# Process Control Strategies for Biological Nutrient Removal in an Oxidation Ditch

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The occurrence of same-time nitrification and denitrification, within a single reactor and without distinct aerated and nonaerated zones, is commonly referred to as simultaneous nitrification/denitrification (SND). Wastewater treatment systems exhibiting SND typically have relatively long mean cell residence times (MCRT); aeration equipment that creates nonuniform flows, such as mechanical aerators; and an operating procedure to limit oxygen input (Daigger, 2014). Three mechanisms have been proposed for SND, including the existence of: 1) aerobic and anoxic zones within the reactor, 2) aerobic and anoxic zones within floc particles, and 3) novel microorganisms with alternative biochemical pathways. The SND processes can be difficult to control because they depend largely on the bioreactor configuration, bulk dissolved oxygen (DO) concentration, and floc size (Jimenez, 2014); however, significant advantages of SND over conventional biological nitrogen removal (BNR) include: 1) reduced tank requirements and 2) reduced consumption of carbon, oxygen, and alkalinity.

Activated sludge models (ASMs) are used in the design, upgrade, and optimization of wastewater treatment plants. Modeling can be a powerful tool for troubleshooting and increasing understanding of plant operations; however, there have been few published modeling studies of SND systems. The overall goal of this study was to develop, calibrate, and verify a SND process model of the Falkenburg Advanced Wastewater Treatment Plant (AWWTP) in Hillsborough County in Tampa; a preliminary assessment of enhanced biological phosphorous removal (EBPR) was also performed. The AWWTP uses a Carrousel® oxidation ditch system to achieve SND. BioWin model calibration was performed using whole-plant influent, effluent, and operational data. The calibrated model was used to assess the facility's operations and recommend improvements in process control strategies. Although the plant continually meets and exceeds its permit requirements, improvements in process control strategies have the potential to improve energy efficiency and decrease chemical use, sludge production, greenhouse gas emissions, and costs.

## Materials and Methods

### Site Description and Model Setup

The AWWTP is a BNR facility, with an annual average influent flow rate of 9.27 mil gal per day (mgd) and a permitted annual average flow rate of 12 mgd. The plant has permit limits for biological oxygen demand ($BOD_5$), total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP) of 5, 5, 3, and 1 mg/L, respectively. Oxidation ditches are used to achieve SND and phosphorus uptake. The oxidation ditches are preceded by anaerobic selectors, which improve sludge settleability and initiate EBPR. Aluminum sulfate (alum) addition is used for additional phosphorous removal. The oxidation ditches at the AWWTP were modeled as a loop of 10 unaerated, completely stirred tank reactors (Abusam, 2001) and two mechanically aerated reactors, equally dividing the volume of all four trains (Figure 1).

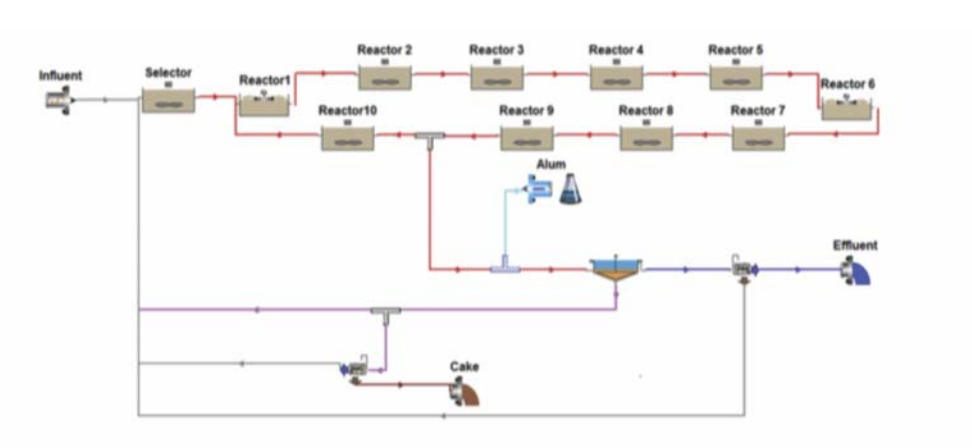


Figure 1. Falkenburg AWWTP Layout in BioWin

### Model Calibration and Verification

Variables chosen for model calibration and verification were mixed liquor suspended and volatile suspended solids (MLSS, MLVSS), and effluent nitrogen species (Total Kjeldahl Nitrogen [TKN], $NH_4^+$, $NO_3^-$, $NO_2^-$). Historical data from a three-year period (Sept. 1, 2010 to Aug. 31, 2013) were exported from the AWWTP Hach WIMS™ system and used for calibration (Sept. 1, 2010 to Aug. 31, 2011) and verification (Sept. 1, 2011 to Aug. 31, 2012). Since only average daily flows were available from historical data, a rough estimate of diurnal influent flow patterns was obtained by viewing supervisory control and data acquisition (SCADA) trends over a 24-hour period. The typical diurnal flow pattern was applied to all average daily flows to create hourly influent data sets.

### Analytical Methods

Measurements of TSS and volatile suspended solids (VSS) and total and filtered chemical oxygen demand (COD) were performed on composite influent and effluent samples using *Standard Methods for the Examination of Water and Wastewater* (APHA et al, 2012). Readily biodegradable COD (rbCOD, flocculated-filtered COD) was measured using the method of Mamais et al (1993). Grab samples were also collected to investigate biological phosphorous removal from four sample points along the treatment train: 1) influent, 2) anaerobic selector, 3) oxidation ditch, and 4) secondary clarifier. A portion of samples 2, 3, and 4 were allowed to settle for several minutes to obtain supernatant samples, which were immediately filtered with 0.45µm syringe filters. Samples were placed on ice and analyzed within eight hours of collection for total and reactive phosphorus using Hach (Loveland, Colo.) TNT 843 and 845 kits (*Standard Methods* 4500E).

**Model Goodness of Fit and Sensitivity**

The average sum of absolute residuals (SAR) in Equation 1 was calculated to determine goodness of fit of modeled to observed concentrations of effluent ammonia, nitrate, and nitrite.

$$Average\ SAR = \frac{\sum^{n}_{(i=1)} |y_{m,i} - y_{o,i}|}{n} \qquad [1]$$

Where $y_m$ is the modeled output, $y_o$ is the observed output and n is the number of SARs that were calculated for each simulation.

These values were compared for several simulations with different arrangements of four kinetic parameters (Table 1). The adjusted parameters were "heterotrophic DO half sat.", "aerobic denit. DO half sat.", "ammonia oxidizer DO half sat.", and "anoxic nitrite half sat." switching functions. The heterotrophic and aerobic denit. DO half-saturation constants were combined into one parameter in the latest BioWin edition; an older edition of BioWin was used in this study, and the two parameters were kept equal for compatibility with newer versions. The number of simulations and combination of parameters were limited due to time constraints. The heterotrophic and aerobic denit. DO half-saturation constants were adjusted based on suggestions in the literature (Envirosim, n.d.) and previously published SND modeling work (Jimenez, 2010). Other model parameters may achieve a better fit to observed data; however, parameter adjustment should be done with care to avoid unrealistic values. Note that the yearlong simulation period resulted in a relatively long simulation time of approximately four to five hours.

Sensitivity analysis of the BioWin model was performed to determine which parameters were the most influential to the outputs of the model. Five parameters were chosen for the sensitivity analysis based on previous modeling by Jimenez et al (2010). A normalized sensitivity coefficient method (Eqation 2; Liwarska-Bizukojc et al, 2010) was used to compare the percent change in output value to a 10 percent change in input values (note that some rounding off was required).

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Table 1. Combination of Kinetic Parameters Tested During Model Calibration

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

*BioWin default kinetic parameter values

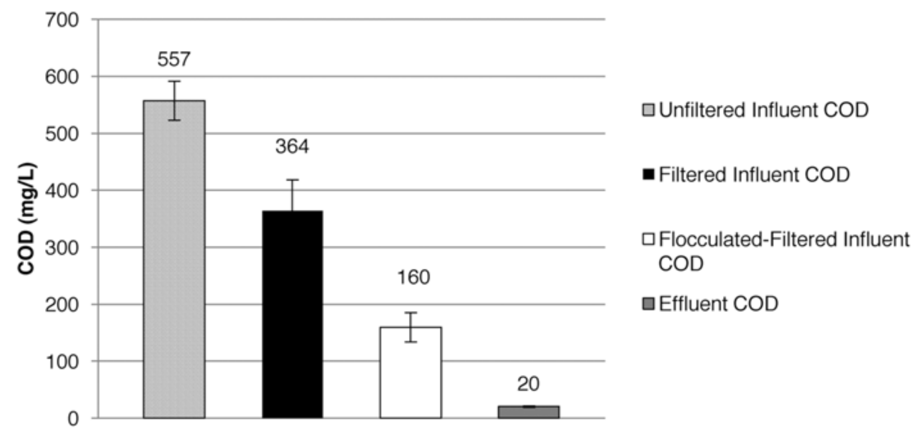


Figure 2. COD Results From Wastewater Characterization for Influent n=5 and Effluent n=2

Table 2. Influent TSS and VSS in Composite and Grab Samples

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

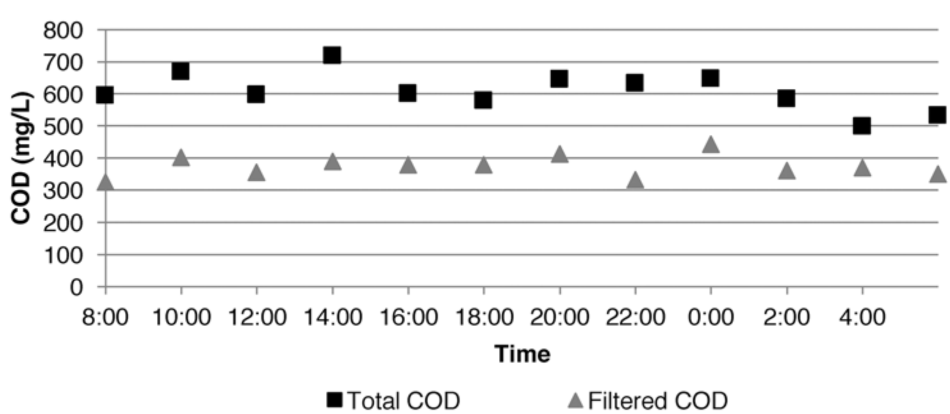


Figure 3. Total and Filtered Influent COD

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$$S = \frac{(\Delta y/y)}{(\Delta x/x)} \quad [2]$$

Where S is the sensitivity coefficient, y is the output value (e.g., nitrate) and x is the input value (e.g., half-saturation coefficients).

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

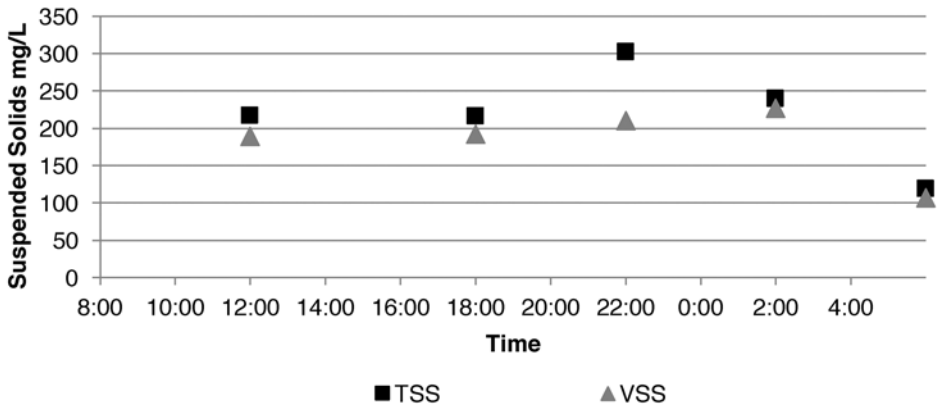


Figure 4. Total and Volatile Influent Suspended Solids

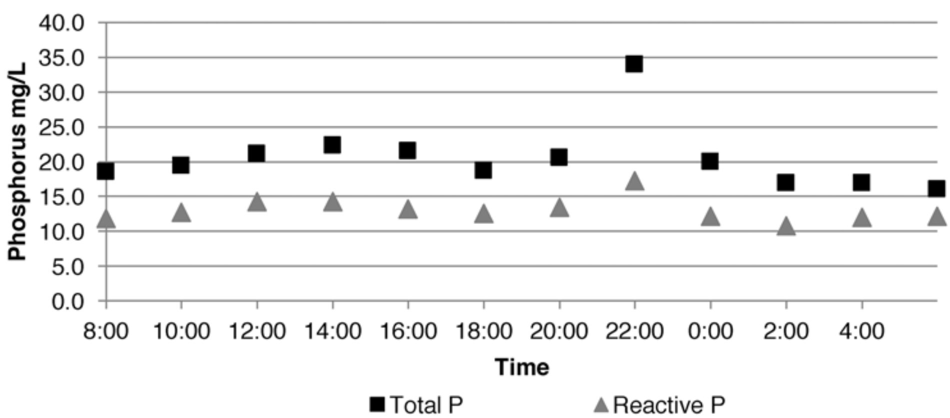


Figure 5. Total and Reactive Influent Phosphorus

Table 3. BioWin Wastewater Fractions

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

## Results and Discussion

The results of the COD analyses on influent and effluent samples are shown in Figure 2. All influent samples were 24-hour composites and secondary effluent samples were grab samples. Note that the effluent grab sample was collected from the secondary clarifier effluent prior to the media filters. The TSS and VSS values for composite and grab samples are shown in Table 2. BioWin requires volatile or inert suspended solids concentrations to be input into the model. As only historical TSS data were available, VSS concentrations were estimated using the average VSS/TSS ratio determined during supplemental sampling.

Total and filtered COD, TSS, VSS, and total and reactive phosphorus concentrations were measured over a 24-hour period, and the results are shown in Figures 3, 4, and 5. The hourly influent flow was also recorded and used to calculate the mass load per day of each constituent (not shown). Noticeable peaks for both phosphorus and TSS were observed at 10 p.m. The color of the sample at that time was uncharacteristically black, and the results from this sample were not used for estimation of influent characteristics. Average values obtained from historical data or during supplemental sampling were compared with typical values from wastewater facilities in the United States. The historical and measured values mainly fell within the medium to medium-high range.

A commonly encountered issue with activated sludge modeling is the lack of needed input data. For this study, some wastewater fraction values were calculated using the results from the COD analyses, while others, such as the unbiodegradable particulate fraction, were estimated using the BioWin Influent Specifier Excel worksheet. The wastewater fractions that were input into BioWin are shown in Table 3. Kinetic parameters that were used to model nitrification and denitrification within the oxidation ditch are shown in Table 4. Tables 3 and 4 also show comparisons between the calculated and calibrated values and the BioWin default parameters.

Modeled and observed MLSS values are shown in Figure 6. A better fit might be possible if the wasting rate were adjusted to more accurately reflect dynamic plant wasting activated sludge (WAS) wasting rather than using a constant average value (0.234 mgd). A poor fit was observed between modeled and observed

MLVSS (data not shown), most likely due to alum addition for phosphorous removal, which was not incorporated into the model. Metal hydroxides, such as those formed during alum addition, are oxidized during VSS analysis in the muffle furnace, which will result in a falsely high MLVSS concentration (Jeyanayagam and Husband, 2009).

Modeled and observed TKN results are shown in Figure 7. The observed data were consistently below the model output, most likely due to additional nitrification in the filters, which was not accounted for in the model. Modeled spikes in effluent TKN concentrations corresponded with high influent TKN loads experienced at the facility. At the Falkenburg facility, operators adjust mechanical aerator speeds based on influent ammonia loads; however, the model maintained a constant DO set point. Fine-tuning model aeration settings to better reflect practices at the facility could improve the model goodness-of-fit.

The results of sensitivity analysis (Table 5) show that ammonia oxidizing bacteria (AOB) maximum specific growth rate has the greatest influence on effluent ammonia concentrations. The maximum specific growth rate for nitrite oxidizing bacteria (NOB) does not influence effluent ammonia, but influences effluent nitrite and nitrate. The combined heterotrophic/aerobic denitrification DO half-saturation constant influences effluent nitrate, as this constant switches on the activity of anoxic heterotrophs at low DO. The AOB DO half-saturation constant was only slightly influential on effluent ammonia, while the anoxic nitrite half-saturation constant mainly influenced effluent nitrate. The anoxic nitrite half-saturation constant switches off anoxic growth process at low nitrate concentrations. Additional bench-scale tests are currently being conducted to understand the fate of nitrogen in the system under varying operating conditions. These studies will also allow a comparison of kinetic parameters for SND models obtained using both model calibration, with whole plant data and experimental studies.

The results of the analysis of total and reactive phosphorus at various points in the treatment train are shown in Figure 8. Although the samples size (n=2) is low, the results indicate that EBPR is taking place at the facility. A characteristic release of phosphorus is observed in

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Table 4. Biowin Kinetic Parameters

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

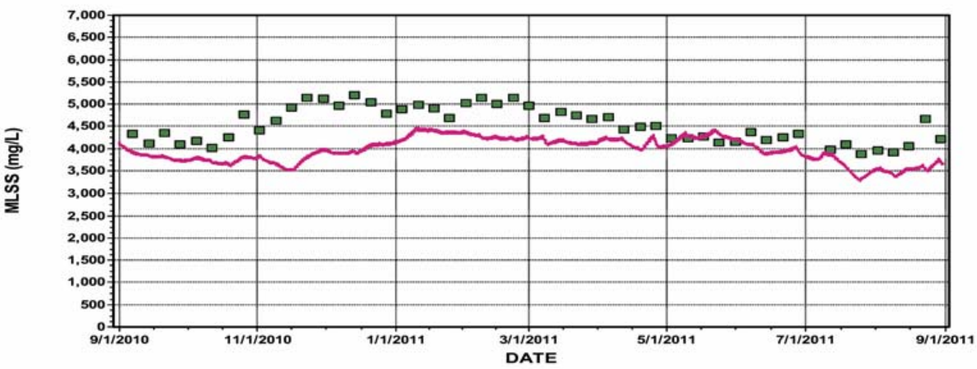


Figure 6. Observed (green squares) and Modeled (pink line) MLSS Concentration

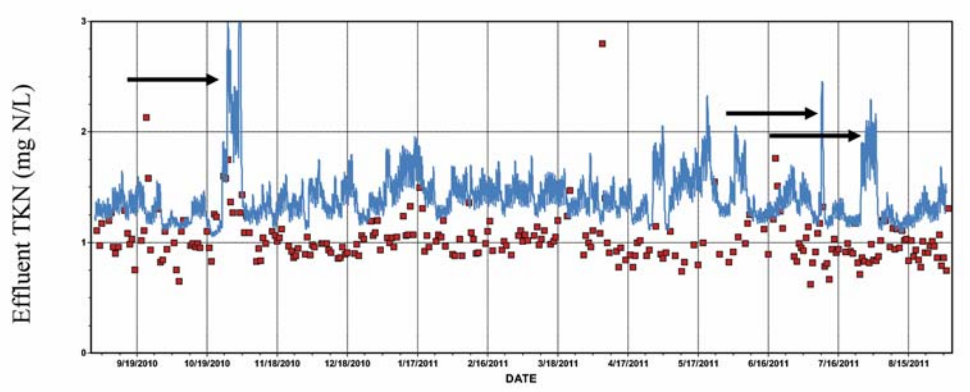


Figure 7. Observed (red squares) and Modeled (blue line) Effluent TKN Concentration

Table 5. Sensitivity Analysis for Kinetic Parameters

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

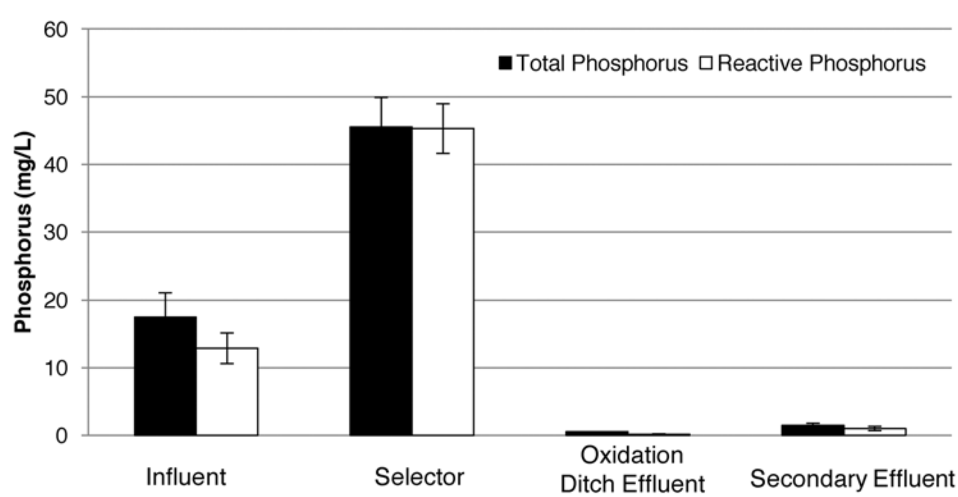


Figure 8. Reactive Phosphorus Profile From Grab Samples Taken Throughout the Treatment Process

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the unaerated selector, followed by very low phosphorus in the effluent of the aerated reactor. The average release of phosphate in the selector was 32 mg/L, and the total amount of phosphate removed was 45 mg/L. Similar phosphate release and uptake were reported by Henze (2008), with a phosphate release of 45 mg/L, uptake of 57 mg/L, and total removal of 12 mg/L. It is not possible to assume the removal of phosphorous in the ditch is fully attributed to EBPR since alum is also added for chemical phosphorus removal. Alum is dosed at a constant rate of ~260 gal per day (gpd) into a splitter box after the oxidation ditches and before the secondary clarifiers. Flow-pacing of alum was recommended to reduce chemical costs, sludge production, and possible impacts of alum on the biological process.

## Conclusions

Operations staff at the AWWTP consistently meet and exceed National Pollutant Discharge Elimination System (NPDES) permit limits; however, improvements in operations have the potential to reduce sludge production and energy and chemical use. These savings will reduce emissions of greenhouse gases and costs to the county ratepayers.

A BioWin model was created for the AWWTP. Data compilation and reconciliation conducted during this study highlighted many good practices in plant operation and monitoring. Three areas where improvements could be made to advance efficiency of operation were identified during this study: 1) flow-pacing alum addition for phosphorus removal, 2) adjusting WAS wasting based on MCRT, and 3) implementation of online aeration control based on ammonia concentration. This project also provides an example of the use of BioWin to model SND processes. Additional bench-scale experiments are currently being conducted to understand SND kinetics under varying temperature and DO concentration conditions.

## Acknowledgments

This work was possible with the help of the Hillsborough County Public Utilities Department and the operations staff at the Falkenburg Advanced Wastewater Treatment Plant. We greatly appreciate the cooperation of Dan Orlosky, plant manager; Marcus Moore, plant supervisor; and the plant operators and their help in collecting samples and data.

This material is based upon work supported by the National Science Foundation

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(NSF) under Grant No. DUE 0965743. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the NSF.

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