Desalination and Water Treatment
www.deswater.com

doi: 10.1080/19443994.2014.880375

54 (2015) 15–27
April

Taylor & Francis
Taylor & Francis Group

Development of an ASM1 dynamic simulation model for an activated sludge process in United Arab Emirates

W.E. Elshorbagy[a,*], M. Shawaqfah[b]

[a] *Civil and Environmental Engineering Department, UAE University, P.O. Box 17555, AlAin, United Arab Emirates*
Email: Walid.shorbagy@uaeu.ac.ae
[b] *Civil Engineering Department, Al al-bayt University, Mafraq, Jordan*

Received 9 December 2012; Accepted 5 December 2013

ABSTRACT

The current study develops an ASM1-based dynamic simulation model for the activated sludge process of a major sewage treatment plant located in the United Arab Emirates. As a first step towards the model calibration and validation, two campaigns of field measurements and lab-controlled experiments were conducted to estimate a number of kinetic and stoichiometric parameters required by the ASM1 model. The first campaign was during the winter season with average temperature of 25˚C and the second was during the summer season with average temperature of 39˚C. Each campaign involved collection of composite samples at every hour for five consecutive days and analyzed for several variables including COD, BOD_5, SS, VSS, TOC, TN, TKN, NH_4-N, NO_3-N, and NO_2-N. Parameters for reaction kinetics and stoichiometry were determined based on winter measurements using respirometry techniques recommended by different studies. The paper reports the values of these parameters in addition to results of the dynamic calibration of ASM1 conducted using GPS-X simulation environment. A steady-state simulation was first conducted to fit the modeled sludge production to the sludge production calculated from plant average data collected during the measurement campaign. The model validation was conducted considering summer measurements after modifying a number of temperature-dependent parameters. The results indicated reasonable agreement between the measured and simulated effluent variables of COD, TSS, and NH_4-N.

Keywords: ASM1; Parameter estimation; Model calibration

1. Introduction

Activated sludge systems (ASS) cover the majority of wastewater treatment plants (WWTPs) all over the world with mounting complexity in the processes' operation over the last decade due to requirements of removing nitrogenous and phosphorus compounds simultaneously with carbonaceous ones. Dynamic simulation of WWTPs and in particular ASS has become a necessary tool for controlling the effluent quality through better operation, studying different aspects in the design phases, and planning and exploring different upgrading options. The modeling of activated sludge process provides a powerful tool to design, operate and predict the upcoming performance of the activated sludge process [1–3].

---

*Corresponding author.

The activated sludge models (ASMs) [4] developed by the International Water Association (IWA, formally IAWQ) have recently been used in sizing and modeling WWTPs. Sizing of new plants is carried out by minimizing the capital and operational costs of the plant adopting different AS processes simulated by different ASMs. Examples of such application can be found in a number of recent studies [5–7]. Modeling of WWTPs is carried out to simulate the static as well as dynamic performance of wastewater biological treatment processes. Careful calibration of these models is a vital component to achieve the above targets and is reached through accurate estimation of different stoichiometric and kinetic model parameters requiring intensive field measurements and lab-controlled experiments. Several studies have adapted these models in dynamic simulation of WWTPs using different calibration approaches [8–11]. The four cited protocols have been recently synthesized by an IWA task group into a new scientific and technical report [12]. ASM1 is the first and most comprehensive version of ASM models used in simulating the carbon and nitrogen removal processes taking place along the various reactors of the ASS from wastewater [13]. The mass balance equations of ASM1 and their associated kinetic and stoichiometric parameters can be found in several references [4,5,13].

The current study developed an ASM1-based dynamic simulation model for the activated sludge process of a major WWTP in United Arab Emirates. The considered WWTP is located in AlAin City that is 160 km east of Abu Dhabi; the capital city of UAE. The collected wastewater is mostly domestic originated from the population equivalent of about 276,000 (based on 244 L/h d). The plant was designed originally to treat an average daily flow of 54,000 m^3/d and a load of 13,800 kg BOD/d. Based on a design flow peaking factor of 2.5, the plant peak instantaneous flow is 135,000 m^3/d. As the plant has been operating over its maximum capacity in the last few years, the quality of its effluent occasionally deteriorates and violates the international standards. Therefore, a number of remedial actions have been taken, among which a recent upgrade project has been scheduled to operate in the first quarter of 2013. The plant employs physical, biological, and chemical means to treat wastewater utilizing the following processes: preliminary treatment (screening and grit removal), suspended growth biological process (extended aeration followed by settling), and tertiary treatment (sand filtration and chemical addition). Developing a dynamic simulation model for the plant can significantly help in controlling the effluent quality and in planning optimal design and operation strategies for the upgrade project. This work focuses on dynamic modeling of the physical and biological processes involved in the considered WWTP.

Several calibration approaches have been adapted in earlier studies to model the dynamic behavior of ASS using ASM models. In general, a calibration process of this type involves major sampling programs, chemical and biological analyses, and laboratory experiments employing different methods to determine large number of stoichiometric and kinetic parameters needed by these models. This is usually regarded lengthy and expensive work requiring great deal of care and precision to produce accurate and representative outputs. Such outcomes may be misleading due to the many uncertainties associated with the measurements, analyses, experimental work, and above all the modeling approach. The Dutch foundation of applied water research (STOWA) has stimulated the development of a protocol assisting the setup and calibration of models for full-scale wastewater treatment plants [8]. The objective was to reflect the present experience and knowledge in a standardized and structured way so that the ASM simulation models can be easily applied while quality control is maintained. Another recent study [14] provides guidelines for the modelers to get them acquainted with the usually changed biokinetic model parameters, the usual ranges for these changes, and what values are typically used for different ASMs.

A key step in developing well-calibrated ASM models is identification of stoichiometric and kinetic parameters that result in good representation of the process's biochemical behavior. Several approaches determine these parameters using lab-controlled and respirometry tests [15–21]. Other approaches identify these parameters from the numerical modeling considering iterative values of the parameters till the produced outputs (usually effluent composite variables) approach the counterpart measured values. This study combines both approaches by identifying a number of kinetic and stoichiometric parameters using respirometry tests then update them along with identification of other parameters using numerical simulation. Section 2.2 provides more details on the respirometry-based methods considered to identify the sought kinetic and stoichiometric parameters.

2. Methods

2.1. Field measurements

As a first step towards the model calibration and validation, two campaigns of field measurements were conducted to gather quantity and quality data about the influent and effluent of WWTP in addition to other information needed to estimate kinetic and stoichiometric parameters required by the ASM1 model. The first campaign was during the winter season (9–14 January 2010) with average temperature of 25˚C and the second was during the summer season (20–24 June 2010) with average temperature of 39˚C. Each campaign involved composite samples, with fixed incremental volumes, collected at every hour for five consecutive days and analyzed for several variables including COD, BOD_5, SS, VSS, TOC, TN, TKN, NH_4-N, NO_3-N, and NO_2-N. Fig. 1 shows a schematic layout of the major components of the considered WWTP with the locations of sampling points (circled letters and numbers). Fig. 1 shows the main components of the plant including grit chamber, aeration tanks, and final settlers. No primary settlers exist in the plant. Winter data and parameters were used to calibrate the developed model while the summer data were used in model validation.

2.2. Parameter estimation by lab-controlled respirometry tests

Six stoichiometric and kinetic parameters needed by ASM1 were estimated using respirometry techniques and based on winter measurement campaign. Description of these parameters and the methods used in their estimation are given below. These selected parameters were among 10 parameters reported in earlier studies [22] as the most influential parameters on ASM1 simulation outputs. Other parameters suggested by these studies and not estimated by respirometry tests (due to limited time and site constraints) were identified via the dynamic calibration. These include autotrophic yield coefficient (Y_A), decay coefficient for autotrophic biomass (b_A), maximum specific hydrolysis rate (K_h), and oxygen saturation coefficient for autotrophic biomass (K_{OA}). All respirometry tests, described below, were undertaken right after collection and inside the WWTP laboratory that had a room temperature identical to the treated wastewater temperateure in the units of 25˚C.

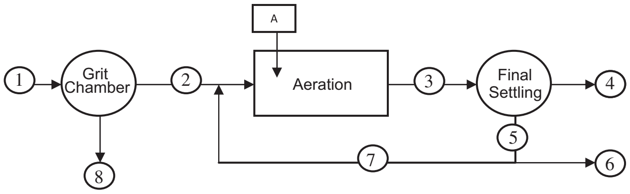

Fig. 1. Layout of the considered WWTP and sampling locations.
Notes: (Description of streams: St. 1: influent of the plant, St. 2: influent to AS, St. 3: effluent of Aeration tank and influent of the final settling, St. 4: effluent of AS and plant, St. 5: sludge underflow of final settling, St. 6: Sludge waste from the plant, St. 7: sludge recycle to AS, and St. 8: sludge removed from the grit chamber, A: air/oxygen supply).

2.2.1. Heterotrophic yield coefficient (Y_H)

The heterotrophic yield coefficient, Y_H, is a proportionality factor that links the maximum specific growth rate and the maximum readily biodegradable substrate utilization rate [13]. The true growth yield used in the ASM No. 1 is defined as the ratio of the rate of cell growth in the absence of maintenance energy requirements. The typical values of Y_H are within the range 0.38–0.75 mg cell COD synthetized/mg COD removed [11] with the default value equal 0.67 mg cell COD/mg COD used in the ASM No. 1.

The method used to identify Y_H is based on the procedure described by Sollfrank and Gujer [17] and summarized as follows: (1) 0.3 L of the mixed liquor and 0.7 L of the filtered primary effluent are added to the beaker, (2) the nitrification inhibitor (ATU) is added, (3) mixing and aerating (DO = 6–8 mg/L), (4) samples for soluble COD and VSS tests are taken at the beginning and at the end of the test, and (5) OURs (oxygen uptake rates) are measured every 5–7 min until it stabilizes at a lower level (related to hydrolysis and endogenous respiration).

Y_H was determined from the equation:

$$Y_H = \frac{\mathrm{COD_{degrad.}} - \int_{t_o}^{t} \mathrm{OUR_{net}} \times V.dt}{\mathrm{COD_{degrad.}}} \tag{1}$$

$\mathrm{COD_{degrad.}}$ = COD (spiked sample as prepared in step 1 and measured at the beginning of the test) − $\mathrm{COD_{inert}}$ (measured in the secondary clarifier).

2.2.2. Decay coefficient for heterotrophic biomass (b_H)

Decay is a composite term which accounts for a reduction in yield caused by predation, cell lysis, and the need for maintenance energy. It is commonly assumed that decay can be expressed by a first-order equation with respect to the concentration of active heterotrophic biomass.

The method used to identify b_H is based on the procedure described by Henze et al. [13] and summarized as follows: (1) one L of the mixed liquor (withdrawn from the reactor outlet) is added to the beaker, (2) the nitrification inhibitor (ATU) is added, (3) mixing and aerating (DO = 6–8 mg/L), and (4) OUR is measured every day at the same time over a period of several days.

Values of the decay coefficient for heterotrophic biomass were calculated from equation:

$$b_H = \frac{b'_H}{1 - Y_H(1 - f_p)} \tag{2}$$

The "traditional" decay coefficient, b'_H, was determined as the slope of the plot Ln (OUR/OUR$_0$) vs. time in the batch test and assuming f_p (non-biodegradable fraction of biomass) of 0.2.

2.2.3. Maximum specific growth rate for heterotrophic biomass (μ_{H-max})

The maximum specific growth rate defines the rate of cell growth in terms of the concentration of cell present, i.e. the mass of viable cells formed per unit mass of viable cells present. The typical values of the maximum specific growth rate for heterotrophic biomass, μ_{H-max}, were within the range 0.6–13.2 d^{-1} [13] with the default value of 6 d^{-1} at 20˚C used in ASM1.

The method proposed by [18] was considered here to estimate this coefficient by measuring the OURs in a batch reactor containing initially a very high S_0/X_0 ratio (substrate to biomass ratio). The test involves adding 50 ml of the mixed liquor + 0.95 L of the filtered (using the Whatman GF/C filter) primary effluent to the beaker, adding the nitrification inhibitor (ATU), mixing and aerating to maximize the DO level, and measuring the OUR with time till DO diminishes. During the first period of the test, OURs increase exponentially until the point when the growth is starting to be limited by low concentrations of readily biodegradable substrate. Consequently, the OURs decrease to level where the growth is dominated by substrate released in the hydrolysis process. After the acclimation period of approximately 1 h, the heterotrophs in the batch reactor start unlimited growth due to a very high S_0/X_0 ratio. The oxygen uptake rates increase exponentially such that Ln (OUR$_H$ (t)/OUR$_H$ (0)) vs. time had a linear slope equal to the maximum net growth rate of heterotrophic biomass (i.e. μ_{H-max}, $-b_H$). OUR$_H$ (0) is the initial oxygen uptake rate by the heterotrophs at the beginning of the experiment.

2.2.4. Substrate half-saturation coefficient (K_S)

The substrate half-saturation coefficient, K_S, is equivalent to the "Michaelis–Menten constant" in the enzyme kinetics equation. The K_S coefficient determines how rapidly μ_H approaches μ_{H-max} in the Monod equation in terms of the substrate concentration, and is defined as the substrate concentration at which μ_H is equal to the half of μ_{H-max}. The oxygen uptake rate by heterotrophic biomass (r_x) can be expressed by the Monod type equation:

$$r_x = r_{x,\,max} \frac{S_s}{K_s + S_s} \tag{3}$$

where S_s is the readily biodegradable substrate (g COD/m^3). The procedure followed to determine K_s is based on Cech et al. [20] and summarized as follows. A volume of about 1 L of the mixed liquor is aerated to remove any readily biodegradable substrate originating from the primary effluent and mixed with the secondary (or plant) effluent to get the MLVSS concentration equal to approx 500 mg/L. Here, a 750 mL of MLSS (Ss = 1.4 mg/L, VSS = 2011 mg/L) was mixed with 2,250 mL of primary effluent (Ss = 0.2 mg/L, VSS = 9.0 mg/L) producing a mix whose Ss = 5.5 and VSS = 510 mg/L. Few drops of the nitrification inhibitor (ATU) were added to the mix to inhibit nitrification. Take filtered sample for the COD analysis. About 0.5 L of the filtered primary effluent was prepared and six samples of the diluted mixed liquor of equal volume (e.g. 500 mL) were prepared and OUR was measured for one sample. The mixed liquor was left to settle in the other vessels and different volumes (e.g. 20, 40, 70, 100, and 150 mL) of the supernatant were removed from each vessel.

Sequentially, each sample was aerated after adding the volumes of the primary fluent equal to the removed volumes of the supernatant so that the MLVSS concentrations of all samples were equal (500 mg/L). Each time, samples were taken for the OUR test immediately after dosing the filleted primary effluent.

The substrate half-saturation coefficient for heterotrophic biomass, K_s, estimates using the linearized form of the Monod equation (plot r_x vs. r_x/S_{NH}).

$$r_x = r_{x,m} - K_s \frac{r_x}{S_s} \tag{4}$$

2.2.5. Maximum specific growth rate for autotrophic biomass (μ_{A-max})

The maximum specific growth rate for autotrophic biomass, μ_{A-max}, is defined in analogous way as μ_{H-max} for heterotrophic biomass. Based on the procedure described by Novak et al. [19], μ_{A-max} coefficient is determined from the equation:

$$\mu_{A-\,max} = \frac{OUR_{A-\,max}}{f_{XPA}} \cdot \frac{Y_A}{4.57 - Y_A} \tag{5}$$

where Y_A is the autotrophic yield (g VSS/g N), OUR$_{A-max}$ is 4.33 × AUR$_{A-max}$, and AUR$_{A-max}$ is the ammonia uptake rate measured as increase in the nitrate concentration.

2.2.6. Ammonia half-saturation coefficient for autotrophic biomass (K_{NH})

The ammonia half-saturation coefficient, K_{NH}, for autotrophic biomass is equivalent to the substrate half-saturation coefficient for heterotrophic biomass. The typical values were within the range 0.6–3.6 mg/L [13]. K values were found to range from 0.06 to 5.6 mg/L [23]. The default values used in ASM No.1 and ASM No.2 were identical and equal 1.0 mg/L [13,24]. Since ammonia may be considered a single substrate of known concentration, the respirometric method of estimation [20,24] is directly applied to determine the value of K_{NH}.

In this method, linearized forms of the Monod equation are developed for three batch tests as follows:

$$\frac{1}{r_x} = \frac{K_{NH}}{r_{x,\max}}\frac{1}{S_{NH}} + \frac{1}{r_{x,\max}} \tag{6}$$

$$\frac{S_{NH}}{r_x} = \frac{K_{NH}}{r_{x,\max}} + \frac{S_{NH}}{r_{x,\max}} \tag{7}$$

$$r_x = r_{x,\max} - \frac{r_x}{S_{NH}}K_{NH} \tag{8}$$

where r_x is actual reaction rate ($ML^{-3}T^{-1}$) and $r_{x,\max}$ is maximum reaction rate ($ML^{-3}T^{-1}$).

The procedures of the test are summarized as follows: A solution of NH_4Cl whose concentration is 1,000 mg N/L was prepared. The plant effluent was added to the mixed liquor taken from the outlet of the aeration basin to get the MLVSS concentration equal to approx. 500 mg/L. Five to six samples of the diluted mixed liquor of the equal volume (e.g 500 mL) were prepared. The nitrification inhibitor was added to one sample, aerated, and a sample was taken for the OUR test to estimate heterotrophic endogenous respiration. A sample was also taken for the VSS and $N–NH_4$ analysis. Sequentially, different doses of the NH_4Cl solution (e.g. 0.2, 0.5, 0.7, 1.0, and 1.5 mg N/L) were added to the next samples and aerated. Samples were taken for the OUR test immediately after dosing the NH_4Cl solution. Three batch tests were performed to estimate the K_{NH} value and each time all linearized forms of the Monod equation mentioned above were considered.

2.3. Simulation environment (GPS-X software)

A number of commercial simulation environments have been built using the ASM model components, notably GPS-X by Hydromantis, WEST by Hemmis and Biowin by Envirosim; the latter includes elemental mass balances. These simulation environments provide a useful interface between the model and the operator or process engineer.

The basis of the GPS-X simulator is a material balance over each of the state variables in the ASM model over each of the process units, taking into account the flow rates in and out of the process unit as well as the generation or consumption rate specified. The state variables are predominantly COD, oxygen, and nutrient fractions. The GPS-X simulator allows the entry of influent COD, nitrogen, phosphorus, and solids fractions in a number of ways. The simulator Influent Advisor spreadsheet demonstrates the links between user input values and state and composite variables. The choice of library is fairly simple: the CN library contains only COD, oxygen and nitrogen fractions; the CNP library contains phosphorus fractions as well. The exact fractions included in any model are dependent on the ASM model, library and influent model chosen. An advanced nitrogen library also exists, called the C2 N library. This library includes fractions for nitrogen associated with inert fractions, as well as nitrite–nitrogen, which is associated with process models that model nitrification as a two-step process. The IP libraries add industrial pollutant fractions to either the CN or CNP libraries. The industrial pollutants are user-defined fractions. The model calculates the composite variables from the state variables using certain ratios, so-called "stoichiometric constant". The default numerical solver integration method is the Runge–Kutta–Felberg method.

2.4. Final settling simulation

The final settling tank in this study considered two models; one for clarification and one for thickening. Clarification was simulated according to Voutchkov Model [25], where the effluent suspended solids concentration (SS_e) is calculated from the equation:

$$SS_e = \frac{6.21\ln(\text{MLSS} \times \text{SVI})}{0.67\ln(H) - (\text{SR})} - 26.43 \tag{9}$$

where SVI is the Sludge Volume Index in (mL/g), H is the side water depth in the settling tank (m), and SR is the surface overflow rate (m/h). The thickening function is modeled using the deferential thickening technique which is based on the limiting flux theory [26]. This technique proposes that the thickened sludge concentration (SS_5) equals:

$$SS_5(\mathrm{g/L}) = [k(n-1)]^{1/n}\left(\frac{n}{n-1}\right)\left(\frac{A_p}{Q_4}\right)^{1/n} \quad (10)$$

where k and n are settling constants determined from the calibration as 360 m/d and 2.2, respectively.

2.5. Calibration and validation approaches

In the steady-state calibration, the procedure suggested by [27] was followed. The data from the wastewater treatment plant influent from the winter measuring campaign were averaged and used as input to the model. The different flows in the model (Q_i, Q_r, and Q_w) were also adjusted according to data from the measuring campaign during this time period. Such data were obtained from a number of flow meters installed in the plant to record different flows in continuous manner. The settler parameters (k and n) were changed to adjust the output of the model to the measured data of COD and TSS. Also the fractions for calculating the amount of sludge collected in the tanks were adjusted to get the right amount of generated sludge. Upon adjusting the settler parameters, X_I and X_S fractions were changed in the wastewater influent to make a better fit of the model. This change adjusts the outgoing NH_4 concentration. The final step of the steady-state calibration was to adjust b_H, which also affects the outgoing NH_4 concentration. Validation of the calibrated model was then conducted using summer measurements and upon adjusting a number of calibrated parameters known to be temperature dependent, namely $\mu_{H-\max}$, $\mu_{A-\max}$, b_H, and b_A. Such adjustment was implemented internally in the used software to account for the summer temperature using Arrhenius equation:

$$K_{ST} = K_{WT} \times \theta^{(ST-WT)} \quad (11)$$

where K_{ST} and K_{WT} are the kinetic parameters associated with summer and winter temperatures, which are 39˚C and 25˚C, respectively. θ is the temperature coefficient and was assigned a default value of 1.07.

3. Results

3.1. Results of estimated parameters

Values of parameters estimated from the laboratory methods described earlier were initially used as guideline in the simulations and were finalized through comprehensive dynamic calibration considering the actual effluent field winter observations to be presented later.

3.1.1. Results of Y_H estimation

Results of the procedures given earlier to determine Y_H are listed below:

$$\begin{aligned}\mathrm{COD_{spiked}} &= 0.3 \times \mathrm{COD_{MLSS}} + 0.7 \\ &\quad \times \mathrm{COD_{filtered\ primary\ clarifier\ effluent}} \\ &= 0.3 \times 141 + 0.7 \times 185 \\ &= 141.8\mathrm{mg/L} \quad \mathrm{COD_{degraded}} \\ &= 141.8 - 112.2 = 29.6\mathrm{mg/L}\end{aligned}$$

Averages and standard deviations (SD) of DO measurements based on three replicates along with OUR values needed for Y_H estimation are listed in Table 1.

Based on the above records, Y_H was found 0.75 mg cell COD/mg substrate COD.

3.1.2. Results of b_H estimation

Based on the method described in section 2.2 to determine b_H, Avg. and SD of DO observations with time (for three replicates) and corresponding OUR values with time are reported in Table 2.

The slope of Ln (OUR/OUR_0) vs. time relation (b'_H) is found 0.0304 h^{-1} (0.73 d^{-1}), and based on equation (2) this corresponds to b_H of 1.82 d^{-1}.

Table 1
DO averages and standard deviations and OURs for Y_H determination

Time (min)	0	5	10	15	20	25	30	35	40	45
Avg. DO (mg/l)	7.89	6.70	4.20	2.32	1.92	1.42	1.15	0.53	0.03	0.02
Standard deviation of DO (mg/L)	0.68	0.54	0.41	0.32	0.25	0.11	0.16	0.08	0.00	0.00
OUR (mg/L/min)		0.24	0.50	0.38	0.08	0.10	0.05	0.12	0.10	0.00
$OUR_{avg.}$ (mg/L/min)			0.37	0.44	0.23	0.09	0.08	0.09	0.11	0.05
OUR × dt (mg/L)			1.85	2.19	1.14	0.45	0.39	0.45	0.56	0.26
Total (OUR × dt)										7.27

Table 2
Results of respirometric test to determine b_H

Time (hrs)	0	0.16667	2	6	24	48	72	96	120	144
Time (min)	0	10	120	360	1,440	2,880	4,320	5,760	7,200	8,640
Avg. DO (mg/L)	8.13	8.07	7.6	6.33	3.7	2.11	0.93	0.28	0.13	0.02
Standard deviation DO (mg/L)	1.23	1.01	0.93	0.84	0.67	0.56	0.23	0.09	0.06	0.02
OUR (mg/L/min)		0.0056	0.0043	0.0053	0.0024	0.0011	0.0008	0.0005	0.0001	0.0001
OUR/OUR_0		1.000	0.763	0.945	0.435	0.197	0.146	0.081	0.019	0.014
Ln (OUR/OUR_0)		0.00	−0.27	−0.06	−0.83	−1.62	−1.92	−2.52	−3.98	−4.29

3.1.3. Results of $\mu_{H-\max}$ estimation

Table 3 lists the measured OUR values and Fig. 2 reports the plot of Ln (OUR/OUR_0) vs. time. The increasing portion of the plot is fitted to the linear equation $(0.335t - 0.278)$ with t in hours and a slope of $0.335\,h^{-1}$ corresponding to $8.04\,d^{-1}$. Based on the estimated b_H of $1.82\,d^{-1}$, $\mu_{H-\max}$ is estimated at $6.22\,d^{-1}$.

3.1.4. Results of K_s estimation

The results of r_x and S_s for the six-tested mixes are shown in Table 4 and plotted in Fig. 3.

K_s was found from the best-fit equation as 6.42 mg/L with a correlation factor (R^2) of 0.82.

3.1.5. Results of $\mu_{A-\max}$ estimation

The fraction of autotrophs in the biomass (f_{XPA}) is estimated from the steady-state ASM No. 1 as 0.039 (using the default value for $\mu_{A-\max}$). The laboratory experiment for the tested samples resulted in $AUR_{A-\max}$ of 0.068 mg N/mg VSS. Based on a default value of 0.24 mg VSS/mg N for the autotrophic yield coefficient (Y_A) [19], $\mu_{A-\max}$ is estimated at 0.42 mg VSS/mg N d^{-1}.

3.1.6. Results of K_{NH} estimation

The measurements of K_{NH} test indicated that the best-fitted equation was associated with Eq. (7). Such measurements of ammonium substrate (S_{NH}) and the corresponding actual substrate removal rates (r_x) are shown in Fig. 4. The best-fitted linear relation reports a maximum substrate removal rate $(r_{x,\max})$ of $17.42\,h^{-1}$ and K_{NH} of 0.733 mg/L.

The tested batches reflected a best estimate of K_{NH} of 0.11 mg/L based on the second Monod form with $R^2 = 0.811$.

A summary of parameters estimated from the respirometry tests are: $Y_H = 0.75$, $b_H = 1.82\,d^{-1}$, $\mu_{H-\max} = 6.22\,d^{-1}$, $K_s = 6.42$ mg/L, $\mu_{A-\max} = 0.42\,d^{-1}$, and $K_{NH} =$ 0,733 mg/L.

3.2. Influent characterization

Besides the conventional characterization of the influent to identify regular composite variables, a detailed characterization was performed to identify 13 state variables required by ASM1 model. Some statistical results of observed data are shown in Table 5.

It is noted that the concentrations of different types of biomass in the influent are negligible compared to the amount formed within the process and thus were all ignored in this model.

Inspecting the measurements reported in Table 5, one can observe a number of aspects related to the influent quality. First, the organic nitrogen (TKN–NH_4-N) is too low (around 3 mg/L) while the nitrates/nitrites (TN–TKN) are too high (around of 26.5 mg/L). This can be attributed to the nature of wastewater generated in this area and to the significant aeration and oxidation the wastewater undergoes due to the extended time it spends in long routes including the significant pumping, pretreatment

Table 3
Measurements of the maximum specific growth rate for heterotrophic biomass $(\mu_{H-\max})$

Time (min)	0	1	10	30	60	120	180	240	300	360
Avg. DO (Mg O_2/L)	8.64	8.5	8.17	7.76	7.445	6.91	5.35	4.12	0.510	0.02
Standard deviation DO (mg/L)	0.75	0.81	0.79	0.75	0.72	0.74	0.65	0.52	0.08	0.01
OUR (Mg O_2/L × d)		201.6	201.6	201.6	208.8	285.12	489.6	532.8	244.8	230.4
Ln $[OUR/OUR_0]$		0.00	0.00	0.00	0.04	0.35	0.89	0.97	0.19	0.13

Table 4
Reported measurements for the K_s determination test

	Mix 1	Mix 2	Mix 3	Mix 4	Mix 5	Mix 6
Vol. primary effluent (L)	0	20	40	70	100	150
Vol. Mix (L)	500	480	460	430	400	350
S_s (mg/L)	5.5293	11.7482	17.9670	27.2952	36.6235	52.1705
r_x (hr^{-1})	0.0293	0.0351	0.0410	0.0459	0.0527	0.0585
r_x/S_s (L/mg/hr)	0.0053	0.0030	0.0023	0.0017	0.0014	0.0011

processes, and grit removal process. Second, X_I is extremely low with respect to COD_{tot} (X_I is about 1% of COD_{tot} only). This is much below the typical ranges of 10–15% indicating that most of the particulate organic matter introduced to the aeration tank is biodegradable and may be attributed to either the nature of generated wastewater itself or to the high efficiency of grit chamber in removing the inert particulates. The small ratio of COD to BOD_5 (1.58) compared to the typical ratio of 2.1 identified in literature [Metcalf] indicates the low content of inert matter in the influent that can also explain the low X_I content as well. Third, the ratio of $COD_{biodegradable}$ ($S_s + X_s =$ 558.5 mgCOD/L) to BOD_5 is 1.52 that is reasonably near the typical ratio of 1.71 given in the literature. However, the ratio of Ss/COD biodegradable = 0.31 and noticeably smaller than the typical ratio (0.43) available in the literature [28].

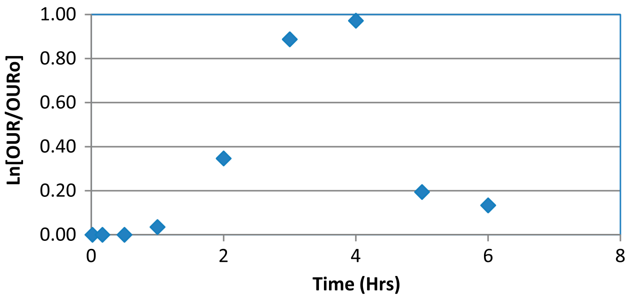

Fig. 2. Logarithmic oxygen uptake rates to determine μ_{H-max}.

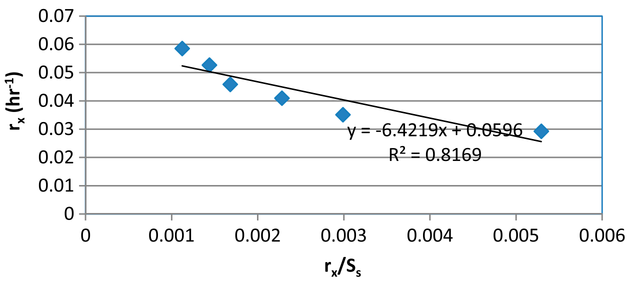

Fig. 3. Measurements of r_x vs. r_x/S_s for estimating K_s.

Some operational parameters of the treatment plant which were entered to the GPS-X software as operational data are given in Table 6.

The model is based on COD fractions and on several stoichiometric coefficients (such as VSS/TSS ratio, soluble fraction of total COD, etc.) which need to be determined to achieve better wastewater characterization. For this purpose, the Influent Advisor developed by Hydromantis Co. was used by inserting the average concentrations. All other data were calculated by the advisor based on material balance.

Initially, the parameters indicated in Table 7 were inserted in the Influent Advisor. The default values were then modified using the influent advisor according to the average concentrations of influent total COD and total suspended solids, which were obtained from the treatment plant. Table 8 lists the default and modified values of the stoichiometric coefficients.

By doing this modification, the concentrations presented in Table 9 are obtained. The concentrations of total COD and total suspended solids obtained using the Influent Advisor were 580.5 g O_2/m^3 and 187.59 g/m^3, respectively; as can be seen extremely close to the actual data from the plant.

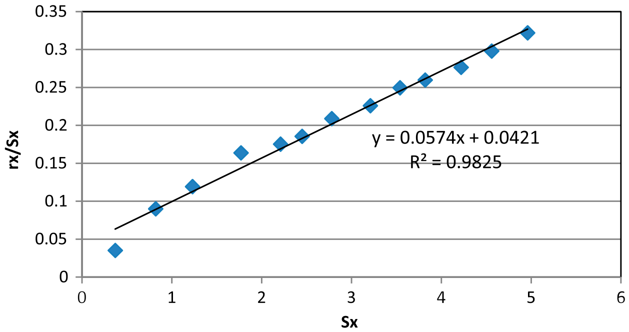

Fig. 4. Fitted r_x/S_x vs. ammonium substrate (S_x) for K_{NH} estimation.

Table 5
Statistical results of measured parameters (all in mg/L) for influent wastewater

	SS	NH_4-N	COD_{tot}	COD_{sol}	TN	TKN	S_s	X_s	S_I	X_I
Mean	187.83	26.55	580.48	192.57	56.04	29.49	175.86	382.64	16.71	5.30
Standard deviation	127.66	3.96	220.39	58.66	5.95	5.33	57.96	244.54	4.28	3.83

Table 6
WWTP operational parameters

Daily mean flow rate	93,165 m^3/d
Sludge age	5.24 d
Waste sludge flow rate	5,904 m^3/d
Recycle flow rate	61,743 m^3/d

3.3. Sludge composition

The average results of the sludge composition analysis, based on ten measurements of the COD, SS, and VSS content of the activated sludge and recycle sludge, are given in Table 10.

3.4. Steady-state calibration

A simple steady-state WWTP configuration was constructed in GPS-X consisting of one aeration tank ($V = 5{,}832\ m^3$), two secondary sedimentation tanks, an internal recycle line, and a constant average sludge waste flow from the recycle line.

The main purpose of the steady-state model calibration is to fit the modeled sludge production to the sludge production calculated from plant data collected during the measuring winter campaign (based on waste flow data and sludge concentration measurements). This is done by adjusting parameters responsible for long-term behavior, i.e. the decay rates b_H and b_A, together with the influent concentration of X_I. The experimental value of b_H determined based on the lab-scale test was 1.82 d^{-1} and was used as a guideline for the model calibration. The influent for the steady-state model was obtained by averaging the dynamic influent data. Initially, these averages were calculated using only the wet weather flow data (influent data from 9 to 10 January 2010 at 8.00 am until the end of the measuring campaign, and with the wastewater characterization described earlier).

The final calibrated value of b_H was 0.62 d^{-1}. Furthermore, initial adjustments of the maximum specific growth rates, $\mu_{-\max H} = 8\ d^{-1}$ and $\mu_{-\max A} = 0.8\ d^{-1}$ were carried out during the steady-state calibration. On the other hand, it is stressed that final values of these parameters were determined in the dynamic model calibration. It is also worth mentioning that b_A had minimal impact on the sludge production so its adjustment in the dynamic calibration was accepted. Table 11 shows the results of the steady-state calibration.

For the 5-day sampling campaign, a total sludge production of 224942.0 kg SS was calculated, with an average waste flow rate of 5,904 m^3/d. The sludge production of 221,035 kg COD resulted from the steady state simulations does closely agree with the sludge production measured at the WWTP.

3.5. Dynamic calibration and validation

The aeration tank was simulated using 2 CSTRs connected in series to represent a truly plug-flow reactor hydraulics based on the proposed equation developed by the Water Research Center in the United Kingdom to estimate the number of equivalent tanks in series:

Table 7
Influent wastewater characteristics inserted in the influent advisor

Composite measurements			
COD	Total COD	g COD/m^3	580.50
TKN	Total TKN	g N/m^3	28.31
Dissolved oxygen			
So	Dissolved oxygen	g O_2/m^3	0.00
Nitrogen compounds			
S_{NH}	Free and ionized ammonia	g N/m^3	26.5
S_{NO}	Nitrate and nitrite	g N/m^3	27.72
S_{NN}	Dinitrogen	g N/m^3	0.00
Alkalinity			
S_{alk}	Alkalinity	$mole/m^3$	7.20

Table 8
Stoichiometric coefficients given in the influent advisor

		Units	Default value	Modified value
Stoichiometric coefficients				
i_{vt}	VSS/TSS ratio	g VSS/ g TSS	0.60	0.94
fr_{scod}	Soluble fraction of total COD	–	0.35	0.33
fr_{si}	Inert fraction of soluble COD	–	0.35	0.09
fr_{xs}	Substrate fraction of particulate COD	–	0.75	0.99
fr_{xu}	Unbiodegradable fraction of particulate COD	–	0.00	0.00
fr_{xbh}	Heterotrophic biomass fraction of particulate COD	–	0.00	0.00
fr_{xba}	Autotrophic biomass fraction of particulate COD	–	0.00	0.00
fr_{xsto}	Stored fraction of particulate COD	–	0.00	0.00
fr_{snh}	Ammonium fraction of soluble TKN	–	0.90	0.90
i_{cv}	XCOD/VSS ratio	g COD/ g VSS	2.20	2.20
f_{BOD}	$BOD_5/BOD_{ultimate}$ ratio	–	0.66	0.66
Nutrient fractions				
i_{xbn}	N content of active biomass	g N/g COD	0.0860	0.0860
i_{xun}	N content of endogenous/inert mass	g N/g COD	0.0600	0.0600

$$N = 7.4 \times \frac{L \times Q(1 + R_R)}{W \times H} \tag{12}$$

where N is equivalent number of tanks in series, L is aeration tank length (m), Q is wastewater flow (m^3/s), R_R is RAS recycle ratio, W is aeration tank width (m), and H is water depth (m). By applying the above equation for the present case study, we got two tanks in series to present the existing WWTP.

Table 12 shows the parameters modified from their default values or estimated values using respirometry tests as a result of the dynamic calibration carried out with respect to COD_{tot}, TSS, and NH_4 in the secondary clarifier effluent based on 5-day winter

Table 9
Concentrations obtained using the influent advisor

Variable	Meaning	Units	Value
S_{COD}	Filtered COD	g COD/m^3	192.57
X_{COD}	Particulate COD	g COD/m^3	387.93
COD	Total COD	g COD/m^3	580.50
S_{BOD}	Filtered carbonaceous BOD_5	g O_2/m^3	116.29
X_{BOD}	Particulate carbonaceous BOD_5	g O_2/m^3	252.19
BOD	Total carbonaceous BOD_5	g O_2/m^3	368.49
S_{BODu}	Filtered ultimate carbonaceous BOD	g O_2/m^3	176.20
X_{BODu}	Particulate ultimate carbonaceous BOD	g O_2/m^3	382.11
BODu	Total ultimate carbonaceous BOD	g O_2/m^3	558.31
S_{TKN}	Filtered TKN	g N/m^3	25.56
X_{TKN}	Particulate TKN	g N/m^3	2.75
TKN	Total TKN	g N/m^3	28.31
TN	Total nitrogen	gN/m^3	56.00
Xiss	Total inorganic suspended solids	g/m^3	11.26
VSS	Volatile suspended solids	g/m^3	176.33
X	Total suspended solids	g/m^3	187.59

Table 11
Steady-state model calibration results

Parameter	COD_{tot} (eff.) (mg/L)	TSS (eff.) (mg/L)	NH_4-N (eff.) (mg/L)
Measurements	42.63	23.28	1.56
Simulation	44.40	21.60	1.55
MAE	0.042	0.072	0.006

Table 10
Analysis results on activated sludge and recycle sludge (average and 95% confidence interval, resulting from 9 measurements)

	SS (g/L)	VSS/SS	COD/SS	COD/VSS	TKN/COD
Activated sludge	2.83 ± 0.56	0.73 ± 0.07	1.12 ± 0.33	1.53 ± 0.4	0.05 ± 0.02
Recycle sludge	7.62 ± 1.31	0.75 ± 0.01	1.33 ± 0.3	1.77 ± 0.39	0.02 ± 0.01

Table 12
List of main modified parameters for dynamic calibration

Parameter	Default (20˚C)	Estimated from respirometry test at 25˚C	Estimated from dynamic calibration at 25˚C	Units
Y_H	0.67	0.75	0.75	mg VSS/mg COD
Y_A	0.24	NA	0.24	mg VSS/mg N
$\mu_{-\max H}$	6.0	6.22	8.0	d^{-1}
$\mu_{-\max A}$	0.8	0.42	0.31	d^{-1}
b_H	0.62	1.82	0.62	d^{-1}
b_A	0.15	NA	0.17	d^{-1}
K_{NH}	20	6.42	5 .0	g COD/m^3
K_S	1.0	0.73	1.0	g NH_3-N/m^3
K_h	3.0	NA	3.0	d^{-1}
K_{OA}	0.4	NA	0.4	g/m^3

measurements. The table also has four parameters not estimated by respirometry tests; Y_A, K_H, K_{OA}, and b_A. Y_A, K_H, and K_{OA} values were kept unchanged from their default values [4] while b_A was set at 0.17 d^{-1}; that is slightly higher than its default value. It is worth mentioning that the b_A is recommended to estimate together with μ_A [29] and increased to 0.19 d^{-1} at $T = 20$˚C to allow keeping the same μ_A value whatever the sludge age. Other study [14] recommended 0.17 d^{-1} which was considered here, as this was found to improve the model performance in terms of outputs agreement with the measurements. Other parameters were also kept at their default values [4] including saturation/inhibition coefficient for nitrate (K_{ON}) of 0.5 g NO_3-N/m^3, oxygn saturation/ inhibition coefficient for heterotrophic biomass (K_{OH}) of 0.2 g O_2/m^3, oxygen saturation coefficient for autotrophic biomass (K_{OA}) of 0.4 g O_2/m^3, and hydrolysis saturation constant (K_X) of 0.03 g slowly biodegradable COD/g cell COD.

Fig. 5 reports the effluent simulated vs. measured effluent variables of COD_{tot}, TSS, and NH_4. The measured variables reported on Fig. 5 are averages of three replicates analyzed for each sample. The variation coefficients (SD/average) of these records ranged from 5 to 34%. The averages of the three simulated variables closely matched those associated with the measured values for the 5-day duration. Also the upper and lower fluctuations of the simulated COD_{tot} closely agreed with the measured counterparts. However, less agreement is reported between the simulated and measured local fluctuations for both TSS and NH_4.

The model validation was conducted using the 5-day summer measurements campaign. Same parameters listed in Table 12 were used here in the summer validation except As mentioned earlier; four temperature-dependent parameters, $\mu_{H-\max}$, $\mu_{A-\max}$, b_H, and b_A, were adjusted inside the model as per the summer temperature and according to Arrhenius equation (see Section 2.5). As a result, this adjustment

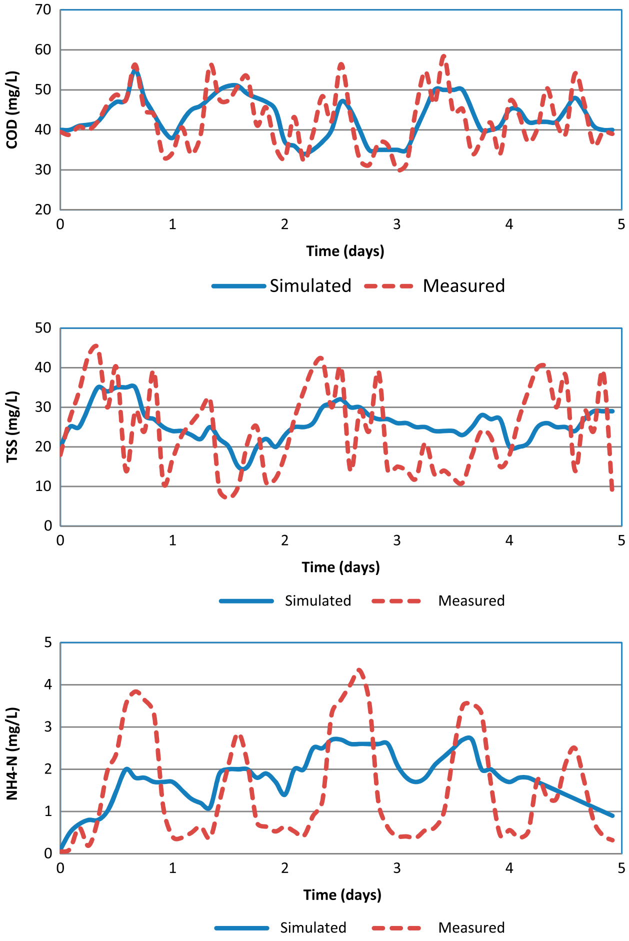

Fig. 5. Dynamic calibration results with respect to COD, TSS, and NH_4-N effluent, respectively.

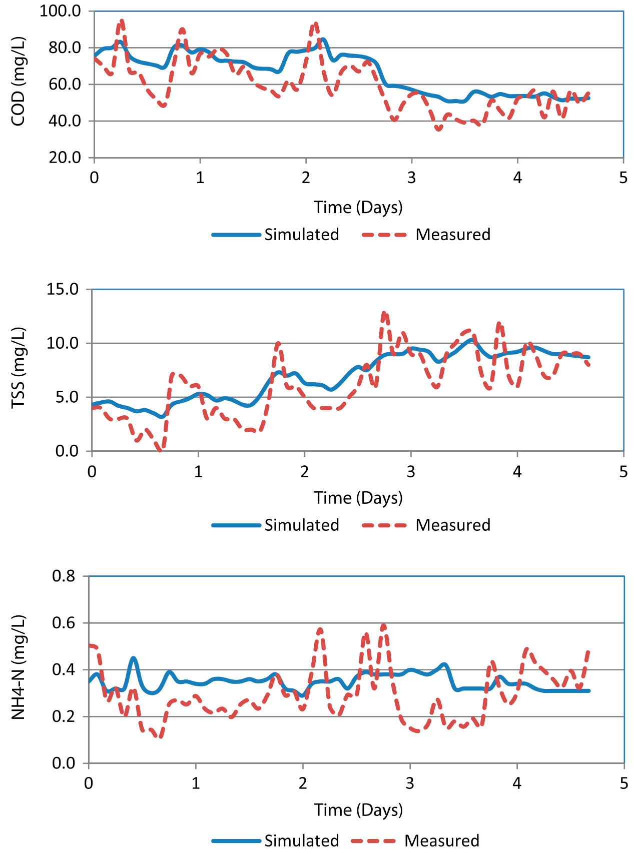

Fig. 6. Validation results with respect to COD, TSS, and NH_4-N effluent, respectively.

involved amplifiying these parameters by a factor of 2.57. Other than these parameters, same values listed in Table 12 and produced from the dynamic calibration were used in the validation.

Fig. 6 shows the temporal variation of the three effluent variables; COD, TSS, and NH_4-N measured in the summer campaign vs. their corresponding simulated counterparts. The shown plots indicate that dynamic behaviors of COD and TSS in the plant effluent are reasonably represented by the calibrated ASM1 model but less accuracy is observed in representing the behavior of NH_4-N in the effluent. The reported deviations may be attributed to uncertainties associated with the measured flows and suggest that more efforts and tuning of a number of parameters related to ammonium concentration in the effluent may still be needed, in particular b_A and K_{AO}.

4. Summary and conclusions

This paper presented the outputs of a dynamic simulation model developed for the activated sludge process of a major WWTP in United Arab Emirates. The model was developed using the GPS-X simulation environment based on the IWRA ASM1 model. The work involved intensive sampling campaigns in the winter and summer seasons. In each campaign, 12 composite samples were collected each day for five consecutive days. The samples were taken from the influent and effluent of the aeration tank and the final effluent following the final clarifier. The variables analyzed in the collected samples include COD, BOD_5, SS, VSS, TOC, TN, TKN, NH_4-N, NO_3-N, and NO_2-N. In addition, bulk volumes of samples were collected from the mixed liquor suspended solids and the final clarifier effluent to run lab-controlled respirometry tests in order to estimate the major kinetic and stoichiometric parameters required by the AM1 model.

Main findings of this study are summarized as follows:

- Parameters estimated using respirometry tests were: b_H = of 1.82 d^{-1}, Y_H of 0.75 mg cell COD synthetized/mg COD removed, μ_{H-max} = of 6.22 d^{-1}, K_s of 6.42 mg/L, μ_{A-max} of 0.42 d^{-1}, and K_{NH} of 0.73 mg/L.
- Other main parameters not estimated by respirometry tests were set at their default values considered for ASM1. This included Y_A of 0.24 mgVSS/mg N, K_h of 3.0 d^{-1}, and K_{OA} of 0.4 g O_2/m^3
- b_A was estimated from the dynamic calibration as 0.17 d^{-1}, that is slightly higher than its default value of 0.15 d^{-1}.
- Initial calibration was carried out for steady-state setting, verifying the sludge production from the 5-day winter sampling campaign against the actual measurements.
- Upon dynamic calibration, considering the winter measurements of effluent COD, TSS, and NH_4-N, further adjustment to all parameters estimated from respirometry tests (except Y_H) were made.
- Dynamic outputs from the calibrated model reflected good agreement between the simulated and measured COD_{tot} and TSS but less agreement in the case of NH_4.
- The calibrated model was validated against summer measurements after amplifying four temperature-dependent parameters (μ_{H-max}, μ_{A-max}, b_H, and b_A) using a linear scaling factor of 2.57.
- The validation again showed strong agreement for COD_{tot} and TSS and less agreement for NH_4-N.

- Considering the process complexity and the potential uncertainty in measurements, such results sound reasonable and promising for further use in process control.

References

[1] A. Nuhoglu, B. Keskinler, E. Yildiz, Mathematical modelling of the activated sludge process—The Erzincan case, Process Biochem. 40(7) (2005) 2467–2473.
[2] A. McHarg, Optimization of Municipal Wastewater Biological Nutrient Removal Using Computer Simulation, Ph.D. dissertation, Department of Chemical Engineering, University of Ottawa, Ottawa, Canada (2002).
[3] C. Fall, J. Loaiza-Navia, M. Esparza-Soto, Full activated sludge model no. 1 calibration experience at a medium-size WWTP in Mexico, Water Sci. Technol. 60 (12) (2009) 3069–3092.
[4] M. Henze, W. Gujer, T. Mino, M. Van Loosdrecht, Activated Sludge Models ASM1, ASM2, ASM2d and ASM3, Sc. and Tech. Report No. 9, IWA Publishing, London, 2000.
[5] W. Elshorbagy, A. Arwani, R.L. Droste, Optimal sizing of activated sludge process considering ASM3 model, Int. J. Civil Environ. Eng. 11(01) (2011a) 19–55 ISSN: 2077-1258.
[6] W. Elshorbagy, N. Nabil, R.L. Droste, Optimization of A^2O BNR processes using ASM and EAWAG Bio-P models: Model formulation, Water Qual. Res. J. Canada 46(1) (2011b) 13–27. doi:10.2166/wqrjc.2011.005.
[7] W. Elshorbagy, N. Nabil, R.L. Droste, Optimization of A^2O BNR processes using ASM and EAWAG Bio-P models: Model performance, Water Environ. Res. 85 (12) (2013) 2237–2250.
[8] J.W. Hulsbeek, J. Kruit, P.J. Roeleveld, M.C.M. van Loosdrecht, A practical protocol for dynamic modeling of activated sludge systems, Water Sci. Technol. 45(6) (2002) 127–136.
[9] P.A. Vanrolleghem, G. Insel, B. Petersen, G. Sin, D. De Pauw, I. Nopens, S. Weijers, K. Gernaey, A comprehensive model calibration procedure for activated sludge models, in: Proceedings 76th Annual Technical Exhibition & Conference (WEFTEC 2003), Los Angeles, CA, October 11–15, 2003.
[10] G. Langergraber, L. Rieger, S. Winkler, J. Alex, J. Wiese, C. Owerdieck, M. Ahnert, J. Simo, M. Maurer, A guideline for simulation studies of wastewater treatment plants, Water Sci. Technol. 50(7) (2004) 131–138.
[11] H. Melcer, P.L. Dold, R.M. Jones, C.M. Bye, I. Takacs, H.D. Stensel, A.W. Wilson, P. Sun, S. Bury, Methods for Wastewater Characterization in Activated Sludge Modeling, Water Environment Research Foundation (WERF), Alexandria, VA, 2003.
[12] L. Rieger, S. Gillot, G. Langergraber, T. Ohtsuki, A. Shaw, I. Takacs, S. Winkler, Guidelines for Using Activated Sludge Models, IWA Publishing, London, 2012, 312 p. ISBN: 9781843391746.
[13] M. Henze, C.P.L. Grady, W. Gujer, G.V.R. Marais, T. Matsuo, Activated Sludge Model No. 1. IAWQ Sc. and Tech. Report No. 1, IAWPRC, London, 1987.
[14] H. Hauduc, L. Rieger, T. Ohtsuki, A. Shaw, Activated sludge modelling: Development and potential use of a practical applications database, Water Sci. Technol. 63(10) (2011) 2164–2182.
[15] P.A. Vanrolleghem, H. Spanjers, B. Petersen, P. Ginestetf, I. Takacs, Estimating (combinations of) activated sludge model no. 1 parameters and components by respirometry, Water Sci. Technol. 39(1) (1999) 195–214.
[16] G.A. Ekama, P.L. Dold, G.V.R. Marais, Procedures for determining influent COD fractions and the maximum specific growth rate of heterotrophs in activated sludge systems, Water Sci. Technol. 18(6) (1986) 91–114.
[17] U. Sollfrank, W. Gujer, Characterization of domestic wastewater for mathematical modeling of the activated sludge process, Water Sci. Technol. 23 (1991) 1057–1066.
[18] J. Kappeler, W. Gujer, Estimation of kinetic parameters of heterotrophic biomass under aerobic conditions and characterization of wastewater for activated sludge modeling, Water Sci. Technol. 25(6) (1992) 125–139.
[19] L. Novák, L. Larrea, J. Wanner, Estimation of maximum specific growth rate of heterotrophic and autotrophic biomass: A combined technique of mathematical modeling and batch cultivations, Water Sci. Technol 30(11) (1994) 171–180.
[20] J.S. Cech, J. Chudoba, P. Grau, Determination of kinetic constants of activated sludge microorganisms, Water Sci. Technol. 17(2/3) (1985) 259–272.
[21] D. Orhon, G. Yildiz, E.U. Cokgor, S. Sozen, Respirometric evaluation of the biodegradability of confectionary wastewaters, Water Sci. Technol. 32(12) (1995) 11–19.
[22] M.A. Sedran, A.S. Mehrotra, A.B. Pincince. The dangers of uncalibrated activaed sludge simulation packages, in: Paper presented at the 2006 Water Environment Foundation Conference, Dallas, TX, 2006.
[23] J.B. Copp, P.L. Dold, Confirming the nitrate-to-oxygen conversion factor for denitrification, Water Res. 32 (1998) 1296–1304.
[24] M. Drtil, P. Nemeth, I. Bodik, Kinetic constants of nitrification, Water Res. 27 (1993) 35–39.
[25] N. Voutchkov, Relationship for clarification efficiency of circular secondary clarifiers, Water Sci. Technol. 26 (9) (1992) 2539–2542.
[26] C. Tang, E.D. Brill Jr, J. Pfeffer, Comprehensive model of activated sludge wastewater treatment system, J. Environ. Eng. 113(5) (1987) 952–969.
[27] B. Petersen, Calibration, identifiability and optimal experimental design of activated sludge models, PhD Thesis in Applied Biological Sciences, Gent University, Belgium (2000).
[28] C.P.L. Grady, G. Daigger, H. Lim, Biological Wastewater Treatment, 2nd ed., Marcel Dekker Inc., New York, NY, 1999.
[29] P.L. Dold, R.M. Jones, C.M. Bye, Importance and measurement of decay rate when assessing nitrification kinetics, Water Sci. Technol. 52(10–11) (2005) 469–477.