CHEMICAL OXYGEN DEMAND (COD) FRACTIONATION FOR PROCESS MODELLING CONSIDERATIONS AND OPTIMIZATION

This dissertation is submitted in the fulfilment of the requirements for the degree of Master of Engineering: Chemical Engineering in the Faculty of Engineering and the Built Environment at Durban University of Technology

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Supervisor: Prof P. Musonge Date: March 2021

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Declaration

I hereby declare that this dissertation presents my original work towards fulfilling the requirements for a Master of Engineering qualification in chemical engineering with the Durban University of Technology. This dissertation has not been submitted in any form to another academic institution. I declare that this dissertation has appropriately acknowledged in the text where use of the work of others was made. This document must not be reproduced, copied or sold without the consent of the author and the authorities of the Durban University of Technology.

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Dedication

I dedicate this work to my pillars of strength and source of inspiration:

- My late father, Felokwakhe Jwara – For always instilling the value of education in me.
- My Mother, Nokulunga Jwara – For always supporting my academic decisions.
- My sisters, Nozipho and Mandisa Jwara - For always looking up to me and making me push myself harder every time.
- My handsome boys, Lwazi and Bandile Mthembu – For always being patient with me and my studies.
- Last but never least, My Lord, Jesus Christ, whom all is possible.

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- My family for their unconditional support throughout my studies.
- My Lord, Jesus Christ, who gave me the strength to complete this thesis.

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Conference Presentations

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Abstract

Wastewater treatment is a critical chain in the urban water cycle. Wastewater treatment prevents the toxic contamination of water bodies. The notable consequences of contamination are the loss of aquatic life, upsurge of eutrophication due to nutrient overload, and potential loss of human life as a result of waterborne diseases. Wastewater works (WWW) are therefore an intrinsic component of protecting the urban water cycle and ensuring that water resources are preserved for future generations. The operation of a WWW is subject to compliance with the national legislative requirements imposed by the Department of Water and Sanitation (DWS) to ensure the preservation of water resources. These requirements oblige water and sanitation departments to employ innovative design, control and optimization of WWW. Wastewater modelling packages have presented the opportunity to simulate the wastewater treatment processes in order to maintain and sustain legal compliance with the DWS.

The successful implementation of a simulation package for wastewater process optimization and modelling depends on an accurate characterization also known as fractionation of the organic fractions of the WWW influents. This thesis is a result of a comprehensive study reported for Darvill wastewater work. Darvill WWW is a 60 ML/D plant which has been receiving flows of up to 120 ML/D. The importance of the study was to motivate for the upgrade of the wastewater work to account for the increased hydraulic, organic and nutrient loading into the plant. The study looked at the application of the World Engine for Simulation and Training (WEST) and all studies required to generate data that will serve as input with the understanding the current state of Darvill WWW in terms of performance.

The study presents the fractionation outcomes of the primary wastewater effluent organic matter as chemical oxygen demand (COD) and the performance by assessing the biological nutrient removal process (BNR) using BNR efficiencies in addition to the development of the Darvill WWW WEST model with the aid of the probabilistic fractionator. The fractionation was achieved through the oxygen uptake rate experiments using the respirometry method. Experiments yielded the following results: biodegradable COD (bCOD) (70.5%) and inert COD (iCOD) (29.5%) of the total COD. Further characterization of the bCOD and iCOD yielded the readily biodegradable fraction (S_S) at 75%, slowly degradable (X_S) at 25%, particulate inert (X_I) was 50.8% and the inert soluble S_I at 49.2%. The COD fractions were used and served as input to the development and evaluation of the Darvill WEST model. Calculations of BNR efficiencies were used to evaluate the effects of high inflow to the

biological treatability of the activated sludge for the period September 2016 - November 2017. It was found that at inflows above design capacity, the nutrient removal efficiency reduced from an expected 80-90% to an average of 40% with an average soluble reactive phosphorus (SRP) removal efficiency being 64%.

A data input file for the period of January – June 2016 was created to serve as input into WEST to develop a baseline average model for the Darvill WWW plant. The model results predicted a mixed liquor suspended solids (MLSS) concentration of 6475 mg/L for the plant during the study period this was comparable with the plant MLSS concentration of 6700 mg/L at the time which was above the design concentration of 4500 mg/L. This was largely due to the plant operating under nutrient overload conditions. The final effluent (FE) concentration in the defractionation model was found to be COD = 41.28 mg/L, ammonia (NH_3) = 22.02 mg/L, Total Suspended Solids (TSS) = 32 mg/L, SRP = 2.16 mg/L. Most of these results were expectedly non-compliant to the discharge limits imposed by the DWS with the exception of COD. The plant FE measurements were COD = 45.1 mg/L, NH_3 = 3.4 mg/L, TSS = 20.9 mg/L, SRP= 6.67 mg/L. The COD and TSS prediction were comparable to the model prediction however there were limitations in the models ability to predict NH_3 and SRP. The model does not account for changes in dissolved oxygen (DO) and temperature as these parameters are kept constant for the purpose of this study.

The model assumes a temperature of 20 °C and a DO concentration of 2 mg/L for the aerobic reactor, 0.01 mg/L for the anaerobic reactor and 0.1 mg/L for the anoxic reactor. The model assumes that with the nutrient overload, oxygen compensation occurs within the reactor to maintain a constant DO concentration within the units. This limits the model in the prediction of actual instance where the overload would deplete the DO and where other competing reactions would give rise to greater non-compliances as well as biological growth's impairment due to cold weather conditions.

Keywords: wastewater modelling, activated sludge models, industrial effluents, COD fractions, domestic effluents

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1. Introduction

1.1 Background

Wastewater treatment is a critical chain in the urban water cycle. Nature has its own means of attenuating wastes; however, in larger volumes and concentrations, the process becomes slower and more problematic. This has led to the design of wastewater works (WWW) which are intended to aid in the faster biodegradation and removal of pollutants found in waste water (Bashide, 2015). The purpose of a WWW is for the treatment of municipal and industrial wastewater (WW) to acceptable effluent quality, devoid of excess organics - chemical oxygen demand (COD), ammonia, nitrogen (N) and phosphorous (P) nutrients. Wastewater treatment prevents toxic contamination of water bodies. The notable consequences of contamination are the loss of aquatic life, upsurge of eutrophication, and potential loss of human life due to waterborne diseases. Wastewater works are therefore an intrinsic component of protecting the urban water cycle and ensuring that water resources are preserved for future generations (Ying Xin Wu, 2015).

The operation of a WWW is subject to compliance with the national legislative requirements of the Department of Water and Sanitation (DWS). Increasing urbanization and scarcity of water resources requires proper design, optimization considerations and treatment methods in order to maintain the quality of the existing and affected water bodies. Darvill wastewater works is a wastewater treatment plant situated in Pietermaritzburg and owned by Umgeni Water, South Africa. It has an average dry weather flow (ADWF) design capacity of 65 ML/D. The WWW is currently undergoing a capacity upgrade.

A study was conducted in 2010 to predict future flows into the Darvill WWW. These flows were predicted using both long and short term trends and data from 2006 to 2009. The findings from the study revealed that there was an increase in the influent flow since 2008 and also a 33% increase in the organic load coming into the plant. The organic load was mainly from industrial discharge into the plant. The ADWF within the Darvill WWW catchment are expected to grow to about 90 ML/D by 2021. The upgrade design makes allowance for expansion in the future to a maximum capacity of 120 ML/D projected to occur in 2029.

It was also seen that the storm water ingress posed a major process constraint on the WWW and urgent intervention was required to mitigate this limitation. Darvill WWW struggled to process the high flows and recycling of the sometimes septic wastewater from the storm dam caused major process upsets which affected process compliance of the waste water works in its COD and nutrient removal efficiencies. The data further suggested that there had been a significant growth in the population in the Pietermaritzburg area and therefore the WWW had to undergo an upgrade to be able to treat the plant's influent in order to meet the DWS compliance standards. The influent wastewater characteristics for the WWW had to be determined as well as COD fractions for design purposes. Ntaka *et al.* (2016) conducted this study and observed that there were high variations in influent COD, varying seasonally and daily, more data was required to assess these trends and specific optimization models were required to predict and simulate plant performance.

COD can be fractionated into biodegradable COD (bCOD) and non-biodegradable (iCOD) COD. The bCOD is oxidized and leaves the process as carbon dioxide gas while the non-biodegradable COD leaves the process unaltered. If high levels of bCOD are discharged into a water body, the aquatic life is threatened as the biochemical reaction of breaking down the bCOD consumes the dissolved oxygen in the water body. The bCOD can be further fractionated into readily biodegradable COD (S_S) and slowly biodegradable COD (X_S). The S_S fraction serves as a substrate for the enhanced biological phosphorus removal (EBPR) process. COD fractionation has been the focal point in wastewater treatment when the biodegradability of incoming raw wastewater is to be understood (Ntaka e*t al.* 2016). Wastewater characterization is considered an indispensable step in generating necessary information for reliable modelling, optimisation and design of the biological treatment processes (Orhon and Cokgor 1997). Optimization, modelling and process design considerations require precise COD fractions and constituents in the plant's influent (Ntaka, *et al.* 2016).

Models are unique to specific systems and are developed using the plant's historic data parameters (Rössle and Pretorius 2001). The modelling of the activated sludge system and the various unit processes linked has expanded with the development of wastewater treatment systems. Previous traditional models (pre-1980s) had a black-box approach, where the biological and physical processes were not fully understood. These models were primarily based on empirical relationships, experiences, and the rule-of-thumb which were established by recognising important parameters that seemed to describe behavioural patterns and observations on the system (van Loosdrecht *et al*, 2008). The traditional models do not provide the necessary depth and accuracy that are required by WWW modellers and designers to meet the modern day environmental pressures, stringent effluent standards, and the economic costs of starting and running a WWW (Ying Xin Wu, 2015). The inaccuracies and inadequacies of the traditional models stimulated developments in the models. The developments were based on the behavioural patterns of the microorganism mediating the wastewater treatment processes (Ikumi *et al*, 2014a). The development of the activated sludge (AS) models such as: ASM1 by Henze *et al.* (1987); ASM2 by Henze *et al.* (1995); and ASM2d by Henze *et al.* (1999) was deemed as the most significant contribution for the modelling of biological wastewater processes (Makinia, 2005).

The current study will determine the COD fractions and develop a baseline model using the dedicated WWW simulator, World Engine for Simulation and Training (WEST) package for the optimization of the Darvill WWW process. Dedicated WWW simulators are software packages that are designed for the simulation of WWW. Examples of dedicated simulators include: *WEST*, *SIMBA*, *BioWin*, *EFOR*, *GPS-X* and *STOAT*. The models generated from these dedicated simulators have unified and streamlined wastewater treatment concepts and additionally standardised the way the processes are modelled. This allowed them to be coded into computer simulation. In these programs, multiple, complex mathematical equations could be solved by running the program. WEST will offer the modelling and simulation environment for the Darvill wastewater treatment process. The model developed will be used to assist in informing the process of setting the conditions for the optimization of the EBNR process.

1.1 Problem statement

Darvill WWW has been experiencing non-compliances in their final effluent nutrient discharge into the river due to the plant being hydraulically and organically overloaded. The plant receives both domestic and industrial effluents. While the characteristics of the domestic stream are predictable, the industrial stream tends to vary considerably (seasonally and daily) depending on the production processes adopted. Darvill WWW has to comply with the same compliance standards set by DWS despite these variations.

1.2 Aim

To determine, develop and test a baseline WEST model for Darvill WWW and evaluate its suitability and applicability for the wastewater works.

1.3 Objectives of study

1. To determine the COD fractions of Darvill WWW settled sewage.
2. To determine the current biological nutrient removal (BNR) efficiency in the activated sludge system.
3. To Develop and Test the WEST (Worldwide Engine for Simulation and Training) model by:
    a. Generating baseline WEST model using COD fractionation and nutrient data obtained from objectives 1&2
    b. Evaluate the performance, applicability and suitability of the Darvill WWW WEST model.

1.4 Significance of study

The COD fractions will assist in understanding the unique characteristics of Darvill settled sewage. The biological nutrient removal efficiencies will give an indication of current plant performance to detect process impairments that exist due to hydraulic overloading. The Darvill WEST model will assist in optimizing plant performance and maintaining legal compliance.

2. Literature review

2.1 Introduction

This chapter presents the review of relevant literature to this study under the following sub-headings:

- Introduction: wastewater treatment
- Preliminary and primary treatment process
- The secondary treatment process
- Wastewater characterization
- Activated sludge modelling
- WEST
- Wastewater legislation and environmental significance

2.2 Wastewater treatment

Wastewater has for a long time been considered a potential health risk and nuisance in urban agglomerations and therefore wastewater management has been developed and improved to match the ever changing needs of society and nature (Henze *et. al*. 2008). In the middle of the 19^{th} century the first combined wastewater system was built in London. The wastewater was directly conveyed into the Thames River. However, it was realised that the direct discharge of wastewater into the river produced an extreme load burden on the water body which lead to detrimental effects on aquatic flora and fauna- eutrophication. Henze *et. al*. (2008) defined eutrophication as the explosive growth of algae and other water plants due to the fertilizing effects of nitrogen and phosphorus present in wastewater. In the early 20th century wastewater treatment plants which treated the wastewater by percolating filter processes were developed. In 1910, the first activated sludge process was used to treat wastewater. This process constituted a milestone in wastewater treatment and resulted in a considerable improvement of the receiving water quality and has been further developed until today (Köhler, 2008).

Today the field of wastewater treatment has grown enormously from simple fill and draw aeration systems to modern activated sludge systems that comprise of intricate interrelated physical-chemical-biological processes (Makinia, 2005). Physical, chemical and biological techniques are used in wastewater treatment processes. These techniques can be classified as physical unit operations, chemical unit processes and biological unit processes. In physical unit operations, physical forces are predominant, and the unit operations include screening, mixing, sedimentation, filtration and adsorption. In chemical unit processes, transformation of constituents or removal occurs as a result of the addition of chemicals or other chemical reactions occurring. Chemical unit processes include oxidation, disinfection and precipitation. In biological unit processes, treatment of wastewater occurs as a result of biological activity, which is mainly responsible for the removal of biodegradable organic matter and nutrients in the waste water. Biological unit processes include activated sludge and trickling filter processes (Mhlanga, 2008).

Wastewater treatment is also sequential processes which convert the harmful constituents present in wastewater. Wastewater treatment is broadly categorized into the following stages: preliminary treatment; primary treatment; secondary treatment; tertiary treatment and solids treatment (Naidoo, 2013). These stages will be discussed in the next sections. The selection of wastewater treatment process or subsequent processes depend on a number of factors, i.e. the influent wastewater characteristics e.g. pH, percentage solids, COD, required effluent quality, cost and availability of land, presence of toxic material, required effluent quality.

2.3 The Preliminary and Primary Treatment Process

Preliminary and primary treatment of wastewater is the removal of suspended solids, debris and grit for the subsequent process through neutralization and/or equalization (Ramalho, 1983). It protects subsequent treatment from blockages, overloading and prevents damage to mechanical equipment. Large solids in the influent are either removed from the flow by screens or sand degritters (Hester and Harrison, 1995).

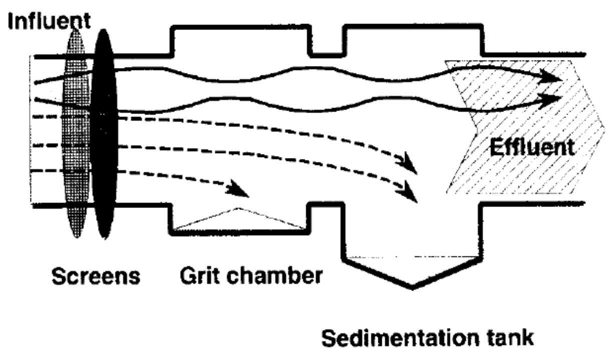


Figure 1: Preliminary and primary treatment schematic representation (EPA.1998)

2.3.1 Screening

Screening is a method employed for the removal of solids of various sizes.it is the first unit operation in the preliminary treatment process of wastewater influent. The screening assists in the removal of objects such as rags, paper, plastics, and metals to prevent damage and clogging of downstream equipment, piping, and appurtenances (Pybus *et al*. 2002). Some modern wastewater treatment plants use both coarse screens and fine screens. Coarse screens remove large solids, rags, and debris from wastewater, and typically have openings of 10 mm or larger. Types of coarse screens include mechanically and manually cleaned bar screens (see Figure 2). Fine screens are normally used for the removal of material that may create operational and maintenance problems in downstream processes, mainly in systems that lack primary treatment (see Figure 3). Typical opening sizes for fine screens are 1.5 to 6 mm they are usually made of steel mesh or perforated steel plates, 5-25% of suspended matter is removed. Very fine screens with openings of 0.2 to 1.5 mm placed after coarse or fine screens are able to reduce suspended solids to concentrations close to those achieved by primary settling (Nozaic and Freese, 2008). At times fine screens are used instead of primary sedimentation tanks however Ramalho (1983) argued that besides the removal efficiency of these screens, clogging is frequently a problem thus their use is uncommon as they increase maintenance cost as compared to the primary settling tank.

Table 1 Classification of screens (Nozaic and Freese, 2008)

“Screening device classification	**Size classification**	**Size range of screen opening**
Bar Screen		
Manually cleaned	Coarse	25 - 50 mm
Mechanically cleaned	Coarse	15 - 75 mm
Fine bar or perforated coarse screen (mechanically cleaned)		
Fine bar	Fine to coarse	3 - 12.5 mm
Perforated plate	Fine to coarse	3 - 9.5 mm
Rotary drum	Fine to coarse	3 - 12.5 mm
Fine screen (mechanically cleaned)		
Fixed parabolic	Fine	0.25 - 3.2 mm
Rotary drum	Fine	0.25 - 3.2 mm
Rotary disc	Very fine	0.15 - 0.38 mm”

Figure 2: Typical wastewater manually raked course screen (Park, 2015)

Figure 3: Typical wastewater fine screen (Schreiber, 2018)

2.3.2 Degritting

Degritting is the removal grit in the preliminary treatment stage in wastewater. Grit (detritus) includes sand, gravel, ashes, or other heavy solid materials that are "heavier" (higher specific gravity) than the organic biodegradable solids in the wastewater. Other grit normally removed during degritting includes bone chips, eggshells, coffee grounds, large organic particles e.g. food waste, and seeds (Nozaic and Freese, 2008). Pybus (2002) infers that the basic principle of grit removal is that grit has a higher density and settles out more rapidly than the less dense organic solids that are carried through with the wastewater. Removal of grit prevents unnecessary abrasion and wear of mechanical equipment, grit deposition in pipelines and channels, and accumulation of grit in downstream process units (Nozaic and Freese, 2008).

2.3.3 Primary sedimentation

Primary sedimentation is utilized to separate suspended solids which could be of organic or inorganic origin from wastewater. This reduces the organic load entering into the activated sludge reactor. The reduction in load is achieved as the result of the solid material in the wastewater settling out to the bottom of the tank based on the difference in specific gravity between solid particles and the bulk of the liquid under the influence of gravity (Hester and Harrison, 1995). According to Pybus (2002) volume of primary sludge removed represents about 2% of the influent wastewater volume, it constitutes about 30-40% of the COD received and about 40-60% of the suspended solids loading.

Over the years, the primary treatment process alone has been unable to treat pollutants in wastewater. This led to secondary treatment, and in some cases, the additional use of advanced treatment processes to remove nutrients and other contaminants to ensure compliance and water resource preservation (EPA.1998).

2.4 The Secondary Treatment Process

The secondary stage of treatment removes about 85 percent of the organic matter in sewage by making use of the bacteria in it (EPA.1998). Bacteria play a primary role in the capture of energy from the sun. Furthermore, their biological activities also complete critical segments of the cycles of carbon, oxygen, nitrogen, and other elements necessary for life (Coker, 2001). The principal secondary treatment techniques used in secondary treatment are the trickling filter, aerated lagoons, contact-stabilization and the activated sludge process (EPA.1998). In this section we are going to discuss the activated sludge process. Basically all the aerobic biological processes mentioned above operate on the same principle. They differ in only in the conditions under which the biological reactions are constrained to operate, called system constraints (Henze *et al*. 2008).

2.4.1 Activated Sludge Process

Activated sludge process has been employed to treat a wide variety of wastewater, and over 90% of the municipal wastewater treatment plants use it as the core part of the treatment process. The basic function of a wastewater biological treatment process is to convert organics to carbon dioxide, water and bacterial cells (Liu, 2003). There are two basic types of activated sludge processes i.e. conventional activated sludge where the process comprises treatment of settled sewage, and extended aeration where the feed is raw or unsettled sewage. Conventional activated sludge is normally used for larger plants, usually 10 MLD and larger. Extended aeration tends to be used on smaller plants usually less than 10 MLD. The extended aeration process consists of an aeration tank and a settling tank (clarifier) for the liquid treatment and on a sequencing batch reactor (SBR) plant only an aeration tank is required (Nozaic and Freese 2008). However, in this study we will be focusing on the conventional activated sludge process as Darvill WWW operates on the conventional activated sludge process. The activated sludge in a wastewater treatment works is a complex ecosystem of competing microorganisms (see Figure 4). The dominant microorganisms are the bacteria, which are about 300 species ubiquitous. Each bacteria is unicellular varying in size from about 0.5 – 2 μm. The outside of the cell is surrounded by a membrane that adjusts the inflow of ions and molecules from the surrounding water. This, in turn is surrounded by a hard cell wall, created of a sugar polymer. The interior of the cell contains the cytoplasm and the thousands of different chemicals whose reactions are regulated by enzymes (see Figure 5) (Ahansazan *et al*. 2014).

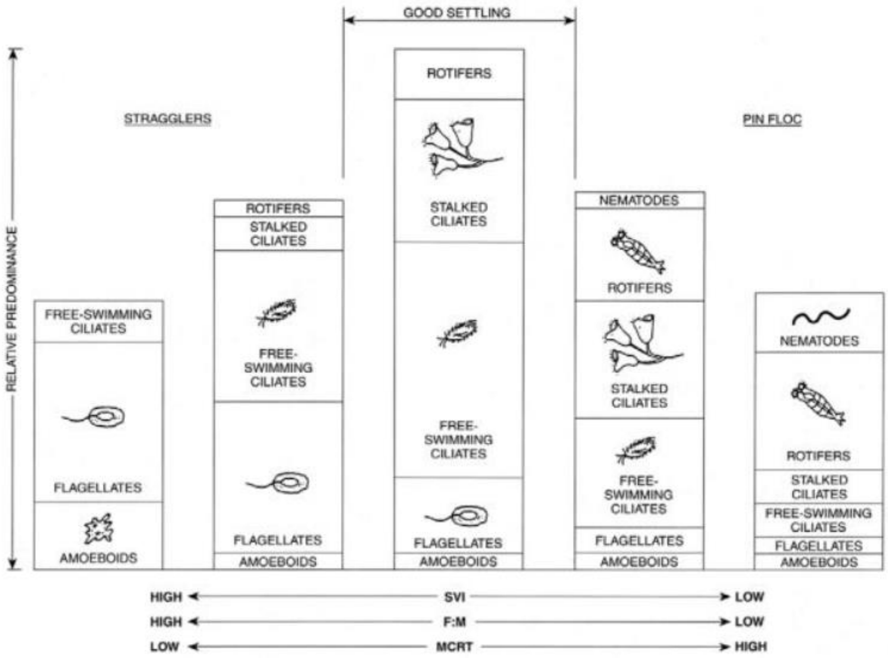


Figure 4: Types of activated sludge microorganisms (Tomasik, 2017)

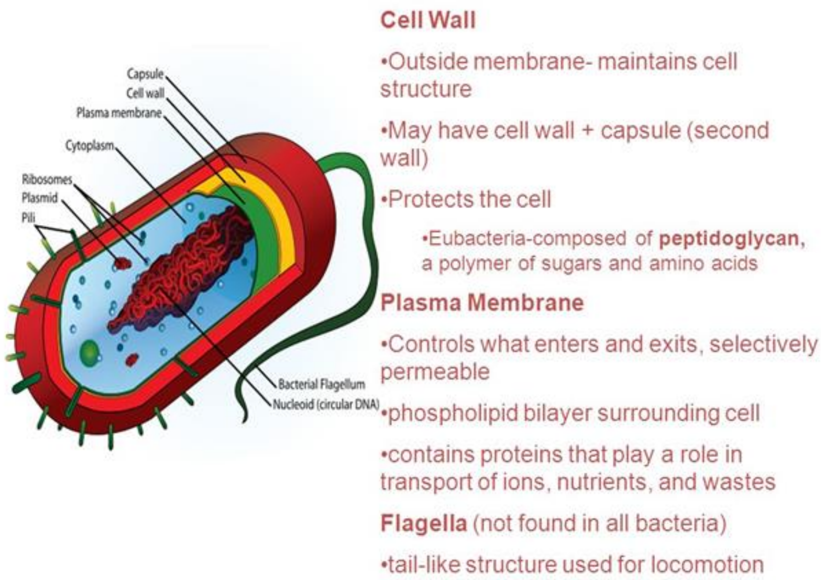


Figure 5: A typical cell structure of bacteria (Berry 2008)

In the activated sludge system aerobic refers to the presence of oxygen, anoxic refers to the absence of oxygen, but the presence of nitrates whereas anaerobic refers to the absence of both oxygen and nitrates. According to Dickinson (1974) the activated sludge process proceeds in distinct steps- adsorption, oxidation of organic matter and oxidation of ammonia and its derivatives to nitrate. The bacteria in the activated sludge are referred to as mixed liquor suspended solids (MLSS) which in practice represents all suspended matter in the system both biological and inert. Another method of expressing the active portion of the activated sludge is by the use of volatile mixed liquor suspended solids which is a measure of the organic fraction and has a value of approximately 80% of the MLSS (James, 1984).

2.4.2 Microbial growth kinetics

The performance of biological processes used for wastewater treatment depends on the dynamics of substrate utilization and microbial growth. The kinetics of microbial growth govern the oxidation i.e. utilization of substrate and production of biomass, which contributes to the total suspended solids in the bioreactor (Tchobanoglous, 2004). The biomass requires nitrogen and phosphorus in order to perform synthesis, metabolism and removal of organics. Nitrogen is available to the biomass in the preferred form as ammonium or as nitrate. Phosphorus must be in the form of soluble orthophosphate in order to be assimilated by the biomass. The maximum nutrient mass ratio to assure adequate nitrogen and phosphorus for COD removal is 100:5:1 (BOD: N: P also known as C: N: P) (Eckenfelder et al. 1995).

2.4.3 Rate of utilization of the soluble substrate

This primary discussion in the current chapter is the removal of the substrate in wastewater i.e. COD. Tchobanoglous *et al.* (2000) describes this as the depletion of the electron donor (COD), this is the case for OHOs and for ANOs it is ammonia or nitrite that is reduced. The substrate utilization rate in biological systems can be modelled with the following expression for soluble substrates.

$$r_{su} = -\frac{kXS}{K_s + S} \quad (1)$$

Where:

r_{su}	rate of substrate concentration change due to utilization. g/.m^3.d
k	maximum specific substrate utilization rate, g substrate/g microorganisms
X	biomass concentration, g/m^3
S	growth limiting substrate concentration in solution, g/m^3
K_s	Half velocity constant, substrate concentration at one half the maximum specific substrate utilization rate, g/m^3

The negative value denotes that the mass of the substrate is decreasing with time due to substrate utilization.

2.4.4 Rate of oxygen uptake

The rate of oxygen uptake is related stoichiometrically to the organic utilization and growth rate. Therefore, the oxygen uptake rate can be defined as:

$$r_o = r_{su} - 1.42 r_g \quad (2)$$

Where:

r_o	oxygen uptake rate, g O_2/m^3.d
r_{su}	rate of substrate utilization, g bsCOD/m^3.d
1.42	The COD of the cell tissue, g bsCOD/g VSS
r_g	rate of the biomass growth, g VSS/m^3·d

The negative sign is required in front of the term r_{su} because the rate of substrate utilization as given by equation 1 is negative. This means that the substrate concentration decreases with time (Tchobanoglous *et al.* 2000).

2.4.5 Oxygen requirement in the Activated sludge system

Atmospheric oxygen is the most common oxidant and is used extensively in the activated sludge treatment processes. It is used to strip out volatile compounds from the wastewater and to initiate the oxidation of inorganic and organic compounds where the wastewater entering the plant may be anaerobic or limited in its available dissolved oxygen. The presence of dissolved oxygen in wastewater as a result of aeration initiates a number of oxidation reactions, some of which are almost immediate whereas others are slow and might need the assistance of a stronger oxidant (Freese et al. 2004). The air can be supplied into the reactor by a variety of means, including:

a. Mechanical surface aerators

b. Blowing air into the mixed liquor through diffusers

c. Allow air from a spurge pipe to be broken up into fine bubbles by a rotating impeller which also provides turbulence required to keep the sludge in suspension.

2.4.6 Activated sludge process advancements

The evolution of the design and working of the activated sludge process for phosphate removal has been reviewed in depth (Barnard, 1976; Grady and Lim, 1980: Toerien *et al.* 1990) and a brief explanation given in this dissertation. Although a number of activated sludge systems have been developed (e.g. Bardenpho, UCT, VIP, Biodenipho, etc), only one process will be discussed in detail i.e. the Phoredox system, also known as the modified Bardenpho system or the 5-Stage Bardenpho. This system has evolved for biological removal of carbon, nitrogen and phosphorus. The system consists of an anaerobic zone followed by a primary anoxic zone, a primary aerobic zone, a secondary anoxic zone, a secondary aerobic zone and lastly a clarifier (settler). In this system sludge is returned from the clarifier to re-enter the anaerobic zone with the influent. Mixed liquor is returned from the primary aerobic zone to the primary anoxic zone (Processes).

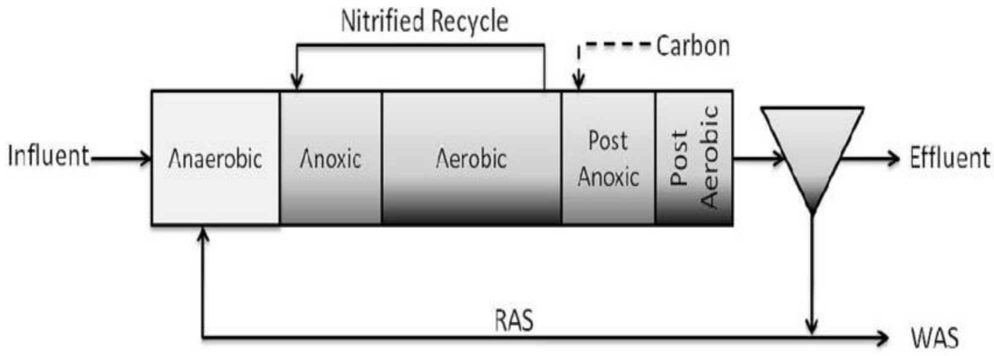


Figure 6: Five stage Bardenpho process (Jimenez et al 2015)

2.5 Wastewater Characterization

Changing wastewater characteristics and the imposition of stricter limits on wastewater discharges have placed greater emphasis on wastewater characterization (Tchobanoglous, *et al*. 2003). Wastewater characterisation prior to treatment is a valuable tool towards the implementation of an effective wastewater management framework (Pybus, 2002). It is also widely used in process modelling for design and optimization of biological treatment processes (e.g. activated sludge process), particularly in wastewaters containing industrial waste (Tchobanoglous, *et al*. 2003).

The constituents of wastewater include a varying range of potential contaminants such as silt; sand; chemical residues; industrial cooling waters; industrial process waters; biodegradable organic wastes; detergents; pesticides; fats; oil; greases; solvents; phenols; cyanides; nutrients (nitrogen (N), phosphates (P), ammonia (NH_3); metals (Hg, Pb, Cd, Cr, Cu, Ni) and microorganisms (pathogenic bacteria, viruses and worm eggs and a range of naturally occurring and xenobiotic organic compounds, including bioactive pharmaceuticals (Naidoo, 2014). These constituents can either be measured as COD, NH_3, P, N, VFA, etc. Some of these constituents will be discussed in detail in the next sections.

The compositions of typical municipal wastewater constituents are shown below (see Table 2), where concentrated wastewater (high) represents cases with low water consumption/infiltration. Diluted wastewater (low) represents high water consumption and/or infiltration. Storm water will further dilute the wastewater as most storm water constituents have lower concentrations compared to highly diluted wastewater (Henze *et al*. 2008). The strength of sewage arriving at treatment works varies considerably, depending largely upon the domestic living and standards of the contributing population and industries (Nozaic & Freese 2008).

Table 2 Typical composition of raw municipal wastewater with minor contributions of industrial wastewater (Henze *et al*. 2008)

Constituent		**Concentration, mg/l**	
	High	**Medium**	**Low**
COD total	1200	750	500
COD soluble	480	300	200
COD suspended	720	450	300
BOD	560	350	230
VFA (as acetate)	80	30	10
Total N	100	60	30
Ammonia-N	75	45	20
Total Phosphorus	25	15	6
Ortho-Phosphorus	15	10	4
Total suspended solids (TSS)	600	400	250
Volatile suspended solids (VSS)	480	320	200

The compositions of typical municipal wastewater nutrients are shown below (see Table 3). The fraction of nitrogen and phosphorus has an influence in treatment options for wastewater. Since most nutrients are generally soluble, they cannot be removed by settling, filtration flotation or other forms of solid liquid separation (Henze *et al*. 2008).

Table 3 Typical content of nutrients in raw municipal wastewater with minor contributions of industrial wastewater (Henze *et al*. 2008)

Constituent		**Concentration, mg/l**	
	High	**Medium**	**Low**
Total N	100	60	30
Ammonia-N	75	45	20
Nitrate + Nitrite N	0.5	0.2	0.1
Organic N	25	10	15
Total Kjeldahl N	100	60	30
Total Phosphorus	25	15	6

Ortho-Phosphorus	15	10	4
Organic-Phosphorus	10	5	2

The distribution between soluble and suspended matter is important in relation to the characterization of wastewater (see Table 4)

Table 4 Distribution of soluble and suspended material for medium concentrated municipal wastewaters (Henze *et al*. 2008)

Constituent		**Concentration, g/m^3**	
	Soluble	**Suspended**	**Total**
COD	100	60	30
BOD	140	210	350
Total N	0.5	0.2	0.1
Total P			

2.6 Inorganics and Metals

Wastewater is comprised of a variety of organic, inorganic and mineral matter some of which are dissolved while some is particulate. Some dissolved inorganic substances include: magnesium (Mg), potassium (K), sodium (Na), chloride (Cl-) and sulphates (SO_4^{2-}), which influences the performance of the wastewater treatment plants. These substances are required as trace elements for biotic growth (Wentzel and Ekama, 2006). Municipal wastewaters also are made of potentially poisonous metals and elements such as cadmium, lead, chrome, arsenic, zinc, copper, nickel, cobalt, mercury, boron, selenium, and fluorine. Greater fractions of these metals are in particulate form and usually adsorb to the sludge produced at the wastewater treatment plant (Sneyders *et al*. 1998). Most of the inorganics and metals are not the direct target for treatment, but they are contributors to the toxicity of wastewater, either in relation to the biological treatment processes or in relation to receiving waters (Henze *et al*. 2008). The typical values for metals and other inorganics are given in Table 5.

Table 5 Typical content of metals in municipal wastewater with minor contributions of industrial wastewater adopted from Henze *et al.* 2008

Metal		**Concentration, mg/m^3**	
	High	**Medium**	**Low**
Aluminium	1000	600	350
Cadmium	4	2	1
Chromium	40	25	10
Copper	100	70	30
Lead	80	60	25
Mercury	3	2	1
Nickel	40	25	10
Silver	10	7	10
Zinc	300	200	100
Sulphide	10000	500	100
Cyanide	50	30	20
Chloride	600 000	400 000	200

2.7 Biological Nutrient Removal

Biological nutrient removal (BNR) activated sludge system has become an established technology in wastewater treatment practice to control eutrophication. An improved understanding of nitrification, denitrification and excess biological phosphorus (P) removal (EBPR) has facilitated this advancement (Ekama and Wentzel 1999). Traditionally, the complexity associated with implementing BNR in WWW has been primarily in terms of balancing competing requirements for nitrogen and EBPR, particularly with respect to the use of influent COD as a carbon source for the microorganisms (Hu *et al.* 2012). In the BNR process, there is a requirement for S_S to achieve EBPR and denitrification. Primary sludge fermentation for production of volatile fatty acids (VFAs) is a widely used method to meet this requirement. A COD/P ratio of more than 40 to 50 g/g is reported as necessary to achieve less than 1 mg/L P in the effluent (Ydstebø et al. 2000).

During seasons of low temperatures, a BNR plant must operate at a high SRT to support sufficient nitrifier growth. High mixed liquor suspended solids (MLSS) and high hydraulic loading during rain and snow melt may result in overloading the clarifier and sludge washout. Reduced hydraulic retention time (HRT) in the reactor also results in insufficient time for completion of the biological reactions (Randall, *et al.* 1992).

Industrial effluent has a high concentration of nitrogen and a low C/N ratio. The main problems in maintaining high nitrification efficiency when treating low C/N wastewaters are changes in influent concentration and flow, which may also affect to the dissolved oxygen level in the reactor, and pH due to fluctuating industrial operations (Campos et al. 2007). This section will be reviewing the nutrients (N, COD and P) that are in the activated sludge process and their removal.

2.8 Total Nitrogen

The nitrogen that enters the wastewater system is found as NH_3, ammonium ions (NH_4^+), nitrites (NO_2^-) and nitrates (NO_3^-). Ammonia and ammonium ions are the most reduced forms of nitrogen and are often products of organic decomposition. Nitrite is, however, the intermediate, with nitrate being the end product of organic oxidation - nitrification (DWAF, 1996). Nitrogen removal is partially achieved by nitrification and completely removed by denitrification. Nitrification is performed by two groups of gram-negative, autotrophic bacteria. Further classification is based on a limited number of characteristics, including cell shape, internal membrane structures, flagella, inclusion bodies, salt tolerance, urease activity, and substrate affinity and inhibition (Prosser, 2005)

These bacteria are known as ammonia oxidizers (ANOs) and nitrite oxidizers (NNOs). The ANOs and NNOs transform ammonia to nitrite and nitrite to nitrate, respectively (Genestet *et al.* 1998). Henze *et al.* (2008) infers that these bacteria derive their carbon requirement (anabolism) from dissolved carbon dioxide (CO_2). They thereafter derive their energy requirement (catabolism) for biomass synthesis from oxidizing the ammonia nitrogen to nitrite and nitrate (see nitrogen cycle Figure 7) The ANOs and NNOs have much lower growth coefficients (1/5th) as compared to the heterotrophic bacteria (OHOs) which are also present in the activated sludge reactor for the breakdown of organic waste (COD).

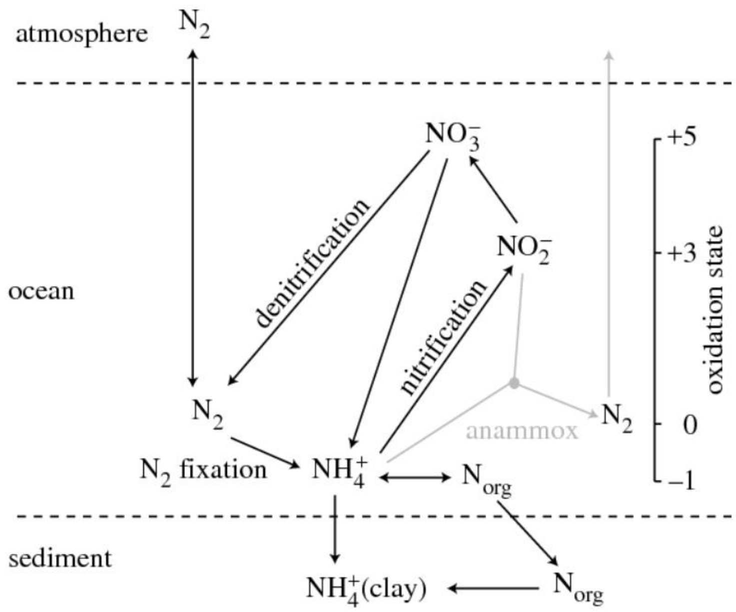


Figure 7: Nitrogen cycle (Falkowski, *et al.* 2008)

Nitrification redox reactions:

$$NH_4^+ + \frac{3}{2}O_2(ANOs) \rightarrow NO_2^- + H_2O \ + 2H^+ \qquad (3)$$

$$NO_2^- + \frac{1}{2}O_2(NNOs) \rightarrow NO_3^- \qquad (4)$$

The NH_3 and NH_4^+ that are found in wastewater are both commonly referred to as "ammonia nitrogen". These two forms can change very rapidly from one to the other depending on temperature and pH. Temperature has a major effect on the nitrification reaction rate, which approximately doubles for every 10 ^oC rise in temperature between 6 ^oC and 25 ^oC. A minimum dissolved oxygen level of 1 mg/l and pH of 7 - 8.5 is adequate for nitrification (Guyer, 2011). According to Henze *et al.* (2008) High dissolved oxygen concentrations, up to 33 mg O_2/L, do not affect nitrification rates significantly. However, low oxygen concentrations reduce nitrification rates. Henze *et al.* (2008) further noted that the effect of pH becomes noted as that the activities of both the hydrogen (H^+) and (OH^-) ions act as inhibitory when their respective concentrations increase are too high. The experimental investigations by Downing *et al.* (1964) showed that the nitrification rate can be formulated in terms of the Monod equation.

Henze *et al.* (2008) indicated that the Monod equation was applied to nitrification before it was applied to model the kinetics of organic material breakdown by heterotrophic bacteria. The successful application of the Monod equation to nitrification prompted Lawrence and McCarthy (1972) to apply it on activated sludge. Monod established that:

1. The mass of organisms generated is a fixed fraction of the mass of substrate (in this case the ammonia utilized)
2. The specific growth, i.e. the rate of growth per unit mass of organism per unit time, is related to the concentration of the substrate surrounding the organism.

From point (1):

$$M\Delta X_{BA} = Y_A M\Delta N_A \qquad (5)$$

Where:

$M\Delta X_{BA}$ = Mass of nitrifiers generated (mgVSS)

$M\Delta N_A$ = Mass of ammonia as N utilized (mgFSA-N)

Y_A = Nitrifier yield coeficient (mgVSS/mgN)

Taking the respective changes over time interval t and assuming that the changes are very small, the equation can be written as:

$$\frac{dX_{BA}}{dt} = Y_A[-\frac{dN_a}{dt}] \qquad \text{(mgANOVSS/L.d)} \qquad (6)$$

From (point (2) Monod developed the following relationship, known as the Monod equation:

$$\mu_A = \frac{\mu_{Am}N_a}{K_{nT}+N_a} \qquad \text{(mgVSS/mgVSS.d)} \qquad (7)$$

Where: μ_A = Specific growth rate at ammonia concentration (L/d) Na (mgANOVSS/mgANOVSS.d)

μ_{Am} = Maximum specific growth rate Na (mgANOVSS/mgANOVSS.d)

K_n = Half saturation constant, i.e. the concentration at which $\mu_A = ½\ \mu_{Am}$ (mgN/L)

N_a = Bulk liquid ammonia concentration (mgN/L)

The Monod constants μ_{Am} and K_n are also known as the affinity coefficients. K_n for the ANOs are sensitive to temperature and generally decreasing as temperature decreases. An additional T as a subscript refers to temperature in °C (Henze *et al.* 2008). The application of the Monod equation continues to eventually calculate the oxygen utilization rate associated with nitrification which is based on the stoichiometric requirement of 4.57 mg O_2/mgFSA-N nitrified to nitrate.

$$O_n = 4.57\frac{dN_a}{dt} = 4.57\frac{dN_n}{dt} \quad \text{(mg O}_2\text{/L.d)} \quad (8)$$

Henze *et al*. (2008) further infers that the application of the Monod growth kinetics to nitrification is one of the most successful applications of microbial kinetic research to wastewater treatment.

The removal of nitrogen is an essential process in the activated sludge system. Nitrogen as free ammonia is toxic to aquatic invertebrate and vertebrate species at concentration as low as 0.5 mg/l (Burghate & Ingole, 2013). When it is as ammonium ion or ammonia, it is an oxygen-consuming compound, which has a potential to deplete the dissolved oxygen in the receiving water. In all forms, nitrogen can be available as a nutrient to aquatic plants and consequently contribute to eutrophication (Ni, *et al*. 2016). Adonadaga, (2014) alluded to the fact that ammonia as the nitrate ion is a potential public health hazard in water consumed by infants and that thyroid dysfunction and formation of carcinogenic compounds have also been linked to nitrates in drinking water. It is for such reasons that Darvill WWW has to comply with the DWS maximum acceptable limits of final effluent discharge into river of 15 mg/L and 6 mg/L for NO^{3-} and NH_3 respectively.

Denitrification is the absolute expulsion of nitrogen from the wastewater system that is performed by heterotrophic anaerobic organisms (OHOs). Heterotrophic denitrification converts nitrate generated from autotrophic nitrification to nitrogen gas, thus removing nitrogen from wastewater (Ni, *et al*. 2016). Denitrification requires an organic carbon source and anoxic (absence of dissolved oxygen) conditions for efficient the removal of nitrogen. The dissolved oxygen concentrations of up to 0.5 mg/l, pH of 6.5 -7.5, 5 minutes of effluent aeration; and sludge recycle from 50 to 100 percent of average flow are sufficient for an effective denitrification process (Guyer, 2011). The OHOs, use nitrate nitrogen as a terminal electron acceptor and the organic carbon source (methanol, acetate, ethanol, lactate and glucose) being the electron donors, (Xu, 2018).

The reaction is as below:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \quad (9)$$

Denitrification is the only nitrogen biotransformation process that removes nitrogen from the eco system and is irreversible as the nitrogen becomes a gas and is released into the atmosphere (Moura *et al*. 2017). The current study looks at the characterization of COD. Bashide, (2015) inferred that nitrogen can also be fractionated into organic and inorganic nitrogen however there generally is no need to fractionate nitrogen as much as for organic matter (COD). This is because the larger part of nitrogen in wastewater is usually present as ammonium, which has no relationship with organic matter while the remaining part is coupled with an organic component.

In municipal waters, nitrogen exists mainly as organic nitrogen and ammonia nitrogen. Typically, it can be distributed as 40% organic nitrogen and 60% ammonia nitrogen. The sum of the two is what is referred to as the "Total Kjeldahl Nitrogen" (TKN). Total nitrogen (TN) in wastewater can be characterized as:

$$\mathrm{TN} = \mathrm{TKN} + \mathrm{SNOX} = \mathrm{XTKN} + \mathrm{SNHX} + \mathrm{SNOX} \tag{10}$$

Where: XTKN = Total Organic Nitrogen

SNOX = Oxidized Nitrogen

TKN = Influent Total Kjeldah Nitrogen

SNHX = Ammonia Nitrogen

If the BNR processes are working optimally, the final effluent should have nitrate levels of less than 1.5 mg/L as per standard regulatory requirements. Monitoring of the nitrate levels in the final effluent assists in determining the efficiency of the BNR process.

2.9 Phosphorus Removal

Domestic wastewater is relatively rich in phosphorus compounds. Research has shown that nitrogen and phosphorus are essential for the growth of algae and cyanobacteria and that limitation in amounts of these elements is usually a factor that controls their rate of growth. The Department of Water Affairs (DWAF) which is the custodian of all water resources in South Africa has regulated that wastewater final effluent shall not contain soluble orthophosphate (as P) in a higher concentration than 1.0 mg/L. Phosphates that respond to colorimetric test without preliminary hydrolysis or oxidative digestion of the sample are termed reactive phosphorus. The form of phosphate determined by this method is termed soluble reactive phosphate (SRP). Phosphorus does not need to be fractionated in as much detail as organic matter because most it occurs in orthophosphate form (Bashide, 2015). The total phosphorus (CTP) is fractionated in two categories such as soluble and particulate.

$$CTP = XTP + STP \tag{11}$$

Where: XTP = Particulate Total Phosphorus

STP = Soluble Total Phosphorus

Particulate total phosphorus includes organic and inorganic phosphorus. The influent phosphorus can be determined from the Total Phosphorus (CTP) test, and Orthophosphate test. The total phosphorus test (CTP) measures soluble orthophosphate, condensed orthophosphates and the phosphorus bound to organic compounds. The orthophosphate test measures the orthophosphates and a small fraction of some condensed phosphates may be included. The difference in the P concentration between TP and orthophosphate test gives the organic P concentration (Mhlanga, 2008). In process operations, it is essential to monitor the mixed liquor for soluble orthophosphates to control the processes related to EBPR and ensure legal compliance. The EBPR process is phenomenon in activated sludge system where wastewater biomass removes phosphorus beyond its anabolic requirement by accumulating intracellular polyphosphate reserves.

In addition to P removal for cell synthesis, further P removal may take place in chemical precipitation either by chemicals present in the wastewater or added to the chemical system.

2.10 Chemical Oxygen Demand Removal

Chemical oxygen demand (COD) is the measure of the amount of oxygen required to oxidise the (oxidisable fraction of) organic carbon contained in the effluent. This is measured in mgO_2/L and is sometimes referred to as the organic strength of the wastewater (Mnguni, 2010). According to Henze *et al*. (2008); the theoretical COD of a given substance can be calculated from an oxidation equation (see example below **Error! Reference source not found.**). Ethanol is calculated based on the following equation:

$$C_2H_6O + 3O_2 \rightarrow 2CO_2 + 3H_2O \tag{12}$$

Which can be interpreted as, 46 g of ethanol requires 96 g of oxygen for the full oxidation to carbon dioxide and water. The theoretical COD of ethanol is therefore 96/46 = 2.09. The adoption of secondary and tertiary treatment processes to remove nutrients and refractory organic matter requires a classification of the subordinate fractions of wastewater organic matter (Choi *et al*. 2017). Fractionation of COD into soluble and particulate constituents that are seen in Table 6 is now used to optimize the performance of both existing and proposed new biological treatment plants designed to achieve nutrient removal (Tchobanoglous, 2003). In another sense, Borg and Ekström (2015) discussed that the fractions are viewed as the biodegradable organic fraction, which is transformed during the biological process and another inert (unbiodegradable) organic fraction (see breakdown on Figure 8**).**

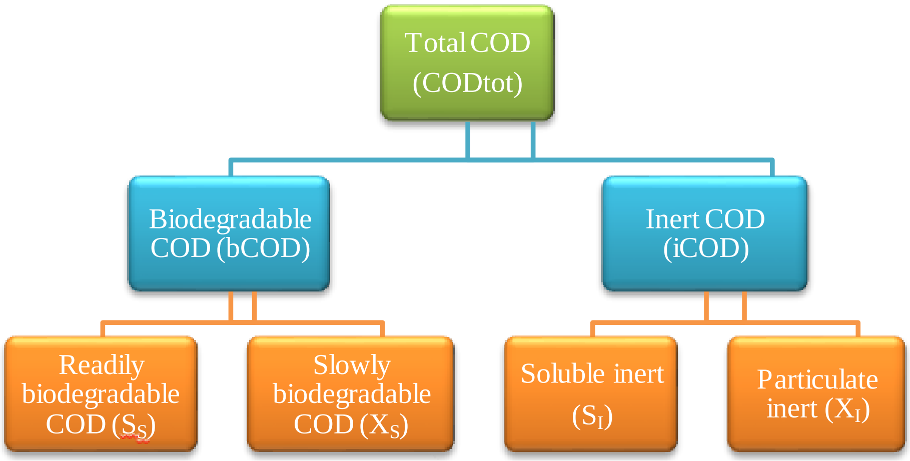


Figure 8: COD fractions

Myszograj *et al*. (2017) states that this fractionation allows for subsequent calculations of the content of individual forms of COD present in wastewater. The constituent elements of total COD are presented by:

$$COD_{Tot} = S_S + S_I + X_S + X_I + X_H + X_A + X_P \qquad gO_2/m^3 \qquad (13)$$

Where: S_S - Soluble readily biodegradable substrates, gO_2/m^3

S_I - inert soluble organic material, gO_2/m^3

X_S - particulate slowly biodegradable substrates, gO_2/m^3

X_I - inert particulate organic material, gO_2/m^3

X_H - heterotrophic organisms, gO_2/m^3

X_A - autotrophic nitrifying organisms, gO_2/m^3

X_P - decay products, gO_2/m^3

Unless the biomass fraction is not included, this model is simplified to the form:

$$COD_{Tot} = S_S + S_I + X_S + X_I \qquad gO_2/m^3 \qquad (14)$$

The fractions of equation 14 is only here discussed. The biodegradability and conversion of COD can be seen in Table 4.

Table 6 Degradability of medium concentrated municipal wastewater influent adopted from Henze *et al.* (2008)

Constituent			**Concentration, g/m³**		
	Biodegradable	**% Biodegradable**	**Inert**	**% Inert**	**Total g/m³**
COD Total	570	**76**	180	**24**	750
COD soluble	270	47	30	17	300
COD particulate	300	53	150	83	450
Total %		100		100	

2.10.1 Readily biodegradable COD

Readily biodegradable COD (S_S) is the most easily accessible COD fraction. S_S is assimilated and mineralized by OHOs. Henze *et al.* (2008) explains that it comprises of small simple dissolved organic compounds that can pass directly through the cell walls and into the OHOs. This fraction is measured in order to determine:

- The easily available organic carbon sources in wastewater, both during design and operation,
- The ratio of each COD fraction in wastewater for the purpose of mathematical modelling of the activated sludge process,
- The kinetic parameters required for efficient operation of the bioreactor (Myszograj *et al*. 2017).

The amount of S_S can be a determining factor for anaerobic and anoxic reactor volumes in model-based design because the phosphate release and denitrification processes are very sensitive to the easily accessible substrate fractions. Scientific data suggests that the readily biodegradable substrate ranges between 3–47 % in raw wastewater and 14–57 % in the settled wastewater (Pasztor *et.al*. 2008). The S_S is degraded via different mechanisms by the OHOS (Henze *et. al*. 2008).

In ASM2 and ASM2d models, this component is divided into volatile fatty acids (VFA = acetic acid, propionic acid, butyric acid, etc.) and non-VFA components (alcohols, lower amino acids, simple carbohydrates) (Henze, 1992). The VFA fraction ranges between 0–8.8 % in raw wastewater and 0–16 % in primary wastewater (Pasztor *et.al*. 2008).

2.10.2 Slowly biodegradable COD

The slowly biodegradable COD (X_S) component comprises complex organic compounds which need to be hydrolyzed by extra cellular enzymes of bacteria prior to utilization. It is also generated by the biomass itself through the death and lysis of organism mass (also known as endogenous mass loss/respiration) (Henze *et al*. 2008).

It is in the form of colloidal and suspended COD. It is for this reason that it is not separated from influent samples solely by physical separation. From the design point of view, this fraction usually has the highest oxygen demand and therefore it greatly influences the air flow required for aeration tank. This component ranges between 28–74 % in raw sewage and 24.5–65 % in primary wastewater (Pasztor *et.al*. 2008). It is degraded via different mechanisms by the OHOS (Henze *et al*. 2008).

2.10.3 Soluble Inert COD

Soluble inert COD fraction (S_I) cannot be further biologically degraded in treatment plants and therefore the influent S_I COD leaves the plant without any significant change in its concentration. Because of the soluble inert fraction of sewage, WWTPs treating strong municipal wastewaters or septic tank effluents (CODTOT > 1500 mg/L) with even perfect carbon oxidation may find it hard to meet the strict COD effluent standards. S_I fraction, relative to the CODTOT, is in the range of 2–15 % in raw wastewater and 3–14.3 % in primary sewage (Henze, 1992; Xu and Hultman, 1996; Satoh *et al*. 2000).

2.10.4 Particulate Inert COD

Particulate unbiodegradable (X_I) component is not degraded biologically during the treatment process and hence it can be removed only by clarification. From designers' point of view, this fraction significantly influences the quantity of primary and secondary sludge and therefore it determines the required dewatering and sludge treating capacity. Raw wastewater and primary wastewater contain 8-39 % and 4–20 % X_I, respectively (Ekama *et al*. 1986).

The main objectives of wastewater treatment include**:**

- The reduction of organic matter (COD) in wastewater to a level which no longer sustains heterotrophic growth and thereby avoid de-oxygenation of the receiving fresh water body
- The oxidation of ammonia to nitrate to reduce its toxicity and de-oxygenation effects
- The reduction eutrophic substances i.e. ammonia, nitrates and phosphates

Wastewater may be classified into the following components:

• Domestic or sanitary wastewater - Wastewater discharged from residences, commercial

(e.g., banks, restaurants, retail stores), and institutional facilities (e.g., schools and hospitals).

• Industrial wastewater - Wastewater discharged from industries (e.g., manufacturing and chemical processes).

• Infiltration and inflow -Water that enters the sewer system from groundwater infiltration and storm water that enters from roof drains, foundation drains, and submerged manholes.

• Storm water - Runoff from rainfall and snow melt (Davis, 2010).

2.11 Activated sludge modelling

Mathematical modelling is the representation of the interactions in the system by a set of equations. Various types of mathematical models have been developed and designed for various purposes. These models have been classified for their specific purpose under 3 general headings i.e. computer design aid, optimization and simulation (James, 1984).

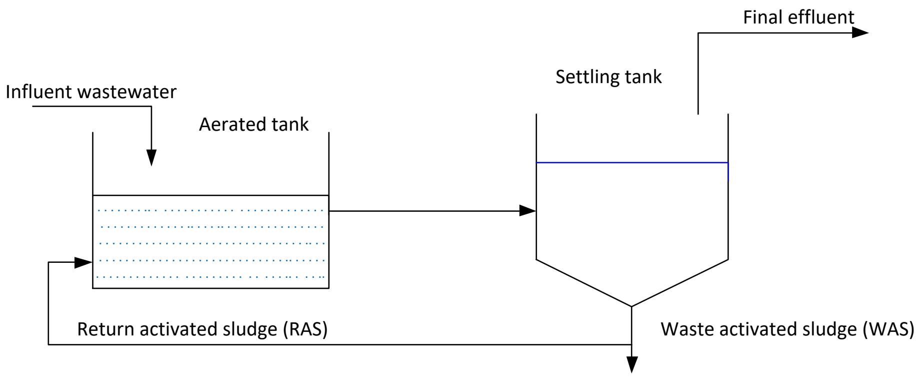


Figure 9: Basic diagrammatic representation of the activated sludge process

Mathematical modelling of activated sludge systems has become a widely accepted tool for research, plant design and optimisation and training of process engineers and plant operators. The mathematical models are only useful if the model predictions are reliable (Mhlanga, 2016). Different simulators for wastewater treatment plants have been developed over the years and are commercially available. We use models to understand and describe our perception of the reality surrounding us (Bolmestedt, 2000). In terms of the framework of these models, it is essential to fractionate the influent COD into a number of fractions (Figure 8). The fractionation is bio-kinetic and not a physical separation. For the activated sludge processes, a simulation model is a necessity to avoid numerous tests which lead to operating conditions that may be difficult or even impossible to realize. In terms of automatic control, a simple model, non-representative of all the dynamics of the process, may be sufficient. However, the more important the domain of validity is, the more efficient the control law will be (Cadet, 2014).

Modellers from around the world have approached the task of modelling in different ways over the years, resulting in numerous guiding principles being developed. The range of approaches and insufficient documentation of methods in some cases makes it challenging to carry out quality comparisons and assessments of simulation results from different modelling projects (Mhlanga, 2016). The International Water Association (IWA) responded to this challenge by forming a task group, the Good Modelling Practice (GMP) Task Group (Rieger *et al.* 2012). The group reviewed the various modelling protocols in an effort to form an internationally recognised standard, the GMP Unified Protocol.

Mhlanga, (2016) discussed the protocols that were considered by the GMP Task Group which included the STOWA (Hulsbeek *et al*. 2002; Roeleveld and van Loosdrecht, 2002), Water Environment Research Foundation (WERF) (Melcer *et al*. 2003), BIOMATH (Vanrolleghem *et al,* 2003) and HSG (*Langergraber et al.* 2004) protocols. These protocols were being compared by different researchers including Sin *et al.* (2005) and Corominas (2006). The modelling approaches of all protocols considered by the GMP Task Group were similar and proposed for the unified protocol as follows:

- Project definition;
- Data collection and reconciliation;
- Plant model set-up;
- Calibration and validation; and
- Simulation and results interpretation.

2.11.1 The Activated sludge models

The activated sludge model No. 1 (ASM1), established by the International Water Association, is intended to promote and facilitate the practical methods of designing and operating the biological treatment systems for wastewater systems. The ASM1 model is able to represent the behaviour of wastewater influent, loaded in carbonaceous and nitrogenous compounds (Mhlanga, 2016). When the ASM1 was published, EBPR was in use to some degree in a number of WWWs. During the mid-1980s to the mid-1990s the use of EBPR processes increased and so was its understanding. Therefore, in 1995, the ASM2 was published. The ASM2 included nitrogen-removal and EBPR processes. In 1999, the ASM2 model was extended to ASM2d which included denitrifying phosphorus-accumulating organisms (PAOs) after it was observed that denitrifying PAOs were required for the simulation of many results from research and practice (Henze *et al.* 2000). In 1998, ASM3 was developed. The ASM3 was developed for biological nitrogen removal like ASM1. It was developed to correct defects that have been identified by users of ASM1 (Gujer *et al,* 1999). The main difference between ASM1 and ASM3 is that the latter includes storage polymers in the utilisation of substrate by heterotrophic biomass in the activated sludge. The ASM3 model presented storage of organic substrates as a new process. It replaced the death-decay process for heterotrophic organisms with an endogenous respiration process (Gujer *et al.* 1999).

The most important measure by which the model can be judged is by its ability to predict real time and space-time dependant changes and he requirement of the electron acceptor. It is because of this that the substrate i.e. COD is fractionated (Henze *et al.* 2000). For practical reasons, activated sludge models (ASMs), irrespective of whether they are complex dynamic or simple steady state models, consider bacteria not as individual organisms. In ASMs, the mass of bacteria cells (X) is modelled as a major organic mass fraction of volatile suspended solids (VSS) and grouped with respect to their metabolism and function within the activated sludge process using the Monod equation (refer to equation 7). ASMs transfer growth related characteristic properties of the bacterial cell to the mass fraction of the particular organism group (Friedrich *et al.* 2015).

2.12 Bacterial growth and biomass Yield

Prior to obtaining the S_S fraction, the biomass yield must be determined (Henze *et al*. 2000). In biological treatment processes, the cell growth and respiration occurs concurrently with the oxidation of organic and inorganic compounds like carbohydrates, proteins and lipids to the end products carbon dioxide (CO_2) and water (H_2O). The microorganisms are able to produce energy by the process of respiration. Energy becomes available to the microorganisms through a series of internal oxidation-reduction reactions. This involves electron and proton transfers from an organic substrate through a number of intermediate enzyme complexes to the final electron acceptor (Naidoo, 1999). The ratio of amount of biomass produced to amount of substrate consumed is defined as the biomass yield, and typically defined relative to the electron donor used (Tchobanoglous *et al.* 2000). In this section we will be discussing COD as a substrate and the biomass yield is known as the heterotrophic yield (Y_H). According to Henze *et al.* (2000), Y_H can be estimated by observing the mass of cell material formed during the removal of the soluble substrate (S_S). An aliquot of wastewater should be settled and filtered to remove the particulate material. The filtrate which contains only the soluble substrate should be seeded lightly with acclimated biomass from the one of the completely mixed reactors. Aliquots should be removed periodically and both soluble COD and total COD are determined.

$$Cell\ COD = Total\ COD - Soluble\ COD \tag{15}$$

$$Y_H = \frac{\Delta\ cell\ COD}{\Delta soluble\ COD} \tag{16}$$

Where: Y_H - heterotrophic yield

2.13 Hydraulic Retention time

According to the activated sludge theory, the volume of the biological reactor per unit volume of influent flow is known as the nominal hydraulic retention time which is:

$$HRT_n = \frac{V_p}{Q_i} \tag{17}$$

Where: HTR_n - Nominal hydraulic retention time

V_p – Volume of the biological reactor (L/d)

Q_i – Daily average influent flowrate

The distinction between biodegradable and unbiodegradable is governed by the biomass in the system and the length of time this biomass has to degrade to organics. It has been observed that the difference in the soluble effluent COD concentration from a short period (2 - 3 hrs) to a very long (18 - 24hr) hydraulic retention time system is very small only 10 -20 mg COD/L. this means that slowly biodegradable soluble organics seem to be very low in concentration in normal municipal wastewater. Therefore, it is reasonable to accept that the soluble organics in municipal wastewater comprise of two groups – the biodegradable which is all almost readily biodegradable and the unbiodegradable. This means that a very short hydraulic time of a few hours, utilization of the biodegradable COD is complete leaving only the soluble unbiodegradable organics in the effluent.

2.14 WEST

Darvill wastewater works uses the WEST (Worldwide Engine for Simulation, Training and automation) simulator for the prediction and optimization of the performance of the wastewater treatment process. WEST is a software package that is managed by MSL (Model Specification Language) and is used for simulations of environmental systems and processes that occur in wastewater treatment plants. It will be used in this study to develop and test the Darvill model. This section will discuss aspects of the WEST model and its operation.

2.14.1 Background

Mhlanga (2008) suggested that one of the main reasons for the use of the WEST simulator is the relative ease with which models can be modified or extended to accommodate new components like industrial wastewater components. WEST consists of four different subprograms where the user accesses the different utilities in order to set up a configuration for a system, launch an experiment and manage the project of concern. (Mhlanga, 2008).

The subprograms are:

- WEST manager

- WEST configuration builder
- WEST experimental environment
- WEST model editor

WEST manager: The WEST manager gives the user an overview of the different projects created and allows new projects to be created. For each created project, a list of the different configuration and or experiments assigned to that project can be viewed in the WEST manger (WEST Tutorial, 2004).

Simulator Company Web address: WEST Hemmis www.hemmis.com/product/west; Efor DHI www.efor.dk/efor/; Simba IFAK www.ifak.fgh.de/regelung; JASS Uppsala University www.it.uu.se/research/project/jass/

The tools are:

- Block editor

The block editor is an application that allows for the creating and managing icon and palette libraries for use with the WEST software. An icon library is a collection of icons that can be used to represent blocks in a WEST layout. The palette library is a collection of pallets, where each palette contains references to a set of icons. The block editor allows for the opening a most one icon library and one palette library simultaneously at any point in time.

- Model editor

The model editor is an application that allows for managing the WEST Model library.

- Unit editor

The Unit Editor is a small application that allows for visualizing and managing the WEST unit table. The WEST unit table is used to perform unit conversion within the WEST software, and the Unit Editor allows for adding, removing and modifying entries in this table.

2.14.2 WEST Operation

WEST solves coupled Ordinary Differential Equations (ODEs). The steady-state solutions are ODEs integrated over long periods of time with constant inputs (Brouckaert and Brouckaert, 2018). The simplest version of an ODE is one of a single variable with constant coefficients:

$$\frac{dy(t)}{dt} = Ay(t) + Bu(t) \tag{18}$$

Where A and B are constants. $y(t)$ is called the *state variable* and $u(t)$ is the *input variable*.

For this simple case, the solution to the equations is:

$$y(t) = e^{At}y(0) + e^{At}\int_0^t Be^{-A\theta}\,u(\theta)d\theta \tag{19}$$

So, in order to obtain a solution, we need to know the *initial value* $y(0)$ at $t = 0$ and the input as a function of time over the whole time interval from 0 to t.

In the simple ODE example, *y(t)* is the **derived variable**, *u(t)* is the **input variable**, A and *B* are **parameters**. Other variables related to *y(t)* , for example $z(t) = y^2/2$, are called **algebraic variables.**

2.15 Parameters and variables

Parameters in WEST remain constant during a simulation run; though can be varied between runs in a virtual experiment. Variables are quantities that change during a simulation run. Manipulated variables transmit information between unit models. In addition to input variables, WEST have two basic types of variables: derived variables that are determined by integration of the ODEs; and algebraic variables that are calculated from the derived variables by algebraic equations at each time step.

2.15.1 Initial Values

The process model consists of multiple state variables, which are typically the masses of each component in each process unit. Every derived state variable requires an initial value. The solution for an ODE over a time interval requires that the initial variables and input variables for the whole time interval are known. However, for wastewater treatment plants the initial condition is strictly the state when the process first started up. It is impractical to simulate from the beginning therefore a method of estimating initial conditions at the start of the time interval of interest is required. This is the primary motivation for the steady-state solution on WEST. If the equations are stable, the influence of errors in the initial conditions will be eliminated out as the simulation precedes hence a steady-state solution with constant average inputs often provides an acceptable estimate of initial conditions when simulating a running plant (Brouckaert and Brouckaert, 2018). The aerobic model is generally stable, and the steady-state solution can still converge from an extremely poor estimate of the initial conditions. However, the anaerobic model has regions of instability, therefore is more difficult to initialize. This is largely associated with pH inhibition – when the pH drops below 6.5, methanogenesis step in anaerobic digestion is inhibited. This results to a state that the model cannot recover.

2.15.2 Process unit models

The block library contains the set of models in various categories. The model library is designed for the unit models to be as far as possible independent of the models of chemical transformation that occur within them.

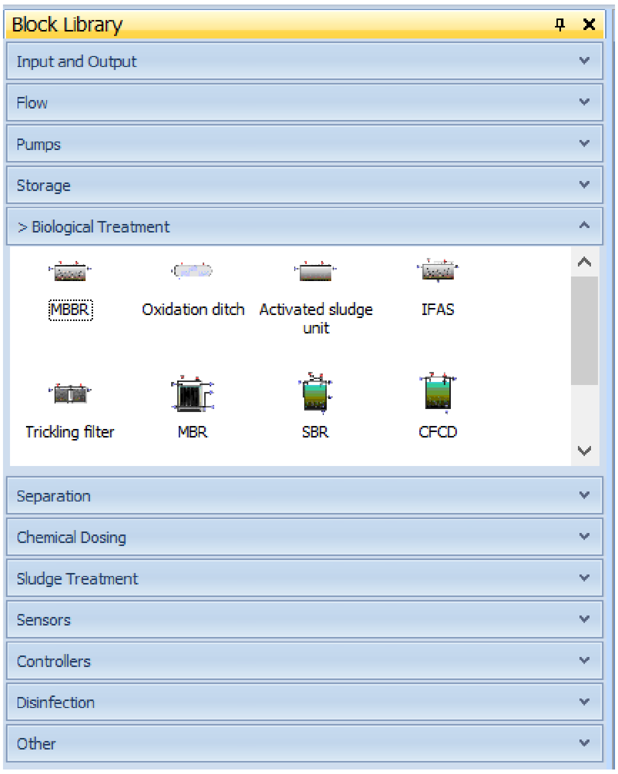


Figure 10: WEST Model block library

A chemistry model is referred to as a Category in WEST i.e. ASM1, ASM2d, etc. The category is defined by a set of model components, and a set of reactions which they take part in. There is another structure called an Instance, which is a category with extra structural parameters which cannot be varied. Every plant model has to work within an Instance.

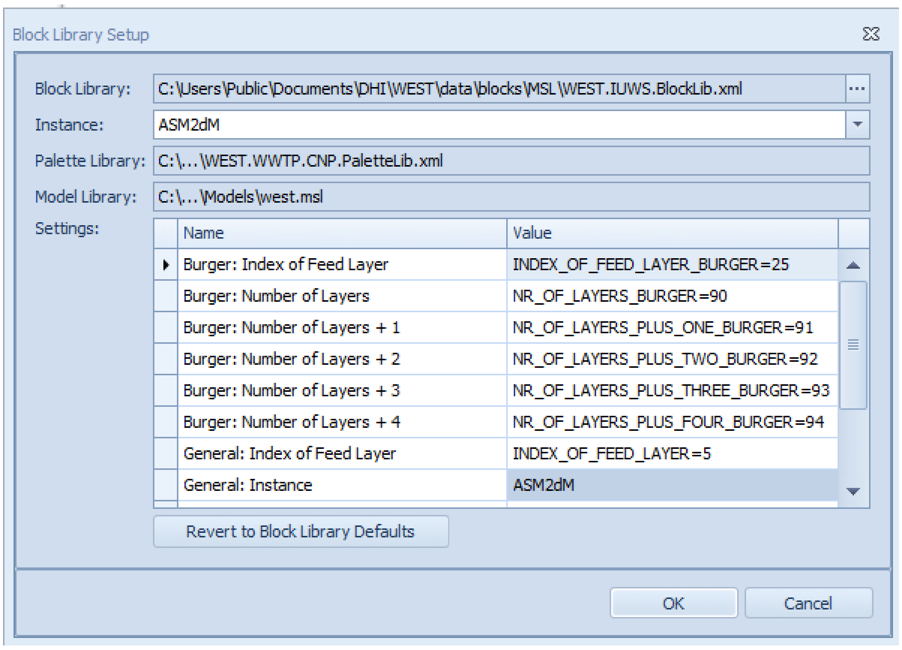


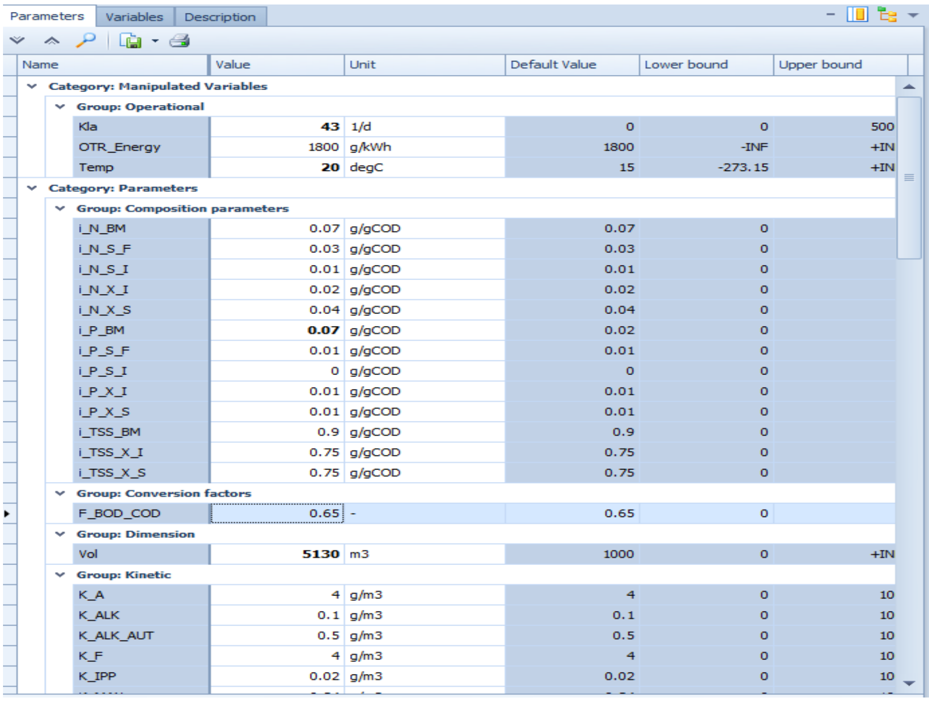
Figure 11: Block Library Instance setup

2.15.3 Components

Components are the chemical entities that are represented in a model. In the software, they are a numbered list of names, grouped according to the physical nature of the substances they represent. These classifications determine aspects of how they are handled in the unit models – e.g. particulates settle in the clarifier models, gases evolve into a digester headspace. The different models have different components which allow them to represent particular processes of interest (Brouckaert and Brouckaert, 2018).

2.15.4 Reactions

The other major aspect of the chemistry model (Category, Instance) is the list of reactions that the components take part in.



Parameters | Variables | Description

Name	Value	Unit	Default Value	Lower bound	Upper bound
Category: Manipulated Variables					
Group: Operational					
Kla	43	1/d	0	0	500
OTR_Energy	1800	g/kWh	1800	-INF	+IN
Temp	20	degC	15	-273.15	+IN
Category: Parameters					
Group: Composition parameters					
i_N_BM	0.07	g/gCOD	0.07	0	
i_N_S_F	0.03	g/gCOD	0.03	0	
i_N_S_I	0.01	g/gCOD	0.01	0	
i_N_X_I	0.02	g/gCOD	0.02	0	
i_N_X_S	0.04	g/gCOD	0.04	0	
i_P_BM	0.07	g/gCOD	0.02	0	
i_P_S_F	0.01	g/gCOD	0.01	0	
i_P_S_I	0	g/gCOD	0	0	
i_P_X_I	0.01	g/gCOD	0.01	0	
i_P_X_S	0.01	g/gCOD	0.01	0	
i_TSS_BM	0.9	g/gCOD	0.9	0	
i_TSS_X_I	0.75	g/gCOD	0.75	0	
i_TSS_X_S	0.75	g/gCOD	0.75	0	
Group: Conversion factors					
F_BOD_COD	0.65	-	0.65	0	
Group: Dimension					
Vol	5130	m3	1000	0	+IN
Group: Kinetic					
K_A	4	g/m3	4	0	10
K_ALK	0.1	g/m3	0.1	0	10
K_ALK_AUT	0.5	g/m3	0.5	0	10
K_F	4	g/m3	4	0	10
K_IPP	0.02	g/m3	0.02	0	10

Figure 12: List of reactions in the chemistry model

It can be seen that five of these reactions involve the growth or death of a micro-organism group. Each group grows on a particular component as substrate. Each reaction has stoichiometric coefficients for components (parameters) and a rate expression (a variable).

A typical biological growth rate expression:

$$GrowthRate = \mu_{max} \left(\frac{C_S}{K_S + C_S} \cdot I_T \cdot I_C\right) Y \cdot C_{org} \tag{20}$$

The variables are: C_S = Concentration of the substrate

C_{org} = Concentration of the organisms that grow on the particular substrate

$I_T \cdot I_C$ = Correction factors related to temperature and concentration

The parameters are:

μ_{max} = Maximum specific growth rate of the organisms

Y = Reaction yield (i.e. organisms produced per substrate consumed)

K_S = Monod half-saturation constant. Each inhibition term involves some additional parameters. There is a set of such parameters for each growth process. There is a biomass death rate associated with each micro-organism of the form.

$$DeathRate = b_{org} \cdot C_{org} \quad (21)$$

In this case the parameter is b_{org} (WEST Models guide).

2.15.5 Activated sludge tank with constant volume

The model describes an ideally mixed, activated sludge tank with constant volume:

$$Q_{in} = Q_{out} \quad (22)$$

The specific instance defines the conversion model that is in place and therefore the processes taking place inside the tank. Energy consumption for aeration and mixing is estimated as follows:

$$P_{Aer} = \frac{S_{O,sat} \times K_{La} \times V}{24 \times OTR} \quad (23)$$

$$P_{Mix} = \frac{ME \times V}{24} \quad (24)$$

Where: SO,sat = oxygen concentration at saturation (g/m^3)

KLa = oxygen transfer coefficient (L/d)

OTR = oxygen Transfer Rate (g/kWh)

ME = mixing energy per unit volume (kWh/m^3/d) respectively

V = tank volume (m^3)

Table 7 Parameters of the category specific conversion model

Name	Description	Value	Units
Vol	Volume of the tank	1000.0	m^3
OTR_Energy	Oxygen Transfer Rate per unit energy inputted	1800.0	g/kWh
ME_unit	Mixing energy per unit volume	0.005	kWh/m^3/d
Kla_Min	Lowest kLa that ensures adequate mixing	20.0	L/d
Mixing_When_Aerated	Mixing is guaranteed while aerating? (0=yes, 1=no)	0	---

Parameters of the Category-specific Conversion Model

Table 8 State variables

Name	Description	Units
C	Concentration of the state components (vector)	g/m^3
V	Volume of the tank	m^3
Q_In	Influent flow rate	m^3/d
Kla_Actual	Oxygen Transfer Coefficient	L/d
Temp_Actual	Temperature	*°C*

State variables of the Category-specific Conversion Model

Table 9 Derived state variables

Name	**Description**	**Units**
M	Mass of the state components (vector)	---

Table 10 Interface variables

Name	**Terminal**	**Description**	**Value**	**Units**
Inflow	in_1	Inflow vector	---	n/a
Outflow	out_1	Outflow vector	---	n/a
AerationEnergy	out_2	Energy for aeration	---	kWh
AerationPower	out_2	Power consumption for aeration	---	W
MixingEnergy	out_2	Energy for mixing	---	kWh
MixingPower	out_2	Power consumption for mixing	---	W
Temp	in_2	Temperature	15.0	°C
Kla	in_2	Oxygen Transfer Coefficient	0	L/d

Interface variables (sensors) of the Category-specific Conversion Model

2.16 The WEST Model

A mathematical model in WEST which is the model underlying the plant layout, consists of three parts: The Input Model, the Plant Model and the Output Model (Figure 13).The input to the overall model is the form of an input file; optionally an output file can be present too. The Input and Output model can get implemented by going through the influent and effluent generator tool (WEST User guide). A WEST model may contain any number of influent and effluent data sets (i.e. wastewater output blocks: at least one of each category) ad any number of input data sets (i.e. measurement data).

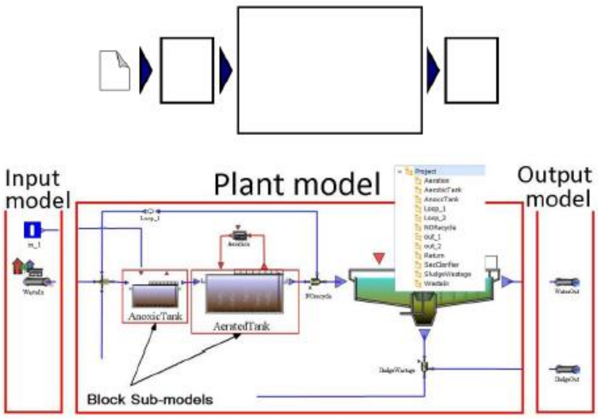


Figure 13: The overall concept of the WEST Model

2.16.1 The Input Model

A WEST Plant Model requires input data specified as model components (typically a vector containing the components as defined by the active model category, e.g. ASM1Temp and fluxes (g/d). In practice however data is generally available in terms of COD, BOD, TSS and TKN, etc. and measurements are expressed in flow rates of (m^3. d) and concentrations (mg/L). Therefore, every input (sub) Model (i.e. the model behind an input block) consists of the following elements:

- **A Fractionation Model:**

Which expresses how the measured quantities are broken down into the model components;

- **A Conversion Model:**

Which translates the flow rate and concentration units into flux (load) units; and

- **Two input files:**

Two input files is a dynamic time series that is generated by combining individual data sources and the corresponding steady-state time series.

2.16.2 The Plant Model

The plant layout is a graphical representation of the plant. WEST automatically translates it in the corresponding plant model by combining the individual sub-models that describe every block.

2.16.3 Flattened Model

Once the plant layout is complete, it is model generated, and ready for simulation while the model compiler creates an executable model that ensures optimal simulation speed.

The following steps are carried out to generate the executable model:

- Translation of the plant layout to the model in MSL
- Flattening and optimization of the MSL model
- Translation of the MSL Model to C
- Compilation of the C Code

2.16.4 Parameters

In the WEST Model Library, a functional distinction is made between parameters and manipulated variables. In essence, both are model parameters but manipulated variables can be altered during the simulation and therefore behave as any other model variables. Parameters can only be altered prior to starting a simulation (their value is constant during a simulation).

2.16.5 Top-level parameters

In WEST, it is possible to group he parameters that are common to different blocks into a set of corresponding parameters at the level of the overall plant model as opposed to the block level. These sets of parameters are called Top-level parameters. A value assigned to Top-level parameters is automatically transferred to the sub-model parameters that are linked to hat parameter.

2.16.6 The Output Model

A WEST plant model generates output data in the same format as input data i.e. specified as model components (ASM1 Temp variables) and expressed in fluxes (g/d); However effluent data is compared against legislation standards to evaluate the performance of the plant. It is therefore convenient that they are expressed in terms of concentrations (mg/l) of COD, BOD, TSS, TKN, etc. Therefore, every output (sub) model (i.e. the model behind an output block) consists of the following elements:

- **A De-Fractionation Model**

It expresses how the model components are combined into the measured quantities in the output/final effluent

- **A Conversion Model**

It translates the flux (load) units into flow rate units and concentration units.

2.16.7 Experiment types

WEST has six types of virtual experiments. The term virtual experiment is used to refer to a computational procedure that is performed on the executable version of a plant model. The most straightforward virtual experiment type is simulation; however, several other more advanced procedures are also available in WEST. When a new project is created, two separate simulation experiments are created in the background. The first is intended for simulating the model under steady-state conditions and the other is intended to perform dynamic simulations. Both experiments share the same model, but can be configured separately in terms of parameter values, initial conditions, solver settings, etc. the steady state and dynamic experiments automatically compute a set of predefined criteria on a selected number of time series. Also these time serious criteria can be configured separately for both simulation experiments. A project can be upgraded from its original to include one of the following advanced computational procedures:

- **Scenario Analysis (SA)**

It runs a series of simulations experiments using different values for a selected set of parameters.

- **Uncertainty Analysis (UA)**

It runs a series of simulation experiments using parameter vectors sampled through Latin Hypercube sampling and computes various types of aggregated data.

- **Global Sensitivity Analysis (GSA)**

It runs a series of simulation experiments using parameter vectors sampled through Latin Hypercube sampling and performs linear regression.

- **Local Sensitivity Analysis (LSA)**

It uses finite difference approach to study the effect of small changes in parameters to output variables.

- **Parameter Estimation (PE)**

It minimizes an objective by running simulations using different values for a selected set of parameters, typically to match simulated to measured data. When a project is upgraded to support one of the above virtual experiment types, a choice is to be made whether to base the upgrade project on the original steady-state or the original dynamic simulation experiment. As of that moment, the settings of the selected simulation experiment are effecting the newly created virtual experiment. A project can also be downgraded back to the original simulation with the steady-state and dynamic simulation experiment not including any of the advanced computational procedures listed above.

2.16.8 Steady-state and Dynamic Experiment

For any layout composed in WEST, two simulation experiments will be automatically created. One is named Steady state and is intended for studying the system under steady state conditions. The other experiment is named Dynamic and is intended to study the system under dynamic conditions. These experiments use separate influent files, which are generated by the Influent Tool. Also, the values of derived variables obtained at the end of the steady state simulation, will be used as initial conditions to the dynamic simulation (Wu, 2015).

2.16.9 Scenario Analysis experiment

The Scenario Analysis (SA) Experiment will run a series of simulations on the basis of the same model, using different values for a selected set of parameters for each simulation run. These parameters can be either of the following model items:

- Manipulated variables
- Parameters
- Derived variables (more especially: initial conditions)
- Input variables

2.16.10 Parameter Estimation Experiment

The Parameter Estimation (PE) Experiment will run a series of experiments on the basis of the same model, using automatically generated values for a selected set of parameters for each simulation run, in order to minimize an objective function. These parameters can be either of the following model items:

- Manipulated variables
- Parameters
- Derived variables (more especially: initial conditions)
- Input variables

On the basis of the simulated time series, time series criteria can be computed on a selected set of variables, through the aggregation of data over time. The aggregation of these time series criteria is the objective function that is used during the optimization process.

2.17 Wastewater Legislation and Environmental Significance

All industries have a duty of care to the communities, and to the environment in general. Caution is required in both the design and operation of WWW to ensure that legal standards are met (Sinnott, 2005). This section will look at Wastewater legislation; Environmental Significance and Legislatorial Requirements for Wastewater Works.

2.17.1 Wastewater Sludge Disposal

Wastewater sludge can be derived from numerous sources. They are:

1. Raw or primary sludge (from a primary settling tank)
2. Anaerobically digested sludge
3. Oxidation pond sludge
4. Septic tank sludge

5. Waste activated sludge (sludge wasted from an activated sludge plant)
6. Humus tank sludge
7. Composted sludge

Most primary, secondary treatment processes as well as tertiary treatment processes yield sludge which requires proper disposal. Sludge solely from solid-liquid separation processes (sedimentation and flotation) is referred to as primary sludge while that which is resulting from biological processes is referred to as secondary sludge. Primary sludge consists mainly of solid particles which are of organic and inorganic matter while secondary sludge consists of excess biomass produced during biological treatment (Ramalho, 1983). Niessen, (2002) defined the tertiary treatment sludge as sludge consisting of biomass generated in advanced wastewater treatment such as denitrification. There are other solid streams often generated in the progression of the treatment process however these are in lesser quantities than the sludge. These streams include:

- Scum - the floatable material accumulating on clarifiers and activated sludge reactor;
- Screenings - the rags, twigs, and other large solids screened from the entering wastewater, and ‘‘grit’’ -the coarse sandy, silty solids removed following screening.

Mulalo (2004) suggested that therefore a conditioning step is required to enhance water removal and improve solids recovery as wastewater sludge typically exhibits resistance to mechanical dewatering. This involves the physical or chemical treatment of the sludge solid to prepare the sludge for dewatering. Once the sludge is dewatered it can be disposed of or incinerated. Nozaic and Freese, 2008, suggested that the type of sludge needs to be taken into consideration when choosing a management option for the disposal of the sludge. Darvill WWW applies land disposal onto agricultural land for their generated secondary and anaerobically digested sludge.

2.17.2 Land disposal of sludge for agricultural purposes

Land application of wastewater sludge is a controlled spreading of sludge on or just below soils surface. The sludge may be applied to agricultural land, forestland, disturbed land, dedicated land and land disposal sites (Tchobanoglous *et al*. 2003). Ekama (1993) discussed some of the reasons for the increasing preference towards land disposal:

• Stringent regulatory constraints on the other options e.g. ocean disposal, incineration and lagooning which were other common methods of disposal.

- Awareness of the usefulness of the sludge as a soil fertilizer.
- Containment of the sludge thereby providing a useful sink for the sludge and its constituents
- Its greater economy

Darvill WWW uses Duzi Turf for land application of sludge through irrigation. The sludge serves as fertilizer for the grass grown by Duzi Turf. According to Hester and Harrison (1995), the high level of ammonia nitrogen in digested sludge ensures that grasses grow rapidly. Land application of wastewater sludge is a cost-effective and excellent way of recycling nutrients and organic matter into the soil-plant ecosystem, while on the other hand providing organic matter which improves the soil physical properties. However, care has to be taken to prevent runoff from polluting surface and ground water sources. Mulalo (2004) stated that the application of wastewater sludge is restricted by the presence of harmful constituents in the sludge, such as heavy metals, organic pollutants and pathogenic microorganisms. This has forced authorities to restrict the amount and type of sludge that can be applied to land.

Table 11 Wastewater sludge characteristics (Hester and Harrison, 1995)

Advantages
The water content varies from 99% down to 5%, dependant on the amount of thickening of dewatering which takes place, and can be of benefit to the crop/grass
Provision of some essential trace elements for crop growth
Nitrogen and Phosphorus ,at levels 3% and 6% of dry matter present, are valuable for crops
Organic matter, which varies between 50% and 80% of dry matter present, improves oil structure and its ability to retain moisture

2.17.3 Impact of regulations on wastewater engineering

During the 1900s to the early 1970s, treatment objectives were concerned primarily with the removal of colloidal, suspended, and floatable materials, the treatment of decomposable organic material and the eradication of pathogenic organisms. Implementation in the United State of the Federal Water Pollution Control Act Amendments of 1972 (Public Law 92-500) also known as the Clean Water Act (CWA), stimulated significant changes in wastewater treatment to achieve the objectives of "fishable and swimmable" waters. Unfortunately, these objectives were not consistently met. From the early 1970s to around the 1980s, wastewater treatment objectives were founded primarily on aesthetic and environmental concerns (Tchobanoglous, 2004). The earlier objectives involving the reduction of biological oxygen demand (BOD), TSS and pathogenic organisms continued but at higher levels. Removal of nutrients, such as nitrogen and phosphorus, also began to be addressed. Major programs were undertaken by both mate and federal agencies to achieve more effective and widespread treatment of wastewater to improve the quality of the surface waters. These programs were based, in part, on an increased understanding of the environmental effects caused by wastewater discharges, a greater appreciation of the adverse long-term effects caused by the discharge of some of the specific constituents (Tchobanoglous, 2004). Till date, countries have stringent legislation governing the treatment and discharge of wastewater into fresh water bodies.

2.17.4 Key legislation directed at controlling wastewater discharges in South Africa

Waste Legislation: The Constitution 1994: The bill of rights within the constitution is a cornerstone of democracy in South Africa. It enshrines the rights of all people in our country and affirms the democratic values of human dignity, equality and freedom. **Relevant sections: The Bill of Rights (Chapter 2) Section 24.**

Explanation of Acts/Regulations: Everyone has the right to an environment that is not harmful to their health or well-being and to have the environment protected for the benefit of present and future generations, through reasonable legislative and other measures that prevent pollution and environmental degradation.

Waste Legislation: The Environment Conservation Act 73 of 1989. The aim of this Act is to provide for the effective protection and controlled utilisation of the environment and for matters incidental thereto

Relevant sections:

1. Part IV S19 - Prohibition of littering
2. Part IV S19A - Removal of litter
3. Part IV S20 - Waste management
4. Regulations GN. R 1986
5. Regulations GN. R 1182, S21 - 1(n) & 8
6. Regulations GN. R 1183, S28

7. Regulations GN. R 1064 – Policy on Hazardous Waste Management

8. Regulations GN. R 1196

9. Regulations to control the Generation and Transportation of Hazardous Waste.

10. Part VII Section 29

Explanation of Acts/Regulations:

1. Control of pollution from littering, waste disposal, and various other activities

2. Landfill site permit

3. Identification of matter as waste

4. Identification of activities with a substantial detrimental effect on the environment

5. Environmental Impact Assessment Regulations

6. The management of hazardous waste (classification)

7. Application for a disposal site permit

8. Introduction of registration and manifest systems, procedures regarding the manifest system. Umgeni Water, "Cradle to Grave" process and documentation

9. Offences, Penalties and Forfeiture

Waste Legislation: The National Environmental Management Act No. 107 of 1998 (NEMA). The purpose of this Act is to promote co-operative environmental governance by establishing principals for decision making on matters affecting the environment

Relevant sections:

1. Section 28

2. Section 30

3. Section 31(1(a&b))

4. Section 32, 33, 34

Explanation of Acts/Regulations:

1. Duty of care and remediation of environmental damages, Waste Avoidance

2. Control of emergency incidents and clean ups

3. Right to information on waste handling, transportation, storage and disposal

4. Legal standing to enforce environmental laws, private and criminal prosecutions

Waste Legislation: The National Water Act 36 of 1998

The purpose of this Act is to meet the basic human needs of present and future generations, promoting equitable access to water, redressing the results of past racial and gender discrimination, promoting efficient and sustainable use of water in the public interest, facilitating social and economic development, providing for the growing demand of water users, protecting aquatic and associated eco-systems and their biological diversities, and reducing and preventing the pollution and degradation of water resources and relevant Regulations.

Relevant sections :

1. Section 19

2. Section 20

3. Section 21g

4. Section 29d

5. Chapter 16

Explanation of Acts/Regulations:

1. Control and prevention of surface and ground water pollution resulting from waste & compliance with waste standards and management practices

2. Emergency incidents

3. Disposal of waste which could detrimentally impact on water sources

4. Condition s for authorizations and licenses of waste treatment activities and management practices

5. Offences and remedies, Compensation, Award of damages

Waste Legislation: Water Act 54 of 1956

The National Water Act, 36 of 1998 has taken the place of the Water Act 54 of 1956. Notwithstanding the above, certain sections of this Act and Regulations made thereunder are still in operation.

The Minimum Requirements

Minimum Requirements are standards by means of which environmentally acceptable waste disposal practices can be differentiated from environmentally unacceptable waste disposal practices.

Relevant sections :

1. Section 22
2. Section 26g
3. Section 170
4. Document 1: Minimum Requirements for Waste Disposal by Landfill
5. Document 2: Minimum Requirements for the Handling and Disposal of Hazardous Waste
6. Document 3: Minimum Requirements for the monitoring at Waste Management Facilities

Explanation of Acts/Regulations:

1. Pollution of water
2. Registration for a waste disposal site
3. Offences and penalties
4. To improve the standard of waste disposal in South Africa
5. To provide a framework of minimum waste disposal standards within which to work and upon which to build
6. To avoid degradation of the environment and to prevent pollution of surface and ground water by leachate produced

Waste Legislation: Municipal by-laws

Numerous different by-laws exist. By-laws control the provision of waste services such as provision and movement of containers and placing of waste (disposal).

3. Research Methodology

3.1 Introduction

The research approach that will be used for this research includes quantitative approach and interpretation of secondary data. The first objective of determining COD fractions from settled usage data will encompass the use of quantitative data obtained from experiments and sample testing. The second objective of determining the biological nutrient removal efficiencies was used in evaluating secondary data of Darvill WWW. The third objective of developing a Darvill WEST Model used quantitative data obtained from experiment as well as secondary data from Darvill WWW.

3.2 Determination of COD Fractions from Darvill WWW settled sewage – Experimental Method

This section outlines the experimental considerations in fulfilment of the first objective. It is essential to determine wastewater components and parameters that are specific to Darvill WWW so that an accurate and specific model can be developed. The experimental work carried out was to determine the COD fractions in the settled sewage (characterisation). A BM-Advance respirometer was used to conduct the experiments. The percentage fractions of the COD i.e. the bCOD and readily-biodegradable fraction COD (S_S) were determined and thereafter using a set of equations and mass balance, the fractions of bCOD and the iCOD were determined. The bCOD was fractionated into (S_S) and slowly-biodegradable COD (X_S) and the iCOD was fractionated into X_I and S_I.

3.3 The BM-Advance respirometer

The principle of the BM-Advance operation was fully designed by Surcis team engineers. Its principle is exclusive and in general the BM operation is a state of the art design that permits a practical and fast ways of carrying out a wide set of respirometry tests and applications. It can be a fundamental tool for design, modelling, control and protection of the biological process of a wastewater treatment plant.

The respirometry principle is based on the oxygen uptake by the microorganisms contained in the genuine activated sludge from the own biological reactor of the treatment plant. Its operation lies in a closed circuit batch method, by means of continuous dissolved oxygen measurements from the activated sludge, mixed liquor and mixing formed with sludge and sample to be analysed in a single reactor glass. The consumed oxygen becomes as a resultant effect of the microorganisms' respiration in the activated sludge, from the biological oxidation of substrate (organic or ammonium) and from its own survival consumption (endogenous respiration).

Figure 14: BM-Advance respirometer

3.4 The reactor vessel

The reactor consists of a 1 litre continuously stirred vessel. The reactor consists of a dissolved oxygen (DO) probe, air diffuser, stirring paddle, peristaltic pump and temperature controller connected to a meter. The diffuser ensures the continuous and consistent supply of air. The air supply into the reactor is controlled based on the input set point into the analyzer (expressed as a percentage of full aeration). The DO probe measures the dissolved oxygen in the system which will drop should there be any addition of a biodegradable substrate and will return to original value once the substrate has been completely broken down. The peristaltic pump ensures that there is adequate mixing to maintain temperature and prevent settling of suspended solids. The pH probe measures the pH inside the reactor vessel. The stirrer paddle ensures mixing so that the substrate and biomass are in contact and to prevent settling of the suspended solids. The temperature controller is programmed to 20 °C and controlled by a peltier-device.

3.5 Experimental Preparations

3.5.1 Wastewater Sample collection

The sludge inoculum was collected at the end of the aerobic reactor since there was a low risk of having residual bCOD. An auto sampler was programmed to take samples of settled sewage at hourly intervals for 24hrs. The composite samples (50 mL filtered and 50 mL unfiltered) were used as substrates for the sludge inoculum.

3.5.2 Activated sludge

The activated sludge was the reagent that was used to perform the experiments. The requirements that the activated sludge had to fulfil were as follows:

- The sludge had to be collected at the final end of the activated sludge reactor
- A litre of the final sludge volume was required for the analysis therefore at least 2 litres were required at the time of collection.
- The activated sludge had to be of 2 – 5 g/L of MLSS concentration and in the case that this concentration was higher; the sludge had to be diluted using the secondary settling tank supernatant.
- The sludge had to be strained using a 600 µm strainer as precaution against particulate heavy solids or waste which could damage the apparatus.
- The sludge had to be aerated over 24 hours from the previous day so as to allow for endogenous respiration to occur. The aeration was achieved by the use of a simple aquarium air pump.
- A nitrifier inhibitor Allyl Thiourea (ATU) had to be added at 2 mg/g MLSS at half an hour before the experiment.

3.5.3 Total endogenous respiration rate

When activated sludge is under endogenous respiration, it does not have any bCOD or ammonium. The required time to reach the endogenous respiration state can vary from a few hours to 24 hours. It depends on the type of process type, actual loading rate state, physic-chemical conditions and the biodegradability of the material under treatment. In order to verify if the sludge is already under endogenous respiration, the sludge is added into the reactor vessel so as to check for a stable DO reading. If the DO value is still increasing, it is an indication that the sludge is not yet under the endogenous respiration phase and still requires more aeration.

3.6 Determination of COD fractionation from Darvill WWW - Methodology

3.6.1 Introduction

The respirometry principle is based on the oxygen uptake rate (OUR) by the microorganisms contained in the activated sludge in a biological reactor of the treatment plant to biodegrade a carbonaceous substrate. It works on a closed circuit batch mode. A litre of the filtered activated sludge was used in the respirometer. The test was done in 3 stages:

Stage 1: Determination of the heterotrophic yield using Acetate

The heterotrophic yield determines both the mass of electron acceptor utilized and the new cell biomass produced in the aerobic reactor. A solution of 400 mg of sodium acetate in 1 litre of distilled water was prepared. For this solution, a COD value (CODac ≈ 300 mg/L) was obtained from the lab using standard methods. The respirometric tests were carried out in order to determine the consumed oxygen (CO) due to the breakdown of acetate by the use of a 50 ml sample of acetate reacting with the 1 litre endogenous activated sludge. Once the respirometer had finalized, consideration that the whole acetate had been broken-down is upheld.

$$YH_{COD} = 1 - COac \, / \, CODac \tag{25}$$

Where: CO: Consumed oxygen = Δ O_2 (mg/L)

$Y_{H,COD}$ - Heterotrophic yield coefficient (mg O_2/mg COD).

Stage 2: Determination of bCOD by the addition of the 50 ml 24h composite sample. The composite sample (50 mL) was used as substrate and its total COD concentration was determined using standard methods. 1-Allyl-2-Thiourea was used at a ratio of 2:1 to sludge MLSS in order to inhibit nitrification so that only the carbonaceous oxygen consumption was derived. The mixture of the sludge and samples were continuously aerated with dissolved oxygen were measured every 2 seconds until the process was observed to be completed. The dissolved oxygen concentration decreased as the amount of consumed oxygen increased. The consumed oxygen becomes a resultant effect of the microorganisms' respiration in the activated sludge, from the biological oxidation of substrate (organic or ammonium) and from its own survival consumption (endogenous respiration). The test was observed to be complete when the amount of initial dissolved oxygen (DO) was equal to the final DO. Each batch runs as long as there is still consumed oxygen (approx. 50 min).

$$bCOD = CO_{bCOD}/1 - YH_{COD} \tag{26}$$

where:

CO_{bCOD} - Consumed oxygen due to the breakdown of the 24 hr composite sample

From the amount of bCOD determined, a percentage could thereafter be derived from the total COD concentration of the 24hr composite sample that was analysed in the lab using standard methods.

$$\%bCOD = (bCOD/COD_{COMP}) * 100 \tag{27}$$

The iCOD was deduced from the COD_{COMP} and bCOD concentration

$$iCOD = COD_{COMP} - bCOD \quad (28)$$

Where:

COD_{COMP} – Total COD in the 24 hr composite sample

and

$$\%iCOD = 100 - \%bCOD \quad (29)$$

Stage 3: Determination of SS by the addition of the filtered 24h composite sample

The second stage procedure was followed to determine the S_S concentration however a 50 ml filtered (using 0.45 µm filter) composite sample was used as substrate. The experiment yielded a consumed oxygen concentration through the breakdown of the filtered sample. The concentration of Ss was determined:

$$S_S = CO_{rbCOD}/1 - YH_{COD} \quad (30)$$

where:

CO_{rbCOD} - Consumed oxygen due to the breakdown of the 24 hr filtered composite sample. The bCOD concentration and the S_S concentration were used to determine the S_S percentage ($\%S_S$).

$$\%S_S = (S_S/bCOD) * 100 \quad (31)$$

It is from here that the X_S was deduced from the bCOD and S_S concentration as:

$$X_S = S_S - bCOD \quad (32)$$

Resulting X_S percentage ($\%X_S$)

$$\%X_S = X_S/bCOD \quad (33)$$

Stage 4: X_I and S_I fractions are deduced through mathematical reduction from all the experimentally/lab determined fractions.

$$S_I = COD_{TS} - S_S \quad (34)$$

Where: COD_{TS} – The total soluble COD in the composite sample as per lab analysis using standard methods.

Resulting S_I percentage ($\%S_I$)

$$\%S_I = S_I/iCOD \quad (35)$$

$$X_I = iCOD - S_I \quad (36)$$

Resulting X_I percentage ($\%X_I$)

$$\%X_I = X_I/iCOD \quad (37)$$

3.6.2 Biodegradability for a specific activated sludge process

This biodegradability as seen from the activated sludge respirometry view, under equivalent conditions to the actual ASP, should be considered not only from the biodegradable character of the wastewater sample to be analysed but also from the sludge activity health and sample adaptation to the biomass. For that reason, this type of biodegradability should be specific for the activated sludge responsible of the organic matter oxidation of the influent wastewater. Here, the biodegradable fractions with the total COD were compared.

Table 12 bCOD/COD Comparison

bCOD/COD	Character
> 0.8	Very biodegradable
0.7-0.8	Biodegradable
0.3-0.7	Little biodegradability
<0.3	Unbiodegradable

3.7 Biological Nutrient removal efficiencies

3.7.1 Introduction

This section outlines the sampling process to obtain the data that is evaluated in fulfilment of the second objective. It also details the method of evaluation used. It is essential to evaluate the nutrient removal efficiency to understand the Darvill activated sludge performance during this period of overloading.

3.7.2 Sampling

A one litre representative grab sample of settled sewage and final effluent to the river were collected and appropriately labelled every morning at 8:00 am at Darvill WWW. Settled sewage is the supernatant sludge from the primary settling tank that goes into the activated sludge reactor. The collection times and sampling points were kept constant throughout the time of the study. Once samples were collected, they were stored in a cooler box with ice and transported immediately to the laboratory where they were handled within three days after drop-off.

3.7.3 Laboratory Analysis

The concentrations of ammonia and SRP were determined as outlined by the standard methods (1995) for ammonia and SRP determination. Briefly, the samples were analysed accordingly. The instrument used to determine the concentration of NH_3 and SRP was the DR900 Spectrophotometer.

3.7.4 Statistical Analysis and calculations

The concentrations from the lab analysis were used to calculate the BNR efficiency of each parameter in the settled sewage and final effluent sample (see equation 38). The plant inflow was recorded hourly and an average for the day was also recorded.

$$Removal\ Efficiencies = \frac{(\text{Settled sewage concentration} - \text{Final effluent concentarion})}{(\text{Settled sewage concentration})} \qquad (38)$$

Microsoft Excel, was used to plot the scatter plots and comparing the BNR efficiencies with the plant inflow.

3.8 Developing a Darvill WWW model methodology

3.8.1 Introduction

This section outlines the data collection and reconciliation process to obtain the Darvill WWW WEST model. The data used is process data and COD fractionation. information. The WEST software was used to develop the model. Section 4.8.1 outlines the data collection process while 4.8.2 is the data clean-up which includes the interpolation, estimation and fitting of missing data points, 4.8.3 shows data reconciliation for input into WEST and 4.8.4 shows the determination of the WEST model.

3.8.2 Data collection

WEST requires the input of plant settled sewage process data which includes settled sewage flowrate which is assumed to be equal to the plant inflow rate (steady state conditions), COD, TKN, TSS, TP, NH_3 and SRP.

3.8.3 Data clean-up

Sampling was not always conducted daily due to weekends, public holidays or shutdowns. Data gaps had to be filled using means of interpolation and estimation. A probabilistic fractionator which is an estimation tool was used to complete the missing data sets. It improves and ensures quality and consistency of the data used as input in biological wastewater treatment models. Use of the Probabilistic Fractionator procedure comprises of five steps:

a. Inputting the available plant measurements in the appropriate columns in the "Measurements" sheet.
b. Interpolating to fill in gaps in the measurement data in the "Interpolation" sheet using the INTERPOLAT macro. Interpolated values are generated for all variables for which some measurements are available but only interpolated inflow and COD values are actually used in the subsequent steps. The interpolation macro also generates weights for each variable to be used in the fitting procedure based on the type of measurement and whether there is an actual measurement or just an estimate.
c. Generating estimates of the missing measurements in the "Estimation" sheet. The calculated estimates are also used to calculate the upper and lower bounds for the fitted results.
d. Calculate the "inferred measurements" in the "Fitting" and "Fitting_SetSew" sheets. Inferred measurements are actual measurements where available or interpolated COD and Inflow values and estimates for all other measurement variables.
e. Calculate the composition in terms of components which best fits the inferred measurements (weighted least squares) using the FITSolve or FITSolveSETSEW macro (User Initiated) and calculate the corresponding fitted measurements.

3.8.4 Probabilistic Fractionator Approach

The probabilistic fractionator uses weighted least-squares optimisation to arrive at the composition estimate. A particular advantage that the least-squares approach has over a straightforward algebraic fractionator is that constraints can be introduced to prevent infeasible compositions, such as negative component concentrations, which tend to cause numerical problems in WEST. The probabilistic fractionator algorithm principle is to begin with a general estimate of the wastewater compositions based on literature or plant process data, and to adjust the compositions when one or more measurements are encountered. The measurement values are input to the algorithm as time series. The anticipated values for all measurements (plant process data) considered by the algorithm can be calculated from the model component

concentrations. The component concentrations are the fitting variables, which are adjusted by the probabilistic fractionator solver to match the inferred measurements as closely as possible. When input measurements are encountered, their values are compared to the corresponding predicted values. This way, any combination of measurements can be taken into account (including none at all). A secondary mechanism uses correlations between measurements to generate estimates of certain measurements when the actual measurements are not available.
These estimates are then used as second class measurements in the algorithm, with lower weightings in the objective function e.g. iCOD is hardly ever measured, but is essential for modelling purposes. The total COD is normally measured and total suspended solids (TSS) and TKN provide no information on this fraction. However, the fraction can be estimated from the total COD using a literature correlation. In this sense, the algorithm will generate a value that at least conforms to what is expected from experience.

3.8.5 Probabilistic Fractionator Measurements

All measurements considered by the organic fractions model are the ones that are related to the elemental content of the components, which means that they can be calculated from a known model composition. Measurements on settled sewage, primary sludge and secondary influent are also considered, because they can provide information on the composition of the influent. The measurements currently included in the user input options (“Measurement” page) are listed in Table 13.These are measurements are typically collected for monitoring and operation purposes and are therefore the most likely to be available.

Table 13 User Input Measurements

Symbol	Description	Units
Raw Sewage		
Inflow	Raw sewage influent flow	m^3/d
COD	Total COD	$gCOD/m^3$
TKN	Total Kjeldahl nitrogen	gN/m^3
TSS	Total suspended solids	$gTSS/m^3$
TP	Total phosphorus	gP/m^3
FSA	Free and saline ammonia	gN/m^3*
OrthoP	(Soluble) ortho-phosphate (SRP)	gP/m^3*
pH_in	Influent pH. Not currently used in the Excel fractionator	
Alk_in		$gCaCO_3/m^3$

Temperature	Influent alkalinity. Not currently used in the Excel fractionator Influent temperature. Not currently used in the Excel fractionator	°C
Settled Sewage		
Flow_SetSew	Settled sewage flow	m^3/d
COD_SetSew	Settled sewage COD	$gCOD/m^3$
TKN_SetSew	Settled sewage TKN	gN/m^3
TSS_SetSew	Settled sewage TSS	$gTSS/m^3$
TP_SetSew	Settled sewage TP	gP/m^3
FSA_SetSew	Settled sewage FSA	gN/m^3
OrthoP_SetSew	Settled sewage OrthoP	gP/m^3
Primary Sludge		
Flow_PS	Primary sludge flow	m^3/d
PS_%TS	Primary sludge total solids concentration	%
PS_%VS	Primary sludge % volatile solids	% of TS
PS_%IS	Primary sludge %inorganic solids	% of TS
Additional "measurements" calculated directly from plant measurements		
Non-settlable solids	Calculated from TSS_SetSew	g/m^3
Settlable solids	Calculated from Flow_PS and PS_%TS	g/m^3
Settlable IS	Calculated from Flow_PS , PS_%TS and PS_%IS	g/m^3
COD_us	Unbiodegradable soluble COD calculated as Effluent COD – (gCOD/g OHO)*Effluent TSS	g/m^3

Table 14 Other measurements used in the Probabilistic Fractionator algorithm

Symbol	**Description**	**Units**
Raw Sewage		

COD_filtered	Filtered (soluble) COD	gCOD/m^3
COD_oho	COD of micro-organisms (not in Steady State Model	gCOD/m^3
COD_up	input version)	gCOD/m^3
COD_up_ns	Un-biodegradable particulate COD	gCOD/m^3
COD_bp_ns	Un-biodegradable non-settlable particulate COD	gCOD/m^3
ISS	Biodegradable non-settlable particulate COD	gISS/m^3
TKN_filtered	Inorganic suspended solids	gN/m^3
TP_filtered	Filtered (soluble) TKN	gP/m^3
VFA	Filtered (soluble) total phosphorus	gCOD/m^3*
	Volatile fatty acids (as acetic acid).	

* Note that the units for the measurements FSA, OrthoP and VFA (gN/m^3, gP/m^3 and gCOD/m^3) are different from the units of the corresponding components (g/m^3 of NH_3, PO_4 and HAc respectively)

3.8.6 Probabilistic Fractionator Estimates

The probabilistic fractionator generates estimates as ratios to other measurements when measured data is missing, usually COD, to use as surrogate measurements for the least squares fitting procedure. Estimates are given lower weights in the fitting. The estimates are generated in the "*Estimates*" sheet using the correlation factors listed in the "*Factors*" sheet. The correlation factors used in the current version of the Excel fractionator are listed in Table 3. Note: all the correlation factors refer to the composition of the influent.

Table 15 Correlation Factors used to generate estimates

parameter	Description	Default values
f_tss	Ratio of TSS to total COD	
f_codus	Un-biodegradable soluble fraction of	
f_codup	total COD	
f_tkn	Un-biodegradable particulate fraction of	
f_fsa	total COD	
f_vfa	Ratio of TKN to total COD (mg N/mg	
f_codf	COD)	
f_tknf	Ratio of FSA/TKN (mg N/mg N)	
f_tp	Fraction of total COD contributed by	
f_tpf	VFAs (taken to be acetic acid)	

f_iss	Soluble fraction of total COD	0.1
f_bp_ns	Ratio of filtered TKN to total COD	0.6
f_up_ns	Ratio of total phosphorus to total COD	0.4
f_iss_ns	(mg P/mg COD)	0.1
f_setsewflow	Ratio of filtered total phosphorus to total	0.985
f_bu	phosphorus	4.7
	Inorganic fraction of total suspended solids	
	Non-settlable fraction of the biodegradable particulate COD	
	Non-settlable fraction of the biodegradable particulate COD	
	Non-settlable fraction of ISS	
	Fraction of raw sewage going to settled sewage	
	Ratio of biodegradable to biodegradable particulate COD	

Depending on which measurements is available, the following ratios can typically be calculated directly from the data in the "*Measurement*" sheet:

$$f_{\_tss} = \frac{Average\ TSS}{Average\ COD} \qquad (39)$$

$$f_{\_TKN} = \frac{Average\ TKN}{Average\ COD} \qquad (40)$$

$$f_{\_fsa} = \frac{Average\ FSA}{Average\ TKN} \qquad (41)$$

$$f_{\_tp} = \frac{Average\ TP}{Average\ COD} \qquad (42)$$

$$f_{\_tpf} = \frac{Average\ OrthoP}{Average\ TP} \qquad (43)$$

$$f_{\_codus} = \frac{Average\ COD\_us}{Average\ COD} \qquad (44)$$

where COD_us = Effluent COD – (gCOD/g OHO) * Effluent TSS

$$f_{\_tss} = \frac{Average\ TSS}{Average\ COD} \qquad (45)$$

$$f\_iss = \frac{Average\ Settlable\ IS}{Average\ TSS*(1-f_{iss_{ns}})} \tag{46}$$

where the default value for f_iss_ns = 0.1

$$f\_setsewflow = \frac{Average\ Flow\_setsew}{Average\ Inflow} \tag{47}$$

Relationships 1-7 (equation 39 – 45) are calculated in the factors page. Relationship 8 (equation 46) is calculated in the PST solids balance page and relationship 9 (equation 47) is calculated either directly from measurement data of from the PST Solids Balance page.

In addition, f_codup and f_codf are calculated as follows to ensure internally consistent estimates of the COD fractionation:

$$f_{codup} = (1 - f\_iss)\frac{f\_tss}{\left(\frac{1}{X\_U\_COD}+\frac{f\_bu}{X\_B\_COD}\right)} \tag{48}$$

Where X_U_COD and X_B_COD are the COD per g for UPO and BPO respectively

$$f_{codf} = 1 - (1 + f_{bu})f\_codup \tag{49}$$

3.8.7 Probabilistic Fractionator Weights

The fractionator algorithm uses weighted least squares fitting approach to calculate the fractionated wastewater composition

$$min \sum_i w_i * \left[1 - \frac{inferred_i}{fitted}\right]^2 \tag{50}$$

Direct measurements of TSS, TKN, FSA are weighted more heavily than values calculated from measurements e.g. total settlable solids, COD_us, etc and both are weighted more heavily than estimates.

Table 16 default measurement weight values

Actual measurements	1
Derived measurements	0.5
Estimates	0.1 * measurement or derived measurement weight

Weights for inferred measurements for which there are typically no actual measurements e.g. VFA, CODup are specified directly in the weights columns in the fitting sheet.

3.8.8 WEST Model components

The probabilistic fractionator components are a subset of the components of the WEST model These are listed in Table 17.

Table 17 West model components used in the Probabilistic Fractionator

Component	Description	Units	WEST model component
s_NH	Ammonia	mg/L NH_3	Ammonia
s_VFA	Acetate	mg/L HAc	Acetate
s_PO4	Phosphate	mg/L PO^4	Orthophosphate
s_U	Un-biodegradable soluble organics	mg/L S_U	USO
s_F	Biodegradable soluble organics	mg/L S_F	FSO
x_U_Inf	Un-biodegradable particulate organics	mg/L X_U_Inf	UPO
x_OHO	Ordinary heterotrophic organisms	mg/l OHO	Not currently in the influent fractionation of the SS model
x_B_Inf	Biodegradable particulate organics	mg/L X_B_Inf	BPO
x_ISS	Inorganics suspended solid	mg/L X_ISS	ISS

All organic components are characterised by a stoichiometric formula of the form CxHyOzNnPp. The coefficients x, y, z, n and p allow the calculation of the contribution of each component to any of the analytic measurements considered in the model, such as COD, TSS, total phosphate, and total nitrogen. The probabilistic fractionator does not consider the full set of the WEST model components, but just the ones that are expected to be present in large amounts in influent wastewater. The other remaining components are assumed to have zero concentration.

3.9 WEST Model

The purpose of the model developed is to run simulations to show the compliance or non-compliance of the final treated effluent to the effluent discharge standards. The following steps provide a systematic approach for developing the WEST model (GMP unified Protocol):

1) Project definition

2) Data collection and reconciliation

3) Plant model set-up

4) Calibration and validation

5) Simulation and result interpretation

3.9.1 Project definition

The scope of modelling focused on the activated sludge process at Darvill WWW. The expectations of the model would be to predict the WWW's performance statistics rather than real time dynamics of an activated sludge system.

3.9.2 Data collection and reconciliation

The output of the data collection (Probabilistic Fractionator) and reconciliation step is reconciled data for use in the subsequent steps of the WEST model. The input data and quality of the data is dependent on the purpose of the model as it has an impact on the accuracy targeted in the model calibration step. The GMP guidelines highlight that data collection and reconciliation step is one of the steps that requires most of the effort in a modelling project and further emphasises the used of data of high quality in a process model (Rieger, 2013).

In this modelling project, this particular step required most effort in order to adhere to the GMP guidelines.

4. Discussion of Results

4.1 Introduction

This chapter discusses the results of this study. Section 0 discusses the COD fractionation experimental results; section 4.2 discusses the biological nutrient removal efficiency and section 4.3 delves on the Darvill WWW WEST modelling outcome.

4.1.2 COD Fractionation

Table 18 Sample of experimental results of study

CODcomp (mg/L)	**440**	**COD_{TS} (mg/l)**	**205**
	Consumed Oxygen (mgO_2/L)	**COD Concentration (mg/l)**	**Percentage fraction (%)**
Y_H,acetate	117.71	320	63
bCOD	97.9	266.2	60.5
iCOD		173.8	39.5
Ss	62.63	170.26	63.96
Xs		173.8	36.04
S_I		34.74	20
X_I		139.06	80

Table 18 presents the heterotrophic yield as used as basis for all experimental calculations on the endogenous sludge sample for finding the COD fractions as the heterotrophic yield denotes the biomass available in the reactor used for the experiment.

4.1.3 The heterotrophic yield using Acetate

4.1.4 Oxygen consumption response

Stage 1: The heterotrophic yield using Acetate

Acetate, which is a readily biodegradable substrate, was used as a reference for assessing the bioactivity of the endogenous sludge under aerobic conditions.

The typical heterotrophic response is presented in Figure 15 and Figure 16 respectively. Upon addition of acetate solution to the endogenous sludge, a rise in the consumption of oxygen is observed while a decline in dissolved oxygen is noted from about 82 s to about 1569 s. This is indicative of microbial breakdown on the added acetate. The oxygen utilization is triggered by the presence of acetate and/or the electron acceptor. Since no exogenous substrates other than acetate was added to the batch reactors, all responses were considered to be due to the utilization of endogenous products released by the bacterial cells attached to the sludge.

At about 1685 s, there is a notable steady increase in the dissolved oxygen and at 2525 s, the consumed oxygen curve becomes constant and no further changes in consumed oxygen are noted. This is indicative of the complete breakdown of acetate in the system and therefore from the experimental data, a value of the amount of consumed oxygen due to the breakdown of acetate can be determined.

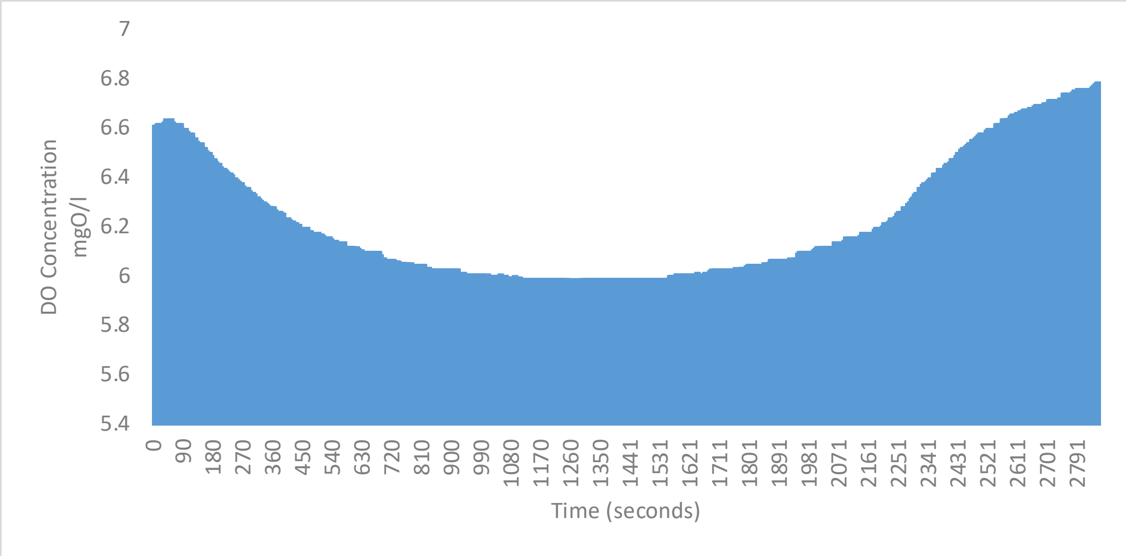


Figure 15: Dissolved oxygen response due to the addition of acetate

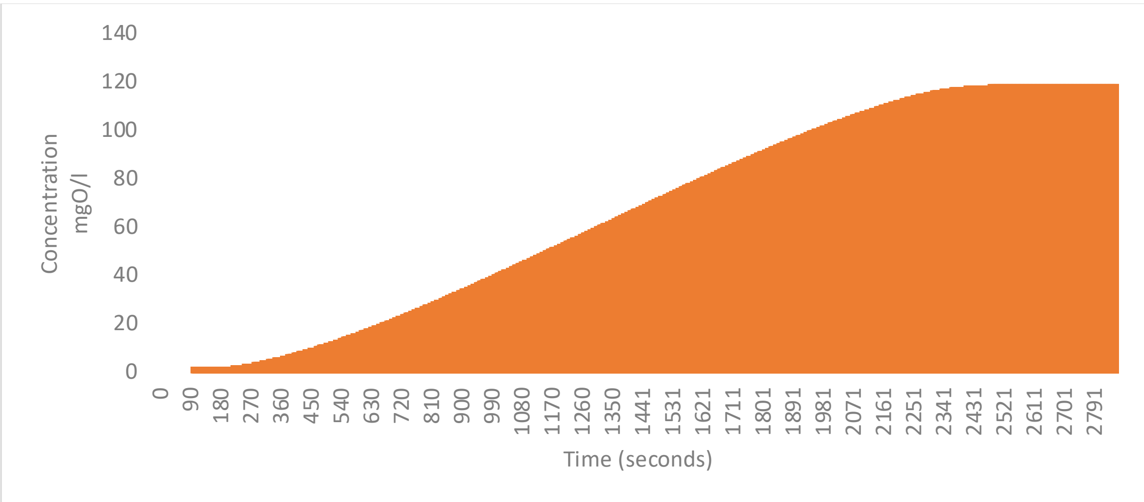


Figure 16: Consumed Oxygen response due to the addition of acetate

The extracted consumed oxygen of 117.71 mg/l and the lab analysed acetate COD concentration (320 mg/l) were used to determine the heterotrophic yield (see equation 25) which was found to be 0.63 mgCOD/mgCOD (see Table 18).

The accepted standard heterotrophic yield value of 0.67 mgCOD/mgCOD is used in ASMs and similar models (Muller et al. 2004) and the value of 0.63 mgO_2/mgO_2 is the yield coefficient suggested by Henze et al., (1995) for aerobic/anoxic reactions involving activated sludge. The experimental yield was found to be 0.63 mgCOD/mgCOD which is well within range and comparable to the expected literature values. High yield coefficients calculated are due to the presence of polyphosphate accumulating organisms which take up and store some of the available acetate. This is believed to be prevalent in the biomass that has been growing under dynamic conditions (Naidoo, 1999).

Stage 2: Determination of the bCOD fraction

The response to the addition of the sample is similar to the response to the addition of acetate. There was a notable decrease in the amount of dissolved oxygen and a noted increase in the amount of consumed oxygen. This is also due to the biodegradation of the COD present in the 24 hr composite sample. The added allyl thio urea inhibitor ensured that the response observed was solely due to the breakdown of the bCOD. The response curves to the addition of the 24 hr composite sample are noted in Figure 17 andFigure 18.

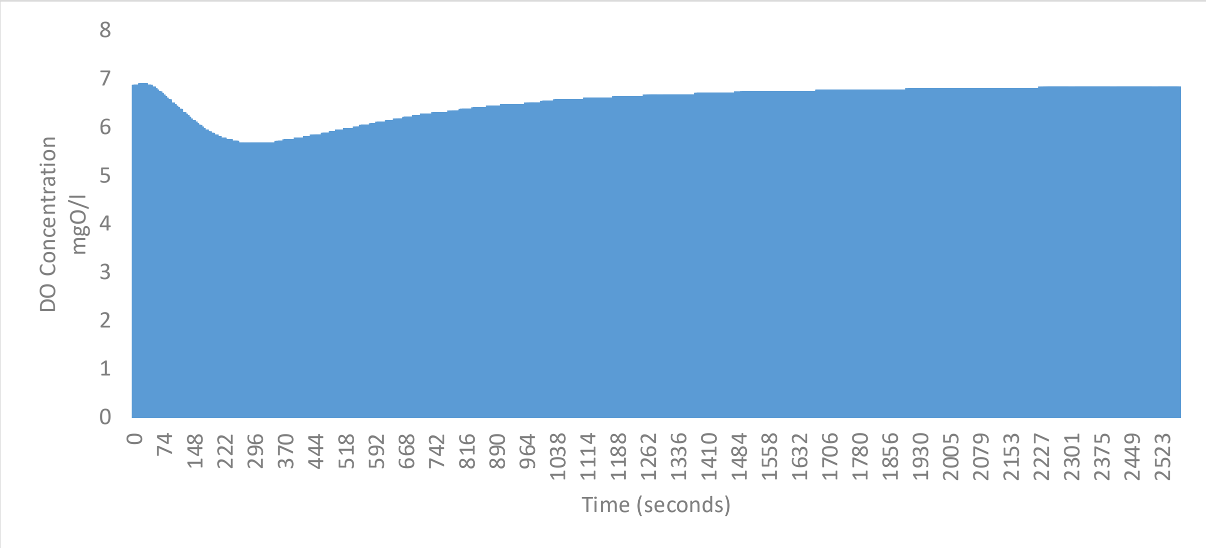


Figure 17: Dissolved oxygen response due to the addition of 24hr composite sample

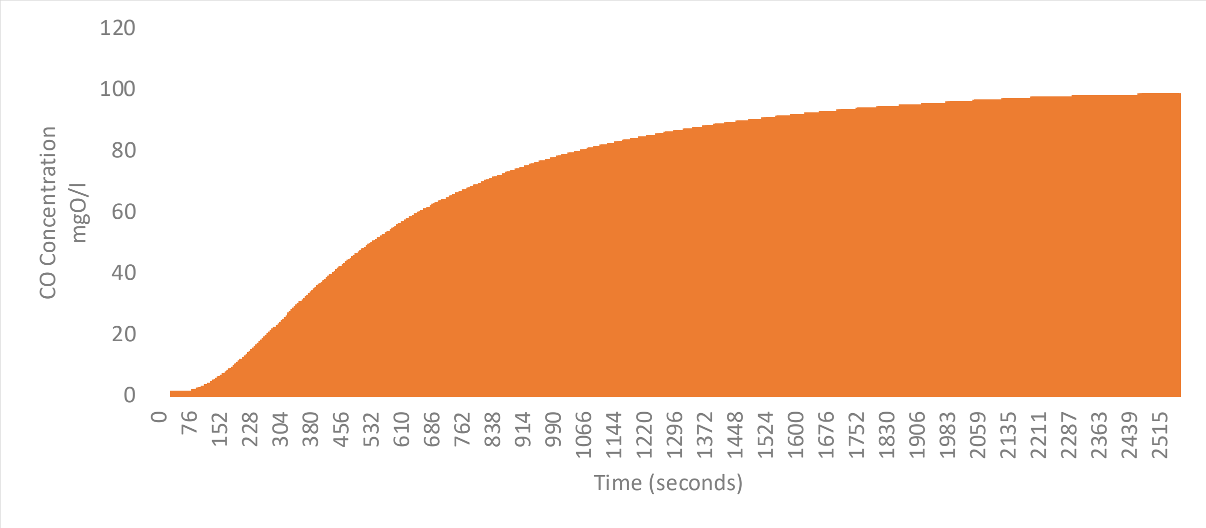


Figure 18: Consumed oxygen response due to addition of 24hr composite sample

The consumed oxygen due to the breakdown of the 24hr composite sample was found to be 97.92 mg/l while the bCOD concentration was found to be 266.2 mg/L (equation 26).

Stage 3: The determination of Ss COD

The response curves due to the addition of the filtered composite sample (Figure 19 and 20) are similar to those of the acetate and the bCOD. There is an observed drop in the dissolved oxygen in the endogenous sludge from 102 s to 632 s where the lowest DO were observed (6.71 mgO/L). The DO steadily increases denoting the decreased concentration of the readily biodegradable COD in the system whereas at 1821 s, the DO becomes constant which indicates a complete breakdown of Ss.

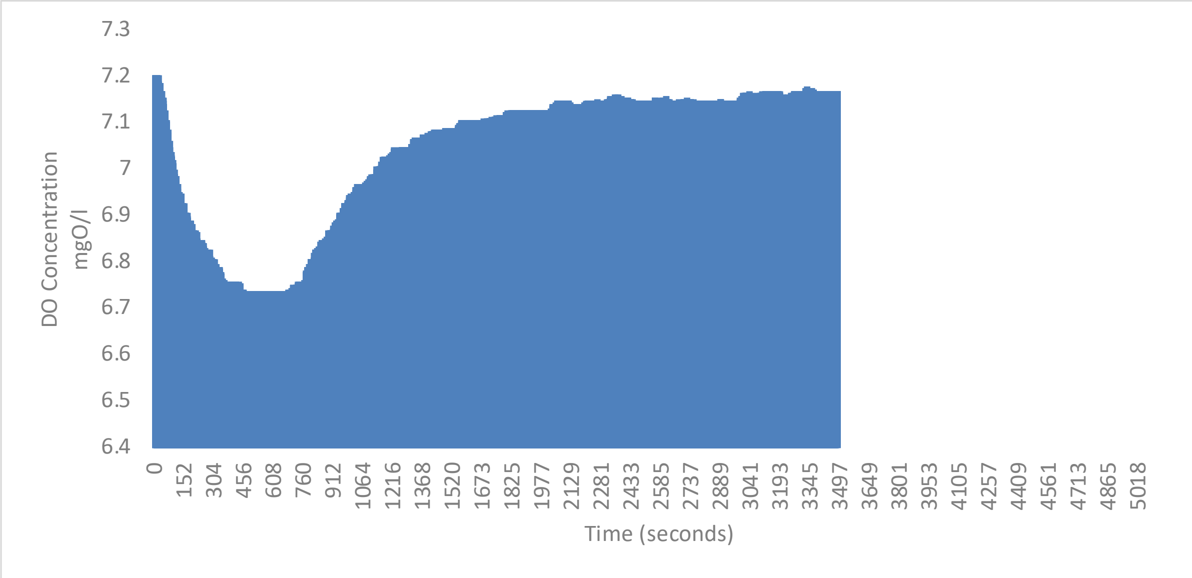


Figure 19: Dissolved oxygen response due to the addition of the 24hr filtered composite sample

The behaviour of consumed oxygen due to the addition of the 24hr filtered composite sample is observed in Figure 17. It is seen that there is some activity two seconds after the addition of the sample but there is a measure of consumed oxygen. There is an increase from 14 s to 4763 s due to the breakdown of substrate, after point 4763 s, there is an observed constant consumption of oxygen. Once there is a constant trend, biodegradation is said to be complete; as at this point, the DO response curve is also seen to be constant at 7.18 mgO_2/L.

The difference in the 3 experiments that were performed are the times taken to run each experiment to completion. This is simply due to the nature of COD in the 3 instances. Acetate has a shorter carbon compared to the bCOD but the readily biodegradable COD is a filtered sample (no particulate matter) and is a soluble readily biodegradable fraction of the bCOD.

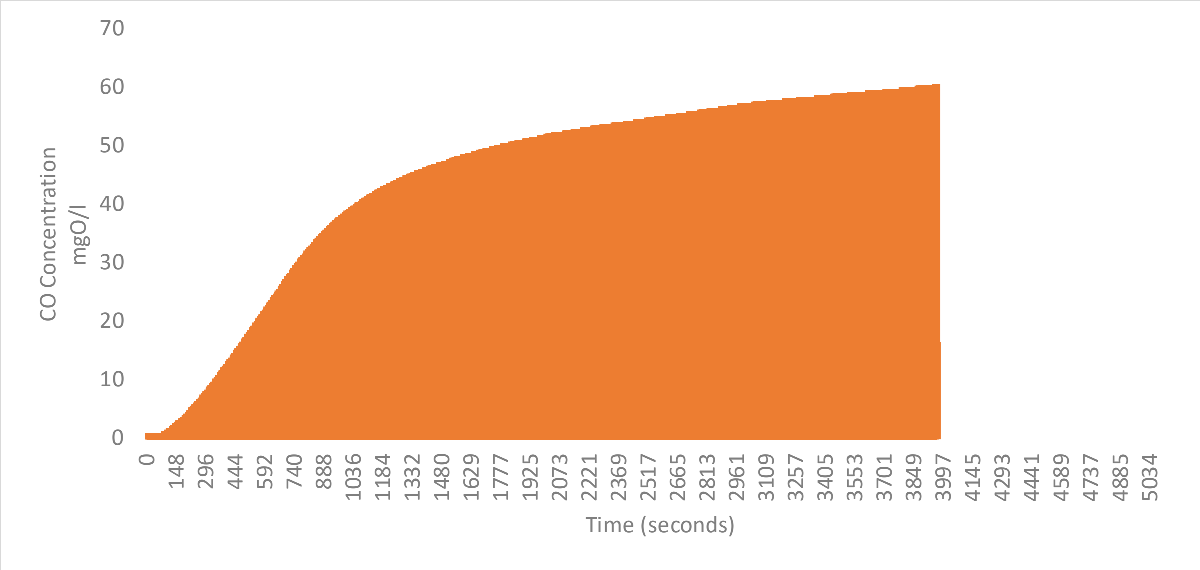


Figure 20: Consumed oxygen response due to the addition of 24hr filtered composite sample

The experiment yielded a consumed oxygen concentration of 62.63 mg/L through the breakdown of the filtered sample. The S_S concentration was found to be 170.26 mg/L (see equation 30). The S_S concentration found is a higher fraction of the bCOD and this is beneficial for the ASR. Choi *et al*. (2017) discussed that S_S is one of the most important components of COD as it is readily degraded by microbial metabolism. The $\%S_S$ in this instance is 63.96% (see Table 18) and is significantly higher than the 47% of degradability of medium concentrated domestic wastewater influent adopted from Henze *et al*. (2008).

4.1.5 Characterization

On the average, the primary effluent bCOD was found to be higher than the iCOD. S_S was higher than X_S. The S_S is the most valued substrate as it is easier for the microorganisms to be broken down. This therefore means that Darvill primary effluent has sufficient substrate for the COD removal and the biological nutrient removal process. The primary effluent of Darvill WWTW is seen to be similar to the domestic wastewater as the amount of bCOD was 70.5% and domestic wastewater was averaged by Henze et al. (2008) to be 75%; however, it is seen that the S_S fraction is significantly higher (75%) than the X_S fraction (25%). This can be directly linked to the industrial effluent coming into the plant. Oil (VFA) and Dairy plants have a high S_S content.

Particulate organic matter which is X_S was found to be 25%. The lower fraction is beneficial for the treatment process as this fraction is degraded slowly by a series of microbial actions, such as adsorption, hydrolysis, and metabolism. The S_I, was found to be 49.2% of the iCOD.

The S_I, is quite susceptible to biodegradation and is contained mostly in industrial effluents. Aromatic compounds which are used in various industries and are typical examples of S_I includes a varieties of soluble compounds, which can pass through the microbial wall but cannot be degraded due to their susceptible refractory nature. As a result, the S_I leaves the activated sludge reactor unaltered in concentration and characteristics (Wu *et al*. 2014). Darvill WWTP S_I fraction is significantly higher than that of medium concentrated domestic wastewater influent adopted from Henze *et al*. (2008) which is 17%. This shows that Darvill receives both industrial and domestic effluents.

Non-biodegradable Particulate COD (X_I) fraction was found to be 50.8%. This fraction becomes absorbed in the activated sludge and is removed by sludge wasting in a WWTP. The X_I significantly affects the volume of wasted sludge and forms part of the considerations for WWTP reactor design (Choi *et al*. 2017). Both S_I and X_I cannot be biologically degraded further in a WWTP and therefore can pass through the activated sludge system unchanged.

Hence, precise COD characterization is important for the efficient operation of biological nutrient removal wastewater treatment process. Several methods have been developed for COD characterization, but the two most commonly used processes are the biological and physical-chemical characterizations. The COD characterization results are represented in Figure 21.

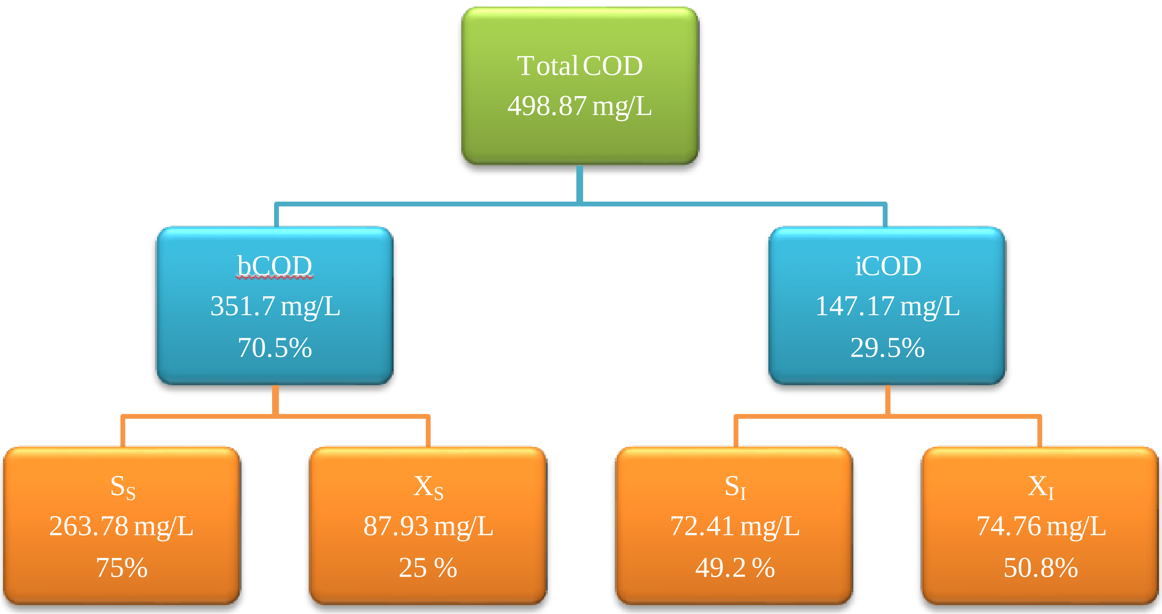


Figure 21: COD characterization

4.1.6 Summary

Wastewater characteristics are unique to every treatment plant and cannot be predicted without comprehensive analysis especially in cases where the influent is made up of both industrial and domestic wastes. The characteristics of Darvill primary effluent COD have been presented and are quite comprehensible. The plant receives process favorable COD fractions as there is a higher bCOD (70.5%) component and S_S (75%) component compared to the iCOD (29.5%). Domestic wastewater is reported to have the following average compositions: X_I = 4–20%, slowly biodegradable COD) = 24.5–65%, S_I = 3 –14.3% and S_S = 14–57%. Darvill wastewater characteristics show more resemblance to domestic wastewater effluents in terms of the biodegradable fractions, however the non-biodegradable fractions did show that Darvill receive mixed effluents. The most important fractions in this study are the bCOD fractions and are significantly higher than the iCOD which means that the process microorganisms in the ASR have sufficient substrates for the current biological nutrient removal process and no chemical additions are required to supplement the process while COD fractions can serve as input for an ASM. It is recommended that COD should be analyzed and characterized into its relative fractions on a routine basis for ease of process optimization through ASM.

4.1.7 iCOD and bCOD

The total settled sewage COD is characterised into two groups: bCOD and iCOD. In both groups, there exist soluble and particulate fractions of each. The experimental results of the ICOD and bCOD are discussed.

The bCOD and iCOD in Figure 22 shows that the two fractions have no definite predictability with time as all R^2 values were found to be far below unity (1). The highest R^2 values for bCOD and iCOD were 0.0769 and 0.0957 respectively. This indicates that the model equations for these graphs do not explain the variability of the response data around its mean. This holds true as wastewater influent COD content varies and has no proper way of prediction as Darvill wastewater influent is composed of domestic and industrial wastes.

The data collected through this study as presented in Figure 22 suggests that bCOD has an inverse proportionality to iCOD. The amount of bCOD on the average were found to be 70.5% and iCOD 29.5% of the total COD in the settled sewage. The design settled sewage load for Darvill waste water works are 420 mg COD/L. Through the study, it was observed to be at an average of 500 mg COD/L. The data suggests that there is sufficient “substrate” for the biomass in the activated sludge system for a functioning system and adequate biomass growth.

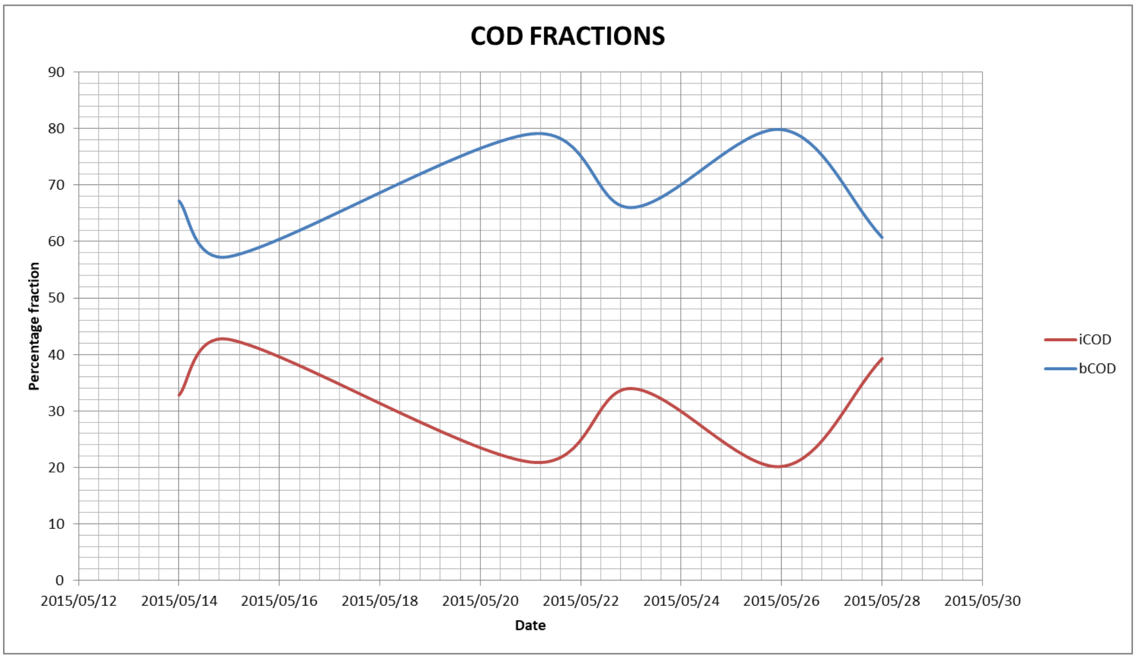


Figure 22: Plot of iCOD and bCOD % age fractions vs Time

4.1.8 S_S and X_S

S_S is the most easily accessible COD fraction for heterotrophic microorganisms. The amount of S_S is the determining factor for anaerobic and anoxic reactor volumes during model-based design because the processes of phosphate release and denitrification are very sensitive to the readily biodegradable fractions (Pasztor et al. 2009). X_S component is made up of colloidal and suspended COD fractions thus cannot be separated from influent samples solely by physical separation. Pasztor et al. (2009) infers that from an AS modelling point of view, this fraction typically has the highest oxygen demand and therefore, it greatly influences the airflow required in the aeration tank. In this section, the results of these two fractions were discussed. Figure 23 shows that the S_S over the time of the study is not easily predictable as the linear regression R^2 values are far below unity (1) with the highest being of a polynomial regression model with the R^2 of 0.0796. These results show the expected behaviour of wastewater influent particularly those comprising the industrial waste contents. The average S_S through the study was found to be 76%. The X_S graph also shows no predictability with polynomial regression of highest value of $R^2 = 0.08$. According to the scientific data, S_S in settled sewage ranges between 14–57 %.

The X_S is inversely proportional to the S_S graph with the S_S being higher than the amount of X_S. According to Henze *et al.* (2008), this X_S fraction in settled sewage ranges between 24.5–65 %. The lowest data points for S_S were slightly below 60% of the total bCOD. Henze *et al.*

(2008) therefore established that phosphorus release appears to be induced effectively if the S_S concentration in the reactor is 70-220 mg COD/L. The 76% average S_S yields a concentration of 319 mg COD/L and the lowest point being 57% yields a concentration of 239 mg COD/L. These are above the range established; therefore, suggesting that there is adequate S_S in the system for efficient EBPR.

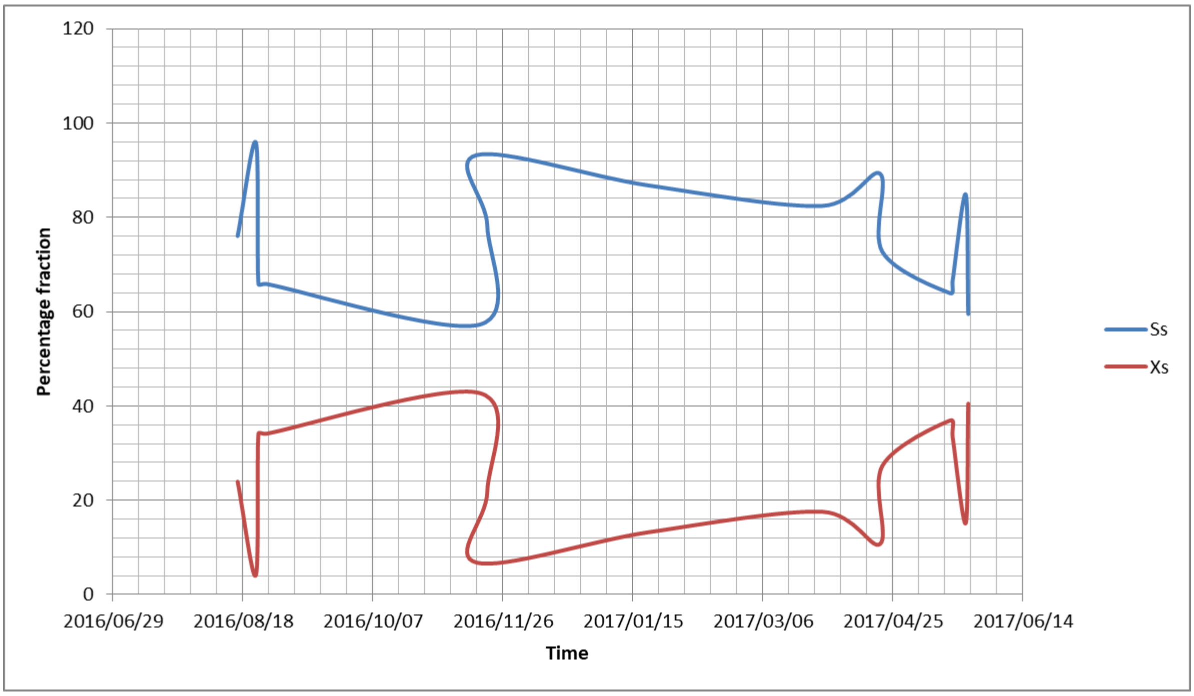


Figure 23: Plot of S_S and X_S %age fraction vs Time

4.1.9 S_I and X_I

The S_I fraction cannot be further biologically degraded during the treatment process and therefore leaves the process without any changes in its concentration. Similarly, the X_I component cannot be degraded biologically during the treatment process due to its non-reactive nature. The S_I fraction leaves with the plant's final effluent to the river and the X_I fraction is removed during the sludge wasting process. In this section the S_I and X_I is discussed. According to Henze *et. al*. (2008), the X_I fraction significantly influences the quantity of primary and secondary sludge and therefore it determines the required dewatering and sludge treating capacity while the Si fraction affects final effluent compliance. Figure 24 has the polynomial model for both the X_I and S_I plots having the highest at coefficient of determination, R^2 of 0.1988. This also shows that there is no predictability as the parent plot of bCOD and iCOD in Figure 22 was the same. This study implies that the average X_I was a bit higher than the S_I at 51 % and 49 % respectively of the total iCOD. The X_I has an inverse proportionality to the S_I. These percentages will both leave the system unaltered. Studies showed that the settled sewage and S_I fractions in domestic wastewater contains 4 – 20 % and 3–14.3 % respectively. This fraction is higher at Darvill WWW in the study due to the industrial component in its influent.

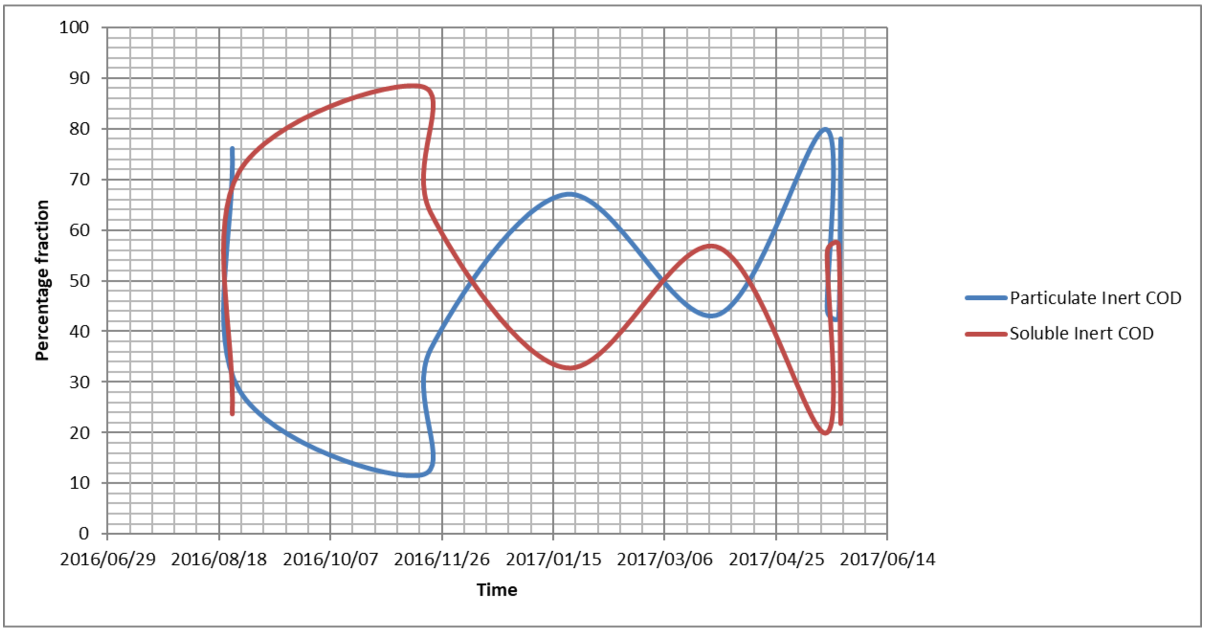


Figure 24: Plot of X_I and S_I percentage fractions vs Time

4.1.10 Average COD Fractions

Figure 25 presents the average COD fractionation outcome of the study. On the average, the primary effluent bCOD was found to be higher than the X_I and S_S which are higher than X_S. The S_S is the most valued substrate as it is easier for the microorganisms to break down. This therefore means that Darvill primary effluent has sufficient substrate for the COD removal and the BNR process. The settled sewage of Darvill WWW shows more resemblance to domestic wastewater. The amount of bCOD during the study was found to be 70.5% while domestic wastewater was averaged by Henze *et al.* (2008) to be 75% in their study. It is also observed that the S_S fraction is significantly higher (75%) than the X_S fraction (25%); this can be directly linked to the industrial effluent coming into the plant. Oil (VFA) and Dairy have high S_S contents. It also shows to be a diluted wastewater (see Table 2) which represents high water consumption and/or infiltration as the average total COD is almost 500 mg/L.

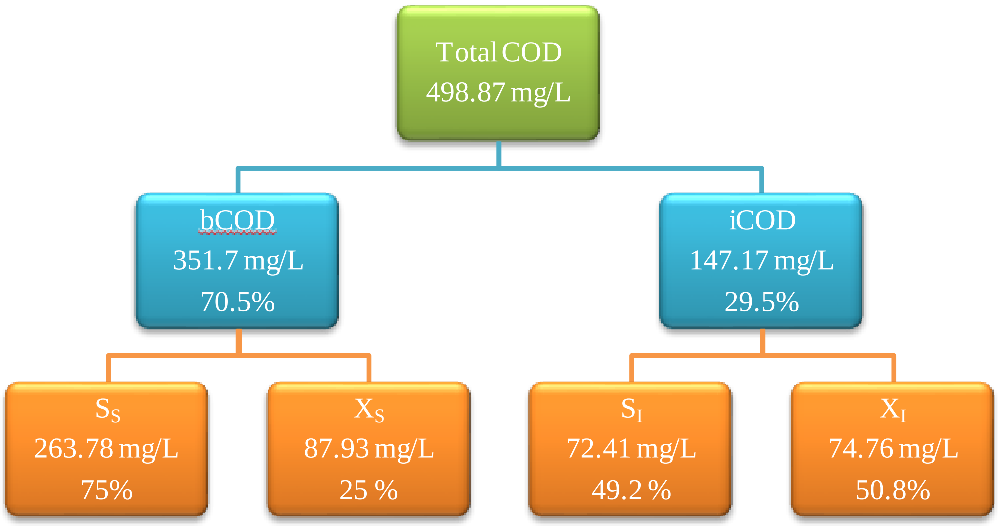


Figure 25: Average COD fractionation results presented graphically

4.2 Biological nutrient removal efficiencies

4.2.1 Introduction

This section aims at discussing the BNR efficiencies as per the study conducted. The plant flows for the period 2016 -2017 were plotted and compared to the removal efficiencies of ammonia and SRP. This was to establish the effects of hydraulic overloading on the activated sludge process.

4.2.2 Discussion

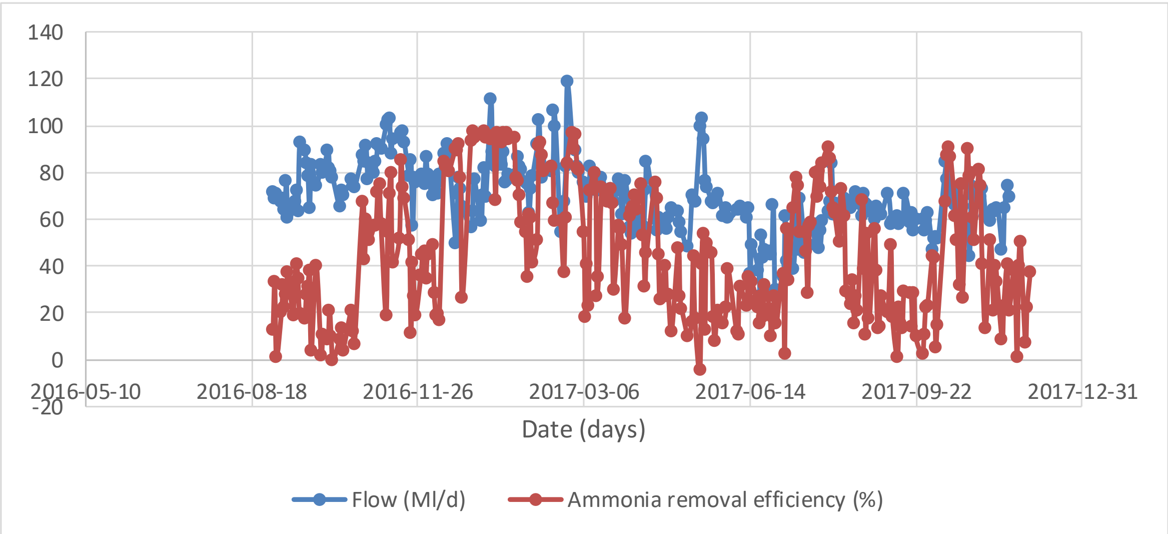


Figure 26: Ammonia removal efficiency and inflow comparison

The trend on the graph (Figure 26) suggests that the removal efficiency of NH_3 has a predominantly inverse proportionality relationship to the plant inflows. The average removal efficiency of ammonia during the period of 2016 and 2017 was 42% and 38% respectively. Darvill WWW has a design capacity of 65 ML/d however it can be seen that there have been numerous periods where the plant was operating above capacity, with peaks as high as 120 ML/d. November 2016, on average had the highest inflow of 88ML/d; however, the lowest removal efficiency was observed in June 2017 (23.2%) with a flow of 48.9 ML/d. This was due to procurement delays of faulty subsurface aerators which resulted in process upsets. This could also be related to the cold Pietermaritzburg's winter temperatures which could reach a minimum of 5°C. Previous studies indicated that denitrification rate was significantly affected by lower temperatures for biological treatments and denitrification rate at 5 °C was decreased by 10 times linearly as compared to 20 °C (Guo *et al*., 2013; Choi *et al*.,1998; Hocaoglu *et al.*, 2011). The temperature effects on BNR are primarily related to the low growth rate of the nitrifying bacteria at low temperatures (Ying, 2015). Ydstebo, *et al*. (2000) however found that nitrogen removal at low temperatures was achieved by operating at a high mixed liquor suspended solids (MLSS) concentration of 6000 - 8000 mg/L and with a sufficient supply of organic matter for denitrification. This however cannot be operated for Darvill WWW as this concentration of MLSS would lead to other process challenges like sludge bulking. Wastewater treatability depends largely on the characteristics of the sewage and the activated sludge conditions (Henze *et. al*. 2008). A higher inflow comes with higher organic and nutrient loads hence compromising the BNR process. It was observed that for the most part of the months of the study the ammonia removal efficiency was below 75% (see Table 19). This shows that the Darvill WWW treatment process was strained and the final effluent compliance was compromised.

Table 19 Average monthly inflow and Ammonia biological removal efficiencies

Month/Year	**Flow average (ML/d)**	**Removal efficiency average (%)**
Sep-16	73,7	24,1
Oct-16	79,3	27,6
Nov-16	88	51,9
Dec-16	75	63
Jan-17	81,1	87
Feb-17	84,5	68,9

Mar-17	74,7	55,1
Apr-17	60,9	50,2
May-17	70,8	25,7
Jun-17	48,9	23,2
Jul-17	52,2	59,2
Aug-17	66,4	42,3
Sep-17	60,8	19,8
Oct-17	64,9	60
Nov-17	64,2	29

The average removal efficiency of SRP during the period of 2016 and 2017 was 61% and 67% for the two years respectively. The trend shows a predominantly inverse proportionality relationship (see Figure 27). The results from this study suggest that the higher flows have a higher detrimental effect on the removal efficiency of ammonia than SRP. This is largely due to the fact that ammonia is removed during the nitrification process in the activated sludge reactor. The amount of dissolved oxygen available in the system becomes inadequate and the growth of the autotrophic nitrifying bacteria is compromised when the plant receives flows above capacity. At flows above design capacity, there is also a significant increase in organic loading which also requires and competes for the available dissolved oxygen in the activated sludge reactor. The POAs are able to compete successfully with other organisms for substrate in completely aerobic activated sludge systems (Wentzel et al., 1986). The removal efficiency of SRP is mostly affected by the hydraulic overloading as some of the POA's get washed out with the final effluent.

Reddy, et al. (2009) also found reduced SRP removal at low temperatures, which is related to reduced biological reaction rates however, Darvill WWW SRP removal efficiency was not significantly adversely affected by temperature. In addition, it has been found that the monitoring and regulation of online parameters such as pH, DO, ORP and nutrient pollutants (phosphorus and NH_4^+–N/NO_2^- –N /NO_3^- –N) can be helpful in achieving and maintaining high nutrient removal for treating real municipal wastewater (Wang et al. 2009). Therefore, any process upsets can also affect the performance of the microorganisms in the activated sludge reactor.

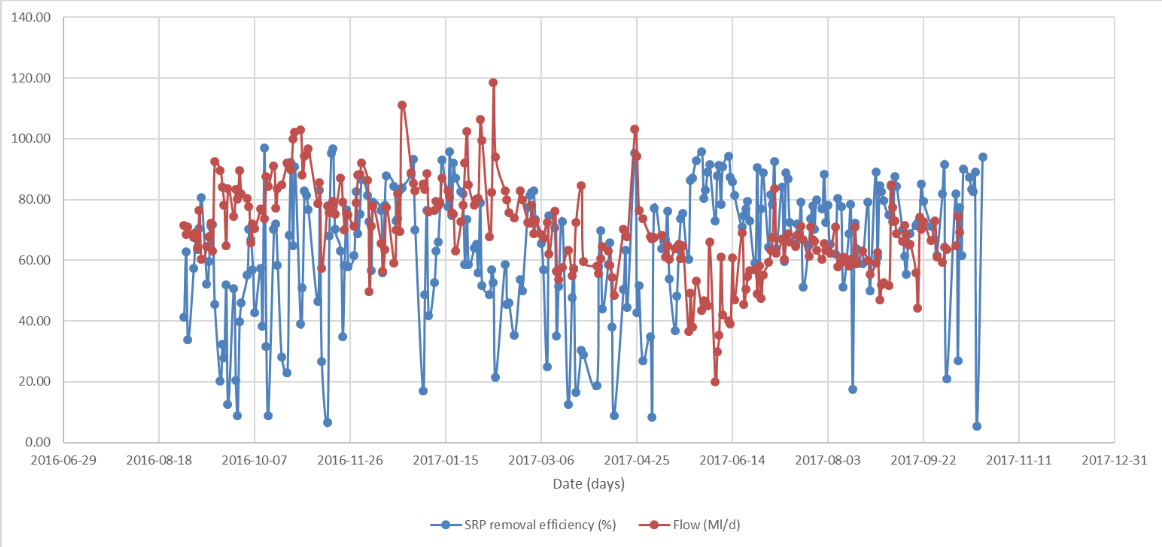


Figure 27 SRP removal efficiency and inflow comparison

The average monthly data as presented in Table 20 shows that the highest removal of SRP (80.8%) was achieved in June 2017 at a low flow of 49 ML/d. This shows that the process upset during this time had little or no effect on the POA's ability in the uptake of phosphorus in the system. This is mostly due to the fact that POAs are facultative microorganisms which means that they can survive in both the presence and absence of oxygen.

Table 20 Average monthly inflow and SRP biological removal efficiencies values

Month/Year	Inflow average (ML/d)	Removal efficiency average (%)
Sep-16	74,7	47,4
Oct-16	84,0	56,6
Nov-16	80,3	64,4
Dec-17	76,5	77,8
Jan-17	80,9	69,0
Feb-17	84,6	54,3
Mar-17	66,3	52,0
Apr-17	67,8	49,1
May-17	57,3	66,6
Jun-17	49,0	80,8
Jul-17	66,3	74,0
Aug-17	60,5	69,1
Sep-17	66,5	73,0
Oct-17	69,3	70,6

4.2.3 Summary

This part of the study was aimed at understanding the effects of inflows to the biological nutrient removal process of Darvill WWW. BNR removal was found to have an inverse proportionality relationship to influent flows – with NH_3 removal being the most affected process compared to SRP removal.

4.3 Darvill WEST Model results and discussion

The third objective of this project was to apply the WEST package for steady state modelling of Darvill WWW for optimization. This was achieved by developing a baseline model for Darvill WWW using COD & nutrient data obtained from objectives 1 & 2 and thereafter evaluating its performance, applicability and suitability for the WWW. The work undertaken on the Darvill model highlighted a number of challenges relating to modelling plants operating above capacity, receiving industrial effluents as well as general problems relating to data availability which were expressed as limitations and thus providing room for further research studies.

4.3.1 Preliminary data

The preliminary data set consisted of flow and composition data for the settled sewage and final effluent quality for the period 01 January 2016 to 30 June 2016.

Table 21 summary of measurement data

Settled Sewage	Final Effluent
COD – assumed to be total COD	COD – assumed to be total COD
NH_3 (FSA)	NH_3 (FSA)
pH	pH
SRP (orthophosphate)	SRP (orthophosphate)
Temperature	Temperature
TKN	TKN
Inflow (assumed to be settled sewage flow for now but presumably also includes primary sludge flow)	NO_3 (NO_x)
TSS	

This data was used in the probabilistic fractionator to generate an input file to test the baseline model of the activated sludge train at Darvill WWW as shown in Figure 33.

4.3.2 The Probabilistic Fractionator vs COD Fractionation

The probabilistic fractionator requires the specification of the set of fractionation parameters listed in Table 22 which are calculated using COD fractions obtained from objective 1. These parameters were used by the probabilistic fractionator to ensure the completeness and accuracy of the Darvill specific stoichiometric parameters and input values for WEST.

Table 22 Darvill WWW coefficients derived from COD fractionation data

Description	**Equation**	**Value**
f_codus	S_I/CODcomp	0.258488714
f_codup	X_I/CODcomp	0.298724239
f_codf	CODTS/CODcomp	0.571736997
f_bu	X_S/X_I	0.433639947
Soluble biodegradable fraction coefficient	f_codf - f_codus	0.313248283
Particulate biodegradable fraction coefficient	f_bu * f_Codup	0.129538763
Should be= 1	(1+f_bu)*f_Codup+f_Codf	1

4.3.3 The Probabilistic Fractionator Results

The probabilistic fractionator was used to estimate the concentrations of the model components for which there are no or incomplete measurement data and to derive coefficients (Table 22 & Table 28) which were used in WEST as well as to generate an input file (see in appendices). The discussion of the parameter estimation and completion generated by the probabilistic fractionator are presented.

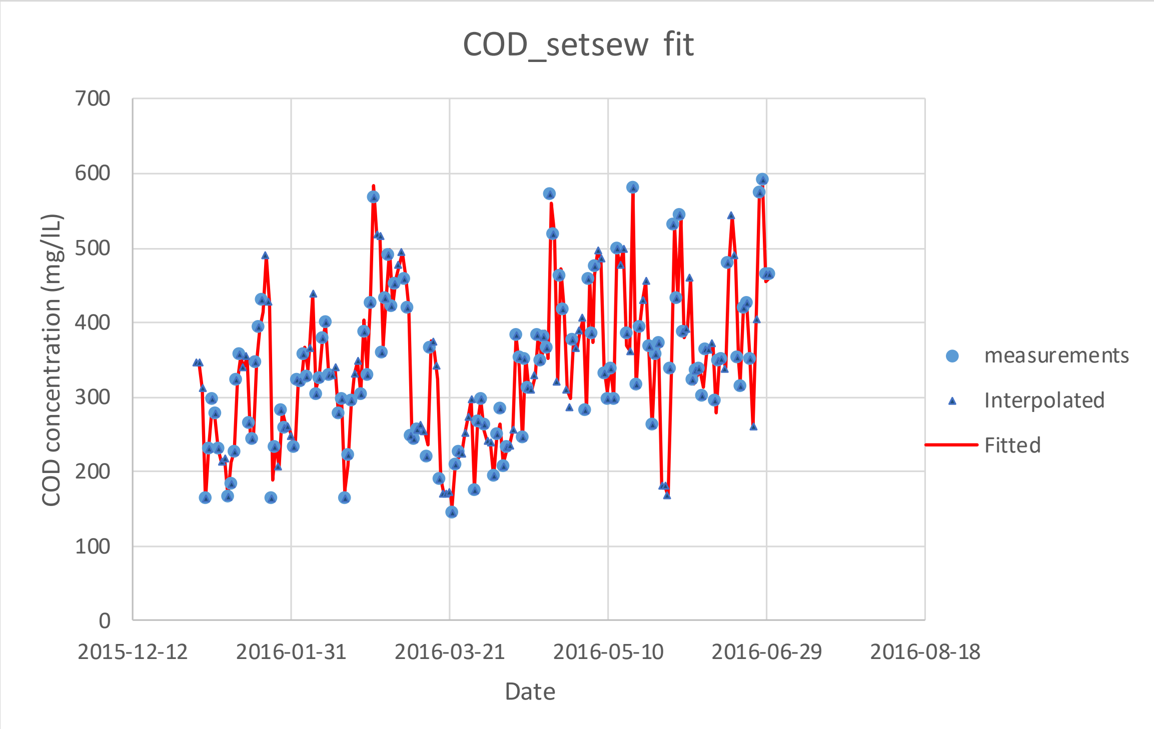


Figure 28: settled sewage COD probabilistic fractionator data

The graph in Figure 28 represents the Darvill WTP settled sewage measured, interpolated and fitted COD data from the probabilistic fractionator. COD is interpolated as it is a baseline of all other estimates and cannot be derived from anything else other than itself hence there is no estimated value on the graph. The graph shows that the inflow COD fluctuated during the period of study which was largely due to the fact that wastewater characteristics vary depending on the source of influent at the time. This is also generally unpredictable as Darvill receives both domestic and industrial effluents.

The average COD values for the period are shown in Table 23. In the probabilistic fractionator, a weighting is assigned for inferred measurements for which there are typically no actual measurements where: 0.1 is estimates, 0.5 is derived measurements and 1 is the actual measurements. The average percent (%) of objective function is a product of the overall error and weight measurements calculated by the fractionator to gauge the accuracy of the fitted value.

The average weighting of the overall fit is 0.68; this value is just above the 0.5 weighting for derived measurements as some of the data is interpolated from actual measurements. The % of objective function is 0.03 which is very close to 0. This can be seen in Table 23; the measured, estimated and inferred values are almost identical at 340.2, 340,18 and 340.18 mg/L respectively giving a fitted value of 339.76 mg/L.

Table 23 COD Averages

	Average
Weight	0.68
Measured	340.20
Estimated	340.18
Inferred	340.18
Fitted	339.76
% of objective function	0.028

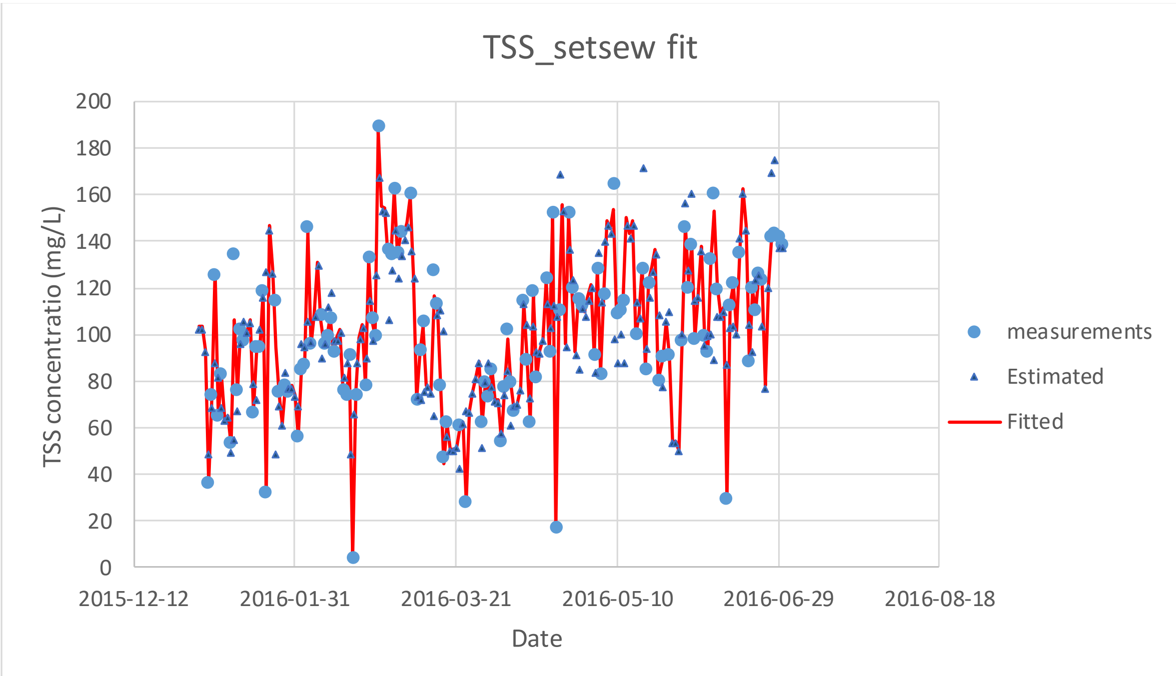


Figure 29: Settled sewage TSS probabilistic fractionator data

Figure 29 represents the Darvill WTP settled sewage measured, estimated and fitted TSS data from the probabilistic fractionator. The average TSS for the period is shown in Table 24. The average weight was 0.67 which is an indication of having estimated values in the TSS data set. There is an average % objective function of 0.0298 which shows that there was a good fit in the quality of measured, estimated and inferred TSS average data which were 100.21, 100.20 and 100.72 mg/L respectively giving a fitted value of 100.88 mg/L.

Table 24 Average TSS

	Average
Weight	0.67
Measured	100.21
Estimated	100.20
Inferred	100.72
Fitted	100.88
% of objective function	0.0298

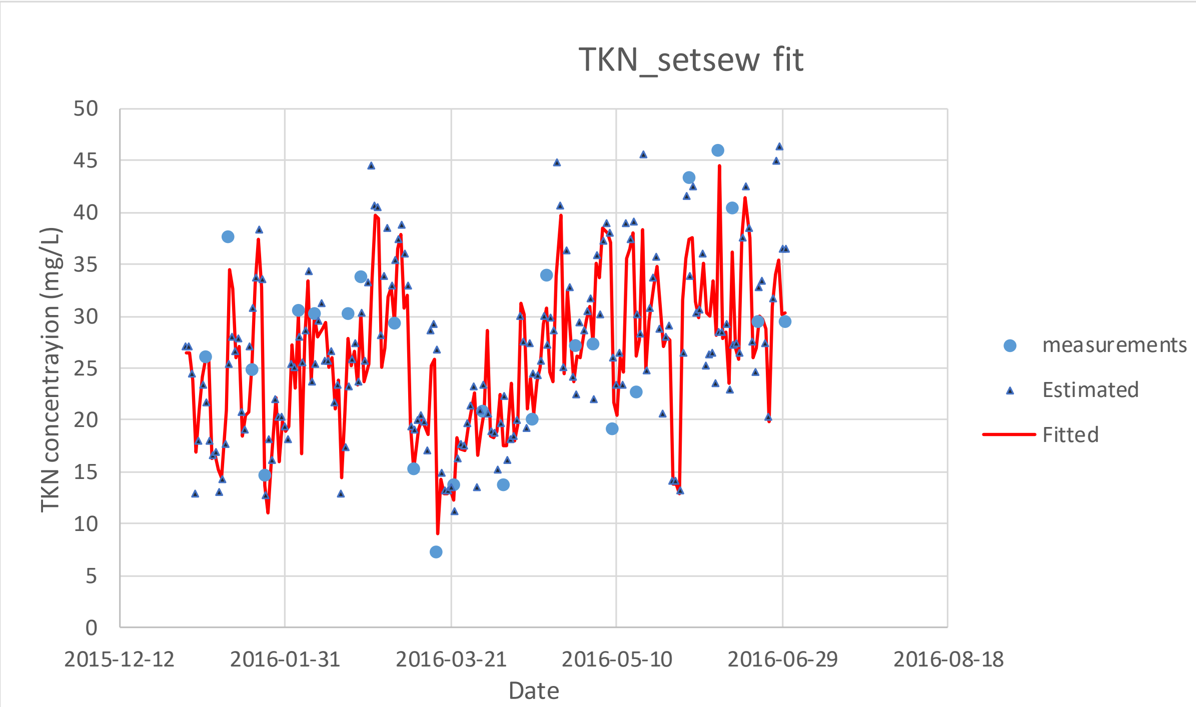


Figure 30: Settled sewage TKN probabilistic fractionator data

Figure 30 represents the Darvill WTP settled sewage measured, estimated and fitted TSS data from the probabilistic fractionator. The average TKN for the period is shown in Table 25. The overall weight for TKN had a value of 0.22 which was closer to 0.1 indicating that most of the data were estimated. The fractionator assisted with ensuring the completeness of the data. TKN can be estimated using COD data therefore its prediction is fairly possible in the case of missing or limited data. The average % objective function was 0.209 which was higher than that of COD and TSS as TKN had limited measured data however the probabilistic fractionator was able to accurately estimate as the average % objective function was close to zero. The measured, estimated and inferred values were 26.7, 26.69, 26.8 mg/L respectively and were relatively close to each giving a fitted parameter of 26.04 mg/L.

Table 25 Average settled sewage TKN data

	Average
Weight	0.22
Measured	26.695
Estimated	26.693
Inferred	26.796
Fitted	26.040
% of objective function	0.209

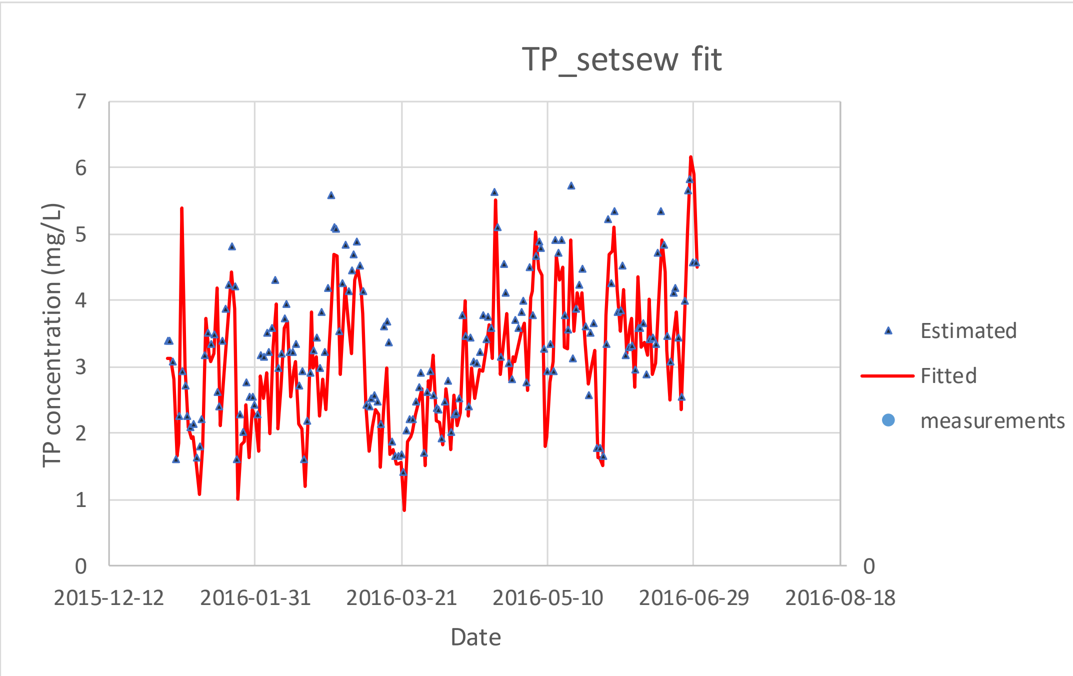


Figure 31: Settled sewage TP probabilistic fractionator data

Figure 31 represents the Darvill WTP settled sewage measured, estimated and fitted TP data from the probabilistic fractionator. The average TP for the period is shown in Table 26. The weight of TP is 0.1 as there were no measurements available for the influent at Darvill WWW. The probabilistic fractionator assisted with producing estimates based on literature and available COD data. The noted % objective function was relatively low at 0.201 due to data unavailability. Since there was no measured data, the estimated and inferred values were both 3.36 mg/L giving a fitted value of 3.07 mg/L.

Table 26 Average settled sewage TP data

	Average
Weight	0.1
Measured	Nil
Estimated	3.36
Inferred	3.36
Fitted	3.07
% of objective function	0.201

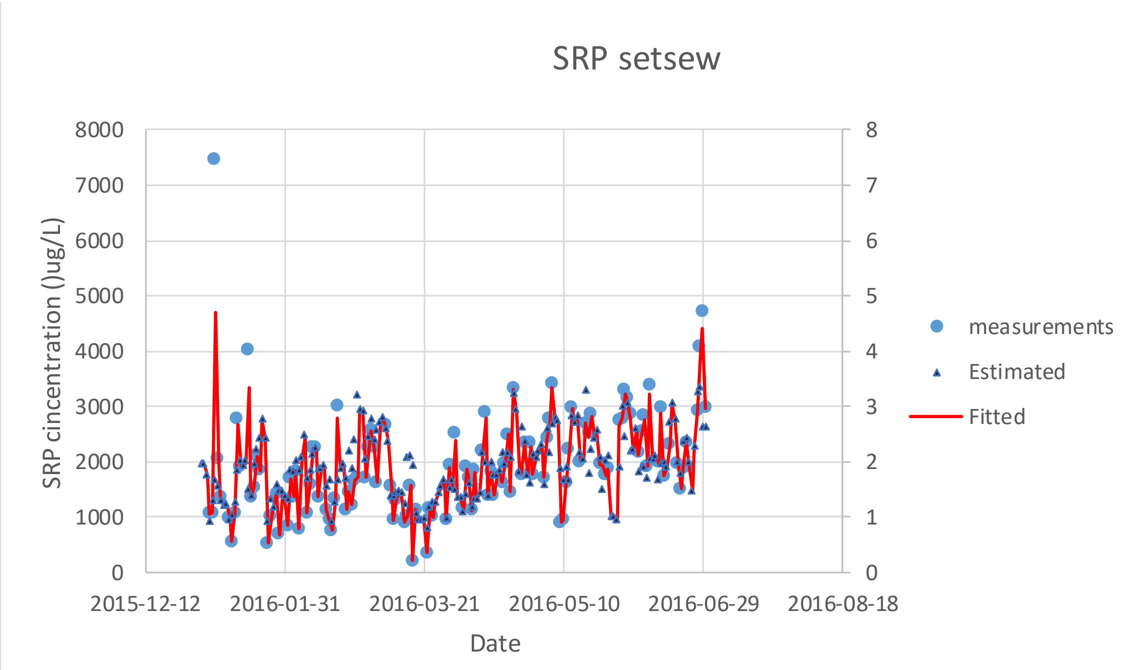


Figure 32 Settled sewage SRP probabilistic fractionator data

Figure 32 represents the Darvill WWW settled sewage measured, estimated and fitted SRP data from the probabilistic fractionator. The average SRP for the period is shown in Table 27. The weight of SRP is 0.69 as there were measurements available for the influent at Darvill WWW and some values had to be estimated. The noted % objective function was 0.051 which indicated a good fit as it was close to 0. The measured, estimated and inferred values were 2.02,1.93,1.94 mg/L respectively giving a fitted value of 1.901 mg/L.

Table 27 Average settled sewage SRP data

	Average
Weight	0.69
Measured	2.02
Estimated	1.93
Inferred	1.94
Fitted	1.901
% of objective function	0.051

4.3.4 Probabilistic Fractionator vs WEST

The PF assists in generating stoichiometric values seen in Table 28 were employed to build the Darvill WWW baseline model in WEST.

Table 28 Probabilistic Fractionator stoichiometry values for Darvill wastewater works Baseline WEST Model

Default ASM2d Stoichiometry							
Name	Type	Default Value	Adjusted value	Description	Unit	Group	
Group: Composition parameters							
i_N_BM	ConversionFactor	0.07	0.068582152	Nitrogen content of biomass X_H, X_PAO, X_AUT	g/gCOD	Composition parameters	
i_N_S_F	ConversionFactor	0.03	0.016206295	Nitrogen content of soluble substrate S_F	g/gCOD	Composition parameters	
i_N_S_I	ConversionFactor	0.01	0.02344022	Nitrogen content of inert soluble COD S_I	g/gCOD	Composition parameters	
i_N_X_I	ConversionFactor	0.02	0.067559928	Nitrogen content of inert particulate COD X_I	g/gCOD	Composition parameters	
i_N_X_S	ConversionFactor	0.04	0.023368892	Nitrogen content of particulate substrate X_S	g/gCOD	Composition parameters	
i_P_BM	ConversionFactor	0.02	0.023696707	Phosphorus content of biomass X_H, X_PAO, X_AUT	g/gCOD	Composition parameters	
i_P_S_F	ConversionFactor	0.01	0.004925751	Phosphorus content of soluble substrate S_F	g/gCOD	Composition parameters	
i_P_S_I	ConversionFactor	0	0	Phosphorus content of inert soluble COD S_I	g/gCOD	Composition parameters	
i_P_X_I	ConversionFactor	0.01	0.016867383	Phosphorus content of inert particulate COD X_I	g/gCOD	Composition parameters	
i_P_X_S	ConversionFactor	0.01	0.002161799	Phosphorus content of particulate substrate X_S	g/gCOD	Composition parameters	
i_TSS_BM	ConversionFactor	0.9	0.855190802	TSS to biomass ratio for X_H, X_PAO, X_AUT	g/gCOD	Composition parameters	
i_TSS_X_I	ConversionFactor	0.75	0.844928486	TSS to X_I ratio	g/gCOD	Composition parameters	
i_TSS_X_S	ConversionFactor	0.75	0.643804148	TSS to X_S ratio	g/gCOD	Composition parameters	
f_iss		0.313275575	0.2		g/g		

4.3.5 The Darvill Layout and model setup

The Darvill WWW baseline model plant layout is shown in Figure 33. It is a setup of the current Darvill WWW secondary treatment train. After setting up the layout of the WWW, sub-models were assigned to the different units in the layout. Design data from the Darvill WWW Operating Manual (O&M) were used to set some of the operating parameters and design data/dimensions. The Darvill WWW secondary treatment train consists of the pre-anoxic basin, anaerobic basin and two aeration basins with the DO concentrations in each being: 0.01 mg/L, 0.001 mg/L and 2 mg/L respectively (the low non-zero DO in the anoxic and anaerobic zone are to account for back mixing from the aerated zones.)

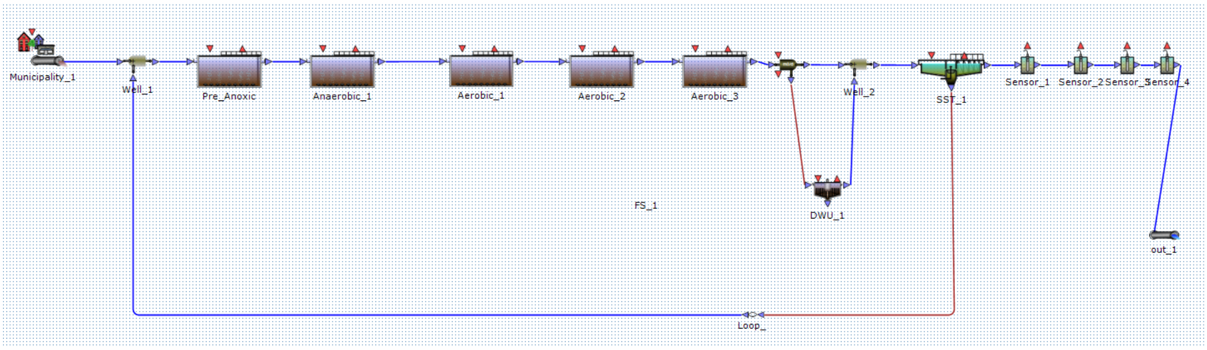


Figure 33 The Darvill WWW model layout

Table 29 Model description

Block	**Description**
Municipality 1	Settled sewage/Primary effluent from Primary Settler
Well 1	Combiner of settled sewage and recycled activated sludge
Pre Anoxic 1	First pre anoxic cell of activated sludge unit
Anaerobic 1	First anaerobic cell of activated sludge unit
Aerobic 1	First aerobic cell of activated sludge unit
Aerobic 2	Second aerobic cell of activated sludge unit
Aerobic 3	Third aerobic cell of activated sludge unit
Inlet 2	Inlet for solids concentration data
Splitter 1	Splitter of secondary clarifier underflow
Well 2	Combiner of mixed liquor/activated sludge from reactor and DAF supernatant
DWU 1	Combination of 2 Dewatering Units represented as one
SST 1	Combination of 5 secondary settlers
Multi sensor unit (1-4)	Unit measuring composition of the final effluent
Outlet 1	Outlet of the final effluent
Loop_1	Loop break introducing time delay for simulations

A flow of 3800 m^3/d of the mixed liquor from the activated sludge units (ASUs) is sent to the dissolved air flotation (DAF) dewatering unit while the rest is sent to the secondary settling tank. There are two DAF units and 5 secondary settlers which are configured as one unit since it is assumed that they operate in the same way. The secondary settler is currently modelled as a point settler and the underflow is set at 60 000 m^3/d. The return activated sludge (RAS) is mixed with 10 % of the incoming settled sewage and fed to the pre-anoxic basin where removal of the nitrates in the RAS by denitrification is supposed to occur. The rest of the settled sewage is combined with the effluent of the pre-anoxic basin in the feed to the anaerobic basin.

4.3.6 Activated sludge basins

The pre-anoxic and anaerobic basins have been modelled as unaerated activated sludge units (ASU) units while the anoxic and aerobic zones have been modelled as aerated ASU with constant DO levels and fixed volume. The relevant parameters are listed in Table 3.

Table 30 Volumes and DO levels

Unit	Volume	DO
Pre-anoxic basin	680 m^3	-
(Selector*)	680 m^3	-
Anaerobic basin	2040 m^3	0.001 mg/L
Anoxic zone	6000 m^3	0.01 mg/L
Aerobic zone	4400 m^3	2 mg/L
Aerobic zone	11800 m^3	2 mg/L

The selector was not included in the model but added to the anaerobic basin volume.

4.3.7 Secondary settler

The secondary settler is modelled as a point settler with the underflow set at 60 000 m^3/d and an assumed f_ns (fraction of suspended solids with is non-settleable) of 0.005.

4.3.8 Sludge splitting

Darvill WWW has two DAF units which were represented as one unit. There is 3 800 m^3/d of the mixed liquor sent to the DAF unit based on a safe hydraulic load (at MLSS = 4 400 mg/L) of 1.9 ML/d for each of the two DAF units.

4.3.9 DAF (Dewatering Unit, DWU_1)

The DAF unit is modelled as a simple efficiency thickening unit with a concentrate flow of 225 m^3/d and a solids removal efficiency e_X = 0.5 (50 % of the solids goes to the concentrated sludge). The concentrate flow was selected based on a float volume of 150 – 300 m^3/d. The removal efficiency was selected to give a ~3.5 % solids concentration float for MLSS = 4400 mg/L and the specified flows.

4.3.10 Model Results

A steady state simulation was run for 100 days; the plot (Figure 34) shows the mixed liquor suspended solids (MLSS) in the aerobic reactor.

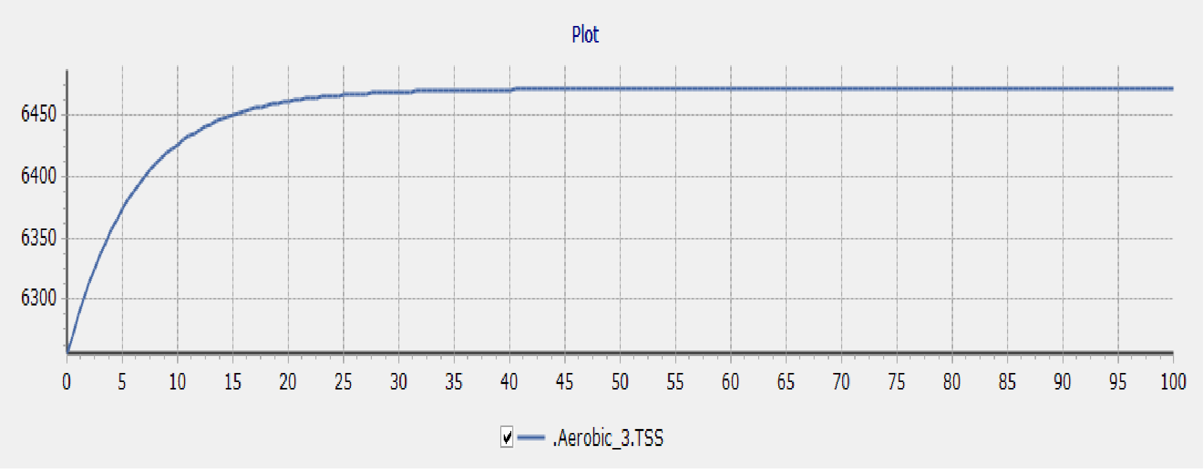


Figure 34 MLSS concentration in the activated sludge system - x-axis(time- days), y-axis (concentration – mg/L) (WEST 2017)

The plant design mixed liquor suspended solids (MLSS) concentration is 4400 mg/L and at the current state, it shows that Darvill WWW is operating above its design capacity due to nutrient overloading and poor biological nutrient removal. The model results predicted a MLSS concentration of 6475 mg/L for the plant during the study period this was comparable with the plant MLSS concentration at the time which was 6700 mg/L.

Table 31 Model results vs plant and compliance results

Determinants	Units	Influent	SteadyState Model input	Plant Effluent	Model Output	Compliance
COD	mg O_2 /L	339.76	286.498	45.1	41.28	75

NH_3	mg N/L	18.79	18.469	3.4	22.02	6
TSS	mg SS/L	100.88	-	20.9	32	25
SRP	mg P/L	1.90	1.67	6.76	2.16	1

The final effluent (FE) concentration in the defractionation model is presented in Table 31. Most of these results were expectedly non-compliant to the discharge limits imposed by the DWS with the exception of COD. The model was able to predict the COD concentration in the final effluent relatively well (45.1 mg O_2/L) as it is comparable to the plant measurement data of the FE (41.28 mgO_2/L). The NH_3 concentration in the FE was predicted to be 22.02 mg N/L and is higher than the plant measurement of 3.4 mg N/L. This can be due to the fact that the model considers TKN in the system and not just NH_3 thus over estimating the NH_3 in the FE as it was seen that the average TKN in the influent was 26 mg/L.

The TSS concentration of 32 mg SS/L in the final effluent predicted by the model is rather comparable to the plant measurement value of 20.9 mg SS/L and with further tuning of the secondary settler solids separation percentage where an optimized value can be ascertained for the pointer settler model.

The model predicted an SRP concentration of 2.16 mg/L while the plant final effluent concentration was 6.76 mg/L. This shows that there are model deficiencies. Assuming that alkalinity (S_Alk) term in the precipitation and dissolution reactions is supposed to represent the production or consumption of 3 mol OH^-, the implied change in alkalinity due to the reaction is not correct because it neglects the contribution of phosphate. For the reaction, Equation 1 was considered.

$$PO_4^{3-} + Fe(OH)_3 \rightarrow FePO_4 + 3OH^- \quad (51)$$

Two equivalents of alkalinity are consumed in the form of PO_4^{3-} because phosphate is mainly present in the form of the species $H_2PO_4^-$ at the alkalinity titration endpoint) while 3 equivalents are produced in the form of OH^- for a net production of 1 equivalent or 0.5 mol as $CaCO_3$ for each mole of phosphate precipitated. There are presumably similar issues with other biochemical transformations in the model. The precipitation/dissolution reactions are currently set up assuming MeOH = $Fe(OH)_3$ and MeP = $FePO_4$. The alum dosing unit is supposed to convert the alum dose into an equivalent number of moles of $Fe(OH)_3$ precipitate. The alum dosing unit currently assumes that 1 mol alum produces 1 mol MeOH whereas in fact Equation

2 calls to mind that;

$$Al_2(SO_4)_3 + 6H_2O \rightarrow 2Al(OH)_3 + 6H^+ + 3SO_4^{2-} \qquad (52)$$

Assuming 1 mol $Al_2(SO_4)_3$ produces 1 mol $Fe(OH)_3$ underestimates the ISS and SRP increase by ~ 30 %.

The alum dosing unit assumes alum has MW = 342 which corresponds to $Al_2(SO_4)_3$. In reality, dosing solutions at treatment plants are usually made up of hydrated alum salts ($Al_2(SO_4)_3.14H_2O$ - $Al_2(SO_4)_3.18H_2O$) and the solution concentration is expressed as a w/w % as the hydrated salt. Therefore, it is essential to know the dosing solution concentrations to see if they are calculated on the same basis when comparing plant and model data which however was not part of the scope of this study and there was limited time to explore the complex subject of quantifying the relevant uncertainty. Methods do exist for quantifying the uncertainty in mathematical modelling (Sin et al., 2011; Belia et al., 2009) and they include sensitivity analysis, scenario analysis, Gaussian error propagation and other methods (Mhlanga, 2016).

4.3.11 Summary

- Darvill WWW baseline WEST model was able to predict plant conditions (MLSS and COD) with noted uncertainties in the prediction of NH_3, SRP and TSS.
- WEST software suitability and applicability for Darvill WWW is subject to further studies being conducted accounting for the limitations presented in the baseline model. The software is mostly recommended for use in WWW concentrating on treating only domestic effluents.

5. Conclusion and Recommendations

This chapter will highlight the conclusion on the study and the recommendations for future research.

5.1 Conclusion

Darvill wastewater works COD characteristics are similar to that of domestic effluent having 70.5 % bCOD, however an industrial effluent characteristic is revealed in the inert soluble and inert particulate fractions which were 49.2% and 50.8% respectively. It is therefore concluded that Darvill WWW primary effluent has good COD characteristics having a higher bCOD component. This means that the microorganisms have sufficient substrate for the BNR process and that these components can now serve as input for the Darvill WWW WEST Model. The findings on the BNR efficiencies investigation suggest that such a plant operating above capacity becomes compromised in its ability to biologically treat the nutrients that come with the wastewater inflow. Darvill WWW has treated flows as high as 100-120 ML/d which are approximately double its design capacity. High flows reduce the nutrient removal efficiency and consequently the legal compliance in terms of the discharge limits set by DWS. This is largely due to the increased organic load and water that comes into the plant. The growth rate of bacteria and available dissolved oxygen in the biological reactor is reduced hence the poor BNR rate. Nitrification and denitrification were also predominantly influenced by temperature as there were observations of low flows but poor NH_3 removal during the winter months (June – August). Darvill WWW has the lowest ammonia removal efficiency (23.2%) in June 2016. Radical measures should be undertaken to increase aeration capacity and overall plant capacity needs upgrading to improve the BNR process. In the WEST model study, the procedure for the development of a baseline WEST model for Darvill WWW receiving domestic and industrial effluent was presented. It was based on a combination of laboratory tests and plant operating data. The Darvill WWW model was developed as a baseline model using the WEST 2017 32-bit version of the software package. The model was able to predict plant conditions (MLSS and COD) with noted uncertainties in the prediction of NH3, SRP and TSS.

5.2 Recommendations

It is recommended that COD fractionation is conducted at least every two years to observe the characteristics of the plant influents. In the case of BNR process, it is suggested that additional aeration is sort to assist the nitrification process, there should also be considerations of plant capacity upgrade to account for increase in hydraulic and nutrient loading.

Further studies can be conducted in WEST modelling to study the effects of variations in DO, temperature, and hydraulic loading. The following limitations/assumptions and recommendations for the baseline model are listed:

- **Fixed DO and Temperature**

The model does not account for changes in dissolved oxygen (DO) and temperature as these parameters are kept constant for the purpose of this study. The model assumes a temperature of 20 ºC and a DO concentration of 2 mg/L for the aerobic ASU. The model assumes that with the nutrient overload, oxygen compensation occurs within the reactor to maintain a DO of 2 mg/L. This limits the model in the prediction of actual instance where the overload would deplete the DO and where other competing reactions would give rise to greater non-compliances as well as biological growth is impaired due to cold weather conditions.

- **Fixed solids separation**

The secondary settler assumes a 0.5% fixed solids separation which does not consider hydraulic overloading. Tuning of the pointer settler model could be done and is recommended for future studies to optimize the TSS prediction in the final effluent.

- **Challenges related to overloaded plants**

When wastewater works are overloaded, it is expected that the dissolved oxygen levels in the activated sludge basins will vary with the influent loads. However, it turns out to be quite difficult to model these situations. The profile is very sensitive to the assumed oxygen transfer coefficients (Kla's), which probably also vary with changes in operating conditions and concentrations. Biological processes such as nitrification are strongly dependent on the dissolved oxygen levels therefore it is essential to get this aspect of the modelling right. Future work should focus on a more robust approach to modelling this type of situation.

- **Challenges relating to the industrial content of the influent**

The probabilistic fractionator tool equations, components, stoichiometry and typical fractions have been set up based on wastewater characterisation data for typical domestic wastewaters and may not adequately handle wastewaters with a substantial industrial component. Darvill WWW influent receives both domestic and industrial wastes in the same pipeline and there is no flow measurement or quality data of the two streams. It may be necessary to adjust the ASM2d components' stoichiometry or possibly even define new industry specific components with their own stoichiometries and degradation kinetics for some plants receiving industrial effluents. This would of course require additional and substantial modelling effort as well as additional industry specific measurements.

- **Hydraulic Overloading**

Darvill WWW was no longer operating as per initial design or as described in the operating manual. The settled sewage bypassed the anaerobic selector, aerators were installed in the anoxic zone to aid the BNR process and phosphorous removal was being achieved using chemical precipitation. This made it harder to make reasonable assumptions about the plant operation and performance with such changes in configuration. Processes with highly variable performance are inherently more difficult to model than processes that are relatively stable. Extreme situations are more likely to occur which may fall outside the range of validity of the various models and typical modelling assumptions made within WEST itself.

All the noted conditions present the possibility that the presence of industrial effluent, DO and temperature variations may affect kinetic parameters of the model such that their true values for certain ASU are different from the default values recommended in Henze et al. (1987) and those determined by the probabilistic fractionator. The different kinetic parameters model may have a strong or weak influence on the model's predictions. This type of uncertainty was not evaluated in this study due to the scope of the project and is recommended for future projects. The COD prediction was however not significantly affected. It is further recommended that WEST software package be used for a WWW that receives only domestic effluent and is operating within its design capacity to eliminate uncertainties.

Modelling requires an extensive investment in time and skill for the effort and expertise required to set up the model. It is recommended that a study that only encompasses WEST modelling is undertaken in future in a WWW.

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Appendices

A. Award certificate

Figure 35 Oral Best Paper Certificate at the 18th SOUTH AFRICA International Conference on Agricultural, Chemical, Biological and Environmental Sciences (ACBES-20). Johannesburg, South Africa. 16 -17 November 2020. Johannesburg, South Africa

B. WEST input file

Table 32 WEST input file generated by the probabilistic fractionator

	#.t	H_2O	S_NHX	S_VFA	S_PO	S_U	S_F	X_U	XC_B	x_TSS
	#d	m^3/d	g/m^3	g/m^3	g/m^3	g/m^3	g/m^3	g/m^3	g/m^3	g/m^3
2016/01/01	1	72180	19.01483	13.85194	1.937808	21.79049	123.1836	11.6872	174.4366	103.4157
2016/01/02	2	72180	19.01483	13.85194	1.937808	21.79049	123.1836	11.6872	174.4366	103.4157
2016/01/03	3	79860	17.18621	12.51982	1.751452	19.69494	111.3372	10.56327	157.6613	93.47034
2016/01/04	4	75540	13.6149	6.870372	1.055408	10.47148	78.81482	5.222251	58.5672	36.32372
2016/01/05	5	72540	15.83217	9.199258	1.077498	14.37341	75.62537	7.914441	125.2535	73.70948

2016/01/06	6	77600	17.61155	13.69299	4.692075	20.48661	13.48837	7.673721	228.8569	127.7602
2016/01/07	7	62900	20.25845	11.47482	1.999827	17.79419	124.3292	8.886512	107.4768	65.7466
2016/01/08	8	76500	20.87584	9.487766	1.369745	14.44736	56.23072	7.534288	142.7792	82.49383
2016/01/09	9	82900	11.73135	8.546059	1.195546	13.44381	75.99906	7.210511	107.62	63.80309
2016/01/10	10	81500	11.93287	8.692863	1.216083	13.67475	77.30456	7.334372	109.4687	64.89909
2016/01/11	11	74200	11.61021	6.783897	0.987333	10.5179	53.28404	5.53469	90.30208	53.00462
2016/01/12	12	93600	9.300007	5.600619	0.559347	9.849513	0	7.142691	189.8585	106.3561
2016/01/13	13	90600	15.95735	9.009082	1.06781	14.05091	67.80447	7.735701	129.1834	75.54635
2016/01/14	14	86250	27.48522	13.29697	2.686714	21.0505	99.81842	10.73556	176.537	103.3411
2016/01/15	15	83310	25.21562	14.45449	1.919566	22.4222	142.9355	11.8825	162.053	97.32498
2016/01/16	16	87440	18.71907	13.63648	1.907667	21.45156	121.2675	11.50542	171.7234	101.8071
2016/01/17	17	83800	19.53216	14.2288	1.990529	22.38334	126.535	12.00517	179.1825	106.2293
2016/01/18	18	78000	13.18902	11.79529	3.329343	17.97263	100.5076	7.391767	113.4281	67.47784
2016/01/19	19	78330	14.8581	9.742261	1.35516	15.28454	52.53905	8.334939	161.6849	93.02295
2016/01/20	20	82220	13.37954	13.50235	1.538819	21.69249	145.4013	12.19382	155.8211	94.17235
2016/01/21	21	86000	17.06474	15.8379	2.102862	24.65009	139.4792	13.06543	198.9388	117.8054

2016/01/22	22	87820	25.46325	17.29852	1.847722	27.11596	324.1845	14.36254	30.7809	32.18936
2016/01/23	23	77060	26.96882	19.64626	2.748401	30.90555	174.7118	16.57601	247.4042	146.6749
2016/01/24	24	88160	23.57324	17.17265	2.402357	27.01431	152.7143	14.48897	216.2542	128.2074
2016/01/25	25	87400	9.008181	5.008837	0.539018	9.025201	6.44E-11	6.588067	168.4711	94.47784
2016/01/26	26	81110	5.894438	8.946769	1.00821	14.59224	79.79441	8.418824	126.4812	74.71467
2016/01/27	27	91320	11.38921	8.296815	1.160678	13.05173	73.78256	7.000218	104.4813	61.94228
2016/01/28	28	28200	16.2341	11.25278	1.423397	17.70229	112.7323	9.608038	130.2304	78.18706
2016/01/29	29	71800	10.32917	9.816374	0.687621	15.93255	105.6606	9.541501	123.9808	74.63615
2016/01/30	30	71380	14.33761	10.44467	1.46115	16.43052	92.88316	8.812412	131.5291	77.97771
2016/01/31	31	75250	13.60024	9.907514	1.386006	15.58552	88.10631	8.359202	124.7647	73.96742
2016/02/01	32	75250	14.53331	9.164396	0.857089	14.44668	110.13	8.119727	90.90474	56.12273
2016/02/02	33	82100	20.53371	13.03776	1.70042	20.36893	136.0304	10.88687	141.1675	85.34285
2016/02/03	34	80890	16.26614	12.56989	1.379147	19.9797	133.5167	11.17407	144.2341	87.09979
2016/02/04	35	84450	21.83128	14.05249	1.824348	22.35505	67.09932	12.61779	251.0854	143.9237
2016/02/05	36	99700	9.59239	12.24373	0.788652	20.11144	133.1827	12.24784	158.6111	95.45091
2016/02/06	37	89180	20.11891	14.65624	2.050325	23.05574	130.3361	12.36581	184.5651	109.4204

2016/02/07	38	74440	24.10269	17.55834	2.456314	27.62105	156.1442	14.81439	221.1112	131.0869
2016/02/08	39	68610	16.77014	11.70945	1.073142	18.66356	86.35593	10.88113	182.8552	106.6734
2016/02/09	40	36000	22.99701	12.91035	1.586962	20.34811	120.0861	11.21935	161.5169	96.11476
2016/02/10	41	82000	20.22818	15.26074	2.236195	23.97201	157.739	12.61976	164.9204	99.58949
2016/02/11	42	78170	20.48014	16.04575	2.253058	25.30272	164.5543	13.49856	178.4263	107.5205
2016/02/12	43	79020	22.48962	13.04746	1.370751	20.42102	131.5648	11.33443	153.1131	91.95576
2016/02/13	44	79010	18.10839	13.19161	1.845432	20.75174	117.3114	11.13007	166.1212	98.48581
2016/02/14	45	76320	18.74665	13.65657	1.910477	21.48316	121.4462	11.52237	171.9763	101.9571
2016/02/15	46	73870	15.28424	10.87857	1.13879	17.24152	114.567	9.680054	126.0334	76.02105
2016/02/16	47	65880	17.58837	11.59466	0.949251	18.39231	138.6023	10.56159	120.3507	74.03302
2016/02/17	48	87610	10.26592	6.269146	0.771476	10.00395	0	5.913542	152.4919	85.50461
2016/02/18	49	109760	17.05304	9.314756	1.316028	14.24962	181.7401	3.690152	0	4.333501
2016/02/19	50	110440	21.89781	12.39378	2.79536	19.52939	121.3131	9.411365	125.1672	75.40478
2016/02/20	51	98810	18.20733	13.26369	1.855515	20.86512	117.9523	11.19088	167.0288	99.02389
2016/02/21	52	93620	19.21669	13.99898	1.958379	22.02181	124.4912	11.81127	176.2884	104.5135
2016/02/22	53	95910	17.05802	11.86275	1.139683	18.80922	134.8615	10.63604	127.9696	78.08063

2016/02/23	54	73780	21.41773	14.48889	1.455681	24.25684	123.5677	14.84456	225.5588	132.6474
2016/02/24	55	77890	16.47432	12.74211	1.233229	20.368	110.1935	11.76009	180.1973	106.2782
2016/02/25	56	81560	16.62066	16.51586	1.671305	26.57034	208.4725	15.05843	159.7714	99.38229
2016/02/26	57	71050	20.79017	21.49008	1.730823	34.95332	189.0423	20.90792	317.4025	187.1835
2016/02/27	58	77610	28.5666	20.81021	2.911232	32.73656	185.0626	17.55806	262.0617	155.3647
2016/02/28	59	77960	28.43835	20.71678	2.898162	32.58959	184.2318	17.47924	260.8852	154.6672
2016/02/29	60	78160	16.92448	14.08121	1.734174	22.47004	86.31853	12.68998	232.6267	134.4741
2016/03/01	61	79890	17.61567	17.04062	2.274696	27.22584	149.9165	14.9893	226.6386	133.9881
2016/03/02	62	76750	21.02849	19.35194	2.568484	30.88408	154.2319	17.07196	275.4583	161.5658
2016/03/03	63	75100	23.6449	16.80219	2.264413	26.45313	136.1923	14.35756	229.35	134.8007
2016/03/04	64	76070	19.41135	17.52553	1.637814	27.9373	156.633	16.06763	241.7214	142.9537
2016/03/05	65	72080	26.25222	19.12424	2.675373	30.08435	170.0695	16.13557	240.8304	142.7775
2016/03/06	66	69400	27.266	19.86275	2.778687	31.24611	176.637	16.75867	250.1304	148.2911
2016/03/07	67	89800	20.49581	18.24738	2.668526	28.99152	127.0775	15.75101	274.102	159.4535
2016/03/08	68	44600	23.11041	16.83932	2.354808	26.48909	150.1517	14.2045	212.1959	125.7868
2016/03/09	69	44560	14.2425	10.03276	1.556618	15.68066	89.44542	8.171815	121.8546	72.30883

2016/03/10	70	63460	9.614915	9.4079	0.951514	15.08919	59.77419	8.766888	158.1211	91.52774
2016/03/11	71	88640	13.04457	10.06072	1.310265	15.98227	47.11367	8.968785	180.388	103.3279
2016/03/12	72	86500	14.43719	10.51721	1.471299	16.54464	93.52827	8.873618	132.4426	78.5193
2016/03/13	73	89540	13.94703	10.16014	1.421346	15.98292	90.35287	8.572347	127.946	75.85346
2016/03/14	74	93700	12.89419	7.966114	0.914924	12.95339	4.55E-10	8.021432	207.4022	116.2888
2016/03/15	75	88160	17.12916	13.95519	1.031967	22.4899	136.3415	13.38124	188.3385	112.1989
2016/03/16	76	86300	18.51417	14.70227	1.578598	23.39137	186.8481	13.02274	123.2726	78.3509
2016/03/17	77	94100	2.848147	13.1918	0.215373	13.733	228.2885	13.89376	54.46011	44.73603
2016/03/18	78	90500	10.20835	7.635908	1.136376	12.01102	58.6542	6.405115	105.8468	61.98156
2016/03/19	79	101300	9.341611	6.805182	0.952007	10.70524	60.51765	5.741692	85.69725	50.80606
2016/03/20	80	101300	9.341611	6.805182	0.952007	10.70524	60.51765	5.741692	85.69725	50.80606
2016/03/21	81	99690	9.492478	6.915086	0.967382	10.87813	61.49501	5.83442	87.08127	51.62658
2016/03/22	82	98860	8.862389	5.263679	0.36501	8.838451	28.92357	5.680583	103.7316	59.73699
2016/03/23	83	88420	13.84595	8.433028	1.149949	13.1356	74.59777	7.000241	105.1159	62.30671
2016/03/24	84	97840	12.96113	9.056344	0.999945	14.23236	150.8478	7.609287	37.92282	28.21373
2016/03/25	85	98560	12.34677	8.994379	1.258263	14.14907	79.98591	7.588769	113.2657	67.15015

2016/03/26	86	87570	13.89628	10.12317	1.416175	15.92477	90.02411	8.541156	127.4805	75.57747
2016/03/27	87	80900	15.04199	10.9578	1.532935	17.23773	97.44637	9.245353	137.9909	81.80864
2016/03/28	88	74650	16.30137	11.87523	1.661278	18.68094	105.605	10.01941	149.5441	88.65799
2016/03/29	89	75010	12.76556	7.017383	0.953063	10.87848	44.99644	5.846986	106.2575	61.59312
2016/03/30	90	81560	13.17992	10.9443	1.894773	17.1688	94.24047	8.778549	134.4014	79.49884
2016/03/31	91	67110	14.68853	11.93403	1.549457	18.77915	134.8908	10.05733	119.868	73.36752
2016/04/01	92	66770	23.13288	11.31978	2.384217	16.99576	71.18113	7.840454	147.8556	85.69885
2016/04/02	93	72750	13.28413	9.677232	1.35379	15.22326	86.05843	8.164907	121.8648	72.24818
2016/04/03	94	73310	13.18266	9.603309	1.343449	15.10697	85.40105	8.102537	120.9339	71.69629
2016/04/04	95	78750	14.89265	7.98777	1.154336	12.29238	74.93931	6.365306	90.75485	54.22231
2016/04/05	96	89940	17.21195	10.42725	1.858569	16.05039	80.74461	7.991058	131.9279	77.42838
2016/04/06	97	94560	11.70989	13.35434	1.714395	17.02036	55.89221	6.204262	172.4203	98.11163
2016/04/07	98	99810	12.85225	8.259507	1.125131	12.95096	45.95268	7.078651	135.7454	78.18962
2016/04/08	99	90090	18.657	9.815339	1.798396	14.92254	80.88085	7.230775	114.3203	67.52449
2016/04/09	100	89100	12.96534	9.444997	1.321302	14.85793	83.9932	7.968965	118.9403	70.51436
2016/04/10	101	81870	14.11032	10.27909	1.437987	16.17005	91.4107	8.672711	129.444	76.74154

2016/04/11	102	76580	23.02105	15.43568	2.185648	24.15304	136.7477	12.84652	192.7193	114.2252
2016/04/12	103	79050	23.1202	14.76982	2.781582	22.70192	144.057	10.97745	149.4323	89.98154
2016/04/13	104	69800	16.12304	9.94414	1.384772	15.46301	105.5467	8.11203	102.7148	62.35225
2016/04/14	105	60950	16.23193	13.81246	1.817241	21.97993	105.5493	12.15078	200.9591	117.5437
2016/04/15	106	69820	14.20568	12.54145	1.397825	19.4851	134.9475	10.48357	133.4927	81.00306
2016/04/16	107	70350	17.09579	12.45395	1.742238	19.59132	110.7514	10.50769	156.8318	92.97857
2016/04/17	108	66500	18.08554	13.17497	1.843104	20.72555	117.1634	11.11603	165.9116	98.36154
2016/04/18	109	64750	20.79666	15.0021	1.6345	23.81146	126.602	13.45823	209.4349	123.365
2016/04/19	110	66130	23.47672	13.64874	1.971762	22.18805	148.0236	12.37071	153.3401	92.90311
2016/04/20	111	66690	15.88299	15.21475	2.487493	24.17847	71.91865	12.99486	263.1476	150.7109
2016/04/21	112	68060	17.44016	14.53113	1.449519	23.04275	295.2895	12.38176	6.055532	17.06998
2016/04/22	113	66190	22.70225	22.92863	3.300373	36.42638	309.4111	19.17084	172.908	111.0479
2016/04/23	114	50460	28.55787	20.80859	2.90987	32.73294	185.5446	17.5527	262.2136	155.4365
2016/04/24	115	81720	17.63662	12.84793	1.797354	20.2111	114.2551	10.8401	161.7932	95.91997
2016/04/25	116	73200	22.2444	17.92693	1.778734	28.69845	152.6173	16.55553	256.2945	150.9615
2016/04/26	117	50800	19.16921	16.66457	2.335964	26.43138	159.0982	14.25272	201.7457	120.3748

2016/04/27	118	68580	17.04008	12.41336	1.73656	19.52748	110.3905	10.47345	156.3208	92.67558
2016/04/28	119	73880	19.42942	11.75259	2.271195	18.5	56.98749	9.500973	202.0446	115.2071
2016/04/29	120	72840	17.83053	14.78998	1.769655	23.54951	139.4932	13.10736	188.1007	111.9496
2016/04/30	121	74900	20.12357	14.65963	2.0508	23.06108	130.3663	12.36867	184.6078	109.4457
2016/05/01	122	70450	21.39468	15.58561	2.180339	24.51774	138.6009	13.14994	196.2687	116.3589
2016/05/02	123	67510	22.3264	16.26435	2.275291	25.58546	144.6369	13.72261	204.816	121.4262
2016/05/03	124	66340	21.26582	11.42896	1.712909	17.80782	87.3952	9.424484	155.531	91.11245
2016/05/04	125	66380	25.46468	18.31278	2.421558	28.82563	180.0783	15.55133	214.2614	128.358
2016/05/05	126	60930	26.07214	16.03474	2.724516	24.67907	185.0483	12.16147	135.1094	83.89592
2016/05/06	127	68880	28.86304	19.62348	3.339164	30.42617	204.5206	15.23414	194.5923	118.1076
2016/05/07	128	65810	27.36049	19.93159	2.788317	31.3544	177.2492	16.81675	250.9973	148.805
2016/05/08	129	67380	26.72297	19.46717	2.723347	30.62382	173.1191	16.42491	245.1489	145.3378
2016/05/09	130	90850	13.71476	13.39201	0.915043	19.7087	23.01377	10.80148	272.7271	153.8117
2016/05/10	131	84080	13.71014	11.36583	0.952262	18.32252	82.50639	10.90688	184.3561	107.4018
2016/05/11	132	71630	18.58648	13.39444	1.629932	21.18437	109.6919	11.75052	186.4234	109.602
2016/05/12	133	58940	17.86257	12.21546	2.185267	18.99708	59.6959	9.733461	198.1188	113.5741

2016/05/13	134	67810	24.92458	19.99964	2.966845	31.5465	175.2138	16.78989	254.1355	150.396
2016/05/14	135	70610	26.32	19.17361	2.68228	30.16202	170.5086	16.17722	241.4521	143.1461
2016/05/15	136	67760	27.42703	19.98006	2.795098	31.43064	177.6802	16.85764	251.6076	149.1669
2016/05/16	137	63690	18.49658	16.61374	2.00442	23.6248	151.9094	10.87491	165.81	99.25157
2016/05/17	138	67630	19.95362	14.53582	2.03348	22.86632	129.2653	12.26421	183.0487	108.5214
2016/05/18	139	67500	26.63937	22.96641	2.696706	36.54725	292.2447	20.04505	205.5311	128.7734
2016/05/19	140	76740	18.42142	13.17942	2.444706	20.42362	122.8237	10.05478	143.6173	85.83623
2016/05/20	141	71880	21.21686	16.10354	2.817077	25.15035	126.8446	12.8086	208.8164	122.6045
2016/05/21	142	65620	23.69155	17.25883	2.414414	27.14989	153.4807	14.56168	217.3394	128.8508
2016/05/22	143	62000	25.07483	18.26653	2.555384	28.73509	162.442	15.4119	230.0293	136.374
2016/05/23	144	50400	23.56729	14.85895	1.964352	23.15259	181.3214	12.1878	129.0253	80.57955
2016/05/24	145	62730	21.34011	10.97755	1.934107	16.72813	68.15744	8.320146	155.6157	90.09935
2016/05/25	146	59690	20.42459	14.1913	1.757217	22.36043	132.2023	12.24596	177.0369	105.3449
2016/05/26	147	35310	20.01944	14.84394	1.871753	23.41274	169.3209	12.65781	149.2731	91.4678
2016/05/27	148	72610	9.955733	7.252558	1.014592	11.40901	64.49611	6.119154	91.33103	54.14608
2016/05/28	149	72610	9.955733	7.252558	1.014592	11.40901	64.49611	6.119154	91.33103	54.14608

2016/05/29	150	78000	9.267766	6.751388	0.944481	10.62061	60.03927	5.696304	85.01983	50.40445
2016/05/30	151	66790	24.4649	14.23196	2.657193	21.77623	118.9662	10.58007	165.454	97.82258
2016/05/31	152	68420	24.31605	21.08414	2.791393	33.4935	214.7004	18.21867	243.6934	146.4968
2016/06/01	153	82600	28.18906	17.1625	3.213578	28.10562	168.6153	15.12539	203.4985	121.8364
2016/06/02	154	70310	26.27852	21.8113	3.133647	34.44337	235.4424	18.27102	228.6168	138.871
2016/06/03	155	76490	23.51288	16.05958	2.794567	24.88609	162.2715	12.4038	163.7662	98.93214
2016/06/04	156	75830	21.53894	15.6907	2.195041	24.68306	139.5355	13.23861	197.5921	117.1435
2016/06/05	157	64470	25.33423	18.4555	2.58182	29.03236	164.1225	15.57134	232.409	137.7849
2016/06/06	158	75560	23.53159	13.26535	2.135972	20.41731	105.2837	10.41233	168.9362	99.31376
2016/06/07	159	81540	23.16552	13.99646	2.479501	21.52026	126.8422	10.63242	155.6661	92.76513
2016/06/08	160	76850	25.75335	14.16721	2.75326	21.62642	58.55632	10.62167	230.7816	131.6487
2016/06/09	161	64400	20.60005	11.85832	1.926222	18.59342	0	10.24795	272.6154	152.6544
2016/06/10	162	66820	36.70131	15.31678	3.231677	23.62912	99.583	11.51622	207.4519	120.4505
2016/06/11	163	66820	20.03228	14.59313	2.041497	22.95646	129.7749	12.31256	183.7704	108.9492
2016/06/12	164	65270	20.508	14.93968	2.089977	23.50162	132.8567	12.60496	188.1345	111.5365
2016/06/13	165	73876	18.31999	12.24165	1.940854	18.91532	202.9443	9.248961	35.97455	29.26257

2016/06/14	166	68780	28.64357	14.1946	2.881165	22.63615	103.8877	11.67179	194.256	113.4849
2016/06/15	167	72450	19.09112	13.83824	1.750388	21.88395	98.81548	12.1273	208.0793	121.2269
2016/06/16	168	74790	18.65915	13.59283	1.901561	21.38289	120.8794	11.46859	171.1737	101.4812
2016/06/17	169	73100	27.5854	19.05103	2.329132	30.00511	188.1577	16.41069	225.6661	135.1891
2016/06/18	170	64490	29.88062	21.76745	3.045144	34.24239	193.5752	18.36571	274.1162	162.5112
2016/06/19	171	71300	27.02666	19.6884	2.754296	30.97184	175.0865	16.61156	247.9348	146.9894
2016/06/20	172	74080	18.73274	14.15625	1.97267	22.27324	155.1129	11.82384	145.3371	88.53653
2016/06/21	173	73400	19.83336	12.42602	1.522814	19.52798	71.36168	10.88303	205.489	118.5102
2016/06/22	174	62220	21.23273	16.69879	1.899956	26.14739	178.7147	14.228	181.7684	110.123
2016/06/23	175	64680	20.5383	16.97897	2.348675	26.88966	155.9363	14.53878	212.4749	126.2492
2016/06/24	176	66620	20.9386	13.98125	1.905708	21.94877	94.93877	11.93336	210.4859	122.3234
2016/06/25	177	89600	14.32167	10.43306	1.459526	16.41226	92.77992	8.802617	131.3829	77.89103
2016/06/26	178	57500	22.3169	16.25743	2.274323	25.57458	144.5753	13.71677	204.7288	121.3745
2016/06/27	179	65680	22.10194	22.67206	2.921086	36.26579	261.1787	19.83215	233.1267	142.7721
2016/06/28	180	62490	23.41291	24.02567	4.015856	38.02139	265.3291	19.5507	236.3377	144.382
2016/06/29	181	73300	20.4201	19.58911	4.413107	30.49345	144.5268	14.388	245.8218	143.7492

2016/06/30	182	78310	20.51717	19.08874	2.954669	29.42734	162.3809	14.95051	233.7546	138.1468

C. Publication

Table 33 Biological nutrient removal efficiencies for hydraulically overloaded wastewater works

BIOLOGICAL NUTRIENT REMOVAL EFFICIENCIES FOR HYDRAULICALLY OVERLOADED WASTEWATER WORKS

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ABSTRACT

In this current age the environmental laws have become more stringent towards health, economy and reduction of pollution at the point of sources. Results of a comprehensive study are reported for Darvill wastewater works (WWW) inflow in relation to biological nutrient removal process. The incoming and outgoing nutrient (ammonia and soluble reactive phosphorus (SRP)) concentrations were determined using standard testing methods. Calculations of biological nutrient removal (BNR) efficiencies were used to evaluate the effects of high inflow to the biological treatability of the activated sludge for the period 2016–2017. At inflows above design capacity the nutrient removal efficiency was found to be at an average of 40% and SRP removal efficiency being 64% for the period of the study. The nutrient removal efficiency had an inversely proportional relationship to the inflow into the plant with ammonia removal being mostly affected. When the plant is overloaded the BNR process is adversely affected.
Keywords: biological nutrient removal, hydraulically overloaded plants, nutrient removal efficiency.

1 INTRODUCTION

Wastewater treatment is a critical chain in the urban water cycle. Nature has its own means of attenuating waste however in larger volumes and concentrations the process becomes slower and more problematic. This has led to the design of wastewater works (WWW) which are intended to aid in the faster biodegradation and removal of pollutants found in waste water [1]. The purpose of a WWW can be elaborated as the treatment of municipal and industrial wastewater (WW) to acceptable effluent quality, devoid of excess organics chemical oxygen demand (COD), ammonia (NH_3), and nitrogen (N) and phosphorus (P) nutrients.

When these nutrients are released into a water resource they cause an unnatural stimulation of organisms such as algae and certain aquatic vegetation such as water hyacinths. This in turn can affect the suitability of water for farming, recreation or potable (drinking) use. It is for this reason that much attention has been given lately to biological nutrient removal (BNR)+ [2]. A new deal for wastewater engineers is now to stretch the performance of existing infrastructure, which represents one of the most significant challenges to the practice of wastewater engineering [3]. Traditionally, the complexity associated with implementing BNR in WWW has been primarily in terms of balancing competing requirements for nitrogen and phosphorus removal, particularly with respect to the use of influent COD as a carbon source for the microorganisms [4].

The nitrogen that enters the wastewater system is found as NH_3, ammonium ions (NH^{4+}), nitrites (NO^{2-}) and nitrates (NO^{3-}) in wastewater. Ammonia and ammonium ions are the most reduced forms of nitrogen and are often products of organic decomposition. Nitrite is, however, the intermediate, with nitrate being the end product of organic oxidation during nitrification [5]. Ammonia and Ammonium ions are both commonly referred to as "ammonia nitrogen". This form of ammonia will be observed in this study. Nitrification is mediated by specific chemical autotrophic nitrifying bacteria (Fig. 1). Henze et al. [6] infer that these bacteria obtain their carbon requirement (anabolism) from dissolved carbon dioxide (CO_2)

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