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Research Article

Troubleshooting a Full-scale Wastewater Treatment Plant for Biological Nutrient Removal

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Abstract: The International Association of Water Quality (IAWQ) Activated Sludge Model No.2 (ASM2) was applied to troubleshoot an existing underperforming full-scale wastewater treatment plant (WWTP) built for biological nutrient removal. The WWTP is operated in a 3-stage pho-redox process configuration (A^2O). This study was undertaken with the aim of finding optimal operating conditions that will meet TP and TN concentration requirements in the effluent of the WWTP under study without the use of either chemical or external carbon sources and also to verify the applicability, capability and predictability of ASM2 as implemented in STOAT software. ASM2 was successfully used to troubleshoot bottle neck areas and to define the operational schedule for optimal performance of the wastewater treatment plant. Consequently, the costs of chemical and external carbon sources were eliminated and the effect of residual chemicals on the environment reduced.

Keywords: Biological nutrient removal, IAWQ_ASM2, optimal performance, troubleshoot, wastewater treatment

INTRODUCTION

The idea of Biological Nutrient Removal (BNR) from Wastewater Treatment Plants (WWTPs) was initiated in the early 1960s, spearheaded by Ludzack and Ettinger 1961, Wuhrman 1964, (EPA, 2010; Barnard, 1975). This initiative resulted in the development of mathematical models for BNR wastewater treatment systems. Early 1970s witnessed the major process development breakthroughs for biological removal of both nitrogen and phosphorus in South Africa through the work of Barnard (1973). The practice of steady state modeling with activated sludge model began in the early 1970s (Smith and Dudley, 1998). The late 1970s witnessed the development of dynamic modeling, which was aimed at evaluating the effect of long-term variations in flow and composition of wastewater (Smith and Dudley, 1998). The need to control Nutrient enrichment of surface water (eutrophication) in the receiving water bodies, has led to the promulgation of strict rules and regulations guiding the effluent discharge limit by different governing bodies, such as the United States Environmental Protection Agency (USEPA) and the European Union (EU), with the publication of the Urban Wastewater Treatment Directive (1991). Achieving these stringent regulations requires tools to assess the nutrient removal capability of wastewater treatment processes. Consequently, a group of engineers and scientists under the auspices of the IAWQ developed ASM1 (Henze *et al*., 1987), which focused on the biological removal of carbon and nitrogen in activated sludge systems. A perceived limitation in phosphorus removal prompted a further improvement of ASM1 with the inclusion of Phosphorus Accumulating Organisms (PAO). The improved model, now known as ASM2 (Henze *et al*., 1995), enhances our understanding of the behavior of biological nutrient removal in activated sludge systems. However, concern about denitrifying PAO was quickly raised within a short period into the released of ASM2 from different researchers based on their experimental observations of the role of PAO in denitrification (Mino *et al*., 1995; Meinholt *et al*., 1999). These concerns subsequently lead to a further improvement on ASM2, which is referred to as ASM2d. Incorporated into ASM2d is PAO has the ability to denitrify and reproduce under anoxic condition, this was accomplished by adding two processes i.e., anoxic growth of poly-phosphate (X_{pp}) and anoxic growth of PAO. Other components of ASM2d are basically the same as ASM2. However, since the publication of ASM2d, its predecessor, ASM2 is rarely used.

The Activated Sludge Models (ASM1, ASM2-ASM2d, ASM3) by the IAWQ task group on Mathematical Modeling for Design and Operation of Biological Wastewater Treatment are the dominant mathematical models used for modeling biological compartments of wastewater treatment plants. These models have been successfully applied to full-scale

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treatment plants and have proved to be a good compromise between the complexity of the activated sludge processes and prediction of the plant behavior under dynamic conditions (Vanrolleghem *et al*., 2003).

The introduction of the Activated Sludge Model No.1 (ASM1) (Henze *et al*., 1987) popularized dynamic modeling of WWTPs, targeting the optimization of treatment plant design and operations, in order to improve the effluent quality at a minimal operating cost. Many researchers have conducted detailed studies on biological nutrient removal (Wentzel *et al*., 1985; Smolders *et al*., 1995; Van Loosdrecht *et al*., 1997), accomplished with different reactor process configurations (AO, A^2/O, UCT, MUCT, MLE, VIP), which metamorphosed from the ideas of Ludzack and Ettinger and Wuhrman (Barnard, 1973). Better knowledge of the complex microbiological populations participating in the combination of processes and biochemical routes is gained through operational optimizations of the WWTPs via laboratory and pilot-scale experiments and mathematical simulations of treatment plant components (Mino *et al*., 1998; Barnard, 1983; Ekama and Wentzel, 1999; Henze *et al*., 1999).

Model calibration is required before it can be used for dynamic modeling exercises; thus, guidelines for the calibration of activated sludge treatment plant models are provided by many researchers based on their experiences (Melcer *et al*., 2003; Vanrolleghem *et al*., 2003; Muschalla *et al*., 2008). However, the approach adopted in calibrating the model depends on the modelling objectives. This study aimed at determine the optimal operating conditions such that the effluent of the wastewater treatment plants under study will meet the TP and TKN concentration requirements without the use of either chemical or external carbon sources and consequently reduces the effect of residual chemicals on the aquatic organism and the environment. Further, to verify the applicability, capability and predictability of ASM2 as implemented in STOAT software. Extensive review of general wastewater treatment plant modeling and simulation can be found in Krist *et al*. (2004) and Peterson *et al*. (2002) for detailed methodology. This study will present the readers with the results of the case study.

MATERIALS AND METHODOLOGY

Description of the WWTP: The WWTP is located in Changzhou, Jiangsu province People Republic of China (PRC). It consists of two parallel plug-flow bioreactors and two secondary sedimentation tanks (clarifiers) operated in a 3-stage pho-redox process configuration (Anaerobic, Anoxic and Oxic). The volumes of the anaerobic, anoxic and aerobic tanks are; 1143 m^3, 3440.8 m^3 and 9179 m^3, respectively for the first lane and 1465.8 m^3, 4408.53 m^3 and 11759.7 m^3, respectively, for the second lane. The flow streams are combined before entering the two secondary sedimentation tanks. The surface area of the clarifier is 1252 m^2, which comes from a 40-m diameter, excluding the area occupied by the centre well and a 3-m depth. There are two screens and two vortex grit chambers for pre-treatment. The influent wastewater stream from the reservoir to the reactor is divided into two with each lane receiving one stream. The plant is designed to treat 50,000 m^3/d of mainly domestic wastewater. Influent wastewater is collected in a reservoir and pumped through the screens and grit chambers before it is finally distributed to the reactors. The WWTP was designed to treat the following influent concentrations: BOD = 180, COD = 400, NH3-N = 35, SS = 250, TP = 4 and TN = 45, all in mg/L. It is expected to produce the following effluent concentrations: BOD≤10, COD≤50, NH3-N≤5, TN≤15, TP≤0.5 and SS≤10, all in mg/L. Although the observed effluent TP and TN concentrations of the Changzhou WWTP meets the target discharge effluent concentrations, however, acetate is added to the anaerobic reactor to facilitate Phosphorus Accumulating Organisms (PAO) activity and Al_2 $(SO4)_3$ is used for chemical polishing for the removal of TP. The addition of acetate and Al2 (SO4) 3 is an extra cost that needs to be optimized and possibly eliminated.

Evaluation of the Changzhou WWTP historical data: A site visit to the wastewater treatment plant was conducted in the accompaniment of operations and management staff of the plant to familiarize with the design of the WWTP layout, to identify the locations of significant sampling and monitoring stations, to obtain input from plant operations staff regarding equipment hydraulics and process limitations in the plant based on their operating experience and to evaluate the plant performance based on the available historical data. Further, to design personal data campaign programs for the purpose of proper influent wastewater characterization.

Figure 1 showed the average historical influent COD, BOD, COD: TP and BOD: TP ratios for the WWTP. This is to verify the assumption that the phosphorus removal treatment plant can either be P deficient or carbon deficient (Randall *et al*., 1992). According to Sedlak (1991), a carbon deficient treatment plant cannot achieve a complete phosphorus removal, whereas, the phosphorus deficient systems can achieve a near-complete phosphorus removal. Randall *et al*. (1992) and Grady *et al*. (1998) reported that 40 mg/L and 20 mg/L are the requirement for COD/TP and BOD/TP ratios, respectively for phosphorus deficient systems. It is obvious from Fig. 1, in recent times, from late 2009 to the time this study was conducted that this treatment plant fulfilled these conditions.

Analytical methods: In order to characterize the composition of the influent wastewater, two weeks

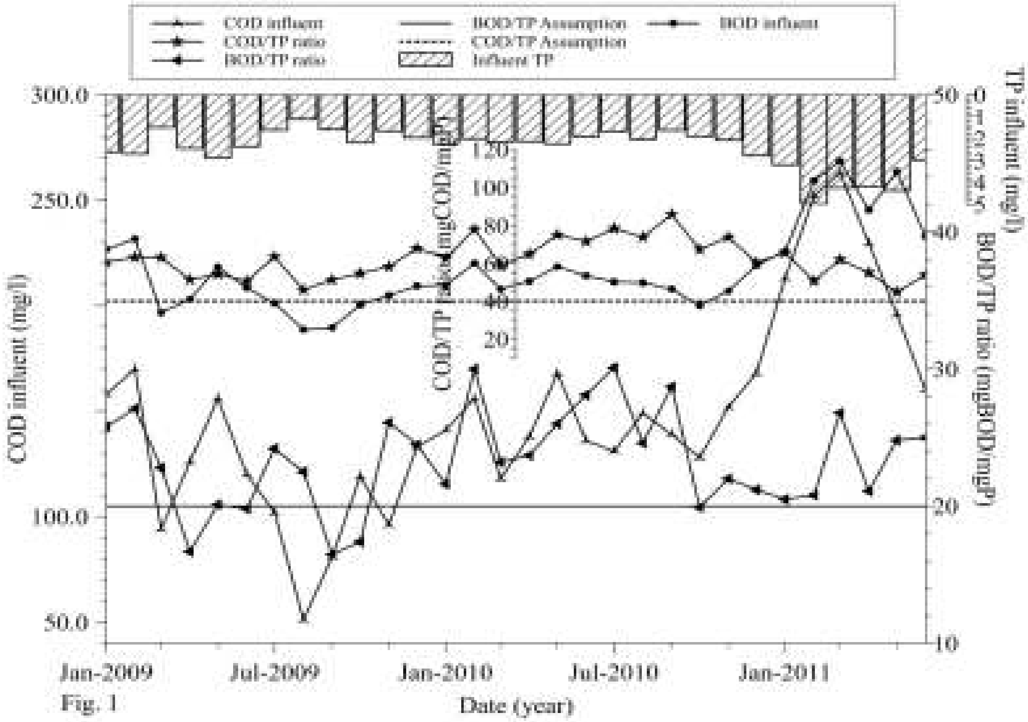


Fig. 1: Average influent BOD and COD, BOD/TP and COD/TP ratios for 2 and half years

Table 1: Operating parameters and influent wastewater characteristics for Changzhou WWTP, January, 2011

Parameter	Unit	Mean	S.D.	Min.	Max.
Flow rate	m^3/d	26600	2464	21425	31676
SRT	d	17	0.4	16	18.5
RAS flow rate	m^3/d	18027	1543	14998	19907
Temperature	°C	13	1.12	11.2	15.1
pH	--	7.5	-	7.5	7.5
CODtot	mg/L	213	42.8	117	266
BOD_5	mg/L	66.4	18.62	43.4	82.7
TN	mg/L	29.7	5.82	17.2	37.1
TP	mg/L	3.35	0.78	2	5.46
N-NH3	mg/L	26	5.74	13.7	33.9

Table 2: Operating parameters and influent wastewater characteristics for Changzhou WWTP, April, 2011

Parameter	Unit	Mean	S.D.	Min.	Max.
Flow rate	m^3/d	33985	2376	28500	39269
SRT	d	18	0.6	16	19
RAS flow rate	m^3/d	23312	1366	19950	25629
Temperature	°C	16.6	1.18	14.7	18.7
pH	--	7.5	-	-	7.5
VSS	mg/L	152	22	126	183
BOD_5	mg/L	88.2	12.9	56	98
CODtot	mg/L	229.6	26	151	246
CODi-sol	mg/L	75	9.2	48	91
VFA	mg/L	22	1.7	16	29
TP	mg/L	4.18	1.12	2.32	8.68
SPO_4	mg/L	2.25	0.53	1.64	6.8
TN	mg/L	31.48	2.9	15.2	35.2
TNi-sol	mg/L	29.1	3.2	13.4	31.5
N-NH3	mg/L	27.5	4.16	18.2	35.8
$N\text{-}NO_3$	mg/L	0.22	0.1	0.12	0.26
$N\text{-}NO_2$	mg/L	0.11	0.12	0.04	0.42

intensive sampling was performed in April, 2011. The influent characterization of the month of April was determined according to the STOWA method (1996) and Meijer *et al*. (2001). A 2-hourly composite data for 24 h daily and some days 1-hourly grab samples were used to characterize the influent. The samples were analyzed based on Chemical Oxygen Demand (COD) of non filtered (total COD) and filtered through 0.45-µm and 1.2-µm diameter pores (COD influent soluble, filtered, flocculated), Biochemical Oxygen Demand (BOD) of non filtered and filtered, Volatile Fatty Acids (VFA), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), Total Kjeldahl Nitrogen (TKN), Total Phosphorus (TP), ortho-phosphate ($PO^3{}_4$), Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS) concentrations. The experiment was conducted in the Changzhou WWTP laboratory in accordance with the standard methods (Standard Methods 5220, 5210, 4500-NH3, 4500-Nitrogen, 4500-P, 2540-D and 2540-E, respectively) (APHA, American Water Works Association and Water Environment Federation, 1995). The influent wastewater characteristics of Changzhou WWTP for the month of January, 2011 and April, 2011, are presented in Table 1 and 2, respectively.

Initial estimates of COD fractions were made from the CODt (total COD) to set-up the model from the existing historical data before the detailed measurement campaign.

MODEL CONSTRUCTION

The decision on how the WWTP should be represented in the simulator are crucial to the success of any simulation or modeling exercise (Hulsbeek *et al*.,

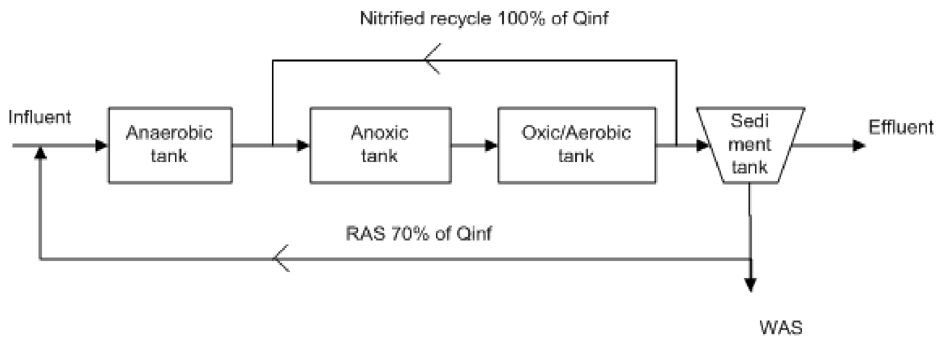


Fig. 2a: Changzhou WWTP simplified schematic flow diagram

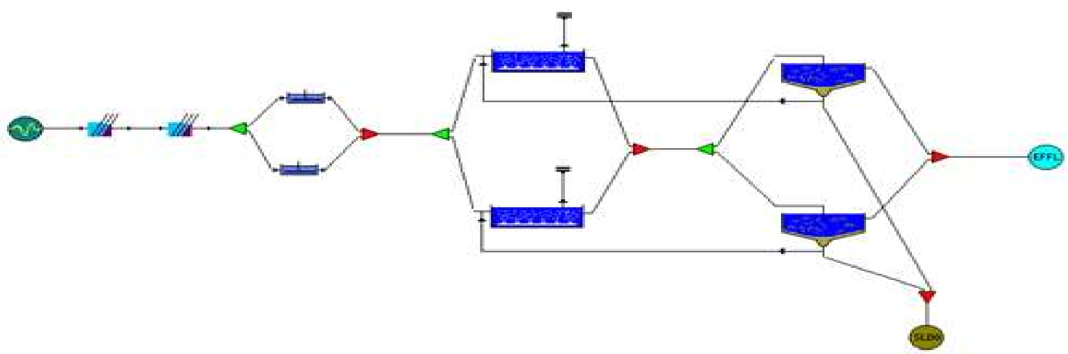


Fig. 2b: STOAT configuration of Changzhou WWTP

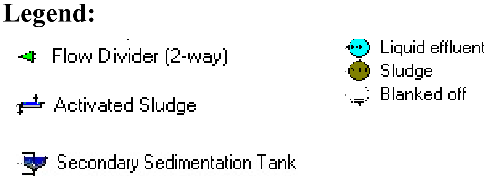
2002) attributed considerable deviation from default model parameters values to errors due to these decisions. Decisions, including should each reactor be modeled as a separate entity or should the entire reactor be represented in the model and simulated at the same time and, for an activated plant, how the mixing characteristics of the aeration tank should be depicted in the model. In this study, because the influent discharge is divided equally between the two lanes and the two Clarifiers are of the same size, one lane was modeled.

Model calibration and validation: The simulation was carried out by using STOAT software version 4.3 and its implementation of ASM2. STOAT implementation of ASM2 has two main changes:

- The production of nitrogen gas through denitrification is not explicitly modeled
- In STOAT, ASM2 uses alkalinity as part of the switching functions. It is assumed that alkalinity will never be limiting. It should be recalled that in ASM1, alkalinity conservation was modeled, but it was never used. Default values of Kinetic and stoichiometric parameters for ASM2 as implemented in the STOAT can be found in (Gujer *et al*., 1995).

A simplified schematic flow diagram of the Changzhou WWTP biological reactors is shown in Fig. 2a and the STOAT configuration of the Changzhou WWTP is shown in Fig. 2b.

Legend:



Data from January, 2011 as shown in Table 1 were used for model set-up and calibration, while the data collected during the April, 2011 data collection campaign as shown in Table 2 were used to validate the model against experimental results.

It is an expensive and time-consuming process to determine all of the model components (Vanrolleghem *et al*., 2003), thus, default values reported in previous applications should be assigned to the model parameters (Henze *et al*., 2000). The model was first run with the default values for steady state and the results were compared with the measured data to evaluate the performance of the existing plant (Gujer *et al*., 1995). The modeler adjusted one parameter at a time to observe the differences between the measured data and the model prediction. The procedure was repeated until a reasonable match between the measured data and the model prediction was attained. This procedure also helps the modeler to determine

Table 3: Calibrated parameter

Parameter	Value used	Model default
Heterotroph yield (mg COD/mg COD)	0.67	0.63

which parameter is sensitive to change because of the interactive nature of the parameters. Changing two or three parameters simultaneously will make it difficult to assess their impacts. The calibrated model parameter is presented in Table 3 alongside the model default value.

RESULTS

One stoichiometric parameter was adjusted to calibrate the model for Changzhou WWTP. Having run the model with the default values, an examination of the results revealed differences between the simulated and the observed MLSS in the aeration tank, thus necessitating the calibration of the heterotrophic yield (Y_H) which is among the parameters responsible for long-term behavior (Liwarska-Bizukojc and Biernacki, 2010), because the system SRT is known with certainty. It should be noted that, in ASM2, there is no difference between the yield aerobic and anoxic. The value of $Y_{H,}$ which was increased from 0.63 to 0.67, corresponds with values found in the literature, which ranges from 0.5 to 0.74 (Henze *et al*., 1986; Kappeler and Gujer, 1992; Barker and Dold, 1995; Strotmann *et al*., 1999; Henze *et al*., 2000; Liwarska-Bizukojc and Biernacki, 2010), showing that the value used in this study is within the acceptable value. The adjustment of one parameter for this calibration exercise is in agreement with previous findings (Henze *et al*., 1995; Van Veldhuizen *et al*., 1999; Petersen *et al*., 2002), indicating that the treatment plant is well configured in the simulation package used. Observed and simulated graphs for the model calibration are presented in Fig. 3a to d.

A model is said to be valid if it's predictions matches measured values from an independent dataset within the permissible deviation from the calibrated values (U.S. EPA, 1993). Therefore, the data obtained from the April data campaign was used to test the validity of the model. The validated results are shown in Fig. 4a to d.

Figure 3a to d and 4a to d show the comparisons between observed and simulated graphs for model calibration and validation, respectively. It is evident from these figures, observed and simulated trends in the treatment plant were reproduced in the model. However, the simulated NH3-N validation result deviated far from the observed value as shown in Table 4. This could be attributed to elevated amount of carbon source added to aid denitrification which is not accounted for in the model. Additionally, it exemplifies the difficulties in aeration control to maintain required cost effective DO in the aeration tank (Cinar *et al*., 1998). The effluent limits, as shown on the graphs, are the allowable discharge limits in effluent concentrations of different constituents as prescribed by the-Chinese government. Average observed and simulated values are presented in (Table 4) for further comparison.

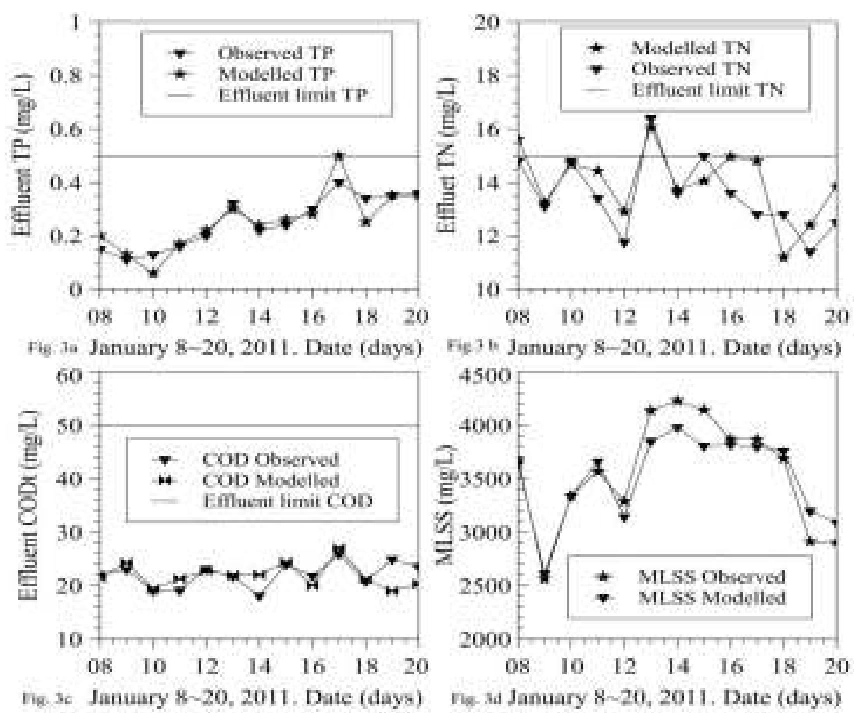


Fig. 3: a-d observed and simulated graphs for model calibration

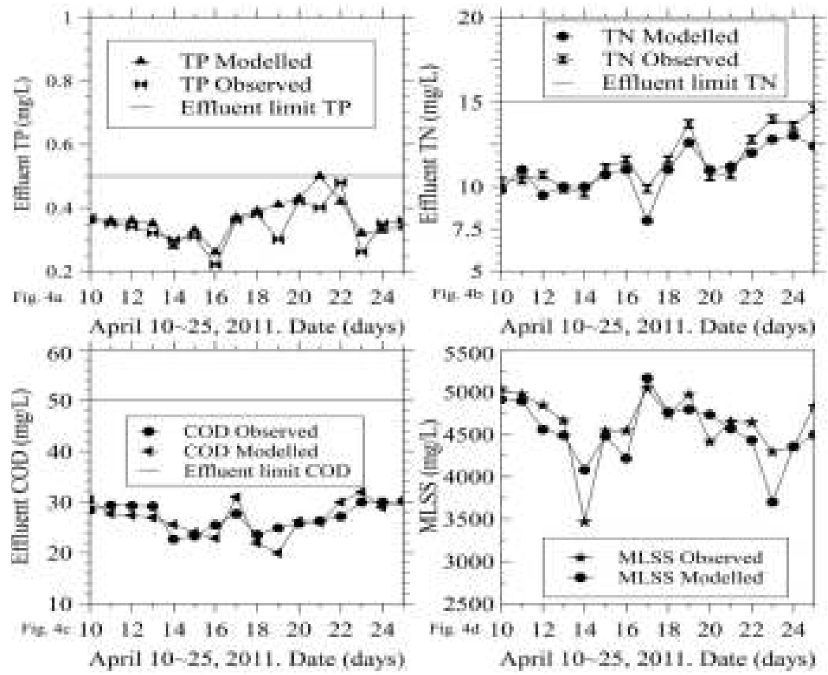


Fig. 4: a-d observed and simulated graphs for model validation

Table 4: Average observed and modeled values

Month		TP	TN	NH_3-N	TSS	MLSS	COD
January	obs	0.25	13.5	1.56	-	3554	22.4
	Mod	0.26	14	1.7	4	3516	22.2
April	obs	0.34	11.6	0.4	10	4624	27.1
	Mod	0.36	11	1.1	7	4440	26.9

obs = observed; mod = modeled; - = no data

To further evaluate the model performance, one of the statistical tests suggested by Power (1993) was used, that is the Janus coefficient. The choice of Janus coefficient over other statistical methods is because; it indicates the change in the predictive quality of the model between the calibration and the validation data sets, in other words, it relates to how valid the model is outside calibration (WEF (Water Environment Federation), 2010; Sin *et al*., 2008). This coefficient is calculated thus:

$$J^2 = \frac{\frac{1}{m}\sum_{i=1}^{m}\left(C_{ob}^{n+i} - C_{\text{mod}}^{n+i}\right)^2}{\frac{1}{n}\sum_{i=1}^{n}\left(C_{ob}^{i} - C_{\text{mod}}^{i}\right)^2} \quad (1)$$

where,

- J = Janus coefficient
- n = Number of values in the calibration data set
- m = Number of values in the validation data se
- C_{ob}^{i} = Observed value at time i
- C_{mod}^{i} = Modeled value at time i.

The statistical evaluation result is presented in Table 5.

Table 5: Statistical values of the model predictive ability

	TP	TN	NH_3-N	COD
Janus coefficient	0.99	0.82	1.2	0.91

The value of J varies between 0 and ∞. If the predictive ability of the model remains more or less constant outside the calibration period, then J will equal approximately 1. The higher J becomes, that is above 1, the poorer the predictive ability of the model with respect to that constituent (WEF (Water Environment Federation), 2010).

The predictive ability of the model outside its calibration with respect to TP, TN and COD are quite good as shown in Table 5, indicating the stability of the model in predicting these constituents and confirming the agreement between the averages observed and modeled values presented in Table 4. On the other hand, the value, 1.2 for NH3-N as shown in Table 5 is on the high side. However, this value could be considered acceptable given the complexity involved in the DO control system. Experiences from the application of activated sludge models have shown wide range of model deviations considered acceptable with relative errors ranging from 10~40% (Melcer *et al*., 2003).

The performance of the model is attributed to changes made in the operating parameters. The use of a

100% internal recycle rate for the simulation instead of a 200% internal recycle rate used in the existing plant resulted in improving the TP concentrations in the effluent in both cases. This outcome is due to the possibility of introducing sufficient dissolved oxygen and nitrate-nitrogen into the anoxic reactor when a high recycle rate is used (Smolders *et al*., 1994). When a 200% internal recycle rate was simulated, it resulted in a small improvement on only TN and nitrate, while it worsened ammonia effluent concentrations and it, required twice the needed energy compared to the 100% internal recycle rate.

The effect of using a proportional RAS flow ratio and a fixed RAS flow rate was modeled, proportional RAS flow ratio produced improved results in all the situations evaluated for the treatment plant when compared to a fixed RAS flow rate. A sensitivity analysis was performed starting with a 25% and ending with a 100% proportional RAS flow ratio to determine which ratio will yield the desired result. From an examination of the sensitivity analysis, a 70% proportional RAS flow ratio produced the best and the most desired results. Ratios below 70% yielded better results in terms of ammonium concentrations; however, the TP concentration in the effluent exceeded the discharged limits. The use of 70% RAS ratio employed here could have played an important role in making up for the limiting nutrient in the anaerobic reactor and thereby facilitates micro-organism activities. Additionally, 1.6~1.8mg/L DO and 0.01~0.05 mg/l DO were maintained in the oxic and anoxic tanks, respectively, for all of the simulations. These values are much lower than the values 3~6mg/L and 0.5mg/l DO for the oxic and anoxic tanks, respectively, currently maintained in the aeration tank, which results in substantial energy savings.

Vital to the success of this simulation is the schedule optimization of the wastage pump run time of 1.5 (h), wastage cycle time of 4.5 (h) and low wastage pump flow rate. The scheduling of the wastage pump run time of 1.5 h and wastage cycle for 4.5 h as against the common practice of scheduling of 3 h wastage pump run time and 6 h wastage cycle, respectively yielded better results. It was observed that the 3 h wastage pump run time and wastage cycle of 6 h as practiced in this treatment plant and many other treatment plants, is sufficient to allow for the second phosphorus release which eventually results in higher TP concentration in the effluent.

CONCLUSION

ASM2 was successfully used to identify operational lapses, evaluate the complex biological reactions and interactions that occur in WWTPs and simulated results were calibrated and verified against experimental data from the WWTPs. It is apparent from the result that ASM2 is capable of depicting the dynamic behavior in the BNR treatment plant with reasonable accuracy. ASM2 is a valid tool for BNR treatment plant process troubleshooting.

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