise the problem of the backflow. Thereafter, the modelled and measured data agreed well in the profile test in March 2015 (AOAO mode), indicating that backflow of mixed liquor is likely to be the major factor for this discrepancy.

Lastly, the measured phosphate concentration in the Anoxic 1B tank is always higher than that modelled in Biowin. This seems to agree with the jar tests that Biowin tends to underpredict the initial rate of phosphate release. Similar to the jar tests, this could be due to the fact that Biowin is not able to model Bio-P accurately due to the presence of APAOs that do not follow the classical model of Bio-P removal.

The validation of the calibrated data with the profile studies indicates that Biowin is able to model the processes occurring in the actual treatment process fairly accurately, even though it is more commonly used for simulations of wastewater treatment at lower temperatures. The only major shortcoming is the inability to model ammonia adsorption. The other major discrepancies are found to be due to the non-ideal operation of the plant. Hence, Biowin will be used for the subsequent optimisation of the IVP.

## 4.2 Optimal MBR configuration

After calibration, the Biowin model is used to optimise the MBR B-stage of the IVP by determining the configuration and operating conditions that require the least influent COD for adequate nutrient removal. The TKN and TP concentrations in the MBR influent are fixed in this section although these values are expected to be correlated to the influent COD value. A sensitivity analysis of these parameters will be performed in section 4.3.3. The following parameters were standardised across all configurations for comparison.

- MBR influent quality
  - Total COD (tCOD): Variable
  - Total Kjeldahl Nitrogen (TKN): 33 mg/l
  - Nitrate (NO<sub>3</sub>-N): 0 mg/l
  - Total Phosphorus (TP): 8 mg/l
  - Calcium (Ca<sup>2+</sup>): 25 mg/l
  - Magnesium (Mg<sup>2+</sup>): 4.2 mg/l
- Operating data
  - Temperature: 31°C
  - MBR influent flow: 37 m<sup>3</sup>/h (888 m<sup>3</sup>/d)
  - RAS flow: 74 m<sup>3</sup>/h (1776 m<sup>3</sup>/d)
  - WAS flow: 1.36 m<sup>3</sup>/h (32.64 m<sup>3</sup>/d)
  - DO in aerated tanks: 2.0 mg/l (unless otherwise stated)
  - DO in deoxygenation tank: Unaerated
  - MBT air flow rate: 295 m<sup>3</sup>/h
  - SRT: ~ 5d
- MBR effluent quality
  - Soluble COD (sCOD): < 60 mg/l
  - Phosphate (PO<sub>4</sub>): < 0.3 mg-P/l
  - Total Nitrogen (TN): < 10 mg-N/l
  - Ammonia (NH<sub>4</sub>): < 1 mg-N/l

### 4.2.1 Optimal configuration for existing MBR

The existing MBR is designed such that it is able to operate in several configurations. Figure 18 shows the design of the MBR. The influent and RAS can be added to the start of the Anoxic 1, Swing 1 and/or Swing 2 tanks. The anoxic tanks can only operate in the unaerated mode whereas the aerobic tank can only operate in the aerated mode. The swing tanks are equipped with both stirrers and aerators, and can hence operate in both aerated and unaerated modes.

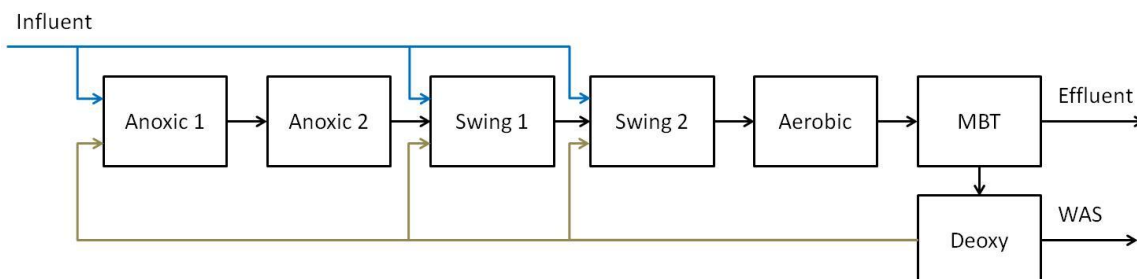


Figure 18: Design of existing MBR

## Configurations

Table 7 shows the MBR configurations that were considered in the optimisation process, where all RAS flows were directed to the beginning of the Anoxic 1 tank. Besides the three MBR configurations that had been tested out in the IVP (MLE1, AOA02, and AAO), three other configurations were simulated in Biowin.

From the jar tests in section 4.1.4, it can be observed that phosphate release in the unaerated phase lasts for over 90 min, and phosphate uptake in the aerobic phase slows down after about 60 min. Taking into account both influent and RAS flow, the mixed liquor in the MLE1 configuration spends only about 40 min in the unaerated tanks and about 90 min in the aerated tanks. Hence, MLE2 converts the Swing 1 tank from aerated to unaerated, bringing the residence time in the unaerated and aerated tanks to approximately 60 min and 70 min respectively and hence allowing more time for COD assimilation. Nitrification is not a concern as full nitrification could be observed to occur within 60 min of aeration.

In addition, the AOA01 configuration was also simulated. This configuration has a region of high nitrate, low DO and low rbCOD in the second anoxic tank, favouring DPAOs and possibly selecting for them. DPAOs are desirable in the MBR as they can denitrify and remove phosphate simultaneously, reducing the influent COD required to achieve acceptable effluent quality. The AOA03 configuration was also considered to determine whether a different proportion of step feed improves nutrient removal in the MBR, especially since it reduces the flowrate through the anoxic tanks and hence increases the anaerobic residence time.

**Table 7: Configurations considered in MBR optimisation (\*refers to configurations tested in the IVP)**

Configuration	Tank					
	Anoxic 1	Anoxic 2	Swing 1	Swing 2	Aerobic	MBT
↙ 100% influent						
MLE1*	Unaerated	Unaerated	Aerated	Aerated	Aerated	Aerated
↙ 100% influent						
MLE2	Unaerated	Unaerated	Unaerated	Aerated	Aerated	Aerated
↙ 75% influent      ↙ 25% influent						
Anoxic-Anaerobic-Aerobic (AAO)*	Unaerated	Unaerated	Unaerated	Aerated	Aerated	Aerated
↙ 100% influent						
AOA01	Unaerated	Unaerated	Aerated	Unaerated	Aerated	Aerated
↙ 75% influent      ↙ 25% influent						
AOA02*	Unaerated	Unaerated	Aerated	Unaerated	Aerated	Aerated
↙ 50% influent      ↙ 50% influent						
AOA03	Unaerated	Unaerated	Aerated	Unaerated	Aerated	Aerated



## Results

Table 8 summarises the optimisation results while keeping to the existing MBR layout. The effluent phosphate is found to be the limiting factor as all the other effluent limits could be met easily. Hence, all influent COD values were adjusted to attain an effluent phosphate concentration of 0.30 mg-P/l. As expected, the AOA01, AOA02, and AOA03 configurations achieved much lower effluent TN concentrations than the others due to a higher amount of denitrification. This denitrification is mainly performed by heterotrophic denitrifiers in the anoxic zone as denitrification by DPAOs is not very significant in the MBR. The higher amount of denitrification would consume a higher amount of COD in the anoxic zone, resulting in less COD available for uptake by PAOs. Hence, it would be better to compare the AOA0 configurations with similar effluent TN concentrations, as will be done in section 4.2.3.

**Table 8: Optimisation results for existing MBR layout**

<b>Configuration</b>	<b>MLE1</b>	<b>MLE2</b>	<b>AAO</b>	<b>AOA01</b>	<b>AOA02</b>	<b>AOA03</b>
Influent:						
<b>COD (mg/l)</b>	<b>365</b>	<b>328</b>	<b>334</b>	<b>350</b>	<b>362</b>	<b>371</b>
Effluent:						
PO <sub>4</sub> (mg-P/l)	0.30	0.30	0.30	0.30	0.30	0.30
NH <sub>4</sub> (mg-N/l)	0.02	0.02	0.03	0.02	0.02	0.05
TN (mg-N/l)	8.92	9.15	9.10	7.02	5.98	5.11
COD (mg/l)	31.9	29.2	29.7	31.21	32.18	32.72
Waste activated sludge:						
OHO (mg-COD/l)	1912.1	1748.8	1767.5	1791.0	1866.9	1932.6
AOB (mg-COD/l)	53.26	55.42	54.6	54.71	53.12	51.05
NOB (mg-COD/l)	32.4	33.7	33.2	32.17	31.36	30.77
PAO (mg-COD/l)	372.9	458.4	468.3	519.3	499.2	468.3
<b>% of PAO</b>	<b>15.7%</b>	<b>20.0%</b>	<b>20.2%</b>	<b>21.7%</b>	<b>20.4%</b>	<b>18.9%</b>
Stored poly-P (mg-P/l)	134.9	139.4	138.3	136.1	133.9	132.2
<b>Stored poly-P (mg-P/mg-PAO)</b>	<b>33.9%</b>	<b>30.2%</b>	<b>29.6%</b>	<b>27.1%</b>	<b>27.6%</b>	<b>28.6%</b>

Among the other configurations, MLE1 was the least efficient, requiring 365 mg/l of influent COD for satisfactory effluent quality. The best performing configuration would be MLE2, which requires only 328 mg/l of influent COD. Table 8 also shows the amount of PAOs as a percentage of organisms present in the WAS, as well as the amount of poly-P stored within the PAOs in the WAS. Using a conversion factor of 1.42 mg-COD/mg-VSS, the percentage of stored poly-P is found to range from 29.6% to 33.9%, which are slightly below the maximum value of 38% that have been found in literature (van Haandel & van der Lubbe, 2012). These values vary with the influent COD, with a lower COD value causing a decrease in the proportion of PAOs and an increase in the amount of stored poly-P in the WAS (see section 4.2.4).

The MLE2 and AAO configuration has a much lower influent COD requirement than the MLE1 configurations. However, the percentage of poly-P in the WAS is found to be about 5% lower. One possible reason for this is the lower residence time of the mixed liquor in the aerobic zones, which has the highest rate of phosphate uptake. The mixed liquor is aerated for about 90 min in the MLE1 configuration compared to approximately 70

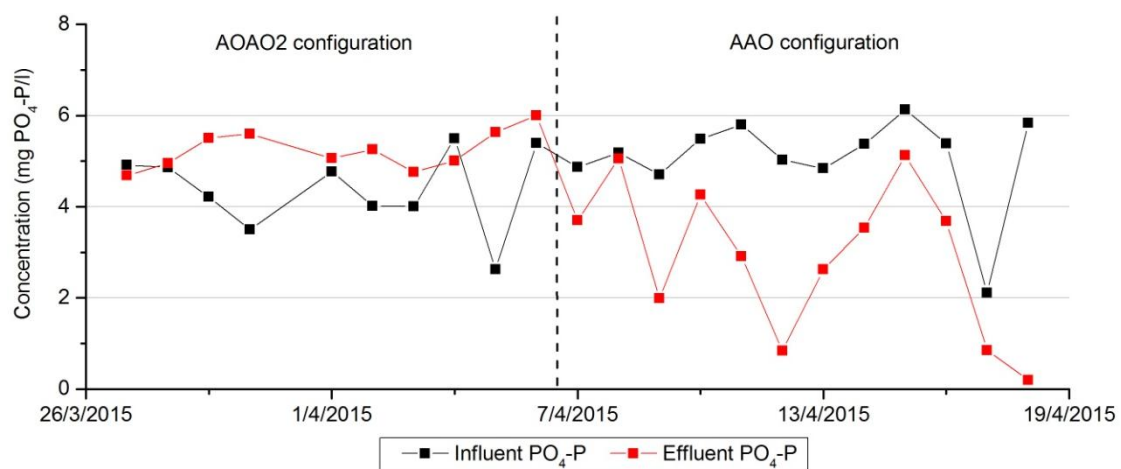


min for the other configurations, and the jar test results in section 4.1.4 show that phosphorus uptake ceases only after about 120 min of aeration. Hence, the MLE2 and AAO configurations results in a lower poly-P content in the PAOs in WAS as they allow less time for phosphorus uptake to occur.

On the other hand, the MLE2 and AAO configurations have a higher proportion of PAOs despite a lower influent COD because of a longer anaerobic residence time. As denitrification essentially completes in the Anoxic 1 tank, the Anoxic 2 tank operates as an anaerobic zone for all configurations. The Swing 1 tank operates as an additional anaerobic tank for the MLE2 and AAO configurations. Overall, the higher percentage of PAOs allowed a better Bio-P removal performance in these two configurations.

### *Comparison of modelling results with MBR performance*

Figure 19 shows the phosphorus concentration in the effluent after the MBR switched from the AOA2 configuration to the AAO configuration on 6 April 2015. According to Table 8, AAO performs significantly better than AOA2 as it reduces the required influent COD from 362 mg/l to 334 mg/l, keeping all other influent parameters constant. Although the MBR was operated for only about 2 weeks in this configuration before it was shut down for cleaning and maintenance, it is evident from the results that the effluent phosphorus was generally lower than the preceding AOA2 configuration. This validates the Biowin results that the AAO configuration has a better Bio-P performance than the AOA2 configuration.



**Figure 19: Effluent PO<sub>4</sub>-P concentration for the AAO configuration**

### **4.2.2 Optimal configuration not limited to existing MBR pipe layout**

In addition to the optimisation performed in section 4.2.1, simulations were also conducted to optimise the MBR, but not limited to the current pipe layout for influent and RAS flows. The volumes of the various tanks were kept the same for optimisation, but the sludge wastage rate may be adjusted to maintain an SRT of 5 days.

### University of Cape Town configuration (UCT)

The first configuration is the University of Cape Town (UCT) configuration, as shown in Figure 20. This configuration is common in EBPR plants around the world as it prevents DO or nitrate from recycling back to the anaerobic zone, provided that complete denitrification occurs in the anoxic zones.

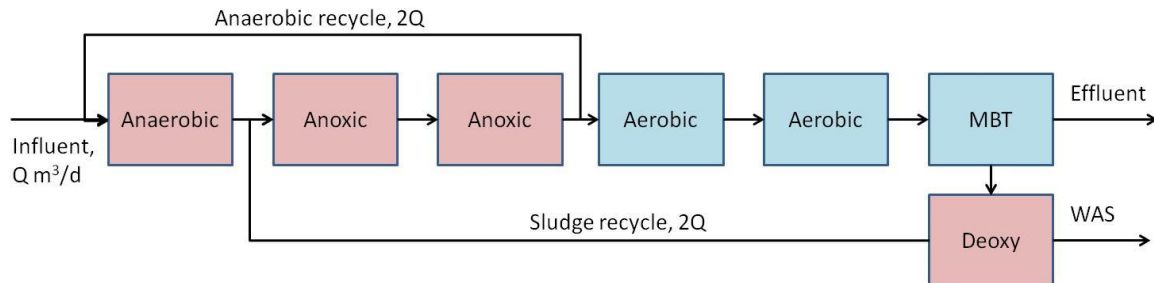


Figure 20: UCT configuration for MBR

### Modified Johannesburg configuration (MJHB)

Denitrification by DPAOs can be observed in the unaerated deoxygenation tank (see section 5.1.2). However, the tank is too small and the residence time is too short for significant amounts of denitrification to occur. Hence, the following configuration (Figure 21) is simulated, with the first anoxic tank used as an extension of the deoxygenation tank. This configuration is similar to the Johannesburg configuration that is sometimes used for EBPR systems, but without an aerobic recycle and a second anoxic reactor after the anaerobic zone.

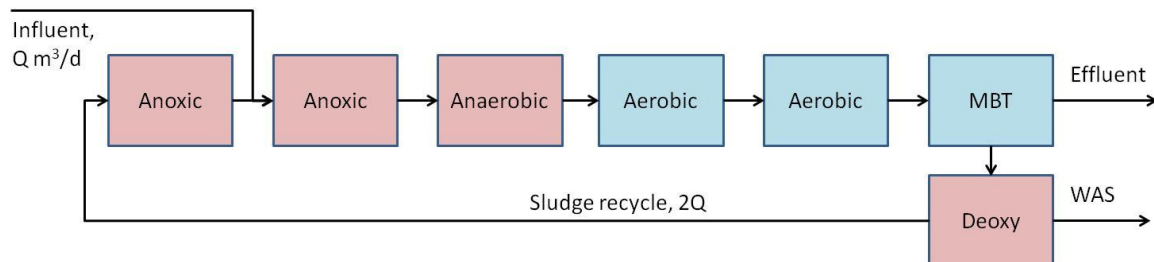


Figure 21: MJHB configuration for MBR

### Results

Table 9 summarises the results for the UCT and MJHB configurations, with the MLE2 results added for comparison. Surprisingly, the UCT configuration performed the worse of the group although it is designed for Bio-P removal. One possible explanation for this is that size of the anoxic and aerobic zones are inadequate for phosphorus uptake. Indeed, a typical value of the aerobic residence time in a UCT configuration is 4 to 6 times that of the anaerobic residence time (Tchobanoglous et al., 2003), compared to a value of approximately 2 based on the above UCT configuration. Adjustments to convert part of the anoxic zone to an aerobic zone is not feasible as it would lead to incomplete denitrification in the anoxic zone, resulting in the presence of nitrate in the anaerobic recycle and a deterioration of Bio-P performance.

One of the disadvantages of the UCT configuration is that there is inefficient use of denitrification capacity due to the need to remove all nitrate in the anaerobic recycle (van Haandel & van der Lubbe, 2012). As a result, it requires larger reactors and a longer HRT than other configurations. A typical HRT for the UCT configuration is

between 7 to 18 hours (Tchobanoglous et al., 2003), compared to a HRT of about 6 to 7 hours for the current MBR. Hence, the MBR can be considered to be undersized for the optimal operation of a UCT configuration. Other configurations that use the available capacity more efficiently would be able to achieve a better nutrient removal performance.

The MJHB configuration showed similar performance to the MLE2 configuration. Although it has a higher proportion of PAOs at the same influent COD concentration, the WAS contains a lower amount of poly-P stored per PAO, resulting in a similar overall Bio-P performance. Hence, MLE2 is considered to be the optimal configuration under the current operating conditions as it can be implemented without modifications to the existing layout of the MBR.

**Table 9: Optimisation results not limited to existing MBR layout**

<b>Configuration</b>	<b>UCT</b>	<b>MJHB</b>	<b>MLE2</b>
Influent:			
<b>COD (mg/l)</b>	<b>350</b>	<b>328</b>	<b>328</b>
Effluent:			
PO <sub>4</sub> (mg-P/l)	0.30	0.30	0.30
NH <sub>4</sub> (mg-N/l)	0.02	0.03	0.02
TN (mg-N/l)	8.98	9.20	9.15
COD (mg/l)	31.6	29.5	29.2
Waste activated sludge:			
OHO (mg-COD/l)	1813	1612.7	1748.8
AOB (mg-COD/l)	58.8	52.45	55.42
NOB (mg-COD/l)	35.8	31.89	33.7
PAO (mg-COD/l)	701.5	455.8	458.4
<b>% of PAO</b>	<b>26.9%</b>	<b>21.2%</b>	<b>20.0%</b>
Stored poly-P (mg-P/l)	148.3	132.3	139.4
<b>Stored poly-P (mg-P/mg-PAO)</b>	<b>23.1%</b>	<b>29.2%</b>	<b>30.2%</b>
<b>tCOD (kg/day)</b>	<b>123.3</b>	<b>117.6</b>	<b>118.0</b>

#### 4.2.3 Further optimisation

Having identified MLE2 as the optimal configuration for the MBR at the current operating conditions, further optimisation of the operating conditions was carried out to identify conditions that can further reduce that required influent COD while achieving the required effluent quality. Two such operating conditions are the RAS flow and DO in the aerated tanks.

##### *RAS flow*

The RAS recycles biomass from the deoxygenation tank back to the Anoxic 1 tank, maintaining a high biomass concentration in the MBR by decoupling the SRT from the HRT. Since the start of operations, the MBR has been operating with an RAS flow of about 2Q, or twice the influent flow. The RAS flow will affect the performance of the MBR by changing factors such as the MLSS. Most importantly, the RAS flow determines the effluent TN by controlling the amount of nitrate that is recycled back to the anoxic zone for denitrification.

From Table 8, the AOA configurations show better nitrogen removal than the other configurations with the same RAS flow due to higher denitrification. For a fairer comparison, the RAS flows were adjusted to achieve similar effluent TN concentrations to the MLE2 configuration (9.15 mg-N/l). Using the AOA1 configuration as an example, Figure 22 shows how the required influent COD and effluent TN concentration typically varies as a function of the RAS flow, keeping the effluent phosphate concentration fixed at 0.30 mg-P/l.

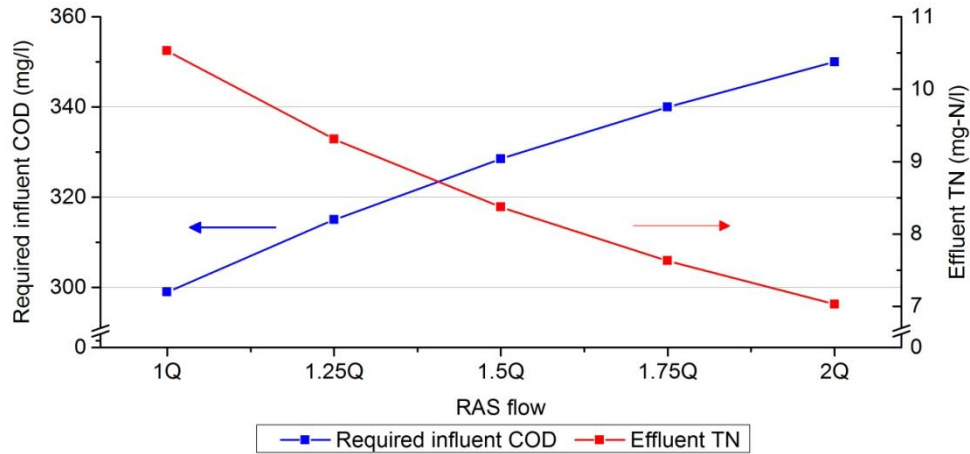


Figure 22: Variation of required MBR influent COD and effluent TN with RAS in AOA1 configuration

Table 10: Optimisation results of MLE2 and AOA at various RAS flow

Configuration	MLE2	AOA01	AOA02	AOA03
RAS flow	2Q	1.3Q	1Q	0.78Q
Influent:				
COD (mg/l)	328	318	311	302
Effluent:				
PO <sub>4</sub> (mg-P/l)	0.30	0.30	0.30	0.30
NH <sub>4</sub> (mg-N/l)	0.02	0.02	0.02	0.05
TN (mg-N/l)	9.15	9.10	9.17	9.13
COD (mg/l)	29.2	28.6	28.0	27.1
Waste activated sludge:				
OHO (mg-COD/l)	1748.8	1876.4	1972.1	2029.9
AOB (mg-COD/l)	55.42	66.08	69.23	70.21
NOB (mg-COD/l)	33.7	38.90	41.07	42.56
PAO (mg-COD/l)	458.4	611.2	603.4	568.9
% of PAO	20.0%	23.6%	22.5%	21.0%
Stored poly-P (mg-P/l)	139.4	168.5	177.1	181.7
Stored poly-P (mg-P/mg-PAO)	30.2%	28.1%	29.4%	31.2%
Anoxic 2 tank:				
Stored PHA (mg/l)	14.43	22.95	26.51	30.68
Stored PHA (mg/mg-VSS <sub>PAO</sub> )	6.7%	9.4%	10.9%	12.6%

Table 10 shows the updated results after taking the effluent TN into account, with the results of the MLE2 configuration included for comparison. The SRT of the MBR was kept at about 5 days by adjusting the WAS flow if necessary. With similar effluent TN concentrations, AOA03 is found to be the optimal MBR configuration. This is likely to be due to higher phosphorus release and COD uptake in the anaerobic phase. As

AOAO3 has the lowest RAS flow and the lowest influent flow to the Anoxic tanks (as only 50% of influent enters Anoxic 1 tank), it has the longest anaerobic residence time, allowing higher phosphorus release and COD uptake. In addition, the lower nitrate return also reduces the COD consumed for heterotrophic denitrification in the first anoxic zone. Indeed, Table 10 shows that the PAOs in the AOAO3 configuration contain the highest amount of PHA just prior to exiting the anaerobic Anoxic 2 tank. With the lowest influent COD requirement, the AOAO3 configuration with a RAS flow of 0.78Q is thus found to be the optimal configuration for the MBR.

### *DO in aerated tanks*

Another operating condition that influences nutrient removal in the MBR is the DO concentration in the aerated tanks as both nitrification and phosphorus uptake occurs faster at higher DO concentrations. Results from section 4.2.1 suggest that an aerobic residence time of around 70 minutes may be insufficient for phosphorus uptake to occur, as indicated by the lower stored poly-P in the PAOs in the MLE2 and AAO configurations. Hence, different levels of DO were simulated to investigate whether operating the aerated tanks at a higher DO concentration would lead to better nutrient performance due to faster phosphorus uptake. The membrane tank is kept at a constant aeration rate of 295 m<sup>3</sup>/h as this rate is influenced by the need for membrane scouring rather than for biological treatment.

**Table 11: Optimisation results for the AOAO3 configuration at various DO concentrations**

<b>DO concentration (mg/l)</b>	<b>0.5</b>	<b>1</b>	<b>2</b>	<b>3</b>
RAS flow	0.64Q	0.73Q	0.78Q	0.81Q
Influent:				
<b>COD (mg/l)</b>	<b>286</b>	<b>296</b>	<b>302</b>	<b>306</b>
Effluent:				
PO <sub>4</sub> (mg-P/l)	0.30	0.30	0.30	0.30
NH <sub>4</sub> (mg-N/l)	0.13	0.07	0.05	0.05
TN (mg-N/l)	9.17	9.14	9.13	9.11
COD (mg/l)	25.8	26.6	27.1	27.4
Waste activated sludge:				
OHO (mg-COD/l)	2044.8	2029.0	2029.9	2034.6
AOB (mg-COD/l)	78.52	72.81	70.21	68.81
NOB (mg-COD/l)	45.65	44.25	42.56	41.56
PAO (mg-COD/l)	660.4	597.6	568.9	553.9
<b>% of PAO</b>	<b>23.3%</b>	<b>21.8%</b>	<b>21.0%</b>	<b>20.5%</b>
Total stored poly-P (mg-P/l)	200.9	187.4	181.7	178.4
<b>Stored poly-P (mg-P/mg- PAO)</b>	<b>30.2%</b>	<b>30.8%</b>	<b>31.2%</b>	<b>31.4%</b>
Aerated tanks:				
<b>N<sub>2</sub> production (g-N/hr)</b>	<b>57.4</b>	<b>30.7</b>	<b>15.7</b>	<b>10.5</b>
<b>N<sub>2</sub>O production (g-N/hr)</b>	<b>7.5</b>	<b>2.6</b>	<b>1.7</b>	<b>1.4</b>
<b>Required aeration (m<sup>3</sup>/hr)</b>	<b>111.5</b>	<b>134.6</b>	<b>176.1</b>	<b>233.9</b>

Table 11 summarises the results for the different DO concentrations. Surprisingly, higher DO concentrations resulted in slightly poorer nutrient removal. Further investigations revealed that this is due to simultaneous

nitrification and denitrification (SND) in the aerated tanks, which was examined by monitoring the nitrogen gas production in the aerated tanks as an indication of the denitrification rate. SND in the aerated tanks can be quite significant in tanks operating at low DO concentrations. Most denitrifiers can accept both oxygen and nitrate as an electron acceptor, and numerous strains of denitrifiers have been found to be able to consume both simultaneously in a process known as aerobic denitrification (Ahn, 2006). A lower DO leads to lower availability of oxygen and hence a higher consumption of nitrate.

The inhibition of denitrification by dissolved oxygen can be accounted for by multiplying the maximum rate with the correction factor  $\left(\frac{K_0}{K_0 + DO}\right)$ , where  $K_0$  is the DO inhibition coefficient for nitrate reduction. Using a typical  $K_0$  value of 0.1 mg/l (Tchobanoglous et al., 2003), the denitrification rate at DO concentrations of 0.5, 1, 2, and 3 mg/l would be 17%, 9.1%, 4.8%, and 3.2% of the maximum denitrification rate respectively. These values agree well with the rate of nitrogen production in the aerated tanks that is reported in Table 11.

With a higher denitrification in the aerated tanks, less nitrate needs to be recycled back to the first anoxic zone for denitrification. This reduces the nitrate load in the Anoxic 1 tank and hence more COD is available for uptake by PAOs, improving the Bio-P removal performance. As expected, simulation results show that the proportion of PAOs in the MBR decreases as the DO in the aerated tank increases. However, a higher amount of stored poly-P is found at higher DO concentrations, indicating that higher DO concentrations do increase the rate of phosphorus uptake. However, the effect of the lower proportion of PAOs dominated and led to worse overall Bio-P removal at higher DO concentrations.

In addition to better nutrient removal performance, a low DO in the aerated tanks has an additional advantage in aeration energy savings. Operating the aerated tanks at a DO of 0.5 mg/l was found to require 17%, 37%, and 52% less aeration than operating the tanks at a DO of 1 mg/l, 2 mg/l and 3 mg/l respectively. Hence, a DO of 0.5 mg/l in the aerated tanks would be the optimal operating condition from the point of view of net energy requirement.

However, it must be noted that a low DO concentration in the aerated tanks may have undesirable side effects. For example, Martins, Heijnen, and van Loosdrecht (2003) found that a low DO concentration (< 1.1 mg/l) results in reduced sludge settleability due to the proliferation of filamentous bacteria. Similar observations were also reported by Wilén and Balmér (1999). Although sludge settleability is not a concern for an MBR, filamentous sludge is undesirable as it tends to form a scum layer on top of the mixed liquor. This would remain trapped in the MBR until it is manually removed. In addition, a low DO in the aerated tanks also leads to higher production of  $N_2O$  through nitrifier denitrification. Indeed, Biowin simulations show that a larger amount of  $N_2O$  is produced as DO in the aerobic tanks decreases (Table 11), and the  $N_2O$  production at a DO of 0.5 mg/l is almost triple the amount produced at a DO of 1 mg/l. As this drastic increase in  $N_2O$  production is highly undesirable for the environment, a DO of 1 mg/l is considered the most suitable for the MBR.

### Influent step feed

During the optimisation of RAS flows, the RAS flows of the various AOA configurations were varied together with the step feed of influent between the first and second anoxic tanks. However, only three step feeds were investigated – 100%, 75%, and 50% of the influent entering the first anoxic tank. The AOA3 configuration, which had 50% of the influent entering the first anoxic tank, performed the best as it requires the lowest influent COD to achieve satisfactory nutrient removal in the MBR. As these step feeds were performed in large intervals of 25%, further optimisation was carried out with smaller intervals of 5% to determine the optimal step feed for the AOA configuration at a DO of 1 mg/l in the aerated tanks.

**Table 12: Optimisation results at different influent step feed**

<b>Proportion of influent entering first anoxic tank</b>	<b>40%</b>	<b>45%</b>	<b>50%</b>	<b>55%</b>
RAS flow	0.75Q	0.72Q	0.73Q	0.76Q
Influent:				
<b>COD (mg/l)</b>	<b>297</b>	<b>296</b>	<b>296</b>	<b>298</b>
Effluent:				
PO <sub>4</sub> (mg-P/l)	0.30	0.30	0.30	0.30
NH <sub>4</sub> (mg-N/l)	0.16	0.11	0.07	0.05
TN (mg-N/l)	9.15	9.14	9.14	9.14
COD (mg/l)	26.7	26.5	26.6	26.8
Waste activated sludge:				
OHO (mg-COD/l)	1944.9	2011.7	2029.0	2033.1
AOB (mg-COD/l)	68.65	72.00	72.81	73.27
NOB (mg-COD/l)	41.64	43.76	44.25	44.35
PAO (mg-COD/l)	549.3	582.8	597.6	611.7
<b>% of PAO</b>	<b>21.1%</b>	<b>21.5%</b>	<b>21.8%</b>	<b>22.1%</b>
Stored poly-P (mg-P/l)	177.0	185.6	187.4	188.3
<b>Stored poly-P (mg-P/mg-PAO)</b>	<b>31.4%</b>	<b>31.1%</b>	<b>30.8%</b>	<b>30.4%</b>

Table 12 summarises the optimisation result of the proportion of step feed between the first and second anoxic tanks. After adjusting the RAS flows to obtain similar effluent TN concentrations (around 9.15 mg/l), it was found that the required influent COD was not very sensitive to the different step feeds over this range. A step feed of 50% entering the first anoxic tank (AOA3 configuration) was one of the better performing ones. Hence, the AOA3 configuration operating with an RAS of 0.73Q and DO of 1 mg/l in the aerated tanks is found to be the optimal MBR configuration. Compared to an influent COD requirement of 365 mg/l for the MLE1 configuration, this optimised AOA3 configuration represents a 19% decrease in the amount of influent COD required.

#### 4.2.4 Sensitivity of effluent quality to MBR influent COD

A sensitivity analysis was performed to investigate how changes in the influent COD will affect the effluent quality for the optimal configuration. It was found that influent COD has the largest impact on the phosphate concentration in the effluent. The effluent TN and COD concentrations are affected to much smaller extents, whereas ammonia was almost unaffected.

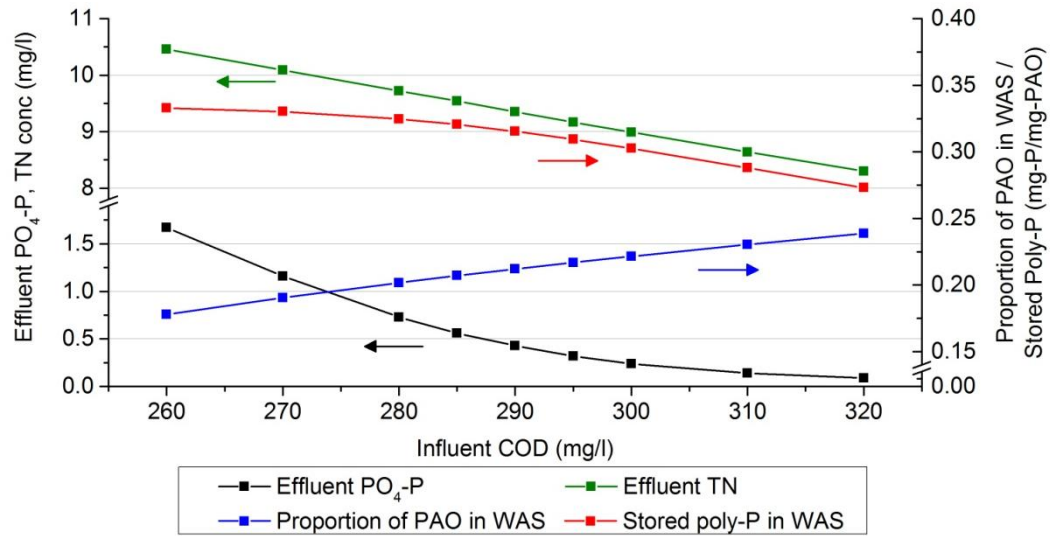


Figure 23: Sensitivity analysis of influent COD on effluent  $PO_4\text{-P}$

Figure 23 shows how the effluent phosphate concentration varies with the influent COD. As the influent COD increases, more COD is available for uptake by PAOs in the anaerobic zone. Hence, the proportion of PAOs increases and the Bio-P performance improves. However, the amount of stored poly-P in the PAOs decreases due to phosphorus limitation. The Bio-P performance is less sensitive to the changes in influent COD at higher COD values as an increase in the PAO proportion is accompanied by a decrease in the amount of stored poly-P. On the other hand, at low influent COD, the remaining PAOs get saturated with poly-P and the decline in PAO proportion when COD decreases results in a much larger change in effluent phosphate concentration.

Starvation is the process whereby the PAOs consume endogenous products such as PHA and poly-P for cell maintenance due to a lack of external substrates (Lu, Keller, & Yuan, 2007). It is unlikely to be occurring in the MBR due to the short aerobic residence time of about 70 minutes. Lopez, Pons, and Morgenroth (2006) found that it takes 4 hours of aerobic starvation for PHA to deplete, and PAOs can still utilise glycogen for cell maintenance for several days after that. The simulation results also revealed that there is still 0.021 mg-PHA / mg-VSS<sub>PAO</sub> at the end of aeration (in the MBT), much higher than the values of < 0.01 mg-PHA / mg-VSS<sub>PAO</sub> observed in the study under starvation conditions.



## 4.3 Effect of biosorption A-stage

### 4.3.1 Effect of designed biosorption conditions

The effect of the A-stage on the overall treatment process was investigated by adding the calibrated biosorption model to the optimal MBR configuration. The following parameters used in the simulation are typical of the raw influent from the equalization tank. The operating conditions are the designed conditions for the biosorption process at IVP or adapted from the biosorption process at the 'Energy+' project.

- Raw influent
  - Flow rate      1000 m<sup>3</sup>/day
  - tCOD:          760 mg/l
  - sCOD:          250 mg/l
  - TKN:          47 mg/l
  - TP:            12 mg/l
- Biosorption tank
  - Volume:        21 m<sup>3</sup>
  - DO:            0.3 mg/l
  - HRT:          ~ 30 min
  - SRT:          0.5 days
- Primary Clarifier:
  - Volume:        85 m<sup>3</sup>
  - Depth:        4 m
  - Underflow:    200 m<sup>3</sup>/d

Table 13 shows the modelled MBR influent and effluent quality after the addition of the biosorption tank, as well as the effluent quality limit used for optimisation in section 4.2 for comparison. Due to differences in the MBR influent in the current model than the MBR influent used for optimisation, the RAS is increased to 1.1Q to ensure sufficient nitrogen removal in the MBR. With the exception of phosphate, all other effluent parameters were found to be lower than their respective limits.

This result is within expectations as biomass in the biosorption tank consumes COD rapidly, decreasing their availability for Bio-P removal in the MBR. In some cases, this can even lead to incomplete denitrification due to insufficient COD. The relatively high COD/TKN value of 16.2 in the raw influent, together with the high hydrolysis rate due to the high temperature, ensured that there is sufficient COD for complete denitrification and some Bio-P activity even after biosorption.

**Table 13: MBR influent and effluent quality with BPT**

Parameter	MBR influent	Effluent	Effluent limit
sCOD (mg-COD/l)	73.7	32.4	60
rbCOD (mg-COD/l)	10.00	1.68	-
VFA (mg-COD/l)	7.66	0	-
NH <sub>4</sub> -N (mg-N/l)	24.13	0.09	1
TN (mg-N/l)	36.96	10.00	10
PO <sub>4</sub> -P (mg-P/l)	3.80	2.63	0.3

### 4.3.2 Operating changes to meet effluent limits

In order to meet the effluent limits as given in Table 13, more COD must be made available in the MBR influent for adequate Bio-P performance. This can be achieved by decreasing the amount of COD removed during the A-stage and there are several alternatives to do so.

#### *Partial bypass of BPT*

The first alternative is to split the raw influent into two streams, one stream entering the biosorption process and the other bypassing biosorption to enter the primary clarifier directly. On one hand, this may increase the COD in the MBR influent as part of the raw influent does not undergo the biosorption process. On the other hand, bypassing part of the flow results in a longer residence time in the biosorption tank and leads to less COD in the mixed liquor leaving the biosorption tank.

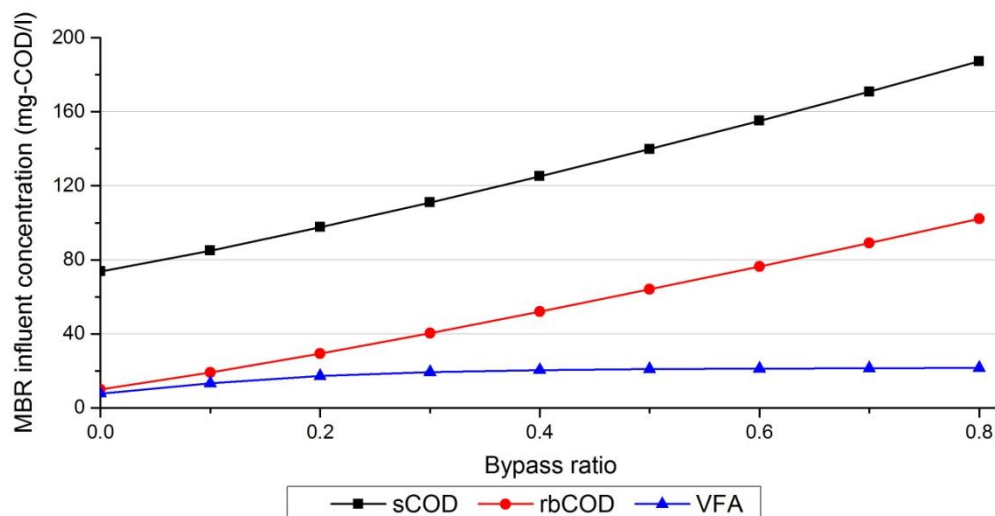


Figure 24: Effect of bypass ratio on sCOD, rbCOD and VFA concentrations in MBR influent

Figure 24 shows how the sCOD, rbCOD (including VFA) and VFA concentrations in the MBR influent varies according to the bypass ratio, which is the proportion of the raw influent that bypasses the biosorption tank and enters the primary clarifier directly. The biosorption SRT was kept constant at 0.5 days by altering the primary sludge wastage rate. The effect of the bypass is found to be larger than the effect of the increased residence time in the biosorption tank, leading to an overall increase in the amount of sCOD in the MBR influent as the bypass ratio increases.

#### *Lower DO in BPT*

The amount of COD uptake is directly related to the growth rate of heterotrophs in the biosorption process, which is influenced by the DO concentration in the biosorption tank. A lower DO in the tank would lead to less COD uptake, and hence more COD present in the MBR influent for nutrient removal in the MBR.

#### *Higher sludge wastage*

The amount of COD uptake is also dependent on the amount of heterotrophs in the biosorption tank. This is controlled by the primary sludge wastage rate, which influences both MLSS in the tank and SRT of the

biosorption sludge. A higher primary sludge wastage rate will result in a lower MLSS in the biosorption tank, which leads to lower COD removal in the A-stage.

## Results

Table 14 summarises the results of the various alternatives mentioned above, with the effluent limit included for comparison. The RAS and sludge wastage of the MBR were adjusted to maintain a MBR RAS flow rate of 0.9Q and an MBR SRT of about 5 days. The main changes are indicated in the remarks, with the initial value given in brackets. All of the alternatives could decrease the COD uptake in the biosorption tank sufficiently for adequate nutrient removal in the MBR.

**Table 14: Biowin results for the different alternatives**

Parameter	Alternative			Limit
	Partial bypass	Lower DO	Higher wastage	
Effluent:				
sCOD (mg/l)	32.9	33.0	33.0	60
NH <sub>4</sub> -N (mg-N/l)	0.08	0.07	0.08	1
TN (mg-N/l)	9.63	9.93	9.80	10
PO <sub>4</sub> -P (mg-P/l)	0.30	0.30	0.30	0.3
MBR influent:				
tCOD (mg/l)	401.1	332.1	365.3	
sCOD (mg/l)	107.9	150.9	122.6	
rbCOD (mg/l)	34.84	66.64	49.87	
VFA (mg/l)	22.51	21.1	20.25	
Recovered tCOD:				
Primary WAS <sup>2</sup> (kg/day)	340.9	306.1	395.2	
Secondary WAS <sup>2</sup> (kg/day)	164.0	185.1	139.4	
<b>Total tCOD (kg/day)</b>	<b>504.9</b>	<b>491.2</b>	<b>534.5</b>	
<b>Response time for effluent PO<sub>4</sub> conc. to decrease (h)</b>	<b>26</b>	<b>74</b>	<b>78</b>	
Remarks	Bypass ratio = 0.38 (0)	BPT DO = 0.08 mg/l (0.3 mg/l)	WAS = 17.8 m <sup>3</sup> /d (8.89 m <sup>3</sup> /d) SRT = 0.32 d (0.5 d)	

Table 14 also shows the COD components in the MBR influent. Among the alternatives, a higher primary sludge wastage results in an MBR influent with the highest amount of sCOD and rbCOD, but the lowest amount of total COD. One likely reason for this is that more COD is wasted in the primary sludge due to the higher wastage flow, resulting in less hydrolysable COD entering the MBR. A higher proportion of rbCOD is hence needed to achieve the required MBR effluent quality.

To maintain a COD balance, a lower amount of COD entering the MBR would lead to a larger amount of COD in the primary WAS. However, it also results in a lower COD content in the secondary WAS. Overall, a higher

<sup>2</sup> Primary WAS and Secondary WAS refers to the sludge wastage from the A-stage primary clarifier and B-stage MBR respectively

primary sludge wastage rate recovers the most COD from the wastewater, 1.7% and 8.1% higher than the partial bypass and lower DO alternatives respectively. As the IVP aims to maximise COD recovery to generate biogas, having a higher primary sludge wastage would be the best alternative during steady state operations to attain the required MBR effluent quality while maximising COD recovery.

In addition to steady state modelling, dynamic modelling was carried out to determine the response speed of the various alternatives. This would be useful for treatment plant operators to decide on the best response in the event that the treatment plant effluent does not meet the required quality due to sudden increases in the nitrogen or phosphorus load in the raw influent. To assess the response speed, the model was initially set up as a steady state as given in section 4.3.1. Changes were then made and the time for the effluent phosphate concentration to decrease to 0.5 mg-P/l was recorded. The results show that the partial bypass has the fastest response to reach the target effluent phosphate concentration. This is expected as the partial bypass will have an immediate effect on COD in the MBR influent, whereas it will take some time for the biosorption process to respond to the changes of a lower DO or a higher primary sludge wastage. Hence, in the short term, controlling the bypass ratio is the best method to respond to sudden changes in the raw influent.

#### 4.3.3 Sensitivity of MBR influent quality to biosorption process changes

The TKN and TP concentrations in the MBR influent were held constant during optimisation in section 4.2. However, any operational changes for the biosorption process to modify the influent COD concentration is expected to change the TKN and TP values too. This is because changes in the biosorption process will affect nitrogen and phosphate uptake for anabolic cell growth and metabolism. Figure 25 shows how the influent COD, TKN, and TP changes with the bypass ratio of the biosorption process. It can be observed that the TKN and TP concentrations are much less sensitive to the bypass ratio than the sCOD concentration, hence validating the procedure of holding TKN and TP concentrations in the MBR influent constant. Similar trends were observed for changes in DO and sludge wastage rate, and these graphs can be found in Appendix D.

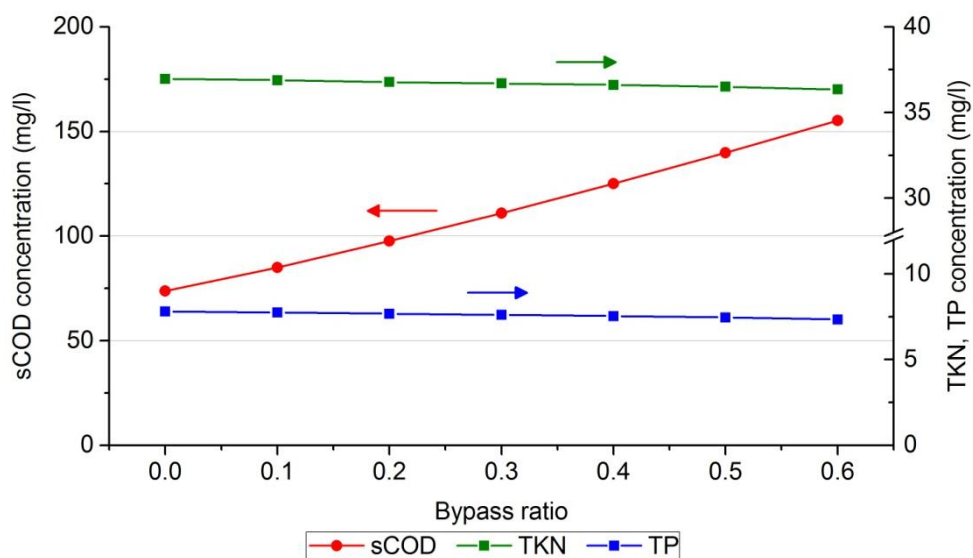


Figure 25: Sensitivity of sCOD, TKN, and TP in the MBR influent to bypass ratio

## 4.4 Discussion of optimisation results

This research aims to answer the question of whether a change in MBR configuration and operating conditions can lead to a reduction of net energy consumption while achieving an acceptable effluent quality. The reference case is where the MBR operates in the MLE1 configuration, which is the most common configuration for wastewater treatment in Singapore. A DO concentration of 2 mg/l in the aerated tanks and an RAS of 2Q are used as these were the starting conditions in the IVP. The IVP is also assumed to be working as designed, with fully functioning biosorption and deoxygenation tanks. The optimised case is where the MBR operates in the AOA03 configuration, with a RAS of 0.9Q and an aerated DO of 1 mg/l. In addition, the biosorption process operates with increased sludge wastage to achieve the required effluent quality.

Table 15 shows a comparison of the results, where the optimised case shows much better performance than the reference case. Firstly, the effluent quality is much better in terms of TN and phosphate concentrations. In addition, the net energy consumption of the optimised case is also much lower, with the tCOD recovered in waste sludge increased by 11% and the total aeration requirements decreased by 20%. This would greatly reduce the total energy footprint of the WRP.

**Table 15: Comparison of reference and optimised cases**

	Reference case	Optimised case
Effluent:		
sCOD (mg/l)	32.0	33.0
NH <sub>4</sub> (mg-N/l)	0.03	0.08
TN (mg-N/l)	11.95	9.80
PO <sub>4</sub> (mg-P/l)	4.64	0.30
Recovered tCOD in sludge:		
Primary WAS (kg/day)	314.4	395.2
Secondary WAS (kg/day)	168.3	139.4
<b>Total tCOD recovered (kg/day)</b>	<b>482.7</b>	<b>534.5</b>
Aeration requirement (m <sup>3</sup> /hr)		
Non-MBR	316.9	195.3
MBR	295.0	295.0
<b>Total</b>	<b>611.9</b>	<b>490.3</b>

## 5 Further discussion

### 5.1 Identification of issues in the IVP

#### 5.1.1 Bio-P removal

Over the course of the past year, it is found that there is usually a high phosphate concentration in the MBR effluent, suggesting a lack of Bio-P removal. However, low effluent phosphate concentrations were observed during certain periods, and tests revealed that these were indeed due to Bio-P activity in the MBR. In addition, the main treatment train at South Works in UPWRP, which takes in the same influent and has an MLE configuration with 5 day SRT, managed to consistently achieve low effluent phosphate concentrations due to Bio-P removal. Hence, it would be interesting to determine the cause of this difference in performance.

The main difference between the IVP and South Works is that the IVP has a biosorption A-stage for primary treatment whereas South Works uses only a primary clarifier. Even though the biosorption process is not functioning well, Table 3 shows that a significant portion of VFA does get removed in the A-stage most of the time. Hence, it is likely that the poor Bio-P removal is due to insufficient VFA.

Consistently good Bio-P removal was observed during 2 periods – May 2014 and Sep 2014. These incidents coincided with certain events occurring in the IVP.

#### 8 – 22 May 2014

The biosorption tank was bypassed from 8 to 22 May 2014 for maintenance. During this period, only the primary clarifier was operating and the IVP worked in the same configuration as South Works. Without the biosorption process, the MBR influent would have had a higher concentration of VFA and other types of rbCOD for secondary treatment. Figure 26 shows the phosphate concentrations in the influent and effluent during that period. The effluent phosphate concentration was mostly below 1.5 mg-P/l, compared to concentration above 2 mg/l before and after the biosorption bypass. The fluctuations in effluent phosphate is most likely due to variation in the COD of the raw influent from the equalization tank.

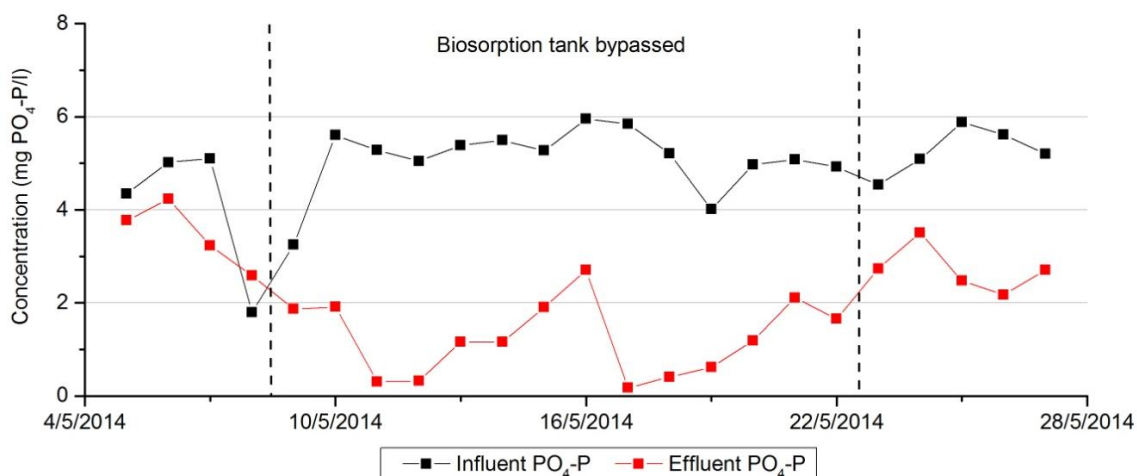


Figure 26: Bio-P removal in May 2014

## 8 - 15 September 2014

Under normal operations, the dewatering centrate from sludge digestion is pumped back to the headworks, where it subsequently enters the equalization tank. However, this centrate was diverted to North Works, another main treatment train, from 8 - 15 September, hence reducing in the nitrogen load to the IVP. This would result in less COD utilised by heterotrophic denitrification in the anoxic zone and more COD available for Bio-P removal. Figure 27 shows the phosphate concentrations in the MBR influent and effluent, as well as the nitrate concentration in the effluent as an approximate indication of the nitrogen load. It can be observed that there is a slow decrease in the effluent phosphate and nitrate during the diversion, supporting the hypothesis that the decreased nitrogen load increased the amount of COD available for Bio-P removal.

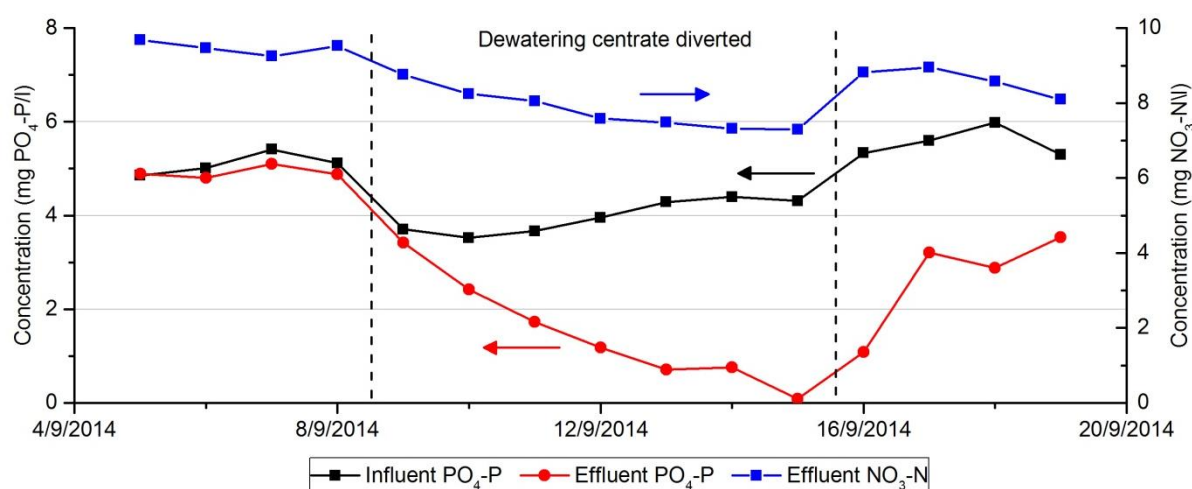


Figure 27: Bio-P removal in Sep 2014

### 5.1.2 DO concentration in deoxygenation tank

The mixed liquor in the membrane tanks contains a high DO concentration due to the aeration required to scour the membranes. On the other hand, the RAS back to the anoxic zone needs to have a low DO concentration for effective denitrification and phosphorus release. Hence, a deoxygenation tank is installed after the membranes tanks for the DO in the mixed liquor to deplete before it is recycled back to the start of the MBR.

Unfortunately, the deoxygenation tank is not functioning very well. The DO concentration in the tank is found to range from 0.2 mg/l to 3.9 mg/l, with an average of about 2 mg/l. Figure 28 shows two typical DO profiles, which tend to fluctuate in 20 min cycles. The fluctuations can be attributed to changes in the water level in the deoxygenation tank, the DO in the aerated membrane tanks, as well as membrane operations. The poor performance of the deoxygenation tank is highly likely to be due to the presence of cascade overflows from the membrane tanks into the deoxygenation tank (Figure 29) that introduces large amounts of oxygen into the mixed liquor. Hence, a DO balance of the deoxygenation tank will be performed in this section to determine the impact of the cascade. Subsequently, Biowin simulations of a well-functioning deoxygenation tank will be performed to determine its impact on nutrient removal, and potential modifications to the design of the deoxygenation tank to minimise the effect of the cascade will also be discussed.

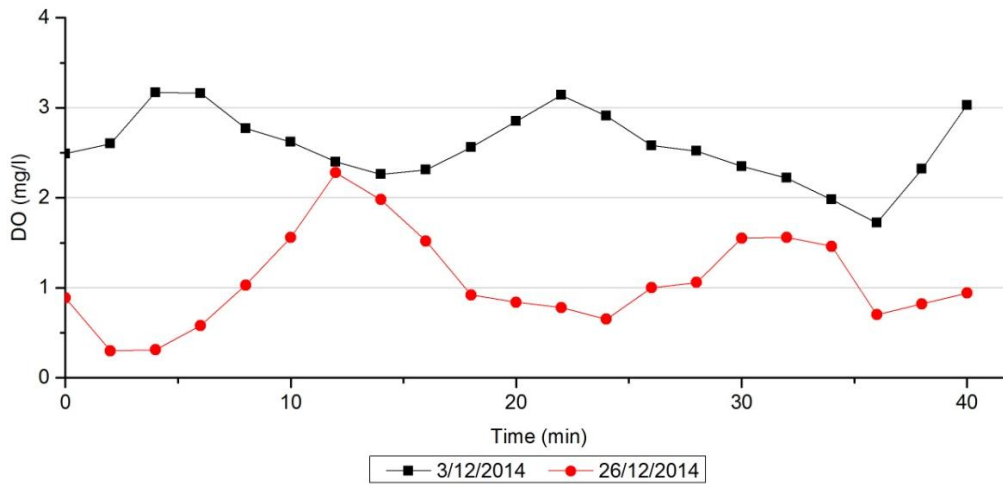


Figure 28: Typical DO profile in the deoxygenation tank



Figure 29: Cascades in the deoxygenation tank

#### Contribution of cascade to DO in deoxygenation tank

A DO balance was performed in the deoxygenation tank to investigate the introduction of oxygen through the cascade overflow. The mixed liquor enters from the membrane tanks at an average flowrate of  $82 \text{ m}^3/\text{h}$  ( $22.8 \text{ l/s}$ ) and an average DO of  $4 \text{ mg/l}$ . The effluent flowrate is  $35 \text{ m}^3/\text{h}$  and the volume of the deoxygenation tank is  $13 \text{ m}^3$  ( $13,000 \text{ l}$ ).

DO balance (per litre of mixed liquor entering deoxygenation tank):

$$\text{Residence time in deoxygenation tank} = \frac{13000}{22.8} = 571 \text{ s}$$

$$\text{OUR in the deoxygenation tank} = 23.1 \times \frac{82+35}{82} = 33.0 \text{ mg/hr} = 0.00916 \text{ mg/s}^3$$

$$\text{DO consumption} = 571 \times 0.00916 = 5.23 \text{ mg}$$

<sup>3</sup> Adapted from the  $\text{OUR}_{\text{en}}$  from Table 5 (Nitrifier OUR tests). The OUR tests was performed on mixed liquor from the aeration tank. As the mixed liquor in the deoxygenation tank is more concentrated due to effluent withdrawal in the MBT, the above equation increases the OUR accordingly. The typical effluent and RAS flows are  $35 \text{ m}^3/\text{h}$  and  $82 \text{ m}^3/\text{h}$  respectively



Through DO balance,  $DO_{in} + DO_{cascade} = DO_{consumed} + DO_{out}$

$$4 \text{ mg} + DO_{cascade} = 5.23 \text{ mg} + 2 \text{ mg}$$

Hence, 1 litre of mixed liquor introduces 3.23 mg of DO through the cascade.

#### Cascade calculations:

Saturated DO in water at 31°C,  $c_s = 7.43 \text{ mg/l}$  (APHA et al., 1999)

Actual DO in deoxygenation tank,  $c_w = 2 \text{ mg/l}$

$$\text{Rate of DO transfer, } \frac{dc_w}{dt} = k_2 \times (c_s - c_w) = 0.001 \times (7.43 - 2) = 0.00543 \text{ mg/(l.s)}^4$$

$$\text{DO introduced through aeration} = 0.00543 \times 571 = 3.10 \text{ mg/l}$$

The above calculations show that the cascades contribute a large amount of DO into the deoxygenation tank. Without it, the OUR of the mixed liquor is sufficient to deplete the DO in the tank. However, it must be noted that the gas transfer coefficient ( $k_2$ ) used in the cascade calculations is a device-dependent parameter depending on the contact surface area and amount of turbulence (TU Delft, 2014). It is also dependent on the height of the cascades, which varies depending on the water level in the deoxygenation tank. The value of  $0.001 \text{ s}^{-1}$  is an estimate that gives results which agree with the DO balance. Experiments would need to be conducted to more accurately determine the gas transfer coefficient as a function of the cascade height.

The gas transfer coefficient is much lower than the value of  $0.02 \text{ s}^{-1}$  given by Rietveld (2013). One reason would be that this cascade was not specifically designed for mass transfer and hence has a much lower efficiency. Another reason would be that the given value is for cascade aeration of clean water, whereas the oxygen transfer efficiency in wastewater would be lower (Mueller, Boyle, & Popel, 2002).

#### *Effect of a functioning deoxygenation tank*

The previous profile studies had been simulated with a DO concentration of 2 mg/l in the deoxygenation tank. These studies were simulated again in Biowin with a DO concentration of 5 mg/l, as well as with an unaerated deoxygenation tank, to investigate how DO concentrations in the deoxygenation tank affects the performance of the MBR. The unaerated deoxygenation tanks for all the simulations have DO concentrations below 0.25 mg/l, supporting the above calculations that the deoxygenation tank is adequately sized for depleting DO from the mixed liquor had the cascades been absent. Similar trends were found in all the profile studies, and the results of the study carried out on 10 June 2014 (MLE configuration, 5d SRT) are presented below. Figures 30 to 33 show the effect of the DO concentration in the deoxygenation tank on the  $\text{PO}_4$ ,  $\text{NO}_3$ ,  $\text{NH}_4$  and COD concentrations respectively in the MBR. The measured data from the 10 June profile study is added for comparison, but there was no measured data for the deoxygenation tank.

---

<sup>4</sup>  $k_2$  denotes the gas transfer coefficient, which is assumed to be  $0.001 \text{ s}^{-1}$  in the calculations

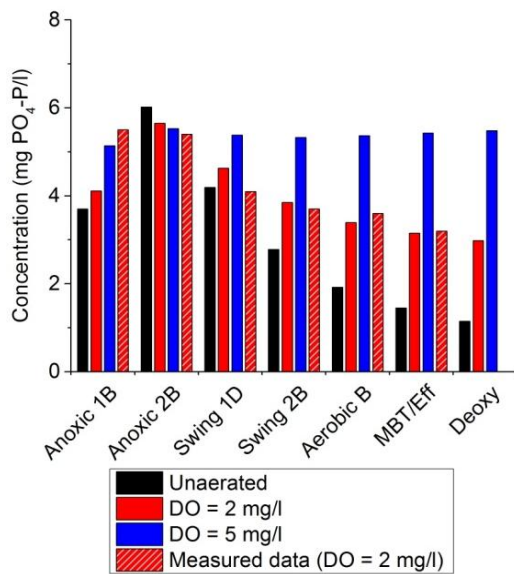


Figure 30: Effect of DO in deoxygenation tank on PO<sub>4</sub> concentrations in the MBR

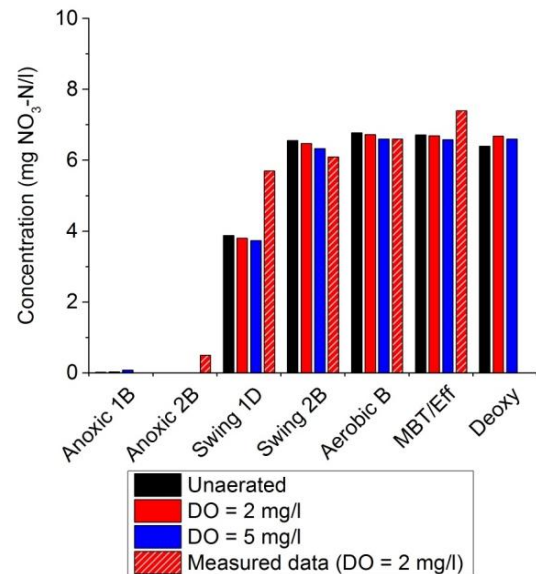


Figure 31: Effect of DO in deoxygenation tank on NO<sub>3</sub> concentrations in the MBR

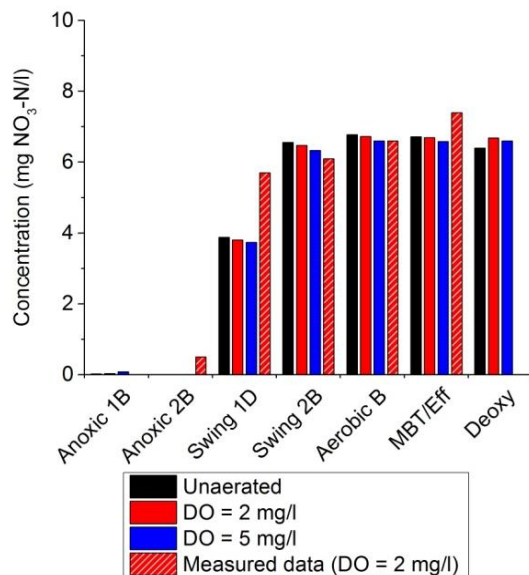


Figure 32: Effect of DO in deoxygenation tank on NH<sub>4</sub> concentrations in the MBR

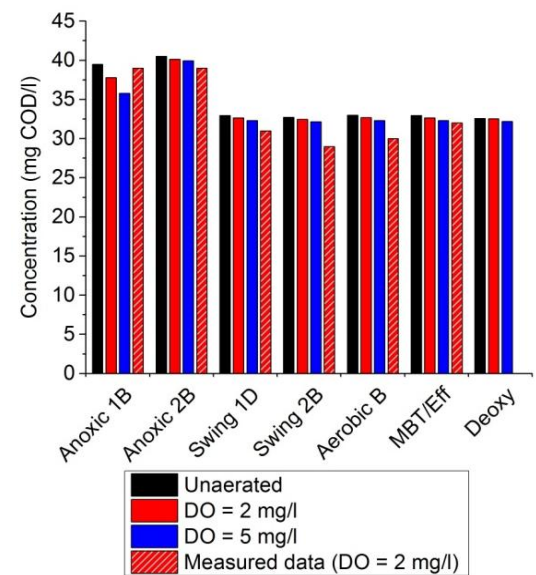


Figure 33: Effect of DO in deoxygenation tank on COD concentrations in MBR

Table 16: Bacterial population in WAS

DO (mg/l)	PAO (mgCOD/l)	OHO (mgCOD/l)	AOB (mgCOD/l)	NOB (mgCOD/l)
Unaerated	233.7	1873	47.2	28.8
2	137.9	1936	46.7	28.2
5	24.7	2023	46.4	27.8

The major effect of the DO concentration in the deoxygenation tank is on the Bio-P performance of the MBR, where a lower DO concentration results in a higher phosphate release during the anaerobic zone and a higher phosphate uptake in the aerobic zone. This can be attributed to an increase in the PAO population at a lower DO concentration (Figure 8) and indicates that less DO in the deoxygenation tank is conducive for PAO growth.

Firstly, an unaerated deoxygenation tank selects for DPAOs. The condition in this tank would be high in nitrate but low in rbCOD, making it unlikely for heterotrophic denitrification to be occurring significantly. On the other

hand, this condition would be conducive for DPAOs and it is likely that they are performing denitrification in the deoxygenation tank. However, it must be noted that the overall denitrification by DPAOs is not very significant compared to denitrification by heterotrophs due to the small size of the deoxygenation tank, which is only about 5% of the total MBR volume.

Next, a lower DO in the deoxygenation tank also leads to a lower nitrate load in the Anoxic 1 tank. From Figure 31, a small amount of denitrification could be observed to occur in an unaerated deoxygenation tank, thus reducing the nitrate load in the RAS. In addition, Figure 32 shows that an unaerated deoxygenation tank leads to a slightly higher ammonia concentration in the Anoxic 1 tank, suggesting that a small part of the influent ammonia can be nitrified in the Anoxic 1 tank by the residual DO in the RAS and an unaerated deoxygenation tank would minimise this nitrification. These two factors leads to a lower amount of COD consumed by the denitrifying heterotrophs in the Anoxic 1 tank, increasing the anaerobic residence time and making more COD available for uptake by PAOs.

The following DO and COD balances in the Anoxic 1 tank were performed, with influent and RAS flowrates of 36.6 m<sup>3</sup>/h and 82 m<sup>3</sup>/h respectively. The difference in nitrogen concentrations in the Anoxic 1 tank were analysed, comparing between the cases of an unaerated deoxygenation tank and a deoxygenation tank with a DO concentration of 5 mg/l.

#### DO balance:

$$\text{Difference in DO recycled to Anoxic 1 tank} = \frac{82}{36.6+82} \times 5 = 3.46 \text{ mg/l}$$

Assuming all nitrified NH<sub>4</sub> is present as NO<sub>2</sub>, 1g of NH<sub>4</sub>-N consumes 3.43g of O<sub>2</sub> for nitrification<sup>5</sup>. For the Anoxic 1 tank with an unaerated deoxygenation tank,

$$\text{Difference in NH}_4\text{-N in the tank} = 7.94 - 7.5 = 0.44 \text{ mg-N/l}$$

$$\text{Difference in O}_2 \text{ consumed for nitrification} = 0.44 \times 3.43 = 1.51 \text{ mg-O}_2\text{/l}$$

$$\text{Difference in O}_2 \text{ consumed by non-denitrifying heterotrophs} = 3.46 - 1.51 = 1.95 \text{ mg-O}_2\text{/l}$$

#### COD balance:

For full denitrification, 1g of NO<sub>3</sub>-N and 1g of NO<sub>2</sub>-N requires 2.86 and 1.71g of COD respectively. For the Anoxic 1 tank with an unaerated deoxygenation tank,

$$\text{Difference in COD consumption in the tank} = 39.5 - 35.8 = 3.7 \text{ mg-O}_2\text{/l}$$

$$\text{Difference in NO}_3\text{-N in the tank (due to recycle)} = \frac{82}{36.6+82} (6.6 - 6.4) = 0.14 \text{ mg-N/l}$$

$$\text{Difference in NO}_2\text{-N in the tank (due to partial nitrification)} = 0.44 \text{ mg-N/l}$$

$$\text{Difference in COD consumed by denitrifiers} = 0.14 \times 2.86 + 0.44 \times 1.71 = 1.16 \text{ mg-O}_2\text{/l}$$

$$\text{Difference in COD consumed by non-denitrifying heterotrophs} = 3.7 - 1.16 = 2.54 \text{ mg-O}_2\text{/l}$$

---

<sup>5</sup> The influent NH<sub>4</sub> is assumed to be partially nitrified into NO<sub>2</sub> due to the limited availability of DO in the Anoxic 1 tank.

The above calculations show that in addition to nitrifiers, a large portion of the recycled DO is consumed by the non-denitrifying heterotrophs. The amount of COD consumed by non-denitrifying heterotrophs calculated through the COD and DO balances agree fairly well, and the difference is likely due to factors such as anabolic processes and hydrolysis that were not considered in the above balances.

In summary, a low DO in the deoxygenation tank would lead to better Bio-P performance by increasing the PAO population in the MBR. It provides an environment that is conducive for the growth of DPAOs and selects for them, and also increases the amount of COD available in the anaerobic zone for assimilation by PAOs as less is consumed by heterotrophs in the anoxic zone.

### *Possible modifications to the deoxygenation tank*

The design of the deoxygenation tank could be modified slightly to minimise unwanted aeration of the deoxygenation tank through the cascade overflow. Possible modifications include the installation of baffle walls (Figure 34) or a slope (Figure 35) to minimise the effect of the cascade. Another option would be to install overflow pipes that lead below the water surface to eliminate the cascade, similar to what has been done at the start of the Anoxic 1 tank (Figure 36).

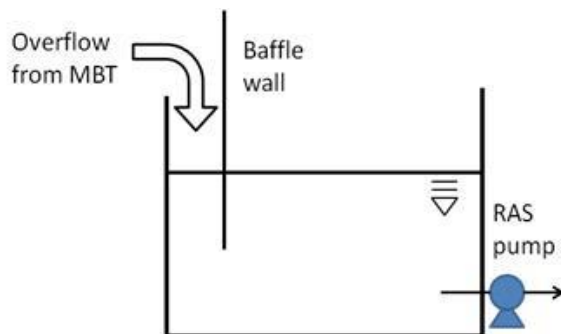


Figure 34: Installation of baffle walls in the deoxygenation tank to minimise cascade

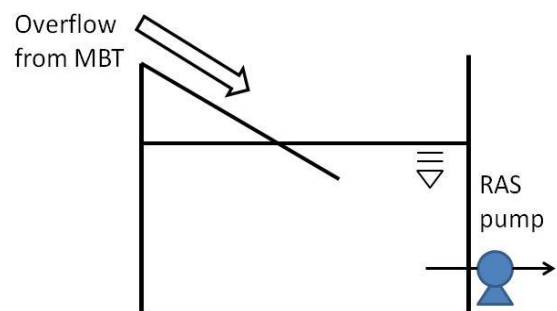


Figure 35: Installation of slope in the deoxygenation tank to minimise cascade

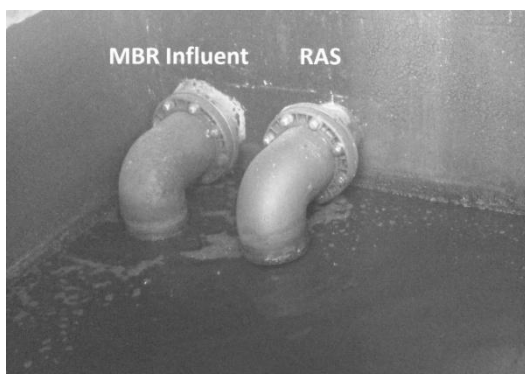


Figure 36: Overflow pipes at beginning of Anoxic 1 tank

## 5.2 Mainstream Anammox

In Singapore, current research is focused on reducing the energy requirements of the WRPs, with the ultimate aim of achieving energy self-sufficiency. One of the processes that is under consideration is mainstream Anammox (Lee et al., 2013), which is one of the most energy-efficient methods of nitrogen removal from wastewater as it requires 60% less oxygen and 89% less COD than the conventional nitrification-denitrification process. However, as Anammox performs best at high temperatures, it is currently widely used only in the sidestream treatment of the supernatant from sludge dewatering. With the high wastewater temperature of about 31°C throughout the year, mainstream Anammox may prove to be a viable option in Singapore. One disadvantage of this process is that since the bacteria involved do not consume COD, Bio-P removal is not possible. As a result, phosphorus removal through chemical dosage will be needed for plants utilising mainstream Anammox but require a low effluent phosphorus concentration.

In addition, most of the WRP effluent in Singapore will be further treated and reclaimed as potable water by tertiary treatment using reverse osmosis (RO). As the MBR effluent can be sent directly to the RO without an intermediate treatment process, an analysis had been carried out to compare the aeration energy required to scour the MBR membranes with the energy required for a conventional ultrafiltration (UF) treatment step. The results showed that operating an MBR is more energy efficient, requiring an aeration energy of 0.11 kWh/m<sup>3</sup> for membrane scouring, compared to an average of 0.13 kWh/m<sup>3</sup> required for UF treatment (Lee et al., 2013). Hence, any future wastewater treatment processes in Singapore will be based on the MBR technology.

Therefore, Biowin is used to investigate the feasibility of mainstream Anammox in Singapore, incorporating an MBR where possible. The SRT of the Anammox process is maintained at 60 days, which is similar to an SRT of 59.5 days used in the Biowin simulations for Dokhaven treatment plant, Netherlands (Fei, 2012), and agrees with an SRT of more than 50 days as recommended by Wett et al. (2010) for Anaerobic Ammonia Oxidisers (Anammox bacteria; AAO). In addition, the biosorption tank volume is doubled from 21 m<sup>3</sup> to 42 m<sup>3</sup> and the solids removal efficiency in the primary clarifier is increased from 96% to 99.5%. This is to reduce the amount of COD entering the MBR as COD will promote the growth of heterotrophic denitrifiers at the expense of AAO. The simulation results for the designed biosorption process (section 4.3.1) show that there is adequate COD in the MBR influent for denitrification and hence no Anammox activity will develop. However, it must be noted that more research will be required to determine if a removal efficiency of 99.5% can be achieved under the local conditions. The raw influent quality, the membrane tank, and the deoxygenation tank were kept the same as before where applicable, and several different Anammox configurations were simulated to investigate their feasibility in Singapore.

### 5.2.1 SHARON-Anammox

The SHARON-Anammox process is a common two-stage process where some ammonia is partially nitrified into nitrite in the aerated SHARON reactor, then undergoes Anammox to produce nitrogen gas in the unaerated Anammox reactor. As this configuration works using suspended sludge, the Anammox process can be operated

in an MBR to achieve an effluent that is of sufficient quality for use as RO feed. Although DO inhibits Anammox activity, this process is reversible (Jetten et al., 2001) and the aerated membrane tanks is hence not expected to inhibit Anammox activity in the MBR significantly. Figure 37 shows the Biowin setup to model the SHARON-Anammox configuration.

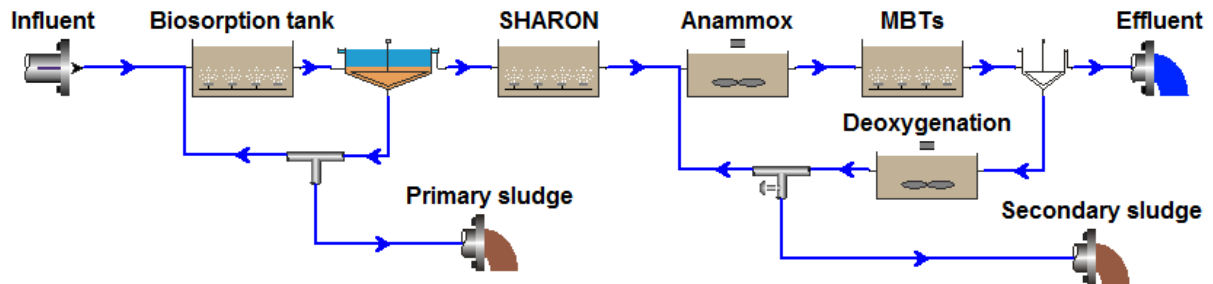


Figure 37: Biowin setup for the SHARON-Anammox configuration

## 5.2.2 CANON

A one-stage configuration that is currently being used is CANON, which allows nitrification and Anammox to occur within the same aerated reactor. One common form of CANON is to use sludge granules to generate the necessary DO profile, and this method is on trial at the Dokhaven WWTP in the Netherlands (Fei, 2012). AOB will occupy the aerated outer core of the granules, utilising DO to oxidise ammonia to nitrite. The unused ammonia and nitrite will then diffuse to the anoxic granule core, where AAO perform Anammox to produce nitrogen gas.

In order to generate well-formed granules, a selection pressure must be applied to retain fast-settling granules within the reactor. An MBR does not provide the necessary selection pressure and is hence unsuitable for the CANON process using granular sludge. Figure 38 shows the Biowin setup used to simulate the CANON configuration. As Biowin is unable to simulate granular sludge directly, several kinetic parameters for AAO were adapted from Fei (2012) and implemented in the model to account for the difference between granular sludge and suspended sludge (Table 17).

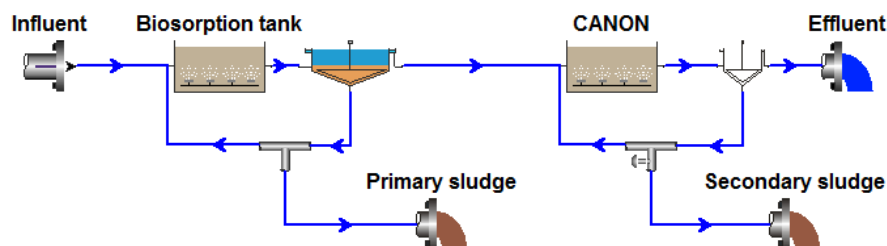


Figure 38: Biowin setup for the CANON configuration

Table 17: Modified kinetic parameters for Anammox in CANON configuration

Parameter	Default value	Modified value
$\text{NH}_4^+$ half-saturation (mg-N/l)	2	0.2
$\text{NO}_2^-$ half-saturation (mg-N/l)	1	0.1
DO half saturation (mg- $\text{O}_2$ /l)	0.01	1

### 5.2.3 DEMON

The DEMON configuration is another design which is widely implemented for sidestream nitrogen removal. In fact, a pilot DEMON plant has been set up in UPWRP to investigate sidestream Anammox treatment of the dewatering centrate. This process is usually carried out in an SBR, where intermittent aeration is provided to allow for cycles of partial nitrification and Anammox. The alternative DEMON configuration at the Glarnerland WWTP in Switzerland using consecutive aerated and unaerated tanks (Wett et al., 2010) could be implemented in an MBR, hence providing effluent of adequate quality as an RO feed without an intermediate treatment step. Figure 39 shows the Biowin setup for this DEMON configuration.

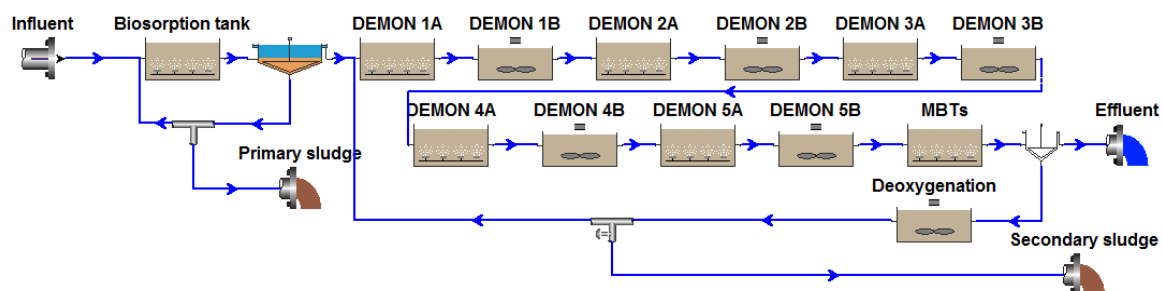


Figure 39: Biowin setup for the DEMON configuration

### 5.2.4 Results

Table 18 compares the results of the three Anammox configurations, together with the AOA03 configuration with higher primary sludge wastage. The tank dimensions and other operating conditions for the Anammox configurations are summarised in Appendix C. As expected, the Anammox configurations are able to achieve much lower effluent nitrogen concentrations at the expense of higher phosphate concentrations. The required aeration rate (excluding membrane scouring) is more than 35% lower than the optimised AOA03 configuration due to lower organic load and lower nitrification in the secondary treatment. The total COD recovered in the waste sludge is also about 11% higher, mainly due to the improved efficiency of the primary clarifier.

Among the Anammox processes, it was found that all of them are able to achieve comparable effluent quality. The CANON process was the most efficient at removing nitrogen through Anammox as it has the highest AAO proportion in the biomass. It also has the highest percentage of nitrogen gas produced by AAO, which indicates the proportion of nitrogen removed through Anammox compared to the total amount removed through Anammox and denitrification. This could be attributed to the compact nature of granular sludge, where the nitrite produced by the AOB is in close proximity with the AAO and quickly consumed for Anammox.

However, there are also some disadvantages to the CANON process as well. For instance, a CANON process cannot be used in conjunction with an MBR, resulting in the need for an intermediate treatment step using ultrafiltration before reverse osmosis. In addition, a CANON system also involves more complex reactor designs to apply a selection pressure for sludge settling. For example, a laminar settler or a three-phase separator needs to be added on top of the CANON reactor such that granules can settle back into the reactor and the suspended sludge leaves with the effluent. The SHARON-Anammox and DEMON configurations are able to work with much simpler reactors, like those currently used in the IVP.

Altogether, CANON and DEMON can be considered to be the most suitable configurations for mainstream Anammox for implementation in Singapore. Although CANON requires a smaller tank volume for reaction, which is a huge plus in land-scarce Singapore, extra space is required for the laminar clarifier or three phase separator to select for Anammox granules, as well as the subsequent UF treatment. The energy requirements are also similar, where the extra energy required for UF treatment of effluent in CANON is mostly offset by the energy savings as membrane scouring is not needed. More research is required to determine the better configuration, such as by examining the stability of the Anammox granules and the relative growth rates of AOB and NOB at local conditions. For the DEMON configuration, the performance could be further enhanced by having a cyclone in the waste sludge stream to select the denser Anammox bacteria for return to the MBR (Wett et al., 2010). The SHARON-Anammox process requires the least energy for aeration, agreeing with L. Zhang et al. (2008) that this process has lower operational costs than CANON. However, its considerable land footprint due to the large SHARON tank required makes it an unattractive option for Singapore.

**Table 18: Comparison of Anammox alternatives**

<b>Parameter</b>	<b>SHARON-Anammox</b>	<b>DEMON</b>	<b>CANON</b>	<b>Optimised AOA03</b>
MBR influent COD (mg/l)	89	89	89	365
Effluent:				
sCOD (mg/l)	31.8	31.9	32.8	33.0
NH <sub>4</sub> (mg-N/l)	0.03	0.03	0.17	0.08
TN (mg-N/l)	3.94	2.76	2.69	9.80
PO <sub>4</sub> (mg-P/l)	3.69	3.71	3.69	0.30
% of AAO in secondary biomass	10.7%	11.5%	18.4%	-
<b>% N<sub>2</sub> production by AAO</b>	<b>69.8%</b>	<b>62.5%</b>	<b>90.8%</b>	-
<b>Total tank volume (A + B-stage) (m<sup>3</sup>)</b>	<b>1040</b>	<b>370</b>	<b>247</b>	<b>346</b>
<b>Required aeration (excluding membrane scouring) (m<sup>3</sup>/hr)</b>	<b>101.3</b>	<b>122.9</b>	<b>124.7</b>	<b>195.3</b>
tCOD in WAS:				
Primary WAS (kg/day)	582.3	582.3	582.3	395.2
Secondary WAS (kg/day)	14.4	13.4	14.1	139.4
<b>Total tCOD (kg/day)</b>	<b>596.7</b>	<b>595.7</b>	<b>596.4</b>	<b>534.5</b>

An interesting observation that can also be made from Table 18 is the percentage of nitrogen gas produced through denitrification in the SHARON-Anammox and DEMON processes. Despite having a biosorption A-stage and an excellent 99.5% removal efficiency in the primary clarifier, there is still a large amount of COD present in the secondary treatment resulting in over 30% of the nitrogen removed through denitrification. Hence, mainstream Anammox may not actually be required or achievable in Singapore. Unless the high removal efficiency can be achieved consistently, Anammox will not develop as the slow-growing AAO will be outcompeted by the fast-growing denitrifiers for nitrite due to the availability of COD as substrate. In this case, it may be better to design an MBR to optimise the use of available COD for denitrification as this can provide a more robust and stable performance. In addition, Bio-P removal could also be achieved, eliminating the need for chemical phosphorus removal to achieve the required effluent quality.



## 5.3 Downstream RO process

### 5.3.1 Introduction

In Singapore, the effluent from the WRP will undergo RO to be reclaimed as potable water (known as NEWater). Most NEWater plants are located next to or within the WRP, and the effluent is piped to the NEWater plants at a constant rate. The rest of the effluent is reclaimed as industrial water or discharged into the sea. This section explores possible synergistic effects between the wastewater treatment and water reclamation processes.

At the NEWater plants, the effluent is first dosed with anti-scalants to prevent scaling. Sodium hypochlorite, together with ammonium if necessary, is also added to form monochloramine at a concentration of 2 mg/l to prevent biofouling. As the existing WRPs do not use MBRs, the effluent is pre-treated with microfiltration (MF) or UF before entering the RO process. After the RO process, the water is further disinfected with ultraviolet light to ensure that all bacteria and other organisms are inactivated. Lastly, the pH balance is restored through the use of chemicals before the water is piped to industries and reservoirs.

### 5.3.2 Combining secondary clarifier and ultrafiltration

The most important synergistic effect is that the MBR combines the functions of the secondary clarifier and UF treatment steps. This allows the treatment plants to have a much smaller land footprint, which is a huge plus in land-scarce Singapore. The energy needed to operate an MBR is also slightly lower than that required for a UF process.

### 5.3.3 Reduction of ammonia usage

The next synergistic potential would be options to reduce chemical dosage in the NEWater production process, specifically ammonia and anti-scalants. All drinking water in Singapore is chlorinated through the addition of sodium hypochlorite to disinfect the water and prevent regrowth of pathogens in the distribution system. The addition of the disinfectant before the RO process also prevents biofouling in the RO membranes. However, as RO membranes are vulnerable to free chlorine (Malaeb & Ayoub, 2011), free chlorine needs to be first converted to chloramine through reaction with ammonia, which is added to the MBR effluent if the ammonia concentration is too low. Hence, a possible improvement would be to reduce the amount of nitrification in the MBR, leaving an effluent ammonia concentration of about 2 mg-N/l for the formation of chloramine.

Figure 40 shows the Biowin model to investigate the feasibility of producing the above effluent ammonia concentration. It is adapted from the AOA03 configuration, where the first aerobic zone is reduced in size and the second aerobic zone omitted to reduce the amount of nitrification. With the MBR influent quality used in section 4.2, a DO of 0.4 mg/l in the aerated tank results in an effluent ammonia concentration of 1.96 mg-N/l, with acceptable TN and phosphate concentrations (Table 19). However, the production of nitrous oxide may be a concern due to the low DO in the aerated tank. In addition, the nitrite concentration in the effluent is also found to be a little high at 1.78 mg-N/l, and Tallec, Garnier, Billen, and Gousailles (2006) reported that a higher

concentration of nitrite in the mixed liquor also increases the production of nitrous oxide. However, they used a much higher nitrite concentration of up to 10 mg-N/l in the study.

In addition, Tchobanoglous et al. (2003) reported that nitrite is oxidised into nitrate by sodium hypochlorite, requiring 4 g-chlorine/g  $\text{NO}_2\text{-N}$ . Its presence in the MBR effluent would mean that a higher dosage of sodium hypochlorite is needed to obtain the same concentration of chloramine in the RO feed. Hence, although it is possible to increase the amount of ammonia in the MBR effluent by decreasing nitrification, whether it is cost-effective to do so would depend on factors such as the relative cost of ammonia and sodium hypochlorite.

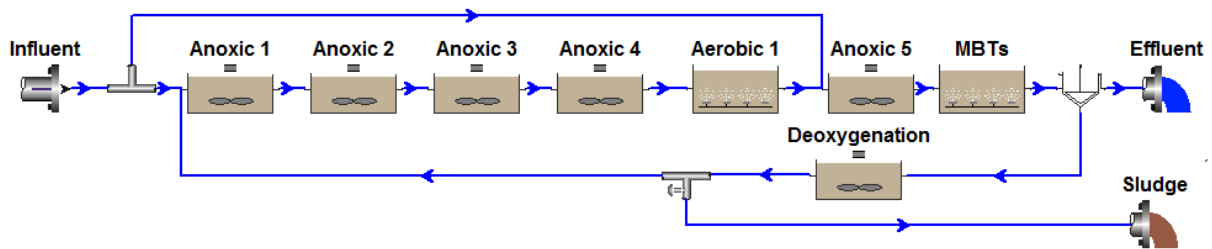


Figure 40: Biowin setup to produce 2 mg/l  $\text{NH}_4\text{-N}$  in effluent

Table 19: Effluent quality of setup to produce 2 mg/l  $\text{NH}_4\text{-N}$  in effluent

Parameter	Concentration
Influent:	
COD (mg/l)	295
Effluent:	
sCOD (mg/l)	29.02
$\text{NH}_4\text{-N}$ (mg-N/l)	1.96
$\text{NO}_2\text{-N}$ (mg-N/l)	1.78
$\text{NO}_3\text{-N}$ (mg-N/l)	4.57
TN (mg-N/l)	9.86
$\text{PO}_4\text{-P}$ (mg-P/l)	0.30

#### 5.3.4 Reduction of anti-scalant usage

Major scalants of RO membranes include calcium carbonate and calcium phosphate (Bryne, 1995). Operational experience in Singapore revealed that calcium phosphate is the main scalant and anti-scalants are dosed during NEWater production to minimise scaling in the RO process. A good bio-P process would be able to lower the phosphate concentration in the MBR effluent, hence reducing the amount of anti-scalants required. However, with an effluent calcium concentration of 25 mg/l, even a very low phosphorus concentration of 0.01 mg-P/l would result in a solubility product ( $K_{sp}$ ) of  $2.5 \times 10^{-23}$ , which is higher than the solubility constant of calcium phosphate (about  $1.26 \times 10^{-29}$ ) (Kucera, 2015). Hence, precipitation in the RO membranes will still occur to some extent and some anti-scalants will be needed to minimise fouling.

#### 5.3.5 Conclusion

The above analysis demonstrates that the secondary treatment process can be modified to achieve synergistic effects with the downstream RO process. Firstly, the membrane filtration process in the MBR can replace the UF treatment process, achieving energy savings and reducing the land footprint. The ammonia concentration

in the effluent can also be increased to about 2 mg/l by decreasing the aerobic DO and aerobic residence time in the MBR. This can eliminate the dosage of ammonia to produce chloramine that prevents biofouling in the RO process, although this also results in a higher sodium hypochlorite requirement to oxidise nitrite in the effluent. Lastly, a lower phosphate in the effluent can also reduce the amount of anti-scalant needed to minimise scaling in the RO membranes.

## 6 Further studies

### *Calibration of Biowin model*

Due to the large number of parameters involved, their default values were used in this study as the starting values for calibration and changed where the experimental data differ significantly from the modelled data. Other methods of calibration are also available and could possibly be implemented to further refine the calibration results. One such example is the General Likelihood Uncertainty Estimation (GLUE) proposed by Beven and Binley (1992) which is widely used in calibrating environmental models.

In GLUE, the parameters are represented by distributions instead of point estimations to factor in uncertainty of the values. A large number of runs is then made with a random selection of these parameters, and they are then ranked based on how well they match the experimental values. The parameter sets that give the best simulation results are then analysed to obtain confidence intervals for the calibrated parameters. (Singh & Frevert, 2002)

Most of the default parameter values in Biowin are based on experimental results and hence may not subject to large uncertainties. However, a number of values is not directly measured in experiments but are calibrated using the Biowin model. In addition, some of the kinetic parameters are temperature dependant, and the Arrhenius values used to correct for this temperature-dependency is mostly based on experiments carried out at lower temperatures. These parameters would have larger associated uncertainties and GLUE could possibly be used for calibrating them.

### *IVP optimisation*

The parameters for the biosorption tank were calibrated with results from the 'Energy+' project at Kranji WRP. However, that pilot plant has an influent flow rate of only 48 m<sup>3</sup>/day, much smaller than the designed capacity of the IVP (1000 m<sup>3</sup>/day). As a result, the biosorption tank and primary clarifier there are also much smaller, being only about 5% the size of the tank and clarifier at the IVP. Hence, the results from the biosorption model in Biowin may not be very accurate due to errors involved in scaling up. In addition, the simulations with a lower DO, increased sludge wastage, and partial bypass were performed assuming a constant primary clarifier removal efficiency of 96%. In reality, the operating conditions of the biosorption process will affect the removal efficiency of the clarifier as the ability of biomass to bind colloidal particles would change. Hence, further studies would be needed to investigate the impact of these changes on the removal efficiency of the primary clarifier and the downstream MBR process.

The AOA03 configuration with an aerated DO of 1 mg/l is the optimal MBR configuration from the point of view of the biological treatment of wastewater. However, besides lower sludge settleability and increased nitrous oxide production, a low DO also generally leads to poorer sludge filterability and higher membrane fouling rates, possibly due to changes in particle sizes or microbial community in the mixed liquor (Le-Clech, Chen, & Fane, 2006). Since this configuration is to be implemented in an MBR, further studies would be required to assess the suitability of this configuration after taking membrane fouling into consideration.

### Mainstream Anammox

Xu, Xue, Wang, Wang, and Yang (2014) proposed a configuration for Anammox removal of nitrogen using an MBR. It consists of an unaerated Anammox tank, followed by an aerated membrane tank where partial nitrification takes place. A high return ratio feeds nitrite back from the membrane tank to the Anammox tank. Biowin simulations using a slightly modified configuration (Figure 41) revealed that this configuration is indeed able to remove nitrogen through Anammox. With a return RAS of 4Q, the effluent ammonia, nitrite, and nitrate concentrations are found to be 1.38, 1.28, and 4.19 mg-N/l respectively, with 35% of nitrogen removed through Anammox.

This configuration is a novel solution and can prove to be more energy efficient than the CANON and DEMON configurations due to the reduced aeration requirement. However, more research needs to be conducted to examine its stability and the required operational controls. Furthermore, the nitrite concentration in the effluent is a little high and studies could be conducted to find adjustments in the controls or configuration to lower the effluent nitrite concentration.

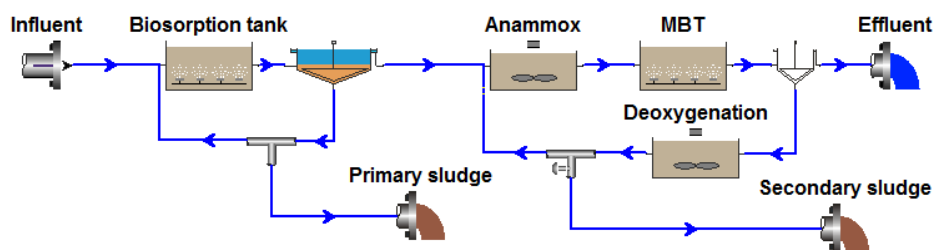


Figure 41: Biowin setup for the Anammox - MBR process

### Tuas WRP (TWRP)

The main aim of the IVP is to identify suitable technologies and treatment processes for the domestic wastewater treatment train at the upcoming Tuas WRP (TWRP). The results from this research project will contribute towards the final design for the WRP. However, there are several factors that need to be considered when using the current results in designing TWRP.

Firstly, although the raw influent to UPWRP is expected to be similar to TWRP, the raw influent to the IVP is influenced by several factors within UPWRP and may differ in quality. For example, the dewatering centrate at UPWRP is returned directly to the headworks without treatment whereas sidestream Anammox treatment is planned for TWRP. Van Loosdrecht and Salem (2006) reported that the high ammonia concentration in the dewatering centrate contains up to 30% of the nitrogen load to the plant. Most of this can be removed by sidestream Anammox, reducing the nitrogen load on the main treatment train. This would increase the amount of COD that can be recovered in the biosorption A-stage as less denitrification is required to achieve the required effluent quality, reducing the energy footprint of the WRP.

In addition, UPWRP is an old treatment plant and the dewatering centrifuges are not functioning well, causing a small amount of digested sludge to be returned to the headworks with the dewatering centrate. This is the

main cause of sedimentation in the UPWRP equalisation tank and the daily TSS peaks in the IVP influent. As a result, the influent COD entering the IVP would also be higher than in the typical domestic wastewater in Singapore. Indeed, the average tCOD value of 760 mg/l is almost double of the wastewater received by KWRP (about 400 mg/l), although the influent to KWRP is expected to include a higher proportion of industrial wastewater due to the numerous light industries in the vicinity. Plans are in place to divert the dewatering centrate away from the equalisation tank at the UPWRP and further studies can then be performed to investigate the treatment performance of the IVP without influence of the dewatering centrate and the daily TSS peaks.

The next factor that needs to be considered is the size of the treatment plant. The domestic wastewater treatment train of TWRP has a designed capacity of 600,000 m<sup>3</sup>/d, 600 times larger than the IVP. As a result, there may be certain issues to consider when scaling up the plant. For instance, full-scale plants are found to respond slower than pilot-scale plants due to a larger inertia (Henze et al., 2008). Kraume, Wedi, Schaller, Iversen, and Drews (2009) also found the membrane fouling results obtained from pilot scale often differs considerably from full-scale plants. A demonstration plant with a similar layout to the IVP is scheduled to be completed at the end of 2016. With a capacity of 20,000 m<sup>3</sup>/day, this plant can shed more light into the challenges that may be encountered in scaling up the treatment processes.

This report provides a method for plant optimisation in Singapore using Biowin modelling. The calibrated parameters can be fine-tuned as more experiments are conducted and more data is available, for example when the demonstration plant is completed or when TWRP is operational. Optimisation can then be performed using the outlined method to identify configurations and operating conditions that can be implemented to reduce the energy footprint.

## 7 Conclusion

A Biowin model was developed to simulate the biological wastewater treatment process at the IVP. For the A-stage biosorption process, the model was calibrated with data from the 'Energy+' project at the Kranji WRP. For the B-stage MBR process, calibration was done using jar tests and OUR tests, and the results verified by profile studies carried out when the MBR was at pseudo steady state. The calibrated values are close to those reported in literature, and simulation results from the calibrated model agree well with the measured data from the profile studies.

The calibrated model was first used to optimise the MBR by determining the configuration and operating conditions that requires the least COD in the MBR influent for adequate nutrient removal (in terms of COD, nitrogen and phosphorus) in the B-stage. A number of configurations were simulated, both limited and not limited to the current pipe layout in the MBR. Several operating parameters, such as the RAS flow and DO in the aerated tanks, were also considered. It was found that the best-performing configuration is the AOA configuration with a step feed, where 50% of the MBR influent enters the first zone and 50% enters the second anoxic zone. The optimal operating conditions are a RAS flow of 0.73Q and a DO of 1 mg/l in the aerated tanks.

After obtaining the optimal MBR configuration, the A-stage biosorption process was added to the Biowin model. With a biosorption tank that is functioning as designed and the optimal MBR configuration, it was found that there was inadequate phosphorus removal in the MBR due to insufficient COD entering the B-stage. Increasing the primary sludge wastage would be the best option under steady state to achieve adequate nutrient removal while maximising COD recovery in the waste sludge. On the other hand, altering the bypass ratio of the biosorption tank would be the best option to respond to sudden increases in the nitrogen and phosphorus load in the raw influent as it has the shortest response time.

The calibrated MBR model was then used to investigate the effect of DO in the deoxygenation tank on the biological treatment process in the MBR. It was found that the deoxygenation tank in the IVP was adequately sized for DO removal in the mixed liquor, but it is not working as designed due to cascade overflows introducing large amounts of oxygen into the tank. With a functioning deoxygenation tank, the low DO in the RAS would lead to low COD consumption by heterotrophs in the anoxic tank, increasing the availability of COD for PAOs and hence improving the Bio-P performance of the MBR. In addition, a functioning deoxygenation tank would also have conditions of high nitrate, low rbCOD, and low DO, providing a selection pressure for DPAOs and possibly further promoting Bio-P removal in the MBR.

Next, various configurations were simulated to investigate the possibility of mainstream Anammox in Singapore. Both the CANON configuration using granular sludge and the DEMON configuration using alternate aerated and unaerated tanks could be suitable for local conditions, although the efficiency of the A-stage in removing COD would have to be increased before mainstream Anammox can occur using these configurations. Additional research is required to determine if it is possible or feasible to obtain the required removal efficiency under local conditions.

Lastly, it was determined that the operating conditions of the MBR could be modified to reduce chemical dosage in the downstream RO process. Firstly, the ammonia concentration in the effluent could be increased by decreasing nitrification in the MBR. This would reduce ammonia dosage required in the RO pretreatment to produce chloramine. However, doing so would increase the required sodium hypochlorite dosage to oxidise the higher concentration of nitrite in the effluent. Next, the amount of anti-scalants dosed in the RO pretreatment process could also be reduced with a low effluent phosphate concentration due to good Bio-P removal. However, it cannot be totally eliminated as calcium phosphate will still precipitate on the RO membranes due to the high calcium concentration.



## 8 References

- Ahn, Y.-H. (2006). Sustainable nitrogen elimination biotechnologies: a review. *Process Biochemistry*, 41(8), 1709-1721.
- Antonelli, M., Bialek, K., Teli, A., Citterio, S., & Malpei, F. (2011). Influence of thermal extraction of extracellular polymeric substances on cell integrity in activated sludge and membrane bioreactor samples. *Water Environment Research*, 83(2), 100-106.
- APHA, AWWA, & WEF. (1999). *Standard methods for the examination of water and wastewater* (20th ed.). Washington DC: American Public Health Association.
- Barnard, J., Houweling, D., & Steichen, M. (2011). Fermentation of mixed liquor for Removal and Recovery Phosphorus. *Proceedings of the Water Environment Federation*, 2011(1), 1-17.
- Bassin, J., Pronk, M., Kraan, R., Kleerebezem, R., & van Loosdrecht, M. (2011). Ammonium adsorption in aerobic granular sludge, activated sludge and anammox granules. *Water research*, 45(16), 5257-5265.
- Batstone, D. J., Keller, J., Angelidaki, I., Kalyuzhnyi, S., Pavlostathis, S., Rozzi, A., . . . Vavilin, V. (2002). The IWA Anaerobic Digestion Model No 1(ADM 1). *Water Science & Technology*, 45(10), 65-73.
- Beer, M., Stratton, H. M., Griffiths, P., & Seviour, R. J. (2006). Which are the polyphosphate accumulating organisms in full - scale activated sludge enhanced biological phosphate removal systems in Australia? *Journal of applied microbiology*, 100(2), 233-243.
- Beven, K., & Binley, A. (1992). The future of distributed models: Model calibration and uncertainty prediction. *Hydrological processes*, 6(3), 279-298.
- Bryne, W. (1995). *Reverse Osmosis: A practical guide for industrial users*. USA: Tall Oaks Publishing.
- Cao, Y. S., Wah, Y. L., Ang, C. M., & Raajeevan, K. S. (2008). *Biological Nitrogen Removal Activated Sludge Process in Warm Climates*. London, UK: IWA Publishing.
- Daigger, G. T., & Nolasco, D. (1995). Evaluation and design of full-scale wastewater treatment plants using biological process models. *Water Science and Technology*, 31(2), 245-255.
- De Kreuk, M., Picioreanu, C., Hosseini, M., Xavier, J., & van Loosdrecht, M. (2007). Kinetic model of a granular sludge SBR: influences on nutrient removal. *Biotechnology and bioengineering*, 97(4), 801-815.
- EnviroSim Associates Ltd. *BioWin Help Manual*.
- Fan, F., Zhou, H., & Husain, H. (2006). Identification of wastewater sludge characteristics to predict critical flux for membrane bioreactor processes. *Water research*, 40(2), 205-212.
- Fei, C. (2012). Cold CANON: Anammox at low temperatures. The Netherlands: Delft University of Technology.
- Filipe, C. D. M., Daigger, G. T., & Grady, C. P. L. (2001). A metabolic model for acetate uptake under anaerobic conditions by glycogen accumulating organisms: Stoichiometry, kinetics, and the effect of pH. *Biotechnology and bioengineering*, 76(1), 17-31. doi: 10.1002/bit.1022
- Foley, J., De Haas, D., Yuan, Z., & Lant, P. (2010). Nitrous oxide generation in full-scale biological nutrient removal wastewater treatment plants. *Water research*, 44(3), 831-844.
- Grunditz, C., & Dalhammar, G. (2001). Development of nitrification inhibition assays using pure cultures of Nitrosomonas and Nitrobacter. *Water research*, 35(2), 433-440.
- Gujer, W., Henze, M., Mino, T., Matsuo, T., Wentzel, M., & Marais, G. (1995). The activated sludge model No. 2: biological phosphorus removal. *Water Science and Technology*, 31(2), 1-11.
- Gujer, W., Henze, M., Mino, T., & van Loosdrecht, M. (1999). Activated sludge model no. 3. *Water Science and Technology*, 39(1), 183-193.
- Hao, X., Heijnen, J. J., & van Loosdrecht, M. C. M. (2002). Sensitivity analysis of a biofilm model describing a one-stage completely autotrophic nitrogen removal (CANON) process. *Biotechnology and bioengineering*, 77(3), 266-277. doi: 10.1002/bit.10105
- Henze, M., Grady, C., Gujer, W., Marais, G., & Matsuo, T. (1987). Activated Sludge Model No. 1: IAWPRC Scientific and Technical Report No. 1. *IAWPRC, London*.
- Henze, M., Gujer, W., Mino, T., Matsuo, T., Wentzel, M. C., vR Marais, G., & van Loosdrecht, M. C. (1999). Activated sludge model no. 2d, ASM2d. *Water Science and Technology*, 39(1), 165-182.
- Henze, M., van Loosdrecht, M., Ekama, G., & Brdjanovic, D. (2008). *Biological Wastewater Treatment: Principles, Modelling and Design*. London: IWA Publishing.
- Jaroszynski, L., & Oleszkiewicz, J. (2011). Autotrophic ammonium removal from reject water: partial nitrification and anammox in one - reactor versus two - reactor systems. *Environmental technology*, 32(3), 289-294.
- Jetten, M. S., Wagner, M., Fuerst, J., van Loosdrecht, M., Kuenen, G., & Strous, M. (2001). Microbiology and application of the anaerobic ammonium oxidation ('anammox') process. *Current opinion in biotechnology*, 12(3), 283-288.
- Jördening, H.-J., & Winter, J. (2005). *Environmental Biotechnology: Concepts and Applications*. Germany: Wiley-VCH.
- Kampschreur, M. J., Temmink, H., Kleerebezem, R., Jetten, M. S., & van Loosdrecht, M. C. (2009). Nitrous oxide emission during wastewater treatment. *Water research*, 43(17), 4093-4103.
- Kappeler, J., & Gujer, W. (1992). Estimation of kinetic parameters of heterotrophic biomass under aerobic conditions and characterization of wastewater for activated sludge modelling. *Water Science & Technology*, 25(6), 125-139.
- Kimura, K., Yamato, N., Yamamura, H., & Watanabe, Y. (2005). Membrane fouling in pilot-scale membrane bioreactors (MBRs) treating municipal wastewater. *Environmental science & technology*, 39(16), 6293-6299.

- Kong, Y., Nielsen, J. L., & Nielsen, P. H. (2005). Identity and ecophysiology of uncultured actinobacterial polyphosphate-accumulating organisms in full-scale enhanced biological phosphorus removal plants. *Applied and environmental microbiology*, 71(7), 4076-4085.
- Kong, Y., Xia, Y., Nielsen, J. L., & Nielsen, P. H. (2006). Ecophysiology of a group of uncultured Gammaproteobacterial glycogen - accumulating organisms in full - scale enhanced biological phosphorus removal wastewater treatment plants. *Environmental microbiology*, 8(3), 479-489.
- Kraume, M., Wedi, D., Schaller, J., Iversen, V., & Drews, A. (2009). Fouling in MBR: what use are lab investigations for full scale operation? *Desalination*, 236(1), 94-103.
- Kristiansen, R., Nguyen, H. T. T., Saunders, A. M., Nielsen, J. L., Wimmer, R., Le, V. Q., . . . Calteau, A. (2013). A metabolic model for members of the genus *Tetrasphaera* involved in enhanced biological phosphorus removal. *The ISME journal*, 7(3), 543-554.
- Kucera, J. (2015). *Reverse Osmosis: Design, Processes, and Applications for Engineers*. USA: John Wiley & Sons.
- Le-Clech, P., Chen, V., & Fane, T. A. (2006). Fouling in membrane bioreactors used in wastewater treatment. *Journal of Membrane Science*, 284(1), 17-53.
- Lee, Y., Tao, G., Lay, W., Lin, L., Kwok, Y. L., & Koh, K. (2013). *Towards Energy Self-Sufficient Water Reclamation Plants*. Singapore: PUB.
- Lesjean, B., Rosenberger, S., Laabs, C., Jekel, M., Gnirss, R., & Amy, G. (2005). Correlation between membrane fouling and soluble/colloidal organic substances in membrane bioreactors for municipal wastewater treatment. *Water Science & Technology*, 51(6-7), 1-8.
- López-Vázquez, C. M., Hooijmans, C. M., Brdjanovic, D., Gijzen, H. J., & van Loosdrecht, M. (2008). Factors affecting the microbial populations at full-scale enhanced biological phosphorus removal (EBPR) wastewater treatment plants in The Netherlands. *Water research*, 42(10), 2349-2360.
- Lopez, C., Pons, M., & Morgenroth, E. (2006). Endogenous processes during long-term starvation in activated sludge performing enhanced biological phosphorus removal. *Water research*, 40(8), 1519-1530.
- Lowry, O. H., Rosebrough, N. J., Farr, A. L., & Randall, R. J. (1951). Protein measurement with the Folin phenol reagent. *J. Biol Chem*, 193(1), 265-275.
- Lu, H., Keller, J., & Yuan, Z. (2007). Endogenous metabolism of *Candidatus Accumulibacter phosphatis* under various starvation conditions. *Water research*, 41(20), 4646-4656.
- Malaeb, L., & Ayoub, G. M. (2011). Reverse osmosis technology for water treatment: state of the art review. *Desalination*, 267(1), 1-8.
- Martins, A., Heijnen, J., & van Loosdrecht, M. (2003). Effect of dissolved oxygen concentration on sludge settleability. *Applied microbiology and biotechnology*, 62(5-6), 586-593.
- Mota, C., Head, M. A., Ridenoure, J. A., Cheng, J. J., & Francis, L. (2005). Effects of aeration cycles on nitrifying bacterial populations and nitrogen removal in intermittently aerated reactors. *Applied and environmental microbiology*, 71(12), 8565-8572.
- Mueller, J., Boyle, W. C., & Popel, H. J. (2002). *Aeration: Principles and Practice*. USA: Taylor & Francis Group.
- Mulkerrins, D., Dobson, A., & Collieran, E. (2004). Parameters affecting biological phosphate removal from wastewaters. *Environment International*, 30(2), 249-259.
- Nielsen, P. H. (1996). Adsorption of ammonium to activated sludge. *Water research*, 30(3), 762-764.
- Oehmen, A., Carvalho, G., Lopez-Vazquez, C., van Loosdrecht, M., & Reis, M. (2010). Incorporating microbial ecology into the metabolic modelling of polyphosphate accumulating organisms and glycogen accumulating organisms. *Water research*, 44(17), 4992-5004.
- Oehmen, A., Lemos, P. C., Carvalho, G., Yuan, Z., Keller, J., Blackall, L. L., & Reis, M. A. (2007). Advances in enhanced biological phosphorus removal: from micro to macro scale. *Water research*, 41(11), 2271-2300.
- Oehmen, A., Saunders, A. M., Vives, M. T., Yuan, Z., & Keller, J. (2006). Competition between polyphosphate and glycogen accumulating organisms in enhanced biological phosphorus removal systems with acetate and propionate as carbon sources. *Journal of Biotechnology*, 123(1), 22-32.
- Pomory, C. M. (2008). Color development time of the Lowry protein assay. *Analytical biochemistry*, 378(2), 216-217.
- Rietveld, L. (2013). *Fundamentals of water treatment: Mass transport & aeration*.
- Singh, V. P., & Frevert, D. K. (2002). *Mathematical Models of Large Watershed Hydrology*. USA: Water Resources Publications.
- Solomon, S. (2007). *Climate change 2007-the physical science basis: Working group I contribution to the fourth assessment report of the IPCC* (Vol. 4): Cambridge University Press.
- Spellman, F. R. (2009). *Handbook of Water and Wastewater Treatment Plant Operations*. USA: CRC Press.
- Suneethi, S., Sri Shalini, S., & Joseph, K. (2014). State of The Art Strategies for Successful ANAMMOX Startup and Development: A Review. *Int J Waste Resources*, 4(168), 2.
- Tallec, G., Garnier, J., Billen, G., & Gossailles, M. (2006). Nitrous oxide emissions from secondary activated sludge in nitrifying conditions of urban wastewater treatment plants: effect of oxygenation level. *Water research*, 40(15), 2972-2980.
- Tchobanoglous, G., Burton, F. L., & Stensel, H. D. (2003). *Wastewater Engineering: Treatment and Reuse* (4th ed.): McGraw Hill.
- TU Delft. (2014). *Water treatment: Aeration and gas stripping*.

- van Dongen, U., Jetten, M., & van Loosdrecht, M. (2001). The SHARON-Anammox process for treatment of ammonium rich wastewater. *Water Science & Technology*(44), 153-160.
- van Haandel, A. C., & van der Lubbe, J. G. M. (2012). *Handbook of Biological Wastewater Treatment* (2nd ed.). Great Britain: IWA Publishing.
- van Kempen, R., Mulder, J., Uijterlinde, C., & van Loosdrecht, M. (2001). Overview: full scale experience of the SHARON® process for treatment of rejection water of digested sludge dewatering. *Water Science and Technology*, 44(1), 145-152.
- van Loosdrecht, M., & Salem, S. (2006). Biological treatment of sludge digester liquids. *Water Science & Technology*, 53(12), 11-20.
- Verdenius, F., & Broeze, J. (1999). Generalised and instance-specific modelling for biological systems. *Environmental modelling & software*, 14(5), 339-348.
- Waki, M., Tokutomi, T., Yokoyama, H., & Tanaka, Y. (2007). Nitrogen removal from animal waste treatment water by anammox enrichment. *Bioresource technology*, 98(14), 2775-2780.
- Wett, B., Nyhuis, G., Takács, I., & Murthy, S. (2010). Development of enhanced deammonification selector. *Proceedings of the Water Environment Federation*, 2010(10), 5917-5926.
- Wilén, B.-M., & Balmér, P. (1999). The effect of dissolved oxygen concentration on the structure, size and size distribution of activated sludge flocs. *Water research*, 33(2), 391-400.
- Wisniewski, C., & Grasmick, A. (1998). Floc size distribution in a membrane bioreactor and consequences for membrane fouling. *Colloids and Surfaces A: Physicochemical and Engineering Aspects*, 138(2), 403-411.
- Woods, L., Riccobono, M., Mehan, N., Hestekin, J., & Beitle, R. (2011). Synergistic effect of abrasive and sonication for release of carbohydrate and protein from algae. *Separation Science and Technology*, 46(4), 601-604.
- Xu, X., Xue, Y., Wang, D., Wang, G., & Yang, F. (2014). The development of a reverse anammox sequencing partial nitrification process for simultaneous nitrogen and COD removal from wastewater. *Bioresource technology*, 155, 427-431.
- Zeng, R. J., Saunders, A. M., Yuan, Z., Blackall, L. L., & Keller, J. (2003). Identification and comparison of aerobic and denitrifying polyphosphate - accumulating organisms. *Biotechnology and bioengineering*, 83(2), 140-148.
- Zengin, G. E., Artan, N., Orhon, D., Satoh, H., & Mino, T. (2011). Effect of aspartate and glutamate on the fate of enhanced biological phosphorus removal process and microbial community structure. *Bioresource technology*, 102(2), 894-903.
- Zhang, H., Hashmi, M. Z., Zhang, J., & Zhang, M. (2014). Ammonia Nitrogen Removal by Single-stage Process: A Review. *J. Chem. Soc. Pak*, 36(4), 775.
- Zhang, L., Zheng, P., Tang, C.-j., & Ren-cun, J. (2008). Anaerobic ammonium oxidation for treatment of ammonium-rich wastewaters. *Journal of Zhejiang University Science B*, 9(5), 416-426.
- Zheng, X., Ernst, M., & Jekel, M. (2009). Identification and quantification of major organic foulants in treated domestic wastewater affecting filterability in dead-end ultrafiltration. *Water research*, 43(1), 238-244.

## Appendix A: Membrane fouling

Membrane fouling is one of the major issues facing the operation of MBRs as it reduces flux through the membranes and necessitate recovery cleaning. The most common method to minimise membrane fouling is the energy-intensive process of increasing aeration to scour the membranes. Since the IVP aims to minimise energy use, investigations were also carried out to determine how operational changes influence factors that affect membrane fouling. This may allow identification of operational changes that reduces aeration energy requirements without compromising on the frequency of membrane cleaning.

### A1. Literature review

Membrane fouling is one of the major issues facing the operation of MBRs. It occurs when there is a deposition of particles on the membrane surface, forming a gel or cake layer that partially blocks the membrane pores and cause an increase in the hydraulic resistance. Unfortunately, this cannot be avoided and MBRs need to be properly designed and operated to minimise its effect. For example, the influent is often pre-treated to remove the larger solid particles and the membranes are scoured using aeration to physically dislodge the deposited cake layer. Once fouled, the membranes can be partially regenerated through maintenance cleaning such as relaxation and backflushing, and through recovery cleaning using chemicals.

#### Physicochemical factors

There are many factors that affect membrane fouling, and many of them are related to one another. One major physicochemical factor has been determined to be the presence of soluble microbial products (SMP), which are cell components released during cell lysis, diffuse through cell membrane, lost during synthesis, or excreted by cells. It consists of many compounds such as polysaccharides, proteins and humic substances. Various studies have concluded that SMP contributes to up to 81% of the membrane fouling. They are first adsorbed to the membrane inside the pores and partially block them, then subsequently form a gel layer on the membrane surface and cause even higher hydraulic resistance. The biopolymer content in SMP, including polysaccharides and proteins, is found to be mostly responsible for membrane fouling, with the former playing a larger role. Humic substances and other lower molecular weight substances do not contribute much to fouling due to their smaller size that allows them to pass through the membrane pores. (Le-Clech et al., 2006; Lesjean et al., 2005; Zheng, Ernst, & Jekel, 2009)

Another physicochemical factor in membrane fouling is the extracellular polymeric substances (EPS), which are large polymeric molecules generated by the cells, surrounding them to form a protective layer and facilitate their aggregation into flocs. Similar to SMP, EPS is made up of many compounds, including polysaccharides and proteins, and high EPS contributes significantly to membrane fouling. However, very low levels of EPS are also associated with increased membrane fouling, due to poor floc formation in the mixed liquor resulting in the presence of more non-aggregated colloids. Unlike SMP, EPS tends to contribute to membrane fouling through the formation of a cake layer on the membrane surface. As backwashing and scouring by coarse air aeration

are effective at dislodging the cake layer, EPS plays a smaller role than SMP in the fouling of MBR membranes. (Le-Clech et al., 2006)

Particle sizes also play a role in membrane fouling. Wisniewski and Grasmick (1998) found that soluble compounds contribute to half of the total membrane resistance, and smaller particles from floc breakage contribute more to fouling compared to larger settleable solids. In particular, colloidal particles, measured in terms of total organic carbon (TOC), have been found to be positively correlated to both SMP concentrations and membrane fouling (Fan, Zhou, & Husain, 2006).

Other physicochemical factors include the flux of the membrane. The critical membrane flux is the maximum flux value that can be maintained in the short term at constant TMP. At fluxes above the critical value, the membranes undergo rapid fouling and would require frequent cleaning (Le-Clech et al., 2006). MBRs operate at sub-critical fluxes where membrane fouling occurs much slower, and the fouling propensity has been found to decrease with decreasing flux. Membrane fouling in an MBR can be well managed with an appropriate cleaning regime and flux control.

### Operational factors

The physicochemical factors mentioned above are influenced by a wide range of operational factors. One of these is the SRT of the MBR system, which affects the mixed liquor suspended solids (MLSS) concentration and the F/M ratio in the system. At very high SRT values, the high MLSS concentration results in a viscous mixed liquor, causing significant filtration resistance and a reduction in flux. On the other hand, low SRTs leads to high F/M ratios and cause higher fouling too, most likely due to an increase in EPS and a change in the nature of foulants (Le-Clech et al., 2006). The foulants are predominantly carbohydrate, but more proteinaceous foulants have been found in reactors with a higher F/M ratio (Kimura, Yamato, Yamamura, & Watanabe, 2005).

Another operational factor is the aeration rate and the dissolved oxygen (DO) concentration in the mixed liquor. Generally, a lower DO leads to a higher fouling rate. Significant differences could be found in the microbial communities and this could have led to the smaller particle sizes in the MBR at low DO concentrations. Furthermore, floc deterioration was also observed at low DO conditions due to a lowering of cell surface hydrophobicity. On the other hand, a higher aeration rate in the membrane tank up to an optimal point have been generally found to decrease fouling, beyond which there is no effect on the fouling rate. Besides increasing DO, the air bubbles also scour membranes and minimise the deposition of a cake layer. However, a breakup of flocs may occur when the aeration is too intense, leading to smaller particle sizes and a release of EPS into the MBR. The air bubbles may also become trapped within the lumens of hollow fibre membranes, reducing the filtration rate. (Le-Clech et al., 2006)

Other operational factors that can influence membrane fouling include the configuration of the treatment process. A long term study of two parallel MBRs with different configurations found a higher fouling rate in the pre-denitrification reactor compared to the post-denitrification reactor, although no conclusion is drawn whether this difference in fouling can be attributed to the different configurations. (Lesjean et al., 2005)

## A2. Methods and materials

From December 2014 to March 2015, the effect of operational changes on membrane fouling was investigated. The SCADA program monitors the system performance, including the TMP across the membranes. In addition, the following parameters in the MBT 1 and MBT 2 tanks were also monitored weekly:

- EPS and SMP in terms of protein and polysaccharide concentrations
- Organic compounds in SMP using a LCOCD-OND-UVD

### SMP and EPS Extraction

The heat extraction method is used to extract SMP and EPS from the sludge for analysis. Similar to literature (Antonelli, Bialek, Teli, Citterio, & Malpei, 2011), SMP was obtained by centrifuging the sludge sample at 10,000 g for 10 min and filtering the supernatant through a 0.45 µm membrane filter. The sludge pellet was then resuspended in Milli-Q water before heating in a water bath at 80°C for 10 min. The resultant solution was then centrifuged again at 10,000 g for 10 min and the supernatant filtered through a 0.45 µm membrane filter to obtain the EPS.

### Carbohydrate analysis

Carbohydrates were analysed using a similar method to the one used by Woods, Riccobono, Mehan, Hestekin, and Beitle (2011). 1 ml of 5 wt% phenol and 5 ml of concentrated (98 wt%) sulphuric acid were added to each 2 ml sample of SMP and EPS and mixed. After leaving to stand for 30 min, the polysaccharide concentrations were determined by measuring the absorbance at 490 nm using a spectrophotometer (Hach DR5000, USA). Glucose was used as a standard for the analysis of polysaccharides.

### Protein analysis

The protein concentration was analysed using a modified Lowry procedure. As there are many variations of the procedure, the method used in this study is described below (Lowry, Rosebrough, Farr, & Randall, 1951; Pomory, 2008), with Bovine serum albumin (BSA) as a standard.

1. Prepare stock solutions of the following:
  - a. Alkaline reagent – 0.1M NaOH, 2 wt% Na<sub>2</sub>CO<sub>3</sub>, 0.02 wt% sodium potassium tartrate, 1 wt% Sodium Dodecylsulfate
  - b. Copper reagent – 0.5 wt% CuSO<sub>4</sub>·5H<sub>2</sub>O
  - c. Folin-Ciocalteu Reagent – dilute to 1N
2. Add the Alkaline reagent and Copper reagent in a 25:1 ratio
3. Add 5 ml of the mixture (from step 1) to 1 ml of sample
4. Mix well and leave to stand for 10 minutes
5. Add 0.5 ml of 1N Folin-Ciocalteu Reagent
6. Mix well and leave to stand for 30 minutes
7. Measure absorbance at 650 nm

## LCOCD-OND-UVD

In addition to analysing for protein and carbohydrate, the SMP samples were also examined with liquid chromatography coupled with organic carbon, organic nitrogen, and ultraviolet detectors (DOC-Labor LC-OCD Model 8, Germany). The ChromCALC software (DOC-Labor, Germany) was used to analyse the raw data.

### A3. Results and discussion

Investigations into membrane fouling were conducted from December 2014 to March 2015. During this period, the MBR initially operated in an MLE configuration with 3d SRT, then changed to an AOA configuration with 5d SRT. Figure A1 and A2 below shows a comparison of the SMP and EPS results. The subscripts 'c' and 'p' refers to carbohydrates and proteins respectively, and the error bars denote the standard deviation. Figure A3 shows the LCOCD analysis of SMP, which breaks down SMP into its components of macromolecular biopolymers, humic acids, and low molecular weight compounds.

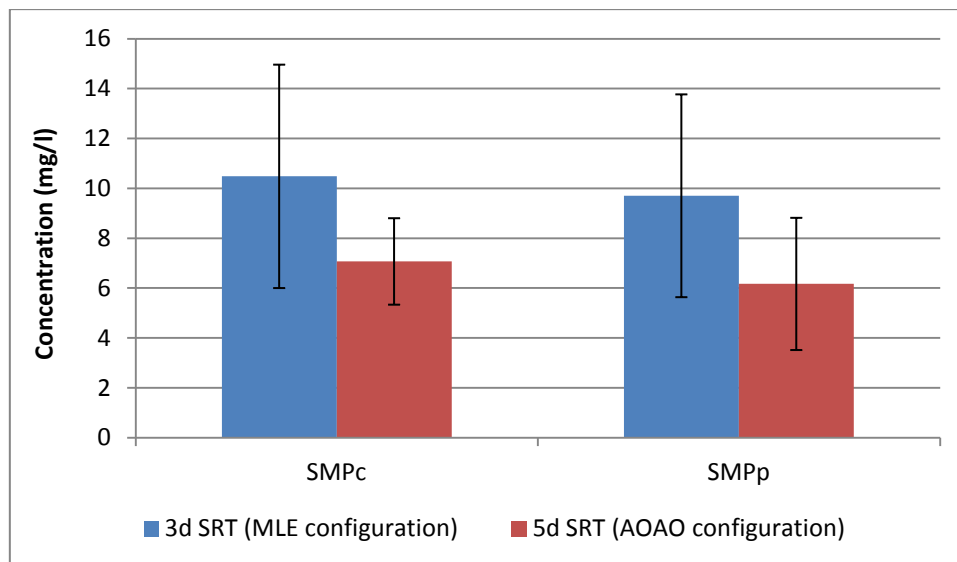


Figure A1: Comparison of SMP concentrations

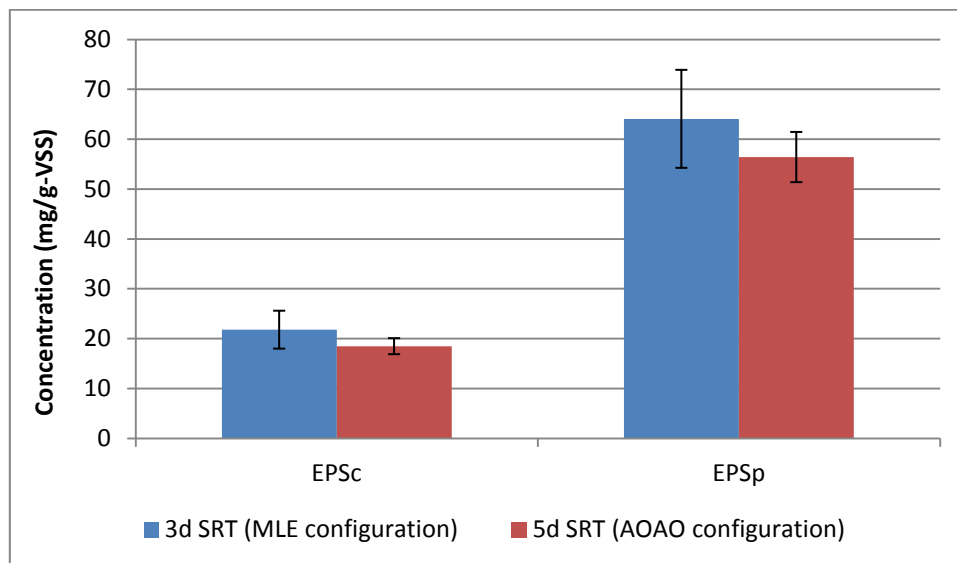


Figure A2: Comparison of EPS concentrations

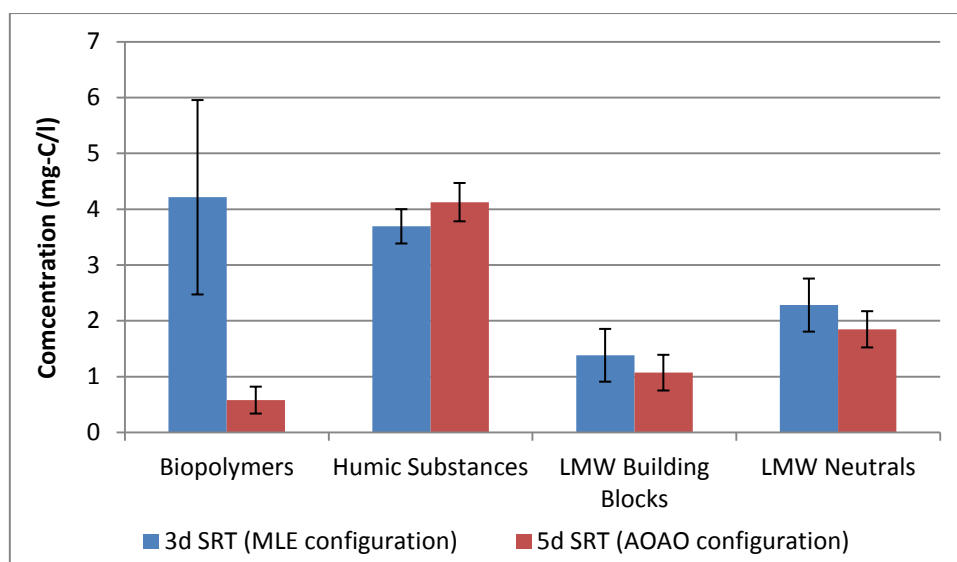


Figure A3: LCOCD analysis of SMP

Figures A1 and A2 seem to indicate that the SMP and EPS concentrations in the mixed liquor decreased with increasing SRT. However, this difference is not significant due to the errors involved. On the other hand, a breakdown of the SMP in Figure A3 shows that the biopolymers in the SMP decreased considerably, although no conclusion could be drawn as to whether this change is due to the difference in SRT or change in MBR configuration.

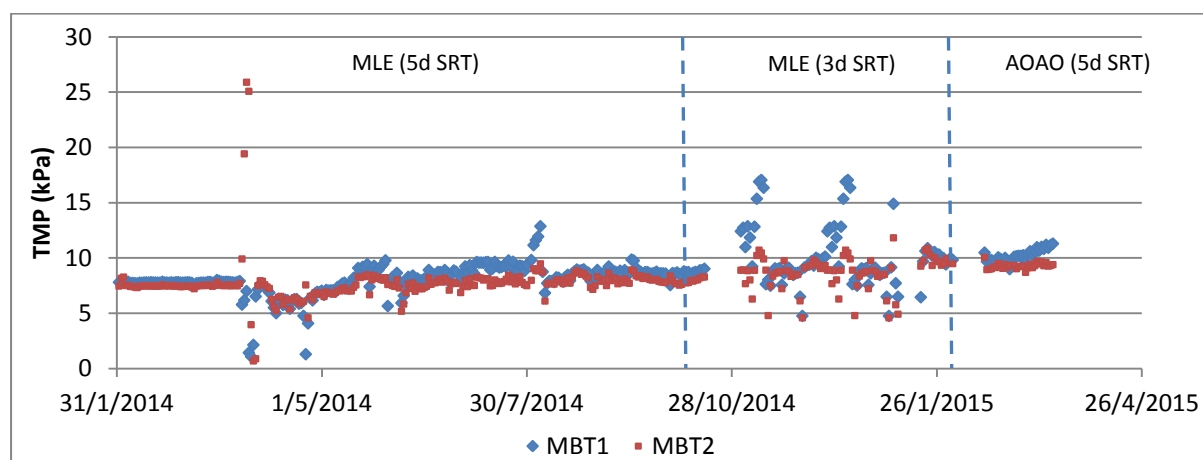


Figure A4: TMP trend

Figure A4 shows the TMP trend of the membranes over time, which can be used to indicate the extent of membrane fouling. A chemical clean of both membranes is conducted whenever there is a sharp increase in the TMP. Comparing the MLE (3d SRT) and AOA (5d SRT) configurations, it can be observed that the membranes fouled much more rapidly with the MLE configuration at 3d SRT.

This observation is consistent with the LCOCD analysis that the biopolymer concentration in the SMP is significantly higher in the MLE configuration than the AOA configuration. Lesjean et al. (2005) found through an LCOCD analysis that biopolymers, consisting of polysaccharides, proteins, and colloidal organic molecules, are retained by the membranes and are likely to be responsible for membrane fouling. Smaller molecules such



as humic acids and low molecular weight compounds pass through the membranes and play a much smaller role in fouling.

Figure A4 also shows that the membrane hardly fouled when it is operated in the MLE configuration with 5 day SRT. This suggests that the increase in fouling propensity is more likely due to the change in SRT than the change in configuration, although more studies will be required to verify this. In addition, it also shows that the TMP is generally higher, and the membranes foul faster, in MBT 1 compared to MBT 2. This is most likely due to higher aeration and scouring in MBT 2. The aeration rates of the individual membrane tanks are controlled by the highly imprecise method of looking at rotameters and adjusting manual valves. Indeed, the DO in MBT2 tends to be higher than in MBT1 (Figure A5), which supports the hypothesis that there is a higher aeration and scouring rate in MBT2.

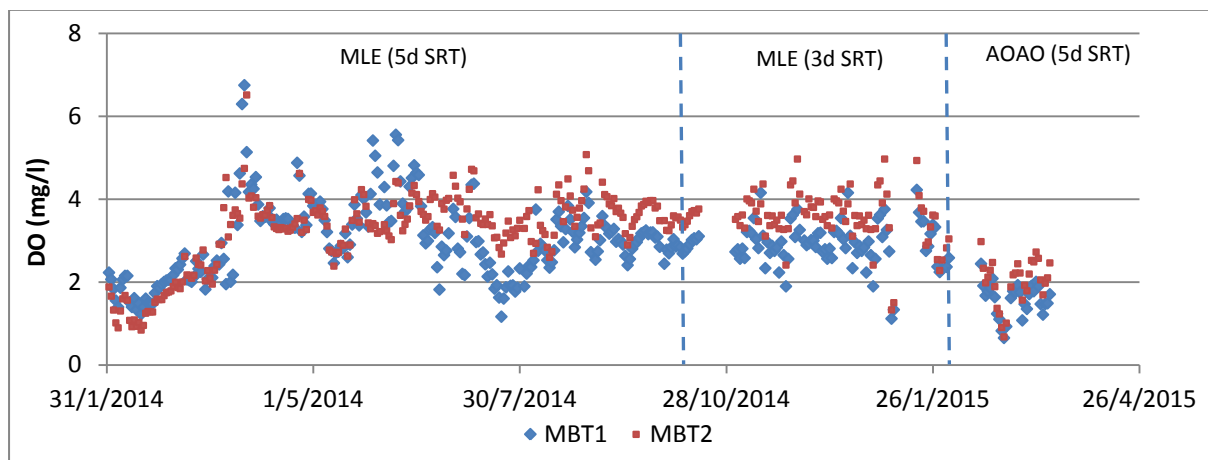


Figure A5: DO concentrations in membrane tanks

## A4. Conclusion

The IVP operated with the MLE configuration (3d SRT) and AOA configuration (5d SRT) during the sampling period from December 2014 to March 2015. It was found that the switch from the MLE to AOA configuration corresponded to a large decrease in the amount of biopolymers in the SMP. In addition, the membranes fouled less frequently with the change in operating conditions, indicating that biopolymers in the mixed liquor play a significant role in membrane fouling. No conclusion could be drawn as to whether the change in fouling propensity is due to the change in SRT or configuration, although the results seem to suggest the change in SRT as the main contributing factor.

## Appendix B: Influent sampling data

This section contains the raw data for the influent VFA and cation concentrations that are summarised in Tables 3 and 4.

**Table B1: Measured VFA data for raw influent**

	2-Feb	5-Feb	23-Feb	27-Feb	5-Mar	9-Mar	16-Mar
Acetate (C2)	26.81	83.99	52.05	11.13	29.26	32.64	20.45
Propionate (C3)	1.33	5.99	3.07	0.25	1.71	4.37	1.74
Butyrate (C4)	0.77	1.69	1.04	0.18	0.57	0.63	0.32
Valerate (C5)	1.27	1.90	1.62	0.45	0.72	0.92	0.54

**Table B2: Measured VFA data for MBR influent**

	26-Dec	29-Dec	2-Feb	5-Feb	23-Feb	27-Feb	5-Mar	9-Mar	16-Mar
Acetate (C2)	15.98	4.91	3.97	37.68	48.02	1.81	19.12	9.57	5.36
Propionate (C3)	1.45	0.00	0.10	0.75	5.95	0.00	0.00	1.07	0.92
Butyrate (C4)	0.26	0.10	0.77	0.75	1.12	0.00	0.20	0.31	0.24
Valerate (C5)	0.54	0.17	0.74	1.11	1.49	0.27	0.76	0.66	0.23

**Table B3: Measured cation data for raw influent**

	2-Feb	5-Feb	6-Mar	9-Mar	16-Mar
Na	93.04	114.12	105.19	101.72	87.99
K	7.80	7.50	14.88	14.16	14.54
Mg	2.12	2.47	5.68	5.54	5.70
Ca	20.98	19.99	24.06	25.09	23.63

**Table B4: Measured cation data for MBR influent**

	8-Dec	16-Dec	19-Dec	2-Jan	6-Jan	2-Feb	5-Feb	6-Mar	9-Mar	16-Mar
Na	98.45	107	100.56	88.7	107.14	103.07	116.02	106.32	106.46	89.3
K	15.23	17.15	18.65	9.87	10.46	7.38	7.34	15.1	14.28	14.08
Mg	5.86	6.32	5.90	0.96	1.38	2.29	2.67	5.66	5.69	5.54
Ca	28.21	29.02	29.14	24.01	25.56	21.42	19.72	23.75	25.65	24.59

## Appendix C: Specifications for Biowin simulations

The following section summarise the specifications and operating conditions used in Biowin for simulating of the profile studies in section 4.1.6 and mainstream Anammox in sections 5.2 and 6.

### C1. Profile studies for model verification

Table C1: MBR operating conditions for profile studies

	23 Sep 2014 (MLE 3d SRT)	10 Jun 2014 (MLE, 5d SRT)	24 Jun 2014 (MLE, 5d SRT)	23 Mar 2015 (AOAO, 5d SRT)
DO in Swing 1 tank (mg/l)	2.00	2.30	2.50	1.80
DO in Swing 2 tank (mg/l)	1.80	1.00	0.90	Unaerated
DO in aerated tank (mg/l)	1.65	1.40	0.85	1.55
Influent flow (m <sup>3</sup> /h)	35.2	36.6	36.6	36.6
RAS flow (m <sup>3</sup> /h)	81	82	75	81
WAS flow (m <sup>3</sup> /h)	2.21	1.63	1.63	1.63

### C2. Mainstream Anammox

#### *MBR influent characteristics (after A-stage treatment):*

tCOD: 88.87 mg/l  
sCOD: 61.11 mg/l  
rbCOD: 4.72 mg/l  
VFA: 1.15 mg/l  
TN: 26.96 mg/l  
TP: 3.96 mg/l

#### *SHARON-Anammox:*

Volume of SHARON tank: 780 m<sup>3</sup>  
HRT of SHARON tank: 19.1 hours  
DO in SHARON tank: 1 mg/l  
Volume of Anammox tank: 80 m<sup>3</sup>  
HRT of Anammox tank: 1.96 hours  
RAS flow: 979 m<sup>3</sup>/d (1Q)  
WAS flow: 1.17 m<sup>3</sup>/d  
SRT of SHARON tank: 19.1 hours  
SRT of Anammox tank: 60 days

#### *CANON:*

Volume of CANON tank: 120 m<sup>3</sup>  
HRT in CANON tank: 2.95 hours  
DO in CANON tank: 0.2 mg/l  
RAS flow: 979 m<sup>3</sup>/d (1Q)  
WAS flow: 1.00 m<sup>3</sup>/d  
SRT: 60 days

#### **DEMON:**

Volume of each unaerated DEMON tank:	10 m <sup>3</sup>
Volume of each aerated DEMON tank:	30 m <sup>3</sup>
DO in aerated DEMON tanks:	0.2 mg/l
RAS flow:	979 m <sup>3</sup> /d (1Q)
WAS flow:	1.47 m <sup>3</sup> /d
SRT:	60 days

#### **Anammox-MBR:**

Volume of Anammox tank:	120 m <sup>3</sup>
HRT in Anammox tank:	2.95 hours
RAS flow:	3907 m <sup>3</sup> /d (4Q)
WAS flow:	2.22 m <sup>3</sup> /d
SRT:	60 days

## Appendix D: Biosorption sensitivity analysis

This section contains the sensitivity analysis of COD, TKN, and TP concentrations in the MBR influent to changes in operating conditions of the biosorption process.

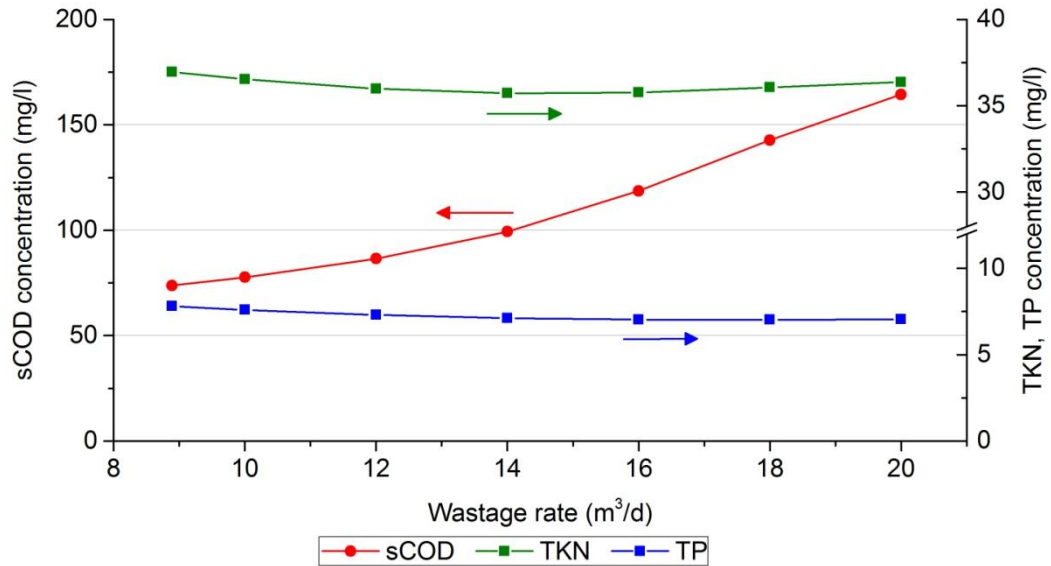


Figure D1: Sensitivity of sCOD, TKN and TP in MBR influent to primary sludge wastage rate

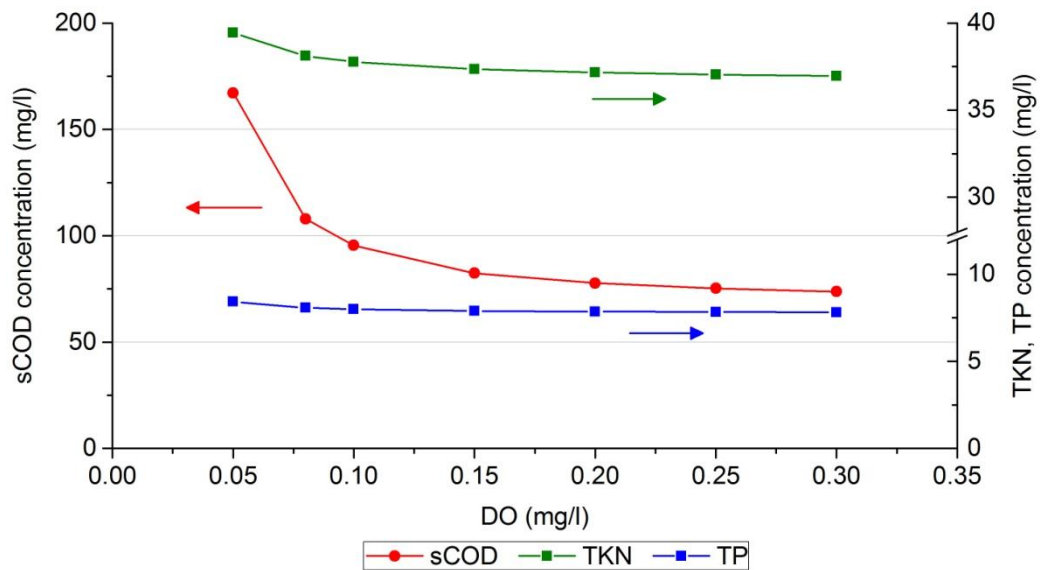


Figure D2: Sensitivity of sCOD, TKN and TP in MBR influent to DO in biosorption tank