Development of an evaluative framework promoting a paradigm shift towards resource recovery in wastewater treatment

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Abstract

Resource recovery from wastewater is a rapidly emerging research area promoting the development of several technologies that have the potential to increase the sustainability of process operations in wastewater treatment. With the implementation of appropriate treatment technologies, nutrients such as phosphorus and nitrogen can be recovered from nutrient rich sludge dewatering liquor in various forms to be used as agricultural fertilisers; portion of power demands at WWTPs can be offset by extracting energy from wastewater; the undesirable impact of wastewater treatment processes on air quality from emission of greenhouse gases, on land from sludge disposal and on downstream water bodies via effluent discharge can be mitigated.

The performance of WWTP design and control strategies can be evaluated, based on their impact on downstream water bodies and the associated net operational costs of each unit operation, by using the performance indices developed by the IWA BSM task group. With the impact of products generated from WRRFs on air quality and land, the value of resources recovered, and other operational costs being overlooked in the current evaluation system, it is unclear whether the fate of products generated from WRRFs would influence the design and operation of WWTPs. This dissertation addresses the hypothesis that a WWTP model evaluative framework that considers the fate of WWTP products and value of resources that can potentially be recovered from WRRFs will generate a useful tool for design and operation of WWTPs in South Africa and aid in their transition to future WRRFs.

To fill the gaps in the current evaluation system, it was important to extend the term “effluent” to be now inclusive of greenhouse gases that evolve from WWTPs and sludge disposed. As such, two new evaluative equations were formulated accordingly (EQI_{gas} and EQI_{sludge}) in addition to the existing effluent quality index now known as EQI_{water}. The latter has been modified such that the weighting assigned to each pollutant is a function of the region-specific or WWTP-specific pollutant concentration limits. This flexibility allows the EQI_{water} evaluation to be tailored towards the effluent quality goals specific to each plant or region. The EQI_{gas} is function of the greenhouse gasses that evolve from WWTPs with weighting factors set equal to their respective global warming potentials. The EQI_{sludge} was formulated based on the South African sludge disposal guidelines taking into consideration the stability, pollutant and microbiological classes. Additionally, the existing operational cost index was modified to include

fines for violation of pollutant limits in the effluent, the monetary benefits of nutrient recovery and dosing of additives such as lime or magnesium.

The PWM_SA by Ikumi *et al.* (2015a) was used to replicate Module 4 of Waterval WCW for the simulation of two scenarios: (1) without side-stream treatment and (2) with side-stream treatment including recycle of side-stream effluent to head of works. Due to model limitations, the effluent quality evaluation was limited to the EQI_{water}. On the other hand, fines for non-compliance to effluent regulations and cost recovered from the sale of struvite were not considered in the OCI evaluation since the determination of an appropriate future unit cost for each component in a South African context was outside of the scope of this project. From the simulated results, it was noted that Module 4 of Waterval WCW has sufficient capacity to treat the influent wastewater as well as the recycled sludge dewatering liquors generated from that module to acceptable quality with respect to the standards of the Waterval Water Use License (08/C22C/fg/646). Thus, the implementation of side-stream struvite precipitation in this case significantly increases the net operational costs for marginal benefits in terms of effluent quality.

The extended performance indices framework is currently being implemented, as an evaluative protocol, in the simulation of all considered strategic scenarios in order to determine the most impactful in future decision making for the given WRRF systems. However, in order for this evaluative framework to be more effective, the models will require modifications, such that the predictions made regarding system performance include the variables of significance, related to prioritised criteria for the products generated by the system.

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AAD	Anoxic-aerobic digestion
AD	Anaerobic digestion
AE	Aeration energy
AerD	Aerobic digestion
ANO	Autotrophic nitrifying organisms
AOO	Ammonia oxidising organisms
AS	Activated sludge
ASM2	Activated sludge model No. 2
BEPR	Biological excess phosphorus removal
BNR	Biological nutrient removal
BPO	Biodegradable particulate organics
BSM	Benchmark Simulation Modelling
BSO	Biodegradable soluble organics
C	Carbon
Ca	Calcium
$\mathbf{C_{limit}}$	Concentration limit of pollutant in effluent
$\mathbf{CH_4}$	Methane
CHP	Combined heat and power
$\mathbf{CO_2}$	Carbon dioxide
COD	Chemical oxygen demand
DAF	Dissolved air floatation
DO	Dissolved oxygen
DWS	Department of Water and Sanitation

EC	External carbon addition
EPA	United States Environmental Protection Agency
EQI	Effluent Quality Index
FBSO	Fermentable biodegradable soluble organics
FOc	Carbonaceous oxygen demand
FOHO	Facultative ordinary heterotrophic organisms
FOn	Nitrification oxygen demand
FOt	Total oxygen demand
FSA	Free and saline ammonia
GWP	Global Warming Potential
H	Hydrogen
HAc	Acetic acid
HE	Total heating energy required for anaerobic digestion of sludge
HPr	Propionic acid
HRT	Hydraulic retention time
ISS	Inorganic settleable solids
IWA	International Water Association
K	Potassium
KPI	Key Performance Indicator
K_{spm}'	Thermodynamic solubility product for struvite
LCA	Life Cycle Assessment
ME	Mixing energy
Mg	Magnesium
MLE	Modified Ludzack-Ettinger

MP	Energy from methane produced
N	Nitrogen
NDBEPR	Nitrification-denitrification biological excess phosphorus removal
N_2O	Nitrous oxide
NO	Nitrates and nitrites
NO_3	Nitrate
NOO	Nitrite oxidising organisms
NR	Nutrients recovered
O	Oxygen
OCI	Operational Cost Index
OHO	Ordinary heterotrophic organisms
OP	Ortho-phosphates
P	Phosphorus
PE	Pumping energy
PAO	Phosphorus accumulating organisms
p_{CO2}	Partial pressure of CO_2
PCR	Polymerase chain reaction
pH_2	Hydrogen partial pressure
PHA	Poly-ß-hydroxyalkanoates
PS	Primary sludge
PST	Primary settling tank
PWM_SA	Plant-Wide Model
Q_e	Effluent flow rate
RAS	Return activated sludge

RBCOD	Readily biodegradable COD
S	Sulphur
SBCOD	Slowly biodegradable COD
SCFA	Short chain fatty acids
SDG	Sustainable Development Goals
SHARON	Single reactor for High activity Ammonia Removal Over Nitrite
SOTE	Standard Oxygen Transfer Efficiency
SP	Sludge produced
SRC	Standardised Regression Coefficients
SRT	Solid retention time
SS	Steady state
SST	Secondary settling tank
TKN	Total Kjeldahl Nitrogen
TP	Total phosphorus
TSS	Total settleable solids
UCT	University of Cape Town
UCTADM	University of Cape Town anaerobic digestion model
UPO	Unbiodegradable particulate organics
USO	Unbiodegradable soluble organics
VFA	Volatile fatty acids
VSS	Volatile settleable solids
WAS	Waste activated sludge
WCW	Wastewater Care Works
WQE	Water Quality Engineering

WRG	Water Research Group
WRRF	Water and Resource Recovery Facility
WW	Wastewater
WWTP	Wastewater Treatment Plant

1. Introduction

1.1 Subject and motivation for report

The overarching aim of this research project is to develop performance indices that can be used to perform system-wide evaluations of wastewater treatment plants (WWTP) in South Africa. This goal will be achieved by modifying the existing evaluative performance indices adopted by the IWA (International Water Association) BSM (benchmark simulation modelling) task group to include the fate of WWTP products and potential recovery of resources, as a step towards transitioning from WWTP to water and resource recovery facilities (WRRF). This research project has been conducted in the Water Research Group (WRG) of the Department of Civil Engineering at the University of Cape Town (UCT) and has been supervised by A/Prof David S. Ikumi.

1.2 Background to research

Wastewater treatment forms an integral part of the urban water cycle. The objectives of wastewater treatment comprise the removal of excess organics and nutrients (nitrogen (N) and phosphorus (P)) from municipal and industrial wastewater such that an effluent of acceptable quality may be obtained. As a result, the harmful impacts of discharging wastewater effluent into water bodies, such as deoxygenation of water, eutrophication, loss of aquatic life and impact on human health via waterborne diseases, may be mitigated.

Driven by the need to mitigate undesirable environmental impacts of engineering and by global issues such as climate change, water scarcity and the fast depletion of the Earth's nutrient reserves, resource recovery from wastewater has become a rapidly emerging research area. It promotes the development of several technologies that hold the promise to increase the sustainability of process operation in wastewater treatment systems, and also promotes a circular economy through the reuse of wastewater-derived products. Different types of organic compounds with industrial relevance can be obtained through controlled fermentation or chain elongation processes. Nutrients (such as N and P) can be recovered in various forms for use in agricultural fertilizers; energy, in the form of heat and electricity, can be extracted from the

wastewater to offset a portion of the power demands of the treatment facility; and water can be reclaimed through treatment or reuse of WWTP effluents.

One of the by-products of the sludge treatment process at WWTPs, is the dewatering liquors that are high in ammonium and phosphate concentrations. With strict effluent regulations, many WWTPs recycle the dewatering liquor back to the head of works. As the return liquor is combined with the influent wastewater, the nutrient load on the biological reactor is inevitably increased. The increase in nutrients entering the reactor is accompanied by an increase in oxygen demand, which adds to the energy cost of the WWTPs. This is due to the need for oxygen to nitrify ammonia and for phosphorus uptake by phosphorus accumulating organisms (PAOs) in the aerobic zone of the reactor. Additionally, most WWTPs are not designed for the extra capacity required due to increased nutrient load, and as a result, the overload of the system impacts the effluent quality. Hence, non-compliance to effluent restrictions is often observed. (Nsengiyumva *et al.*, 2020; Ikumi *et al.*, 2019)

Side stream treatment of sludge dewatering liquor can potentially mitigate the negative impact on the system discussed above. With the implementation of suitable side stream treatment technologies, the opportunity arises for nutrients (ammonia and phosphates) to be removed or recovered without having to recycle the untreated liquor back to the biological reactor. With the benefit of nutrient recovery, improved effluent quality and potentially lower energy consumption, side stream treatment can prove to be important in aiding the transition from WWTP to WRRF.

The evaluation of treatment technologies in wastewater engineering is traditionally carried out using simplistic rules of thumb or mathematical models through software simulation environments (e.g., Matlab/Simulink, WEST®, BIOWIN, SUMO, etc.). Nevertheless, most of the state-of-the-art models used in wastewater engineering have been developed for conventional systems that focus on ensuring good effluent quality, rather than extraction of valuable compounds. In promotion of the paradigm shift from WWTPs to WRRFs, mathematical models require a more comprehensive and rigorous evaluation process to assist in strategic decisions of reshaping the current WWTP systems to future WRRFs. A comprehensive evaluation process that includes the fate of products generated by the WRRFs may result in benefits such as: (1) promoting the recovery of resources (such as nutrients, energy and water) from wastewater instead of simply “removing pollutants”, (2) reducing energy needs and consequently greenhouse

gas emissions and the risk of climate change, and (3) the optimisation of energy production from residual organic material, thus reducing the expenditures derived from plant operation.

1.3 Research problem

The performance of various WWTP design and control strategies can be assessed using the performance indices developed by the IWA BSM task group (Jeppsson *et al.*, 2007). This approach is based on the impact of those control strategies on water bodies, into which wastewater effluent is discharged, and the associated net operational costs at each WWTP unit operation. In the application of waste treatment process mathematical models, as decision making tools that promote resource recovery and sustainability, the limitations in the existing evaluative performance indices become more apparent. The latter overlooks the impact of products generated from WRRFs on air quality, land, other operational costs and the prospect of cost reduction via resource recovery strategies (Figure 1-1). A more appropriate evaluation framework is required to include the fate of products generated from WRRFs to influence their design and operation, within the system-wide context (i.e., consideration of resource recovery potential).

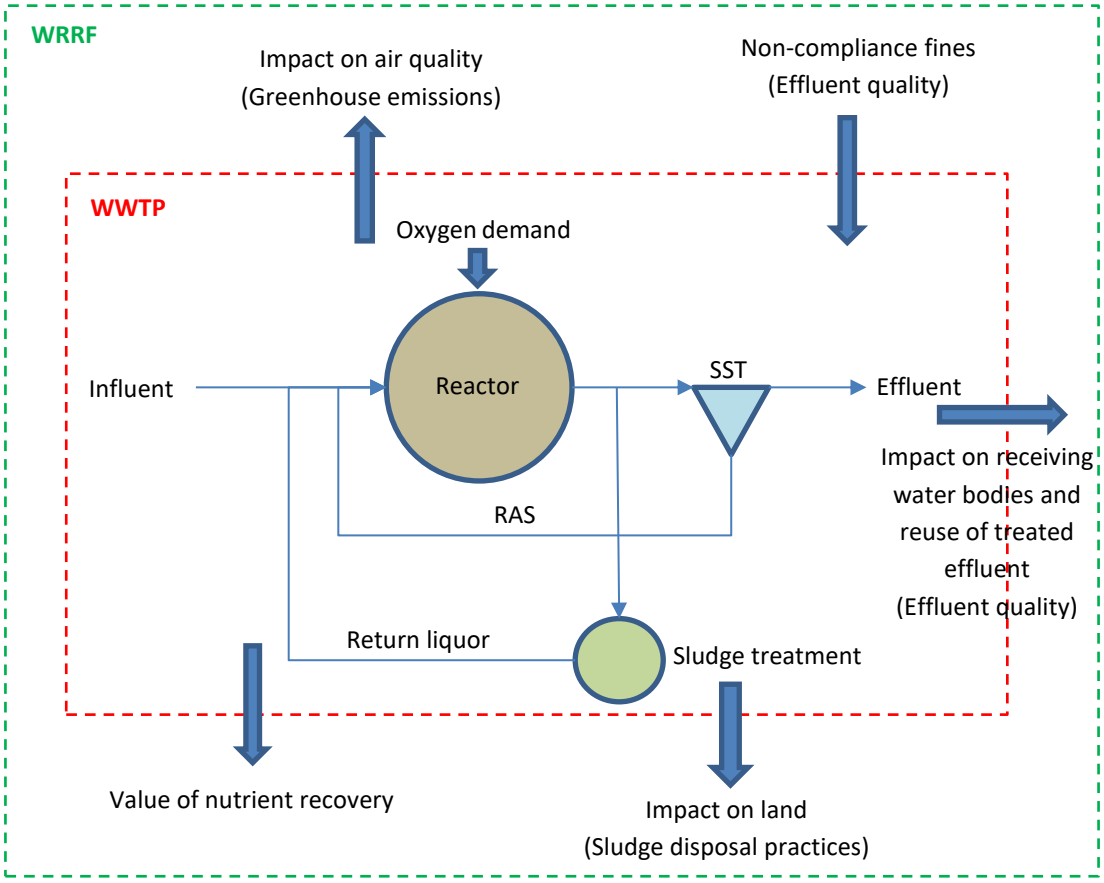

Figure 1-1: WWTP-WRRF comparison overview

1.4 Research hypothesis

Implementation of side stream treatment technologies to treat the nutrient-rich dewatering liquor from the sludge treatment process at WWTPs comes at a cost. However, the potential recovery of nutrients allows for some of the overall costs at WWTPs to be offset. In an age when Earth's nutrient reserves are fast depleting and when water scarcity is a reality in many countries around the world, the value of resource recovery is rapidly increasing. This dissertation therefore hypothesises that:

A WWTP model evaluative framework that considers the fate of WWTP products and value of resources that can potentially be recovered from WRRFs will generate a useful tool for design and operation of WWTPs in South Africa and aid in their transition to future WRRFs.

1.5 Key questions

Based on the hypothesis, several key research questions have been formulated such that the answers to these questions address the proposed objectives.

i) What are the currently available nutrient recovery technologies in the field of wastewater treatment?

ii) What are the fates of the different products generated from the treatment of wastewater at WRRFs and what WRRF products need to be taken into consideration?

iii) How can the existing evaluative performance indices (EQI and OCI) be modified to fit the concept of WRRFs?

1.6 Project objectives

The overall objective of this research project is to develop performance indices that can be used to perform system-wide evaluations of WRRFs in South Africa. The following are specific project objectives:

i) Review of conventional wastewater treatment plant unit operations, together with common side stream treatment technologies and the fate of products generated by the treatment process, with the aim of determining the best use of the resources recovered from waste in industrial applications.

ii) Development of formulated performance indices that include the fate of WRRF products as the evaluative framework (which is an extension of the one developed IWA BSM task group; Jeppsson *et al*., 2007) used to assess strategies in system design or operation, during application of mathematical models that virtually replicate WRRFs. The performance indices shall be included as evaluative equations in the UCT plant wide model (PWM_SA; Ikumi *et al.*, 2015a). The utilisation of the modified PWM_SA model for the simulation of a full scale South African wastewater treatment works shall be prepared as a case study that showcases the potential for utilisation of the model as WRRF evaluative tool.

1.7 Scope and limitations of project

This project is essentially a comparative study of the impact of side-stream nutrient recovery on WWTP performance. This desk study therefore does not involve any form of physical investigation such as collection of wastewater samples at the WWTP and laboratory experiments among others.

For the purposes of this study, the modelling of scenarios for comparison is limited to steady state MS excel modelling and WEST® using the PWM_SA. The latter was developed by the UCT WRG and integrates various sub-models that allow the simultaneous modelling of processes such as nutrient removal, sludge digestion and ionic speciation among others.

The performance indices developed by the IWA BSM task group were modified and used as a basis for the evaluation of the impact of side-stream nutrient recovery on the overall WWTP performance. While this study is limited to the aforementioned method of evaluating WWTP performance, other Key Performance Indicators (KPIs) such as the Life Cycle Assessment (LCA) may be used in future studies as a means of comparison.

This study does not involve the development of new sub-models such as modelling of greenhouses gases from unit processes, modelling of micro-pollutant concentrations in sewage sludge, and modelling of particle sizes and purity of nutrients precipitated. Thus, the outcome of the evaluation is limited with respect to the current outputs of the PWM_SA. The extensions to the IWA BSM task group performance indices is therefore an evaluative framework whose depth of evaluation is intended to increase in the future when there will be more developments in WWTP modelling.

1.8 New knowledge contribution

It is expected that from this research project, the existing performance indices used to evaluate WWTPs are extended to fit the concept of WRRFs. Including system-wide evaluative performance indices, which comprise the fate of products generated by WRRFs, into PWM_SA developed by Ikumi *et al.* (2015a) allows the evaluation of different WWTP design and control strategies towards ensuring that good effluent quality is maintained, recovery of resources is

maximised, energy production is optimised and at the same time, keeping operational cost and environmental impact as low as possible.

1.9 Sustainable development goals (SDG)

This study is aligned with goal 14 of the United Nations SDGs. With the modification of the OCI to include fines for non-compliance with effluent standards, it is expected that non-compliance to effluent standards (which is partly due to operational issues at WWTPs) will be mitigated. Therefore, the development of the new OCI encourages reduction in marine pollution. Additionally, goal 13 of the SGDs is also addressed through the formulation of an EQI specifically for greenhouse gas emissions.

1.10 Plan of development

The overall objective of this dissertation is addressed through various sub-topics grouped into 6 main chapters. Chapter 1 of the report introduces the background to the research, the research problem, the hypothesis and key questions, and the objectives of the research. Thereafter in Chapter 2, a critical review of literature pertaining to the aim of this dissertation is presented and the gaps in the research are identified. Additionally, Chapter 2 also addresses the above-mentioned objective (i) through a synthesis of literature relating to WWTP unit operations, side-stream treatment technologies and the fate of products generated from WRRFs. Chapter 3 then details the development of evaluative performance indices, building on the existing ones, with the inclusion of elements pertaining to WRRFs. Chapter 4 presents a case study of a South African WWTP where the impact of implementing a chosen side-stream treatment process was simulated. Chapter 5 presents the findings of model simulations including the new performance indices as well as a discussion on the results. Lastly, Chapter 6 concludes the dissertation with the synthesis of the findings in preceding chapters and recommendations are made. Figure 1-2 illustrates an overview of the research report.

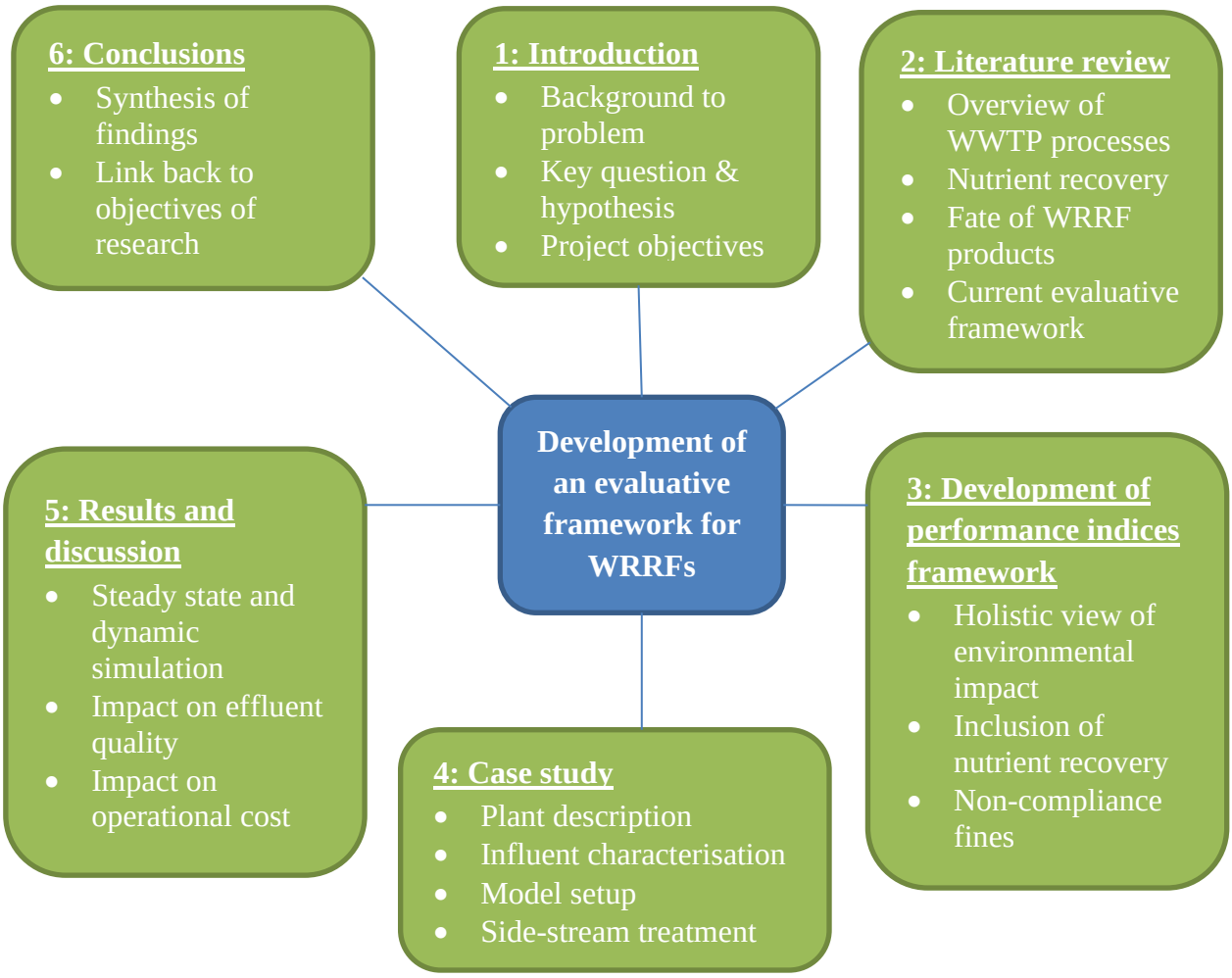

Figure 1-2: Overview of the research report

2. Literature review

The literature review investigates the current state of knowledge pertaining to the objectives of this research. This chapter begins with a brief description of the nature and characteristics of wastewater in Section 2.1. This is followed by an overview of treatment processes at WWTPs (Section 2.2) comprising primary treatment, organic and nutrient removal, and sludge treatment. Thereafter in Section 2.3, a review of side stream nutrient recovery technologies that could potentially be suitable for implementation in South Africa is presented. Subsequently, Section 2.4 presents a review of literature pertaining to the fate of products generated from WRRFs. Section 2.5 describes the current framework used to quantitatively evaluate the performance of WWTPs in terms of effluent quality and operational cost. Finally, the findings of the literature review are summarised and gaps in research are identified in Section 2.6.

2.1 Characteristics of wastewater

The design and operation of the various unit processes in the treatment of wastewater requires the physical and biological characterisation of influent wastewater. Physical characterisation of wastewater involves the determination of organic and inorganic fractions, and also the soluble and particulate fractions. (Ekama & Wentzel, 2008a)

Inorganic matter is that which is not derived from biological sources - i.e., does not originate from living organisms. Inorganic matter, which is typified by the lack of carbon-hydrogen bonds, includes rocks, metal, minerals, etc. and may also be in soluble form. (Henze *et al.*, 2008)

Organic matter, on the other hand, is carbon-based matter that originates from living organisms generally containing carbon-hydrogen bonds. Whether in soluble or particulate form, organic matter can be subclassified into unbiodegradable and biodegradable. The implication of the latter is that living things such as bacteria and microorganisms are able to biologically break down such organic matter into simple products such as carbon dioxide and water. (Henze *et al.*, 2008)

Depending on its characteristics, a constituent of influent wastewater may undergo either physical or biological transformations in the biological reactor or may escape with effluent

undergoing no transformation (Ekama & Wentzel, 2008a). Figure 2-1 summarises the possible transformations that take place in the reactor based on the wastewater characteristics.

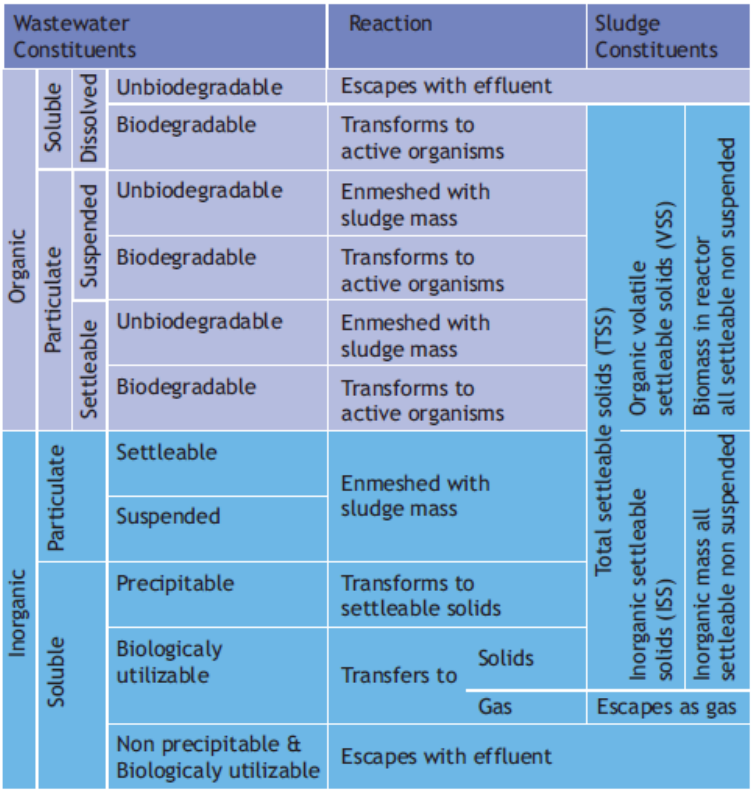

Figure 2-1: Reactions in the biological reactor (Ekama & Wentzel, 2008a:54)

2.2 Overview of WWTP processes

Each of the unit operations in the wastewater treatment process comprise physical and/or biochemical processes that facilitate the removal of various pollutants from wastewater. The removal of dissolved pollutants necessitates a phase transformation from either liquid to gas, or liquid to solid. On the other hand, solids can be removed from the liquid through sedimentation processes— primary sedimentation removing solids from the influent wastewater prior to treatment in the biological reactor and secondary sedimentation after biological treatment. The sludge collected from the sedimentation processes can then be thickened and is usually stabilised via aerobic or anaerobic digestion.

Figure 2-2 shows a depiction of a typical layout of a WWTP with either aerobic digestion or anaerobic digestion of sludge.

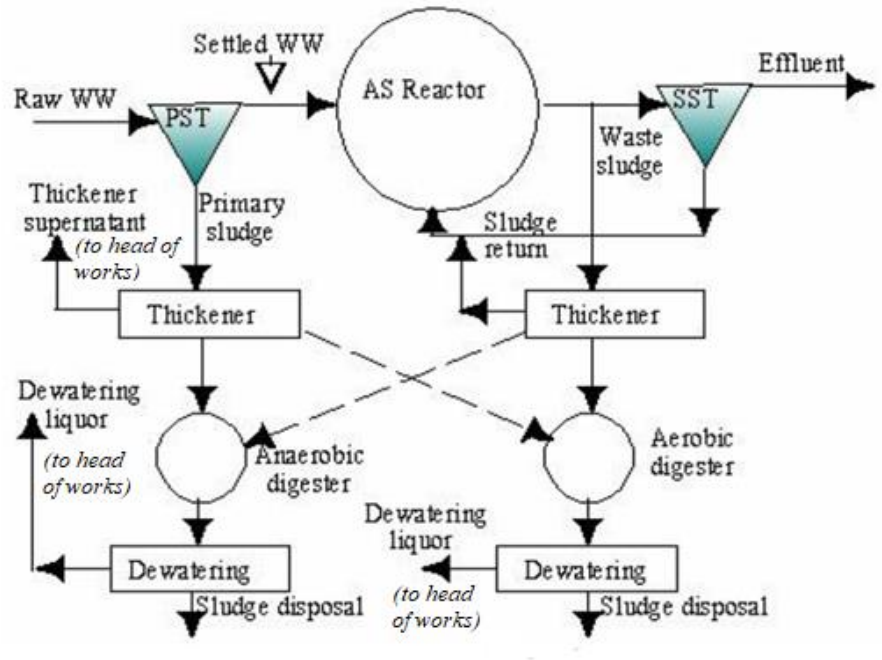

Figure 2-2: Typical layout of a WWTP

2.2.1 Primary treatment

Primary treatment is a physical process which involves the removal of certain solids that can be easily collected from raw wastewater influent. This stage of treatment comprises two major steps. Firstly, the raw wastewater undergoes a screening process in which large solids, which could interfere with mechanical equipment or cause blockages downstream, are removed. Secondly, grit is removed from the influent to prevent abrasion-damage to machinery as well as accumulation in other process units. (Henze *et al.*, 2008)

2.2.1.1 Primary sedimentation

Prior to the biological treatment, some WWTPs may be designed such that the influent wastewater undergoes the additional step of primary sedimentation once the screening and

degritting processes are complete. Readily settleable particles, removed from the wastewater by the primary settling tank (PST), form primary sludge at the bottom of the tank. The supernatant, comprising dissolved pollutants and non-settleable solids in suspension, is subsequently treated in the biological reactor. With settleable solids removed by the PST, the organic load on the biological reactor is reduced. Consequently, the required reactor volume, aeration power requirement and secondary sludge production are all reduced accordingly. (Henze *et al.*, 2008)

2.2.2 Secondary treatment

Once primary treatment is completed, the influent wastewater undergoes secondary treatment. During this stage of treatment, pollutants are removed from the wastewater biologically in the reactor by the action of microorganisms. Thereafter, solids are separated from the water by secondary sedimentation. The result is a water of very high quality relative to the product of primary treatment alone. (Henze *et al.*, 2008)

2.2.2.1 Biological treatment

Biological treatment involves the removal of organics and nutrients (nitrogen and phosphorus) from wastewater by the action of mediating microorganisms. During this stage biodegradable soluble organics become newly formed biomass while endogenous residue and particulate organics, regardless of their biodegradability, get enmeshed within the sludge mass. Unbiodegradable soluble organics that enter the WWTP in the influent escape to the effluent as they undergo no change in the biological reactor. Some of the nutrients are taken up by microorganisms to synthesise cell mass, while the rest can be removed by using the bioprocesses described below. The biological sludge mass is then separated from the water by secondary sedimentation. (Marais & Ekama, 1976)

Organic removal

Organic removal is the process whereby biodegradable organic material, whether in dissolved or particulate form, is broken down in the biological reactor by mediating microorganisms known as ordinary heterotrophic organisms (OHOs). A fraction of the organics that are broken down is

utilised by the microorganisms to synthesise cell mass (anabolism) resulting in an increase in bacterial population. The remaining fraction of the organic material is used to generate energy for cell maintenance (catabolism). The catabolism process entails the passing of electrons from the organic material to a terminal electron acceptor, which in aerobic conditions is oxygen. In the absence of oxygen however, nitrates and sulphates can be used as electron acceptors, albeit at a lower efficiency. Besides the growth process, the second process in biological treatment is the death process of microorganism known as endogenous respiration. During this process, about 80% of the live biomass that dies is utilised for catabolism while the remaining 20% becomes unbiodegradable endogenous residue. (Marais & Ekama, 1976)

Nitrogen removal

The ammoniacal nitrogen content of wastewater can be traced to two principal forms. It can either be in the form of free and saline ammonia (FSA) in dissolved (NH_3) and ionic (NH_4^+) form where its speciation is dependent on the pH of the wastewater; or in organically bound nitrogen form where NH_4^+ is bound to a carbonaceous molecule forming a protein (Marais & Ekama, 1976). The removal of nitrogen from influent wastewater is a two-step process— firstly nitrification and subsequently, denitrification.

Nitrification is a biological process by which the nitrogen content of wastewater, in the form of FSA, is converted into nitrates. FSA is oxidised sequentially into nitrite (NO2-) and nitrate (NO3-) by two groups of autotrophic nitrifier organisms (ANOs) that are obligate aerobes (Ekama & Wentzel, 2008b). During the first step of nitrification, a group of ammonia oxidising organisms (AOOs) known as Nitrosomonas convert NH_4^+ to NO_2^- (Eq. 2-1). The second step of nitrification is facilitated by a group of nitrite oxidising organisms (NOOs) known as Nitrobacter converting NO_2^- to NO_3^- (Eq. 2-2). The first step of nitrification is the limiting process as NOOs are much faster than AOOs such that the oxidation of NO_2^- to NO_3^- happens relatively rapidly. Thus, build-up of NO_2^- in the effluent in uncommon (Marais & Ekama, 1976). The overall stoichiometry for the nitrification process is shown in Eq. 2-3 below.

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 2H_2O + 4H^+ \qquad \textbf{Eq. 2-1}$$

$$2NO_2^- + O_2 \rightarrow 2NO_3^- \quad \textbf{Eq. 2-2}$$

$$2NH_4^+ + 4O_2 \rightarrow 2NO_3^- + 2H_2O + 4H^+ \quad \textbf{Eq. 2-3}$$

The second step in the nitrogen removal from wastewater entails the conversion of nitrate produced from nitrification into nitrogen gas, in a biological process known as denitrification. Denitrification by the action of facultative OHOs (FOHOs) necessitates anoxic conditions. In the absence of oxygen, FOHOs are able to use the nitrates generated in the nitrification process as electron acceptors while breaking down organic material. In this process, the nitrate acting as terminal electron acceptor is reduced to nitrogen gas, which escapes into the atmosphere. Thus, nitrogen is removed from wastewater. (Ekama & Wentzel, 2008b)

A common WWTP configuration for nitrogen removal is the Modified Ludzack-Ettinger (MLE) system as shown in Figure 2-3 below. The first unit in this plant configuration is the anoxic zone, which is the recipient to the influent wastewater, the return sludge from the SST and the mixed liquor recycled back from the aerobic tank. To carry out their metabolic activities, the OHOs from the return activated sludge (RAS) recycle feed on the biodegradable organic material in the influent wastewater while using the nitrate-rich mixed liquor as electron acceptor. In the anoxic tank the nitrates are thus reduced to nitrogen gas. The second unit, being aerobic, allows the production of nitrates by the oxidation of FSA. Concurrently, biodegradable organic material is removed in the aerobic zone and transformed into biomass. (Ekama & Wentzel, 2008b)

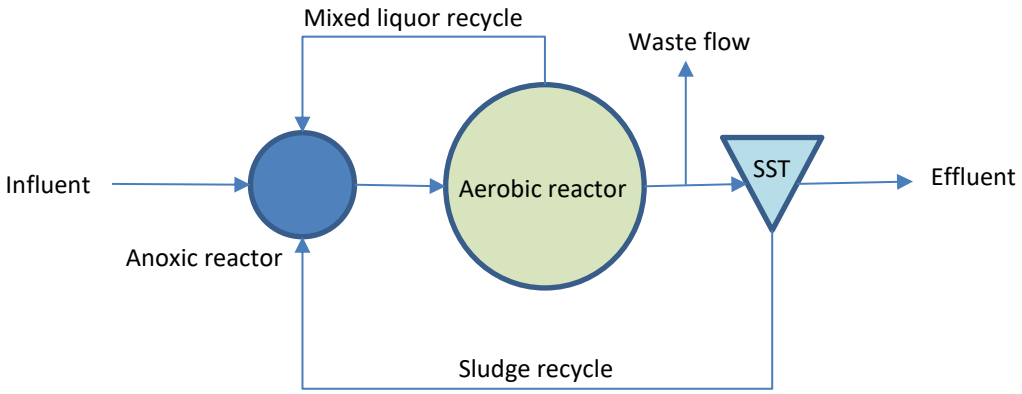

Figure 2-3: The MLE system

Phosphorus removal

Phosphorus removal at WWTP is achieved by transforming dissolved phosphorus into the solid phase chemically and/or biologically. The chemical removal of phosphorus involves the precipitation of phosphates via the addition of aluminium sulphate or Iron salts. The phosphate precipitate, being in the solid phase, becomes part of the sludge mass and is separated from the water in the secondary settling process. (Henze *et al.*, 2008)

Biological phosphorus removal in activated sludge systems is achieved by stimulating the growth of a group of organisms known as phosphorus accumulating organisms (PAOs). The PAOs have the ability to internally store large quantities of phosphorus in long polyphosphate chains. Unlike OHOs which incorporate up to 0.025 mgP/mgVSS, PAOs can accumulate phosphorus in excess up to 0.38 mgP/mgVSS. Since both types of microorganism coexist in the reactor, the larger the PAO to OHO ratio, the greater the phosphorus removal that can be achieved. Stimulation of PAO growth requires two vital conditions to be met— having an anaerobic zone preceding the aerobic (or anoxic) zone in the reactor and addition or generation of VFAs in the anaerobic zone. (Wentzel *et al.*, 2008)

With the lack of external electron acceptors in the anaerobic reactor, OHOs are unable to utilise biodegradable organics (FBSO) for growth. OHOs acid-ferment the fermentable

biodegradable soluble organics fraction of biodegradable organics forming volatile fatty acids (VFAs). The VFAs formed by fermentation add to the pool of VFAs already present in the wastewater. The PAOs in the anaerobic zone sequester the available VFAs from the bulk liquid forming long chain molecules of poly-ß-hydroxyalkanoates (PHAs) internally. To provide the energy required for the formation of PHAs, polyphosphates and glycogen are degraded. As a result of polyphosphate degradation, ortho-phosphates are released into the bulk liquid in the anaerobic reactor (Wentzel *et al.*, 2008). This behaviour of PAOs in the anaerobic reactor is illustrated in Figure 2-4.

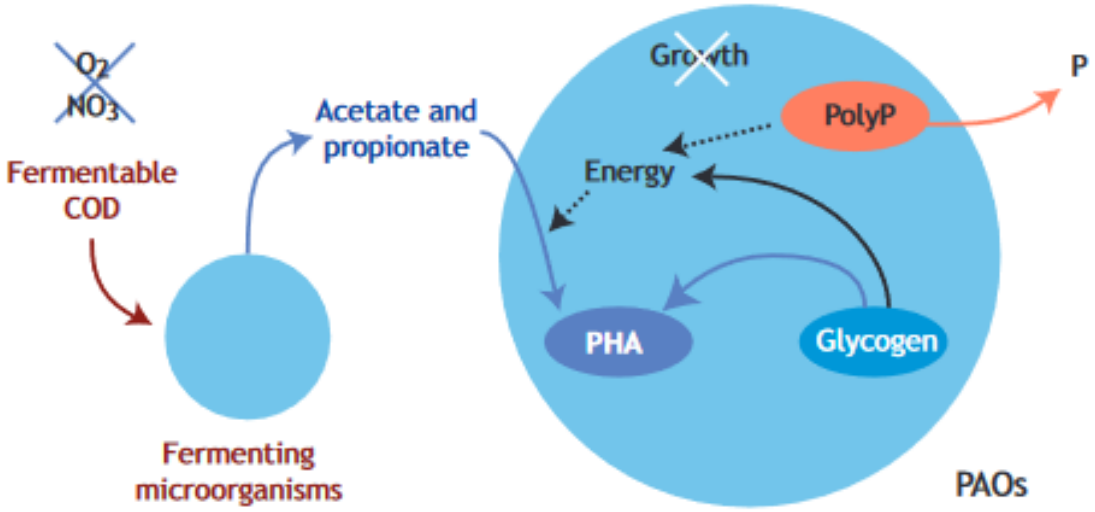

Figure 2-4: Biochemical model of PAOs under anaerobic conditions (Wentzel *et al.*, 2008:159)

Subsequently in the aerobic zone, with oxygen as external electron acceptors (or nitrate in the anoxic zone), the stored PHA formed in the previous step is utilised by PAOs for energy generation and synthesis of new cell mass as well as regeneration of glycogen. Additionally, the energy generated from PHA is used to form phosphate bonds as ortho-phosphates are taken up to regenerate the poly-phosphates degraded in the anaerobic zone. With the new PAO cells grown in the aerobic reactor, more ortho-phosphates are taken up than are released anaerobically thus resulting in a net removal of phosphorus from wastewater (Wentzel *et al.*, 2008). Figure 2-5 shows a depiction of PAO behaviour under aerobic/anoxic conditions.

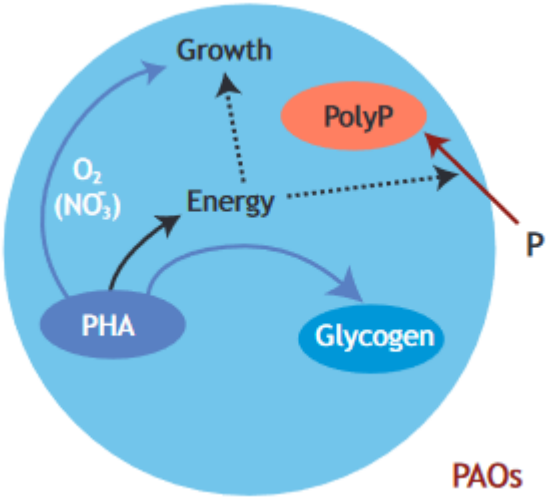

Figure 2-5: Biochemical model of PAOs under aerobic/anoxic conditions (Wentzel *et al.*, 2008:160)

NDBEPR systems

Nitrification-Denitrification Biological Excess Phosphorus Removal (NDBEPR) systems are activated sludge (AS) systems that comprise anaerobic, anoxic and aerobic zones in the biological reactor such that removal of both nitrogen and phosphorus from wastewater is achieved. In order to prevent the inhibition phosphorus removal, it is crucial to prevent the recycling of oxygen and nitrates to the anaerobic tank. With the presence of oxygen or nitrate as external electron acceptors in the anaerobic reactor, OHO would then be able to metabolise biodegradable soluble organics. Consequently, the amount of VFAs available to be taken up by PAOs would be reduced and release of ortho-phosphates in the anaerobic zone would also be reduced accordingly. As a result, the uptake of ortho-phosphates in the aerobic zone (or anoxic) would also be reduced resulting in a decrease in the net phosphorus removal (Wentzel *et al.*, 2008).

Configurations for WWTP AS systems, such as the UCT system or the JHB system (Figure 2-6) are designed to facilitate removal of both nitrogen and phosphorus from wastewater while preventing recycle of nitrate and oxygen to the anaerobic reactor. Both the UCT and JHB consist of a sequence of anaerobic, anoxic and aerobic reactors, with mixed liquor recycled from the

aerobic zone to the anoxic zone for denitrification. The difference between the two configurations lies in the RAS flow. In the JHB system, the RAS from the SST passes through an underflow anoxic reactor and thereafter, the RAS now void of nitrate and oxygen is recycled to the anaerobic reactor. Conversely, in the UCT system the RAS is recycled straight to primary anoxic reactor where any oxygen or nitrate present in the RAS is utilised. Then, the denitrified activated sludge is recycled from the anoxic reactor to the anaerobic reactor.

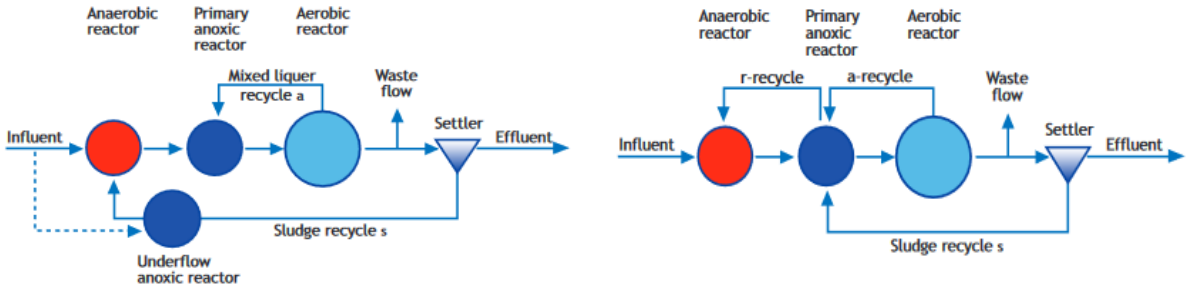

Figure 2-6: JHB (left) and UCT (right) WWTP configurations (Wentzel *et al.*, 2008:165)

2.2.2.2 Secondary sedimentation

Secondary treatment of wastewater is concluded with the clarification process in the secondary settling tank (SST). This step begins with the discharge of mixed liquor (treated water and sludge) from the biological reactor into the SSTs. The hydraulic conditions in the SSTs allow the sludge to settle by gravity to the bottom of the tank while along the perimeter of the SST, clarified effluent overflow over the weirs (Henze *et al.*, 2008). The activated sludge settled at the bottom of the tank is then pumped back into the biological reactor to continue treatment in a process known as the RAS recycle.

2.2.3 Tertiary treatment

Due to presence of pathogenic bacteria, the effluent from SSTs is unsafe to be discharged into receiving water bodies. In order to stabilise and disinfect the effluent, shallow maturation ponds may be used in which biological activity continues, albeit at a much slower rate than in a biological reactor. Over the retention time of the maturation pond, algae grow, and pathogenic

bacteria die off. Additionally, any particulate material that escape from the SST settle to the bottom of the pond. Maturation ponds make use of the sun's ultraviolet rays for disinfection (Henze *et al.*, 2008). A more modern approach to disinfection is through chlorine contact or ultraviolet light. With this approach the water may be fed chlorine which kills pathogens and diminishes odours. Alternatively, the effluent may be disinfected by using ultraviolet light or ozone as excess chlorine could be detrimental to aquatic life. (United States Environmental Protection Agency [EPA], 1998)

2.2.4 Sludge treatment

Sludge treatment is an essential part of the wastewater treatment process involving the treatment of primary sludge (PS), which is the product of sedimentation of raw wastewater, and secondary sludge, which is derived from waste activated sludge (WAS). Wastewater sludge is naturally rich in pathogenic organisms, decayable and foul-smelling. Unstabilized sludge is susceptible to attracting vector such as flies, which can carry infectious diseases. Sludge stabilisation involves reducing the biodegradable fraction of sludge, which as a result reduces risk of putrefaction, vector attraction and reduced the concentration of pathogenic organisms. It must be noted that while a fraction of the WAS is biodegradable, it more stable than PS. Additionally, the longer the sludge age at which the activated sludge reactor is operated, the more stable the WAS produced. There exists a number of sludge treatment options such as aerobic or anaerobic digestion amongst others, with the latter being the most common method used to achieved biological stabilisation of sludge. (Luduvice, 2007)

2.2.4.1 Aerobic digestion

While it is possible to aerobically digest both PS and WAS, aerobic digestion (AerD) of the latter is the most common out of the two types of wastewater sludge. This is because PS contains biodegradable organics that are utilised in AerD by OHOs for growth, resulting in a high oxygen demand. Consequently, the oxygen demand for endogenous respiration also increases due to the increase in OHO population. Additionally, PS is more useful in AD (explained below) for the production of methane gas which can be used to generate energy. (Henze *et al.*, 2008)

With only WAS being treated, no substrate is available for microorganisms to grow and multiply. Aerobic digestion essentially becomes a continuation of the endogenous respiration process from the activated sludge system. As the OHOs die from endogenous respiration, organically bound nitrogen and phosphorus are released as ammonia and ortho-phosphates respectively. Ammonia is then converted into nitrate by ANOs, consuming alkalinity in the process. To prevent the digester pH from dropping too low and consequently inhibiting nitrification, lime needs to be dosed to maintain the pH in the optimal range of 7 to 8 for nitrification. Alternatively, the digester can be operated with intermittent aeration such that the sludge becomes periodically anoxic. Nitrate produced by ANOs are then denitrified and alkalinity consumed during nitrification is recovered. Additionally, denitrification allows 63% recovery of nitrification oxygen demand. (Henze *et al.*, 2008)

2.2.4.2 Anaerobic digestion

Anaerobic digestion (AD) is multi-stage biochemical process whereby sewage sludge is stabilised in an oxygen-free environment by the action of four groups of microorganisms—acidogens, acetogens, acetoclastic methanogens and hydrogenotrophic methanogens (Batstone *et al.*, 2002). The anaerobic digestion of organic material in sewage sludge takes place in four main metabolic steps (Figure 2-7):

1. Hydrolysis: During this step, complex organic compounds such as proteins, carbohydrates and lipids are broken down by enzymes into simpler amino acids, sugars and fatty acids respectively.
2. Acidogenesis: The products of the hydrolysis stage are converted by acidogens into carbon dioxide, hydrogen, acetic acid and other short chain fatty acids (SCFA) such as propionate and butyrate.
3. Acetogenesis: This step involves the conversion of SCFAs with more than two carbon atoms into acetic acid, hydrogen and carbon dioxide by the action of acetogenic bacteria.
4. Methanogenesis: This final metabolic step in anaerobic digestion is the conversion of acetic acid and hydrogen into methane gas by acetoclastic methanogens and hydrogenotrophic methanogens respectively. (van Lier *et al.*, 2008; Luduvice, 2007)

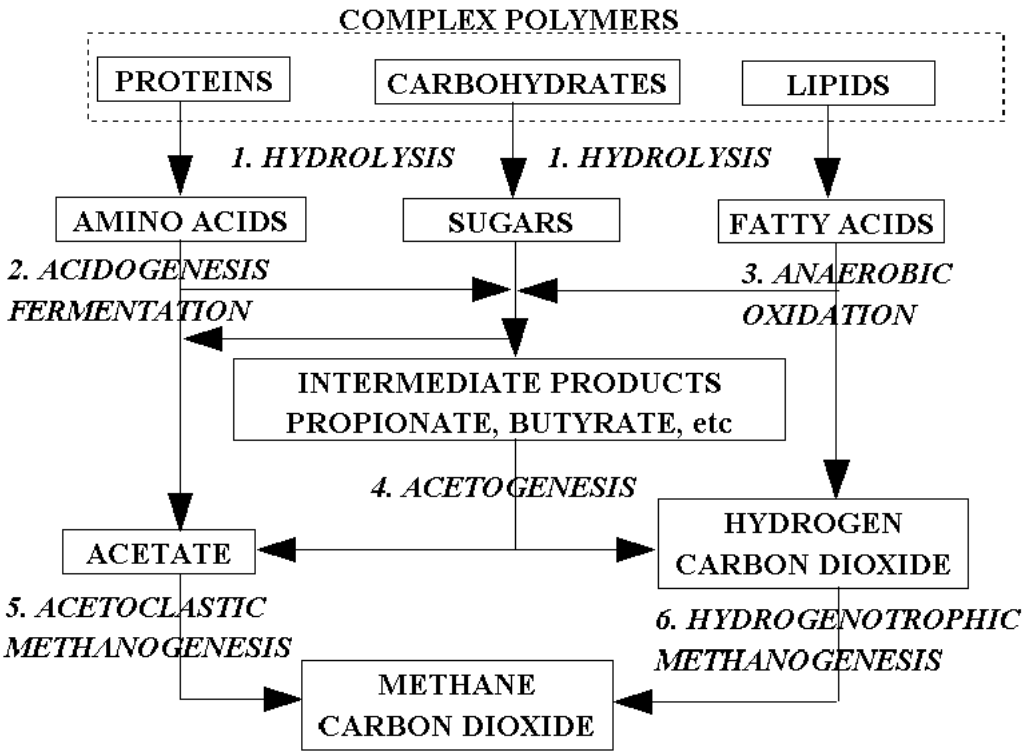

Figure 2-7: Overview of the anaerobic digestion process (Adapted from Gujer & Zehnder, 1983:129)

Sludge stabilisation through anaerobic digestion results in the production of biogas, which comprises mainly methane and carbon dioxide as well as small concentrations of other gases such as nitrogen, oxygen and hydrogen sulphide. Due to the presence of methane and carbon dioxide, if allowed to escape into the atmosphere, biogas will contribute to global warming. However, with adequate piping to minimise leakages, biogas can also be used to generate energy which can offset the power demand at WWTPs. The greater the concentration of methane in biogas, the higher the energy that can be generated. (Luduvice, 2007)

2.3 Nutrient recovery

Stabilisation of sewage sludge (PS and WAS) through aerobic or anaerobic digestion involves the breakdown of biodegradable organic material. As proteins and microorganisms in the sludge

are broken down, high concentrations of organically bound nitrogen and phosphorus are released into the bulk liquid in form of ammonia and ortho-phosphates respectively. Should the sludge mixture be aerobically digested, the ammonia released would be oxidised and turned into nitrate (Ikumi & Harding, 2020). Nitrates in turn can be converted to nitrogen gas with intermittent aeration, leaving a sludge liquor rich in ortho-phosphate. On the other hand, if the sludge mixture is anaerobically digested, both ammonia and ortho-phosphates would exist in the sludge dewatering liquor. In addition to the organically bound phosphorus released during the lysis of microorganisms, anaerobic digestion provides the optimal conditions for the release of large amounts of phosphorus due to degradation of polyphosphate chains inside PAOs. (Ikumi & Ekama, 2019)

Due to stringent regulations, many WWTPs recycle the nutrient-rich sludge dewatering liquor to the head of works for further treatment prior to disposal into water bodies. This recycle flow results in an increase in the nutrient load on biological reactor which can potentially have a negative impact on the wastewater treatment process. If WWTPs are not designed to have extra capacity do treat the additional nutrient load, the system will be overloaded, and this can result in non-compliance to effluent regulations. In addition to that, since nitrification and phosphorus uptake by PAOs both require oxygen, an additional oxygen demand is generated for the removal of the extra nutrient load and consequently adding to the aeration costs. Furthermore, denitrification of extra nitrate produced as well as PAO growth in the anaerobic reactor to subsequently remove the extra phosphorus load aerobically both require addition of readily biodegradable organics such as acetate, which further adds to operational cost. (Ikumi *et al.*, 2019)

Side stream treatment of sludge dewatering liquor mitigates the abovementioned negative impacts on the wastewater treatment process as nutrients (N and P) are removed prior to recycling. Additionally, the soluble nutrients in the sludge dewatering liquor can be recovered in useful forms such as agricultural fertiliser. The following subsections give an overview of nutrient removal/recovery technologies that could potentially be suitable for implementation in a South African context.

2.3.1 Nitrogen removal

2.3.1.1 ANAMMOX

Anaerobic ammonium oxidation (ANAMMOX) is a potential side-stream treatment technology that can be used to remove nitrogen from sludge dewatering liquor at WWTPs. This biochemical reaction is mediated by autotrophic ANAMMOX bacteria which have the metabolic ability to oxidise ammonium anaerobically, to produce nitrogen gas, by utilising nitrites as terminal electron acceptors. (Kotay *et al.*, 2013)

$$NH_4^+ + 1.32NO_2^- + 0.066HCO_3^- + 0.13H^+ \rightarrow 0.066CH_2O_{0.5}N_{0.15} + 1.02N_2 + 0.26NO_3^- + 2.03H_2O \quad \textbf{Eq. 2-4}$$

Eq. 2-4 above (Strous *et al.*, 1998) describes the stoichiometry of the ANAMMOX process. It can be seen that for each mole of ammonia oxidised, the ANAMMOX bacteria require 1.32 moles of nitrite, used as terminal electron acceptors, to mediate the process. While being a necessity for the reaction to take place, nitrite has been observed to hinder the performance of ANAMMOX bacteria at certain concentrations (Kotay *et al.*, 2013).

To meet the nitrite requirement of the reaction, the dewatering liquor can be supplemented with nitrites (e.g., sodium nitrite) added externally. However, external supplementation of nitrites at the scale of a real WWTP is not feasible due to cost and operational reasons (Kotay *et al.*, 2013). Alternatively, the partial oxidation of ammonia under oxygen limited conditions, can be induced. This can be done simultaneously with the ANAMMOX process in a single stage partial nitrification-anammox reactor, by maintaining low dissolved oxygen (DO) concentration in the reactor which prevents nitrites from being oxidised (Dasgupta *et al.*, 2017). Since the presence of DO and nitrite can affect the growth of ANAMMOX bacteria, having the partial nitritation and ANAMMOX processes taking place in a single reactor requires close monitoring and control of DO and nitrites. At an actual WWTP, ensuring efficient ANAMMOX process for this setup would therefore demand skilled workers and high maintenance (Kotay *et al.*, 2013). Therefore, having the partial nitrification and ANAMMOX in separate reactors renders operation simpler and allows for more efficient process control. Partial nitrification can be achieved using the

SHARON (Single reactor for High activity Ammonia Removal Over Nitrite) process. By having a short hydraulic retention time, an alkaline pH, a high temperature and a low DO concentration in the SHARON reactor, the growth of NOOs is suppressed and thus production of nitrate is mitigated (Kotay *et al.*, 2013; Hellinga *et al.*, 1998). From the SHARON process, a mixture of 50% ammonium and 50% nitrite mixture is fed into a separate reactor for the ANAMMOX process whereby ammonium is converted into nitrogen gas as shown in Figure 2-8. (EPA, 2007)

The ANAMMOX process allows for the removal of about 90% nitrogen from the sludge dewatering liquor without going through the conventional processes of nitrification and denitrification (Kotay *et al.*, 2013; EPA, 2007). While the reaction still results in production of nitrates as a by-product, a much smaller quantity is produced. The oxidation of ammonia by the conventional process of nitrification under aerobic conditions produces one mole of nitrates for each mole of ammonium oxidised (Marais & Ekama, 1976). With the ANAMMOX process, approximately 76% less nitrates are expected to be produced as a result of the reaction (Strous *et al.*, 1998). Residual nitrates produced by the ANAMMOX process can either be reduced by autotrophic denitrification or by heterotrophic denitrification. The heterotrophic denitrification, in anoxic conditions, of the nitrate produced would therefore require less readily biodegradable organics as electron donors compared to nitrate produced if conventional nitrification-denitrification were the treatment method. Alternatively, autotrophic denitrification by using elemental sulphur as electron donors eliminates the need for dosing of readily biodegradable organics which would result in savings with respect to operational cost and smaller carbon footprint (Dasgupta *et al.*, 2017).

Unlike conventional oxidation of ammonia in biological nutrient removal (BNR) reactors, ANAMMOX requires approximately 60% less oxygen (Kotay *et al.*, 2013). With aeration being the most demanding process with regards to energy consumption (Musvuto & Ikumi, 2016), such a significant reduction in oxygen demand naturally translates into savings in the energy cost at WWTPs implementing side stream ANAMMOX process.

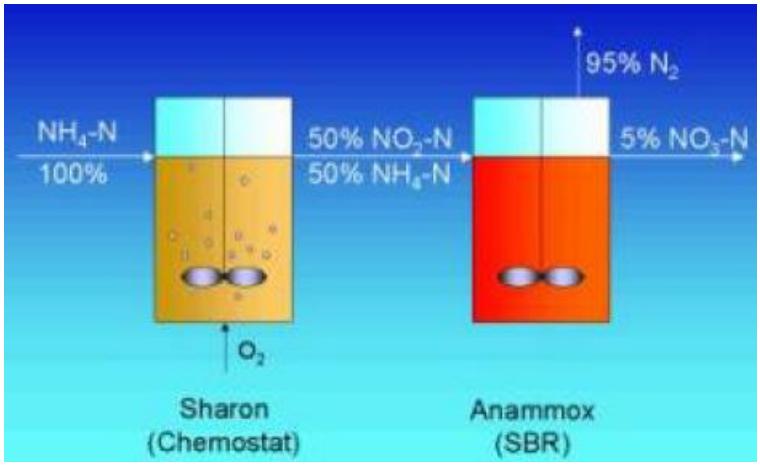

Figure 2-8: The SHARON/ANAMMOX process (EPA, 2007:3)

Due to the abovementioned benefits pertaining to oxygen consumption, operational cost, carbon footprint, nitrogen removal efficiency and simplicity of operating a SHARON/ANAMMOX system along with the fact that ANAMMOX bacteria has previously been detected in some South African WWTPs (Tikilili & Chirwa, 2014), it can be said that ANAMMOX may have potential for implementation as a side stream treatment process in South Africa. However, it must be noted that the ANAMMOX bacteria that may be found in some South African WWTPs are present in extremely low quantities. Extended pre-enrichment is thus required to increase the bacterial population to levels detectable by nested PCR analysis (Gokal, 2017). In order to successfully implement ANAMMOX as a side-stream treatment option, mass cultivation of the bacteria would be required. However, South Africa does not currently have any operational ANAMMOX systems. Hence, the challenge would be to have access to a sufficiently large quantity of seed sludge to start up the system. This would likely come at a substantial cost due to the need for importation.

2.3.1.2 BABE

The BABE (bio-augmentation batch enhanced) process, developed by DHV water (Zilverentant, 1999), is a side stream treatment process designed to remove ammonia from sludge dewatering liquor and improve nitrification in the activated sludge system through the bio-augmentation of nitrifiers in a side stream reactor (Figure 2-9). It is operated in a batch reactor where each batch cycle comprises five steps: (1) aeration, (2) mixing, (3) second aeration, (4) settling and (5)

discharging. Steps (1) and (3) involve nitrification of ammonia in the reactor while denitrification of nitrate produced takes place during steps (2), (4) and (5) (Salem *et al.*, 2004). Once the cycle completed, the sludge with augmented nitrifiers is mixed with the RAS and re-enters the activated sludge system. (Salem *et al.*, 2002)

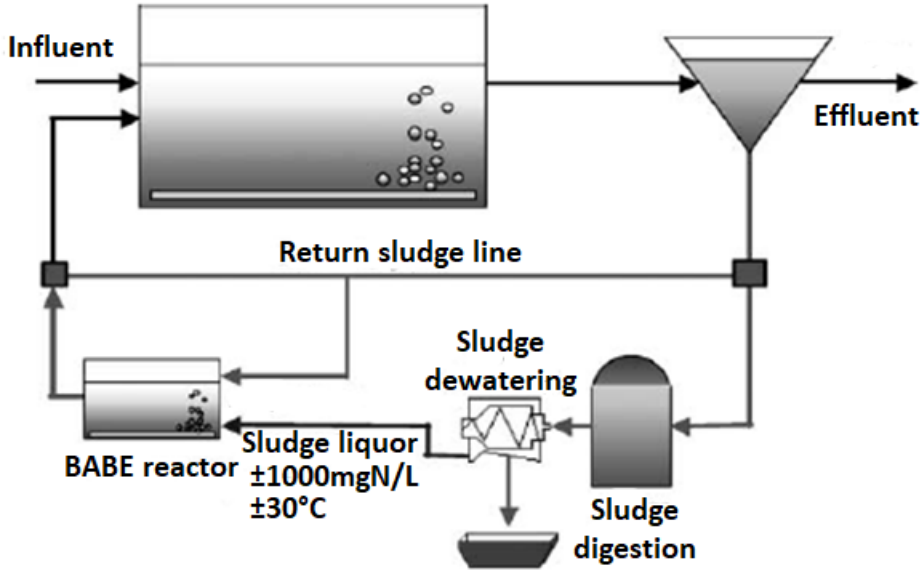

Figure 2-9: The BABE process (van Loosdrecht & Salem, 2006:16)

The influent to the BABE reactor comprises high temperature sludge dewatering liquor (about 30˚C) from the sludge treatment process and a fraction of the RAS. The RAS flow, being at ambient temperature, affects the temperature inside the BABE reactor. Thus, the fraction of RAS added to the side stream reactor needs to be selected such that optimal operation of the BABE reactor can be achieved. A low RAS fraction allows the temperature inside the BABE reactor to be high. Under these conditions bacterial growth rates are high and hence the enrichment of nitrifiers and removal of ammonia from the sludge dewatering liquor are maximised. Since nitrification consumes alkalinity, the pH of the BABE reactor can be controlled through the introduction of denitrification at steps (2), (4) and (5) mentioned above when the batch reactor is not aerated (Salem *et al.*, 2003).

Simulations run by Salem *et al.* (2004) showed that the optimal sludge retention time (SRT) for the BABE reactor lies between 0.3 to 0.8 days (Figure 2-10), albeit the actual value is dependent on the ammonia load on the BABE reactor and on the actual death rate of nitrifiers.

With a short SRT, the endogenous nitrifiers in the system are ones enriched by the side stream treatment process. A long SRT would lead to growth nitrifier populations that might not be adapted to the mainstream reactor. (Salem *et al.*, 2003)

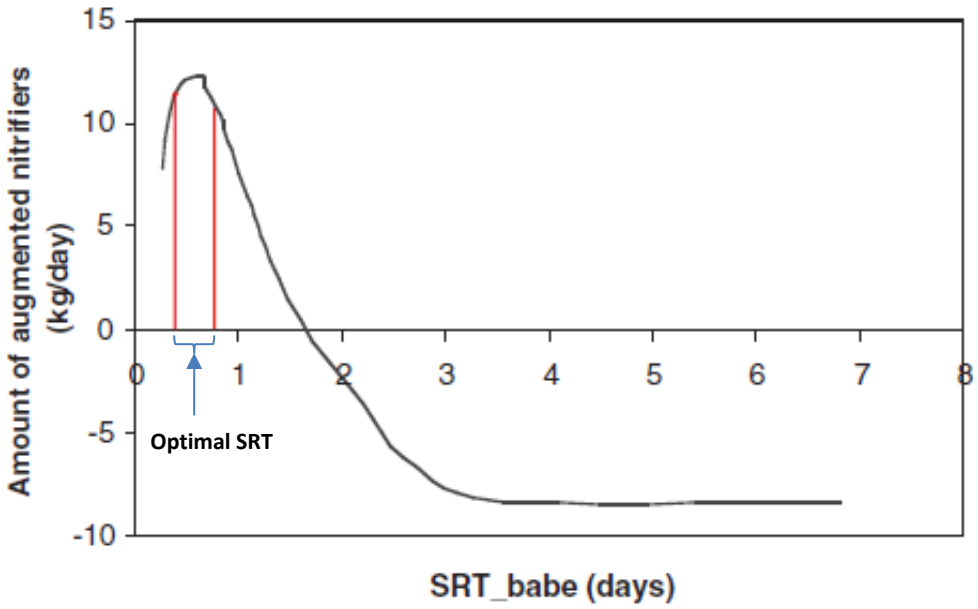

Figure 2-10: Amount of nitrifiers augmented as a function of SRT (Adapted from Salem *et al.*, 2004:95)

The presence of augmented nitrifiers implies that mass balance for the rate of change of nitrifiers in the mainstream process now includes the addition rate of nitrifiers from the BABE process. As a result, the minimum SRT for nitrification in the mainstream process as well as the ammonia concentration in the effluent decreases. The BABE process has a greater influence on the mainstream treatment process at lower ambient temperatures. For an addition rate of 0.1 g/gd, the minimum SRT is reduced by approximately 45% at 8°C compared to 20% at 20°C. Likewise, the effluent ammonia concentration decreases by 80% at 8°C and 60% at 20°C (Salem *et al.*, 2003). The decrease in SRT implies that a smaller aerobic volume is required to achieve complete nitrification and thus, the extra volume could be allocated to the anoxic denitrification of nitrate. (Salem *et al.*, 2004)

At low temperatures (< 15°C), the minimum SRT for nitrification to take place increases due to the much slower growth rate of nitrifier. Consequently, a larger aerobic volume is required to nitrify ammonia. With augmented nitrifiers from the BABE process, the minimum SRT decreases and nitrification is able to take place at low temperatures in winter. Conversely, at high ambient temperatures in summer, the impact of the BABE process on the mainstream treatment process is marginal as the system already nitrifies. Thus, at high temperatures the BABE process essentially only aids in reducing the extra ammonia load on the mainstream reactor (Salem *et al.*, 2003).

In the case where effluent standards still cannot be met with side stream bio-augmentation, upgrade of the activated sludge reactor would be necessary. Model simulations of Walcheren WWTP in the Netherlands by Salem *et al.* (2002) predicted that upgrading the WWTP to meet effluent standards without a side stream BABE reactor would require an 88% increase in the aerobic tank volume, 1300% increase in anoxic tank volume and increasing the SRT from 4.8 days to 20 days. For the scenario where side stream BABE is implemented, the model predicted a 22% increase in aerobic tank volume, 370% increase in anoxic tank volume and an increased SRT from 4.8 to 7.8 days. Upgrade of the WWTP with side stream BABE resulted in a smaller footprint with approximately 50% saving in area required for reactors. Additionally, it was estimated that upgrade of Walcheren WWTP with side stream BABE would result in 750,000 Euros savings on the construction costs and net savings of 11,500 Euros per annum.

In developing countries such as South Africa, the BABE process could prove to be a viable side stream treatment method due ease of retrofitting, small footprint and low cost. In addition to removal of ammonia from sludge dewatering liquor and the drastic enhancement of nitrification in cold climate during winter, a WWTP with side stream bio-augmentation has shown to be a cost-effective method of upgrading WWTPs. Therefore, implementation of BABE process is a financially manageable means of dealing with future load increases at WWTPs, especially in the context of a developing country.

2.3.2 Phosphorus removal

2.3.2.1 Struvite precipitation

Struvite, also known as magnesium ammonium phosphate hexahydrate ($MgNH_4PO_4 \cdot 6H_2O$), is a phosphate mineral that can be precipitated in conditions where the concentrations of magnesium, ammonium (NH_4^+) and dissolved ortho-phosphates are unusually high (Loewenthal *et al.*, 1994). Formation of one mole of struvite requires one mole of each of the components (i.e., magnesium, ammonium and phosphate). Therefore, the precipitation of struvite enables the removal of ortho-phosphates and concomitantly NH_4^+ from the nutrient-rich sludge dewatering liquor.

Struvite precipitation occurs when the product of Mg^{2+}, NH_4^+ and PO_4^{3-} concentrations exceeds the thermodynamic solubility product for struvite (K_{spm}') such that there is a state of supersaturation with respect to struvite. Should the product be less than K_{spm}', mineral struvite will re-dissolve back into magnesium, ammonium and phosphate species in aqueous phase (Eq. 2-5). Until a state of equilibrium is achieved between the solid and aqueous phases of magnesium, ammonium and phosphate species, struvite will either precipitate or dissolve into solution. (Loewenthal *et al.*, 1994)

$$[Mg^{2+}][NH_4^+]\left[PO_4^{3-}\right] = K_{spm}' \qquad \textbf{Eq. 2-5}$$

Various side stream treatment technologies have been developed for the precipitation of struvite mineral. From case studies involving four different technologies, Sikosana *et al.* (2016) concluded that Ostara Pearl® and Multiform Harvest, based on operational and economic aspects, were the most sustainable struvite precipitation technologies.

The Ostara Pearl® process entails the recovery of nutrients from sludge dewatering liquors by means of controlled precipitation of high-purity struvite fertiliser granules in an up-flow fluidised-bed reactor as illustrated in Figure 2-11 (Gysin *et al.*, 2018). The struvite crystallisation process starts at the bottom of the reactor where the nutrient-rich dewatering liquor enters. Magnesium salts are also added to meet the magnesium requirements for struvite precipitation. Additionally, sodium hydroxide is added when required to maintain pH. A fraction of the treated effluent from the top of the Pearl® reactor is recycled backed to the bottom of the reactor,

allowing control of struvite crystal size. The struvite crystals formed in the reactor are dewatered, heat dried and sorted by size. The end result is a high-value fertiliser with 99.6% purity without pathogens and relatively lower in salts and heavy metals. (Gysin *et al.*, 2018)

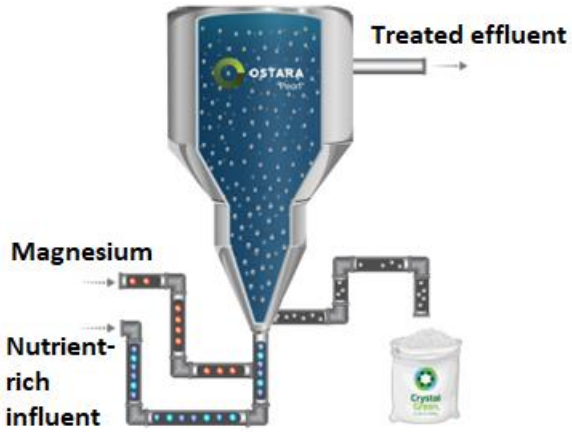

Figure 2-11: Ostara Pearl® process (Ostara, 2020)

Similar to Ostara's Pearl® process, struvite precipitation in the Multiform Harvest process is achieved in conical fluidised bed reactors (Figure 2-12). The sludge dewatering liquor is pumped up from the bottom of the reactor into the reactive zone. There, magnesium chloride and sodium hydroxide are dosed providing favourable conditions for the precipitation of struvite (Multiform Harvest, 2015). Unlike the Pearl® process, the treated effluent from the Multiform Harvest reactor is not recycled back to the bottom and the retention time is short. The struvite produced from this process ends up being of low quality and sand-like. (Sikosana *et al.*, 2016)

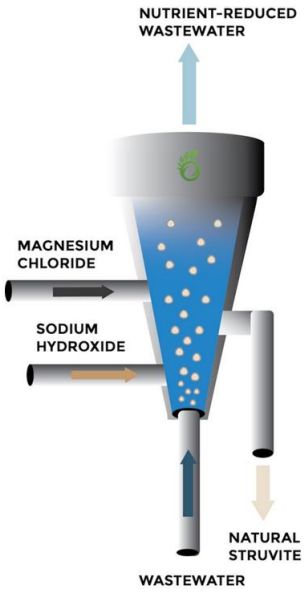

Figure 2-12: Multiform Harvest process (Multiform Harvest, 2015)

2.4 Fate of products generated from WRRFs

As a result of the biological and physical treatment processes, WWTPs generate various products which include greenhouse gasses, sewage sludge and treated effluent. With the view of transitioning towards WRRFs, the need arises to reduce the harm that the traditional pathways of the WWTP products may cause to the environment. In addition to reduction of the polluting effect of those products, it becomes important to consider the potential recovery options, where possible, for the said products for beneficial uses including nutrient recovery. This section thus entails a review of literature pertaining to the fate of products generated from WRRFs (*viz.* greenhouse gases, sewage sludge, nutrients recovered and treated effluent) and their respective potential management options.

2.4.1 Greenhouse gases

Greenhouse gases comprising carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are generated from WWTPs as a result of the various bioprocesses involved in the treatment of wastewater. During the biological treatment of wastewater, organically bound carbon from biodegradable organics is either used by the biomass for cell synthesis or is emitted as CO_2.

Carbon incorporated in the biomass is converted to CO_2 and CH_4 during the anaerobic digestion process. Subsequently, the CH_4 is converted to CO_2 via the combustion of biogas. (Campos et *al.*, 2016)

Anaerobic digestion of PS and WAS contribute to approximately three quarters of the total CH_4 emission from WWTPs. The potentially adverse effects of this CH_4 on the atmosphere is mitigated by using the biogas to generate energy (Daelman *et al.*, 2012). Other than from the AD process, Daelman *et al.* (2012) observed that CH_4 formed in sewers enters the WWTP via the influent at a load of approximately 1% of the influent COD load. A negligible amount of the influent CH_4 is stripped at the head of works while a significant amount (80%) is oxidised in the biological reactor. The remaining unoxidized CH_4 is then emitted from the reactor as a minor contributor to the total emission from the WWTP. The main sources of CH_4 emission at WWTPs are the PS thickener, the centrifuge, the exhaust gas of the cogeneration plant, the buffer tank for digested sludge and the storage tank for dewatered sludge. (Daelman *et al.*, 2012)

Nitrous oxide is biologically produced during the denitrification process at WWTPs in areas of high biological oxygen demand and low oxygen concentration such as influent lines, PSTs, SSTs, sludge holding tanks and sludge transfer lines (Czepiel *et al.*, 1995). It was observed by Czepiel *et al.* (1995) that the N_2O generated is stripped from the liquid as it is agitated by aeration and from turbulent flows in grit tanks for instance. From the study, it was noted that a significant portion of the total N_2O emission at WWTPs emanated from the aerobic tanks.

Campos *et al.* (2016) presented three strategies for the reduction greenhouse gases emission from WWTPs — (1) minimisation, (2) treatment, and (3) prevention.

2.4.1.1 Minimisation

Minimisation of greenhouse gas emissions is achieved by firstly identifying the operational factors contributing to the emissions and then modifying these operational conditions to reduces the emissions. (Campos *et al.*, 2016)

It is known that operational conditions that favour N_2O emissions include: (1) low DO concentration in the aerobic zone for nitrification and presence of DO in the anoxic zone during denitrification, (2) high nitrite concentrations during the biological nitrogen removal processes, (3) low COD/N ratio during denitrification, (4) sudden shifts in pH, DO, NH_3 and nitrite

concentrations, and (5) transient anoxic and aerobic conditions (Kampschreur *et al.*, 2009). N2O minimisation can therefore be achieved by operating WWTPs at high SRTs to maintain low ammonia and nitrite concentrations, and larger reactor volumes to reduce the risk of short-term oxygen depletion. (Campos *et al.*, 2016)

CH_4 emissions may be minimised by capturing the emissions from each unit of the sludge line using hoods. The captured biogas can then the burned together with the biogas generated in the AD and hence contribute towards energy generation at the WWTP. (Campos *et al.*, 2016)

It has been established that the breakdown of organics in the AS system and the combustion of biogas directly contribute to CO_2 emissions from WWTPs. The SRT of the AS system is a key factor with regards to CO_2 emissions. A high SRT would result in higher CO_2 emissions from the biological reactor due to the endogenous respiration process, and concurrently a lower CO_2 emission from the combustion of biogas as a result of a smaller sludge wastage (Campos *et al.*, 2016). A decrease in CO_2 emissions can be achieved by operating the WWTP at the shortest SRT possible without compromising effluent quality as shown be Campos *et al.* (2016).

2.4.1.2 Treatment

The second possible greenhouse gas management option involves capture and treatment. While various technologies are available, the need remains to develop efficient low-cost treatment technologies to make it a viable option. (Campos *et al.*, 2016)

N_2O has been shown to be successfully removed at high efficiencies (>75%) using biofilter technologies based on denitrification by nitrifying and denitrifying bacteria. However, achieving high removal efficiencies of N_2O requires high hydraulic retention times (HRT) due to greenhouse gases having low aqueous solubility (which limits transfer of greenhouse gases to the liquid phase). Consequently, large biofilter volume are required resulting in increased capital costs (Campos *et al.*, 2016). Alternatively, N_2O can be collected from the gas flow above the aerobic zone and used as an oxidiser in the combustion of methane. (Campos *et al.*, 2016)

CH_4 can be removed in biofilter systems under aerobic conditions by the action of methanotrophic bacteria in a process whereby the CH_4 is converted to CO_2. Alternatively, bacteria are able to oxidise CH_4 using sulphate, nitrite or nitrate as electron acceptors in anoxic

conditions (Campos *et al.*, 2016). Similar to the N_2O case, biofilters used in the removal of CH_4 are required to operate at high HRTs due to the low solubility of CH_4.

Microalgae cultivation is regarded as an attractive means of CO_2 sequestration. CO2 can be sequestered by microalgae pre-combustion as a means of biogas purification, or post-combustion thereby maximising microalgae growth for use as biofertilizer or as substrate to increase biogas production. (Campos *et al.*, 2016)

2.4.1.3 Prevention

Campos *et al.* (2016) suggests that prevention would require that WWTPs in the future prioritise anaerobic pathways for organic removal, usage of microalgae or partial nitritation-Anammox for ammonia removal.

2.4.2 Sewage sludge

Sewage sludge generated from WWTPs is essentially a mixture of organic matter, water, both dead and alive microorganisms, and organic and inorganic toxic pollutants. Modern sludge management, which revolves around sustainability principles, favours the beneficial use of sewage sludge over the traditional unsustainable options such as use of sewage sludge as landfill. More sustainable management options include (1) utilisation of sludge for its calorific energy value, (2) utilisation of the useful constituents of sewage sludge (carbon and nutrients) and (3) extraction of useful constituents such as phosphorus from sewage sludge (Snyman & Herselman, 2006a).

While the landfill pathway for sludge is a simple and economical management option, there are significant downsides to this option. At landfills, the biodegradable organics in the sludge disposed get broken down anaerobically releasing CO_2 and CH_4 greenhouse gases into the atmosphere. The organically bound nutrients are also released as leachate and can potentially contaminate the surrounding environment including groundwater and surface water.

Agricultural use of sewage sludge is deemed to be the one of the preferred sludge management options (Snyman & Herlselman, 2006a). With the nutrient content of sewage sludge, its use as fertiliser was shown to be promising as crop yields were found to be similar to

yields achieved with chemical fertiliser at equivalent nitrogen rate applied. Additionally, the biological and physical properties of soil showed improvements with the use of sewage sludge as fertiliser. (Suss, 1994)

Depending on the classification of the sludge, it may not be suitable for use in agricultural practices. In this scenario, the remaining two sustainable pathways can be considered as possible option for sludge management (Snyman & Herselman, 2006a). One such alternative pathway is incineration. The advantages of incineration comprise the significant reduction in sludge mass, the complete destruction of pathogens and other harmful substances, and is a means of recovering energy (Đurđević *et al.*, 2019). While the generation of electricity from sewage sludge incineration does contribute to greenhouse gas emissions, Đurđević et al. (2019) noted that the emissions compared to that of combined heat and power (CHP) systems using natural gas and hard coal power plants are 58% and 80% less respectively.

2.4.3 Nutrients recovered

The increasing world population and the resulting rise in demand for food supply, causes the demand for nutrients to increase accordingly. The latter is not helped by the fact that the Earth's nutrient reserves are fast depleting. Indeed, the P used is primarily obtained from non-renewable phosphorus rock and is predicted to be depleted within 50-100 years (Daneshgar *et al.*, 2018; Mayer *et al.*, 2016; van der Hoek et *al.*, 2018). In addition to that, the fact that phosphorus rock is geographically concentrated in a few countries increases cost of food production for countries requiring import of fertilisers (Mayer *et al.*, 2016). Due to the abovementioned issues regarding rock phosphates, recovery of nutrients from wastewater is becoming an area of increasing interest. The various methods of nutrient recovery from wastewater include (1) chemical crystallisation, (2) gas stripping and absorption, acidic air scrubbing, (4) membrane separation and (5) biomass production and harvest (Vanrolleghem & Vaneeckhaute, 2014).

Nutrients can be recovered from wastewater in the form of struvite. The latter is a slow-release fertiliser that has been thoroughly studied with regards to its solubility, inorganic fertiliser equivalence, pathogen and metal content, salinity and nutrient availability (Sikosana *et al.*, 2016). Sikosana *et al.* (2016) noted that struvite is likely to be within toxicity, pathogen and metal content regulations in spite of the associated health and safety concerns and is comparable to

most phosphate fertiliser on the market. While organic agriculture is not a viable market route for struvite recovered, more viable alternatives could be commercial inorganic fertiliser production, animal feed and community scale gardening (Sikosana *et al.*, 2016).

Kim *et al.* (2021) proposes that instead of using struvite as fertiliser (which brings low revenues), a more economically viable alternative is to use struvite in the manufacture of fire retardants. The performance of struvite-based fire retardants was found to be comparable to the conventional polyphosphate retardants. Struvite-based retardants, which require 60% less total phosphorus than the conventional retardants, can aid in reducing the dependency on rock phosphates which is a finite and fast depleting resource (Kim *et al.*, 2021). The study by Kim *et al.* (2021) indicates that in the United States of America, WWTPs can produce sufficient struvite-based fire retardants to meet the demand for wildfire management and bring significantly greater revenue than struvite fertiliser sales. While using struvite in the manufacture of fire retardants looks like an attractive option, its viability in different countries will depend on the need and demand for such products.

2.4.4 Treated effluent

The common practice for treated effluent is its discharge into water bodies. In most countries, less than 10% of the effluent from WWTPs is recycled (Mayer *et al.*, 2016), with the most common form of reuse being through agriculture (Drechsel *et al.*, 2015). With water scarcity becoming a growing concern worldwide, the need for water reclamation becomes important to alleviate the stress on existing potable water sources. There are many potential applications of wastewater effluent reuse that help in decreasing potable water use. These applications, each having their respective water quality standards to be met, include agricultural reuse, industrial reuse, urban reuse, and indirect and direct potable reuse. Current public perception and potential health hazards are hurdles yet to be overcome when it comes to potable uses of treated effluent. Until there is an improvement in public perception and health hazards are eliminated, the reuse of wastewater effluent will be limited to non-potable applications which exclude its use on crops intended for food production. For successful water reclamation it is also important to ensure the removal of toxic contaminants (from industrial wastewater), heavy metals, pathogens or

micropollutants as these could have a detrimental impact on the environment and on human health.

2.5 Evaluation of design or control strategies

Numerous wastewater treatment technologies are available to meet almost any effluent quality regulation. However, the obstacle to meeting the requirements remains the cost of the treatment processes. As effluent quality required become higher, the costs to meet these requirements escalate (Vanrolleghem *et al.*, 1996). Cost-efficiency of WWTPs is generally dependent on the process choices and dimensioning (Benedetti, Bixio & Vanrolleghem, 2006). Studying the various possible process or control choices that can be implemented at WWTPs allows the investigation of their potential financial benefits while mitigating negative environmental impact (De Ketele *et al.*, 2018). Assessment of these control strategies can be done using the evaluative performance indices adopted by the IWA BSM task group (Jeppsson *et al.*, 2007; De Ketele *et al.*, 2018).

2.5.1 Effluent quality index (EQI)

The EQI is a means of methodically describing the effluent quality at a WWTP (De Ketele et *al.*, 2018). It quantifies the pollution load to a receiving water body into a single term by applying weighting factors to the pollutants based on their relative environmental impact (Copp, 2002; De Ketele *et al.*, 2018).

The EQI used in the BSM2 is formulated as shown in Eq. 2-6 below, with the weighting factors for each pollutant shown in Table 2-1 (Copp, 2002; Jeppsson *et al.*, 2007). The original pollutant weightings, determined based on Vanrolleghem *et al.* (1996), allocated to nitrate and ammonium were both 20. With this weight applied, there is no incentive to prioritise nitrification. Because the environmental impact of ammonia is more severe than nitrates, the weights were modified to promote removal of ammonia (Nopens *et al.*, 2010; Gernaey *et al.*, 2014) as shown in Table 2-1.

$$EQI = \frac{1}{T \cdot 1000} \int_{t=245\ days}^{t=609\ days} \left(\beta_{TSS} \cdot TSS_e(t) + \beta_{COD} \cdot COD_e(t) + \beta_{TKN} \cdot S_{TKN,e}(t) + \beta_{NO} \cdot S_{NO,e}(t) + \beta_{BOD5} \cdot BOD_{5,e}(t)\right) \cdot Q_e(t) \cdot dt \quad \textbf{Eq. 2-6}$$

Where:

T : Total length of evaluation period (days)

β : Pollutant weighting factor

Q_e : Effluent flow rate (m^3/d)

Table 2-1: Pollutant weighting factors

Weighting factor	BSM2	BSM2-modified
β_{TSS}	2	2
β_{COD}	1	1
β_{TKN}	20	30
β_{NO}	20	10
β_{BOD5}	2	2

De Ketele *et al.* (2018) modified the existing EQI as shown in Eq. 2-7. With this new formula for EQI, the pollutant weighting factor is now determined based on site-specific regulations with regards to effluent emission limits. These weighting factors are calculated using the COD emission limit as a reference point. The weighting factor for COD is thus always equal to 1 while the weighting factor for ortho-phosphates (OP), for instance, is the ratio of the COD emission limit to the OP emission limit. The new weighting factors used by De Ketele *et al.* (2018), shown in Table 2-2 below, are based on the Water Act No. 54 of 1956 Regulation No. 991 (1984).

With the formula set up in this format, when a pollutant is above the emission limit, its component (i.e, β*(Limit - pollutant(t)) in the formula will be negative. In the case where the effluent concentration is within the limit, the component in the formula will be positive. Since each pollutant is independent of each other, there is a possibility that some terms cancel each

other out. It is therefore required that the negative terms and positive terms are grouped separately into EQI negative and EQI positive (De Ketele *et al.*, 2018).

An EQI negative equal to zero implies that the regulatory limits have not been exceeded by any of the pollutants (acceptable effluent). Improvement in effluent quality is translated to an increase in the EQI positive value. This is because the better the effluent quality, the lower the pollutant concentration and hence, the larger the difference between the limiting concentration and the actual concentration becomes (De Ketele *et al.*, 2018).

Conversely, an EQI positive of zero would imply that the concentration of pollutants in the effluent have all exceeded their respective regulatory limits (EQI negative <0).

If EQI positive and EQI negative are both not zero, it means that the regulatory emission limit for at least one pollutant has been exceeded. Analysis of the data can then help identifying the pollutant above the discharge limit. When striving for better effluent quality, the emphasis should be to reduce to value of EQI negative to as close to zero as possible (De Ketele *et al.*, 2018).

$$EQI = \frac{1}{T \cdot 1000} \int_{t=245\ days}^{t=609\ days} (\beta_{TSS} \cdot (TSS_{limit} - TSS(t)) + \beta_{COD} \cdot (COD_{limit} - COD(t)) + \beta_{FSA} \cdot (FSA_{limit} - FSA(t)) + \beta_{NO} \cdot (NO_{limit} - NO(t)) + \beta_{OP} \cdot (OP_{limit} - OP(t))) \cdot Q_e(t) \cdot dt \quad \textbf{Eq. 2-7}$$

Table 2-2: Modified EQI pollutant weighting factors

Pollutant	Concentration limit (mg/l)	Weighting factor (β)
COD	30	1
FSA	1	30
OP	1	30
NO	1.5	20
TSS	10	3

2.5.2 Operational cost index (OCI)

Design and operation cost analysis of WWTPs is performed using OCI, which includes cost factors pertaining to operation and potential savings that could be realised through strategic plant design or control operations. The OCI gives an overview of the variables that have the most substantial impact on the total costs at WWTPs (De Ketele *et al.*, 2018).

Originally, only 3 criteria were used to assess the costs at WWTPs in the BSM1- sludge production, aeration energy and pumping energy (Copp, 2002). However, with the development of the BSM2, more variables were considered when evaluating cost, pertaining to the new unit processes added to the model. The formula used by the BSM2 is described in Eq. 2-8 (Jeppsson *et al.*, 2007; Nopens *et al.*, 2010).

$$OCI = AE + PE + 3 \cdot SP + 3 \cdot EC + ME - 6 \cdot MP + HE^{net} \qquad \textbf{Eq. 2-8}$$

Where:

AE : Aeration energy (kWh/d)
PE : Pumping energy (kWh/d)
SP : Sludge produced (kgTSS/d)
EC : External carbon addition (kgCOD/d)
ME : Mixing energy (kWh/d)
MP : Methane produced (kgCH_4/d)
HE^{net} : Net heating energy required by anaerobic digester for sludge treatment (kWh/d)

With no homogeneity in the units used in Eq. 2-8, the resulting OCI is dimensionless (Gernaey *et al.*, 2014). To achieve uniformity in the units, De Ketele *et al.* (2018) modified the formula as shown in Eq. 2-9 below. This method of calculating the OCI group the variables into three categories: energy cost, sludge disposal cost and carbon cost. These costs are naturally country or region dependent. For instance, the unit energy cost will depend on the tariff set by the local authorities.

$$OCI = (AE + PE - MP + ME + HE) \cdot Energy\ cost + SP \cdot Sludge\ disposal\ cost + EC \cdot Carbon\ cost \quad \textbf{Eq. 2-9}$$

Where:

AE : Aeration energy (kWh/d)

PE : Pumping energy (kWh/d)

SP : Sludge produced (kgTSS/d)

EC : External carbon addition (kgCOD/d)

ME : Mixing energy (kWh/d)

MP : Energy from methane produced (kWh/d)

HE : Total heat energy required be anaerobic digester for sludge treatment (kWh/d)

In both Eq. 2-8 and Eq. 2-9, it can be observed that the methane production is the only negative term. This is because methane production is an internal source of energy for WWTPs, and therefore reduces the overall energy demand.

2.5.2.1 Aeration energy

The aeration energy used in the OCI calculation above is computed using Eq. 2-11. This method of determining the contribution of aeration relates the oxygen transfer rate (Eq. 2-10) to power consumption, assuming a Standard Oxygen Transfer Efficiency (SOTE) of 1.8 kgO_2/kWh (Nopens *et al.*, 2010). This negates the dominating effect that aeration energy had on the OCI calculation (Nopens *et al.*, 2010; Gernaey *et al.*, 2014) formerly used in the BSM1 (Copp, 2002).

$$OTR = V \cdot K_L a_{15} \cdot (S_{O,15}^{sat} - 0)/1000 \quad \textbf{Eq. 2-10}$$

$$AE = \frac{S_{O,15}^{sat}}{T \cdot 1.8 \cdot 1000} \int_{t_{start}}^{t_{end}} \sum_i V_i \cdot K_L a_{i,15}(t) dt \quad \textbf{Eq. 2-11}$$

Where

OTR : Oxygen transfer rate (kgO_2/d)

V_i : Volume of reactor *i* (m^3)

$K_L a_{i,15}$: Oxygen transfer coefficient in reactor *i* at 15°C (/d)

$S_{O,15}^{sat}$: Saturated dissolved oxygen at 15°C (=8 gO_2/m^3)

AE : Aeration energy (kWh/d)

T : Total length of evaluation period (days)

2.5.2.2 Pumping energy

The pumping energy calculated as shown in Eq. 2-12 is essentially a sum of pumped flows, with the pumping energy conversion coefficients factored in, over the length of the evaluation period (Gernaey *et al.*, 2014). It is an extension of the original BSM1 definition (Copp, 2002), such that more pumped flows in the system are included in the calculation and each flow has its own weighting factor.

$$PE = \frac{1}{T} \int_{t_{start}}^{t_{end}} \left(f_{PE,int} \cdot Q_{int} + f_{PE,r} \cdot Q_r + f_{PE,w} \cdot Q_w + f_{PE,pu} \cdot Q_{pu} + f_{PE,tu} \cdot Q_{tu} + f_{PE,do} \cdot Q_{do} \right) dt \quad \textbf{Eq. 2-12}$$

Where:

Q_{int} : Internal recycle (m^3/d)

Q_r : Return sludge recycle (m^3/d)

Q_w : Waste sludge (m^3/d)

Q_{pu} : Primary clarifier underflow (m^3/d)

Q_{tu} : Thickener underflow (m^3/d)

Q_{do} : Dewatering overflow (m^3/d)

PE : Pumping energy (kWh/d)

f_{PE} : Pumping energy conversion factors (kWh/m^3)

T : Total length of evaluation period (days)

Table 2-3: Pumping energy conversion factors (Gernaey *et al.*, 2014)

Parameter	Value
$f_{PE,int}$	0.004
$f_{PE,r}$	0.008
$f_{PE,w}$	0.050
$f_{PE,pu}$	0.075
$f_{PE,tu}$	0.060
$f_{PE,do}$	0.004

2.5.2.3 Energy from methane production

Energy is harnessed from methane production via a gas motor, which converts the gas flow originating from the anaerobic digester to electricity and heat (Jeppsson *et al.*, 2007). As a result, the energy generated from methane production is calculated using Eq. 2-13, in which the efficiency of the motor and the energy density of methane are factored (De Ketele *et al.*, 2018; Gernaey *et al.*, 2014).

$$MP = \frac{ED_{CH4} \cdot \varepsilon_{motor} \cdot P_{atm} \cdot 16}{T \cdot 1000 \cdot R \cdot T_{op}} \int_{t_{start}}^{t_{end}} \frac{1}{p_{gas,tot}(t)} \cdot p_{gas,CH4}(t) \cdot Q_{gas}(t) \cdot dt \quad \textbf{Eq. 2-13}$$

Where:

MP : Methane production energy (kWh/d)

ED_{CH4} : Energy density of methane (=15.42 kWh/kg)

ε_{motor} : Efficiency of motor

T : Total length of evaluation period (days)

P_{atm} : Atmospheric pressure (=1.013 bar)

R : Universal gas constant (=8.3145×10^{-2} bar · m^3 · $kmol^{-1}$ · K^{-1})

T_{op} : Operating temperature of AD (=308.15 K)

$p_{gas,tot}$: Total gas pressure (bar)

$p_{gas,CH4}$: Partial pressure of CH_4 produced (bar)

Q_{gas} : Normalised gas flow rate (m^3/d)

2.5.2.4 Mixing energy

Mixing in activated sludge reactor is only required if the oxygen transfer coefficient for that reactor is less than 20 d^{-1}. It follows that reactors with oxygen transfer coefficient greater than 20 d^{-1} are provided sufficient mixing from the aeration process. On the other hand, for anaerobic digesters, it is assumed that mixing is constant (Gernaey *et al.*, 2014).

The total mixing energy at a WWTP comprises mixing in the activated sludge reactor and in the anaerobic digester, and can be calculated using Eq. 2-14, Eq. 2-15 & Eq. 2-16.

$$ME = ME_{AS} + ME_{AD} \quad \textbf{Eq. 2-14}$$

$$ME_{AS} = \frac{24}{T} \int_{t_{starts}}^{t_{end}} \sum_{i} \begin{bmatrix} for\ t\ when\ K_L a_i(t) < 20\ d^{-1}\ then\ f_{ME,AS} \cdot V_i \\ for\ t\ when\ K_L a_i(t) \geq 20\ d^{-1}\ then\ 0 \end{bmatrix} \cdot dt \quad \textbf{Eq. 2-15}$$

$$ME_{AD} = 24 \cdot f_{ME,AD} \cdot V_{AD} \quad \textbf{Eq. 2-16}$$

Where:

ME : Total mixing energy (kWh/d)

ME_{AS} : Mixing energy for activated sludge reactor (kWh/d)

ME_{AD} : Mixing energy for anaerobic digester (kWh/d)

T : Total length of evaluation period (days)

$K_L a_i$: Oxygen transfer coefficient in reactor *i* (/d)

$f_{ME,AS}$: Mixing power consumption factor for activated sludge tanks (=0.005 kW/m^3)

$f_{ME,AD}$: Mixing power consumption factor for anaerobic digester (=0.005 kW/m^3)

V_i : Volume of the i^{th} tank (m^3)

V_{AD} : Liquid volume of the anaerobic digester (m^3)

2.5.2.5 Heating energy

The heating energy required for anaerobic digestion, calculated using Eq. 2-17, is the average energy required to increase the temperature of the anaerobic digester influent to match the operational temperature of the digester (Gernaey *et al.*, 2014).

$$HE = \frac{24}{T \cdot 86400} \int_{t_{start}}^{t_{end}} \rho_{H2O} \cdot c_{H2O} \cdot (T_{op} - T_{AD,i}(t)) \cdot Q_{AD}(t) \cdot dt \qquad \textbf{Eq. 2-17}$$

$$T_{AD,i}(t) = \frac{T_{pu}(t) \cdot Q_{pu}(t) + T_{tu}(t) \cdot Q_{tu}(t)}{Q_{AD}(t)} \qquad \textbf{Eq. 2-18}$$

$$Q_{AD}(t) = Q_{pu}(t) + Q_{tu}(t) \qquad \textbf{Eq. 2-19}$$

Where:

HE : Heating energy (kWh/d)

T : Total length of evaluation period (days)

ρ_{H2O} : Density of water (=1000 kg/m^3)

c_{H2O} : Specific heat capacity of water (=4.186 $kJ \cdot kg^{-1}K^{-1}$)

T_{op} : Operating temperature of AD (=308.15 K or 35℃)

$T_{AD,i}$: Temperature of anaerobic digester influent (K or ℃)

Q_{AD} : Flow rate to the anaerobic digester (m^3/d)

T_{pu} : Temperature of primary clarifier underflow (K or ℃)

Q_{pu} : Primary clarifier underflow rate (m^3/d)

T_{tu} : Temperature of thickener underflow (K or ℃)

Q_{tu} : Thickener underflow rate (m^3/d)

2.5.2.6 External carbon addition

The mass of external carbon added daily is calculated using Eq. 2-20 below (Gernaey *et al.*, 2014).

$$EC = \frac{COD_{EC}}{T \cdot 1000} \int_{t_{start}}^{t_{end}} Q_{EC}(t) \cdot dt \qquad \textbf{Eq. 2-20}$$

Where:

EC : External carbon added (kgCOD/d)

COD_{EC} : External carbon source concentration (gCOD/m^3)

T : Total length of evaluation period (days)

Q_{EC} : Sum of flow rate of external carbon source into the reactor (m^3/d)

2.5.2.7 Sludge production

The sludge production for disposal, as used in the BSM2, is calculated using Eq. 2-21. Since the BSM2 is an extension of BSM1 with the inclusion of primary clarifier and anaerobic digestion in addition to existing bioreactors and secondary clarifier (Jeppsson *et al.*, 2007), the sludge produced for disposal no longer refers to the waste activated sludge. Instead, it refers to the dewatered sludge mass, which comprises each of the components described by Eq. 2-22 (Gernaey *et al.*, 2014).

$$SP = \frac{1}{T \cdot 1000}\left(M_{TSS}(t_{end}) - M_{TSS}(t_{start}) + \int_{t_{start}}^{t_{end}} TSS_x(t) \cdot Q_x\,(t) \cdot dt \right) \qquad \textbf{Eq. 2-21}$$

$$M_{TSS,x}(t) = M_{TSS,AS}(t) + M_{TSS,SC}(t) + M_{TSS,PC}(t) + M_{TSS,AD}(t) + M_{TSS,SS}(t) \qquad \textbf{Eq. 2-22}$$

Where:

SP : Sludge produced for disposal (kg/d)

T : Total length of evaluation period (days)

$M_{TSS}(t_{end})$: Total mass of solids at the end of evaluation period (kg)

$M_{TSS}(t_{start})$: Total mass of solids at the start of evaluation period (kg)

TSS_x : Total settleable solids concentration in sludge flow (g/m^3)

Q_x : Waste sludge flow rate (m^3/d)

AS : Activated sludge reactors

SC : Secondary clarifier

PC : Primary clarifier

AD : Anaerobic digester

SS : Sludge storage tank

2.6 Closure

Wastewater treatment is an integral part of the urban water cycle, mitigating harmful impact on downstream water bodies and on human health. During wastewater treatment process, the constituents of influent wastewater undergo physical and biological transformations with the exception of soluble constituents that are not biologically utilisable escaping with the effluent. The removal of pollutants from influent is achieved through several steps: (1) primary treatment involving the removal of large solids from the influent by screening and degritting and removal of settleable solids by primary sedimentation; (2) secondary treatment involving the biological removal of biodegradable organic material and nutrients from wastewater through different WWTP configurations (such as UCT and JHB systems) that include anaerobic, anoxic and aerobic zones in the reactor; (3) the mixed liquor from the biological reactor is discharged into a clarifier for secondary sedimentation resulting in a clear effluent void of solids; (4) tertiary treatment involving the disinfection of the effluent prior to discharge into water bodies; (5) sludge treatment, most commonly by anaerobic digestion, which is essentially the stabilisation of wastewater sludge such that a reduction in the biodegradable fraction is achieved which results in reduced risk of putrefaction, vector attraction and pathogenic organisms.

Sludge dewatering liquor, a by-product of the sludge treatment process, contains high concentrations of ammonia and ortho-phosphates and therefore cannot be discharged into water

bodies. The dewatering liquor, which is often recycled back to the head of works, can potentially overload the system resulting in non-compliance to effluent regulations. Additionally, the latter can also result in the increase in operational costs due to increased aeration requirement and external addition of readily biodegradable organics. Side stream treatment of the sludge dewatering liquor allows the mitigation of the abovementioned negative impacts as well as the recovery of nutrients in useful forms such as agricultural fertiliser. Treatment technologies that hold potential for implementation in South Africa include, but are not limited to, ANAMMOX and BABE processes for nitrogen removal, and struvite precipitation for the recovery of both nitrogen and phosphorus nutrients.

WWTPs generate various products which include greenhouse gasses, sewage sludge and treated effluent. With the view of transitioning towards WRRFs, the need arises to reduce the harm that the traditional pathways of the WWTP products may cause to the environment while considering potential recovery options for beneficial uses including nutrient recovery. Emission of greenhouse gases can be managed through three strategies — (1) minimisation, (2) treatment, and (3) prevention. Sustainable pathways for sewage sludge include agricultural use as fertiliser and incineration. Nutrients can be recovered in the form of struvite to either be used as a fertiliser or to manufacture fire retardants. Lastly, water reclamation through its various applications helps in decreasing the stress on potable water sources.

Design and operation strategies implemented at WWTPs can be assessed using the evaluative performance indices adopted by the IWA BSM task group known as the EQI and OCI. The EQI is a means of quantifying the pollution load to a receiving water body into a single term by applying weighting factors to the pollutants based on their relative environmental impact. On the other hand, the OCI is a measure of the total costs associated with each unit process pertaining to energy cost, sludge disposal cost and external carbon addition cost.

In an era where a holistic view of the environment is a necessity, it does not suffice to only evaluate the impact of wastewater treatment on water, as is currently the case with the EQI. It is therefore suggested that the term “effluent” needs to be expanded to include other products that may harm the environment, i.e., greenhouse gases and sludge production. This would then require the development of additional EQI protocols for gases and sludge.

Moreover, since the OCI does not include the potential monetary benefits of nutrient recovery, it is inherently inadequate to perform evaluation of WWTPs with side stream nutrient

recovery as this could potentially dissuade decision makers from implementing nutrient recovery technologies. Lastly, it is suggested that the EQI should be included in the OCI formulation such that the design and operation of WWTPs does not result in compromise of effluent quality for lower operating costs.

3. Development of performance indices framework

3.1 Introduction

The evaluation of the performance of WWTPs under different design and operational options is traditionally performed using the performance indices adopted by the IWA BSM task group (Jeppsson *et al.*, 2007). More recently, works by De Ketele *et al.* (2018) provided improvements on the original formulation of the EQI and OCI equations. In spite of the improvements, the performance indices are still lacking from a point of view of resource recovery and holistic environmental impact. This chapter attempts to address the gaps identified through further modification of the EQI and OCI.

3.2 Effluent quality index

The EQI is a useful tool to quantitatively describe effluent quality at WWTPs. It allows for the total pollutant load to a receiving water body to be quantified into a single term by applying weighting factors to each pollutant.

With the aim of becoming more environmentally conscious, it was deemed necessary to take into consideration the impact of wastewater treatment processes on the atmosphere and on land, instead of solely focusing on the impact on water. In light of the latter, three separate EQI equations were developed as described in Sections 3.2.1, 3.2.2 and 3.2.3 below.

3.2.1 EQI water

The discharge of wastewater effluent to water bodies or its reuse by irrigation in South Africa is regulated by the National Water Act, No. 36 of 1998 (1998: s39). The revision of General Authorisation under Section 39 of the Act (*National Water Act, No. 36 of 1998. Regulation*, 2004) specifies the volumes of wastewater effluent that can be discharged to water bodies or used for irrigation within the stipulated limitations of their respective effluent quality requirements (Nozaic & Freese, 2009).

The limiting concentrations of pollutants in wastewater effluent that may discharged into water resources at a rate not exceeding 2 ML/d (i.e., for small WWTPs) are shown in Table 3-1 below. The special limit applies to all the listed water resources under the revision of General Authorisation under Section 39 of the Act (*National Water Act, No. 36 of 1998. Regulation*, 2004).

Table 3-1: Pollutant limit values for wastewater discharges not exceeding 2 ML/d (National Water Act, No. 36 of 1998. Regulation, 2004)

Pollutant/Parameter	General limit	Special limit
Faecal Coliforms (per 100 ml)	1 000	0
Chemical Oxygen Demand (mg/l)	75	30
pH	5.5-9.5	5.5-7.5
Ammonia (ionised and un-ionised) as Nitrogen (mg/l)	36	2
Nitrate/Nitrite as Nitrogen (mg/l)	15	1.5
Chlorine as Free Chlorine (mg/l)	0.25	0
Suspended Solids (mg/l)	25	10
Electrical Conductivity (mS/m)	70 mS/m above intake to a maximum of 150 mS/m	50 mS/m above background receiving water, to a maximum of 100 mS/m
Ortho-Phosphate as phosphorus (mg/l)	10	2.5
Fluoride (mg/l)	1	1
Soap, oil or grease (mg/l)	2.5	0
Dissolved Arsenic (mg/l)	0.02	0.01
Dissolved Cadmium (mg/l)	0.005	0.001
Dissolved Chromium (VI) (mg/l)	0.05	0.02
Dissolved Copper (mg/l)	0.01	0.002

Pollutant/Parameter	General limit	Special limit
Dissolved Cyanide (mg/l)	0.02	0.01
Dissolved Iron (mg/l)	0.3	0.3
Dissolved Lead (mg/l)	0.01	0.006
Dissolved Manganese (mg/l)	0.1	0.1
Mercury and its compounds (mg/l)	0.005	0.001
Dissolved Selenium (mg/l)	0.02	0.02
Dissolved Zinc (mg/l)	0.1	0.04
Boron (mg/l)	1	0.5

The EQI_{water} is formulated as shown in Eq. 3-1 below. The original EQI expression (See Eq. 2-6), developed by Jeppsson *et al*. (2007), calculates the impact of effluent discharge on water bodies as a sum over time of the amount of each pollutant discharged per day. The EQI_{water} expression of the current EQI formulation is similar to the original EQI expression, with some added modifications, which include:

1. The addition of the term X(t). This term represents any other pollutant that may be taken into consideration in future evaluations where more advanced models can predict and track more pollutants than current models do; and
2. Allowing flexibility in the calculation of pollutant weighting factors. This acknowledges that different plants or regions within the same country (or different countries) may have different effluent goals, thus allowing the EQI to be a better representation of plant performance with respect to plant-specific or regional goals. The weighting factors are calculated based on the environmental impact of each pollutant with respect to COD. (e.g $\beta_{TSS} = \frac{COD\ limit}{TSS\ limit}$)

$$EQI_{water} = \frac{1}{T \cdot 1000} \int_{t_0}^{t_{end}} \left(\beta_{TSS} \cdot TSS(t) + \beta_{COD} \cdot COD(t) + \beta_{FSA} \cdot FSA(t) + \beta_{NO} \cdot NO(t) + \beta_{OP} \cdot OP(t) + \beta_{x} \cdot X(t) \right) \cdot Q_e(t) \cdot dt \quad \textbf{Eq. 3-1}$$

To illustrate the above, two types of WWTPs are considered—Plant A being a small WWTP with effluent limits stipulated in Table 3-1, and a larger WWTP Plant B with stricter effluent quality standards based on the special standard of the Water Act No. 54 of 1956 Regulation No. 991 (1984). From Table 3-2, it can be seen that the Beta weighting factors for each plant are different. This allows the evaluation of the effluent quality using Eq. 3-1 to be tailored towards the effluent quality goals specific to each plant.

Table 3-2: Comparison of Beta factors for different WWTPs

Parameter	**Plant A**		**Plant B**	
	C_{limit} (mg/l)	**Beta**	**C_{limit} (mg/l)**	**Beta**
COD	30	1	30	1
FSA	2	15	1	30
OP	2.5	12	1	30
NO	1.5	20	1.5	20
TSS	10	3	10	3

While in the example above, the weighting factors for the EQI calculation were determined on the premise that the effluent would be discharged to water bodies, it is acknowledged that different applications of water reclamation from WRRFs have different standards to be met. The flexibility in the weighting factors and the term X(t) (which is intended to be used as a generic term for any additional component/pollutant to be considered in the EQI calculation) allow for application specific EQI evaluations.

3.2.2 EQI gas

Greenhouse gases, gases that trap heat in the atmosphere, are regarded as a major driver of human-induced climate change. Greenhouse gases comprise carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) and fluorinated gases such as chlorofluorocarbons. The impact of each greenhouse gas on climate change is essentially dependent on three factors:

i) The concentration of greenhouse gases in the atmosphere

ii) The length of time the gases remain in the atmosphere

iii) The Global Warming Potential (GWP) of each gas (EPA, 2017).

The expression for EQI_{gas} specific to the impact of wastewater treatment processes on the atmosphere is formulated as shown in Eq. 3-2 below. This expression includes greenhouse gases that evolve from WWTPs bioprocesses such as CO_2, CH_4 and N_2O that escape into the atmosphere. The weighting factors assigned to each gas are set equal to their respective average global warming potentials (Table 3-3)—a measure of the amount of heat greenhouse gases trap relative to CO_2 (EPA, 2017).

$$EQI_{gas} = \frac{1}{T \cdot 1000} \int_{t_0}^{t_{end}} \left(\beta_{CO_2} \cdot FCO_2(t) + \beta_{CH_4} \cdot FCH_4(t) + \beta_{N_2O} \cdot FN_2O(t) \right) \cdot dt \quad \textbf{Eq. 3-2}$$

Where:

T : Total length of evaluation period (days)

β : Pollutant weighting factor

F : Flux of gas evolved (kg/d)

Table 3-3: Pollutant weighting factors (gas)

Gas	GWP range	Beta value
CO_2	1	1
CH_4	28-36	32
N_2O	265-298	281

3.2.3 EQI sludge

The handling of sludge generated at South African WWTPs is regulated by the *Guidelines for the Utilisation and Disposal of Wastewater Sludge* (Snyman & Herselman, 2006a). The

guidelines offer various sustainable sewage sludge management options dependent on the quality of the sludge. The selection of appropriate sludge management options requires the classification of the sludge. More specifically, characteristics of sewage sludge pertaining to microbiological parameters, stability indicators and pollutant content need to be determined.

The microbiological classification is a function of the concentration of faecal coliforms and helminth ova in the sludge. It is divided into three classes A, B and C, where class A sludges have the least allowable concentrations and class C sludges have the highest allowable concentrations. (Snyman & Herselman, 2006a)

The stability classification is deemed to be of high importance when it comes to beneficial use of sewage sludge. This is because odour is recognised as the most important factor that has an impact on public perception. Since the amount of volatile solids in the sludge directly influences the odour, it is desirable to maximise reduction of volatile solids. The stability is divided into three classes (1, 2 and 3), where each class is determined by the level of compliance to a 38% minimum reduction of volatile solids. (Snyman & Herselman, 2006a)

The pollutant classification is related to the concentration of potentially toxic metals and elements in the sludge which limit the applications of sewage sludge. The pollutant characterisation is split into three classes a, b and c, with class a having the lowest concentration limits and class c having the highest concentration limit. (Snyman & Herselman, 2006a)

Table 3-4 below gives a summary of every possible sludge classification and their suitability relative to each available management options. In this diagram, “i” indicates that for this specific sludge characteristic class (i.e., microbiological, stability or pollutant), usage of the sludge is permitted without any restriction; “ii” indicates that the sludge may be used with restrictions and good management practices; “iii” indicates that the sludge may only be used under strict restrictions involving costly and major management practices; “iv” indicates that the sludge may not be used unless unique conditions are met; and “v” indicates that usage of the sludge is not permitted. While each sludge management option has distinct sludge characteristic requirements, the preferred option as stipulated in the guidelines is agricultural use (Snyman & Herselman, 2006a). It can be seen that the poorest quality sludge that may be used in agriculture with restriction is B2b classification, which would be a suitable basis for the formulation of a EQI for sludge. However, since the goal is to transition towards resource recovery, it is important to consider options such as beneficial use of sludge to rehabilitate degraded soils (Herselman &

Moodley, 2009), and the use of sludge to produce commercial products such as fertiliser or certain construction materials (Herselman *et al.*, 2009). From Table 3-4, it can be seen that sludge with classification A1a is the only one that may be used in agriculture, for beneficial use and to produce saleable products. Therefore, A1a classification is most suitable for use as a basis for the formulation of the EQI for sludge (i.e., to benchmark WRRFs that include sludge re-use).

Table 3-4: Management options for each sludge classification (Adapted from Snyman & Herselman, 2006a: 31)

Sludge classification	Available management options for each sludge classification														
	Agricultural use at agronomic rates			On-site or off-site disposal			Beneficial use (other than agricultural use at agronomic rates)			Thermal treatment methods			Produce saleable products		
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
A1a	i	i	i	iii	i	iv	i	ii	i	v	iii	i	i	i	i
A1b	i	i	ii	iii	i	iii	i	ii	iii	v	iii	ii	i	i	iii
A1c	i	i	v	iii	i	iii	i	ii	iv	v	iii	ii	i	i	iii
A2a	i	ii	i	iii	ii	iv	i	iii	i	v	ii	i	i	iv	i
A2b	i	ii	ii	iii	ii	iii	i	iii	iii	v	ii	ii	i	iv	iii
A2c	i	ii	v	iii	ii	iii	i	iii	iv	v	ii	ii	i	iv	iii
A3a	i	v	i	iii	iv	iv	i	iv	i	v	i	i	i	v	i
A3b	i	v	ii	iii	iv	iii	i	iv	iii	v	i	ii	i	v	iii
A3c	i	v	v	iii	iv	iii	i	iv	iv	v	i	ii	i	v	iii
B1a	ii	i	i	iii	i	iv	iii	ii	i	iv	iii	i	iv	i	i
B1b	ii	i	ii	iii	i	iii	iii	ii	iii	iv	iii	ii	iv	i	iii
B1c	ii	i	v	iii	i	iii	iii	ii	iv	iv	iii	ii	iv	i	iii
B2a	ii	ii	i	iii	ii	iv	iii	iii	i	iv	ii	i	iv	iv	i
B2b	ii	ii	ii	iii	ii	iii	iii	iii	iii	iv	ii	ii	iv	iv	iii
B2c	ii	ii	v	iii	ii	iii	iii	iii	iv	iv	ii	ii	iv	iv	iii
B3a	ii	v	i	iii	iv	iv	iii	iv	i	iv	i	i	iv	v	i
B3b	ii	v	ii	iii	iv	iii	iii	iv	iii	iv	i	ii	iv	v	iii
B3c	ii	v	v	iii	iv	iii	iii	iv	iv	iv	i	ii	iv	v	iii
C1a	iv	i	i	i	i	iv	iv	ii	i	i	iii	i	v	i	i
C1b	iv	i	ii	i	i	iii	iv	ii	iii	i	iii	ii	v	i	iii

Sludge classification	Available management options for each sludge classification														
	Agricultural use at agronomic rates			On-site or off-site disposal			Beneficial use (other than agricultural use at agronomic rates)			Thermal treatment methods			Produce saleable products		
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
C1c	iv	i	v	i	i	iii	iv	ii	iv	i	iii	ii	v	i	iii
C2a	iv	ii	i	i	ii	iv	iv	iii	i	i	ii	i	v	iv	i
C2b	iv	ii	ii	i	ii	iii	iv	iii	iii	i	ii	ii	v	iv	iii
C2c	iv	ii	v	i	ii	iii	iv	iii	iv	i	ii	ii	v	iv	iii
C3a	iv	v	i	i	iv	iv	iv	iv	i	i	i	i	v	v	i
C3b	iv	v	ii	i	iv	iii	iv	iv	iii	i	i	ii	v	v	iii
C3c	iv	v	v	i	iv	iii	iv	iv	iv	i	i	ii	v	v	iii
1 = Microbiological class 2 = Stability class 3 = Pollution class															

Table 3-5, Table 3-6 and Table 3-7 below show the characteristics of wastewater sludge with A1a classification.

Table 3-5: Microbiological class A (Adapted from Herselman *et al.*, 2009:45)

Microbiological class A	Target value	Maximum permissible value
Faecal coliform (CFU/g_{dry})	<1000	10 000
Helminth $_{ova}$ (Viable ova/g_{dry})	<0.25 (or 1 ova/4g_{dry})	1
Compliance requirements	90% compliance	The 10% samples that exceed the Target value may not exceed this value

Table 3-6: Stability class 1 (Adapted from Herselman *et al.*, 2009:46)

Stability class 1	
Product must **always** comply with one of the options below.	
Applicable Vector attraction reduction options	
Option 1	Reduce the mass of volatile solids by a minimum of 38 percent
Option 2	Demonstrate vector attraction reduction with additional anaerobic digestion in a bench-scale unit
Option 3	Demonstrate vector attraction reduction with additional aerobic digestion in a bench-scale unit

Stability class 1	
Product must **always** comply with one of the options below.	
Applicable Vector attraction reduction options	
Option 4	Meet a specific oxygen uptake rate for aerobically treated sludge
Option 5	Use aerobic processes at a temperature greater than 40°C (average temperatures 45°C) for 14 days or longer (e.g., during sludge composting)
Option 6	Add alkaline material to raise pH under specific conditions
Option 7	Reduce moisture content of sludge that do not contain unstabilised solids (from treatment processes other than primary treatment) to at least 75 percent solids

Table 3-7: Pollutant class a (Adapted from Herselman et al., 2009:46)

Aqua regia extractable metals (mg/kg)	**Pollutant class**
	a
Arsenic (As)	<40
Cadmium (Cd)	<40
Chromium (Cr)	<1200
Copper (Cu)	<1500
Lead (Pb)	<300
Mercury (Hg)	<15
Nickel (Ni)	<420
Zinc (Zn)	<2800

The expression used for the evaluation of sludge quality (EQI_{sludge}) is formulated as shown in Eq. 3-3 below. It is based on the consideration of the pollutant class, microbiological class and stability class. The weighting factors used in the expression (Table 3-8) are based on the concentration limits of pollutants and microorganisms in wastewater sludge. Similar to EQI_{water}, the weighting factors in EQI_{sludge} reflect the regional sewage sludge standards and would therefore be different for another region/country.

On the basis of the stability classification described above, a minimum reduction of volatile solids mass of 38% was chosen as the sludge stability criterion for the formulation of the EQI_{sludge}. Thus, the term $\frac{38}{x(t)}$ was introduced in the expression such that any reduction in volatile solids *x(t)* less than 38% will result in an increase in the EQI_{sludge} by a factor greater than 1.

Conversely, a desirable reduction in volatile solids greater than 38% translates into a decrease in the EQI_{sludge} by a factor less than 1.

$$EQI_{sludge} = \frac{1}{T \cdot 1000} \int_{t_0}^{t_{end}} \frac{38}{x(t)} \cdot \left(\beta_{As} \cdot As(t) + \beta_{Cd} \cdot Cd(t) + \beta_{Cr} \cdot Cr(t) + \beta_{Cu} \cdot Cu(t) + \beta_{Pb} \cdot Pb(t) + \beta_{Hg} \cdot Hg + \beta_{Ni} \cdot Ni + \beta_{Zn} \cdot Zn + \beta_{CFU} \cdot CFU + \beta_{ova} \cdot ova\right) \cdot F\ Sludge(t) \cdot dt \quad \textbf{Eq. 3-3}$$

Where:

T : Total length of evaluation period (days)

β : Pollutant weighting factor

F : Flux of sludge produced (kg/d)

Table 3-8: Pollutant weighting factors (sludge)

Weighting factor	Value
β_{As}	70
β_{Cd}	70
β_{Cr}	2.33
β_{Cu}	1.87
β_{Pb}	9.33
β_{Hg}	186.67
β_{Ni}	6.67
β_{Zn}	1
β_{CFU}	0.28
β_{ova}	2800

3.3 Operational cost index

The new OCI is formulated by modifying the existing one used by De Ketele *et al.* (2018) (See Eq. 2-9) through considerations of post-treatment investments and benefits together with WRRF processes. The OCI includes the operating cost factors as well as any potential savings arising

from the implementation of WWTP design or control operating strategies and non-compliance fines (Eq. 3-4).

Recovery of resources, for instance struvite or calcium phosphate, is incorporated into the OCI as a benefit in terms of tangible costs. Additionally, any operational cost incurred related to side stream nutrient recovery are now included in the OCI. For instance, precipitation of struvite may require dosing lime to maintain pH and may also require dosing magnesium chloride.

Each of the costs in the OCI function vary regionally. For instance, the market price of additives and WRRF recovered products may not be the same in different regions/countries with different economies. Thus, similar to the EQI, the OCI allows operational cost evaluations tailored to specific regions. This was implemented in the model through parameterization of the market price values.

$$OCI = (AE + PE - MP + ME + HE) \cdot Energy\ cost + SP \cdot Sludge\ disposal\ cost + EC \cdot Carbon\ cost + Metals\ dosed \cdot Metal\ cost + Lime\ dosed \cdot Lime\ cost - NR \cdot Market\ price + Fines \quad \textbf{Eq. 3-4}$$

Where:

AE : Aeration energy (kWh/d)
PE : Pumping energy (kWh/d)
SP : Sludge produced (kgTSS/d)
EC : External carbon addition (kgCOD/d)
ME : Mixing energy (kWh/d)
MP : Energy from methane produced (kWh/d)
HE : Total heat energy required by anaerobic digester for sludge treatment (kWh/d)
NR : Nutrient recovered, e.g., Struvite (kg/d)

3.3.1 Energy tariffs

In South Africa, the energy cost is determined by the tariffs set by Eskom. The latter makes use of "time of use" tariffs which vary during peak, standard and off-peak times during the day, and

vary during high demand (June to August) and low demand (September to May) seasons. In addition to the aforementioned, Eskom tariffs also vary based on transmission zones. The average tariffs set by Eskom for 2019/2020 for the low demand season are shown in Table 3-9 below.

The time-of-use tariff employed by Eskom is applied in the OCI formula through the addition of a time dependent price function P(t) as described by De Ketele *et al.* (2018). To illustrate the latter, the aeration energy cost is calculated with consideration of time-of-use tariffs as $AE\ cost = \int_{t_{begin}}^{t_{end}} AE \cdot \frac{P(t)}{100} dt$. Similarly, other energy costs relating to pumping, mixing and heating, as well as savings from methane production is calculated with the inclusion of the term "P(t)".

Table 3-9: Eskom 2019/2020 tariffs (Eskom, 2019)

Hour of the day	Time of use	P(t), Price (incl. VAT) (c/kWh)
0h – 6h	Off-peak	53.20
6h – 7h	Standard	83.83
7h – 10h	Peak	121.78
10h – 18h	Standard	83.83
18h – 20h	Peak	121.78
20h – 22h	Standard	83.83
22h – 24h	Off-peak	53.20

3.3.2 Non-compliance fines

To ensure that effluent quality is not compromised for lower operating costs while aiming to achieve optimal design and operation of WWTP, effluent violation charges are included in the OCI. Once a "unit fine" is decided, the total fines incurred is calculated as the product of the unit fine and the $EQI_{negative}$. The latter, as described by using the approach described by De Ketele *et al.* (2018), disregards all non-negative terms in Eq. 3-5 below. As a result, the only terms left in the formula are pollutants that have exceeded the regulatory effluent limit. Therefore, as the pollutants are all standardised by application of weighting factors, the product of unit fine and EQI negative gives the total fine. This method of calculating fines is similar to the approach used in Flanders (Vanrolleghem *et al.*, 1996). While currently in South Africa WWTPs do not get fined for non-compliance to effluent standards, this framework for calculating fines remains a

potential solution to prevent compromise in effluent quality should lawmakers opt to implement fines in the future.

$$EQI_{water} = \frac{1}{T \cdot 1000} \int_{t_0}^{t_{end}} (\beta_{TSS} \cdot (TSS_{limit} - TSS(t)) + \beta_{COD} \cdot (COD_{limit} - COD(t)) + \beta_{FSA} \cdot (FSA_{limit} - FSA(t)) + \beta_{NO} \cdot (NO_{limit} - NO(t)) + \beta_{OP} \cdot (OP_{limit} - OP(t))) \cdot Q_e(t) \cdot dt \quad \textbf{Eq. 3-5}$$

$$Fines = Unit\ fine \times EQI_{negative} \quad \textbf{Eq. 3-6}$$

3.4 Closure

The development of performance indices that can be used to evaluate wastewater treatment systems was done by modifying the existing EQI & OCI adopted by the IWA BSM group (Jeppsson *et al.*, 2007; Solon *et al.*, 2017) in line with the concept of WRRFs. The modified performance indices therefore incorporated the fate of WRRFs products and a broader consideration of environmental impact and resource recovery.

The existing EQI from the IWA BSM2P task group (Solon *et al.*, 2017) was previously used in simulation models to only evaluate the impact of treated effluent on water bodies. In order to include evaluations on the impact of products generated by the WRRFs, some extensions were proposed to this EQI formulation: (i) Firstly, the EQI as used in the BSM2P (Solon *et al.*, 2017) was extended to include all pollutants according to South African effluent discharge limits. Similar to the approach taken by De Ketele *et al.* (2018), the weighting factors assigned to each pollutant were based on its impact on the environment relative to COD. (ii) Secondly, an expression for EQI_{gas} specific to the impact of wastewater treatment processes on the atmosphere was formulated. This expression included greenhouse gases that evolve from WWTPs such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The weighting factors assigned to each gas were set equal to their respective global warming potentials—a measure of the amount of heat greenhouse gases trap relative to CO_2 (EPA, 2017). (iii) Lastly, an expression for EQI_{sludge} specific to the impact of sludge disposal on land was formulated. The pollutant class for

South African wastewater sludges (Synman & Herselman, 2006) were used as a basis for this EQI. Weightings for each pollutant were determined based on the relative pollutant limits.

The new OCI was formulated by modifying the existing one used by De Ketele *et al.* (2018) through considerations of post-treatment investments and benefits together with WRRF processes. The new OCI was made to include operating cost factors and potential savings arising from the implementation of WWTP design or control operating strategies. In addition, recovery of resources, for instance struvite or calcium phosphate, was incorporated into the OCI as a benefit in terms of tangible costs.

To ensure that effluent quality is not compromised for lower operating costs while aiming to achieve optimal design and operation of WWTP, effluent violation charges were included in the OCI. Using the approach described by De Ketele *et al*. (2018) coupled with EQI modifications described above, an 'EQI negative' value can be calculated. Total fines can then be calculated by the product of the latter and a unit fine with respect to pollutant. This method of calculating fines is similar to the approach used in Flanders (Vanrolleghem *et al*., 1996).

Once formulated, the EQI and OCI equations were included into the PWM_SA model. Then, the performance of case studies including side stream nutrient recovery were evaluated through steady state modelling using the mass balanced models in Microsoft Excel and WEST® (Vahnooren *et al*., 2003) simulation software. This is detailed in the following Chapter 4.

4. Case study: Waterval Wastewater Care Works (WCW)

This chapter details the methodology used to test the research hypothesis. The PWM_SA by Ikumi *et al.* (2015a) was used to replicate a South African WWTP for the simulation of two scenarios: (1) without side-stream treatment and (2) with side-stream treatment including recycle of side-stream effluent to head of works. The WWTP was firstly run as a mass-balanced steady state (SS) model on Microsoft Excel, and thereafter simulated on the WEST® simulation software (Vanhooren *et al.*, 2003). The evaluative performance indices described in Chapter 3, which were integrated into WEST®, was used to assess the performance of the WWTP under the two scenarios. The steps taken to virtually replicate the WWTP for simulation are summarised in Figure 4-1 below.

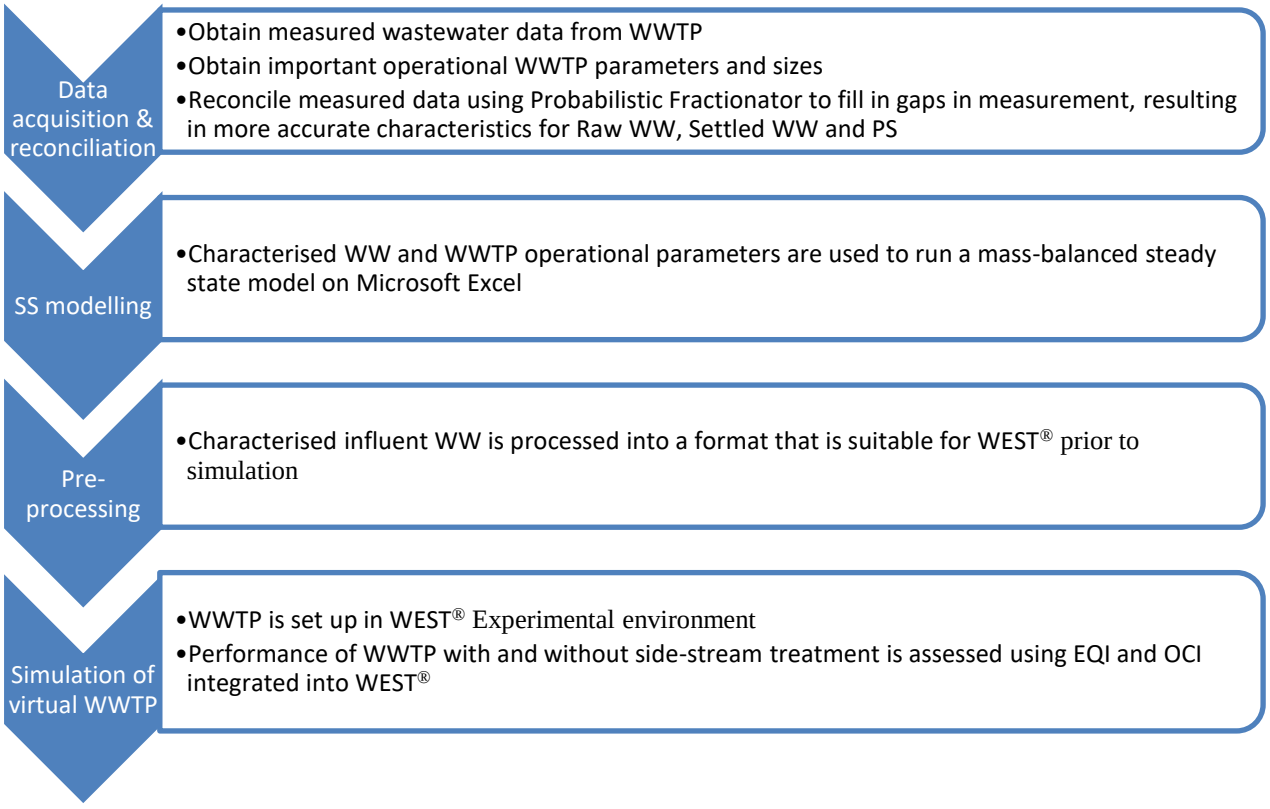

Figure 4-1: Steps taken to simulate virtual WWTP

4.1 Plant description

The chosen WWTP to simulate side stream nutrient recovery was the Waterval WCW located in Gauteng, South Africa (Figure 4-2). Commissioned in 1980, the Waterval WCW originally consisted of a first module with a capacity of 25 ML/d which was subsequently increased to 30 ML/d. In 1989 and 1993, modules 2 and 3 were respectively constructed each with a capacity of 35 ML/d. A fourth module was then commissioned in 2009 with a capacity of 50 ML/d. (Ekurhuleni Water Care Company [ERWAT], 2021)

A capacity estimation study conducted by Mott MacDonald in 2015 revealed that the capacities of modules 1, 2, 3 and 4 are now 40, 40, 40 and 50 ML/d respectively, resulting in a total plant capacity of 170 ML/d. (Mott MacDonald, 2016)

Figure 4-2: Aerial view of Waterval WCW (Google Maps, 2021)

4.1.1 Module 1

Module 1 of Waterval WCW, with a capacity of 40 ML/d, comprises two identical units with each including one PST, one primary reactor (anaerobic and aerobic zones), two primary clarifiers, one secondary reactor (aerobic zone) and two secondary clarifiers (Figure 4-3).

Module 1 was designed to have a dual sludge age system whereby the primary reactors are the recipients of 40% of the influent flow and, the secondary reactors are recipients to 60% of the influent flow as well as the effluent from the primary clarifiers. Sludge is wasted from the underflow of both the primary and secondary clarifiers separately which therefore allows the sludge age of each set of reactors to be independently controlled. The RAS from the primary and secondary clarifiers are pumped back to the start of the primary and secondary reactors respectively. This process model, while outdated, continues to be operated in order to reduce infrastructural changes to module 1.

While nitrification occurs in the secondary reactor, no provisions were made in the design of module 1 for denitrification. Rather, it was primarily designed for phosphorus and COD removal. However, since partial nitrification occurs in the system, RAS containing nitrates is pumped back into the unaerated zone. This turns the intended anaerobic zone anoxic and inhibits biological phosphorus removal. (Mott MacDonald, 2016)

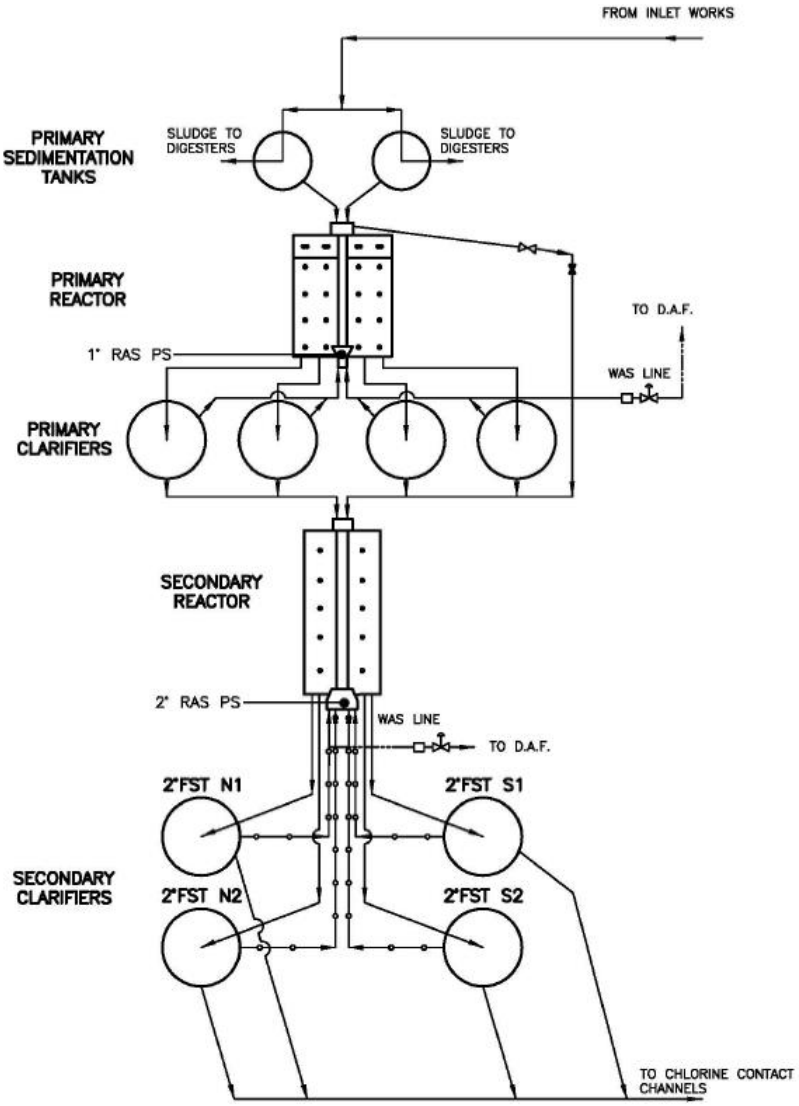

Figure 4-3: Waterval WCW Module 1 (Mott MacDonald, 2016:37)

Each unit of module 1 consist of a 530 m^3 unaerated zone and a 2160 m^3 aerobic zone in the primary reactor a 3250 m^3 secondary aerobic reactor. Therefore, the total volumes for the unaerated, primary aerobic and secondary aerobic zones of module 1 are 1060 m^3, 4320 m^3 and 6500 m^3 respectively.

4.1.2 Modules 2 and 3

Modules 2 and 3 at Waterval WCW are two functionally identical modules each having a capacity of 40 ML/d. Each module comprises two identical units, each including one PST, one

BNR reactor (anaerobic, anoxic and aerobic zones) and two SSTs as shown in Figure 4-4. The modules were designed operate either as a 3-Stage Phoredox, Johannesburg system or Modified UCT system.

Both modules 2 and 3 are currently operated as a 3-Stage Phoredox system which consists of a sequence of anaerobic, anoxic and aerobic zones. In the configuration, the settled wastewater firstly enters the anaerobic zone where PAOs take up readily biodegradable organics and release ortho-phosphates into the bulk liquid (see Section 2.2.2.1 for description of PAO metabolism). Thereafter in anoxic zone, nitrates produced via nitrification in the subsequent aerobic zone (recycled to anoxic zone through the a-recycle) are denitrified and converted to nitrogen gas. Finally, in the aerobic zone, removal of organics, oxidation of ammonia and a net uptake of ortho-phosphates occurs.

The RAS is recycled back from the SSTs to the anaerobic zones of the reactors. Since nitrates are present in the RAS, recycling to the anaerobic zone inhibits biological phosphorus removal to a certain extent. With the presence of nitrates, facultative OHOs are then able to utilise a portion of readily biodegradable organics present in the anaerobic zone for denitrification. Consequently, less readily biodegradable organics are available for uptake by PAOs. The reduced substrate for growth of active PAOs ultimately results less ortho-phosphates (OP) aerobic uptake, hence higher effluent OP. (Bradford, Conning & Partners, 1994)

Each module has a total volume of 15988 m^3 split among the anaerobic, anoxic and aerobic zones as 2702.66 m^3, 6359.2 m^3 and 6836.14 m^3 respectively.

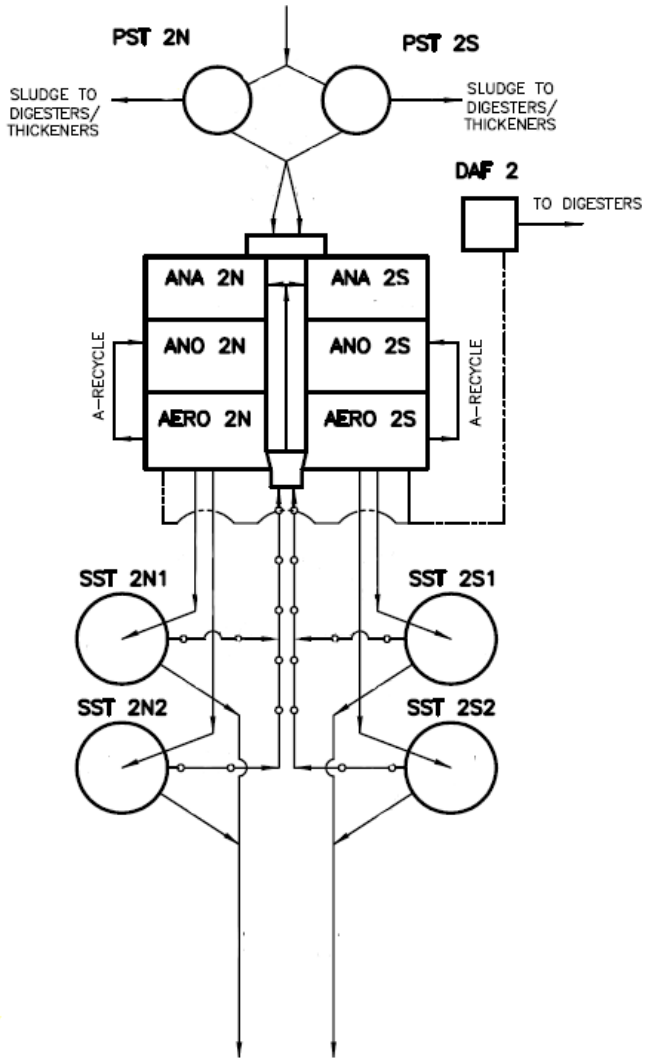

Figure 4-4: Waterval WCW Module 2 (Mott MacDonald, 2016)

4.1.3 Module 4

Module 4 at Waterval WCW, with a design capacity of 50 ML/d, comprises two identical units. Similar to modules 2 and 3, each unit of module 4 consists of one PST, one reactor (anaerobic, anoxic and aerobic zones) and two SSTs. The biological reactor was designed such that either a 3-Stage Phoredox, Johannesburg or Modified UCT system may be operated.

Module 4 is currently operated as a 3-Stage Phoredox as it expected to be the most suitable configuration under most loading conditions.

The total volume of the reactor is 21688 m^3, of which 12680 m^3 is allocated to the aerobic zone and 9008 m^3 to unaerated volume. The unaerated volumes are split into seven

compartments, each having an approximate volume of 1286 m^3. The anaerobic zone may be made up of two or three unaerated compartments while the anoxic zone may be made up of four or five unaerated compartments. The anaerobic and anoxic volumes may thus be varied from 2572 to 3858 m^3 and 5144 to 6430 m^3 respectively provided that the sum does not exceed the allocated unaerated volume. (Sintec, 2008)

4.1.4 Sludge management

Primary sludge along with WAS (thickened in dissolved air floatation (DAF) units) are anaerobically digested in a total of sixteen units as follows:

- Module 1: four digesters
- Module 2 and 3: six digesters
- Module 4: four digesters

The nutrient-rich sludge dewatering liquor from DAF units and from dewatering digested sludge are returned to the head of works for further treatment.

4.1.5 Effluent requirements

The effluent quality requirement for discharge into the Klip River is regulated by the Waterval Water Use License (08/C22C/fg/646) issued in 2011 by the Department of Water and Sanitation (DWS). The pollutant limits stipulated in the license are summarised in Table 4-1 below.

Table 4-1: Effluent concentration limits

Parameter	Limit	Unit
COD	70	mgCOD/l
FSA	4	mgN/l
NO	9	mgN/l
OP	0.7	mgP/l
TSS	20	mgTSS/l

Parameter	Limit	Unit
pH	6.0 - 8.5	-
E.Coli	500	CFU/100 ml
Chlorine	0.25	mg/l
Fluoride	0.7	mg/l
Electrical conductivity	80	mS/m
Soap, oil, grease	2.5	mg/l

Based on the above effluent requirements, the corresponding Beta factors for the EQI water were determined as summarised in Table 4-2 below (Section 3.2.1 details the methods used to determine Beta factors).

Table 4-2: Beta factors for Waterval WCW

Pollutant	Concentration limit (mg/l)	Weighting factor (β)
COD	70	1
FSA	4	17.5
OP	0.7	100
NO	9	7.77
TSS	20	3.5

4.2 Influent characterisation

The raw wastewater data from Waterval WCW (*See Appendix A*) was characterised into its components by using flow weighted average concentrations. This was achieved by firstly determining the average pollutant fluxes. Where gaps in measured data for pollutant concentrations were encountered, the corresponding fluxes were calculated from the product of the measured flow rate and the average pollutant concentration. Thereafter, the average fluxes for each pollutant were divided by the average influent flow rate resulting in the flow weighted average concentrations for each pollutant.

The influent wastewater for Module 4 Waterval WCW was characterised and tabulated in Table 4-3.

Table 4-3: Waterval WCW wastewater characteristics

Sewage Characteristic	Symbol	Units	Raw WW	Settled WW	PS
Flow	Qi	Ml/d	55.38	55.26	0.12
Total COD	Sti	mgCOD/l	729.00	437.40	135387.11
Total Soluble COD (filtered COD)	Stsi	mgCOD/l	291.60	291.60	291.60
Total Particulate COD	Stpi	mgCOD/l	437.40	145.80	135095.51
Unbiodegradable Soluble COD	Susi	mgCOD/l	70.00	70.00	70.00
Unbiodegradable Particulate COD	Supi	mgCOD/l	80.19	21.91	26991.29
Biodegradable Particulate COD	Sbpi	mgCOD/l	357.21	123.89	108104.22
Biodegradable Soluble COD	Sbsi	mgCOD/l	221.60	221.60	221.60
Fermentable Biodegradable Soluble COD	Sbsfi	mgCOD/l	193.15	193.15	193.15
COD in Volatile Fatty Acids	Sbsai	mgCOD/l	28.45	28.45	28.45
Total Biodegradable COD	Sbi	mgCOD/l	578.81	345.49	108325.82
Total Unbiodegradable COD	Sui	mgCOD/l	150.19	91.91	27061.29
Total Kjeldahl Nitrogen (N)	Nti	mgN/l	58.03	50.85	3372.26
Total Soluble N	Ntsi	mgN/l	47.65	47.65	47.65
Total particulate N (Organic)	Ntpi	mgN/l	10.37	3.20	3324.61
Unbiodegradable Soluble Organic N	Nousi	mgN/l	4.27	4.27	4.27
Unbiodegradable Particulate Organic N	Noupi	mgN/l	5.44	1.49	1832.58
Biodegradable Particulate Organic N	Nobpi	mgN/l	4.93	1.71	1492.03
Biodegradable Soluble Organic N	Nobsi	mgN/l	9.08	9.08	9.08
Free and Saline Ammonia	Nai	mgN/l	34.30	34.30	34.30
Influent Nitrate/Nitrite (NO_3-N/NO_2-N)	NO_{xi}	mgN/L	0.00	0.00	0.00
Total Phosphorus (P)	Pti	mgP/l	8.43	6.06	1088.14
Total Soluble P	Ptsi	mgP/l	5.03	5.03	5.03
Total particulate P (Organic)	Ptpi	mgP/l	3.40	1.02	1083.11
Unbiodegradable Soluble Organic P	Pousi	mgP/l	1.24	1.24	1.24
Unbiodegradable Particulate Organic P	Poupi	mgP/l	1.58	0.39	533.21
Biodegradable Particulate Organic P	Pobpi	mgP/l	1.82	0.63	549.89
Biodegradable Soluble Organic P	Pobsi	mgP/l	0.89	0.89	0.89
Ortho Phosphate	Pai	mgP/l	2.90	2.90	2.90
Influent Magnesium Concentration	Mg_i	mg/l	10.00	10.00	10.00
Influent Potassium Concentration	K_i	mg/l	10.00	10.00	10.00

Sewage Characteristic	Symbol	Units	Raw WW	Settled WW	PS
Influent Calcium Concentration	Ca_i	mg/l	5.00	5.00	5.00
Influent VSS	X_{vi}	mg/l	273.72	90.94	84677.69
Influent TSS	X_O	mg/l	305.86	94.61	97863
Influent Inorganic Settleable Solids	Xioi	mg/l	32.15	3.66	13185

4.3 Mathematical modelling of unit processes at Waterval WCW

A WWTP processes model is an essential tool that could be used for design and process optimisation. From the study of wastewater treatment, explicit equations based on fundamental kinetic and stoichiometric relationships have been developed. Consequently, SS and dynamic models can be developed for the various purposes, among others, summarised below et al., 2004):

- Tools for establishing design criteria through the identification of parameters that considerably affect system response;
- Tools for process optimisation and evaluation through comparison of predicted results and observed responses; and
- Tools for system diagnosis and monitoring such that early warnings are provided to prevent system malfunction.

4.3.1 Steady state models

SS WWTP models, based on rate-limiting kinetics and stoichiometry of biological processes, are often useful in the determination of system design parameters to meet specific WWTP performance criteria (Wentzel *et al.*, 2006). SS models include explicit equations that compute reactor sizes, interconnecting system flows, and other design parameters such as sludge retention time and reactor mass fractions. Once the system has been designed and operating parameters are known, dynamic models can then be used to optimise the design and operation of the WWTP through simulations under varying flows and loads. The main uses of SS models are summarised as follows (Wentzel *et al.*, 2006):

- To provide simple estimations of WWTP design and operational parameters.

- To investigate the sensitivity of system performance with design and operational parameters.
- To provide estimates for upstream products which are used as inputs to the ensuing processes.
- To be used as a basis for the verification of the simulation results from the PWM_SA.

4.3.1.1 UCT activated sludge steady state model

The UCT based research group developed SS activated sludge (AS) models for the removal of organics and nutrients from wastewater. These models incorporate the various biological processes described hereunder, which are responsible for the removal of organics, N and P from influent wastewater. By virtue of their nature, unbiodegradable organics cannot be removed from wastewater by biological processes. As a result, soluble unbiodegradable organics undergo no change and escape in the effluent. On the other hand, particulate unbiodegradable organics may be removed from wastewater by means of primary sedimentation and/or via the WAS. The concentration of biodegradable organics in the effluent is very low provided that the sludge retention time of the AS system is longer than 3 days (Ekama & Wentzel, 2008a). The AS SS model was therefore built on the assumption that biodegradable organics are completely utilised by OHOs in the biological reactor (as long as the sludge age is greater than 3 days and the system is stable; Henze *et al*., 2008). With the growth of microorganisms being complete, its kinetics may be ignored in the SS model. Conversely, Marais and Ekama (1976) included the kinetics of the death process into the SS model since this process takes a very long time to reach completion.

Biomass growth and death

The biological behaviour of microorganisms is modelled based on their growth and death processes (Ekama & Wentzel, 2008a). The study of microorganism growth kinetics by Monod (1949) resulted in a formulation for the specific growth rate as shown in Eq. 4-1. The Monod kinetics assume that the substrate is the only limiting factor in the growth rate of microorganism and that other factors affecting growth, specifically N and P, are plentiful. Under SS conditions, the rate of change of substrate in the bulk liquid is zero and the rate of consumption of COD by

the organisms is governed by their metabolisms. Thus, it was suitable for Marais and Ekama (1976) to use Monod kinetics as the basis for modelling the growth process under SS conditions.

$$\mu = \frac{\mu_m S_b}{K_s + S_b} \qquad \textbf{Eq. 4-1}$$

Where:

μ : Specific growth rate of organisms (g/g.d))

μ_m : Maximum specific growth rate

K_s : Substrate half maximum saturation coefficient

S_b : Concentration of biodegradable organic material (gCOD/*l*)

The death and disintegration of biomass is modelled based on the concept of endogenous respiration. This concept suggests that 24% of the biomass population dies per day. Out of the dead biomass, 80% is biodegradable and is used to generate energy for cell maintenance, and the remaining 20% is unbiodegradable and accumulates in the reactor.

The equations used to model the biological processes involved in the removal of organics were derived conceptually, with mass concentrations of WAS components being constant under SS conditions, by performing mass balances for each component over the AS system (Ekama & Wentzel, 2008a). The equations describing the stoichiometry of organic removal are summarised in Table 4-4 below.

Table 4-4: Stoichiometry of organic removal (Terms defined in Appendix B)

SN	Equation	Reference
AS1.1	$FS_{ti} = Q_i S_{ti}$	(Ekama & Wentzel, 2008a)
AS1.2	$FS_{bi} = FS_{ti}(1 - f_{S\prime us} - f_{S\prime up})$	
AS1.3	$FX_{Ii} = \frac{FS_{ti} f_{S\prime up}}{f_{cv}}$	
AS1.4	$FX_{IOi} = Q_i X_{Ioi}$	
AS1.5	$MX_{BH} = FS_{bi} \frac{Y_H R_s}{(1 + b_H R_s)}$	
AS1.6	$MX_{EH} = f_H b_H MX_{BH} R_s$	
AS1.7	$MX_I = R_s \frac{FX_{Ii}}{f_{cv}}$	
AS1.8	$MX_V = MX_{BH} + MX_{EH} + MX_I$	
AS1.9	$MX_{IO} = FX_{IOi} R_s + f_{iOHO} MX_{BH}$	
AS1.10	$MX_t = MX_V + MX_{IO}$	
AS1.11	$FO_c = FS_{bi}[(1 - f_{cv} Y_H) + (1 - f_H) b_H \frac{Y_H f_{cv} R_s}{(1 + b_H R_s)}$	
AS1.12	$FX_t = \frac{MX_t}{R_s}$	
AS1.13	$S_{te} = S_{use} + f_{cv} X_{ve}$	

Nitrification and denitrification

The nitrification is a two-step process described more fully in Section 2.2.2.1. For modelling purposes, Marais and Ekama (1976) deemed it acceptable to model the process as a single step. Consequently, *Nitrosomonas* which are the slowest growing organisms involved in the conversion of FSA to nitrite set the overall rate for the nitrification process.

Nitrates produced during the nitrification process is converted into nitrogen gas by means of biological denitrification under anoxic conditions. In addition to decreasing effluent nitrate concentration, denitrification allows for the recovery of 3.5 mg alkalinity as $CaCO_3$ and 2.86 mg of oxygen for every 1 mg of nitrate denitrified (Ekama & Wentzel, 2008b). Denitrification requires a substrate to act as an electron donor, which is in the form of influent readily biodegradable COD (RBCOD), influent slowly biodegradable COD (SBCOD) and SBCOD generated from the endogenous respiration process. The basis for the denitrification model is the

kinetics of the conversion of nitrate to nitrogen gas with respect to the type of substrate available (i.e., RBCOD or SBCOD). (Ekama & Wentzel, 2008b)

Denitrification in a primary anoxic reactor is a two-phase process. The first phase is characterised by a rapid decrease in nitrate concentration due to the simultaneous utilisation of RBCOD and SBCOD, with specific denitrification rates K_1 and K_2 respectively. In the second phase, only SBCOD is available for utilisation as all RBCOD have been used up in the first phase. Thus, phase 2 occurs at a slower denitrification rate (K_2). (Ekama & Wentzel, 2008b)

In cases where the system includes a secondary anoxic reactor, the only substrate available for denitrification is the SBCOD generated from the endogenous respiration process, which itself is a slow process. Therefore, denitrification in a secondary anoxic reactor occurs at a slower rate K_3 (about 2/3 of K_2). (Ekama & Wentzel, 2008b)

Table 4-5 provides a summary of the equations used in the SS nitrogen removal model.

Table 4-5: Stoichiometry of nitrogen removal (Terms defined in Appendix B)

SN	Equation	Reference
AS2.1	$N_a = N_{ae} = \frac{K_{nT}(b_{AT} + \frac{1}{R_s})}{\mu_{AmT} - (b_{AT} + \frac{1}{R_s})}$	(Ekama & Wentzel, 2008b)
AS2.2	$N_{ae} = \frac{K_{nT}(b_{AT} + \frac{1}{R_s})}{\mu_{AmT}(1 - f_{xt}) - (b_{AT} + \frac{1}{R_s})}$	
AS2.3	$N_{ae} = \frac{K_{nT}}{(S_f - 1)}$	
AS2.4	$N_{ae} = N_{ai} + N_{obsi} + N_{obpi} - (N_s - N_{oupi})$	
AS2.5	$N_{te} = N_{ouse} + N_{ae}$	
AS2.6	$N_s = f_n \frac{MX_v}{Q_i R_s}$	
AS2.7	$N_{te} = N_{ti} - N_s$	
AS2.8	$N_{ne} = N_c = N_{ti} - N_{te} - N_s$	
AS2.9	$D_{p1RBCOD} = \frac{f_{Sb's} S_{bi}(1 - f_{cv} Y_H)}{2.86}$	
AS2.10	$D_{p1SBCOD} = \frac{K_2 f_{x1} S_{bi} Y_H R_s}{(1 + b_h R_s)}$	

SN	Equation	Reference
AS2.11	$D_{p1} = D_{p1RBCOD} + D_{p1SBCOD}$	
AS2.12	$D_{p3SBCOD} = \frac{K_3 f_{x3} S_{bi} Y_H R_s}{(1 + b_h R_s)}$	
AS2.13	$D_{p3} = 0 + D_{p3SBCOD}$	
AS2.14	$FO_n = 4.57 N_c Q_i$	
AS2.15	$FO_d = 2.86(N_c - N_{ne}) Q_i$	
AS2.16	$FO_t = FO_c + FO_n - FO_d$	

Biological excess phosphorus removal

Biological excess phosphorus removal (BEPR) is a process that is facilitated by the action of PAOs through P uptake and release mechanisms as described more fully in Section 2.2.2.1. The BEPR is based on the substrate allocation between PAOs and OHOs. Once the latter has been determined, the masses of each of the organisms can be calculated. With the P content of the organisms known, the overall P removal can be determined. (Wentzel *et al.*, 2008)

A summary of the equations used in the SS BEPR model is presented in Table 4-6 below.

Table 4-6: Stoichiometry of phosphorus removal (Terms defined in Appendix B)

SN	Equation	Reference
AS3.1	$MX_{PAO} = \frac{Y_{PAO}}{(1 + b_{PAO,T} R_s)} FS_{bs,PAO} R_s$	(Wentzel *et al.*, 2008)
AS3.2	$MX_{E,PAO} = f_{XE,PAO} * b_{PAO,T} * MX_{PAO} * R_s$	
AS3.3	$MX_{OHO} = \frac{Y_{OHO}}{(1 + b_{OHO,T} R_s)} FS_{b,OHO} R_s$	
AS3.4	$MX_{E,OHO} = f_{XE,OHO} * b_{OHO,T} * MX_{OHO} * R_s$	
AS3.5	$MX_I = \frac{f_{XI,COD,i} FS_{ti} R_s}{f_{cv}}$	
AS3.6	$S_{bsf,i,conv} = S_{bsf,i} - 8.6(sS_{NO3,s} + S_{NO3,i}) - 3.0(sS_{O2,s} + S_{O2,i})$	
AS3.7	$S_{bsf,ANn} = \frac{\frac{S_{bsf,i,conv}}{(1+s)}}{[1 + k_{F,T} \frac{MX_{OHO}}{Q_i(1+s)} \frac{f_{AN}}{N}]^n}$	
AS3.8	$FS_{bs,PAO} = Q_i[S_{bsf,i,conv} - (1+s) S_{bsf,ANn}] + Q_i S_{bsai}$	
AS3.9	$\Delta P_{PAO} = f_{P,PAO} \frac{MX_{PAO}}{R_s} \frac{1}{Q_i}$	

SN	Equation	Reference
AS3.10	$\Delta P_{OHO} = f_{P,OHO} \frac{MX_{OHO}}{R_s} \frac{1}{Q_i}$	
AS3.11	$\Delta P_{XE} = f_{P,XE} \frac{(MX_{E,PAO} + MX_{E,OHO})}{R_s} \frac{1}{Q_i}$	
AS3.12	$\Delta P_{XI} = f_{P,XI} \frac{MX_{I,i}}{R_s} \frac{1}{Q_i}$	
AS3.13	$\Delta P_{SYS,POT} = \Delta P_{PAO} + \Delta P_{OHO} + \Delta P_{XE} + \Delta P_{XI}$	
AS3.14	$\Delta P_{SYS,ACT} = \min(\Delta P_{SYS,POT}; T_{P,i})$	
AS3.15	$MX_{VSS} = MX_{PAO} + MX_{OHO} + MX_{E,PAO} + MX_{E,OHO} + MX_I$	
AS3.16	$MX_{FSS} = f_{FSS,OHO} MX_{OHO} + f_{FSS,PAO} MX_{PAO} + FX_{ISS,i} R_s$	
AS3.17	$MX_{TSS} = MX_{VSS} + MX_{ISS}$	
AS3.18	$f_{p,TSS} = \frac{\frac{f_{P,OHO} MX_{OHO}}{f_{VT}}}{MX_{TSS}} + \frac{\frac{f_{P,XE}(MX_{E,OHO} + MX_{E,PAO})}{f_{VT}}}{MX_{TSS}} + \frac{\frac{f_{P,XI} MX_{I,i}}{f_{VT}}}{MX_{TSS}} + \frac{\frac{f_{P,PAO} MX_{PAO}}{f_{VT,PAO}}}{MX_{TSS}} + \frac{f_{P,FSS,i} MX_{FSS}}{MX_{TSS}}$	
AS3.19	$X_{P,e} = f_{P,TSS} TSS_e$	
AS3.20	$T_{P,e} = T_{P,i} - \Delta P_{SYS,ACT} + X_{P,e}$	
AS3.21	$FO_{2,PAO} = (1 - f_{CV} Y_{PAO}) FS_{bs,PAO} + f_{CV}(1 - f_{E,PAO}) b_{PAO,T} MX_{PAO}$	
AS3.22	$FO_{2,OHO} = (1 - f_{CV} Y_{OHO}) FS_{b,OHO} + f_{CV}(1 - f_{E,OHO}) b_{OHO,T} MX_{OHO}$	
AS3.23	$FO_C = FO_{2,OHO} + FO_{2,PAO}$	

4.3.1.2 Anaerobic digestion model

The SS AD model was developed by Sötemann *et al.* (2005) based on COD, N and P mass balances over the AD. Depending on the hydrogen partial pressure (pH_2), AD of sludge is carried out by the action of three or four types of microorganisms. At equilibrium (i.e., low pH_2), the AD process is carried out by three microorganisms — acidogens, acetoclastic methanogens, and hydrogentrophic methanogens. Acidogens firstly convert complex biodegradable organics into acetic acid (HAc) and H_2, releasing organically bound N into the bulk liquid. Thereafter, acetoclastic methanogens convert the Hac into CH_4 and CO_2 is released in the gaseous phase. Hydrogenotrophic methanogens then convert the H_2 produced into CH_4. At high pH_2, acidogens also produce propionic acid (HPr) and VFAs. Acetogens are then able to convert the HPr into HAc and H_2. Once HPr is converted to HAc and H_2, the same mechanisms involved in methane production at low pH_2 take over and methane is produced. (van Lier *et al.*, 2008)

Optimum operation of the digester requires its pH to be maintained between 7 and 8 by controlling the HAc concentration in the system. Since hydrolysis/acidogenesis is the slowest process and affects the pH of the digester, the AD model uses the kinetics of acidogens. The AD model developed by Sötemann *et al.* (2005) thus comprises three parts: (i) a kinetic model to determine rate of COD utilisation and methane production; (ii) a stoichiometric model to predict ammonium and alkalinity generated, and the gas composition of the fraction of COD removed; and (iii) a weak acid/base chemistry model which uses the partial pressure of CO_2 (p_{CO2}) and alkalinity generated to predict the pH of the digester.

The AD model developed by Sötemann *et al.* (2005) was later extended by Ikumi *et al.* (2015b) to include digestion of WAS from BNR systems. This introduced a higher level of complexity to the existing AD model, with metal and nutrient release due to poly-phosphate degradation which significantly affects the performance of the digester. Ikumi and Ekama (2019) noted that PAOs would exhibit the same behaviour during AD as in the anaerobic zone of the AS system. However, with the lack of an alternating aerobic zone in the digester, PAOs are unable to grow and can only undergo lysis thereby resulting in breakdown of all their stored products (PHAa and remaining polyphosphates), and further nutrient release into the bulk liquid.

Table 4-7 provides a summary of the equations used in the SS AD model.

Table 4-7: Stoichiometry of anaerobic digestion (Terms defined in Appendix B)

SN	Equation	Reference
AD1.1	$R = \frac{V}{Q}$	(Sötemann *et al.*, 2005)
AD1.2	$S_{bpi} = (1 - f_{PS'up})S_{ti} - S_{bsai}$	
AD1.3	$S_{upi} = f_{PS'up}S_{ti}$	
AD1.4	$S_{bp} = \frac{K_s(\frac{1}{R} + b_{AD})}{Y_{AD}K_m - (\frac{1}{R} + b_{AD})}$	
AD1.5	$S_{bp\,hydrolysed} = S_{bpi} - S_{bp}$	
AD1.6	$Z_{AD} = \frac{Y_{AD}(S_{bpi} - S_{bp})}{[1 + b_{AD}R(1 - Y_{AD}\{1 - f_{AD}\})]}$	
AD1.7	$Z_E = f_{AD} * b_{AD} * Z_{AD} * R_S$	
AD1.8	$r_h = \frac{K_m S_{bp}}{[Z_{AD}(K_s + S_{bp})]}$	

SN	Equation	Reference
AD1.9	$S_m = (1 - Y_{AD})Rr_h + S_{bsai}$	
AD1.10	$Q_m = [(1 - Y_{AD})Rr_h + S_{bsai}]\frac{24}{64}$	
AD1.11	$S_{te} = S_{up} + Z_{AD} + S_{bp} + Z_E + S_m$	
AD1.12	$C_xH_yO_zN_a + \left(2x + a - z - \frac{ED_S}{D_B}(n + 2k - m) - \frac{(1-E)D_S}{4}\right)H_2O \rightarrow \left(x - a - \frac{ED_S}{D_B}(n-k) - \frac{(1-E)D_S}{8}\right)CO_2 + \left(\frac{(1-E)D_S}{8}\right)CH_4 + \left(\frac{ED_S}{D_B}\right)C_kH_lO_mN_n + \left(a - \frac{nED_S}{D_B}\right)NH_4^+ + (a - n\frac{ED_S}{D_B})HCO_3^-$	
AD1.13	$CH_3COOH \longrightarrow CH_4 + CO_2$	
AD1.14	$CH_3COO^- + H_2O \longrightarrow CH_4 + HCO_3^-$	
AD1.15	$S_{bsHAci} = \frac{S_{bsai}}{(1 + 10^{pH_i - pK'_a})}$	
AD1.16	$S_{bsAci} = \frac{S_{bsai}}{(1 + 10^{pK'_a - pH_i})}$	
AD1.17	$p_{CO2} = \frac{[HCO_3^-](1 + 10^{pK'_{c1} - pH} + 10^{pH - pK'_{c2}})}{10^{-pK'_{HCO2}}(1 + 10^{pH - pK'_{c1}} + 10^{2pH - pK'_{c1} - pK'_{c2}})}$	

4.3.2 Dynamic models

Dynamic models, unlike SS models, are more complex types of mathematical modelling of WWTPs. Dynamic models have the ability to predict the time dependent system response to varying flows and loads (Ekama & Wentzel, 2008a). The uses of dynamic models are as follows:

- To perform sensitivity analysis of the model and to assess WWTP operation and control strategies.
- For accurate sizing of WWTP unit processes, and for selection of best design parameters and configurations for optimal plant performance in terms of effluent quality and operational cost.
- To provide training to plant operators through the illustration of the impact of operating conditions on plant performance.

4.3.2.1 UCT plant-wide dynamic simulation

Based on a complete elemental mass balance (i.e., Carbon (C), Hydrogen (H), Oxygen (O), N, P and Sulphur (S)), the three-phase plant-wide model developed by Ikumi *et al.* (2015a) comprises biological N and P removal AS based on the ASM2d (Henze *et al.*, 1995), AD of PS and anoxic-aerobic digestion (AAD) of WAS along with the non-reactive unit operation of physical thickening. The PWM_SA defines influent wastewater organics (*viz*. VFAs, biodegradable soluble organics (BSO), biodegradable particulate organics (BPO), unbiodegradable soluble organics (USO) and unbiodegradable particulate organics (UPO)) in the generic form $C_xH_yO_zN_aP_bS_c$. The model separates the slow (biological and physio-chemical) and fast (aqueous) processes in wastewater treatment by means of an external algebraic equation equilibrium speciation sub-routine. Additionally, the PWM_SA includes the effects of non-ideal aqueous solution as well as mineral precipitation on the calculation of pH.

The PWM_SA comprises three sub-models which allow for the simulation of various WWTP configurations (e.g., BNR AS system with either AD or AAD as sludge treatment options). The three sub-models are as summarised below:

1. The ionic speciation model developed by Brouckaert *et al.* (2010) which includes pairing of ionic components and inter-phase transfer of component species.
2. The ASM2-3P model which extends the Activated Sludge Model No. 2 (ASM2, Henze *et al.*, 1995) to include the ionic speciation model of Brouckaert *et al.* (2010), the Inorganic Settleable Solids (ISS) model developed by Ekama and Wentzel (2004) and modelling of mineral precipitation (Musvuto *et al.*, 2000).
3. The UCTSDM3P model which extends the UCT Anaerobic Digestion Model (UCTADM, Sötemann *et al.*, 2005) to include the hydrolysis of organic sludge, the ISS model of Ekama and Wentzel (2004), modelling of mineral precipitation (Musvuto *et al.*, 2000) and the ionic speciation model (Brouckaert *et al.*, 2010).

4.3.2.2 Model evaluation

The evaluation of the PWM_SA model included:

i) Verification of the model via a systematic method of checking material (i.e., COD, C, H, O, N, P, magnesium (Mg), potassium (K) and calcium (Ca)) mass balances, as proposed by Hauduc *et al.* (2010).

ii) Uncertainty analysis, which involved (a) determination of initial kinetic and stoichiometric parameter values and assigning of their typical ranges in accordance with the method devised by Brun *et al.* (2002). (b) Monte Carlo simulations were performed, with the assumption of the value ranges being uniformly distributed. (Ikumi *et al.*, 2014; Ikumi, 2020)

iii) Sensitivity analysis, using two main methods of (a) determination of the Standardised Regression Coefficient (SRC) and (b) the Morris Screening methods (Ikumi, 2020)

iv) The main reactions included in PWM_SA model were taken through a rigorous calibration against data generated by laboratory scale systems (Ikumi *et al*., 2014; Ikumi and Harding 2020; Ghoor, 2020; Matesun *et al*., 2021).

4.4 Model setup

In order to test the research hypothesis, it was necessary to set up virtual models of Waterval WCW with and without side stream nutrient recovery respectively. Waterval WCW comprises four modules with Module 1 using an outdated dual sludge age plant configuration and Modules 2, 3 and 4 using the three-stage Phoredox configuration. According to the raw data, Modules 2 and 3 were significantly overloaded while in the average flow into Module 4 was closest to its design capacity in comparison. Therefore, for the purposes of this research, Module 4 was chosen for modelling and struvite precipitation was the chosen method of removing both ammonia and phosphates from the nutrient-rich sludge dewatering liquor to be included in the model. The simulation software WEST® was used to model Module 4 of Waterval WCW. In addition to that, mass balanced SS models of Module 4 were set up on Microsoft Excel to compare the outputs from the SS model and from the WEST® experiment. From the simulation results, the impact of implementing side-stream struvite precipitation on energy demand, operational costs and effluent

quality were assessed *(See Chapter 5).* Two main scenarios were modelled as described in the following sub-sections 4.4.1 and 4.4.2. The icons representing the different unit processes in WEST® are described in Appendix C. Module 4 of Waterval WCW was simulated as a three-stage Phoredox configuration with operational parameters as shown in Table 4-8.

Table 4-8: Waterval WCW module 4 operational parameters

Parameter	Value
Raw WW flow rate (ML/d)	55.38
PST underflow rate (ML/d)	0.12
Wastewater temperature (°C)	20
Sludge retention time (d)	10
a-recycle ratio	3
s-recycle ratio	1
Aerobic tank volume (m^3)	12680.00
Anaerobic tank volume (m^3)	2573.71
Anoxic tank volume (m^3)	6434.29
Anaerobic digester volume (m^3)	10408.45
AD temperature (°C)	35

4.4.1 Scenario 1

The first scenario to be modelled on WEST® was a replication of Module 4 of Waterval WCW as shown in Figure 4-5 below. Scenario 1 did not include side-stream treatment of sludge dewatering liquor and was used as the base scenario against which other scenarios were evaluated.

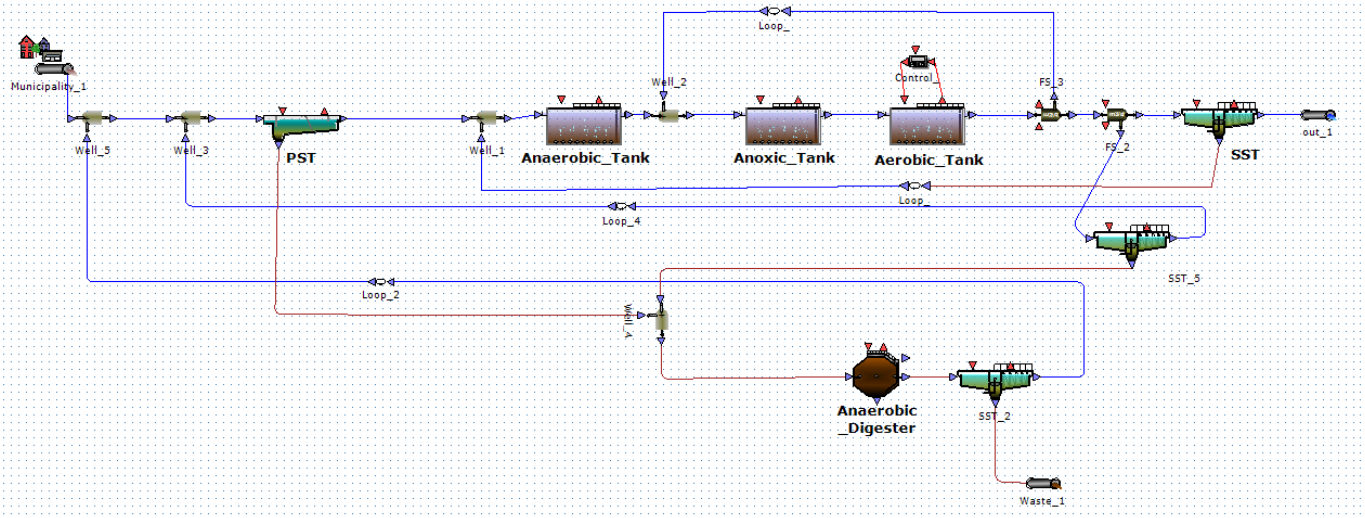

Figure 4-5: Scenario 1 — No side-stream treatment

4.4.2 Scenario 2

Scenario 2, as illustrated in Figure 4-6 below, includes the addition of a side-stream struvite precipitation unit thereby treating the nutrient rich sludge dewatering liquor, and a dosing unit for the addition of lime or magnesium hydroxide should it be needed for optimal struvite precipitation. In this scenario, the remaining liquid once separated from the struvite is recycled back to the head of works.

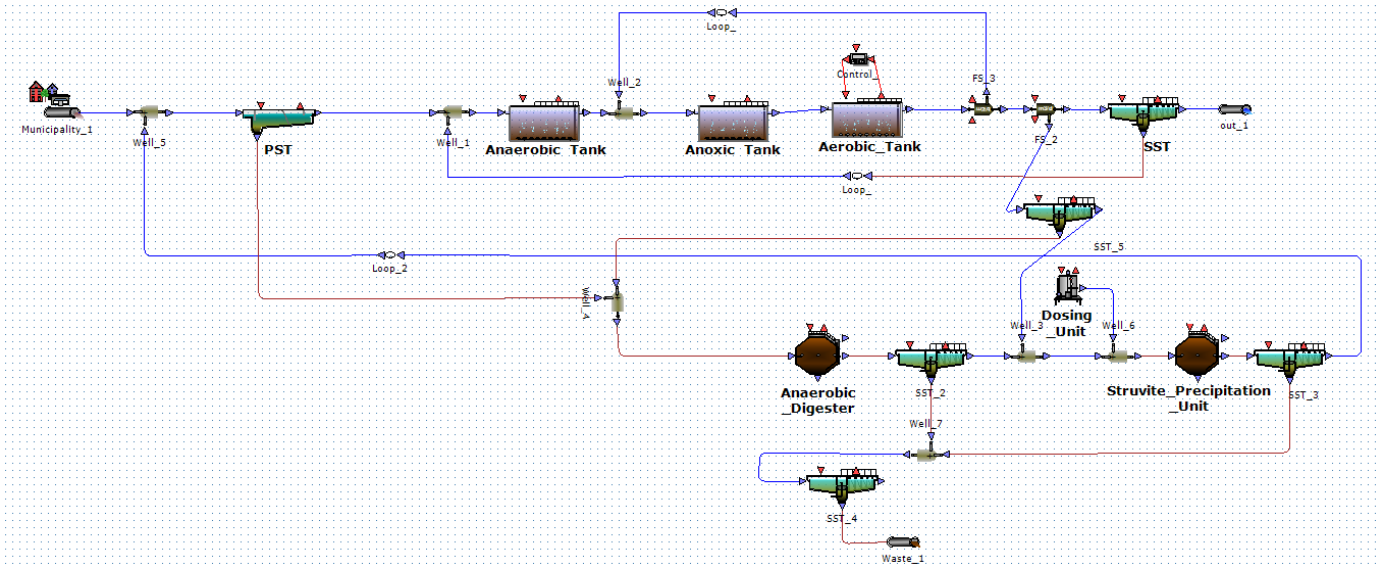

Figure 4-6: Scenario 2 — Side-stream treatment with recycle

4.5 Closure

Waterval WCW was the chosen WWTP for the simulation of nutrient recovery. The WWTP consists of four modules with maximum capacities of 40, 40, 40 and 50 ML/d respectively. Module 1 of Waterval WCW is operated as an outdated dual sludge age system to minimise infrastructural changes. Conversely Modules 2, 3 and 4 are operated as 3-stage Phoredox systems.

For the purposes of this project, Module 4 was chosen for simulation of two main scenarios — (1) no side-stream treatment of sludge dewatering liquor and (2) side-stream treatment through struvite precipitation with recycle of effluent from side-stream unit to head of works. Mass balanced SS models of each scenario were firstly set up on Microsoft Excel and subsequently simulated on the WEST® platform.

5. Results and discussion

This chapter discusses the results of the modelling of the two scenarios described in Section 4.4. For each scenario, the results from the SS model and the WEST® simulation are presented in Section 5.1. From the results, the extended EQI and OCI formulae developed in Chapter 3 were used to quantitatively assess the impact of implementing side-stream struvite precipitation has on Module 4 of Waterval WCW (Sections 5.2 and 5.3). *(See Appendix E for detailed model results)*

5.1 Modelling results

5.1.1 Scenario 1 — No side-stream treatment

5.1.1.1 AS results

The recycle of untreated sludge dewatering liquor to the head of works has a diluting effect on the influent COD concentration which is reduced by 3.2% and 2.3% according to the MS Excel SS and PWM_SA predictions respectively. From Figure 5-1, it can be observed that the predictions from the MS Excel SS model and the PWM_SA closely match each other in terms of COD concentrations — the predicted influent COD concentrations were found to be within 1% of each other and the predicted effluent COD concentrations were within 5% of each other. The overall COD removal via the AS system was determined to be approximately 83% for both models.

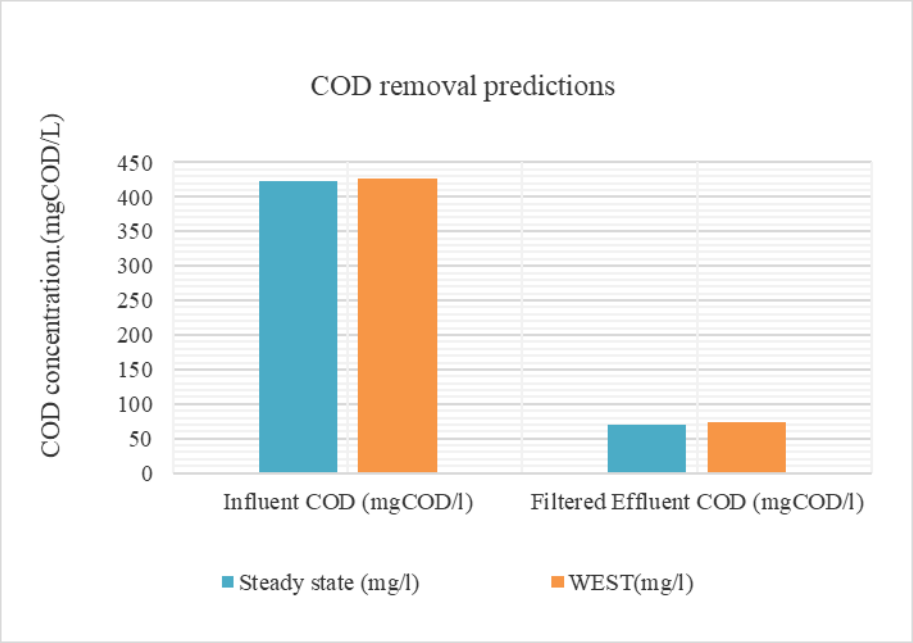

Figure 5-1: Scenario 1 — MS Excel model and PWM_SA WEST AS predictions for COD removal

With the recycle of nutrient-rich sludge dewatering liquor, the influent Total Kjeldahl Nitrogen (TKN) concentration was predicted to increase by 6% and 4% by the MS Excel SS model and the PWM_SA respectively. The corresponding increases in influent FSA concentrations due to the recycle flow for each modelling platform were 11% and 9%. The model predictions for the overall TKN removal via the AS system were determined to be approximately 91% for both the MS Excel SS model and the PWM_SA. As far as FSA was concerned, removal of 98% and 99% were predicted by the MS Excel SS model and the PWM_SA respectively.

The MS Excel SS model predicted that the nitrate (NO_3) generated in the aerobic zone would amount to 40.1 mg NO_3-N/l, out of which 31.3 mg NO_3-N/l would be denitrified in the anoxic zone, resulting in an effluent NO_3 of 8.0 mg NO_3-N/l. On the other hand, the PWM_SA predicted that the NO_3 generated would amount to 35.6 mg NO_3-N/l, out of which 27.8 mg NO_3-N/l would be denitrified, resulting in an effluent NO_3 of 7.7 mg NO_3-M/l. In both cases, 78% of the NO_3 generated was predicted to be denitrified in the anoxic zone. Figure 5-2 illustrates the predicted nitrogen removal graphically.

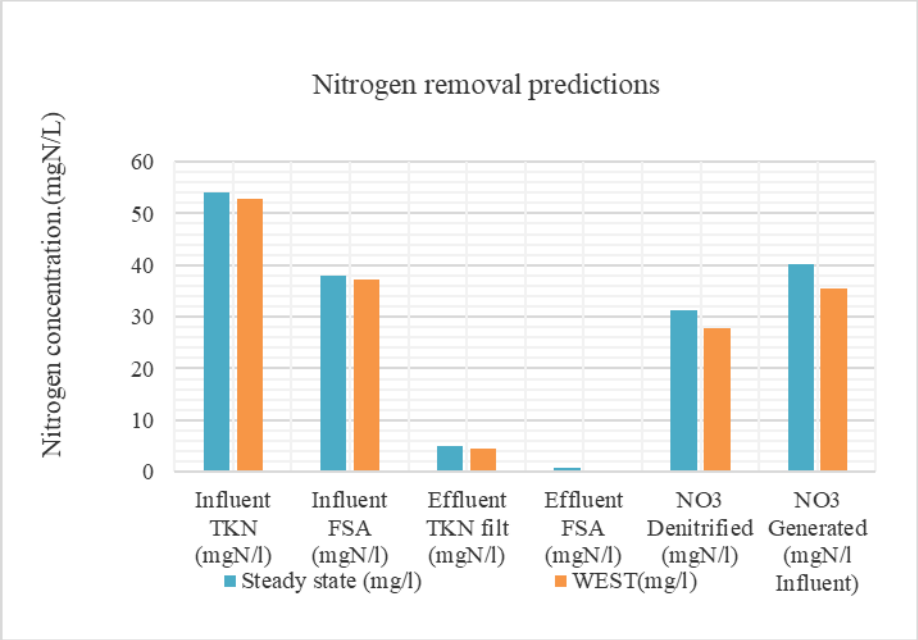

Figure 5-2: Scenario 1 — MS Excel model and PWM_SA WEST AS predictions for nitrogen removal

Due to the recycle of sludge dewatering liquor, the influent total phosphorus (TP) concentration was predicted to increase by 90% by the MS Excel SS model, and by 69% by the PWM_SA. The influent OP concentration was predicted to increase by 190% and 146% according to the MS Excel SS model and the PWM_SA respectively. As illustrated in Figure 5-3, the overall TP removal was predicted to be 89% and 84% according to the MS Excel SS model and the PWM_SA respectively. The overall OP removal was predicted to be 100% and 95% according to each model respectively.

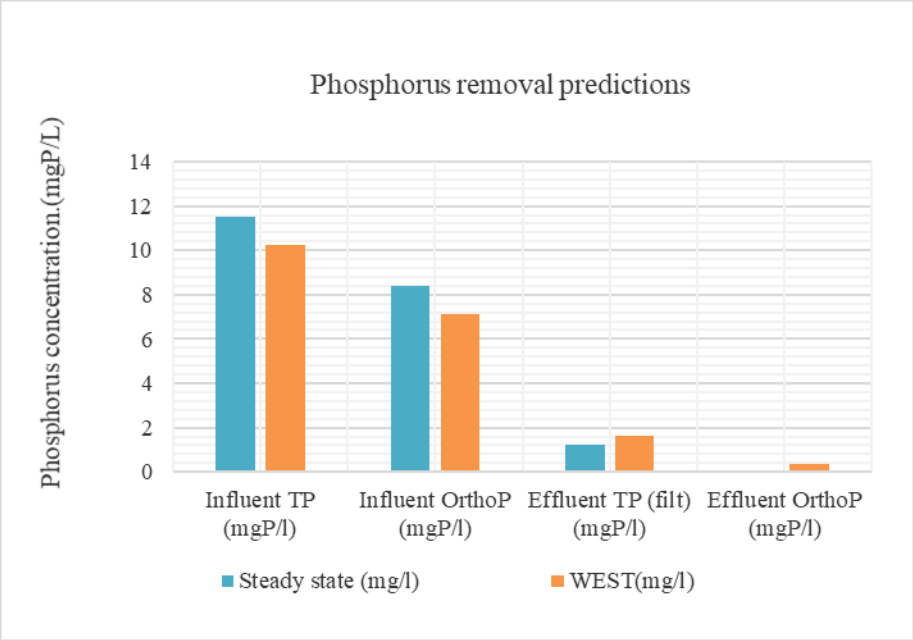

Figure 5-3: Scenario 1 — MS Excel model and PWM_SA WEST AS predictions for phosphorus removal

The carbonaceous oxygen demand (FOc) from OHOs and PAOs was predicted to be 12070.5 KgO/d and 12132.2 KgO/d according to the MS Excel SS model and the PWM_SA respectively. The nitrification oxygen demand (FOn) was predicted to be 10339.0 KgO/d and 9271.3 KgO/d by each model respectively. Taking into account the oxygen demand recovered from the denitrification process, the total oxygen demand (FOt) was predicted to be 17258.6 KgO/d and 16828.2 KgO/d. The above is illustrated in Figure 5-4 below.

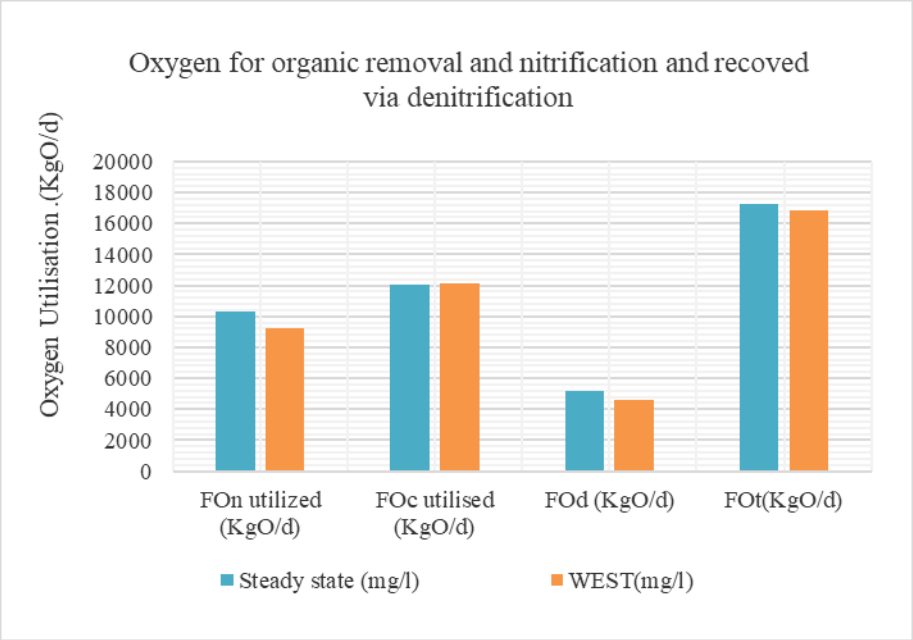

Figure 5-4: Scenario 1 — MS Excel model and PWM_SA WEST AS predictions for oxygen utilisation

5.1.1.2 AD results

The influent COD to the AD, which originates from the PS and the WAS, comprises USO, BSO, BPO and UPO. The USO and BSO were not significant contributors to the COD influx as they made up only approximately 0.2% of the influent COD to the AD. The influent BPO COD concentration was predicted to be 70932.7 mgCOD/l and 69203.6 mgCOD/l by the MS Excel SS model and the PWM_SA respectively. The second largest contributor to the COD, i.e., UPO, had concentrations of 22820.1 mgCOD/l and 22366.2 mgCOD/l according to each model respectively. The stabilisation of the sludge through anaerobic digestion resulted in approximately 89% decrease in BPO COD. As shown in Figure 5-5 below, the decrease in BPO COD is accompanied by production of CO_2 and CH_4. The latter, which can be used to generate heat and electricity and offset the energy demand, was determined to have a COD concentration of 64186.8 mgCOD/l feed and 61511.6 mgCOD/l feed according to the MS Excel SS model and the PWM_SA (in WEST®) respectively.

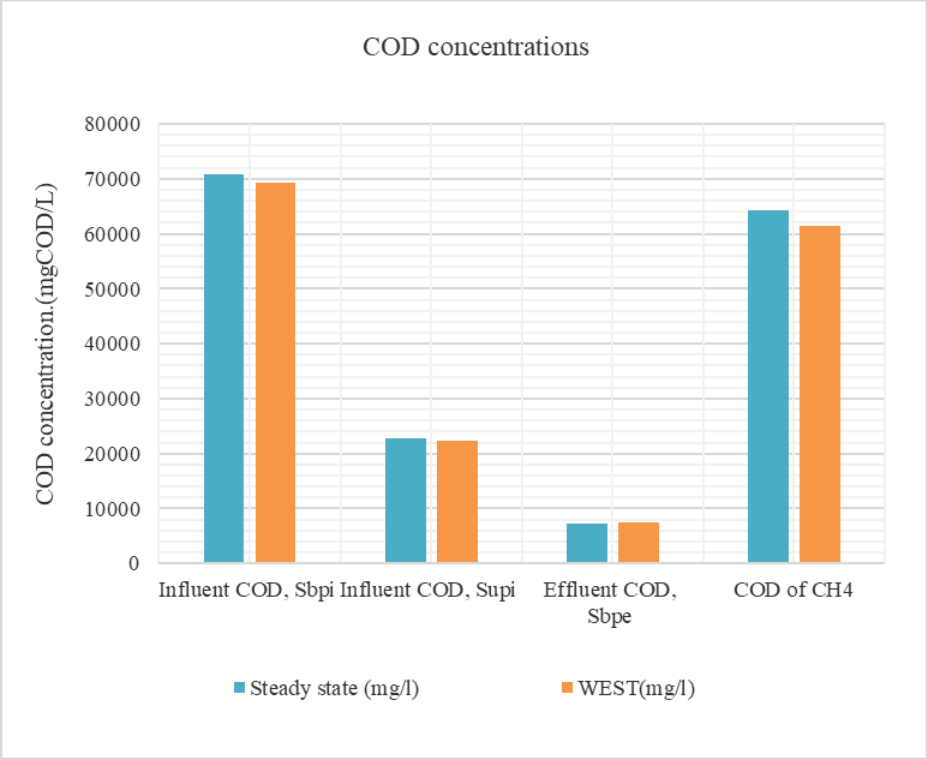

Figure 5-5: Scenario 1 — MS Excel model and PWM_SA WEST AD predictions for COD concentrations

With the decrease in volatile mass, the organically bound N and P are released into the bulk liquid. Consequently, the TP and TKN remain unaffected by the digestion processes. As illustrated in Figure 5-6 and Figure 5-7, the release of organically bound N and P into the bulk liquid results in a significant increase in the effluent FSA and OP concentrations. The effluent FSA concentration as predicted by the MS Excel SS model and the PWM_SA model was found to be 1516.4 mgN/l and 1264.3 mgN/l respectively. The effluent OP concentration was predicted to amount to 1685.8 mgP/l and 1431.1 mgP/l according to each model respectively.

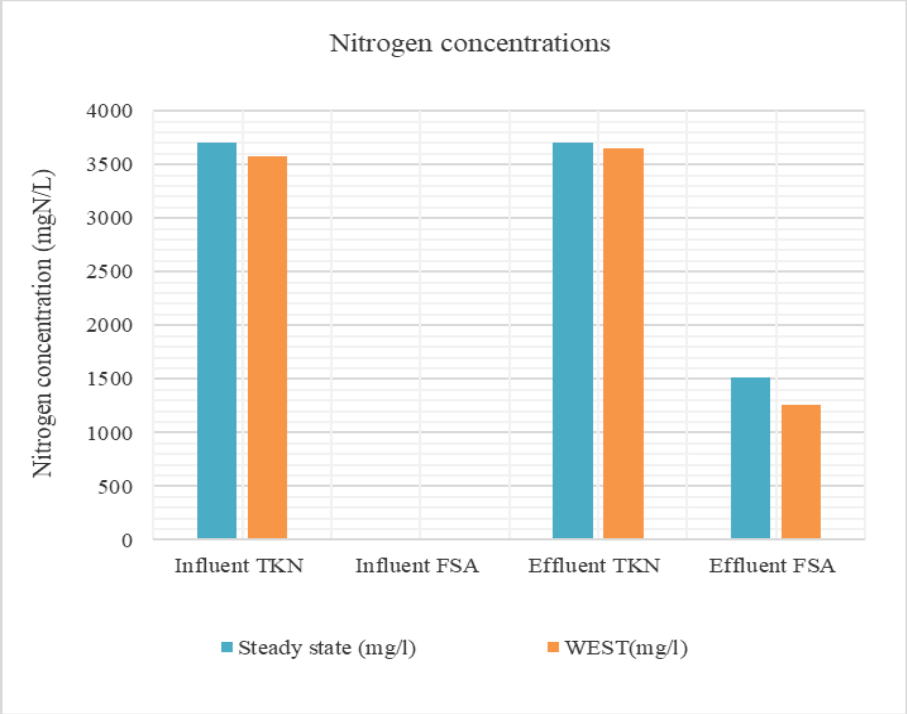

Figure 5-6: Scenario 1 — MS Excel model and PWM_SA WEST AD predictions for nitrogen concentrations

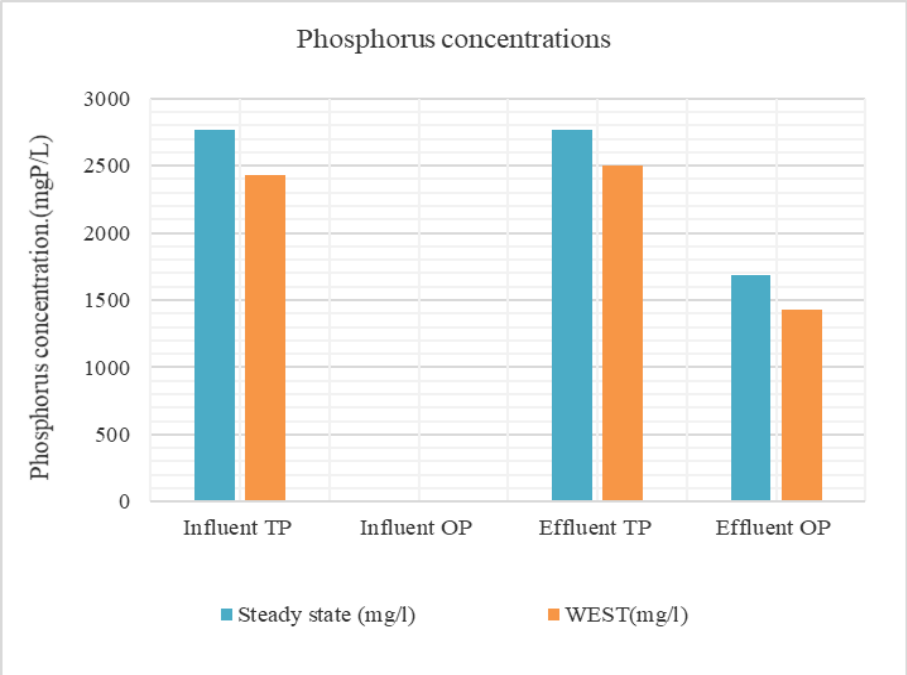

Figure 5-7: Scenario 1 — MS Excel model and PWM_SA WEST AD predictions for phosphorus concentrations

5.1.2 Scenario 2 — Side-stream treatment with recycle

5.1.2.1 AS results

This scenario involves the recycle of sludge treated sludge dewatering liquor from the side-stream struvite precipitation unit to the head of works. As a result, the influent COD concentration which is diluted by 3.3% and 2.2% according to the MS Excel SS and PWM_SA predictions respectively. From Figure 5-8, it can be observed that the predictions from the MS SS model and the PWM_SA closely match each other in terms of COD concentrations — the predicted influent COD concentrations were found to be within 1% of each other and the predicted effluent COD concentrations were within 3% of each other. The overall COD removal via the AS system was determined to be approximately 83% for both models.

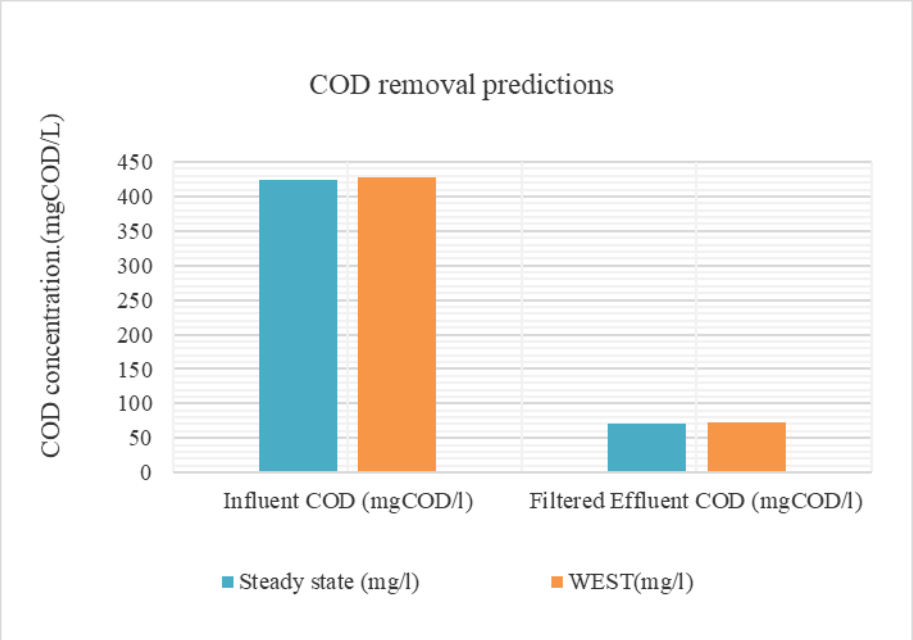

Figure 5-8: Scenario 2 — MS Excel model and PWM_SA WEST AS predictions for COD removal

With the recycle of treated sludge dewatering liquor, the influent TKN concentration was predicted to increase by 5% and 2% by the MS Excel SS model and the PWM_SA respectively. The corresponding increases in influent FSA concentrations due to the recycle flow for each modelling platform were 9% and 6%. The model predictions for the overall TKN removal via the AS system were determined to be approximately 91% for both the MS Excel SS model and

the PWM_SA. As far as FSA was concerned, removal of 98% was predicted by both the MS Excel SS model and the PWM_SA.

The MS Excel SS model predicted that the NO_3 generated in the aerobic zone would amount to 39.4 mg NO_3-N/l, out of which 30.8 mg NO_3-N/l would be denitrified in the anoxic zone, resulting in an effluent NO_3 of 7.9 mg NO_3-N/l. On the other hand, the PWM_SA predicted that the NO_3 generated would amount to 34.2 mg NO_3-N/l, out of which 26.8 mg NO_3-N/l would be denitrified, resulting in an effluent NO_3 of 7.4 mg NO_3-M/l. In both cases, 78% of the NO_3 generated was predicted to be denitrified in the anoxic zone. Figure 5-9 illustrates the predicted nitrogen removal graphically.

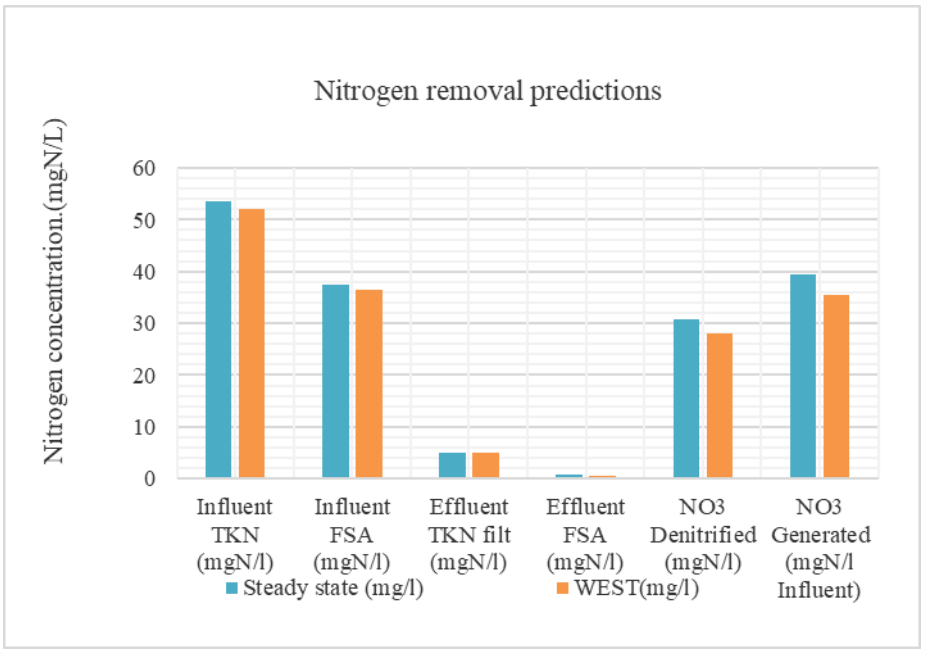

Figure 5-9: Scenario 2 — MS Excel model and PWM_SA WEST AS predictions for nitrogen removal

Since the recycled treated sludge dewatering liquor was low in phosphorus, the influent TP concentration was predicted to decrease by 1.5% by the MS Excel SS model, and by 0% by the PWM_SA. The influent OP concentration was predicted to decrease by 2% and increase by 2% according to the MS Excel SS model and the PWM_SA respectively. As illustrated in Figure 5-10, the overall TP removal was predicted to be 79% according to both the MS Excel SS model

and the PWM_SA. The overall OP removal was predicted to be 100% and 99% according to each model respectively.

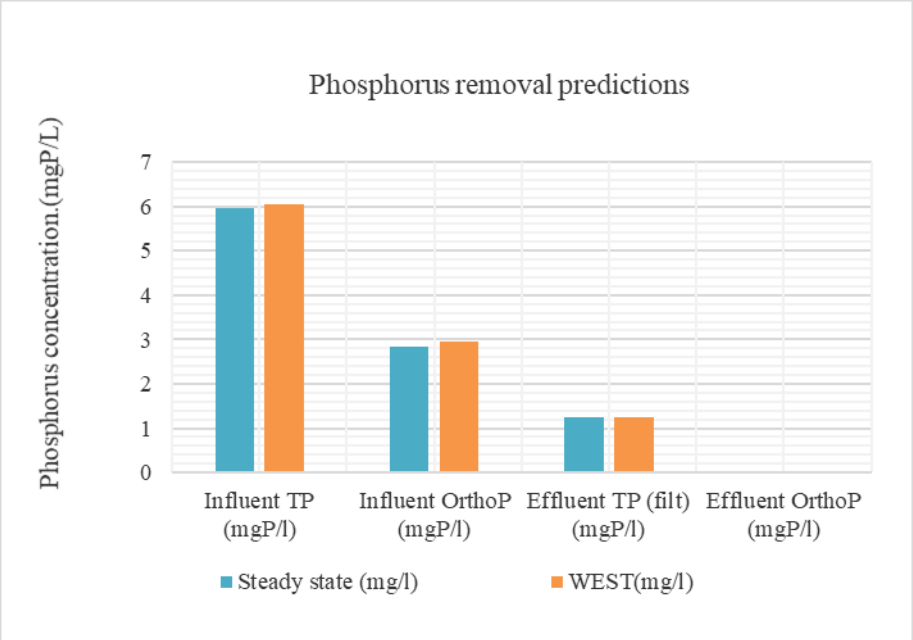

Figure 5-10: Scenario 2 — MS Excel model and PWM_SA WEST AS predictions for phosphorus removal

The FOc from OHOs and PAOs was predicted to be 12070.5 KgO/d and 12296.6 KgO/d according to the MS Excel SS model and the PWM_SA respectively. The FOn was predicted to be 10159.5 KgO/d and 8819.6 KgO/d by each model respectively. Taking into account the oxygen demand recovered from the denitrification process, the FOt was predicted to be 17169.0 KgO/d and 16711.5 KgO/d. The above is illustrated in Figure 5-11 below.

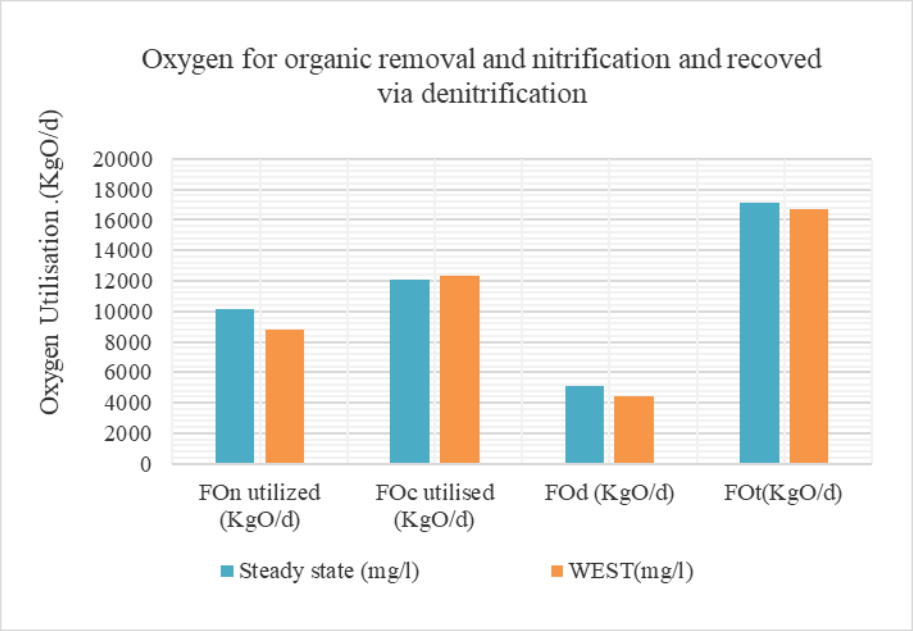

Figure 5-11: Scenario 2 — MS Excel model and PWM_SA WEST AS predictions for oxygen utilisation

5.1.2.2 AD results

The influent COD to the AD, originating from the PS and the WAS, comprises USO, BSO, BPO and UPO. The USO and BSO were not significant contributors to the COD influx as they made up only approximately 0.2% of the influent COD to the AD. The influent BPO COD concentration was predicted to be 70932.3 mgCOD/l and 70590.6 mgCOD/l by the MS Excel SS model and the PWM_SA respectively. The second largest contributor to the COD, i.e., UPO, had concentrations of 22820.2 mgCOD/l and 22351.6 mgCOD/l according to each model respectively. The stabilisation of the sludge through anaerobic digestion resulted in approximately 89% decrease in BPO COD. As shown in Figure 5-12 below, the decrease in BPO COD is accompanied by production of CO_2 and CH_4. The latter, which can be used to generate heat and electricity and offset the energy demand, was determined to have a COD concentration of 63933.2 mgCOD/l feed and 62083.1 mgCOD/l feed according to the MS Excel SS model and the PWM_SA respectively.

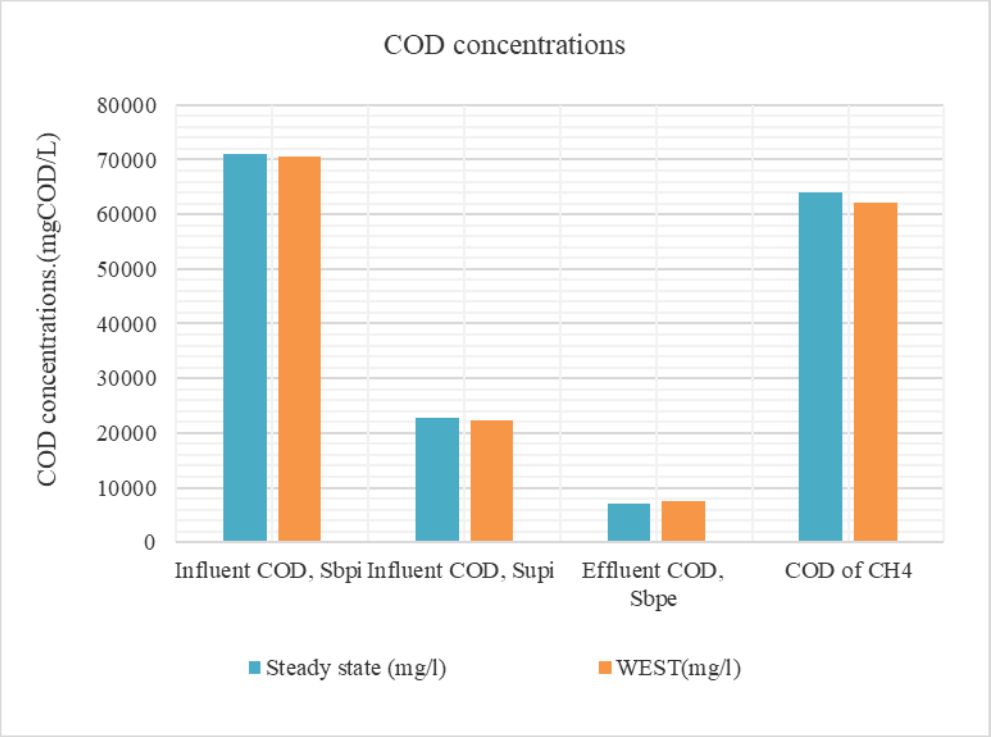

Figure 5-12: Scenario 2 — MS Excel model and PWM_SA WEST AD predictions for COD concentrations

The decrease in volatile mass results in the release of organically bound N and P into the bulk liquid as FSA and OP. Consequently, the TP and TKN remain unaffected by the digestion processes. As illustrated in Figure 5-13 and Figure 5-14, the release of organically bound N and P into the bulk liquid results in a significant increase in the effluent FSA and OP concentrations. The effluent FSA concentration as predicted by the MS Excel SS model and the PWM_SA was found to be 1687.4 mgN/l and 1439.3 mgN/l respectively. The effluent OP concentration was predicted to amount to 831.7 mgP/l and 904.0 mgP/l according to each model respectively.

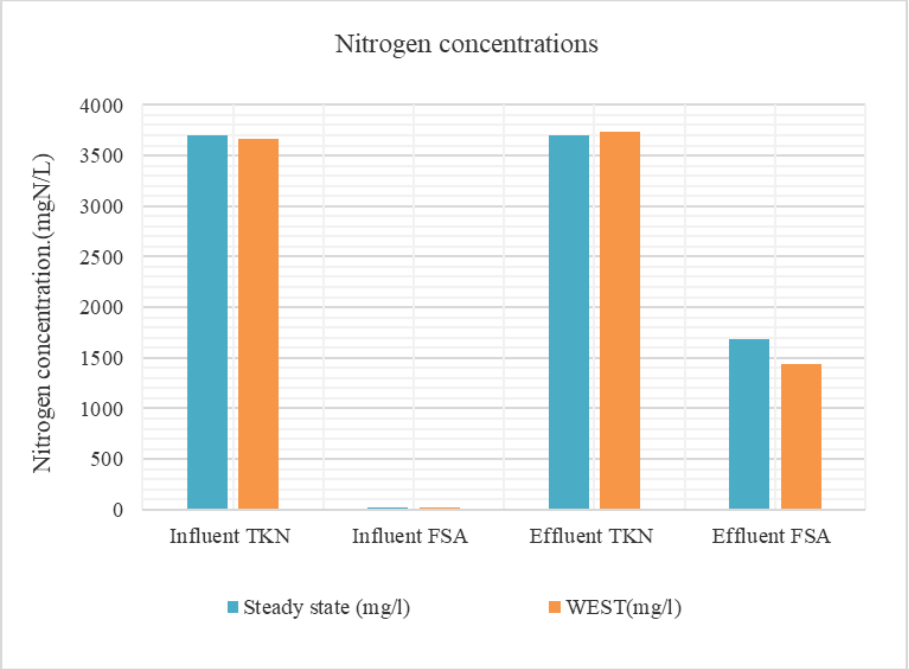

Figure 5-13: Scenario 2 — MS Excel model and PWM_SA WEST AD predictions for nitrogen concentrations

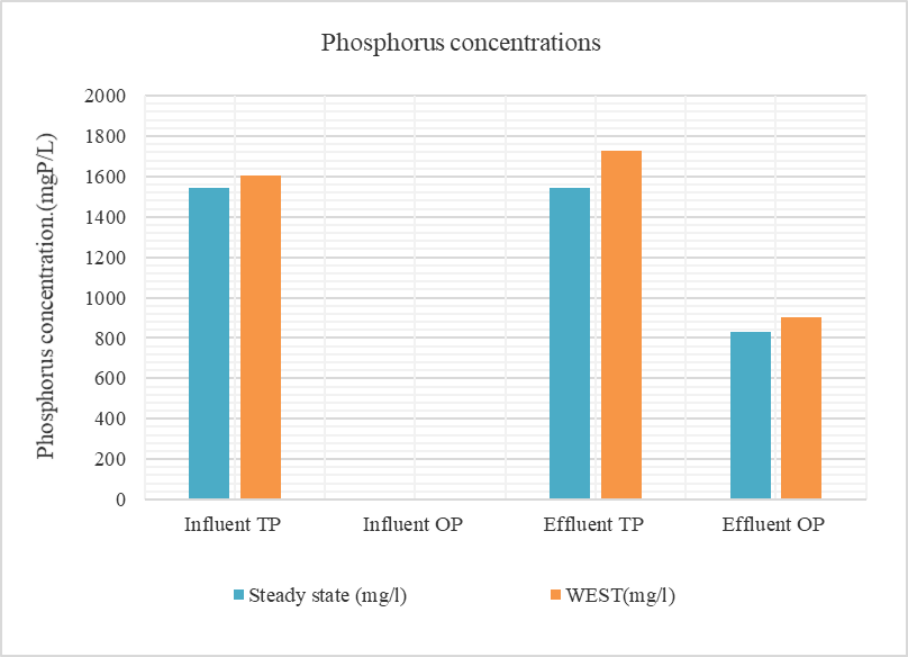

Figure 5-14: Scenario 2 — MS Excel model and PWM_SA WEST AD predictions for phosphorus concentrations

5.1.2.3 Side-stream struvite precipitation

The side-stream struvite precipitation unit was modelled such that it was the recipient to the AD effluent and to the dissolved air floatation (DAF) effluent. The latter being low in FSA and OP had a diluting effect on the influent to the precipitation unit. Consequently, the FSA concentration entering the precipitation unit was diluted to 144.6 mgN/l and 122.0 mgN/l according to the MS Excel SS model and the PWM_SA respectively. Likewise, the influent OP was diluted to 71.1 mgP/l and 76.3 mgP/l according to both models respectively.

In order to precipitate struvite, it was necessary to dose 168.49 mg/l of magnesium in the form of magnesium hydroxide (i.e., 45272.28 mg$Mg(OH)_2$/l). This resulted in the precipitation of 550.6 mg/l and 570.9 mg/l of struvite according to the MS Excel SS model and the PWM_SA respectively as shown in Figure 5-15 below. Following the precipitation of struvite, the effluent FSA was predicted to be 113.2 mgN/l and 89.4 mgN/l according to each model respectively. The effluent OP was predicted to amount to 1.5 mgP/l and 4.3 mgP/l by the MS Excel SS model and the PWM_SA respectively.

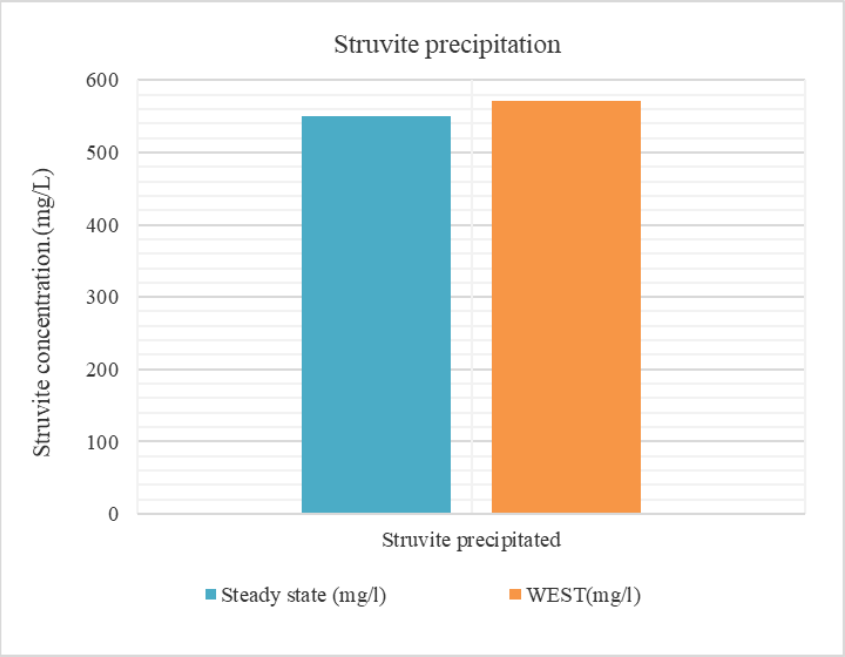

Figure 5-15: Scenario 2 — MS Excel model and PWM_SA WEST predictions for struvite precipitation

5.1.3 General comments on modelling results

The purpose of the comparison between steady-state and dynamic modelling results is for the verification of the output of the dynamic model. Dynamic models are significantly more complex than steady models and can predict time dependent system response to varying loads. In order to have confidence in the output of dynamic models for the simulation of various scenarios, it is important to ensure that the output is aligned with that of the steady state model.

As notable in the AS results, the SS model predictions of system performance match reasonably well with that simulated by the WEST® model, with some minor discrepancies. It was noted that the main reasons for the few discrepancies observed were (i) the PAO substrate allocation was difficult to match between the WEST® model and the SS spreadsheet models. In WEST®, the AS model relies on the kinetics of anaerobic fermentation of FBSO to VFA together with the kinetics of BPO hydrolysis to calculate the quantity of VFA that is available in the anaerobic zone for utilisation by biomass. This calculated VFAs is then either sequestered by PAOs (for storage as PHA; Smolders *et al.*, 1995) or taken up by facultative denitrifying OHOs (depending on the concentration of nitrates present in the 'anaerobic' zone and the kinetics of VFA uptake by the different cultures of biomass in the PHA storage or denitrification processes). In the SS spreadsheet model, the substrate to be sequestered by PAOs is calculated with the assumption that the FBSO conversion to VFA (i.e., fermentation) rate is rapid enough for all the FBSO to get utilised in the unaerated zones. However, in all the SS spreadsheet models, the nitrates recycled back into the 'anaerobic' zone are assumed to be utilised for denitrification (by OHOs) faster than the PAOs could sequester VFAs. Hence the VFAs available to PAOs anaerobically are the ones present after the reduction of nitrates. This discrepancy between the WEST® and the SS spreadsheet model has further implications on the model predictions. Firstly, if the predicted PAOs are lower (due to less anaerobic PHA uptake), the reactor VSS is reduced, since the PAOs have a much lower endogenous death rate than the OHOs (according to the death regeneration model, which is implemented in WEST®, the OHO death rate is 0.62/, while that of PAOs is 0.04/d; Henze *et al.*, 2008). Also, with less PAOs present, there is less potential for polyphosphate uptake in the aerobic zone, resulting in higher effluent OrthoP concentrations. Also impacted, is a reduced nutrient uptake, due to lower active biomass concentrations, which results in marginally higher effluent FSA.

Likewise, there are marginal differences in the AD influent data between the PWM_SA WEST® model and the SS Excel model. The main reason for these differences is the marginal discrepancies from the results in the AS virtual parent systems. In the plant-wide models (both SS Excel and PWM_SA WEST®), the AS and AD unit operations are linked directly, such that the characteristics of the WAS generated from the AS unit process are reflected as the input sludge characteristics in the AD unit process.

5.2 Impact of side-stream treatment on effluent quality

The impact on effluent quality was evaluated and quantified using the EQI developed in Section 3.2. Due to current limitations in WWTP modelling, predictions for greenhouse gas production from WWTP unit processes other than from the AD, and sludge micro-pollutant concentrations could not be made. The EQI_{gas} and EQI_{sludge} require further developments in WWTP modelling before they can be used for the respective effluent quality evaluations. Therefore, for the purposes of this project, evaluation of effluent quality was limited to EQI_{water}.

Table 5-1 shows a comparison between the predicted effluent quality for Scenarios 1 and 2 and effluent standards (*viz.* SA special standards and Waterval Water Use License).

Table 5-1: Comparison of effluent quality predictions against effluent standards

Pollutant	SA special standards (1984)	Waterval Water Use License (2011)	Scenario 1	Scenario 2
	Limit concentration (mg/l)	Limit concentration (mg/l)	Concentration (mg/l)	Concentration (mg/l)
COD	30	70	73.59	71.95
FSA	1	4	0.26	0.57
OP	1	0.7	0.39	0.02
NO	1.5	9	7.73	7.35
TSS	10	20	0.00	0.00

In order to illustrate the effect of plant-specific or regional effluent quality standards on the EQI_{water}, two cases were considered as shown in Table 5-2 below — (1) using the South African special standards for WWTP effluent discharge (Water Act, No. 54 of 1956. Regulation, 1984)

as the basis for the weighting factors in the EQI calculation, and (2) using the 2011 Waterval Water Use License (08/C22C/fg/646) as a basis for the weighting factors in the EQI calculation.

Table 5-2: EQI_{water} comparison for scenarios 1 and 2

Scenarios	SA special standards			Waterval water use license		
	EQI	EQI_{pos}	EQI_{neg}	EQI	EQI_{pos}	EQI_{neg}
Scenario 1	13696.36	3898.94	-9299.76	9782.18	9772.47	-198.37
Scenario 2	13079.65	3998.33	-8788.49	7800.82	11648.93	-107.59

For both scenarios, it was observed that the EQI calculated based on the Waterval Water Use License was significantly lower than when calculated based on the South African special standards. This is explained by the fact that the latter has stricter effluent quality requirement for discharge into water bodies. The magnitude of the EQI thus gives an indication of the amount of pollutants discharged per day based on the effluent quality requirements. For instance, the EQI for Scenario 1 based on the Waterval Water Use License was determined to be 9782.18 while when based on the South African special standards, the EQI was determined to be 13696.36. This implies that the effluent from Scenario 1 is more polluting to receiving water bodies according to the South African special standards than it is according to the Waterval Water Use License. Therefore, flexibility in pollutant weighting factors allows for the assessment and comparison of effluent quality for plant-specific or region-specific effluent quality standards.

From Table 5-2, it can be observed that side-stream struvite precipitation has a positive impact on the improvement of effluent quality. This is illustrated by Scenario 2 whereby a decrease of 20% in EQI (based on the Waterval Water Use License) was observed. The improvement in effluent quality is further apparent in the EQI_{pos} and EQI_{neg}. The simulation of Scenario 2 shows that the EQI_{pos} increases by 19%, and the EQI_{neg} decreases by 46%. The increase in EQI_{pos} indicates that the margin of difference between pollutant concentrations that are within the effluent limits and the said limits increase for Scenario 2, i.e., there is a decrease in pollutant concentrations in comparison with Scenario 1. Similarly, the decrease in EQI_{neg} is an indication that, while there are still pollutants exceeding the regulatory limits, the concentrations of these pollutants are significantly lower for Scenario 2 than for Scenario 1.

While there is merit to the EQI_{pos} and EQI_{neg} in giving and overall quantification of pollutants below and above the regulatory limits respectively, their outputs do not allow for the determination of the magnitude by which each individual pollutant is below of exceeds the regulatory limits. Therefore, in addition to the EQI_{pos} and EQI_{neg}, it was necessary to analyse each component individually as illustrated in Figure 5-16. The latter graphically represents the difference between the effluent regulatory limits and the predicted effluent concentrations. Consequently, the impact of each scenario on the level of exceedance of each pollutant can be determined. From Figure 5-16, it can be observed that the components contributing to the decrease in EQI for Scenario 2 are COD, ortho-phosphates (OP) and nitrates (NO).

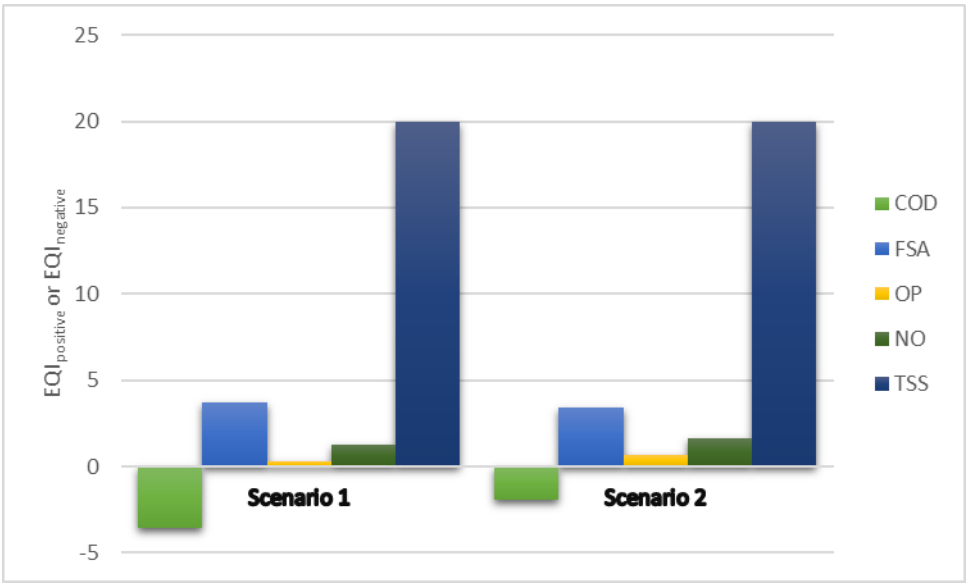

Figure 5-16: Breakdown of the components of EQI_{pos} and EQI_{neg}

5.3 Impact side-stream treatment on operational cost

The impact operational cost was evaluated using the OCI formulation developed in Section 3.3. For the purposes of this project, the parameters considered for the evaluation of operational cost include (1) aeration energy, (2) pumping energy, (3) methane production, (4) mixing energy, (5) heating energy, (6) sludge disposal, and (7) magnesium hydroxide dosing.

While consideration of fines for non-compliance was included in the evaluative framework for operational cost, the OCI calculation for the two scenarios simulated excluded non-

compliance fines. This was because South African WWTPs do not currently incur any fines for the violation of effluent limits. Calculation of fines would require the determination of an appropriate ‘unit fine’ in a South African context for pollutants exceeding the regulatory limits, which is outside the scope of this project. For other countries where non-compliance fines are applied, the proposed framework for calculation of fines may be suitable for application.

Additionally, the struvite precipitated in the side-stream precipitation unit was not taken into consideration in the OCI calculation in spite of its inclusion in the operational cost evaluative framework. This is because presently, struvite is not marketable on the South African organic agriculture market. Instead, struvite precipitated at South African WWTPs is usually donated to local farmers for animal fodder. While there is certainly a possibility that in the future struvite could become marketable as a fertiliser or in the manufacture of fire retardants, the determination of a future potential price for struvite for the South African market is not within the scope of this project.

Figure 5-17 shows a graphical breakdown of the cost components contributing to the OCI evaluation. This breakdown of costs was plotted for Scenarios 1 and 2 so that comparisons could be made, and the most economically suitable scenario could be determined.

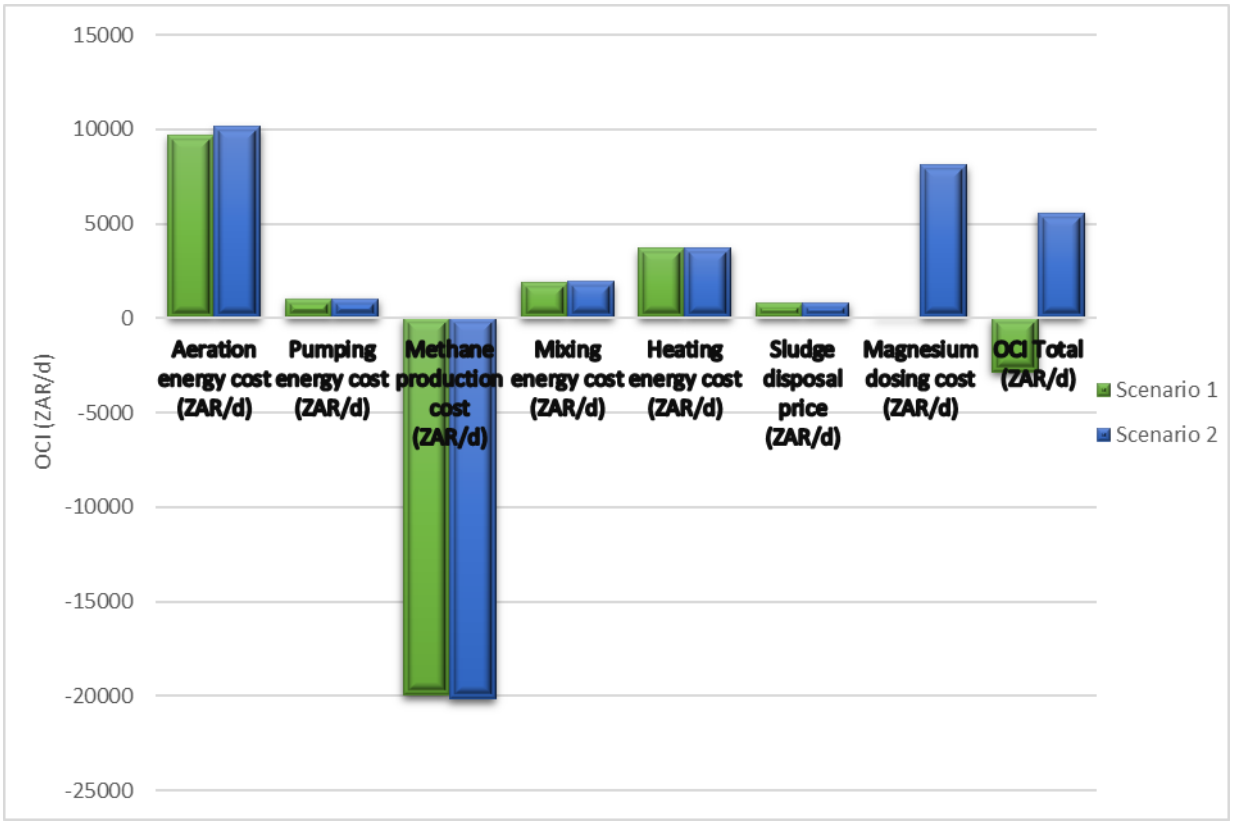

Figure 5-17: Cost comparison for scenarios 1, 2 and 3

From Figure 5-17 above, it is observed that the differences between scenarios for each cost component is marginal. While the aeration energy, methane production and magnesium dosing were the major contributors to the OCI, it is clear that the determining component in the OCI comparison is the magnesium dosing.

The OCI for Scenarios 1 and 2 were determined to be -2839.96 ZAR/d and +5557.70 ZAR/d respectively. The negative OCI value for Scenario 1 indicates that the equivalent cost recovered from methane production exceeds the total cost of the remaining components. While this is also the case for Scenario 2, the reason for the positive values is due to the cost of dosing magnesium hydroxide at a unit cost of 8.35 ZAR/kg dosed. From these results, it can be concluded that Scenario 1 is the most economical solution due to the high dosing costs of Scenario 2. However, it must be noted that fines for effluent quality violations and cost recovered from the sale of struvite products (e.g., fertiliser, fire retardants, etc.) were not included in the OCI calculations for the abovementioned reasons. Should these components become applicable

in South Africa, the results obtained for the OCI could potentially change in the favour of Scenario 2 based on associated cost parameters for those components.

5.4 Closure

In this chapter, the simulation results for the two scenarios comparing the impact of implementing side-stream struvite precipitation were presented. While an improvement in effluent quality was observed from Scenario 2 when compared to Scenario 1, the said improvement was marginal. In addition to that, the outcome of the OCI evaluation was that Scenario 1 was the most economical scenario due to the high dosing costs in Scenario 2. The EQI and OCI results indicate that Module 4 of Waterval WCW has sufficient capacity to treat the influent wastewater and that implementation of side-stream struvite precipitation in this case significantly increases the net operational costs for marginal benefits in terms of effluent quality.

6. Conclusions

The primary aim of this research was to develop an evaluative framework to perform system-wide evaluations of WRRFs in South Africa based on effluent quality and operational cost. In order to accomplish this goal, the following specific objectives were identified:

i) Review of conventional wastewater treatment plant unit operations, together with common side stream treatment technologies and the fate of products generated by the treatment process, with the aim of determining the best use of the resources recovered from waste in industrial applications.

ii) Development of formulated performance indices that include the fate of WRRF products as the evaluative framework (which is an extension of the one developed IWA BSM task group; Jeppsson *et al*., 2007) used to assess strategies in system design or operation, during application of mathematical models that virtually replicate WRRFs. The performance indices shall be included as evaluative equations in the PWM_SA (Ikumi *et al.*, 2015a). The utilisation of the modified PWM_SA model for the simulation of a full scale South African wastewater treatment shall be prepared as a case study that showcases the potential for utilisation of the model as WRRF evaluative tool.

6.1 Potential side-stream treatment technologies

Side-stream treatment is a useful means of preventing the potentially negative impacts of recycling nutrient-rich sludge dewatering liquors in WWTPs. In addition to removing nutrients from the dewatering liquor, some side-stream treatment technologies allow for the recovery of these nutrient in usable forms such as agricultural fertiliser.

The ANAMMOX and BABE processes were identified as nitrogen removal technologies that hold potential. While implementation of ANAMMOX in South Africa would require extended pre-enrichment of bacteria, successful ANAMMOX systems have shown high nitrogen removal efficiency, relatively low carbon footprint, oxygen requirements and operational costs. On the other hand, the BABE process was shown to significantly improve removal of ammonia in AS systems and lower the minimum SRT required especially at lower temperatures and is a

cost-effective upgrade to WWTPs to deal with future load increases that is expected in developing countries such as South Africa.

Lastly, struvite precipitation using Ostara's Pearl® process and the Multiform Harvest process was identified as an effective treatment method for the removal and recovery of both N and P nutrients from sludge dewatering liquor. The former produces high purity struvite crystals while the latter produces low quality sand-like struvite.

6.2 Fate of products generated

The transition towards WRRFs requires consideration of the fate of products generated and identifying pathways that reduce harm to the environment and potential recovery of products.

Greenhouse gases comprising CO_2, CH_4 and N_2O are emitted from WWTPs as a result of the various bioprocesses involved in the wastewater treatment process. From literature, three strategies have been identified to reduce greenhouse gas emissions — (1) minimisation, (2) treatment, and (3) prevention which consist of the modification of operational conditions to reduce emissions, the capture and treatment of gases and the prioritisation of anaerobic pathways for organic removal and usage of microalgae or partial nitritation-Anammox for ammonia removal (Campos *et al.*, 2016).

The disposal of sewage sludge to landfill is not a sustainable sludge management option due to the release of greenhouse gases into the atmosphere as it decomposes and due to the contamination of the surrounding environment from leachate. A more sustainable option is the use of sewage sludge in agriculture. With this option, the nutrients in the sludge mass allow it to be used as fertiliser achieving similar crop yields as chemical fertiliser at equivalent nitrogen rate applied (Suss, 1994). An alternative sludge management option is incineration which significantly reduces sludge mass, eliminates pathogens and allows for electrical energy generated while emitting 80% less greenhouse gases than hard coal power plants. (Đurđević *et al.*, 2019).

N and P nutrients can be recovered from wastewater in the form of struvite — a slow-release fertiliser. While struvite is comparable to most phosphate fertilisers on the market, organic agriculture is not a viable market route for struvite and more viable alternative include

commercial inorganic fertiliser production, animal feed and community scale gardening (Sikosana *et al.*, 2016). Alternatively, depending on the demand and need, studies about use of struvite in the manufacture of fire retardants have shown that this form of struvite can potentially generate significantly more revenue than struvite fertilisers while requiring 60% less phosphate that conventional fire retardants (Kim *et al.*, 2021).

Due to the growing concerns associated with water scarcity, the need for water reclamation has grown accordingly out of the need to alleviate the stress on existing potable water sources. Potential applications of water reclamation, each having their respective water quality standards to be met, include agricultural reuse, industrial reuse, urban reuse, and indirect and direct potable reuse. However, until the obstacles caused by public perception and health hazards are overcome, the reuse of wastewater effluent will be limited to non-potable applications which exclude its use on crops intended for food production.

6.3 Extensions to existing performance indices

The performance indices developed by the IWA BSM task group were extended to fit the concept of resource recovery. The effluent quality which previously only considered the impact of effluent discharge into water bodies now considers greenhouse gas emissions as well as sludge quality. Thus, three formulations for the EQI were developed which comprise the EQI_{water}, EQI_{gas} and EQI_{sludge}.

On the other hand, the OCI was extended to include fines for effluent quality violations, costs of dosing metals or lime in the nutrient recovery process and the potential cost recovered from the nutrients recovered.

6.4 Waterval WCW case study

The comparative evaluation of control strategies, using performance tools has proved to be an effective procedure to evaluate WRRF performance on a system-wide level. Hence, providing ways to troubleshoot prospective scenarios and suggest feasible control options. The extended performance indices framework is currently being implemented, as an evaluative protocol, in the

simulation of all considered strategic scenarios in order to determine the most impactful in future decision making for the given WRRF systems.

This report presents the scenario analysis for one treatment module of a full-scale system of Waterval WCW, whereby the two scenarios comparing the impact of implementing side-stream struvite precipitation were presented. From the simulated results, it was noted that Module 4 of Waterval WCW has sufficient capacity to treat the influent wastewater as well as the recycled sludge dewatering liquors generated from that module to acceptable quality with respect to the standards of the Waterval Water Use License (08/C22C/fg/646). Thus, the implementation of side-stream struvite precipitation in this case significantly increases the net operational costs for marginal benefits in terms of effluent quality.

6.5 Future modelling prospects

It has been noted that wastewater treatment systems are part of the larger water and sanitation systems and for holistic decision making to occur, the mathematical models often have to be extended to beyond the fence of the wastewater treatment plant. This is important to promote the changing paradigms that encourage sustainable infrastructure, such as ensuring the change of wastewater treatment systems to WRRFs, and the conversion of water stressed regions to water sensitive zones. In this project, the evaluative framework for models has been extended to include the fate of products generated by waste treatment processes. This is to ensure that decisions on development of future systems considers the products (resources) generated by waste treatment. The evaluative framework is an extension of the IWA BSM version, which uses the effluent quality and operational cost indices (i.e., EQI and OCI). However, it was noted that in order for this evaluative framework to be more effective, the models shall require modifications, such that the predictions made regarding system performance include the variables of significance, related to prioritised criteria for the products generated by the system. For example, the sludge generated as fertiliser would require for the model to predict its quality in terms of the heavy metal content to present the quality for use (which can be connected to environmentally sustainability) and prospective market price (which is linked to economic sustainability). Similarly, the quality of the final effluent could be predicted in terms of the environmental impact and (for countries that promote fines) the cost implications and also, where tertiary treatment for water reclamation is

included, the cost benefit of recovering effluent water. Hence it can be noted, for system-wide models, the EQI is linked to the OCI and the manner in which these are weighed against each other (i.e., does it cost more or less to have poor design and operated infrastructure if it does then there may be some modifications needed for policies that govern water and sanitation infrastructure). This way the mathematical models could begin revealing the aspects that need to be addressed for improvement in sustainable wastewater treatment systems, hence providing guidance towards decision making at various levels. For the mathematical models to be trusted, they must be developed from a scientifically sound basis, hence the extension of these models is an ongoing process that requires continuous development and calibration of components that make up the water and sanitation infrastructure.

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Appendix A : Raw data

Table A-1: Waterval WCW influent flow rate data (September 2015 - August 2016)

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
01/09/2015	270	21.7	97.8	101.4	49.1
02/09/2015	258.3	23.6	89.9	98.2	46.6
03/09/2015	254.8	21	91.2	95.1	47.5
04/09/2015	242	19.8	85.3	89.9	47
05/09/2015	208	22.9	72.1	71.2	41.8
06/09/2015	200.7	24.3	60.2	72.9	43.3
07/09/2015	259.4	19.8	86.7	102.4	50.5
08/09/2015	273.9	16.4	105.2	107.2	45.1
09/09/2015	274.8	18.4	102.7	107.6	46.1
10/09/2015	271.8	18.3	100.9	105.4	47.2
11/09/2015	282.9	18.8	104.6	109.5	50
12/09/2015	258.6	16.5	97.5	95.9	48.7
13/09/2015	250.9	19.2	86.9	97.8	47
14/09/2015	267.1	20.3	93.6	103.5	49.7
15/09/2015	245.8	19.8	75.2	100.2	50.6
16/09/2015	300.2	21.8	112	123.1	43.3
17/09/2015	295.9	24.4	109.8	115.1	46.6
18/09/2015	292.6	23.3	110.5	109.5	49.3
19/09/2015	290	22.1	104.3	115.8	47.8
20/09/2015	275.2	22.1	98.5	107.3	47.3
21/09/2015	262.5	21.8	96.9	95.7	48.1
22/09/2015	230.1	16.6	76.7	83.7	53.1
23/09/2015	199.5	16.4	66.4	65.4	51.3
24/09/2015	213.4	17.9	70.4	71.6	53.5
25/09/2015	194.7	31.9	53.9	52.1	56.8
26/09/2015	161.9	26.8	45.7	41.4	48
27/09/2015	182.5	17.6	54.1	62.7	48.1
28/09/2015	197.8	17.6	60.1	66.6	53.5
29/09/2015	184.5	18.9	59.7	60.3	45.6
30/09/2015	189.4	15.1	62.7	63.4	48.2
01/10/2015	203.1	17.5	64.5	69.9	51.2
02/10/2015	208	22.9	72.1	71.2	41.8

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
03/10/2015	200.7	24.3	60.2	72.9	43.3
04/10/2015	259.4	19.8	86.7	102.4	50.5
05/10/2015	273.9	16.4	105.2	107.2	45.1
06/10/2015	274.8	18.4	102.7	107.6	46.1
07/10/2015	271.8	18.3	100.9	105.4	47.2
08/10/2015	282.9	18.8	104.6	109.5	50
09/10/2015	258.6	16.5	97.5	95.9	48.7
10/10/2015	250.9	19.2	86.9	97.8	47
11/10/2015	267.1	20.3	93.6	103.5	49.7
12/10/2015	245.8	19.8	75.2	100.2	50.6
13/10/2015	300.2	21.8	112	123.1	43.3
14/10/2015	295.9	24.4	109.8	115.1	46.6
15/10/2015	292.6	23.3	110.5	109.5	49.3
16/10/2015	290	22.1	104.3	115.8	47.8
17/10/2015	275.2	22.1	98.5	107.3	47.3
18/10/2015	262.5	21.8	96.9	95.7	48.1
19/10/2015	230.1	16.6	76.7	83.7	53.1
20/10/2015	199.5	16.4	66.4	65.4	51.3
21/10/2015	213.4	17.9	70.4	71.6	53.5
22/10/2015	194.7	31.9	53.9	52.1	56.8
23/10/2015	161.9	26.8	45.7	41.4	48
24/10/2015	182.5	17.6	54.1	62.7	48.1
25/10/2015	197.8	17.6	60.1	66.6	53.5
26/10/2015	184.5	18.9	59.7	60.3	45.6
27/10/2015	189.4	15.1	62.7	63.4	48.2
28/10/2015	203.1	17.5	64.5	69.9	51.2
29/10/2015	212.4	15.93	72.8	73.57	50.1
30/10/2015	181.8	13.6	62.1	61.2	44.9
31/10/2015	188	14.1	62.4	65.4	46.1
01/11/2015	184.9	17	53.5	64.3	50.1
02/11/2015	183.7	18.9	56.9	60.2	47.7
03/11/2015	197.2	19.9	58.9	68	50.4
04/11/2015	168.8	20.3	48.9	48.9	50.7
05/11/2015	205.9	20.8	65.4	69.4	50.3
06/11/2015	211.1	20.9	67.3	71.2	51.7

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
07/11/2015	233.8	25.3	75.2	85.2	48.1
08/11/2015	214.6	23.1	68.9	72.6	50
09/11/2015	192.5	18.6	62.1	65.2	46.6
10/11/2015	180.8	20.1	62.1	56.7	41.9
11/11/2015	164.5	18.8	47.3	53.7	44.7
12/11/2015	169.2	18.8	48.1	55	47.3
13/11/2015	155.7	20.4	43.1	47.4	44.8
14/11/2015	161.3	28.6	42.1	46	44.6
15/11/2015	149.5	27	36.9	42.1	43.5
16/11/2015	209.8	27	54.6	62.4	65.8
17/11/2015	246.9	29	78.9	83.7	55.3
18/11/2015	205.62	15.42	63.7	64.4	62.1
19/11/2015	243.14	18.24	78.9	86.9	59.1
20/11/2015	212.4	15.93	72.8	73.57	50.1
21/11/2015	181.8	13.6	62.1	61.2	44.9
22/11/2015	188	14.1	62.4	65.4	46.1
23/11/2015	179.7	13.5	60.1	61	45.1
24/11/2015	170.1	12.8	55.1	59.6	42.6
25/11/2015	176.6	13.3	56.7	63.1	43.5
26/11/2015	178.1	13.4	58.9	61.8	44
27/11/2015	176.9	13.3	59.7	60.7	43.2
28/11/2015	188.3	14.1	63.2	64.4	46.6
29/11/2015	199.9	14.9	65.8	72.2	47
30/11/2015	166	16.7	50	52	47.3
01/12/2015	176.2	16.7	57.7	59.1	42.7
02/12/2015	212.1	21.1	71.4	66.4	53.2
03/12/2015	215.8	18.3	74.2	69.9	53.4
04/12/2015	175	16.3	58.9	56.4	43.4
05/12/2015	166.3	16.5	45.4	62.6	41.8
06/12/2015	152.1	16.4	45.1	50.3	40.3
07/12/2015	153.9	24.8	45.2	47.2	36.7
08/12/2015	167.3	20.5	52.1	51.2	43.5
09/12/2015	190.5	21.9	71.6	53.2	43.8
10/12/2015	150.3	17.6	45.4	45.7	41.6
11/12/2015	150.4	17.2	42.9	44.5	45.8

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
12/12/2015	142.9	15.9	40.4	43.6	43
13/12/2015	148.4	14	47.8	43.9	42.7
14/12/2015	164.7	28.8	49.8	36.4	49.7
15/12/2015	157.6	14.6	50.3	54.2	38.5
16/12/2015	150.9	18.2	42.5	39.4	50.8
17/12/2015	195	16.6	57.7	64.3	56.4
18/12/2015	185.6	14.4	60.3	57.3	53.6
19/12/2015	201.9	22.1	64.5	57	58.3
20/12/2015	164.6	14.8	53.2	48.1	48.5
21/12/2015	221.2	18.1	72.4	67.3	63.4
22/12/2015	168.4	16.1	52.8	51.2	48.3
23/12/2015	150.6	15.1	43.5	46.3	45.7
24/12/2015	135.8	14.2	41.7	42.9	37
25/12/2015	131.76	14.6	41.3	39.98	35.88
26/12/2015	173.8	15.6	55.2	54.4	48.6
27/12/2015	151.8	16.3	45.3	43.9	46.3
28/12/2015	150.5	15	48.7	46.6	40.2
29/12/2015	146.7	14.8	44.2	45.1	42.6
30/12/2015	138.8	14.4	40.6	43.4	40.4
31/12/2015	134.32	13.82	41.9	38.8	39.8
01/01/2016	161.9	15.7	51.4	49.1	45.7
02/01/2016	149.6	13.2	52.5	46.6	37.3
03/01/2016	141	12.9	40.6	44.2	43.3
04/01/2016	139.2	12.8	43.9	42.5	40
05/01/2016	138.8	14.3	40.5	43.9	40.1
06/01/2016	139	11.2	46.1	42.8	38.9
07/01/2016	138.8	14.8	37	40	47
08/01/2016	175.1	17.4	59.9	51.4	46.4
09/01/2016	159.9	15.8	51.4	45.6	47.1
10/01/2016	169.5	16.9	54.3	49.4	48.9
11/01/2016	193.3	16.5	65.4	53.2	58.2
12/01/2016	230.8	9.9	94.2	62.4	64.3
13/01/2016	178.9	13.2	59.9	52.3	53.5
14/01/2016	168.8	12.3	57.6	48.7	50.2
15/01/2016	173.9	12.4	61.7	48.8	51

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
16/01/2016	165.9	12.4	57.2	47.4	48.9
17/01/2016	161.8	12.1	54.7	46.9	48.1
18/01/2016	151.1	11.9	46.8	47.9	44.5
19/01/2016	139.4	11.3	46.8	41.5	39.8
20/01/2016	172.7	12.8	57.3	49.9	52.7
21/01/2016	182.2	13	61.8	49.6	57.8
22/01/2016	191.2	11.9	73.8	48.5	57
23/01/2016	222.8	22.8	70.4	56.6	73
24/01/2016	268.1	22.3	88	80.3	77.5
25/01/2016	206.1	18.5	64.9	57.8	64.9
26/01/2016	187.1	17.6	57.8	55.6	56.1
27/01/2016	192.2	18.9	56.1	57.5	59.7
28/01/2016	174.5	17.3	50.4	52.6	54.2
29/01/2016	164.9	17.6	48.9	48	50.4
30/01/2016	163.1	16.2	48.9	50.1	47.9
31/01/2016	165	18.4	47.9	50.4	48.3
01/02/2016	159.7	20.5	50.2	42.6	46.4
02/02/2016	129.4	11.3	49	29.5	39.6
03/02/2016	150	16.5	48.6	40.1	44.8
04/02/2016	146.5	15.4	42.3	42.3	46.5
05/02/2016	172.6	16.7	59	44.8	52.1
06/02/2016	183.9	18.4	58.5	50.6	56.4
07/02/2016	158.3	16.8	51.2	42.9	47.4
08/02/2016	171.6	17.2	45.5	56.2	52.7
09/02/2016	154.8	16.6	49.9	40.9	47.4
10/02/2016	156.3	20	48.5	38	49.8
11/02/2016	156.7	20.3	49.4	35.5	51.5
12/02/2016	158.8	19.3	49.7	39.2	50.6
13/02/2016	166.9	19	47.9	34.4	65.6
14/02/2016	234.4	17.2	54.5	91.2	71.5
15/02/2016	212.1	14.8	64.2	73.9	59.2
16/02/2016	166.4	13.9	57.3	39.6	55.6
17/02/2016	188.5	20.1	68	33.1	67.3
18/02/2016	227.1	12.4	75.1	65	74.6
19/02/2016	201.7	24.6	69.4	52.9	54.8

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
20/02/2016	180.5	24.1	64.4	41.3	50.7
21/02/2016	198.7	25.5	67.4	48.4	57.4
22/02/2016	181.3	23.3	64.4	44.9	48.7
23/02/2016	186.7	23.3	60.5	58.4	44.5
24/02/2016	214.9	26.5	72.5	60.8	55.1
25/02/2016	208.3	24.9	72.6	56.9	53.9
26/02/2016	191.9	24.3	63.5	53.2	50.9
27/02/2016	200.2	25.1	64.8	56.2	54.1
28/02/2016	174.1	22.5	58.9	45.2	47.5
29/02/2016	173.6	22.4	59.8	44.4	47
01/03/2016	171.5	22.1	54.4	48.6	46.4
02/03/2016	172.1	22.7	54.3	48.7	46.4
03/03/2016	167.8	22.1	52.2	48.1	45.4
04/03/2016	166.3	20.4	55.2	46	44.7
05/03/2016	172.9	19.5	58.2	50.7	44.5
06/03/2016	178.5	18.7	61.9	49.5	48.4
07/03/2016	163	21.8	51.7	48.5	41
08/03/2016	175.3	22.7	56.6	49.4	46.6
09/03/2016	233.6	24.3	81.4	59.7	68.2
10/03/2016	190.5	17.7	28	67.6	77.2
11/03/2016	368.4	15.7	173.1	112	67.6
12/03/2016	257.2	15	108.1	66.7	67.4
13/03/2016	255.1	14.8	107.1	68.1	65.1
14/03/2016	217.6	19	80.8	61.4	56.4
15/03/2016	220.4	17.6	87.9	57.4	57.5
16/03/2016	253.7	19.9	102.6	66.6	64.6
17/03/2016	311.5	14	143.2	80.4	73.9
18/03/2016	269.6	17.7	111.7	70.9	69.3
19/03/2016	260.4	17.6	101.3	74.3	67.2
20/03/2016	266.8	18.4	109.4	72.4	66.6
21/03/2016	253.9	17.8	97.4	73.2	65.5
22/03/2016	234.4	16.1	79.3	62.4	76.6
23/03/2016	234.4	15.1	75.6	56.9	86.8
24/03/2016	186.4	15.9	46.2	57.5	66.8
25/03/2016	223.9	16.4	72.4	62	73.1

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
26/03/2016	176.7	13.9	55.1	49.1	58.6
27/03/2016	191.4	14.3	61.1	52.6	63.4
28/03/2016	199.7	15.8	68.4	55.4	60.1
29/03/2016	180	15.1	55.8	50.5	58.6
30/03/2016	193	14.1	59.9	53.9	65.1
31/03/2016	173.3	14.7	45.9	52.3	60.4
01/04/2016	186.8	18.8	55.7	47.6	64.7
02/04/2016	171.9	19.9	43.5	48.6	59.9
03/04/2016	215.7	21.5	59.8	55.5	78.9
04/04/2016	196.3	20.2	50.2	51.2	74.7
05/04/2016	183	19.1	47.9	50.1	65.9
06/04/2016	183.2	19.1	54.3	48.1	61.7
07/04/2016	163.1	17.9	50.7	48.2	46.3
08/04/2016	188.5	20.8	54.8	51.8	61.1
09/04/2016	170.6	21.6	44.5	47.1	57.4
10/04/2016	171.2	18.6	42.8	47.4	62.4
11/04/2016	159.5	20.4	35.5	47.4	56.2
12/04/2016	150.6	17.3	40.5	43.8	49
13/04/2016	163	17.9	42	48.6	54.5
14/04/2016	143.9	13.2	36.2	42.9	51.6
15/04/2016	173.1	26	49.2	38.9	59
16/04/2016	153.4	15.1	49.2	30.5	58.6
17/04/2016	159.4	15.5	38.7	45.1	60.1
18/04/2016	157.9	16.8	38	41.5	61.6
19/04/2016	153.4	13.3	38.8	39.2	62.1
20/04/2016	160.2	17.2	34.2	39.1	69.7
21/04/2016	174.9	18.9	39.5	39.3	77.2
22/04/2016	170.5	12.1	32.5	38.2	87.7
23/04/2016	184.6	14.5	54.4	50.5	65.2
24/04/2016	165.6	13.3	48.8	48.6	54.9
25/04/2016	164.4	13.3	49	48.3	53.8
26/04/2016	185.4	14.1	56.8	52.3	62.2
27/04/2016	233	10.3	70.4	77.9	74.4
28/04/2016	271.3	9.4	66.5	112.8	82.6
29/04/2016	275.2	9.4	63	122.2	80.6

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
30/04/2016	276.2	8.5	64.4	122.7	80.6
01/05/2016	156.9	13.4	50.2	41.6	51.7
02/05/2016	164.6	13.9	51.6	47.6	51.5
03/05/2016	162.9	13.3	47.3	46.9	55.4
04/05/2016	163.2	13.7	48.3	46.4	54.8
05/05/2016	169.3	13.6	51.9	48	55.8
06/05/2016	164.9	13.4	51.7	45.3	54.5
07/05/2016	165.6	13.9	51.9	45.6	54.2
08/05/2016	163	13.5	48.4	47.8	53.3
09/05/2016	170.4	13.8	52.5	48.3	55.8
10/05/2016	165.9	13	52.4	47.3	53.2
11/05/2016	168.3	14.5	53.6	46.3	53.9
12/05/2016	168.9	13.6	54.2	46.7	54.4
13/05/2016	165.6	13.8	53.1	45.7	53
14/05/2016	259.4	16.7	78.1	80.5	84.1
15/05/2016	267.8	16.5	81.9	80.9	88.5
16/05/2016	213.6	14.5	68.6	61.8	68.7
17/05/2016	181.6	14.6	58.9	49.6	58.5
18/05/2016	182.3	14.5	58.9	50.5	58.4
19/05/2016	180.7	13.9	58.1	50.9	57.8
20/05/2016	176.9	13.7	56.5	50	56.7
21/05/2016	180.9	14.3	57.9	51.1	57.6
22/05/2016	182.2	14.5	57.5	54.3	55.9
23/05/2016	184	10.4	59.1	59.6	54.9
24/05/2016	179.5	13	60.1	50.1	56.3
25/05/2016	184.1	12.5	63.7	50.2	57.7
26/05/2016	198.5	13.6	69	52.4	63.5
27/05/2016	196.6	13.2	66.7	52	64.7
28/05/2016	197	12.9	68	55.5	60.6
29/05/2016	203	13.2	70.4	53.4	66
30/05/2016	189.8	13.1	62.1	55.2	59.4
31/05/2016	191.9	12.1	67	53.5	59.3
01/06/2016	187.4	13.3	63.5	52	58.6
02/06/2016	185.9	15.5	63.5	47.1	59.8
03/06/2016	187.4	13.3	63.5	52	58.6

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
04/06/2016	185.9	15.5	63.5	47.1	59.8
05/06/2016	202.7	13.8	64.1	65.6	59.2
06/06/2016	206.9	14.3	66.4	65.7	60.5
07/06/2016	219.3	14.6	68.1	66.7	69.9
08/06/2016	190.9	11.9	64	63.6	51.4
09/06/2016	200.6	14.4	65.4	61.5	59.3
10/06/2016	193.1	13	60.5	61.6	58
11/06/2016	208.8	14.4	67.9	62.4	64.1
12/06/2016	212.9	13.1	67.9	72	59.9
13/06/2016	200.2	14.2	62	66.3	57.7
14/06/2016	184.7	13.8	61.4	54.4	55.1
15/06/2016	201.2	13.7	62	65.6	59.9
16/06/2016	180.3	15.3	54.5	55.7	54.8
17/06/2016	171.5	16.4	55.2	48	51.9
18/06/2016	188.7	17.9	60.6	53.5	56.7
19/06/2016	185.2	17.3	58.8	54.2	54.9
20/06/2016	183.1	16.7	55.6	56.1	54.7
21/06/2016	186.8	16.8	57.2	57.7	55.1
22/06/2016	186	17.2	52.1	60.5	56.2
23/06/2016	188.4	14.5	48.2	69.1	56.6
24/06/2016	183.9	13	60.2	58.2	52.5
25/06/2016	195.1	10.7	61.7	65.9	56.8
26/06/2016	185.2	12.4	72.1	43.8	56.9
27/06/2016	181.4	10.8	54.9	57.8	57.9
28/06/2016	182.6	6.5	57.4	64.1	54.6
29/06/2016	192.1	10.5	59.4	61.1	61.1
30/06/2016	184	10.5	54.9	59.4	59.2
01/07/2016	186.1	10.1	60.2	54.9	60.9
02/07/2016	207.6	5.1	64.8	75.8	61.9
03/07/2016	198	10.1	62.4	66	59.5
04/07/2016	201.6	11.4	63.9	66.2	60.1
05/07/2016	198.3	11.6	64.8	70.2	51.7
06/07/2016	218.6	9.9	67.9	74.3	66.5
07/07/2016	233	10.3	70.4	77.9	74.4
08/07/2016	271.3	9.4	66.5	112.8	82.6

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
09/07/2016	275.2	9.4	63	122.2	80.6
10/07/2016	276.2	8.5	64.4	122.7	80.6
11/07/2016	256.5	6.8	65.5	124.3	59.9
12/07/2016	281.1	10.7	66.3	123.1	81
13/07/2016	247.4	7.7	60.4	105.9	73.4
14/07/2016	228.6	13.5	54.4	98.1	62.6
15/07/2016	227.9	15.5	57.4	95.5	59.5
16/07/2016	224.2	14.1	58.7	91.2	60.2
17/07/2016	267	11.4	71.4	119.6	64.6
18/07/2016	259.7	16.2	70.5	107.7	65.3
19/07/2016	264.4	9.5	71.8	118.2	64.9
20/07/2016	255.8	13.4	73.1	115.2	54.1
21/07/2016	342	8.6	62.7	211.3	59.4
22/07/2016	239.5	13.8	65.3	99.7	60.7
23/07/2016	193.6	12.5	67.2	54.4	59.5
24/07/2016	211.6	16.4	68.2	69.1	57.9
25/07/2016	217.9	13.6	70.1	76.3	57.9
26/07/2016	237.6	9.9	75.3	89.6	62.8
27/07/2016	205.9	10.6	67.1	78	50.2
28/07/2016	199.8	13.9	64.3	70.1	51.5
29/07/2016	201.7	10.5	65.1	74.7	51.4
30/07/2016	209.8	9.1	68.1	80.1	52.5
31/07/2016	211.3	9.8	70.3	78.8	52.4
01/08/2016	204.8	9.5	71.8	72.2	51.3
02/08/2016	187.1	11.6	73.9	51.8	49.8
03/08/2016	219.7	11.7	81	64.9	62.1
04/08/2016	211.2	11.6	75.6	68.7	55.3
05/08/2016	208.1	10.6	73.5	68.7	55.3
06/08/2016	219.4	12.5	78	70.4	58.5
07/08/2016	212.9	12.3	79	67.3	54.3
08/08/2016	196.7	11.3	67	66	52.4
09/08/2016	211.8	10.7	75.7	70.5	54.9
10/08/2016	209	10.4	70.3	73.6	54.7
11/08/2016	204.9	10.9	68.2	71.2	54.6
12/08/2016	214	10.7	74.9	71.2	57.2

Dates	Total inflow (Ml/d)	Module 1 (Ml/d)	Module 2 (Ml/d)	Module 3 (Ml/d)	Module 4 (Ml/d)
13/08/2016	214	10.7	74.3	72.2	56.8
14/08/2016	214.1	11.8	75.9	70.7	55.7
15/08/2016	203	11.2	71.9	66.8	53.1
16/08/2016	199.8	11.3	68	66.6	53.9
17/08/2016	213.5	10.9	75.4	72.3	54.9
18/08/2016	218.3	11.2	73.8	75.2	58.1
19/08/2016	204.2	9.8	67.3	73.2	53.9
20/08/2016	205.9	10.7	65.7	76.2	53.3
21/08/2016	210.8	9.9	71.6	74.2	55.1
22/08/2016	204.3	10	70.6	72.5	51.2
23/08/2016	205.6	9.4	73.9	69.6	52.7
24/08/2016	209	9.9	69.7	74.5	54.9
25/08/2016	204.6	9.7	68.4	73.3	53.2
26/08/2016	204.1	9.8	69	70.7	54.6
27/08/2016	205.6	9.8	69.8	71.8	54.2
28/08/2016	208.1	9.6	72.3	72.8	53.4
29/08/2016	194.9	10.3	63.4	71.2	50
30/08/2016	185.9	10	58.5	70	47.4

Table A-2: Raw concentrations (September 2015 - August 2016)

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
01/09/2015	33.8	1149	7.1	4.5	299		321		270
02/09/2015	28.7		7	3.4	492		273		258.3
03/09/2015	27.2	879	6.9	2.6	377		287		254.8
04/09/2015	25.7		6.9	4.2	382		234		242
05/09/2015	22.3	460	6.9	2.2	71		178		208
06/09/2015	22	809	6.9	2.1	314		205		200.7
07/09/2015	25.5		7	2.8	335		261		259.4
08/09/2015	25.6	761	6.9	3.2	182		269		273.9
09/09/2015	31.1	977	7	3.9	349	55.5	305	8.9	274.8
10/09/2015	28	755	6.9	3.1	184		276		271.8
11/09/2015	33.3	1381	6.6	5	371		284		282.9

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
12/09/2015	31.7	1248	6.6	5			321		258.6
13/09/2015	26.9	744	7	3.1	281		265		250.9
14/09/2015	29.9	487	7.3	3.3	34		288		267.1
15/09/2015	27.5	1070	7	3.9			269		245.8
16/09/2015	29.4	1326	6.9	2.8	441	45.1	291	6.6	300.2
17/09/2015	30.4	876	6.8	3.1	90		294		295.9
18/09/2015	29.9	722	6.9	2.8			298		292.6
19/09/2015	26.7	1331	6.6	2.7			285		290
20/09/2015	27.4	1496	7	3.7	500		289		275.2
21/09/2015	26.8	1237	7	3	321		291		262.5
22/09/2015	27.4	872		3.2	488		331		230.1
23/09/2015	31	952	7.2	3.9	441		317		199.5
24/09/2015	27.4	963	6.7	3.6	175		312		213.4
25/09/2015	34	648		4.1					194.7
26/09/2015	29.1	630	6.8	3.8			311		161.9
27/09/2015	28.8	1057	7	3	312		288		182.5
28/09/2015	30.1	617	7	3.5			262		197.8
29/09/2015	28.4	412	7	3.1	94		284		184.5
30/09/2015	21.5	1521	6.8	2.2	450	48.6	298	7.5	189.4
01/10/2015	36.5	1395	7	3.1	280		292		203.1
02/10/2015	26.3	735	6.8	3	250		288		208
03/10/2015	26.3	727	6.8	2.4	322		266		200.7
04/10/2015	27.3	1203	6.9	2.9	313		276		259.4
05/10/2015	30.2	808	7.1	3.2	275		279		273.9
06/10/2015	23.3	529	6.7	3.3	297		237		274.8
07/10/2015	26.2	629	7.3	2.7	210	34.6	241	3.7	271.8
08/10/2015	26.9	1123	7.2	3	235		281		282.9
09/10/2015	23	538	6.4	2.5	143		231		258.6
10/10/2015	15.4	612	6.7	1.2	166		191		250.9
11/10/2015	30.3		6.9	3.5	344		284		267.1
12/10/2015	30.1	768	7.3	3.7	187		303		245.8
13/10/2015	29.7	1400	6.6	3.4	267		303		300.2
14/10/2015	28.3		7	3.1	321		295		295.9

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
15/10/2015	21.1	882	7.3	2.5	294		298		292.6
16/10/2015	28.1	1060	6.9	2.7	431		288		290
17/10/2015	25	1210	6.7	3	354		268		275.2
18/10/2015	23.5	802	6.7	2.1	113		245		262.5
19/10/2015	33.6	607	7.5	3.8	180		292		230.1
20/10/2015	27.5	1188	6.9	4	323		306		199.5
21/10/2015	29.3	799	7.2	3.5	225	55.5	301	5.1	213.4
22/10/2015	27.4	1560	7	3.4	128		341		194.7
23/10/2015	23.8	531	6.8	2.1	123		271		161.9
24/10/2015	29.7	763	6.8	3	303		290		182.5
25/10/2015	26.9	1209	7.3	4.9	61		263		197.8
26/10/2015	33.8	684	7	3.4	147		303		184.5
27/10/2015	25.2	657	7.4	3.7	125		342		189.4
28/10/2015	27.4	529	7.2	3.1	98	38.8	287	5.3	203.1
29/10/2015	21.2	664	7	2.8	213		324		212.4
30/10/2015	24	828	7.1	3.2	154		305		181.8
31/10/2015	26.3	1142	7	4.4	83		299		188
01/11/2015	30.2	956	7.2	5.3	379		327		184.9
02/11/2015		715	7.3		163		322		183.7
03/11/2015	26.7	1064	7.5	3.7	279		276		197.2
04/11/2015	26.6	639	7.7	3.2	146	56.5	315	7.3	168.8
05/11/2015	29.5	1098	7.3	4.2	274		282		205.9
06/11/2015	25.9	1453	6.4	3.5	473		297		211.1
07/11/2015	27.6	1083	7.2	3.5	447		309		233.8
08/11/2015	30.9	1479	7.3	3.9	300		293		214.6
09/11/2015	31.6	750	7.4	4.3	170		310		192.5
10/11/2015	29.1		7.1	3.8	278		301		180.8
11/11/2015	31.8		7.5	4.2	322	33.2	282	6.1	164.5
12/11/2015	34.3	838	7.6	1.8	288		255		169.2
13/11/2015	27.9	790	6.6	2.5	324		289		155.7
14/11/2015	28.9	799	6.3	3.5	383		273		161.3
15/11/2015	25.9	680	7.4	2.4	419		235		149.5
16/11/2015	28.4	770	7.4	4.2	328		237		209.8

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
17/11/2015	25.4		7.5	3	232		171		246.9
18/11/2015	25.7	1387	7.4	3.1	403		224	7.2	205.62
19/11/2015	25.6	1169	6.8	2.8	454		243		243.14
20/11/2015		797	6.9		388		227		212.4
21/11/2015	26.6	748	6.9	3.2	335		233		181.8
22/11/2015	27.1	1464	7.4	3.2	289		269		188
23/11/2015	26.4	660	7.1	2.8	209		233		179.7
24/11/2015	36	921	7.2	3.6	275		234		170.1
25/11/2015	28.8	687	7.4	3.4	190	55.5	219	7.1	176.6
26/11/2015	26.4	919	7.3	2.8	248		249		178.1
27/11/2015	25.6	740	7.1	3.5	130		258		176.9
28/11/2015	25.9	809	7.3	3.4	144		262		188.3
29/11/2015	27.7	1023	7.1	3.8	266		273		199.9
30/11/2015	25.4	1133	7.3	3.1	274		303		166
01/12/2015	25.5	1038	7.5	3.1	332		277		176.2
02/12/2015	25.8	858	7.3	2.9	245	66.5	265	7.5	212.1
03/12/2015	20.4	1090	6.7	2.5	357		254		215.8
04/12/2015	24.8	554	7.4	2.6	115		243		175
05/12/2015	26.2	644	7.1	3.1	220		274		166.3
06/12/2015	24.5	616	7.3	5.5	271		280		152.1
07/12/2015	41.1	949	7.4	5.2	366		300		153.9
08/12/2015	28.9	680	7.4	3.6	270		262		167.3
09/12/2015	32.9	762	6.6	3.7	197	70.5	267	9.2	190.5
10/12/2015	30.7	581	6.7	3.6	150		261		150.3
11/12/2015	29.9	687	6.9	4	167		258		150.4
12/12/2015	25.8	728	6.5	4.6	240		256		142.9
13/12/2015	30.3	1092	7.3	5.4	188		241		148.4
14/12/2015	29.5	671	7.4	3.9	151		259		164.7
15/12/2015	26.9	785	6.7	3.4	294		268		157.6
16/12/2015	18.2	593	5.8	2.3	198	53.5	179	7.2	150.9
17/12/2015	29.4	611	7.2	3.4	212		262		195
18/12/2015	23.1	407	6.9	2.7	112		257		185.6
19/12/2015	27.5	636	6.6	3.3	220		314		201.9

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
20/12/2015	26.9	606	7	2.9	158		268		164.6
21/12/2015	29.1	541	7.4	3.3	65		294		221.2
22/12/2015	20.3	361	7	2.3	139		214		168.4
23/12/2015	23.2	643	7.1	2.7	98	48.7	242	6.5	150.6
24/12/2015	24.7	538	6.9	3.2	149		263		135.8
25/12/2015	20.3	389	7	2.4	224		264		131.76
26/12/2015	21.4	422	6.9	2.7	158		268		173.8
27/12/2015	18.9	325	7	2.4	149		196		151.8
28/12/2015	18.7	255	7.2	2	126		213		150.5
29/12/2015	18.9	481	6.2	2.3	273		230		146.7
30/12/2015	21.3	472	7.2	2.5	266		235		138.8
31/12/2015	24.6	488	6.9	3.3	241		204		134.32
01/01/2016	25.7	449	6.8	3.1	175		269		161.9
02/01/2016	17.4	445	7.3	2.1	187		258		149.6
03/01/2016	12.7	510	6.9	1.1	287		197		141
04/01/2016	22.5	528	7.3	3	174		255		139.2
05/01/2016	22	776	6.7	2.9	218		225		138.8
06/01/2016	23.3	461	7.3	2.4	159	54.5	208	5.4	139
07/01/2016	24.5	457	7	3.3	84		239		138.8
08/01/2016	22.5	634	6.1	3	218		256		175.1
09/01/2016	16.3	475	6	1.7	135		198		159.9
10/01/2016	10.6	471	6.9	1.9	101		206		169.5
11/01/2016	16	437	6.9	1.6	203		172		193.3
12/01/2016	18.3	576	6.9	2	200		200		230.8
13/01/2016	19.8	612	7	2.2	126	34.2	214	4.8	178.9
14/01/2016	18.6	925	6.9	2.5	205		267		168.8
15/01/2016	21.9	592	7.1	2.4	147		227		173.9
16/01/2016	19.8	760	7.2	2.4	224		275		165.9
17/01/2016	24.8	848	7	2.9	139		297		161.8
18/01/2016	22.3	995	7.3	2.3	204		273		151.1
19/01/2016	23	199	7.1	2.7	180		244		139.4
20/01/2016	18.7	881	6.9	2.2	133	53.5	256	5.6	172.7
21/01/2016	14	1215	6.8	1.6	207		256		182.2

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
22/01/2016	21	417	7.2	2.5	125		210		191.2
23/01/2016	17.8	432	7.1	2.3	118		218		222.8
24/01/2016	17.4	535	7.3	2.1	240		200		268.1
25/01/2016	13.9	397	6.8	1.4	107		181		206.1
26/01/2016	16.5	467	7	2.1	185		269		187.1
27/01/2016	18.8	672	7.1	1.7	230	25.1	217	4.6	192.2
28/01/2016	19.9	570	7	2.3	188		213		174.5
29/01/2016	19.6	629	7.3	2.3	266		299		164.9
30/01/2016	19.3	514	7.1	2.5	73		270		163.1
31/01/2016	20.2	507	7.3	2.9	144		275		165
01/02/2016	20.9	787	7.3	2.9	209		280		159.7
02/02/2016	21.5	586	7.1	3.5	126		296		129.4
03/02/2016	19	769	7.4	2.4	200	35.9	324	3.9	150
04/02/2016	19.8	845	7.2	2.5	187		317		146.5
05/02/2016	21.9	777		2.9	346		305		172.6
06/02/2016	21.9	594	6.4	2.9	72		304		183.9
07/02/2016	21.1	641	7.1	3	144		296		158.3
08/02/2016	19.9	577	7.4	2.4	87		300		171.6
09/02/2016	19.4	803	7.6	2.3	171		316		154.8
10/02/2016	19.4	588	7.4	2.6	76	47.8	315	5.2	156.3
11/02/2016	21.7	844	7.3	3.3	280		304		156.7
12/02/2016	21.1	725	7	3.7	268		286		158.8
13/02/2016	21.5	449	7.4	3.5	78		344		166.9
14/02/2016	20.3	571	6.9	2.7	156		271		234.4
15/02/2016	17.9	592	7.4	2.4	199		245		212.1
16/02/2016	19.4	483	7.4	2.5	153		200		166.4
17/02/2016	24.4	450	7.1	2.9	88	76.5	299	3.8	188.5
18/02/2016	22.2	607	6.9	4.3	193		238		227.1
19/02/2016	16.5	400	6.9	1.9	234		211		201.7
20/02/2016	15.9	388	7	1.6	151		220		180.5
21/02/2016	22.2	547	7.1	2.7	142		204		198.7
22/02/2016	23.9	476	6.9	2.9	368		258		181.3
23/02/2016	22.2	867	7	3.1	298		249		186.7

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
24/02/2016	22.6	916	7.2	3.4	200	53	252	6.8	214.9
25/02/2016	13	864	6.8	1.7	247		224		208.3
26/02/2016	18.1	444	7.2	2	199		235		191.9
27/02/2016	18.9	472	6.8	2.1	170		254		200.2
28/02/2016	21.1	745	6.9	2.5	193		260		174.1
29/02/2016	22.4	635	6.9	2.7	196		253		173.6
01/03/2016	27.4	686	6.4	3.3	219		263		171.5
02/03/2016	21.8	572	6.5	3	245	53.5	288	5.6	172.1
03/03/2016	20.8	637	6.2	2.7	157		275		167.8
04/03/2016	19.3	618	6.9	2	201		291		166.3
05/03/2016	23.4	394	7	2.5	95		294		172.9
06/03/2016	18.6	522	7.2	1.7	128		268		178.5
07/03/2016	26.1	564	7	3.5	177		277		163
08/03/2016	23.3	777	6.3	3.3	248		267		175.3
09/03/2016	27.1	757	7.2	3.6	248		268		233.6
10/03/2016		508	7.1		234		186		190.5
11/03/2016	15.1	523	6.6	0.9	166		191		368.4
12/03/2016	14.7	424	7	1.6	76		247		257.2
13/03/2016	18.7	544	7	2.1	56		267		255.1
14/03/2016	18.3	780	6.7	2.3	229		283		217.6
15/03/2016	19.7	845	7.2	3	242		224		220.4
16/03/2016	19.6	626	6.6	2.6	187	36.4	217	5.6	253.7
17/03/2016	13.8	562	6.9	1.5	237		199		311.5
18/03/2016		159	6.6	0.8	381		137		269.6
19/03/2016	10.1	292	6.5		205		165		260.4
20/03/2016	13.1	375	6.9	1.4	113		183		266.8
21/03/2016	16.4	326	7	2.3	207		222		253.9
22/03/2016	18.6	388	6.9	2.3	243		239		234.4
23/03/2016	17.3	404	7.1	1.9	242	61	232	4.4	234.4
24/03/2016	17.9	584	6.7	1.6	158		249		186.4
25/03/2016	17.2	343	7.3	1.9	88		258		223.9
26/03/2016	20.6	474	7	2.6	178		248		176.7
27/03/2016	16.6	700	7	2	233		240		191.4

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
28/03/2016	19	640	6.8	2.4	169		303		199.7
29/03/2016	22	751	7.3	2.4	215		340		180
30/03/2016	17.4	356	7.3	2.3	235	51.5	276	5.9	193
31/03/2016	22.9	895	7	2.9	213		248		173.3
01/04/2016	23.7	902	7.2	2.9	183		235		186.8
02/04/2016	19.6	516	6.9	2.3	139		260		171.9
03/04/2016	25.5	417	7.2	2.9	208		291		215.7
04/04/2016	25.2	1022	7	3.7	284		293		196.3
05/04/2016	24.1	673	7.2	3.2	174		264		183
06/04/2016	20.7	948	6.6	2.3	262	57.5	254	6.21	183.2
07/04/2016	17.5	709	7	1.7	301		181		163.1
08/04/2016	20.8	555	6.2	1.9	253		219		188.5
09/04/2016	20.7	631	5.9	2	302		203		170.6
10/04/2016	22.6	494	7.1	2.9	105		260		171.2
11/04/2016	24	650	7.3	2.8	244		268		159.5
12/04/2016	22.4	816	7	2.6	236		235		150.6
13/04/2016	22	991	7.4	2.7	312		267	7.8	163
14/04/2016	23.7	851	7.4	3.6	339		230		143.9
15/04/2016	17.9	898	7.4	1.5	446		258		173.1
16/04/2016	21	858	7.1	2.9	354		247		153.4
17/04/2016	23.2	912	6.7	4.3	321		308		159.4
18/04/2016	23.5	794	7.3	3	274		305		157.9
19/04/2016	20.6	775	6.9	2.7	250		267		153.4
20/04/2016	24.7	597	6.8	3.2	111	34.6	268	3.4	160.2
21/04/2016	24.2	889	6.9	3.5	291		221		174.9
22/04/2016	21.6	813	6.6	2.1	224		217		170.5
23/04/2016	22	543	6.8	2.7	91		256		184.6
24/04/2016	22.2	590	7	3	158		276		165.6
25/04/2016	23.4	492	7.2	3.4	111		249		164.4
26/04/2016	21.4	584	7	2.8	115		266		185.4
27/04/2016	18.8	430	6.8	2.3	127	39.8	249	1.5	233
28/04/2016	20.4	673	6.7	2.5	215		246		271.3
29/04/2016	20	784	6.9	2.2	274		194		275.2

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
30/04/2016	22.5	1058	6.9	3.6	204		234		276.2
01/05/2016	15.2	774	6.5	1.7	201		127		156.9
02/05/2016	25.3	442	6.3	2.5	90		223		164.6
03/05/2016	24.5	693	6.7	3.6	167		229		162.9
04/05/2016		677	6.1		122	30.2	236	4	163.2
05/05/2016	25.5	600	6.5	2.8	69		229		169.3
06/05/2016		900	6.8		294		213		164.9
07/05/2016	21	666	6.7	2.5	261		205		165.6
08/05/2016	22.2	671	6.4	3.2	173		235		163
09/05/2016	24.2	619	6.4	2.7	212		229		170.4
10/05/2016	23	576	6.6	2.3	57		218		165.9
11/05/2016	24.8	548	7.1	2.3	96	71.5	367	10.2	168.3
12/05/2016	22.3	772	6.8	2.9	383		227		168.9
13/05/2016	24.4	919	6.9	4.4	194		293		165.6
14/05/2016	20.1	1445	6.5	2.7	363		312		259.4
15/05/2016	13.4	371	6.9	1.6	136		304		267.8
16/05/2016	32.3	398	7.1	4	158		151		213.6
17/05/2016	17.6	595	6.7	1.7	232				181.6
18/05/2016	19.3	720	6.7	1.8	228		299	3.4	182.3
19/05/2016	19.5	539	6.8	1.9	179		286		180.7
20/05/2016	23.2	736	7.1	2.6	290		237		176.9
21/05/2016	25.4	922	6.9	3.2	390		230		180.9
22/05/2016	22.7	804	6.8	3.2	281		249		182.2
23/05/2016	25.5	586	6.9	2.6	183		283		184
24/05/2016	20.4	612	6.7	2.2	98		219		179.5
25/05/2016	27.1	591	6.9		60	50.5	275	6.5	184.1
26/05/2016	20.5	631	6.8	1.9	77		302		198.5
27/05/2016	20.2	626	6.6	1.9	127		266		196.6
28/05/2016	22.2	550	7.3	2.1	114		235		197
29/05/2016	22.9	561	6.9	1.5	155		270		203
30/05/2016	24	572	7.3	2.2	115		263		189.8
31/05/2016	24.7	617	7	2.4	52		241		191.9
01/06/2016	26.1	626	6.8	2.7	98		258		187.4

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
02/06/2016	20.8	600	7.1	2.3	128	41.6	254	4.4	185.9
03/06/2016	21.8	611	6.8	2.2	116		238		187.4
04/06/2016	25.2	743	6.9	2.7	180		252		185.9
05/06/2016	22.3	664	6.8	2.3	99		274		202.7
06/06/2016	25.1	538	6.9	2.6	97		274		206.9
07/06/2016	22.6	597	6.9	2.8	111		234		219.3
08/06/2016	22.9	592	7.1	2.6	116		282		190.9
09/06/2016	24.9	603	7	2.7	202		269		200.6
10/06/2016	23.1	548	6.9	2	123	44.4	252	3.9	193.1
11/06/2016	27	624	6.9	1.9	129		269		208.8
12/06/2016	24.7	512	6.9	2.4	66		261		212.9
13/06/2016	27.3	562	7.2	1.7	165		263		200.2
14/06/2016	19.4	456	6.9	1.8	94		252		184.7
15/06/2016	21.8	603	7	2.3	89		132		201.2
16/06/2016	26.5	472	6.8	2.8	69		283		180.3
17/06/2016	26.9	912	7.1	1.5	58	42.9	242	14.1	171.5
18/06/2016		504	7	1.7	80		255		188.7
19/06/2016		578	6.9		113		260		185.2
20/06/2016	21.6	601	6.9	2.6	110	77.5	295	17.4	183.1
21/06/2016	25.6	862	6.4	3	147		287		186.8
22/06/2016	27.4	578	6.9	2.3	141		304		186
23/06/2016	25.4	585	6.9	2.7	121		229		188.4
24/06/2016	18.8	614	7	2.5	135		243		183.9
25/06/2016	24.1	598	6.7	2.4	117		268		195.1
26/06/2016	23.9	557	6.8	2.2	91		268		185.2
27/06/2016	28.9	606	7.2	3.2	114		272		181.4
28/06/2016	23.2	665	7.1	2.1	139	30.75	260	5.1	182.6
29/06/2016	25.3	821	7.2	2.6	173		260		192.1
30/06/2016	24.8	494	7	2.7	127		267		184
01/07/2016	17.4	662	6.5	1.1	292		204		186.1
02/07/2016	25.8	733	6.8	2	187		250		207.6
03/07/2016	23.1	618	6.7	2.5	121		259		198
04/07/2016	26.1	375	7.3	2.4			242		201.6

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
05/07/2016	25.6	605	7	2.4	63		204		198.3
06/07/2016	30.6	577	7.1	3.7	117	78.5	246	7.4	218.6
07/07/2016	24	598	6.9	3	83		225		233
08/07/2016	27.6	775	6.7	3.2	177		204		271.3
09/07/2016	25.5	671	6.8	2.5	147		255		275.2
10/07/2016	25.6	592	7.2	3	149		233		276.2
11/07/2016	27.4	528	6.8	3.2	306		231		256.5
12/07/2016	23.3	553	7	2.9	70		211		281.1
13/07/2016	24	879	6.9	2.7	278		208		247.4
14/07/2016	19	622	7.2	1.9	81	32.5	178	3.5	228.6
15/07/2016	24.5	627	7	2.3	52		246		227.9
16/07/2016	22.4	528	6.6	2.2	97		225		224.2
17/07/2016	26.2	448	6.8	2.8	86		248		267
18/07/2016	23.8	1270	7	2.7	186		237		259.7
19/07/2016	23.4	789	7	2.8	146		202		264.4
20/07/2016	31	823	7.1	3.1	121	44.4	233	5.2	255.8
21/07/2016	26.3	599	7.2	2.7	50		213		342
22/07/2016	27.2		7.1	3.3	130		258		239.5
23/07/2016	24.8	604	7	3	96		208		193.6
24/07/2016	26.7	577	6.9	3.3	94		258		211.6
25/07/2016	26.5	810	6.8	3.1	40		244		217.9
26/07/2016	24.4	615	7	2.6	73		324		237.6
27/07/2016	18.7	855	7.3	2.2	15		246		205.9
28/07/2016	21.8	483	7.2	2.5	34	50.5	276	6.6	199.8
29/07/2016	24.3	911	7.1	1.9	133		240		201.7
30/07/2016	27	855	6.5	2.3	101		223		209.8
31/07/2016	25	661	6.4	3.5	142		203		211.3
01/08/2016	24.6	692	6.6	3.2	61		277		204.8
02/08/2016	23	645	7.1	3.1	62		228		187.1
03/08/2016	23.7	690	6.6	3.1	119		199		219.7
04/08/2016	25.7	1059	7	2.9	63		273		211.2
05/08/2016	24.7	654	7	3.2	58	40.9	256	4.1	208.1
06/08/2016	29.5	560	6.8	3.6	56		281		219.4

Sample Date	Ammonia (mg/l)	COD (mg/l)	pH	PO4 (mg/l)	Suspended Solids (mg/l)	TKN (mg/l)	Total Alkalinity (mg/L CaCO3)	TP (mg/l)	Flow (Ml/d)
07/08/2016	27.2	582	7.1	2.7	117		267		212.9
08/08/2016	28.4	528	7.2	4.1	98		283		196.7
09/08/2016	26.6	700	6.9	3.9	230		217		211.8
10/08/2016		694	7.1	5.3	127		218		209
11/08/2016	19.5		7.3	3.3	205	41.8	370	8.4	204.9
12/08/2016	28.7	686	7.1	3.6	98		317		214
13/08/2016	26.5	602	6.4	3.5	117		264		214
14/08/2016	29.1	589	6.9	4.1	111		334		214.1
15/08/2016	24.6	533	7.1	4.5	73		289		203
16/08/2016	24.9	619	6.9	3.4	110		269		199.8
17/08/2016	24.9	829	6.9	3.3	107	53.5	270	6.6	213.5
18/08/2016	21.8	819	6.7	3.8	129		237		218.3
19/08/2016	28.5	742	6.9	2.9	111		311		204.2
20/08/2016	24.7	644	7	3.9	129		381		205.9
21/08/2016	24.5	597	6.9	4	116		319		210.8
22/08/2016	14.1	646	6.9	1	136		304		204.3
23/08/2016	23.7	650	7	3	105		309		205.6
24/08/2016	26.3	672	6.9	3.2	84		306		209
25/08/2016	27.8	648	6.8	3	141	47.5	274	4.9	204.6
26/08/2016	24.5	1009	6.8	3.2	102		276		204.1
27/08/2016	28.7	702	6.2	3.5	70		197		205.6
28/08/2016	19	1348	6.7	2.3	233		156		208.1
29/08/2016	23.5	610	7	3.8	78		309		194.9
30/08/2016	27.5	687	7	4.3	86		308		185.9
31/08/2016	26.4	584	6.9	3	76		256		

Appendix B: Definition of steady state model terms

Table B-1: Steady state model terms

Term	Definition	Unit
$[HCO_3^-]$	Bicarbonate concentration	mol/l
ΔP_{OHO}	P removal due to OHOs	gP/m^3
ΔP_{PAO}	P removal due to PAOs	gP/m^3
$\Delta P_{SYS,\ ACT}$	Actual total P removal for the system	gP/m^3
$\Delta P_{SYS,\ POT}$	Potential total P removal by the system	gP/m^3
ΔP_{XE}	P Removal due to endogenous residue mass	gP/m^3
ΔP_{XI}	P Removal due to influent inert mass	gP/m^3
μ_{AmT}	Maximum specific growth rate of nitrifiers at T°C	d-1
b_{AD}	Acidogen endogenous respiration rate	d-1
b_{AT}	Specific endogenous respiration rate for nitrifiers at 20°C	d-1
b_H	Specific rate of endogenous mass loss of OHOs	d-1
$b_{OHO,T}$	OHO specific endogenous mass loss rate constant at temperature T°C	d-1
$b_{PAO,T}$	PAO specific endogenous mass loss rate at temperature T°C	gEVSS/gVSS.d
D_B	4k+l-2m-3n	e^-eq/mol biomass
$D_{p1RBCOD}$	Denitrification potential of the influent RBCOD in the primary anoxic reactor	mgN/l
D_S	4x+y-2z-3a	e^-eq/mol biodegradable organics
E	$\frac{V(Z_{AD}+Z_E)}{R_s Q_i(S_{bpi}-S_{bp})}=\frac{Y_{AD}(1+f_{AD}b_{AD}R_s)}{1+b_{AD}R_s[1-Y_{AD}(1-f_{AD})]}$	Unitless
f_{AD}	Acidogen biomass unbiodegradable fraction	Unitless
f_{AN}	Anaerobic mass fraction (gVSS/gVSS)	Unitless
f_{cv}	COD to VSS ratio of the sludge	Unitless
$f_{FSS,OHO}$	Fraction of FSS in the OHO active biomass	gFSS/gVSS
$f_{FSS,PAO}$	Fraction of FSS in the PAO active biomass	gFSS/gVSS
f_H	Unbiodegradable fraction of the OHOs	mgCOD/mgCOD
f_{iOHO}	Inorganic content of OHOs	mgISS/mgCOD
f_n	Nitrogen content of VSS	mgN/mgVSS
$FO_{2,\ OHO}$	Daily mass of oxygen consumed by OHOs	gO_2/d
$FO_{2,\ PAO}$	Daily mass of oxygen consumed by PAOs	gO_2/d
FO_C	Daily carbonaceous oxygen demand	gO_2/d

Term	Definition	Unit
FO_c	Daily flux of oxygen utilised	mgO_2/d
FO_n	Flux of oxygen required for nitrification	kgO_2/d
FO_t	Total flux of O_2 required including the oxygen recovered by denitrification	kgO_2/d
$f_{P,FSS}$	P fraction of fixed (inorganic) suspended solids mass	gP/gFSS
$f_{P,OHO}$	fraction of OHO active mass that is P	gP/gAVSS
$f_{P,PAO}$	Fraction of PAO active mass that is P	gP/gAVSS
$f_{P,TSS}$	P fraction of total suspended solids mass	gP/gTSS
$f_{P,XE}$	fraction of endogenous residue mass that is P	gP/gEVSS
$f_{P,XE}$	fraction of inert mass that is P	gP/gEVSS
$f_{PS'up}$	Unbiodegradable particulate fraction of the primary sludge	Unitless
$f_{S'bs}$	RBCOD fraction with respect to influent biodegradable COD	Unitless
$f_{s'up}$	Particulate unbiodegradable fraction of total influent COD	Unitless
$f_{s'us}$	Soluble unbiodegradable fraction of total influent COD	Unitless
$FS_{b,OHO}$	Daily mass of biodegradable substrate available to OHOs	gCOD/d
FSbi	Daily flux of influent biodegradable COD	mgCOD/d
$FS_{bs,PAO}$	Daily mass of readily biodegradable substrate stored by PAOs in the anaerobic reactor	gCOD/d
FS_{ti}	Daily mass of influent COD	gCOD/d
f_{x1}	Primary anoxic sludge mass fraction	Unitless
f_{x3}	Secondary anoxic sludge mass fraction	Unitless
$f_{XE,OHO}$	Fraction of endogenous particulate residue of OHO	gEVSS/gAVSS
$f_{XE,PAO}$	Fraction of endogenous particulate residue of PAOs	gEVSS/gAVSS
$FX_{FSS,i}$	Daily mass of influent FSS	gFSS/d
$f_{XI,COD,i}$	Fraction of influent COD that is particulate inert	gCOD/gCOD
FX_{Ii}	Daily flux of influent particulate unbiodegradable COD	mgCOD/d
FX_{IOi}	Daily flux of influent particulate inorganic matter	mgISS/d
FX_t	Daily flux of total solids produced	mgTSS/d
K_2	Second specific rate of denitrification in primary anoxic reactor	$mgNO_3$-N/mgOHOVSS.d

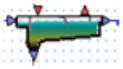
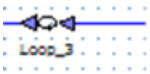
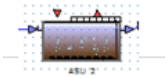
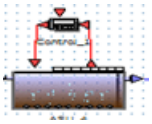
Term	Definition	Unit
K_3	Specific rate of denitrification in secondary anoxic reactor	$mgNO_3$-N/mgOHOVSS.d
K_m	Kinetic constant	gCOD organics/gCOD biomass.d
K_{nT}	Half saturation constant for nitrifiers at T°C	mgN/l
K_s	Kinetic constant	gCOD/l
MX_{BH}	Mass of OHO in the bioreactor	mgVSS
$MX_{E,\ PAO}$	Mass of PAO endogenous residue in the system	gEVSS
$MX_{E,OHO}$	Mass of OHO endogenous residue in the system	gEVSS
MX_{EH}	Mass of endogenous residue in the bioreactor	mgVSS
MX_{FSS}	mass of fixed suspended solids in the system	gFSS
MX_I	mass of inert organic matter in the system coming from the influent	gIVSS
MX_{IO}	Mass of influent particulate inorganic matter in the bioreactor	mgISS
MX_{OHO}	OHO active biomass	gAVSS
MX_{PAO}	Mass of PAO in the system	gAVSS
MXt	Mass of TSS in the reactor	gTSS
MX_V	Mass of volatile suspended solids in the bioreactor	mgVSS
MX_{VSS}	VSS mass in the system	gVSS
N	Total number of anaerobic reactors of equal volume in the series n = 1, 2…N	Unitless
N_a	Bulk liquid ammonia concentration	mgN/l
N_{ae}	Effluent ammonia concentration	mgN/l
N_{ai}	Influent ammonia concentration	mgN/l
N_c	Nitrification capacity	mgN/l
N_{ne}	Effluent nitrate concentration	mgN/l
N_{obpi}	Influent biodegradable particulate organic nitrogen	mgN/ l
N_{obsi}	Influent biodegradable soluble organic nitrogen	mgN/l
N_{oupi}	Influent unbiodegradable particulate organic nitrogen	mgN/l
N_{ouse}	Effluent unbiodegradable soluble organic nitrogen	mgN/l
N_s	Concentration of TKN in influent required for sludge production	mgN/l
N_{te}	Effluent TKN concentration	mgN/l
p_{CO2}	Partial pressure of CO_2 in the gas phase	mol fraction
pH	Negative log10 of the (H^+) activity	Unitless

Term	Definition	Unit
pH_i	Influent pH	Unitless
pK'a	Negative $\log_{10}$ of acid dissociation constant	Unitless
pK'_{C1} & pK'_{C2}	Negative $\log_{10}$ of 1st and 2nd carbonate system apparent dissociation constant (where "apparent" means corrected for ionic strength effects)	Unitless
pK'_{HCO2}	Negative $\log_{10}$ of the apparent Henry's law constant	Unitless
Q_i	Influent flow rate	m^3/d
Q_m	Methane gas flow per litre influent	l methane/d/l influent flow/d
R	Retention time	d
r_h	Monod hydrolysis rate	gCOD/l.d
R_s	Sludge retention time	d
s	Sludge recycle ratio to anaerobic reactor based on influent flow	Unitless
Sbi	Influent biodegradable COD	mgCOD/l
S_{bi}	Influent biodegradable COD	mgCOD/l
S_{bp}	Residual biodegradable COD concentration	gCOD/l
S_{bpi}	Hydrolysable COD concentration from the influent	gCOD/l
S_{bsAci}	Dissociated acetate species concentration in the influent	gCOD/l
S_{bsai}	Concentration of VFA in the influent	$gCOD/m^3$
S_{bsai}	Volatile fatty acid concentration in the influent	gCOD/)
$S_{bsf,ANn}$	Concentration of fermentable COD in the effluent of the n^{th} anaerobic reactor	$gCOD/m^3$
$S_{bsf,i}$	Fermentable COD influent concentration	$gCOD/m^3$
$S_{bsf,i,conv}$	Fermentable COD available for conversion per volume of influent	$gCOD/m^3$
S_{bsHAci}	Undissociated acetate species concentration in the influent	gCOD/l
S_m	Effluent methane concentration	gCOD/l
$S_{NO3,i}$	Nitrate concentration in the influent to anaerobic reactor	$gNO_3\text{-}N/m^3$
$S_{NO3,s}$	Nitrate concentration in the sludge recycle to anaerobic reactor	gNO_3^-N/m^3
$S_{O2,i}$	Oxygen concentration in the influent to anaerobic reactor	gO_2/m^3
$S_{O2,s}$	Oxygen concentration in the sludge recycle to the anaerobic reactor	gO_2/m^3
S_{te}	Effluent COD	mgCOD/l

Term	Definition	Unit
Sti	Influent total COD	mgCOD/l
S_{upi}	Unbiodegradable COD concentration in the influent	gCOD/l
$T_{P,e}$	Effluent total P concentration	gP/m^3
$T_{p,i}$	Influent total P concentration	gP/m^3
TSS_e	Total suspended solids concentration of the effluent	$gTSS/m^3$
V	Volume of the AD	m^3
X_{Ii}	Influent unbiodegradable matter in activated sludge	mgVSS/l
X_{IOi}	Influent inorganics concentration	mgISS/l
X_t	TSS concentration is reactor	mgTSS/l
X_{ve}	Effluent VSS concentration	mgVSS/l
Y_{AD}	Pseudo-acidogen yield coefficient	gCOD biomass/gCOD organics hydrolysed
Y_H	Yield of OHOs	0.45mgVSS/mgCOD
Y_{OHO}	OHO yield	gAVSS/gCOD
Y_{PAO}	PAO biomass yield	gAVSS/gCOD
Z_{AD}	Concentration of acidogen biomass	gCOD/l
Z_E	Acidogen endogenous residue concentration	gCOD/l

Appendix C : WEST® icon descriptions

Table C-1: WEST® icon descriptions

Unit operation	Class name	Comment	Symbol
SSTs, PSTs & thickeners	Primary/Secondary point settlers	The settling tank has no volume (i.e., no retention time) and acts as an ideal phase separator.	
Loop breaker	Differential loop breaker	This component is used to 'break' algebraic loop (e.g., return flows) for WWTP type terminals.	
Anaerobic digester	UCTAD_BSM3P	This icon represents the AD model as described in Section 4.3.2.1.	
Struvite precipitation unit	UCTAD_BSM3P	Based on AD model. A variable "K_ONOFF" was added to the kinetics to allow switching off all processes except struvite precipitation.	
Universal dosing unit	-	This unit allows dosing of multiple chemicals (e.g., Mg, Ca, VFA, etc.)	
Activated sludge unit	Fixed volume ASU	This icon is used for anaerobic and anoxic zones.	
		This icon is used for aerobic zones.	

Appendix D : WEST® model parameters

Table D-1: Kinetic and stoichiometric parameters for AS system

Parameter		Description	Value	Unit
Kinetic Parameters	A_20	Activation/inhibition term at 20 degrees Celsius	1	1/d
	A_35	Activation/inhibition term at 35 degrees Celsius	1	1/d
	A_55	Activation/inhibition term at 55 degrees Celsius	1	1/d
	KA_CO2	CO2 liquid to gas phase equilibrium constant	1.64346E-05	-
	KA_H_N	Inhibition of bacteria growth due to high pH	0.0000001	-
	KI_H_N	Inhibition of bacteria growth due to low pH	0.00085	-
	KLa_CO2	Rate of Active CO2 gas exchange rate with aeration	2	1/d
	KS_fPP_PAO_PHAstor	Saturation coeff for poly-phosphate	0.01	-
	kM_fPP_PAO_PHAstor	maximum rate for PP release with anaerobic PHA storage	0.05	
	K_F_OHO	Saturation/inhibition coeff for growth on S_F	4	-
	K_I_PP_PAO	Inhibition coeff for X_PP storage	0.2	-
	K_MAX_fPP_PAO	Maximum ratio of X_PP/X_PAO	1	-
	K_NOx_OHO	Nitrate Half-Saturation Coefficient For Denitrifying Heterotrophic Biomass	0.5	gNO3-N/m3
	K_O2	Saturation/inhibition coeff for oxygen	0.2	-
	K_O2_ANO	Saturation/inhibition coeff of autotrophs for oxygen	0.2	-
	K_S_ALK	Saturation coeff for alkalinity (HCO3-)	0.1	-
	K_S_ALK_ANO	Saturation coeff of autotrophs for alkalinity	0.1	-
	K_S_BInf_OHO_hyd	Half Saturation Coefficient For Hydrolysis Of Slowly Biodegradable Substrate	0.1	gCOD/gCOD
	K_S_F_OHO_ferm	Saturation coeff for fermentation on S_F	20	-
	K_S_NHx	Ammonia Half-Saturation Coefficient For Organisms Growth	0.05	gNH3-N/m3
	K_S_NHx_ANO	Saturation coeff of autotrophs for ammonium	0.3	-

Parameter		Description	Value	Unit
	K_S_PHA_PAO	Saturation coeff for PHA	0.01	-
	K_S_PO4	Saturation coeff for phosphorus (nutrient)	0.01	-
	K_S_PO4_PAO_PPstor	Saturation coeff for phosphorus in PP storage	0.2	-
	K_S_VFA	Saturation coeff for S_A (acetate)	4	-
	Q_OHO_F_VFA_ferm	Maximum rate for fermentation	20	1/d
	Q_PAO_PO4_PPstor	Rate constant for storage of PP	4.5	1/d
	Q_PAO_PP_PHAstor	Rate constant for storage of PHA (base: X_PP)	1.5	1/d
	S_O_Sat	Oxygen saturation concentration	8.9	g/m3
	Th_d_XG_20	Temperature correction factor for decay of PAOs at 20°C	1.12	1/d
	Th_d_XG_35	Temperature correction factor for decay of PAOs at 35°C	1.12	1/d
	Th_d_XH_20	Temperature correction factor for decay of OHOs at 20°C	1.12	1/d
	Th_d_XH_35	Temperature correction factor for decay of OHOs at 35°C	1.12	1/d
	Th_d_XN_20	Temperature correction factor for decay of ANOs at 20°C	1.072	1/d
	Th_d_XN_35	Temperature correction factor for decay of ANOs at 35°C	1.072	1/d
	Th_h_20	Temperature correction factor for hydrolysis at 20°C	1.116	1/d
	Th_h_35	Temperature correction factor for hydrolysis at 35°C	0.05	1/d
	Th_m_XH_20	Temperature correction factor for growth of OHOs at 20°C	1.072	1/d
	Th_m_XN_20	Temperature correction factor for growth of ANOs at 20°C	1.103	1/d
	b_ANO	Decay rate	0.15	1/d
	b_OHO	Decay Coefficient For Heterotrophic Biomass	0.62	1/d
	b_PAO	Rate constant for lysis of X_PAO	0.04	1/d
	b_PHA	Rate constant for lysis of X_PHA	0.04	1/d
	b_PP	Rate constant for lysis of X_PP	0.04	1/d
	k_M_BInf_OHO_hyd	Maximum Specific Hydrolysis Rate	3	gCOD/(gCOD*d)
	mu_ANO	Maximum specific growth rate for ANOs	1	1/d
	mu_OHO_max	Maximum Specific Growth Rate For Heterotrophic Biomass	6	1/d

Parameter		Description	Value	Unit
	mu_PAO	Maximum growth rate for PAOs	1	1/d
	n_NO_Het	Reduction factor for denitrification	0.3	-
	n_OHO_BInf_ferm	Anaerobic hydrolysis reduction factor	0.1	-
	n_OHO_BInf_hyd	Anoxic hydrolysis reduction factor	0.6	-
	TempCoeff	Rate temperature coefficient	0.0667	-
	Temperature	System Temperature	20	degC
	Tref	Reference temperature for kinetics	20	degC
	kdis_cal	Dissolution/ precipitation of calcite	0.5	
	kdis_cap	Dissolution/ precipitation of calcium phosphate	150	
	kdis_mag	Dissolution/ precipitation of magnesite	50	
	kdis_mgkp	Dissolution/ precipitation of K-struvite	100	
	kdis_newb	Dissolution/ precipitation of newberyite	0.05	
	kdis_stru	Dissolution/ precipitation of struvite	0.0001	
Stoichiometric Parmeters	f_XU_Bio_lysis	Unbiodegradable fraction of biomass that accumulates on lysis with death regeneration model	0.08	dUnit/dUnit
	ISS_BM	ISS to biomass for OHO and PAO	0.15	g/gCOD
	f_SU_SF	Inert Fraction in Fermentable Soluble Organics	0	-
	i_Ca_PP_mol_perP	Molar fraction of Ca/P in polyphosphate	0.028	dUnit/dUnit
	i_K_PP_mol_perP	Molar fraction of K/P in polyphosphate	0.330	dUnit/dUnit
	i_Mg_PP_mol_perP	Molar fraction of Mg/P in polyphosphate	0.307	dUnit/dUnit
	i_H_Org_mol_perC	H/C : organisms	1.535	dUnit/dUnit
	i_H_SF_mol_perC	H/C : fermentable soluble	1.952	dUnit/dUnit
	i_H_SU_mol_perC	H/C: unbiodegradable soluble	1.747	dUnit/dUnit
	i_H_XBInf_mol_perC	H/C: PS biodegradable particulate	2.307	dUnit/dUnit
	i_H_XBOrg_mol_perC	H/C: biodegradable particulate	1.535	dUnit/dUnit
	i_H_XUInf_mol_perC	H/C: unbiodegradable particulate	1.535	dUnit/dUnit
	i_H_XUOrg_mol_perC	H/C: endogenous residue	1.535	dUnit/dUnit
	i_N_Org_mol_perC	N/C : organisms	0.167	dUnit/dUnit
	i_N_SF_mol_perC	N/C: fermentable soluble	0.110	dUnit/dUnit
	i_N_SU_mol_perC	N/C: unbiodegradable soluble	0.152	dUnit/dUnit
	i_N_XBInf_mol_perC	N/C: PS biodegradable particulate	0.040	dUnit/dUnit
	i_N_XBOrg_mol_perC	N/C: biodegradable particulate	0.167	dUnit/dUnit

Parameter		Description	Value	Unit
	i_N_XUInf_mol_perC	N/C: unbiodegradable particulate	0.167	dUnit/dUnit
	i_N_XUOrg_mol_perC	N/C: endogenous residue	0.167	dUnit/dUnit
	i_O_Org_mol_perC	O/C : organisms	0.420	dUnit/dUnit
	i_O_SF_mol_perC	O/C : fermentable soluble	0.476	dUnit/dUnit
	i_O_SU_mol_perC	O/C: unbiodegradable soluble	0.508	dUnit/dUnit
	i_O_XBInf_mol_perC	O/C: PS biodegradable particulate	0.601	dUnit/dUnit
	i_O_XBOrg_mol_perC	O/C: biodegradable particulate	0.420	dUnit/dUnit
	i_O_XUInf_mol_perC	O/C: unbiodegradable particulate	0.420	dUnit/dUnit
	i_O_XUOrg_mol_perC	O/C: endogenous residue	0.420	dUnit/dUnit
	i_P_Org_mol_perC	P/C : organisms	0.022	dUnit/dUnit
	i_P_SF_mol_perC	P/C: fermentable soluble	0.005	dUnit/dUnit
	i_P_SU_mol_perC	P/C: unbiodegradable soluble	0.020	dUnit/dUnit
	i_P_XBInf_mol_perC	P/C: PS biodegradable particulate	0.007	dUnit/dUnit
	i_P_XBOrg_mol_perC	P/C: biodegradable particulate	0.022	dUnit/dUnit
	i_P_XUInf_mol_perC	P/C: unbiodegradable particulate	0.022	dUnit/dUnit
	i_P_XUOrg_mol_perC	P/C: endogenous residue	0.022	dUnit/dUnit
	Y_ANO	Yield for autotrophic nitrifying biomass (ANO)	0.24	gCOD/gN
	Y_OHO	Yield for heterotrophic (OHO)biomass	0.67	gCOD/gCOD
	Y_PAO	Yield coeff for PAO biomass/PHA	0.67	-
	Y_PP_Stor_PAO	PP requirement (S_PO4 release) per PHA stored	0.4	-
	Y_Stor_PP_PAO	PHA requirement for PP storage	0.2	-
	Y_f_PP_VFA	fraction of PP taken up with VFA uptake for PHA	0.5	-

Table D-2: AS reactor parameters

AS system reactor parameters				
	Anaerobic	**Anoxic**	**Aerobic**	**Unit**
Category: Manipulated Variables				
Group: Operational				
Kla	0	0	19.846753	1/d
Temp	20	20	20	degC
Category: Parameters				
Group:				
f_XU_Bio_lysis	0.08	0.08	0.08	dUnit/dUnit
Group: Aeration				
OTR_Energy	1800	1800	1800	g/kWh
Group: Composition parameters				
ISS_BM	0.15	0.15	0.15	g/gCOD
Group: Conversion factors				
F_TSS_COD	0.75	0.75	0.75	-
Group: Dimension				
Vol	29979	29979	29979	L
Group: Effluent Quality Index				
Limit_COD	50	50	50	
Limit_FSA	1	1	1	
Limit_NO	6	6	6	
Limit_OP	1	1	1	
Limit_TSS	10	10	10	
Group: Kinetic				
A_20	1	1	1	1/d
A_35	1	1	1	1/d

AS system reactor parameters				
	Anaerobic	**Anoxic**	**Aerobic**	**Unit**
A_55	1	1	1	1/d
b_ANO	0.15	0.15	0.15	1/d
b_OHO	0.62	0.62	0.62	1/d
b_PAO	0.04	0.04	0.04	1/d
b_PHA	0.04	0.04	0.04	1/d
b_PP	0.04	0.04	0.04	1/d
f_gly_PAO_max	0.32	0.32	0.32	
K_A	4	4	4	g/m3
K_A_PO4	5	5	5	
K_ALK	0.1	0.1	0.1	g/m3
K_destruct	5	5	5	
K_F	4	4	4	g/m3
K_IPP	0.02	0.02	0.02	g/m3
k_M_BInf_OHO_hyd	3	3	3	gCOD/(gCOD*d)
k_M_BOrg_OHO_hyd	3	3	3	
K_MAX	0.34	0.34	0.34	g/m3
K_NH	0.05	0.05	0.05	gNH3-N/m3
K_NO	0.5	0.5	0.5	gNO3-N/m3
K_O2	0.2	0.2	0.2	g/m3
K_O2_ANO	0.2	0.2	0.2	g/m3
K_O2_PP	0.1	0.1	0.1	
K_P	0.01	0.01	0.01	g/m3
K_PHA	0.01	0.01	0.01	g/m3
K_PP	0.01	0.01	0.01	
K_PS	0.2	0.2	0.2	gCOD/gCOD
K_S_ALK_ANO	0.1	0.1	0.1	g/m3
K_S_F_OHO_ferm	20	20	20	g/m3
K_S_Glyco	0.001	0.001	0.001	
K_S_NH_ANO	0.3	0.3	0.3	g/m3

AS system reactor parameters				
	Anaerobic	**Anoxic**	**Aerobic**	**Unit**
K_X	0.1	0.1	0.1	
KA_CO2	1.64E-05	1.64346E-05	1.64346E-05	g/m3
KA_H_N	1E-07	0.0000001	0.0000001	g/m3
KI_H_N	0.00085	0.00085	0.00085	g/m3
KLa_CO2	0	0	0	1/d
KS_fPP_PAO_PHAstor	0.0001	0.0001	0.0001	g/m3
mu_ANO	1	1	1	1/d
mu_OHO_max	6	6	6	1/d
mu_PAO	1	1	1	1/d
n_NO_Het	0.3	0.3	0.3	-
n_OHO_BInf_ferm	0.1	0.1	0.1	-
n_OHO_BInf_hyd	0.6	0.6	0.6	-
n_OHO_BOrg_ferm	0.1	0.1	0.1	
n_OHO_BOrg_hyd	0.6	0.6	0.6	
PCO2_atm	0.000347	0.0003467	0.0003467	
Q_Glycogen	0.2	0.2	0.2	
Q_OHO_F_VFA_ferm	20	20	20	1/d
Q_PAO_PO4_PPstor	4.5	4.5	4.5	1/d
Q_PAO_PP_PHAstor	1.5	1.5	1.5	1/d
S_O_Sat	8.9	8.9	8.9	g/m3
Th_d_XG_20	1.12	1.12	1.12	1/d
Th_d_XG_35	1.12	1.12	1.12	1/d
Th_d_XH_20	1.12	1.12	1.12	1/d
Th_d_XH_35	1.12	1.12	1.12	1/d
Th_d_XN_20	1.072	1.072	1.072	1/d
Th_d_XN_35	1.072	1.072	1.072	1/d
Th_h_20	1.116	1.116	1.116	1/d
Th_h_35	0.05	0.05	0.05	1/d
Th_m_XH_20	1.072	1.072	1.072	1/d
Th_m_XN_20	1.103	1.103	1.103	1/d

AS system reactor parameters				
	Anaerobic	**Anoxic**	**Aerobic**	**Unit**
Group: kinetics				
deltaH_cal	-2297	-2297	-2297	
deltaH_cap	54000	54000	54000	
deltaH_mag	-20000	-20000	-20000	
deltaH_mgkp	20000	20000	20000	
deltaH_newb	15000	15000	15000	
deltaH_stru	22600	22600	22600	
K_I_O2_pha	0.01	0.01	0.01	
kdis_cal	0	0	0	
kdis_cap	0	0	0	
kdis_mag	0	0	0	
kdis_mgkp	0	0	0	
kdis_newb	0	0	0	
kdis_stru	0	0	0	
logK_cal	-8.48	-8.48	-8.48	
logK_cap	-28.92	-28.92	-28.92	
logK_mag	-7.46	-7.46	-7.46	
logK_mgkp	-10.62	-10.62	-10.62	
logK_newb	-5.8	-5.8	-5.8	
logK_stru	-13.26	-13.26	-13.26	
TempCoeff	0.0667	0.0667	0.0667	-
Temperature	20	20	20	degC
Tref	20	20	20	degC
Group: Mixing energy				
Kla_Min	20	20	20	1/d
ME_unit	0.005	0.005	0.005	kWh/m3/d
Mixing_When_Aerated	0	0	0	kWh/d

Table D-3: Kinetic and stoichiometric parameters for AD

Parameter		Description	Value	Unit
Kinetic parameters	K_I_H_AD	H+ inhibition for acidogens	0.0155	
	K_I_NH3	NH_3 Inhibition for AD organisms	0.0018	
	KS_fPP_PAO_PHAstor	Saturation coeff for poly-phosphate	0.01	-
	K_O2	Saturation/inhibition coeff for oxygen	20	-
	K_S_ALK	Saturation coeff for alkalinity (HCO3-)	0.0001	-
	K_S_NHx	Saturation coeff for Ammonia (nutrient)	0.0001	gNH3-N/m3
	K_S_PO4	Saturation coeff for phosphorus (nutrient)	0.0001	-
	K_S_VFA	Saturation coeff for Acetate	10	-
	KS_AC	Half Sat coeff for acetogens	0.089	g/m3
	KS_AD	Half Sat coeff for acidogens	0.78	g/m3
	KS_AM	Half Sat coeff for acetoclastic methanogens	0.013	g/m3
	KS_OB	Half Sat coeff for acetate oxidizing bacteria	0.29	g/m3
	KS_BInf_AD_hyd	Half sat coeff for BPO_PS (for sewage organics)	10.12	gCOD/gCOD
	KS_BOrg_AD_hyd	Half sat coeff for BPO (for organics from biomass death)	10.37	gCOD/gCOD
	KS_HM	Half Sat coeff for hydrogenotrophic methanogens	0.156	g/m3
	K_CO2	Rate constant for CO2 exchange in AD	0.1	
	K_CO2_eq	equilibrium constant for CO2 liquid - gas phase exchange	1.21E-08	
	K_I_H2	Inhibition coefficient for H2 in acidogenesis	9.999375	g/m3
	K_I_H_AM	H+ inhibition for acetoclastic methanogens	1.15E-06	Mol.kg-1
	K_I_H_OB	H+ inhibition for acetate oxidizing bacteria	0.00053	Mol.kg-1
	K_I_H_HM	H+ inhibition for hydrogenotrophic methanogens	0.00053	Mol.kg-1
	TempCoeff	Rate temperature coefficient	0.0667	-
	Temperature	System Temperature	35	degC
	Tref	Reference temperature for kinetics	35	degC
	b_AC	Decay rate constant for Xac	0.015	1/d
	b_AD	Decay rate constant for Xad	0.041	1/d
	b_AM	Decay rate constant for Xam	0.037	1/d
	b_OB	Decay rate constant for Xob	0.041	1/d
	b_HM	Decay rate constant for Xhm	0.01	1/d
	b_OHO_AD	Decay rate constant for X_OHO in AD	20	1/d

Parameter		Description	Value	Unit
	b_PAO_AD	Decay rate constant for X_PAO in AD	20	1/d
	kH_F_AD_hyd	Hydrolysis rate constant for FSO	10	1/d
	kH_PHA_AD_hyd	Hydrolysis rate constant for PHA	5	1/d
	kH_PP_AD_hyd	Hydrolysis rate constant for PP	1	1/d
	kM_BInf_AD_hyd	Hydrolysis rate constant for BPO_PS	2.00	1/d
	kM_BOrg_AD_hyd	Hydrolysis rate constant for BPO	1.95	1/d
	kM_fPP_PAO_PHAstor	maximum rate for PP release with anaerobic PHA storage	0.3	1/d
	kdis_cal	Dissolution/ precipitation/ precipitation of calcite	0.5	
	kdis_cap	Dissolution/ precipitation of calcium phosphate	150	
	kdis_mag	Dissolution/ precipitation of magnesite	50	
	kdis_mgkp	Dissolution/ precipitation of K-struvite	100	
	kdis_newb	Dissolution/ precipitation of newberyite	0.05	
	kdis_stru	Dissolution/ precipitation of struvite	300	
	mu_AC	Max specific growth rate for acetogens	1.15	1/d
Stoichiometric parameters	mu_AD	Max specific growth rate for acidogens	0.8	1/d
	mu_AM	Max specific growth rate for acetoclastic methanogens	4.39	1/d
	mu_OB	Max specific growth rate for acetate oxidizing bacteria	3.25	
	mu_HM	Max specific growth rate for hydrogenotrophic methanogens	1.2	1/d
	f_SU_SF	Inert Fraction in Fermentable Souble Organics	0	-
	f_XU_Bio_lysis	Unbiodegradable fraction of biomass that accumulates on lysis with death regeneration model	0.08	-
	ISS_BM	ISS to biomass for OHO and PAO	0.15	g/gCOD
	i_Ca_PP_mol_perP	Molar fraction of Ca/P in polyphosphate	0.0279	
	i_K_PP_mol_perP	Molar fraction of K/P in polyphosphate	0.3302	
	i_Mg_PP_mol_perP	Molar fraction of Mg/P in polyphosphate	0.3070	
	i_H_Org_mol_perC	H/C : organisms	1.5352	dUnit/dUnit
	i_H_SF_mol_perC	H/C : fermentable soluble	1.9518	dUnit/dUnit
	i_H_SU_mol_perC	H/C: unbiodegradable soluble	1.7467	dUnit/dUnit
	i_H_XBInf_mol_perC	H/C: PS biodegradable particulate	2.3066	dUnit/dUnit
	i_H_XBOrg_mol_perC	H/C: biodegradable particulate	1.5352	dUnit/dUnit
	i_H_XUInf_mol_perC	H/C: unbiodegradable particulate	1.5352	dUnit/dUnit

Parameter		Description	Value	Unit
	i_H_XUOrg_mol_perC	H/C: endogenous residue	1.5352	dUnit/dUnit
	i_N_Org_mol_perC	N/C : organisms	0.1670	dUnit/dUnit
	i_N_SF_mol_perC	N/C: fermentable soluble	0.1100	dUnit/dUnit
	i_N_SU_mol_perC	N/C: unbiodegradable soluble	0.1524	dUnit/dUnit
	i_N_XBInf_mol_perC	N/C: PS biodegradable particulate	0.0396	dUnit/dUnit
	i_N_XBOrg_mol_perC	N/C: biodegradable particulate	0.1670	dUnit/dUnit
	i_N_XUInf_mol_perC	N/C: unbiodegradable particulate	0.1670	dUnit/dUnit
	i_N_XUOrg_mol_perC	N/C: endogenous residue	0.1670	dUnit/dUnit
	i_O_Org_mol_perC	O/C : organisms	0.4202	dUnit/dUnit
	i_O_SF_mol_perC	O/C : fermentable soluble	0.4756	dUnit/dUnit
	i_O_SU_mol_perC	O/C: unbiodegradable soluble	0.5080	dUnit/dUnit
	i_O_XBInf_mol_perC	O/C: PS biodegradable particulate	0.6006	dUnit/dUnit
	i_O_XBOrg_mol_perC	O/C: biodegradable particulate	0.4202	dUnit/dUnit
	i_O_XUInf_mol_perC	O/C: unbiodegradable particulate	0.4202	dUnit/dUnit
	i_O_XUOrg_mol_perC	O/C: endogenous residue	0.4202	dUnit/dUnit
	i_P_Org_mol_perC	P/C: organisms	0.0219	dUnit/dUnit
	i_P_SF_mol_perC	P/C: fermentable soluble	0.0049	dUnit/dUnit
	i_P_SU_mol_perC	P/C: unbiodegradable soluble	0.0200	dUnit/dUnit
	i_P_XBInf_mol_perC	P/C: PS biodegradable particulate	0.0066	dUnit/dUnit
	i_P_XBOrg_mol_perC	P/C: biodegradable particulate	0.0219	dUnit/dUnit
	i_P_XUInf_mol_perC	P/C: unbiodegradable particulate	0.0219	dUnit/dUnit
	i_P_XUOrg_mol_perC	P/C: endogenous residue	0.0219	dUnit/dUnit
	Y_AC	Acetogenesis yield (COD/COD)	0.039714	-
	Y_AD	Lo H2 Acetogenesis yield (COD/COD)	0.0895	-
	Y_AH	Hi H2 Acetogenesis yield (COD/COD)	0.0895	-
	Y_AM	Acetoclastic Methanogenesis yield (COD/COD)	0.03925	-
	Y_HM	Hydrogenotrophic Methanogenesis yield (COD/COD)	0.04	-
	Y_OB	Acetate oxidizing bacteria yield	0.104	-

Table D-4: AD model parameters

AD Parameters	Value	Unit
i_Ca_PP_mol_perP	0.05	
i_K_PP_mol_perP	0.33	
i_Mg_PP_mol_perP	0.33	
i_H_Org_mol_perC	1.542114926	dUnit/dUnit
i_H_SF_mol_perC	1.957981978	dUnit/dUnit
i_H_SU_mol_perC	1.749857969	dUnit/dUnit
i_H_XBInf_mol_perC	2.307258453	dUnit/dUnit
i_H_XBOrg_mol_perC	1.542114926	dUnit/dUnit
i_H_XUInf_mol_perC	1.538862293	dUnit/dUnit
i_H_XUOrg_mol_perC	1.542114926	dUnit/dUnit
i_N_Org_mol_perC	0.166435506	dUnit/dUnit
i_N_SF_mol_perC	0.036474164	dUnit/dUnit
i_N_SU_mol_perC	0.153124083	dUnit/dUnit
i_N_XBInf_mol_perC	0.038600521	dUnit/dUnit
i_N_XBOrg_mol_perC	0.166435506	dUnit/dUnit
i_N_XUInf_mol_perC	0.166435506	dUnit/dUnit
i_N_XUOrg_mol_perC	0.166435506	dUnit/dUnit
i_O_Org_mol_perC	0.420608108	dUnit/dUnit
i_O_SF_mol_perC	0.676989123	dUnit/dUnit
i_O_SU_mol_perC	0.508134869	dUnit/dUnit
i_O_XBInf_mol_perC	0.745483503	dUnit/dUnit
i_O_XBOrg_mol_perC	0.420608108	dUnit/dUnit
i_O_XUInf_mol_perC	0.420811398	dUnit/dUnit
i_O_XUOrg_mol_perC	0.420608108	dUnit/dUnit
i_P_Org_mol_perC	0.022549327	dUnit/dUnit
i_P_SF_mol_perC	0.007466731	dUnit/dUnit
i_P_SU_mol_perC	0.01990013	dUnit/dUnit
i_P_XBInf_mol_perC	0.007902023	dUnit/dUnit
i_P_XBOrg_mol_perC	0.022549327	dUnit/dUnit
i_P_XUInf_mol_perC	0.022549327	dUnit/dUnit
i_P_XUOrg_mol_perC	0.022549327	dUnit/dUnit
V_liq	18000	L

Appendix E: Modelling results

Table E-1: AS simulation results

Parameter		Scenario 1		Scenario 2	
		MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
Retention Time (d)		10	10	10	10
Volumes (litres)	Anaerobic Vol.	2573714.3	2593660.5	2573714.3	2623711.5
	Anoxic Vol.	6434285.7	6494168.1	6434285.7	6582509.5
	Reaeration Vol.	0.0	0.0	0.0	0.0
	Fully Aerated	12680000.0	13269033.1	12680000.0	14003513.3
	Total Aerobic	12680000.0	13269033.1	12680000.0	14003513.3
	Total Vol.	21688000.0	22356861.7	21688000.0	23209734.2
Temp.(ºC)		20.0	20.0	20.0	20.0
Sludge Recycle (ratios)	a-recycle	3.0	3.0	3.0	3.0
	r-recycle	0.0	0.0	0.0	0.0
	S-recycle	1.0	1.0	1.0	1.0
Flow Rates (litres/day)	Influent (Qi)	57475600.8	57472450.4	57485600.8	57432093.8
	Waste (Qw)	2168800.0	2168800.0	2168800.0	2168800.0
	Effluent (Qe)	55306800.8	55303650.4	55316800.8	55263293.8
Influent	Influent COD (mgCOD/l)	423.2	427.4	423.2	427.7
	Influent TKN (mgN/l)	54.1	52.8	53.4	51.9
	Influent TP (mgP/l)	11.5	10.2	6.0	6.1
	COD	283.0	281.5	283.0	281.7

Parameter		Scenario 1		Scenario 2	
		MS Excel steady state (mg/l)	PWM_SAWEST (mg/l)	MS Excel steady state (mg/l)	PWM_SAWEST (mg/l)
Filtered Influent	TKN	51.1	49.1	50.4	48.3
	Influent FSA (mgN/l)	38.1	37.3	37.4	36.4
	TP	10.5	9.1	4.9	4.9
	Influent OrthoP (mgP/l)	8.4	7.1	2.8	3.0
	Mg	9.9	10.0	14.5	14.4
	K	12.3	11.5	10.6	10.7
	Ca	5.2	5.1	5.0	5.0
Anaerobic Tank	TSS (mg/l)	3561.2	3310.3	3084.2	3048.4
	VSS (mg/l)	2565.8	2499.8	2565.8	2555.3
	ISS (mg/l)	995.4	810.4	518.5	493.1
	Nitrates (mgN/l)		0.4		0.4
	OrthoP (mgP/l)		14.4		11.5
Anoxic Tank	TSS (mg/l)	3561.2	3324.1	3084.2	3059.5
	VSS (mg/l)	2565.8	2491.5	2565.8	2547.6
	ISS (mg/l)	995.4	832.6	518.5	511.9
	Nitrates (mgN/l)		0.9		0.8
	OrthoP (mgP/l)		8.3		6.5
Aerobic Tank (total)	COD (Stml) (mgCOD/l)	3865.0	3775.9	3864.9	3859.6
	TKN (mgN/l)	262.7	265.9	262.7	271.3
	TSS (mg/l)	3561.2	3337.7	3084.2	3068.2
	VSS (mg/l)	2565.8	2478.7	2565.8	2534.5
	ISS (mg/l)	995.4	859.0	518.5	533.7
	OUR (mgO/l/h)	56.7	52.8	56.4	49.7

Parameter		Scenario 1		Scenario 2	
		MS Excel steady state (mg/l)	PWM_SAWEST (mg/l)	MS Excel steady state (mg/l)	PWM_SAWEST (mg/l)
	Nitrates (mgNO3-N/l Influent)	8.0	7.7	7.9	7.4
	TP (mgP/l)	273.7	237.5	126.6	135.6
	Effluent TP (filt) (mgP/l)	1.2	1.6	1.2	1.3
	Effluent OrthoP (mgP/l)	0.0	0.4	0.0	0.0
	Mg (mg/l)	55.6	47.5	26.2	28.0
	K (mg/l)	91.2	76.5	30.7	34.4
	Ca (mg/l)	12.0	10.7	6.8	7.1
	VFA (mgCOD/l)	0.0	0.0	0.0	0.0
	Effluent TKN filt (mgN/l)	5.0	4.6	5.0	4.9
	Effluent FSA (mgN/l)	0.8	0.26	0.8	0.6
	Filtered Effluent COD (mgCOD/l)	70.0	73.6	70.0	71.9
	OHO (mgCOD/l)	1146.9	1162.6	1147.0	1127.1
	PAO (mgCOD/l)	1399.2	1216.7	1399.1	1360.0
	PolyP (mgP/l)	197.5	162.2	50.3	59.1
	XER (mgCOD/l)	690.5	658.2	690.5	643.5
	Supe (mgCOD/l)	558.3	557.9	558.3	547.9
	Alkalinity				
	pH				
	Filtered Mg (mg/l)	8.1	8.4	14.1	13.8

Parameter		Scenario 1		Scenario 2	
		MS Excel steady state (mg/l)	**PWM_SAWEST (mg/l)**	**MS Excel steady state (mg/l)**	**PWM_SAWEST (mg/l)**
	Filterd K (mg/l)	9.2	9.0	9.8	9.7
	Filtered Ca (mg/l)	4.9	4.9	5.0	4.9
Denitrification	NO3 Denitrified (mgN/l)	31.3	27.8	30.8	26.8
	NO3 Generated (mgN/l Influent)	40.1	35.6	39.4	34.2
Oxygen requirements	FOn utilised (KgO/d)	10339.0	9271.3	10159.5	8819.6
	FOc utilised (KgO/d)	12070.5	12132.2	12070.6	12296.6
	FOd (KgO/d)	5151.0	4575.3	5061.1	4404.7
	FOt(KgO/d)	17258.6	16828.2	17169.0	16711.5
Mass Balances	COD balance (%)	100.0	99.3	100.0	100.3
	Nitrogen balance (%)	100.0	94.6	100.0	94.5
	Phosphorous balance (%)	100.0	102.8	100.0	104.6
	Mg balance (%)	100.0	98.8	100.0	99.7
	K balance (%)	100.0	99.9	100.0	100.0
	Ca balance (%)	100.0	99.9	100.0	100.0

Table E-2: AD simulation results

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	**PWM_SA WEST (mg/l)**	**MS Excel steady state (mg/l)**	**PWM_SA WEST (mg/l)**
Influent	Influent flow	Ml/d	0.26	0.26	0.26	0.26
	Influent COD	mgCOD/l	93911.60	91718.96	93911.23	93091.51
	Influent COD, Susi	mgCOD/l	70.00	69.91	70.00	69.89
	Influent COD, Sbsfi	mgCOD/l	88.82	79.32	88.82	79.43
	Influent VFA, Sasi	mgCOD/l	0.00	12.60	0.00	12.61
	Influent COD, Stsi	mgCOD/l	158.82	161.84	158.82	161.93
	Influent COD, Sbpi	mgCOD/l	70932.71	69203.57	70932.26	70590.64
	Influent COD, Supi	mgCOD/l	22820.08	22366.15	22820.15	22351.55
	Influent COD, Stpi	mgCOD/l	93752.78	91569.73	93752.42	92942.19
	Influent TKN	mgN/l	3700.41	3576.96	3700.39	3667.67
	Influent filt TKN	mgN/l	24.07	21.50	24.07	21.33
	Influent FSA	mgN/l	16.16	17.23	16.16	17.06
	Influent OrgN	mgN/l	3684.25	3559.73	3684.22	3650.61
	Influent TOD	mgO/l	110822.48	108065.67	110822.00	109852.77
	Influent TP	mgP/l	2770.75	2427.30	1545.08	1604.64
	Influent filt TP	mgP/l	2.93	5.17	2.93	2.61
	Influent OP	mgP/l	1.33	3.93	1.33	1.37
	Influent OrgP	mgP/l	2769.42	2423.37	1543.75	1603.27
	Influent TSS	mg/l	74682.93	70102.31	70707.75	68283.18
	Influent VSS	mg/l	60323.59	56973.11	60323.35	57760.13

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Influent ISS	mg/l	14359.34	13129.19	10384.40	10523.05
	Influent Carbon	mgC/l		29655.07		30136.67
	Influent Hydrogen	mgH/l		4806.56		4868.05
	Influent Oxygen	mgO/l		22850.01		21815.06
	Influent Magnesium	mg/l	405.24	331.54	113.20	132.51
	Filt. Influent Magnesium	mg/l	8.98	9.15	12.20	14.08
	Influent Potassium	mg/l	695.17	568.18	184.63	215.09
	Filt. Influent Potassium	mg/l	9.54	10.38	9.88	10.17
	Influent Calcium	mg/l	64.25	53.26	20.10	22.70
	Filt. Influent Calcium	mg/l	4.96	5.03	4.99	4.98
	Influent Alk	mg/l as CaCO3	200.00	0.00	200.00	0.00
	Influent pH		7.50	0.00	7.50	0.00
	Effluent COD, Ste	mgCOD/l	30060.70	32589.38	30060.70	32768.65
	Effluent COD, Suse	mgCOD/l	70.00	70.76	70.00	70.69
	Effluent COD, Sbsfe	mgCOD/l	0.00	0.00	0.00	0.00
	Effluent VFA, Sase	mgCOD/l	0.00	0.02	0.00	0.02
	Effluent COD, Stse	mgCOD/l	70.00	70.76	70.00	70.69
	Effluent COD, Sbpe	mgCOD/l	7170.62	7451.04	7170.55	7608.49
Effluent	Effluent COD, Supe	mgCOD/l	22820.08	25086.66	22820.15	25107.51

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Effluent COD, Stpe	mgCOD/l	29990.70	32537.69	29990.70	32716.00
	Effluent TKN	mgN/l	3698.59	3652.80	3698.57	3732.69
	Effluent filt TKN	mgN/l	1520.71	1268.65	1688.93	1443.59
	Effluent FSA	mgN/l	1516.44	1264.33	1684.66	1439.28
	Effluent OrgN	mgN/l	2182.15	2388.46	2013.91	2293.41
	Effluent TOD	mgO/l	46963.27	49282.66	46963.16	49827.03
	Effluent TP	mgP/l	2771.54	2500.30	1544.84	1727.45
	Effluent filt TP	mgP/l	1687.08	6986.43	832.92	7135.05
	Effluent OP	mgP/l	1685.84	1431.11	831.68	904.03
	Effluent OrgP	mgP/l	1085.70	1069.19	713.16	823.42
	Effluent TSS	mg/l	33288.52	32240.97	30339.57	30445.96
	Effluent VSS	mg/l	20172.24	22007.05	20172.24	22128.13
	Effluent ISS	mg/l	13116.28	10233.92	10167.33	8317.83
	Effluent Carbon	mgC/l		30407.01		30671.96
	Effluent Hydrogen	mgH/l		2183.42		2111.77
	Effluent Oxygen	mgO/l		32532.05		30799.59
	Effluent Magnesium	mg/l	405.27	350.28	113.21	149.73
	Filt. Effluent Magnesium	mg/l	0.02	26.93	0.04	16.70
	Effluent Potassium	mg/l	695.24	560.68	184.67	229.19
	Filt. Effluent Potassium	mg/l	695.24	550.34	184.67	226.08

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Effluent Calcium	mg/l	64.26	52.46	20.11	17.28
	Filt. Effluent Calcium	mg/l	64.26	51.53	20.11	13.65
	Effluent Alk	mg/l as CaCO3; no P	4423.10	3260.97	4791.20	3260.97
	Effluent Total Alk	mg/l as CaCO3	8329.89	4021.93	6691.85	4021.93
	Digester pH		7.29	7.00	7.33	7.00
Gas Production	Gas production	litres/d	9897646.41	10284552.39	9850957.88	10284552.39
	Gas prod.	l gas/l influent	38036965.40	39464897.88	37857539.94	39464897.88
	CH4 gas component	l gas/l influent	24451634.06	25629738.36	24355022.83	25629738.36
	CO2 gas component	l gas/l influent	13585331.34	13835159.52	13502517.12	13835159.52
	COD of CH4	mgCOD/l feed	64186.77	61511.60	63933.16	62083.13
	pCO2	atm	0.36	0.35	0.36	0.35
COD and nutrient removal	% FSA released	%	40.54	34.86	45.09	38.78
	% OP released a	%	60.80	58.80	53.74	56.25
	%COD removed	%	89.90	89.27	89.90	89.26
	Struvite precipitated	mg/l	4091.70	3204.52	1142.62	1325.02
	ACP precipitated	mg/l	0.00	0.00	0.00	26.00
	Calcite precipitated	mg/l	0.00	0.07	0.00	0.00
Mass balances	COD balance	%	100.36	102.60	100.09	101.89
	Nitrogen balance	%	99.95	102.12	99.95	101.77

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Phosphorous balance	%	100.03	103.01	99.98	107.65
	Magnesium balance	%	100.01	105.65	100.01	113.00
	Potassium balance	%	100.01	98.68	100.02	106.56
	Calcium balance	%	100.02	98.48	100.05	76.14

Table E-3: Struvite precipitation model results

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Influent flow	Ml/d	-	-	2.2	2.2
	Influent COD	mgCOD/l	-	-	69.4	37.4
	Influent COD, Susi	mgCOD/l	-	-	69.4	69.3
	Influent COD, Sbsfi	mgCOD/l	-	-	0.0	0.00
	Influent VFA, Sasi	mgCOD/l	-	-	0.0	0.0
	Influent COD, Stsi	mgCOD/l	-	-	69.4	37.4
	Influent COD, Sbpi	mgCOD/l	-	-	0.0	0.0
	Influent COD, Supi	mgCOD/l	-	-	0.0	0.0
	Influent COD, Stpi	mgCOD/l	-	-	0.0	0.0
Influent	Influent TKN	mgN/l	-	-	148.9	133.2

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Influent filt TKN	mgN/l	-	-	148.9	126.2
	Influent FSA	mgN/l	-	-	144.6	122.0
	Influent OrgN	mgN/l	-	-	4.2	11.3
	Influent TOD	mgO/l	-	-	749.7	646.3
	Influent TP	mgP/l	-	-	72.3	77.5
	Influent filt TP	mgP/l	-	-	72.3	77.5
	Influent OP	mgP/l	-	-	71.1	76.3
	Influent OrgP	mgP/l	-	-	1.2	1.2
	Influent TSS	mg/l	-	-	0.0	0.0
	Influent VSS	mg/l	-	-	0.0	0.0
	Influent ISS	mg/l	-	-	0.0	0.0
	Influent Carbon	mgC/l	-	-		200.1
	Influent Hydrogen	mgH/l	-	-		293.7
	Influent Oxygen	mgO/l	-	-		2889.0
	Influent Magnesium	mg/l	-	-	181.2	182.6
	Filt. Influent Magnesium	mg/l	-	-	181.2	182.6
	Influent Potassium	mg/l	-	-	24.6	27.9
	Filt. Influent Potassium	mg/l	-	-	24.6	27.9
	Influent Calcium	mg/l	-	-	6.2	5.6
	Filt. Influent Calcium	mg/l	-	-	6.2	5.6

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Influent Alk	mg/l as CaCO3	-	-	200.0	0.0
	Influent pH		-	-	7.5	0.0
Effluent	Effluent COD, Ste	mgCOD/l	-	-	69.4	37.4
	Effluent COD, Suse	mgCOD/l	-	-	69.4	69.3
	Effluent COD, Sbsfe	mgCOD/l	-	-	0.0	-31.9
	Effluent VFA, Sase	mgCOD/l	-	-	0.0	0.0
	Effluent COD, Stse	mgCOD/l	-	-	69.4	69.3
	Effluent COD, Sbpe	mgCOD/l	-	-	0.0	0.0
	Effluent COD, Supe	mgCOD/l	-	-	0.0	0.0
	Effluent COD, Stpe	mgCOD/l	-	-	0.0	0.0
	Effluent TKN	mgN/l	-	-	148.5	133.3
	Effluent filt TKN	mgN/l	-	-	117.4	93.6
	Effluent FSA	mgN/l	-	-	113.2	89.4
	Effluent OrgN	mgN/l	-	-	35.3	43.9
	Effluent TOD	mgO/l	-	-	748.0	646.5
	Effluent TP	mgP/l	-	-	72.2	77.6
	Effluent filt TP	mgP/l	-	-	2.7	5.5
	Effluent OP	mgP/l	-	-	1.5	4.3
	Effluent OrgP	mgP/l	-	-	1.2	73.3
	Effluent TSS	mg/l	-	-	550.6	570.9

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Effluent VSS	mg/l	-	-	0.0	0.0
	Effluent ISS	mg/l	-	-	550.6	570.9
	Effluent Carbon	mgC/l	-	-		200.2
	Effluent Hydrogen	mgH/l	-	-		71.7
	Effluent Oxygen	mgO/l	-	-		1126.8
	Effluent Magnesium	mg/l	-	-	181.3	182.7
	Filt. Effluent Magnesium	mg/l	-	-	126.7	126.1
	Effluent Potassium	mg/l	-	-	24.6	27.9
	Filt. Effluent Potassium	mg/l	-	-	24.6	27.9
	Effluent Calcium	mg/l	-	-	6.2	5.6
	Filt. Effluent Calcium	mg/l	-	-	6.2	5.6
	Effluent Alk	mg/l as CaCO3; no P	-	-	5002.1	3261.0
	Effluent Total Alk	mg/l as CaCO3	-	-	5005.0	4021.9
	Digester pH		-	-	7.8	7.0
COD and nutrient removal	% FSA released	%	-	-	-21.1	-24.4
	% OP released a	%	-	-	-96.2	-92.9
	%COD removed	%	-	-	0.0	0.0
	Struvite precipitated	mg/l	-	-	550.6	570.9
	ACP precipitated	mg/l	-	-	0.0	0.0

Group	Parameter	Unit	Scenario 1		Scenario 2	
			MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)	MS Excel steady state (mg/l)	PWM_SA WEST (mg/l)
	Calcite precipitated	mg/l	-	-	0.0	0.0
Mass balances	COD balance	%	-	-	100.0	100.0
	Nitrogen balance	%	-	-	99.8	100.0
	Phosphorous balance	%	-	-	99.9	100.0
	Magnesium balance	%	-	-	100.0	100.0
	Potassium balance	%	-	-	100.0	100.0
	Calcium balance	%	-	-	100.2	100.0