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Study of Process Control Strategies for Biological Nutrient Removal in an Oxidation Ditch

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Study of Process Control Strategies for Biological Nutrient Removal in an Oxidation Ditch

by

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ABSTRACT

Advanced wastewater treatment plants must meet permit requirements for organics, solids, nutrients and indicator bacteria, while striving to do so in a cost effective manner. This requires meeting day-to-day fluctuations in climate, influent flows and pollutant loads as well as equipment availability with appropriate and effective process control measures. A study was carried out to assess performance and process control strategies at the Falkenburg Road Advanced Wastewater Treatment Plant in Hillsborough County, Florida.

Three main areas for control of the wastewater treatment process are aeration, return and waste sludge flows, and addition of chemicals. The Falkenburg AWWTP uses oxidation ditches where both nitrification and denitrification take place simultaneously in a low dissolved oxygen, extended aeration environment. Anaerobic selectors before the oxidation ditches help control the growth of filamentous organisms and may also initiate biological phosphorus removal. The addition of aluminum sulfate for chemical phosphorus removal ensures phosphorus permit limits are met. Wasting is conducted by maintaining a desired mixed liquor suspended solids (MLSS) concentration in the oxidation ditches.

For this study, activated sludge modeling was used to construct and calibrate a model of the plant. This required historical data to be collected and compiled, and supplemental sampling to be carried out. Kinetic parameters were adjusted in the model to achieve simultaneous nitrification-denitrification. A sensitivity analysis found maximum specific growth rates of nitrifying organisms and several half saturation constants to be influential to the model.

Simulations were run with the calibrated model to observe relationships between sludge age, MLSS concentrations, influent loading, and effluent nitrogen concentrations.

Although the case-study treatment plant is meeting discharge permit limits, there are several recommendations for improving operation performance and efficiency. Controlling wasting based on a target MLSS concentration causes wide swings in the sludge age of the system. Mixed liquor suspended solids concentration is a response variable to changes in sludge age and influent substrate. Chemical addition for phosphorus removal could also be optimized for cost savings. Finally, automation of aeration control using online analyzers will tighten control and reduce energy usage.

CHAPTER 1: INTRODUCTION

1.1 Background

Amendments to the Clean Water Act in 1972 established a system of permitting for point source discharges to surface water bodies in the United States (EPA, 2002). This permitting system, known as the National Pollutant Discharge Elimination System (NPDES), applies to wastewater treatment plants (WWTPs) that treat municipal and industrial wastewater. Generally, the issuing of permits is the responsibility of state regulatory agencies, and each WWTP must apply for and receive a specific permit tailored to the individual facility based on the characteristics of the receiving water body. At a minimum, most WWTPs must meet permit requirements for organics, solids, and indicator bacteria. Stricter permits limit the amount of nutrients, namely nitrogen (N) and phosphorus (P), that may be discharged, and these stricter permits require design and operation of advanced wastewater treatment plants (AWWTPs) that have additional treatment technologies.

Complying with permit limits requires meeting day-to-day fluctuations in influent flows, pollutant loads, temperature, and equipment availability with effective process control measures. Design engineers strive to create appropriate and robust treatment systems; however, the performance of WWTPs is ultimately dependent on the operating practices and decisions made by treatment plant operators and managers. In addition to legally complying with NPDES permits, many WWTPs are focusing on reducing carbon footprints and even becoming energy neutral or net-energy positive (Schwarzenbeck et al., 2008; Mo and Zhang, 2012; Gori et al., 2013; Jenicek et al., 2013). As the level of treatment needed to meet stricter N and P limits increases, the emissions of greenhouse gases and other air pollutants associated with energy and chemical usage also increases (Falk et al., 2011). Process monitoring and control is critical to efficient operation that will save energy and decrease operating costs while ensuring that the requirements of discharge permits are met.

Throughout 2013 and 2014, special conferences are being held to celebrate the 100th anniversary of the activated sludge process for the treatment of wastewater. The term activated sludge refers to wastewater that has been aerated to allow for the growth of microorganisms that consume soluble organic matter (Grady et al., 1999). Modifications to the activated sludge process can be made to achieve biological nutrient removal (BNR) of N and P. The following excerpt is from a study published in 1914 by Ardern and Lockett, who are credited with the development of the activated sludge process.

> *"The main feature of the experimental work was the satisfactory purification of sewage by tank treatment alone, with the production of a sludge, which owing to its oxidised and flocculent condition, could be readily dealt with and turned into a valuable fertilising agent." –(Ardern and Lockett*, 1914)

This excerpt highlights two important functions of the activated sludge process that operators are attempting to control: transformation of wastewater constituents through oxidation (and reduction) and the ability of activated sludge bacteria to flocculate and settle. Mixed liquor suspended solids (MLSS) is the term given to the solids in the biological treatment system and refers to the mixture of newly formed solids and settled solids that are returned to the reactor. Generally, there are three main areas where the treatment plant operator makes adjustments to control the activated sludge process: 1) aeration and mixing, 2) return and waste sludge flows, and 3) chemical addition. Operators collect samples, perform tests, use readings from online analyzers and meters, and analyze data to determine how the plant is performing and what actions need to be taken to achieve desired performance. While knowledge of the activated sludge processes has greatly increased over the past 100 years, continual efforts to minimize plant upsets and increase efficiency are still needed.

Because the operator is ultimately responsible for the performance and efficiency of the WWTP, operator training and the information disseminated for process control is paramount. The Office of Water Programs (OWP) at California State University of Sacramento received a federal grant in 1968 to establish a correspondence training program for wastewater treatment plant operators (Austin et al. 1970). For the past four decades, OWP has been providing correspondence courses and training manuals, colloquially known as “the Sacramento Manuals”, for all levels of WWTP operator. The Sacramento Manuals have been widely used over the years, with most states listing them as approved training material to qualify operators to sit for required certifying exams. The content of these manuals has gone largely unchanged since their inception, and while much effort has been made to be operator-friendly, some information is contradictory to other literature and reference texts. For example, Volume II of the *Operation of Wastewater Treatment Plants* manual (OWP, 2008) states: “Usually, it is necessary to vary the amount of MLSS in the ditch as seasons change. Because the microorganisms are not as active in winter at low temperatures, the MLSS will need to be higher in the winter than in the summer if complete nitrification is desired.” In contrast, other reference materials and researchers stress that the nitrifying biomass is dependent on only two variables: mean cell residence time (MCRT) and the average ammonia load (Grady et al. 1999; Rieger et al., 2014). Another Sacramento Manual, *Advanced Waste Treatment* (OWP, 2006), states “The operator is usually working with a fixed reactor volume and will need to determine the desired MLSS concentration and overall MCRT to meet one or more treatment objectives”. This implies that the MCRT and the MLSS concentration may be controlled at the same time. Neglecting the MCRT by focusing on increasing MLSS concentration may result in poor sludge quality leading to negative effects on sludge flocculation, settling, and compaction.

Advancements in computer technology, data storage, and sensor capability have made it possible to easily store and retrieve large quantities of data. Some data collection and reporting is mandatory for permitting requirements, while other data are collected for in-plant process control purposes. Data collection is an attempt to come as close as possible to understanding the processes occurring in the WWTP. Data must also be as accurate as possible to be truly representative. Potential error in data collection can be determined using mass balances (Puig et al, 2008), comparisons of parameter ratios and ranges, and statistical tests. Graphing of data can also be used to quickly visualize outlying values. While sampling and laboratory analysis are essential to the successful operation, time and cost factors must be considered when choosing the appropriate sampling regimen.

Technological advancements have also contributed to the development and increased use of mathematical models of the activated sludge system, which can be powerful tools for the design and operation of WWTPs. Mathematical models use equations that represent the uptake and conversion of substrates by bacteria, as well as physical processes such as sedimentation and chemical precipitation. The International Water Association (IWA) published Activated Sludge Model 1 (ASM1) in 1987, which set the stage for the evolution of more complex models that simulate nitrogen and phosphorus removal processes. An indirect benefit of modeling is that the use of ASM models has highlighted existing gaps in research and helped to guide scientific investigation of wastewater treatment processes. In addition to the knowledge gained from running simulations to test varying conditions, the need for ample and accurate data used in model construction and calibration can also help draw attention to existing errors at individual WWTPs (Henze et al., 2000).

1.2 Research Objectives

The overall goal of this project is to improve operator knowledge, process control and system performance using analysis of historical data and activated sludge modeling at a full-scale AWWTP. Specific objectives are to:

- Investigate process control best practices for advanced wastewater treatment plants and their applicability to the case-study AWWTP;
- Analyze influent, effluent and operating data over a three-year period (September 1, 2010 to August 31, 2013) to further understand process performance, determine gaps in knowledge, and suggest possible improvements for future data collection;
- Construct and calibrate a BioWin model of the plant following published guidelines for good modeling practice;
- Use the calibrated plant model to simulate the effect of changes in MCRT and influent pollutant loads on plant performance.

CHAPTER 2: LITERATURE REVIEW

This literature review focuses on nutrient removal processes, process control strategies, and modeling of activated sludge systems. Specific attention was given to oxidation ditches to highlight the case study WWTP.

2.1 Nitrogen Removal

Nitrification and denitrification are widely used biological processes for the removal of nitrogen from wastewater. Nitrification is understood to occur under aerobic conditions, where predominately autotrophic bacteria oxidize ammonia to nitrite and then nitrate using oxygen as an electron acceptor. Denitrification occurs when heterotrophic bacteria reduce nitrate to nitrogen gas under anoxic conditions (absence of free oxygen). Denitrifying bacteria will use oxygen as an electron acceptor in preference to nitrate because it is more thermodynamically favorable, thus making anoxic conditions imperative for denitrification to take place. The overall reactions for nitrification and denitrification are given in equations [1] and [2] (Henze et al, 2002).

- Autotrophic oxidation of ammonium:

$$NH_4^+ + 1.86O_2 + 1.98HCO_3^- \rightarrow 0.020C_5H_7O_2N + 0.98NO_3^- + 1.04H_2O + 1.88H_2CO_3 \quad [1]$$

- Heterotrophic reduction of nitrate (with ammonia assimilation for growth):

$$0.52\,C_{18}H_{19}O_9N + 3.28NO_3^- + 0.48NH_4^+ + 2.80H^+ \rightarrow C_5H_7NO_2 + 1.64N_2 + 4.36CO_2 + 3.8H_2O \quad [2]$$

These equations show that alkalinity, in the form of HCO_3^-, is consumed during nitrification. This is important because nitrifiers will be inhibited at low pH. Some alkalinity is recovered during denitrification.

A number of different treatment plant configurations have been invented and successfully implemented for nutrient removal over the years. Some systems, such as the Modified Ludzack-Ettinger (MLE) and Bardenpho ® processes, provide dedicated zones or tanks for nitrification and denitrification (Barnard, 1975; Ludzack and Ettinger, 1962). The MLE process (Figure 2.1) consists of an anoxic zone followed by an aerated zone. Internal mixed liquor recycle returns nitrified mixed liquor to the anoxic zone, where the influent wastewater provides carbon for the denitrifying bacteria. The 4-stage Bardenpho configuration (Figure 2.2) consists of an anoxic/aerobic layout similar to the MLE process, with an additional anoxic (and optional external carbon source) and aerobic zones for further nitrogen removal and sludge conditioning.

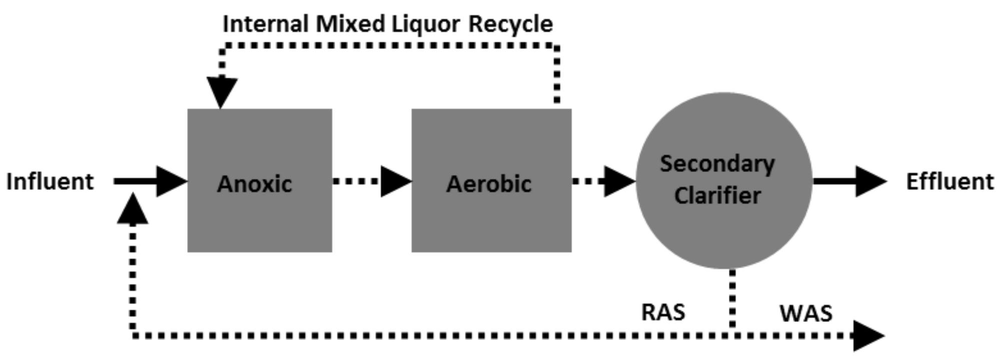


Figure 2.1 Schematic of a Modified Ludzack-Ettinger process.

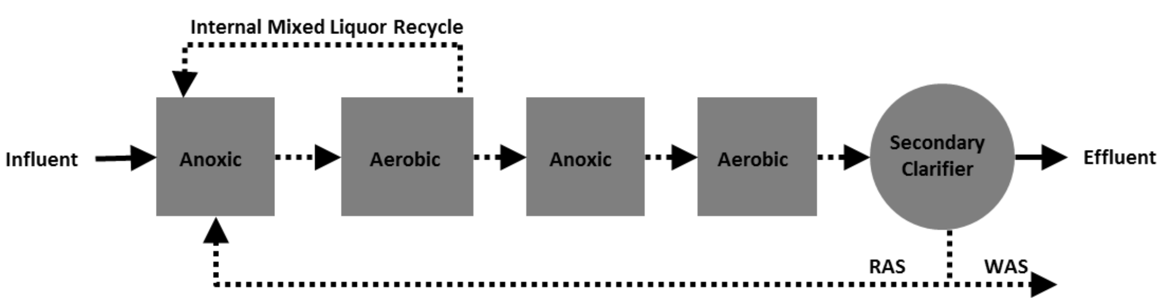


Figure 2.2 Schematic of a 4-stage Bardenpho process.

The term "oxidation ditch" is used loosely to refer to a variety of operating schemes and physical configurations. In general, all oxidation ditch systems are loop-shaped reactors operated in an extended aeration mode at relatively long hydraulic retention time (HRT) and

MCRT (Mandt and Bell, 1982). Operating with a longer MCRT makes nitrification possible even at low dissolved oxygen (DO) levels (Stenstrom and Poduska, 1980). The oxidation ditch is typically configured in a race-track style, with mechanical aerators or brushes placed at one or more points along the ditch. Mechanical aeration entrains oxygen and provides mixing in a horizontal flow pattern around the ditch. This flow pattern allows the mixed liquor to be recirculated around the ditch, presumably between aerobic and anoxic zones. Oxidation ditches exhibit simultaneous nitrification and denitrification, which will be discussed in detail in the next section. A schematic of a single-pass Carrousel oxidation ditch is shown in Figure 2.3.

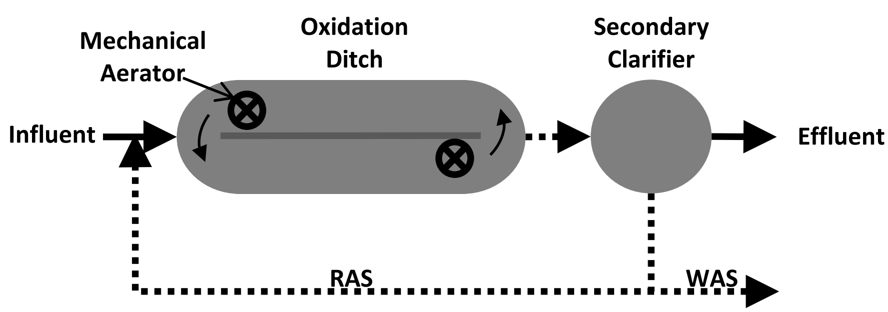


Figure 2.3 Schematic of an oxidation ditch.

Because nitrification and denitrification are occurring concurrently in an oxidation ditch, the consumption of alkalinity by nitrifying bacteria is partially offset by the alkalinity production of denitrifiers. The circular flow pattern in the ditch also supplies denitrifying bacteria with influent carbon and nitrate without the need for supplemental carbon addition or internal mixed liquor recycle. Oxidation ditches are sometimes paired with additional reactors to create Bardenpho or other type systems, so attention should be given to the exact nature of the WWTP.

2.1.1 Simultaneous Nitrification-Denitrification

Although many operating schemes use separate basins for aerobic and anoxic processes, substantial denitrification has been observed in aerated bioreactors without dedicated anoxic zones. In fact, the observation of denitrification within the aeration basin was the impetus for the creation of specific zones for nitrification and denitrification in an attempt to enhance removal rates (Barnard, 1998; Ludzack and Ettinger, 1962). The occurrence of nitrification and denitrification at the same time in a single reactor without distinct aerated and non-aerated zones is commonly referred to as simultaneous nitrification-denitrification (SND). Treatment systems exhibiting SND typically have relatively long SRTs, aeration equipment that creates non-uniform flows, such as mechanical aerators, and an operating procedure to limit oxygen input (Daigger, 2013). Recently, some WWTPs that were designed with separate aerobic and anoxic zones have been reconfigured to lower the DO concentration within the aerobic portion of the system to achieve high levels of SND (Jimenez et al, 2010; 2013). Operating at low DO concentrations has the potential to decrease overall energy usage, as supplying oxygen is often the most costly and energy-intensive process in the WWTP (WEF, 2010).

Three mechanisms for the occurrence of SND have been investigated previously (Daigger et al., 2007): (1) occurrence of aerobic and anoxic zones within the reactor, (2) occurrence of aerobic and anoxic zones within the floc particle, and (3) existence of novel microorganisms with alternative biochemical pathways. The literature is inconclusive as to whether the macro environment, the presence of aerobic and anoxic zones within the reactor, plays an important role in SND processes. Rittmann and Langeland (1985) measured DO, nitrate, and nitrite concentrations in full-scale Carrousel oxidation ditches. The authors found that denitrification occurred continuously in the reactor without evidence of distinct anoxic zones. Dissolved oxygen profiles in an Orbal oxidation ditch showed low DO concentrations (0.2 mg/L) before and after the mechanical aerator, suggesting that a DO gradient within the floc instead of heterogeneity in the reactor was the principal mechanism for SND (Daigger and

Littleton, 2000). Although difficult to measure in the field, a later study of the same Orbal reactor using computational fluid dynamics suggested that areas of higher and lower DO concentration can occur (Littleton et al., 2007).

In regards to the micro-environment in SND reactors, a study comparing SND performance with varying floc particle size found denitrification diminished with smaller floc sizes, possibly due to the diffusion of oxygen into the inner areas of the floc (Pochana and Keller, 1999). Daigger et al. (2007) further investigated DO gradients within individual floc particles using a "microprobe". The concentration of DO within floc particles decreased steadily with depth and ultimately reached near-zero levels in the interior of the larger flocs (≥ 2mm), as shown in Figure2.4.

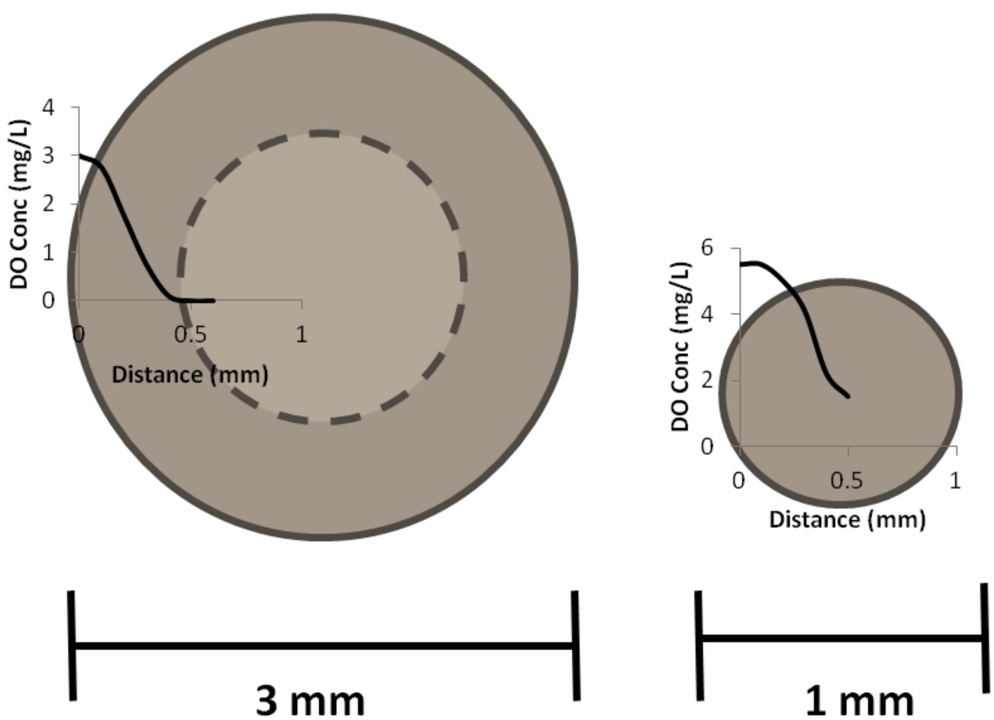


Figure 2.4 Dissolved oxygen gradients inside floc particles of varying size.

Nitrite Shunt refers to the conversion of ammonia to nitrogen gas without the intermediary step of the oxidation of nitrite to nitrate. Instead, ammonia is oxidized to nitrite, and nitrite is directly reduced to nitrogen gas by anoxic heterotrophic or autotrophic metabolism. Operating at low DO may result in limited nitrification and the occurrence of nitrite shunt (Ju et al., 2007). Littleton et al. (2003) further investigated the role of novel microorganisms in SND and found the contribution of alternative biochemical pathways to nitrification/denitrification to be insignificant.

In a typical nitrification reactor, blower or mechanical aerator speed is increased in response to an increase in ammonia concentration. However in a single reactor with SND, increasing the DO concentration will ultimately inhibit denitrification. Therefore, maintaining sufficient DO for nitrification without negatively impacting denitrification is critical. A bulk DO concentration of 0.4-0.5 mg/L has been found to be optimal for SND (Münch et al, 1996; Insel, 2007; Dey, 2010), with a decrease in denitrification rate occurring at greater than 0.8 mg/L (Pochana and Keller, 1999). Because diffusion of oxygen into the floc particle is one of the mechanisms of SND, the optimal bulk DO concentration may be dependent on the size and characteristics of the floc, as discussed previously. Another factor that may be especially influential on the rate of denitrification is the shearing of floc particles by mechanical aerators (Barnard et al., 2004). As the mixed liquor comes into contact with the aerator, the shearing of the floc allows carbon necessary for denitrification to be absorbed before re-flocculation takes place.

2.2 Phosphorus Removal

If NPDES permits set limits for phosphorus, treatment systems for phosphorus removal must be implemented. While complete chemical removal of nitrogen is usually prohibited by cost, phosphorus is commonly removed by a combination of both chemical and biological processes. Biological phosphorus removal is less reliable and understood than other nutrient removal processes, and addition of chemical removal processes is often needed to ensure permit compliance (Ingildsen et al., 2006; Oehmen et al., 2007).

2.2.1 Biological Phosphorus Removal

Phosphorus is essential to life, making up important molecules such as adenosine triphosphate (ATP), DNA and RNA, and the phospholipids that form cellular membranes. While all bacteria in the activated sludge process must have sufficient amounts of phosphorus to meet energy and growth needs, some species of bacteria can take up more phosphorus than needed for metabolism. Many species appear to be capable of excess uptake of phosphorus (Mino et al., 1998, Bond et al., 1999) and these bacteria are collectively called phosphorus accumulating organisms (PAOs). Enhanced biological phosphorus removal (EBPR) is the term given to treatment systems that take advantage of the phosphorus accumulation by PAOs. Generally, the design of EBPR systems includes an anaerobic zone followed by an aerobic zone as shown in Figure 2.5.

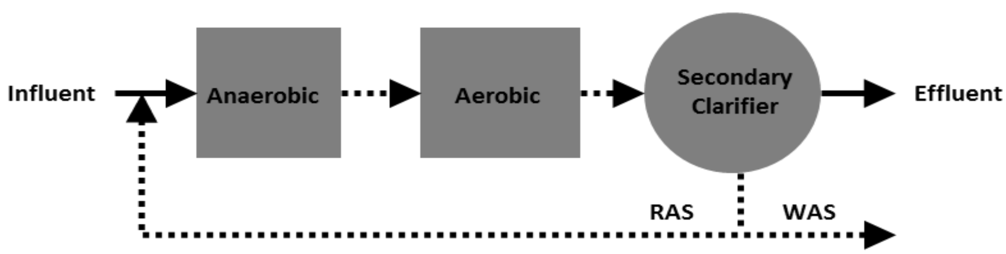


Figure 2.5 Schematic of a system for enhanced biological phosphorus removal.

An anaerobic environment is first used to encourage the growth of PAOs. In the absence of oxygen and other electron-accepting compounds, such as nitrate, heterotrophic bacteria ferment, instead of oxidize, influent organic material, creating volatile fatty acids (VFAs). The PAOs uptake and store the VFAs in the form of polyhydroxyalkanoic acids (PHAs) using energy from the hydrolysis of intracellular polyphosphate, resulting in a release of orthophosphate from the cell. Therefore, the first step of EBPR is accompanied by an increase in mixed liquor dissolved phosphorus concentrations. The removal of phosphorus occurs in the subsequent aerobic stage, when PAOs oxidize stored PHA using oxygen or nitrate as an electron acceptor. Oxidation of PHA is accompanied by uptake of the phosphate that was released along with additional phosphorus that was present in the raw influent wastewater. Phosphorus removal is accomplished when sludge is wasted from the system. Figure 2.6 shows the expected profile of phosphate and soluble, biodegradable COD over time through the anaerobic and aerobic reactors designed for EBPR.

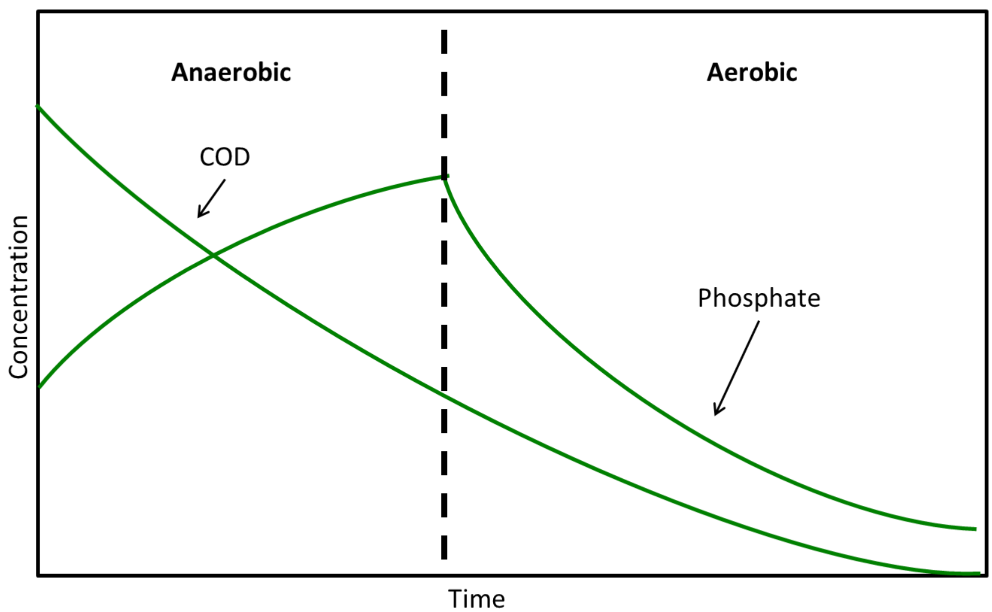


Figure 2.6 Phosphate and biodegradable COD profiles over time in the anaerobic and aerobic reactors of an EBPR system.

If nitrate is present in the RAS that enters the anaerobic zone, denitrifiers may outcompete PAOs and hinder EBPR. Several process configurations have been developed to over-come this problem. For example, the University of Cape Town (UCT) process eliminates the presence of nitrate in the anaerobic reactor by returning settled sludge to the anoxic tank and supplying microbes to the anaerobic reactor through internal anoxic mixed liquor recycle as shown in Figure 2.7.

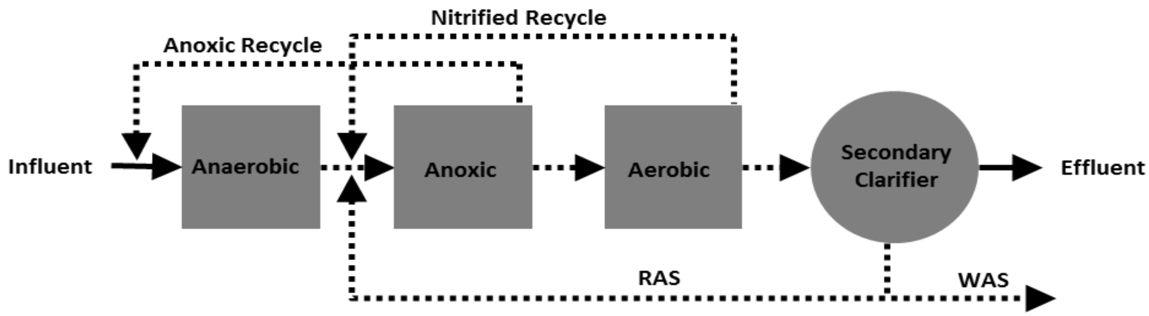


Figure 2.7 Schematic of the UCT process.

Although EBPR systems are designed with aerobic zones for phosphorus uptake, at least some PAOs can use nitrate as an electron acceptor during phosphorus uptake (Hu et al., 2002; Mino et al., 1998; Stevens et al., 1999). Substantial removal of phosphorus has also been observed in systems without anaerobic-aerobic configurations. Jimenez et al. (2010) observed significant removal of phosphorus (93.75%), without chemical addition, in a pilot plant operated at low DO for SND without a dedicated anaerobic stage. Removal of phosphorus was also observed in Orbal oxidation ditch reactors at full-scale plants without dedicated anaerobic zones or chemical addition (Daigger and Littleton, 2000). The authors suggested that mixing patterns may create anaerobic areas within the Orbal reactor, but that any anaerobic areas existing within the floc would likely not receive diffused readily biodegradable organic material. A CFD model of the same Orbal oxidation ditch reactor demonstrated the occurrence of varying DO environments that could result in anaerobic zones where PAOs could compete with other heterotrophs (Littleton et al., 2007).

Clarifier design and operating conditions can have an impact on the performance of EBPR systems. There is potential for a secondary release of phosphorus in the secondary clarifier if settled sludge is subject to anaerobic conditions in the absence of VFAs (Mikola et al., 2009). Effluent phosphorus concentration will also be impacted if suspended solids escape over the clarifier weir due to poor settling.

2.2.2 Chemical Phosphorus Removal

Phosphorus removal can also be achieved with the addition of chemicals at different stages of the treatment process, such as in the primary clarifier or the mixed liquor for precipitation in the secondary clarifiers. Ferric chloride and aluminum sulfate (alum) are examples of metal salts that are added to wastewater to precipitate phosphorus. The optimal dosage is usually determined on-site with jar tests and is dependent on the species of phosphorus present and the plant permit requirements (WEF, 2011). Bowker and Stensel (1990) point out that increased sludge production and effect on thickening and dewatering characteristics of sludge are two considerations when using aluminum salts for phosphorus removal.

2.3 Settling

The ability of bacteria to flocculate and settle is a necessary component of suspended growth activated sludge treatment systems. Mixed liquor suspended solids settle in the secondary clarifier and are returned to the aeration basin or wasted. Poor settling mixed liquor will decrease the capacity of the secondary clarifiers and may result in excessive loss of solids over the weirs in the secondary effluent. Sludge Volume Index (SVI) is commonly used as a measure of sludge flocculation and settling ability.

2.3.1 Sludge Bulking

Sludge bulking caused by filamentous organisms is a frequently-encountered problem in activated sludge systems, resulting in poor settling sludge in the secondary clarifier. Defining the exact conditions responsible for the proliferation and control of filamentous organisms can be difficult, as bulking occurs at numerous plants with a range of operating conditions (Ekama and

Wentzel, 1999). WWTPs that operate at low DO, long MCRTs, and low F:M ratios are particularly susceptible to filamentous bulking (Jenkins et al., 1993). In addition to achieving nutrient removal goals, manipulating reactor environments also serves to promote the growth of floc-forming organisms and reduce the population of filamentous organisms. Control of bulking uses some of the same principles as those used in the design of EBPR systems. In fact, an anaerobic reactor for EBPR is considered to be a selector, and PAOs are classified as floc-forming bacteria (Grady et al., 1999). A selector tank is introduced before the main aeration reactor to create feed-starve conditions, taking advantage of readily available organic matter. Filamentous organisms have been found to be less able than floc-formers to store substrate during the "feed" stage for subsequent use in the "starve" stage (van Niekerk et al., 1989). Chlorine can also be added to RAS to temporarily reduce the population of filamentous organisms, although this practice has had negative effects on biological phosphorus removal (Diagger et al., 1988). Microscopic examination of the mixed liquor can confirm the presence of filaments, and resources are available to help with identification of the particular species and type of filament present (Jenkins et al., 1993).

2.3.2 Measures of Sludge Quality

Although good sludge quality, activated sludge that flocculates, settles, and compacts, is critical for the successful operation of the WWTP, its measurement varies between plants. Sludge Volume Index (SVI) is a regularly calculated value by wastewater treatment operators in attempt to determine sludge quality, and it is used as an indirect indicator of bulking sludge. To calculate SVI, a sample of mixed liquor is first collected and allowed to settle for 30 minutes in a "settlometer" (Figure 2.8). Then, the height of the settled sludge is measured, divided by the

MLSS concentration, and multiplied by 1000 (Equation 3). Mixed liquor with SVIs greater than 150 ml/g are generally considered to be experiencing sludge bulking (Grady et al., 1999).

$$SVI\ \left(\frac{ml}{g}\right) = \frac{30\ minutes\ settled\ sludge\ volume\ \left(\frac{ml}{L}\right)}{MLSS\ concentration\ \left(\frac{mg}{L}\right)}\ x\ \frac{1000\ mg}{g} \qquad [3]$$

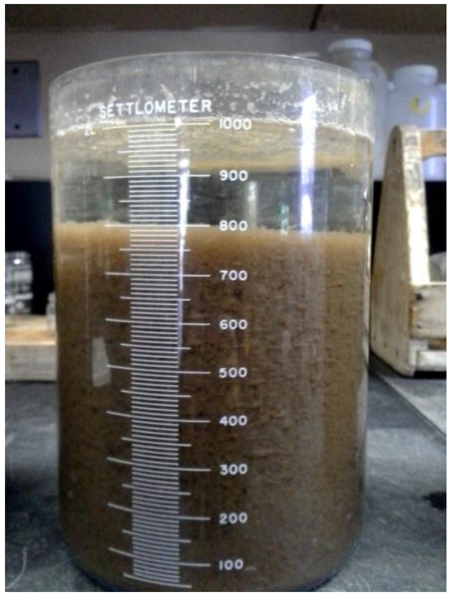


Figure 2.8 A Mallory Settlometer and a sample of mixed liquor after conducting the 30 minute settled sludge volume test.

Despite its prevalence, the validity of SVI as a measure of sludge quality has been debated. Bye et al. (1998) found that sludge samples with varying compactabilities had identical SVI values. Assuming that varying degrees of compactability indicate different extents of bulking, the authors suggested that SVI may not be a good indicator of sludge bulking.

2.3.3 Sludge Blankets

The sludge blanket is the layer of settled sludge residing in the bottom of the clarifier. While sensors are available to measure the depth of the sludge blanket, operators still typically measure this manually several times a day using a simple apparatus known as a “sludge judge” (Figure 2.9). The sludge judge is a clear, plastic tube that is slowly inserted into the clarifier until it reaches the bottom and then pulled back out. A check valve in the bottom of tube traps the contents of the clarifier inside, essentially taking a core sample. The height of the sludge blanket inside the tube is measured and recorded.

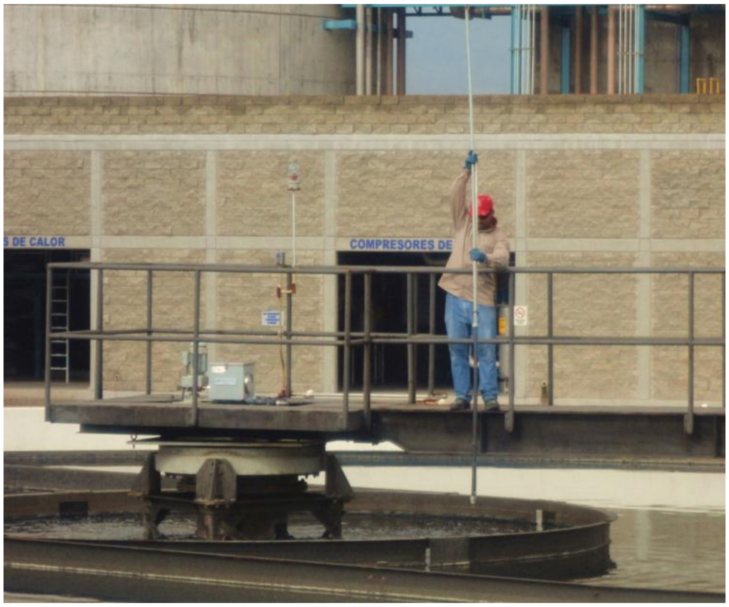


Figure 2.9 An operator at a WWTP in Mexico uses a "sludge judge" to measure the clarifier's blanket depth.

If blankets are allowed to accumulate, the clarifier will eventually fail and solids will exit over the clarifier weirs with the secondary effluent. This "wash out" scenario is particularly likely during high flow events. In general, suggested blanket levels are between 0 and 3 feet (WEF, 2002). Sludge blankets can also result in rising sludge. Nitrate present in the sludge blanket can undergo denitrification due to the development of anoxic conditions. The subsequent release of nitrogen gas to the surface of the clarifier can cause sludge to rise. Rerelease of phosphorus may also occur due to absence of oxygen within the blanket. Unlike the phosphorus release that takes places in the anaerobic tank before aeration, phosphorus released within the blanket will not undergo re-uptake.

2.4 Process Control for Biological Nutrient Removal

Many textbooks and trade manuals on wastewater treatment highlight the importance of process control and attempt to outline and define the process control strategies that are available to the wastewater treatment plant operator. In general, the three main operational areas for the control of the activated sludge process are return activated sludge flow, waste activated sludge flow, and dissolved oxygen concentration. Manipulation of internal recycle flows and the addition of external sources of carbon may also be considered in process control strategies but will not be addressed here due to the characteristics of the oxidation ditch technology.

2.4.1 Sources of Variability in WWTPs

Wastewater treatment plants are subject to many sources of variability. Influent flows and loads fluctuate diurnally, weekly, and seasonally. More sporadic fluctuations in flow may result from pumping at lift stations in the collection system or during periods of high flow variability, such as large sporting events or heavy rains. Periodic discharges of industrial wastewater, septage or landfill leachate can greatly alter the loading to the plant. To comply with NPDES permits and gain insight into plant operation, grab and 24-hour composite samples are collected and analyzed in certified on-site or contracted laboratories. Variability in plant data could result from something as simple as poor sample collection if an operator fails to sufficiently agitate the composite-sample container before collecting the sample. The results of a bench-top analysis in a mixed liquor sample may be grossly misrepresentative if the operator lets too much time pass before filtering the sample as the bacteria will continue to act on the constituents of interest.

Knowledge about the WWTP process can be gained from datasets using a variety of visual and statistical methods. First, the integrity of data can be assessed using simple "common sense" checks. For example, the MLVSS can never be higher than the MLSS. While it may be easy to spot an unusually high or low value in a dataset, determining if the outlier represents a true value can be difficult. Examining ratios of parameters (COD/total phosphorus, BOD/TSS, COD/TKN) can aid in identifying erroneous outliers (Bratby and Fevig, 2012). If a flow meter does not accurately measure the rate of WAS wasted daily, calculation of MCRT will be inaccurate. Conducting mass balances will expose discrepancies in data such as the amount of sludge wasted that would ultimately affect the calculation of MCRT. For example, the influent loading of phosphorus should equal the phosphorus in the effluent and the WAS.

2.4.2 Instrumentation, Control, and Analysis

Arthur (1983) lists three key factors for effective process control: (1) controllability of plant components, (2) capable on-line sensors, and (3) management of data. Controllability of plant components refers to the ability to make adjustments to aeration equipment, RAS, and wasting. For example, control may be limited by the available speeds (both minimum and maximum) of aerators or if wasting is hindered by downstream processes such as dewatering. On-line analyzers must be dependable and produce quality data. Finally, the performance of the system must be assessed using data collected by operators and analyzers in order to make process control decisions. In addition to these factors, the setting of priorities, such as minimizing cost and meeting effluent requirements, will help to guide the plant operator

There are many types of on-line sensors on the market today, and their reliability is continually improving. While this study will not investigate different sensors, it should be noted that there are sensors available for measuring operating parameters such as oxidation-reduction potential (ORP), MLSS, ammonia, nitrate and DO concentration. Myers et al. (2006) found that ORP probes are effective to control aeration and ammonia concentration in an extended aeration oxidation ditch. ORP-probes measure the ability of a solution to accept or donate electrons, and may be particularly useful in low-DO extended aeration processes.

Olsson et al. (1999) list four components of control for wastewater systems: the process, the measurement, the decision-making, and the implementation. These components are arranged depending on whether the control is feedback or feedforward. In feedback control, a "disturbance" is measured after it affects the process, and decisions and adjustments are made accordingly to correct the impact of the disturbance. Feedforward control involves measuring disturbances directly before they impact the process and making decisions and implementations to the process that will off-set or eliminate the disturbance. Both feedback and feedforward control can be used to improve process stability, reduce operating costs, and ensure permit compliance.

2.4.3 Aeration Control for Simultaneous Nitrification-Denitrification

Jimenez et al. (2013) investigated and compared two aeration control strategies (constant low DO and ammonia-based control) for SND in bench scale sequencing batch reactors (SBRs) and at several full-scale WWTPs. For the bench scale experiment, two SBRs were operated in parallel. The bulk DO concentration in one of the SBRs was maintained at a constant low DO of between 0.25 and 0.5 mg/L. In the second SBR, ammonia was monitored and aeration was turned off and on when the ammonia concentration reached 0.5 and 1.0 mg/L, respectively, with a maximum DO set point of 0.8 mg/L. The performance (% removal of nitrogen) of both strategies was compared, with constant low DO producing lower TN values overall but with higher SVI values. A comparison of eight full-scale WWTPs using either constant low DO or ammonia-based control for SND showed that both strategies produced good nitrogen removal, but ammonia-based control resulted in better sludge quality (lower SVI values). However, the authors noted that other variables, such as sludge age, were not taken into account.

2.4.4 Wasting Control

While the majority of sludge that settles in the secondary clarifier is returned to the bioreactor with the RAS, a fraction of the biomass in the activated sludge system must be wasted regularly. The three strategies for determining the amount of sludge to be wasted frequently given are (1) SRT control, (2) MLSS control, and (3) F:M control. Solids Retention Time (SRT) and MCRT are terms given to the average amount of time that bacteria remain in the wastewater treatment system. Sometimes these terms are used interchangeably, but both should be defined when used to make clear what solids are being included in the calculation. Sometimes the mass of solids in the clarifier is included with the mass in the aeration basin. Other times, only the mass under aeration is calculated.

Previous studies have been conducted to compare the effects of operating at constant MLSS, constant SRT, and constant F:M. Wahlberg et al., (1996) used BioWin to model an existing WWTP and run simulations over a period of a year using three wasting strategies: constant SRT, constant MLVSS, and constant F:M ratio. Simulations to maintain a constant MLVSS concentration resulted in variation in the SRT ranging from 8.8 to 20 days and the most variability in the WAS flow rate of the three strategies, illustrating that MLVSS, SRT, and F:M cannot be held constant simultaneously.

Sludge age is especially important for nitrification. Ammonia oxidizing bacteria (AOBs) have relatively low growth rates and require a higher minimum sludge age to sustain an adequate nitrifying population than heterotrophs. The minimum SRT is approximately the reciprocal of the maximum specific growth rate as in equation [4].

$$SRT_{min} \cong \frac{1}{\mu_{max}} \qquad [4]$$

The method of control is important for proper operation of the WWTP because MLSS and SRT are related as in equation [5]. This equation is used during plant design, and illustrates that MLSS is a response variable to SRT and substrate. The hydraulic retention time (HRT) also shows that variable flows affect the system assuming tank volume is constant. At a fixed SRT, the MLSS will vary with changing substrate concentration and flow. The bacteria, at a given growth rate determined by the operating SRT, will grow when food (substrate) is available and decrease as food decreases. Maintaining a constant MLSS concentration causes forces the SRT to change as substrate concentration and flow vary.

$$[MLSS] = \frac{SRT}{HRT}\left(\frac{Y(S_o - S)}{1 + (k_d)SRT}\right) \qquad [5]$$

where Y is a yield coefficient, kd is a decay constant, and S is the substrate concentration.

The SRT or MCRT is the most important variable for the successful operation of biological suspended growth processes (Grady et al., 1999). Operating at a long SRT results in the accumulation of inert biomass, which has been shown to increase linearly with SRT (Moussa, 2005). Inert biomass occupies space in the system without providing treatment. Wasting control using the constant MLSS method is only recommended for small WWTPs that do not have the means and technology in place to accurately calculate sludge age (WEF, 2002).

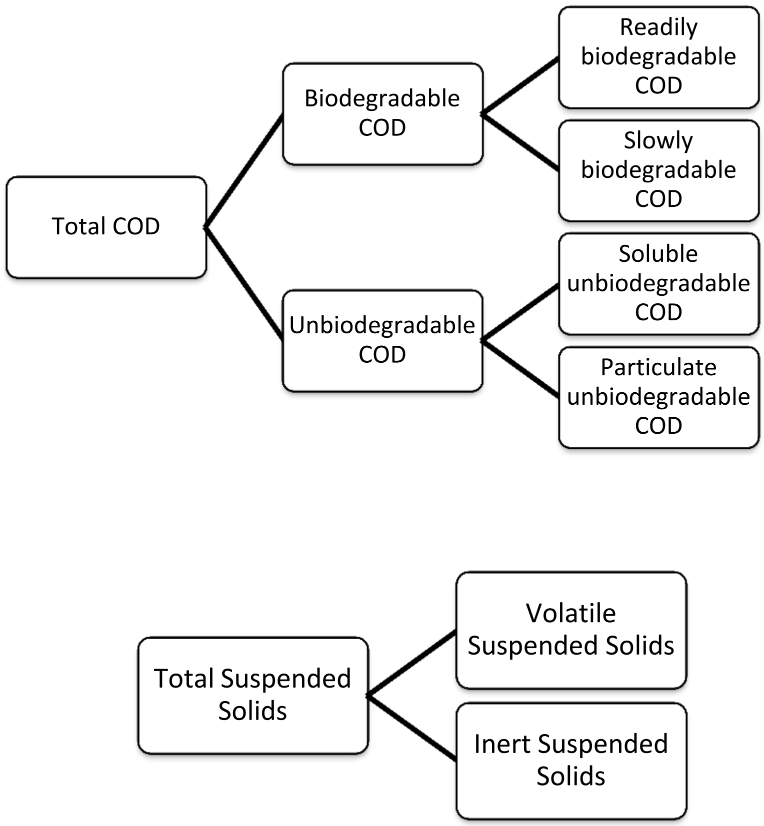
2.5 Modeling

Activated sludge models (ASMs) are used in the design, upgrade, and optimization of wastewater treatment plants. Modeling can be a powerful tool for troubleshooting and increasing understanding of plant operations. Models are used to assess the effect of projected flow increases on effluent quality, oxygen demand, and clarifier capacity and to make decisions about process control and capital investment. In a survey of model users by Hauduc et al., (2009), plant optimization was found to be the most common use for ASMs, while modeling as a training tool was the least common use.

Although models have been used successfully in many applications, modeling is still widely performed by self-taught modellers without formal training who may be misapplying or creating inadequate models (Hauduc et al., 2009). The IWA Task Group on Good Modelling Practice was created in 2004 with the goal of establishing and promoting a set of guidelines for using activated sludge modeling. These guidelines, labeled the GMP Unified Protocol, were published in a 2012 report, along with examples of modeling in practice (Rieger et al., 2012). The GMP Unified Protocol took into account previously published modeling guidelines such as the HSG protocol (Langergraber et al.,2004), WERF guidelines (Melcer et al., 2003), and a generic calibration procedure from Ghent University (Vanrolleghem et al., 2003). The 5 steps of the GMP Unified Protocol are (1) Project Definition, (2) Data Collection and Reconciliation, (3) Plant Model Set-up, (4) Calibration and Validation, and (5) Simulation and Result Interpretation.

2.5.1 Wastewater Characterization and Model Calibration

Wastewater characterization is essential to the modeling process and is also useful in routine data checks and troubleshooting. The constituents in influent wastewater, such as COD, phosphorus, nitrogen and suspended solids can be broken down to different components as shown in Figure 2.10. Two of the most influential wastewater constituents are readily biodegradable and unbiodegradable particulate fractions of COD (Melcer, 2003). Readily biodegradable COD (rbCOD) is the fraction of organic matter that is most available for use by bacteria and will determine if processes such as EBPR are possible. The unbiodegradable particulate fraction of COD will impact the level of volatile suspended solids concentration and the oxygen uptake rate in the mixed liquor. Although not routinely measured at most WWTPs, there are three methods for determining the fraction of rbCOD. Two methods for determining rbCOD use physical-chemical methods (Dold et al., 1980; Mamais et el, 1993). The third method involves respirometry and biological methods (Melcer et al., 2003)



(a)

(b)

Figure 2.10 (a) Partitioning of COD and (b) influent suspended solids.

The challenges and complexity of using dynamic activated sludge models were captured by Ekama (2009): “Dynamic models always demand more information than available and prompt more questions than can be answered”. To calibrate an activated sludge model, simulated data is compared to historical data. Calibration sometimes requires adjusting kinetic and stoichiometric parameters. This is true with modeling of simultaneous nitrification-denitrification systems. Jimenez et al. (2010) attempted to model SND in a continuous-flow activated sludge pilot plant. To adequately predict SND performance, the model calibration required changes to values for several maximum specific growth rates and half saturation coefficients. The authors indicated that the aerobic denitrification and nitrite oxidizer DO half saturation coefficients were the most important for simulating SND performance.

CHAPTER 3: METHODS

3.1 Site Overview

The Falkenburg Road AWTP, shown schematically in Figure 3.1, is a Biological Nutrient Removal (BNR) plant located in Hillsborough County, Florida, with an annual average influent flow of 9.27 MGD and a permitted annual average flow of 12 million gallons per day (MGD). In addition to domestic wastewater, the plant receives some landfill leachate and wastewater from local industry. The plant's NPDES permit requires the removal of carbonaceous BOD, total suspended solids, total nitrogen and total phosphorus to levels of 5, 5, 3, and 1 mg/L (annual averages), respectively.

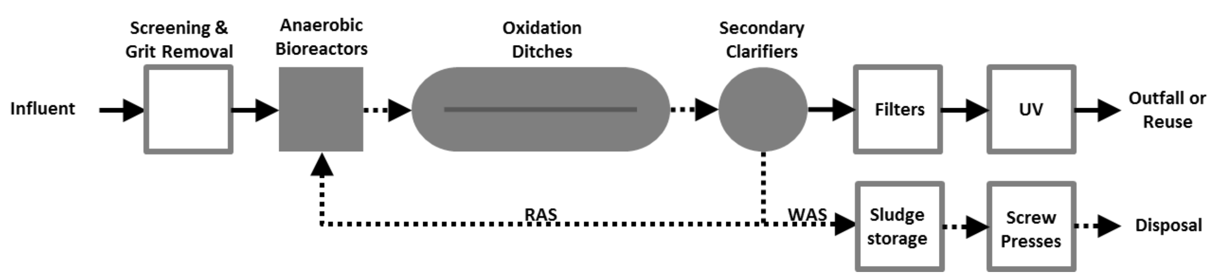


Figure 3.1 Layout of the Falkenburg AWWTP.

In the liquid train, influent wastewater first passes through the headworks, where screening and grit removal take place. The facility uses Carrousel® oxidation ditch systems for BOD removal, nitrification, and denitrification, preceded by anaerobic tanks that were designed to promote phosphorus removal. The mechanical aerators in the oxidation ditches have variable frequency drives (VFDs) that can be manually or automatically controlled based on DO or by using NH_4^+-N and NO_3^--N measurements. Mixed liquor leaving the oxidation ditches enters a splitter box where aluminum sulfate is added for chemical phosphorus removal, and the flow is divided between five circular secondary clarifiers. Return activated sludge is returned to the anaerobic tanks where it mixes with incoming screened influent. Further removal of suspended solids from secondary clarifier effluent is achieved with deep bed filtration followed by UV disinfection. Finished effluent is either discharged to the Palm River/Hillsborough River Bypass Canal or used directly as reclaimed water. In the solids train, WAS is diverted from the RAS line from the secondary clarifiers and sent to a holding tank prior to screw press dewatering. Dewatered biosolids are then trucked to a landfill or incinerated in a neighboring resource recovery facility. The dimensions of the anaerobic basins, oxidations ditches, and clarifiers were obtained from the Falkenburg Operations and Maintenance (O & M) Manual and are given in Table 3.1.

Table 3.1 Physical WWTP data

<table>
<tr><th>Tank</th><th colspan="3">Dimensions</th><th>Number of tanks</th><th>Total Volume (gallons)</th></tr>
<tr><td rowspan="2">Anaerobic</td><td>Length (ft)</td><td>Width (ft)</td><td>Depth (ft)</td><td rowspan="2">4</td><td rowspan="2">1,215,800</td></tr>
<tr><td>48</td><td>51</td><td>16.6</td></tr>
<tr><td rowspan="2">Oxidation ditch</td><td>Area (ft^2)</td><td>Width of Pass (ft)</td><td>Depth (ft)</td><td rowspan="2">4</td><td rowspan="2">7,130,000</td></tr>
<tr><td>15,890</td><td>30</td><td>15</td></tr>
<tr><td rowspan="2">Clarifier</td><td>Diameter (ft)</td><td colspan="2">Depth (ft)</td><td rowspan="2">5</td><td rowspan="2">4,112,300</td></tr>
<tr><td>100</td><td colspan="2">14</td></tr>
</table>

3.2 Following the GMP Unified Protocol Steps

The IWA Task Group on Good Modelling Practice was created in 2004 with the goal of establishing and promoting a set of guidelines for using activated sludge modeling. These guidelines, labeled the GMP Unified Protocol, consists of five steps to help direct the modeler: (1) Project definition, (2) Data colleciton and reconciliation, (3) Plant model set-up, (4) Calibration and validation, and (5) Simulation and result interpretation.

3.2.1 Step 1: Project Definition

Meetings were held with the Hillsborough County Public Utilities Department to further define the goal of the modeling project. It was determined that an overall working model of the plant would be constructed to serve as a benchmark to aid in future process control decisions. The variables chosen for model calibration and validation were MLSS, MLVSS, and effluent ammonia, TKN, nitrate, and nitrite.

3.2.2 Step 2: Data Collection and Reconciliation

Microsoft Excel files containing data from a 3-year period (September 1, 2010 to August 31, 2013) were exported from the Falkenburg AWWTP Hach WIMS™ system. Hach WIMS is propriety software that serves as a central database for laboratory, SCADA, and operator-entered data. A dashboard with programmed calculations and reports is available in WIMS to make organizing, analyzing, and viewing data easier on the user. Files were available in monthly increments, and 36 months of data were compiled into a master spreadsheet. After compilation, values that were entered as less than the detection limit were entered as zero. Each parameter dataset was screened for outliers, which were detected using several methods. First, columns of data were ordered from smallest to largest, exposing unusually low or high values. Data that were clearly entered in error were deleted. For example, a column containing daily volumes of WAS contained two relatively high values of 9 and 15 mgd. These values were undoubtedly invalid, as it would by physically impossible to waste such high volumes in a 24-hour period. Time series were also plotted to reveal potential errors.

The only influent flow data that were available for export to Excel were average daily flows. Flow meters measure the actual flow and current and historical diurnal trends are available in SCADA. A rough estimate of a typical diurnal influent flow was made by reviewing these SCADA trends, recording the hourly flow for one day and creating a flow chart manually in Excel (Figure 3.2). The flow was normalized by the average daily flow (Figure 3.3), and then this normalized flow was multiplied by the average daily flow for each day to develop an hourly data set for entry into BioWin.

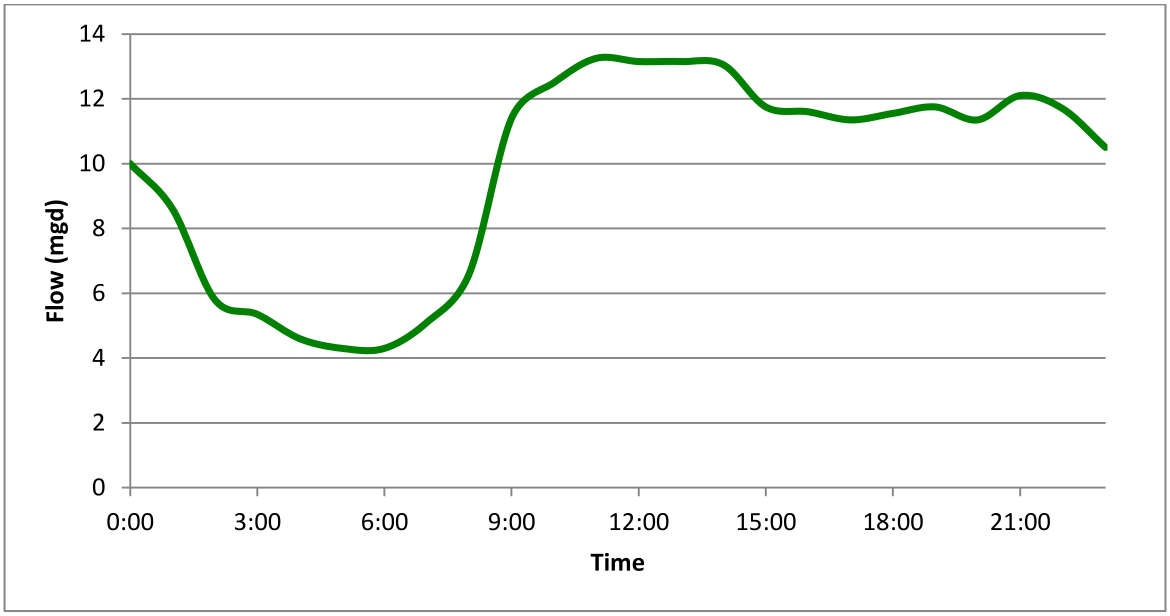


Figure 3.2 Typical diurnal influent flow pattern.

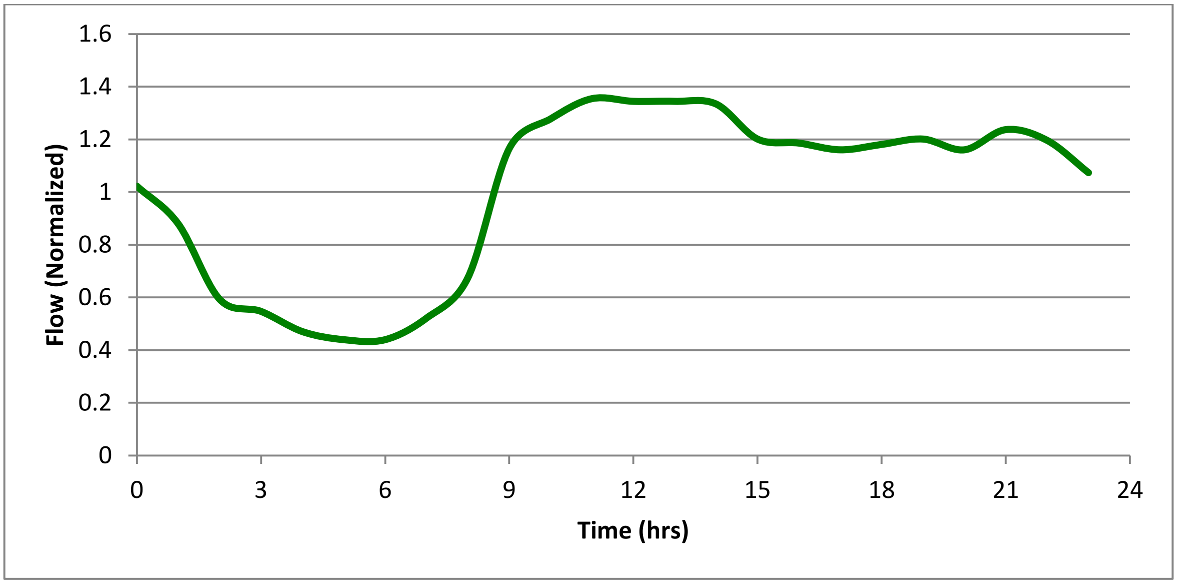


Figure 3.3 Normalized diurnal influent flow.

Although this normalization method allows an hourly profile to be created to better represent the diurnal pattern of the influent flow, it does not take into account atypical flow patterns such as those due to storm events. In addition, the daily maximum flow that is recorded in WIMS was not represented in the influent flow data that was developed for input to the model. Historical daily flow trends are captured in SCADA and may be viewed as a visual trend line, but actual values are not exportable in a usable format. Hillsborough County is working towards a system to store data from SCADA that may later be exported and used.

In addition to the hourly flow, the BioWin influent data set also required concentrations of BOD, TKN, Total P, VSS, TSS, pH, calcium, magnesium, alkalinity, and DO. A 24-hour sampling event was conducted to observe changes in influent COD, total P, and suspended solids. A 24-hour sampling campaign was carried out on April 14 to April 15, 2014. During the sampling event, grab samples of influent were collected from the influent sampling sink every two hours beginning at 8:00 AM on April 14th. Total and filtered COD and total and volatile suspended solids were measured using the methods described in Section 3.2.4.1. Although there was some variation in influent concentrations during the 24-hour period, one sampling event was not sufficient to estimate typical diurnal variations. Therefore, the average influent concentrations of BOD, TKN, and TSS in the BioWin influent file were used and held constant over the 24-hour period. Future modeling work should further investigate diurnal changes in influent concentrations. Influent DO was assumed to be 0 mg/L at all times. Default values for calcium, magnesium, and alkalinity were used.

3.2.3 Step 3: Plant Model Set-up

The layout of the Falkenburg AWWTP that was constructed in the BioWin simulator is shown in Figure 3.4. The oxidation ditches where modeled using a loop of 8 unaerated completely stirred tank reactors (CSTR) and 2 mechanical aerator reactors, equally dividing the volume of all 4 trains. This loop configuration was needed to develop the DO gradient that occurs within the oxidation ditch. A splitter element was placed in the loop, which allowed the horizontal flow velocity within the ditch to be adjusted. Abusam (2001) found 10 CSTRs to be ideal after evaluating the number of CSTRs needed to model an oxidation ditch with two mechanical aerators. The alpha factor value was raised from the default setting of 0.5 to 0.85 to better represent aeration with surface aerators. The default alpha factor is more typical for diffused aeration systems (Envirosim, n.d.). All five clarifiers were modeled as one ideal clarifier, and the anaerobic selectors were modeled as one completely mixed, unaerated bioreactor. The underflow rate for the secondary clarifier, the RAS flow, was flow-paced at 100 percent of the influent flow and the WAS flow rate was set at a constant rate of the average daily value. Dewatering elements were used for the screw presses and media filters, and assumptions were made for the percent solids removal and underflow values (Table 3.2).

Table 3.2 Assumptions for plant model set-up

ELEMENT	ASSUMPTIONS AND SETTINGS
Aerated Reactors (Reactors 1 & 6)	The DO set point was set at a constant concentration of 2.0 mg/L.
Clarifier	An ideal clarifier was used with a sludge blanket height of 0.3 (fraction of settler height). The "biological reaction" box was left unchecked for the calibration for simplicity. The RAS flow (underflow) was paced at 100% of influent flow. Actual data for plant RAS flow was missing. Operations staff confirmed that the plant is operated with a return rate of 100% of influent flow.
WAS Splitter	The splitter element for WAS flow was set at a constant rate of 0.234 mgd. This was the average waste flow rate from September 1, 2010 to August 31, 2011. This WAS flow rate along with the influent inputs resulted in an SRT at steady state of approximately 20 days.
Temperature	The temperature was held constant at 20°C.
Screw Presses	The dewatering element underflow was set at 0.05 mgd and a percent removal of 95% based on previous modeling conducted during the 2009 plant expansion.
Media Filters	The dewatering element underflow was set at 0.0003 mgd and a percent removal of 94% based on previous modeling conducted during the 2009 plant expansion.

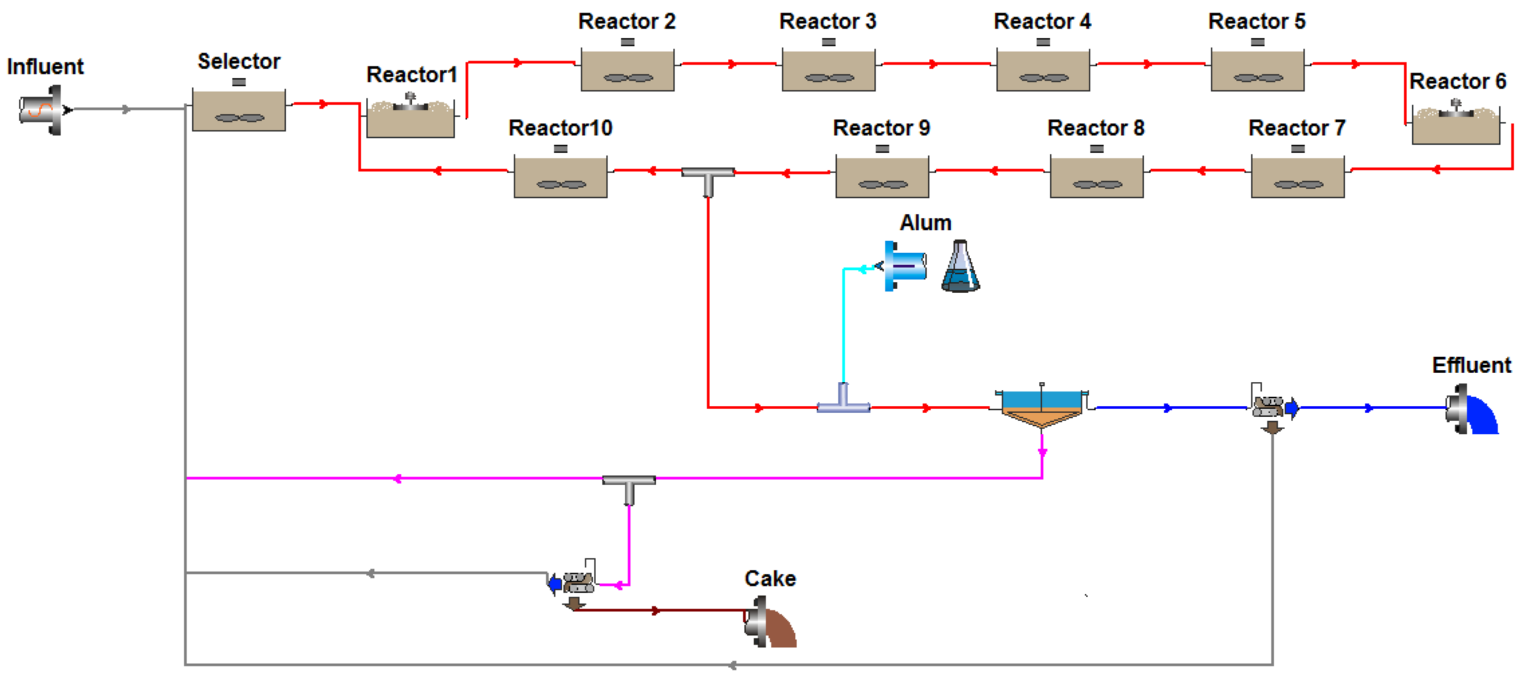


Figure 3.4 Falkenburg AWWTP layout in BioWin.

3.2.4 Step 4: Calibration and Validation

A commonly encountered issue with activated sludge modeling is the lack of needed input data. For the Falkenburg plant, influent $cBOD_5$, TSS, TKN, NH_3, and PO_4 are measured two times per week in 24-hour composite samples. Influent COD and VSS are not measured. Activated sludge models require designation of COD fractions and inert suspended solids (calculated by subtracting VSS from TSS) that will impact how the model functions. For example, the particulate unbiodegradable COD fraction impacts sludge production and oxygen demand, and the influent ISS also contributes to total sludge production (Melcer et al., 2003). For this reason, the calibration step required additional wastewater characterization to determine fractions of COD and estimations of volatile and inert suspended solids. Using historical and measured data, the BioWin Influent Specifier Excel worksheet (Appendix A) was used to calculate the wastewater fractions that were entered into the BioWin model. Kinetic and other parameters were adjusted on a trial and error basis, while also taking into account previous modeling of SND processes found in the literature (Jimenez et al, 2010; Envirosim, n.d.). Goodness of fit analyses were used to determine the best arrangement of kinetic parameters, and sensitivity analyses indicated which parameters had the greatest effect on the model outputs. Finally, historical influent data from September 1, 2011 to August 31, 2012 were used to validate the calibrated model.

3.2.4.1 Wastewater Characterization

Total, filtered, and flocculated-filtered COD were measured in influent and secondary effluent. Refrigerated 24-hour composite samples of influent wastewater (post-screening and grit removal) were collected from the Falkenburg AWWTP on five days (Appendix E). Each sample was placed on ice and analyzed within 8 hours of collection. The flocculated-filtered fraction of influent COD was determined using a physical-chemical method developed by Mamais et al. (1993). First, the influent sample was flocculated by adding 1mL of a 100 g/L zinc sulfate solution to 100 mL of influent wastewater and mixing with a magnetic stirrer for 1 min. Next, the pH of the sample was adjusted to 10.5 using a 6M NaOH solution. After pH adjustment, stirring was stopped and the sample was allowed to settle for approximately 5 minutes. Forty milliliters of supernatant were removed, taking care not to disturb the settled portion of the sample, and vacuum filtered through a 0.45µm membrane filter (Fisherbrand 0.45µm, 47mm, MCE membrane filters). The COD of the flocculated-filtered sample and total and filtered COD of the influent sample were determined using Standard Methods 5220D (APHA et al, 2012). Total and filtered COD were also measured in grab samples of secondary effluent in order to determine the amount of unbiodegradable soluble COD.

Washed and dried 47mm diameter glass fiber filters (Whatman, 1.5µm, 47mm glass microfiber filters) were used to measure TSS and VSS according to Standard Methods 2540D and 2540E, respectively. Both influent and secondary effluent were analyzed for TSS and VSS. For the influent wastewater, the sample was shaken vigorously and 50mL of sample were vacuum-filtered through the glass fiber filter. For secondary effluent, approximately 1000 mL were filtered.

3.2.4.2 Goodness of Fit

The average sum of absolute residuals (Equation 6) was calculated to determine goodness of fit of modeled to observed concentrations of effluent ammonia, nitrate, and nitrite. These values were compared for several simulations with different arrangements of four kinetic parameters (Table 3.3). A lower number indicated a better fit of modeled to observed data. The adjusted parameters were “heterotrophic DO half sat.”, “aerobic denit. DO half sat.”, “ammonia oxidizer DO half sat.", and "anoxic NO2 half sat." switching functions. The heterotrophic and aerobic denit. DO half saturation constants have been combined into one parameter in the latest BioWin edition; an older edition of BioWin was used in this study, and the two parameters were kept equal for compatibility with newer versions. The number of simulations and combination of parameters was limited due to time constraints. The heterotrophic and aerobic denit. DO half saturation constants were adjusted based on suggestions in the literature (Envirosim, n.d.) and previously published SND modeling work (Jimenez, 2010). Other model parameters may achieve a better fit to observed data; however parameter adjustment should be done with care to avoid unrealistic values. The year-long simulation period resulted in a relatively long simulation time of approximately 4-5 hours. Shortening the simulation period to a six months or one month would increase the amount of time available to run simulations.

$$Average\ SAR = \frac{\sum_{i=1}^{n} |y_{m.i} - y_{o,i)}|}{n} \quad [6]$$

where y_m is the modeled output and y_o is the observed output.

Table 3.3 Combination of kinetic parameters tested during model calibration

Simulation	**heterotrophic DO half sat. (mgO_2/L)**	**aerobic denit. DO half sat. (mgO_2/L)**	**ammonia oxidizer DO half sat. (mgO_2/L)**	**anoxic NO2 half sat. (mg N/L)**
1*	0.05	0.05	0.25	0.01
2	0.05	0.05	0.15	0.05
3	0.25	0.25	0.25	0.05
4	0.25	0.25	0.35	0.01
5	0.3	0.3	0.15	0.05
6	0.3	0.3	0.25	0.01
7	0.3	0.3	0.25	0.05
8	0.5	0.5	0.25	0.01
9	0.5	0.5	0.15	0.05

*BioWin default kinetic parameter values

3.2.4.3 Sensitivity Analysis

Sensitivity analysis of the BioWin model was performed to determine which parameters were the most influential to the outputs of the model. Modeling professionals advise performing the sensitivity analysis early on in the modeling process to determine where time and resources should be directed (Hulsbeek et al., 2002). Five parameters were chosen for the sensitivity analysis based on previous modeling by Jimenez et al. (2010). A normalized sensitivity coefficient method (Equation 7) was used by Liwarska-Bizukojc et al. (2010) to compare the percent change in output value to a 10 percent change in input values. Changing the parameters by 10 percent was not feasible in BioWin, because the model required an input in the tenths place. For example, the value of the AOB max spec growth rate was 0.9. A 10 percent change would be 0.99, but the model would only find a steady state solution if 0.8 or 1.0 were entered.

$$S = \left|\frac{\Delta y/y}{\Delta x/x}\right| \qquad [7]$$

The half saturation parameters are located under a heading entitled "switches" in the BioWin simulator. These parameters act as on-off switches by either turning on or off activity of groups of bacteria under certain environmental conditions. For example, the heterotrophic DO half saturation constant turns off the activity of ordinary heterotrophic organisms under low DO conditions. Similarly, the anoxic NO_3 half saturation parameter turns off anoxic growth that uses nitrate under low nitrate conditions.

3.2.5 Step 5: Simulation and Result Interpretation

Two sets of steady state simulations were run with the calibrated model while changing one variable at a time and observing and recording the effect on response variables, such as effluent ammonia and nitrate and MLSS concentrations. For the first set of simulations, SRT was adjusted between 2 and 60 days using the "control SRT" feature in BioWin. For the second set of simulations, the SRT was held constant at 20 days and the influent COD concentration was varied between 200 and 600 mg/L. During all simulations, aeration, RAS, and influent settings other than the manipulated variable were kept constant. A “COD Influent” element with a “constant” input type was chosen for all steady state simulations using historical, measured, and estimated data (Table 3.4).

Table 3.4 BioWin COD influent parameters for steady state simulations

Parameter	Units	Value
Flow	mgd	8.83
Total COD	mg/L	557
TKN	mg/l	51.2
Total P	mg/L	10
Nitrate	mg/L	0
pH		7.4
Alkalinity	mmol/L	6
Inorganic Suspended Solids	mg/L	24
Calcium	mg/L	160
Magnesium	mg/L	25
Dissolved Oxygen	mg/L	0

3.3 Investigation of Phosphorus Removal

Grab samples were collected from 4 sampling points shown in Figure 3.5. A portion of samples 2, 3, and 4 were allowed to settle for several minutes to obtain a sample of supernatant. The supernatant was immediately filtered with a 0.45µm syringe filter (Fisherbrand syringe filters, 0.45 µm, 33 mm). Samples were placed on ice and analyzed within 8 hours of collection. Hach (Loveland, CO) TNT 843 and 845 kits were used for analysis of low and ultra high range phosphorus, respectively. The Hach kits use the ascorbic acid method of Standard Methods 4500E. Depending on the sample preparation steps that were carried out, either total phosphorus or reactive phosphorus was determined with the kits; for total phosphorus an additional digestion step at 100°C for 1-hr is required.

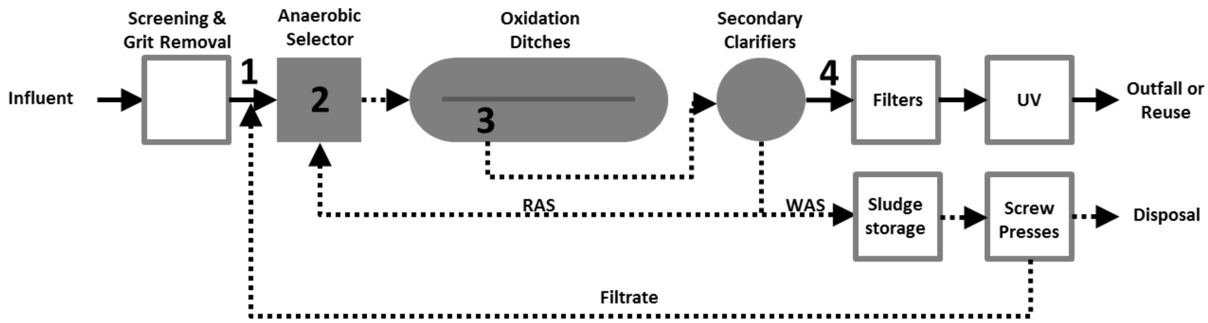


Figure 3.5 Sampling points for phosphorus analysis.

CHAPTER 4: RESULTS AND DISCUSSION

4.1 Review of Plant Operation

A review of historical influent, effluent, and operating data showed that the Falkenburg has consistently met NPDES permit limits. The effluent nitrogen concentrations (shown as 30-day moving average concentrations) over a three year period beginning September 1, 2010 are shown in Figure 4.1. Relatively low ammonia and nitrate/nitrite concentrations indicate that the plant nitrifies and denitrifies adequately through-out the year. A large portion of the total N present in the effluent is attributed to dissolved organic nitrogen (DON), which does not undergo transformation in the wastewater treatment plant (Pagilla et al., 2008). From the data obtained for the three year period in this study, DON constituted almost 48 percent of total effluent N. Although the permit limit for total nitrogen has historically been met, optimizing aeration in the oxidation ditch will likely result in cost and energy savings.

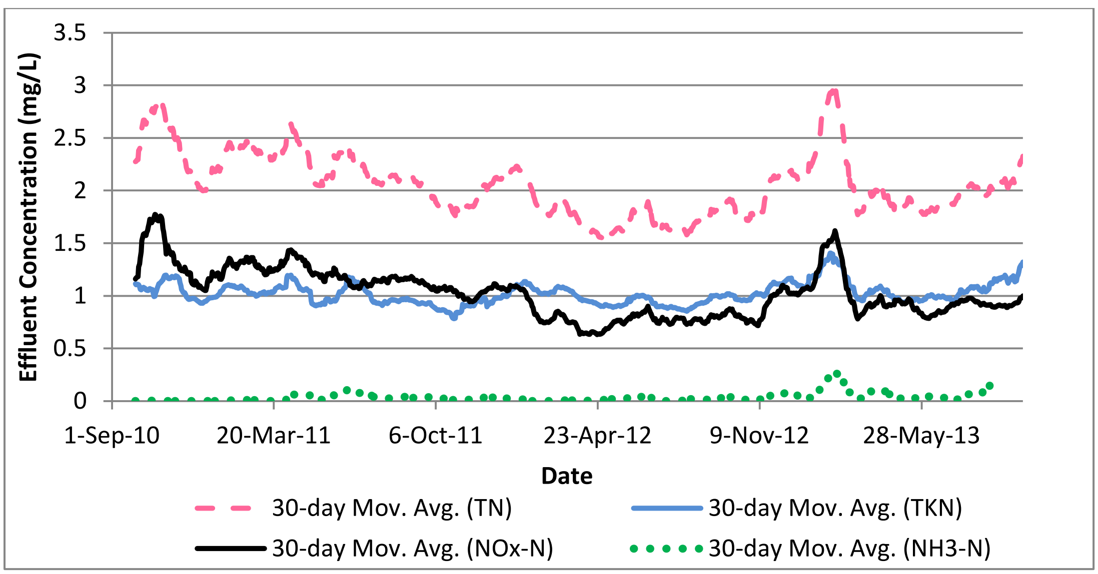


Figure 4.1 30-day moving average of effluent N concentrations at Falkenburg AWWTP.

4.1.1 Aeration Control

The operations staff at the Falkenburg AWWTP typically collect samples of mixed liquor from the effluent of the oxidation ditch 6 times in a 24 hour period (roughly every 4 hours). The mixed liquor sample is filtered through paper towels to remove suspended solids, ammonia, nitrate, and phosphate concentrations are measured in the filtrate using bench top reagents and spectrophotometry. Operators then make adjustments to the speed of the mechanical aerators based on the ammonia and nitrate measurements. All adjustments to the mechanical aerators are made manually. Each oxidation ditch is equipped with two 200-hp mechanical aerators, located on either end of the ditch. The aerator closest to the inlet is usually operated at 100%, while adjustments are made to the second aerator. During very low flows, the second aerator may be turned off and the speed of the inlet-side aerator may be adjusted.

In addition, a Chemscan® -UV4100 analyzer was added with the last plant expansion in 2009. Mixed liquor is pumped from the oxidation ditch effluent to the Chemscan unit, filtered, and analyzed for ammonia, nitrate, and phosphate. Generally, aeration adjustments are made based on the bench readings. The operators have found discrepancies between the Chemscan and bench measurements and are not comfortable using Chemscan for control of aeration. However, the Chemscan results from the SCADA system are used to observe overall responses to changes (For example, “Is the ammonia concentration continuing to increase?”).

4.1.2 Wasting Control

The operations staff at the Falkenburg AWWTP has a targeted MLSS concentration that varies with the seasons due to a perceived effect of temperature. The volume of WAS wasted daily from the RAS line is calculated to meet these target MLSS concentrations. It is important to note that the amount of space in the WAS storage tank is also a factor that limits the ability to waste as needed.

Once the total mass of solids to be wasted has been calculated, the gallons to be wasted daily and the pump flow rate are estimated. Currently, Falkenburg AWWTP staff are working towards a 24-hour wasting regimen instead of only wasting a portion of the day. The calculated 7-day moving average MCRT (including solids in the clarifiers) and the average MLSS concentration for all four oxidation ditches are shown in Figure 4.2. The MCRT oscillates as wasting is performed to meet a target MLSS concentration.

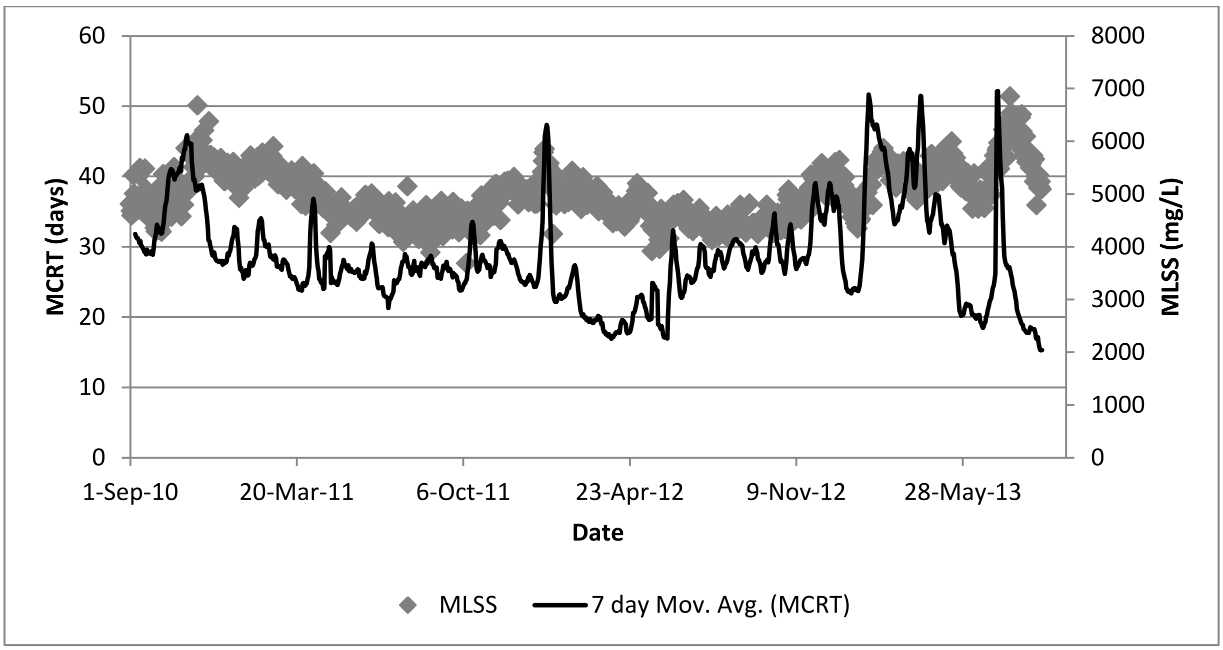


Figure 4.2 Average MLSS concentration and MCRT from Sept 1, 2010 to August 31, 2013.

The optimal MCRT for the Falkenburg AWWTP is one that achieves the necessary level of treatment, specifically for nitrification, and also produces sludge that settles well. Operating at unnecessarily high MCRTs and high MLSS concentration will increase the amount of inert solids in the system, increasing loading on the clarifiers, and will have a negative impact on sludge quality at a certain point. Conversely, if the MCRT is too low, nitrifying bacteria will be washed out of the system and ammonia concentration will increase in the effluent. One concern that was mentioned by the operating staff was not knowing when nitrifying and denitrifying organisms are being wasted because both organisms are present in a single sludge.

4.1.3 Data Comparisons between Laboratories

Concentrations of total P, ammonia, and nitrate in the final effluent measured by the County laboratory for reporting purposes and in filtered mixed liquor samples from the effluent of the oxidation ditch measured by operators in the plant laboratory are shown in Figure 4.3. Total P results were similar, while discrepancies between ammonia and nitrate values were observed. This suggests that further nitrification is occurring after the oxidation ditch. Biofilms located after the ditch in the deep bed media filters could further nitrify the effluent with sufficient DO. Further research to measure DO, as well as ammonia and nitrate before and after the filters could aid in identifying the source of the data discrepancies. Another possibility is that the two methods used by the respective labs produce varying results.

The BioWin model outputs were compared to the County laboratory data. If the data recorded by the operators are correct, the model predictions should be calibrated to these values. It is recommended that a filtered sample of mixed liquor be taken to the County lab for analysis. This may be especially important since the operators use the values obtained from their bench-top lab to make adjustments to the mechanical aerators.

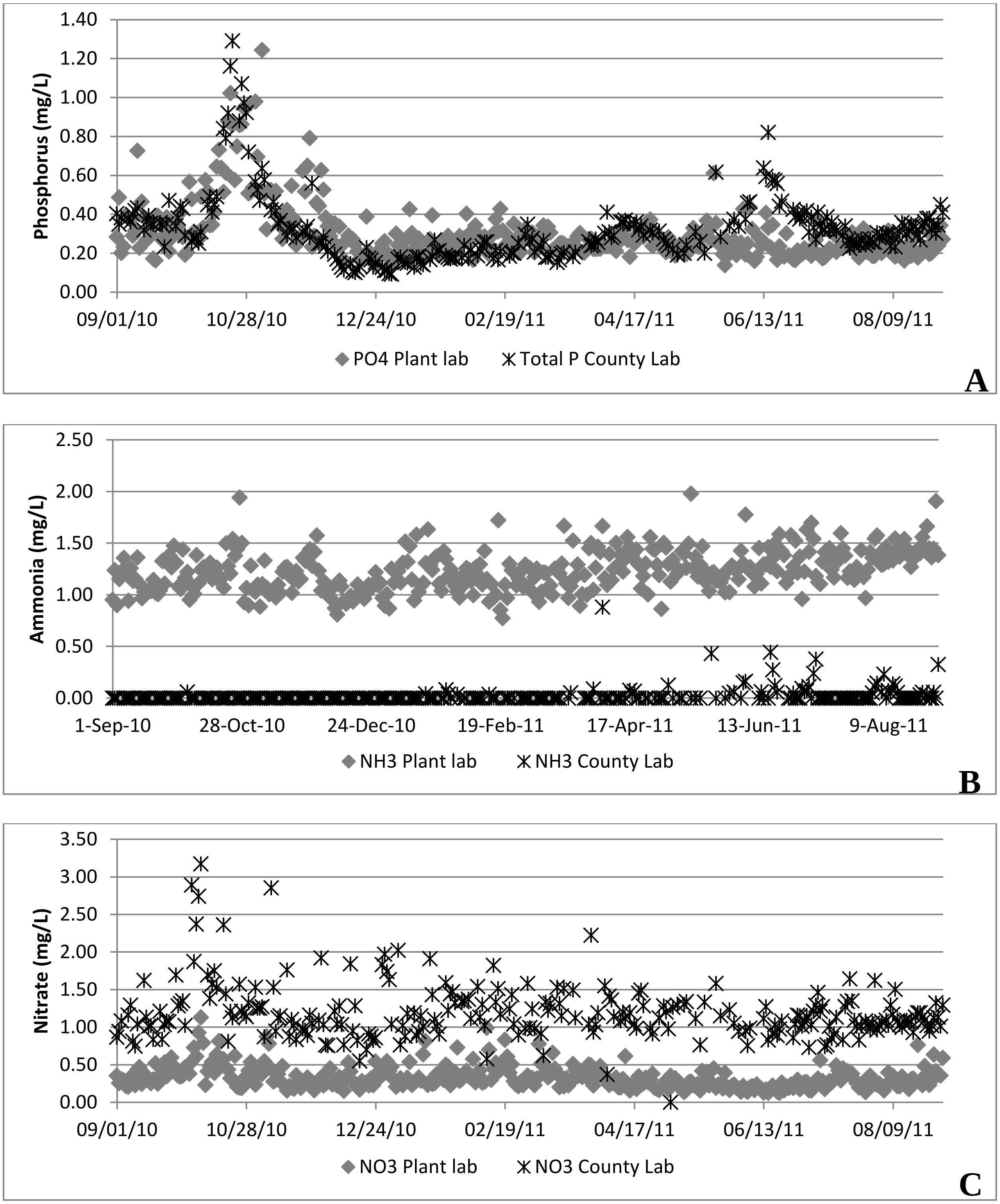


Figure 4.3 Concentration of total phosphorus (A), ammonia (B), and nitrate (C) measured in the final effluent by the County lab and in filtered mixed liquor at the effluent of the oxidation ditch measured by operators in the plant lab.

4.2 BioWin Calibration and Validation

A calibration and validation were performed for the Falkenburg BioWin model. This required supplemental sampling to be carried out for wastewater characterization to obtain values for COD and VSS. These values were used along with historical plant data to determine influent wastewater fractions for the model. Several kinetic parameters were adjusted during the calibration and goodness of fit analyses were used to select the best parameter arrangement. Finally, the model was validated using a dataset from a time period other than that used during calibration.

4.2.1 Wastewater Characterization

The results of the COD analyses on influent and effluent samples are shown in Figure 4.4. All influent samples were 24-hour composites and secondary effluent samples were grab samples. It is important to note that the effluent grab sample was collected from the secondary clarifier effluent before the media filters.

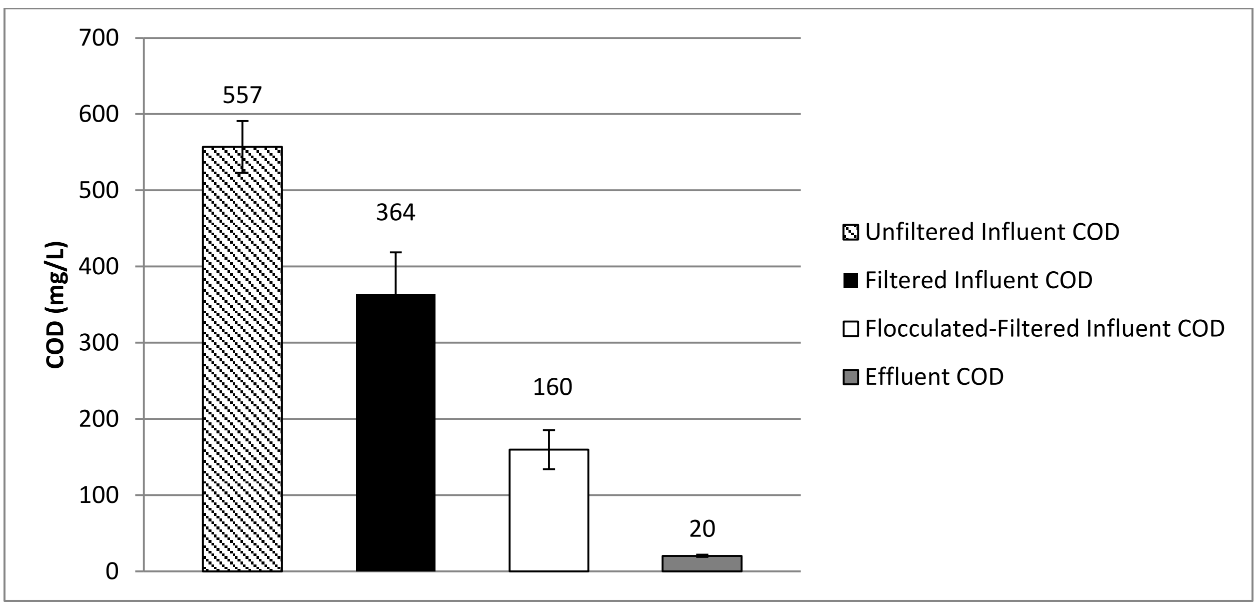


Figure 4.4 COD results from wastewater characterization. For influent n=5, for effluent n=2.

Total and volatile suspended solids in both composite and grab samples are shown in Table 4.1. BioWin requires volatile or inert suspended solids concentrations to be input into the model. Only historical TSS data were available, therefore VSS concentrations were estimated using the average VSS/TSS ratio determined during supplemental sampling.

Table 4.1 Influent TSS and VSS in composite and grab samples

Date	Time	TSS	VSS	VSS/TSS	Type
3-Mar-14	-	195	170	0.874	Composite
14-Apr-14	12:00 PM	217	189	0.871	Grab
14-Apr-14	6:00 PM	216	192	0.889	Grab
15-Apr-14	2:00 AM	240	227	0.946	Grab
15-Apr-14	6:00 AM	119	107	0.899	Grab
16-Apr-14	-	195	168	0.865	Composite
14-May-14	-	212	176	0.828	Composite
Average		**199**	**176**	**0.882**	
SD		**38.5**	**36.3**	**0.036**	

Total and filtered COD, TSS, VSS, and total and reactive phosphorus concentrations were measured over a 24-hour period and the results are shown in Figures 4.5, 4.6, and 4.7. The hourly influent flow was also recorded and used to calculate the mass load per day of each constituent (Appendix E). Noticeable peaks for both phosphorus and suspended solids were observed at 22:00. The color of the sample at time 22:00 was black, and the results from this sample were not used for estimation of influent characteristics. The reactive phosphorus concentrations (average of 12.6 mg/L) were consistently higher than the orthophosphate concentrations (average 6.1 mg/L) measured by the County laboratory. This should be investigated further. Table 4.2 compares the average values obtained from historical data or during supplemental sampling to typical values. The historical and measured values fell within the medium to medium-high range except for the calculated value for inorganic suspended solids, equal to the difference between TSS and VSS, which was very low.

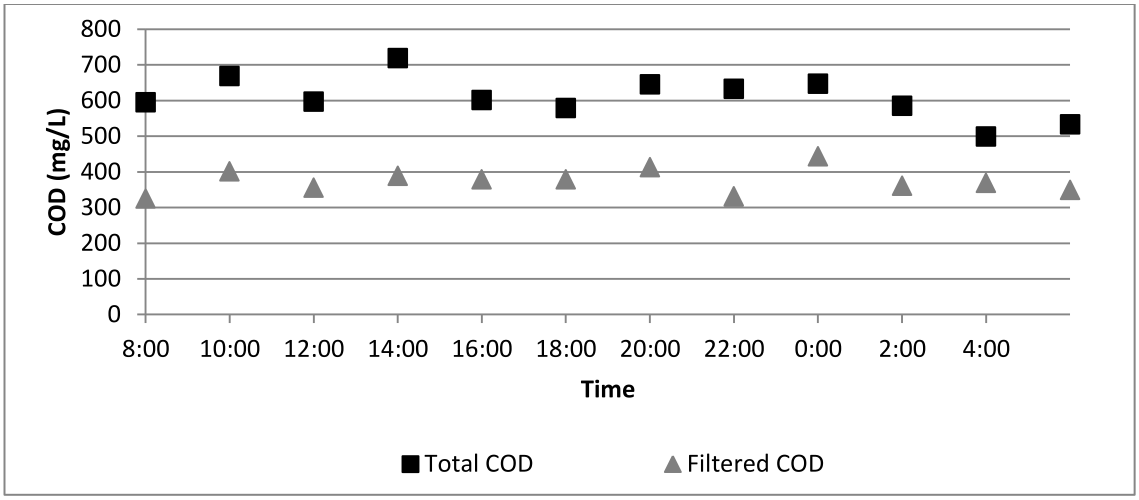


Figure 4.5 Total and filtered influent COD.

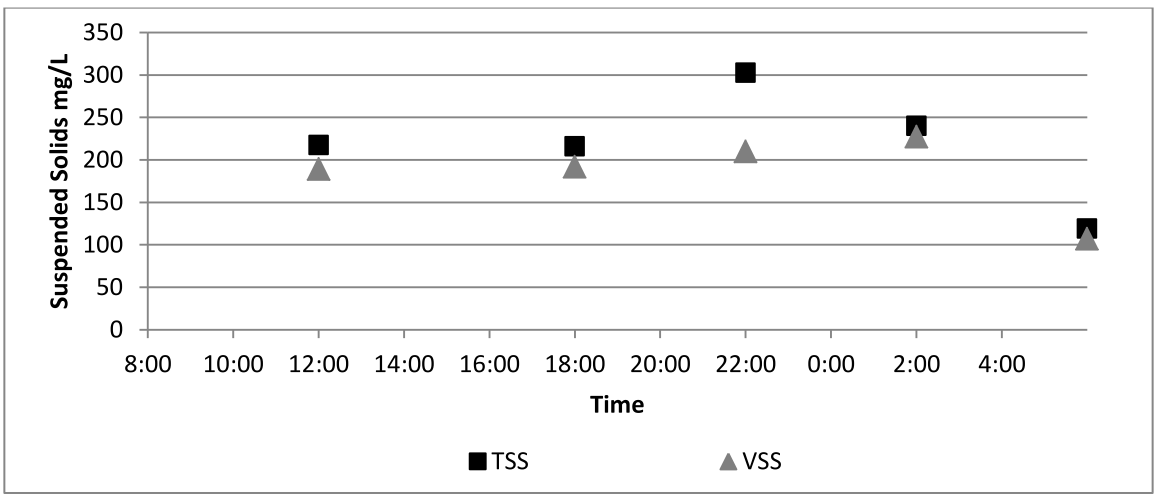


Figure 4.6 Total and volatile influent suspended solids.

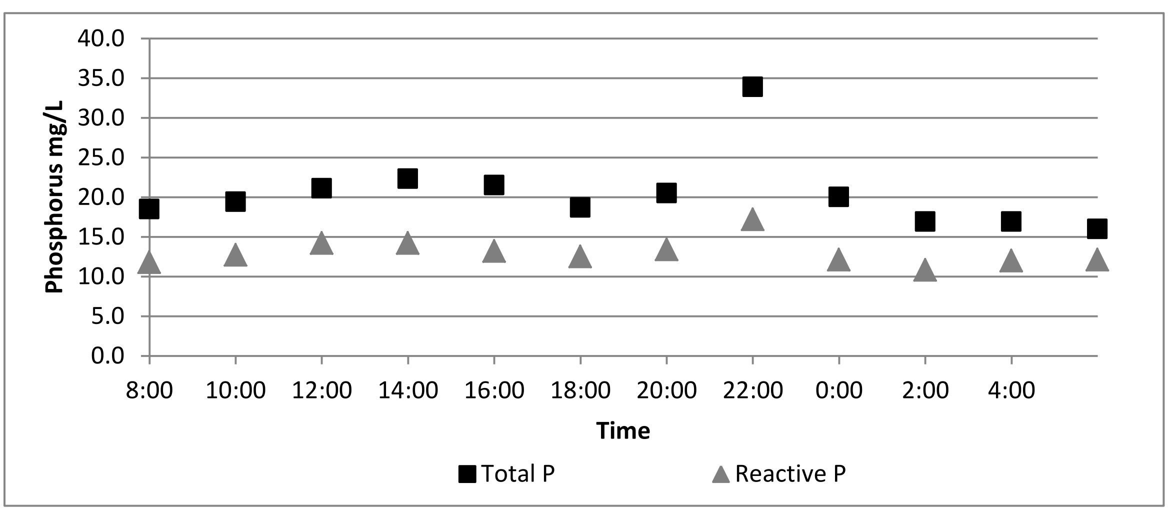


Figure 4.7 Total and reactive influent phosphorus.

Table 4.2 Comparison of literature values to average values of influent parameters from historical data and supplemental sampling

Parameter	Range in mg/L (Metcalf & Eddy, 2003) Low Strength	Medium Strength	High Strength	Falkenburg Average Data (mg/L)	Data Source	Strength
COD	250	430	800	**557**	Measured in USF lab	**Medium-High**
BOD	110	190	350	**290**	Historical County data	**Medium-High**
TSS	120	210	400	**192 / 205**	Historical County data / Measured in USF lab	**Medium**
VSS	95	160	315	**169 / 181**	Measured in USF lab/ Estimated from historical TSS data	**Medium**
ISS	25	50	85	**24**	Calculated from TSS & VSS measured in USF lab	**Low**
TKN	20	40	70	**51.2**	Historical County data	**Medium-High**
TP	4	7	12	**6.1**	Historical County data	**Medium**

The calculated values from BioWin Influent Specifier Excel worksheet are given in Table 4.3. Some values were calculated using the results from the COD analyses (Figure 4.4), while others such as the unbiodegradable particulate fraction had to be estimated.

Table 4.3 BioWin wastewater fractions

Fraction	Symbol	Units	BioWin Default Value	Calculated Value
Readily biodegradable COD	F_{bs}	g COD/g COD_{total}	0.16	0.254
Acetate	F_{ac}	g COD/g rbCOD	0.15	0.141
Non-colloidal slowly biodegradable COD	F_{xsp}	g COD/ g slowly biodegradable COD	0.75	0.400
Soluble unbiodegradable COD	F_{us}	g COD/g COD_{total}	0.05	0.033
Particulate unbiodegradable COD	F_{up}	g COD/g COD_{total}	0.13	0.110
Ammonia	F_{na}	g NH_3-N/g TKN	0.66	0.743
Particulate organic N	F_{nox}	g N/g organic N	0.5	0.500
Soluble unbiodegradable TKN	F_{nus}	g N/g TKN	0.02	0.000
N:COD ratio for unbiodegradable particulate COD	F_{upN}	g N/g COD	0.35	0.35
Phosphate	F_{po4}	g PO_4-P/ g TP	0.5	0.638
P:COD ratio for unbiodegradable particulate COD	F_{upP}	g P/g COD	0.011	0.011

4.2.2 Goodness of Fit

The average SAR results are given in Table 4.4. A lower number indicated a better fit of modeled to observed data. Although Simulations 8 and 9 resulted in lower SAR, Simulation 5 parameters were selected as the best fit. The values in Simulation 5 for the heterotrophic and aerobic denit. DO half saturation constants were close to the value of 0.25 mgO_2/L recommended by EnviroSim to model SND in an oxidation ditch. It was not known if a value of 0.5 mg O_2/L was a reasonable adjustment, and the improvement in results between Simulation 5 and 8 or 9 were less than those observed with the change seen between Simulations 3 or 4 and Simulation 5. The final calibrated kinetic parameters are shown in Table 4.5.

Table 4.4 Average sum of absolute residuals of effluent N concentrations with varying kinetic parameters

Simulation	Effluent NH_3-N (mg/L)	Effluent TKN (mg/L)	Effluent NO_3 (mg/L)	Effluent NO_2 (mg/L)
1*	1.01	1.37	1.90	0.865
2	0.781	1.14	1.84	0.833
3	0.665	0.999	1.23	0.336
4	0.784	1.12	1.02	0.397
5	0.437	0.778	0.908	0.291
6	0.567	0.905	0.884	0.355
7	0.614	0.934	1.09	0.284
8	0.436	0.769	0.789	0.204
9	0.368	0.680	0.807	0.193

*BioWin default kinetic parameter values

Table 4.5 BioWin kinetic parameters

Parameter	Default Value	Calibrated Value
AOB Maximum Specific Growth Rate	0.9	0.9
NOB Maximum Specific Growth Rate	0.7	0.7
OHOs Maximum Specific Growth Rate	3.2	3.2
Heterotrophic DO half saturation constant	0.05	**0.3**
Aerobic denitrification DO half saturation constant	0.05	**0.3**
Ammonia oxidizer DO half saturation constant	0.25	**0.15**
Nitrite oxidizer DO half saturation constant	0.5	0.5
Anaerobic ammonia oxidizer DO half saturation constant	0.01	0.01
Anoxic NO3-N half saturation constant	0.1	0.1
Anoxic NO2-N half saturation constant	0.01	**0.05**
NH3-N nutrient half saturation constant	1.00E-04	1.00E-04

Charts were prepared in the BioWin model to plot modeled values against data observed at the Falkenburg AWWTP from September 1, 2010 to August 31, 2014. The modeled MLSS concentration in the calibrated model was charted against the observed MLSS (Figure 4.8). These concentrations resulted from a constant average wasting rate of 0.234 mgd. Changing the daily WAS flow to more accurately reflect plant wasting could produce a better fit of modeled to observed data.

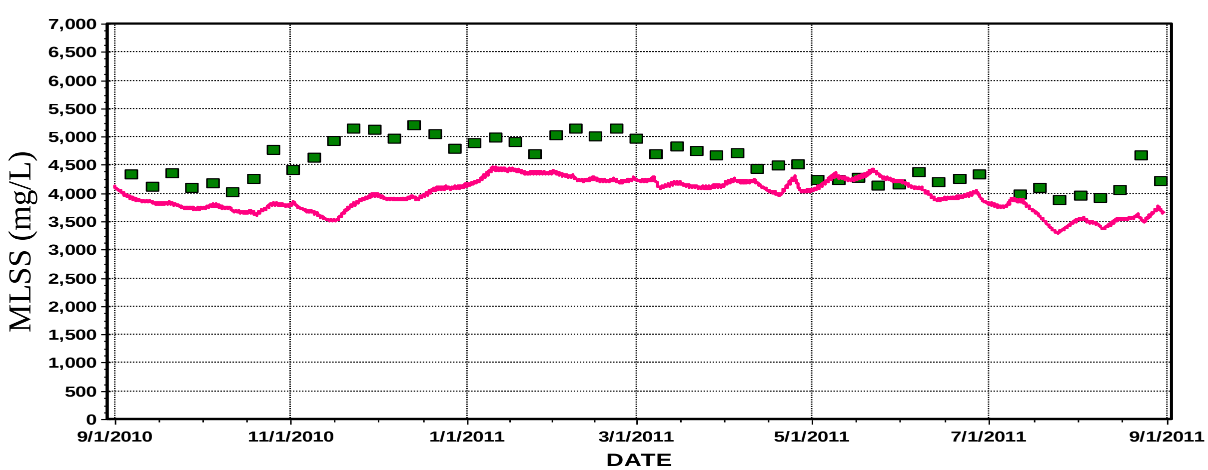


Figure 4.8 Observed (green squares) and modeled (pink line) MLSS concentration.

A poor fit was observed between modeled and observed MLVSS concentrations (Figure 4.9). The model-predicted MLVSS is usually adjusted by changing the influent unbiodegradable particulate COD fraction. However, it was not known if the discrepancy in MLVSS was solely attributed to model input or if addition of alum played a role. Metal hydroxides, such as those formed during alum addition, are oxidized during VSS analysis in the muffle furnace, which will result in a falsely high MLVSS concentration (Jeyanayagam and Husband, 2009). Adjustments to influent COD fractions could be made once it is determined if alum is contributing to the VSS concentration.

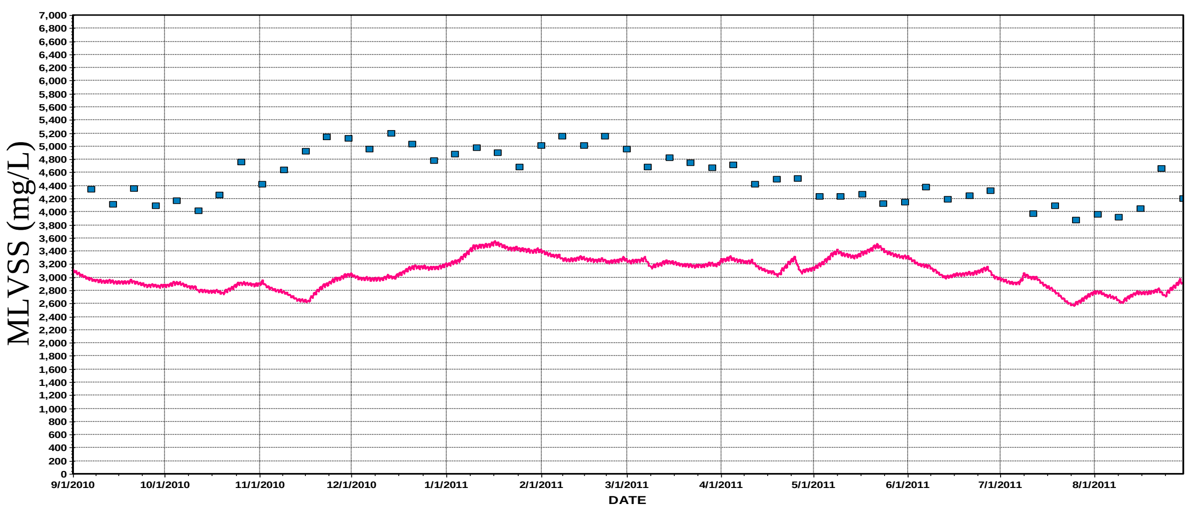


Figure 4.9 Observed (blue squares) and modeled (pink line) MLVSS concentration.

Spikes in effluent nitrogen species were observed several times during the year-long simulation (Figure 4.10). These spikes correspond with the influent loading of TKN (Figure 4.11). The model is sensitive to N loading at higher flow or higher influent TKN concentration. In the WWTP, operators make changes to mechanical aerator speed in response to oxidation ditch ammonia concentration. The model maintained a constant DO set point in the mechanical aerator reactors. Fine-tuning the model aeration settings to better reflect the aeration practices of the WWTP could reduce the modeled effluent spikes, and produce improved goodness-of-fit of modeled to observed parameters.

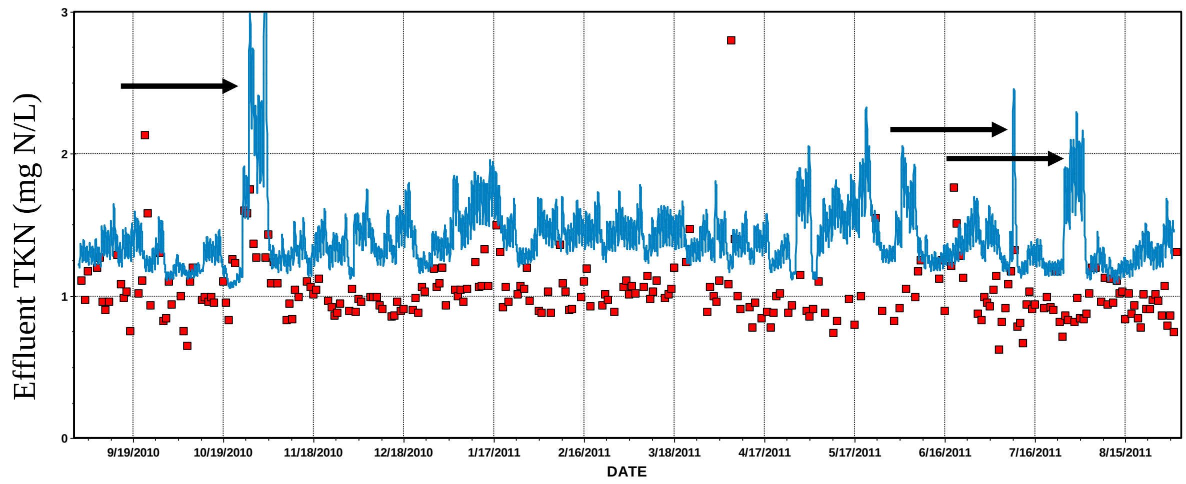


Figure 4.10 Observed (red squares) and modeled (blue line) effluent TKN concentration.

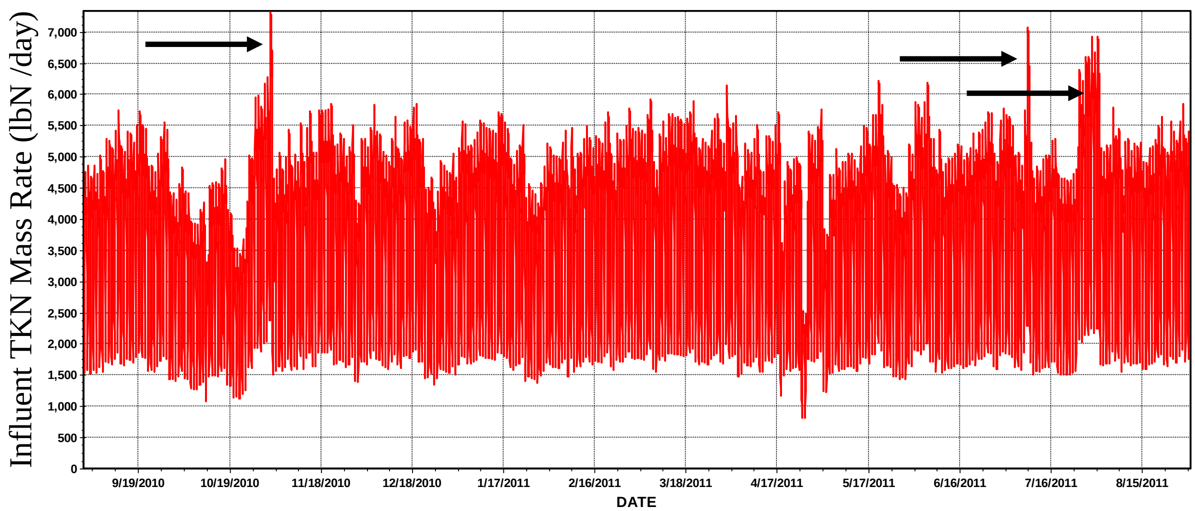


Figure 4.11 Influent TKN mass load.

4.2.3 Sensitivity Analysis

The results of sensitivity analysis show that altering kinetic parameters affects the resultant effluent concentrations differently (Table 4.6). AOB maximum specific growth rate is very influential for effluent ammonia concentrations. The maximum specific growth rate for nitrite oxidizing bacteria does not influence the effluent ammonia concentration, but is very influential on effluent nitrite and nitrate concentration. The combined heterotrophic/aerobic denit DO half saturation constant is most influential on effluent nitrate concentration. This is expected because this constant switches on the activity of anoxic OHOs under low DO conditions. By increasing this parameter, the anoxic OHO activity is turned on at a higher concentration of DO. The ammonia oxidizer DO half saturation constant was only slightly influential on the ammonia concentration, while the anoxic NO_2 half saturation constant mostly influenced the effluent nitrate concentration. The anoxic NO_2 half saturation constant switches off anoxic growth process at low nitrite concentrations.

Table 4.6 Sensitivity analysis for kinetic parameters

	AOB max spec growth rate	**NOB max spec growth rate**	**Heterotrophic/ Aerobic denit DO Half Sat**	**Ammonia Oxidizer DO Half Sat**	**Anoxic NO2 Half Sat**
Δx	**0.1**	**0.1**	**0.1**	**0.1**	**0.01**
Ammonia	2.45	0	0.273	0.409	0
Nitrate	1.14	2.43	1.21	0.237	0.71
Nitrite	0.474	1.84	0.474	0	0.263

4.2.4 Validation

Data from September 1, 2011 to August 31, 2012 were entered into the calibrated BioWin model and the model was simulated for a one year period. The modeled effluent concentrations were compared to observed effluent concentrations measured by the County laboratory for the same period. Results of the goodness of fit comparison using the average sum of absolute residuals are shown in Table 4.7. The goodness of fit was comparable to the fit obtained during the calibration except for nitrate. Nitrate concentrations modeled during the validation were relatively high. The concentration of effluent nitrate is important and the calibrated model should be investigated further before future modeling simulations are carried out.

Table 4.7 Average sum of absolute residuals of effluent N concentrations for validation simulation

Effluent NH_3-N (mg/L)	**Effluent TKN (mg/L)**	**Effluent NO_3 (mg/L)**	**Effluent NO_2 (mg/L)**
0.212	0.487	2.09	0.179

4.2.5 Simulation and Result Interpretation

Figure 4.12 shows the change in MLSS concentration in the oxidation ditch as SRT increases. During this simulation, the influent COD load was kept constant while the average wasting rate per day, and the SRT, was manipulated. This shows that at a constant influent load, MLSS will increase with increasing SRT. When SRT is kept constant and influent COD fluctuates, the MLSS concentration increases with increasing influent COD. During the same simulation, effluent ammonia was also recorded, and the results were plotted against SRT (Figure 4.13). The results show that once the minimum SRT is reached for ammonia oxidizing bacteria (approximately 10 to 20 days in this case), effluent ammonia concentrations remain low. At SRTs of 5, 10, 15 and 20 days, the effluent ammonia concentrations were 1.46, 0.46, 0.29 and 0.23, respectively.

For the second set of simulations, the SRT was kept constant at 20 days and the influent COD was changed to values between 200 and 600 mg/L. Figure 4.14 shows the response of the MLSS to changing influent COD. When SRT is kept constant and influent COD fluctuates, the MLSS concentration increases with increasing influent COD. Effluent ammonia and nitrate were also plotted against influent COD (Figure 4.15). Effluent ammonia remained low (0.18-0.24 mg/L) at all influent COD concentrations, while effluent nitrate concentration decreased with increasing influent COD. Denitrifying bacteria are heterotrophic and require sufficient carbon.

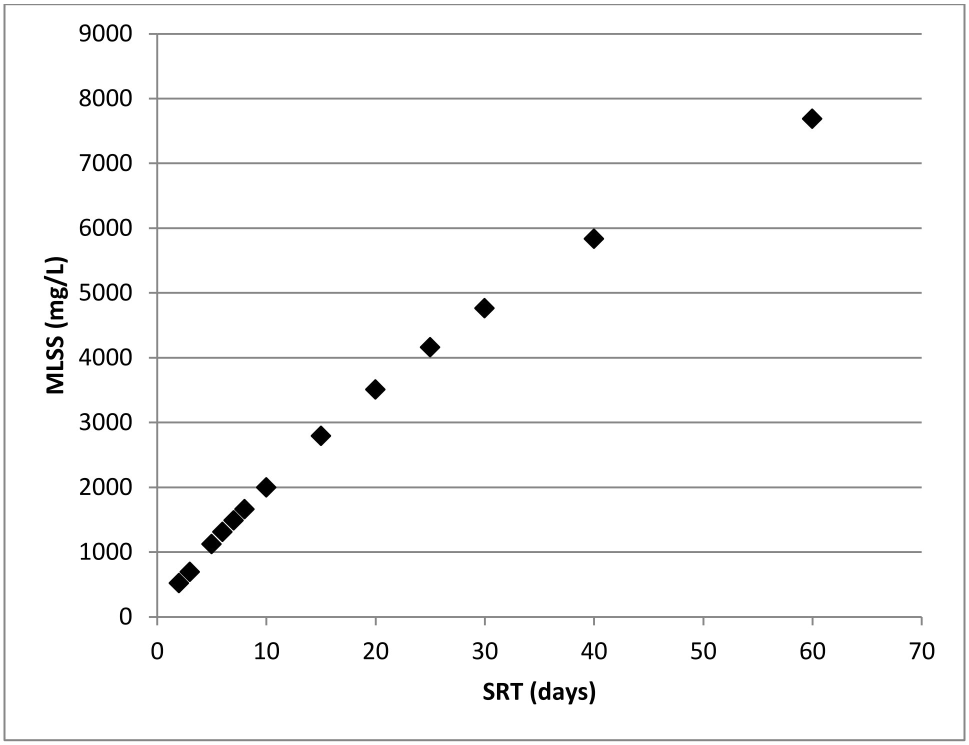


Figure 4.12 Response of MLSS concentration to changing SRT during steady state simulations with constant influent parameters and aeration regimen.

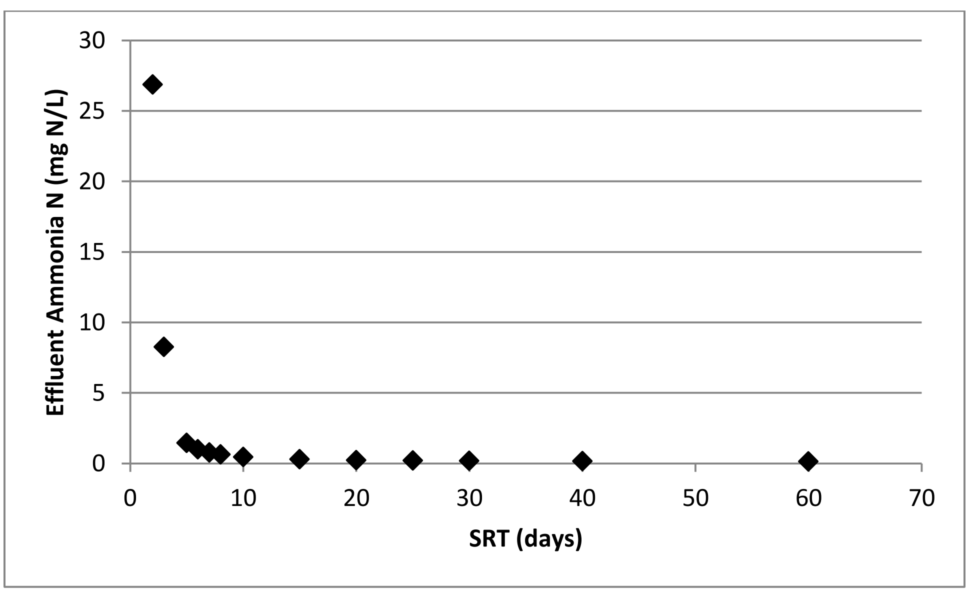


Figure 4.13 Response of effluent ammonia concentration to changing SRT during steady state simulations with constant influent parameters and aeration regimen.

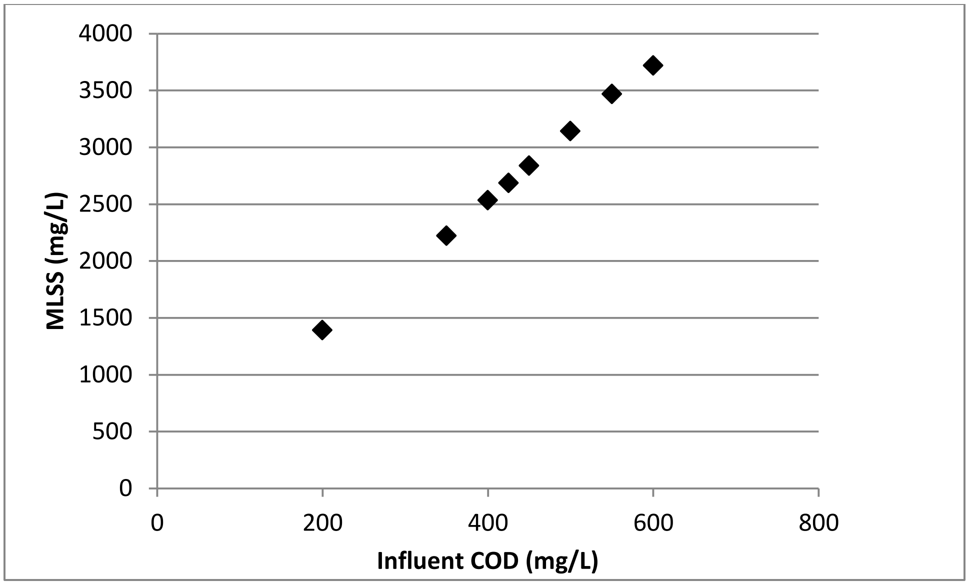


Figure 4.14 Response of MLSS concentration to changing influent COD concentration during steady state simulations with constant SRT (20 days) and aeration regimen.

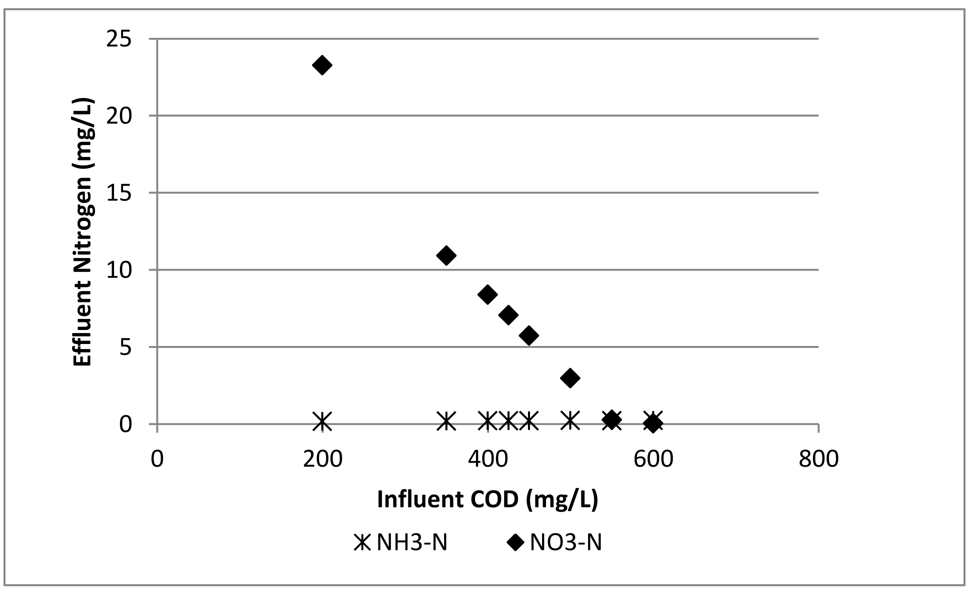


Figure 4.15 Response of effluent ammonia and nitrate concentrations to changing influent COD concentration during steady state simulations with constant SRT (20 days) and aeration regimen.

4.3 Investigation of Phosphorus Removal

The results from the analysis of total and reactive phosphorus at various points in the treatment train are shown in Figure 4.16. Although the samples size (n=2) is low, the results indicate that biological phosphorus removal is taking place. A characteristic release of phosphorus is observed in the unaerated selector, followed by very low phosphorus in the effluent of the aerated reactor. The average release of phosphate in the selector was 32 mg/L, and the total amount of phosphate removed was 45 mg/L. Similar phosphate release and uptake were reported by Henze (2008) with a phosphate release of 45 mg/L, uptake of 57 mg/L, and total removal of 12 mg/L. It is not possible to assume the removal of P in the ditch is fully attributed to EBPR since aluminum sulfate (alum) is also added for chemical phosphorus removal. Alum is dosed at a constant rate (~260 gpd) into a splitter box after the oxidation ditches and before the secondary clarifiers. Jar testing to determine the optimum dose of alum was discussed with plant staff, but this was later determined to be unfeasible due to the presence of alum in the RAS and mixed liquor. Flow-pacing of alum and slowly decreasing alum dosing while observing the effect on effluent total phosphorus concentration are suggested. This will reduce chemical costs, sludge production, and possible impacts of alum on the biological process.

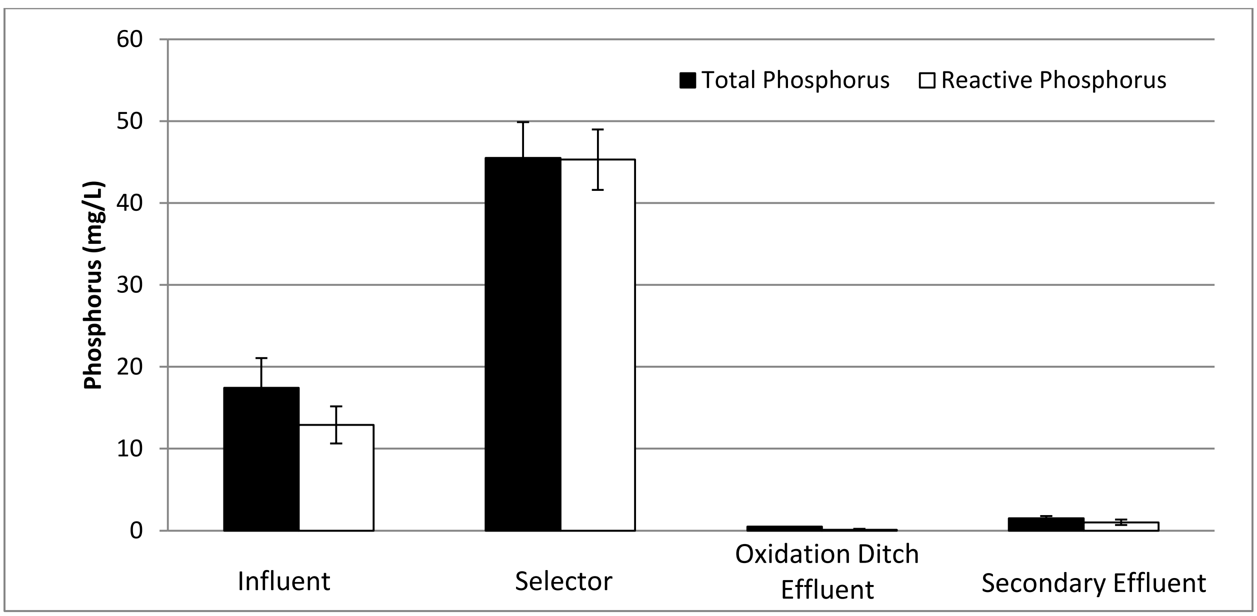


Figure 4.16 Reactive phosphorus profile from grab samples taken throughout the treatment process.

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

The Operations Staff at the Falkenburg AWWTP consistently meet and exceed NPDES permit limits. In addition to meeting permit, the goal of the operator is to also reduce costs and energy usage associated with plant operation. These savings will reduce emissions of greenhouse gases and costs to County ratepayers. There are three areas where improvements could be made to improve efficiency of operation: 1) alum addition for phosphorus removal 2) wasting based on MCRT and 3) online aeration control based on ammonia concentration. Activated sludge modeling enables the user to observe the virtual effect of changes in operation and influent loading relatively quickly and without risk of permit violation. A BioWin model was created for the Falkenburg AWWTP, and several simulations were run to observe relationships between MLSS, MCRT, and influent loads. Finally, data compilation and reconciliation conducted during this study highlighted many good practices in plant operation and monitoring, with only a few gaps for future improvement.

The addition of alum for phosphorus removal at the Falkenburg AWWTP should be optimized to ensure that NPDES permits are met without overdosing. In addition to reducing costs and chemical usage, improvements in sludge dewatering may result from optimization of alum addition. A brief investigation of phosphorus throughout the treatment train showed a release of phosphorus in the anaerobic selector characteristic of enhanced biological phosphorus removal. The Operations Staff at the Falkenburg AWWTP is planning to reduce the amount of alum that is added for chemical phosphorus removal by flow-pacing and observing the effects of an overall decrease in dose. Because alum addition also improves settling in the clarifiers, a reduction in alum addition may increase secondary effluent turbidity if sludge quality is poor and further necessitate finding the MCRT where sludge quality is best.

The time that bacteria remain in the system, the SRT or MCRT, is one of the most important factors affecting plant performance. The optimal MCRT for the Falkenburg AWWTP should be found and wasting practices should be changed to maintain this target MCRT. Currently, wasting is controlled by keeping the constant MLSS method, while disregarding fluctuations in MCRT. The optimal MCRT for the Falkenburg AWWTP is one that achieves the necessary level of treatment, specifically for nitrification, and also produces sludge that settles well. Operating at unnecessarily high MCRTs and high MLSS concentration will increase the amount of inert solids in the system, increasing loading on the clarifiers, and can have a negative impact on sludge quality. For nitrification, it is important to know the MCRT of solids that are under aeration. Currently, the MCRT calculation at the Falkenburg AWWTP incorporates the mass of solids in the clarifiers as well. It is recommended that both the MCRT of the total system and the aerobic MCRT be calculated to ensure that the minimum sludge age for nitrification is being met. One challenge in calculating sludge age, as well as SVI, is accurately measuring the MLSS concentration. MLSS concentrations can fluctuate over the course of a day, and collecting and analyzing one grab sample per day may not adequately represent the MLSS concentration. A composite sample of MLSS will give a more accurate estimate of the average MLSS concentration. Another option is a MLSS analyzer that can measure concentrations quickly and more frequently and can be coupled with the SCADA system.

Implementing aeration control using online analyzers is increasingly being used at AWWTPs. The use of online analyzers allows for more frequent monitoring of aeration basin conditions and automatic adjustment based on pre-defined parameters. The Falkenburg

AWWTP has ChemScan units that monitor ammonia, phosphate, and nitrate. Automatic control of mechanical aerators should be reevaluated as a process control option for nitrification and denitrification. The online analyzers will be able to respond more quickly to changes in influent ammonia loads. Although aeration is critical, it is important to remember the role of MCRT in nitrification as well. A minimum MCRT must be met to allow nitrifiers to grow in the system.

One objective of this study was to construct and calibrate a BioWin model of the Falkenburg AWWTP. The BioWin model was able to achieve simultaneous nitrification and denitrification in the modeled oxidation ditch by adjusting several half saturation constants. The model was calibrated for nitrification and denitrification. If phosphorus removal is to be investigated, the model should be reevaluated. The model was found to be sensitive to changes in the maximum specific growth rates of ammonia and nitrite oxidizing bacteria. These parameters should be measured experimentally for future model calibration. A limited number of simulations were run with the model. Steady state simulations were run with the model to observe the effect of variable and constant SRT (or MCRT) and changing influent loads. In each simulation only one variable was manipulated. While this is not reflective of normal plant operation, it allows the effect of each variable change to be observed. Plotting effluent ammonia concentration at varying SRT shows that SRT has little effect at decreasing ammonia concentration after the minimum MCRT needed for nitrification is reached. As SRT increases, the MLSS increases due to the buildup of inert solids. At constant SRT, MLSS varies with influent COD concentration, because bacteria grow as food becomes available. Mixed liquor suspended solids is a response variable and should not be used as a control variable.

Comparisons of oxidation ditch effluent collected and analyzed by plant operators and final effluent samples analyzed by the Hillsborough County certified laboratory showed relatively good agreement, especially for phosphorus. Small differences in ammonia and nitrate were observed, which may be due to further biological activity after the oxidation ditch. Ammonia, nitrate, and DO should be measured at points between the oxidation ditch and final effluent to further investigate these discrepancies. BioWin modeling requires additional inputs, such as VSS and fractions of COD, which are not currently measured at the Falkenburg AWWTP. Periodically monitoring these parameters will aid in future modeling projects. It was also not possible to fully assess the operation of the clarifiers and sludge quality, because no data were available on the amount of suspended solids leaving in the secondary clarifier effluent. A simple turbidity test with secondary clarifier effluent will give the operators more information about the ability of the sludge to flocculate and settle in the clarifiers. This will be especially important when assessing the effect of MCRT control on sludge quality.

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APPENDICES

Appendix A List of Acronyms

AOB	Ammonia Oxidizing Bacteria
ASM	Activated Sludge Model
AWWTP	Advanced Wastewater Treatment Plant
BNR	Biological Nutrient Removal
$cBOD_5$	5-day Carbonaceous Biological Oxygen Demand
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
EBPR	Enhanced Biological Phosphorus Removal
F:M	Food to Microorganism Ratio
HRT	Hydraulic Residence Time
IWA	International Water Association
MCRT	Mean Cell Residence Time
MGD	Million Gallons Per Day
MLE	Modified Ludzack-Ettinger
MLSS	Mixed Liquor Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
NOB	Nitrite Oxidizing Bacteria
NPDES	National Pollutant Discharge Elimination System
OHO	Ordinary Heterotrophic Organisms
ORP	Oxidation Reduction Potential
OWP	Office of Water Programs (Sacramento, California)
PAO	Phosphorus Accumulating Organisms
RAS	Return Activated Sludge
SND	Simultaneous Nitrification Denitrification
SRT	Solids Retention Time
SSV	Settled Sludge Volume
SVI	Sludge Volume Index
TKN	Total Kjeldahl Nitrogen
TSS	Total Suspended Solids
VFA	Volatile Fatty Acids
VSS	Volatile Suspended Solids
WAS	Waste Activated Sludge
WEF	Water Environment Federation
WWTP	Wastewater Treatment Plant

Appendix B Influent Specifier Worksheet

Measurements	Value	Unit
Main influent concentrations		
Flow	8.8	mgd or m3/d
Total COD	557.0	mgCOD/L
Total Kjeldahl Nitrogen	51.2	mgN/L
Total P	20.0	mgP/L
Other influent concentrations		
Nitrate N	0.0	mgN/L
pH	7.3	
Alkalinity (CaCO3 equivalent)	300.0	mgCaCO3/L
Calcium	160.0	mg/L
Magnesium	25.0	mg/L
Dissolved oxygen	0.0	mgO2/L
Other measurements		
Effluent filtered COD	20.0	mgCOD/L
Influent filtered COD (GFC)	364.0	mgCOD/L
Influent FF COD	160.0	mgCOD/L
Influent acetate	20.0	mgCOD/L
Influent ammonia	38.0	mgN/L
Influent ortho-phosphate	6.4	mgP/L
Influent carbonaceous BOD5	318.0	mgO2/L
Influent filtered cBOD5	246.0	mgO2/L
Influent VSS	181.0	mgVSS/L
Influent TSS	205.0	mgTSS/L

GUIDE

- Enter measured lab data in column on left (**BOLD**)
  (If data is missing, estimate. May need to repeat after Step 2)
- Check resulting fractions **(BOLD)**

Parameter	Value	Unit	Typical range
Alkalinity (molar)	**6.0**	**meq/L**	**2 - 6**
Fus	**0.03**	-	**0.03 - 0.08**
CODp	**193.0**	**mgCOD/L**	
Fbs	**0.25**	-	**0.12 - 0.25**
Fac	**0.14**	-	**0.0 - 0.3**
Fna	**0.74**	-	**0.5 - 0.8**
Fpo4	**0.32**	-	**0.3 - 0.6**
COD/BOD5	**1.75**	-	**1.9 - 2.2**
Fcv	**1.07**	**mgCODp/mgVSS**	**1.5 - 1.7**
ISS	**24.0**	**mgISS/L**	**15 - 45**

Figure B.1 BioWin influent specifier worksheet.

Appendix C Distribution of Mass

The distribution of mass between the clarifiers, oxidation ditches, and selectors is shown in Figure C.1. During the dynamic simulations, the clarifiers were modeled with a 4-foot deep sludge blanket, which is typical for the Falkenburg AWWTP. This pie chart shows that sludge blankets can have a large impact on the mass of solids residing outside the aeration basin. The mass of solids in the clarifiers is currently included in the calculation of MCRT. For nitrification, the MCRT of solids under aeration should be calculated as well.

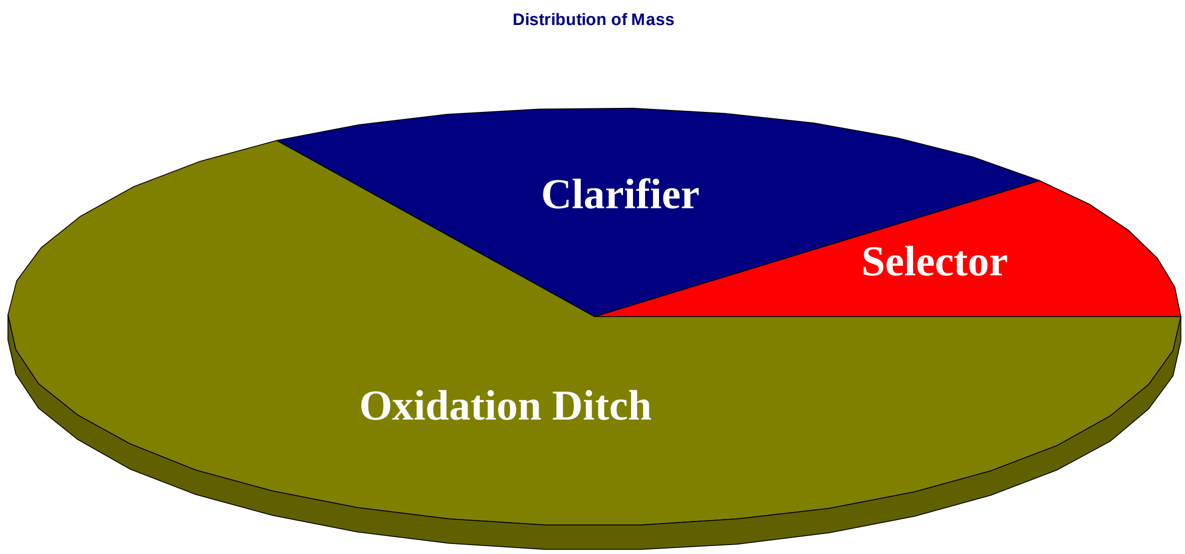

Figure C.1 Percent distribution of mass between clarifier (blue), selector (red), and oxidation ditch (green).

Appendix D Influent Loading

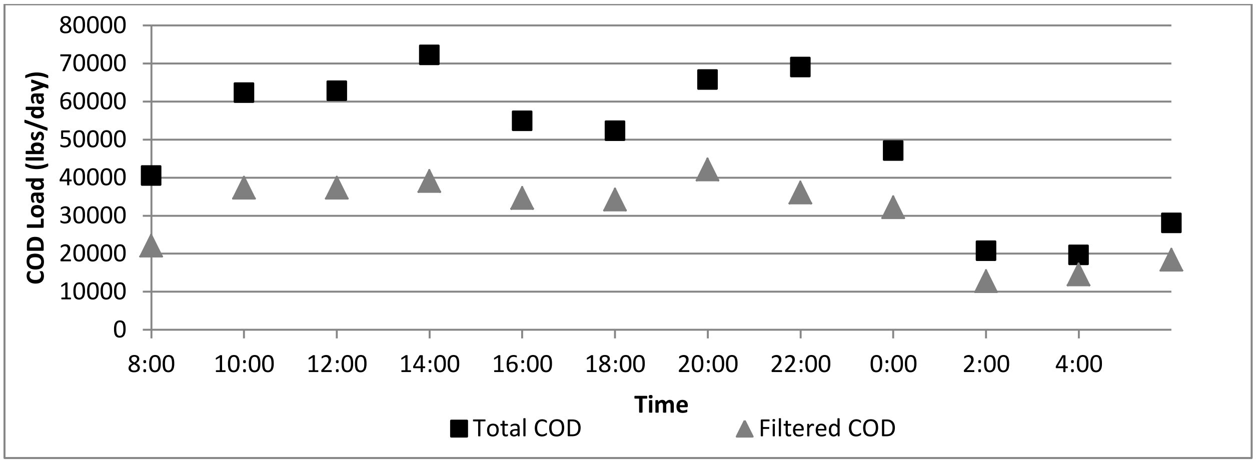

Figure D.1 Mass load of influent COD calculated from a 24-hour sampling event.

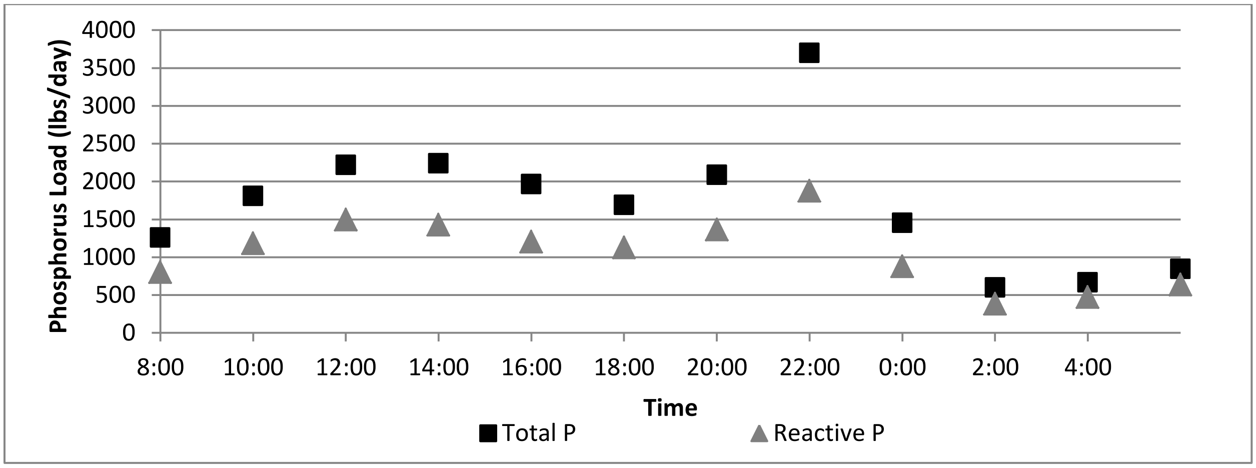

Figure D.2 Mass load of influent phosphorus calculated from a 24-hour sampling event.

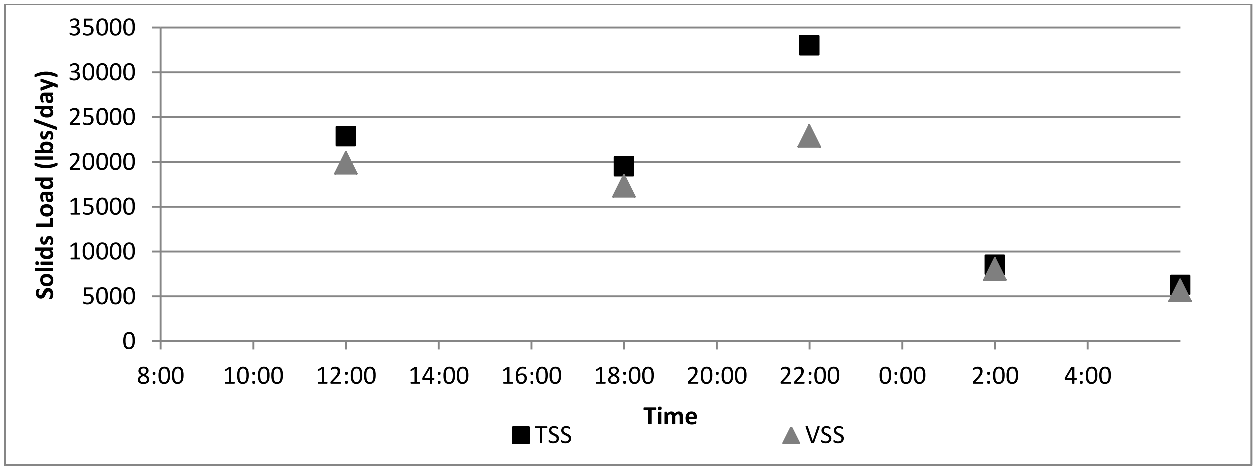

Figure D.3 Mass load of influent TSS and VSS calculated from a 24-hour sampling event.

Appendix E Composite Influent and Grab Effluent COD Analyses

Table E.1 Results from influent and effluent COD analyses

	Influent (Composite)			Effluent (Grab)	
Date	**Unfiltered COD (mg/L)**	**Filtered COD (mg/L)**	**Flocculated-filtered COD (mg/L)**	**Unfiltered COD (mg/L)**	**Filtered COD (mg/L)**
2/3/2014	509.6	291.6	121.6	-	-
2/19/2014	572.6	400.6	-	-	-
3/3/2014	543.8	427.8	167.8	-	-
4/18/2014	601.0	371.0	175.5	21.5	18.3
5/14/2014	557.8	326.8	173.8	19.3	20.8
Average	**557.0**	**363.6**	**159.7**	**20.4**	**19.5**
SD	**33.9**	**55.0**	**25.6**	**1.5**	**1.6**