Chinese Journal of Oceanology and Limnology

http://dx.doi.org/10.1007/s00343-015-4201-z

Achieving low effluent NO_3-N and TN concentrations in low influent chemical oxygen demand (COD) to total Kjeldahl nitrogen (TKN) ratio without using external carbon source*

CAO Jiashun (操家顺)[1, 2, 3], OLEYIBLO Oloche James[2, **], XUE Zhaoxia (薛朝霞)[1, 2, 3], OTACHE Y. Martins[4], FENG Qian (冯骞)[1, 2]

[1] *Key Laboratory of Integrated Regulation and Resource Development on Shallow Lake of Ministry of Education, College of Environment, Hohai University, Nanjing 210098, China*

[2] *College of Environment, Hohai University, Nanjing 210098, China*

[3] *National Engineering Research Center of Water Resource Efficient Utilization and Engineering Safety, Hohai University, Nanjing 210098, China*

[4] *Department of Agricultural & Bioresources Engineering, Federal University of Technology, P.M.B 65, Minna, Niger State, Nigeria*

Received Jul. 19, 2014; accepted in principle Oct. 2, 2014; accepted for publication Nov. 21, 2014

Abstract Two mathematical models were used to optimize the performance of a full-scale biological nutrient removal (BNR) activated treatment plant, a plug-flow bioreactors operated in a 3-stage pho-redox process configuration, anaerobic anoxic oxic (A^2/O). The ASM2d implemented on the platform of WEST2011 software and the BioWin activated sludge/anaerobic digestion (AS/AD) models were used in this study with the aim of consistently achieving the designed effluent criteria at a low operational cost. Four ASM2d parameters (the reduction factor for denitrification ($\eta_{NO_3,H}$), the maximum growth rate of heterotrophs (μ_H), the rate constant for stored polyphosphates in PAOs (q_{pp}), and the hydrolysis rate constant (k_h)) were adjusted. Whereas three BioWin parameters (aerobic decay rate (b_H), heterotrophic dissolved oxygen (DO) half saturation (K_{OA}), and Y_P/acetic) were adjusted. Calibration of the two models was successful; both models have average relative deviations (ARD) less than 10% for all the output variables. Low effluent concentrations of nitrate nitrogen (N-NO_3), total nitrogen (TN), and total phosphorus (TP) were achieved in a full-scale BNR treatment plant having low influent chemical oxygen demand (COD) to total Kjeldahl nitrogen (TKN) ratio (COD/TKN). The effluent total nitrogen and nitrate nitrogen concentrations were improved by 50% and energy consumption was reduced by approximately 25%, which was accomplished by converting the two-pass aerobic compartment of the plug-flow bioreactor to anoxic reactors and being operated in an alternating mode. Findings in this work are helpful in improving the operation of wastewater treatment plant while eliminating the cost of external carbon source and reducing energy consumption.

Keyword: anaerobic anoxic oxic (A^2/O) process; activated sludge; ASM2d; BioWin AS/AD; WEST2011

1 INTRODUCTION

One important issue for wastewater treatment plants (WWTPs) is low influent organic concentration. To ameliorate this problem, several strategies such as the addition of external carbon source, pH control, process improvement have been performed (Tong and Chen, 2007; Wang et al., 2012, 2013, 2014). WWTPs is relatively complex, comprising a series of unit processes. Each of the unit processes is designed to achieve a specific goal. Nevertheless, design engineers, in most cases, are limited to using their experiences and trial and error. Consequently, only limited consideration is given to interactions among the unit processes during the design procedure.

* Supported by the College of Scientific Innovation Significant Cultivation Fund Financing Projects (No. 708047), the Key Special Program for the Pollution Control (No. 2012ZX07101-003), the National Natural Science Foundation of China (No. 51208173), and the Priority Academic Program Development of Jiangsu Higher Education Institutions (PAPD)

** Corresponding author: evangjamesa1@gmail.com, evangjamesa1@yahoo.com

However, improvements on activated sludge models in the last decades have made it possible for complete and compatible set of unit process models for use in comprehensive, systematic design procedures. These models allow good predictions of the effluent compositions of chemical elements in WWTPs. The models are useful tools for troubleshooting and optimization of existing WWTPs, and for the development of control strategies (Buris, 1981; Harleman, 1990; Parker et al., 2000; Guisasola et al., 2006; Kennedy and McHarg, 2007; Griborio et al., 2008; Andres et al., 2008; Zhou et al., 2009b; Oleyiblo et al., 2014). Nevertheless, the calibration of multifaceted WWTPs model is required before it can be applied for process optimization. A calibrated model should be able to predict and describe reactions going on in a WWTP under varying operational conditions, within acceptable error tolerance considering the uncertainties in the system.

Because of discoveries of new processes in recent decades as well as advances in research, the complexity of model structures have increased (Barker and Dold, 1997; Henze et al., 2000). Therefore, a lot of time is required for ad hoc calibration of model parameters. The characterization of essential processes taking place in the bioreactors demands that a compromise be established between difficulties in parameter estimation to calibrate models with numerous parameters. Model validation is required prior to its application for real-time operation. Therefore, four notable systematic protocols for model calibration have been established [STOWA (Hulsbeek et al., 2002; Roeleveld and van Loosdrecht, 2002), BIOMATH (Petersen et al., 2002; Vanrolleghem et al., 2003), WERF (Melcer et al., 2003), and HSG (Langergraber et al., 2004)]. These protocols are structurally synonymous. Each of the protocol begins by defining the calibration goals, which also influences the remaining steps in the procedure. Sin et al. (2005) discussed the strength, weakness, opportunity, and threat (SWOT analysis) of the existing calibration protocols. In order to simplify these calibration protocols, the International Water Association (IWA) Task Group on Good Modelling Practice (GMP) — guidelines for activated sludge modeling based on engineering experiences — has been elaborated (Gillot et al., 2009; IWA Task group, 2011). The GMP unified protocol comprises of five major steps as shown below:

1) project definition;
2) data collection and reconciliation;
3) model setup;
4) model calibration and validation;
5) simulation and result interpretation.

The GMP protocol is used for the purpose of model calibration.

2 METHOD AND PROCEDURE

2.1 Project definition

This study aims at calibrating two different models for a full-scale biological nutrient removal (BNR) treatment plant as well as to optimize the operational strategies of the WWTP under varying influent loads in order to achieve consistent effluent quality. Therefore, the impacts of cycle time, wastage pump run time, internal recycles, and return activated sludge (RAS) on the effluent quality were assessed; and operating the treatment plant in alternative configuration and the influence of solids retention time (SRT) on the effluent quality was evaluated. The description of the WWTP for this study is given in (Oleyiblo et al., 2013a, b).

2.2 Data collection and reconciliation

Available historical data on operation and performance were evaluated to comprehend plant operations and determine the treatment efficiency of the existing processes. Table 1 presents three-year monthly average effluent concentrations of some important variables, whereas Table 2 presents the average monthly influent concentrations and flowage for Changzhou city WWTP. From Table 1, it is obvious that the WWTP has been performing well in terms of meeting effluent discharge standards of China. However, the performance came with extra operating costs incurred from, the addition of external carbon source, and other chemicals supplements to improve denitrification and phosphorus removal. The effluent permit limits for Changzhou WWTP are BOD≤10, COD≤50, NH_3-N≤5, TN≤15, TP≤0. 5 and SS≤10, all in mg/L.

Nonetheless, evaluation on the chemical oxygen demand to total phosphorus ratio (COD/TP) and biochemical oxygen demand to total phosphorus ratio (BOD/TP), as well as chemical oxygen demand (COD) to total Kjeldahl nitrogen (TKN) ratio (COD/TKN), with respect to nutrient requirement for BNR treatment plants were performed. However, the result revealed that, the WWTP could be operated without chemical polishing nor the addition of external carbon

Table 1 Three-years monthly average effluent concentrations (mg/L) in Changzhou WWTP

Month	Chemical item														
	2010					2011					2012				
	COD	TN	TP	NH_3	NO_3	COD	TN	TP	NH_3	NO_3	COD	TN	TP	NH_3	NO_3
Jan.	19.6	17.4	0.1	0.68	16.1	23	12.8	0.27	1.56	9.4	20.3	12.1	0.18	1	11.1
Feb.	15.8	16.8	0.14	0.65	15.7	22.2	12.8	0.35	2	10	20	10.8	0.24	2.2	6.8
Mar.	15	11.8	0.09	0.56	10.5	29.2	10.6	0.29	1.1	8.2	20.8	13.6	0.1	0.55	12.1
Apr.	18.6	11.6	0.11	0.46	10	27.3	12.5	0.39	0.4	11.6	21	12.9	0.16	0.5	12.4
May	21.4	10.9	0.12	0.27	10.6	24.4	11.7	0.38	0.38	10.9	22.3	11.7	0.2	0.3	11.2
Jun.	19.4	13	0.15	0.25	12.2	28.2	8.8	0.29	0.21	7.3	20.7	12.5	0.44	0.3	12
Jul.	13.6	11.3	0.14	0.1	10.9	24.1	8.3	0.34	0.3	7.7	20.6	11	0.32	0.25	10.6
Aug.	11.3	11.1	0.21	0.14	10.5	21	8.5	0.3	0.21	8.2	21	8.54	0.3	0.21	8.2
Sep.	10.4	12	0.23	0.21	10.7	22	10.8	0.31	0.3	10.4	21.7	11.8	0.38	0.17	9.9
Oct.	9.2	12.4	0.32	0.32	11.3	19.9	14	0.27	0.31	12.9	21.2	12.3	0.35	0.18	11.4
Nov.	10	13	0.32	0.34	11.5	14.8	12.2	0.27	0.46	11.6	20	13	0.3	0.32	12.5
Dec.	18.4	13.1	0.28	0.32	12.6	19.6	12	0.23	0.4	10.9	18.7	12	0.32	0.26	11.7

Note: COD: chemical oxygen demand; BOD: biochemical oxygen demand; TN: total nitrogen; TP: total phosphorus; NH_3-N: ammonia nitrogen; NO_3-N: nitrate nitrogen.

Table 2 Designed and three-year recorded data in Changzhou WWTP

Load	Designed 2008	Recorded		
		2010	2011	2012
Mean flow (m^3/d)	50 000	34 342.2	35 207	43 273.4
Standard Deviation (m^3/d)	-	5 562	9 000	7 687
Minimum flow (m^3/d)	-	17 956	10 620	17 057
Maximum flow (m^3/d)	-	49 162	52 100	58 860
Raw sewage composition				
BOD_5 (mg/L)	180	53	74.2	62
COD (mg/L)	400	143	174	165.5
TN (mg/L)	45	26.75	30.53	29.5
NH_3-N (mg/L)	35	21.34	23	25.5
TP (mg/L)	4	2.1	3.26	2.82
SS (mg/L)	250	122	142	160

Note: COD: chemical oxygen demand; BOD: biochemical oxygen demand; TN: total nitrogen; TP: total phosphorus; SS: suspended solid; NH_3-N: ammonia nitrogen.

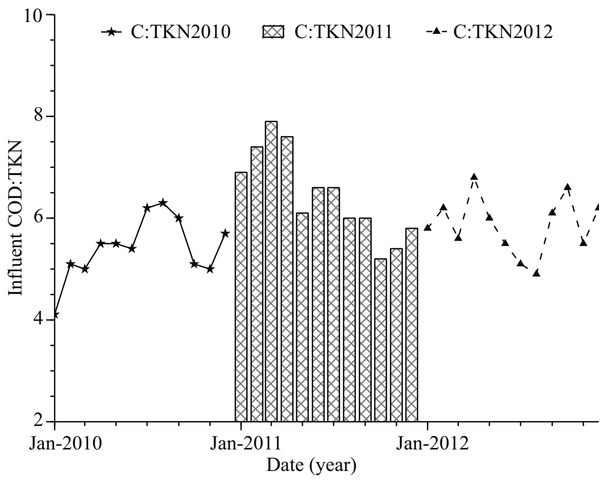

Fig.1 Three-year average influent COD/TKN ratios

source, based on some known hypotheses (Randall et al., 1992; Grady et al., 1999; Henze et al., 2002; Metcalf and Eddy Inc, 2003; Neethling et al., 2005; Barnard, 2006; EPA, 2010). This could result in considerable reduction in operating costs. According to Grady et al. (1999), nitrogen can be removed at COD/TKN ratio of 5, while the optimal ratio in most cases is about 9. However, the COD of NO_3^- as an electron acceptor is 2.86 g COD per g NO_3^-. Additional COD could be required to achieve complete denitrification due to COD and N incorporation into biomass, and the fraction of COD that are slowly biodegradable. Nonetheless, other authors (Henze et al., 2002; Metcalf and Eddy Inc, 2003; Yagci et al., 2006; Feng et al., 2009; EPA, 2010) reported COD/TKN ratio of at least 8 and the minimum BOD/TKN ratio for BNR for complete denitrification ranges between 3 and 4.3. However, the average monthly influent COD/TKN (Fig.1) for the WWTP under study fluctuates in season from 8.1 to 5.34, in average at 7.1.

Mass balance methods were used to check for data reliability, and to identify possible systematic errors (Meijer et al., 2002; Langergraber et al., 2004; Puig et al., 2008; Thomann, 2008). In addition, the unit

Table 3 Operation parameters and the influent characteristics of the WWTP during model calibration and validation

Parameter	April 2011				September/October 2011			
	Mean	SD	Min	Max	Mean	SD	Min	Max
Flow rate (m^3/d)	33 985	2 376	28 500	39 269	42 401	3 249	37 096	48 070
SRT (d)	18	0.6	16	19	14	0.4	13	16
RAS flow rate (m^3/d)	23 312	1 366	19 950	25 629	17 692	600	16 692	18 620
Temperature (°C)	16.6	1.18	14.7	18.7	26.2	1	24.5	28.6
pH	7.5	-	-	7.5	7.5	-	-	7.5
TSS (mg/L)	152	22	126	183	168	19.5	115	227
VSS (mg/L)	89	18	71	130	125	12.2	104	180
ISS (mg/L)	61	12	30	69	43	9	36	58
BOD_5 (mg/L)	88.2	12.9	56	98	48.6	15.2	42.7	71
COD_{tot} (mg/L)	229.6	26	151	246	123	22.3	76	178
S_F (mg/L)	39	3	30	43	18	1.4	12	24
S_A (mg/L)	22	1.7	16	29	15	0.7	9	17.9
X_S (mg/L)	56	5.2	43	66	27	3.4	20.3	35
S_I (mg/L)	23.6	1.3	17.8	26	12.2	0.9	8	15
X_I (mg/L)	60	6.2	48	74	32	4.3	25	39
X_H (mg/L)	30	2.4	24	41	18.8	0.6	16.4	21
TP (mg/L)	4.18	1.12	2.32	8.68	2	0.4	1.32	2.62
SPO_4 (mg/L)	2.25	0.53	1.64	6.8	0.94	0.1	0.53	1.16
TN (mg/L)	31.48	2.9	15.2	35.2	26	2.5	22	28.1
TNi-sol (mg/L)	29.1	3.2	13.4	31.5	23.7	1.4	18.9	25.2
N-NH_3 (mg/L)	27.5	4.16	18.2	35.8	16.6	2.48	14.2	22.5
N-NO_3 (mg/L)	0.22	0.1	0.12	0.26	0.4	0.21	0.1	1.25
N-NO_2 (mg/L)	0.11	0.12	0.04	0.42	0.14	0.1	0.1	0.52

Note: SRT: solids retention time; RAS: return activated sludge; TSS: total suspended solids; VSS: volatile suspended solids; ISS: inert suspended solids; BOD_5: five-day biochemical oxygen demand; COD_{tot}: total chemical oxygen demand; TP: total phosphorus; SPO_4: soluble phosphate; TN: total nitrogen; TNi-sol: influent soluble total nitrogen; NH_3-N: ammonia nitrogen; NO_3-N: nitrate nitrogen; NO_2-N: nitrite nitrogen; S_F: fermentable COD; S_A: acetate; X_S: slowly biodegradable COD fraction; S_I: inert soluble COD fraction; X_I: inert particulate COD fraction; X_H: heterotrophic biomass.

processes and additional sampling campaign aimed at missing data for specific compounds, and key parameters of interests were undertaken. The influent, effluent, sludge recirculation flow rates, internal recycle ratio, pH, and temperature data were evaluated and measured during special sampling campaigns.

The influent to the treatment plant was measured online. Samples were taken every 2 h from the influent, raw influent with recycles, secondary effluent, and final effluent for 24 h, in two separate special sampling campaigns (April, and September/October 2011), for model calibration and validation, respectively. The 24-hour composite data were used for the analysis. The influent characterization was determined by STOWA method (STOWA, 1996; Meijer et al., 2001). Samples were analyzed on COD of non-filtered (total COD), and filtered through 0.45-μm and 1.2-μm diameter pores (COD influent soluble, filtered, flocculated), BOD of non-filtered and filtered, volatile fatty acids (VFA), ammonium nitrogen (NH_3-N), nitrate nitrogen (NO_3-N), total Kjeldahl nitrogen (TKN), total phosphorus (TP), ortho-phosphate (PO_4^3), total suspended solids (TSS) and volatile suspended solids (VSS) concentrations. The experiment was performed in the Changzhou WWTP laboratory in accordance with the standard methods for the examination of water and wastewater (APHA, 1995). The average data (Table 3) collected during the two sampling campaigns were used for steady state calibrations, whereas daily dynamic influent profiles were used for dynamic simulations.

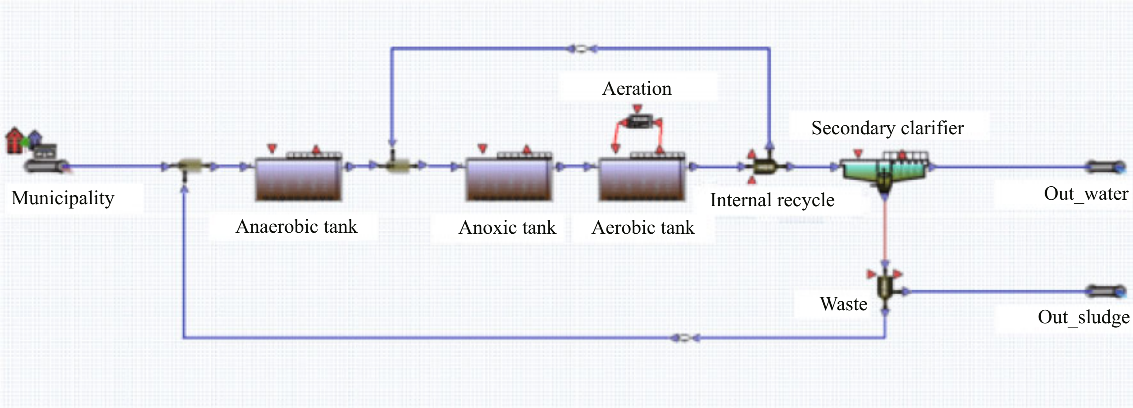

Fig.2 Schematic flow diagram of the biological reactor of Changzhou WWTP in WEST

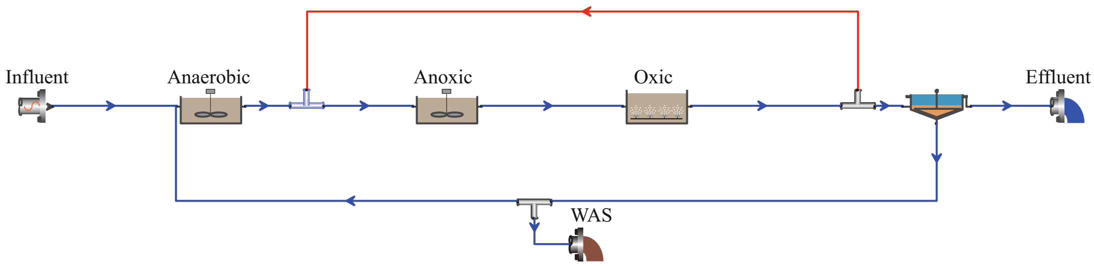

Fig.3 Schematic flow diagram of the biological reactor of Changzhou WWTP in BioWin

2.3 Plant model set-up

ASM2d (Henze et al., 1999) on the platform of WEST2011, and the BioWin® model (Barker and Dold, 1997) were chosen for this study. Although both models use different approaches, nevertheless, they can simulate the metabolic processes of microorganism under anaerobic, anoxic, and aerobic environment in the BNR treatment processes. Incorporated in the WEST2011 is the Takács's settling model (Takács et al., 1991). Whereas BioWin® has three models to evaluate settling and separation processes, a flux based model, an ideal separation model, and a point separation model (User Manual for BioWin v3.0, 2008). In this study, the ideal separation model was used. Moreover, both models have been used extensively for treatment plant process design and optimizations (Guisasola et al., 2006; Kennedy and McHarg, 2007; Griborio et al., 2008; Andres et al., 2008; Zhou et al., 2009b; Liwarska et al., 2013; Oleyiblo et al., 2014). The WEST configuration of Changzhou WWTP is shown in Fig.2.

ASM2d model is an extension of ASM2 (Henze et al., 1995) and ASM1 (Henze et al., 1987). ASM1 have proved very well for modeling nitrification and denitrification, while ASM2 can simulate phosphorus utilization by phosphorus accumulating organisms (PAOs) only under aerobic conditions. On the other hand, ASM2d is capable of simulating phosphorus uptake under both anoxic and aerobic conditions (Mino et al., 1998; Meinhold et al., 1999). The influent total COD (COD_{tot}) is introduced into the WEST model as $COD_{tot}=S_A+S_F+S_I+X_I+X_S+X_H$, through the WEST inbuilt influent fractionation tool.

On the other hand, the BioWin® AS model differentiates five fractions of carbonaceous substrates. These are: (1) fraction of total influent COD, which comprises of soluble and readily biodegradable including acetate (F_{bs}); (2) fraction of readily biodegradable that is volatile fatty acid (VFA) or fermentation product — acetate (F_{ac}); (3) fraction of total influent COD that is soluble non-biodegradable (F_{us}); (4) fraction of total influent COD that is particulate non-biodegradable (F_{up}); and (5) fraction of slowly biodegradable influent COD that is particulate and colloidal (F_{xsp}) (User Manual for BioWin v3.0, 2008). The BioWin® configuration of Changzhou WWTP is shown in Fig.3.

2.4 Model calibration and validation

According to Petersen et al. (2002), model calibration is fitting a model to a set of data collected from a full-scale WWTP. This will required, in most cases, the modification or adjustment of some model parameters from their original or default values. However, model validation with an independent data set collected under entirely different condition from the same WWTP is required prior to its application for process optimization and scenarios analysis. In this study, the data used for model calibration was collected in April 2011, whereas the data for model validation was collected in September/October 2011.

Simple steady state calibration aimed at solids balance over the entire system was performed first. Having a good knowledge of the SRT of the WWTP under study, thus, the wastage sludge was estimated in steady-state loading. It should be stated here that the calibration parameters were selected based on the knowledge of the WWTP rather than on sensitivity analysis. The simulation was performed with the model default parameters; and then the measured and simulated sludge concentrations were compared. The discrepancies between measured and simulated sludge concentrations was negligible, consequently, no parameter associated with biomass concentration was calibrated. This is because the fractionation of the influent compositions were properly performed. More so, the measured and simulated ammonium concentrations matched well, indicating that both defaults values for nitrifier decay rate (b_{AUT}) and oxygen saturation coefficient ($K_{\mathrm{NO_2}}$) needed no adjustment.

However, the simulated effluent nitrate concentration was much lower than the measured nitrate concentration; thereby requiring the calibration of the reduction factor for denitrification ($\eta_{\mathrm{NO_3,H}}$) and the maximum growth rate of heterotrophs (μ_{H}). It should be noted that in ASM2d, denitrification is accomplished by both heterotrophic and phosphorus removing bacteria. Moreover, a reduction factor for denitrification is influent specific dependent. To fit the measured effluent nitrate concentrations to the simulated nitrates concentrations, the reduction factor was reduced from 0.8 to 0.4, and the growth rate of heterotrophs was reduced from 6 to 4.5. Nonetheless, there was disparity between the measured and simulated phosphate concentrations. In order to fit the measured phosphate to the simulated phosphate, the rate constant for stored polyphosphates in PAOs (q_{pp}) was calibrated, the default value was reduced from 1.5 to 1.3. Hence, the steady state calibration was complete, and all the simulated output variables fitted well with the measured output variables.

Dynamic simulation was necessary in order to evaluate whether the calibrated model can mimic the dynamic trend in the WWTP. However, the value of hydrolysis rate constant (k_{h}) had to be increased from 3 to 3.2/day to achieve denitrification during the dynamic simulation. Increasing the rate constant for hydrolysis increases the availability of soluble substrates, which resulted from the conversion of particulate substrates through hydrolysis process. It should be noted, the calibrated parameters are within the range used in ASM models. In a case by Makinia et al. (2006), a larger k_{h} value (4.5) was used, whereas a study by Ginestet et al. (2002), gave different values for different wastewater sources ranging from 7 to 21/day. Makinia et al. (2006), assigned a higher value (0.9) for ($\eta_{\mathrm{NO_3,H}}$), whereas 0.58 and 0.35 were used by Manga et al. (2003) and Russel et al. (2002), respectively. Yagci et al. (2006) used four different q_{pq} values: 1.3, 1.65, 1.86, and 2, respectively, whereas Makinia et al. (2006) used a lower value, 1.2. On the other hand, Insel et al. (2004) used 1.3 for q_{pp}, which is the same with the value used in this study. The operating parameters and wastewater characteristics are given in (Table 3) and the calibrated parameters are given in Table 4.

Table 4 Values of parameters adjusted during the calibration of WEST ASM2d and BioWin AS model

Parameter definition	Symbol	Unit	Default	Calibrated
ASM2d Reduction factor for denitrification	$\eta_{\mathrm{NO_3,H}}$		0.8	0.4
Maximum growth rate of heterotrophs	μ_{H}	/d	6	4.5
Hydrolysis rate constant	k_{h}	/d	3	3.2
Rate constant for storage of X_{PAO}, pp	q_{pp}	gP. (g COD/d)	1.5	1.3
BioWin Y_{P}/acetic		mg P/mg COD	0.49	0.41
Aerobic decay rate	b_{H}	/d	0.62	0.6
Switches Heterotrophic DO half saturation	K_{OA}	mg O_2/L	0.05	0.1

In addition to the specific raw wastewater COD fractions (Table 5) determined according to Mamais et al. (1993), three BioWin® parameters were calibrated. Two parameters, aerobic decay rate (b_{H}) and heterotrophic dissolved oxygen (DO) half saturation (K_{OA}) were adjusted during the steady state simulation. b_{H} was reduced from 0.62 to 0.6 whereas K_{OA} was increased from 0.05 to 0.1, during the steady state simulation to fit the simulated mixed liquor

Table 5 specific raw wastewater COD fractions for BioWin

Name	Description	Model default	Calculated
F_{bs}	Fraction of total influent COD which is soluble and readily biodegradable including acetate	0.15	0.14 (plant data)
F_{ac}	Fraction of readily biodegradable which is VFA or fermentation product — acetate (very essential for bio-P removal)	0.16	0.13 (plant data)
F_{us}	Fraction of total influent COD which is soluble non- biodegradable	0.05	0.061 (plant data)
F_{up}	Fraction of total influent COD which is particulate non- biodegradable	0.13	default
F_{xsp}	Fraction of slowly biodegradable influent COD which is particulate and colloidal	0.75	default
F_{na}	Ammonia	0.66	0.88 (plant data)
F_{PO_4}	Phosphate	0.5	0.65 (plant data)

Table 6 Measured and simulated values of variables under steady state simulation and validation

Model	TP	TN	NH_3	NO_3	COD	TSS	MLSS
April 2011 for model calibration							
Observed	0.34	11.6	0.4	8.9	27.1	10	4 624
BioWin	0.36	11.3	0.5	9.3	25.8	8	4 671
WEST	0.38	12.2	0.44	8.6	27.5	10.86	4 540
Sept/Oct 2011 for model validation							
Observed	0.31	12.6	0.3	8.7	23.2	5	2 722
BioWin	0.35	13.1	0.33	10	21.7	6.6	2 745
WEST	0.37	15.2	0.36	8.3	22.5	9	2 680

Note: COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus; NH_3-N: ammonia; NO_3-N: nitrate; TSS: total suspended solids; MLSS: mixed liquor suspended solids.

suspended solid (MLSS) and NH_3 to their observed values. Furthermore, one parameter (Y_P/acetic) was calibrated during the dynamic simulation in order to reproduce the phosphorus trend in the effluent concentration. Nonetheless, the calibrated parameters are within the range used in the ASM models (Hulsbeek et al., 2002; Liwarska and Biernacki, 2010). The measured and simulated steady state effluent variables are given in Table 6 for model calibration and validation.

The aim of steady state calibration is for the model to have average simulated effluent concentrations in the same order of magnitude with the plant-measured data, and to get the solids balance right in order to have good initial conditions for the dynamic simulation. However, dynamic simulation, on the other hand, aimed at imitating the trend in the WWTP for all the output variables, with an acceptable error margin.. Therefore, targets were set for parameters of interests as follows: effluent TP=10%, TN=10%, COD=10%, NO_3-N=10% and NH_3-N=15%, average relative difference (ARD) between simulated and observed values for the dynamic simulations. ARD

Table 7 ARD results for both calibration and validation period

Month	TP	TN	COD	NO_3-N	NH_3-N	MLSS
April 2011, calibration BioWin	0.89	2.4	2.2	5	5.2	3.5
WEST	0.91	2.7	2.7	4.3	3.5	3.4
Sept/Oct 2011, validation BioWin	1.12	3.1	2.2	4.4	4.8	3.1
WEST	1.14	3.6	2.7	3.1	5.5	3.3

Note: COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus; NH_3-N: ammonia; NO_3-N: nitrate; MLSS: mixed liquor suspended solids.

can be expressed as Eq.1:

$$\mathrm{ARD}=\frac{1}{N}\sum_{i=1}^{N}\frac{|(\mathrm{Ob}_i-\mathrm{Mod}_i)|}{\mathrm{Ob}_i}100\%, \tag{1}$$

where ARD=average relative deviation, N=number of observations, Ob_i=observed value at time i, Mod_i=modeled value at time i.

The calculated ARD results are given in Table 7.

The trend in the WWTP versus the model prediction are given in Figs.4a–d and 5a–d for model calibration and validation, respectively to support the results in Table 7.

The values of ARD (Table 7) varied from 0.89 to 5.5, which is within the calibration targets set for every variable. More importantly, the ARD values are all below 20%, which is the upper discrepancy limit acceptable for a calibrated model (Petersen et al., 2002). This shows that the calibration of the two models were performed properly (Makinia et al., 2006; Makinia, 2010).

Furthermore, the model predictive ability was evaluated by determining its predictive quality and stability between the calibration and the validation data sets, using Janus coefficient (J) (Power, 1993; Sin et al., 2008). The J value varies between 0 and ∞.

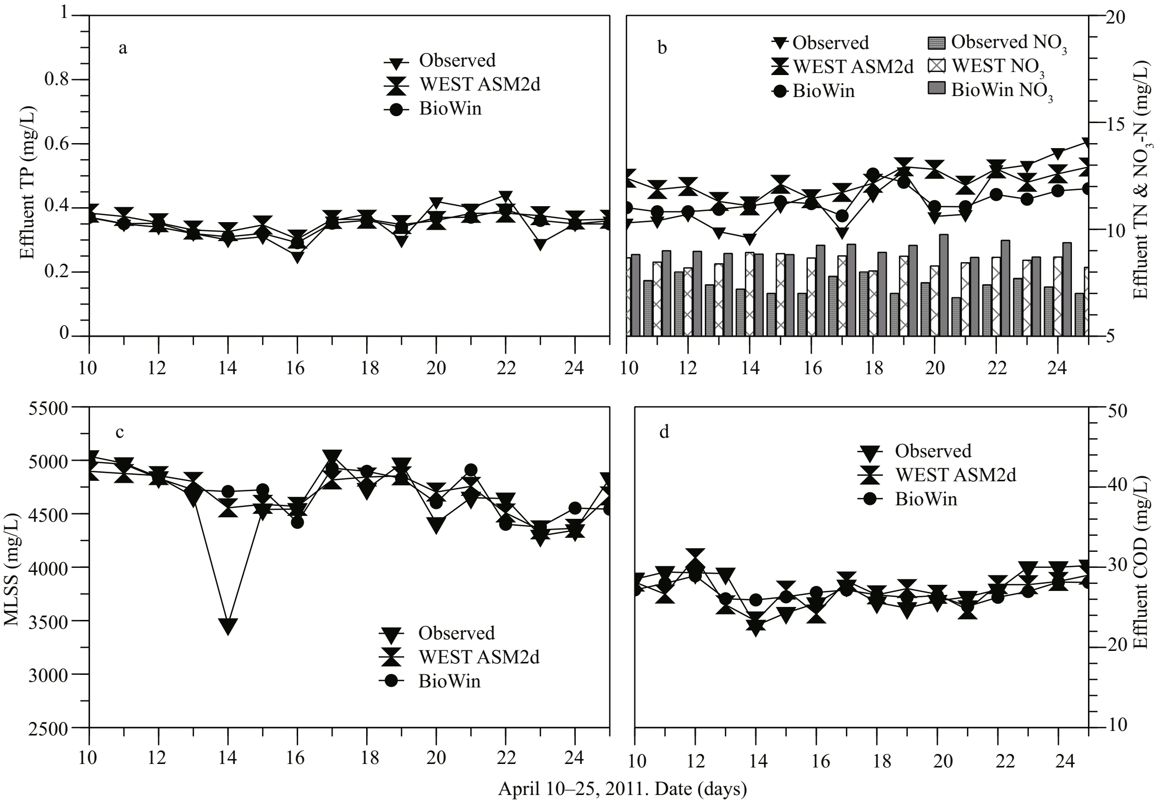

Fig.4 Observed and simulated variables under dynamic simulation for model calibration

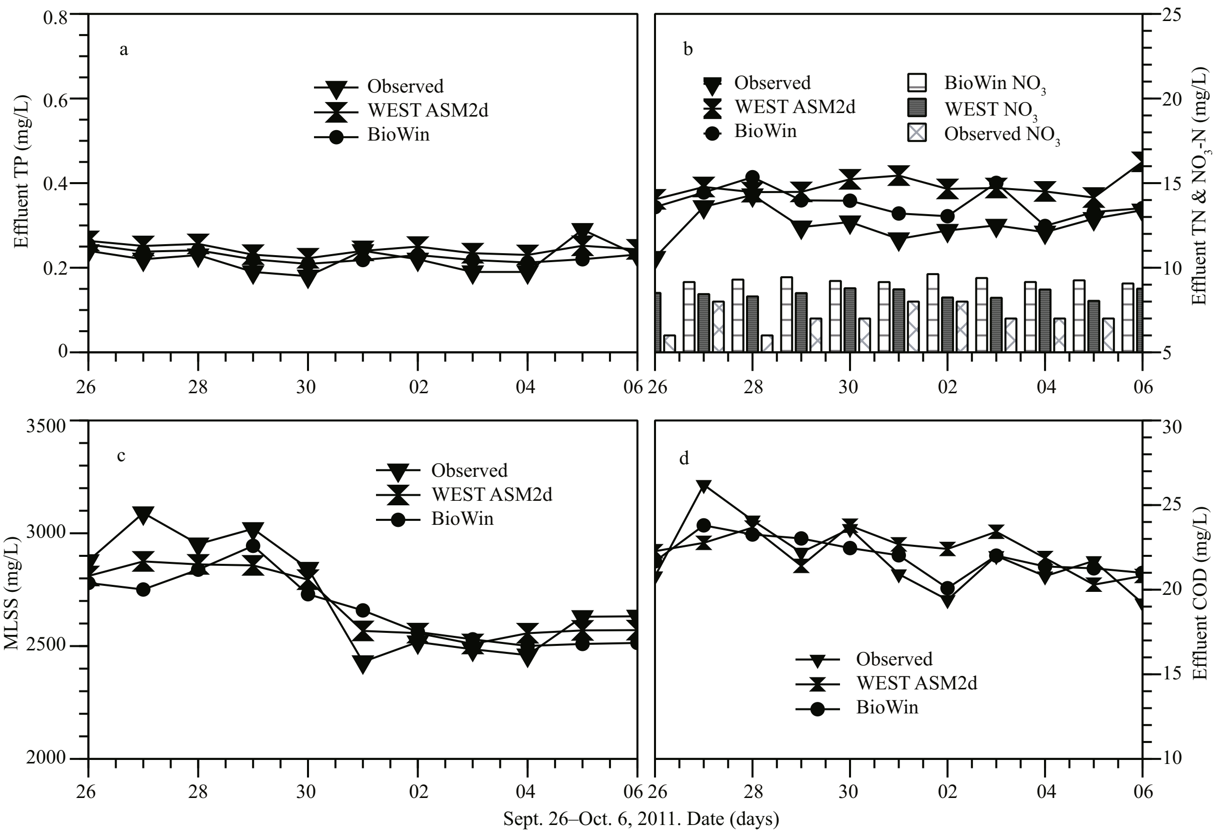

Fig.5 Observed and simulated variables under dynamic simulation for model validation

Table 8 Values of Janus coefficient on model predictive ability

Month	TP	TN	COD	NO_3-N	NH_3-N	MLSS
BioWin	0.94	0.81	0.92	0.97	0.81	0.98
WEST	0.95	0.83	0.94	0.95	0.97	0.94

Note: COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus; NH_3-N: ammonia; NO_3-N: nitrate; MLSS: mixed liquor suspended solids.

The higher J becomes or greater than 1, the poorer the predictive ability of the model with regard to a particular variable. If J is approximately equal to 1, then the predictive ability of the model remains more or less constant outside the calibration period.

Janus coefficient is calculated as Eq.2:

$$J^2 = \frac{\frac{1}{m}\sum_{i=1}^{m}(C_{\text{ob}}^{n+i} - C_{\text{mob}}^{n+i})^2}{\frac{1}{n}\sum_{i=1}^{n}(C_{\text{ob}}^{i} - C_{\text{mob}}^{i})^2}, \tag{2}$$

where J=Janus coefficient, n=number of values in the calibration data set, m=number of values in the validation data set, C_{ob}^{i}=observed value at time i, C_{mob}^{i}=modelled value at time i.

The predictive ability of the model outside its calibration (Table 8) are within the neighborhood of 1, indicating that the model is stable, and it also confirmed the agreement between the averages observed and modeled (Table 6), and the ARD results (Table 7), respectively. However, it is obvious in Tables 5 and 6 and Figs.4 and 5 that the main fraction of total nitrogen concentration in the effluent was nitrate, having concentrations varied from 8.3 to 10 mg N/dm^3, whereas the concentration of ammonium varied from 0.3 to 0.5 mg N/dm^3. This implies that the treatment plant had difficulties in denitrifying properly. Consequently, there is need to optimize the operating conditions in order to improve the effluent concentrations of this WWTP.

2.5 Simulation and result interpretation

To achieve the project goals as stated under project definition, a number of scenarios simulations were performed with the aid of the validated model, and they are discussed as follows:

2.5.1 The impacts of cycle time and wastage pump run time on effluent quality

An interval between wastage events defines the cycle time, and it is equal to the sum of pump on-and-off times. A sensitivity analysis was performed to evaluate the impacts of cycle time, the wastage pump run time, and the internal recycle ratios. The result reveals that the longer the cycle time, the greater the risk and possibility of secondary TP release in the sedimentation tank, and the shorter the cycle time, the less the risk of secondary TP release (Fig.6a). When sludge was wasted for 3 h in every 6 h [3 h, 6 h], effluent TP concentrations deteriorated compared to when sludge was wasted for 1.5 h in every 4.5 h [1.5 h, 4.5 h]. This phenomenon is due to deflocculating of the floc in the sedimentation tanks causing cell lysis and the release of stored poly-phosphate into the water phase, and consequently, increasing the effluent TP concentrations (Fig.6a). However, Wouters-wasiak et al. (1994) attributed secondary P release to low nitrate (below 0.5 mg N/L) in the clarifier in the absence of oxygen.

To confirm that the high TP observed did not arise from the rising sludge caused by undesired denitrification in the secondary clarifier (Henze et al., 1993), nor that of low nitrate in the clarifier (Wouters-wasiak et al., 1994), the effluent concentrations of NH_3-N, NO_3-N, and TN were evaluated to compare their disparity under the same scenario Oleyiblo et al. (2013a). However, there was no visible difference in TN concentration (Fig.6c), a very negligible improvement in NH_3-N concentration (Fig.6b), and a negligible increase in NO_3-N concentration (Fig.6d). Meanwhile, NO_3-N concentration was much higher than what was postulated by Wouters-wasiak et al. (1994), thus crediting the increase in TP effluent concentration to a possible cell lysis resulting in the second TP release due to longer cycle time.

2.5.2 The effect of internal recycles on the effluent quality

A 200%–400% internal recycle results (not shown) showed little improvement on TN effluent concentration and the ammonium concentration, however, there was no obvious improvement on TP effluent concentration. Nevertheless, the use of 300%–400% internal recycle would not be economically prudent as it requires 3–4 times energy consumption compared to a 100% internal recycle, thereby increasing the operating costs (Baeza et al., 2003). Moreover, an increase in nitrate load to the anoxic reactor indicates a higher COD removal in the reactor; however, in this study, the influent COD concentration was low and thus a higher internal recycle is not necessary. Besides, the most effective internal nitrate recycle depends on the COD/TKN

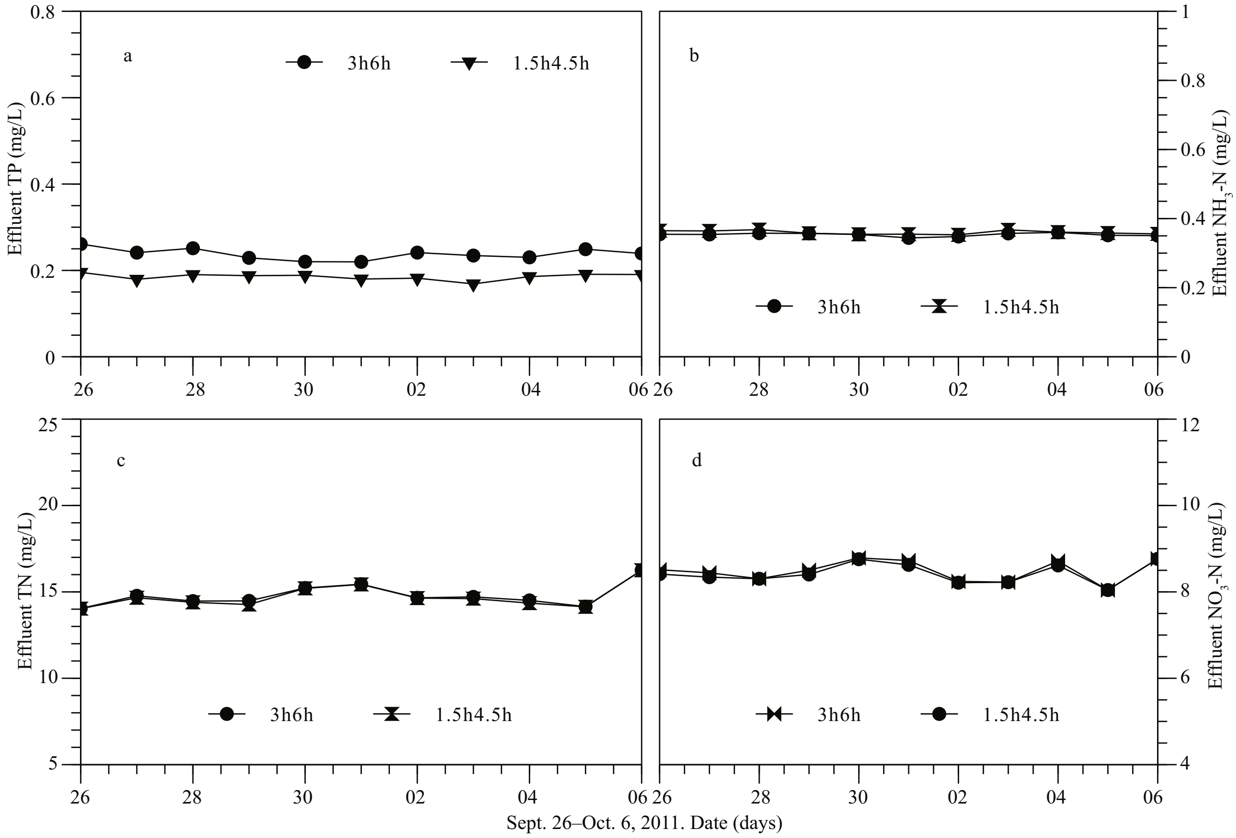

Fig.6 Effluent TP, TN, NO_3-N and NH_3-N concentrations

ratio in the anoxic zone. In addition, if there is insufficient BOD to reduce the quantity of nitrates entering the anoxic zone, additional recycle may rather be detrimental instead of being beneficial in the treatment process because internal recycle normally recycles DO as well as nitrates. Hence, the use of a 100% internal recycle prevented the possibility of excessive dissolved oxygen (DO) being introduced to anoxic reactor, which might have negative impact on the reactor's denitrifying efficiency.

2.5.3 The effect of RAS on effluent quality

Sensitivity analysis was performed starting with a 25% and ending with a 100% proportional RAS flow ratio to determine the optimal ratio. The result of the sensitivity analysis shows that 70% proportional RAS flow ratio gave the best effluent concentrations for the output variables during the model calibration on the data sampled in the April campaign. However, for model validation on September/October campaign, 45% gave the best effluent concentrations. It is important to note (Table 1), the influent pollutants or nutrient concentrations from these periods.

Therefore, it can be seen that influent nutrients concentrations to the WWTP plays vital role in determining the %RAS. Higher RAS indicates larger amount of active microorganism returns into the tank, and vice versa. In certain operational situations, however, returning more competitive active microorganism into the anaerobic tank may happen when there is insufficient food to support their activities. Similarly, there is also a likelihood of introducing less microorganism into the tank when there is surplus food beyond their uptake capability, hence, the organism might not utilize the nutrients completely (Oleyiblo et al., 2013a). This principle was employed in this work, as can be seen (Tables 2 and 3): the COD, BOD, and other nutrient concentrations are higher in the month of April, compared to September and October; therefore, 70% and 45%RAS were used, respectively.

2.5.4 Alternating the aerobic compartment of plug flow reactor

Each stream or lane of the plug flow reactor comprises of one anaerobic pass, three anoxic passes, and four aerobic passes. In this configuration, the anaerobic and anoxic compartments remained unchanged, and the first aerobic pass was assigned DO value of 1 mg/L, whereas the second and the third

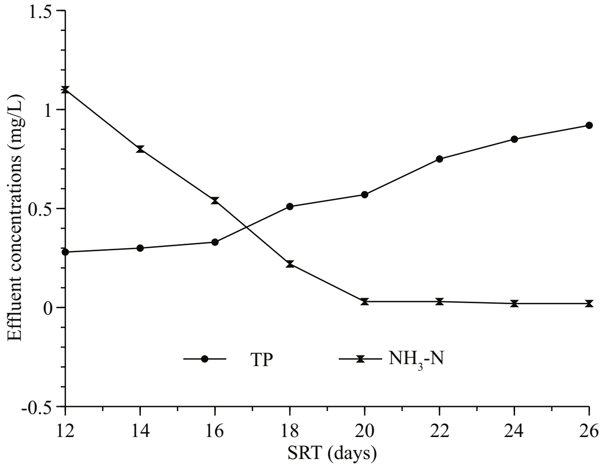

Fig.7 Effect of SRT on effluent NH_3-N and TP

Table 9 Simulated results when the aerobic compartment was alternated

Variable	TP	TN	NO_3-N	NH_3-N	COD	TSS	MLSS
Values	0.35	6.7	4.6	0.26	19.3	6.5	2692

Note: COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus; NH_3-N: ammonia; NO_3-N: nitrate; TSS: total suspended solids; MLSS: mixed liquor suspended solids.

passes were converted to anoxic tank, and the fourth aerobic pass was assigned DO value of 2.2 mg/L. In other words, the aerobic compartment is simply operated in an alternating way (aerobic, anoxic, anoxic, and aerobic) and thus, resulting in a multiple combinations of aerobic and anoxic phases. The overall aerobic volume was reduced when two of the aerobic passes were converted to anoxic compartments. This reduction of aerobic volume is favorable for simultaneous nitrification-denitrification, and consequently improved the effluent concentrations of both nitrate and total nitrogen. The simulation results for this configuration achieved the best effluent concentrations of the output variables as shown in Table 9. This result corroborate with the finding of Wang et al. (2008, 2012). According to Wang et al., the aerobic extended-idle (AEI) regime has some benefits, such as tolerance of higher nitrate level (Wang et al., 2012), and reduced dependence upon wastewater VFA content (Wang et al., 2008).

Reduction in aerobic compartment reduces the aerobic hydraulic retention time (HRT) from 10.67 h to 9 h and thereby reduces the risk of aerobic P-release that resulted from endogenous biomass destruction, whereas anoxic HRT increased from 4 h to 5.5 h, which favors simultaneous nitrification and denitrification. The performance indicates that the

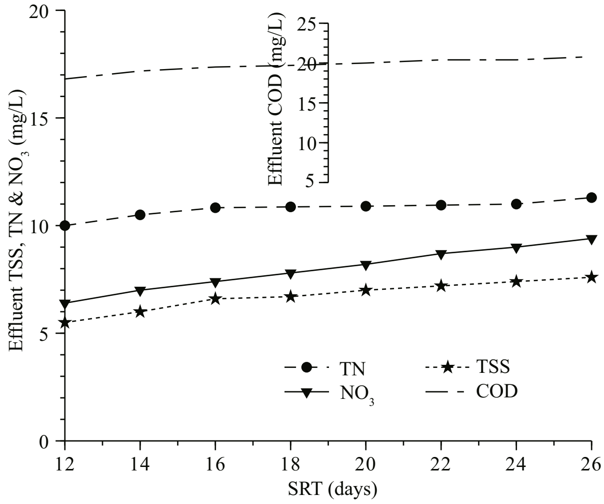

Fig.8 Effect of SRT on COD, TN, NO3-N, and TSS

aeration tank was over designed, which affected the denitrification capability of the treatment plant. It also shows that both limited anoxic HRT and DO were the major factors affecting denitrification, given that the washout of microorganisms associated with SRT limitation was not observed because of sufficient SRT in the systems.

2.5.5 The influence of SRT on effluent quality

The influence of SRT on the efficiency of biological nutrient removal was evaluated in the range of 12 to 26 days, and the temperature was maintained at 14°C. The simulation results are shown in Figs.7 and 8. The influence of the change in SRT on the effluent quality is obvious. The concentration of ammonium nitrogen improved substantially with increasing SRT, while effluent concentrations of other variables deteriorated with increasing SRT, especially the total phosphorus (Fig.7).

Ammonium nitrogen concentration approaches zero from 20 d SRT onward, whereas the total nitrogen concentration and COD remained almost constant; and the nitrate nitrogen and TSS increased linearly with increase in SRT. The result suggests that operating the Changzhou WWTP between 12 and 18 days SRT is long enough to sustain nitrification and is suitable for biological nutrient removal WWTPs (Liwarska et al., 2013).

3 CONCLUSION

In this study, ASM2d implemented on the platform of WEST2011 software and the BioWin® activated sludge/anaerobic digestion (AS/AD) models were successfully calibrated. None of the variables exceeded the target effluent limits in both steady state and dynamic simulations. The maximum difference between observed/measured and the simulated output variables based on average relative deviation (ARD) was 5.5, which means none of the variables did exceed 20%. The predictive ability of the model outside its calibration period for all output variables are in the neighborhood of 1 based on Janus coefficient, which implies that the models are stable and they can be used for real-time WWTPs operations. TP effluent concentration was found better at shorter cycle times. Hence, the control of cycle time and wastage pump run time are paramount in meeting consistent effluent TP concentration. More so, there was linear correlation between SRT and effluent quality, indicating that operating the WWTP at the minimum required SRT to sustain nitrification is most beneficial whereas a longer SRT is detrimental to its performance. In addition, the concentrations of nutrients in the influent to the WWTPs should be considered when determining the %RAS, especially in WWTPs experiencing low influent nutrient concentrations. Most importantly, alternating the aerobic compartment of an A2/O plug flow configuration has proved successful in improving effluent concentrations without any capital work, and has resulted in considerable reduction in energy cost by converting the aerobic reactor to anoxic reactor. Finally, the design of BNR treatment plants should take into consideration the influent BOD/TP, COD/TP, and COD/TKN ratios.

4 ACKNOWLEDGMENT

The authors are grateful to Hohai University for supporting this work. Also included in the acknowledgment is the staff of Changzhou WWTP, the Editors, and the anonymous reviewers for their valuable observations and suggestions.

References

Andres H, Newbigging M, Bicudo J. 2008. Using a Wastewater Process Model to Evaluate the Capacity of the Galt WWTP Under Increased Load from an Industrial Source. Proceedings of the 37th Annual WEAO Technical Symposium, Blue Mountain, Ontario, 2008.

APHA. 1995. Standard Methods for the Examination of Water and Wastewater. 19th edn. American Public Health, Washington DC, USA.

Baeza J A, Gabriel D, Lafuente J. 2003. Effect of internal recycle on the nitrogen removal efficiency of an anaerobic/anoxic/oxic (A^2/O) wastewater treatment plant (WWTP). *Process Biochem*., **39**(11): 1 615-1 624, http://dx.doi.org/10.1016/S0032-9592(03)00300-5.

Barker P S, Dold P L. 1997. General model for biological nutrient removal activated-sludge systems: model presentation. *Water*. *Environ*. *Res*., **69**(5): 969-984, http://dx.doi.org/10.2175/106143097X125669.

Barnard J L. 2006. Biological nutrient removal: Where we have been, where we are going? *In*: WEFTEC 2006, Water Environment Federation, Alexandria, VA, USA. p.1-25.

Buris B E. 1981. Energy conservation for existing wastewater treatment plants. *J. Water Pollut*. *Control Fed*., **53**(5): 536-545.

EPA. 2010. Nutrient Control Design Manual: State of Technology Review Report by Cadmus Group, Inc. Office of Research and Development/National Risk Management Research Laboratory, EPA/600/R-10/100|August 2010, United States Environmental Protection Agency, Cincinnati, Ohio 45268.

Feng L Y, Wang H, Chen Y G, Wang Q. 2009. Effect of solids retention time and temperature on waste activated sludge hydrolysis and short-chain fatty acids accumulation under alkaline conditions in continuous-flow reactors. *Bioresour*. *Technol*., **100**(1): 44-49, http://dx.doi.org/10.1016/j.biortech.2008.05.028.

Gillot S, Ohtsuki, T, Rieger L, Shaw A R, Tackas I, Winkler S. 2009. Development of a unified protocol for good modeling practice in activated sludge modeling. Influents, 4, 70.

Ginestet P, Maisonnier A, Sperandio M. 2002. Wastewater COD characterization: biodegradability of physico-chemical fractions. *Water Sci*. *Technol*., **45**(6): 89-97.

Grady Jr, C P L, Daigger G T, Lim H C. 1999. Biological Wastewater Treatment. 2nd edn. Marcel Dekker, New York.

Griborio A, Rohrbacher J, Taylor R, Pitt P, Litmer R. 2008. Evaluation of Wet Weather Strategies and Clarifier Optimization Using State-of-the-Art Tools. *In*: Proceeding of the 81th Annual Water Environment Federation Technical Exposition and Conference, Chicago, IL. p.4 957-4 968.

Guisasola A, Pijuan M, Baeza J A, Carrera J, Lafuente J. 2006. Improving the start-up of an EBPR system using OUR to control the aerobic phase length: a simulation study. *Water Sci*. *Technol*., **53**(4-5): 253-262, http://dx.doi.org/10.2166/wst.2006.130.

Harleman D R F. 1990. Cutting the waste in wastewater cleanups. *Technology Review*, **93**(3): 60-68.

Henze M, Dupont R, Grau P, De La S A. 1993. Rising sludge in secondary settlers due to denitrification. *Water Res*., **27**(2): 231-236, http://dx.doi.org/10.1016/0043-1354(93)90080-2.

Henze M, Grady C P L, Gujer W, Marais G V R, Matsuo T. 1987. A general model for single-sludge wastewater treatment systems. *Water Res*., **21**(5): 505-515.

Henze M, Gujer W, Mino T, Matsuo T, Wentzel M C, Marais G

V R. 1995. Activated Sludge Model No. 2. IAWQ Scientific and Technical Report No. 3. IAWQ, London.

Henze M, Gujer W, Mino T, Matsuo T, Wentzel M C, Marais G V R, van Loosdrecht M C M. 1999. Activated sludge model No.2D, ASM2D. *Water Sci*. *Technol*., **39**(1): 165-182, http://dx.doi.org/10.1016/S0273-1223(98)00829-4.

Henze M, Gujer W Mino T, van Loosdrecht M C M. 2000. Activated sludge models ASM1, ASM2, ASM2D and ASM3. IWA Scientific and Technical Report No. 9. IWA Publishing, London, UK.

Henze M, Harremoes P, Jansen J, Arvin E. 2002. Wastewater Treatment: Biological and Chemical Processes. Verlag, Berlin.

Hulsbeek J J W, Kruit J, Roeleveld P J, van Loosdrecht M C M. 2002. A practical protocol for dynamic modelling of activated sludge systems. *Water Sci*. *Technol*., **45**(6): 127-136.

Insel G, Sin G, Lee D S, Vanrolleghem P A. 2004. A calibration methodology and model-based system analysis for nutrient removing SBRs, 3rd IWA International conference on SBR: Feb. 22-26, Noosa, Queensland, Australia.

IWA Task Group. 2011. On Good Modelling Practice: Guidelines for use of activated sludge models. http://www.iwahq.org/fc/networks/task-groups/task-group-on-good-modelling-practice.html.

Kennedy K J, McHarg A M. 2007. Optimization of municipal wastewater biological nutrient removal using ASM2d. *J*. *Environ Eng*. *Sci*., **6**(1): 31-43.

Langergraber G, Reiger L, Winkler S, Alex J, Wiese J, Owerdieck C, Ahnert M, Simon J, Maurer M. 2004. A guideline for simulation studies of wastewater treatment plants. *Water Sci*. *Technol*., **50**(7): 131-138.

Liwarska-Bizukojc E, Biernacki R. 2010. Identification of the most sensitive parameters in the activated sludge model implemented in BioWin software. *Bioresour*. *Technol*., **101**(19): 7 278-7 285, http://dx.doi.org/10.1016/j.biortech.2010.04.065.

Liwarska-Bizukojc E, Olejnik D, Biernacki R, Ledakowicz S. 2013. Improving the operation of the full scale wastewater treatment plant with use of a complex activated sludge model. *Environ*. *Prot*. *Eng*., **39**(1): 183-195.

Makinia J. 2010. Mathematical Modelling and Computer Simulation of Activated Sludge Systems. IWA Publishing, London.

Makinia J, Rosenwinkel K, Spering V. 2006. Comparison of two model concepts for simulation of nitrogen removal at a full-scale biological nutrient removal pilot plant. *J*. *Environ*. *Eng*., **132**(4): 476-487, http://dx.doi.org/10.1061/(ASCE)0733-9372(2006)132:4(476).

Manga J, Ferrer J, Seco A, Garcia-Usach F. 2003. Design of nutrient removal activated sludge systems. *Water Sci*. *Technol*., **47**(11): 115-122.

Mamais D, Jenkins D, Pitt P. 1993. A rapid physical-chemical method for the determination of readily biodegradable soluble COD in municipal wastewater. *Water Res*., **27**(1) 195-197, http://dx.doi.org/10.1016/0043-1354(93)90211-y.

Meijer S C F, van der Spoel, H, Susanti S, Heijnen J J, van Loosdrecht M C M. 2002. Error diagnostics and data reconciliation for activated sludge modelling using mass balances. *Water Sci*. *Technol*., **45**(6): 145-156.

Meijer S C F, van Loosdrecht M C M, Heijnen J J. 2001. Metabolic modelling of full-scale biological nitrogen and phosphorus removing WWTP's. *Water Res*., **35**(11): 2 711-2 723, http://dx.doi.org/10.1016/S0043-1354(00)00567-4.

Melcer H, Dold P L, Jones R M, Bye C M, Takács I, Stensel H D, Wilson A W, Sun P, Bury S. 2003. Methods for wastewater characterization in activated sludge modelling. WERF Report 99-WWF-3. Water Environment Research Foundation, Alexandria, VA, USA.

Meinhold T, Filipe C D M, Daigger G T, Isaacs S. 1999. Characterization of the denitrifying fraction of phosphate accumulating organisms in biological phosphate removal. *Water Sci*. *Technol*., **39**(1): 31-42, http://dx.doi.org/10.1016/S0273-1223(98)00773-2.

Metcalf and Eddy Inc. 2003. Wastewater Engineering Treatment and Re-use. 4th edn. McGraw-Hill, Higher Education, New York. 1848p.

Mino T, van Loosdrecht M C M, Heijnen J J. 1998. Microbiology and biochemistry of the enhanced Biological phosphate removal process. *Water Res*., **32**(11): 3 193-3 207, http://dx.doi.org/10.1016/S0043-1354(98)00129-8.

Neethling J B, Bakke B, Benisch M, Gu A, Stephens A, Stensel H D, Moore R. 2005. Factors Influencing the Reliability of Enhanced Biological Phosphorus Removal, WERF Report, 01-CTS-03, Water Environment Research Foundation, Alexandra, VA, USA.

Oleyiblo O J, Cao J S, Feng Q, Wang G, Xue Z X, Fang F. 2014. Evaluation and improvement of wastewater treatment plant performance using BioWin. *Chin*. *J*. *Oceanol*. *Limnol*., Accepted for publication. http://dx.doi.org/10.1007/s00343-015-4108-8.

Oleyiblo O J, Cao J S, Lu X G. 2013a. The use of simulation modelling for optimisation of phosphorus removal in sewage treatment under varying influent loading. *RJASET*, **6**(24): 4 663-4 670.

Oleyiblo O J, Cao J S, Lu X G. 2013b. Troubleshooting a full-scale wastewater treatment plant for biological nutrient removal. *RJASET*, **7**(4): 745-753.

Parker D, Esquer M, Hetherington M, Malik A, Robinson D, Wahlberg E, Wang J. 2000. Assessment and optimization of a chemically enhanced primary treatment system. *In*: WEFTEC 2000. Proceedings of the 73rd Annual Conference on Water Quality and Wastewater Treatment. Anaheim. p.560-573.

Petersen B, Vanrolleghem P A, Gernaey K, Henze M. 2002. Evaluation of an ASM1 model calibration procedure on a municipal-industrial wastewater treatment plant. *J*. *Hydroinform*., **4**: 15-38.

Power M. 1993. The predictive validation of ecological and environmental models. *Ecol*. *Modelling*., **68**(1-2): 33-50, http://dx.doi.org/10.1016/0304-3800(93)90106-3.

Puig S, van Loosdrecht M C M, Colprim J, Meijer S C F. 2008.

Data Evaluation of full-scale wastewater treatment plants by mass balance. *Water Res*., **42**(18): 4 645-4 655, http://dx.doi.org/10.1016/j.watres.2008.08.009.

Randall C W, Barnard J L, Stensel H D. 1992. Design and Retrofit of Wastewater Treatment Plants for Biological Nutrient Removal. Technomic Publishing Company, Lancaster, Pennsylvania, USA.

Roeleveld P J, van Loosdrecht M C M. 2002. Experience with guidelines for wastewater characterisation in the Netherlands. *Water Sci*. *Technol*., **45**(6): 77-87.

Russel B M, Henriksen J P, Jørgensen S B, Gani R. 2002. Integration of design and control through model analysis. *Comput*. *Chem*. *Eng*, **26**(2): 213-225, http://dx.doi.org/10.1016/S0098-1354(01)00742-6.

Sin G, De Pauw D J W, Weijers S, Vanrolleghem P A. 2008. An efficient approach to automate the manual trial and error calibration of activated sludge models. *Biotechnol*. *Bioeng*., **100**(3): 516-528, http://dx.doi.org/10.1002/bit.21769.

Sin G, van Hulle S W H, De Pauw D J W, van Griensven A, Vanrolleghem P A. 2005. A critical comparison of systematic calibration protocols for activated sludge models: a SWOT analysis. *Water Res*., **39**(12): 2 459-2 474, http://dx.doi.org/10.1016/j.watres.2005.05.006.

STOWA. 1996. Methods for influent characterization: inventory and guidelines. Report STOWA 96-08. Utrecht, the Netherlands. (in Dutch)

Takács I, Patry G G, Nolasco D. 1991. A dynamic model of the clarification-thickening process. *Water Res*., **25**(10): 1 263-1 271, http://dx.doi.org/10.1016/0043-1354(91)90066-Y.

Thomann M. 2008. Quality evaluation methods for wastewater treatment plant data. *Water Sci*. *Technol*., **57**(10): 1 601-1 609, http://dx.doi.org/10.2166/wst.2008.151.

Tong J, Chen Y G. 2007. Enhanced biological phosphorus removal driven by short-chain fatty acids produced from waste activated sludge alkaline fermentation. *Environ*. *Sci*. *Technol*., **41**: 7 126-7 130, http://dx.doi.org/10.1021/es071002n.

User Manual for BioWin v3.0. 2008. EnviroSim Associates Ltd., Hamilton, Canada. EnviroSim Associates Ltd.

Vanrolleghem P A, Insel G, Petersen B, Sin G, De Paw D, Nopens I, Dovermann H, Weijers S, Gernaey K. 2003. A comprehensive model calibration procedure for activated sludge models. *In*: WEFTEC 2003: 1-28. Water Environment Federation, Alexandria, VA, USA. p.210-237.

Wang D B, Li X M, Yang Q, Zeng G M, Liao D X, Zhang J. 2008. Biological phosphorus removal in sequencing batch reactor with single-stage oxic process. *Bioresour*. *Technol*., **99**(13): 5 466-5 473, http://dx.doi.org/10.1016/j.biortech.2007.11.007

Wang D B, Li X M, Yang Q, Zheng W, Wu Y, Zeng T J, Zeng G M. 2012. Improved biological phosphorus removal performance driven by the aerobic/extended-idle regime with propionate as the sole carbon source. *Water Res*., **46**(12): 3 868-3 878, http://dx.doi.org/10.1016/j.watres.2012.04.036.

Wang D B, Xu Q X, Yang W Q, Chen H B, Li X M, Liao D X, Yang G J, Yang Q, Zeng G M. 2014. A new configuration of sequencing batch reactor operated as a modified aerobic/extended-idle regime for simultaneously saving reactor volume and enhancing biological phosphorus removal. *Biochem*. *Eng*., **87**: 15-24, http://dx.doi.org/10.1016/j.bej.2014.03.009.

Wang D B, Zheng W, Liao D X, Li X M, Yang Q, Zeng G M. 2013. Effect of initial pH control on biological phosphorus removal induced by the aerobic/extended-idle regime. *Chemosphere*, **90**(8): 2 279-2 287, http://dx.doi.org/10.1016/j.chemosphere.2012.10.086.

Wouters-Wasiaka K, Héduit A, Audic J M, Lfevre F. 1994. Real-time control of nitrogen removal at full-scale using oxidation reduction potential. *Water Sci*. *Technol*., **30**(4): 207-210.

Yagci N, Insel G, Tasli R, Artan N, Randall C W, Orhon D. 2006. A new interpretation of ASM2d for modeling of SBR performance for enhanced biological phosphorus removal under different P/HAc ratios. *Biotechnol*. *Bioeng*., **93**(2): 258-270, http://dx.doi.org/10.1002/bit.20701.

Zhou X F, Zhang Y L, Hu M D, Shi W. 2009b. Optimization design for nutrients removal process with activated sludge 2D model. *CIESC J*., **60**(12): 3 122-3 129. (in Chinese with English abstract)