# RAS Ratio Impacts to Biological Phosphorus Removal and Secondary Clarifier Denitrification

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## Authorization to Submit Thesis

This thesis of Brent D. Deyo, submitted for the degree of Master of Science with a Major in Civil Engineering and titled "RAS Ratio Impacts to Biological Phosphorus Removal and Secondary Clarifier Denitrification" has been reviewed in final form. Permission, as indicated by the signatures and dates below, is now granted to submit final copies to the College of Graduate Studies for approval.

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## Abstract

Return Activated Sludge (RAS) is the cornerstone of activated sludge treatment and research presented herein demonstrates the considerable impact it has on BPR process stability. Data collected at Moscow's full-scale and scale model Water Resource Recovery Facilities (WRRFs), as well as a process model demonstrate that increasing the RAS ratio, in turn increases the RAS nitrate load through decreased secondary clarifier denitrification and increased RAS flow containing nitrate. Impacts of the increased nitrate load are apparent within the anaerobic basins in terms of decreased phosphorus release and PHA synthesis. The results indicate that RAS rate is an important process control parameter that can be used to increase BPR process stability and success. Additionally, the BPR Operator Survey responses gathered as part of this research indicate many highly regarded WRRFs practicing BPR are beginning to use novel RAS control techniques to enhance their systems. Further discussion with the operators of these WRRFs would undoubtedly lead to valuable insight that could then be analyzed and disseminated to benefit the BPR community as a whole. The findings from this research will hopefully shine light on an otherwise largely neglected process control parameter, the RAS rate.

## Acknowledgements

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## Dedication

To my wife Aliesha and son Glen for helping me smile and remember to have some fun throughout this process.

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## List of Abbreviations

| | |
|---|---|
| 2TP | Secondary Clarifier Effluent TP |
| AE | Aerobic |
| AE1 | Scale Model Aerobic Basin 1 |
| AE2 | Scale Model Aerobic Basin 2 |
| AN | Anaerobic |
| AN1 | Moscow WRRF or Scale Model Anaerobic Basin 1 |
| AN2 | Moscow WRRF or Scale Model Anaerobic Basin 2 |
| AN3 | Moscow WRRF or Scale Model Anaerobic Basin 3 |
| AOB | Ammonia Oxidizing Bacteria |
| AX | Anoxic |
| AX1 | Moscow WRRF Anoxic Basin 1 |
| AX2 | Moscow WRRF Anoxic Basin 2 |
| BNR | Biological Nutrient Removal |
| BOD | Biochemical Oxygen Demand |
| BPR | Biological Phosphorus Removal |
| COD | Chemical Oxygen Demand |
| DO | Dissolved Oxygen |
| ffCOD | Filtered Flocculated COD |
| GAO | Glycogen Accumulating Organisms |
| gpm | Gallons per Minute |
| HAc | Acetic Acid |
| HRT | Hydraulic Residence Time |
| lb/d | Pounds per Day |

| | |
|---|---|
| mg/L | Milligrams per Liter |
| mgd | Million Gallons per Day |
| MLE | Modified Ludzak Ettinger |
| MLR | Mixed Liquor Return |
| MLSS | Mixed Liquor Suspended Solids |
| OHO | Ordinary Heterotrophic Organism |
| OD | Oxidation Ditch |
| P | Phosphorus |
| PAO | Phosphorus Accumulating Organism |
| PHA | Polyhydroxyalkanoate |
| PHB | poly-b-hydroxybutyrate |
| PHV | poly-b-hydroxyvalerate |
| PO4 | Phosphate |
| RAS | Return Activated Sludge |
| rbCOD | Readily Biodegradable COD |
| Redox | Oxidation-Reduction |
| sbCOD | Slowly Biodegradable COD |
| SCADA | Supervisory Control and Data Acquisition |
| SC-SRT | Secondary Clarifier Solids Retention Time |
| SDNR | Specific Denitrification Rate |
| SM | Scale Model |
| SRT | Solids Residence Time |
| SC-SRT | Secondary Clarifier Solids Residence Time |
| TS | Total Solids |

| | |
|---|---|
| TSS | Total Suspended Solids |
| VFA | Volatile Fatty Acid |
| VSS | Volatile Suspended Solids |
| WAS | Waste Activated Sludge |
| WEF | Water Environment Federation |
| WRRF | Water Resource Recovery Facility |

# 1 Introduction

Treatment of human waste is a cornerstone of a stable human society. Wastewater treatment has saved countless lives and allowed civilizations and their adjacent environments to prosper. The spread of disease carried in human waste has decimated cultures past and present. Beyond health of humans, wastewater left untreated can be incredibly detrimental to the natural environment. Cultural eutrophication, or excessive availability of nutrients in waterways associated with anthropogenic activities, is credited as the most widespread water quality problem on the planet (Schindler, 2012). Eutrophication can lead to devastating impacts to the environment including anaerobic conditions and the release of deadly algal toxins that are fatal to nearly anything that encounters them (Chorus et al., 2000). Regulators are continually increasing restrictions on nutrient discharge from treatment facilities for these reasons. While it would be ideal to return wastewater to a natural state perfectly in balance with the receiving water, it is not always technically feasible to do so. Moreover, treating wastewater is expensive and energy intensive.

There are various approaches to wastewater treatment depending on its characteristics, the sensitivity of the receiving water, NPDES permit criteria, and potentially desired end uses (Tchobanoglous et al., 2014). Typically, the purer the effluent water, the higher the treatment cost. However, there are opportunities to improve existing treatment systems – with new process knowledge and/or insight – such that they operate more efficiently with little to no financial investment and thereby benefit the adjacent environment and those that pay for operation and upgrades of the system. One area with high potential for optimization is systems involving activated sludge and biological nutrient removal (BNR). Wastewater Resource Recovery Facilities (WRRFs) that use activated sludge use naturally occurring microorganisms to remove contaminants, including nutrients, from wastewater (Tchobanoglous et al., 2014). Microorganisms can perform a variety of tasks much more efficiently than humans can achieve synthetically, of which nutrient removal from wastewater is one. Engaging microorganisms to do the work of synthetic chemicals or physical barriers, such as filters, can save significant amounts of chemical inputs and energy. However, employing organisms to do work can be difficult as they have their "own agendas" within the context of their environments, and thus desired outcomes can be challenging to manage.

Targeted outcomes of activated sludge wastewater treatment processes require creating the right environments that selectively promote the appropriate organisms. One outcome of activated sludge treatment is biological nutrient removal (BNR), which commonly includes carbon, nitrogen, and phosphorus. Within a BNR configuration, biological phosphorus removal (BPR) makes use of phosphorus accumulating organisms (PAOs) that internally store phosphorus (P) obtained externally

from the environment. After storing P, the PAOs readily settle out of the wastewater in the process removing phosphorus from the waste stream. The exact biological mechanisms occurring during BPR are somewhat controversial and not entirely understood. The most well accepted characteristic of BPR is the necessity of creating an anaerobic zone in which readily available electron acceptors, commonly in the form of nitrate, nitrite, or dissolved oxygen, are unavailable, followed by an aerobic or anoxic zone. An aerobic environment is one in which oxygen is readily available and an anoxic environment is defined as one where oxygen is not present but nitrate and/or nitrite are accessible. Within the anaerobic zone, the BPR specific metabolism is initiated within the PAOs causing them to release P and synthesize PHA. As the PAOs are cycled into the aerobic or anoxic zones they then use the PHA for energy and take up more P than they released anaerobically, thus leading to a net decrease in P concentration within the waste stream (Tchobanoglous et al., 2014). Controlling the amount of oxygen provided within the aerobic environment is relatively straightforward as operators control the only source of dissolved oxygen via the aeration system, but nitrate can be more challenging as it is often produced within the treatment system. The detrimental impacts of nitrate to the BPR process have been noted for decades (Barnard, 1975; Siebritz et al., 1983). Nitrate can be present within the raw wastewater, but the most common source of nitrate is from the internal process of nitrification, converting ammonia to nitrate, followed by recirculation of nitrate rich sludge.

The presence of nitrate in RAS cannot be easily avoided in a conventional BNR system. First and foremost, ammonia has been found to be acutely toxic to aquatic life and has been regulated in wastewater discharge for decades – thus requiring removal in wastewater treatment (Schroeder and Tchobanoglous, 1985). BNR via nitrification of ammonia is the most popular form of ammonia removal and is typically employed in all facilities performing BPR. The regulated removal of the nitrate from wastewater is much less common and therefore nitrate is present within recirculated sludge at varying concentrations. Secondary clarifiers follow the biological treatment system and serve to separate solids from the waste stream. The process of recirculating activated sludge, primarily composed of bacteria, that has been settled in the secondary clarifier back to the biological treatment system to be mixed with influent wastewater is known as Recycled or Return Activated Sludge (RAS). RAS is pulled from the bottom of the secondary clarifier where the solids have settled and condensed to a high concentration (Figure 1). The rate at which RAS is returned to the beginning of the treatment system is one of the few operational parameters within a BNR system over which an operator has direct control. Tank sizes are literally cast in concrete, piping layouts are not easily manipulated; influent flow rates and influent wastewater quality are controlled by the weather through variation in precipitation and temperature, and those discharging waste into the sanitary sewer system. RAS is the very basis of activated sludge treatment and is one of the few aspects an operator can

control, yet there is relatively little guidance on optimization and impacts of RAS rate manipulation. Indeed, it appears RAS rates employed by operators are typically based on trial and error or historic values that "work" with little to no standard operation procedure (SOP) or guidance for altering.

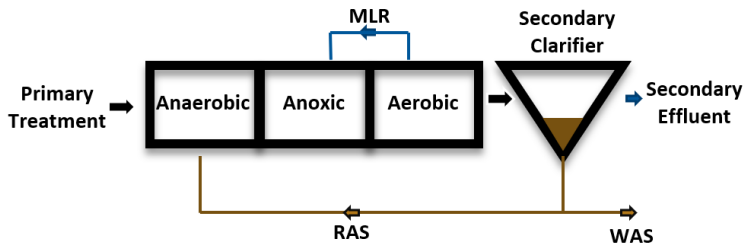


Figure 1. Common layout of a BPR WRRF.

Beyond controlling the rate at which flow and activated sludge is returned to the head of the treatment system, the RAS rate also directly controls how long solids are held within the secondary clarifier. The RAS rate controls the rate at which solids are removed from the clarifier. The environmental conditions at the bottom of a secondary clarifier typically contain low concentrations of dissolved oxygen which creates an environment conducive to denitrification if nitrate is available (Tchobanoglous et al., 2014). Yet little to no research has sought to quantify RAS rate impacts to secondary clarifier sludge denitrification and subsequent effects of RAS nitrate load on BPR. Operators are the only ones who may know why their specific plants maintain a particular RAS rate. Understanding to what degree RAS rate can affect secondary clarifier denitrification could serve to enable operators to make informed decisions related to manipulation of RAS rates.

Research presented and discussed in this thesis specifically interrogates RAS rate impacts on secondary clarifier denitrification and BPR performance. This thesis was driven by the following Research Questions.

**Research Question 1: Do WRRF operators practicing BPR consider RAS rate an important process control parameter?**

> **Hypothesis 1:** RAS rates are an underutilized operational parameter by operators of BPR facilities.
>
> **Objective 1:** Analyze responses to determine the success of each WRRF relative to BPR and what they consider to be important operational parameters, including approach to managing RAS rate.
>
> **Task 1:** Develop and deploy a survey to WRRFs practicing BPR requesting information about the capacity and efficiency of their BPR system and how they manage their RAS rate as well as other operational parameters.

**Research Question 2: How does RAS rate impact secondary clarifier sludge denitrification?**

**Hypothesis 2:** RAS rate affects secondary clarifier sludge denitrification through varying solids retention time within the clarifier.

**Objective 2.1:** Research will establish relationships between RAS rate, sludge residence time in secondary clarifier and sludge denitrification.

**Objective 2.2:** Demonstrate findings through manipulation of RAS rates at Moscow WRRF to better enhance secondary sludge denitrification.

**Task 2:** RAS rate control of denitrification will be assessed through analysis of continuous RAS nitrate, RAS flow, influent flow and solids data collected via an autosampler. Demonstrate that nitrate concentration within the RAS varies due to RAS rate fluctuation using summer 2020 pilot plant data.

**Research Question 3: How does RAS rate and nitrate mass load to the anaerobic basin impact BPR stability?**

**Hypothesis 3:** BPR performance can be stabilized through RAS rate control by limiting the mass of nitrate recycled to the anaerobic zone.

**Objective 3:** Establish relationships between RAS rate, mass of RAS nitrate, and BPR performance/stability (P:C ratio, effluent P).

**Task 3.1:** Perform plant profiles twice a day, during the highest and lowest nitrate load to the anaerobic basin. Request modifications to the RAS rate and monitor impacts.

**Task 3.2**: Create a BioWin model of Moscow's WRRF to compare with actual data collected at the WRRF. Compile established model parameters for denitrification to compare with values generated at Moscow's WRRF.

# 2 Literature Review

## 2.1 Biological Nutrient Removal

There are four primary elements to activated sludge wastewater treatment. They include primary treatment, secondary treatment, tertiary treatment, and disinfection and discharge. Primary treatment largely consists of screening, sedimentation, and primary clarification. During this stage large solids, inorganics and settleable organics are removed from the waste stream. After primary treatment, the wastewater then flows on to secondary treatment. Secondary treatment is the biological stage within an activated sludge system. Secondary treatment is key to removal of nutrients down to levels necessary for permit limits. In order to remove and recycle the microbes essential to BNR, a secondary clarifier or membrane system is installed after the secondary treatment basins. The backbone of secondary treatment is capturing and recycling the biomass such that they can continuously treat incoming wastewater. After secondary treatment, some systems use tertiary treatment to further reduce levels of nutrients or other wastewater constituents, typically through chemical addition and filtration. Finally, the wastewater is then disinfected and discharged. Secondary treatment and clarification are the subjects of this research.

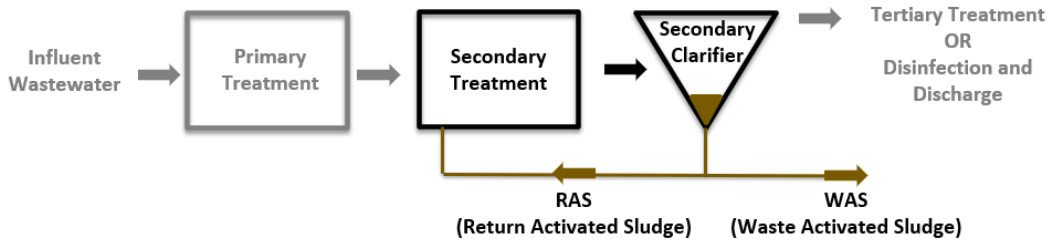


Figure 2. Basic diagram of the primary elements of wastewater treatment.

Secondary treatment can further be broken down by different zones engineered according to the nutrient removal desired. The zones are largely broken down according to the electron acceptor available. The mechanism through which the microbes remove nutrients from wastewater is primarily oxidation/reduction also known as redox. Redox requires an electron acceptor and an electron donor. The primary electron acceptors within wastewater treatment are oxygen, nitrate and nitrite. Other electron acceptors do exist, but the energy yield is substantially smaller and their relevance to this research is minimal. The electron donors are largely carbon-based molecules, often referred to as COD or BOD, as well as ammonia. The zones are referred to aerobic/oxic, anoxic and anaerobic. Within the aerobic/oxic zone dissolved oxygen is maintained through a blower system diffusing air or pure oxygen into the basin and is the primary electron acceptor utilized by microbes. While nitrate and nitrite are also present within the aeration/oxic zone if nitrification is occurring, the microbes prefer to use oxygen as it yields the highest amount of energy of the electron acceptors

present. The aerobic zone is responsible for removal of dissolved carbon and ammonia from the waste stream through endogenous respiration and nitrification, respectively. Within the anoxic basin, little to no dissolved oxygen is present, but nitrate and nitrite are available. The absence of oxygen forces the microbes to utilize nitrate and nitrite as the electron acceptors leading to denitrification. An ideal anaerobic basin has no electron acceptor and therefore minimal redox occurs. The purpose of the anaerobic basin is to initiate BPR specific metabolisms within PAOs (Tchobanoglous et al., 2014).

Ammonia is acutely toxic to aquatic organisms, and effluent limitations are commonly applied to WRRFs (Tchobanoglous et al., 2014). One common method to remove ammonia is through nitrification, a process by which ammonia oxidizing bacteria (AOB), a group of chemo lithotrophic organisms, convert ammonia to nitrite followed by nitrate in the presence of oxygen (Figure 3). While ammonia concentrations and loads are highly regulated across the U.S., few facilities have any discharge limits related to nitrate or nitrite. Impacts of nitrate on waterways is debatable but is largely considered to be less than that of other nutrients like phosphorus (Carpenter et al., 2001; Schindler, 2012; Schindler et al., 2008). However, to remove phosphorus biologically it has been found to be beneficial to limit the amount of nitrate recycled to the anaerobic basins (Barnard, 1975; Kuba et al., 1993; Siebritz et al., 1983).

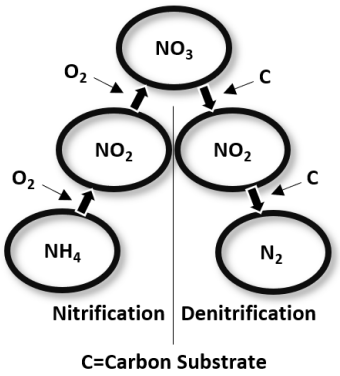


Figure 3. Diagram of nitrification and denitrification.

The most ubiquitous approach to denitrification for WRRF's practicing BPR is the use of pre-anoxic basins, wherein anoxic basins are installed between the anaerobic and aerobic basins (

Figure 4). The use of pre-anoxic basins is typically done as a replication or variant of the Modified Ludzak Ettinger (MLE) process. The key to the MLE process is a high rate of recycle from the nitrate rich wastewater at the end of the aerobic basin to an anoxic basin (Figure 3). This process has many benefits to the overall treatment system but is limited in the amount of nitrate that can be removed, typically around 80%

(Tchobanoglous et al., 2014), ultimately leaving 20% or more of the nitrate to be recycled back to the anaerobic basin within the RAS.

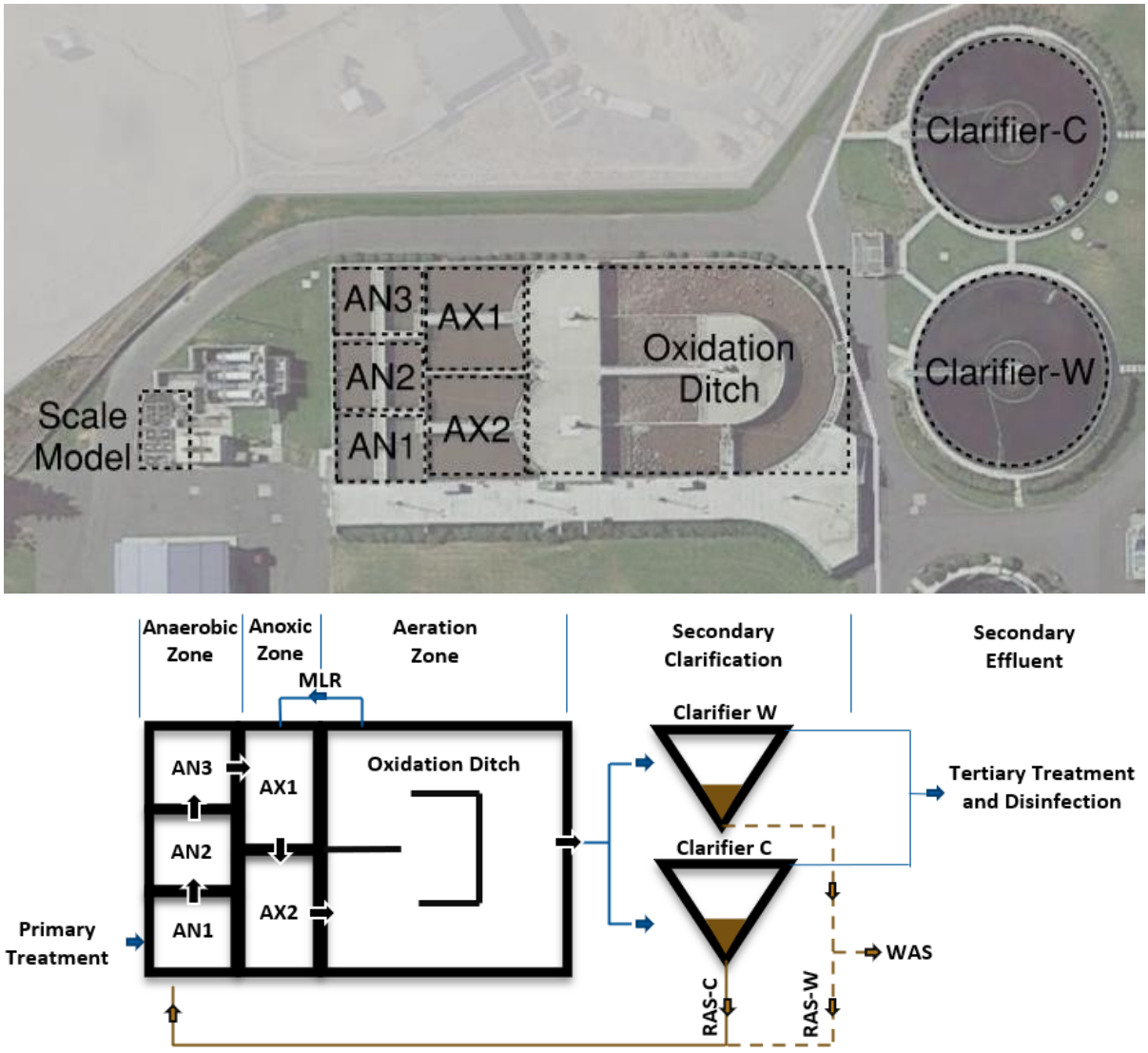


Figure 4. Diagram and aerial of the Moscow BPR WRRF. Background aerial image from Google Earth 6-2021.

## 2.2 Nitrate Impacts to BPR

Most, if not all, facilities practicing BPR also remove ammonia and therefore must deal with nitrate. Nitrate has long been known to be detrimental to BPR (Barnard, 1975; Kuba et al., 1993; Siebritz et al., 1983). One of the most universally accepted and appreciated aspects of BPR is the necessity of the anaerobic-aerobic cycle. It is crucial that wastewater be cycled through an anaerobic environment which does not contain any electron acceptors (e. g., oxygen, nitrate, nitrite), followed by an aerobic or anoxic environment. In doing so, PAOs are given a distinct advantage over other organisms. Such that accumulation of phosphorus within the PAOs can be consistently accomplished (Figure 5). Within the anaerobic zone, PAOs are the only organisms with the ability to utilize volatile fatty acids

(VFAs). This energy rich carbon source is taken up by PAOs so they can transform it into an internal carbon reserve, PHA. Once the carbon is stored internally, PAOs are no longer dependent on external carbon sources as they move into either aerobic or anoxic environments. This is considerable advantage over other organisms considering carbon is commonly scarce within the waste stream in the aerobic and anoxic zone given that the purpose of wastewater treatment is to remove this nutrient to minimal concentrations. Once in the presence of an electron acceptor, PAOs can utilize their PHA for cell growth and maintenance. In addition to an anaerobic zone, the PAOs need a consistent source of volatile fatty acids (VFAs) for uptake and conversion to PHA, as the PHA is then utilized aerobically and anoxically for uptake of P. A strong correlation between the quantity of PHA synthesized anaerobically and P removed aerobically has been found (Coats et al., 2021). Without the VFAs fed anaerobically, PAOs struggle to generate PHA and overall P uptake is degraded. Degradation of the anaerobic basin (e.g., excess nitrate in the RAS) removes the PAOs advantage and encourages growth of other organisms that do not remove excess P from the wastewater. The state of the anaerobic basin is conditional on the inputs which are typically RAS, raw wastewater, liquor from a solids fermenter, or a combination thereof. The challenge in maintaining the anaerobic conditions at the head of the plant are largely due to variations in influent properties and the RAS.

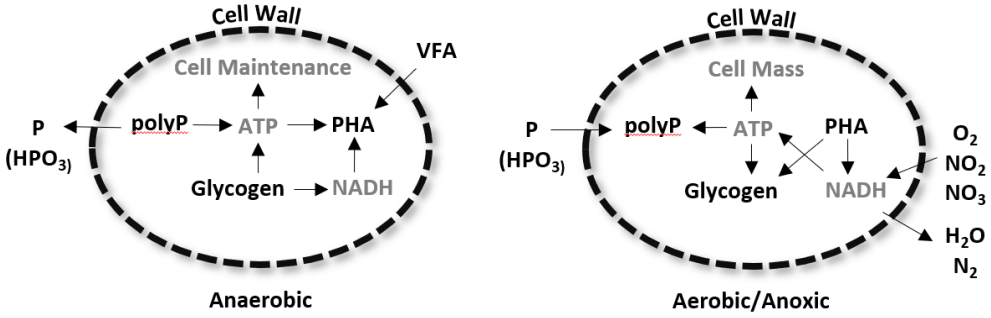


Figure 5. Illustration of PAO metabolism under anaerobic and aerobic/anoxic conditions (by author).

Nitrate in the anaerobic basin has two negative impacts on BPR. The main issue arises from the use of nitrate as an electron acceptor leading PAOs to metabolize VFAs for growth rather than conversion to PHA for aerobic uptake of P. The other is potential competition by denitrifying Ordinary Heterotrophic Organisms (OHO's) that leads to further metabolism of VFA's. In the absence of an electron acceptor, some OHO's may ferment readily degradable organic matter into VFA's (Brown and Koch, 2005). Since typical OHO's cannot utilize the VFA's, this synergistic relationship would afford PAOs additional substrate to generate PHA (Grady et al., 2011). The impacts of increasing RAS nitrate load have generally been associated with an increase in effluent P concentrations (Grady et al., 2011). This relationship is not so clearly seen in full-scale facilities or modeling programs, but recent trends in BPR process optimizations have focused on increasing the anaerobic SRT (i.e., the

anaerobic mass fraction) (Coats et al., 2018; Onnis-Hayden et al., 2020; Wang et al., 2019) which can help to mitigate the impacts of RAS nitrate by allowing additional time for fermentation of substrate in the influent wastewater.

While it is a common understanding that nitrate loads entering the anaerobic zone are detrimental to BPR, specifics remain largely unquantified. Measuring the health of the BPR system via the anaerobic zone is achieved through various stochiometric measurements that demand extensive and regular sampling – interrogations typically beyond that executed in the operation of a full-scale BNR system. One common method is comparing the amount of P released anaerobically to the concentrations of VFAs in the influent wastewater; typically acetic acid is the only VFA measured and considered (Coats et al., 2017). The ratio of P release to influent VFAs consumed, referred to as the P:C ratio, gives insight to the activity of PAOs within the system. If the ratio is high, then it is indicative of PAOs utilizing VFAs and releasing P, which is the main goal of the anaerobic basins. Researchers have postulated that the P:C ratio can be correlated with the amount of PAOs relative to a competitive anaerobic bacteria, Glycogen Accumulating Organisms (GAOs), present in the system (Coats et al., 2017; Onnis-Hayden et al., 2020) and that it is dependent on pH (Filipe et al., 2001; Smolders et al., 1994). However, these perceptions were developed either in lab experiments void of nitrate or without considering the impacts of nitrate on the anaerobic zone of full-scale WRRFs.

The anaerobic basin is essential to BPR and extended SRT and HRT have been correlated to better P removal (Siebritz et al., 1983). In recent years, several methods of storing activated solids longer anaerobically have been developed under the umbrella of Side-Stream Enhanced Biological Removal (S2EBPR) (Onnis-Hayden et al., 2020; Wang et al., 2019). S2EBPR processes utilize existing or new anaerobic basins to store the activated sludge for extended periods of time, yet none consider the option of increasing the secondary clarifier SRT which is ultimately a different means to the same end. Multiple S2EBPR layouts rely on internal carbon reserves or endogenous decay for carbon, which is no different within the secondary clarifier. Additional risks, as given in Section 2.4, are present within the secondary clarifier but to a manageable degree. Ultimately it is on a case-by-case basis whether the secondary clarifier could accomplish the same end as S2EBPR processes. The benefits of leveraging the secondary clarifier to enhance the BPR anaerobic zone integrity is that minimal effort is required for adjusting secondary clarifier SRT, no rerouting of piping or installation of additional tankage is necessary, and nitrate loads can be reduced in a manner that does not demand raw wastewater organic carbon.

### 2.3 Potential Value of Secondary Clarifier Denitrification

Secondary clarifiers create extended periods of anoxic and/or anaerobic environments during which denitrification can occur prior to recirculation to the anaerobic basin. Siegrist and Gujer (1994) found 15-30% of the total plant-wide denitrification occurred within the secondary clarifier at two plants in Germany, one with a substantial sludge blanket and the other holding minimal sludge in the secondary clarifier. They also noted the potential for denitrification in the secondary clarifier to increase within WRRFs with diluted wastewater or small anoxic volumes. Mikola et al. (2009) found that 40% of the total plant nitrogen load was denitrified within the secondary clarifier at the Pihlajaniemi WRRF in Finland. Koch et al. (1999) found 19% of total inlet nitrogen was denitrified within the secondary clarifier or 37% of the total denitrification capacity at the Zurich-Werdholzli WRRF in Switzerland. Guidance published by the Water Environment Federation suggests lowering RAS rates to generate a sludge blanket within the secondary clarifier that will in turn reduce nitrate in the RAS (Brown and Koch, 2005). Ultimately the potential for denitrification within the secondary clarifier could allow WRRFs practicing BPR to enhance their system by reducing RAS nitrate load with little effort and no capital expenditures.

### 2.4 Secondary Clarifier Denitrification Challenges

Opposing the potential positive impacts of denitrification within the secondary clarifier, storing PAO-rich sludge anaerobically for too long can lead to release of stored P, commonly referred to as secondary P release. The primary concern with P release in the secondary clarifier is a potential increase in secondary effluent P. Indeed Mikola et al. (2009) found that secondary P release took place under anaerobic conditions or low nitrate concentrations, below 1 mg-N/L, within the secondary clarifier sludge blanket. However, nearly all phosphorus released was reabsorbed in parts of the sludge blanket where the nitrate concentrations were higher leading to no impacts on the secondary effluent P concentrations. These finding indicate that the presence of nitrate in the upper regions of the clarifier and sludge blanket can function as a protective layer against secondary P release. Mikola et al. (2009) further found a strong correlation between secondary clarifier SRT and nitrate concentration within the sludge blanket. Secondary P release did not impact effluent quality at the studied SRTs of up to 14 hours. Of course, SRT is only one piece of the equation, as clarifier geometry and design, wastewater temperature, settling characteristics, carbon source, and nitrate load on the clarifier are all important elements to consider (Henze et al., 1993). Similarly, Wouters-Wasiak and Ho (1996) found that in batch tests, P release did not occur until nitrate concentration was below 0.5 mg-N/L. They further found that any P released within the clarifier was not detrimental to further BPR processes and all P was taken up aerobically. Guidance from WEF also indicates that release of phosphorus within the lower region of the clarifier, likely within the sludge blanket, is of

little concern as little liquid from this region will pass to the upper region to be removed with the effluent (Brown and Koch, 2005). In theory, a small amount of P release could actually be considered an indicator that the RAS is fully anaerobic which is the best state for discharge to the anaerobic basins. During a thorough analysis at an anaerobic/aerobic WRRF in Texas, it was found that TSS, P, and $NH_3$ concentrations in the effluent sporadically increased when the sludge blanket was over 4 feet deep. However, with the sludge blanket maintained around 3 feet, consistently low effluent concentrations were maintained and the RAS nitrate concentration was near 0 mg/L (Robinson et al., 2019).

A primary issue with denitrification in the secondary clarifier can be floating sludge caused by nitrogen gas bubbles carrying solids to the top of the clarifier. The sludge could potentially be released with the effluent leading to TSS and P permit discharge limitation exceedance. Henze et al. (1993) noticed a sporadic increase in suspended solids at higher secondary clarifier nitrate loads within pilot scale clarifiers. A proposed limit of 6-8 mg-N/L at 20°C was developed for maximum nitrate load to avoid saturation of nitrogen and formation of bubbles. The formation in small clarifiers, bench or pilot scale, is even more common due to the reduced depth of water decreasing the pressure as well as the rapidity with which the solids can settle into a blanket. The Henze et al. (1993) experiments relied on scale model clarifiers with depths of 3.6 and 4.1 m while testing sludge blanket heights of 1.5 – 3 m depths. A sludge blanket height of 37 – 80% of the overall depth within a full-scale clarifier is unheard of and casts some doubt on the absolute applicability of these findings to full-scale WRRFs. Siegrist and Gujer (1994) note that issues of rising sludge are rare within full-scale facilities and can be amended by simply adjusting the RAS rate. Indeed adjustment of RAS rate to amend these conditions is also recommended by WEF (Brown and Koch, 2005). Koch et al. (1999) also confirmed that denitrification within the sludge blanket did not cause rising sludge due to nitrogen gas bubbles even with 19% of the total inlet nitrogen denitrified within a full-scale secondary clarifier. While increased effluent TSS and P are potential risks, neither appear to be prevalent within full-scale facilities and the issues can potentially be easily remedied by increasing the RAS rate.

### 2.5 RAS Impacts on Secondary Clarifier Denitrification

Denitrification is a function of sludge mass, contact time (SRT), and availability of carbon source. Several models have been developed to predict the level of nitrate removal within the secondary clarifier applying these parameters. RAS and, to a much smaller extent, WAS are the primary means of moving sludge through the secondary clarifier. In this way, the RAS rate has a strong influence on the secondary clarifier SRT as well as the depth of sludge blanket. Siegrist et al. (1995) proposed a model for denitrification in the secondary clarifier based on RAS ratio and clarifier scraper interval

within a rectangular clarifier utilizing suction lift RAS pumps, indicating that denitrification could be improved by controlling these two parameters. Mikola et al. (2009) found a strong correlation, $R^2$=0.97, between sludge retention time and nitrate concentration within the secondary clarifier within a full-scale facility. The sludge retention times typically varied from 2.4 to 4.0 hours during the summer and winter, respectively, but was observed to be as high as 14 hours. This makes sense as the nitrate concentration likely varies according to the specific denitrification rate (SDNR). SDNR is calculated as the change in nitrification over time with respect to mass of biosolids, g-N/day*g-VSS. In theory an established SDNR could be multiplied by the secondary clarifier SRT (SC-SRT) to determine the amount of denitrification occurring in the clarifier. SC-SRT is calculated as total mass of solids in clarifier divided by the RAS solids removal rate (Schuyler, 2010). A major challenge in utilizing SDNR is estimating the SRT which requires approximating the amount of solids within the clarifier. Solids concentrations within the majority of the secondary treatment system are largely constant throughout the day, but changes in influent flow lead to varying solids loading and removal rates from the secondary clarifier.

WEF discusses two WRRF's on the extreme ends of RAS control (Brown and Koch, 2005). One plant was able to reduce the 3 mg-NO3-N/L within their effluent to 0 mg-N/L in the RAS simply by reducing their RAS rate. In turn this plant was able to consistently obtain effluent P concentrations below 0.1 mg/L. Another plant was forced to run at a high RAS ratios, 150% of influent flow, due to mechanical limitations of their system leading to significant amounts of nitrate recycled to the anaerobic zone ultimately leading to poor overall P removal.

### 2.6 Guidance on RAS Control

Little guidance is available to WRRF operators quantifying the impacts of RAS rate adjustment on BNR. One common approach to RAS rate control is to base it on the influent wastewater, here the RAS flow divided by the influent flow is referred to as the RAS ratio. It makes sense that feeding solids in proportion within the influent flow would allow for a steady solids concentration throughout the system assuming the solids concentration within the RAS is constant. EPA developed guidance in the past, but it is limited to primarily maintain consistent MLSS and speaks little to aspects of BNR (West, 1973). Grady et al. (2011) cautions running the RAS rate too high due to the potential negative impacts of nitrate on the anaerobic system. It is also suggested to not run it too low, but without any reasoning. Grady et al. (2011) also present results indicating that as RAS rate is increased, so too is nitrate mass load which directly increases the amount of P within the effluent. Indeed WEF recommends decreasing the RAS rate to enhance denitrification within the secondary clarifier particularly during warmer weather as nitrifying organisms tend to be sensitive to cold

temperatures (Brown and Koch, 2005). Research by Schuyler (2010) presents results from a wide range of facilities that benefitted from reduced sludge flow rates. Schuyler's work is not focused on BPR facilities and the discussion is primarily focused on developing a famine stage during a single pass of sludge through the treatment system. The famine stage is thought to help develop sludge with high settleability thereby increasing secondary clarifier performance. Settleability of sludge is rarely a concern for BPR facilities due to the presence of the anaerobic basins and its tendency to select for well-settling PAOs.

# 3 Methods

## 3.1 BPR Operator Survey

A survey was developed and sent out to over 40 WRRFS in order to gauge the health of the BPR system as well as gain insight into operational approaches such as RAS rate; the survey instrument is provided in Appendix A. The survey did not request any information about individuals that would necessitate a review by Human Resource Protections (IRB). All WRRFs were contacted by email and provided a brief introduction to the purpose of the survey; the contextualization was purposely written to avoid influencing or skewing responses. Of the 40 WRRFs invited to participate in the survey, 20 responded. Respondents were typically the lead operators of the WRRF. Responses were analyzed both quantitatively and qualitatively.

Quantitative survey responses were statistically assessed against two survey questions considered the response variable: i) the reported frequency of failures or process upsets each year, and ii) the average effluent TP concentration from the secondary treatment system (2TP) prior to chemical polishing or filtration. This quantitative approach was taken because these two responses represent the consistency and capacity of BPR within the WRRF. There was some variability in reported frequency of failure - most reported as frequency of failures per year while a few others responded in a variety of ways such as weeks per year, days per month or struggling constantly in a given season; all responses were converted to frequency of failures per year to make them statistically comparable. As an example, if a response indicated monthly failures, then it was considered to have 12 failures per year. Analysis of variance (ANOVA) was employed to evaluate the potential significance of a response to the various responses utilized as explanatory variables. ANOVA allows for a statistical examination of relationships within data that may be hard to otherwise quantify or interpret with the naked eye. Applying ANOVA, relationships between several parameters and the responses of frequency of failure and 2TP were analyzed to gain insight into potential influences on the response variables; additionally, the potential relationship between the two response variables was statistically evaluated. P-values below 0.05 were considered to indicate that a specific parameter has a statistically significant relationship with the response while values between 0.05 and 0.10 have a potentially statistically significant relationship. The statistical software program R was used to perform the ANOVA (R Development Core Team, 2017).

Qualitative responses that provided BPR operational insights were more difficult to compare, yet potentially provide the most value to the BPR field. These responses were analyzed for overlapping trends between facility responses and organized under the topic of Operational Approaches. Operational insights were the primary purpose for conducting this survey, but the complexity of BPR

system management and decades of experiences is not easily summarized in a few sentences. This highlights the importance of regular conversations between operators, engineers, and academics.

### 3.2 Moscow WRRF Sampling

The Moscow WRRF services a community of approximately 25,000 people with an average influent flow rate of approximately 2.0 mgd (predominantly domestic wastewater). The system operates as a typical A2O process, anaerobic-anoxic-oxic, and consists of three anaerobic basins (each 208,000 gallons), two pre-anoxic basins (each 360,000 gallons), an aerobic oxidation ditch (1,800,000 gallons), and two secondary clarifiers (each 950,000 gallons), with influent wastewater pre-treated through a 6 mm perforated plate screen followed by a vortex grit basin (Figure 4). The RAS flow rate is typically set at 36% of influent flow, while the aerobic basin MLR flow rate to the first pre-anoxic basin is controlled using a nitrate probe in the second pre-anoxic basin (target anoxic effluent NO3-N of 0.5 mg/L). The Moscow WRRF operates one of their clarifiers by continuously discharging RAS (Clarifier-C discharging RAS-C), while the other clarifier stops discharging for one hour then releases WAS for 3 hours before resuming RAS flow (Clarifier-W discharging WAS and RAS-W). The RAS rates are the same for both RAS lines when in operations. WAS flow rate being a steady 250 gpm with minimal variation in flow time, +/- 10 minutes, to manage MLSS concentrations.

A scale model of the Moscow WRRF was also operated during summer 2020 to support various topics within the Coats research group (Figure 6). The system was operated from 8/3/2020 to 10/10/2020 at two different RAS rates with no other process or input changes. Influent flow was set at a constant 2 gpm with RAS rate running at 0.65 gpm from 8/3/2020 to 8/19/2020 at which point the RAS was increased to 2 gpm for the remainder of the season. The scale model sits adjacent to the full-scale Moscow WRRF and processes screened and de-gritted wastewater from the Moscow WRRF. The system includes an activated primary fermentation system (Krause, 2010) with a 240 gallon CSTR fermenter, a primary clarifier (approximate volume of 1000 L), and a positive displacement pump driven by a variable frequency drive (VFD) to return settled sludge to the fermenter. The secondary biological treatment system was operated to achieve post-anoxic BPR (Coats et al., 2011; Winkler et al., 2011), and includes anaerobic (three CSTRs in series at 200 gallons each), aerobic (two CSTRs in series at 350 gallons each), and anoxic (1800 gallon serpentine ditch) environments, with a secondary clarifier (approximate volume of 140 gallons) providing return activated sludge (RAS) to the first anaerobic basin. Influent wastewater and RAS are pumped using positive displacement pumps driven by VFDs. Aeration is achieved using a VFD-driven rotary lobe blower and fine bubble diffusers. The scale model is operated under ammonia-based aeration control (ABAC) using a Hach ANISE probe (Hach, Loveland, CO, USA) and Hach SC-200 controller. The

ANISE probe is installed where aerobic basin wastewater enters the anoxic ditch; the controller operates an electronically actuated valve that provides air to fine bubble diffusers in the anoxic/aerobic ditch immediately downstream of the probe to maintain a maximum $NH_4$–N concentration of 2 mg-N/L. Oxidation ditch DO is controlled using a Hach LDO probe and SC-200 controller; the LDO probe is installed in the 2nd aerobic CSTR, and the controller seeks to maintain a DO concentration of 2 mg-$O_2$/L by varying the blower speed.

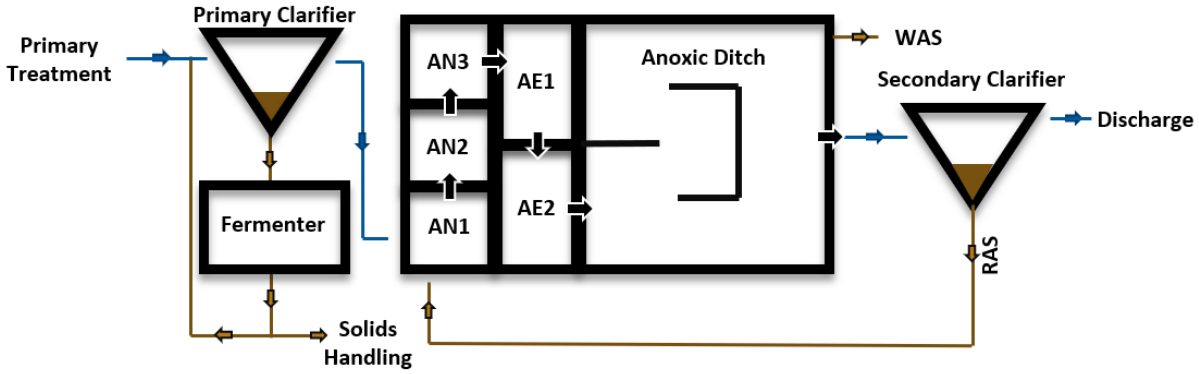


Figure 6. Diagram of the Moscow scale model WRRF.

Sampling at the full-scale Moscow WRRF was performed via grab samples as well as continuous monitoring. Grab samples were once daily, around 10:00, on Monday, Wednesday, and Friday from 5/18/2020 to 9/10/2020 (summer) at both the scale model and full-scale WRRF, then 2 times daily, 8:00 and 2:00 pm, on Monday, Tuesday, and Wednesday from 1/15/2021 to 2/22/2021 (winter) at the full-scale WRRF only. The focal point of the study occurred in winter when multiple grab samples were collected each day and two Hach ANISE probes were installed to monitor nitrate in the oxidation ditch and the RAS lines of the full-scale facility. The time of day for the winter sample collection was chosen after reviewing preliminary ANISE nitrate data to determine the maximum and minimum RAS nitrate load with the morning being at the end of peak nitrate load and the afternoon after RAS nitrate loading had subsided for a few hours, as well as to fit within hours that the WRRF was accessible. Sampling at 11:00 am was also conducted from 1/15/2021 to 1/18/2021 but was not continued due to time constraints. During summer sampling, grab samples were from 10 points along the treatment system; 3 anaerobic basins, 2 anoxic basins, inlet and outlet of the oxidation ditch, RAS line, influent after screening and near the secondary clarifier weir outfall. During the winter, the anoxic basins were not collected from and only one point in the oxidation ditch, reducing the total sampling points to 7. Grab samples for nutrients and VFAs were immediately filtered through a 0.22 micrometer filter before transportation to the lab for analysis.

Continuous monitoring of nitrate, ammonium and temperature was conducted via HACH ANISE meters connected to HACH SC-200 controllers with data stored every 5 minutes. The meters were installed at the outlet of the oxidation ditch and within the RAS line sample station during Fall 2020 and Winter 2021. The RAS feeding the RAS sample station was changed weekly by opening/closing sample port valves on the RAS main line. Calibration of probes was performed as needed after lab analysis of nitrate. Data that did not appear to align well with the lab data was discarded if no reasonable explanation for the discrepancy could be found. Nitrate concentrations from the ANISE within the RAS line appeared noisy at initial viewing. The nitrate data was smoothed out by taking hourly averages using the 30 minutes before and after the specific time of interest. Averages of each hour of the day were then taken for graphical representation found in the *Results and Discussion*. Originally the RAS sample station was equipped with field DO and ORP sensors as well, but they were found to quickly clog leading to inaccurate readings. The operators at the Moscow WRRF provided hourly SCADA data for influent, plant, effluent, and RAS flow rates as well as oxidation ditch MLSS and DO, and nitrate within the second anoxic basin. Hourly SCADA data is often presented as a 24-hour period with averages at each hour. The n values shown in Chapter 4 shows the minimum number of points within the averaged data sets as each hour of the day did not always have an equal number of data points due to missing or erroneous readings. A solids analysis of the RAS lines was conducted via an ISCO 3700 autosampler drawing individual hourly samples from the RAS sample station. The autosampler collected individual 350 mL samples of RAS on the hour, which were then shaken and poured into 50 mL bottles for solids analysis as outlined in *Analytical Techniques*.

### 3.3 Batch Operating Conditions

Two different series of batch tests intended as preliminary scans were conducted with one focused on clarifier conditions and the other anaerobic basins. Clarifier batch tests were designed to mimic clarifier conditions with a MLSS concentrations around 5,000 mg-TSS/L. Concentrations were chosen based on the lowest concentration found within the RAS lines at Moscow WRRF. The test was run at two different nitrate concentrations, roughly 5 and 20 mg/L via addition of potassium nitrate. 5 mg/L is typically what was seen within the oxidation ditch and 20 mg/L was meant to capture the mechanisms of denitrification as well as impacts of high nitrate load on the secondary clarifier. Oxidation ditch MLSS was collected and immediately transferred to the lab where it was decanted to reach the desired MLSS concentration at which point the reactors were continuously stirred at the minimum setting to avoid surface oxygen transfer. Hach AN-ISE probes were installed in the reactors and grab samples were collected and analyzed periodically. The reactors were allowed to run until all nitrate had been consumed and significant P release was noticed.

Anaerobic basin batch tests were meant to capture impacts of RAS nitrate load. Preliminary tests consisted of raw wastewater mixed with RAS collected from the Moscow WRRF. One test was spiked with additional nitrate via potassium nitrate to increase the nitrate concentration to around 10 mg-N/L. The reactors were then allowed to run for around 24 hours as was the upper anaerobic HRT seen at the Moscow WRRF. After which time aerators were turned on to determine impacts to P uptake as well. Hach AN-ISE probes were installed in the reactors and grab samples were collected and analyzed periodically.

### 3.4 Analytical Techniques

Samples were collected to monitor TP, $PO_4$, $NO_3$, $NH_3$, TS, TSS, VS, VSS, ffCOD, tCOD, VFAs, PHA and glycogen. For soluble constituents, samples were filtered through a 0.22 μm syringe filter (Millipore Corp., Billerica, MA, USA) while on site and directly after collection. Total P (TP) was determined in accordance with Hach (Loveland, CO, USA) method 8190 while $PO_4$ was determined in accordance with Hach (Loveland, CO, USA) method 8048 - both methods equivalent to Standard Methods 4500-PE (APHA. et al., 2012).

Soluble $NO_3$ was determined in accordance with Hach method 10020 and soluble $NH_3$ testing followed Hach method 10031 - both consistent with Standard Methods (APHA. et al., 2012). A Spectronic® 20 Genesys™ spectrophotometer (Thermo-Fisher Scientific Corp, Waltham, MA, USA) was utilized to measure the absorbance of the reacted sample at a wavelength of 410 nm for $NO_3$ and 655 nm for $NH_3$. $NO_3$ and $NH_3$ concentrations were determined utilizing a standard curve ($R^2$>0.99). TS, VS, TSS and VSS were measured in accordance with Standard Methods 2540 D and 2540 E (APHA. et al., 2012), respectively. Filtered flocculated COD (ffCOD) was determined in accordance with the technique outlined by Mamais et al. (1993).

VFAs (acetic, propionic, butyric, isobutyric, valeric, isovaleric, and caproic acids) and methanol were quantified using a Hewlett-Packard 6890 series gas chromatograph (GC) (Agilent Technologies, Inc., Santa Clara, CA, USA) equipped with a flame-ionization detector (FID) and a Hewlett-Packard 7679 series injector. The system was interfaced with the Hewlett-Packard GC ChemStation software version A.06.01. VFA separation was achieved using a capillary column (Heliflex® AT™-AquaWax-DA, 30 m x 0.25 mm ID, W. R. Grace & Co., Deerfield, IL, USA) which was ramped from an initial 50°C to 200°C in three steps (2 min at 50°C, ramp to 95°C at 30°C min-1 then to 150°C at 10°C min-1 and hold for 3 min; finally, ramp to 200°C at 25°C min-1 and hold for 12 min) with helium as the carrier gas (1.2 mL min-1). The split/splitless injector and detector were operated isothermally at 210 and 300°C, respectively. Prior to analysis, samples were acidified to a pH of 2 using nitric acid. 0.5 μL of each sample was injected in 20:1 split mode. VFA concentrations were determined through

retention time matching with known standards (Sigma-Aldrich Co., St. Louis, MO, USA; Thermo Fisher Scientific Inc., Waltham, MA, USA) and linear standard curves ($R^2$>0.99).

Biomass PHA content was determined by gas chromatography/mass spectrometry (GC-MS) as described in Braunegg, et al. (1978). Dried biomass samples were digested at 100°C in 2 mL of acidified methanol (3% v/v sulfuric acid) and chloroform. Benzoic acid was added as an internal standard to the chloroform at 0.25 mg/mL. After digestion, 2 mL of deionized water was added and vortexed to separate into chloroform and water phases. The chloroform phase was extracted and filtered through sodium sulfate anhydrous to remove excess moisture and particulates. GC-MS was performed on a ThermoScientific ISQ7000-Trace1300 GC-MS instrument. The sample was introduced using split injection. Separation was achieved on a ZB1 (15 m, 0.25 mm ID) capillary column (Phenomenex, Torrance, California, USA) with helium as the carrier gas (1.2 mL $min^{-1}$) and an initial temperature of 40°C (2 min) ramped to 200°C at 5°C $min^{-1}$. The compounds were confirmed by retention time and mass spectral matching with known PHA standards (PHB and PHB-co-HV: Sigma Aldrich; NaHB: Alfa Aeser) as methyl ester derivatives, and quantified based on the internal standard. The Xcalibur software program (Thermo Electron Corporation) was used to facilitate PHA quantification, and the optimal molecular weight for PHA quantification was determined to be 103 g $mol^{-1}$. PHB eluted at approximately 5.4-5.6 min, and PHV eluted at approximately 7.9-8.4 min. The benzoic acid standard eluted at 11.9-12.1 min. Total intracellular PHA content was determined on a percent dry weight basis (mass PHA per mass TSS, w/w) and a percent cell weight basis (mass PHA per mass VSS, w/w). Glycogen was determined with dried biomass samples as described by Parrou and Francois (1997). Total intracellular glycogen content was determined on a percent dry weight basis (mass glycogen per mass TSS, w/w) and a percent cell weight basis (mass glycogen per mass VSS, w/w).

### 3.5 Process Modeling

EnviroSim's BioWin V6.0 was utilized for process modeling of the Moscow WRRF. The model was developed with the same layout, basin volumes, and flow paths as the full-scale system. Oxidation ditches are not an option within the software, so six aeration basins in series were utilized to mimic the conditions created by an oxidation ditch. Recorded, bound RAS ratios for the 36% and 50% RAS setpoints were compared as well as steady, unbound RAS ratios of 10%, 25%, 36%, 50%, 70%, 100% and 200%. Each scenario was run for 90 days to reach steady state, after which a single day was run for comparison. The influent flow and concentrations of constituents were the same for all scenarios. Influent flow rates were maintained according to what was measured during the winter 36% analysis. Influent wastewater characteristics were held constant throughout the day to focus attention on the

impact of the varying flows and RAS nitrate load rather than differences in influent characteristics. COD ratios were adjusted to match the averages collected for VFA/rbCOD and rbCOD/tCOD. The wasting strategy during the bound RAS flows at the 36% and 50% matched what occurred at the Moscow WRRF. All unbound RAS ratios had a wasting rate of 9% of the RAS rate which is comparable to what was seen during the bound 50% RAS ratio setpoint, the difference being it was spread out evenly throughout the day within the model rather than occurring over a 3 hour period. The MLSS within the aerobic basins was maintained around 2500 between modeling scenarios, which maintained a near constant SRT and is consistent with several recommendations from the BPR Operator Survey. The SRT this created was around 20 days, which is somewhat high when compared with recommended values for BPR.

# 4 Results and Discussion

The following discussion presents results demonstrating the impacts of RAS ratio on a key aspect of BPR, RAS nitrate load. Data collected at the Moscow WRRF, a scale model, and a process model demonstrate that increasing the RAS ratio, in turn increases the RAS nitrate load through decreased secondary clarifier denitrification and increased flow containing nitrate. Additionally, impacts of the increased nitrate load are apparent within the anaerobic basins in terms of P release and PHA synthesis. The results indicate that RAS rate is an important process control parameter that can be utilized to increase BPR process stability and success.

## 4.1 BPR Operator Survey Results

The general characteristics of each facility that responded to the WRRF operator survey were reviewed before comparing their success with various parameters. Overall, the operators that participated in the survey (n=20) indicated that their WRRFs were generally successful in terms of BPR with all reporting 2TP below 1.1 mg/L; the lowest value was 0.15 mg/L and the average was 0.46±0.33 mg/L (n=18). According to WEF, nearly all respondents would be practicing complete BPR (Brown and Koch, 2005). The average influent TP for all WRRFs was 6.24±1.27 mg/L (n=19), which indicates an impressive 93% recovery biologically. The most common BPR process configuration for responding facilities was the A2O process, but there was a range of process configurations including A/O, Modified Bardenpho, UCT, VIP, JHB, and Westbank (Tchobanoglous et al., 2014). The BPR systems represented in the survey have been in operation from less than a year to over 30 years. The reported frequency of failure ranged from less than once a year to monthly with near constant challenges seasonally. Average influent flow ranged from 0.8 to 90 million gallons per day (MGD), with an average of 12.3 MGD. WRRFs were located across a wide geographic range and climatic regions, including Arizona, Colorado, Idaho, Washington, Montana, Oregon, Virginia, and British Columbia.

Comparison between WRRFs with different layouts and wastewater characteristics may seem a frivolous exercise comparable to comparing apples to oranges. However, taking this step back from the nuances of BPR allows a wider comparison to determine if general themes emerge. It is true that a WRRF may struggle with BPR due to specific loading challenges or relative abundance of specific microbes, but perhaps the solutions can be seen by looking at the system as a whole compared with others across the continent. If all successful facilities have a similar approach to RAS rate control, anaerobic SRT, the presence of a primary solids fermenter, or maintain a similar mixed liquor concentration, then this is noteworthy no matter the layout and specific characteristics. Whether

those facilities function as a modified Johannesburg, post-anoxic or A2O becomes irrelevant to a degree if they succeed utilizing the same strategies or require the same quantity of tankage.

### *4.1.1 Statistical Interpretation of BPR Survey Responses*

Influent and effluent wastewater characteristics were reviewed to see which had the strongest impacts on success of the facilities. Average influent wastewater concentrations of BOD/COD, Total P, Ammonia, and TKN were provided by respondents. It is feasible that those with ideal influent wastewater characteristics - high amounts of carbon, low TP, low TKN/$NH_3$ - could perform more consistent and/or more complete BPR regardless of operation. The ratio of BOD:P or COD:P is also viewed as an important metric for BPR applicability with suggested minimum values of 30 and 60, respectively, for effluent concentrations of less than 0.5 mg-P/L (Tchobanoglous et al., 2014). However, no influent characteristics or combinations thereof exhibited a strong relationship with either frequency of failure or 2TP. Ultimately these results indicate that success of a BPR-configured WRRF is not a primarily a function of influent concentrations.

Effluent characteristics specifically associated with certain permit criteria were found to be an indicator of BPR stability. WRRFs with total nitrogen (TN) permit limits (n=6) reported significantly fewer failures per year (p=0.024) than those without (n=14). The relationship nitrogen, more specifically nitrate, has on BPR performance has been well documented and a major subject of research for decades (Kuba et al., 1993; Siebritz et al., 1983). Presence of nitrate fundamentally changes the metabolic pathways of PAOs, causing them to uptake P rather than release P (Grady et al., 2011); excess nitrate can induce BPR failure associated with impaired anaerobic activity. WRRFs that are required to minimize effluent nitrate to meet TN permit limits are in turn managing a major cause of BPR failure. Conversely, a TN permit limit did not have a statistically significant impact on 2TP. The lack of TN permit limit impacts on 2TP may indicate that nitrogen is not always a consistent issue, and the variable impacts are most apparent as failures. It could be inferred that those without TN permits allow more nitrate into the secondary clarifier which is in turn transferred to the anaerobic basins via RAS. In addition to an effluent limitation, facilities with TN permits are required to maintain an intensified operational approach to manage nitrate.

Survey responses also allowed for a comparison of specific unit processes and theoretical impact toward BPR. Specifically, the impacts of specialized unit processes on BPR were analyzed by comparing the presence of a primary solids fermenter, anaerobic digester, influent equalization tank, or side stream P recovery system with the response variables. Surprisingly, none of these processes had a significant relationship with the frequency of failure, though the presence of an anaerobic digester and a fermenter landed in the potentially significant range (p=0.07 and p=0.09). Of

particular interest, WRRFs with primary solids fermenters (n=8) had a higher frequency of failure than those without (n=12). While the relationship was only within the potentially statistically significant range (p=0.09), the lack of a significant advantageous relationship between fermenters and either frequency of failure or 2TP was unexpected. This result aligns well with recent work by Coats et al. (2021) that revealed increased volatile fatty acid (VFA) loading does not directly lead to more complete or consistent net P removal. This finding demonstrates that simply having access to additional VFAs from a fermenter does not guarantee successful utilization. There is wide consensus that VFAs are critical to the success of BPR. Indeed, several publications have focused on determining the role of specific VFAs for BPR success and PAO biochemical processes, while others have focused on optimization and modeling of fermenters to generate VFAs (Siebritz et al., 1983; Skalsky and Daigger, 1995). However, a void still remains in the literature on how to properly utilize VFAs to gain more complete and consistent BPR in full scale WRRFs. Perhaps part of the discrepancy noticed in the survey is ensuring that PAOs are utilizing the additional VFAs to create PHA rather than for denitrification. WRRFs with influent equalization basins (n=9) tended to produce lower 2TP (p=0.04), potentially owing to the opportunity to create a more consistent flow regime and stable BPR operating environment. With fewer parameters to balance due to reduction of seasonal fluctuations and diurnal flow, operators can focus directly on optimizing BPR, including potential development of an optimum RAS ratio.

### *4.1.2 Operational Insights*

In addition to analyzing numerical operational and wastewater characteristics, general insight from lead operators on RAS rate management as well as maintaining consistent BPR was also performed. The value of operator knowledge cannot be overestimated but it can be hard to put onto paper for general usage. Perhaps key operational parameters are already well known and therefore taken for granted and true operational insight cannot be relayed within a few sentences or paragraphs. It must be assumed that the responses from these operators reflect what they deemed most important to be shared and overlapping themes demonstrate where consensus lies within their approaches no matter system configuration or unit processes involved.

Sludge blanket depths in the secondary clarifier are closely managed by many successful facilities, although there was no general consensus on the ideal blanket depth. The sludge blanket within the secondary clarifier represents the solids that have settled and collected on the bottom of the clarifier. Some operators reported success with a “zero sludge blanket policy” and utilize this policy for managing the RAS rate. If any sludge blanket is noticed, then the RAS rate is increased to remove the accumulated solids. The buildup of excess sludge mass can have downsides within the secondary

clarifier, including potential for rising sludge and some secondary P release (Henze et al., 1993; Mikola et al., 2009; Wouters-Wasiak and Ho, 1996). However, others seek to maintain a consistent blanket of sludge, potentially capitalizing on the ability of the sludge to complete denitrification of any remaining nitrate that would otherwise return to the anaerobic basin via RAS (Brown and Koch, 2005; Koch et al., 1999; Robinson et al.; Siegrist and Gujer, 1994; Siegrist et al., 1995). This discrepancy highlights potential misunderstanding on the impacts of sludge blankets and solids within the secondary clarifier.

There appeared to be three types of approach to managing RAS rates. The majority of respondents indicated RAS was set to a ratio of influent flow (n=9). The second most common approach was a constant flow rate (n=4), primarily used in facilities with influent flow equalization. The least common approach was management through monitoring of characteristics measured throughout the facility – anaerobic P (n=1), RAS P (n=1) and sludge blanket depth (n=1). Average RAS rates ranged from 10% to 100% of the influent flow. There was no ideal RAS ratio that led to lower frequency of failure or 2TP. The facilities that managed according to P release either sought to run the RAS as low as possible while monitoring for P release in the RAS or to increase P release within the anaerobic basins by reducing the RAS flow rate. The potential theory is that as RAS rate is reduced so to is the nitrate load delivered to the anaerobic basin which would ultimately lead to an increase in anaerobic P release. Alternatively, the WRRF that monitored sludge blanket sought to keep zero sludge in the secondary clarifier and would increase the RAS rate if any was noticed.

Operator also directly commented on the importance of maintaining low nitrate in RAS. This may seem like an obvious statement, but the fact that this aspect of BPR was mentioned multiple times - statistically - TN permitted facilities operating with less frequency of failure, unique RAS approaches, and qualitative feedback from operators – combined the emphasis on RAS nitrate control was highlighted more than any other aspect of maintaining consistent BPR. Operator feedback demonstrates the importance of managing RAS nitrate within a BPR system.

## 4.2 Variability of RAS Rate and Ratio at the Moscow WRRF

### *4.2.1 Background*

A forensic evaluation of the Moscow WRRF was conducted to gain insight on the impacts of RAS ratio on secondary clarifier denitrification and the impacts of RAS nitrate load on BPR. According to operations staff, the RAS rate at the Moscow WRRF has historically been set by a constant RAS flow ratio of 36% (18% for each RAS line from each secondary clarifier) of the influent flow rate via their SCADA system. However, even with VFDs on the RAS pumps, an analysis of the recorded RAS pump rates indicates the actual hourly RAS ratio varied from 0% to over 200% from 5/1/2020 to

2/1/2021, which places it within but also well outside typically recommended ratios for BPR facilities (Grady et al., 2011). Figure 7 shows the RAS ratio from 1/15/2021 to 2/22/2021, the focal point of the study. The RAS ratios above the setpoints are primarily due to a limitation of control on minimum RAS pump rates creating bottom bound flow, which is not unique to the Moscow WRRF (Brown and Koch, 2005). Indeed, typical pump and valve sizing is based on average and maximum demands rather than daily minimums. This demonstrates a major design flaw of RAS control, RAS pumps, and associated apparatus. Examination of flow data revealed that during the low flow periods, at night and early in the morning, the RAS ratio spikes due to the inability of the pumps to operate at sufficiently low flow rates. RAS ratios below the setpoint are due to the wasting procedures Moscow employs wherein one RAS pump is shut down while a wasting pump transfers sludge from the associated clarifier to be dewatered and transported offsite. This strategy essentially cuts the RAS flow in half while operating two clarifiers and eliminates all RAS flow when operating only one clarifier (as was observed during summer 2020 process monitoring).

Fortunately, Moscow's wasting strategy allows for studying the impacts of reducing/stopping the flow of solids leaving the secondary clarifier intermittently against continuous removal in addition to the daily fluctuation of the RAS ratio. The operators of the Moscow WRRF were also interested in adjusting the RAS ratio setpoint from 36% to 50% of influent flow in support of this study as well as for other operational curiosities, so the effects of an overall RAS ratio setpoint change were also analyzed. The setpoint was adjusted on 2/1/2021 at 9:45 am from 36% to 50% of the influent flow which is apparent in Figure 7. The setpoint change caused the median and minimum values to obviously change, but maximums remained in the same range barring a few occasional spikes.

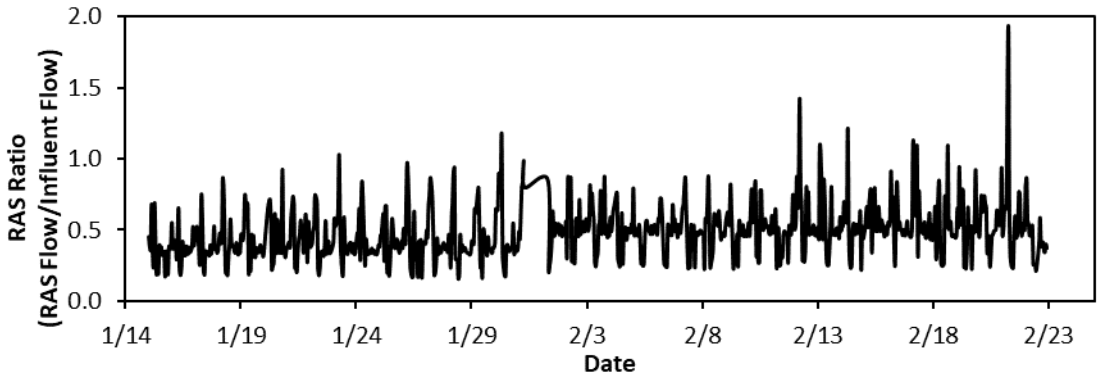


Figure 7. RAS ratio during focal point of study.

### *4.2.2 Results*

Contextualization of the RAS data was done by averaging continuous data such that 24-hour periods could be compared. A primary interest was on daily variations due to the bound flow conditions, and it was suspected that results from overall RAS setpoint changes would be apparent rather quickly. To capture these variations, it seemed appropriate to compare profiles over 24-hour periods as well as daily averages at different setpoints within the two RAS lines, RAS-W and RAS-C. Figure 8 and Figure 9 show the average RAS flow rates by hour for the winter sampling period at the 36% and 50% RAS setpoints. As shown, RAS-W drops substantially due to the wasting strategy from 8:00 am to 11:00 am. Rather than following the setpoint RAS ratios, the RAS flow rates are the same for the 36% and 50% RAS setpoints from 3:00 am to 8:00 am due to the limitations of the pumping system causing the minimum flow rate to be around 265 gpm.

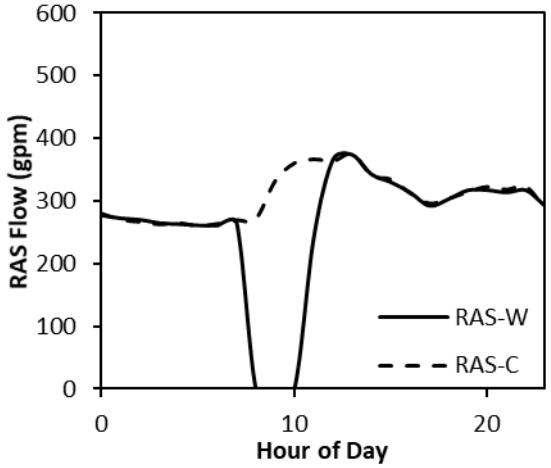


Figure 8. Average RAS-C and RAS-W flow rate from 1/15/2021-2/1/2021, 36% RAS setpoint (n=15).

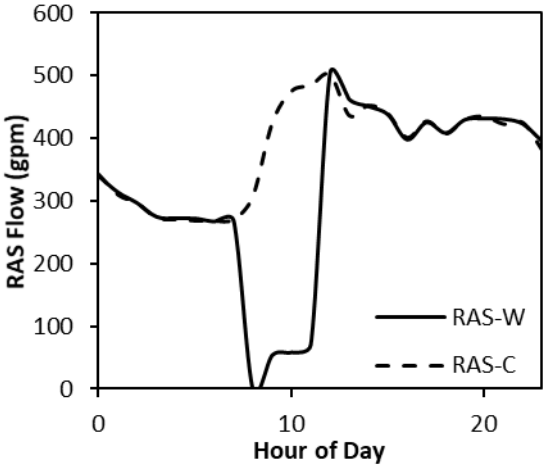


Figure 9. Average RAS-C and RAS-W flow rate from 2/1/2021-2/22/2021, 50% RAS setpoint (n=20).

There did not appear to be any major change in influent flow during the comparison periods for 36% and 50% RAS setpoints, but the RAS ratio in the afternoon obviously changed (Figure 10 and Figure 11). During the late morning and afternoon, the RAS ratio closely matched with the setpoint, but during the early morning the RAS ratio steeply increased during low influent flow due to the minimum RAS flow rate. Around 8:00 am the RAS ratio drastically dropped off as influent flow began to increase, and RAS-W was shut off for wasting. The change in setpoint from 36% to 50% was most apparent in the afternoon and evening when influent flows were high enough to control the RAS rate, but in the early to late morning the RAS ratio and flow rates were nearly identical as they are set by the minimum flow rate of the RAS pumping system.

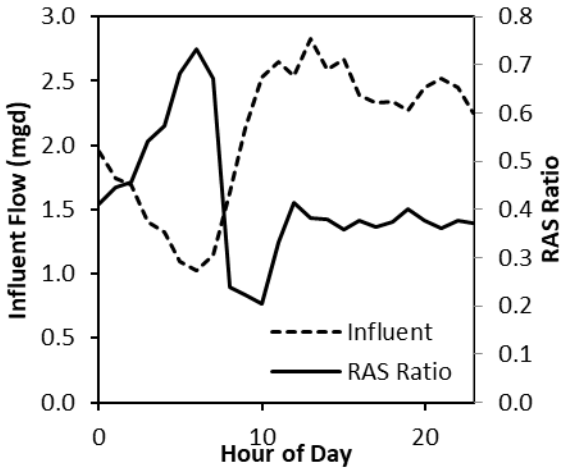


Figure 10. Average RAS ratio and influent flow rate from 1/15/2021-2/1/2021, 36% RAS setpoint (n=15).

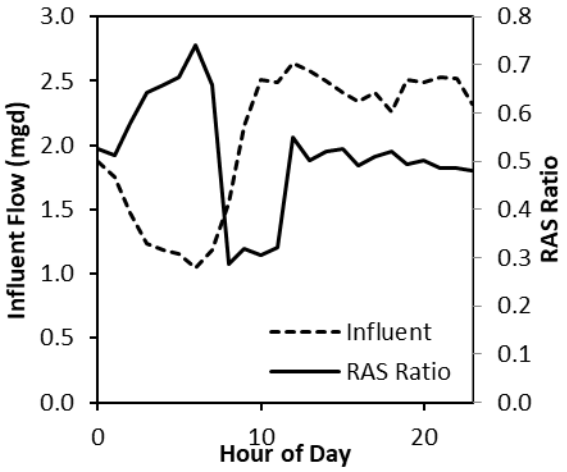


Figure 11. Average RAS ratio and influent flow rate from 2/1/2021-2/22/2021, 50% RAS setpoint (n=20).

### 4.3 RAS Ratio Impacts on RAS TSS

As would be expected of any activated sludge basin, variations of secondary clarifier SRT and total solids led to fluctuations in microbial performance, specifically denitrification. Solids concentrations within the RAS were analyzed to determine how varying RAS ratios throughout the day and at different setpoints affected secondary clarifier SRT. The variation in RAS ratio and influent flow throughout the day led to variation in secondary clarifier HRT, as well as SRT through fluctuating solids loading, removal, and accumulation within the secondary clarifier. A cyclical pattern of solids storage and removal was evidenced within the RAS Total Suspended Solids (TSS) concentrations (Figure 12 and Figure 13). The RAS TSS data demonstrated a high amount of variability hour to hour over the study period, therefore a 4-hour running average over multiple 24-hour periods was used to see the general trends. The solids concentration was significantly reduced at night when the RAS ratio was high, causing the solids removal rate from the secondary clarifier to be greater than solids loading rate. During the morning, the RAS ratio returned to the setpoint, and solids accumulated, leading to a somewhat steady concentration until evening when the RAS ratio again increases. The concentration within RAS-W varies more drastically due to shutting down of the clarifier followed by removing solids at a relatively slower rate via the wasting pump, around 250 gpm. Once RAS-W resumes pumping the solids concentration has drastically increased and is slowly reduced to a steady state concentration around 8 g/L, similar to RAS-C. MLSS within the oxidation ditch typically varied less than 100 mg/L, or ~5%, throughout the day, so the influence on RAS TSS was considered negligible.

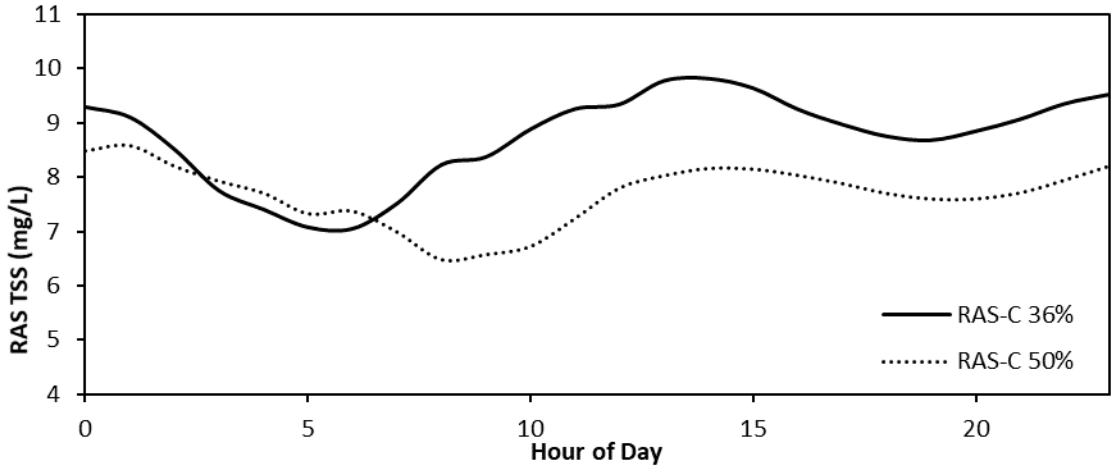


Figure 12. Average RAS-C TSS concentration at 36% (n=3) and 50% (n=5) RAS setpoint.

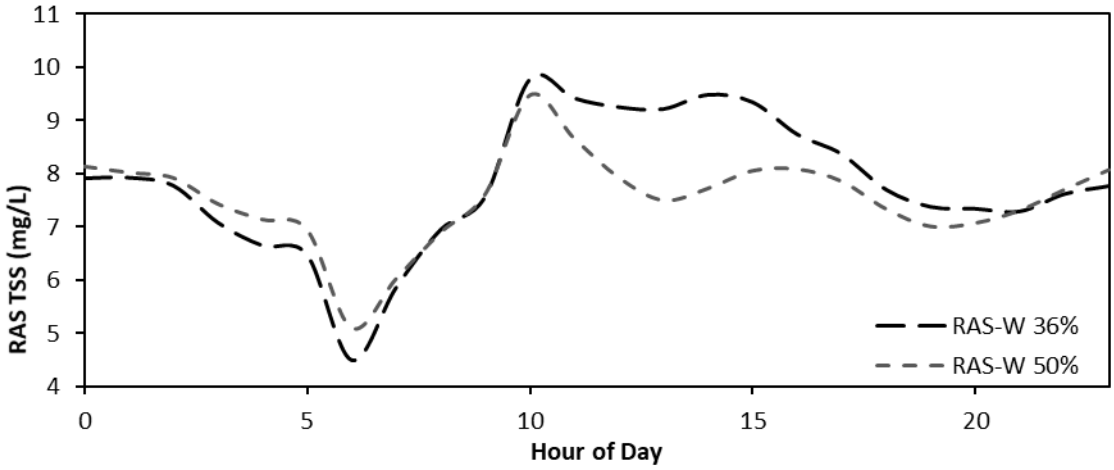


Figure 13. Average RAS-W TSS concentration at 36% (n=3) and 50% (n=3) RAS setpoints.

The increase of the RAS ratio setpoint from 36% to 50% led to noticeably different RAS TSS concentrations within RAS-C, but had a lesser impact to RAS-W. The higher RAS ratio during the day led to less solids accumulating in both clarifiers which reduced the quantity of biomass available for denitrification. The decrease in TSS concentration is less apparent within RAS-W, as the solids accumulate to a high concentration during wasting events at both RAS ratio setpoints. Oddly, the solids concentration was near 0 mg/L at 8 am for all samples on RAS-W and had the appearance of effluent quality water; during this time both the RAS-W and WAS pump are shut down for approximately 1 hour to allow the solids to settle before wasting. Note that solids concentrations shown in Figure 12 and Figure 13 do not represent the actual concentrations measured, as they are 4-hour running averages. It is not entirely clear how clear water could make it to the sample location located on the RAS line. In addition to being free of solids, the sample also contained high concentrations of nitrate similar to what was found within the secondary clarifier effluent.

An increase of solids within the RAS is indicative of an increase of solids accumulated within the secondary clarifier. To reach a higher concentration of solids, the sludge within the clarifier must be given sufficient time and sludge mass to go through compression settling (Tchobanoglous et al., 2014). Based solely on a single daily measurement, operators did not notice a change in the sludge blanket which was typically checked around 9:00 between the 36% and 50% RAS ratio setpoints. Typical sludge blanket measurements were quite low at around 1 foot within Clarifier-W and 6 inches within Clarifier-C. The measurements are taken near the outside of the feedwell approximately 16 feet from the center of the clarifier. Beyond the feedwell, no measurable sludge blanket was found. The increasing solids concentration due to compression may offset an increase in overall blanket depth. These results were echoed within the BioWin modeling (see 4.7 Process Modeling). The variation in solids concentration is indicative of changing SRT within the clarifier, but also affords varying masses of biosolids to perform denitrification as well as endogenous decay.

### 4.4 RAS Ratio Impacts on Nitrate Load

Utilization of ANISE meters for continuous monitoring of the RAS nitrate concentrations allowed comprehensive analysis of nitrate concentration changes due to fluctuating RAS ratios and setpoints (Figure 14 and Figure 15). The RAS nitrate concentration as measured by the ANISE meters varied throughout the day largely corresponding to variations in secondary clarifier SRT and total solids mass therein. The concentration of RAS nitrate was highest in the morning after high RAS ratios depleted the biomass in the clarifier, as evidenced by a low RAS TSS concentration, leading to reduced secondary clarifier denitrification capacity. RAS nitrate concentrations then decreased throughout the late morning and afternoon as the solids concentrations increased in the RAS, indicating higher denitrification capacity in the secondary clarifier. The impact of shutting down Clarifier-W for wasting is obvious, as the nitrate concentrations drop drastically from around 7 to 2 mg-N/L during the wasting cycle.

The increase in RAS ratio setpoint generally caused higher concentrations of nitrate within both RAS lines. The RAS nitrate concentrations increased more rapidly within RAS-W after the wasting cycle, as the accumulated solids are more quickly removed due to the higher RAS flow rates at the 50% setpoint than 36% setpoint. Again, it is of interest to note that the RAS-W TSS was near 0 mg/L at 8:00 am indicating a bypass of the sludge blanket by clarifier liquid. It is not likely that the blanket was completely depleted as the TSS concentration drastically increases once the RAS line begins pumping. This potential bypassing of the sludge blanket by clarifier liquid line would explain why RAS-W nitrate concentrations are higher than RAS-C at this time of day. It also is not surprising that RAS-W nitrate concentrations fluctuate more than RAS-C, as the mass of solids stored within the

Clarifier-W fluctuate more due to the higher variations in flow throughout the day caused by the wasting procedures.

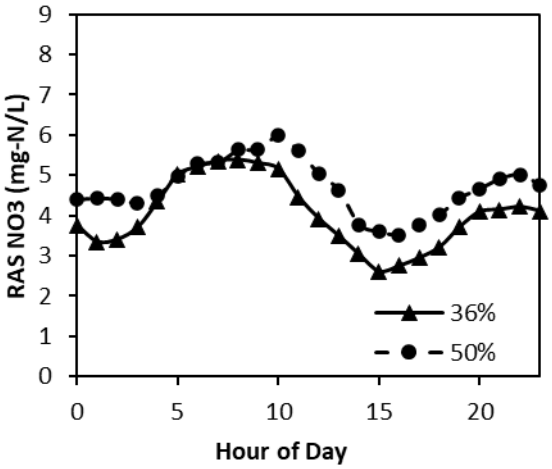


Figure 14. Average RAS-C nitrate concentration for 36% (n=7) and 50% (n=9) setpoints.

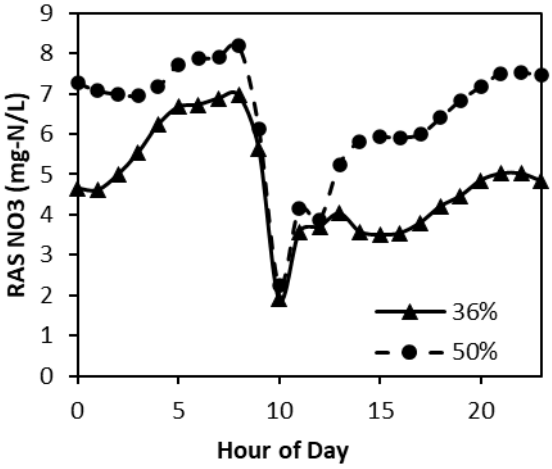


Figure 15. Average RAS-W nitrate concentration for 36% (n=9) and 50% (n=7) setpoints.

The RAS nitrate concentrations are contradictory to what would be expected from an HRT perspective. Indeed, for an activated sludge process a longer amount of time would indicate more denitrification according to typical kinetics (Tchobanoglous et al., 2014). However, typical kinetics based on HRT rely on the assumption that SRT is closely bound with HRT, but this is not the case in the secondary clarifier, as evidenced by the RAS TSS data presented herein. The longest HRTs occur during the night and into the morning, but RAS nitrate actually increased at the same time as HRT did from around 1:00 am to 7:00 am (Figure 16). The HRT of both clarifiers is entirely governed by the loading rate as water continues to flow through the clarifiers whether solids are removed or not. Indeed, both clarifiers have identical HRTs, because influent flow is evenly split between them at all times yet the nitrate data indicates a drastically different level of denitrification in each. The lack of a relationship between secondary clarifier HRT and nitrate concentrations is reasonable in that the solids are mostly separated from the liquid stream and do not experience the same retention time as most of the liquid.

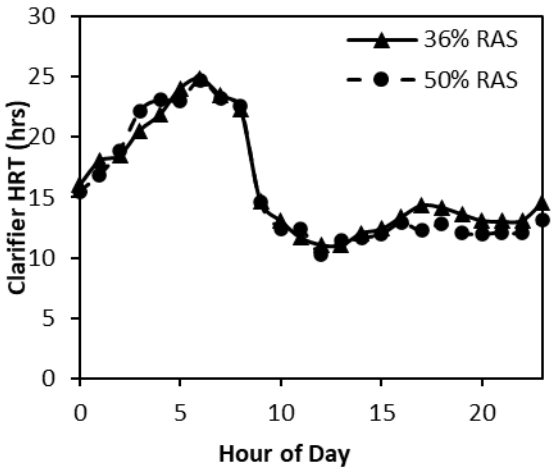


Figure 16. Secondary clarifier HRT at 36% (n=13) and 50% (n=18) setpoints.

While RAS nitrate concentrations are interesting to consider related to denitrification within the secondary clarifier, the mass load of RAS nitrate is equally, if not more, important to the BPR process. Total RAS nitrate load increased throughout the day due to the RAS setpoint change (Figure 17). The change is most apparent in the afternoon when flow rates were controlled by the setpoint. In the early morning, the nitrate load was nearly identical for both setpoints as flow rates became controlled by the minimum boundary condition. Total RAS nitrate load at each hour of the day normalized to plant flow, as influent plus RAS, can be seen in Figure 18. Normalizing the nitrate load to flow demonstrates when the highest concentration of RAS enters the anaerobic basin relative to the amount of influent and substrate. The flow normalized nitrate ranges from less than 1 mg/L to over 3 mg/L throughout the day. Depending on the timing of the day for performance sampling, the impact of RAS nitrate could be vastly over or underestimated. Assessing the capacity of secondary clarifier denitrification and RAS nitrate load is therefore highly dependent on the time-of-day at which samples are taken.

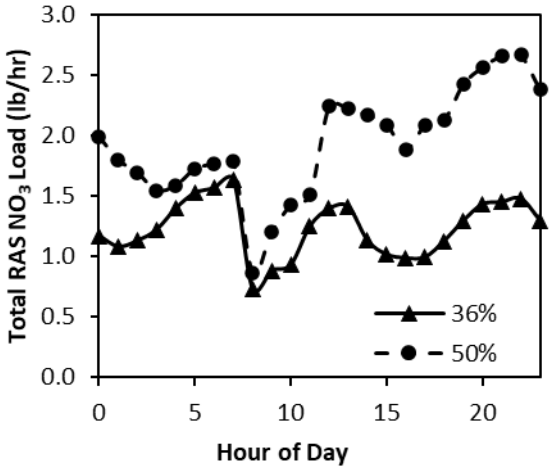


Figure 17. Total RAS $NO_3$ load for 36% and 50% setpoints.

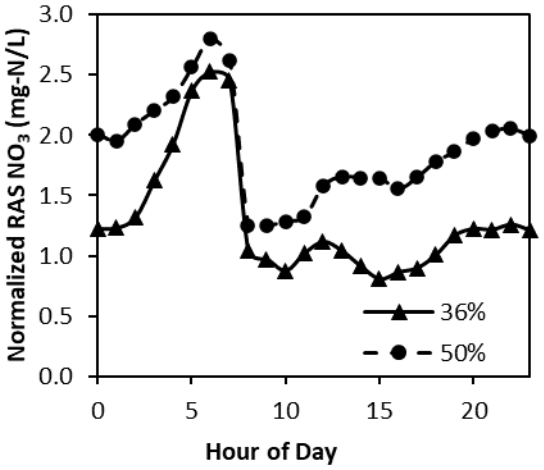


Figure 18. Normalized average total RAS nitrate concentration for 36% and 50% setpoints.

Normalized nitrate concentrations fluctuated considerably throughout the day, but from 8:00 am to 12:00 am the fluctuation was relatively minor compared to 3:00 am to 7:00 am. The sharp increase in nitrate load in the early morning corresponds with the increase in RAS ratio due to the bound flow conditions. If the average concentration from 8:00 am to 12:00 am is considered a baseline for stable RAS ratio conditions, then the overall change due to bound conditions can be compared. The baseline average was 1.1 and 1.7 mg-N/L for the 36% and 50% RAS setpoints, respectively. The average during the increased period from 3:00 am to 7:00 am is 2.2 and 2.5 mg-N/L at the 36% and 50% RAS setpoints, respectively (Table 1). This represents approximately a 200% and 150% increase in RAS nitrate due to RAS ratio variability. The percent change during the 50% RAS setpoint is considerably less due to the impacts of the bound conditions being less drastic when operating at a higher overall RAS ratio. If this increase in RAS nitrate could be lessened through process control, it could be a considerable benefit to BPR systems (discussed further in Section 4.5).

Table 1. Average normalized RAS nitrate increase due to bound flow conditions.

| | **Average Nitrate Concentration (mg-N/L)** | | |
|---|---|---|---|
| **RAS Setpoint** | **8:00 am - 2:00 am** | **3:00 am - 7:00 am** | **Percent Change** |
| 36% | 1.1 | 2.2 | 203% |
| 50% | 1.7 | 2.5 | 146% |

Averaging the oxidation ditch effluent and RAS nitrate concentrations throughout the day indicates the clarifiers removed approximately 7.6% and 4.5% of the total nitrate removed plant-wide over a 24-hour period for the 36% and 50% RAS ratio setpoints, respectively (Table 2). This is lower than what was found in other studies (Koch et al., 1999; Mikola et al., 2009; Robinson et al.; Siegrist and Gujer, 1994) indicating the potential opportunity for further denitrification within the secondary clarifier. Calculation of denitrification within the anoxic basin for comparison assumes the average

influent ammonia is entirely converted to nitrate and is the sole source of nitrate; considering the size of the aerobic ditch and associated operations, this assumption is valid.

Table 2. Total denitrification in clarifiers and anoxic zone.

| | RAS Ratio Setpoint | |
|---|---|---|
| **Denitrification (lbs-N/day)** | **36%** | **50%** |
| Clarifier-C | 13.8 | 10.8 |
| Clarifier-W | 9.6 | 2.1 |
| Clarifier Total | 23.4 | 12.9 |
| Anoxic Basin | 282.8 | 272.9 |
| Total System | 306.2 | 285.8 |
| Percent Clarifier | 7.6% | 4.5% |

The change in RAS ratio setpoint from 36% to 50% led to a 50% increase in RAS nitrate load. At the RAS setpoints of 36% and 50%, the average nitrate load was 29.5 and 46.1 lbs-N/day, respectively (Table 3). Nitrate load is both impacted by the flow rate and the amount of secondary clarifier denitrification and therefore does not change on a 1:1 basis with the amount of secondary clarifier denitrification. Impacts of the RAS rate on nitrate load are therefore two-fold: higher RAS rates carry more water containing nitrate and decrease the capacity of the secondary clarifier to perform denitrification. Additionally, kinetics within the anaerobic zone become lessened due to the dilution effects of higher RAS flows.

Table 3. Average RAS nitrate load at 36% and 50% setpoints within RAS-W and RAS-C.

| | RAS Ratio Setpoint | |
|---|---|---|
| **RAS Nitrate Load (lbs-N/day)** | **36% RAS** | **50% RAS** |
| RAS-C | 14.8 | 21.5 |
| RAS-W | 14.7 | 24.6 |
| **Total RAS Nitrate Load** | **29.5** | **46.1** |

Quantification of denitrification within the secondary clarifier is challenging; nitrate concentration changes must either be measured by averaging concentrations throughout the day or by offsetting the instantaneous readings such that a portion of the sample of water leaving the oxidation ditch is approximately the same sample as tested within the RAS. Pairing concentrations within the oxidation ditch and the RAS line carries several difficulties as flow rates and therefore offset times vary throughout the day. Ideally tracer studies could be done, utilizing something inert such as a dye or sodium chloride, throughout the day to determine SC-SRT. Unfortunately, a tracer test was not completed during sampling at Moscow WRRF. However, during testing at the Moscow WRRF an upset occurred, and the aeration system failed causing a spike in ammonia. The ammonia rose from below 0.3 mg-N/L to over 32 mg-N/L within 24 hours. The increase in ammonia concentration was

obvious within both the oxidation ditch and RAS probe (Figure 19). It took approximately 3 hours for the ammonia spike in the oxidation ditch to be noticed within the RAS line. The upset occurred around 10:00 am. During this time the RAS pump rate was near the average, so it was reasonable to assume that the average offset between a portion of a slug of water entering the clarifier and exiting is around 3 hours. This value will of course vary throughout the day. It was later found that this is relatively close to the SRT predicted using results from BioWin (see *4.7 Process Modeling*) as well as typical values seen at other WRRFs (Mikola et al., 2009). Utilizing an offset of 3 hours, the amount of denitrification in the secondary clarifier was quantified throughout the day (Figure 20 and Figure 21). Further, an SC-SRT of 3 hours would indicate an additional 3 hours of comparable anaerobic SRT. Typical BPR design suggests a minimum of 1.5-3 days anaerobic SRT, so SC-SRT could contribute an additional 4-8% to the overall anaerobic SRT.

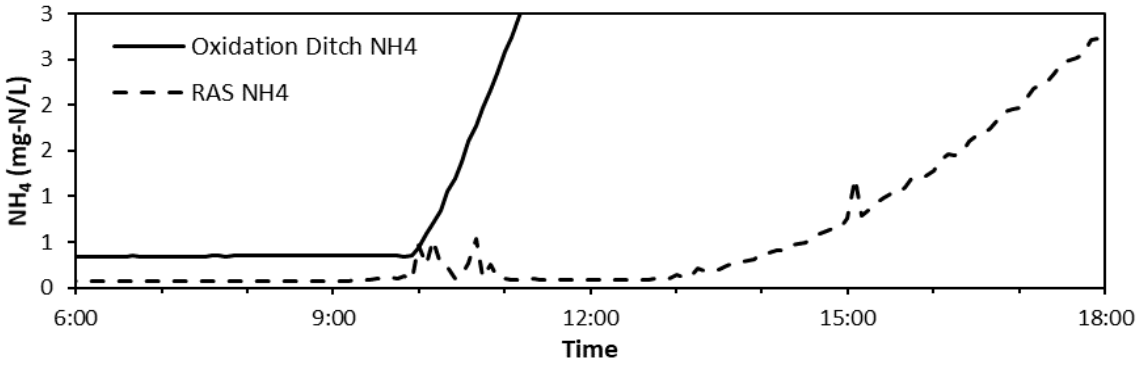


Figure 19. NH4 concentrations during aeration failure within RAS and oxidation ditch.

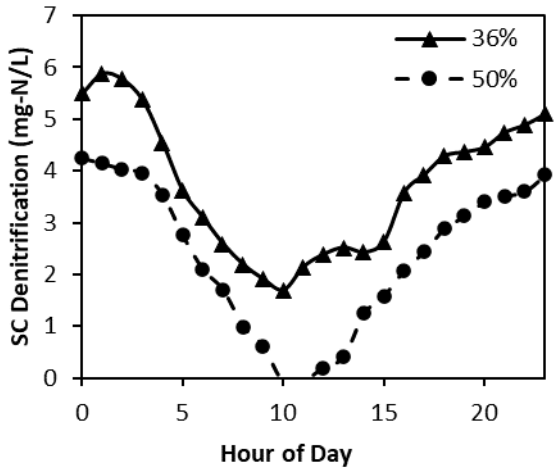


Figure 20. Clarifier-C denitrification for 36% (n=7) and 50% (n=9) setpoints.

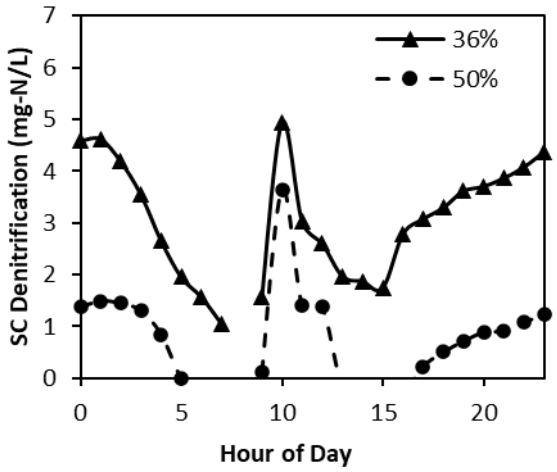


Figure 21. Clarifier-W denitrification for 36% (n=9) and 50% (n=7) setpoints.

Utilization of denitrification estimates presented in Table 2 as well as solids data allowed for estimation of specific denitrification rates (SDNR) within the secondary clarifier. Concentration of RAS TSS was applied to estimate the total amount of solids within the clarifier assuming that the

sludge blanket depth of 0.5 to 1 foot varied linearly from the RAS TSS concentration to 3,000 mg/L. A TSS concentration of 3,000 mg/L was the assumed concentration at which the blanket began to form, which was in agreement with work by Koch et al. (1999) as well as how the blanket height was measured with BioWin. Solids still in suspension actually make up the majority of the solids in the clarifier and were quantified assuming a linear variation throughout the height of the clarifier above the sludge blanket from 3,000 to 0 mg-TSS/L. The solids were only assumed to be present within the feedwell of the clarifier as no sludge blanket was ever detected outside of this area. The resulting average SDNR for the 36% and 50% RAS setpoints in Clarifier-C were 0.010 and 0.007 g-N/g-VSS*day, respectively. Complementary clarifier batch tests were slightly higher than those estimated in the full-scale system at 0.026 g-N/g-VSS*day. Collectively these values are in agreement with literature values presented for post-anoxic denitrification, which is substantially lower than pre-anoxic denitrification as demonstrated in Table 4. It has been postulated that the primary driver for pre-anoxic denitrification is PHA and readily available organic carbon where post-anoxic denitrification is driven by glycogen (Coats et al., 2011) or endogenous decay (Siegrist et al., 1995; Tchobanoglous et al., 2014). Due to the significantly more complicated flow pattern of Clarifier-W as well as the potential bypassing of the sludge blanket, SNDR's were not quantified utilizing RAS-W nitrate data.

Table 4. SDNR from literature and this study.

| Scenario | SDNR ($g\text{-}NO_3\text{-}N/g\text{-}VSS*day$) | Source |
|---|---|---|
| Pre-Anoxic | 0.04-0.25 | (Tchobanoglous et al., 2014) |
| Post-Anoxic/No Additional Carbon | 0.01-0.03 | |
| | 0.017-0.022 | (Coats et al., 2011) |
| Secondary Clarifier | 0.033 | (Koch et al., 1999) |
| 36% Clarifier-C | 0.010 | This study |
| 50% Clarifier-C | 0.007 | |
| Clarifier Batch Tests | 0.026 | |

Peak nitrate concentrations within RAS-W exceeded those within the oxidation ditch on three days in a row during the 50% RAS setpoint data collection, albeit typically by less than 1 mg-N/L. While this was unexpected, it is potentially possible through endogenous decay within the clarifier and the presence of oxygen carried over from the oxidation ditch allowing ammonia release and nitrification, but no denitrification. Theoretically, destruction of biomass could lead to release of ammonia. The ammonia would be oxidized with the dissolved oxygen producing nitrate at a stoichiometric ratio of 0.06 $g\text{-}NO_3\text{-}N/g\text{-}O_2$ (Tchobanoglous et al., 2014). Assuming all available dissolved oxygen were utilized to oxidize ammonia, an initial concentration of 2 $mg\text{-}O_2/L$ would yield 0.12 $mg\text{-}NO_3$ as N/L.

However, if endogenous decay were leading to ammonia it would also likely lead to additional carbon which would be oxidized by dissolved oxygen and decrease the amount going toward nitrification.

Understanding the carbon source utilized for denitrification within the secondary clarifier allows for more complete understanding of the driver and what the potential implications may be. For example, if PHA is the main carbon source utilized, the benefits to BPR may be compromised by utilizing the carbon source for P uptake instead for denitrification in the secondary clarifier. A potential carbon source for denitrification could be internal carbon storage reserves, namely PHA and glycogen. In order to assess this possibility, PHA and glycogen samples were taken from the oxidation ditch and the RAS lines twice daily during winter at the full-scale system, 8:00 am and 2:00 pm, and once daily at 10:00 am during the summer in the full-scale and scale model systems. The results from the ANISE probe indicate that comparison of grab samples taken at the same time can be misleading, therefore biokinetic data would potentially be inaccurate. Nevertheless, the data and associated analyses does allow preliminary insight into carbon sources for denitrification. No significant variations between the 8:00 am and 2:00 pm grab samples were noticed, so they were averaged together in hopes of gaining a more representative value. Table 5 displays the change in concentration of PHA and glycogen from the oxidation ditch to the RAS line. The negative values indicate there was an increase in concentration whereas positive values indicate a decrease. The winter PHA concentration tended to decrease slightly, but also saw an accumulation within the RAS-C line during the 36% RAS setpoint. This is likely due to an outlier raising the average but does indicate the possibility of PHA synthesis within the clarifier, which could be driven by glycogen consumption. The highest PHA (10 mg-COD/g-TSS) concentration occurred when a large RAS P-release (2.42 mg-P/L) was also noticed. The glycogen concentrations generally trended downward with the exception of the RAS-W line at the 36% RAS setpoint. Overall, the data indicate that both PHA and glycogen are utilized within the secondary clarifier for denitrification. Similarly, the scale model data indicate that both PHA and glycogen are utilized for denitrification, particularly at the 33% RAS setpoint. At the 100% setpoint, neither PHA nor glycogen percentages change, potentially indicating insufficient time within the secondary clarifier to utilize internal carbon storage.

Table 5. PHA and glycogen change from oxidation ditch to RAS line.

| Carbon Source | Setpoint | RAS Line | Change from OD to RAS (mg-PHA/g-TSS) | Samples |
|---|---|---|---|---|
| **PHA** | 36% | C | -1.52 | 6 |
| | | W | 0.82 | 4 |
| | 50% | C | 0.90 | 9 |
| | | W | 0.50 | 6 |
| | Summer | C | 0.00 | 21 |
| | SM 33% | | 0.39 | 7 |
| | SM 100% | | 0.00 | 7 |
| **Glycogen** | 36% | C | 0.87 | 8 |
| | | W | -2.89 | 4 |
| | 50% | C | 4.51 | 10 |
| | | W | 2.12 | 5 |
| | Summer | C | 4.00 | 24 |
| | SM 33% | | 0.21 | 7 |
| | SM 100% | | -0.01 | 6 |

Note: Negative values indicate an increase in concentration from the OD to RAS line. Scale Model (SM) changes are measured from the post-anoxic basin to the single RAS line.

Another potential source of carbon for denitrification is endogenous decay (Siegrist et al., 1995; Tchobanoglous et al., 2014). Direct measurement of endogenous decay was not a component of this research, but $NH_4$ could potentially be used as a proxy for endogenous decay. As bacteria decay they release various nutrients, notably carbon and organic nitrogen that is converted to $NH_4$. As presented previously, it has been proposed that roughly half of the ammonia within VSS is released as $NH_4$ during endogenous decay or 0.06 g $NH_4$-N/g VSS-decayed, with the other half remaining inert (Tchobanoglous et al., 2014). Assuming 1.42 g-COD/g-VSS-decayed and 2.86 g-COD/g-$NO_3$-N (Tchobanoglous et al., 2014), this converts to 8.28 g-$NO_3$-N denitrified per gram $NH_4$-N released. Therefore, a minimal amount of ammonia release could indicate sufficient decay of VSS to generate the necessary carbon for the denitrification measured. It is recommended that future work attempt to correlate ammonia release with endogenous decay. During the clarifier batch tests, a slow but steady increase of $NH_4$ was noticed. Utilizing the previously presented stoichiometric ratio, the ammonia released could account for approximately 22-54% of the nitrate denitrified in the batch test experiments. Further batch testing is recommended to gain more insight into the source of carbon utilized for denitrification within the secondary clarifier.

### 4.5 RAS Nitrate Impacts on BPR

Ultimately a critical element of this research was to better understand the potential impact of RAS ratio, and in turn RAS nitrate, on performance of Moscow's BPR system. As described, daily RAS ratio fluctuation as well as the overall RAS setpoint increase led to high RAS nitrate loads. The consequence of the RAS ratio increase was observed as decreased phosphorus concentrations in the

anaerobic zone. Grab samples taken 2-3 times daily during the winter in the full-scale system at 8:00 am, 11:00 am and 2:00 pm demonstrated the impacts of variable nitrate load throughout the day (Figure 22). The largest flow-normalized nitrate loading to the anaerobic zone occurred from 5:00 to 7:00 am, and at 8:00 am the impacts to the anaerobic zone were most apparent in the first anaerobic basin, AN1 (Figure 23 and Figure 24). The impacts are evidenced by both low P release and low PHA synthesis when compared with the concentrations at 2:00 pm within AN1. By 11:00 am the decreased anaerobic activity has caused lower P and PHA concentrations in AN1, AN2, and AN3. At 2:00 pm the concentrations within all basins have increased, but most substantially within AN1 and AN2 indicating that AN3 is still recovering (Figure 25 and Figure 26). The impacts of the RAS ratio setpoint change were also clearly demonstrated in the P and PHA profiles. Both the 8:00 am and 2:00 pm P and PHA concentrations were decreased during the RAS ratio setpoint of 50% when compared to that of 36%. Both RAS ratio setpoints created similar morning and afternoon P and PHA profiles due to the RAS ratio being controlled by flow limitations rather than the setpoint. The 2:00 pm P profile displays more variation due to the RAS setpoint because the flows in the late morning and afternoon were closely tied to the setpoint.

The changes in P and PHA concentrations within the anaerobic basins largely mirrored one another. Specifically, as the P concentration decreased so too did the amount of PHA synthesized. These results affirm the metabolic understanding of PAOs and OHOs within the anaerobic zone as the presence of nitrate introduces an electron acceptor thereby switching the metabolism of PAOs and OHOs to both consume VFAs for growth and energy production instead of forcing PAOs to release P and store VFAs as PHA (Grady et al., 2011). The ability to store VFAs as PHA for use aerobically/anoxically is unique to the PAOs and gives them a competitive advantage. A system with a degraded anaerobic system risks decreasing the metabolic advantage PAOs have to uptake VFAs anaerobically and store as PHA and in turn could decrease the overall population of PAOs – ultimately leading to BPR failure.

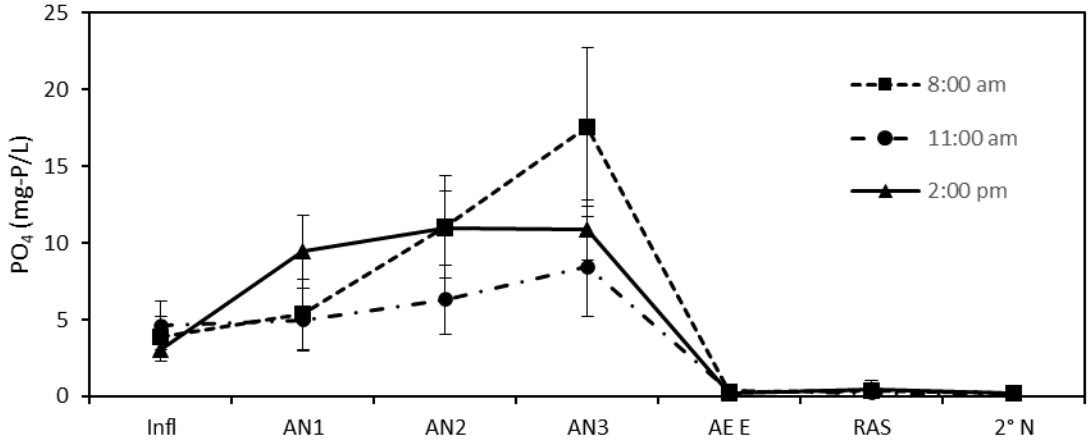


Figure 22. Moscow WRRF $PO_4$ profiles at 8:00 am, 11:00 am, and 2:00 pm.

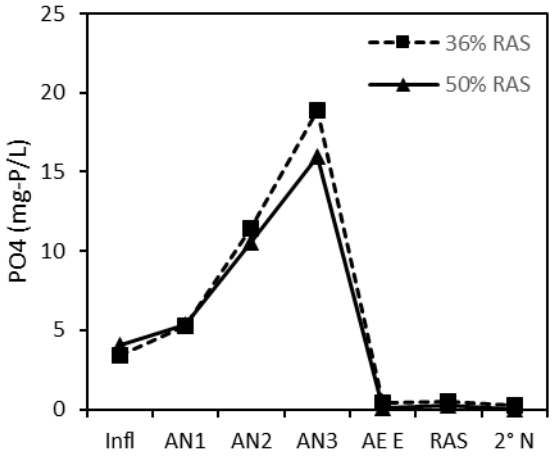


Figure 23. $PO_4$ profiles at 8:00 am for 36% and 50% RAS setpoints.

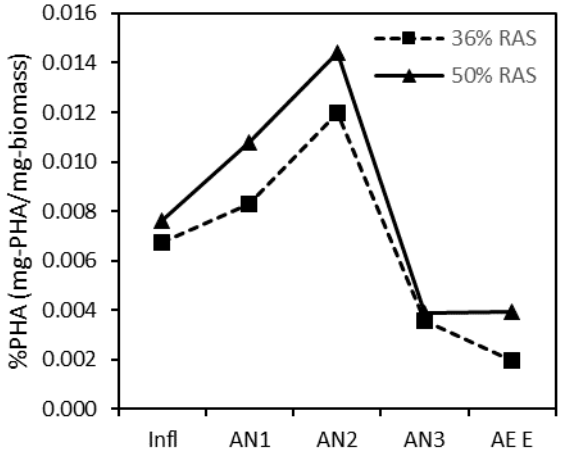


Figure 24. PHA profiles at 8:00 am for 36% and 50% RAS setpoints.

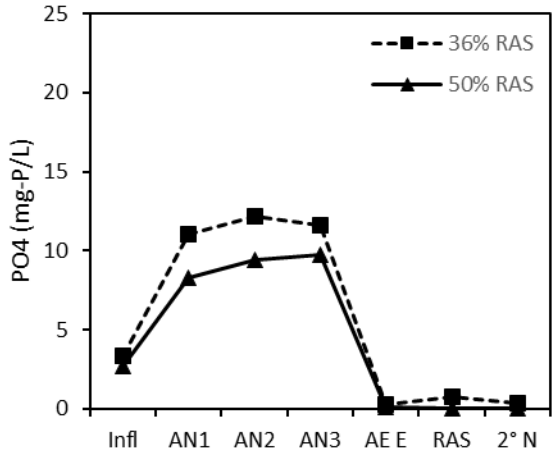


Figure 25. $PO_4$ profiles at 2:00 pm for 36% and 50% RAS setpoints.

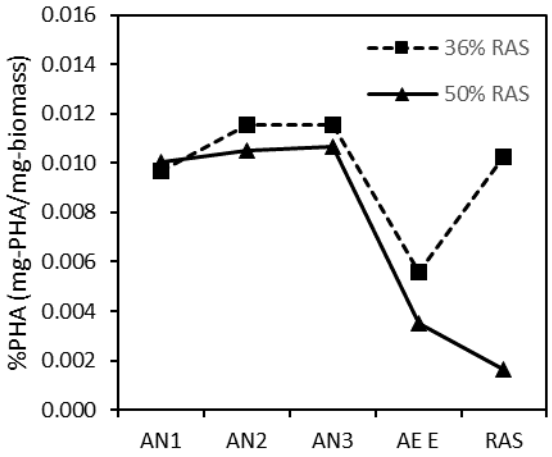


Figure 26. PHA profiles at 2:00 pm for 36% and 50% RAS setpoints.

Similar reductions in P release and PHA synthesis were observed within the scale model study corresponding with RAS ratio increase. A large decrease in anaerobic P release occurred when the RAS rate was increased from 33% to 100% of the influent flow. The scale model was operated at constant influent flow and utilized primary clarification followed by a primary solids fermenter. The loading on the basins was not anticipated to change as drastically throughout the day due to the constant flow rates and substrate equalization effect of the primary clarifier and fermenter and therefore was not measured at different times of day. Here the flatness of the P profile could be due to the fermenter degrading slowly biodegradable COD (sbCOD) to readily biodegradable COD (rbCOD) that is taken up within AN1 instead of degrading in the anaerobic basins and causing additional P release within AN2 and AN3.

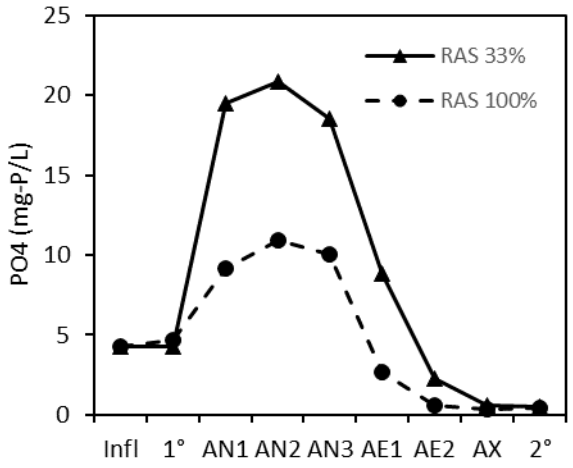


Figure 27. Scale model $PO_4$ profiles at 33% and 100% RAS ratios.

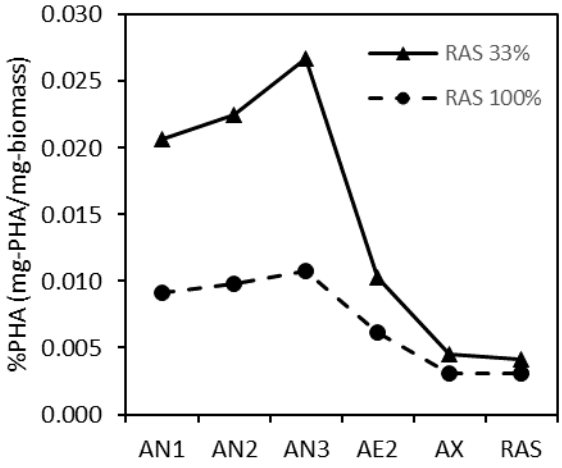


Figure 28. Scale model PHA profiles at 33% and 100% RAS ratios.

Results presented thus far indicate anaerobic P release and PHA production are both considerably impacted by the RAS nitrate load. A quantifiable metric in terms of P release to nitrate load change would allow operators to potentially correlate changes in the anaerobic basin to RAS nitrate loading. Quantification of the degree to which P release is impacted due to the varied nitrate load requires consideration of the timing of the nitrate load and when P samples were taken. One approach utilized is a comparison between P concentrations at 2:00 pm for the different RAS ratio setpoints. The P release change in the morning is minimally impacted by the setpoint change because RAS flow and ratio are governed by the minimum RAS flow during the early morning not the setpoint. Therefore, the largest impacts of the setpoint change are most noticeable in the afternoon – specifically the 2:00 pm sample time – when RAS flow is governed by the setpoint. The next comparison considered is the change in RAS nitrate load in the early morning versus the average throughout the rest of the day. The impacts of this change are most noticeable within AN1 at the 8:00 sample time as the nitrate load has primarily impacted only AN1; here AN1 in the afternoon can serve as a comparison for the lower nitrate load at both RAS setpoints. The scale model change in RAS rate also correlated with increase in RAS nitrate load and subsequent decrease in P release within the anaerobic zone. To calculate the impacts of nitrate load on P release, the nitrate and P release need to be put onto similar terms, here COD is utilized as the common denominator.

Anaerobic metabolisms of PAOs leading to PHA synthesis are not entirely understood. One metric that has been utilized is the P:C ratio, defined as the amount of P released to VFAs consumed where VFA is typically only acetate. Filipe et al. (2001) and Smolders et al. (1994) have found this ratio to be dependent on pH. Both authors developed equations to generate the stochiometric ratio given pH. Assuming a pH of 7.2 based on Moscow WRRF daily monitoring reports, the ratio for Moscow

would be around 0.55 mg-P/mg-COD. Smolders et al. (1994) also present the work of several other authors where the P:C values ranged from 0.24 to 0.73 mg-P/mg-COD as HAc. All studies reviewed were fed synthetic wastewater and controlled denitrification by inhibiting nitrifiers. However, synthetic wastewater does not always behave like raw wastewater wherein a large variety of VFAs can be found (Romenesko and Coats, 2018). In an analysis of past data collected, Coats et al. (2017) found that when the P:C ratio was at least 0.18, the effluent P was less than 0.5 mg-P/L. All data presented by Coats et al. (2017) was collected while inhibiting nitrifiers and therefore purely focused on P released per carbon utilized, neglecting any impacts of nitrate. As further reference, the default value for P:C within BioWin is 0.51 mgP/mgHAc.

Denitrification is largely a straightforward redox reaction, and the stoichiometry indicates that 5.2 mg-COD are consumed per mg-$NO_3$ as N. Utilizing the common factor of COD, P:C and the stoichiometry of denitrification yields a ratio of P release reduction due to nitrate load of 0.94 to 3.80 mg-P/mg-N. Quantification of the impacts during this study indicate the reduction in P release was on the high end of this range and at times beyond, with values ranging from 2.9-5.2 mg-P/mg-N (Table 6). The impact on P release exceeding the theoretical maximum could be due to variations in influent characteristic changes (carbon, P, nitrate), RAS TSS, and the intricacies of microbial metabolisms not captured in these analyses. In any case, the results demonstrate the significant influence nitrate can have on anaerobic zone performance. Larger mg-P/mg-N ratios within AN1 than AN2 and AN3 both in the scale model and full-scale system indicate that impacts of the nitrate load could become diluted as other sbCODs are converted to VFAs during the anaerobic retention time allowing further P release.

Table 6. P release reduction at different flow normalized nitrate concentration changes.

| P Release Comparison | P Release Change (mg-P/L) | RAS Nitrate Comparison | RAS $NO_3$ Change (mg-N/L) | P/N (mg-P/mg-N) |
|---|---|---|---|---|
| AN1 8:00 vs. AN1 2:00 pm – 36% RAS | 5.73 | Baseline vs. Peak – 36% | 1.1 | 5.2 |
| AN1 8:00 vs. AN1 2:00 pm – 50% RAS | 2.87 | Baseline vs. Peak – 50% | 0.8 | 3.6 |
| RAS 36% vs. RAS 50% at 2:00 pm – AN1 | 2.78 | Average 36% vs. 50% | 0.57 | 4.8 |
| RAS 36% vs. RAS 50% at 2:00 pm – AN2 | 2.81 | | 0.57 | 4.9 |
| RAS 36% vs. RAS 50% at 2:00 pm – AN3 | 1.90 | | 0.57 | 3.3 |
| RAS 33% vs. 100% – AN1 Scale Model | 10.3 | Average 33% vs. 100% | 5.15 | 3.5 |
| RAS 33% vs. 100% – AN2 Scale Model | 9.92 | | 5.15 | 3.4 |
| RAS 33% vs. 100% – AN3 Scale Model | 8.47 | | 5.15 | 2.9 |

Results from this research demonstrate substantial fluctuation of P:C, 0.45 to 0.98 mg-P/mg-COD, can be largely due to RAS nitrate loading and are highly dependent on timing of sampling (Table 7). Lower P:C ratio are often credited to the presence of GAOs (an anaerobic competitor for VFA uptake), with little attention given to the impacts of nitrate load (Onnis-Hayden et al., 2020). It is unlikely this variability could be due to fluctuation in the microbial population due to the rapidity of changes and consistent cyclical nature corresponding with RAS nitrate loading. The P:C ratio is calculated with the carbon substrate, C, as mg-VFA rather than HAc where VFA is the sum of acetic, propionic, butyric, and valeric acids. P release was calculated as the average AN3 P less the average influent P during the specified scenario. VFA loading did not appear to significantly change between the different sampling scenarios and was therefore averaged to create a more representative value, as composite sampling was not completed.

Table 7. P:C ratios during winter sampling.

| RAS Setpoint | Time of Day | P:C |
|---|---|---|
| 36% RAS (AN3) | 8:00 | 0.98 |
| | 2:00 pm | 0.53 |
| 50% RAS (AN3) | 8:00 | 0.75 |
| | 2:00 pm | 0.45 |
| 33% RAS (AN3) – Scale Model | - | 0.38 |
| 100% RAS (AN3) – Scale Model | - | 0.14 |

The high P:C ratios in the morning are potentially due to the exceptionally long anaerobic HRTs around 11-14 hours from 3:00-7:00, leading to fermentation of the influent wastewater and development of additional VFAs. A comparison between the P:C ratios within the full-scale and scale model systems may also demonstrate the influence of sbCOD on P:C ratios. For the scale model, it is likely that the sbCOD is largely broken down to VFAs within the primary solids fermenter. This increases the measured VFA concentration thereby decreasing the P:C ratio, whereas the sbCOD is likely broken down within the anaerobic basins of the full-scale system and is not quantified within the measured influent VFAs. This discrepancy also demonstrates the influence of primary solids fermenters and anaerobic retention time on P:C ratios.

The presence of nitrate in the anaerobic basin is thought to switch both PAO and OHO metabolisms such that they utilize VFAs for growth rather than PHA storage while denitrifying. Indeed, the batch test results demonstrate this. During the anaerobic batch tests, no P was released until nitrate was removed to near zero concentrations. In fact, P was actually removed from bulk solution in the presence of nitrate just as it typically is within the anoxic zone of many WRRFs including Moscow. Influent wastewater utilized for the batch tests appeared to contain little VFAs and changes throughout the test were negligible. Denitrification likely occurred utilizing other carbon sources, as evidenced by the minimal change in PHA while P was stored.

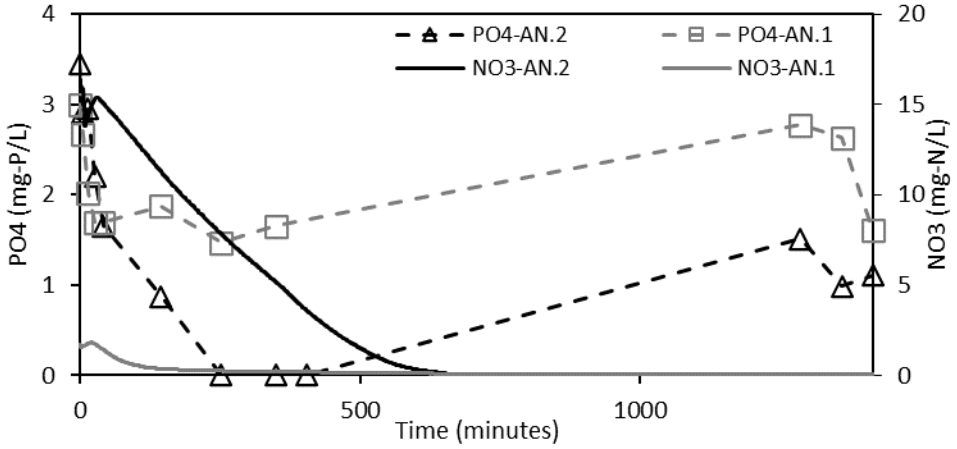


Figure 29. Anaerobic batch test nitrate (mg-N/L) and PO4 (mg-P/L) concentrations over time.

### 4.6 RAS Ratio Impacts on Secondary Clarifier Effluent

It has been postulated that denitrification within the secondary clarifier can lead to deleterious effects on the secondary clarifier effluent either through increases in TSS or P (Henze et al., 1993). Effluent grab samples were collected throughout the analysis period to determine if these postulations held under the scenarios tested. Changes in the RAS ratio, nitrate load, and anaerobic P release appeared to have minimal impact on secondary clarifier effluent quality (Table 8). Ultimately it is evident the full-scale and scale model anaerobic systems have sufficient capacity to remove nitrate at the loads tested while maintaining consistent effluent P values. This result echoes the analytical analysis of the BPR survey wherein it was found that having a TN permit yielded lower frequency of failure but had no impact on average secondary TP concentrations. Generally, a BPR system, and more specifically the anaerobic basins, potentially have a specific threshold to which they can denitrify before the effluent P increases causing a failure. This threshold is dependent on influent carbon concentrations as well as the anaerobic volume. Effluent P remained quite low, but did appear to slightly decrease at the higher RAS ratio setpoint within the full-scale and scale model systems contradictory to data presented by Grady et al. (2011). While a longer-term study would demonstrate whether decreases in effluent P are sustainable after RAS rate changes, results indicate that short-term reductions in effluent P may occur after an increase in RAS rate. Somewhat contrary to findings presented by Henze et al. (1993), there was a small increase in TSS at the higher RAS ratio setpoint which caused a decrease in secondary clarifier denitrification.

Table 8. Effluent characteristics for Moscow WRRF and the scale model.

| RAS Setpoint | 2° $PO_4$ (mg-P/L) | RAS $PO_4$ (mg-P/L) | TSS (mg/L) |
|---|---|---|---|
| 36% | 0.20 ± 0.06 (n=19) | 0.52 ± 0.18 (n=12) | 463 ± 68 (n=8) |
| 50% | 0.10 ± 0.04 (n=30) | 0.18 ± 0.05 (n=14) | 547 ± 20 (n=14) |
| 33% SM | 0.51 ± 0.03 (n=7) | - | - |
| 100% SM | 0.40 ± 0.03 (n=8) | - | - |

Batch test data further sheds light on the potential impacts of extended SC-SRT to effluent quality through secondary P release. During the clarifier batch tests, the presence of nitrate inhibited the release of P (Figure 30). Similar to the anaerobic batch tests, P release did not occur until directly after all nitrate was reduced. Clarifier batch test C.1 was spiked with additional nitrate as is apparent by the concentration at time t=0. The increased initial concentration appears to delay P release until the nitrate is removed, but the P release and denitrification rates appear consistent between both tests. The lack of P release in the presence of nitrate corresponds with previous findings (Mikola et al., 2009; Wouters-Wasiak and Ho, 1996) and demonstrates the propensity of nitrate to inhibit potential secondary P release within the sludge blanket from entering the secondary effluent.

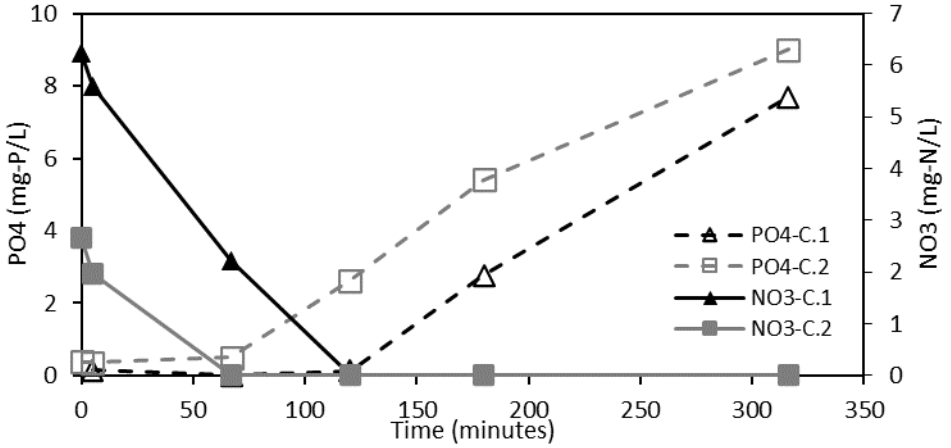


Figure 30. Clarifier batch test nitrate (mg-N/L) and PO4 (mg-P/L) concentrations over time.

## 4.7 Process Modeling

Coupled with the forensic study of the Moscow WRRF, a model of the Moscow WRRF was created using BioWin. After calibration of the model with winter data, additional RAS rates were tested to determine if the trends noticed at Moscow WRRF and the scale model continued. Modeling also provides insight in areas that were not feasible through traditional sampling techniques, such as continuous monitoring of solids stored within the clarifier and P release from the anaerobic basins. Additionally, modeling allowed more thorough analyses of the impacts of bound vs. unbound RAS flow conditions as well as strategies to optimize RAS rate control.

### *4.7.1 Model Calibration*

The BioWin model was calibrated using the data collected during winter 2021, with the aim to analyze RAS ratios beyond those tested within the full-scale or scale model systems. The primary metrics considered for calibration were RAS nitrate, P, and TSS as well as effluent P. Anaerobic P values were attempted to be calibrated, but adjustment of model parameters did not yield values of the

same magnitude as those measured. The main PAO parameter toggled to attempt anaerobic basin calibration was the P:C ratio, but this parameter was found to be extremely sensitive. Small changes caused substantial fluctuation in effluent P concentrations. Oddly both a decrease and an increase in the P:C ratio caused less P release anaerobically and a higher effluent P, indicating a relatively narrow window in which the model had reasonably representative results. While the relative magnitude of P release did not calibrate well, the model displayed the same trends as those found through grab sampling at the Moscow WRRF. Other PAO specific parameters were explored but none yielded the anaerobic P concentrations seen at Moscow WRRF. The only system-wide biokinetic parameter changed was for nitrite oxidizing bacteria. Nitrite oxidizing bacteria were enhanced via the maximum specific growth rate and substrate half saturation coefficient to reduce nitrite in the effluent to values seen at Moscow WRRF, which were always below 0.2 mg-N/L. Total P recovery was considered reasonably close to the measured values at 90% biological P recovery (0.37 mg-P/L effluent) in BioWin versus an average 95% (0.17 mg-P/L effluent) at Moscow WRRF in the winter.

Initial trials indicated that the biological activity within the modeled secondary clarifier was so high it caused complete reduction of nitrate and substantial P release. Therefore, the secondary clarifier was given local kinetic parameters to adjust the amount of microbial activity locally without impacting the rest of the system. Adjustment of the anoxic hydrolysis factor served to decrease the amount of denitrification as well as reduce the P release to be more in line with Moscow WRRF values. The default BioWin anoxic hydrolysis growth factor is 0.28 but needed to be reduced to 0.07 to generate comparable nitrate concentrations in the RAS given similar oxidation ditch nitrate concentrations. The impact of this factor lends insight into what controls metabolic activity within the secondary clarifier as modeled within BioWin. The hydrolysis rate is a kinetic rate at which particulate substrate is broken down into readily biodegradable substrate in aerobic conditions. The anoxic hydrolysis factor is applied under anoxic conditions to simulate a decrease in microbial activity due to fewer microbes being able to use nitrate/nitrite as an electron acceptor as use oxygen in aerobic conditions. The impact of the anoxic hydrolysis factor on modeled secondary clarifier denitrification indicates the controlling factor is the availability of external substrate, cell decay, rather than internal carbon storage. Further, the base hydrolysis rate is 2.1 $d^{-1}$; multiplying this by the anoxic hydrolysis factor, 0.07, to obtain the hydrolysis rate under anoxic conditions yields 0.15 $d^{-1}$ which is in range of the SDNR found within this research as well as those presented by literature for post-anoxic denitrification (Tchobanoglous et al., 2014). P release within the sludge blanket continued to occur even in the presence of nitrate, so the PAO anoxic growth factor was also reduced from a default value of 0.33 to 0.10 to decrease the amount of P released without further impacting denitrification. The only settling parameter adjusted was the specified TSS concentration utilized for determining the

height of the sludge blanket. The default value for this parameter is 2,500 mg/L, but was increased to 3,000 mg/L to be in line with Koch (1999) and to avoid potential model issues as MLSS within the basins was typically greater than 2,500 mg/L.

Using the RAS rates measured at the full-scale facility for the 36% and 50% setpoints in the BioWin model yielded similar results for RAS TSS within RAS-C and RAS-W (Figure 31 and Figure 32). At 8:00 am the RAS TSS for RAS-W was near zero. This oddity is still unexplained and was removed from Figure 32 to allow a more direct comparison of typical values. It is apparent that RAS TSS concentrations are more sensitive to the flow changes within RAS-W as the system goes through a wasting cycle. The resolution of data within BioWin likely creates the seemingly more sensitive RAS-W TSS fluctuations. Though the magnitudes are nearly identical, the RAS TSS concentration profiles have slightly different timing than what was measured. The highest and lowest TSS values appeared to occur later in the BioWin model than was measured. Adjustment of settling parameters did not seem to significantly realign the timing between BioWin and the measured values. It is not clear why the model and the measured values displayed different timings, but the relative trends are similar and were considered the primary interest of this research.

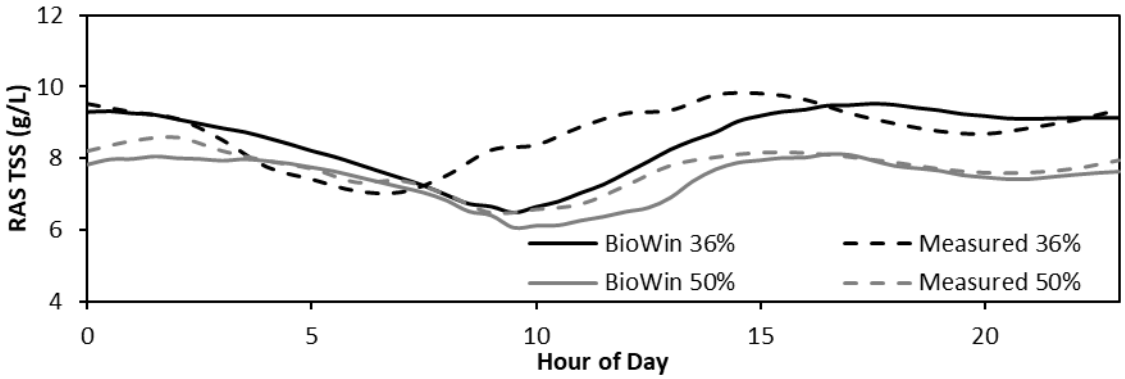


Figure 31. RAS-C TSS concentrations as measured and modeled at bound RAS ratios of 36% and 50%.

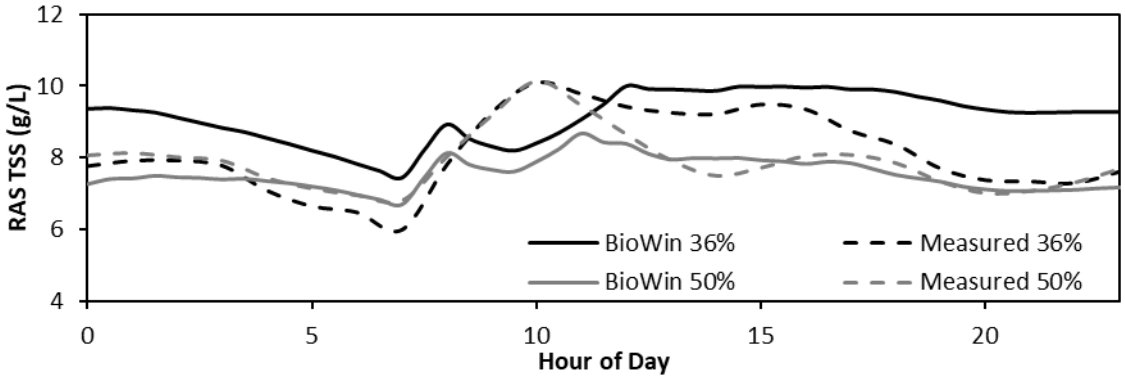


Figure 32. RAS-W TSS concentrations as measured and modeled at bound RAS ratios of 36% and 50%.

The magnitude of the RAS-W nitrate concentrations was in line with those measured, but similar to the RAS TSS concentrations the peaks and troughs were shifted in time (Figure 33 and Figure 34). This should be expected as the RAS nitrate is inextricably tied to the mass of solids in the clarifier. In addition to the timing being off, RAS-W nitrate values were quite different, again potentially owing to bypassing of the solids blanket by clarifier liquid during wasting.

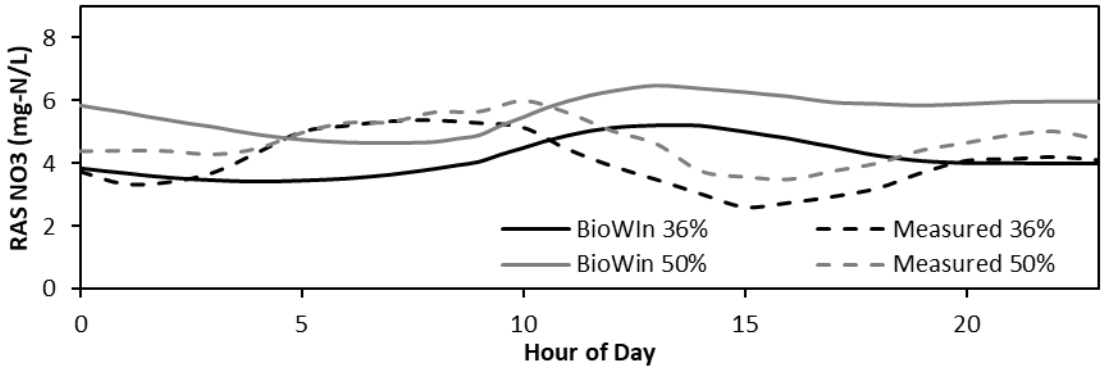


Figure 33. RAS-C $NO_3$ concentrations as measured and modeled at bound RAS ratios of 36% and 50%.

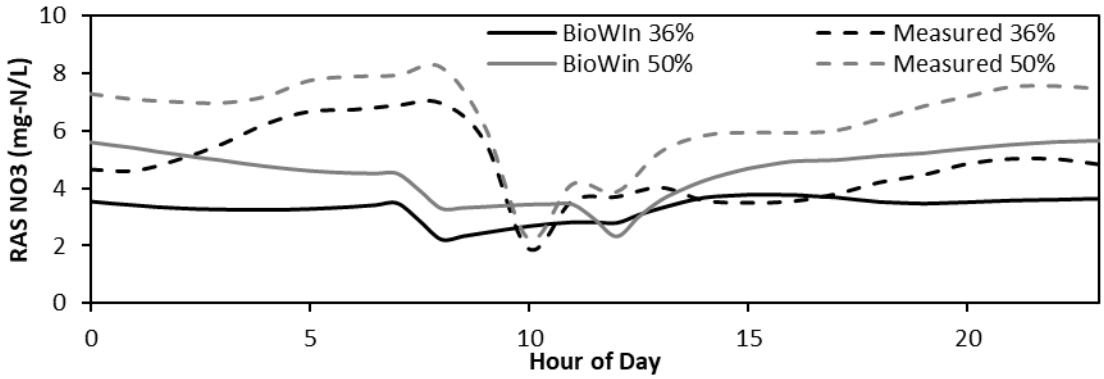


Figure 34. RAS-W $NO_3$ concentrations as measured and modeled at bound RAS ratios of 36% and 50%.

The magnitude of the anaerobic P concentrations did not align well with what was measured, but both the measured and modeled profiles followed a similar trend (Figure 35 and Figure 36). The P concentrations appeared most widely spread in the early morning, followed by reduced concentrations until about noon, then an increase in concentrations through the afternoon and evening. The concentrations appear overall to be decreased when comparing the model results for the 36% and 50% setpoints.

The increased nitrate load in the early morning caused AN1 P concentrations to decrease at 8:00 am to around 11:30 am at which time P release increased as nitrate load reduces. AN2 and AN3 followed shortly behind with only slightly reduced impact. The impact to P concentrations appears to occur later within the model than within Moscow WRRF which would be expected due to the difference in

nitrate load timing. The overall reduction in P concentration appeared to be greater at Moscow WRRF than what was modeled, but so too did the maximum P concentration.

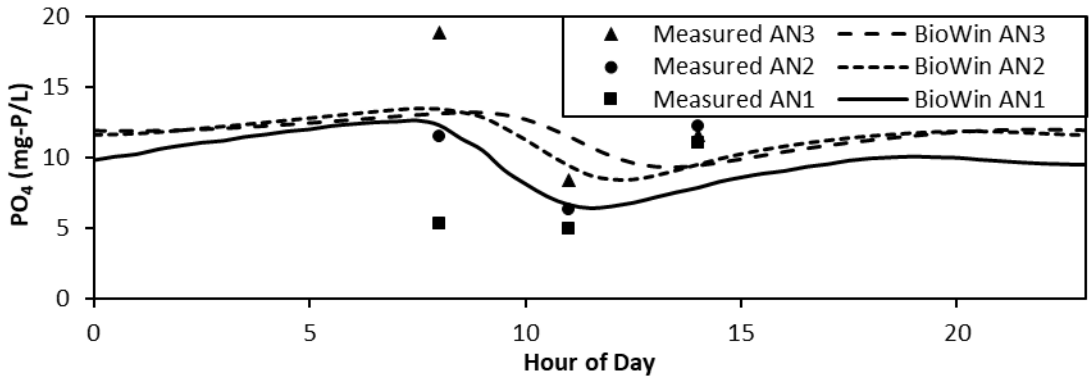


Figure 35. Anaerobic $PO_4$ concentrations (mg-P/L) as measured and modeled at bound 36% RAS ratio.

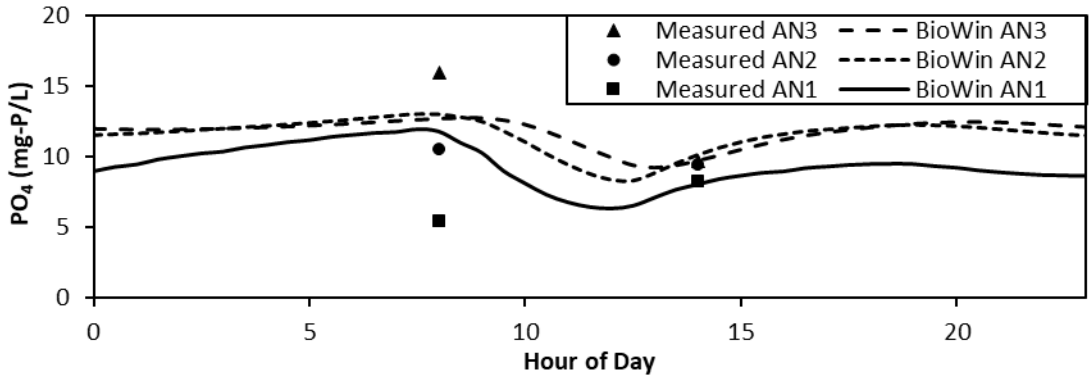


Figure 36. Anaerobic $PO_4$ concentrations (mg-P/L) as measured and modeled at bound 50% RAS ratio.

### *4.7.2 Bound vs. Unbound RAS Flow Conditions*

One of the early interests of this research was the realization that bound RAS flow conditions were causing considerable fluctuations in the RAS ratio. Use of BioWin allows a comparison of bound and unbound RAS flow conditions on system parameters. Using the maximum and minimum flows from the winter study period, 3.05 and 1.08 mgd, respectively, allows bracketing of when the RAS system would be entirely bound and entirely unbound based on the current minimum RAS flow rate, approximately 265 gpm, and assuming no limitations on maximum flow rate. At a RAS ratio setpoint of 25%, virtually all flows would be governed by the minimum RAS flow (bound) and at 70% nearly all flows would be governed by the RAS ratio setpoint (unbound). Variability of solids storage increases with the percent of the RAS flows that are bound (Figure 37). As the setpoint becomes more controlled by the boundary, it approaches a steady flow equal to the boundary which ultimately leads to a larger variance between the maximum and minimum RAS ratio and in turn RAS TSS.

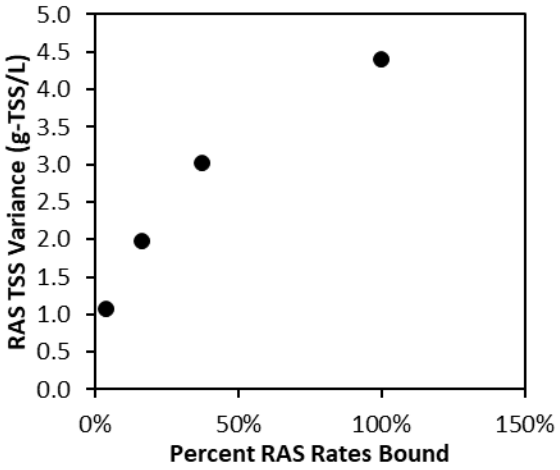


Figure 37. RAS TSS variance (maximum-minimum) vs. percent RAS rates bound.

Variance within RAS TSS does not necessarily correlate with more variance in SC-SRT or nitrate concentration; in fact just the opposite appears to be the case. Comparing Clarifier-C, at low unbound RAS ratios, the SC-SRT varies substantially more than it does for the same RAS ratio at bound conditions (Figure 38). This is due to a balancing act between flow rates and RAS TSS concentrations. During low influent flow periods, bound RAS flow conditions limit the decrease in RAS flow rates and decrease RAS TSS which leads to a relatively steady SRT throughout the day (Figure 39). Unbound RAS flow creates near constant RAS TSS throughout the day and in turn leads to longer SC-SRT during low flow periods. The longer SRT then allows for more denitrification to occur; in turn creating a higher fluctuation in RAS nitrate concentrations throughout the day, but lower overall loading (Figure 40). SC-SRT is calculated as total mass of solids in clarifier divided by the RAS solids removal rate (Schuyler, 2010).

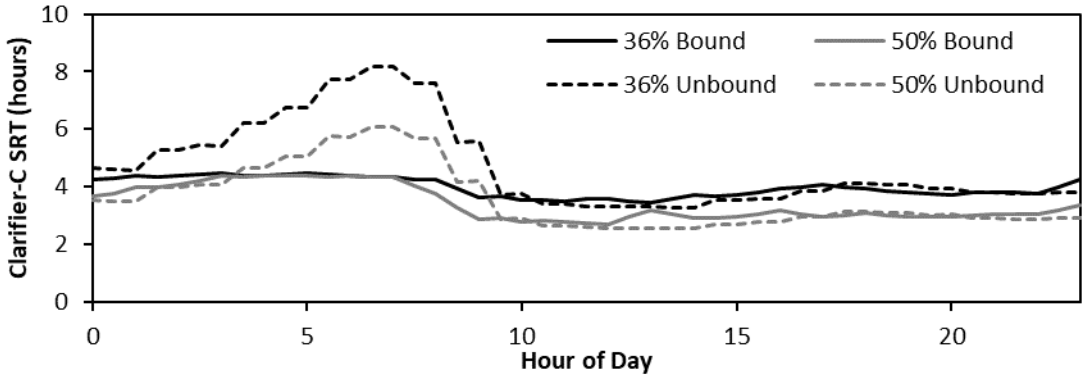


Figure 38. Modeled Clarifier-C SRT throughout the day at bound and unbound RAS ratio setpoints of 36% and 50%.

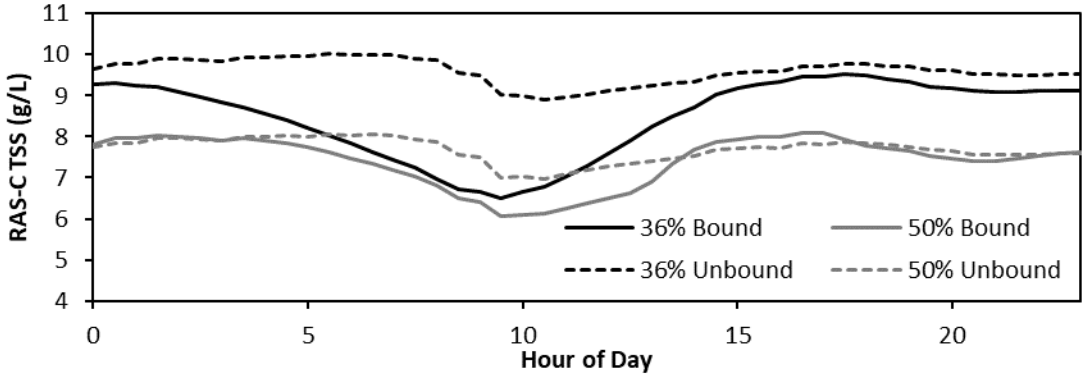


Figure 39. Modeled RAS-C TSS throughout the day at bound and unbound RAS ratio setpoints of 36% and 50%.

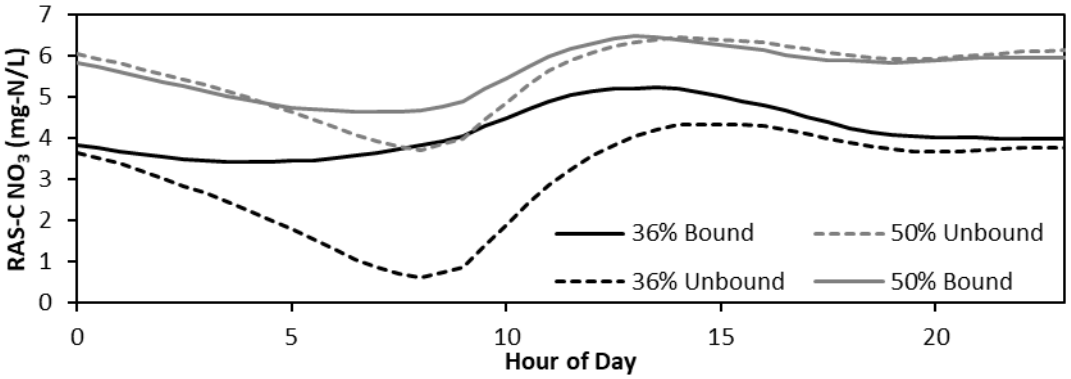


Figure 40. Modeled RAS-C $NO_3$ throughout the day at bound and unbound RAS ratio setpoints of 36% and 50%.

The Moscow WRRF wasting strategy allows for increased secondary clarifier solids storage and reduced RAS ratio, therein mimicking unbound RAS flow within RAS-W. RAS-W for the bound conditions was modeled with the current wasting strategy, shut down for 1 hour then wasted at 250 gpm for three hours, whereas wasting within the unbound models was at a steady rate throughout the day. Shutting down the clarifier and reducing the flow rate allows for solids to accumulate, bringing SC-SRT and RAS-W TSS more in line with the unbound flow conditions (Figure 41 and Figure 42). The similarity in TSS ultimately led to RAS nitrate concentrations that resemble those predicted for unbound RAS (Figure 43). The similarities between the intermittent flow of RAS-W and unbound RAS flow highlight what seems to be of the most importance – that is, the average RAS ratio, rather than a specific setpoint or flow scenario as similar results can be obtained in a myriad of ways.

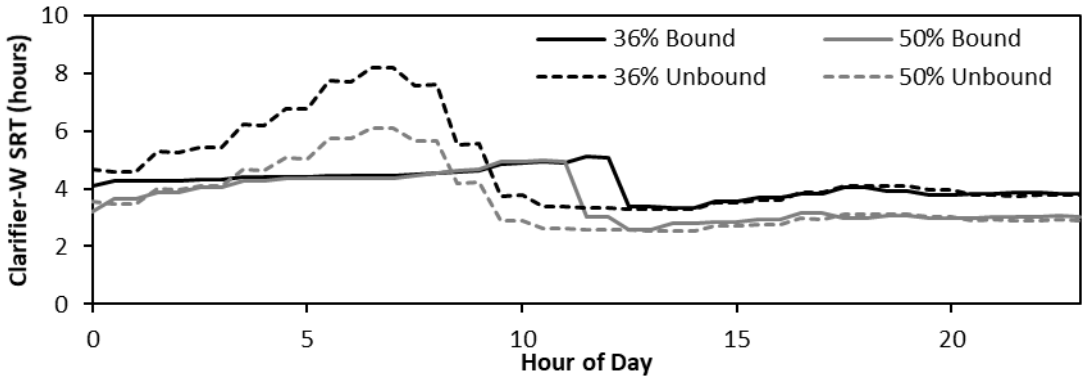


Figure 41. Modeled Clarifier-W SRT throughout the day at bound and unbound RAS ratio setpoints of 36% and 50%.

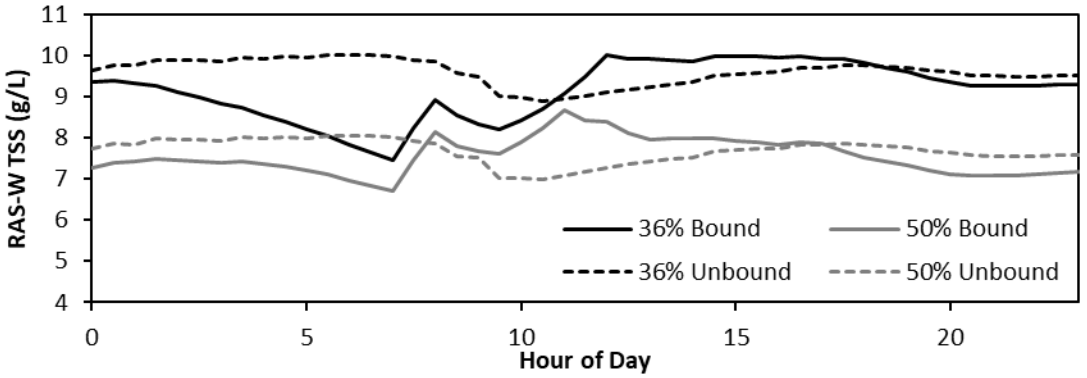


Figure 42. Modeled RAS-W TSS throughout the day at bound and unbound RAS ratio setpoints of 36% and 50%.

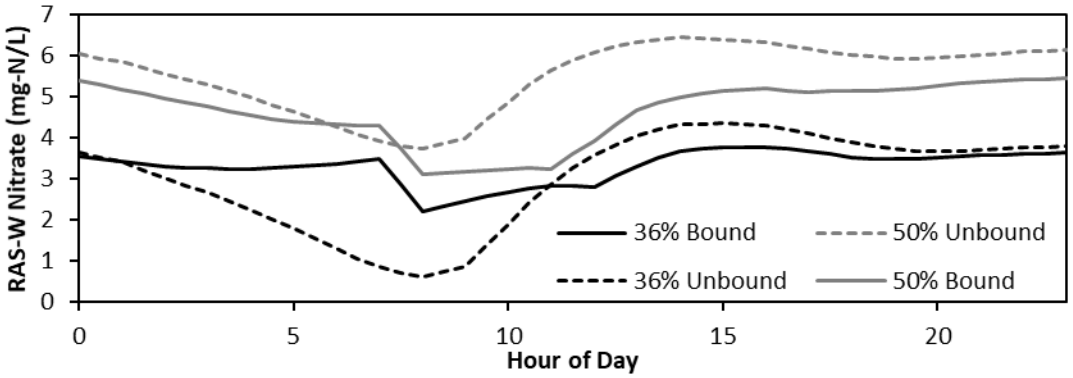


Figure 43. Modeled RAS-W $NO_3$ throughout the day at bound and unbound RAS ratio setpoints of 36% and 50%.

### *4.7.3 Relationships between RAS Ratio and Secondary Clarifier Solids Storage*

The ability to estimate solids stored within the secondary clarifier is key to predicting secondary clarifier denitrification. While this is challenging within a full-scale system, the usage of a model allowed thorough analysis of RAS ratio impact on solids within the secondary clarifier. Only Clarifier-C was considered for the solids storage comparisons as it is more representative of RAS changes as opposed to wasting procedures. As was presented in the bound vs. unbound discussion, the average RAS ratio throughout the day appears to be more important than setpoints or flow conditions. To compare different flow conditions, RAS ratio setpoints were converted to average RAS ratios by totaling the daily RAS flow and dividing by the total daily influent flow. Daily total RAS ratios varied slightly from the RAS setpoints due to losses from wasting as the wasting line is coupled with the RAS line.

The sludge blanket depth was found to correlate well with RAS ratio, $R^2$=0.95, but the changes in depth were minimal – 1.2-0.8 feet for unbound ratios of 25%-200% and bound ratios of 25%-50% (Figure 44). Considering the accuracy of a sludge judge, it would be difficult to notice this small of a change. It appears that total solids stored within the clarifier can largely be predicted based on the average RAS TSS (Figure 45). This is a helpful relationship for operators considering the complexity of solids loading, storage, and removal rates within secondary clarifiers. Additionally, as higher amounts of solids are stored in the clarifier, it is less and less noticeable via sludge blanket measurements (Figure 46). This is reasonable considering the concentration of solids within the sludge blanket increases with time due to compression settling and thus the blanket becomes denser. The total solids within the clarifier is correlated with RAS ratio but not in a linear manner (Figure 47). As the RAS ratio is decreased, substantially more solids become stored within the clarifier. This demonstrates the sensitivity of the secondary clarifier at lower RAS ratios and indicates additional attention should be paid to RAS TSS concentrations when operating at low RAS ratios to avoid excessive storage of solids. Again, the variation between unbound and bound RAS conditions is minimal for comparable average RAS ratios.

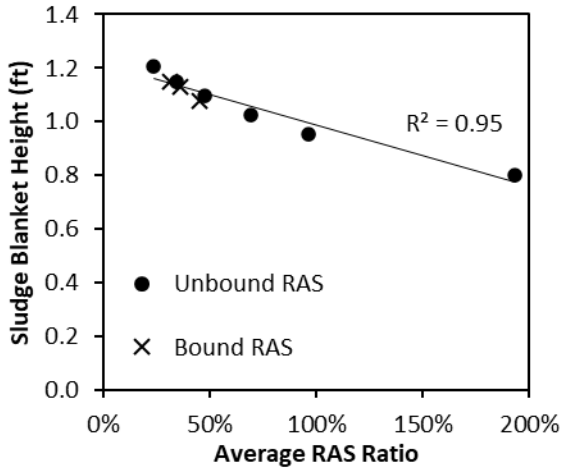


Figure 44. Average sludge blanket height vs. RAS ratio.

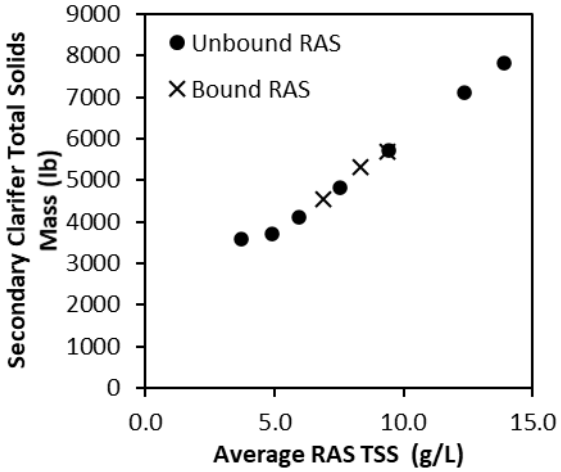


Figure 45. Average total solids mass in clarifier vs. average RAS TSS.

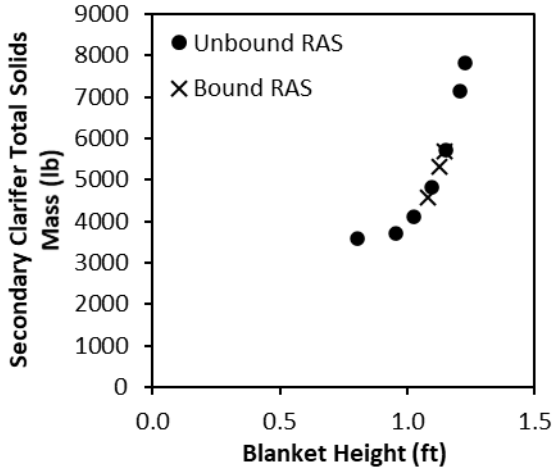


Figure 46. Average total solids mass in clarifier vs. blanket height.

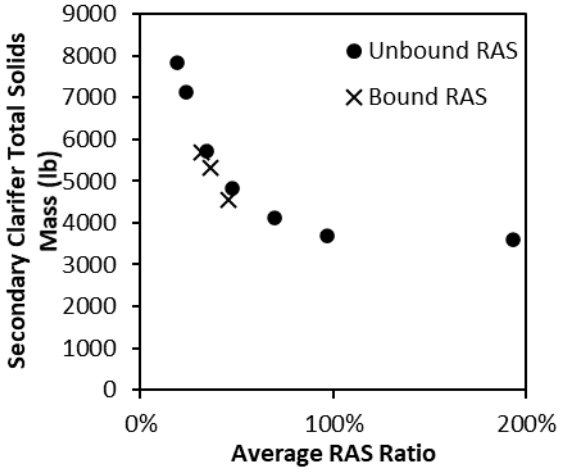


Figure 47. Average total solids mass in clarifier vs. RAS ratio.

### *4.7.4 RAS Ratio, RAS Nitrate Load, and Anaerobic P Release Relationships*

As was preliminarily indicated by the sampling at Moscow WRRF, model results suggest increasing RAS ratios led to increasing nitrate load across a wide range of ratios and differing RAS flow conditions (Figure 48). The relationship between RAS ratio and nitrate load is largely linear, $R^2$=0.94. As was mentioned in the bound vs unbound discussion, the daily average RAS ratio appears to be a more important factor than bound or unbound flow conditions. Modeled nitrate load also aligned well with values from Moscow WRRF. The increasing secondary clarifier solids storage and SC-SRT leads to more denitrification. The SDNR suggests that as more sludge is available and given more time, it can decompose creating additional carbon available for denitrification.

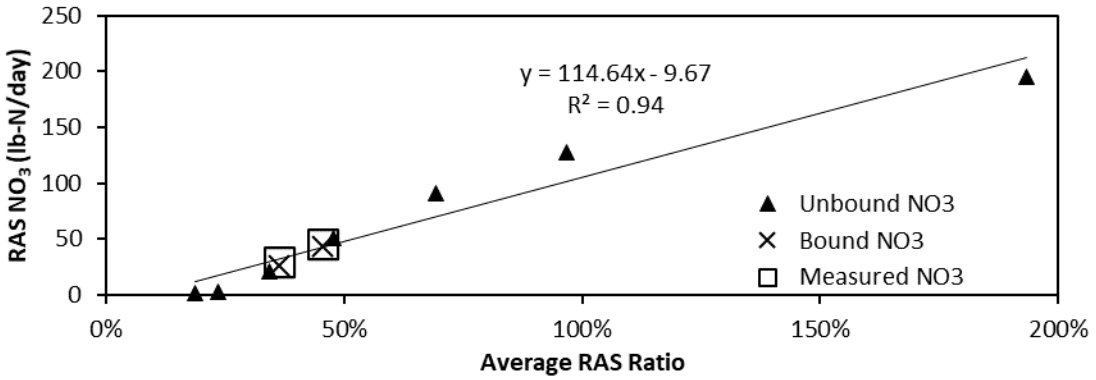


Figure 48. Modeled RAS $NO_3$ for different RAS ratios and conditions.

Sampling at Moscow WRRF and the scale model suggested a significant impact on P release related to RAS nitrate load; BPR theory suggests the same relationship (Grady et al., 2011). However, modeling results present a more complicated view. Modeling allowed generation of continuous concentrations and mass flow in areas where only grab samples were possible at the actual Moscow WRRF. Collection of continuous data allows for anaerobic P release to be analyzed on a mass load basis as well as the daily average concentration (Figure 49). P values within the last anaerobic basin, AN3, were primarily the only ones analyzed as they would typically be the highest within a healthy BPR system. Similar to what was noticed at the Moscow WRRF, P concentrations appear to slightly decrease with increasing RAS nitrate loads. Yet, the mass of P released actually appears to increase until around a RAS nitrate load of 80 lbs/day, above which both the average P concentration and the mass P released begin to fall with increasing nitrate loads. The variability of P mass load should theoretically correspond to nitrate load, but it does not appear to correlate well at low RAS ratios. It is not clear why the model would indicate so and it is suggested that further research be done to understand this phenomenon that seems contradictory to BPR theory. It could be that the denitrifiers are modeled to consume only rbCOD leaving the VFAs to PAOs. This situation may occur until a threshold is exceeded, at which point the denitrifiers start consuming the VFAs. In reality, the denitrifiers likely consume both rbCOD and VFAs simultaneously. To complicate matters further, the bound flow conditions appear to align better with BPR theory with a decrease in average P concentration and mass P release over all nitrate loads. The somewhat contradictory results from the bound and unbound flow conditions on P release may indicate that unbound RAS flow generates a more stable system that can handle additional nitrate loads with little to no impacts on anaerobic basin performance.

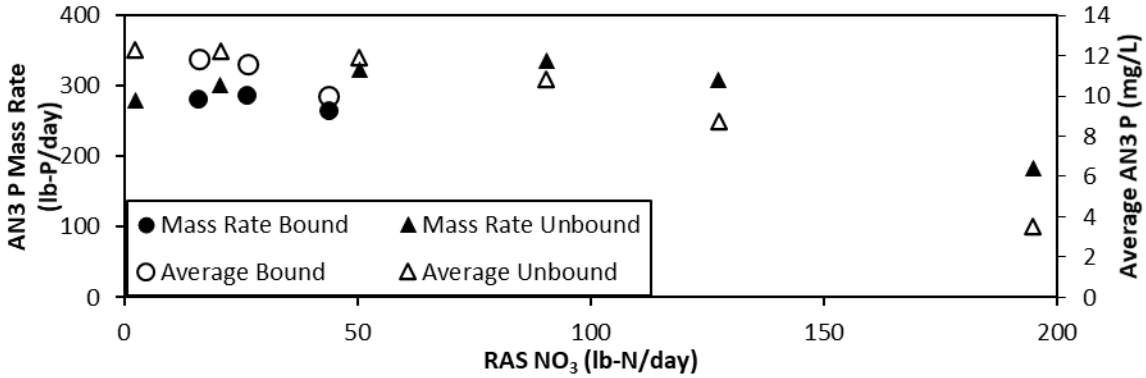


Figure 49. BioWin predictions of AN3 P mass rate and average concentration vs. RAS $NO_3$ load.

The ratio of anaerobic P release reduction to nitrate load above a nitrate load of 90 lbs/day appears largely linear. Fitting a line to the limited set of data points at this range indicates a reduction of P to $NO_3$ of 1.52 lb-P/lb-N, which is consistent with the theoretically estimated values presented in Section 4.5 RAS Nitrate Impacts on BPR. The ratio is also significantly lower than what was measured, which is to be expected as the magnitude of the P release change was considerably smaller in the model than reality (Figure 35 and Figure 36).

### *4.7.5 RAS Ratio Impacts to Secondary Clarifier Effluent*

Anaerobic PAO activity as indicated by P release can be a useful indicator of whether BPR metabolisms are active, however, what really matters is the end goal of attaining a quality effluent that meets permit limits. Some literature has suggested that low RAS ratios can lead to an increase in effluent TP due to secondary P release within the clarifier. However, effluent TP primarily decreased with decreasing RAS ratios and corresponding RAS nitrate loads (Figure 50). At low RAS ratios, these results are somewhat contradictory to what would be anticipated as higher P release would theoretically lead to higher P uptake (Coats et al. 2021). The discrepancy between modeled effluent P and AN3 P release demonstrates either the inability of BioWin to accurately predict effluent P relative to P:C fluctuation or the ineffective nature of P:C as a predictor of effluent P and health of the BPR system for the time frame and scenarios studied. Concerns have also been expressed that low RAS ratios can lead to an increase in TSS due to nitrate gas lifting sludge in the secondary clarifier. Yet, BioWin predicts TSS decreases with lower RAS ratios, potentially due to the increased secondary clarifier solids loading rate at higher RAS ratios. Additional endogenous decay occurring due to the longer SRTs, and additional solids mass stored at lower RAS ratios also does not appear to generate increasing ammonia concentrations in the effluent. The effluent ammonia concentration actually appears to increase at higher RAS ratios, possibly owing to lessening the single pass HRT as proposed by Schuyler (2010).

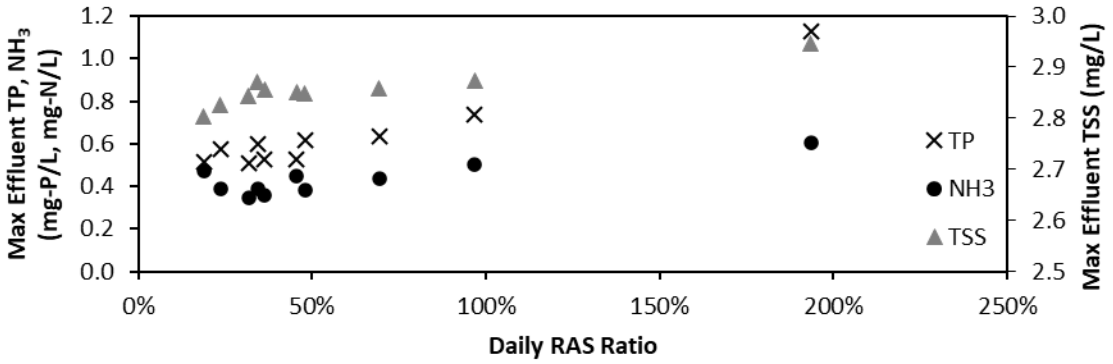


Figure 50. Modeled max effluent TP, $NH_3$ and TSS concentrations vs. average RAS ratio.

### *4.7.6 Bound RAS Optimization*

Ideally a system would be able to run at unbound RAS flow rates that could be optimized to a specific facility's needs, but this may not be the reality for many facilities and achieving such could come with substantial capital costs. For the Moscow WRRF, an unbound RAS ratio of 25% led to a RAS nitrate load of near zero but would require a minimum total RAS flow rate of 179 gpm or approximately 90 gpm per RAS line when running two simultaneously. This reduction would likely require installation of new pumps at the Moscow WRRF to match these minimum flows, which in part defeats the purpose of secondary clarifier denitrification optimization. Another way to reach a similar flow condition would be to intermittently turn off the RAS pumps while alternating between clarifiers so RAS is always being pumped. However, RAS flow would need to cease for a considerable amount of time during the day to substantially reduce the average RAS ratio. For comparison, BioWin was run identically to the bound 36% scenario, but with RAS-C shut down for 3 hours. The reduction in RAS nitrate load from this change is minimal, 26 to 22 lbs/day, as is the change in daily average RAS ratio 36% to 34% (Table 9). To reach an average RAS ratio of 25% under bound flow conditions, the system would need to run at the minimum RAS pump rate, represented by a RAS setpoint of 25%, and the clarifiers would need to be shut down for a combined total of 13 hours during the evening and early morning. Alternating shutdowns between the clarifiers would allow continuous flow of RAS. In this way, both clarifiers are essentially used as one big clarifier during low flow. This cuts the flow by half when RAS ratios are the highest. While this may seem to some a far-fetched idea, it is actually one of the strategies reported in the BPR Operator Survey. One WRRF, which operates two clarifiers, apparently switches the clarifier that RAS is taken from every 30 minutes.

Another option for increasing control over bound RAS flow conditions would simply be to take one clarifier completely offline so that only one RAS line and pump would be used. BioWin models were run using one clarifier at 25% and 36% RAS ratio setpoints to compare this strategy with intermittent

flow. While this seems a comparable method, the model results indicate it generates less favorable conditions. Utilizing a single clarifier leads to roughly half the amount of solids within the clarifier as the two combined, which in turn leads to substantially less secondary clarifier denitrification. The single clarifier model created higher nitrate loads than the bound RAS ratios as well as the intermittently shut down models. Additionally, the single clarifier model led to increased effluent TP, TSS and $NH_3$, when compared with all other modeling scenarios. RAS nitrate load appeared to remain strongly correlated with the average RAS ratio regardless of how it is accomplished.

Table 9. Bound RAS optimization scenarios compared to existing bound and unbound scenarios.

| RAS Condition | Scenario | Setpoint | RAS Average Daily | NO3 Load (lbs/day | Effluent (mg/L) Max TP | Max NH3 | Max TSS |
|---|---|---|---|---|---|---|---|
| Bound | Existing | 25% | 32% | 16 | 0.51 | 0.35 | 2.84 |
| | | 36% | 36% | 26 | 0.53 | 0.36 | 2.86 |
| | | 50% | 46% | 44 | 0.53 | 0.45 | 2.85 |
| | Clarifier-C Shut Down 3:00 am to 6:00 am | 36% | 34% | 22 | 0.57 | 0.35 | 2.86 |
| | Alternating Intermittent Clarifier Shutdown 5:00 pm to 5:00 am | 25% | 24% | 4 | 0.60 | 0.40 | 2.86 |
| | 1 Clarifier | 25% | 23% | 15 | 0.62 | 0.47 | 6.21 |
| | | 36% | 32% | 34 | 0.68 | 0.43 | 6.32 |
| Unbound | - | 25% | 24% | 2 | 0.58 | 0.39 | 2.83 |
| | | 36% | 34% | 21 | 0.60 | 0.39 | 2.87 |
| | | 50% | 48% | 50 | 0.61 | 0.38 | 2.85 |

Shutting down the clarifiers allows for imitation of unbound flow and the modeled $NO_3$ concentrations demonstrate the similarities of these operational approaches (Figure 51). The unbound RAS ratio of 25% led to near complete removal of nitrate from the RAS. To match this, the clarifiers were run in an intermittent, alternating shut down pattern from 5:00 pm to 5:00 am where RAS would be taken from one for an hour then the other for the following hour. For additional comparison, shutting down Clarifier-C under a bound 36% RAS ratio setpoint for three hours at night allowed generation of similar nitrate concentrations as the 36% unbound RAS conditions. To accomplish either of these approaches, electronically controlled valving would need to be installed and integrated into the existing control system which would require capital investment but would be substantially less than a new pumping system.

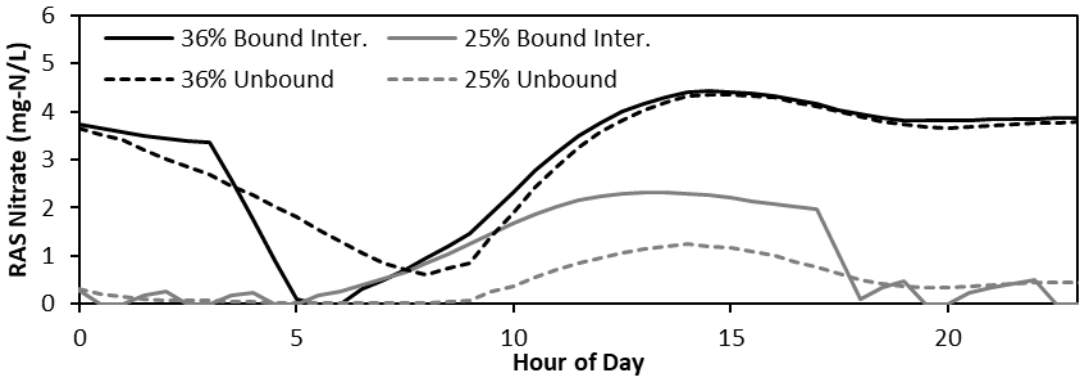


Figure 51. Modeled $NO_3$ for scenarios of unbound RAS and bound RAS intermittently shutdown.

### *4.7.7 Tracer Test*

The transfer of the impact of the nitrate between basins takes time due to the volume of each basin and diffusion through continuous mixing. With varying influent and RAS flow it is challenging to estimate when the impacts of the increased nitrate load would be apparent within each basin. To evaluate the timing of nitrate load impacts on the different anaerobic basins, a tracer test was modeled utilizing a highly concentrated slug of magnesium (Figure 52). The BioWin model was set up to have the same hourly influent and 36% RAS setpoint flows as the Moscow WRRF. From ANISE measurements at the full-scale system, it is apparent the RAS nitrate load began to increase at 3:00 am, with a peak at 6:00 am and a steep decline at 7:00 am when the influent flow increased and RAS-W was shutdown. To mimic the increased load of nitrate, a plug of magnesium (1,000 mg/L) was introduced within the BioWin model at 6:00 am to see when the peak concentrations of magnesium occurred within the anaerobic basins. Magnesium is one of the metals available within BioWin and was used due to its minimal impact on water chemistry and biological metabolisms. The peak concentration occurred at 7:00, 9:00 and 10:30 am within the AN1, AN2, and AN3, respectively. There is potential that the impacts on a metabolic process, such as P release, would take longer to be displayed than that of a metal. However, the tracer results demonstrate that an increased nitrate load could take around 3.5 hours for the impacts to reach AN3. These results were mirrored by the measured P profiles where the lowest concentration occurred in AN3 at 11:00 am approximately 4 hours after the peak nitrate load.

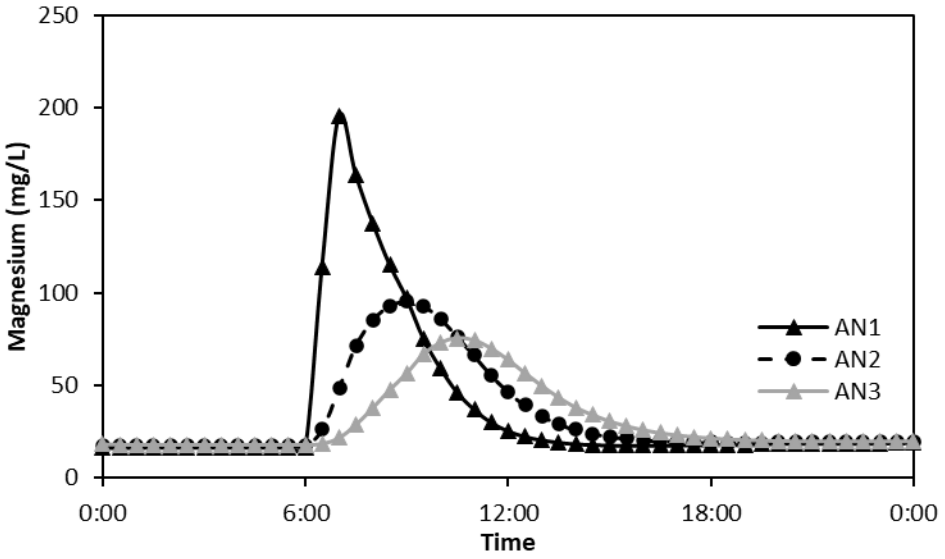


Figure 52. BioWin magnesium anaerobic basin tracer results.

# 5 Summary and Conclusions

Research executed in completing this thesis focused on the impacts of the RAS ratio on secondary clarifier denitrification and BPR. Through extensive sampling at the Moscow WRRF and scale model WRRF, as well as process modeling, it was demonstrated that secondary clarifier denitrification can be enhanced by decreasing the RAS ratio. Enhanced denitrification is largely driven by an increase of solids storage in the secondary clarifier as well as the SC-SRT. Reduction of the RAS ratio in turn reduces the concentration of RAS nitrate as well as the overall flow containing nitrate delivered to the first anaerobic basin in the WRRF. BPR theory and related literature indicate that nitrate deteriorates the function of the anaerobic zone by changing OHO and PAO metabolisms. Indeed, anaerobic P release appeared to be decreased at higher RAS ratios during sampling; BioWin modeling is in agreement at high RAS ratios, but demonstrated the opposite in terms in of overall P load released at low RAS ratios. This is potentially due to the compartmentalization of carbon sources as utilized by PAOs and denitrifiers within the BioWin model. In addition to evaluating overall RAS ratio setpoint adjustments through sampling and process modeling, the daily changes in RAS ratio due to bound RAS flow conditions were also modeled against unbound conditions; the Moscow WRRF currently operates under bound RAS flow conditions, wherein RAS flow rates cannot go below a minimum flow due to mechanical limitations. The bound condition causes considerable fluctuation in RAS ratio throughout the day as the influent flow dips below the minimum RAS rate at times. Sampling at the Moscow WRRF indicated that the RAS nitrate load fluctuation was a result of the bound RAS flow conditions. However, process modeling indicated that though bound flow conditions may cause a slight increase in RAS nitrate load, they actually cause more stable secondary clarifier SRTs and less fluctuation in RAS nitrate. The RAS nitrate concentration within unbound conditions is able to drop lower due to an increase in SC-SRT during low flow conditions. Further, the bound flow conditions in and of themselves do not appear to generate substantially different results than unbound conditions in terms of RAS nitrate load and anaerobic P release. The truly important metric is the overall average RAS ratio which appears to have considerable impacts on RAS nitrate and solids stored within the secondary clarifier. Multiple strategies were presented to allow WRRFs with bound RAS conditions optimize and reduce average RAS ratios and associated RAS nitrate load. It is worth mentioning that while bound RAS flow conditions do not appear as impactful as average RAS ratios, the influence of boundary conditions indicate that the system is nearing the minimum RAS ratio at which it can perform. Therefore, if a system is experiencing RAS flow limitations, it is not likely there is much room to decrease RAS ratios further without initiation of intermittent shutdowns. Moreover, adjustment of SCADA setpoints can be rather meaningless if the reality of mechanical limitations are not considered. For

example, adjusting the RAS ratio at Moscow WRRF from 36% to 25%, assuming the wasting strategy is maintained, is really adjusting the daily average RAS ratio from 36% to 32%. While this seems like a substantial change to the RAS ratio setpoint, the results would likely be negligible.

A review of responses from WRRFs practicing BPR demonstrated the importance of managing nitrate within RAS both through the quantitative analysis of those with a TN permit vs those without and qualitative responses directly from the operators. Yet there appeared to be little guidance on how to manage the nitrate and only a few WRRFs utilize RAS rate control to help manage BPR process stability. Further, contradictory approaches to managing a key aspect of secondary clarifier denitrification – that is, the sludge blanket – were reported. These results demonstrate a lack of common knowledge relative to the utilization of RAS ratios and secondary clarifier denitrification capacity to enhance BPR stability. The respondents that utilized unique approaches to managing RAS rate were some of the most successful and highest regarded facilities that participated in the survey. Additionally, they were all regulated with TN permit limitations. Their approach may demonstrate a coming change to the old method of "set it and forget it" as the benefits of actively managing RAS rate come to light.

The forensic analysis of the Moscow WRRF indicates what decades of effluent data would also tell; the system is operated exceptionally well with average effluent P values that would cause jealousy from other BPR operators. The Moscow WRRF was not chosen to study necessarily to optimize their system, but largely to gain insight on how it is they operate so well and to dive into aspects potentially overlooked. While Moscow has done well developing a RAS ratio that has led to exceptionally stable BPR, the opportunity to optimize the RAS rate is still open as is evidenced by the presence of nitrate at concentrations above 5 mg-N/L and the relatively low overall denitrification occurring within their secondary clarifiers relative to other WRRFs. Indeed, the Moscow WRRF as well as others may benefit from dynamically managing their RAS rate to limit nitrate load returned to the anaerobic zone. Inclusion of an additional operational parameter does not necessarily benefit a facility unless they have the resources and time available to devote to managing it. Considering the consistently low effluent P concentrations Moscow is able to maintain, they may not be the ideal candidate to adopt a thorough and responsive RAS management approach; instead the facilities that noted regular failure within the BPR survey may be better suited.

Even a well-monitored and operated system such as the Moscow WRRF can have operational oddities within their system. Through this research it was highlighted that a setpoint within a SCADA system, such as RAS rate, can in actuality vary significantly depending on the mechanical abilities of the system to match the specified setpoint. The mechanical limitations of a system can be

such that they force operations to swing in and out of the optimal range causing potentially deleterious impacts to other aspects of the facility. The Moscow WRRF is not alone in having a RAS pumping system designed such that it cannot match the lowest pumping rate called for by the RAS ratio setpoint, as it is typical to focus design on maximum and average flows rather than minimum. In part due to the variability of the RAS ratio throughout the day, solids concentrations over a 24-hour period within the RAS varied by 2-3 g/L within the continuously flowing RAS line and by 4-5 g-TSS/L within the RAS line that shuts down for wasting. This variability in RAS TSS has impacts throughout the system, notably so on the denitrification capacity of the secondary clarifier and P release within the anaerobic basin. The variability of RAS solids concentrations and nitrate load throughout the day also demonstrates the impacts daily timing of grab samples within full-scale system without complete equalization can have for gauging the capacity and health of the system. This variation causes considerable changes to the P:C ratio that is often utilized as a metric of health and stability of a BPR system. The lack of discussion of sample timing on facility-to-facility comparison metrics emphasizes a potentially significant, yet exceptionally simple variable that is often not considered.

The changes measured and modeled throughout the BPR system due specifically to fluctuation of the RAS rate/ratio demonstrate the viability of RAS rate as an important process control. SDNRs found within the full-scale secondary clarifier and clarifier batch tests were in line with those published for post-anoxic denitrification. The similarity in SNDR coupled with the clarifier holding a considerable amount of activated sludge signifies the importance of considering the secondary clarifier as a biologically active component of the secondary treatment system. Indeed, the secondary clarifier can offer considerable amounts of anaerobic/anoxic SRT in addition to the capacity afforded in the anaerobic basins with minimal risk to effluent over a wide range of RAS ratios. The additional anaerobic SRT could be a substantial benefit to many WRRFs practicing BPR.

## 5.1 Research Questions Revisited

Research presented and discussed in this thesis specifically interrogates RAS rate impacts on secondary clarifier denitrification and BPR performance. This thesis was driven by the following Research Questions.

**Research Question 1: Do WRRF operators practicing BPR consider RAS rate an important process control parameter?**

BPR Operator Survey results indicated that most operators do not utilize RAS rate as an important process control parameter. Many of the operators simply reported that it is set either at a constant

flow or as a ratio based on the influent flow. However, a few operators mentioned unique approaches to managing the RAS rate to optimize their BPR process. Further interrogation of these unique methods could be a considerable benefit to the BPR community.

**Research Question 2: How does RAS rate impact secondary clarifier sludge denitrification?**

Through sampling at the Moscow full-scale and scale model WRRF, as well as process modeling, it was demonstrated that RAS rate has a direct influence on secondary clarifier sludge denitrification. By lowering the RAS ratio, WRRFs can increase secondary clarifier denitrification and in turn decrease the nitrate load delivered to the anaerobic basin.

**Research Question 3: How does RAS rate and nitrate mass load to the anaerobic basin impact BPR stability?**

Sampling at the Moscow full-scale and scale model WRRFs indicated that RAS rate and nitrate load had a measurable impact on P release within the anaerobic basins. However, process modeling complicated this finding by indicating that P release, in terms of mass load released, actually increased at higher RAS rates and nitrate loads until a threshold was met. After the threshold nitrate load was met, the process model estimated a decrease in anaerobic P release with increasing RAS rate and nitrate mass load. These findings warrant further sampling of WRRFs at a wider range of RAS ratios to determine if the modeled results represent reality or are due to process model limitations.

## 5.2 Future Research

RAS is the cornerstone of activated sludge treatment and deserves its fair share of research attention. The results from this research will hopefully begin to shine light on an otherwise largely neglected process control parameter. The BPR Operator Survey responses indicated many highly regarded WRRFs practicing BPR are beginning to use novel RAS control techniques to enhance their systems. Further discussion with the operators of these WRRFs would undoubtedly lead to valuable insight that could then be analyzed and disseminated to benefit the BPR community as whole. Two particular techniques of interest are actively managing RAS rate to enhance anaerobic P release and intermittent alternating shutdowns to lower RAS ratios for bound RAS rate systems.

To further the specific research presented within this thesis would be further pilot and full-scale analysis of additional RAS ratios to determine if modeled results are indeed accurate. The discrepancy between modeled and sampled anaerobic P release at low RAS ratios may yield interesting insight as well as modeling within a different software program. Specific attention should be paid to effluent concentrations as well as ammonia concentration changes within the secondary clarifier as an indicator of endogenous decay. Enhancement of secondary clarifier denitrification

through RAS ratio appears to be largely driven by solids storage and SRT within the secondary clarifier. Development of metrics for estimating the amount of solids stored and the SRT within the secondary clarifier utilizing easily measured parameters could be of considerable benefit to those looking to utilize RAS ratios control techniques. This approach could be benefitted by performance of tracer study to analyze SRT within the secondar clarifier coupled with various sludge measurement techniques – turbidity, sonar, and RAS TSS. Beyond impacting secondary clarifier denitrification, varying RAS TSS concentrations may carry impacts throughout the system that should be investigated to further the understanding of bound RAS flow conditions on the system as a whole.

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# Appendix A - BPR Operator Survey

*Brent Deyo – deyo0528@vandals.uidaho.edu*

*12/10/2019*

Good morning/afternoon.

My name is Brent Deyo and I am working on my master's degree in Civil Engineering at the University of Idaho. I am conducting research for my thesis focused on enhanced biological phosphorus removal (EBPR) within Dr. Erik R. Coats' research team. One area of my research is focused on EBPR "failure," and seeking to better understand potential causes of EBPR process upset. I believe operators hold valuable insight into EBPR, insight that could potentially benefit other EBPR operators and the field as a whole. Please share your knowledge by completing a short survey, which is available below and as an attachment. The survey should take no more than 15-20 minutes of your time. Results from the survey will be shared with all participants and any significant findings will be submitted to relevant industry journals for publication. All individuals and organizations will remain anonymous unless they grant permission below. Please feel free to contact me with any questions, comments, or concerns.

Thank you in advance for your time.

Sincerely,
Brent Deyo
208-816-0595
deyo0528@vandals.uidaho.edu

**EBPR Survey**

| **Survey Questions** | **Answers** |
|---|---|
| Respondent Name, Title and Email | |
| Name and Location of Facility | |
| Would you like your organization to remain anonymous in the summary and any publications? | |
| Describe your process configuration? (e.g., A2O, UCT, 5 Stage Bardenpho; if unknown please describe the sequence of bio reactors – Anaerobic-Anoxic-Aerobic – and any internal recycle streams) | |
| Does your system utilize any sidestream processes? For example: struvite recovery or annamox | |
| What is your permit $NH_4$ effluent limit? | |
| What is your permit $NO_3$ effluent limit? | |
| What is your permit P effluent limit? | |
| How often does your EBPR process effluent exceed your permitted limit? | |
| Do you employ a backup process to ensure compliance with your permitted effluent P limit? | |
| How long has the EBPR system been operating? | |
| What is your operational SRT? Is there an SRT range you target? | |
| What operational dissolved oxygen concentration do you target? | |

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*12/10/2019*

| **It is well documented that EBPR systems experience occasional upset, and sometimes even extreme failure; often the causes are not well known or understood. Moreover, when EBPR processes recover, the cause for such recovery is not well known or understood. The questions that follow seek to gain a better understanding on your EBPR process, if you have experienced process upsets/failure, and if you determined the cause and remedied the upset/failure.** | |
|---|---|
| How often does your EBPR system experience a process upset or realize significant failure? (in occurrences per year, month or week) | |
| Have you established potential explanations for EBPR upsets/failure? If so, please explain. | |
| Is EBPR upset/failure associated with any known cycles or occurrences? (weekly, seasonally, $NO_3$ in RAS etc.) | |
| What do you consider EBPR upset/failure? | |
| How long does the system take to recover? | |
| Do you follow any specific procedures to recover the EBPR process? If so, what are they? | |
| What operational criteria do you employ to sustain stable EBPR? | |
| Have you established any ideal operational parameters? If so, what are they? | |
| Do you experience any struvite issues? | |
| How do you control your RAS flow rate? (e.g. MLSS concentration, fraction of influent flow, constant flow rate) | |
| Where do you monitor P concentrations within the system? (grab or continuous?) | |
| Would you be willing and able to share plant data related to EBPR? | |

**Physical Plant Data and Influent Characteristics:**

-Please feel free to attach any documents that present this information instead of filling out the table (e.g. permits, plan sheets, facility plan appendices).

-If you have a schematic diagram of your facility, it would be greatly appreciated if you could attach it with your survey response.

-You can add rows by right clicking in a cell or copy a table for asymmetrical trains.

| **Zone** | **Number of Units** | **Units in Operation** | **Volume of each unit (million gallons)** |
|---|---|---|---|
| Primary Clarifier | | | |
| Anaerobic Basin | | | |
| Anoxic Basin | | | |
| Aerobic Basin | | | |
| Secondary Clarifier | | | |

| | | | |
|---|---|---|---|
| Primary Solids Fermenter | | | |
| Anaerobic Digester | | | |
| Equalization Basin | | | |
| Other: | | | |
| | | | |
| | | | |
| | | | |

| **Critical Flow Streams** | **Average Flow Rate (MGD)** | **Avg. MLSS concentration (mg/L)** |
|---|---|---|
| Influent Flow | | ---------- |
| Return Activated Sludge (RAS, Secondary Clarifier to Anaerobic) | | |
| Wasted Activated Sludge (WAS) | | |
| Mixed Liquor Return (MLR, Aerobic to Anoxic) | | |
| Fermenter feed to Anaerobic | | |
| Other: | | |
| | | |
| | | |

| **Wastewater Characteristics** | |
|---|---|
| Average Influent Flow Rate (MGD) | |
| Max and Min Day Influent Flow (MGD) | |
| Influent $BOD_5$ (mg/L) | |
| Influent Total P (mg/L) | |
| Influent $NH_4$ (mg/L) | |
| Influent TKN (mg/L) | |
| Average EBPR Total P Effluent (mg/L) *before any chemical polishing | |

**General Comments**

Are there any improvements you would like to make to the system to enhance/stabilize the EBPR process; in answering this question please ignore financial constraints?

Please use this space to provide any general comments, concerns with your system, curiosities with EBPR or just general ramblings that come to mind.

**Thank you once again for taking the time to complete this survey. Your responses are greatly appreciated.**