Optimized biological nitrogen removal of high-strength ammonium wastewater by activated sludge modeling
Abdelsalam Elawwad

ABSTRACT

Wastewater containing high ammonium concentrations is produced from various industrial activities. In this study, the author used a complex activated sludge model, improved by utilizing BioWin© (EnviroSim, Hamilton, Canada) simulation software, to gain understanding of the problem of instability in biological nitrogen removal (BNR). Specifically, the study focused on BNR in an industrial wastewater treatment plant that receives high-strength ammonium wastewater. Using the data obtained from a nine-day sampling campaign and routinely measured data, the model was successfully calibrated and validated, with modifications to the sensitive stoichiometric and kinetic parameters. Subsequently, the calibrated model was employed to study various operating conditions in order to optimize the BNR. These operating conditions include alkalinity addition, sludge retention time, and the COD/N ratio. The addition of a stripping step and modifications to the configuration of the aerators are suggested by the author to increase the COD/N ratio and therefore enhance denitrification. It was found that the calibrated model could successfully represent and optimize the treatment of the high-strength ammonium wastewater.

Key words | BioWin© modeling, high-strength ammonium, industrial wastewater, model optimization

INTRODUCTION

Wastewater containing high-ammonium concentrations is produced from various industrial activities, such as the petrochemical, steel manufacturing, pharmaceutical, fertilizer, and food industries (Carrera et al. 2005; Pinzón Pardo et al. 2007; Lay-Son & Drakides 2008). Among these activities, coal coking in the manufacturing of steel produces high-strength ammonium wastewater, with complicated characteristics (Lee et al. 2005; Kim et al. 2007, 2008). Wastewater from coke-ovens contains high concentrations of ammonium, phenols, cyanides, and thiocyanates. Thiocyanate nitrogen that is hydrolyzed into ammonia nitrogen increases the ammonium concentration (Kim et al. 2007, 2008; Morling et al. 2012). The disposal of untreated high-strength ammonium wastewater in water bodies is a significant environmental problem because of the harmful effect of the free ammonia (FA) on the aquatic life and environment (Carrera et al. 2005; Lay-Son & Drakides 2008).

The biological nitrogen removal (BNR) process, utilizing nitrification and denitrification, is the most common method for removing ammonium from wastewater (Carrera et al. 2005; Van Hulle et al. 2010). This process is accomplished mainly by two different groups of bacteria, namely, the ammonia-oxidizing bacteria (AOB) and the nitrite-oxidizing bacteria (NOB) (Daims et al. 2000). The inhibition of nitrification occurs in conditions of high levels of ammonium, FA, or nitrite (Kim et al. 2007; Van Hulle et al. 2010). Moreover, the nitrification process in high-strength ammonium wastewater requires an extended

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solids retention time (SRT) (Morling et al. 2012). Treatment of industrial wastewater with ammonium concentrations up to 5,000 mg N-NH₄ using the BNR system is often applied in the literature (Carrera et al. 2003). The presence of inorganic carbon is a crucial factor in the nitrification process (Kim et al. 2009). In addition, phosphate is an important element, though it is usually not present in adequate quantities for nitrification in such wastewater (Pinzón Pardo et al. 2007; Lay-Son & Drakides 2008). However, mathematical modeling of the BNR process can be employed to rectify such problems (Moussa et al. 2004; Pinzón Pardo et al. 2007).

In the recent past, extensive research has been conducted into the use of mathematical models to facilitate understanding of the BNR process. Barker & Dold (1997) developed an improved general model for BNR based on the famous activated sludge models (ASM). As an improvement to the general model, the BioWin® ASM has been introduced (EnviroSim, Hamilton, Canada). This model is a general model intended for biological organics, nitrogen, and phosphorus removal. The BioWin model has many useful features, such as pH modeling and ammonia stripping processes (Liwarska-Bizukojc et al. 2013). Obviously, such features are extremely useful for modeling complicated water treatment systems.

The main advantage of having access to a computer model for a particular wastewater treatment plant (WWTP) is the ability to analyze different operational parameters in order to develop an optimum operational strategy. Various research has studied ASM models for the purpose of optimizing the BNR process in municipal WWTPs (Moussa et al. 2004; Brdjanovic et al. 2007; Elawwad et al. 2017). However, a limited number of studies have used ASM models to optimize the BNR process in a full-scale industrial WWTP that receives high-strength ammonium wastewater. Moreover, there is a lack of information on the application of ASM models to similar industrial WWTPs. Therefore, the aim of this manuscript is to evaluate the capability of activated sludge modeling to optimize the performance of the BNR process in full-scale industrial WWTPs that receive high-strength ammonium wastewater. Over the study period, the author has improved their knowledge about the complex biological processes of wastewater treatment, modeling, and the performance of WWTPs. The author expects this knowledge to be of significant value in the future.

**MATERIALS AND METHODS**

An industrial WWTP, located in Helwan, Egypt, was used in the case study. This facility, in operation since 1989, receives high-strength ammonium wastewater from a coke-oven plant in the vicinity. The WWTP was designed for organic and nitrogen removal and employs the oxidation ditches process. Recently, operational problems have been encountered in this plant relevant to nitrogen removal because the total nitrogen level does not meet the Egyptian standard for industrial effluents. Therefore, the process had to be optimized for BNR. The author used the BioWin built-in model to simulate and optimize the process performance of the plant. The study was conducted according to the ‘Good Modeling Practice’ protocol which was developed by Rieger et al. (2012). The BioWin software facilitates analysis of the effects of various operational parameters (such as SRT, HRT, and the like) on the biological treatment performance.

The first step in the study was to understand the plant configurations. The staff of the WWTP provided us with operational and technical data relating to the WWTP’s performance. The plant includes an equalization lagoon with a total volume of 81,600 m³, a buffering tank, two oxidation ditch modules with a total volume of 7,000 m³, and two secondary clarifiers with a total volume of 3,200 m³. A simplified schematic diagram of the industrial WWTP is presented in Figure 1. The industrial wastewater, which has an average flow rate of 3,680 m³/d, flows into the equalization lagoon and is subsequently pumped to the WWTP at a semi-constant rate. The wastewater from the administration buildings of an industrial facility, with an average flow of 480 m³/d, is pumped in at the inlet of the biological reactors with the industrial wastewater flow to compensate for the lack of nutrients encountered in this type of industrial wastewater.

Because of the wide variations of pH in the influent, the pH level is observed and controlled before the oxidation ditches. pH control includes lime and phosphoric acid addition. The return sludge (Q_{return}) is pumped to the inlet of the biological step and mixed with influent. The excess
sludge (Q_{ex}) is drawn out to be dried. Three brush aerators are used to aerate each of the oxidation ditches. Each brush aerator has a constant power supply of 45 kW. Finally, the treated water is recycled and reused in the coke oven process, while the unneeded water is disposed of at a wastewater treatment facility. Operational data and measurements dating back to the start of the operation of the WWTP were obtained from the plant. The raw sewage temperature was 16–32 °C, with an annual average of 23 °C. Details on the operations of the plant and the average annual measurements for the influent and the effluent are available in Elawwad et al. (2016).

A nine-day sampling campaign was carried out in March 2015, for the purpose of wastewater characterization and calibration. Samples were examined to measure the parameters shown in Table 1. The measured parameters are: filtered chemical oxygen demand (COD), with a 0.45 mm filter (COD_{ filt} ); total COD; total suspended solids; volatile suspended solids (VSS); total Kjeldahl nitrogen (TKN); ammonium (NH_{4}-N); FA (NH_{3}-N); nitrite (NO_{2}-N); nitrate (NO_{3}-N); orthophosphates (PO_{4}-P); alkalinity, and temperature (T). All the measurements were based on Standard Methods (APHA, AWWA & WEF 2005). Additionally, in order to describe the biological conversion as a function of the reactor length, pH and dissolved oxygen (DO) were measured over the length of the oxidation ditches at different places. The average sampling results are presented in Table 1. The characteristics of the mixed influent wastewater

![Figure 1](https://iwaponline.com/jwrd/article-pdf/8/3/393/240762/jwrd0080393.pdf)
for industrial and domestic wastewater are also estimated in Table 1. The sampling program was conducted at a sewage temperature of approximately 22°C.

BioWin© software V5 (EnviroSim, Hamilton, Canada) was used to perform the simulation of the industrial WWTP. A schematic of the biological reactor's representation in BioWin is shown in Figure 2. The industrial wastewater and wastewater from the administration buildings were represented separately. In oxidation ditches, the wastewater is streamed nonstop in a looped ditch, which resulted in a high (15-fold) dilution level (dilution level = flow in the ditches divided by the influent flow). This strong dilution level helped to decrease the inhibition of the effect of bacterial species that could result from exposure to the toxic pollutants that exist in such industrial wastewater (Elawwad et al. 2016).

The oxidation ditches were represented in BioWin as a series of 11 completely mixed bioreactors to consider its plug-flow characteristics as BioWin does not have a built-in plug flow reactor (Figure 2). In the calculation of SRT, the suspended solids escaped in the effluent were considered. The DO concentrations in the model for each reactor were controlled by modifying the power supply for each brush aerator. The DO concentrations for each reactor were subsequently compared to the real DO measurements.

Using the guidelines proposed by Hulsbeek et al. (2002), industrial wastewater was characterized. The calculated COD and nitrogen fractions are presented in Table 2. For domestic wastewater characterization, the default values of BioWin software were applied. COD mass balance was applied in order to validate the data inputted to the software. All the data were entered into the BioWin simulator and the model was calibrated and validated. Two important steps involved in process simulation are the calibration and validation steps (Hulsbeek et al. 2002). The calibration process is a crucial step in wastewater modeling, and is usually done using steady state simulation mode. Here, the stoichiometric and kinetic parameters were adjusted to enable model output close to the measured values. In steady state simulation mode, the model does not consider the effects of time as it assumes that the plant has reached steady operating conditions. The second step is model validation and is usually done in dynamic simulation mode where the model output is verified over a period of time. In dynamic simulation mode, the model response is verified over a period of time data from continuous flow systems under time-varying loading.

Different statistical tests were used in the literature to estimate the accuracy of a model. These tests include mean relative error (MRE), mean absolute error, mean error or linear repression coefficient, root mean squared error, root mean squared scaled error (Makinia 2010). In this study MRE was chosen to estimate the precision of the validation process and the model prediction as suggested by Makinia et al. (2006) and Liwarska-Bizukojc et al. (2013). MRE values could give an idea about the goodness-of-fit of the ASM and was estimated using Equation (1), where \( m_i \) is the measured value for a certain parameter, \( p_i \) is the predicted value from BioWin software.

![Figure 2](https://iwaponline.com/jwrd/article-pdf/8/3/393/240762/jwrd0080393.pdf)
for the parameter, and \( N \) is the number of observations.

\[
MRE = \frac{1}{N} \sum_{i=1}^{N} \left| \frac{mi - p0}{mi} \right| \times 100\% \tag{1}
\]

RESULTS AND DISCUSSION

WWTP current performance

While evaluating the present condition of the Helwan WWTP (Table 1), no problem was detected in the removal of the organic matter. However, there were problems regarding the nitrogen treatment, including nitrification and denitrification. Depending on the current sludge withdrawal rates, the current SRT is 45 days. Despite this high SRT, the total ammonia removal was only approximately 10–20% of TKN. Moreover, the quality of the effluent parameters does not meet the Egyptian standards for industrial wastewater.

While comparing the characterization of the influent wastewater with the typical characterization of domestic wastewater values, the author detected significant differences in the values. The industrial wastewater used in the study was almost soluble, with extremely low particulate values. The presence of phenol, at about 50% of the total COD, was the main characterization difference from the domestic wastewater. Phenol is considered a readily biodegradable organic compound (Lee et al. 2005). The calculated COD/N ratio was 1.43 for the industrial wastewater in this study, which is not sufficient for achieving full biological denitrification.

The oxygen levels were always above 1.5 mg L\(^{-1}\) in the aerobic compartments of the oxidation ditches; therefore, aeration was not the main limiting parameter in the nitrification process. However, from this first evaluation, the aeration capacity would probably have to be increased with the increase in nitrification rates that is expected from this optimization study.

The nitrification of high-ammonium wastewater is a critical process and is sensitive to various parameters including temperature, DO, pH, FA, free nitrous acid, and the presence of aromatic compounds. The existence of toxic constituents in such types of industrial wastewater could hinder the nitrification process (Lee & Park 1998; Kim et al. 2007). This problem is often encountered in WWTPs that treat wastewater containing high ammonium concentrations. In such WWTPs, the nitrogen-loading rate is usually significantly more than the maximum nitrification rate of the reactor; therefore the ammonia accumulates in the system. As a result, high levels of FA in instances of elevated pH concentrations, or free nitrous acid in instances of low pH concentrations, would accumulate. Complete inhibition of the nitrification process can happen at FA concentrations more than 150 g N m\(^{-3}\), whereas inhibition of NOB can occur at FA concentrations more than 2.8 g N m\(^{-3}\) (Jubany et al. 2008). Adaptation of the nitrification process to elevated ammonium and FA concentrations can occur after extended periods of operation under these conditions. The time required for adaptation and establishing a steady state could be up to one year (Morling et al. 2012). In such

| Name     | Description                                      | Default for settled wastewater | Value calculated | Unit            |
|----------|--------------------------------------------------|--------------------------------|------------------|-----------------|
| Fbs      | Readily biodegradable (including acetate)        | 0.27                           | 0.81             | gCOD/g total COD |
| Fxp      | Noncolloidal slowly biodegradable                | 0.5                            | 0.5              | gCOD/g slowly degradable COD |
| Fus      | Unbiodegradable soluble                          | 0.08                           | 0.05             | gCOD/g total COD |
| Fup      | Unbiodegradable particulate                      | 0.08                           | 0.03             | gCOD/g total COD |
| Fna      | Ammonia                                          | 0.75                           | 0.78             | gNH\(_3\)N/gTKN |
| Fnox     | Particulate organic nitrogen                     | 0.25                           | 0.25             | gN/g Organic N  |
| Fnus     | Soluble unbiodegradable TKN                      | 0.02                           | 0.02             | gN/gTKN         |
| FupN     | N:COD ratio for unbiodegradable part. COD        | 0.035                          | 0.035            | gN/gCOD         |
| Fpo4     | Phosphate                                        | 0.75                           | 0.86             | gPO\(_4\)-P/gTP  |

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cases, ammonia stripping pretreatment could be useful in reducing FA concentrations that inhibit nitrification (Kim et al. 2008).

The system of oxidation ditches functions well with high ammonium wastewater and wastewater that contains toxic substances. The dilution of the influent wastewater by the recycled stream of water reduces the inhibition effect, and flexible aeration could help the nitrification-denitrification process.

Model calibration and validation

At the start of the experiment, it was difficult for the model to reach equilibrium for such high-ammonium wastewater. This is because the BioWin Model takes into account the inhibition of the nitrification process by high FA concentrations. Because of the sensitivity of the nitrification process, it is recommended that initially in the operation the increase in the nitrogen-loading rate should be gradual and controlled to treat such wastewater. In this way, the nitrification rate is allowed to remain close to the maximum nitrification rate to facilitate low ammonium concentrations in the reactor (Jubany et al. 2008). The same strategy was used in calibrating the model, with a stepwise application of ammonia loading, and various runs were performed until equilibrium was reached. The same strategy should be employed during actual optimization to avoid the accumulation of high ammonia concentrations in the reactors.

The average annual concentrations of sludge, SRT, sludge production, and N-content sludge were used for the initial calibration of the VSS and COD, as proposed by Hulsbeek et al. (2002). Subsequently, nitrification and denitrification were calibrated. The effluent COD was calibrated by adjusting the parameter of non-biodegradable soluble substrate \( F_{\text{str}} \) in the BioWin software. The final step was calibrating the model stoichiometric and kinetic parameters, for which only the effective parameters defined by the sensitivity analysis in a previous study (Elawwad et al. 2016) were adjusted. For the COD calibration, the maximum specific growth rate of the heterotrophic bacteria \( \mu_{\text{max}} \) in aerobic conditions was calibrated at 2.6 instead of its default value of 3.2 d\(^{-1}\). The aerobic decay rate of the heterotrophic bacteria \( b_{\text{b}} \) was calibrated at 0.7 instead of its default value of 0.62 d\(^{-1}\). Furthermore, the decay rate of the AOB \( b_{\text{aerob,A}} \) was calibrated at 0.23 instead of its default value of 0.17 d\(^{-1}\) and the maximum specific growth rate of the AOB \( \mu_{\text{max}} \) was calibrated at 0.5 instead of its default value of 0.9 d\(^{-1}\), which meant that the nitrification process was calibrated by reducing \( \mu_{\text{max}} \) to almost half its default value. This is in agreement with the range reported by a previous study relevant to the treatment of wastewater from oil refineries (Pinzón Pardo et al. 2007). The other stoichiometric and kinetic parameters related to the nitrification process were kept unchanged. After calibration, the BioWin model was capable of predicting the low performance of the nitrification process. Unlike the rest of the ASM modeling family, parameters such as pH and temperature are included in the BioWin model which suits the present study. The model predicted values were extremely close to the measured values.

Dynamic model validation was performed, based on the routinely measured data over 105 days between August and October 2014. The predicted values for COD and TKN were compared with the measured values. This information is shown in Figure 3. A quality check was done of the validation process by calculating the MRE values. For COD and TKN, the MRE values were 10 and 7.6\%, respectively. These results show the quality of the calibration and validation processes. The deviation between the measured and predicted values of the output variables by BioWin did not exceed 20\%, as shown in Figure 3, which means that the calibration of the BioWin model was performed in a good way (Makinia et al. 2006; Liwarska-Bizukojc et al. 2013).

Optimization of the current situation

The current status of the WWTP is that a high ammonia concentration of about 515 mg/L is present in the effluent. Nitrite is accumulated at 23 mg/L as NOB are probably partially inhibited by the high ammonia concentration. No nitrate is present in the effluent, which means that all nitrate was denitrified. The residual alkalinity and pH in the effluent were 80 mg/L as CaCO\(_3\) and 7.13, respectively. The optimization strategy for nitrogen removal was to study the alternative operational parameters that exert the most influence on nitrogen removal. The optimization was done in several steps and the effluent quality after each step is included in Table 3.
The first step in the optimization process was to modify the plant configuration so that simultaneous nitrification and denitrification can occur. There are two options for the denitrification process in wastewater treatment, which are either pre-denitrification or post-denitrification. Post-denitrification can be performed by creating an anoxic zone at the end of the treatment process and adding an external organic carbon source (ethanol, methanol, and acetic acid for example). The costs and availability of the external carbon source are the main constraints for this technique. The other alternative is using pre-denitrification, which requires creating an anoxic zone at the beginning of the treatment process, a suitable COD/N ratio, and recirculation for nitrate. Oxidation ditches sustain the required recirculation for the pre-denitrification process. The current aerator configuration in the WWTP is not suitable for pre-denitrification as the pre-anoxic zone is small. In order to increase the pre-denitrification rate, the volume of the anoxic zone at the beginning of the oxidation ditches had to be increased. This could be done by moving the brush aerator from reactor T2 to reactor T9 (compare Figures 2 and 4). By performing this simple modification in the aerator configuration, the pre-denitrification zone was increased from 5 to 40% of the total tank volume. As shown in Table 3, the nitrification rate was increased slightly as the...
ammonia content in the effluent decreased to 508 mg/L. Consequently, residual alkalinity and pH decreased to 29.5 mg/L and 6.6, respectively. The pH is below its optimum value for nitrification, which is between 7.5 and 8.0. A minimum residual alkalinity of 50–100 mg/L is required to ensure adequate buffering. The next optimization step should be optimizing the buffering capacity.

Taking into consideration the organic and inorganic constituents in the coke-oven wastewater in this study, an inadequate amount of inorganic carbon is usually the first factor that leads to the instability of the nitrification process (Anthonisen et al. 1976). Alkalinity in water determines the availability of inorganic carbon needed for the metabolism of nitrifying bacteria. The required alkalinity for the removal of 1 mg of ammonia is approximately 7.14 mg as CaCO₃. Mixed influent wastewater has a TKN value of about 823 mg/L (Table 1), which stoichiometrically needs about 5,877 mg/L of CaCO₃ for complete nitrification. This alkalinity is much more than the 560 mg/L that the mixed influent wastewater contains (Table 1). Lack of alkalinity could cause inhibition of the nitrification process (Carrera et al. 2004). Increasing the alkalinity by adding bicarbonates or carbonates as external sources of alkalinity could be needed for the stability of the nitrification process. It should be noted that the added alkalinity will increase the operating cost dramatically. Denitrification will reduce the required alkalinity as 3.57 mg/L of alkali is produced per 1 mg/L of NO₃⁻N nitrified. Thus, performing simultaneous nitrification and denitrification will reduce the alkalinity needed for complete nitrification by half.

In order to optimize the buffering capacity needed for the nitrification process, the alkalinity was increased stepwise by adding lime to the influent raw wastewater. The alkalinity was increased as shown in Table 3 from an existing value of 562 mg/L as CaCO₃ to 2,440 mg/L as CaCO₃. At that level of added lime, the influent pH increased to 10.8, as shown in Table 3. Lime addition after that point can increase the operation cost dramatically. Moreover, it could be difficult to dissolve carbonates and bicarbonates in the wastewater completely at these target concentrations. With such a high pH value in the influent, ammonia stripping could be an economical option. This will be discussed next. During the increase in alkalinity, the software notifies the user that a limitation in the phosphorus concentrations required for the biological process has occurred. To fix this, the phosphorus concentration in the mixed influent wastewater was increased from its current value of 6.23 to 8 mg/L. Phosphorus is an important factor in the growth of the bacteria. Therefore, phosphorus has to be added at the same rate as the alkalinity is increased, else the bacterial growth would be constrained.

Table 3 indicates the effect of the stepwise increase in alkalinity on the effluent characterization. It was found that the nitrification process improved with the increase in alkalinity as the ammonia concentration in the effluent decreased to 179 mg/L at an alkalinity of 2,440 mg/L as CaCO₃ in mixed influent wastewater. Consequently, residual alkalinity and pH decreased to 164 mg/L and 8.2, respectively. NOB are still partially inhibited by the high ammonia concentration as nitrite is accumulated at 77 mg/L. No nitrate is present in the effluent, which means that all nitrate was denitrified.

At that optimization level, the pH in the influent became 10.8. Most of the ammonia is converted to FA between pH...
10.8 and 11.5, which is advantageous to the stripping process. Ammonia stripping will remove a large portion of the FA existing in the influent and reduce its inhibitory effect on the nitrification process. The ammonia stripping process is extremely effective for treating high-strength ammonium (Maranon et al. 2008). Adding an ammonia stripping unit would be feasible and less costly, taking into account that the influent already has a high free-ammonia content. In addition, ammonia stripping would facilitate alteration of the COD/N ratio, which is one of the most effective parameters in nitrogen treatment, relevant to nitrification and denitrification for high-strength ammonium wastewater. For effective nitrification, the COD/N ratio should be kept low to minimize the competition between autotrophic and heterotrophic bacteria. However, the influence of this ratio might be lower in BNR systems than conventional activated sludge systems (Carrera et al. 2003, 2004). As in BNR systems, most of the COD is removed in the pre-denitrification zone, which minimizes the competition between autotrophic and heterotrophic bacteria in the aerobic zone. As regards pre-denitrification, a higher COD/N ratio is preferred and stoichiometrically should be more than 4.2 in order to realize the full BNR (Carrera et al. 2004). Such a COD/N ratio is not common in this type of wastewater. In the WWTP used in the current study, the actual COD/N ratio was 1.43.

Benefiting from the high pH in the influent wastewater, a stripping unit was added to the industrial wastewater stream (Figure 4). There is no built-in ammonia stripper in the BioWin software. However, the ammonia stripping process is represented in the BioWin model. Therefore, the ammonia stripper was represented as an aerated reactor with a high airflow rate. The efficiency of the ammonia stripping unit was controlled by altering the power supply rate of the aerated reactor. The efficiency of ammonia stripping decreases with air temperature (EPA 2000). Efficiencies between 80 and 95% are expected for wastewater temperatures in this study (between 16 and 32 C). The effect of different efficiencies of the ammonia stripper (between 80 and 90%) on the effluent quality were tested, as shown in Table 3. Ammonia stripping would facilitate reducing ammonia contents, which enables the system to remove the rest of the ammonia and organic nitrogen biologically. As shown in Table 3, the nitrification process improved significantly as a result of using the ammonia stripping unit. Ammonia stripping at only 80% efficiency was efficient in reducing ammonia concentration in the effluent to below 3 mg/L. No nitrite was accumulated when using ammonia stripping, which means no inhibition of NOB occurred. The pH in the reactors was about 7.6, which is optimal for the nitrification process. Low nitrate was found in the effluent, which decreases as the efficiency of ammonia stripping increases. This is to be expected as COD/N was optimal for denitrification and varied between 4 and 5.1 for ammonia stripping efficiencies between 80 and 90%. Phosphorus was reduced to its initial value of 6.23 mg/L with no problem as the nitrification rate was reduced by using ammonia stripping.

The disadvantage of the ammonia stripping process is that the ammonia pollutant was transferred from the water to the air (EPA 2000); consequently, it is recommended that the stripped ammonia be recovered by employing a closed loop. In a closed loop, the air is sent to an absorber where concentrated ammonium sulfate is formed. Then, the clean air is recycled back to the ammonia stripper without emitting ammonia into the air.

As a final step in the optimization study, the effect of different SRT values on WWTP performance was tested. The SRT could be a crucial optimization variable, as the nitrifying bacteria are extremely sensitive and require a considerable time for growth (Kim et al. 2011). To study the effect of the SRT, a stepwise modification of SRT between 30 and 60 days was performed, the output of which is presented in Table 4. It is recommended that a high SRT duration of 40–50 days for this type of wastewater treatment is implemented (Morling et al. 2012). Moreover, high SRT is required for growing bacteria species responsible for removing some of the other compounds in this type of industrial wastewater, such as cyanide and phenol. By increasing the SRT, the ammonia removal efficiency improved; however, the COD concentration in the effluent increased. At high SRT, the clarifier would become overloaded and its efficiency would decline; therefore, the COD increased because of the deficient clarification process. The optimum SRT could be 40 days, which is close to the actual current conditions, and this value is similar to the result from a previous study by Morling et al. (2012).

After all these optimization steps, the plant effluent was found to be acceptable according to the specifications of...
Egypt for industrial wastewater discharged into municipal wastewater networks. The final configuration of the optimized industrial WWTP is shown in Figure 4. The performance of the plant was optimized for nitrogen with all the previously mentioned alterations. The optimization was done by applying the following modifications:

- increasing the pre-anoxia by moving the brush aerator from reactor T2 to reactor T9 to increase the pre-denitrification capacity;
- increasing the alkalinity to 2,440 mg/L as CaCO₃ from its current value of 562 mg/L by adding lime to the industrial wastewater;
- using an ammonia stripping unit with a minimum efficiency of 80% for the industrial wastewater, benefiting from the high pH due the added alkalinity;
- SRT to be kept high (between 40 and 50 days) for such types of industrial wastewater.

CONCLUSIONS

In this study, the BioWin model was calibrated and validated, with a good representation of an industrial WWTP that receives high-strength ammonium wastewater. The model was calibrated in steady state simulation mode and validated in dynamic flow simulation mode. The calibration was performed by adjusting the most sensitive stoichiometric and kinetic parameters. The BioWin model proved to be capable of describing the processes performed at the WWTP with regard to the COD and ammonium. The model was helpful in optimizing the plant for almost full nitrogen removal and solving the problems related to the instability of nitrification and denitrification processes in such types of wastewater. This work should support the application of wastewater modeling for operational decision making and optimizing the treatment of industrial wastewater that contains high ammonia concentrations. As a recommendation for future studies, the ASM model should be extended for predicting the inhibitory effect on the nitrification process of the other compounds in this type of industrial wastewater, such as cyanide and phenol.

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