Innovative Nutrient Removal Technologies:

CASE STUDIES OF INTENSIFIED OR ENHANCED TREATMENT

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Innovative Nutrient Removal Technologies: Case Studies of Intensified or Enhanced Treatment

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Technology performance and variability in effluent concentrations, particularly for nutrient removal, is affected by site-specific factors such as process design, wet weather flow, variability in influent flow and concentrations, process control capabilities, presence of biological inhibitors or toxics, presence of equalization tanks, sidestreams, and many other factors. In addition, a plant's actual flow and nutrient loading relative to the design capacity could be a significant factor that impacts performance. As such, the information in this report can be viewed as a guide based on the investigated plants' actual full-scale operation over 36 months but should not be used to translate performance or variability to other plants without careful consideration of the plant's site-specific conditions.

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Foreword

The Office of Wastewater Management supports communities' consideration and adoption of innovative and alternative technologies as part of their infrastructure investments for a resilient, clean, and safe water future. This document provides information on the performance and reliability of several innovative nutrient removal technologies available for municipal wastewater treatment facilities. Specifically, the publication shares information on innovative technologies or approaches for achieving nitrogen and/or phosphorus targets in municipal wastewater treatment plant effluents, evaluates performance and reliability in meeting permit limits, and shares the lessons learned in implementing such technologies.

In the last few years, there has been an increased interest in innovative nutrient removal technologies. This interest is driven by many factors including nutrient pollution impacts on water quality, the need to renew aging infrastructure, and the emergence of new and highly sustainable treatment approaches and practices. These innovations offer significant advantages in terms of treatment performance and resource management efficiency. We at EPA have seen many water resource recovery facilities (WRRFs) lead the way towards a more sustainable and climate resilient future through the adoption of innovative and alternative technologies and solutions.

As communities evaluate infrastructure investment options, there is an opportunity to integrate resource recovery solutions in areas such as nutrient removal and recovery, water reuse, energy recovery, and carbon management; to deliver triple bottom line benefits (i.e., economic, social, environmental) to WRRFs and their communities. Innovative technologies introduced over the last few years have the potential to significantly transform and intensify treatment approaches to nutrient removal. Innovative processes or approaches, when properly designed and operated, can achieve reliable nutrient removal at a lower carbon and economic footprint, and often with a smaller physical footprint as well. They can help wastewater treatment facilities reduce their energy demands, costs, chemical usage, or solids production while reliably meeting discharge permit limits. An additional significant benefit of some of these innovative technologies is that they can also result in a net gain of treatment capacity.

Introduction and wider adoption of new wastewater treatment technologies can be challenging in the North American marketplace and require collaborative efforts by all industry stakeholders in addressing barriers to wider deployment. These barriers can be technical, regulatory, or economic. The risks involved are often borne by innovators and early technology adopters. Measures that can reduce the risk involved in new technology applications include increased transparency of information, provision of independent technology evaluations, and development of new mechanisms for sharing risk more broadly. In order to facilitate the consideration and adoption of such technologies, the Office of Wastewater Management works to provide objective resources on innovative and alternative technologies to the public in response to emerging needs and trends in the sector.

This report includes detailed assessments of several innovative nutrient removal technologies. The assessments are based on actual operational data over a three-year period and under specific operating conditions. It is our hope that this report will be useful to utilities and regulators in informing decision-making related to innovative technology capability and choices as well as in informing the implementation of water quality standards and discharge permits.

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This study was completed by the EPA Office of Water, Office of Wastewater Management. Participation of managers and staff from the six case study facilities who provided data and participated in discussions with the study authors, and their review and concurrence with their case study write-ups, was valuable and much appreciated. In addition to providing operational data, the facility managers and in some cases their consultants provided information on plant process treatment trains and diagrams, operational procedures, design criteria, and operational difficulties encountered. Following completion of the case study performance and statistical analyses, draft case studies were developed and provided to facility managers for review, comments, and concurrence. Final drafts were then reviewed by external experts in nutrient removal prior to finalizing and publishing this report.

Contributions of facility managers and staff and external peer reviewers of this study are recognized below:

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Acronyms and Abbreviations

A^2O	Anaerobic/anoxic/aerobic
ABAC	Ammonia-based aeration control
AWRRF	AlexRenew (Alexandria Renew Enterprises) Advanced Water Resource Recovery Facility
Anammox	Anaerobic ammonium oxidation
AOB	Ammonia oxidizing bacteria
BAC	Biological activated carbon
BNR	Biological nutrient removal
BOD	Biochemical oxygen demand
BPR	Biological phosphorus removal
BRB	Biological reactor basin
cBOD	Carbonaceous biochemical oxygen demand
$cBOD_5$	Five-day carbonaceous biochemical oxygen demand
COD	Chemical oxygen demand
CoV	Coefficient of variation
CPT	Centrate pre-treatment
CV	Coefficient of variation
C/P	Carbon to phosphorus ratio
DO	Dissolved oxygen
EBPR	Enhanced biological phosphorus removal
EPA	United States Environmental Protection Agency
EPRI	Electric Power Research Institute
FE	Final effluent
FWHWRC	F. Wayne Hill Water Resource Center
GAO	Glycogen accumulating organism
GMF	Granular Media Filtration
HDPE	High density polyethylene
HWWTP	Hillsborough Wastewater Treatment Plant
HRT	Hydraulic retention time
IDNR	Iowa Department of Natural Resources
IFAS	Integrated Fixed Film Activated Sludge
IX	Ion exchange
KWTF	Kingsley Wastewater Treatment Facility
MBBR	Moving bed biofilm reactor
MGD	Million gallons per day
ML	Mixed Liquor
MLE	Modified Ludzack-Ettinger
N	Nitrogen
ND	Not detected
NdN	Nitrification and denitrification
NH_3-N	Ammonia species (as nitrogen)
NH_4-N	Ammonium (as nitrogen)
NO_2-N	Nitrite (as nitrogen)

NO_3-N	Nitrate (as nitrogen)
NOB	Nitrite oxidizing bacteria
NO_x-N	Nitrate plus nitrite as nitrogen
NPDES	National Pollutant Discharge Elimination System
NRCY	Internal nitrified recycle
OHO	Ordinary heterotrophic organism
ON	Organic nitrogen (total)
ORP	Oxidation-reduction potential
Ortho-P	Orthophosphate
P	Phosphorus
PAO	Polyphosphate accumulating organism
qPCR	Quantitative polymerase chain reaction
PHA	Poly-β-hydroxyalkanoate
PHB	Poly-β-hydroxybutyrate
PO_4-P	Phosphate (as phosphorus)
PS	Primary sludge
RAS	Return activated sludge
rbCOD	Readily biodegradable chemical oxygen demand
RO	Reverse osmosis
SAGR®	Submerged Attached Growth Reactor
S2EBPR	Sidestream enhanced biological phosphorus removal
SBR	Sequencing batch reactor
SCADA	Supervisory control and data acquisition
SDWRF	South Durham Water Reclamation Facility
SRT	Solids retention time
TKN	Total Kjeldahl nitrogen
TN	Total nitrogen
TP	Total phosphorus
TPSs	Technology performance statistics
TSS	Total suspended solids
UV	Ultraviolet
VFA	Volatile fatty acid
WAS	Waste activated sludge
WERF	Water Environment Research Foundation
WRWTP	Westside Regional Wastewater Treatment Plant
WRF	The Water Research Foundation
WRRF	Water Resource Recovery Facility

CHAPTER 1.0
Project Background and Approach

1.1 Introduction

The U.S. Environmental Protection Agency (EPA) completed a project that developed six detailed case studies of recent innovations in municipal nutrient removal treatment. The project included five facilities in the U.S. and one in Canada. The facilities implemented innovative technologies or process enhancements designed to significantly intensify treatment or enhance the removal of total nitrogen (TN) or total phosphorus (TP), with one of the facilities evaluated for ammonia nitrogen treatment only. Treatment intensification has been defined as any system that significantly outperforms conventional designs, and performance could be defined using effluent quality, energy consumption, or capital expenditures (Sturm, 2016). Technologies that result in treatment intensification have also been described as those that provide reduction in treatment tank volume or footprint compared to those with establisheddesigns.

The focus of the analysis in this project centered on assessing the performance of the selected processes over a three-year period, including assessing their impact on mainstream treatment performance and statistical variability of plant effluent nutrient concentrations. Each case study presents a detailed technical description of the innovative process, an analysis of process performance, an assessment of the process train consistency in meeting permit limits, and the lessons learned by the facility in implementing the process and addressing operational difficulties.

Wastewater treatment facilities are subject to real conditions that impact performance and variability in effluent concentrations. These conditions include seasonal challenges such as lower temperatures, wet weather high flow events, changes in influent characteristics, unavoidable imperfections that are present in every design or operation, mechanical problems, and impacts of toxic discharges into the sewer collection system that impact plant process performance, among others. As such, providing detailed assessments of innovative nutrient removal technologies over a three-year period based on sufficient actual operational data can be quite useful to utilities and regulators in informing decision-making related to innovative technology capability and choices, as well as in informing the implementation of water quality standards and development of discharge permits.

1.2 The Need for Innovation

The water and wastewater industry is facing significant challenges in its ability to maintain safe and sustainable water resources. These challenges include decreased availability and quality of water resources, population growth, emerging contaminants, aging infrastructure, and impacts of climate change related to precipitation, temperature, and flooding. Other challenges include changing workforce dynamics and the need to enhance workforce retention, recruitment, and development. In addition, challenges related to costs associated with meeting water quality objectives, coupled with declining water consumption and associated decline in water revenue at some facilities, are resulting in significant economic challenges for utilities.

In addition, the industry has been transforming from one of known challenges that can be addressed by well-established wastewater engineering solutions, to one with a variety of uncertainties such as in how to project future water use, climate change impacts, and the technical capacity necessary for decision making. These uncertainties will likely require a suite of strategies to mitigate and master the most probable and consequential trends and associated risks. Utility responses to uncertainties, risks, costs, and innovative opportunities will help shape public perceptions of water utilities and their leaders, and in turn, shape the state of the industry. In the face of uncertainty, research and information sharing are critical to industry adaptation to uncertainties and, ultimately, success (Hughes et al., 2013).

The water and wastewater industry is also realizing that utility sustainability is dependent on the ability to explore, evaluate, and implement innovative technologies and practices (Ries and Murthy, 2014). Water and wastewater utilities have historically been more conservative in accepting new ideas and technologies, in part due to traditional procurement practices intended to manage risk and organizational structures resistant to new business practices. Over the last ten years, however, many utilities have begun to take ownership of solution development to key challenges. These utilities are implementing innovation programs focused on accelerating the development of innovative technologies.

Third party process assessments of innovative technologies, such as the assessments in this document, can be of value to many utilities exploring innovative technologies. The assessments provide potential technology adopters information that may not otherwise be obtained with their own resources and that can be useful in better assessing potential risks and benefits in adopting these technologies.

Optimizing existing technologies and introducing new ones can both be effective in mitigating the costs of wastewater treatment for the wastewater industry; however, the introduction of new technologies faces significant obstacles in the North American marketplace. The risks involved with these innovation introductions mean that many municipalities will not participate in initial technology adoption, waiting instead for others to be first. Recommended measures that will reduce the risk involved in new technology applications have been identified, such as increased transparency of information, provision of independent evaluations of technologies, and mechanisms for sharing risk more broadly (Parker, 2011).

1.3 Conventional and Innovative Nutrient Removal

Nutrient control has been required at some municipal treatment plants for many years. In the last few years, there has been an increased interest amongst industry stakeholders in innovative nutrient removal technologies. This interest is driven by a number of factors. These include the need to renew aging infrastructure originally constructed in response to the 1972 Clean Water Act, the emergence of new and highly sustainable treatment approaches and practices, a paradigm shift in the industry's view that wastewater is a resource and not a waste, and increasingly stringent effluent nutrient standards implemented across the U.S. to mitigate eutrophication by managing nitrogen and/or phosphorus. Many States have adopted or are now planning to adopt nutrient criteria into their water quality standards and are considering lowering nutrient limits in renewing discharge permits. Many more plants may soon be required to construct new nutrient control facilities or upgrade their existing facilities to consistently meet lower effluent limits. This has increased interest in information on innovative cost- effective technologies or approaches for achieving lower total nitrogen and/or total phosphorus levels in municipal wastewater treatment plant effluents as well as in a need to evaluate novel technologies or operating strategies and define their performance and reliability in meeting permit limits.

Conventional nitrogen removal practiced over the last 40 years involves the microorganism-mediated autotrophic oxidation of ammonia to nitrite then to nitrate under aerobic conditions in a two-step process called nitrification. This process is followed by heterotrophic reduction of nitrate to nitrite then to nitrogen gas in the presence of an organic carbon source and the absence of dissolved oxygen (DO) in a process called denitrification. Nitrification is an aerobic process requiring energy-intensive and costly aeration and additional aeration volume to maintain required DO levels; it may require the addition of chemicals such as caustic or sodium bicarbonate to maintain desired alkalinity levels. Denitrification requires an adequate amount of carbon, often beyond what is available in the incoming wastewater in order to achieve low total nitrogen limits. Both processes result in sludge production which requires subsequent processing prior to beneficial use or disposal.

Conventional phosphorus removal is achieved by chemical phosphorus precipitation and/or by biological phosphorus removal. Chemical phosphorus precipitation requires the addition of chemicals in the form of metal salts. This process results in significant additional sludge production particularly when very low phosphorus limits need to be met. Also, metal salts react with and consume natural alkalinity in the wastewater. Biological phosphorus removal (BPR) relies on consumption of readily biodegradable chemical oxygen demand (rbCOD) and anaerobic release of phosphate followed by aerobic phosphate uptake by polyphosphate accumulating organisms (PAOs). Plants using BPR typically include dedicated anaerobic zones followed by aerobic zones with DO generally greater than or equal to 1.0 mg/l.

Over the last decade, a number of highly sustainable processes used for nutrient removal or approaches for enhancing existing nutrient removal processes have been introduced to the market, but the rate of introduction and adoption by facilities has been generally slow. In some cases, these innovative processes or approaches can achieve reliable nutrient removal at a lower carbon and economic footprint, often with a smaller physical footprint, thereby helping wastewater treatment facilities in reducing their energy demands, costs, chemical usage, and solids production while reliably meeting their permit limits. In some cases, technologies (such as deammonification) have been implemented under specific favorable sidestream conditions but may not yet be ready for deployment under mainstream full plant flow and varying wastewater influent characteristics and conditions. In other cases, innovative technologies or approaches may not be integral to the mainstream process (e.g., sidestream phosphorus stripping, granular sludge applications) but have a significant positive impact on nutrient removal efficacy and required resources such as aeration energy or carbon addition. An additional significant benefit of some of these innovative technologies is that they can, in some cases, also result in a net gain of treatment capacity.

It should be noted that many technical publications can be found in the literature drawing conclusions or making claims about the capabilities of specific technologies in reaching low nutrient concentrations. While many of these publications are accurate and useful, claims of technology performance should be viewed with a degree of caution unless supported by plant design and operational information, along with statistical analysis of data from longer-term operating periods. Presentation of performance data without stating its statistical characteristics could be misleading to utilities making infrastructure decisions to comply with nutrient permit limits and prevents the comprehensive comparison of data among various studies (Bott et al., 2011).

1.4 Project Background

This project continues previous EPA efforts to share information on municipal wastewater treatment technologies in the area of nutrient removal. The scope and approach of this project was influenced by two previous studies that presented information on the performance of nutrient removal processes. The first was a two-volume EPA report titled "Municipal Nutrient Removal Technologies Reference Document" published in 2008. The report was developed to provide information to assist local decision-makers and regional and state regulators in planning cost-effective nutrient removal projects for municipal wastewater treatment facilities (Kang et al., 2008). The report included performance data and a statistical evaluation of 40 treatment alternatives in service at the time and 30 full-scale treatment facilities achieving various levels of nutrient removal. The statistical analysis evaluated the variability of effluent concentrations based on mostly one-year data sets and included various percentiles (i.e., 50^{th}, 92^{nd}, 98^{th}, and 99.7^{th} percentiles) of nutrient concentration data sets. The report also included information on capital as well as operations and maintenance costs associated with the various technologies and facilities.

The second project was conducted by the Water Environment Research Foundation (WERF, now the Water Research Foundation) through its nutrients research challenge program and in cooperation with the Water Environment Federation (WEF) and resulted in a publication entitled "Nutrient Management Volume II: Removal Technology Performance & Reliability". The project was influenced by the EPA study mentioned above and included a comprehensive evaluation of multiple nutrient removal plants designed and operated to meet very low effluent TN and TP concentrations, several as low as 3.0 mg/l TN and 0.1 mg/l TP. The study focused on determining the TN and TP effluent concentrations achieved by the processes investigated. The investigation also focused on the ability of nitrification technologies to meet low maximum daily limits for ammonia. Three years of operational data from 22 exemplary plants were analyzed using a consistent statistical approach that considered both process reliability and the permit limits applied. Technology Performance Statistics (TPS) were defined as three separate values representing the ideal, median, and reliably achievable performance. Also, monthly average 95^{th} percentiles of effluent data were used to compare the plants in terms of their ability to achieve the 3.0 mg/l TN or 0.1 mg/l TP criteria. Maximum day statistics were used to stratify the ability of plants to meet low maximum day permit levels. The project focused on maximizing what can be learned from existing technologies to provide a database that will inform key decision makers about proper choices for both technologies and rational bases for statistical permit writing (Parker et al., 2011).

This EPA study focuses on innovative nutrient removal treatment processes or process enhancements designed to significantly intensify treatment or enhance the removal of TN or TP from municipal wastewaters. The study presents detailed technical descriptions of the innovative processes, an analysis of process performance, an assessment of the process train reliability in meeting permit limits, and the lessons learned by the facility in implementing the process and addressing operational difficulties encountered. The study also presents the benefits of each innovative technology compared to applicable conventional technologies.

1.5 Participating Facilities and Plant Data

At the start of the project, EPA developed an initial list of 18 candidate facilities. These facilities had relatively recently implemented innovative nutrient removal technologies or process enhancements that intensified treatment or significantly improved existing process performance. EPA reviewed published information on the facility projects to determine if the innovations have been in operation at full scale and for how long, and to get an initial understanding of the benefits resulting from their deployment. EPA selected a short list of six of these facilities and contacted facility managers to assess their interest in participating in this project and inquire about data availability. The main considerations used in selecting the six facilities were the expected technology benefits, availability of a minimum of three years of operating data, the challenges addressed by the facility, and lessons learned in implementing the technology that could be useful to this report's intended audience. All six facilities expressed their interest in participating in the project and providing facility data. Table 1-1 provides a list of the facilities and the technologies implemented.

Table 1-1. Participating Facilities and Technologies Implemented

Facility	Innovative Process or Enhancement
AlexRenew Advanced Water Resource Recovery Facility, City of Alexandria, Virginia	DEMON® Sidestream Deammonification
Westside Regional Wastewater Treatment Plant, District of West Kelowna, British Columbia, Canada	Sidestream Enhanced Biological Phosphorus Removal (S2EBPR)
F. Wayne Hill Water Resource Center, City of Buford, Georgia	WASSTRIP® Stripping and Ostara Pearl® Phosphorus Recovery
City of Kingsley Wastewater Treatment Facility, City of Kingsley, Iowa	Submerged Attached Growth Reactor (SAGR®) Nitrification
South Durham Water Reclamation Facility, City of Durham, North Carolina	ANITAMOX® Sidestream Deammonification
Hillsborough Wastewater Treatment Plant, Town of Hillsborough, North Carolina	Low Nitrogen BNR Enhancement Modification

For the six selected case study facilities, 36 months of final effluent data and innovative process operational data provided by the participating facilities were analyzed and subjected to a detailed statistical analysis. For plants analyzed for nitrogen removal, nitrogen species concentrations evaluated included TN, ammonia nitrogen (NH_3-N), nitrate plus nitrite nitrogen (NO_x-N), and organic nitrogen (ON). ON as referred to in this report means total organic nitrogen. In some cases, data for some species (e.g., ON) was not available. The nitrogen species (i.e., NH_3, NO_2, NO_3, ON) referred to in this report indicate the chemical forms using the molecular weight of only the nitrogen atoms (e.g., NH_3-N) and not the molecular weight of the entire ammonia molecule (in the case of ammonia, 1 nitrogen atom and 3 hydrogen atoms). For plants analyzed for phosphorus removal, TP and orthophosphate (OP) were considered.

To ensure consistency in data representation and usability in statistical analyses, EPA reviewed all data received from the case study facilities prior to analysis. Effluent values for each constituent reported as non-detectable, zero, or as the minimum detection limit were replaced with half of the minimum detection limit. EPA used all data provided by the case study facilities in the analyses with no exclusions.

1.6 Operational Performance

This study assessed the performance of the innovative process as well as the performance and variability of plant effluent concentrations of the nutrient treated. The study also assessed each plant’s ability to consistently meet its nutrient limit for the nutrient investigated.

Depending on operational data and process control information provided by each plant, an attempt was made to identify the factors that impacted the performance of the innovative process investigated. In addition to reviewing available process control data, this included correlating nutrient effluent concentrations with other operational parameters as well comparing actual influent loading and characteristics to design criteria. For the sidestream processes, the study also investigated the impact of the sidestream process on the mainstream nutrient removal process, including the impact on mainstream process stability.

It should be noted that technology performance and variability in effluent concentrations, particularly for nutrient removal, is affected by site-specific factors such as process design, wet weather flow, variability in influent flow and concentrations, process control capabilities, presence of biological inhibitors or toxics, presence of equalization tanks, sidestreams, and many other factors. In addition, a plant’s actual flow and nutrient loading relative to the design capacity could be a significant factor that impacts performance as it can reflect the degree of stress placed on the plant, particularly relative to biological treatment processes when looking to achieve very low nutrient limits. As such, the information in this report can be viewed as a guide based on the investigated plants’ actual full-scale operation over 36 months but should not be used to translate performance or variability to other plants without careful consideration of the plant’s site-specific conditions.

1.7 Statistical Analysis

1.7.1 Summary Statistics and Probability Plots

The use of probability values that relate to permit compliance is very useful in evaluating process performance under an appropriate set of averaging conditions. A statistical analysis was conducted on the complete sets of 3-year final effluent data from each facility. In some cases, the effluent concentrations from the innovative process were statistically assessed as well. Summary statistics were calculated for the full final effluent data set. These included the mean, geometric mean, standard deviation, coefficient of variance (CoV), skew, minimum, and maximum. Time series data plots were prepared showing the individual data points (e.g., daily concentrations) with the plots also generally showing the facility's discharge limits and median values.

To better assess performance and variability of effluent concentrations and the ability of each facility to meet permit limits, a set of percentile statistics was calculated from the data sets including the Technology Performance Statistics (TPSs) evaluated by Neethling et al. (2009) to represent the lowest (3.84th percentile), median (50th percentile), and reliably achievable technology performance (95th percentile). The calculated set of percentile statistics for this project includes the 3.84th, 50th, 90th, 95th, and 99th percentiles. A percentile represents the probability that a data value is less than or equal to the stated concentration. As an example, a 95th percentile effluent concentration of 4 mg/l TN calculated from a data set indicates that 95 percent of the data points are below 4 mg/l and as such, the probability of meeting a 4 mg/l effluent concentration is 95 percent.

Log-transformed probability plots were then developed using the approach presented by Bott et al. (2011) by ranking the effluent concentration data in Microsoft Excel then calculating the corresponding Weibull probability (P) values by dividing P by (n+1), with n being the number of data points in the set. The concentration values were plotted versus the probability (less than or equal to) using SigmaPlot 14.0 (Systat Software, Inc.) with the y-axis converted to a log scale to reflect the lognormal transformation, and the x-axis plotted using the normal distribution probability scale. Other statistical calculations were done using Microsoft Excel.

The TPSs evaluated were used in checking performance in relation to desired effluent limits and their specific averaging periods. The 14-day TPS (3.84th percentile) representing an ideal performance sustained over only 14 days is useful in assessing best possible short-term performance. It is minimally influenced by various factors affecting facility and operational performance variability including influent variability, seasonal variations and loading conditions, the full impact of process control corrections, equipment failures, and occasional industrial discharges. As such, it does not reflect reliable sustained performance. The TPS-50 percent (median) is useful in looking at the performance on an annual basis but does not provide a statistical assessment of performance on a monthly basis. The TPS-95 percent (95th percentile) is often used as a measure of reliable maximum month performance but the appropriate reliable performance percentile should be selected depending on the facility risk of tolerance given existing plant infrastructure, equipment redundancy, and other facility factors. Other statistics such as a 30-day rolling average could be used in assessing performance consistency in meeting a monthly or an annual average under conditions experienced over a longer time period such as 3 years.

In using the probability plots, the appropriate percentile probability to be used in assessing reliable performance should be selected taking into account a number of considerations such as the technology itself, the averaging period used in a plant's National Pollutant Discharge Elimination System (NPDES) permit, and risk tolerance. Additionally, the use of specific percentiles in assessing performance should be considered with an understanding of the associated implications in terms of the resulting frequency of permit violations. For example, the 95th percentile for a daily average limit is exceeded 91 times (5 percent x 365 days x 5 years) in a 5-year NPDES permit, while the 95th percentile for a monthly average limit is exceeded 3 times (5 percent x 12 months x 5 years) in a 5-year NPDES permit.

1.7.2 Data Set Statistical Manipulations

For each facility, a series of data manipulations were conducted based on daily values, 30-day rolling average values, monthly average values, and 12-month rolling average (rolling annual average) values. In most cases, weekly values were also analyzed. A rolling average is a moving average calculated as explained below. For each of these value categories, the manipulations included the summary statistics, percentile calculations (probabilities), and probability plots described above as well as time series plots showing effluent parameter concentrations versus the data category (e.g., daily, monthly, etc.).

The calculation of rolling averages was done by taking the mean of the data points in the initial averaging period to calculate the initial rolling average and then shifting forward by one data point to calculate the next rolling average. For example, the first 30-day rolling average was calculated by averaging the first 30 consecutive daily data points (such as effluent concentrations). The next 30-day average will exclude the previous average's first day data point and add the data point for the next day following the initial 30-day period. Each 30-day average was plotted on the 30th day. As such, the rolling average plotted date represents the plotted date and the previous 29 days. With three years of data generally provided for each plant and one leap year period in 2016, 1,096 daily data points will be subject to the above manipulations if complete daily data is provided.

It should be noted that some plants did not collect daily samples. So, for example, a plant that only collects samples three times a week would generally have 12 data points included in a 30-day rolling average. When 30-day rolling averages were calculated, the averages span gaps in the data. As a result, the rolling average represents a true rolling average of daily data within a 30-day period and not a 30 data point rolling average.

1.7.3 Variability

Variations in effluent quality from nutrient removal processes are the result of various internal and external factors. These could include variations in influent characteristics, environmental conditions such as temperature, presence of toxics or inhibitors in the influent, and process operational parameters and other factors inherent to the treatment process. In many cases, causes of effluent variability may not be explained by analyzing available data. However, determining variability in effluent concentrations is an important consideration in designing nutrient removal facilities as well as in the development of discharge permit limits. Variability measures allow designers assessing technology options for a particular facility to include appropriate levels of conservatism in their design in light of permit effluent concentrations and averaging periods. Effluent variability measures also allow the development of appropriate effluent limits taking into consideration effluent variability as well as receiving water flows and constituent concentrations variability.

Several statistical parameters have been used for assessing variability of effluent concentrations including the variance, standard deviation, CoV, and TPSs. TPSs are used to assess variability in this report, as they provide a precise and practical measure of the capability of a nutrient removal treatment to meet an effluent limit in specific numerical terms particularly at low concentrations where variability is expected to be higher. The 95th percentile, for example, can be useful in evaluating the ability to meet monthly permit limits. It is also useful in informing process design to ensure permit compliance.

The ratio between the 3.84th, 50th, and 95th percentiles can also be used to represent the variability of performance. For example, the ratio of the 95th to 50th percentile can be used to assess the ability of a technology to meet monthly limits compared to annual values. Comparing a particular ratio for different technologies provides a measure of the stability of a particular process compared to another.

1.7.4 Reliability

The reliability of a treatment plant or a treatment process may be defined as the probability of adequate performance for a specified period of time under specific conditions, or, in terms of treatment plant performance, the percent of time that effluent concentrations meet specified permit requirements. Because of the variations in effluent quality, treatment plants must be designed to produce an average effluent concentration below the permit requirements. The question is: what value should be used for process design to be assured that constituent concentrations in the effluent will be equal to or less than a specified limit with a specified degree of reliability? (Tchobanoglous et al., 2003)

Two approaches have been used to estimate the design value needed to meet prescribed standards. One approach developed by Niku et al. (1979, 1981) involves the use of a coefficient of reliability (COR) in relating design values to the standards that must be achieved on a probability basis. Another graphical probability approach for setting the required effluent value at a specified reliability level was described by Tchobanoglous et al. (2003). As nutrient removal as well as many wastewater treatment and other environmental processes are often log-normally distributed, the use of a plotted ideal log-normal distribution line can be practical and may eliminate the need to conduct multiple calculations to determine reliability under different conditions of effluent variability and permit averaging periods as long as log-normal data distribution is valid.

In addition to the individual data points on the case study probability plots in this report, a colored line representing the ideal log-normal distribution was drawn as done by Bott et al. (2011) based on log-normally transforming the data set, determining the expected probabilities by computing the log-normal Z (=LOGNORMDIST(x, mean of ln(x), std. dev. of ln(x) in Excel) using the log-transformed mean and standard deviation. The expected normal probability associated with the Z values (=NORMSDIST(x) function in Excel) were then calculated. The log-normally transformed data were then plotted as a red line versus the expected log-normal probability in each of the probability plots. The reliability in meeting a given effluent concentration can be determined simply by following the red line and reading the percentile on the x-axis as long as the log-normal distribution is valid. While the plotted red lines are not used in describing process performance in this report, we included them in this report for use by practitioners who may be interested in assessing the reliability associated with any given concentration using the ideal log-normal distribution red line in cases where the data is well fitted to the log-normal distribution.

1.8 Report, Data, and Analysis General Limitations

Facility and operational data used in this study were provided by the participating facilities. While considerable effort was conducted by EPA in evaluating the data and discussing it with facility managers, EPA does not assume responsibility for the quality of the data or any issues or circumstances associated with the collection or analysis of the data.

As noted in Section 1.7.2, some plants did not collect daily samples. Also, some facilities did not have a complete data set of influent flow or influent and effluent nutrients species concentrations. In other cases, mainstream biological nutrient removal (BNR) process data or sidestream process operational data was incomplete. Where applicable, this was noted in the case studies and for the most part, did not significantly impact the performance evaluations.

It is important to recognize that infrastructure conditions, operational procedures, and site-specific conditions under which the data for this project were collected significantly impact treatment performance and the technology performance statistics and related analyses. These conditions include internal factors such as process design, process control capabilities, presence of equalization tanks and onsite solids processing including anaerobic digesters, sidestreams, and construction. They also include external conditions such as wet weather flow, ambient temperature, and industrial discharges. As such, while the performance analysis in this study provides a clear picture of the achieved treatment associated with the technologies evaluated at these facilities, the results cannot be directly translated and should be used with significant judgement in relating them to performance at other facilities, taking into consideration site-specific conditions and the factors mentioned above.

The mention of trade names, vendors, or products in this report does not represent an actual or presumed endorsement, preference, or acceptance by EPA or the Federal Government. Stated results, conclusions, usage, or practices do not necessarily represent EPA views or policies.

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CHAPTER 2.0
AlexRenew Advanced Resource Recovery Facility – Alexandria, Virginia Mainstream Biological Nitrogen Removal and Sidestream Deammonification Process (DEMON®) for Centrate Nitrogen Removal – Case Study

2.1 Background

The Alexandria Renew Enterprises (AlexRenew) Advanced Water Resource Recovery Facility (AWRRF) is a 54 million gallons per day (MGD) wastewater treatment facility located in Alexandria, Virginia. The facility currently serves more than 300,000 people in the City of Alexandria and adjacent portions of Fairfax County. The plant discharges into Hunting Creek, a tributary of the Potomac River and subsequently the Chesapeake Bay.

To meet an annual average Total Nitrogen (TN) concentration goal of 8 mg/l, the facility was upgraded in 2002 to include replacement of its rotating biological contactors with a suspended growth activated sludge system and an upgraded tertiary treatment process to remove total suspended solids (TSS) and Total Phosphorus (TP) using inclined-plate settling tanks and deep-bed sand filters.

In 2005 Virginia's Department of Natural Resources enacted new regulations to reduce nutrient levels in the Chesapeake Bay, which included setting nutrient load caps on wastewater treatment plant nutrient discharges. The facility was required to meet a maximum TN waste load allocation of 493,381 lb/yr and a TP average monthly concentration limit of 0.18 mg/l and average weekly limit of 0.27 mg/l. The facility initiated a nitrogen removal enhancement program in 2009, which included addition of anoxic volume to the existing biological reactors, expansion of the methanol addition system, and construction of a centrate pretreatment facility and a nutrient load management facility. The certificate-to-operate, following completion of all the construction activities, was obtained in August of 2016. Starting January 2017, the annual average TN permit limit was lowered to 3.0 mg/l, with the same load cap.

Since May 2015 the facility has been operating a centrate pre-treatment system using the DEMON® sidestream deammonification process to remove nitrogen from the centrate that results from dewatering anaerobically digested sludge. Prior to the upgrade, this stream comprised as much as 20 percent of the total nitrogen load entering the biological reactors.

Cost comparisons with mainstream treatment by other investigators revealed that sidestream treatment using the deammonification pathway can be as much as three times lower in cost per pound of nitrogen removed ($0.93/lb of nitrogen removed for deammonification compared with $2.66/lb of nitrogen removed for mainstream treatment) when capital and operating costs were considered (Bilyk et al., 2017). However, actual savings will vary depending on the process used, existing facility infrastructure, influent characteristics, process control efficiency, and other factors.

2.2 Plant Processes

Liquid treatment processes at the AWRRF are shown in Figure 2-1. Preliminary treatment includes three-inch coarse screening, ¼ inch fine screening, and vortex grit removal. Primary treatment occurs in eight settling tanks. Primary effluent flows into the biological treatment process consisting of six biological reactors and six secondary settling tanks. Primary sludge from the bottom of the primary settling tanks is pumped to gravity sludge thickeners.

The primary effluent can also be diverted to a nutrient load management facility, which is essentially a load equalization facility used to balance the diurnal ammonia-nitrogen loading to the biological reactors during periods of above-average ammonia nitrogen loading. The diverted flow is held in storage and then pumped back to the biological reactors during below-average loading periods. In this manner, the ammonia-nitrogen loading to the biological reactors is balanced and spikes are avoided.

The biological reactors also receive flow from the centrate pre-treatment facility. This facility is designed to treat ammonia-rich dewatering centrate and remove most of the associated ammonia load prior to re-introduction of the recycle stream to the main plant flow ahead of the biological system. The biological reactor system can be operated in parallel (Modified Ludzack-Ettinger (MLE) mode) or in series (step-feed mode). While the system was designed with the step feed BNR configuration, it has relied on the addition of methanol to the last anoxic zone when and as needed to achieve the new effluent requirements.

Ferric chloride is added in the secondary settling tanks for phosphorus removal. Solids from the bottom of the settling tanks are returned to the biological reactor basins as return activated sludge (RAS), and a portion of the solids is diverted to the solids handling system as waste activated sludge (WAS).

Effluent from the secondary settling tanks flows to a tertiary settling process consisting of eight tanks. Each tank is sub-divided into a rapid mix tank where a coagulant (normally alum or alternatively, ferric chloride) is added to the water and thoroughly mixed, a gentle mix flocculation tank, and an inclined plate settling tank for floc gravity settling to further remove suspended solids and phosphorus. This is followed by effluent settling in a filtration system of twenty-two sand gravity filters. Final treatment includes a UV disinfection system in six parallel channels containing low-pressure low-intensity UV lamps. Effluent then enters a post-aeration system to increase the dissolved oxygen (DO) concentration prior to discharge to Hunting Creek.

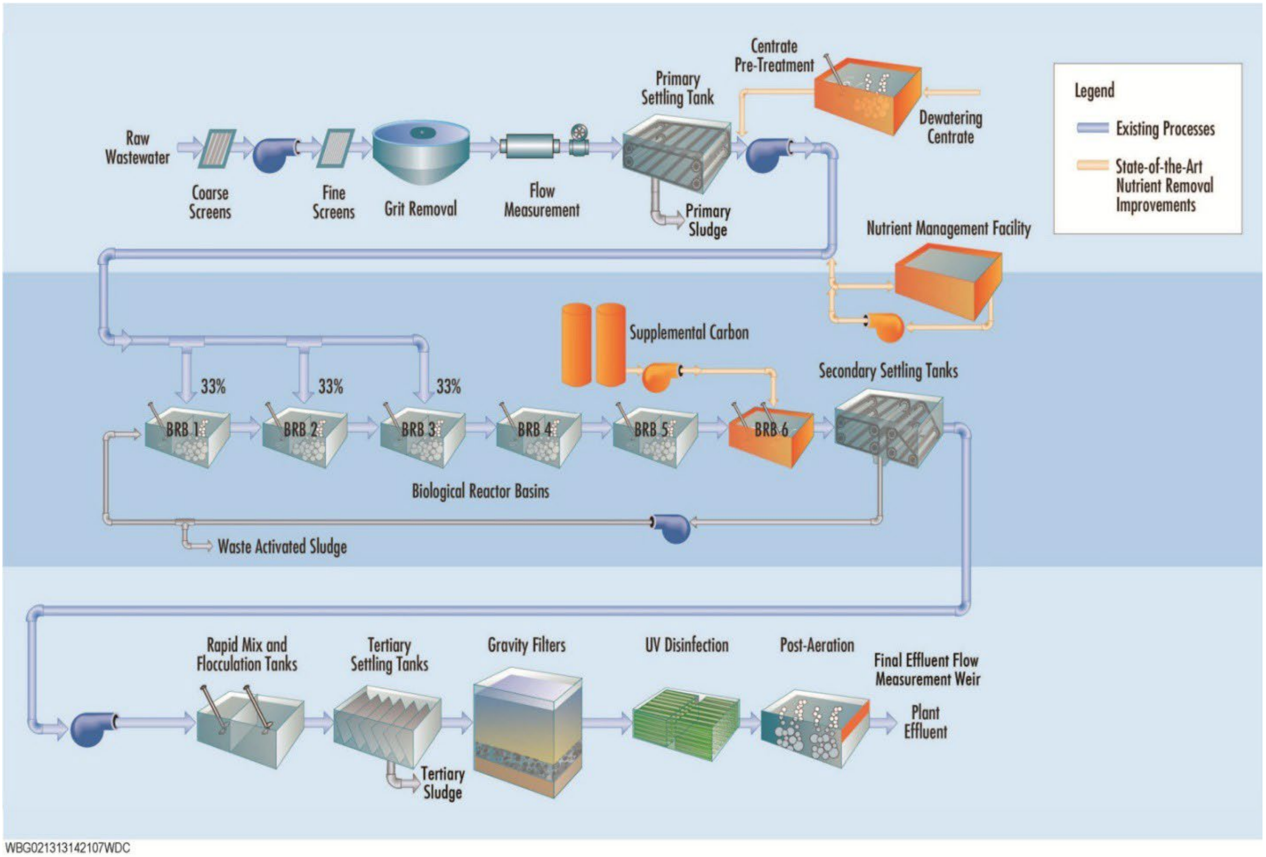


Figure 2-1. Liquid Treatment Process Train

Solids treatment processes at AWRRF are shown in Figure 2-2. The solids handling system includes gravity thickening of primary and tertiary sludge followed by pumping the thickened sludge to the thickened sludge equalization tanks. Waste activated sludge is stored in raw sludge blending tanks and pumped to four thickening centrifuges. Thickened sludge is transferred to sludge equalization tanks where it is blended with the gravity-thickened sludge and pumped to a pre-pasteurization process where the sludge is screened and then pumped through heat exchangers to be heated to a temperature of 70 °C for at least 30 minutes. The sludge is then cooled and sent to four mesophilic anaerobic digesters. After digestion, the sludge is pumped to equalization tanks. Digester gas is returned to the digesters for mixing and excess gas is utilized for operation of the steam boilers or burned in the waste gas flares.

Three dewatering centrifuge trains are used to dewater digested sludge to a dewatered sludge cake of approximately 27 percent total solids content. Polymer is added to the sludge to aid the liquid/solid separation process. The centrate is stored in dewatering centrate tanks and pumped to the centrate pre-treatment facility.

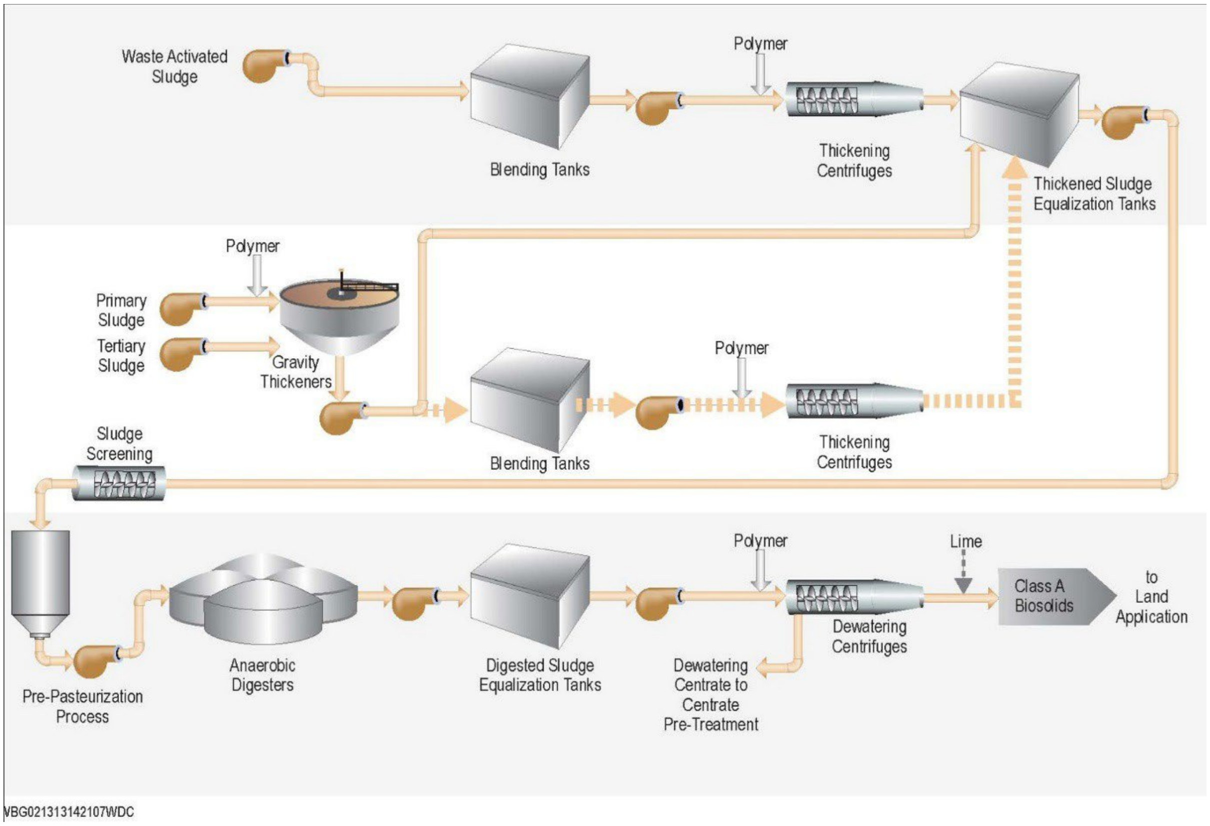


Figure 2-2. Solids Treatment Process Train

Table 2-1. Design and Average Raw Influent Concentrations and Loads, and Percent of Design for the AWRRF from January 2015 to December 2017

Parameter	Raw Influent Design		Average Raw Influent	Percent of Design[1]
	(Annual Average)	(Maximum Month)		
Flow (MGD)	54	70	32.9	61
$cBOD_5$ (lbs/d)	84,600	110,000	72,438	85.6
TSS (lbs/d)	110,000	154,000	85,060	77.3
Ammonia (lbs/d)	N/A	N/A	7,134	N/A
TKN (lbs/d)	15,800	19,000	12,896	81.6
TP (lbs/d)	2,600	3,640	1,756	67.5
Temperature (ºC)	20	14	21.1	N/A

Note:
1. Percent of design values, except for flow, are based on average annual values for the analysis period for influent design loads (lbs./day) and actual influent loads (lbs./day).
N/A: Data not available or applicable.

Table 2-2. NPDES Limits - January 2015 – December 2017 at AWRRF

Parameter	Monthly Average (mg/L)	Monthly Average (kg/day)	Weekly Average (mg/L)	Weekly Average (kg/day)	Annual Load (lb/yr)	Annual Average (mg/L)
					Jan 2015 – Dec 2016	**Jan – Dec 2017**
$cBOD_5$	5	1,000	8	1,600	N/A	N/A
TSS	6	1,200	9	1,800	N/A	N/A
Ammonia (Apr.- Oct.)	1	200	4.4	900	N/A	N/A
Ammonia (Nov.- Jan.)	8.4	N/A	10.0	N/A	N/A	N/A
Ammonia (Feb. - Mar.)	6.9	N/A	8.5	N/A	N/A	N/A
TN	N/A	N/A	N/A	N/A	493,381	N/A
TN	N/A	N/A	N/A	N/A	N/A	3.0
Temperature	N/A	N/A	N/A	N/A	N/A	N/A

Note:
1. N/A: Data not available or applicable.

2.3 Conventional Nitrogen Removal Technology

Nitrogen removal in most wastewater treatment facilities (also known as water resource recovery facilities – WRRFs) is achieved biologically through conventional biological nitrogen removal using nitrification and denitrification processes, with nitrification consuming as much as half the power required for aeration based on typical wastewater carbon to nitrogen ratios (COD/TKN). Nitrification and denitrification occur in two-steps in which autotrophic and heterotrophic bacteria sequentially convert ammonia to nitrogen gas. The first step, nitrification, is aerobic whereby ammonium (NH_4^+) is oxidized to nitrite (NO_2^-) by ammonia oxidizing bacteria (AOBs), and nitrite is converted to nitrate by nitrite oxidizing bacteria (NOBs).

The second step is anoxic whereby nitrate (NO_3^-) is converted to nitrite and then to nitrogen gas by ordinary heterotrophic bacteria (OHOs). In the first step, ammonia conversion consumes oxygen and alkalinity. In the second, no oxygen is consumed, and alkalinity is produced. Additionally, since this reaction is by heterotrophic bacteria, sufficient carbon is needed in the plant influent COD to achieve a minimum BOD:TKN ratio of about four to five or a minimum COD:TKN ratio between nine and ten. Where plant influent carbon is not adequate, supplemental degradable soluble COD needs to be added, typically at a COD:NOx-N removed ratio in the range of approximately 3.5 – 8, with the ratio using methanol addition being at about 3.5.

Energy consumption by public water and wastewater services is about 0.5 percent of total U.S. primary energy and 2 percent of its end-use electricity (Twomey and Webber, 2011, EPRI, 2013). Energy consumption by wastewater treatment facilities and drinking water systems can amount to up to one third of a municipality's total energy bill (EPA, 2009). Typically, at an activatedsludge wastewater treatment facility, 40-70 percent of the energy used is for aeration (WEF, 1997). Plants using biological nitrogen removal are at the higher end of this range.

In plants that use anaerobic digestion for sludge stabilization and reduction, volatile solids are destroyed resulting in the release of significant amounts of ammonia nitrogen, typically amounting to between 15- 20 percent of the plant nitrogen load in many cases, but may be as high as 40-50 percent in some cases depending on the type of sludge stabilization used and the associated degree of volatile solids destruction achieved as well as other factors such as whether other solids are co-digested and whether primary treatment is used. In plants where sludge is dewatered intermittently, this sidestream ammonia load may significantly affect the stability of mainstream biological nutrient removal processes and cause diurnal spikes in effluent ammonia or total nitrogen levels. As such, effective approaches to treating this sidestream load has become an increasingly important treatment objective due to potential cost savings and the positive impact on the mainstream nutrient removal process.

2.4 Deammonification

Several processes have been used over the last decade to treat high ammonia sidestreams generally relying on using biomass for treatment in varying configurations, process control approaches, and control of process parameters such as hydraulic retention times (HRT) and ammonia concentration. However, the discovery in the 1990s (Mulder et al., 1995) of a group of microorganisms known as the anammox (anaerobic ammonium oxidation) bacteria that can convert ammonia and nitrite directly to nitrogen gas has significantly enhanced the attractiveness of sidestream nutrient removal processes due to aeration energy savings, reduced external carbon demand, and reduced sludge production. This is possible as the anammox are anaerobic, autotrophic and the reaction has low biomass yield and only produces small amounts of nitrate.

The deammonification process involves partial nitritation (conversion of ammonia to nitrite) and anaerobic ammonia oxidation. It requires a 50 percent mix of ammonia and nitrite for the anammox bacteria to oxidize ammonia under anoxic conditions using nitrite. The process has been established and used successfully for sidestream treatment at a number of wastewater facilities in the U.S. and overseas. Deammonification is an ideal process for dewatering sidestreams because centrate or filtrate resulting from dewatering of anaerobically digested sludge is warm in temperature and high in ammonia concentration (around 1,000 mg/l or above in most cases) which inhibits nitrite oxidizing bacteria (NOBs) that compete for nitrite for aerobic nitrite oxidation (nitratation). Centrate also has low carbon content which inhibits heterotrophic bacteria from outcompeting the anammox for the available nitrite. Research and testing for stable mainstream deammonification continue but the process has not been used at full scale in the U.S. yet, mainly since full-scale repression of NOBs in mainstream processes is difficult to consistently achieve. Current deammonification systems for sidestream treatment include sequencing batch reactor (SBR) processes, an up-flow granular bed process, a moving bed biofilm reactor (MBBR) process, and a hybrid suspended and attached growth process. The process known as Integrated Fixed Film Activated Sludge (IFAS) has also been tested for sidestream treatment.

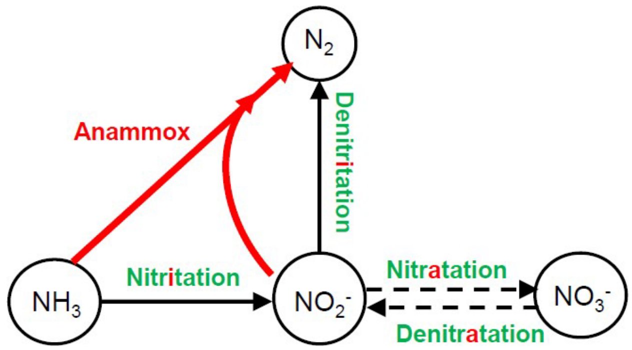


Figure 2-3. Nitrogen Transformations (WERF, 2014)

Deammonification can result in significant savings compared to conventional nitrification-denitrification biological nitrogen removal. This can theoretically amount to 60 percent or more reduction in oxygen demand and associated aeration energy and near complete elimination of costly supplemental carbon addition. As mentioned in section 2.1 above, cost comparisons with mainstream treatment revealed that sidestream treatment using the deammonification pathway could be as much as three times lower in cost per pound of nitrogen removed compared with mainstream conventional biological nitrogen removal. However, actual savings will vary depending on the process used, existing facility infrastructure, influent characteristics, process control efficiency, and other factors. At AWRRF, based on Table 2-1 and assuming about 35 percent BOD removal with primary treatment, the BOD:TKN ratio is less than 4.0 and thus supplemental carbon would be needed. Since the cost per pound of nitrogen removed in the mainstream is related to adding carbon for denitrification in addition to more energy for aeration for full nitrification, sidestream deammonification processes are particularly advantageous when there is a carbon limitation as the anammox bacteria do not use carbon.

2.5 DEMON® Sidestream Deammonification Process at AWRRF

Centrate from dewatering anaerobically digested sludge at AWRRF is stored in centrate tanks and pumped to the centrate pre-treatment facility. This facility uses the DEMON® sidestream deammonification system which is based on a SBR process with the use of hydrocylones for sludge wasting with return of anammox granules to maintain the anammox bacteria in the reactor. The SBR system operates on a fill/react/settle/decant cycle. The process cycles pumps, mixers, and blowers on and off multiple times per hour and includes instrumentation to measure DO, pH, temperature, nitrate nitrogen, and ammonia nitrogen. Decanters remove a portion (approximately 5 – 10 percent at AWRRF) of the settled reactor contents at the end of each batch to free up reactor volume for the next cycle. The clarified supernatant is discharged into the primary settling effluent channel where it blends with the main plant flow ahead of the biological reactor basins. The facility has been operating in DEMON® mode since May 2015. DEMON® SBR is typically designed with a volumetric ammonium loading rate near 0.7 kg/m^3-day. Ammonium and total inorganic nitrogen removal efficiencies of 90-95 percent and 80- 85 percent, respectively, have been reported for full-scale systems at loading rates ranging from 0.3-0.6 kg-N/m^3-day (Bowden et al., 2015). A process schematic for the centrate pretreatment facility is presented in Figure 2-4.

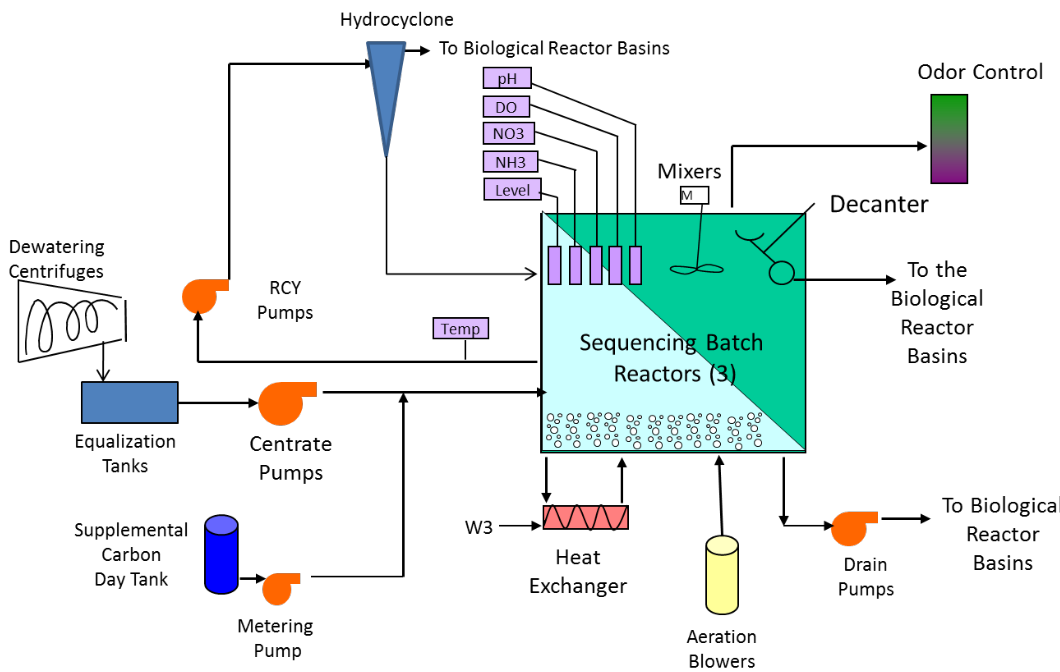


Figure 2-4. Schematic of Deammonification Reactor at AWRRF (Yin and Sanjines, 2017)

The facility uses two full scale SBRs sized to treat all the dewatering centrate produced at AlexRenew at design annual average flow. A third small-scale reactor is available for testing and can also be used to grow and store biomass if needed. The centrate from the centrate storage tanks is pumped to the centrate pre-treatment (CPT) facility which operates in the DEMON® mode. The mixed liquor inside the reactor is aerated by flat panel diffusers mounted on the reactor floor and is kept homogeneously mixed by top-mounted mixers. All reactors are equipped with heat exchangers to cool the contents of the reactor to avoid excessively high temperatures (over 35 °C) which could inhibit the process. Since the anammox bacteria are slow-growing, a portion of the reactor contents is circulated through the hydrocyclones which allows the wasting of lighter weight organisms such as AOBs, NOBs, and heterotrophs while keeping the heavier anammox granules and circulating them back to the reactor and increasing their solids retention time (SRT**).**

2.6 Detailed Statistical Analysis – Plant Effluent Concentrations

Facility operating data from January 2015 to December 2017 were analyzed. Figures 2-5 through 2-8 and Tables 2-3 through 2-6 provide a summary of the statistical analysis performed for the AWRRF facility in Alexandria, Virginia. As explained below, the data shows that the facility consistently met the final effluent treatment objectives for total nitrogen and ammonia shown in Table 2-2 throughout the analysis period.

Figure 2-5 shows the rolling 12-month average TN concentrations and discharge loadings, and the plant's consistency in meeting the concentration and loading limits from month to month. For 2015 through 2016, the annual average TN discharge waste load allocation was 493,381 lbs/yr TN at actual flow. As of January 2017, the facility was required to meet an annual average TN discharge limit of 3.0 mg/l. The actual annual average effluent TN concentration for 2017 was 2.61 mg/l. Additionally, for 2015 and 2016, Figure 2-5 shows that the plant effluent was also consistently well below its maximum TN load limit of 493,381 lbs/day.

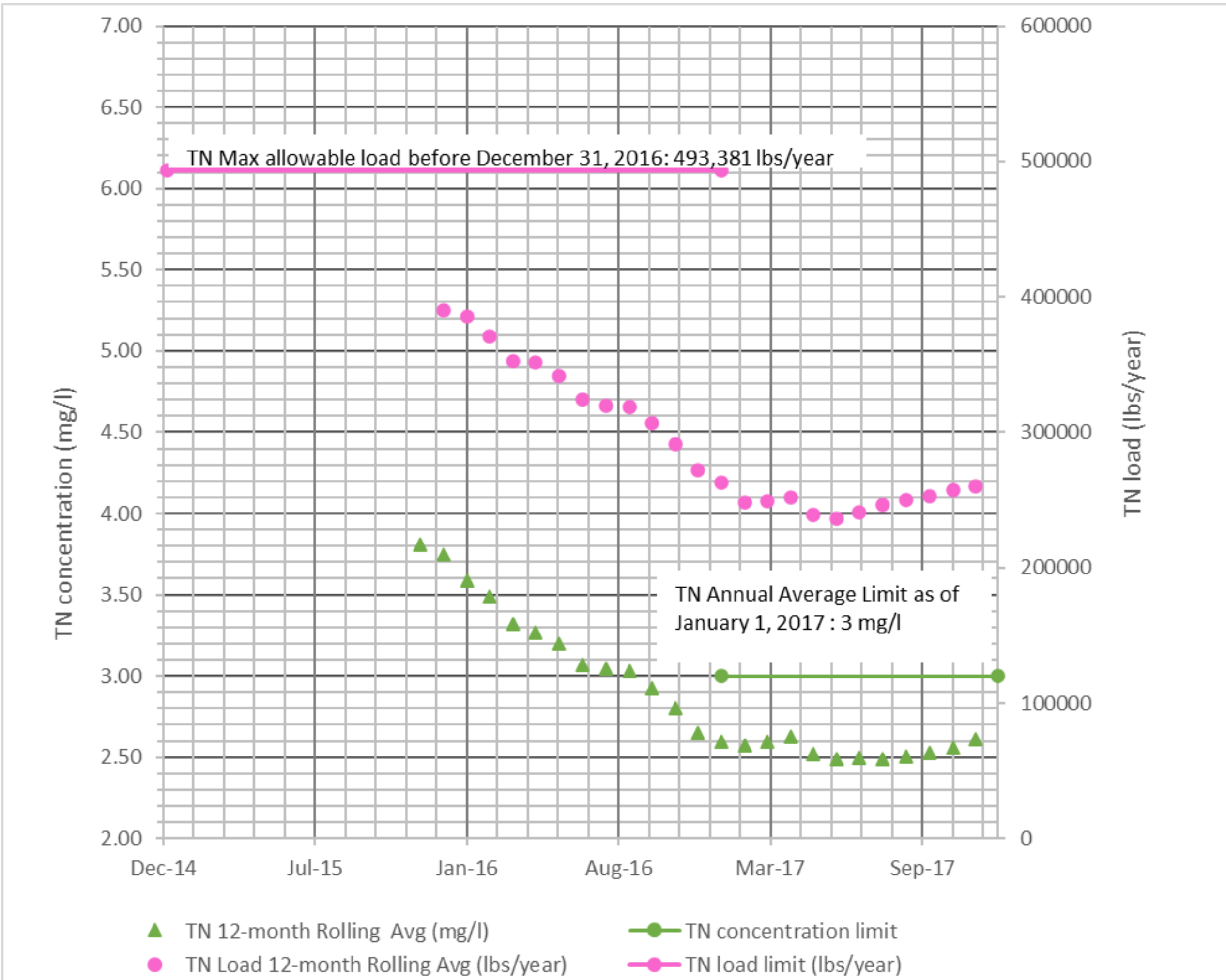


Figure 2-5. 12-Month Rolling Average Time Series Plot for Plant Effluent TN Load

AWRRF's NPDES discharge permit for the 3-year analysis period included weekly average ammonia limits of 4.4 mg/l (April – October), 10 mg/l (November – January), and 8.5 mg/l (February – March). The permit also included monthly average ammonia limits of 1.0 mg/l (April – October), 8.4 mg/l (November – January), and 6.9 mg/l (February – March). In addition, the permit included weekly and monthly ammonia loading limits of 900 kg/day and 200 kg/day, respectively for the periods between April and October.

The weekly average plant effluent ammonia concentrations over the 3-year data set period were as follows: 0.07 mg/l (April – October), 0.17 mg/l (November – January), and 0.35 mg/l (February – March). The monthly average ammonia concentrations calculated from the daily data for these intervals over the 3-year data set period were as follows: 0.08 mg/l (April – October), 0.16 mg/l (November – January), and 0.37 mg/l (February – March). Similarly, the weekly and monthly average ammonia loadings discharged in the effluent between April and October over the 3-year period were 15.24 kg/day and 34.76 kg/day, respectively. In all these cases, the facility was well within the permit requirements for ammonia.

Figure 2-6 shows the 30-day rolling average time series plot for nutrient species effluent concentrations at AWRRF. For the 3-year analysis period, the median 30-day rolling average TN concentration was 2.83 mg/l with a maximum value of 5.12 mg/l. The median 30-day rolling average concentration was 0.085 mg/l for ammonia and 1.94 mg/l for NOx-N, with maximum values of 0.793 mg/l NH_3-N and 3.53 mg/l Nox-N. During the analysis period before DEMON startup (January 2015 – May 2015), the median NOx-N value was 2.87 mg/l and the maximum value was 7.61 mg/l. After DEMON startup (June 2015 – December 2017), the median NOx-N value was 1.78 mg/l, and the maximum value was 5.11 mg/l.

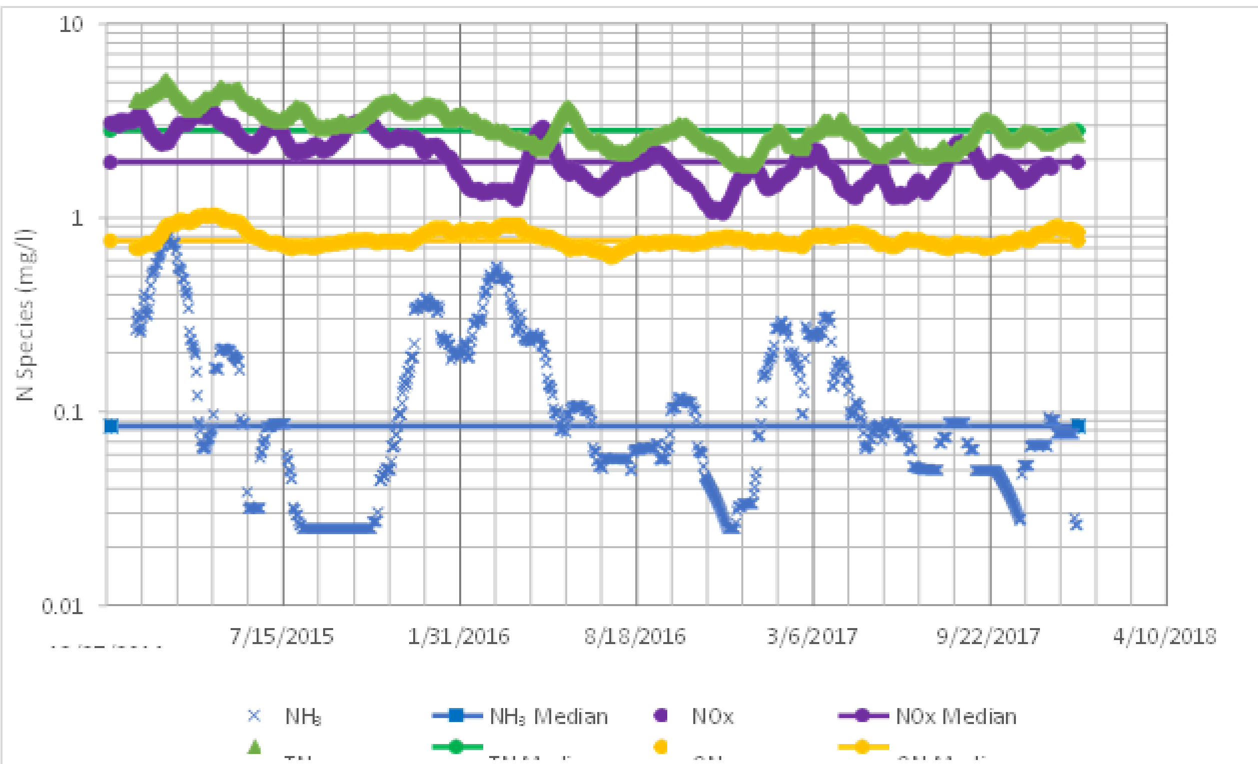
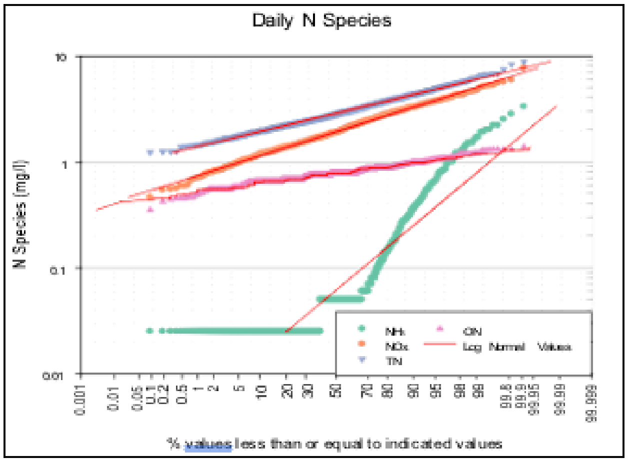


Figure 2-6. 30-Day Rolling Average Time Series Plot

Figures 2-7 A through D include cumulative probability plots for AWRRF's daily, 30-day rolling average, monthly average, and 12-month rolling average data sets. A percentile value on the x-axis represents the probability that the value is less than or equal to the stated corresponding concentration on the plot's y-axis. Figure 2-7A shows that a significant part of the effluent TKN was comprised of organic nitrogen and that most of the effluent TN was due to effluent nitrate. Organic nitrogen concentrations were calculated as the difference between TKN and NH_3 nitrogen.



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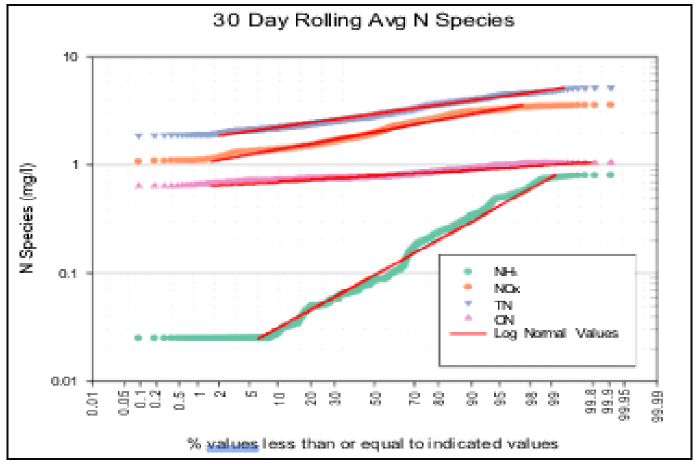

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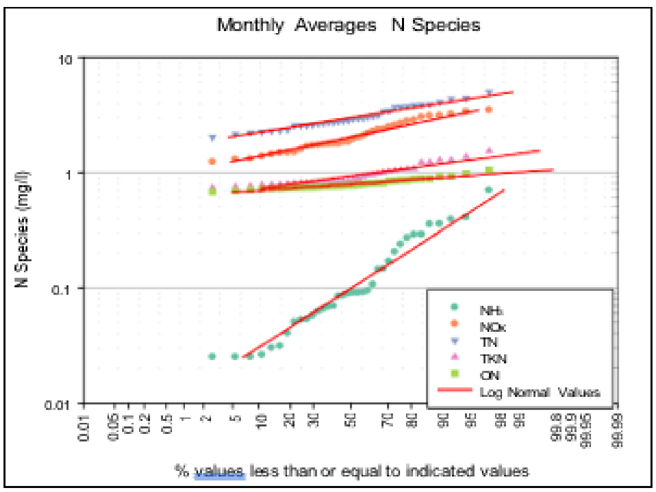

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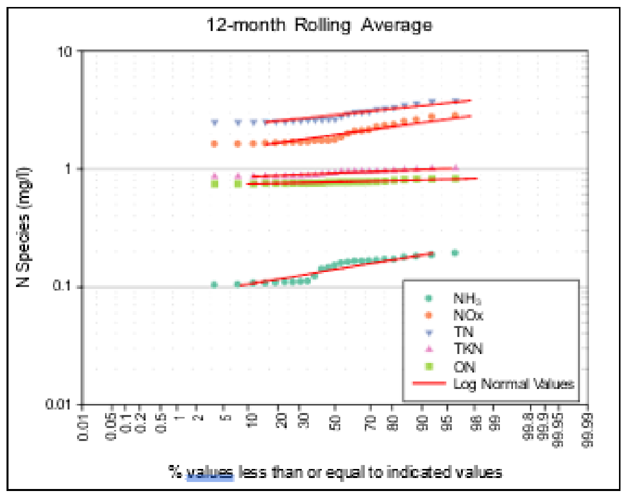


D

Figure 2-7. Probability Plots for AWRRF from January 2015 – December 2017 – (A) Daily Data; (B) 30-day Rolling Average; (C) Monthly Average; (D) 12-Month Rolling Average

The daily average plant effluent TN concentration for the 3-year analysis period was 3.0 mg/l with a 95th percentile of 4.92 mg/l shown in in Figure 2-7 A. For 2017, the year following completion of the plant's upgrades including the sidestream deammonification facility, the daily average TN concentration was much lower at 2.61 mg/l with a 95th percentile daily average TN average concentration was significantly lower at 3.94 mg/l.

Table 2-3. Summary Statistics for Final Effluent Ammonia Nitrogen for AWRRF from January 2015 to December 2017

	NH_3-N Daily Data	NH_3-N Weekly Data	NH_3-N 30-day Rolling Average	NH_3-N Monthly Average	NH_3-N 12-Month Rolling Average
n	1,096	156	1,067	36	25
Mean	0.15	0.15	0.15	0.15	0.14
Geometric Mean	0.06	0.08	0.10	0.10	0.14
Standard Dev.	0.31	0.19	0.15	0.15	0.03
CV	2.10	1.29	1.01	0.99	0.21
Skew	4.77	2.46	2.03	1.87	-0.08
Minimum	0.03	0.03	0.03	0.03	0.10
Maximum	3.30	1.00	0.79	0.69	0.19

Table 2-4. Summary Statistics for Final Effluent NOx-N for AWRRF from January 2015 to December 2017

	NOx-N Daily Data	NOx-N Weekly Data	NOx-N 30-day Rolling Average	NOx-N Monthly Average	NOx-N 12-Month Rolling Average
n	1,096	156	1,067	36	25
Mean	2.09	2.07	2.08	2.09	1.98
Geometric Mean	1.92	1.95	1.99	2.00	1.95
Standard Dev.	0.87	0.73	0.62	0.64	0.38
CV	0.42	0.35	0.30	0.31	0.19
Skew	1.11	0.78	0.49	0.58	0.90
Minimum	0.46	0.79	1.06	1.21	1.62
Maximum	7.61	4.51	3.53	3.41	2.81

Table 2-5. Summary Statistics for Final Effluent Total Nitrogen for AWRRF from January 2015 to December 2017

	TN Daily Data	TN Weekly Data	TN 30-day Rolling Average	TN Monthly Average	TN 12-Month Rolling Average
n	1,096	156	1,067	36	25
Mean	3.02	3.00	3.01	3.02	2.90
Geometric Mean	2.87	2.90	2.93	2.94	2.87
Standard Dev.	1.00	0.84	0.70	0.72	0.43
CV	0.33	0.28	0.23	0.24	0.15
Skew	1.20	1.01	0.71	0.79	0.85
Minimum	1.22	1.62	1.86	2.00	2.49
Maximum	8.73	6.21	5.12	4.93	3.81

Table 2-6. Summary Statistics for Final Effluent ON for AWRRF from January 2015 to December 2017

	ON Daily Data	ON Weekly Data	ON 30-day Rolling Average	ON Monthly Average	ON 12-Month Rolling Average
n	1,096	156	1,067	36	25
Mean	0.79	0.79	0.79	0.79	0.78
Geometric Mean	0.77	0.78	0.78	0.78	0.78
Standard Dev.	0.14	0.10	0.08	0.08	0.02
CV	0.18	0.13	0.10	0.10	0.03
Skew	0.73	0.90	1.06	1.12	0.92
Minimum	0.35	0.58	0.63	0.67	0.75
Maximum	1.38	1.21	1.04	1.04	0.83

Figure 2-8 provides a probability summary for the nitrogen species at the 3.84, 50, 90, 95, and 99th percentiles for the various data sets (i.e., daily, weekly, 30-day rolling, monthly, and annual). The monthly average ammonia concentrations associated with the 3.84, 50, 90, 95, and 99 percentiles were 0.025, 0.089, 0.352, 0.390, and 0.587 mg/l, respectively, all well below the discharge permit monthly average limits of 1.0 mg/l (April – October), 8.4 mg/l (November – January), and 6.9 mg/l (February – March). Similarly, the weekly average ammonia concentrations associated with the 3.84, 50, 90, 95, and 99th percentiles were 0.025 mg/l, 0.057 mg/l, 0.372 mg/l, 0.566 mg/l, and 0.887 mg/l, respectively, all well below the discharge permit monthly average limits of 4.4 mg/l (April – October), 10.0 mg/l (November – January), and 8.5 mg/l (February – March).

Figure 2-8 also highlights the process variability for TN and NH_3-N. Comparing the daily data median (50th) percentile of 2.82 mg/l for TN to the 95th percentile of 4.92 mg/l, the 95th /50th was about 1.74. Calculating this ratio for the period when the 3 mg/l TN limit was effective (January – December 2017), the ratio was lower at 1.57 demonstrating lower variability.

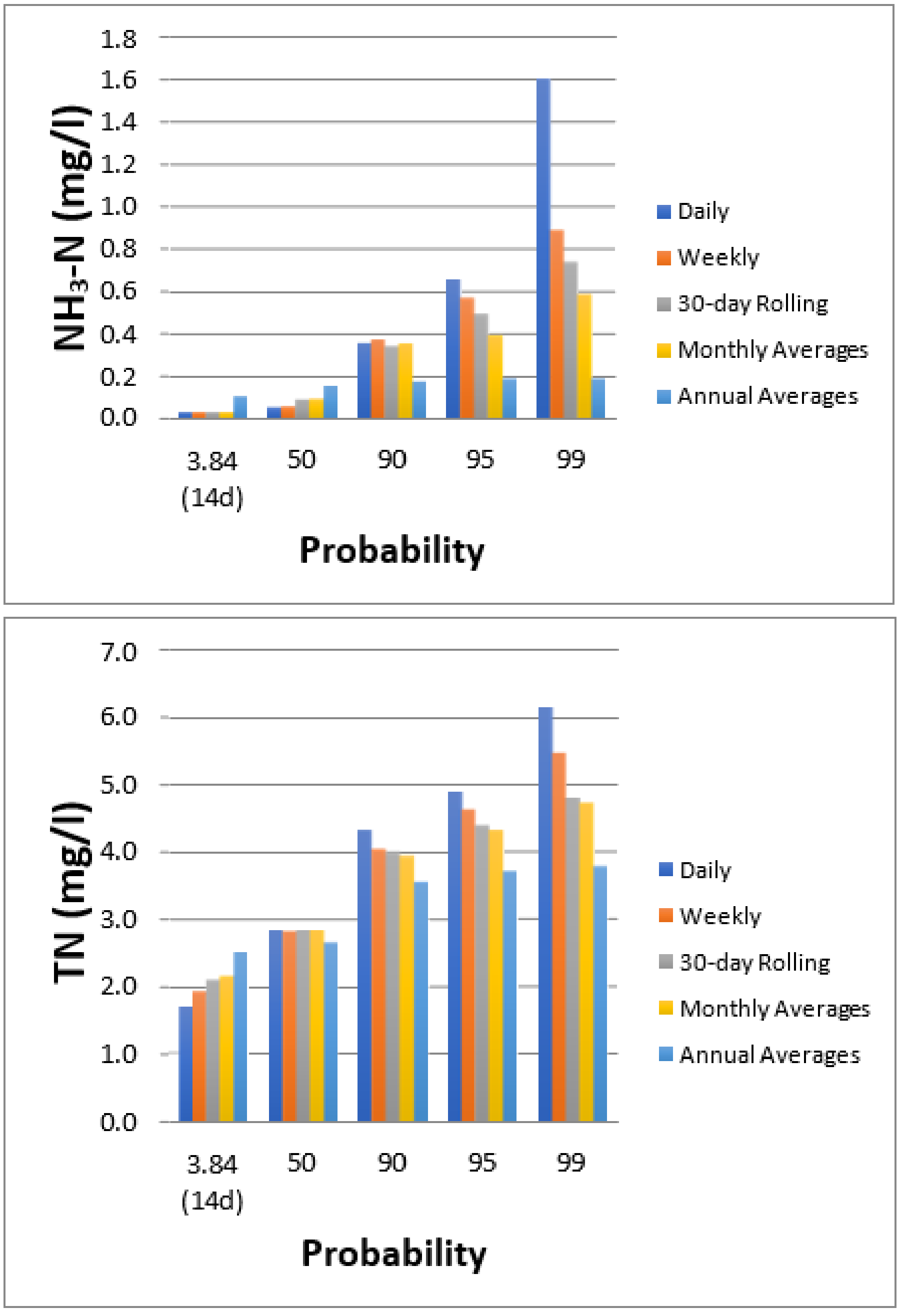


Figure 2-8. Probability Summary for AWRRF

2.7 Process Performance - DEMON® Sidestream Deammonification at AWRRF

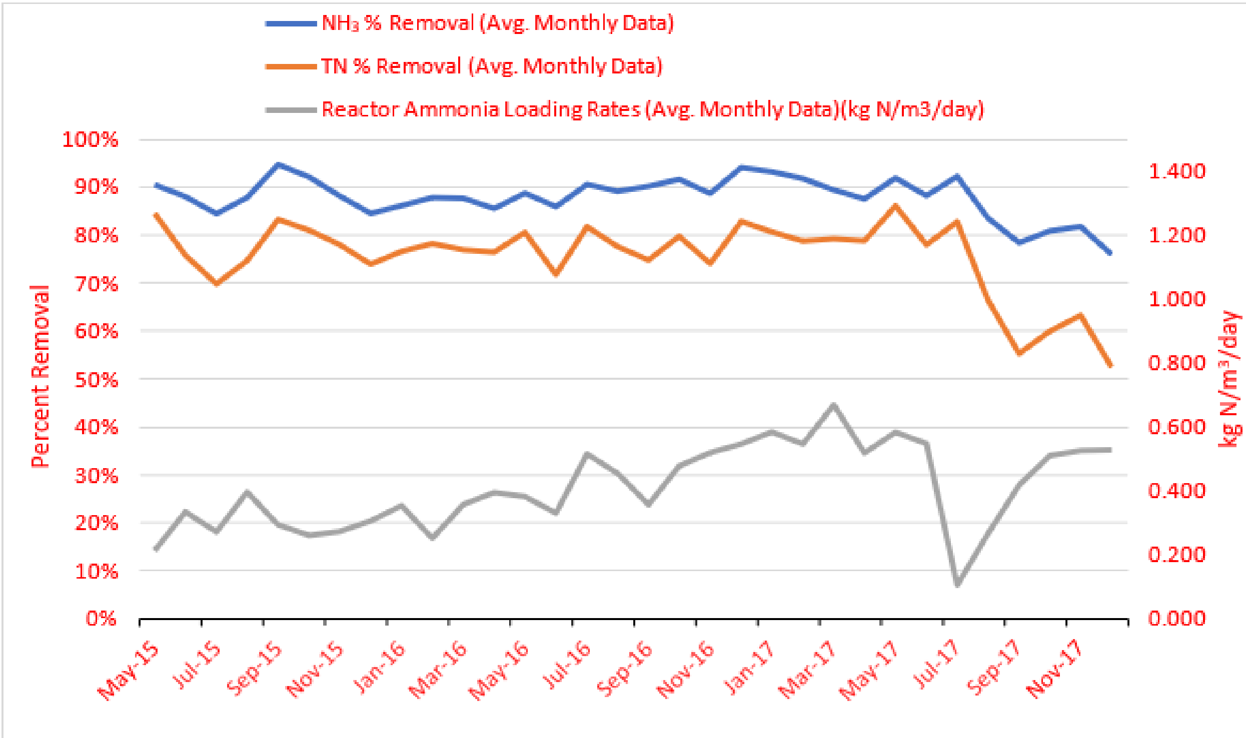
As shown in section 2.5 and figure 2-4 above, the centrate pre-treatment (CPT) facility at AWRRF uses the DEMON® sidestream deammonification system which is based on a SBR process to reduce the ammonia loading to the bioreactors. The system was chosen as a preferred option to treat the ammonia load compared to treatment in the mainstream BNR system, due to reduction of aeration and supplemental carbon requirements which result in cost savings. Sidestream deammonification also results in reduction in alkalinity requirements compared to conventional nitrification-denitrification.

The SBR operational sequence includes a fill/react initial step in anoxic mode with the mixer running until the pH reaches a maximum setpoint or until a set time has passed (usually about 10 minutes). The mixer is then turned off and the aeration is turned on until the pH reaches a minimum setpoint or until a set time has passed (usually about 10 minutes). The anoxic and aerobic cycles are repeated until the end of the fill/react step (generally about 7 hours). A settle step is then started with the mixer and the aeration turned off (generally 15 to 30 minutes). A decant step is then initiated to remove the top layer in the SBR to a desired minimum operating level. The facility is operated remotely by a supervisory control and data acquisition (SCADA) system. Main process control parameters include pH, DO, ammonia, nitrate, and reactor level. The design values for the process are shown in Table 2-7.

Table 2-7. Design Parameters for DEMON® Sidestream Deammonification at AWRRF

Parameter	Design Value
Reactor Dimensions	46 ft x 56 ft x 21 ft depth (14 m x 17 m x 6 m)
Reactor Volume	400,000 gal (1,500 m^3)
Reactor Unit Loading	0.0036 lb NH_3-N/gal-day (0.43 kg NH_3-N/m^3-day)
Loading Per Reactor	1,400 lb/day (636 kg NH_3-N/day)
Average Centrate Concentration	1,200 mg/l
Centrate Flow per Reactor	140,000 gpd
Ammonia Removal	85%
Total Nitrogen Removal	80%

The process was put in service in May 2015. The CPT system was designed with two sets of SBR reactors and associated equipment/instrumentation, with each reactor sized to treat all the dewatering centrate produced at AlexRenew at design annual average flow rates. Based on centrate production and the volume of each SBR reactor, only one of the two reactors was needed for the process. It was seeded with imported Annamox and initially operated at a low ammonia loading rate which was gradually increased over the following 12 months as the process became more stable. Figure 2-9A shows the operational performance of the sidestream deammonification reactor through December 2017 while Figure 2-9B shows the reactor's monthly average temperature. Figure 2-9A plots monthly average reactor nitrogen loading rates as well as ammonia nitrogen and total nitrogen removal values calculated from daily data provided by AWRRF. The facility was able to reach its target ammonia loading rate of 0.43 kg NH_3-N/m^3-day in August 2016 and operate at average monthly loading rates as high as 0.67 kg NH_3-N/m^3-day at times thereafter. Ammonia removal rates after July 2016 were achieved consistently except for a period between July and September 2017 when wasting rates were inadvertently set too high resulting in loss of biomass from the reactor. This was corrected, however it impacted ammonia and total nitrogen removal rates and it took over 3 months for the system to completely recover due to the very slow anammox growing rates.



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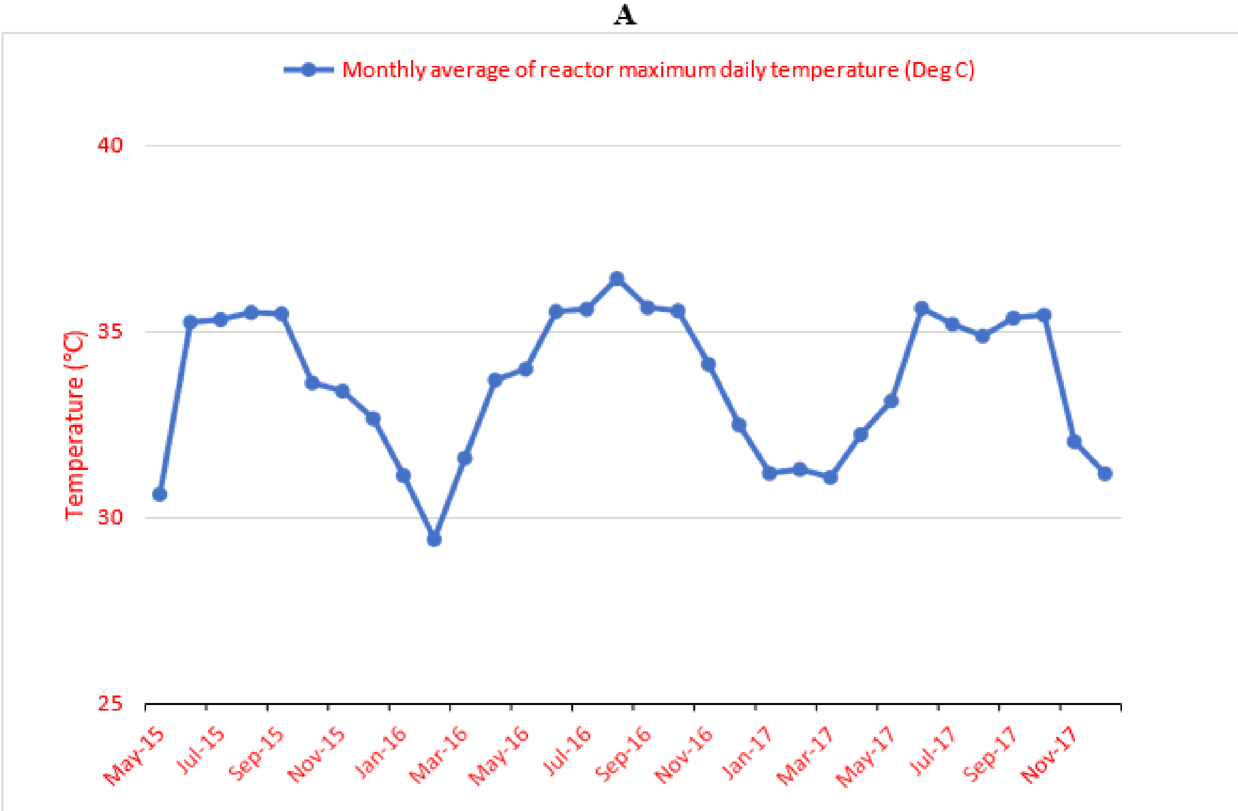

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Figure 2-9. (A) Ammonia and Total Nitrogen Removal and Ammonia Loading of Sidestream Deammonification Process (B) Monthly Average Temperature of the Sidestream Deammonification Reactor

During the time period investigated (May 2015 – December 2017), the facility faced a number of operational challenges which reduced deammonification activity, some of which are explained below. These included problems with centrate quality, micronutrients deficiency, over-aeration, floating granules, pH probe issues, and others. In addition to monitoring DO, pH, temperature, and fill/react/settle/decant cycle periods, reactor nitrite was monitored to assess process stability. Nitrite is a significant parameter as the electron acceptor in the conversion by anammox in the deammonification process. Testing at other facilities has shown that concentrations as low as 4.8 mg/l nitrite nitrogen result in decreased anammox activity (Wett, 2007).

Nitrite concentration control was a significant challenge during the reactor startup. Operational data provided by AWRRF showed that the nitrite concentration in the DEMON® reactor exceeded the target concentration of 2 mg/l during startup and throughout the first year of operation and was as high as 18 mg/l on occasions. About three weeks after startup, it was discovered that high solids in the centrate generated during centrifuge startup and shutdown periods inhibited AOB and anammox resulting in lower ammonia and TN removal rates. This problem was addressed by diverting dirty centrate away from the CPT system.

Reactor performance deteriorated again about eight weeks later with higher reactor nitrite concentrations even at low ammonia loading rates (below 0.15 kg N/m^3 /d). A scan of micronutrients was conducted on influent centrate and reactor mixed liquor to evaluate and compare to threshold values reported in the literature; however, results were inconclusive. In late July 2015, a micronutrient solution was added to the reactor, which immediately lowered the reactor nitrite concentration and allowed a quick increase of influent ammonia loading to 0.4 kg N/m^3/d within one week. In early September 2015, reactor nitrite spiked again and was attributed to over-aeration. As a result, ammonia loading was lowered again to stabilize nitrite. The ammonia loading was then increased but at a very slow rate. After addressing other challenges mentioned below, influent ammonia loading was raised and exceeded the design target of 0.43 kg N/m^3/d in November 2016.

Another challenge encountered during startup was floating granules in the CPT reactor during the reactor nitrite spike periods. This posed a risk of losing anammox bacteria from the reactor and was thought to be caused by excessive polymer in the centrate. The excessive polymer may have coated the granules possibly causing them to trap nitrogen gas and cause floating granules. To address this issue, plant operations staff reduced the polymer dose in the dewatering process by about 30 percent without impacting biosolids cake dryness; however, this did not address the problem. Staff then observed that when nitrite concentrations were high, significantly higher bubble formation occurred which is expected in the deammonification reaction due to nitrogen gas formation when conditions are favorable. Operations staff reported that the floating granules problem was subsequently addressed by maintaining low nitrite concentrations which limited nitrogen gas production during the settling and decanting phases and by keeping theses phases reasonably short.

Additionally, dark solids and foam were observed in the DEMON® reactor for several weeks after startup. After investigating potential causes, AWRRF determined that this was due to large amounts of solids in the centrate during centrifuge startup and stoppage, which escaped the centrate storage tanks that fed the CPT facility. This was determined to be due to difficulties in cleaning the bottom of the centrate receiving tank. As a result, AWRRF implemented operating procedures to periodically drain the bottom of the centrate tanks.

Hydrocyclone operation is very important in the DEMON® process since they result in retaining anammox granules within the reactor and facilitate wasting of undesired species such as AOB, NOB, and heterotrophs out of the system. AWRRF staff addressed a problem related to clogging of the recycle pumps that feed the hydrocyclones resulting in damage of the pumps and casings. This was determined to be caused by rags and other debris escaping the sludge screening system ahead of the sludge pre-pasteurization process mentioned above and ending up in the centrate and not settling in the centrate receiving tank. A strainer was installed by AWRRF staff on the centrate feed line to resolve this problem; however, frequent strainer cleanings were required as a result.

It should also be noted that quantitative polymerase chain reaction (qPCR) analyses were conducted during the DEMON® startup phase to quantify AOB, anammox and NOB abundance (Yin et al., 2018). During initial reactor startup, AOB were present in high abundance and NOB were low. However, after June 2015, AOB decreased significantly coupled with quick accumulation of *Nitrospira spp*. This coincided with the incidents of dirty centrate entering the reactor and micronutrient deficiency mentioned above. After micronutrient addition, a sample in late September 2015 showed low NOB presence. In early 2016, the AOB population was back to desired levels but then decreased again in March and April and were reduced to very low levels in summer 2016. In April 2016, hydrocyclone operation hours were reduced and hydrocyclone overflow decreased. This caused an increase of reactor mixed liquor TSS. Concurrently, CPT reactor nitrite concentration reliably dropped below 5 mg/l despite the process upset periods. These observations matched with reduced anammox fractions observed in the period, which is believed to be associated with increased presence of heterotrophs. It was concluded that while the qPCR analysis showed decreased anammox and AOB populations during the periods of poor centrate quality and reduced hydrocyclone overflow, their microbial activities were higher thereby preventing nitrogen removal from being negatively affected (Yin et al., 2018).

2.8 Impact of DEMON® Sidestream Deammonification Process and Other Upgrades on AWRRF Plant Performance

Figure 2-10 shows the monthly average plant effluent TN concentrations from January 2014 through December 2017. The graph clearly shows the significant reduction in plant effluent concentrations since the startup of the CPT facility in May 2015. Monthly average effluent TN concentrations from January 2014 through April 2015 were 4.00 mg/l compared to 2.87 mg/l from May 2015 through December 2017, the period since startup of the CPT facility. The 95^{th} and 50^{th} percentile monthly average TN concentration for these two periods were 6.08, 4.00 and 3.82, 2.74 mg/l, respectively. The ratio of the 95^{th} to 50^{th} percentiles for these two periods were 1.59 and 1.46, indicating more stable TN removal at AWRRF after startup of the CPT facility.

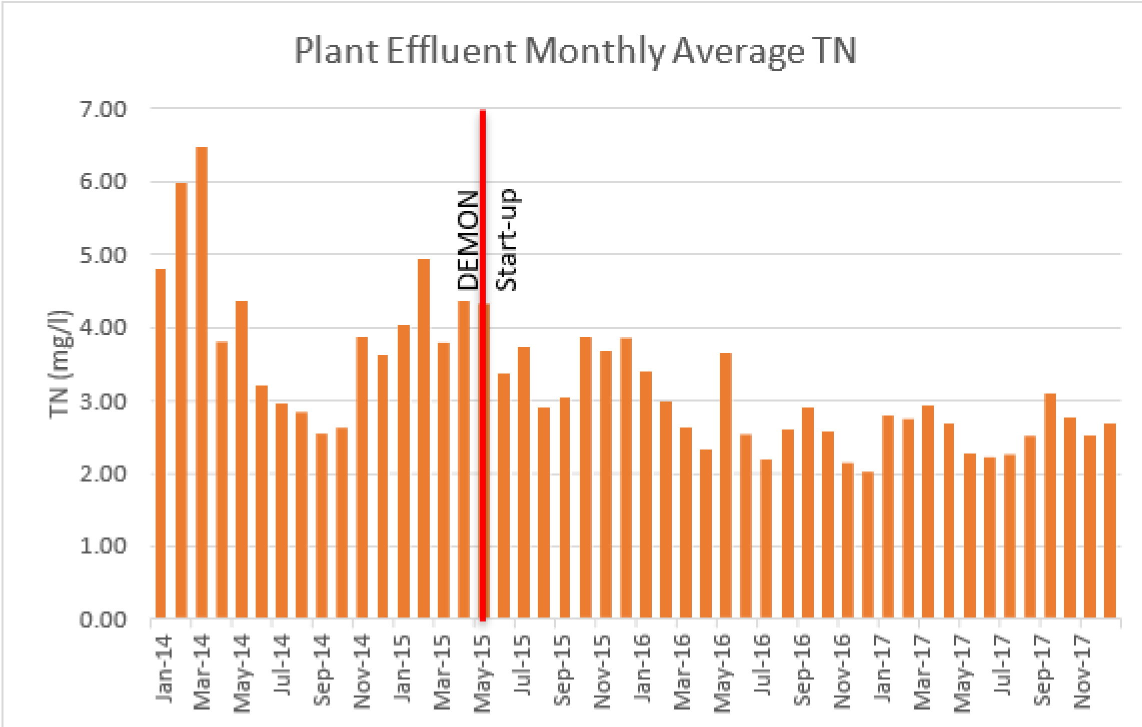

Figure 2-10. Monthly Average Plant Effluent TN Concentrations

While AWRRF believes that the CPT facility's impact on improving the plant's overall nitrogen removal as well as on reducing mainstream aeration requirements and methanol addition is significant, the facility did implement a number of additional mainstream BNR process enhancements which were placed in service in 2016 to further reduce costly aeration requirements and methanol consumption. These enhancements included implementation of ammonia-based aeration control (ABAC), automated methanol dosing, and primary effluent flow and load equalization to reduce fluctuations in the carbon to nitrogen ratio of the bioreactor influent.

In implementing ABAC, a target ammonia concentration range at the end of each biological reactor is set and the ammonia concentration is measured. The plant's supervisory control and data acquisition (SCADA) system automatically adjusts the DO set points in the biological reactor to maintain the ammonia at the target range. If the ammonia concentration is less than the ammonia low end setpoint, the DO setpoint is decreased; if the ammonia concentration is greater than the ammonia high end setpoint, the DO setpoint is increased. SCADA also automatically turns swing zones on and off to maintain the ammonia at the target range. This process control approach can result in significant reduction in aeration energy.

Figure 2-11 below shows the monthly average aeration flow from the blowers to the biological reactors. Comparing the monthly average air use values between the periods of January 2014 through April 2015 and May 2015 through December 2017 (i.e., before and after CPT facility startup) shows a reduction in aeration flow of 28 percent. It is worth noting that for 2017, the lower air and associated lower energy use were accomplished while meeting the more stringent final effluent total nitrogen annual limit of 3 mg/l starting in January 2017.

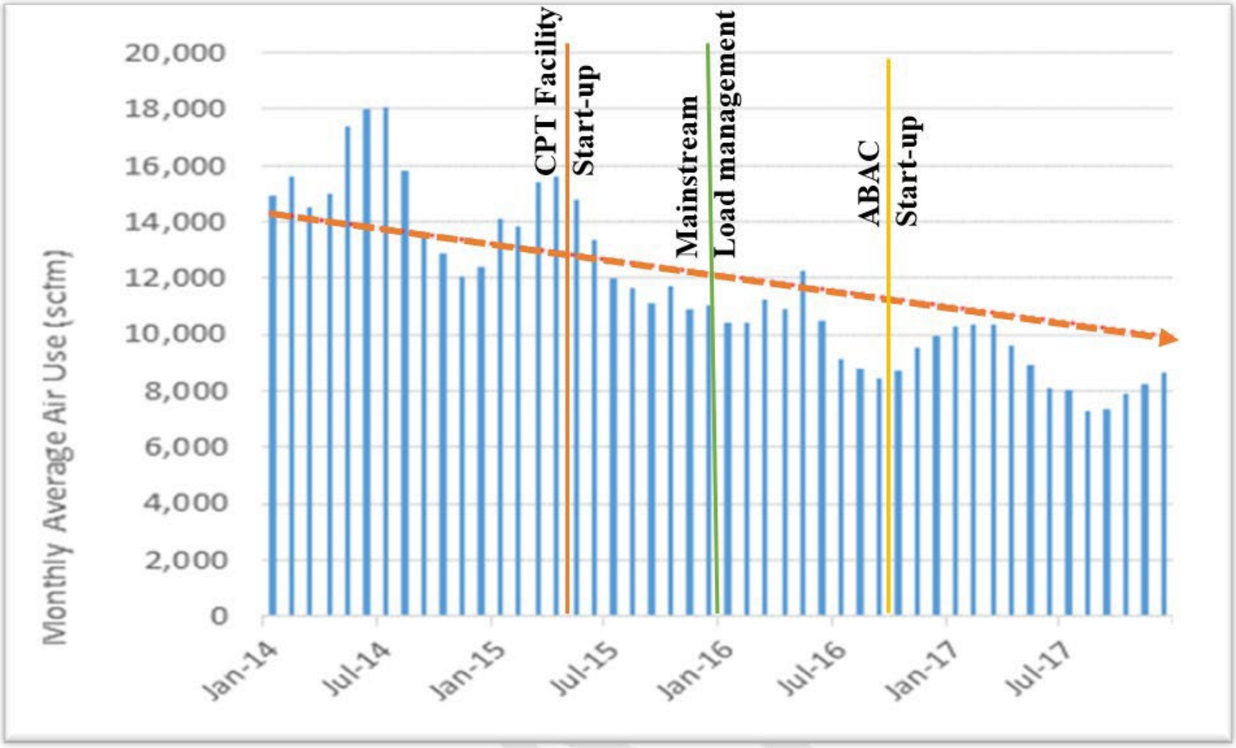

Figure 2-11. Average Monthly BNR Aeration

Automated methanol dosing in the mainstream BNR system was initiated in October 2016 and is controlled through the SCADA system. The methanol dose setpoint is automatically determined using an operator-adjustable lookup table that increases the dose in proportion to the NOx concentration measurement in the influent to the last post-anoxic biological reactor basin (BRB). The methanol dosing is also interlocked with oxidation-reduction potential (ORP) readings in the last BRB. If the ORP readings indicate anaerobic conditions, methanol pumps are turned off and air is turned on just upstream of the post-anoxic BRB. Figure 2-12 shows monthly methanol use for January 2014 through December 2017. Comparing average monthly methanol use before (January 2014 through September 2016) and after automated dosing was initiated (October 2016 through December 2017) shows a significant reduction in methanol use of 46 percent. This was accomplished while meeting the lower annual average TN effluent limit of 3 mg/l starting in January 2017.

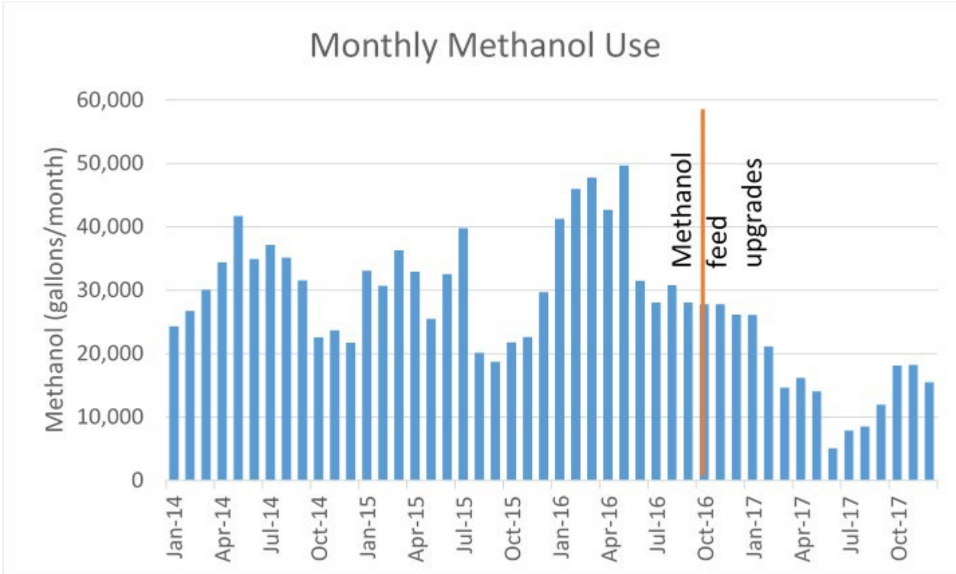

Figure 2-12. Average Monthly Methanol Use

It should be noted that the treatment system upgrades at WRRF to reduce energy and chemicals consumption described above were done as part of an overall treatment strategy that included efforts to redirect the nitrogen removal pathways in the mainstream biological reactors from the conventional nitrification-denitrification to the more efficient nitritation-anammox (deammonification) while maintaining the ability to nitrify and denitrify as needed to meet TN effluent limits. As mentioned in Section 2.4, full-scale repression of NOBs in mainstream processes is difficult to consistently achieve. This is generally due to lower wastewater temperatures and substrate concentrations as compared to sidestream conditions and requires an operational strategy that focuses on these limitations. Based on previous research and testing, the success of mainstream deammonification is thought to depend, to a large extent, on the control of two crucial parameters for NOB repression and prevention of nitrate formation: competition between AOB and NOB for oxygen and competition between anammox and NOB for nitrite (Wett et al., 2013).

The AWRRF BNR system mainstream deammonification upgrades focused on creating the particular environmental conditions necessary to suppress NOB activity and increase deammonification rates. In addition to implementation of ABAC and automated methanol dosage described above, the implementation strategy at AWRRF included planned operational actions such as the following:

1. Aeration process control to create conditions in the biological reactor basins (BRBs) that are favorable for nitritation/denitritation/deammonification.
2. Hydrocyclones on the waste-activated sludge (WAS) flow stream to separate the anammox granules and keep them in the system while wasting NOBs and other smaller particles and improving sludge settleability by keeping larger particles in the system.
3. Seeding anammox bacteria from the centrate pre-treatment (CPT) facility to augment the anammox population and increase its activity in the mainstream process.

4. Primary effluent flow/load equalization to diminish fluctuations in the carbon to nitrogen (C:N) ratio of the bioreactor feed, aeration demand, and methanol dosage requirements.

System testing, tuning and optimization has been ongoing to test implementation of mainstream deammonification at AWRRF. A complete description of this strategy and related upgrades is provided by Sanjines et al. (2019).

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CHAPTER 3.0
The Westside Regional Wastewater Treatment Plant – Sidestream Enhanced Biological Phosphorus Removal (S2EBPR) – Case Study

3.1 Background

The Westside Regional Wastewater Treatment Plant (WRWTP) is a tertiary treatment plant located in the District of West Kelowna, British Columbia, Canada. It is operated by the Regional District of Central Okanagan and receives wastewater from the Districts of West Kelowna and Peachland and from the Westbank First Nation Reserves #9 and #10. Its purpose is to serve the sewered areas of Westbank and Shannon Lake. WRWTP underwent three expansions in 1995, 2006, and 2012 with a final resulting capacity of 3.8 MGD (16,800 m^3/day). The 2012 expansions included a new headworks building with 6 mm perforated plate mechanical screens, fermenter retrofits, additional bioreactors, clarifiers, and fabric filters. Additional UV banks as well as biosolids dewatering centrifuges were also included.

3.2 Plant Processes

Preliminary treatment of influent wastewater includes coarse bar screening, perforated plate mechanical screens, and vortex grit removal. Primary settling occurs in three rectangular clarifiers.

Primary effluent is then treated in a BNR facility designed to remove both nitrogen and phosphorus in addition to BOD. The BNR facility is operated in the Westbank process mode but with only a portion of the fermentate (from a primary sludge fermenter) added to the anerobic zone, and primary effluent is introduced into the mainstream anoxic zone (none to the pre-anoxic zone or anaerobic zone) along with the rest of the fermentate.

Effluent from the bioreactors is settled in the secondary clarifiers for clarification. Alum can be added ahead of the secondary clarifiers for chemical phosphorus removal when needed. Clarified secondary effluent is treated in AquaDisk® 10-micron cloth membrane filters prior to ultraviolet (UV) disinfection and discharge into Okanagan Lake.

Primary sludge solids are treated in fermenting tanks for generating volatile fatty acids (VFAs) which are fed to the sidestream reactor and to the mainstream anoxic zone for phosphorus release. Waste activated sludge (WAS) is thickened in dissolved air flotation units and then mixed with fermented primary sludge prior to centrifuge dewatering. Dewatered sludge cake is hauled offsite to a land application site.

Table 3-1. Current Discharge Limits as of January 1, 2014 at WRWTP

Parameter	Daily Max (mg/L)	Annual Average (mg/L)
BOD_5	10	N/A
$cBOD_5$	N/A	N/A
TSS	10	N/A
TN (mg/l)	10	6
TP (mg/l)	2.0	0.2
Temperature (°C)	N/A	N/A

Note: N/A: Data not available or applicable

3.3 Conventional Enhanced Biological Phosphorus Removal Technology

Enhanced biological phosphorus removal (EBPR) in wastewater treatment is accomplished by encouraging the growth of phosphate accumulating organisms (PAOs). PAOs are heterotrophic bacteria that occur naturally in the environment and in aerobic activated sludge. The growth of PAOs is encouraged by cycling them between anaerobic and aerobic conditions. In the presence of oxygen (i.e., aerobic conditions), PAOs obtain energy from stored food and uptake large amounts of phosphorus into their cells, which they store as polyphosphates. These polyphosphates contain high-energy bonds and function like energy storage batteries.

In the absence of oxygen (i.e., anaerobic conditions), PAOs can break the polyphosphate bonds resulting in the release of orthophosphate and use the resulting energy to uptake easily biodegradable compounds, namely short chain volatile fatty acids (VFAs). PAOs polymerize and store the VFAs in their cells as intermediate products known as poly-β-hydroxy-alkanoates (PHAs), of which the most common is poly-β-hydroxy-butyrate (PHB). When oxygen becomes available again (i.e., aerobic conditions), they can metabolize the PHAs to generate energy and uptake phosphorus (in the form of phosphate) and store the excess amount. (Randall et al., 2010).

In general, it was traditionally accepted that phosphorus could only be removed in conventional plants when the wastewater characteristics were favorable with a COD:TP ratio of at least 37:1 or a BOD:TP ratio about 18:1, with some of the COD consisting of short chain VFAs. More COD may be required if the process also involves denitrification of nitrate (Kobylinski et al., 2008). At wastewater treatment facilities where this ratio is low, external sources of carbon such as methanol, ethanol, or proprietary carbon products can be added. In some cases, carbon-rich waste products such as molasses, sugar wastes or others may also be used. Additionally, many plants have elected to ferment their primary sludge. This often requires measures to limit odors and to ensure stable and consistent performance.

Other microorganisms besides PAOs exhibit a similar metabolism as PAO. For example, Glycogen Accumulating Organisms (GAOs) are similar to PAOs in terms of being able to store readily biodegradable organic matter such as VFAs as PHA in the anaerobic phase (Zeng et al., 2003). GAOs do not contribute to phosphorus removal as their metabolism does not involve anaerobic phosphorus release and subsequent aerobic (or anoxic) phosphorus uptake (López-Vázquez et al., 2007). Therefore, in terms of biological phosphorus removal, GAOs are seen as competitors of PAOs for substrate and, as such, a main cause of process deterioration, or even failure, of EBPR systems (Thomas et al., 2003). Temperature has been identified as having a potential significant impact on the PAO-GAO competition. At temperatures greater than 25 °C, GAOs can outcompete PAOs for organic carbon (Law et al., 2016).

EBPR is considered a sustainable approach to removing phosphorus from wastewater. However, the process can in some cases be unstable particularly at low influent rbCOD/TP ratios and where other wastewater constituents such as dissolved oxygen or nitrate interfere with the anaerobic phosphorus release. To address such difficulties, particularly where low phosphorus limits are in place, many facilities install backup chemical systems and incur the cost of using chemical phosphorus removal as a backup to EBPR.

Generally speaking, the reliability of a nutrient removal process in meeting particular effluent targets varies from plant to plant and depends on various site-specific factors including wastewater characteristics, design, specific process configurations, operational conditions, and control parameters. In a detailed study of the long-term performance of EBPR facilities (Neethling, 2005), five EBPR facilities of various process designs, wastewater characteristics, operation, and other factors were evaluated over a three-year period to determine the biological phosphorus removal efficiency as well as their consistency (termed reliability in the study) of producing effluent concentrations at or below a given treatment goal. Table 3-2 shows the range of frequencies with which the plants achieved effluent orthophosphate (OP) concentrations of 0.5 mg/l, 1.0 mg/l, and 2.0 mg/l. The table also shows the concentrations achieved in 50 percent and 90 percent of the samples as well as the ratio between the 90^{th} and the 50^{th} percentiles (90 percent/50 percent) effluent OP concentrations. The values show that EBPR reliability is significantly reduced with lower desired effluent phosphorus concentrations particularly at lower discharge levels. Moreover, EBPR effluent concentration variability is significant as evidenced in the wide reliability range observed (90 percent/50 percent ratio range of 2 – 24) for the five facilities indicating relatively unstable performance.

Table 3-2. EBPR Reliability at Various EBPR Facilities (Adapted from Neethling et al., 2005)

Effluent Concentration & Percentile	Reliability Range for Five EBPR Facilities	Reliability Average
OP< 0.5 mg/l	24% - 95%	68%
OP< 1 mg/l	64% - 99%	82%
OP < 2 mg/l	85% - 100%	93%
50% (Geometric Mean)	0.05 – 0.76 mg/l	0.26 mg/l
90%	0.2 – 2.5 mg/l	1.6 mg/l
90%/50%	2.0 – 24.0	11.5

3.4 Sidestream Enhanced Biological Phosphorus Removal (S2EBPR)

An earlier study conducted to quantify key processes in full-scale sidestream hydrolysis tanks at two treatment plants concluded that sidestream EBPR is a promising configuration that in a number of cases will have advantages to the conventional mainstream EBPR process, especially when sufficient VFA is not available in the mainstream wastewater (Vollertsen et al., 2006). While many previous research observations showed that fermentation of return activated sludge (RAS) or a portion of the mixed liquor could help in producing low effluent phosphorus, it was assumed that the fermentation-produced VFAs were sustaining the growth of the much-researched PAO, *Candidatus Accumulibacter*, found mostly in conventional BNR plants (Barnard et al., 2017).

Candidatus Accumulibacter is able to store large amounts of polyphosphate (poly-P) aerobically after taking up organic substrates anaerobically, unlike ordinary heterotrophic organisms. *Accumulibacter* PAOs take up VFAs anaerobically and store them as PHAs, with energy obtained from hydrolysis of intracellular poly-P and reducing power from glycolysis of intracellular glycogen. Besides *Accumulibacter*, *Tetrasphaera*-related organisms are also putative PAOs present in a higher abundance than *Accumulibacter* in full-scale EBPR systems. *Tetrasphaera*-related PAOs can take up polyphosphate aerobically and store it intracellularly as poly-P, while assimilating different organic substrates (such as glucose and amino acids) under anaerobic conditions (Marques et al.,2017). Recent research showed that other PAOs may contribute to biological phosphorus removal and that *Tetrasphaera* can contribute and provide stability to the enhanced biological removal process (Nguyen et al., 2011; Barnard et al., 2017).

An alternative approach for improving EBPR process stability, eliminating the need for costly external carbon addition, and minimizing chemical usage includes implementation of a sidestream anaerobic biological solids hydrolysis and fermentation reactor. The reactor would involve sidestream RAS or mixed liquor hydrolysis and fermentation and has been named sidestream EBPR (S2EBPR). Recent full-scale and lab-scale testing performed on S2EBPR showed that it can be implemented in multiple configurations that can be used in a variety of wastewater treatment facilities, including those that do not utilize primary clarifiers (Tooker et al., 2018).

A study published by the Water Research Foundation (Gu et al., 2019) was conducted to elucidate the fundamental mechanisms involved in the S2EBPR process and to develop criteria for effective design and operation of the process. The study included a survey of operational information from a number of full-scale S2EBPR facilities, simulated sidestream reactor batch testing, full-scale pilot testing with side-by-side S2EBPR and conventional EBPR processes, and development of an improved biological process model for S2EBPR. The study concluded that a properly designed S2EBPR process allows the continuous generation of in-situ VFAs from RAS that have more complex composition than those in the conventional EBPR influent and that results in multiple advantages that enable more robust and stable EBPR performance over a wider range of wastewater characteristics and process configurations. The study found that GAOs, generally considered as undesirable with aerobic phosphorus uptake, were generally found at lower abundances in S2EBPR facilities than those in conventional EBPR facilities. The study also found that PAOs can outcompete GAOs with extended anaerobic conditions due to the ability of PAOs to sequentially utilize polyphosphates and glycogen for maintenance prior to their delayed decay, while GAOs' decay occurred relatively quickly following the initial consumption of glycogen. The study also mentioned that it is possible that enrichment of specific types of PAOs, such as those that can obtain energy via fermentation such as *Tetrasphaera,* occurs in the sidestream reactor of an S2EBPR system. The study concluded that additional research on the role of *Tetrasphaera* or other PAOs that are capable of fermentation is needed to better understand their role in the S2EBPR process (Gu et al., 2019).

3.5 Sidestream Enhanced Biological Phosphorus Removal (S2EBPR) at WRWTP

The configuration of the BNR facility at WRWTP is similar to that of the Westbank process (The term "Westbank" here refers to the Westbank process and not the Westbank First Nation Reserves mentioned on page 3-1). A typical Westbank process has a small pre-anoxic zone followed by an anaerobic zone, a mainstream anoxic zone, and a large aerobic zone. The pre-anoxic zone minimizes DO and nitrates entering the anaerobic portion. Primary effluent is divided among the pre-anoxic zone (to denitrify the RAS), the anaerobic zone (to provide some VFAs for phosphorus removal), and the second anoxic zone (to stimulate denitrification). VFAs obtained from fermentation of sludge (typically primary sludge) are passed to the anaerobic zone. However, at WRWTP, only a portion of the fermentate is added to the anerobic zone, and primary effluent is introduced into the mainstream anoxic zone #1 (none to the pre-anoxic zone or anaerobic zone) along with the rest of the fermentate as explained below.

As shown in Figure 3-1, 100 percent of the RAS at WRWTP is first passed through the small pre-anoxic zone (HRT of less than 20 minutes). This is followed by sidestream treatment of the RAS in the anaerobic S2EBPR zone (HRT of 1 to 4 hours) where a portion of the fermentate from a primary sludge fermenter is added. Primary effluent is introduced into the mainstream anoxic zone #1 along with the rest of the fermentate. A benefit of a S2EBPR system configuration where 100 percent of RAS is blended with a source of supplemental carbon (e.g., primary sludge fermenter overflow) in a sidestream reactor, compared to other S2EBPR configurations, is that the addition of fermentate to the S2EBPR reactor significantly shortens the HRT to 1-4 hours compared to an HRT of 16 hours or higher in other S2EBPR configurations. An additional benefit is that a portion of the fermentate can also be used for denitrification (Gu et al., 2018). At WRWTP, Alum is added upstream of the secondary clarifiers as a safety factor when needed for additional phosphorus removal.

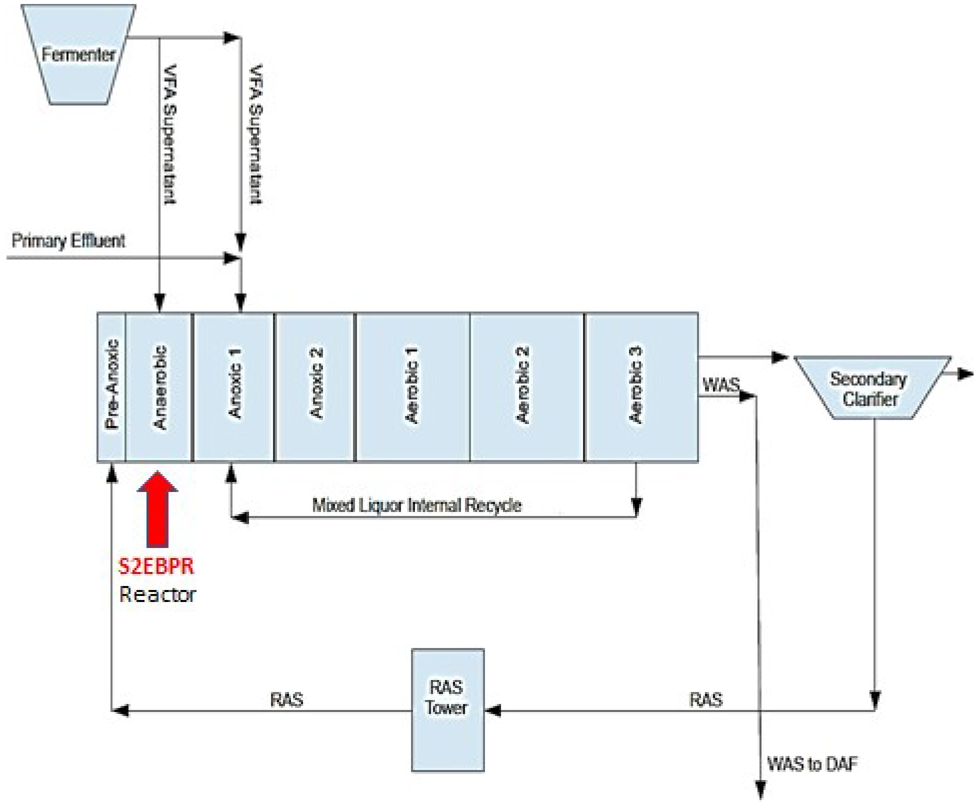


Figure 3-1. Schematic of S2EBPR Reactor and BNR System at WRWTP

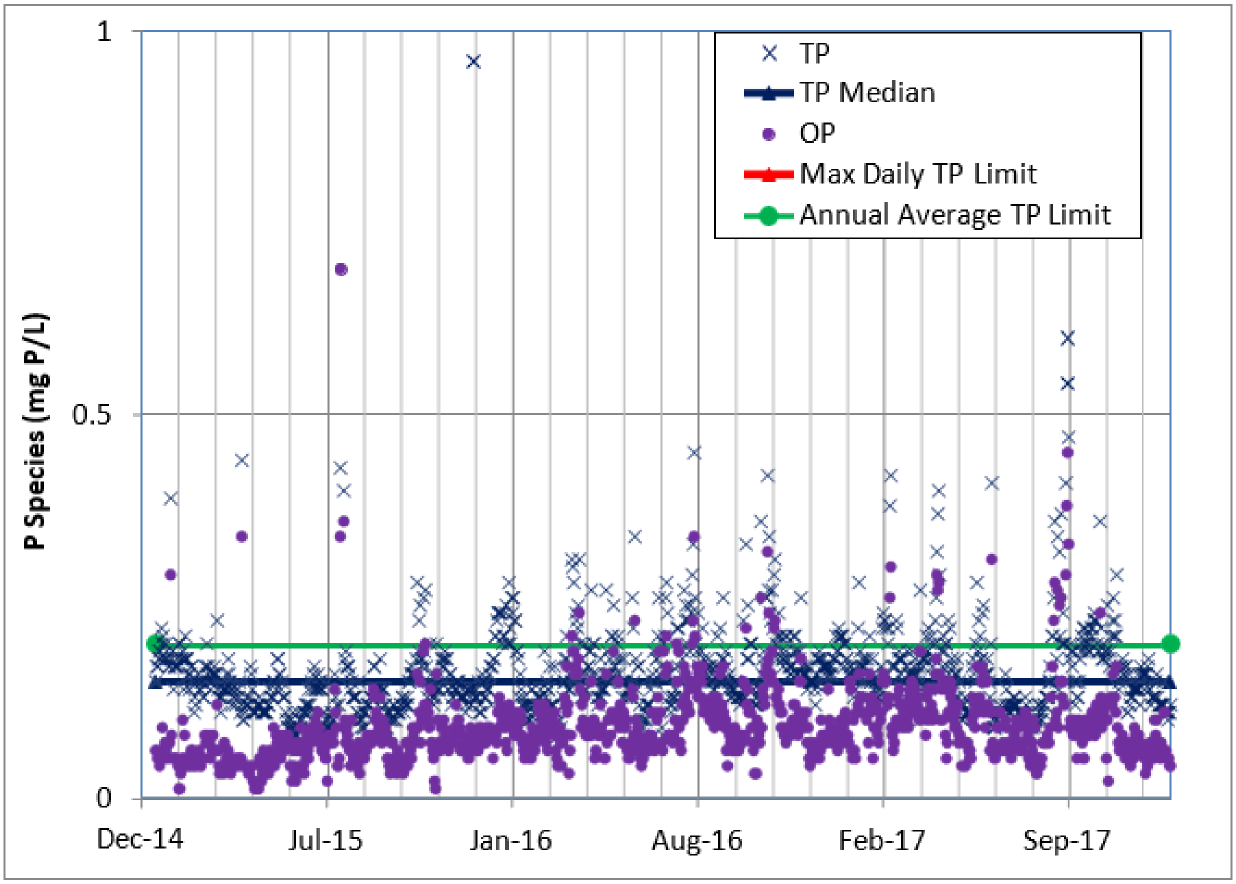
(Two anoxic zones shown here, some of the reactors have 3 but smaller anoxic zones – See Section 3.7)

3.6 Detailed Statistical Analysis – Plant Effluent Concentrations

The facility's performance was evaluated from January 2015 to December 2017. Figures 3-2 through 3-4 and Table 3-3 provide a summary of the statistical analysis performed for the WRWTP facility. The daily maximum TP effluent concentrations for 2015, 2016 and 2017 were 1.8, 0.45, and 0.6 mg/l, respectively, all below the permit limit of 2.0 mg/l. Similarly, annual average TP concentrations for 2015, 2016 and 2017 were 0.16, 0.17, and 0.17 mg/l, respectively, all below the annual average permit limit of 0.2 mg/l. Figure 3-3 shows that the facility consistently met the maximum day and annual average TP discharge. Figure 3-4 shows that the facility can consistently meet low effluent phosphorus concentrations with most of the final effluent daily TP concentrations below 0.3 mg/l (95th percentile of 0.26 mg/l, 50th percentile of 0.15 mg/l).

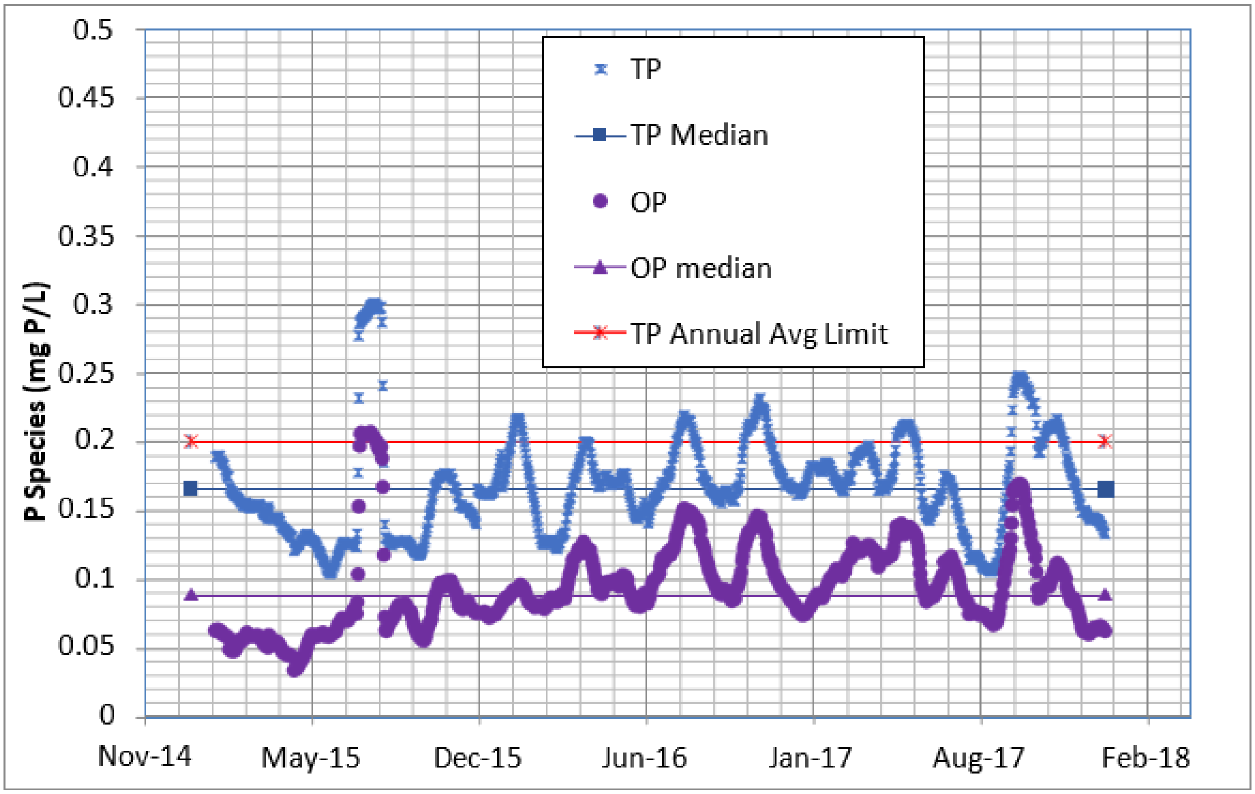
Figure 3-4 also shows that WRWTP's effluent daily median and 30-day median TP concentrations were 0.15 mg/l and 0.17 mg/l, respectively. For the entire analysis period, daily and 30-day rolling maximum TP concentrations shown on Figure 3-3 A and B were 1.80 and 0.30 mg/l, respectively. The maximum 12-month rolling average TP concentration for the analysis period was 0.181 mg/l.

The 95th percentile daily TP of 0.26 mg/l shown on Figure 3-4 is 173 percent of the median value (0.15mg/l) indicating a significant degree in variability. However, based on the above, and comparing the maximum 30-day rolling average TP concentration of 0.17 mg/l for the entire analysis periodto the 0.2 mg/l annual average shows that the treatment objective was consistently met on a rolling 30-day basis. Additionally, in looking at Table 3-3 and Figure 3-2B, it is clear that maximum values for the 30-day rolling average (0.30 mg/l) and 12-month rolling average (0.18 mg/l) are approximately equal to their respective (99 percent) probability.



A* - Daily Plant Effluent Concentrations

* Points not shown on the graph: TP and OP concentrations were 1.8 and 1.57 mg/l, respectively, on July 21, 2015, and 1.50 and 1.42 mg/l, respectively, on July 22, 2015. Max Daily TP limit is 2.0 mg/L.



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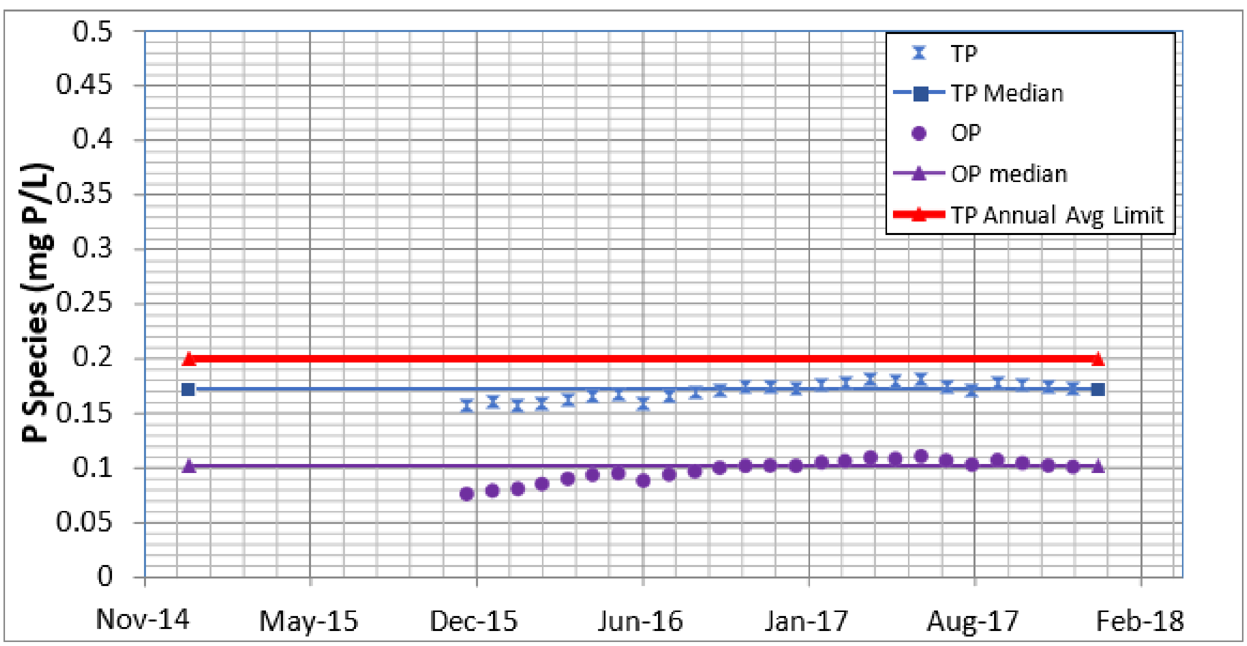

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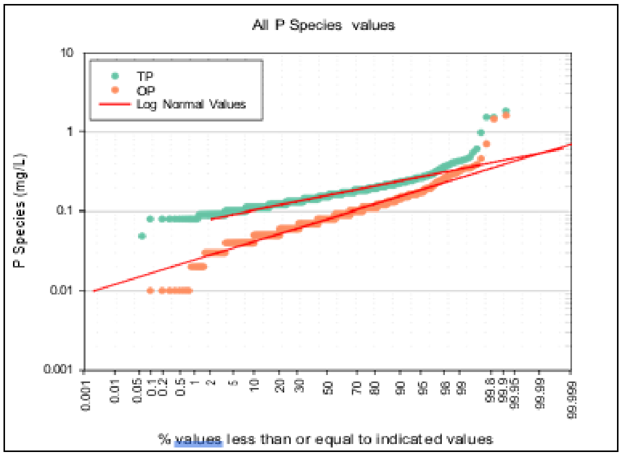
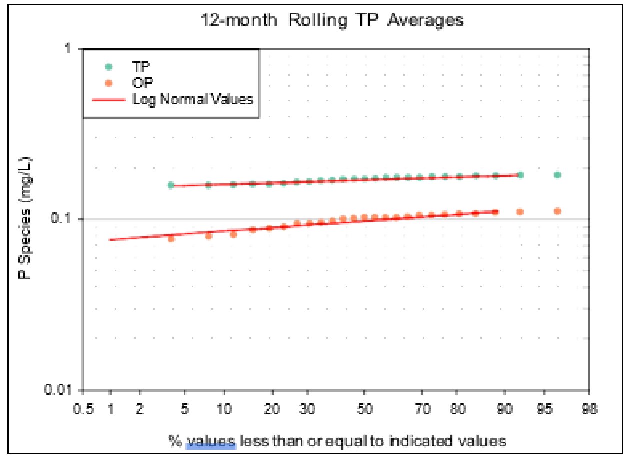
Figure 3-2. (A) Daily Times Series Plot, (B) 30-Day Rolling Average, and (C) 12-month Rolling Average Time Series Plots for WRWTP



A



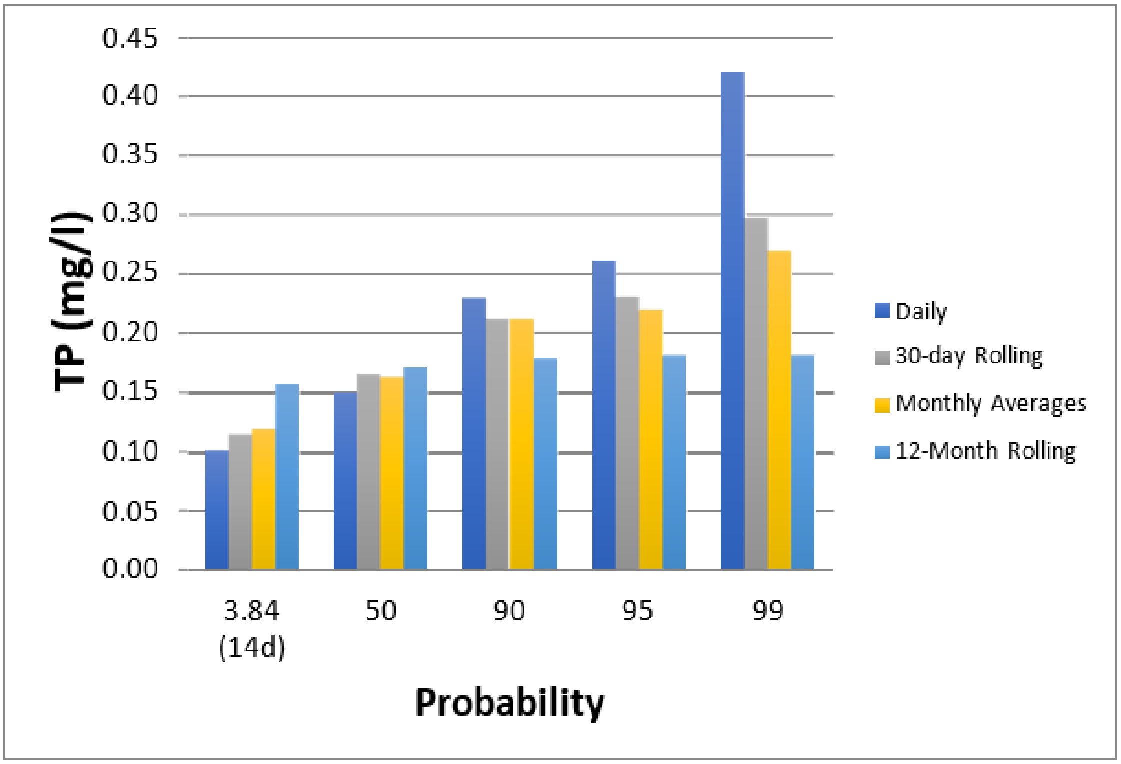
B

Figure 3-3. Probability Plots for WRWTP (A) Daily Data; (B) 12-Month Rolling Average

Table 3-3. Summary Statistics for Final Effluent TP and OP for WRWTP

	TP Daily Data	TP Rolling 30-day Average	TP Monthly Averages	TP 12-month Rolling Average
n	1094	1067	36	25
Mean	0.17	0.17	0.17	0.17
Geometric Mean	0.16	0.16	0.16	0.17
Standard Dev.	0.10	0.04	0.04	0.01
CV	0.58	0.22	0.22	0.05
Skew	9.89	0.97	1.01	-0.44
Minimum	0.08	0.10	0.12	0.16
Maximum	1.80	0.30	0.29	0.18

	OP Daily Data	OP Rolling 30-day Average	OP Monthly Averages	OP 12-month Rolling Average
n	1095	1067	36	25
Mean	0.09	0.09	0.09	0.10
Geometric Mean	0.08	0.09	0.09	0.10
Standard Dev.	0.01	0.03	0.03	0.01
CV	0.10	0.34	0.34	0.10
Skew	11.06	1.07	1.12	-0.84
Minimum	0.01	0.03	0.04	0.08
Maximum	1.57	0.21	0.20	0.11



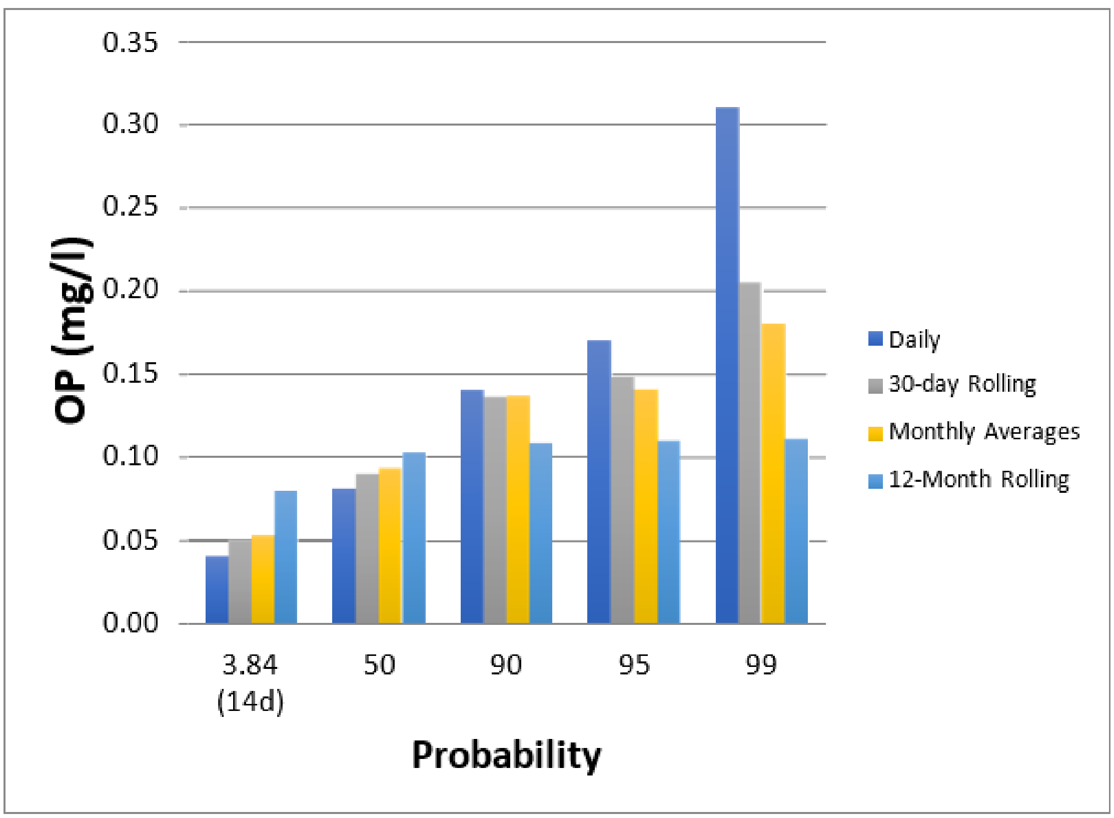


Figure 3-4. Probability Summary for WRWTP Effluent TP and OP

3.7 Process Performance – Mainstream EBPR at WRWTP

The BNR facility at WRWTP includes 6 bioreactors operated in the Westbank process mode but with only a portion of the fermentate added to the anerobic zone, and primary effluent is introduced into the mainstream anoxic zone (none to the pre-anoxic zone or anaerobic zone) along with the rest of the fermentate. One hundred percent of the RAS is passed through a small pre-anoxic reactor with an HRT of about less than 20 minutes. This is followed by sidestream treatment of the RAS in the anaerobic S2EBPR zone where a portion of the fermentate from a primary sludge fermenter is added, with an actual flow HRT of approximately 80 minutes. Primary effluent is introduced into the first mainstream anoxic zone along with rest of the fermentate. As shown in Figure 3-5, bioreactors 1 and 2 include 3 anoxic zones while bioreactors 3 through 6 include only two; however, the total volume of the anoxic zones in each reactor is approximately the same. Under normal flow conditions, only 4 bioreactors are in service.

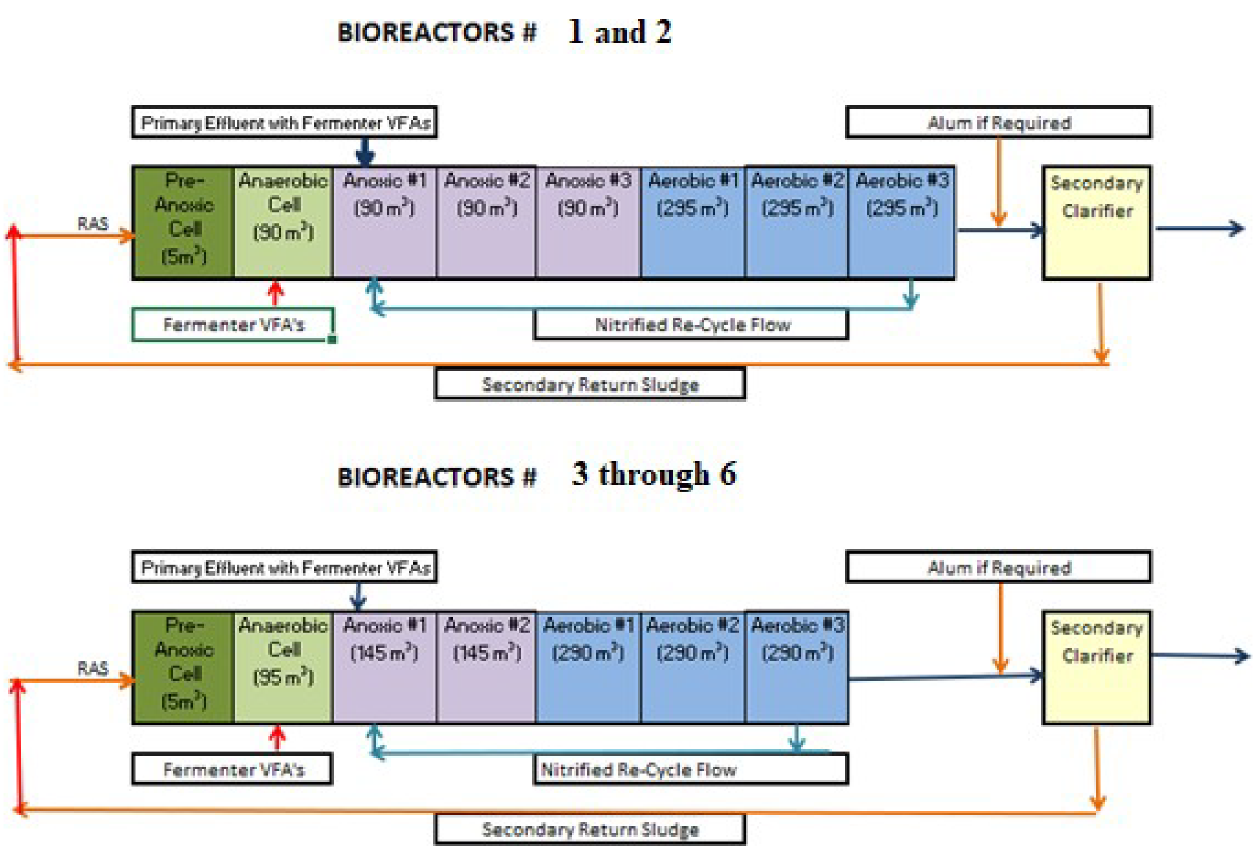


Figure 3-5. Bioreactor Layout at WRWTP

Each bioreactor is dedicated to one clarifier with individual RAS pumps, each with VFD control. A RAS collection well collects RAS from all clarifiers and is believed to result in a significant reduction in the nitrate concentration and in variability of the RAS total solids concentration fed to the bioreactors. The RAS pre-anoxic cell further reduces RAS nitrate to very low levels, typically, below 0.5 mg/l, thereby minimizing negative impacts on the anaerobic S2EBPR zone. This also allows less fermentate addition to the anaerobic zones and more fermentate available to the mainstream anoxic zone. Since additional VFA addition to the anoxic zone may potentially increase OP removal to levels that negatively impact nitrification in the aerobic zone, OP levels in the 1st aerobic zone are carefully monitored; fermenter supernatant flow to the anerobic zone is reduced if needed to ensure that sufficient OP remains to support the growth of nitrifiers and ensure compliance with total nitrogen effluent limits.

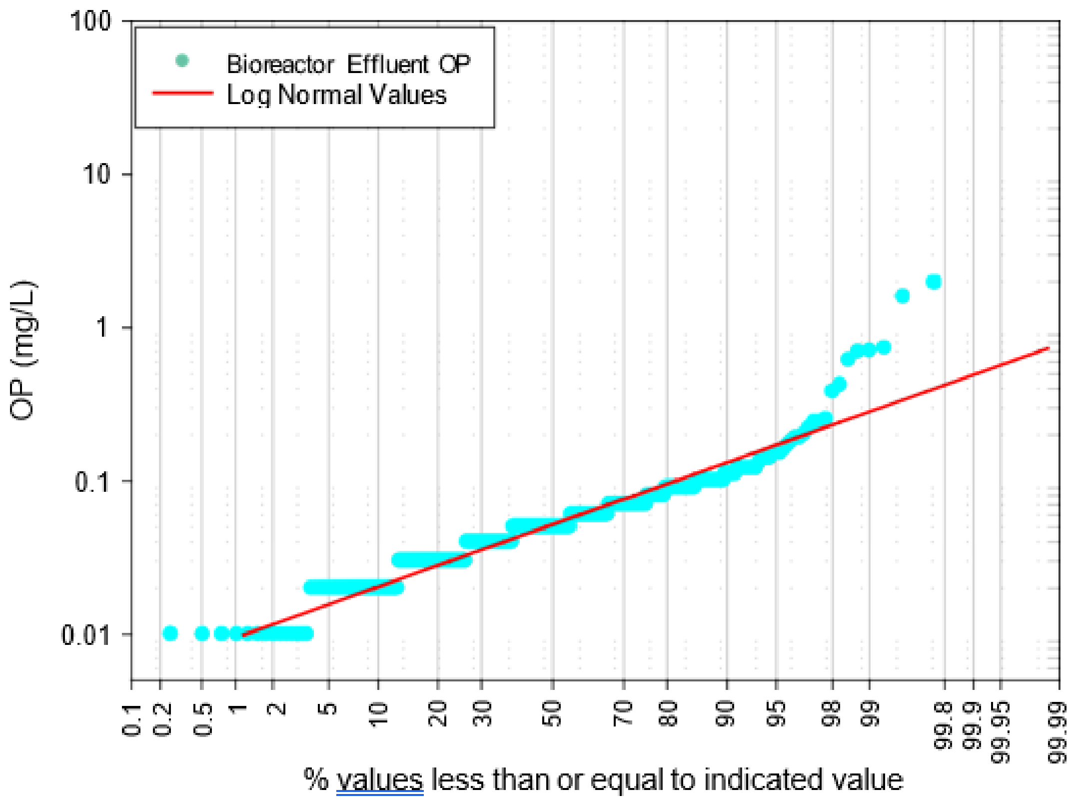


Figure 3-6. Cumulative Probability Plot for Bioreactor Effluent OP at WRWTP

Figure 3-6 shows the cumulative probability plot for bioreactor daily effluent OP concentrations before alum addition for the 3-year analysis period. As shown on this graph, the 90th percentile concentration was 0.11 mg/l OP, significantly lower than the average 90th percentile concentration of 1.6 mg/l reported in Table 3-3 for five conventional EBPR processes. This indicates that S2EBPR can result in significant improved performance compared to conventional EBPR. Additionally, Table 3-4 below shows that the ratio of the 90th to the 50th percentile bioreactor effluent OP concentrations at WRWTP was 2.2, which is significantly lower than the corresponding average ratio of 11.5 reported in Table 3-3 for the conventional EBPR processes. This indicates that S2EBPR can result in significantly improved phosphorus removal process stability and reliability. Table 3-4 also shows the frequency in meeting bioreactor effluent Ortho-P concentrations of 2.0 mg/l, 1 mg/l, and 0.5 mg/l (100 percent, 100 percent, and 99.8 percent respectively) before alum addition, all significantly higher than those reported in Table 3-2 for the conventional biological phosphorus removal facilities.

Table 3-4. S2EBPR-Enhanced BNR Reliability at WRWTP

Effluent Concentration & Percentile	Reliability
OP< 0.2 mg/l	96.8%
OP< 0.5 mg/l	99.8%
OP < 1 mg/l	100%
OP< 2 mg/l	100%
50% (Geometric Mean)	0.05 mg/l
90%	0.11 mg/l
90%/50%	2.2

Finally, it should be noted that WRWTP staff monitor bioreactor effluent orthophosphate and TP to ensure that the optimal levels of phosphorus removal are being achieved. As the plant at times experiences uncontrollable conditions including high influent flows and spikes of influent phosphorus, staff typically add alum to the bioreactor effluent ahead of the secondary clarifiers as a safety factor to ensure that the daily maximum and annual average TP limits shown in Table 3-1 are consistently met. Figure 3-7 shows that the bioreactor effluent orthophosphate concentrations before alum addition were consistently low. Daily data for the three-year analysis period shows that 90 percent of the samples were below 0.11 mg/l TP and 50 percent of the samples were at or below 0.05 mg/l. However, as Figure 3-7 shows, there were occasions where bioreactor effluent orthophosphate concentrations were above desired levels, necessitating the use of alum addition to ensure TP limits are not exceeded. It should be noted that the recirculation of alum in the RAS may also have played a positive role in improving overall phosphorus removal.

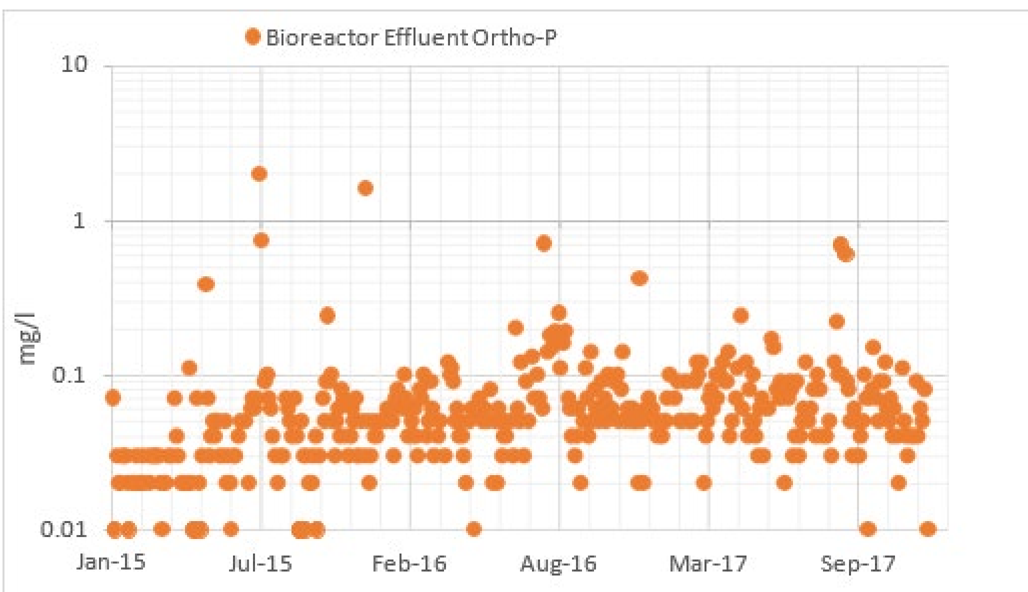


Figure 3-7. Bioreactor Effluent Ortho-P Concentrations at WRWTP

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CHAPTER 4.0
The Kingsley Wastewater Treatment Facility – Enhanced Nitrification – Submerged Attached Growth Reactor (SAGR®) - Case Study

4.1 Background

The City of Kingsley Wastewater Treatment Facility (KWTF) is a 0.13 MGD average dry weather flow facility located in Plymouth County, Iowa and discharges to the West Fork of the Little Sioux River. Prior to 2013, the City anticipated significant growth over a 20-year planning period and was expecting its new NPDES permit to include low single digit ammonia limits which could not be met with their existing lagoon treatment system, particularly during prolonged periods of low water temperatures.

Original lagoon treatment at KWTF consisted of a two-cell aerated lagoon system. Following an alternatives analysis, the City opted for upgrading the existing lagoons and retrofitting its treatment system using a Submerged Attached Growth Reactor (SAGR®) system provided by Nelson Environmental Inc. (now Nexom) of Winnipeg, Manitoba, Canada. In the summer of 2013, construction began on the facility upgrade. New lower NPDES ammonia effluent limits became effective after construction of the SAGR facility.

4.2 Plant Processes

Current wastewater treatment processes at KWTF include two aerated lagoon cells with fine bubble aeration in cells 1 and 2 and an unaerated setting zone in cell 2 which was isolated using a baffle curtain. The total depth of the aerated lagoon is nine feet, and the water depth is six feet. Lagoon effluent flows into a four-cell horizontal flow SAGR treatment system for nitrification, and an ultraviolet (UV) treatment system for disinfection prior to discharge. Figure 4-1 shows the facility's flow scheme including the SAGR system.

A lagoon fine bubble diffused aeration system was installed as part of the upgrade to achieve improved year-round BOD and TSS removal, provide stability, and optimize the SAGR process design for year-round nitrification (ammonia removal). Since the Iowa wastewater design standards did not specifically address the SAGR process, the Iowa Department of Natural Resources (IDNR) worked with the facility and its consultant to approve SAGR as an acceptable alternative technology. This involved developing alternative design criteria which included lagoon BOD treatment requirements, SAGR BOD and TSS loading requirements, minimum DO, minimum HRT, and SAGR media, liner, and insulation requirements.

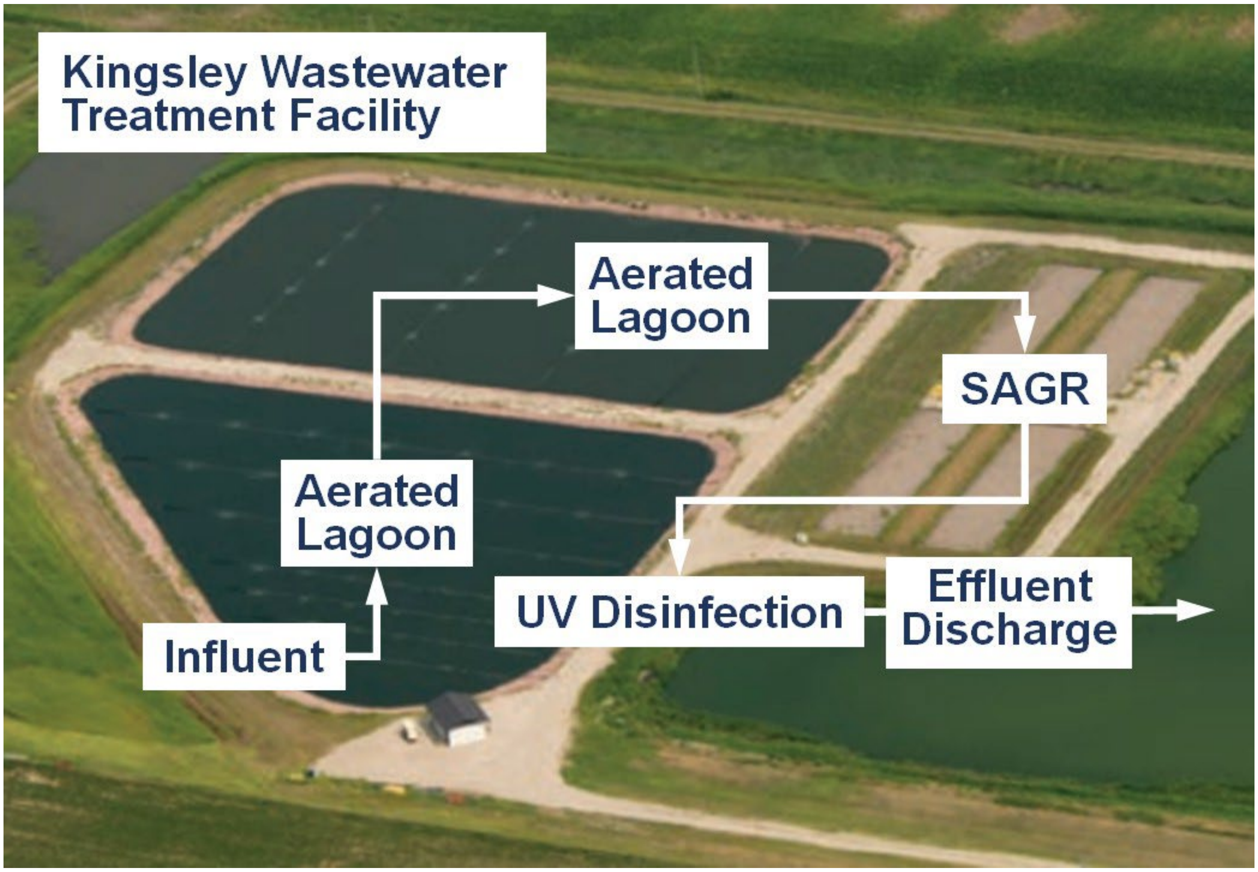


Figure 4-1. Wastewater Flow Scheme at KWTF

Table 4-1. Design and Average Raw Influent Concentrations and Percent of Design Loads for the KWTF from January 2015 to December 2017

Parameter	Raw Influent Design	Average Raw Influent	Percent of Design
Average Flow (MGD)	0.131 (Dry Weather) 0.30 (Wet Weather)	0.150[1] (Avg. for Analysis Period)	N/A
$cBOD_5$ (lbs/day)	262	N/A	N/A
TSS (lbs/day)	300	57.3[1]	19.1
Ammonia (mg/l)	N/A	N/A	N/A
TKN (mg/l)	45	N/A	N/A
TP (mg/l)	N/A	N/A	N/A
Temperature (ºC)	0.5-20[2]	N/A	N/A

Note:

1. Based on limited facility data.
2. Temperature is expected lagoon temperature, not influent temperature.

N/A: Data not available or applicable

Table 4-2. NPDES Permit Limits – January 2015 – December 2017 at KWTF

Parameter	Daily Maximum (mg/l)	Daily Maximum (lbs/day)	7-Day Average (mg/l)	7-Day Average (lbs/day)	30-Day Average (mg/l)	30-Day Average (lbs/day)
$cBOD_5$	N/A	N/A	40	63	25	100
TSS	N/A	N/A	30	75	45	113
Ammonia (January)	20.8	50.8	N/A	N/A	11.9	20.3
Ammonia (February)	9.5	22.2	N/A	N/A	9.5	22.2
Ammonia (March)	4.7	10.6	N/A	N/A	4.7	10.6
Ammonia (April)	5.2	12.3	N/A	N/A	4.4	7.8
Ammonia (May)	3.7	8.6	N/A	N/A	3.7	6.7
Ammonia (June)	3.7	8.6	N/A	N/A	2.6	4.7
Ammonia (July)	3.7	8.6	N/A	N/A	2.6	4.4
Ammonia (August)	3.1	7.1	N/A	N/A	2.4	4.0
Ammonia (September)	3.3	7.3	N/A	N/A	2.9	5.3
Ammonia (October)	3.3	7.3	N/A	N/A	3.3	7.3
Ammonia (November)	3.2	7.2	N/A	N/A	3.2	7.2
Ammonia (December)	4.5	10.4	N/A	N/A	4.5	10.4

Note:
N/A: Not applicable.

4.3 Nitrification

In many watersheds, ammonia loading to receiving waters can be a significant cause of eutrophication and/or toxicity as ammonia can be toxic to certain fish and other aquatic species, even at very low concentrations in some cases.

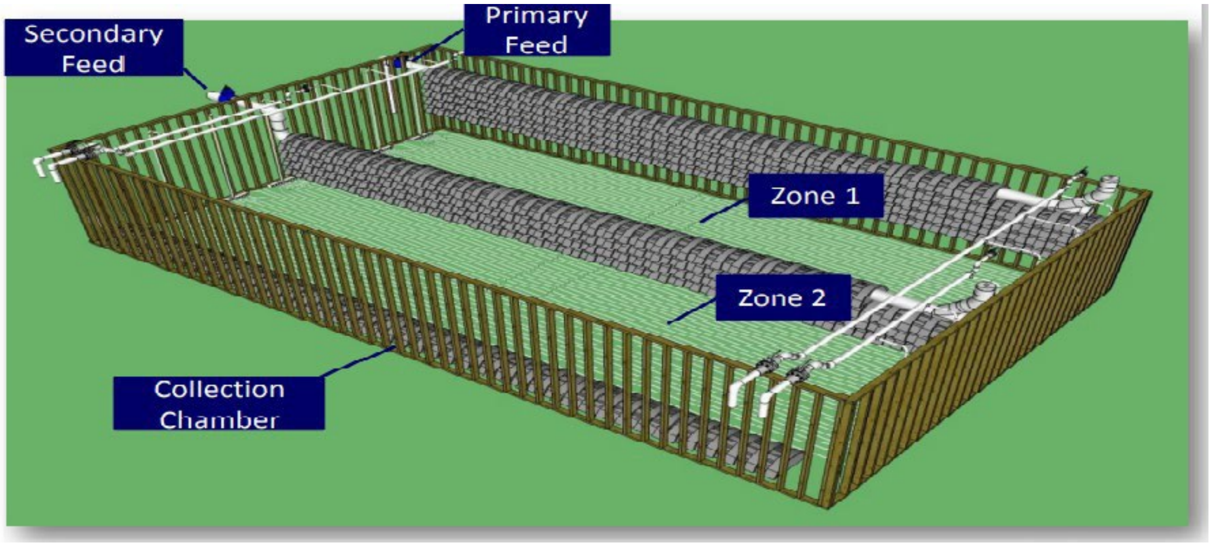
The main effective and most widely used approach for ammonia removal is the process of nitrification, which involves biological oxidation of ammonia nitrogen to nitrate nitrogen under aerobic conditions. Biological oxidation of ammonia is carried out by nitrification organisms (nitrifiers) and occurs in two steps. In the first step, an autotrophic group of ammonia-oxidizing bacteria (AOB) produce nitrite nitrogen ions as an intermediate product (Metcalf and Eddy, 2014). In the second step, another group of autotrophic microorganisms known as nitrite oxidizing bacteria (NOB) oxidize nitrite-nitrogen to nitrate nitrogen (Metcalf and Eddy, 2003).

The nitrification process has been shown to be strongly dependent on temperature and generally occurs over a range of approximately 4-45 °C, with about 35 °C optimum for *Nitrosomonas* and 35-42 °C optimum for *Nitrobacter.* Maximum specific growth rate values for *Nitrosomonas* agree reasonably with the van't Hoff-Arrhenius equation, which predicts the doubling of growth rates with each 10 °C increment in temperature (EPA,1993). Nitrification activity is significantly reduced with colder water temperatures, particularly below 8 ºC. Nitrifiers are very slow-growing microorganisms, and their growth can be inhibited by various other environmental conditions including DO, pH, alkalinity, and the presence of toxic or inhibitory compounds. In adverse conditions, nitrifier growth rates can significantly slow down or even cease leading to nitrifier washout from the treatment system.

When passive treatment systems such as some lagoons are used for nitrification, limited control may be available for consistent performance and control of important parameters such as DO and SRT. When lagoons are used for nitrification treatment in cold climates, wastewater temperatures can reach as low as 1 ºC for prolonged periods. This can significantly inhibit nitrifiers and ammonia removal. However, there is evidence that attached growth nitrification processes can achieve important rates of ammonia removal at temperatures as low as 4 ºC (Delatolla et.al., 2009). Facultative or partially aerated lagoons can be upgraded to activated lagoons by converting the lined earthen basins to aeration basins and adding secondary clarifiers and a return activated sludge system. In addition to conversion to an activated sludge system, lagoon retrofits for ammonia removal can be accomplished by post-lagoon polishing, in-lagoon polishing, and other systems. Fixed film (attached-growth) processes, such as trickling filters, biotowers, and rotating biological contactors (RBCs) may be used to remove ammonia as well as BOD if required. A combination of fixed film and suspended growth processes can also be used.

4.4 Submerged Attached Growth Reactor (SAGR®) at KWTF

As mentioned in section 4.2, the original aerated lagoon system at KWTF was upgraded and retrofitted with a SAGR in 2013. A schematic of a typical SAGR reactor is included in Figures 4-2 A and B.



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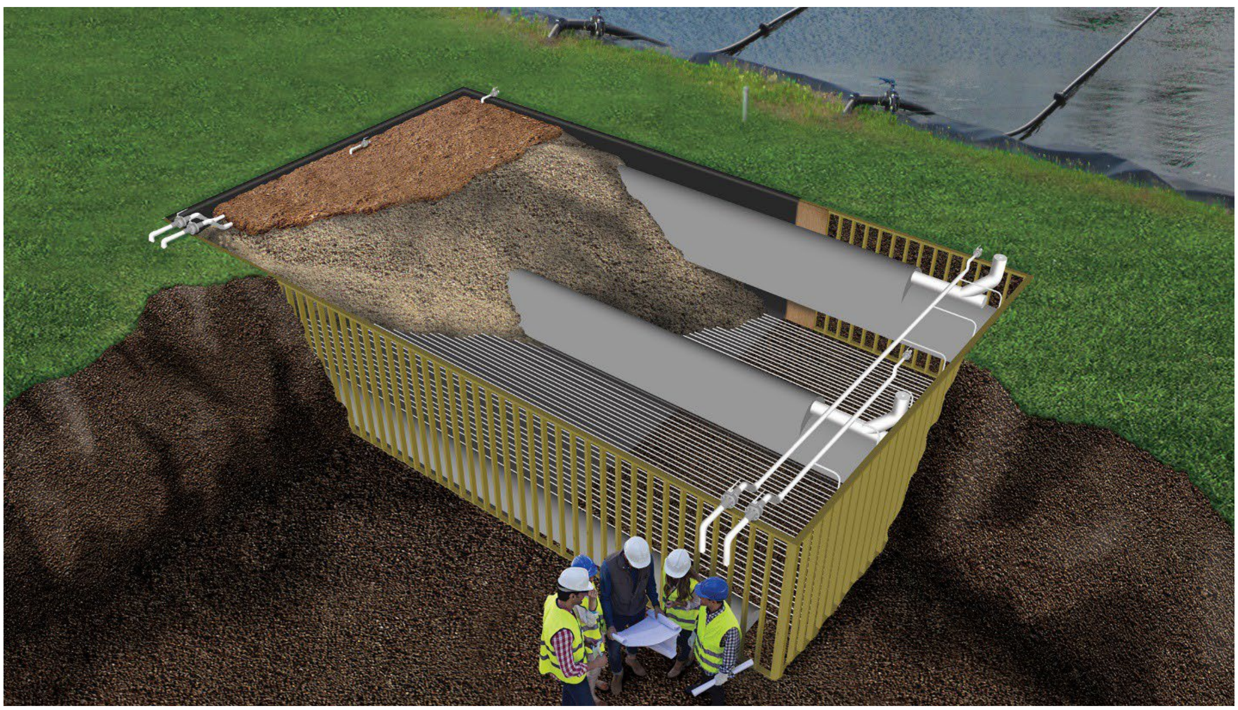


B

Figure 4-2. (A) Schematic of SAGR Reactor (B) Cut-away with air distribution (Both Schematics Courtesy of Nexom)

The SAGR® process by Nexom is a patented tertiary wastewater process that can provide nitrification during prolonged periods of cold-water temperature. The process can be implemented for nitrification following aerated or facultative lagoons. It consists of an aerated flow-through aggregate (gravel) bed with a horizontal distribution structure at the front end of the system to distribute the influent flow across the width of the cell. A linear aeration system with coarse bubble diffusers is used to provide oxygen to the sub-surface flow and enhance sludge digestion. The submerged aggregate provides the necessary surface area for growth of nitrifying biomass within the bed. The aggregate gradation was selected to balance bacterial growth area with hydraulic flow through the pore spaces. As the lagoon effluent flows horizontally through the bed, the high-DO environment encourages nitrifying bacterial growth on the aggregate surface area. A horizontal effluent collection chamber at the end of the treatment zone collects the treated effluent and channels it to the discharge structure.

SAGR aggregate bed depths vary generally from four feet to twelve feet. A layer of mulch, compost, chipped rubber tires, woodchips or other insulating material is placed on the surface of each SAGR cell as an insulation layer and for prevention of aggregate bed freezing. Since the Iowa Wastewater Facilities Design Standards did not specifically address the SAGR process, the Iowa Department of Natural Resources (IDNR) performed a technology analysis in order for KWTF to proceed with the project. Based on the IDNR review, the SAGR system was approved as an acceptable alternative technology after satisfying specific concerns by IDNR. The City of Kingsley then proceeded with one of the first installations in Iowa using the SAGR technology designed in accordance with the IDNR guidelines to meet the ammonia discharge limits listed in Table 4-2 above.

The aggregate (gravel) bed at KWTF consists of eight feet of wetted gravel media with one foot of mulch over the gravel. The aggregate gradation (Table 4-3) is selected to balance the bacterial growth area with hydraulic flow through the pore spaces. Aeration is provided throughout the floor of the reactor to maintain desired aerobic conditions required for nitrification. Based on IDNR requirements, SAGR aggregate media at KWTF was required to meet the following requirements:

Table 4-3. Gravel Media Composition

Sieve Analysis Sieve Size	Percent Passing
1½”	100
1”	80-100
¾”	30-80
½”	10-30
⅜”	0-2
¼”	0-1

Sizing of the bed is based on TKN loading rates to provide year-round nitrification needed to consistently meet ammonia discharge permit requirements. According to Nexom, the SAGR is generally designed to not exceed 0.52 lbs TKN/day per 1000 ft^3 of aggregate for systems with water temperatures below 1 °C and effluent ammonia requirements of <2 mg/l. An important design consideration for a SAGR system is the organic loading rate.

SAGR beds are designed with a minimum of two cells, with each cell including two zones in series. A shallow buried header connects blowers to the SAGR laterals. Aeration is provided through high density polyethylene (HDPE) laterals located in the top layer of the insulating mulch. HDPE drop legs provide aeration to the individual diffuser lines. Influent distribution and effluent collection chambers prevent short-circuiting in the bed which is sized to enable full cBOD polishing as well as full nitrification at cold temperatures.

The SAGR's patented step-feed process at KWTF prebuilds and stores nitrifying bacteria in October while the water is still warm so that they are already in place to compensate for the slow nitrifier growth rate at cold water temperatures in the winter. When the water temperature is warm (> 12 °C), most of the ammonia removal happens in the first zone. But as the water temperature drops, nitrifier activity slows down and more ammonia reaches the second zone for treatment. During fall before temperature drops to below 53 °F, the first zone is bypassed, and the entire influent runs only through the secondary zone (step feed operation). After approximately one month, the influent is sent back to the first zone (regular operation). Through this patented operational strategy, nitrifiers are grown in both zones of SAGR. The increase in the population of nitrifiers compensates for reduced biomass kinetics in low temperature and enables SAGR to provide effective ammonia removal during the winter. This step feeding is critical for optimizing cold temperature nitrification. Aeration remains in operation even for the zones that are not directly receiving lagoon effluent. This allows for enhanced aerobic solids digestion and minimization of any long-term fouling effects. Flow distribution is important to ensure true horizontal flow throughout the aggregate media.

Figure 4-3 below shows a layout of the upgraded lagoon system cells 1 and 2 as well as the added four SAGR cells at KWTF.

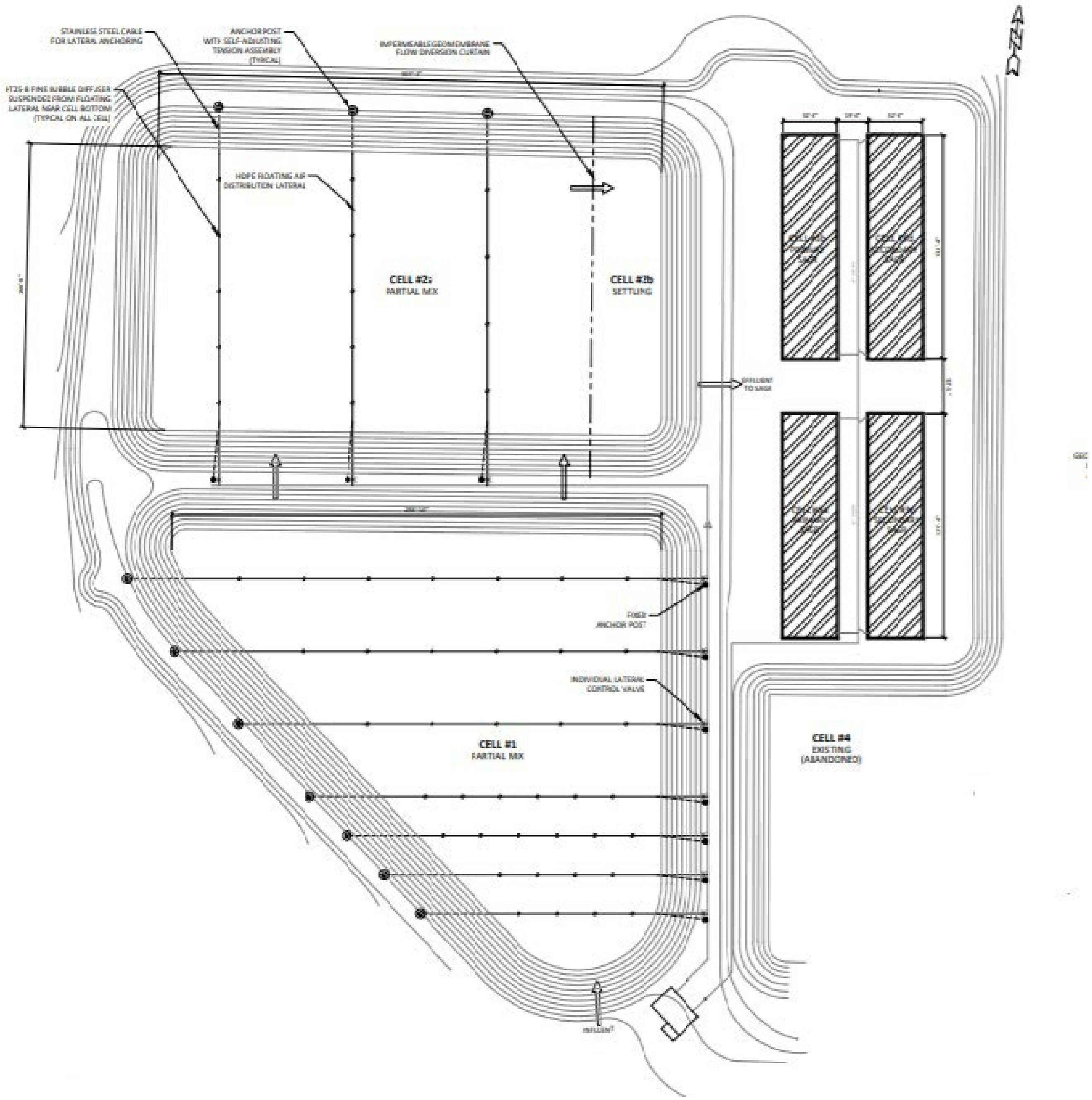


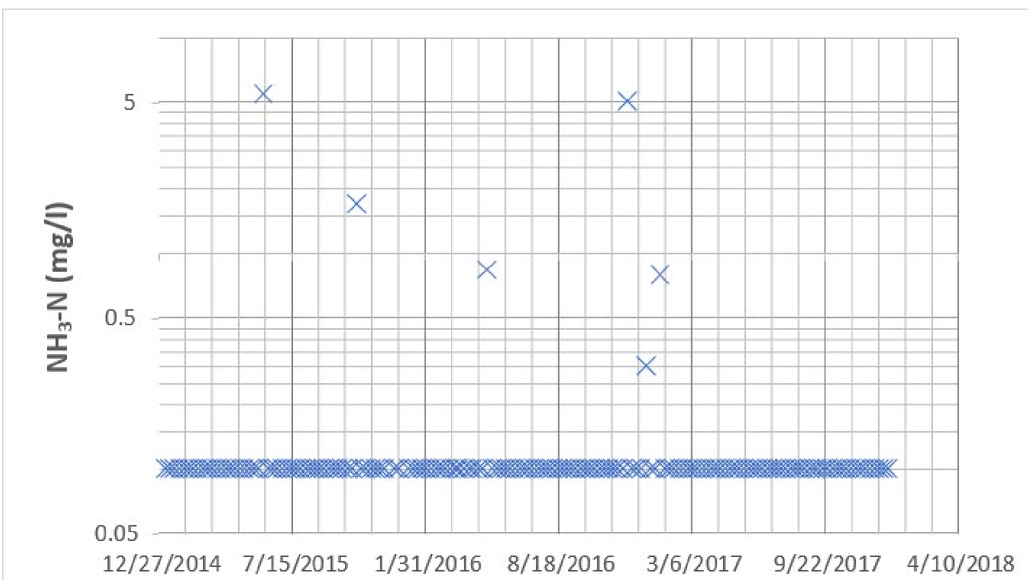
Figure 4-3. Lagoon Upgrades and SAGR Layout at KWTF
(Courtesy of Nexom)

4.5 Detailed Statistical Analysis – Plant Effluent Concentrations

The facility's performance based on operating data from January 2015 to December 2017 was analyzed. Figures 4-4 (A through C), 4-5, and 4-6 and Table 4-3 provide a summary of the statistical analysis performed for the KWTF facility. The facility has daily maximum and 30-day average effluent ammonia concentration and mass loading limits as shown in Table 4-2 and is required to collect a 24-hr composite sample once a week for effluent ammonia (with a minimum of five samples in one calendar month during each 3-month period from March 15 to November 15). Daily maximum and 30-day average ammonia concentration limits vary on a monthly basis from highs in January of 20.8 and 11.9 mg/l, respectively, to lows in August of 3.1 and 2.4 mg/l, respectively. Daily maximum and 30-day ammonia loading limits vary on a monthly basis from highs in January of 50.8 and 20.3 lbs/day, respectively, to lows in August of 7.1 and 4.0 lbs/day, respectively.

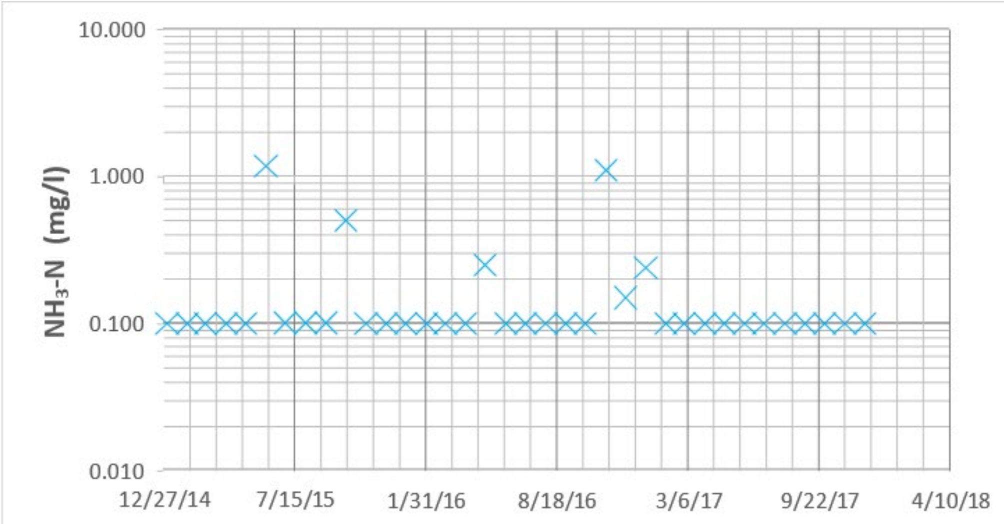
Figure 4-4 A shows that the facility produced low effluent ammonia concentrations even during cold weather months with plant effluent daily ammonia concentrations consistently below the detection limit of 0.2 mg/l except for six days during the entire 3-year period (See Section 4.6).The daily median ammonia value was below detection limit.

Figure 4-4 (B and C) shows the monthly average and 30-day rolling average effluent ammonia concentrations at KWTF. Comparing the 30-day rolling average concentrations to the 30-day ammonia discharge limits in Table 4-2 shows that the facility can consistently meet its 30-day treatment objective. The monthly average and 30-day rolling average ammonia concentrations were all below 1.5 mg/l with a median concentration below the detection limit of 0.2 mg/l.



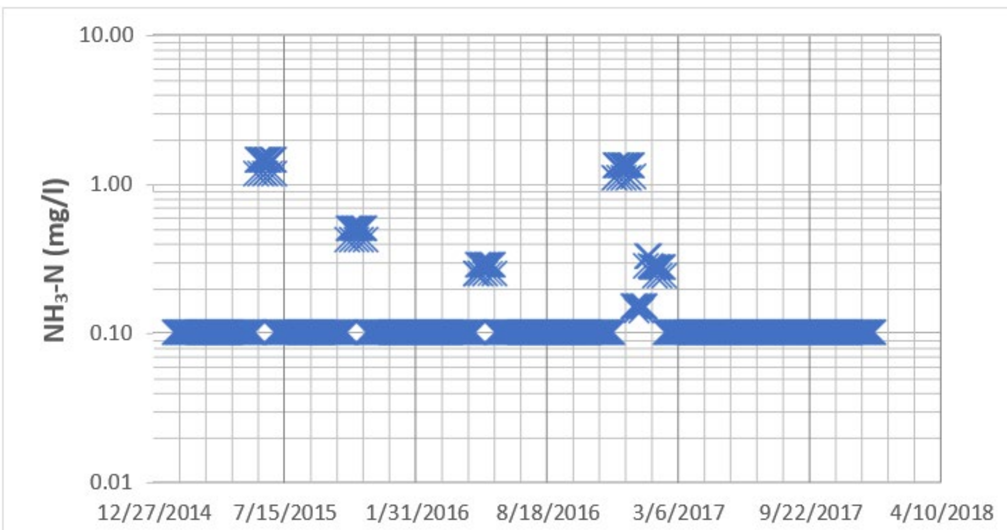
A

Daily Effluent Ammonia Time Series Plot for KWTF



B

Monthly Average Effluent Ammonia Concentrations Time Series Plot



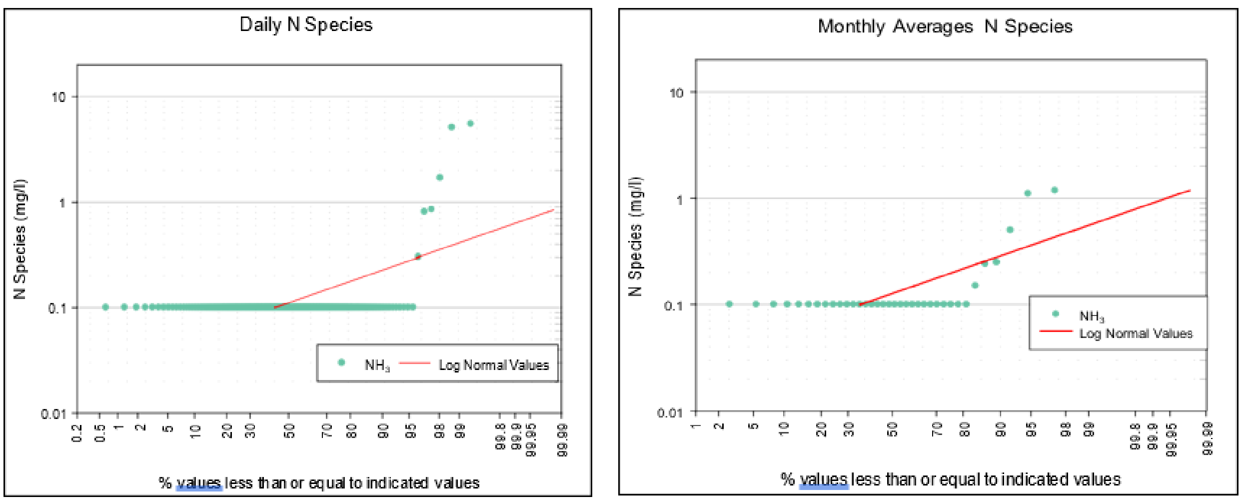
C

30-Day Rolling Average Effluent Ammonia Time Series Plot

Figure 4-4. (A) Daily and (B) Monthly Average, and (C) 30-Day Rolling Average Time Series Plots for Effluent Ammonia at KWTF

Figure 4-5 A and B represents the cumulative probability plots for daily and monthly average effluent ammonia concentrations. Figure 4-5 A and Table 4-3 show that the overall daily effluent 95th percentile ammonia concentration was below the detection limit of 0.2 mg/l (shown on the graph as 0.1 mg/l, half of the detection limit). The 99th percentile daily ammonia concentration was 5.28 mg/l. The median performance (50th percentile) was also below the detection limit. Comparing the 95th percentile concentration to the median indicates consistent achievement of low levels of ammonia at low concentrations. It should be noted that the various technology performance statistics (TPSs) evaluated in this report are affected by various wastewater, site, and technology-specific conditions and upset events. They are used in this report for describing technology performance at the facility. They can also be useful in informing the design of the process at other facilities by taking into consideration site-specific characteristics of each facility, its permit averaging period and permit limit, and how each TPS relates to permit exceedances. For example, a 95th percentile concentration, if used on a daily maximum basis, would be exceeded 91 times in a 5-year permit period while a 99th percentile concentration would be exceeded 18 times. In comparison, a 95th percentile concentration used on a monthly maximum basis would result in three exceedances in a 5-year permit period.

Figure 4-5 B shows that the effluent monthly average 95th percentile ammonia concentration for the three-year period was 1.11 mg/l while the 99th percentile was 1.15 mg/l, both well below the 30-day permit limits in Table 4-2. The median (50th percentile) monthly ammonia concentration was below the detection limit of 0.2 mg/l.



A B

Figure 4-5. Probability Plots for KWTF (A) Daily Data; (B) Monthly Average

Table 4-4. Summary Statistics for Final Effluent Ammonia Nitrogen for KWTF

	Individual Sample Data	NH_3-N Weekly Data	NH_3-N 30-day Rolling Average	NH_3-N Monthly Average	NH_3-N 12-Month Rolling Average
n	154	154	1,067	36	25
Mean	0.189	0.189	0.189	0.178	0.188
Geometric Mean	0.111	0.111	0.129	0.127	0.184
Standard Dev.	0.608	0.608	0.286	0.248	0.039
CV	3.225	3.225	1.509	1.390	0.204
Skew	8.012	8.012	3.577	3.592	-0.963
Minimum	0.100	0.100	0.100	0.100	0.112
Maximum	5.500	5.500	1.450	1.180	0.236

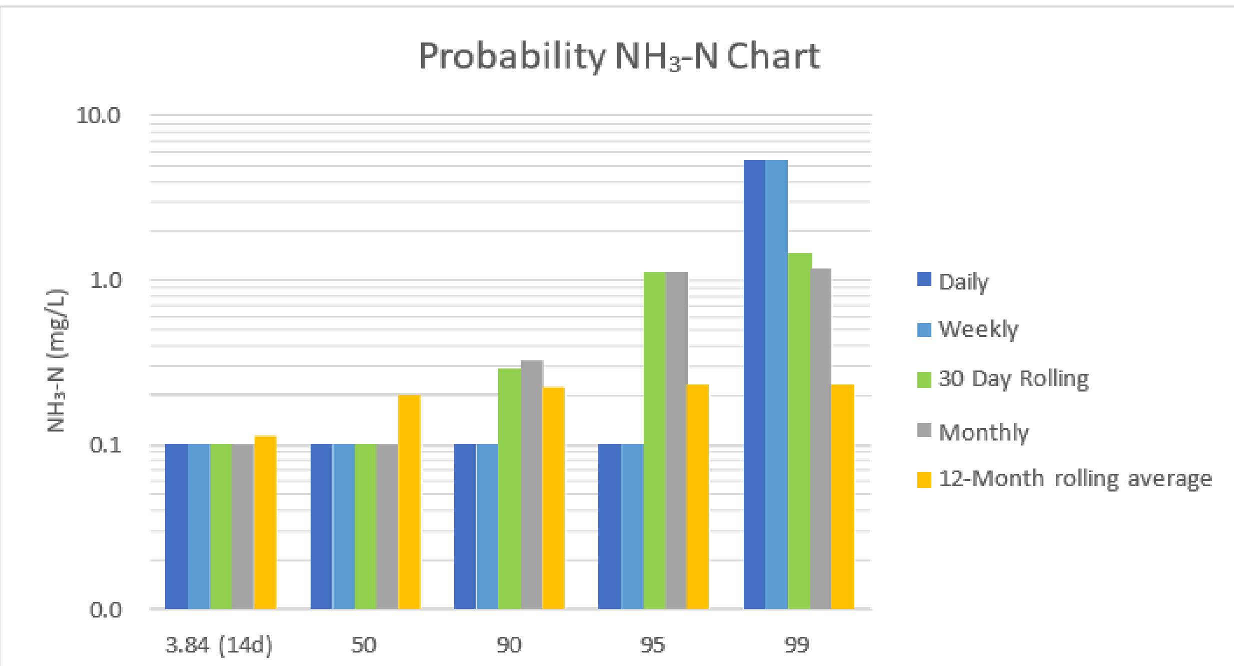


Figure 4-6. Probability Summary for KWTF

4.6 Process Performance - Submerged Attached Growth Reactor (SAGR®) at KWTF

According to the KWTF facility operator, the SAGR process is simple to operate and maintain, with minimal moving parts in the system such as the blowers supplying oxygen to the process and the aerated lagoons at KWTF. Generally, routine actions include performing a system inspection, collecting water samples, and occasionally changing lubricating oil. Dissolved oxygen and pH are measured once a week. Approximately every six months, the filters in the blower units are changed. Twice a year, the operator implements the SAGR step feed procedure by fully opening and fully closing separate valves.

Other actions include adjusting SAGR influent valves periodically and adding ammonium sulfate to SAGR influent in the fall as needed if influent ammonia levels entering the SAGR cells are very low. The facility operator also reported an incident where sulfuric acid was used to clear mineral deposits in air diffusers of a SAGR cell.

SAGR effluent ammonia data was analyzed for the 3-year period of January 2015 to December 2017 along with plant effluent ammonia concentrations. For the entire dataset received from the facility, the SAGR effluent concentrations were mostly the same as the plant effluent concentrations. For the instances where plant effluent ammonia data (after UV disinfection) was not available, the SAGR reactor effluent concentration was used. As shown in Table 4-2, the daily maximum effluent ammonia discharge limit varies on a monthly basis from a high of 20.8 mg/l in January to a low of 3.1 mg/l in August. Average monthly high and low air temperatures in Kinsley, Iowa are approximately 27 °F and 8 °F, respectively, in January and 81 °F and 61 °F, respectively, in August (source: NOAA). During the entire 3-year period, the facility consistently met its effluent daily maximum ammonia discharge limit except for 2 occasions out of 154 as explained below (the facility is required to sample its effluent for ammonia once per week). This is significant given the susceptibility of nitrification to extreme cold temperatures and other factors such as the potential presence of inhibiting constituents in the facility influent.

These excursions occurred as follows:

- June 2, 2015: Effluent ammonia concentration was 5.50 mg/l; Daily maximum limit is 3.7 mg/l. SAGR operating data do not seem to explain this effluent ammonia concentration. The facility attributed this value to likely inappropriate sampling.

- November 29, 2016: Effluent ammonia concentration was 5.1 mg/l; Daily maximum limit is 3.2 mg/l. The facility reported that in order to ensure sufficient and consistent winter nitrification during extreme cold weather and sufficient winter month biomass growth in SAGR while the wastewater was still relatively warm, the facility supplemented lagoon effluent ammonia by adding ammonium sulfate during the SAGR step feed procedure with the dosing rate based on splitting the influent flow between zones 1 and 2. However, the entire flow was passed through the two zones in series resulting in the higher effluent ammonia concentration on 11/29/2016. This was quickly corrected and reported by the operator.

On a 30-day basis, the maximum effluent 30-day ammonia limit varies on a monthly basis from a high of 11.9 mg/l in January to a low of 2.4 mg/l in August. The actual effluent monthly average ammonia concentrations were consistently below these limits with a maximum of 1.18 mg/l in June 2015 and a minimum concentration below the detection limit of 0.2 mg/l in 32 of the 36 months periods.

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CHAPTER 5.0
The F. Wayne Hill Water Resources Center (FWHWRC) – Gwinnett County, GA - Enhanced Nutrient Recovery through WASSTRIP® Phosphorus Stripping and Ostara Pearl® Nutrient Recovery – Case Study

5.1 Background

The F. Wayne Hill Water Resources Center (FWHWRC) plant is a 60 MGD advanced treatment facility located near the City of Buford, Georgia and operated by the Gwinnet County Department of Water Resources. Advanced tertiary effluent from FWHWRC is discharged primarily to Lake Lanier which serves as a recreational resource as well as an important water supply for the Atlanta metropolitan area. A portion of the effluent is also occasionally discharged to another water reclamation facility where the two effluents are combined and discharged to Chattahoochee River.

Enhanced biological phosphorus removal (EBPR) and chemical phosphorus precipitation trim are used at the FWHWRC to meet a monthly average total phosphorus (TP) limit of 0.08 mg/l. The facility's solids processing system includes anaerobic digestion of combined primary sludge and waste activated sludge (WAS). The facility also receives combined primary sludge and WAS from the 22 MGD Yellow River Water Reclamation Facility which significantly increases the phosphorus and TKN load handled at FWHWRC.

In 2009, Gwinnett County initiated the addition of magnesium hydroxide ($Mg(OH)_2$) into its collection system to control odor and corrosion. This addition significantly reduced the need for alkalinity adjustment at the plant and resulted in phosphorus precipitation from the digested sludge centrate, which decreased phosphorus recycle to the liquid treatment train and provided stability to the EBPR process. However, the phosphorus precipitated as struvite accumulated in centrate drain lines, the dewatering centrifuges, and in the digestion facility; it negatively impacted sludge dewatering capacity. With the facility experiencing increasing incidents of undesirable struvite precipitation necessitating significant maintenance efforts and resulting in negative process impacts, Gwinnett County began piloting struvite precipitation phosphorus recovery technologies in 2011. Based on the pilot results and in comparing a phosphorus recovery process versus phosphorus removal via chemical coagulant addition, Gwinnett County chose to pursue a nutrient recovery process for phosphorus recovery and controlling its phosphorus recycle loads at FWHWRC.

In 2015, FWHWRC implemented the WASSTRIP® process, which strips phosphorus from WAS, and the OSTARA Pearl® process for phosphorus and nitrogen recovery and the creation of a slow-release fertilizer (Crystal Green) under controlled conditions.This allowed combining of dewatered sludge centrate with phosphorus and magnesium-rich filtrate from sludge thickening to feed the Pearl® process, thereby increasing Crystal Green production and phosphorus recovery and further reducing struvite precipitation in undesired locations at the plant.

5.2 Plant Processes

Liquid and solids treatment processes at FWHWRC are shown in Figure 5-1.

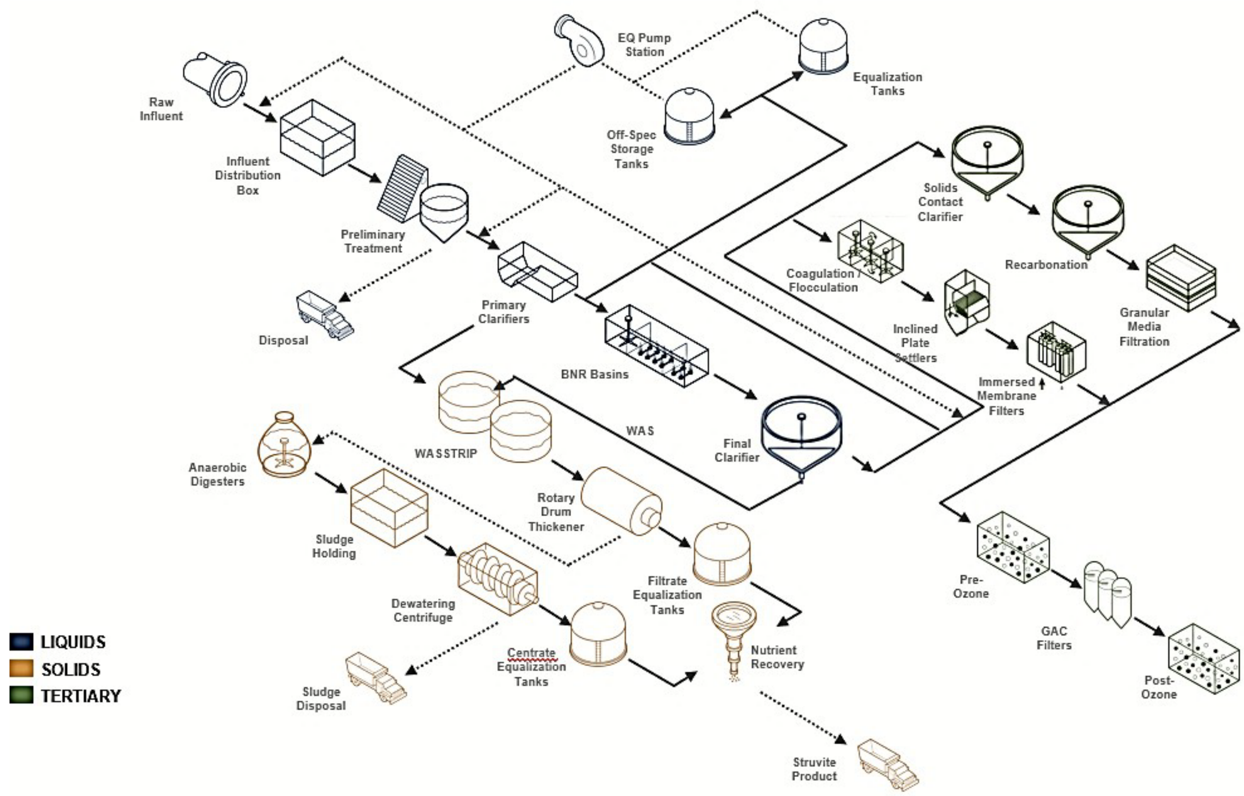


Figure 5-1. Liquid and Solids Treatment Process Train at FWHWRC

Preliminary treatment is comprised of influent screening and grit removal. Seven individual trains are used, each train employing a center-flow perforated plate band screen and a stirred vortex grit removal tank. Primary treatment occurs in ten rectangular primary clarifiers, with primary sludge pumped to the WASSTRIP tank (described below) and subsequently to the plant's solids handling system. Fermentation of primary sludge occurs naturally in the 5-7 ft blankets maintained in the clarifiers. This helps with generation of more volatile fatty acids (VFAs) that are beneficially used subsequently in the bioreactors where EBPR is implemented. Scum thickening and removal equipment are used to concentrate the scum for disposal in the facility’s anaerobic digesters.

Primary effluent flows into a biological treatment system consisting of ten plug-flow biological reactors to achieve BOD removal, nitrification, denitrification, and EBPR. These are followed by ten circular clarifiers for solids separation. Sludge is wasted from the secondary clarifiers and is pumped back to the front of the bioreactors as return activated sludge (RAS). The effluent from the secondary clarifiers flows to a secondary effluent collection box, from which flow can be routed to equalization or to the two downstream tertiary treatment processes.

Eight 20 million-gallon circular tanks are used for storage and flow equalization. Five of the tanks are for flow equalization, three are for emergency storage, and one is dedicated for off-specification flow. Primary effluent is equalized diurnally to maintain a more consistent flow and loading to the bioreactors. Secondary effluent flow can also be equalized to maintain consistent flows to tertiary treatment.

The biological treatment system at FWHWRC is operated in the Anaerobic/Anoxic/Aerobic (A^2O) mode. Low energy mixers are installed in the anoxic and anaerobic zones to maintain mixed liquor solids in suspension. Anaerobic zones allow for BOD reduction and phosphorus release. After secondary treatment, secondary effluent flow is split into two treatment trains. The first treatment train is rated for 20 MGD and includes solids contact clarifiers, recarbonation clarifiers, and granular media filtration (GMF). Alum is added right before the secondary clarifiers to provide chemical phosphorus polishing. The solids contact clarification includes four circular clarifiers for coagulation and flocculation of solids with ferric chloride. Previously used recarbonation clarifiers serve as an additional flow-through point to settle solids. The effluent then flows to multi-media dual-bed type gravity filters which are backwashed for periodic cleaning. The second treatment train is rated for 40 MGD and includes chemical coagulation, flocculation, clarification, and membrane ultrafiltration. This train also allows for ferric chloride addition to the secondary effluent. Chemical sludge is handled in the solids treatment facility as described below. Effluent from the chemical clarifiers flows to the tertiary membrane system for further treatment. The tertiary ultrafiltration membrane facility uses hollow fiber membranes and is comprised of 16 parallel treatment trains.

The effluent from both trains is combined prior to pre-ozone treatment which consists of ozone generators, sidestream ozone dissolution, contactors, and off-gassing system. The purpose of the pre-ozone system is to meet the immediate oxidation demand of the GMF and membrane effluent and also to convert recalcitrant organic compounds to bioavailable organic compounds prior to treatment in the biological activated carbon (BAC) process. This is operated as a biological process and is not used for organic pollutant adsorption. As such, activated carbon is not removed and reactivated.

Effluent then flows to the post-ozone system which operates similarly to the pre-ozone system. The primary purpose of the post-ozone system is disinfection of the BAC effluent, as much of the oxidation demand is satisfied by the pre-ozone system. The effluent pump station provides effluent pumping conveyance to Lake Lanier and the Chattahoochee River.

Solids treatment processes at the FWHWRC are also shown in Figure 5-1. Solids handling includes a primary sludge and WAS stripping, co-thickening using rotary drum thickeners, chemical sludge gravity thickeners, egg-shaped anaerobic digesters, and dewatering centrifuges. The existing facility also includes a co-generation system which includes fats, oil, and grease (FOG) and high strength waste (HSW) receiving station.

Primary sludge and WAS are combined in the WASSTRIP tank for phosphorus (and magnesium) release prior to thickening and subsequent anaerobic digestion. Digested sludge is transferred to sludge storage tanks and is then sent to the centrifuges for dewatering. Chemical sludge from the tertiary treatment train is thickened by gravity and then blended with digested sludge in the pipeline as material is being sent to the dewatering centrifuges. Dewatered cake is loaded onto trucks and hauled to a landfill and used as fill and cover material. Anaerobic digester gas handling consists of compressors, storage tanks, waste gas burners (flares), hydrogen sulfide and siloxane removal, and an engine generator that operates using the cleaned digester gas. The heat from the engine generator is recovered and utilized in the digester heating process, and the energy generated is used as electricity within the plant.

The nutrient recovery facility at FWHWRC uses the Ostara Pearl® process to recover phosphorus, minimize nuisance struvite formation in solids handling unit processes, and minimize phosphorus recycle loading to the head of the facility, while facilitating the production of inorganic struvite pellets that are beneficially used as a slow-release fertilizer. Equalized filtrate and centrate are pumped from thickening and centrifuge dewatering equalization tanks to the nutrient recovery facility.

The Ostara process has two Pearl® 2000 reactors, where optimal pH conditions are maintained using caustic addition to induce struvite precipitation. The struvite pellets are dewatered, dried, classified based on pellet size, and transferred to the storage silos. The finished product is bagged in one-ton super sacks and stored onsite for pickup for final use as fertilizer, labeled as Crystal Green®.

Table 5-1. Design and Average Raw Influent Concentrations and Percent of Design Loads for the FWHWRC from January 2015 to December 2017

Parameter	Raw Influent Design		Average Raw Influent	Percent of Design[1]
	(Annual Average)	(Maximum Month)		
Flow (MGD)	50	60	32.2	64
$cBOD_5$ (lbs/d)	109,671	131,065	61,515	56
TSS (lbs/d)	188,901	226,681	140,138	74
Ammonia (lbs/d)	13,761	16,513	8,536	62
TKN (lbs/d)	20,016	24,019	11,841	59
TP (lbs/d)	3,962	4,754	2,431	61

Note:
1- Percent of design values, except for flow, are based on average annual values for the analysis period for influent design loads (lbs./day) and actual influent loads (lbs./day).

Table 5-2. NPDES Limits - January 2015 – December 2017 at FWHWRC

Parameter	Monthly Average (mg/l)	Monthly Average (Kg/day)	Weekly Average (mg/l)	Weekly Average (Kg/day)
COD	18	2729	27	3412
TSS	3	455	4.5	569
Ammonia (mg/l)	0.4	61	0.6	76
TP (mg/l)	0.08	12	0.19	24[2]

Note:
1. The weekly average is based on FWHWRC design for a monthly phosphorus limit of 0.13 mg/l

5.3 Conventional Enhanced Biological Phosphorus Removal Technology

Enhanced biological phosphorus removal (EBPR) in wastewater treatment is accomplished by encouraging the growth of polyphosphate accumulating organisms (PAOs). PAOs are heterotrophic bacteria that occur naturally in the environment and in aerobic activated sludge. The growth of PAOs is encouraged by cycling them between anaerobic and aerobic conditions. In the presence of oxygen (i.e., aerobic conditions), PAOs obtain energy from stored food and uptake large amounts of phosphorus into their cells, which they store as polyphosphates. These polyphosphates contain high-energy bonds and function like energy storage batteries.

In the absence of oxygen (i.e., anaerobic conditions), PAOs can break the polyphosphate bonds resulting in the release of orthophosphate and use the resulting energy to uptake easily biodegradable compounds, namely short-chain volatile fatty acids (VFAs). PAOs polymerize and store the VFAs in their cells as intermediate products known as poly-β-hydroxy- alkanoates (PHAs), of which the most common is poly-β-hydroxy-butyrate (PHB). When oxygen becomes available again (i.e., aerobic conditions), they can metabolize the PHAs to generate energy and uptake phosphorus (in the form of phosphate) and store the excess amount (Randall et al., 2010).

In general, it was traditionally accepted that phosphorus could only be removed in conventional plants when the wastewater characteristics were favorable with a COD:TP ratio of at least 37:1 or a BOD:TP ratio of about 18:1, with some of the COD consisting of short chain VFAs. More COD may be required if the process also involves denitrification of nitrate (Kobylinski et al., 2008). At wastewater treatment facilities where this ratio is low, external sources of carbon such as methanol, ethanol, or proprietary carbon products could be added, and in some cases, carbon-rich waste products such as molasses, sugar wastes, or others may also be used. Additionally, many plants have elected to ferment their primary sludge. However, this often requires measures to limit odors and to ensure stable and consistent performance.

Other microorganisms besides PAOs exhibit a similar metabolism. For example, Glycogen Accumulating Organisms (GAOs) are similar to PAOs in terms of being able to store readily biodegradable organic matter such as VFAs as poly-β-hydroxyalkanoate (PHA) in the anaerobic phase (Zeng et al., 2003). GAOs do not contribute to phosphorus removal as their metabolism does not involve anaerobic phosphorus release and subsequent aerobic (or anoxic) phosphorus uptake (López-Vázquez et al., 2007). Therefore, in terms of biological phosphorus removal, GAOs are seen as competitors of PAOs for substrate and a main cause of process deterioration or even failure of EBPR systems (Thomas et al., 2003). Temperature has been identified as having a potentially significant impact on the PAO-GAO competition. At temperatures greater than 25°C, GAOs can outcompete PAOs for organic carbon (Law et al., 2016).

EBPR is considered a sustainable approach to removing phosphorus from wastewater. However, the process can in some cases be unstable particularly at low influent rbCOD/TP ratios and where other wastewater constituents such as dissolved oxygen or nitrate interfere with the anaerobic phosphorus release. To address such difficulties, particularly where low phosphorus limits are in place, many facilities incorporate added chemical polishing and incur the cost of using chemical phosphorus removal as a backup to EBPR.

In a detailed study of the long-term performance of EBPR facilities (Neethling, 2005), five EBPR facilities of various process designs, wastewater characteristics, operation, and other factors were evaluated over a three-year period to determine the biological phosphorus removal efficiency as well as their consistency of producing effluent concentrations (termed reliability in the study) at or below a given treatment goal. Table 5-3 shows the range of frequencies with which the plants achieved effluent orthophosphate (OP) concentrations of 0.5 mg/l, 1.0 mg/l, and 2.0 mg/l.

Table 5-3. EBPR Reliability at Various EBPR Facilities (Adapted from Neethling et al., 2005)

Effluent Concentration & Percentile	Reliability Range for Five EBPR Facilities	Reliability Average
OP< 0.5 mg/l	24% - 95%	68%
OP< 1 mg/l	64% - 99%	82%
OP < 2 mg/l	85% - 100%	93%
50% (Geometric Mean)	0.05 – 0.76 mg/l	0.26 mg/l
90%	0.2 – 2.5 mg/l	1.6 mg/l
90%/50%	2.0 – 24.0	11.5

The table also shows the 90^{th} and the 50^{th} percentile concentrations achieved as well as the ratio between the 90^{th} and the 50^{th} percentile (90 percent/50 percent) effluent OP concentrations. The values show that EBPR reliability is significantly reduced with lower desired effluent phosphorus concentrations particularly at lower discharge levels. Moreover, EBPR effluent concentration variability is significant as evidenced in the wide reliability range observed (90 percent/50 percent ratio range of 2 – 24) for the five facilities indicating relatively unstable performance.

5.4 Phosphorus Recovery

Nutrient removal from wastewater represents a significant resource demand for water resource recovery facilities (WRRFs). This can include costs related to consumption of electricity, organic carbon, chemicals, and sludge production, utilization, or disposal. Development of effective and economically feasible nutrient removal options is highly desirable to utilities. This is particularly the case for phosphorus removal as conventional BPR can be unstable and, if the BPR sludge is anaerobically digested and dewatered, can result in the recycle of significant phosphorus loads back into the mainstream processes thereby requiring additional treatment. The other conventional phosphorus removal alternative is chemical phosphorus removal which involves costly chemical addition and produces significant amounts of chemical sludge, particularly for phosphorus removal to low effluent discharge concentrations. From a resource recovery perspective, phosphate rock is a non-renewable resource and current global reserves may be depleted in 50–100 years (Cordell, Drangert, & White, 2009).

Extractive phosphorus recovery represents an alternative potentially attractive strategy for managing a portion of the phosphorus treated at many WRRFs. The approach generally involves using energy and resources to accumulate phosphorus and produce a nutrient product that has value in a secondary market and, if resold, can also potentially help plants offset operating costs (WERF, 2015). The attractiveness of a phosphorus recovery process to a WRRF generally depends on several factors including the loading and concentration in the influent nutrient stream, nutrient recovery efficiency, cost competitiveness with conventional treatment technology, quality of return flow to the plant's mainstream processes, and product quality and purity.

5.5 The WASSTRIP® Process

The waste activated sludge stripping to recover internal phosphate (WASSTRIP®) process is a patented process designed to release phosphates, magnesium, and potassium produced in an EBPR process prior to anaerobic digestion. The process complements and enhances other nutrient recovery processes such as Ostara's Pearl® process described below by providing magnesium and increasing struvite formation through the additional phosphorus made available. WASSTRIP anaerobically reacts primary sludge (PS) with EBPR waste activated sludge resulting in phosphorus release from PAOs. Under WASSTRIP's anaerobic conditions, PAOs in the EBPR sludge readily release stored phosphate, along with magnesium and potassium counter ions. The primary sludge and WAS are then thickened and the filtrate is blended with dewatering centrate or filtrate typically high in ammonia thus allowing increased phosphorus recovery by providing a feed stream with a higher phosphorus content to a nutrient recovery facility. The WASSTRIP process HRT depends on factors including the phosphorus content in the WAS as well as VFA availability to PAOs to allow phosphorus release. VFA availability is typically from WAS fermentation but can be added if needed, such as from a primary sludge fermenter, to enhance phosphorus release and reduce WASSTRIP HRT.

In addition, WASSTRIP can provide other benefits. WASSTRIP controls undesirable struvite precipitation in the solids treatment train by lowering the concentration and bioavailability of OP and magnesium in the digester, two of the major ingredients essential for formation of inorganic precipitates such as struvite and Newberyite (Fabiyi et al., 2016). Such undesired struvite formation in downstream solids handling processes can disrupt operations and is costly to remove and maintain. Another benefit observed by a number of facilities that use EBPR for phosphorus removal is a reduction in the volume of the produced dewatered biosolids cake due to improved dewatering. Additionally, WASSTRIP can in some cases benefit certain facilities that land apply their biosolids by reducing biosolids phosphorus content and potentially increasing availability of sites for land application.

5.6 The Pearl® Process

The Pearl® process is a patented nutrient recovery process by Ostara that recovers phosphorus from nutrient-rich wastewater filtrate and/or centrate through the controlled precipitation of struvite. The Pearl reactor is an up-flow fluidized bed reactor designed to maximize nutrient recovery and production of a high-quality fertilizer. The process includes optimized reactor geometry, flow management, and process control of chemical addition such as soluble magnesium for ionic concentration adjustment and sodium hydroxide for pH adjustment. The influent and chemicals are introduced at the bottom of the reactor, where struvite crystal formation begins. Treated effluent is discharged from the top of the reactor and returned to the mainstream for further treatment. A portion of treated effluent is returned to the bottom of the reactor for product size control and as needed for influent flow variations. The product is dewatered, dried, sorted by size, and bagged or optionally stored in silos as high-purity struvite pellets prior to distribution and sale directly from the facility as a slow-release fertilizer branded Crystal Green®.

5.7 The WASSTRIP® and Pearl® Processes at FWHWRC

Nutrient recovery at FWHWRC is accomplished by using the WASSTRIP® process and the Pearl® process to mitigate nuisance struvite formation in solids handling unit processes and to minimize phosphorus recycle to the head of the facility, while facilitating the production of inorganic struvite pellets that are beneficially used as a slow-release fertilizer. FWHWRC began adding magnesium hydroxide into the collection system in 2009 to control odor and corrosion. As a result, P precipitation from the digester centrate decreased P recycle loads returned to the bioreactors allowing for the stabilization of the EBPR process. However, this precipitation restricted flow in the centrate drain lines and reduced centrifuge dewatering capacity. After evaluating several alternatives, the facility selected the OSTARA Pearl® nutrient recovery process with WASSTRIP® process phosphorus stripping. The main project components included WASSTRIP, centrate and filtrate equalization tanks, transfer pumps, Ostara Pearl nutrient recovery reactors and chemical feed systems, product handling system, and process control. A process flow diagram is provided in Figure 5-2.

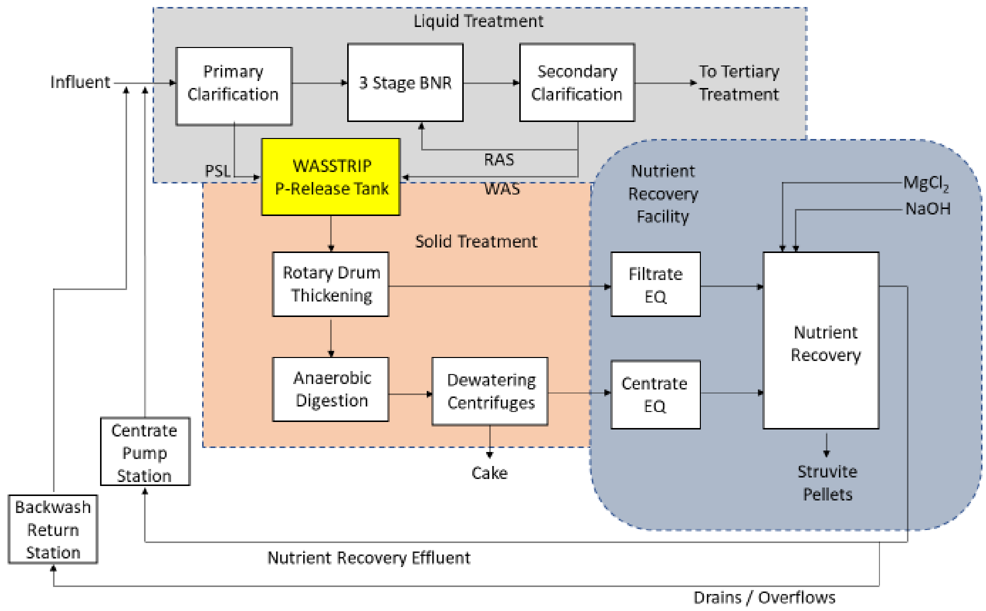


Figure 5-2. Liquid and Solids Treatment Process Train at FWHWRC (Adapted from Latimer et al., 2017)

Primary sludge (PS) and WAS are pumped to the WASSTRIP tank and blended in a constant level tank, and the blended sludge is mixed with a large bubble mixing system and allowed to react anaerobically for at least 6 hours. From there, combined sludge from the WASSTRIP process is thickened in rotary drum thickeners (RDTs). The filtrate, rich in phosphate and magnesium, is stored in an equalization tank and fed along with flow from centrate equalization tanks to the nutrient recovery facility. The equalization tanks were designed to allow heavy solids to settle and automatically drain and wash down. Filtrate and centrate equalization tanks, as well as one swing tank are 500,000 gallons each.

In order to reduce the potential for struvite accumulation in centrate lines, the facility installed PVC centrate pipes with removable sections in the centrifuge dewatering building as well as parallel HDPE centrate pipes into the recovery facility. In addition, an acid feed loop was installed to allow cleaning of the feed pipes as needed.

The Ostara process consists of two Pearl ® 2000 reactors with space for a third reactor in the future. Each reactor has a nominal capacity of 4,400 pounds of daily struvite production. Optimal reactor pH conditions are maintained using caustic addition to induce struvite precipitation at a target pH of 7.8. The struvite pellets are dewatered, dried, classified based on pellet size, and transferred to the storage silos. The finished product is bagged in one-ton sacks and stored onsite for pickup and final use as fertilizer, labeled as Crystal Green ®.

5.8 Detailed Statistical Analysis – Plant Effluent Concentrations

Facility operating data from January 2015 to December 2017 were analyzed. Figures 5-3 through 5-5 and Table 5-3 provide a summary of the statistical analysis performed for the FWHWRC. As explained below, the data shows that the facility consistently met the final effluent treatment objectives for total Phosphorus shown in Table 5-2 throughout the analysis period.

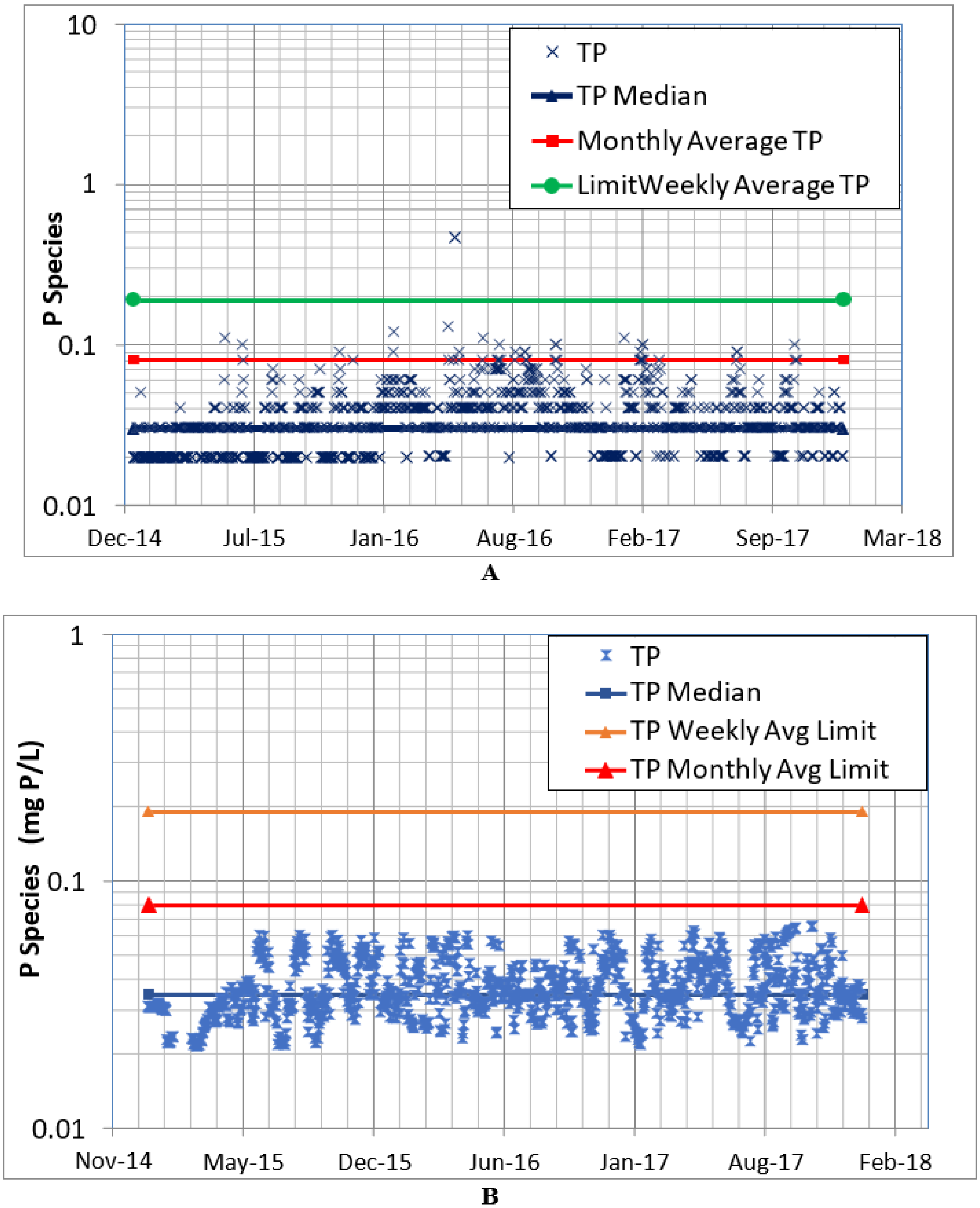
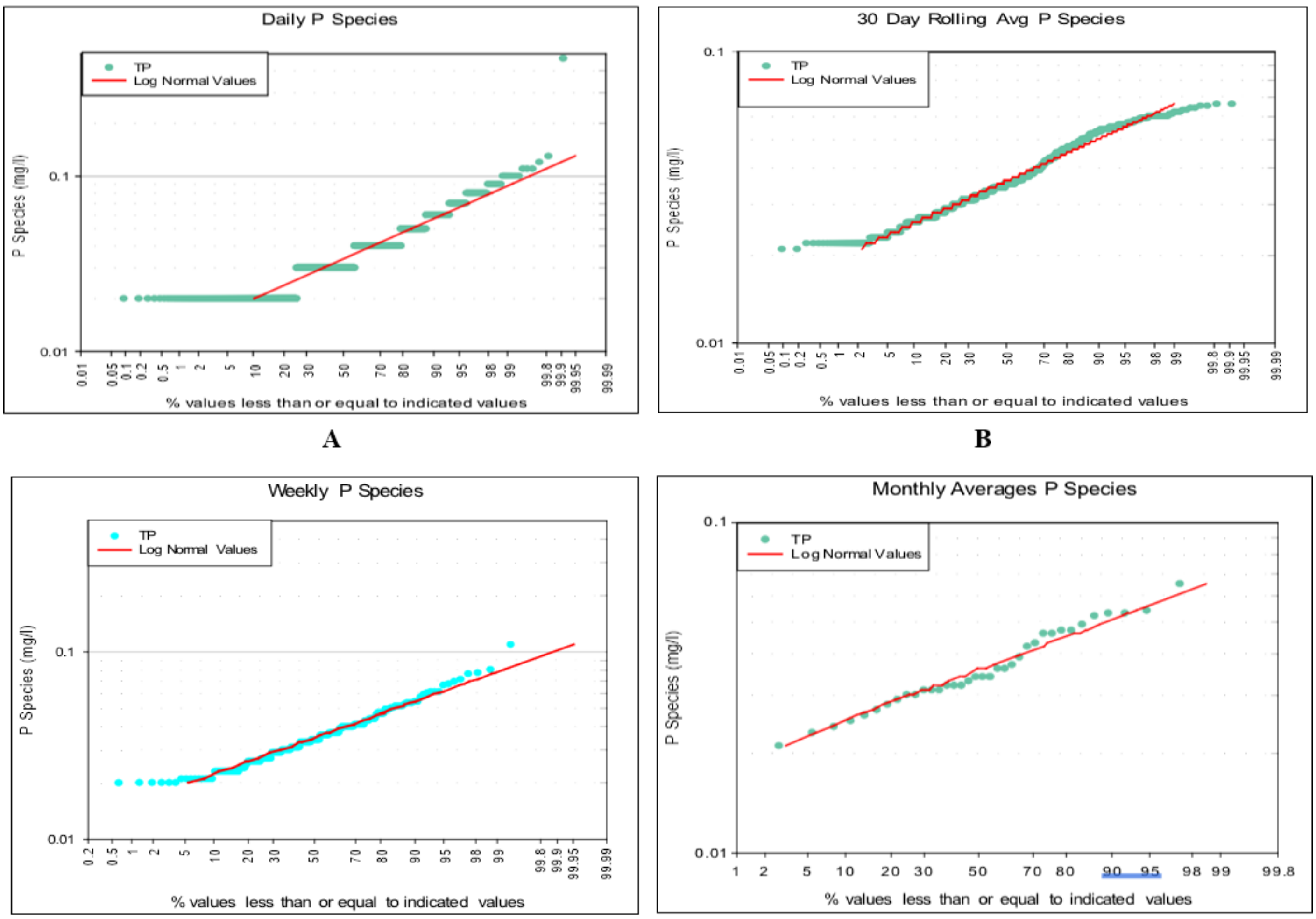


Figure 5-3. (A) Daily and (B) 30-Day Rolling Average Time Series Plots for Effluent TP atFWHWRC

Figures 5-4 A through D include cumulative probability plots for FWHWRC's daily, 30-day rolling average, weekly average, and monthly average data sets. A percentile value on the x-axis represents the probability that the value is less than or equal to the stated corresponding concentration on the plot's y-axis. Figure 5-4 C shows that the weekly average 99 percentile TP concentration was 0.08 mg/l. All 156 weekly average effluent concentrations reported were below the weekly average discharge limit with a maximum value of 0.11 mg/l. Similarly, on a monthly basis, Figure 5-4 D shows that the monthly average 99 percentile TP concentration was 0.06 mg/l. All 36 monthly average effluent concentrations reported were below the TP monthly average discharge limit of 0.08 mg/l, with a maximum value of 0.065 mg/l.



A B C D

Figure 5-4. Probability Plots for FWHWRC–
(A) Daily Data; (B) 30-day Rolling Average; (C) Weekly Average; (D) Monthly Average

Figure 5-5 provides a probability summary for TP effluent concentrations at the 3.84, 50, 90, 95, and 99 percentiles for the various data sets (daily, 30-day rolling, weekly average, monthly average, and annual average). Based on the three-year data set, the daily average concentration at the 95th percentile (as an example) is 0.07 mg/l TP, while at the 50th percentile (median), the concentration is 0.03 mg/l.

Figure 5-5 also can highlight the variability for TP concentrations. Comparing the daily data 14-day (3.84th) percentile of 0.02 mg/l for TP to the 95th percentile of 0.07 mg/l, the 95 percent/3.84 percent is about 3.5 demonstrating low variability.

Table 5-4. Summary Statistics for Final Effluent Total Phosphorus for FWHWRC

	TP Daily Data	TP Rolling 30-day Average	TP Weekly Averages	TP Monthly Averages	TP Annual Average
n	1,081	1,067	156	36	25
Mean	0.04	0.04	0.04	0.04	0.04
Geometric Mean	0.03	0.04	0.03	0.04	0.04
Standard Dev.	0.02	0.01	0.01	0.01	0.00
CV	0.57	0.27	0.38	0.28	0.13
Skew	8.61	0.69	1.58	0.73	-0.44
Minimum	0.02	0.02	0.02	0.02	0.03
Maximum	0.47	0.07	0.11	0.06	0.05

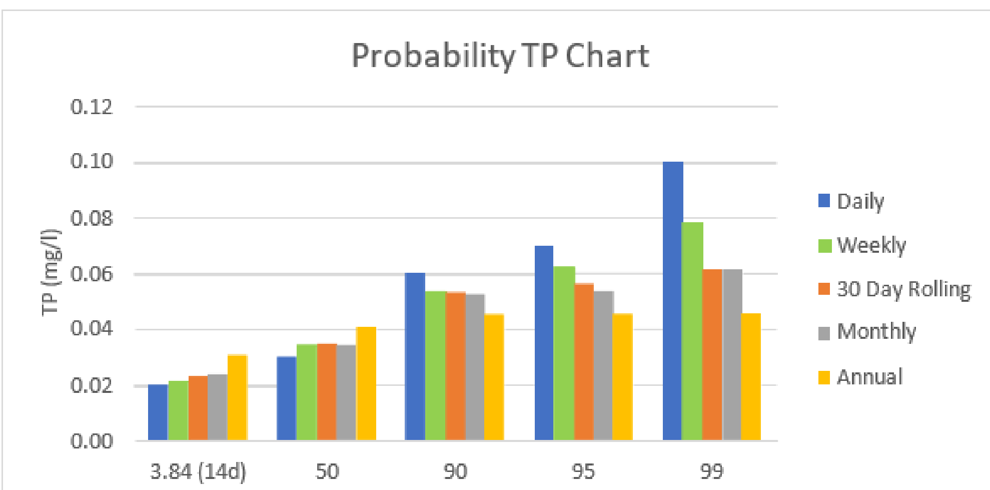


Figure 5-5. Probability Summary for FWHWRC

5.9 WASSTRIP® and Ostara Pearl® Process Performance & Lessons Learned

As mentioned in Section 5.7, The FWHWRC facility implemented the OSTARA Pearl® nutrient recovery process with WASSTRIP® process phosphorus stripping. The OSTARA Pearl® facility came online on July 6, 2015. Figure 5-6 shows the OP percent recovery in each of the two reactors of the OSTARA Pearl facility. The percent recovery values ranged from about 40 percent to 83 percent. The initial lower OP recovery values before December 2016 were attributed to a number of operational factors as explained below. Additionally, the facility reported lower initial total phosphorus recovery due to struvite fines production and loss to the overflow. Struvite fines as small particulates, generally described as less than 0.5 millimeters, are spontaneously precipitated in supersaturated conditions.

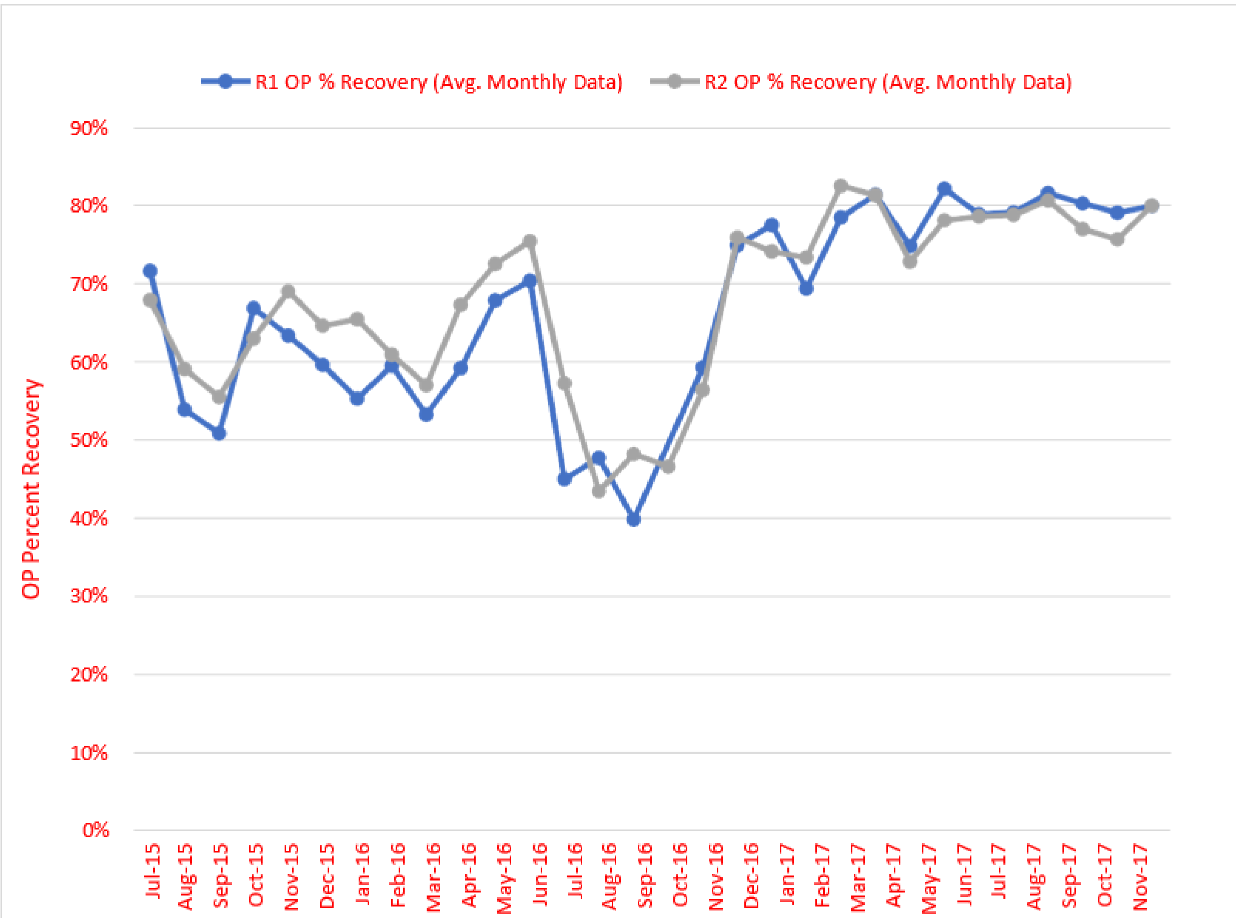


Figure 5-6. Orthophosphate Percent Recovery

In order to address the initial low percent OP removal and the fines production problem, the facility implemented various modifications in the operational pH setpoint and in the product harvesting procedure. This work included optimizing the type, frequency, and mode of seeding, and routine replacement of pH probes every six months. The facility also implemented periodic citric acid cleaning to prevent struvite clogging of pumps and pipes. Additionally, since the WASSTRIP RDT thickening facility operates seven days a week while centrifuge dewatering does not occur on weekends, procedures were implemented to ensure centrate tanks are full ahead of the weekend so as to maintain desired filtrate-to-centrate ratios in the feed to the Ostara facility.

Table 5-5 shows the average daily gross and net production of the fertilizer product at the Ostara facility during the analysis period of January 2015 through December 2017. The process during this period was operating at well less than the nominal capacity of a single reactor. As shown, the facility was able to increase net daily production in 2016 and to almost 1,700 pounds per day in 2017 due to operational enhancements as plant operators gained further experience with operating the nutrient recovery facility and its process-specific requirements.

Table 5-5. Ostara Pearl® Production Data

Operating Period	Average Daily Gross Production*, lb/d	Average Daily Net Production*, lb/d
Startup – Dec 2015	1,416	601
2016	1,116	800
2017	1,978	1,682

*Average daily production values reported by FWHWRC.

Figure 5-7 shows the bioreactor effluent OP concentrations before and after Ostara Pearl process startup. As mentioned previously, FWHWRC utilizes EBPR and chemical polishing to meet a low effluent total phosphorus limit of 0.08 mg/l.

Methods of nutrient recovery such as the use of the Ostara Pearl® process through controlled harvesting of struvite normally can help in the reduction of phosphorus recycle loadings to the mainstream BNR process, thereby improving biological phosphorus removal performance and reducing the need for supplemental chemical phosphorus removal. However, as shown in Figure 5-7, the BNR facility experienced an initial increase in effluent OP concentrations in the effluent from the BNR reactors.

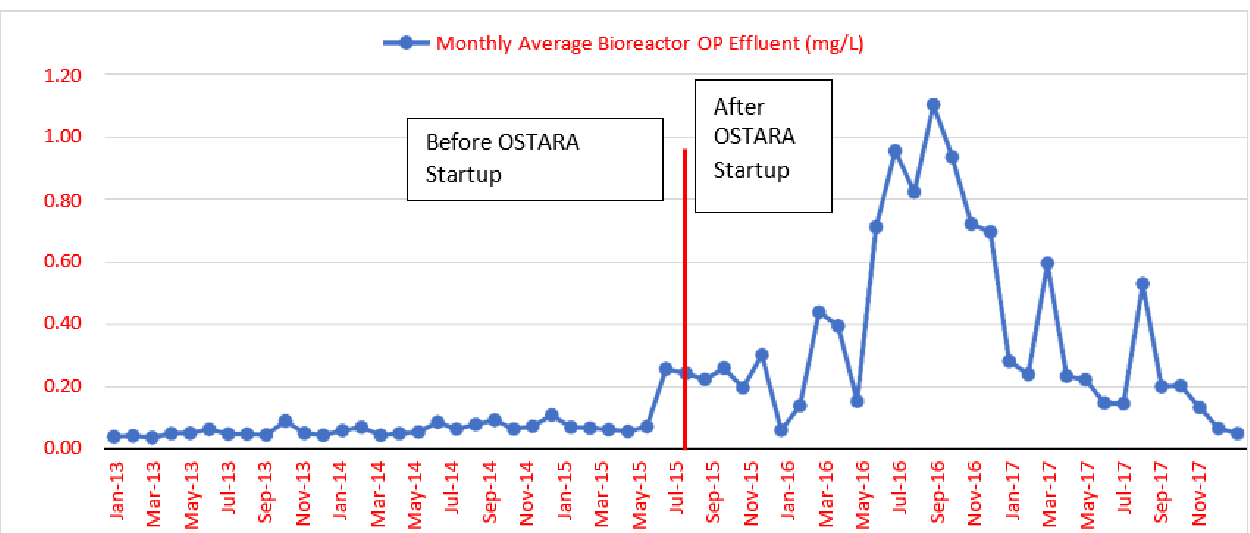


Figure 5-7. Monthly Average Bioreactor OP Effluent

FWHWRC indicated that the initial performance of the nutrient recovery facility startup had a negative impact on mainstream EBPR performance due to the facility's low recovery of particulate TP after its startup. Based on a strong correlation observed between the reduction in EBPR performance, stability, and the facility's reduced TP recovery, the EBPR performance was attributed by FWHWRC mainly to the struvite fines loss after startup of the nutrient recovery facility. The loss of struvite fines is a significant issue since it can dissolve upon return to the head of the plant thereby increasing the phosphorus load to the biological reactors and resulting in higher bioreactor effluent concentrations. This resulted in the sudden breakthrough of OP in the reactor effluent, with average monthly OP concentrations fluctuating between approximately 0.4 mg/l and 1.1 mg/l between February 2016 to August 2017 as shown in Figure 5-7. These periodic episodes of reduced EBPR performance in turn resulted in increased reliance on metal salt addition to the bioreactor effluent and upstream of the tertiary treatment train as explained below.

In order to address this problem, the facility implemented a 7-month field optimization effort in 2017 to improve EBPR performance and reduce metal salt addition, especially during periods of reduced nutrient recovery performance. This effort focused on better understanding of the impact of several operational variables on EBPR performance in the bioreactors including the impact of the internal nitrified recycle (NRCY) operation and the bioreactor configuration that would be less susceptible to upset conditions. Results of the testing showed that higher DO from NRCY streams, higher nitrate back to the anoxic zone, along with back mixing when they entered anaerobic zones, negatively impacted phosphorus release. They also showed that higher secondary clarifier sludge blankets may have resulted in secondary phosphorus release. Additionally, testing showed that RAS short-circuiting and back mixing of flow from aerated zones to unaerated zones also negatively impacted performance.

Based on the data collected in the optimization effort and available historical efforts, changes in the bioreactor configuration and in the zone receiving NRCY were implemented on all bioreactors. This was done to allow proper anaerobic conditions for EBPR and maximize anoxic volume for denitrification. Additional secondary clarifier sludge blanket control measures were also implemented. As a result of these measures, EBPR performance improved significantly starting in August 2017 as shown in Figure 5-7. Alum addition also improved as shown below. However, the main driver for the reduced alum addition at the secondary process was to maximize the bioavailability of OP in the WAS, and a subsequent increase of ferric dosing at the tertiary processes was implemented to remove any OP remaining in the BNR effluent. Facility staff also realized that while implementing these measures would improve performance, limited periodic EBPR upsets are likely to occur in the future, and an additional treatment step for the recycle stream from the nutrient recovery facility may be needed. A complete description of the optimization effort, including testing and results, are illustrated in the paper by Mohan et al., 2018.

Figure 5-8 shows the alum addition to the secondary clarifiers and the Ferric chloride addition at the tertiary process before and after OSTARA start up in July 2015. Notwithstanding periodic fluctuations in Alum addition experienced in the months following startup of the Ostara facility, the monthly average alum addition after OSTARA startup (July 2015 through December 2017) was lower by approximately 60 percent compared to the period between January 2013 through July 2015.

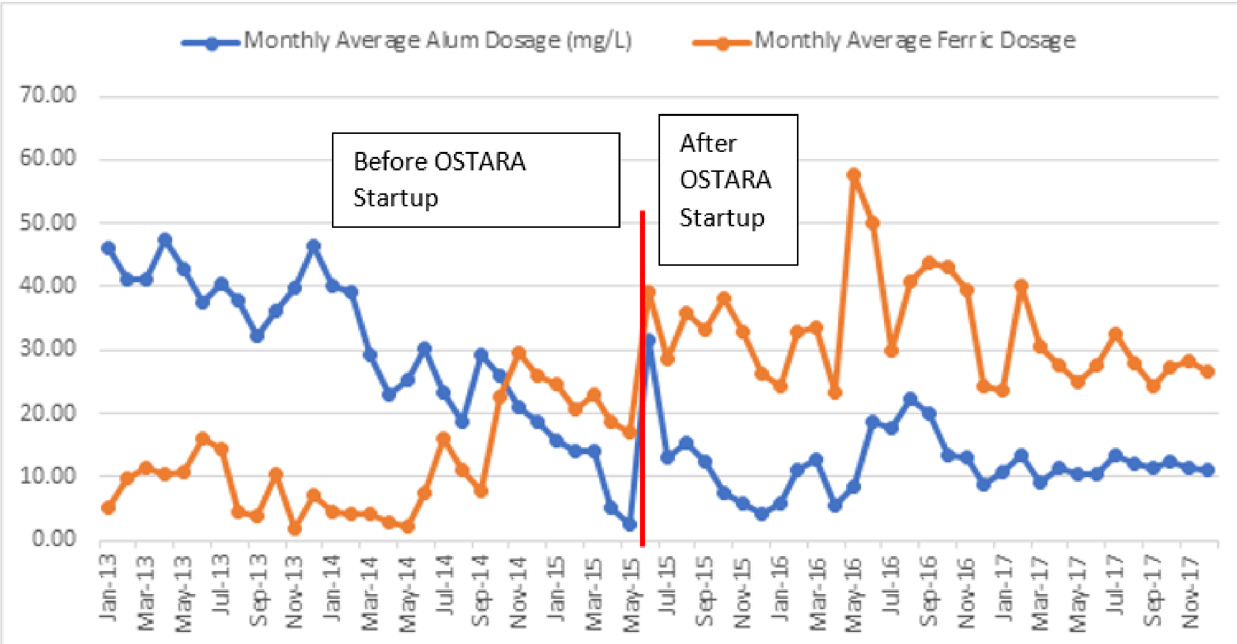


Figure 5-8. Monthly Average Alum and Ferric Chloride Dosage before and after OSTARA Startup

An additional benefit observed since startup of the Nutrient recovery facility in July 2015 along with using the WASSTRIP process is a significant increase in dewatered sludge solids content resulting in reduced volumes of dewatered sludge produced, associated sludge hauling, and landfilling costs. Figure 5-9 shows the average monthly solids content of the dewatered sludge cake from January 2013 through December 2017. As shown, the sludge cake percent total solids (percent TS) increased from an average monthly value of 22.2 percent from January 2013 through June 2015 to an average of 23.7 percent after startup of the Ostara process in July 2015. Research has shown that biological phosphorus removal plants may experience higher monovalent to divalent (M/D) cation ratios in their anaerobic digestion due to release of phosphate and potassium under anaerobic conditions and complexation and precipitation of calcium phosphate and magnesium phosphate species. The resulting increase in M/D cation ratios contribute to poor floc formation and subsequent poor dewatering performance (Higgins et al., 2014). As such, the addition of a process such as WASSTRIP to release and redirect the phosphorus and potassium prior to digestion results in an improvement in the dewatering properties.

Figure 5-9. Sludge Cake Percent Total Solids – Monthly Average

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CHAPTER 6.0
South Durham Water Reclamation Facility - Durham, North Carolina Mainstream Biological Nitrogen Removal and Sidestream Deammonification Process (ANITA Mox®) for Centrate Nitrogen Removal – Case Study

6.1 Background

The South Durham Water Reclamation Facility (SDWRF) is a 20 MGD design flow wastewater treatment facility located in the City of Durham, North Carolina. The facility currently treats an average flow of approximately 10 MGD and discharges to Jordan Lake in Cary, North Carolina. Currently, the facility operates under an annual average total nitrogen (TN) load limit of 334,705 lb/yr, which translates to an equivalent TN discharge limit of approximately 5.5 mg/l at design flow. The TN loading limit may be reduced to 185,345 lb/yr in the near future to comply with the total maximum daily load limits in the Jordan Lake Watershed. This translates to a mass equivalent TN discharge limit of approximately 3.0 mg/l TN at design flow.

In 2011, the city completed a wastewater master planning effort that evaluated different treatment alternatives for meeting a total nitrogen (TN) limit of 3 mg/l and a total phosphorus (TP) limit of 0.23 mg/l at its design flow expected in the near future to comply with total maximum daily load limits in the Jordan Lake Watershed which serves as a source of drinking water in the region.

SDWRF uses anaerobic digesters to process the plant's sludge. Digested sludge is dewatered using belt filter presses and the filtrate sidestream amounts to approximately 20 percent of the load to the BNR process. As part of the master planning study, several mainstream and sidestream treatment alternatives were evaluated to enable the plant to meet its expected TN limits. Sidestream treatment using the ANITA Mox® deammonification process for sidestream nitrogen removal was recommended. Cost comparisons with mainstream treatment revealed sidestream treatment using the deammonification pathway to be three times lower in cost per pound of nitrogen removed ($0.93/lb N removed for deammonification compared with $2.66/lb N removed for mainstream treatment) when capital and operating costs were considered (Bilyk et al., 2017).

6.2 Plant Processes

Liquid and solids treatment processes at SDWRF are shown in Figure 6-1. The major treatment processes at the facility include screening, influent pumping, grit removal, primary clarification, five-stage biological nutrient removal, secondary clarification, alum precipitation for chemical Phosphorus precipitation trim, filtration, ultraviolet (UV) disinfection, solids thickening, anaerobic digestion, belt filter press dewatering, and sidestream ANITA Mox® Deammonification.

Flow enters the plant at the influent pump station where it passes through two bar screens and is then pumped up to four grit collectors before flowing to primary settling tanks. Scum and grit are collected and transported to a landfill. Primary sludge is sent to the anaerobic digesters for treatment. Primary effluent flows to a modified 5-stage Bardenpho BNR system to treat the wastewater for nitrogen and phosphorus removal, which is equipped with the ability to feed carbon though the plant currently does not require carbon addition. Secondary effluent is settled in secondary clarifiers and then passed through dual media filters. Filtered effluent undergoes UV disinfection and is post-aerated prior to discharge.

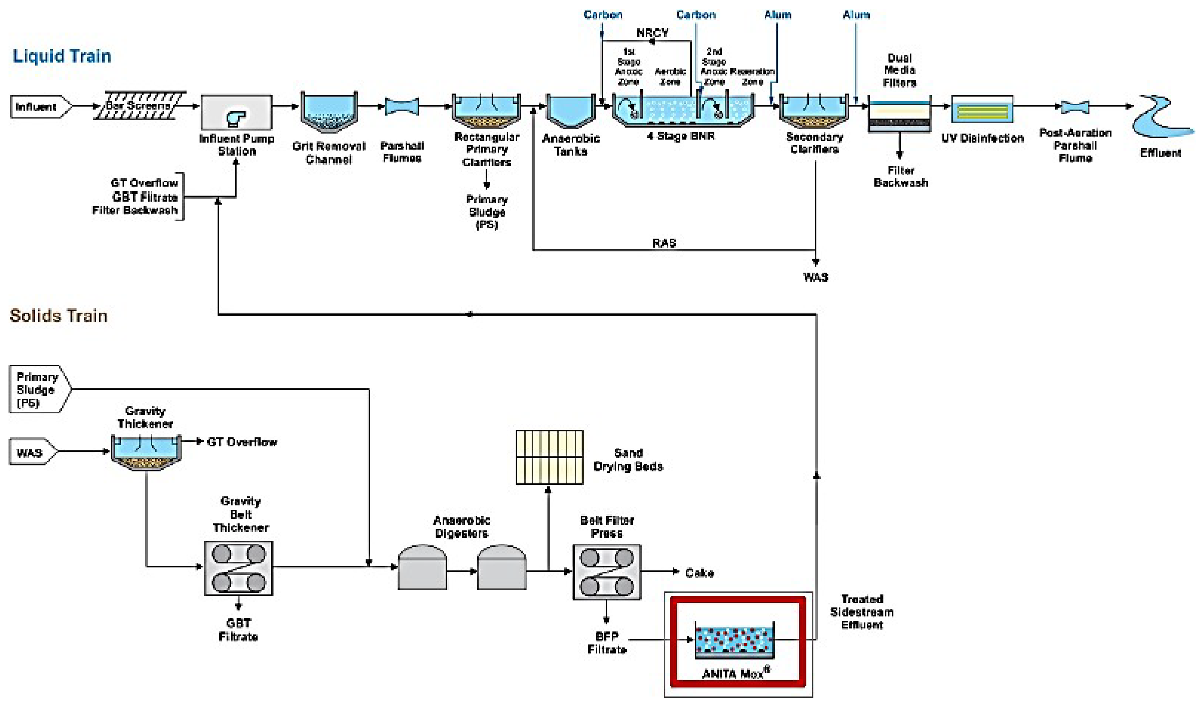


Figure 6-1. Liquid and Solids Treatment Process Train at SDWRF (Bilyk et al., 2017)

Waste activated sludge (WAS) is thickened through gravity thickeners and then through gravity belt thickener. It is then mixed with primary sludge before entering the anaerobic digesters. Digested sludge is dewatered with belt filter presses, and the dewatered cake is placed in a sludge storage pad prior to transportation to land application sites. Sand drying beds are available but are not in service. Biogas produced in the anaerobic digesters is utilized to run two engine-driven blowers. The dewatering filtrate is fed to a sidestream ANITA Mox® deammonification facility for ammonia and total nitrogen load reduction, and the effluent from the deammonification facility is returned to the head of the plant.

Table 6-1. Design and Average Raw Influent Concentrations and Percent of Design Loads for the SDWRF from January 2015 to December 2017

Parameter	Raw Influent Design		Average Raw Influent	Percent of Design[1]
	(Annual Average)	(Maximum Month)		
Flow (MGD)	15.4	NA	10.60[2]	68.8
BOD (lbs/d)	33,907	44,439	27,401	80.81
TSS (lbs/d)	30,182	43,411	26,716	88.52
Ammonia (lbs/d)	NA	NA	2,431	NA
TKN (lbs/d)	3,622	4,739	3,359	92.74
TP (lbs/d)	861	1,130	425	49.36
Temperature (ºC)	NA	NA	NA	NA

Note:
1- Percent of design values, except for flow, are based on average annual values for the analysis period for influent design loads (lbs./day) and actual influent loads (lbs./day).
1- N/A: Data not available or applicable.
2- Raw influent flow data available from 1/1/2015 through 8/1/2016 only

Table 6-2. NPDES Limits - January 2015 – December 2017 at SDWRF

Parameter	Monthly Average (mg/l)	Weekly Average (mg/l)	Annual Load (Lbs/yr)
BOD_5 (Apr - Oct)	5.0	7.5	N/A
BOD_5 (Nov - Mar)	7.0	10.5	N/A
TSS	30.0	45.0	N/A
Ammonia (Apr.- Oct.)	1.0	3.0	N/A
Ammonia (Nov.- Mar.)	2.0	6.0	N/A
TN	N/A	N/A	334,705 (Current) 185,345 (Future expected)
TP	N/A	N/A	14,053
Temperature (ºC)	N/A	N/A	N/A

Note:
N/A: Data not available or applicable.

6.3 Conventional Nitrogen Removal Technology

Nitrogen removal in most wastewater treatment facilities (also known as water resource recovery facilities – WRRFs) is achieved biologically through conventional biological nitrogen removal using nitrification and denitrification (NdN) processes, with nitrification consuming as much as half the power required for aeration based on typical wastewater carbon to nitrogen ratios (COD/TKN). NdN is a two-step process in which autotrophic and heterotrophic bacteria sequentially convert ammonia to nitrogen gas. The first step, nitrification, is aerobic whereby ammonium (NH_4^+) is oxidized to nitrite (NO_2^-) by ammonia oxidizing bacteria (AOBs), and nitrite is converted to nitrate by nitrite oxidizing bacteria (NOBs). The second step, denitrification, is anoxic whereby nitrate (NO_3^-) is converted $NO2^-$ then to nitrogen gas by ordinary heterotrophic organisms (OHOs). In the first step, ammonia conversion consumes oxygen and alkalinity. In the second, no oxygen is consumed, and alkalinity is produced. Additionally, since the denitrification reaction is conducted by heterotrophic bacteria, sufficient carbon is needed from the wastewater COD or by external chemical carbon addition to achieve a COD:TKN ratio of about ten or more depending on the carbon source type.

Energy consumption by public water and wastewater services consumes about 0.5 percent of total U.S.primary energy and 2 percent of its end-use electricity (Twomey and Webber 2011, EPRI 2013). Energy consumption by wastewater treatment facilities and drinking water systems can amountto up to one third of a municipality's total energy bill (EPA, 2009). Typically, at an activated sludge wastewater treatment facility, 40-70 percent of the energy used is for aeration (WEF, 1997). Plants using conventional biological nitrogen removal are at the higher end of this range.

In plants that use anaerobic digestion for sludge stabilization and reduction, volatile solids are destroyed resulting in the release of significant amounts of ammonia nitrogen, typically amounting to between 15 – 20 percent of the plant's influent nitrogen load in many cases. This may be higher depending on the type of digesters used and the associated degree of volatile solids destruction achieved, as well as other factors such as whether other solids are co-digested and whether primary treatment is used. In plants where sludge is dewatered intermittently or if the dewatering process is not operating properly, this sidestream ammonia load may significantly affect the stability of mainstream biological nutrient removal processes and cause diurnal spikes in affluent ammonia or total nitrogen levels. As such, cost-effective approaches to treating this sidestream load has become an increasingly important treatment objective due to potential cost savings and the positive impact on the mainstream nutrient removal process.

6.4 Deammonification

Several processes have been used over the last decade to treat high ammonia sidestreams, generally relying on using biomass for treatment in varying configurations, process control approaches, and operational variables including hydraulic retention times (HRT), temperature, and ammonia concentration. However, the discovery of a group of microorganisms known as the anammox (anaerobic ammonia oxidation) bacteria that can convert ammonia and nitrite directly to nitrogen gas published in the 1990s (Mulder et al.,1995) has significantly enhanced the attractiveness of sidestream nutrient removal processes. This is due to aeration energy savings, reduced external carbon demand, reduced alkalinity demand, and reduced sludge production as the anammox are anaerobic, autotrophic, and the reaction has low biomass yield and only produces small amounts of nitrate.

The deammonification process involves partial nitrification (conversion of ammonia to nitrite) and anaerobic ammonia oxidation. It requires approximately a 50 percent mix of ammonia and nitrite for the anammox bacteria to oxidize ammonia under anoxic conditions using nitrite. The process has been established and used successfully for sidestream treatment at a number of wastewater facilities in the US and overseas. Deammonification is an ideal process for dewatering sidestreams because centrate or filtrate resulting from dewatering of anaerobically digested sludge is warm in temperature and high in ammonia concentration (around 1,000 mg/l or above in most cases). The higher temperature allows the anammox to grow within a reasonable volume and sludge retention time (SRT) and the higher ammonia concentrations are believed to inhibit nitrite oxidizing bacteria (NOBs) that compete for nitrite for aerobic nitrite oxidation (nitrification). Centrate also has low carbon content which inhibits heterotrophic bacteria from outcompeting the anammox for the available nitrite for anerobic denitrification. Research and testing for stable mainstream deammonification continue but the process has not been used at full scale in the United States yet. This is mainly due to the difficulty of consistent full-scale repression of NOBs in mainstream processes as well as the need for retention of the anammox due to their slower growth rates.

Current deammonification systems for sidestream treatment include sequencing batch reactor (SBR) processes, an up-flow granular bed process, a moving bed biofilm reactor (MBBR) process, an Integrated Fixed Film Activated Sludge (IFAS) process for sidestream treatment, and a hybrid suspended and attached growth process.

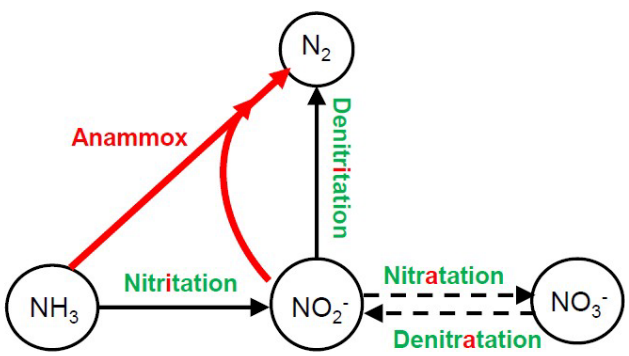


Figure 6-2. Nitrogen Transformations (WERF, 2014)

Deammonification can result in significant savings compared to conventional nitrification-denitrification (NdN) biological nitrogen removal since only a portion of the ammonium is aerobically oxidized by ammonia oxidizing bacteria (AOB) to nitrite and without the need for nitratation, and the subsequent step of anoxic ammonium oxidation (annamox) takes place without the need for costly external carbon addition. This can theoretically amount to 60 percent or more reduction in oxygen demand and associated aeration energy and near complete eliminationof costly supplemental carbon addition. Savings in overall capital and operations and maintenance costs for sidestream deammonification have been reported as high as 65 percent compared to costs for a mainstream BNR system (Farina, 2012); however, actual savings will vary depending on the process used, existing facility infrastructure, influent characteristics, process control efficiency, and other factors.

6.5 ANITA Mox® Sidestream Deammonification Process at SDWRF

The ANITA™ Mox MBBR deammonification process is an ammonia and TN removal biofilm process which combines nitritation with anaerobic ammonia oxidation (Annamox) in a single stage two-step process. The two steps of the process occur in different layers of the biofilm with aerobic nitritation occurring in the outer layer and anammox (anoxic) in the inner layer. This takes place in a Moving-Bed Biofilm Reactor (MBBR) equipped with specially designed plastic carriers for biofilm growth (Figure 6-3), typically retained in the reactor by screens.

Approximately 50 percent of the influent ammonia is oxidized to Nitrite (NO_2^-) in the first step and the nitrite produced, and the remaining ammonia, are utilized by the anammox bacteria and converted to nitrogen gas (N_2) and a small amount of Nitrate (NO_3^-) in the second step. Both steps occur concurrently in the biofilm.

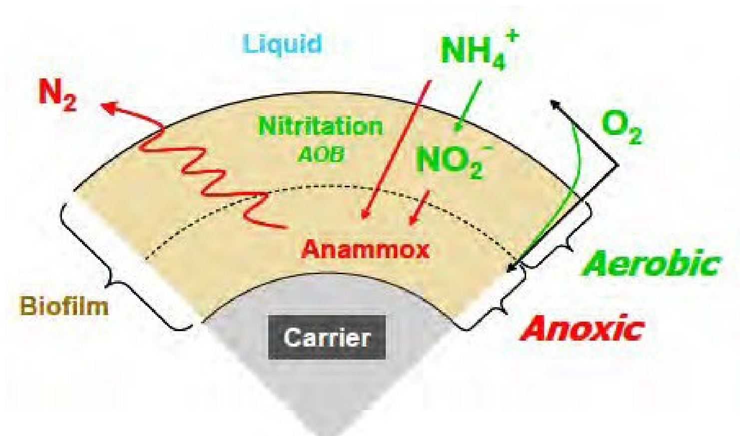


Figure 6-3. ANITA Mox® MBBR Model (Source: Kruger/Veolia)

Operational variables in the reactor such as dissolved oxygen, pH, and temperature are maintained to favor the desired microorganisms (AOB, Annamox) in the biofilm and prevent their washout. Biofilm processes can be limited by diffusion with the limiting factor depending on concentration and diffusivity. In the Anita Mox MBBR process, the energy source for nitritation and annamox is ammonium and the electron acceptors are oxygen and nitrite, respectively. As such, the oxygen concentration in the liquid is critical. A high DO concentration may inhibit the annamox reaction and favor the undesirable oxidation of nitrite to nitrate by NOB while a low concentration will limit nitritation. Other important factors are pH and nitrite concentrations. Ensuring that nitrite is consumed at about the same rate of its production will limit the inhibition of the annamox process by nitrite (Plaza et al., 2009).
As mentioned in Section 6.2, anaerobically digested sludge at SDWRF is dewatered with belt filter presses. An abandoned aerobic digester was repurposed to provide approximately three days of equalization and two 95,000-gallon deammonification reactors. Dewatering filtrate is fed to a sidestream ANITA Mox® deammonification facility for total nitrogen removal. The Anita

Mox® process was in full-scale operation starting in December 2015. The effluent from the deammonification facility is returned to the head of the plant. Main system components include a filtrate equalization tank, two parallel MBBR reactors, reactor feed pumps, and an aeration system with coarse bubble aeration grids in the reactors and independent airflow control in each reactor. Two submersible mixers in the equalization tank provide mixing, and a heat loop provides supplementary heat if needed during winter months. Both reactors and the equalization tank are provided with insulated covers to retain heat. Reactor mixing during anoxic or low DO periods is provided by a vertical mixer. Both reactors contain airlift pumps to control excessive foaming. MBBR media is retained in the reactor by virtue of a stainless-steel screen over the outlet ports and a screen is placed over a sump in each reactor to allow draining when needed. Process influent design parameters are shown in Table 6-3.

Table 6-3. Influent Design Values for the Anita Mox® MBBR Process at SDWRF (Bilyk etal., 2017)

Influent Design Values		
Parameter	**Units**	**Value**
Flow, Current Average	MGD	0.04
Flow, Design	MGD	0.08
Flow, Peak	MGD	0.16
Flow, Max Hydraulic Flush	MGD	0.6-0.7
BOD, Design Flow	mg/L	85
COD, Design Flow	mg/L	500
TSS, Design Flow	mg/L	250-500
NH3-N, Design Flow	mg/L	1000
TKN, Design Flow	mg/L	1100
Alkalinity Design Flow	mg/L	2500
pH	SU	7.4
Min Temperature	C	24

6.6 Detailed Statistical Analysis – Plant Effluent Concentrations

Facility operating data from January 2015 to December 2017 were analyzed. Figures 6-5 through 6-7 and Tables 6-4 through 6-7 provide a summary of the statistical analysis performed for the SDWRF facility.

During the entire analysis period, the data shows that the facility met its final effluent treatment objectives shown in Table 6-2 for total nitrogen and ammonia. SDWRF's discharge permit includes annual effluent TN loading limits of 334,705 lb/yr. Actual discharged TN loads were 217,075, 170,903 and 171,927 lb/yr in 2015, 2016, and 2017, respectively, all well within their discharge limits.

For Ammonia, SDWRF's permit includes monthly and weekly ammonia concentrations of 1 mg/l and 3 mg/l, respectively from April to October, and 2 mg/l and 6 mg/l, respectively from November to March. Calculated actual average monthly and weekly effluent ammonia concentrations are as shown below. They were found to be the same when rounded off to 2 decimal points.

Monthly

November 2014 – March 2015: 0.24 mg/l; April 2015 – October 2015: 0.22 mg/l;
November 2015 – March 2016: 0.23 mg/l; April 2016 – October 2016: 0.03 mg/l;
November 2016 – March 2017: 0.13 mg/l; April 2017 – October 2017: 0.07 mg/l.

Weekly

November 2014 – March 2015: 0.24 mg/l; April 2015 – October 2015: 0.22 mg/l;
November 2015 – March 2016: 0.23 mg/l; April 2016 – October 2016: 0.03 mg/l;
November 2016 – March 2017: 0.13 mg/l, April 2017 – October 2017: 0.07 mg/l.

Comparing the effluent concentrations to the ammonia permit limits shows that the facility consistently met the seasonal ammonia limits throughout the 3-yr analysis period.

Figure 6-4 shows the 12-month rolling average TN concentrations and discharge loadings and shows the plant's consistency in meeting the annual discharge loading limits from month to month.

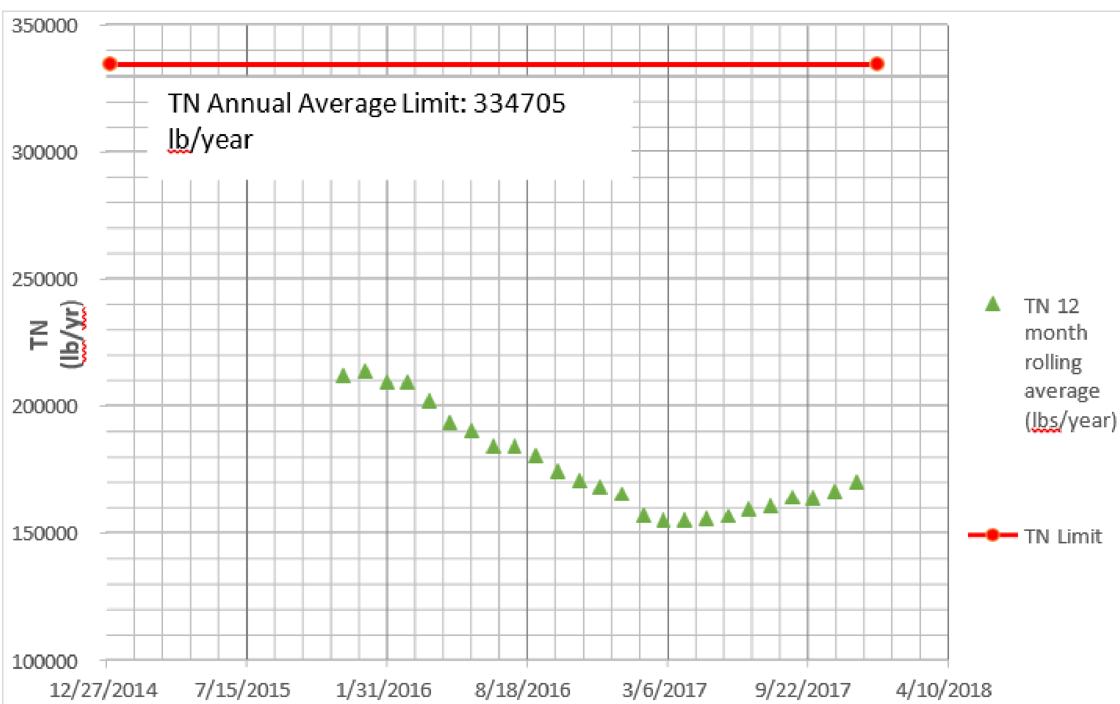


Figure 6-4. 12-month Rolling Average Time Series Plot for TN

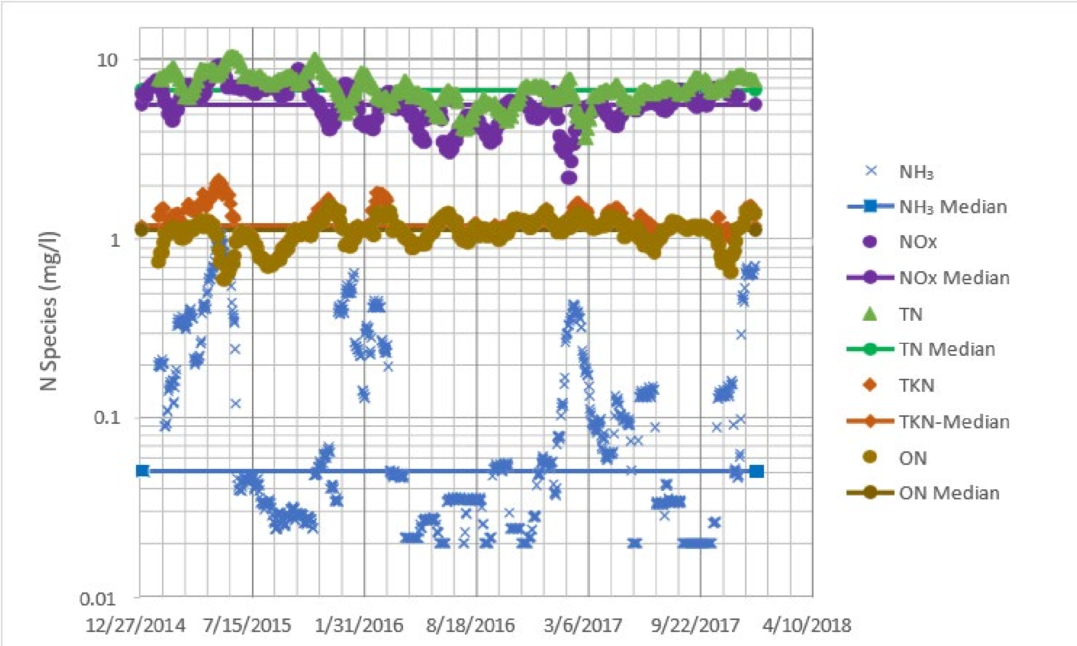
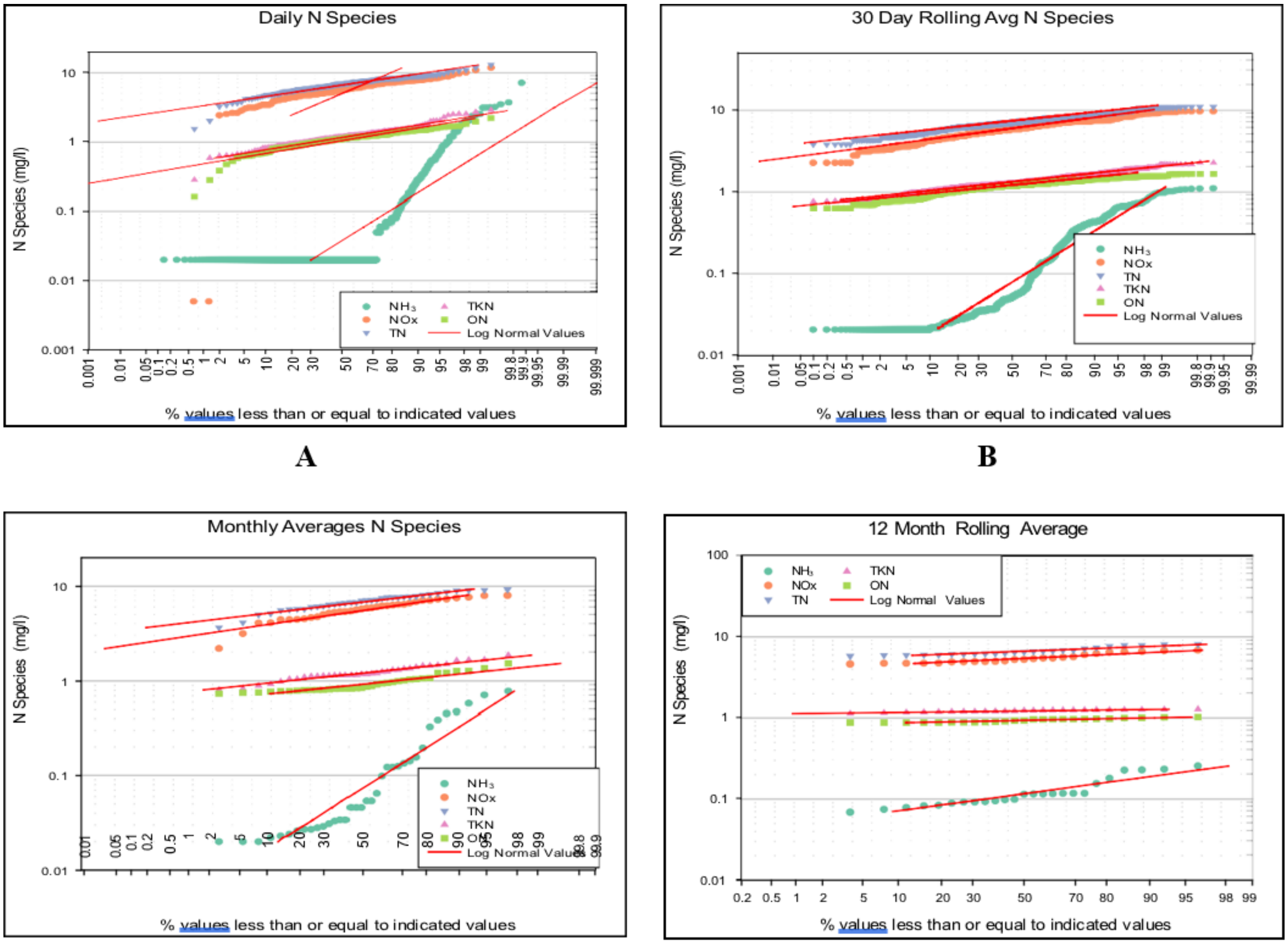


Figure 6-5. 30-Day Rolling Average Time Series Plot

Figure 6-5 shows the 30-day rolling average time series plot for nutrient species effluent concentrations at SDWRF. For the 3-year analysis period, the median 30-day rolling average TN concentration was 6.79 mg/l with a maximum value of 10.38 mg/l. The median 30-day rolling average concentration was 0.05 mg/l for ammonia and 5.60 mg/l for NO_x-N, with maximum 30-day rolling values of 1.06 mg/l NH_3-N and 9.23 mg/l NO_x-N.

Figures 6-6 A through D include cumulative probability plots for SDWRF's daily, 30-day rolling average, monthly average, and rolling annual average data sets. A percentile value on the x-axis represents the probability that the value is less than or equal to the stated corresponding concentration on the plot's y-axis. In looking at the NH_3-N and TKN values on Figure 6-6A, it is clear that for the most part, the effluent TN is comprised of nitrate nitrogen. The 95th percentile TN daily average concentration in Figure 6-6 (A) for the 3-year analysis period is 9.36 mg/l.



C D

Figure 6-6. Probability Plots for SDWRF – (A) Daily Data; (B) 30-day Rolling Average; (C) Monthly Average; (D) 12 Month Rolling Average

For the period of December 2015 through December 2017 corresponding to the implementation of the Anita Mox®, the 95th percentile effluent TN average concentration was lower at 8.85 mg/l.According to SDWRF, this can mainly be attributed to the impact of the Anita Mox process in reducing the TN load to the mainstream BNR process.

The monthly average effluent ammonia concentrations associated with the 3.84, 50, 90, 95, and 99th percentiles during the 3-year analysis period (2015-2017) are 0.02, 0.05, 0.47, 0.61, and 0.75 mg/l, respectively, all well below the discharge permit monthly average limits of 1.0 mg/l (April – October), and 2.0 mg/l (November –March). Similarly, the weekly average ammonia concentrations associated with the 3.84, 50, 90, 95, and 99 percentiles are 0.02, 0.03, 0.45, 0.76, and 1.54 mg/l, respectively, all well below the discharge permit weekly average limits of 3.0 mg/l (April – October) and 6.0 mg/l (November – March).

Table 6-4. Summary Statistics for Final Effluent Ammonia Nitrogen for SDWRF

	NH_3-N Daily Data	NH_3-N Weekly Data	NH_3-N Rolling 30-day Average	NH_3-N Monthly Averages	NH_3-N 12 Month Rolling Average
n	753	156	1067	36	25
Mean	0.152	0.151	0.148	0.153	0.123
Geometric Mean	0.039	0.053	0.074	0.075	0.114
Standard Dev.	0.491	0.305	0.197	0.203	0.054
CV	3.236	2.026	1.331	1.332	0.438
Skew	7.020	3.401	2.141	1.866	1.364
Minimum	0.020	0.020	0.020	0.020	0.069
Maximum	7.080	1.833	1.062	0.771	0.251

Table 6-5. Summary Statistics for Final Effluent NOx-N for SDWRF

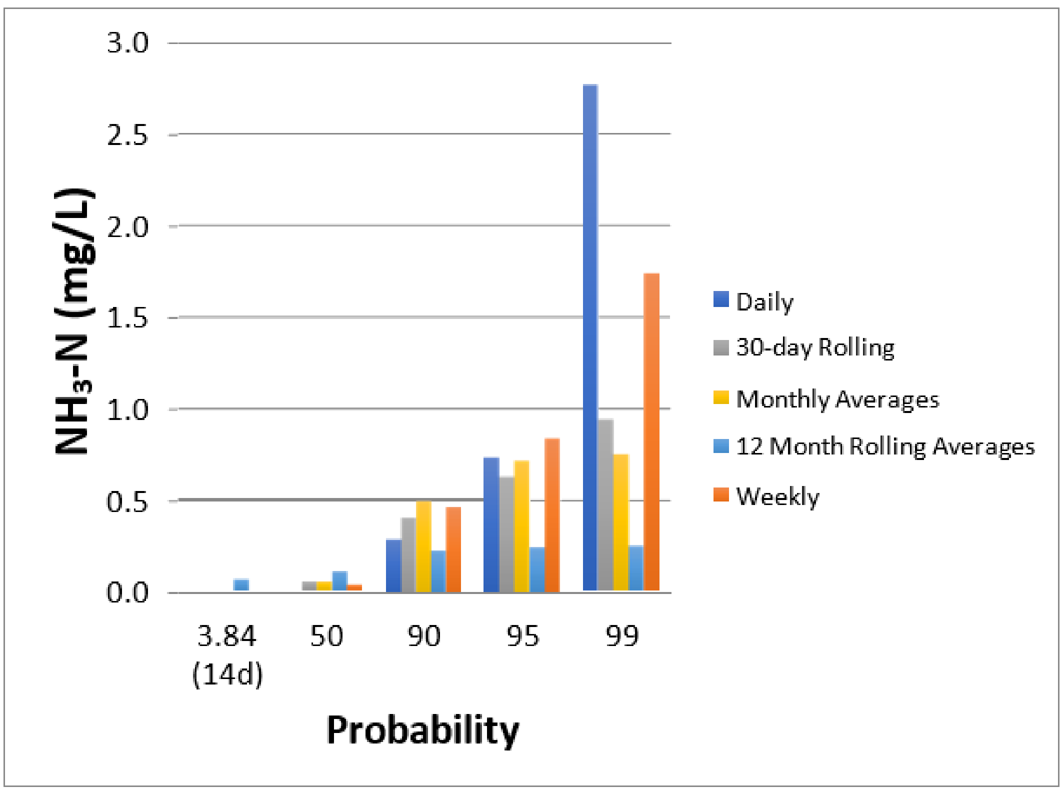
	NOx-N Daily Data	NOx-N Weekly Data	NOx-N 30-day Rolling Average	NOx-N Monthly Averages	NOx-N 12 Month Rolling Average
n	156	156	1067	36	25
Mean	5.684	5.684	5.662	5.697	5.397
Geometric Mean	5.038	5.038	5.513	5.518	5.354
Standard Dev.	1.785	1.785	1.260	1.345	0.716
CV	0.314	0.314	0.223	0.236	0.133
Skew	0.079	0.079	0.086	-0.373	0.731
Minimum	0.005	0.005	2.178	2.178	4.636
Maximum	11.700	11.700	9.225	8.025	6.754

Table 6-6. Summary Statistics for Final Effluent Total Nitrogen for SDWRF

	TN Daily Data	TN Weekly Data	TN 30-day Rolling Average	TN Monthly Average	TN 12 Month Rolling Average
n	156	156	1067	36	25
Mean	6.908	6.908	6.881	6.920	6.600
Geometric Mean	6.647	6.647	6.759	6.775	6.564
Standard Dev.	1.788	1.788	1.279	1.374	0.721
CV	0.259	0.259	0.186	0.199	0.109
Skew	0.080	0.080	0.151	-0.323	0.784
Minimum	1.530	1.530	3.625	3.625	5.817
Maximum	13.030	13.030	10.378	9.303	7.990

Table 6-7. Summary Statistics for Final Effluent ON for SDWRF

	ON Daily Data	ON Weekly Data	ON 30-day Rolling Average	ON Monthly Average	ON 12 Month Rolling Average
n	156	156	1067	36	25
Mean	1.098	1.098	1.100	0.934	0.921
Geometric Mean	1.048	1.048	1.084	0.917	0.920
Standard Dev.	0.306	0.306	0.182	0.195	0.050
CV	0.279	0.279	0.166	0.209	0.055
Skew	0.117	0.117	-0.399	1.303	0.198
Minimum	0.160	0.160	0.590	0.723	0.862
Maximum	2.190	2.190	1.525	1.514	1.009



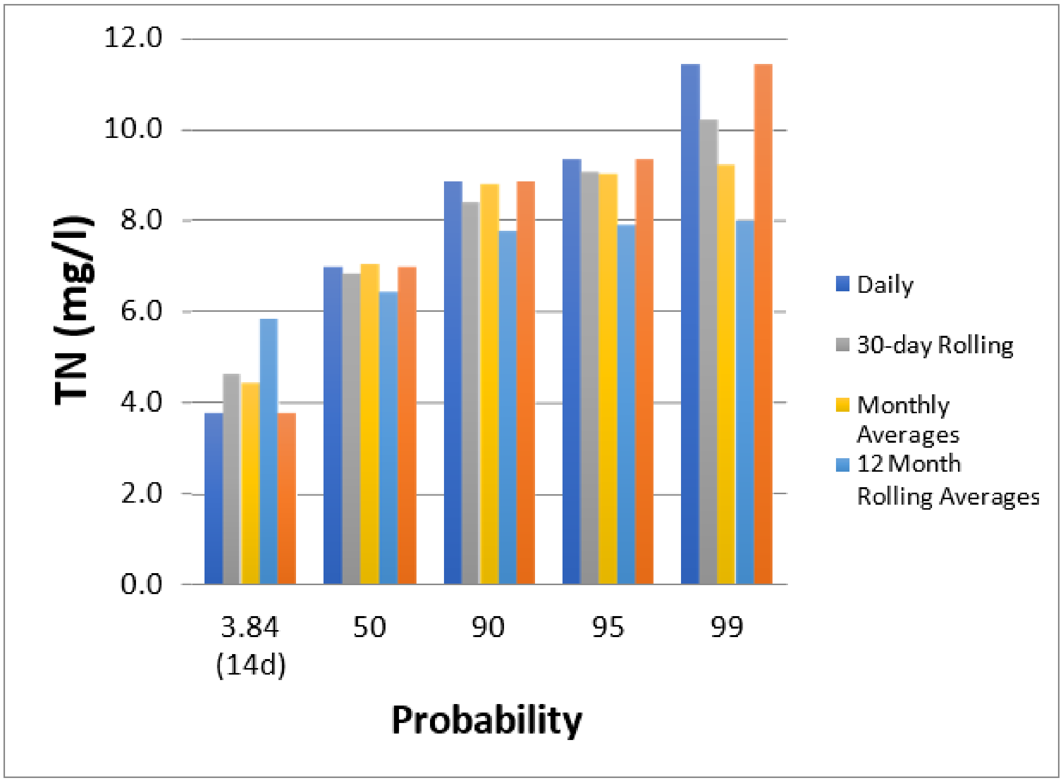


Figure 6-7. Probability Summary for SDWRF

Figure 6-7 can be used to assess the process variability for TN and NH_3-N. For example, the daily data 50th percentile TN concentration for the entire 3-year period was 6.95 mg/l, and the 95th percentile was 9.61 mg/l for TN with a ratio of 95th to 50th percentile of about 1.38. Calculating this ratio for the periods before the Anita Mox process was in service (January 2015 – November 2015) and after it was in service (December 2015 – December 2017), the ratios were 1.43 and 1.38, respectively, demonstrating slightly lower variability when Anita Mox® was in service.

6.7 Process Performance - ANITA Mox® Sidestream Deammonification at SDWRF

As shown in section 6.5 above, the centrate pre-treatment facility (CPT) at SDWRF uses the ANITA Mox® Sidestream Deammonification system to treat anaerobically digested sludge dewatering filtrate for ammonia and total nitrogen load reduction. The Anita Mox® process was infull-scale operation starting in December 2015. The effluent from the deammonification facility was returned to the head of the plant.

A simplified diagram of SDWRF deammonification system is shown in Figure 6-3 (Hollowed et al., 2018).

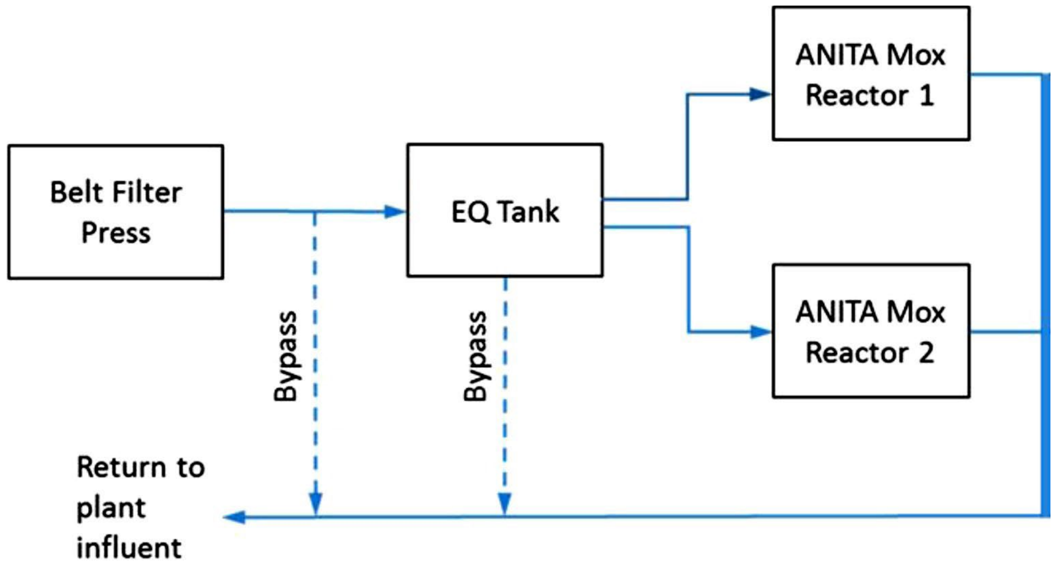


Figure 6-8. Simplified Flow Diagram of SDWRF Deammonification System

Digested sludge is typically dewatered five days a week in one shift, and the filtrate flows to the equalization (EQ) tank. The Anita Mox reactors are normally fed at a steady rate set to ensure that the EQ tank is not full anytime during the weekly dewatering cycle. Any excess filtrate is returned to the head of the plant.

Aeration control can be accomplished in one of three modes: intermittent aeration, continuous aeration, and DO control. Intermittent aeration with constant airflow is the main mode of operation with a specified duration of aerobic and anoxic cycles. The facility indicated that this mode results in stable operation as the filtrate ammonia concentrations are generally stable. However, the facility has used the DO control mode on two separate occasions but decided that intermittent mode results in more stable operation. The process is underloaded with respect to its design loading, and therefore using DO control mode results in too many pounds of oxygen for the system for it to operate in the optimal manner. On-line DO, ammonia, and nitrate probe readings are used to monitor the process. Grab samples are also collected and analyzed in the laboratory for reactor ammonia, nitrite, and nitrate nitrogen. These samples are generally collected at a minimum of once a week with some exceptions.

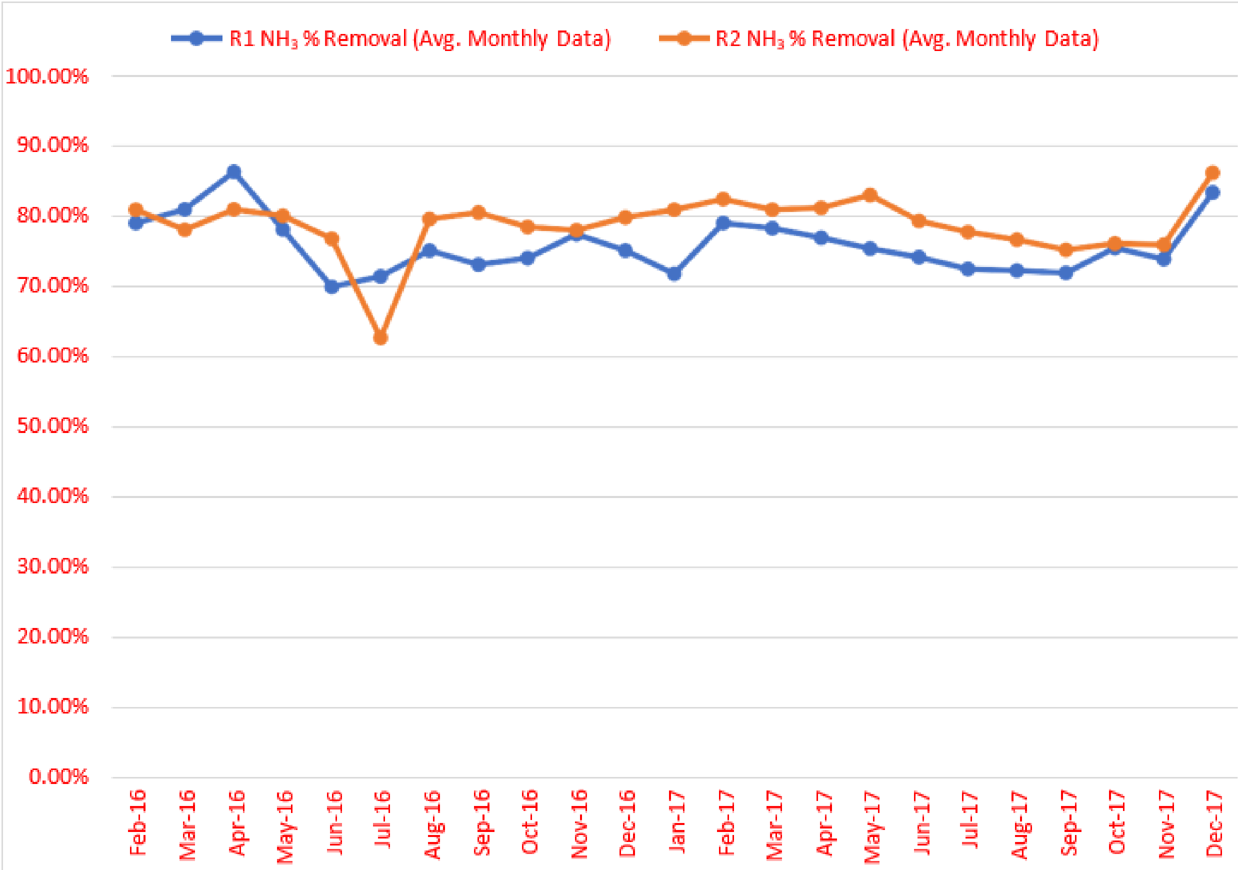


Figure 6-9. Monthly Average Ammonia Percent Removal for Sidestream Reactors 1 and 2

Figure 6-9 shows ammonia percent removal in reactors 1 (R1) and 2 (R2). As shown, both reactors were able to reach ammonia removal rates of as high as 86 percent (April 2016 for R1 and December 2017 for R2). The rates were calculated based on reactor influent and effluent laboratory analysis provided by the facility. However, the facility experienced periods of reduced performance due to process imbalances requiring additional and more frequent process control to stabilize the process.

One incident occurred in June - July 2016, when the facility experimented with the continuous aeration mode which negatively impacted performance and likely favored NOB over anammox based on the process control procedure used at the time, resulting in nitrate elevation. This resulted in a reduction in ammonia removal efficiency to a monthly average of 70 percent in R1 and 63 percent in R2 for July 2016 as shown in Figure 6-4. As a result, the intermittent aeration mode operation was resumed, and the removal efficiency improved as shown in the August 2016 average ammonia removal of 75 percent in R1 and 80 percent in R2.

Another incident occurred in November 2016 which the facility attributed to struvite buildup in the reactor feed piping from the equalization tank. This reduced reactor influent flow and increased reactor HRT and was subsequently corrected by implementing pipe cleaning, flushing the reactors with dilution water to remove excess solids and nitrite, and struvite formation minimization procedures.

Another process difficulty was experienced in R1 in August 2017, attributed to faulty high DO probe readings while operating temporarily in the DO control mode. This resulted in an unnecessary reduction of the DO setpoint by the process control system and an elevation in the ammonia concentration. The operators corrected by temporarily shutting off the reactor feed flow and switching the airflow control mode to an intermittent cycle to allow the reduction of ammonia concentration to normal levels before restarting feed flow, and by cleaning and calibration of the probe. A detailed assessment of the operational difficulties experienced by the facility and the approaches taken to address them is provided by Bilyk et al. (2017).

It should be noted that the SDWRF experienced a significant improvement in plant effluent total nitrogen concentrations with a reduction of approximately 1.5 mg/l TN after the Anita Mox® process was in operation at full scale (December 2015 to December 2017) compared to a period before Anita Mox was in full scale operation (January 2014 to November 2015). This improvement was achieved notwithstanding increases in influent TKN loads in 2016 and 2017 compared to 2015. While the facility has implemented a number of other efforts to improve TN removal (such as implementing mainstream BNR ammonia-based aeration control in 2015 and improvement of baffle walls in the mainstream BNR reactors in 2016 to reduce air back-mixing from the aerobic zone into the 1st anoxic zone), the implementation of the Anita Mox® process andthe associated reduction in the return nitrogen load to the mainstream and its reduced variability played an important role in the reduction in plant effluent total nitrogen concentrations mentioned above.

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CHAPTER 7.0
Town of Hillsborough Wastewater Treatment Plant – Hillsborough, North Carolina – Modification and Enhancement of 5-Stage BNR – Case Study

7.1 Background

The Hillsborough Wastewater Treatment Plant (HWWTP) is a 3.0 MGD wastewater treatment facility located in the Town of Hillsborough, North Carolina. As of January 2016, HWWTP's NPDES permit annual total nitrogen (TN) discharge load was reduced to 10,422 lb/yr to comply with the water quality standards for Falls Lake, the drinking water supply for the City of Raleigh, NC. At the plant's annual average design flow of 2.4 MGD, the permit annual average TN discharge load is equivalent to 1.43 mg/l.

HWWTP implemented the first phase of nutrient removal upgrades by converting the system to a 5-Stage BNR process. The facility had originally planned additional upgrades of the original BNR system to include a reverse osmosis (RO), ion exchange (IX), or equivalent process treatment to be added downstream of the BNR facility. However, since the facility's actual flow is significantly lower than design flow, as shown in Table 7-1, HWWTP implemented innovative modifications to its BNR process. It also implemented process control strategies. This allowed the plant to meet its discharge limits without the need for the RO or IX processes and without the need for carbon or other chemical addition. As a result, the facility reported that it expects further upgrades will be deferred by several years.

7.2 Plant Processes

Liquid treatment processes include preliminary treatment consisting of bar screening and grit removal. This is followed by a five-stage BNR process for removal of BOD as well as TN and total phosphorus (TP), followed by secondary clarification where alum is added for additional phosphorus removal if needed. Figure 7-1 shows the liquid and solids treatment trains at HWWTP before upgrading the biological process to the five-stage BNR configuration (shown in Figure 7-4 below). Since the upgraded BNR system was constructed by modifying two previously existing stages of aeration and secondary clarification, the last two stages (anoxic, reaeration) of the current 5-stage BNR system are physically separated from the previous three stages. The system is equipped with methanol addition capability in the second anoxic zone and alum addition capability to the secondary clarifiers to be used if needed. Denitrification filters with alum addition are also available but are only used to remove residual suspended solids and total phosphorus. No external carbon is added. The effluent is then disinfected by chlorination prior to dechlorination and post aeration, and then discharged to the Eno River. Solids processing includes aerobic digestion of waste activated sludge, digested sludge storage, gravity belt thickening of digested sludge, storage of thickened sludge, and dewatering prior to hauling the sludge offsite.

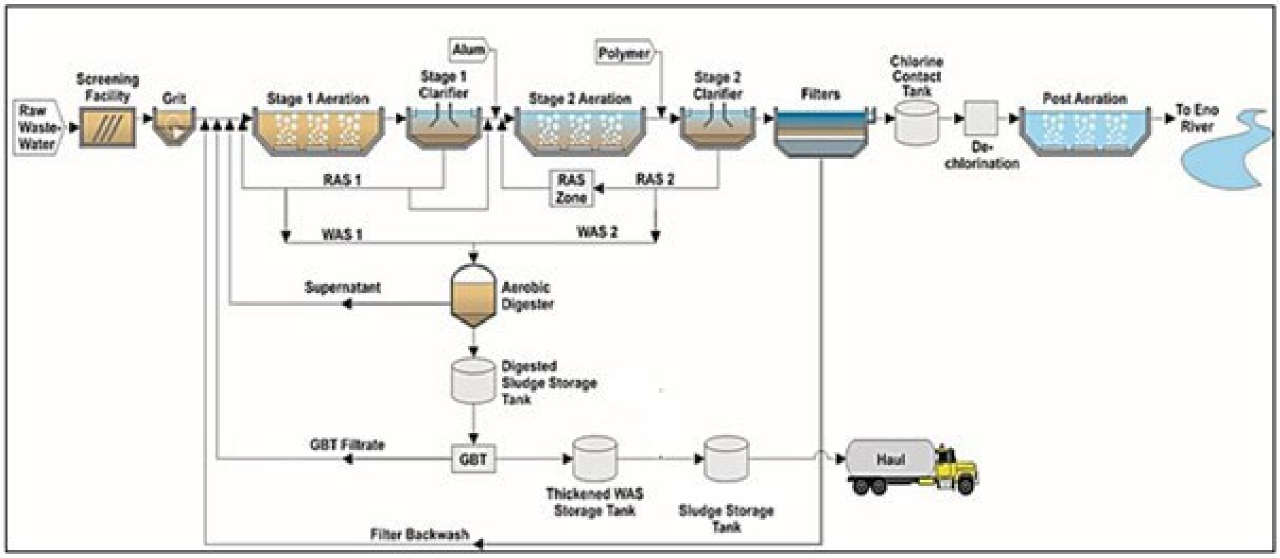

Figure 7-1. Solids and Liquid Treatment Process Trains at HWWTP Before BNR Upgrades
(Adapted from Mahagan and Bilyk, 2016 – With Permission)

Table 7-1. Design and Average Raw Influent Concentrations and Percent of Design Loads for the HWWTP from January 2015 to December 2017

Parameter	Raw Influent Design		Average Raw Influent	Percent of Design[1]
	(Annual Average)	(Maximum Month)		
Flow (MGD)	2.4	3.2	1.04	43
BOD_5 (lbs/d)	4,441	6,000	1941	44
TSS (lbs/d)	4,421	5,400	2055	46
TKN (lbs/d)	697	840	NA	NA
TP (lbs/d)	121	150	NA	NA
Temperature (ºC)	NA	NA	19.13	NA

Note:
1- Percent of design values, except for flow, are based on average annual values for the analysis period for influent design loads (lbs./day) and actual influent loads (lbs./day).
N/A: Data not available or applicable.

Table 7-2. NPDES Limits - January 2015 – December 2017 at HWWTP

Parameter	Monthly Average (mg/l)	Weekly Average (mg/l)	Quarterly Average (mg/l)	Annual Load (lbs/yr)
Flow (MGD)	3.0	N/A	N/A	N/A
BOD_5	5.0	7.5	N/A	N/A
BOD_5 (Nov - Mar)	7.0	10.5	N/A	N/A
TSS	30.0	45.0	N/A	N/A
Ammonia	2.0	6.0	N/A	N/A
TN	N/A	N/A	N/A	50,228 (until 12/31/2015) 10,422 (effective 1/1/2016)
TP	N/A	N/A	2.0	1,352 (effective 1/1/2016)
Temperature (°C)	N/A	N/A	N/A	N/A

Note:
1- N/A: Data not available or applicable.

7.3 Conventional Nitrogen Removal Technology

Nitrogen removal in most wastewater treatment facilities (also known as water resource recovery facilities – WRRFs) is achieved biologically through conventional biological nitrogen removal (BNR) using nitrification and denitrification (NdN) processes, with nitrification consuming as much as half the power required for aeration based on typical wastewater carbon to nitrogen ratios (COD/TKN). NdN is a two-step process in which autotrophic and heterotrophic bacteria sequentially convert ammonia to nitrogen gas. The first step, nitrification, is aerobic whereby ammonium (NH_4^+) is oxidized to nitrite (NO_2^-) by ammonia oxidizing bacteria (AOB), and nitrite is converted to nitrate (NO_3^-) by nitrite oxidizing bacteria (NOB). The second step is anoxic whereby NO_3^- is converted to NO_2^- then to nitrogen gas by ordinary heterotrophic organisms (OHOs). In the first step, ammonia conversion consumes oxygen and alkalinity. In the second, no oxygen is consumed, and alkalinity is produced. Additionally, since this second step reaction is by heterotrophic bacteria, sufficient carbon is needed from the wastewater COD or by external chemical carbon addition to achieve a minimum COD:TKN ratio of ten or more depending on the carbon source type.

Nitrogen removal in wastewater treatment facilities can be accomplished as an integral component of the biological treatment system or as an add-on process to an existing treatment plant. A variety of biological treatment configurations are used by treatment facilities to achieve biological nitrogen removal, and the selection of a specific process depends on treatment requirements, existing process and equipment, and other site-specific conditions. In some systems, nitrification and denitrification can be achieved in one treatment unit while in others, denitrification can be achieved separately in either post-anoxic or pre-anoxic units.

Suspended-growth biological nutrient removal processes can be categorized as single sludge or two-sludge processes. Single sludge refers to systems that use only one solids separation device, normally a secondary clarifier (Metcalf & Eddy, 2003). In such systems, the biological tank is divided into different zones of anoxic and aerobic conditions and mixed liquor can be pumped from one zone to another as internal nitrified recycle (NRCY). These systems are generally categorized depending on whether the anoxic zone is located before, within, or after the aerobic nitrification zone. In the pre-anoxic configuration, initial contact of the wastewater and the return activated sludge (RAS) occurs in the anoxic zone and nitrate produced in the aerobic zone is recycled to the pre-anoxic zone. One example of such a process is the Modified Ludzack- Ettinger (MLE) process shown in Figure 7-2.

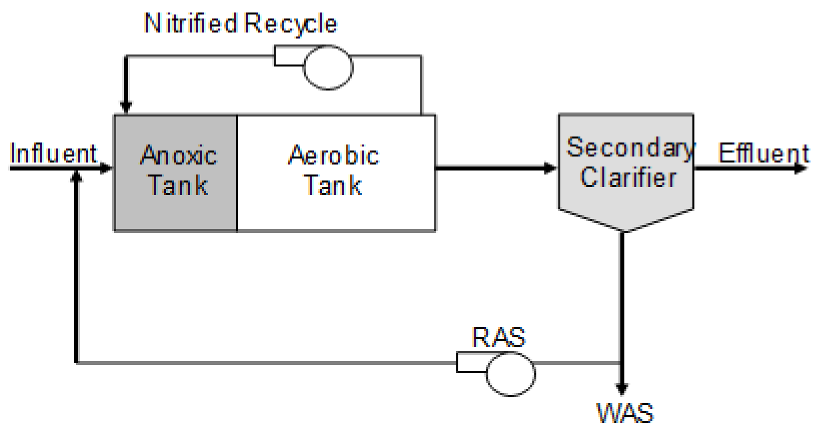

Figure 7-2. Modified Ludzack-Ettinger Process

The MLE is a two-stage process that consists of an anoxic zone upstream of an aerobic zone. An internal recycle carries nitrate created during the nitrification process in the aerobic zone along with mixed liquor to the anoxic zone for denitrification. RAS is mixed with the influent to the anoxic zone (EPA, 2008). This process is generally used to meet intermediate levels of total nitrogen concentrations (generally about 7 – 10 mg/l but as high as 15 mg/l in some cases). This is attributed to nitrate removal being limited by the practical levels of internal recycle to the pre-anoxic zone (Metcalf & Eddy, 2003).

Multi-stage processes are used to achieve higher levels of nitrogen removal such as the 4-Stage Bardenpho Process. This process is essentially an MLE process with subsequent anoxic and oxic zones, but the nitrate nitrogen leaving the last oxic zone is lower, as low as 3 to 4 mg/l with supplemental carbon addition. The five-stage Bardenpho process illustrated in Figure 7-3 uses the same layout but adds an anaerobic zone in front of the four-stage system to allow for biological phosphorus removal. In the anaerobic zone, RAS from the clarifiers and influent wastewater are mixed but not aerated. Both processes (4 and 5-stage Bardenpho) have a relatively longer SRT (10 to 20 days) and enhance nitrifier growth as well as carbon oxidation capability. The second anoxic zone provides additional denitrification using nitrate produced in the aerobic zone as the electron acceptor and the endogenous organic carbon as the electron donor (Metcalf & Eddy, 2003).

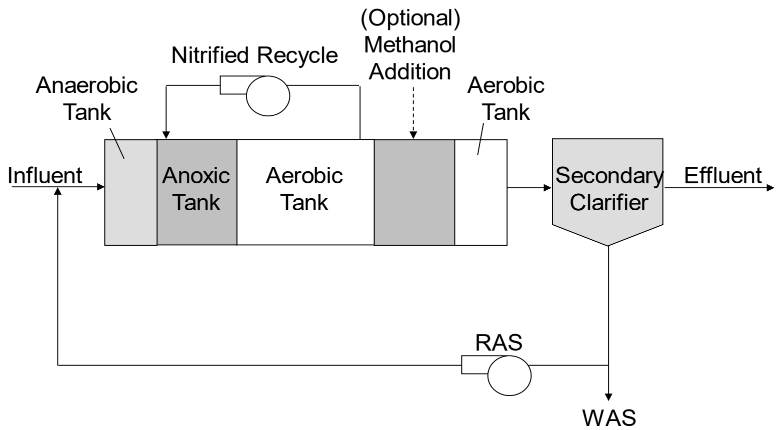

Figure 7-3. 5-Stage Bardenpho Process

7.4 Five-Stage BNR Modifications at HWWTP

In 2010 the North Carolina Division of Water Quality revised the nutrient standards for Falls Lake to significantly lower allowable levels of TN and TP to be implemented starting in January 2016. These standards resulted in a lower annual average TN discharge allocation of 10,422 pounds per year for HWWTP which at a design flow of 2.4 MGD is equivalent to a TN discharge limit of 1.43 mg/l. In 2011, construction began on upgrades to HWWTP's biological treatment system and included reconfiguring existing basins to a conventional 5-stage BNR treatment facility based on a design annual average TN effluent criterion of 3 mg/l. The upgrades included reconfiguring the existing aeration tanks based on a 5-stage Bardenpho system, new surface mixers, baffle walls, and process instrumentation. This was the first part of a three-stage upgrade with an anticipated second stage future upgrade to include either reverse osmosis, ion exchange, or equivalent process treatment to be added downstream of the upgraded BNR system.

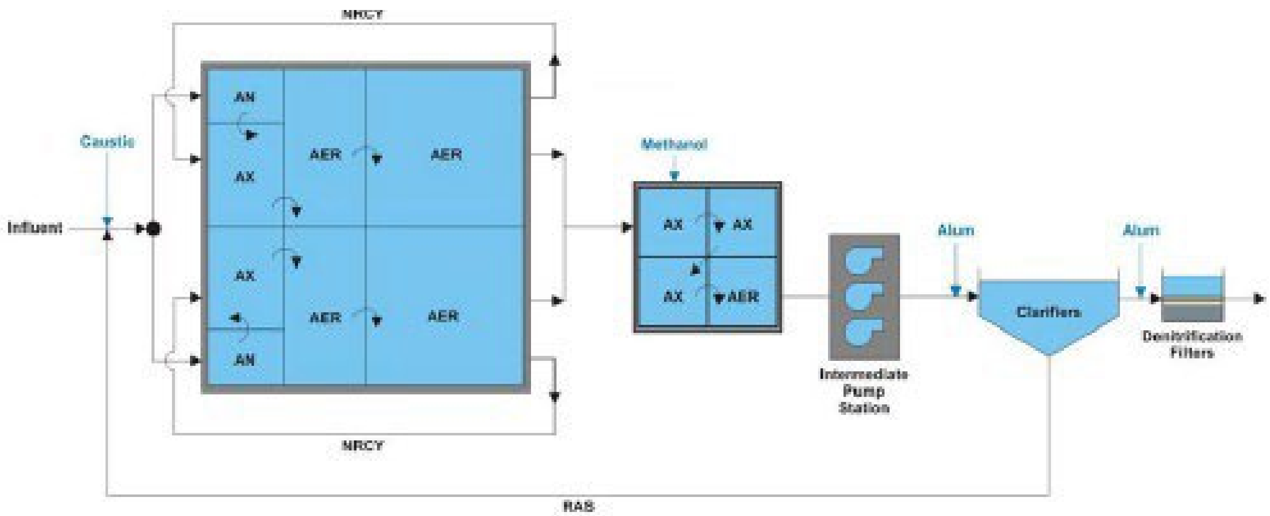

Figure 7-4. HWWTP 5-Stage BNR Configuration (Source: Mahagan and Bilyk, 2016, with permission)

The new 5-stage BNR system was put in service in October 2013. The facility reported that system sizing criteria for each BNR stage were based on a calibrated BioWin model. As explained in section 7.6 below, the facility was able to meet the design criteria for a TN of approximately 3 mg/l starting in mid-May 2014, with effluent concentrations consistently around or below 3 mg/l TN. In addition, HWWTP operations staff took the initiative to further optimize the process and were able to meet further performance improvements while addressing a number of operational challenges explained in section 7.6.

7.5 Detailed Statistical Analysis – Plant Effluent Concentrations

Facility operating data from January 2015 to December 2017, reflecting a period of optimized operations after the improvements were in place, were analyzed. Figures 7-6 through 7-9 and Tables 7-3 through 7-6 provide a summary of the statistical analysis performed for the HWWTP facility in Hillsborough, NC. As explained below, the data shows that the facility consistently met the final effluent permit requirements for total nitrogen shown in Table 7-2 throughout the analysis period.

Figure 7-5 shows the 12-month rolling average TN discharge loadings and shows the plant's consistency in meeting the annual discharge loading limits from month to month. The annual TN discharge loading limit for 2015 was 50,228 lb/yr while for 2016 and 2017, the annual limit was 10,422 lb/yr. At the design flow of 2.4 MGD, these loads correspond to annual average effluent TN concentrations of 6.9 mg/l TN (2015) and 1.4 mg/l TN (2016 and 2017). Actual annual discharged TN loads were 6,535 lb/yr, 4,711 lb/yr, and 5,533 lb/yr in 2015, 2016, and 2017, respectively, all well below their discharge loading limits. The annual average influent flows for 2015, 2016, and 2017 were 1.07, 1.10, and 0.95 MGD, respectively.

For ammonia, HWWTP's permit includes monthly and weekly ammonia concentrations of 2 mg/l and 6 mg/l, respectively. As shown in Figure 7-7 below, all effluent ammonia concentrations were well below limits throughout the 3-year period.

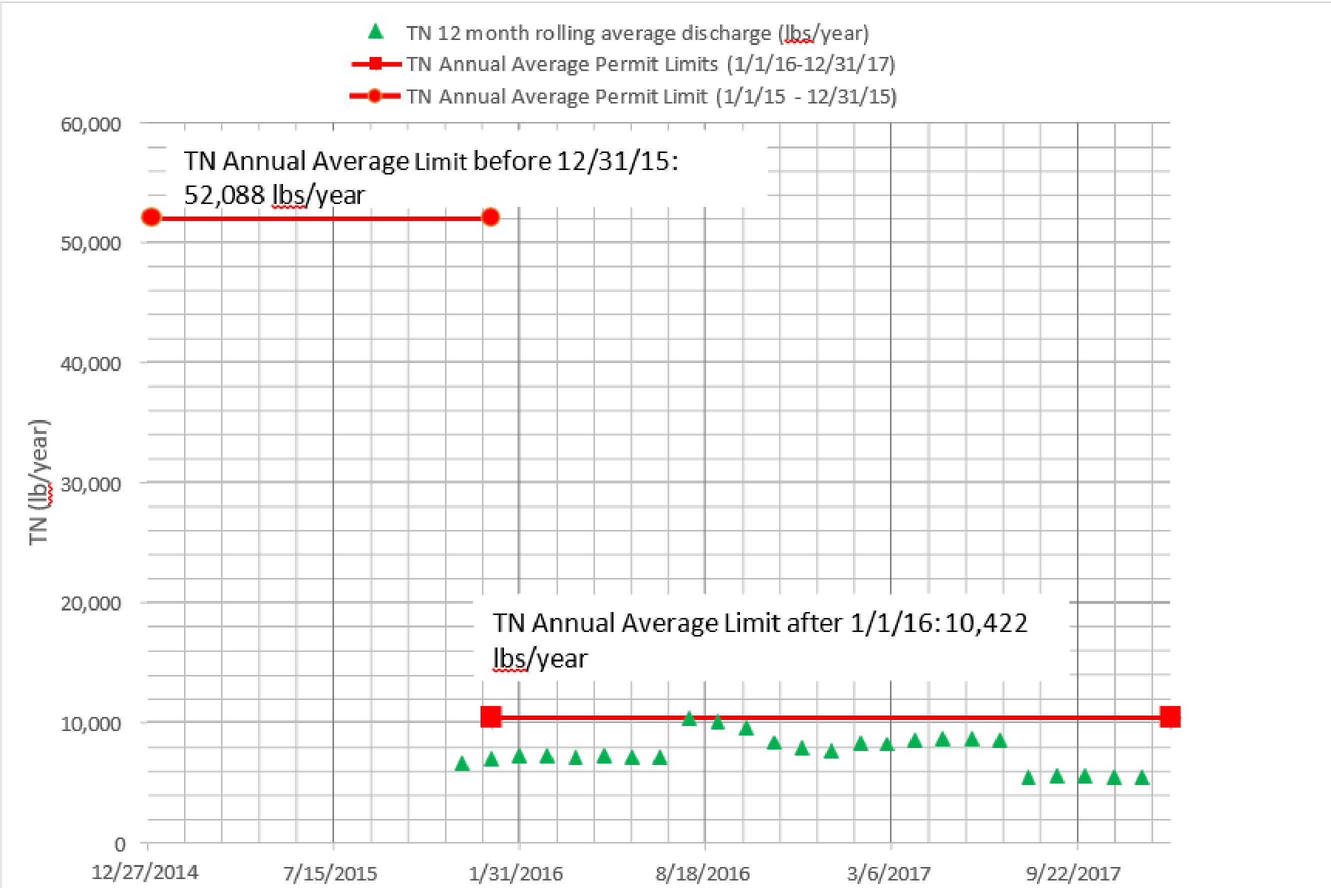

Figure 7-5. 12-Month Rolling Average Time Series Plot for TN

Figure 7-6 shows the 30-day rolling average time series plot for effluent nitrogen species concentrations at HWWTP. For the 3-year analysis period, the median 30-day rolling average TN concentration was 1.57 mg/l with a maximum value of 5.34 mg/l.

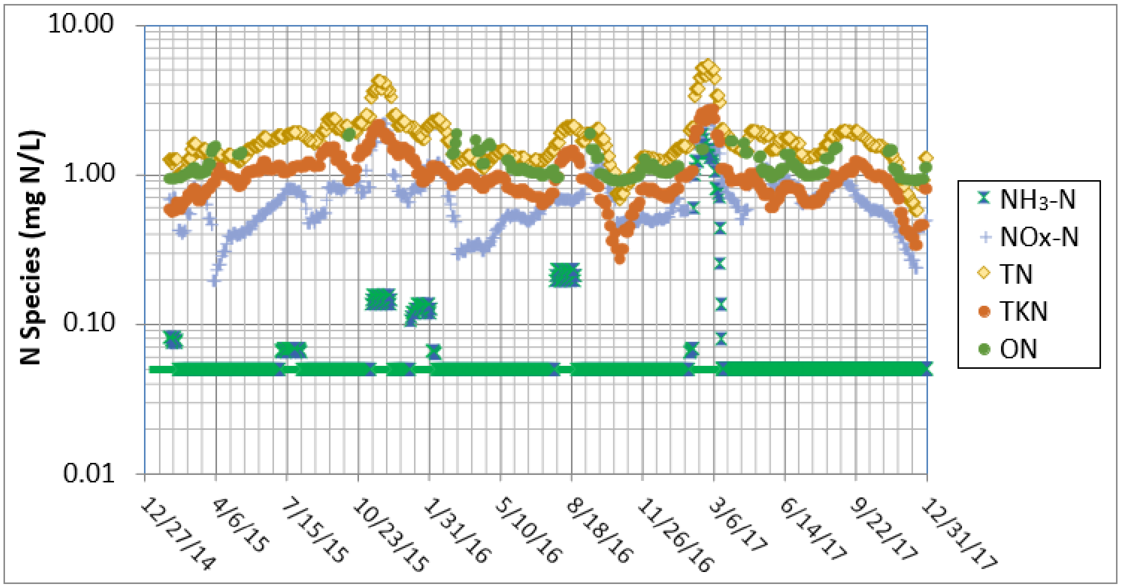

Figure 7-6. 30-Day Rolling Average Time Series Plot

Figure 7-7 shows the individual sample time series plot for effluent nitrogen species concentrations at HWWTP. Samples are generally collected once a week. The chart shows some variability in total nitrogen concentrations. For the 3-year analysis period, the median TN concentration was 1.53 mg/l with a maximum value of 8.33 mg/l.

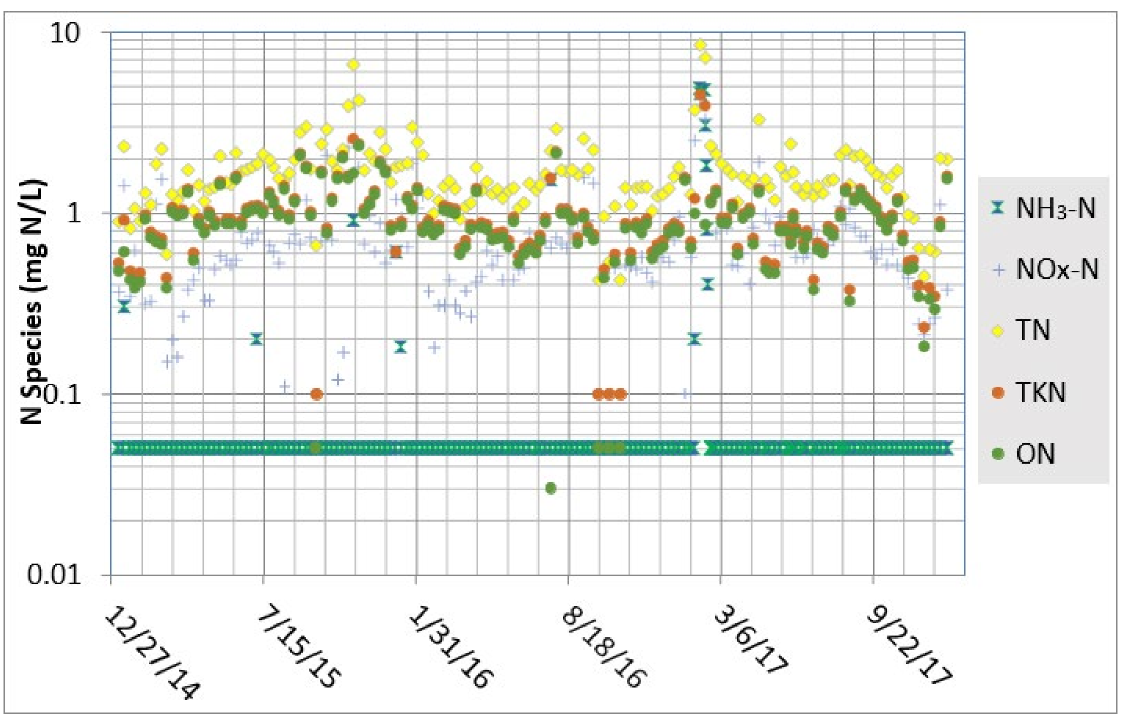

Figure 7-7. Effluent TN Individual Sample Time Series Plot

Figures 7-8 A through D include cumulative probability plots for HWWTP's daily, 30-day rolling average, monthly average, and rolling annual average data sets. A percentile value on the x-axis represents the probability that the value is less than or equal to the stated corresponding concentration on the plot's y-axis. In looking at the nitrogen species values on Figure 7-8A, it is clear that for the most part, the effluent TN is comprised of nitrate nitrogen and organic nitrogen (organic nitrogen values were calculated as the difference between TKN and NH_3-N). The 95th percentile TN daily average concentration for the 3-year analysis period is 2.97 mg/l.

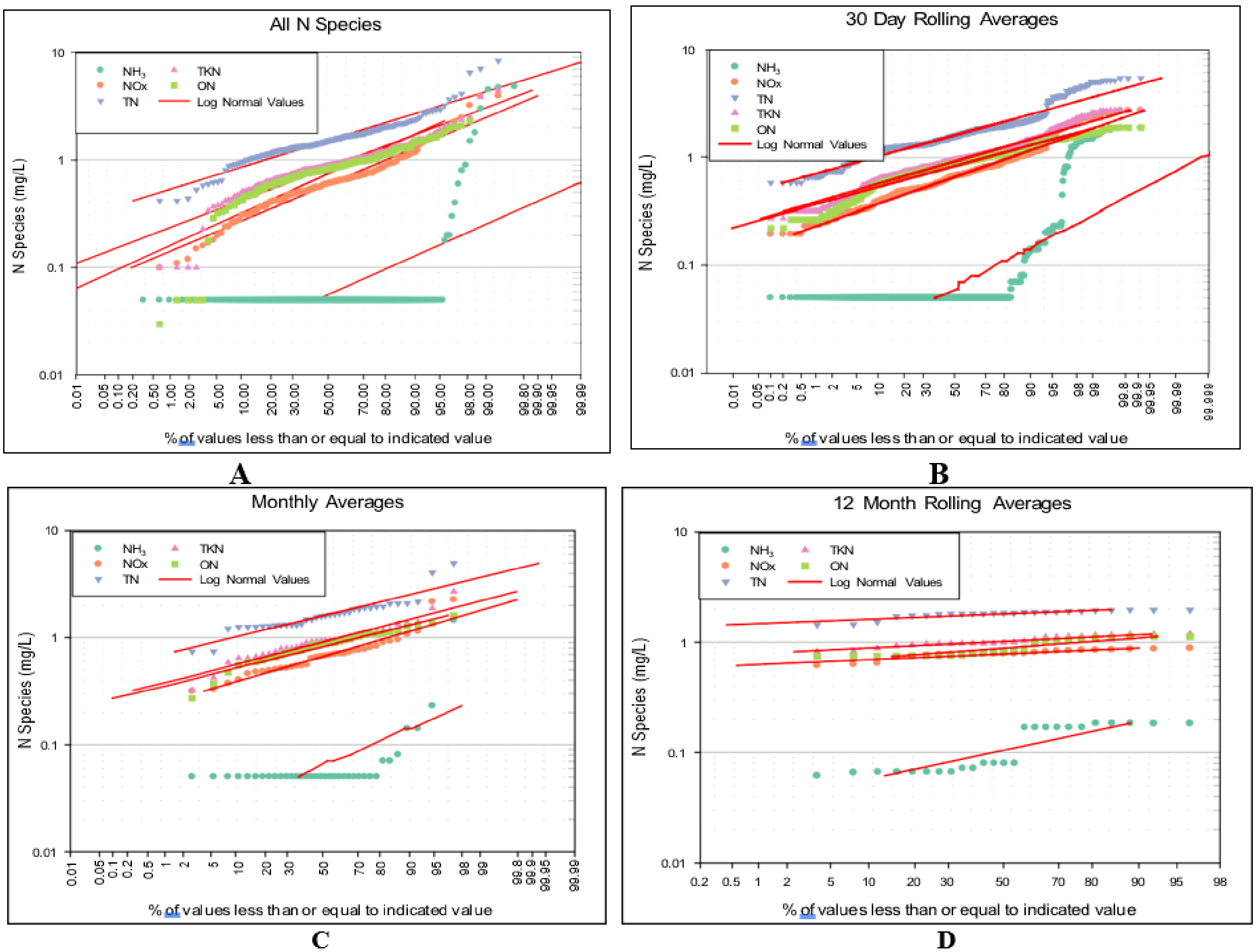

Figure 7-8. Probability Plots for HWWTP – (A) Daily Data; (B) 30-day Rolling Average; (C) Monthly Average; (D) 12-Month Rolling Average

Figure 7-9 below show the various cumulative probability percentiles for the nitrogen species. The graph can also be used to highlight the process variability for TN and NH_3. For example, the daily TN 50th percentile effluent concentration was 1.53 mg/l and the 95th percentile was 2.96

mg/l for TN, with a 95 percent/50 percent of about 1.93 demonstrating significant variability typical of facilities meeting very low TN limits.

Table 7-3. Summary Statistics for Final Effluent Ammonia Nitrogen for HWWTP

	NH_3-N Daily Data	NH_3-N Rolling 30-day Average	NH_3-N Weekly Averages	NH_3-N Monthly Averages	NH_3-N 12-Month Rolling Average
n	319	1,067	156	36	25
Mean	0.12	0.11	0.11	0.10	0.12
Geometric Mean	0.057	0.06	0.06	0.06	0.11
Standard Dev.	0.50	0.24	0.45	0.23	0.053
CV	4.04	2.30	4.00	2.33	0.46
Skew	8.18	5.45	8.88	5.79	0.26
Minimum	0.050	0.050	0.05	0.050	0.062
Maximum	4.80	1.85	4.65	1.45	0.18

Table 7-4. Summary Statistics for Final Effluent NOx-N for HWWTP

	NOx-N Daily Data	NOx-N Rolling30-day Average	NOx-N Weekly Averages	NOx-N Monthly Averages	NOx-N 12-Month Rolling Average
n	157	1,067	146	36	25
Mean	0.73	0.73	0.73	0.74	0.79
Geometric Mean	0.60	0.66	0.59	0.67	0.78
Standard Dev.	0.58	0.389	0.58	0.412	0.070
CV	0.79	0.53	0.80	0.56	0.09
Skew	3.24	2.27	3.24	2.66	-1.07
Minimum	0.100	0.195	0.10	0.315	0.62
Maximum	3.96	2.68	3.96	2.26	0.88

Table 7-5. Summary Statistics for Final Effluent Total Nitrogen for HWWTP

	TN Daily Data	TN Rolling 30-day Average	TN Weekly Averages	TN Monthly Averages	TN 12 month rolling Average
n	157	1,067	146	36	25
Mean	1.71	1.71	1.72	1.71	1.80
Geometric Mean	1.53	1.60	1.53	1.60	1.80
Standard Dev.	1.00	0.72	1.02	0.77	0.14
CV	0.58	0.42	0.59	0.45	0.08
Skew	3.78	2.40	3.75	2.85	-1.51
Minimum	0.42	0.57	0.42	0.74	1.44
Maximum	8.33	5.34	8.33	4.95	1.96

Table 7-6. Summary Statistics for Final Effluent ON for HWWTP

	ON Daily Data	ON Rolling 30-day Average	ON Weekly Averages	ON Monthly Averages	ON 12 month rolling Average
n	155	1,067	144	36	25
Mean	0.87	0.88	0.88	0.87	0.90
Geometric Mean	0.75	0.83	0.76	0.82	0.88
Standard Dev.	0.40	0.28	0.40	0.28	0.15
CV	0.45	0.32	0.46	0.33	0.17
Skew	0.80	0.51	0.80	0.26	0.45
Minimum	0.03	0.22	0.03	0.27	0.74
Maximum	2.31	1.86	2.31	1.61	1.12

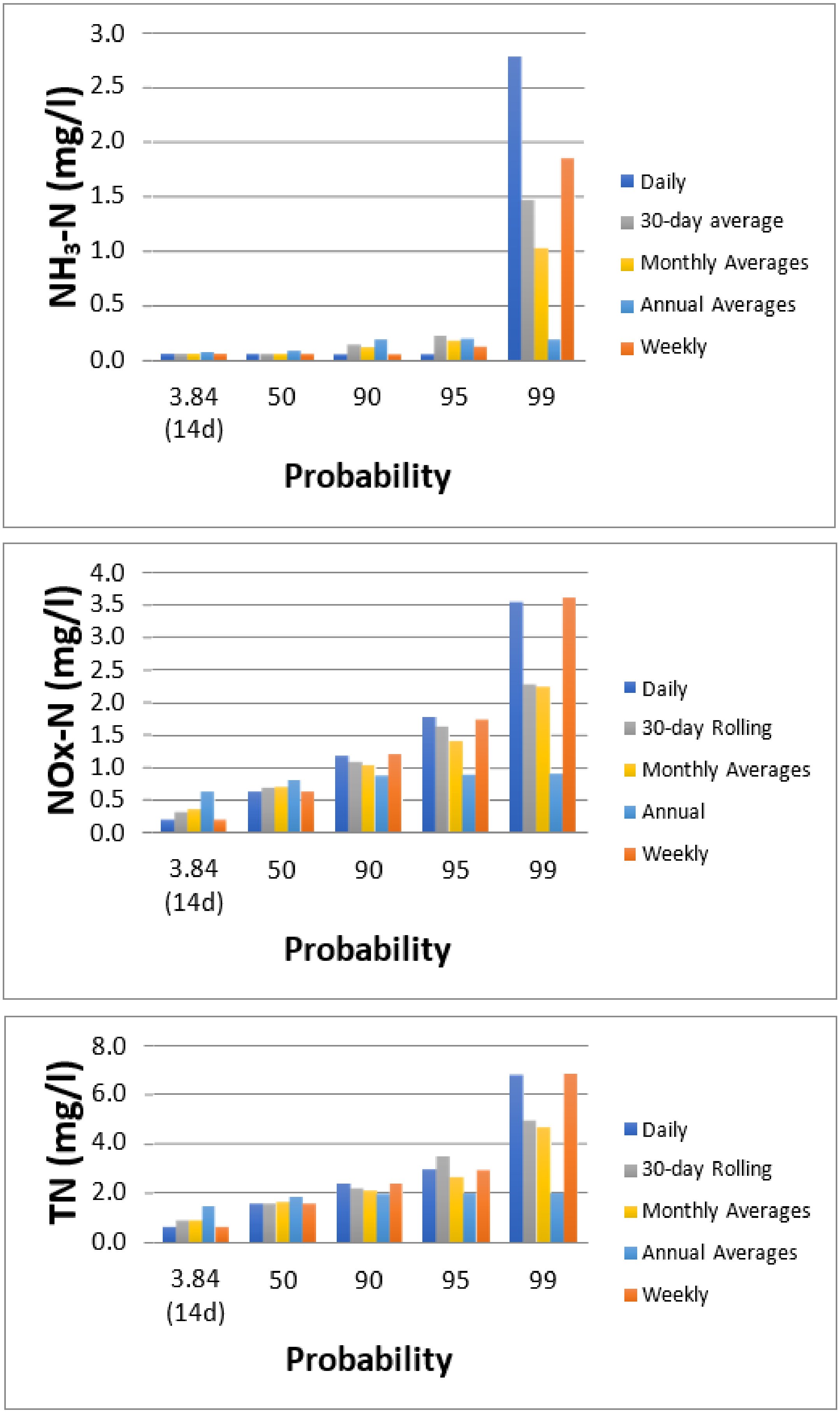

Figure 7-9. Probability Summary for HWWTP

7.6 Process Performance and Lessons Learned - Five-Stage BNR Modifications at HWWTP

In looking at the plant's effluent TN concentrations over the 3-year analysis period as shown in Figures 7-6 through 7-8 above and as explained in Section 7.5, it is clear that the facility maintained exceptional performance for a 5-Stage BNR system for the entire 3-year period. As mentioned in Section 7.4, the new 5-stage BNR system was put in service in October 2013 and system sizing criteria for each BNR stage were based on a calibrated BioWin model. Reported results from the first six months in 2014 plotted in Figure 7-10 below show that the system met the design criteria for a TN of approximately 3 mg/l starting in mid-May 2014, with effluent concentrations consistently around or below 3 mg/l TN.

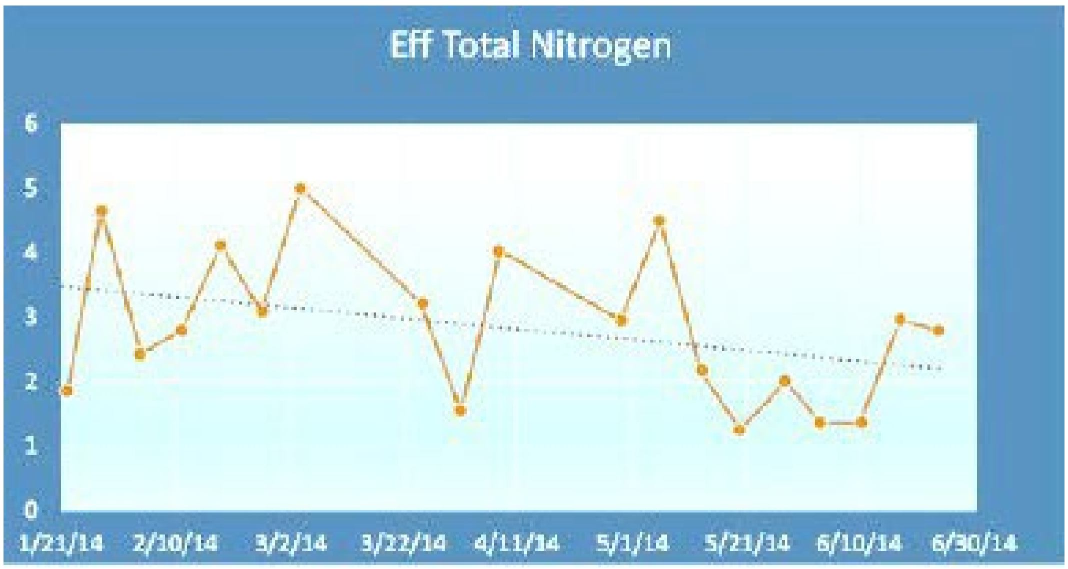

Figure 7-10. Initial BNR Performance in 2014
(Source: Mahagan and Bilyk, 2016, with permission)

While the facility achieved significant improvement in effluent TN concentrations, it should be noted that the BNR system experienced initial variability in effluent TN, lower pH, poor sludge settleability in secondary clarifiers, and higher than expected TP concentrations. As a result, the facility initiated the addition of caustic soda at high doses to control the pH. As the pH was raised above 6.6, TN effluent concentrations stabilized and sludge settleability improved. Biological phosphorus removal was also being observed.

The facility then initiated an effort to evaluate additional system and process improvements to ensure continued high performance and consistent and stable process operation. This involved development of a simple model to determine the detention times at each stage at various flows based on the total flow leaving each zone as an indication of the adequacy of each zone's sizing. After reviewing the model, staff determined that at current flows, aerobic zones volume could be significantly reduced, and the internal nitrogen recycle rates could be significantly increased, resulting in better process control and improved effluent quality. As a result, system modifications were completed at the end of June 2014 and in full operation by July 2014. A summary of the modifications is included below, and a detailed discussion of the 5-stage BNR configuration and equipment modifications and process control strategies is reported by Mahagan and Bilyk, 2016. In general, treatments plants may find opportunities to reconfigure flow through existing basins and effect improvements to enhance BNR treatment allowing the plant to achieve enhanced biological treatment (EPA, 2010). The modifications implemented by HWWTP described below are a good example of successful process enhancements that may be implemented by facilities that are below design capacity. The modifications further improve performance and achieve enhanced denitrification by modifying BNR system zones and internal mixed liquor recycle rates. The biodegradable carbon that would otherwise be oxidized aerobically can be used instead to fuel further nitrate removal.

The modifications to the BNR zones included reallocating a portion of the first aerobic zone to anoxic volume as shown in Table 7-7. In addition, the nitrogen recycle flow rate was increased to 900 percent to keep this new anoxic volume mixed and maximize denitrification capacity. As a result, the facility reported that effluent TN concentrations were dramatically reduced, initially from around 3 mg/l to 1.5 mg/l and averaging 1.38 mg/l over the next 12 months. No pH adjustment was needed during that time, and no additional carbon source or coagulant was used to achieve these results (Mahagan and Bilyk, 2016).

Table 7-7. Comparison of Zone Volumes and Detention Times Before and After Modifications to the Original Design (Adapted from Mahagan and Bilyk, 2016)

	Original (Nov 2013 through June 2014)				Modified (July 2014 through Sep 2015)		
Zone	Volume (MG)	% of Volume Allocated	NRCY % of Inf		Volume (MG)	% of Volume Allocated	NRCY % of Inf
Anaerobic	0.125	6%			0.125	6%	
1st Anoxic	0.375	17%	200%		0.875	39%	900%
Aerobic	1.5	67%			1	44%	
2nd Anoxic	0.1875	8%			0.1875	8%	
Reaeration	0.0625	3%			0.0625	3%	
Avg Influent Flow: 1.038 MGD					*Avg Influent Flow: 0.898 MGD*		

The facility also implemented a process control strategy that included several components. As the first aerobic zone volume was reduced, a tapered aeration approach was implemented with DO setpoints of 2.2 to 3.0 mg/l at the head of the zone and 0.5 - 1.5 mg/l at its end. Additionally, in anticipation of storm events, operators implemented a procedure to pace the nitrogen recycle pumps to maintain a desired anoxic zone detention time. This included gradually reducing the speed of the nitrogen recycle pumps manually to reduce the possibility of recycle flow spikes (that would cause shorter anoxic zone detention times resulting in elevated effluent TN). Additionally, the plant modified detention time setpoints through the plant's supervisory control and data acquisition (SCADA) system that modulate the speed of the recycle pumps to maintain a desired detention time setpoint. The facility reported that this was done to allow for increasing the first anoxic zone detention time. This was based on sampling the end of the first anoxic zone, testing for nitrate concentrations and using the result to increase the detention time setpoint if the nitrate concentration is above the target (usually 0.3 to 0.5 mg/l).

The facility does not fully know the exact scientific cause of the significant enhancement of nitrogen removal to levels well below those typical to 5-Stage BNR processes. It suspects that several causes worked together to attain this significant level of treatment. The treatment level was attained without the addition of costly external carbon addition, except for small amounts of methanol during short periods of time. The increase in the NRCY rate was likely an important contributor to the success. Another potential cause of significant operational improvement was thought to be related to conditions selecting for organisms that are more adept at hydrolyzing slowly biodegradable influent BOD under anoxic conditions. The plant believed this was likely aided by the splitting of the first anoxic zone into two separate compartments, the first compartment serving as an anoxic selector and the second compartment being completely anoxic and approaching anaerobic conditions. Anaerobic conditions are/were demonstrated by the very low nitrate found at the end of the second compartment of the first anoxic zone. The second anoxic zone also has separate chambers with the three chambers resulting in a plug flow and each compartment becoming more anoxic. HWWTP staff believe this is part of the reason for the success in removing nitrate without additional carbon added.

Finally, in considering the challenges encountered and lessons learned by the facility after implementing the above, two brief operational incidents of reduced performance occurred around November 2015 and February 2017 and were addressed by the facility. The first occasion in November 2015 saw elevated TN effluent concentrations (Figure 7-6) starting on November 3rd (3.7 mg/l) and as high as 6.5 mg/l on November 10th before going back to typical concentrations on November 24th (1.69 mg/l). This incident was reported to be caused by the impact of large discharges of drinking water plant sludges which caused organic nitrogen spikes. This was resolved by controlling these discharges, thus eliminating the spike loads.

The second incident occurred in February 2017, with higher plant effluent TN concentrations starting on January 31 (3.7 mg/l) and peaking at 8.3 mg/l on February 7th before subsiding on February 21st (2.3 mg/l). The facility reported that although the specific cause was not completely identified, they suspected that the cause was a chemical discharge to the collection system which inhibited nitrification; ammonia nitrogen levels measured between February 7th and February 14th were in the 7 – 8 mg/l range with no other causes identified. The ammonia levels started to subside gradually reaching typical levels on February 20th.

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