EVALUATION OF NUTRIENT REMOVAL PERFORMANCE FOR AN ORBAL PLANT USING THE ASM2d MODEL

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ABSTRACT

An Orbal plant achieving biological nutrient removal was modeled using nitrate, oxygen, ammonium-nitrogen and phosphate measurements for calibration. It was found that the oxygen and nitrate concentration in outer and middle ring is the bottleneck for modeling enhanced biological phosphorus removal (EBPR) whilst the system is under diurnal variation in the influent load. The simulation results showed that the overall system performances for total Kjeldahl-nitrogen and phosphate removal are around 77% and 94%, respectively. Also, the system is highly influenced by the operational changes rather than model kinetics. On the basis of this modeling exercise by using the ASM2d model, the necessary information for a trustable calibrated model that can be applied to upgrade the plant is discussed.

KEYWORDS

Modeling, Orbal, calibration, simultaneous nitrification-denitrification, EBPR

INTRODUCTION

In recent years, nutrient removal gained more importance in order to prevent eutrophication of receiving waters. As a solution, different plant configurations can be used for the design of wastewater treatment plants to achieve nutrient removal from domestic wastewaters. From an engineering point of view, the reliability of the treatment plant is important to meet nutrient discharge limits at all times. Extended aeration systems are associated with plants for small communities where reliability and simplicity of the operation are of prime importance (Grady *et al.*, 1999; Orhon and Artan, 1994; Randall *et al.*, 1992). The applicability of the system is good, because less sludge is to be disposed of, no further treatment is needed for the sludge, it maintains good effluent quality and allows flexibility and simplicity for the plant operators. However, an increase in the total volume (etc. for an upgrade) by addition of aerobic or anoxic reactors increases the investment and operational/maintenance costs.

An Orbal plant is a type of extended aeration activated sludge plants which claims to achieve simultaneous nitrification and denitrification in a single reactor, offering reduced costs. Generally, three or four channels are recommended for design as shown in Figure 1-left (Drews *et al.*, 1972; 1973). The simultaneous nitrification and denitrification in the outer loop that receives the influent wastewater give

an overall denitrification performance of 80% operated under moderate sludge ages in order to suppress the growth of microorganisms causing bulking and settling problems (Daigger and Littleton, 2000).
In Orbal systems, the outer loop is designed to be the largest channel used for simultaneous nitrification-denitrification and combine the advantages of completely mixed and plug flow reactors in one bioreactor (Smith, 1996). Step-feed in terms of return activated sludge (RAS) and influent wastewater can be used in the optimization of the plant (Applegate *et al*, 1980).
In the literature, different hydraulic layouts were considered for the Orbal plants. A tracer test was conducted for an Orbal plant by Burrows *et al.*(1999). It was concluded from the residence time distribution results that the outer channel could be regarded as two CSTR in series, however, while the middle and inner channels could be configured as single CSTRs. On the contrary, Cinar *et al.* (1998), Daigger and Littleton (2000) considered the outer channel as 6 reactors in series but the other channels as individual CSTRs, as well.

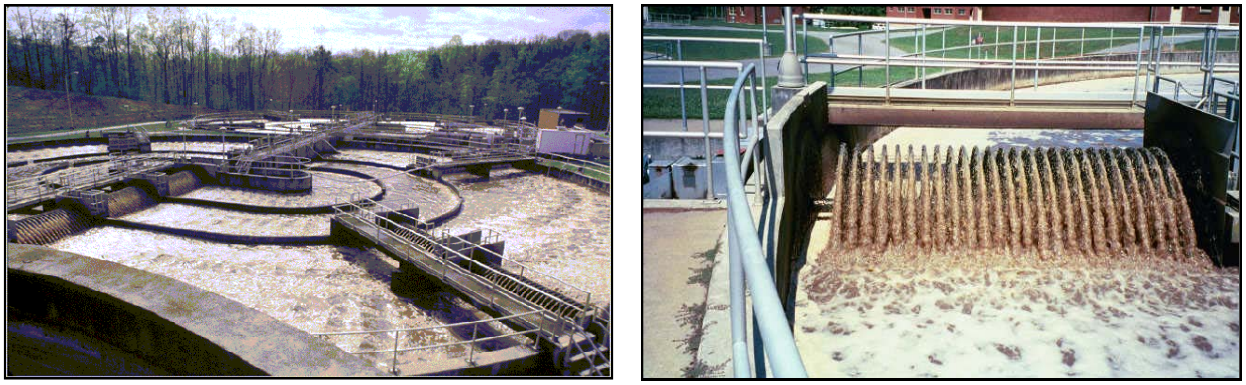

Figure 1. Pictures of biological reactor and vertical disc aerators (Liu, 2000)

Aeration is provided by a number of perforated discs installed perpendicular to the rotating shaft. The discs are partially immersed in the mixed liquor and rotated around their horizontal axes (see Figure 1-right). The main feature for the operation strategy is that the Orbal plants are operated under oxygen deficit conditions (around 0.1-0.75mgO_2/l) promoting the simultaneous nitrification and denitrification together with carbon and biological phosphorus removal (DeSilva and Rittman, 2001; Daigger and Littleton, 2000; Bertanza, 1997; Applegate *et al.*, 1980; Drews *et al.*, 1972). Simultaneous nitrification and denitrification is achieved by supplying oxygen to all channels by mechanical disc aerators (Figure 1-right). The actual oxygen demand of the outer loop might be as high as 75% of the total system, the aeration discs allocated for the channel supply only 30-60% of the oxygen requirements. The dissolved oxygen in the middle loop varies depending upon the organic loading. The last channel works as a polishing mode to remove the remaining COD and ammonia (USFilter/Envirex, 2002). For simultaneous nitrification-denitrification, it was suggested that the dissolved oxygen concentration around 0.5 is suitable to achieve a nitrification rate equal to the denitrification rate (Münch *et al.*, 1996).

It was hypothesized that three general mechanisms are responsible for simultaneous biological nutrient removal. These are (a) bioreactor mixing patterns that allow the anoxic and anaerobic zones necessary for biological nutrient removal (b) anoxic and anaerobic zones within the flocs and (c) the presence of novel microorganisms in the system (Littleton *et al.*, 2002; Daigger and Littleton, 2000; Applegate *et al.*, 1980). The removal efficiency for carbon, nitrogen and phosphorus removal is reported to be in a wide range depending upon the operation strategy, plant configuration, wastewater strength and environmental factors (i.e. temperature). It was stated that the plants operated under low oxygen concentration up to 90-95% nitrogen removal could be attained (Pochana and Keller; 1999; Rittman and Langeland, 1985). The largest denitrification was attained at a COD/TKN ratio of 6.4 under 0.18 mg/l

oxygen concentration in the bulk liquid (DeSilva and Rittman, 2001). Drews *et al.* (1972) reported that the removal efficiency of COD in the range of 78.1%-88.9%. By keeping the dissolved oxygen concentration below 1.0 mg/l, the total nitrogen removal efficiency was in the range of 65.6%-86.3% with elevated ammonia concentrations in the effluent around 5-8 mg NH_4-N/l without EBPR.
A model-based process analysis of an Orbal plant carried out by Daigger and Littleton (2000) via ASM1 simulation showed that the oxygen limited conditions without internal recycle from inner channel to outer channel exerted a minimum effluent nitrate concentration due to the fact that the oxygen carry over through the outer channel probably disturbed the simultaneous biological nutrient removal in outer ring. Depending upon the variation in the organic loading (55%-132%) and the sludge ages (8-33 days), the removal efficiencies for total phosphorus removal efficiency were calculated in the range of 61%-95%. The total nitrogen removal was calculated to be around 85-90% (Daigger and Littleton, 2000). Applegate *et al.* (1980) compared the extended aeration and the step feed operation mode (by diverting a portion of inflow to the second channel) for a full-scale Orbal plant. The removal efficiency for total nitrogen is increased from 76% up to 91%. Enhanced biological phosphorus removal, EBPR has been observed in aerated bioreactors without anaerobic zones prior to aerobic or anoxic reactors (Cinar *et al.*, 1998; Applegate, 1980). For this, the anaerobic zones formed inside the flocs and/or vertical or horizontal flow patterns in the outer channel may promote the phosphorus release resulting in an overall biological phosphorus removal (Daigger and Littleton, 2000). On the contrary, it was also suggested that the influent total phosphorus concentrations are often low so that phosphorus removal may simply be due to biomass synthesis (Daigger and Littleton, 2000). Regarding the EBPR modeling, Cinar *et al.* (1998) was unable to model using the steady state ASM2 model because the anaerobic conditions that are necessary for PHA storage could not be obtained. The simulations using ASM2 (Henze *et al.*, 1995) with 3 CSTR-in-series representing the Orbal also could not yield PAO growth (Littleton *et al.*, 2002; Cinar *et al.*, 1998).

In this study, the behavior of the Georgia/Athens Orbal plant was investigated under dynamic loadings with respect to its nitrogen and phosphorus removal using the ASM2d model (Henze *et al.*, 1999). The first step is to calibrate the model using steady state simulations. In the second step, the behavior of the plant under dynamic loading was investigated.

MATERIALS AND METHODS

Plant Definition

The Orbal plant under study is located in Georgia, Athens treating the wastewater of 50000 PE. The layout used in the calibration consists of a bioreactor and 4 final clarifiers (1 spare for rain events). The biological reactor is composed of three concentrically arranged closed loop bioreactors (see Figure 2-left). Influent wastewater first enters the outer loop and passes through the middle, inner reactor and the final clarifier, respectively. The plant can be operated as a step-feed plant and/or the return sludge can be diverted to the inner rings depending upon the operating strategy. As can be seen from in Figure 2-right, the influent wastewater and return activated sludge, RAS, are fed only to the outer loop without any step feed pattern for the calibration period. The overall plant capacity is 20328 m^3/day (5.37 MG/day) on average.

The total volume of the biological reactor is 12553 m^3 consisting of outer, middle and the inner rings with the volumes of 8665 m^3 (69% of total volume), 2544 m^3 (20%) and 1344 m^3 (11%), respectively. The surface area for each circular final clarifier is 448 m^2 with 4.8 m in height (Figure 2-right). The hydraulic retention time (HRT) and the sludge age of the system is 18 hrs and 9-10 days, respectively. The mixed liquor concentration in the aerobic reactor is kept around 3000-3500 mg/l and the reactor is operated under low oxygen for the simultaneous nitrification-denitrification. Oxygen transfer is provided with surface disc aerators and all loops have disc aerators mounted on a rotor providing the recirculation of mixed liquor within the reactor. Each rotor has 24 aeration discs (see Figure 1-right). Depending upon oxygen requirement and operation the number and the spans between discs, speed and the immersion depth can be adjusted as appropriate. So, this feature gives considerable flexibility in terms of oxygen transfer. The velocity in the channel is in the range of 0.28-0.35 m/sec (0.3 m/sec in average) by keeping the mixed liquor in suspension.

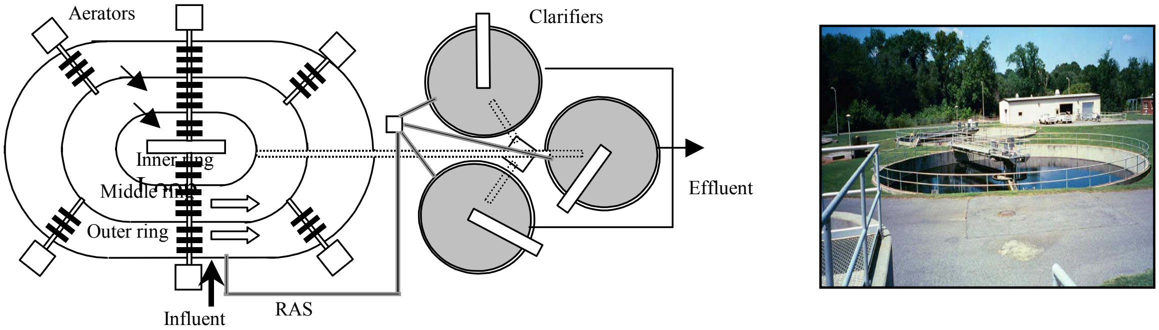

Figure 2. The plant layout for the simulated plant (left) and clarifier (right)

Measurements and calibration

In the course of measuring campaign, the influent flow rate, return activated sludge, RAS; waste activated sludge, WAS flow rate, 0.2 micron-filtered TOC, total filtered organic nitrogen, filtered ammonia nitrogen, orthophosphate, dissolved oxygen concentration in the channels and the sludge blanket level were measured in every 15 minutes with Environmental Process Control Laboratory, EPCL manufactured by Capital Controls-Minworth Systems Ltd. (Liu, 2000). The measuring campaign data for the Athens/Orbal plant was obtained from Liu (2000). The reader should consult to Liu (2000) for more information about the analytical methods, data collection and the hardware facilities used in the measuring campaign. The experimental results are shown in the results and discussion section.

Table 1. Flux based average influent wastewater characterization

Component	Unit	Concentration	% of total COD
Total COD, COD_{tot}	mgCOD/l	404	
Biochemical Oxygen Demand, BOD_5	mgO_2/l	159	
pH^{**}		6.8	
Total inert COD, C_I	mgCOD/l	174	43
Particulate Inert COD, X_I	mgCOD/l	150^*	37
Soluble Inert COD, S_I	mgCOD/l	24	5
Biodegradable COD, C_S	mgCOD/l	230	57
Fermentable COD, S_F	mgCOD/l	60	14
Acetate, S_A	mgCOD/l	30	7
Slowly Biodegradable COD, X_S	mgCOD/l	140	32
Ortho-P, S_{PO4}-P	mgP/l	1.8	
Total Phosphate, TP	mgP/l	5.3	
Ammonium, NH_4-N	mgN/l	12	
Total Kjeldahl Nitrogen TKN	mgN/l	22.4	
TSS (calculated from COD)	mgSS/l	217	
COD_{tot}/TKN (BOD_5/TKN)		18 (7.1)	
COD_{tot}/TP (BOD_5/TKN)		76 (30)	
C_S/TKN		10	
C_S/TP		43	

*calibrated in order to fit the MLSS

The influent wastewater characterization in terms of COD fractions used in the dynamic calibration was generated according to the filtered TOC measurements. The ratio of COD_{tot}/TOC ratio was adopted as 3.1 according to Metcalf and Eddy (1991) and Servais *et al.* (1999). The average percentages for the COD fractions were adopted from Henze *et al.* (1999). The flux based average wastewater characterization and the COD fractionations are given in Table 1.

The BIOMATH calibration protocol (Vanrolleghem *et al.*, 2003) was applied for the calibration of the plant using the existing plant data. As a first step, simple layout composed of 3 reactors in series was used for the preliminary simulations. All simulations were performed in the modeling and simulation software WEST (Vanhooren *et al.*, 2003). The growth of PAOs could not be sustained because they were unable to compete with other heterotrophs. As a result, PAOs were completely washed out at the end of the simulation even tough the parameters pertaining to PAOs were changed (Cinar *et al.*, 1998; Littleton *et al.*, 2002). In order to improve the PAO growth in the system the plant layout shown in Figure 3 was introduced to WEST. The outer, middle and the inner ring are represented by 6, 4 and 2 CSTRs in series based on the reactor geometry. The volumes of each reactor for the outer, middle and inner ring are 1444, 636 and 672 m^3, respectively. As stated above, three out of 4 clarifiers were operational during the measurement campaign period. Because three of the clarifiers were under identical conditions, a simple clarifier with a non-reactive 20 layered-Takács *et al.* (1991) model was assigned for the simulation of the sludge blanket in the clarifier. The overall surface area, A and the water height, H_S in this clarifier are 1345m^2 and 4.8 m, respectively. The flow rates for the influent underflow and the sludge wastage rates were introduced in WEST as input-log files. The flow rate of the influent, return activated sludge (RAS) and activated sludge (WAS) used in the simulations are illustrated in Figure 4.

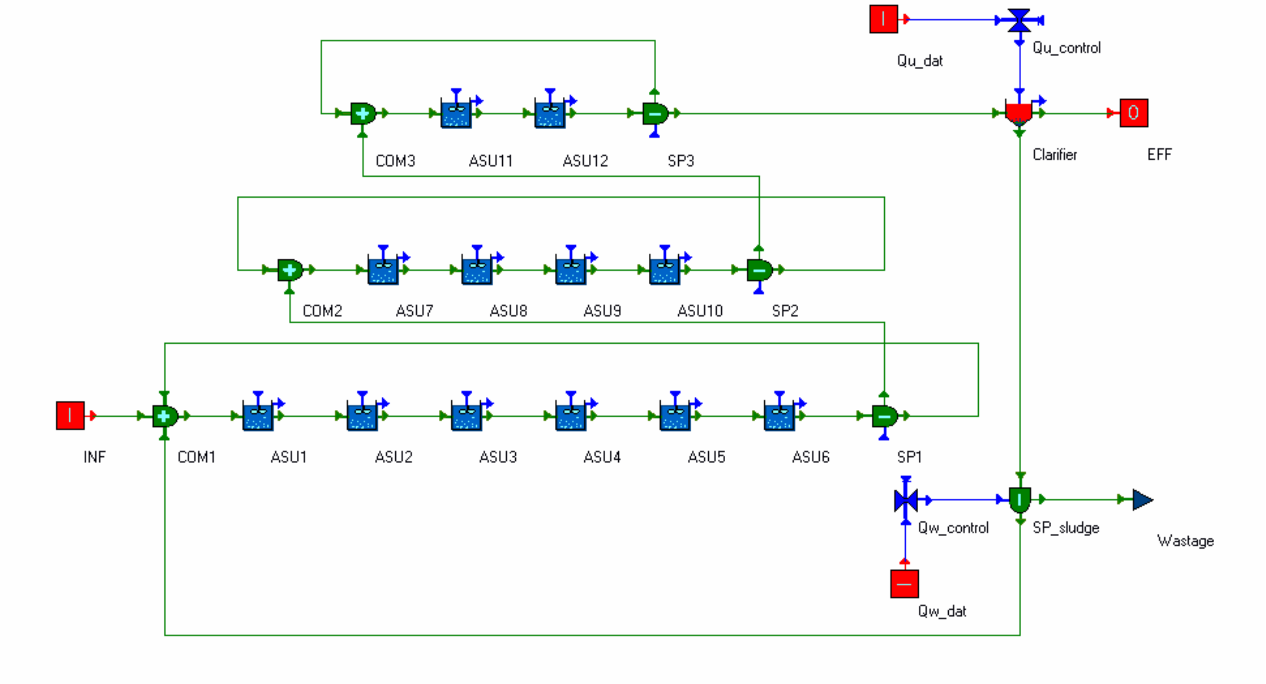

Figure 3. Implementation of plant layout in WEST

A steady state influent file was used for the preliminary simulations in order obtain an appropriate biomass composition that would be a starting point for the dynamic calibration. The ASM2d model was selected for this calibration issue. In order to reflect the wastewater plant performance under steady state average measurement outputs were considered during the steady state calibration. A manual iterative estimation method was used until an appropriate initial biomass composition was obtained. It should be stressed here that the manual iterative method was only terminated when the same values were found considering all measurement outputs for the steady state and dynamic calibration, as well. The scheme of the calibration approach is illustrated in Figure 5. The steady state simulation relates to a 50 days period so that all the components reached their steady state values. The most sensitive parameters with respect to the measurements outputs were selected for adjustment.

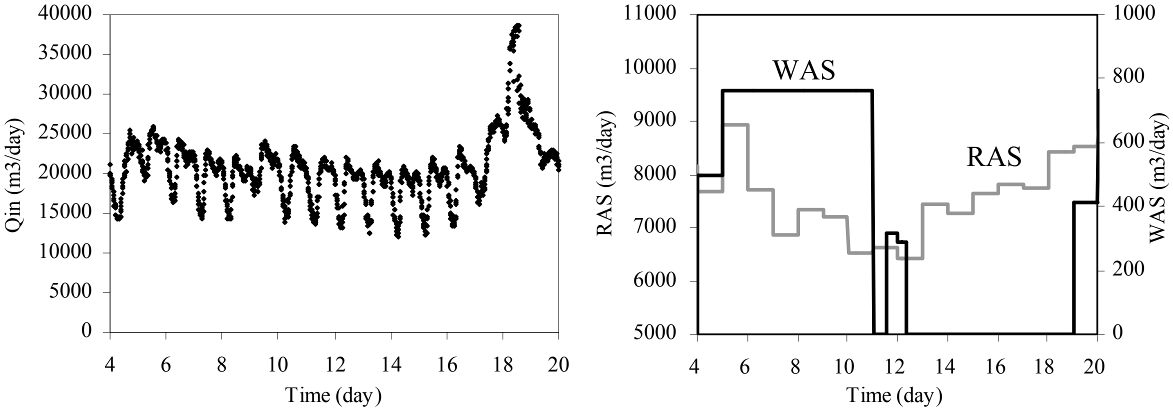

Figure 4. Influent flow rate (left), RAS and WAS flow rates (right)

RESULTS AND DISCUSSION

The steady state simulation not only reflects the overall treatment capacity of a plant but also approximates the appropriate biomass composition of the mixed liquor which is a crucial factor for the calibration and the estimation of the parameters (Vanrolleghem *et al.*, 2003). The reason is that an inappropriate determination of the sludge composition causes bias in the other calibrated parameter values. The results of the steady state simulation based on flux and concentrations are shown in Table 2 and Table 3. It can be inferred from Table 2 that the removal efficiency for TKN and TP is around 77% and 94% respectively. Simultaneous nitrification and denitrification provided a removal of 42% of the total daily influent TKN load was removed by simultaneous nitrification and denitrification. The remaining 35% is entrapped in the sludge and used as a nutrient. The removal efficiency of total nitrogen removal of the plant is in compliance with the counterparts in the literature of which where reported in the range of 65.6%-86.3%.

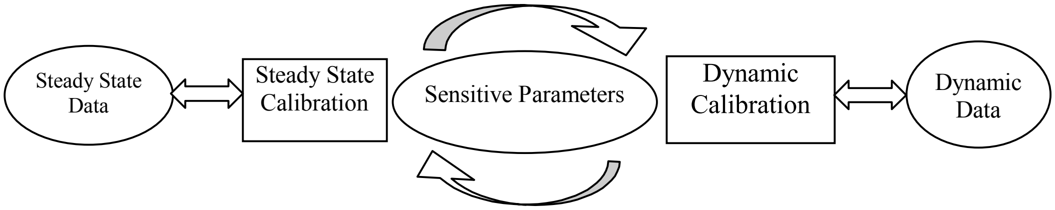

Figure 5. Schematic representation of the calibration methodology

The first step in the calibration study was to fit the solid mass balance over the treatment plant. The sludge age of the system was calculated to be around 10 days from the phosphate balance that is in agreement with the information provided from plant operators.In order to fit the sludge concentration in the reactor and maintain 3000-3500 mg MLSS/l in the middle ring, the particulate inert COD fraction in the influent was calibrated during steady state calibration. According to the value in Table 1, the X_I fraction was found to be comparably higher. Figure 6 shows the measured and the simulated MLSS concentrations in the middle ring. It is obvious that, till day 18, the model captured the trend of MLSS in the reactor successfully. The difference between the measurement and simulation is the operational changes in the process by the plant operators. For instance, at day 18 the mixed liquor is diverted into the middle channel because of the storm event reported. Insufficient information on the changes in flow and made the system difficult to calibrate in terms of sludge blanket in the clarifier. It should be noted here that the sludge wastage is not constant during the calibration period that was also included in the model. The ridges on the MLSS profile were also captured to some extent by the model because the reactor received a much higher TSS load during the morning and afternoon time when the flow rate increased.

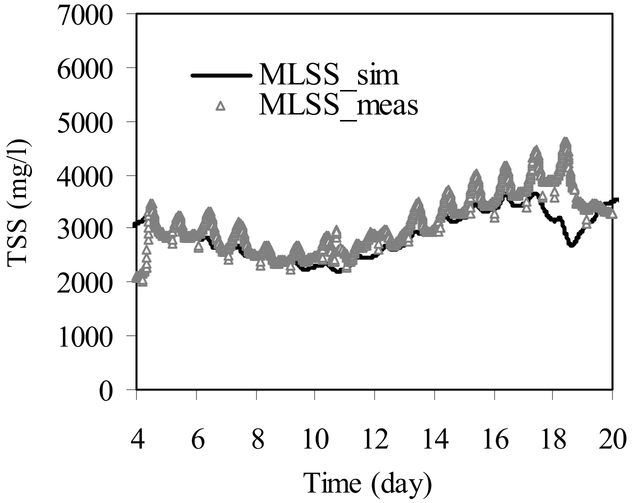

Figure 6. MLSS profiles in middle ring

In the steady state calibration the measured and simulated ammonia concentrations in the effluent are 4.4 and 4.8 mgN/L, respectively. The reported effluent ammonium nitrogen concentrations are typically around 5-8 mgN/l in the effluent (Drews *et al.*, 1972; Drews and Greefs, 1973; Applegate; 1980). The phosphorus removal efficiency of the plant was found to be closer to the upper limit of the range 61%-95% (Daigger and Littleton, 2000; Cinar *et al.*, 1998; Applegate, 1980). The measured and simulated effluent phosphate concentrations are 0.6 and 0.3 mgP/L for steady state simulation.

A higher phosphorus removal efficiency was attained since the NO_3 concentration is nearly zero in the outer ring leadin to the anaerobic condition necessary for PAOs (Comeau *et al.*, 1986; Ekama and Wentzel, 1999; Kerrn-Jespersen and Henze; 1993; Kuba *et al.*, 1994).

Table 2. Simulated steady state plant performance

Component	**Influent** kg/day	**Effluent** kg/day	**Removed** kg/day	**Removal** %
Flowrate	20328*	19878*		
TSS load	4411	198	4213	96
COD load	8944	485	8459	95
TKN load	450	105	345	77
			191**	42**
TP load	108	6	103	94

*units in m3/day, ** removed by simultaneous nitr./denit.

The calibrated ASM2d parameters are visualized in Table 4 together with their default values (Henze *et al.*, 1999). Calibration study revealed that the most sensitive process on the oxygen budget is found to be the hydrolysis process. In the ASM2d report, the anaerobic hydrolysis reduction factor, η_{fe} is recommended to be 0.1 as a default value. According to the calibration results carried out by Satoh *et al.* (2000), a lumped parameter group $\eta_{fe}k_h$ was estimated to be 0.35 where η_{fe} is 0.2. The hydrolysis rates were reported to be much higher for NDBEPR than for ND systems (Clayton *et al.,* 1991). Another statement on the hydrolysis is that the hydrolysis rate is independent from electron acceptor conditions (Gujer *et al.*, 1999; Mino *et al.*, 1995; Goel *et al.*, 1998). However, the hydrolysis rate was also reported to be dependent on the electron acceptor conditions (Sözen *et al.*, 1998; Henze *et al.* 1987; 1995; 1999; Maurer and Gujer; 1994; Henze and Mladenovski, 1991).

Table 4. Parameters used in the calibration for ASM2d

Parameters	Symbol	Unit	Default*	Calibrated**
Biological parameters				
Anaerobic hydrolysis reduction factor	η_{fe}	-	0.1	0.35
Poly-P requirement per PHA stored	Y_{PO4}	gP/gCOD	0.4	0.36
Maximum storage rate for PHA	q_{PHA}	day^{-1}	3.0	2.20
Rate for lysis of X_{PAOs} X_{PP} X_{PHA}	b_{PAO}, b_{PP}, b_{PHA}	day^{-1}	0.2	0.08
Maximum growth rate for X_{AUT}	μ_{AUT}	day^{-1}	1.0	0.70
Half saturation constant of O_2 for X_{AUT}	K_{OAUT}	mgO/l	0.5	0.10
Half saturation constant of O_2 for X_H	K_O	mgO/l	0.2	0.13
Saturation coefficient for phosphorus in storage of PP	K_{PS}	mgP/l	0.2	0.50
Settling parameters				
Maximum theoretical settling velocity	v_0	m/day	-	680

*@ 20^0C ** @ 17^0C

The anaerobic hydrolysis reduction factor, η_{fe} was experimentally found in the range of 0.8-1.0 (Mino *et al.*, 1995) and it was reported that, depending on the rate and the configuration, the η_{fe} has a crucial role on the phosphorus release concomitantly, the X_{PAO}, X_{PP} and effluent phosphate concentrations in the bulk liquid (Larrea *et al.*, 2001; Ekama and Wentzel, 1999; Carucci *et al.*, 1999). The slowly biodegradable COD degradation is a very important issue with respect to the realistic modeling of activated sludge systems because it is primarily responsible for the attainment of realistic space-time dependent electron acceptor profiles (Maurer and Gujer, 1994; Bidstrup and Grady, 1988). To keep it simple, only the anaerobic hydrolysis reduction factor was increased from 0.1 to 0.35. In that way, more carbon source became available for the simultaneous denitrification together with EBPR by keeping the dissolved oxygen at lower levels in the rings.

The yield coefficient for Poly-P requirement per PHA stored, Y_{PO4} was decreased to 0.36 mPO_4/mgCOD to lower the peaks during the morning and afternoon (under high COD loadings). The second issue is that the pH in the outer reactor is around 6.7 resulting in a lower phosphate release as discussed by Smolders *et al.*, (1994) and Filipe and Daigger (1999). It was shown that the energy generated from the observed phosphate release at low pH is not enough to convert acetate to acetylCoA, so the glycogen metabolism is more pronounced. The value of this parameter shows a wide variation in the literature with a range of 0.4-1.4 (P/C) by Wentzel *et al.*(1985); 0.3-0.78 by Mino *et al.*(1987). The studies carried out by Filipe *et al.*, 2001; Romansky *et al.* (1997) and Smolders *et al.* (1994) showed the effect of different pH on phosphorus release. A decrease in pH resulted in a decrease in released phosphate under anaerobic conditions. So, the calibrated value of the Y_{PO4} is in concert with the results of Smolders *et al.* (1994).

The oxygen saturation constant for heterotrophs, K_{OH} was decreased from 0.2 to 0.13 mgO$_2$/l in order to increase the activity of the heterotrophs and PAOs under oxygen limited conditions. The K_{OH} varies significantly depending on the model structure, biomass type and mixing patterns in bioreactors (Orhon and Artan, 1994). Lau *et al.* (1984) reported a value of K_{OH} varying from 0.01 mgO$_2$/l for filamentous bacteria, to 0.15 mgO$_2$/l for floc-forming bacteria. According to the literature, the value of K_{OH} is reported to be 0.002 mgO$_2$/l in the general model of Dold and Marais (1986); 0.2 mgO$_2$/l by Larrea *et al.* (2001) and De la Sota *et al.* (1994); 0.7 mgO/l for a SND (EBPR) modeled by Meijer *et al.* (2001); 0.5 mgO$_2$/l in the full scale modeling studies of Wichern *et al.* (2001) and 0.15 mgO$_2$/l by Carucci *et al.* (1999). In addition, Meijer *et al.* (2001) stressed the relationship between the K_{OH} and the influent slowly biodegradable COD fraction, X_S. The effluent nitrate concentration is highly dependent on the oxygen saturation constant and K_{OH}. Lowering the K_{OH} value together with the $X_S/(X_S+X_I)$ ratio resulted in the increase of the nitrate concentration because of limited denitrification capacity. In IAWPRC (ASM1, ASM2d, ASM3) models, a value of 0.2 mgO$_2$/l is suggested (Henze *et al.*, 1987; 1995; 1999).
The half saturation oxygen constant for autotrophs was reported in wide range of 0.002-2.0 mgO2/l (US/EPA, 1975; Dold and Marais, 1986). On top of that, after an adaptation period, low oxygen concentration was found to have an influence on the cell aggregation where the establishment of smaller aggregates exerted low oxygen affinity constants in an aerated reactor (Park and Noguera, 2002). The value of the maximum autotrophic growth rate, μ_{AUT} and the half saturation constant for ammonium were reported in a wide range of 0.25-1.23 day^{-1}and 0.06-5.6 mgN/l (Cinar *et al.*, 1997; Dold, 2002; Stenstrom and Podushka, 1980; Sözen *et al.*, 1996; Daigger and Nolasco, 1995; Copp and Murphy, 1995). The μ_{AUT} and K_{oaut} was fine-tuned separately with the aid of the calibration method (Figure 5) using the steady state and dynamic data. The μ_{AUT} and K_{OAUT} were adjusted to 0.7 day^{-1} and 0.10 mgO$_2$/l respectively. The values are in the range of their reported bounds. The activity of nitrite oxidizers was strongly limited at 0.5 mgO$_2$/l and nitrite can be accumulated up to 60 mgN/l (Hanaki *et al.*, 1990). However, nitrite concentrations were negligible in the measuring campaign data. The effect of heterotrophs on the nitrifiers was also found to be significant because of the ammonia and the oxygen balance in the bulk liquid. The maximum growth rate for autotrophs was reported in the range of 0.66-78 day^{-1} under low oxygen levels (Hanaki et al., 1990). The nitrification process under low oxygen levels was activated by decreasing the K_{OAUT} during the steady state calibration. The maximum growth rate for nitrifiers was calibrated under dynamic conditions. The results show that the rate was adversely influenced by low oxygen concentrations and/or slightly lower pH (Orhon and Artan, 1994).

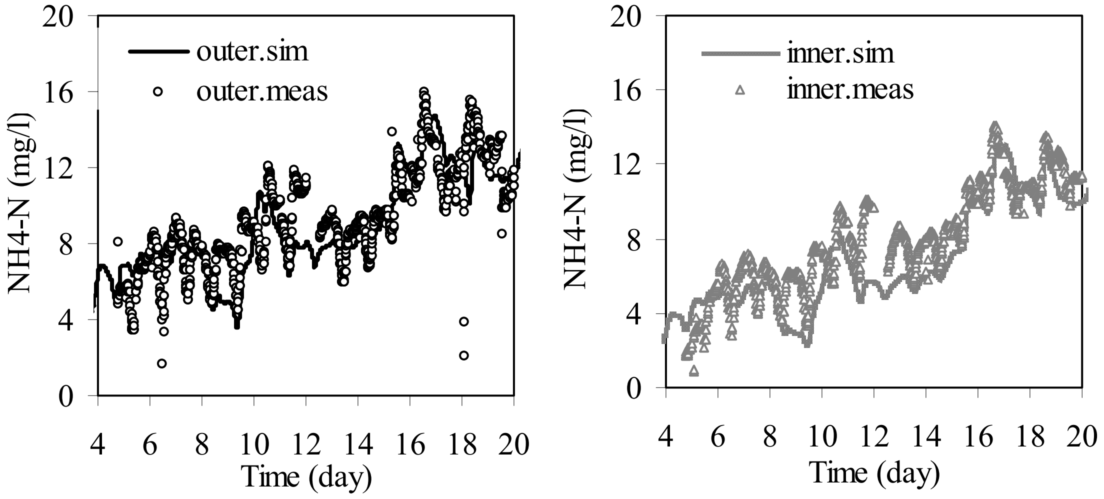

Figure 7. NH$_4$-N profile in outer and inner rings

The maximum storage rate for PHA, q_{PHA} was assigned to 2.2 gCOD/gcellCOD.d also falling in the range of 2.0-3.0 suggested in Henze *et al.* (1999). The steady state calibration studies of Cinar *et al.*, (1998) for oxidation ditches showed that EBPR could only be attained by setting the q_{PHA} to 8.0 gCOD/gCODd using the ASM2 model. Otherwise, X_{PAOs} could not compete with the ordinary heterotrophs and finally washed out from the system because of the fact that they could not build up their PHA pool for biomass growth and Poly-P storage. This may have been due to the over estimation of the denitrification capacity of ordinary heterotrophic biomass. The values of the q_{PHA} in the literature vary between 3.0-8.0 gCOD/gCODd (Siegrist *et al.*, 2002; Cinar *et al.*, 1998; Daigger and Nolasco; 1995, Wentzel *et al.*, 1989).

The lysis rates for X_{PAO}, X_{PHA} and X_{PP} were adjusted to 0.08day^{-1} since the PAOs were washed out from the system. This adjustment in this parameter provided for a relatively constant PAO concentration in the bioreactor. Also, the PHA pool could be efficiently used by PAOs under dynamic loadings. Similar observations and calibration results were also reported for values around 0.12day^{-1} suggested by Siegrist *et al.* (1999) and 0.14day^{-1} by Cinar *et al.* (1998). The hypothesis is that the PAOs slow down their endogenous metabolism in order to compete with other heterotrophs under low oxygen/nitrate concentrations (Cinar *et al.*, 1998; Randall *et al.*, 1992). Another reason could be predation by protozoan activity that exists mainly under aerobic conditions (van Loosdrecht and Henze, 1999; Tijhuis *et al.*, 1993). The saturation coefficient for phosphorus uptake, K_{PS} was increased to 0.5 which allowed to keep the P fluctuations smoother under dynamic conditions. Figure 7 shows that there is an increase in the effluent ammonia concentration because of the suppressed growth due to the oxygen limitation. The autotrophic biomass concentration, X_{AUT} was simulated to be around 15 mg COD/l and showed a decrease towards 10 mgCOD/l during that period because of oxygen limitation and excessive sludge wastage (see Figure 11). It was difficult to calculate the minimum sludge age for autotrophic biomass because of oxygen-limited conditions.

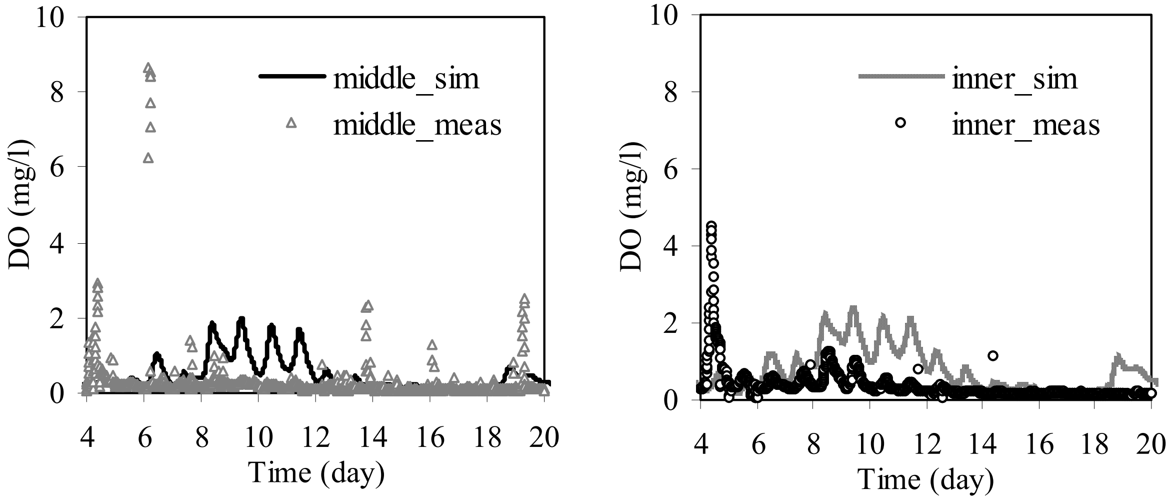

Figure 8. Dissolved oxygen in middle and inner rings

Generally, the oxygen concentrations were simulated to be below 0.2-0.5 mgO_2/l depending upon the organic loadings. During the time interval between 8-12 days, an increase in the oxygen level concomitant to NO_3 peaks can be observed in Figure 8, Figure 9. In reality, the most probable reason is the decrease in the biodegradable COD load causing an oxygen increase or increase in the aeration capacity. As a result, the activity of autotrophs increased under non-limited oxygen conditions. In addition, the aeration capacities of the disc aerator system in all reactors were kept constant despite some minor changes done by plant operators during the calibration period. In spite of these uncertainties, a fairly good fit was attained on the ammonia, the nitrate and the phosphate measurement outputs.

The filtered phosphate concentrations in the effluent stayed below 1.0 mgP/l during the measurement period. However, Figure 10 reflects the phosphorus in the middle ring and in the clarifier. According to the figure, the effluent phosphorus slightly increased and fluctuated until day 12. On the following days, obviously, the removal efficiency again recovered. In addition to that, these fluctuations in the profile are quite visible because of the variations in the incoming organic loading during the day. Depending on that situation, the internal storage polymer, PHA was also found to be varying over time. The exhaustion of the PHA pool also causes phosphorus release due to endogenous decay metabolism. Shortly, two marginal situations could be observed along the day period (a) depending upon the influent COD load, PAOs could release phosphorus in the absence of electron acceptors. This situation inevitably occurs if aeration is not sufficient under elevated organic loadings; (b) phosphorus release can also occur if no PHA pool is available for phosphorus uptake. In the literature, the reason of the phosphate peaks were attributed to the temporal imbalance between P-release and uptake by Isaacs *et al.* (1994) or the depletion of internally storage pool due to the excessive aeration and/or low organic loading (Brdjanovic *et al.*, 1998; Temmink *et al.*, 1996).

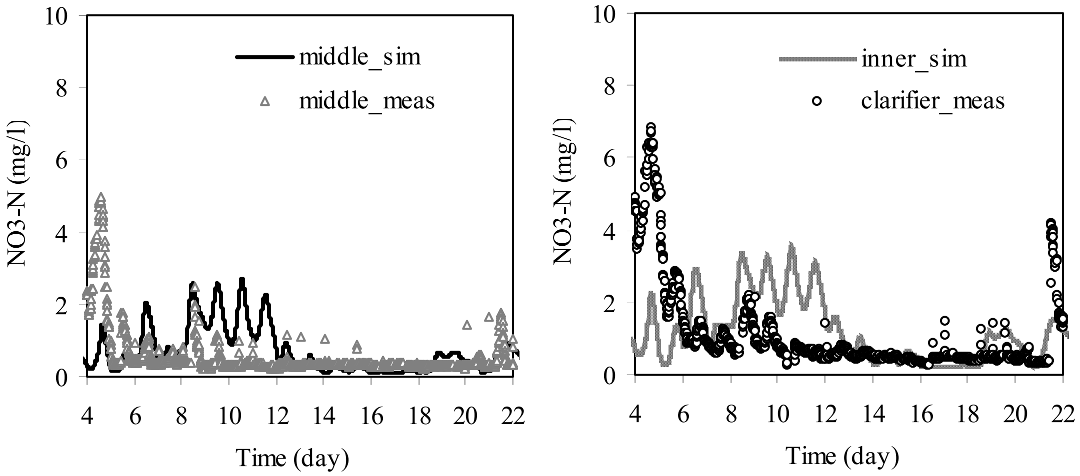

Figure 9. Nitrate nitrogen profile in middle and inner rings

Another point is the difference between the simulated and measured phosphate profiles in the clarifier. The difference may be attributed to a "secondary release of phosphorus" in the final clarifier since the nitrate and the oxygen levels are very low. Wouters-Wasiak *et al.* (1996) stated that the secondary phosphate release occurs at low rates (0.2-0.4 mgP/gVSS.h) when the nitrate concentration is below 0.5 mgN/l in the absence of oxygen, as well.

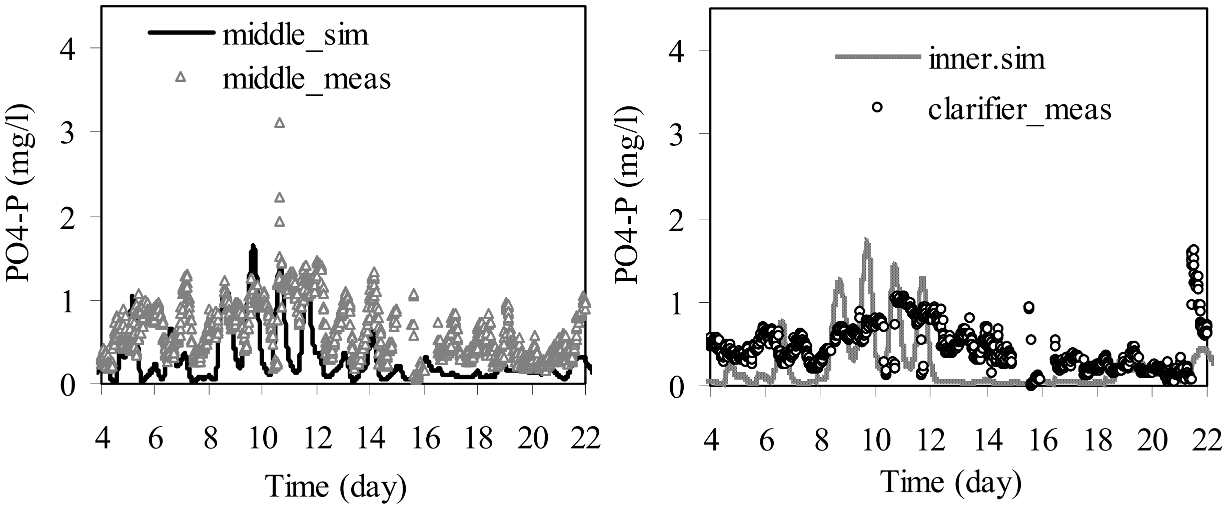

Figure 10. Phosphate profiles in middle and inner rings

The exhaustion of the PHA pool also makes PAOs to release phosphorus into the bulk liquid to derive energy for their maintenance (Siegrist *et al.*, 1999). The adjustment of the aeration and sludge wastage can be the solution to maintain the storage material at some degree during these starvation periods (Miyake and Morgenroth, 2002). Recovery time needed for phosphorus removal can be decreased in that manner. Considering the dual storage phenomena, it was suggested that the glycogen cannot replace PHB for phosphate uptake under aerobic conditions and it is only used for maintenance (Brdjanovic *et al.*, 1998). The trajectories of the simulation pertaining to the biomass components are illustrated in Figure 11

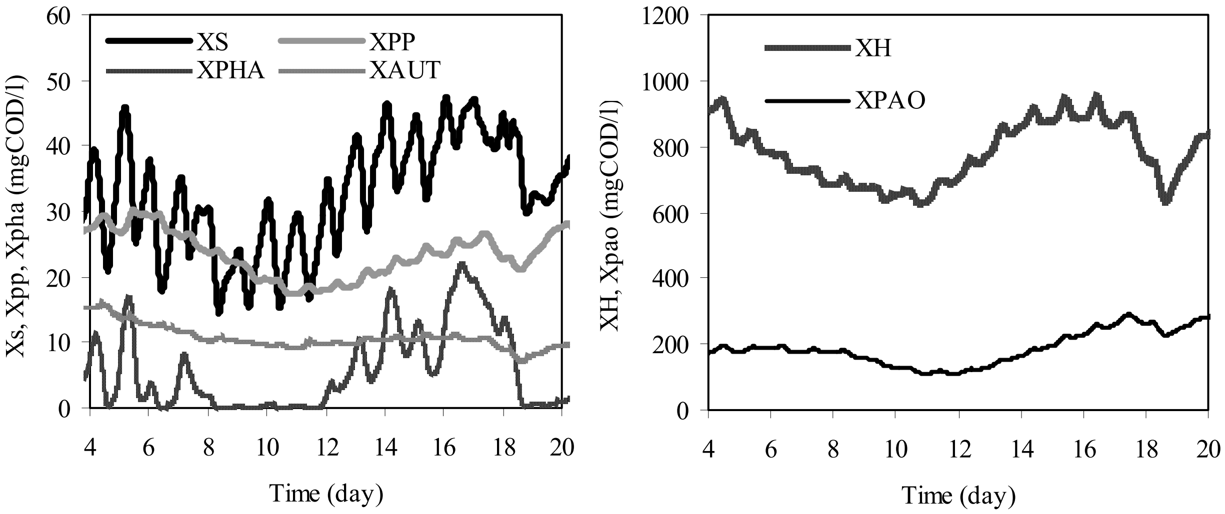

Figure 11. Simulated biomass composition (middle ring)

A non-reactive Takacs *et al.* (1991) settling model with 20 layers was used to simulate the sludge blanket and concentrations in the clarifier. The theoretical settling velocity, V_0 was set to 680 m/day in order to allow faster settling in the clarifier and keep the sludge blanket at low levels ($H_S<1m$). Figure 12 (left) reflects the measured and the simulated sludge blankets in the clarifier. The simulation for the period between 4-18 days successfully predicted the sludge blanket. However, as seen in the figure, the model overestimates the sludge blanket height at 18th days. Diversion of the flow into the middle ring could explain the experimentally obtained sludge blanket at 18th days. Sludge concentration profile measured on day 10 showed a good agreement with the experimental sludge concentration profile along the depth of the clarifier (Figure 12-right).

It should also be noted here that during the rain event some measures had been taken by the plant operators in order to prevent sludge lost from the clarifier which were not very well documented. As a result, it was difficult to conclude and model the effect of low organic loading in last days. The gradual decline in the biomass components during the first 12 days is due to the excessive sludge wastage. In the following period the recovery (increase in concentration) in all components was simulated and the trends of all components are identical. These results also agree with the phosphorus removal recovery stated above. The concentration of PHA was found to be instable. As previously discussed and concluded from the simulations, the recovery of the PHA pool is highly dependent on the COD load and the aeration capacity of the plant.

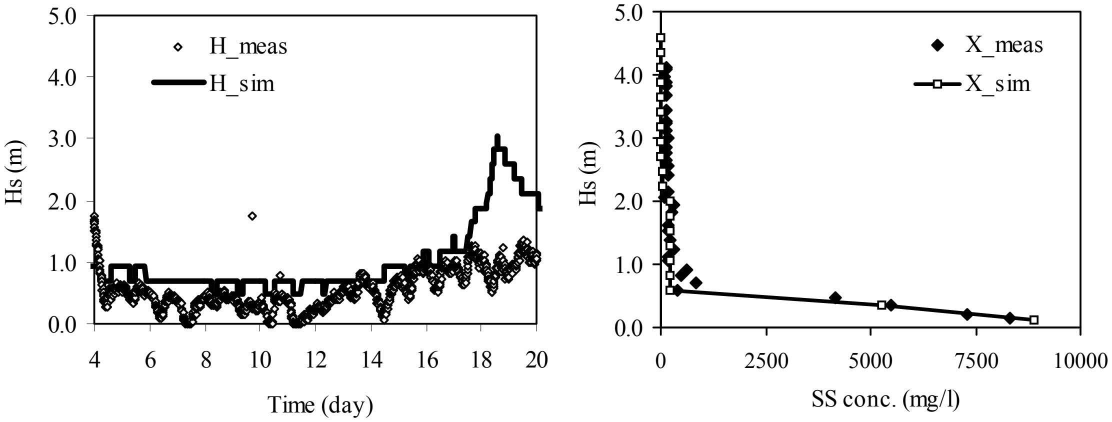

Figure 12. Sludge blanket height (left) and sludge conc. profile (right) in the clarifier (Day 10)

CONCLUSIONS AND PERSPECTIVES

The influent wastewater characterization in terms of COD fractionation plays a crucial role in modeling. For instance, uncertainty in the slowly biodegradable COD caused oxygen elevations in the reactors. As a result all processes depending on oxygen concentration (i.e. P release, nitrification, denitrification) are adversely affected by this fact. So, the hydrolysis mechanism plays an important role not only on the oxygen levels and COD source in the reactors but also on the effluent nutrient concentrations. The measurement outputs were found to be very sensitive to the treatment plant operational parameters. A successful calibration is highly dependent on better understanding of the system in terms of process changes. Otherwise, it causes biased estimation and/or misleading conclusions in the model calibration.

The EBPR mechanism in simultaneous nitrification-denitrification, SND was found to be instable in terms of the storage polymer pool. During daytime, the variation in the COD load may result in P release. On the other hand, the exhaustion of this storage polymer also leads to P release because of endogenous metabolism. This should be investigated further. The system operated under low oxygen levels and under varying input load suffered from oxygen limitation, as a result, nitrification failure. In addition, insufficient aeration and excessive sludge wastage resulted in nitrification loss during the measuring campaign period. With efficient control of aeration, nitrification can be improved. However, EBPR should be taken care of then, because of nitrate will build up in the outer ring. So, the sludge wastage and the aeration control are necessary for stable operation of SND systems.

The changes in the plant operations should be well defined in order to represent them in the simulation. This would be a part of the verification of the model. Otherwise, it is impossible to calibrate or verify the model and/or to interpret the measurement results. A successful model calibration necessitates to collect ample information that reflects the real situation.

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