

Evaluation of a new model for the reduction of excess sludge production by ozonation of return activated sludge: what solids COD fraction is affected?

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ABSTRACT

This paper aims at clarifying the effect of ozone on the RAS solids to model activated sludge systems equipped with RAS-ozonation processes for the reduction of sludge production. A common hypothesis is that ozone only affects active biomass by promoting cryptic growth. Data from a pilot-scale study were used to test this and two other model extensions to IWA-ASM3. All model extensions were able to simulate the observed linear reduction in sludge production with increasing ozone dose when the MLVSS are kept constant. However, model simulations showed the inconsistency of the cryptic growth hypothesis with the extent of sludge reduction. The second tested model extensions assumes that ozone affects all the solids fractions (active biomass, endogenous residue, and influent inert particulate COD) equally. This extension could properly simulate the observed sludge reduction, but it failed to predict the trends in effluent BOD₅, ATP/VSS, and nitrification rates. A third tested model extension, which performed better, assumes that biomass is inactivated at a specific rate higher than the specific rate of transformation by ozone of the other solids fractions. Finally, the predictions from this model extension were most accurate if either (i) the nitrifiers were inactivated at a lower rate than heterotrophs, (ii) the nitrifiers model parameters (e.g., maximum growth rate) were changed under ozone (i.e., metabolic adaptation, (iii) or both.

Key words | nitrification, ozone, process modeling, sludge production, activated sludge model

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INTRODUCTION

The cost of excess sludge disposal around the world has increased considerably, forcing activated sludge treatment plants to reduce sludge production by adopting new technologies. Among them, the ozonation of return activated sludge (RAS) has had some success in a number of installations in Europe and Asia, where a typical installation would guarantee a reduction in sludge production of 40%. However, the explanation of the effect that ozone has on sludge remains a matter of debate among practitioners, limiting the predictability of the performance of new installations. A first hypothesis on the effect of ozone commonly described in the literature is that it promotes biomass cryptic growth by attacking cellular integrity (Chu *et al.* 2009). Although probably valid, this interpretation seems to be incomplete in light of modeling results using current IWA activated sludge

models (e.g., IWA-ASM3; Gujer *et al.* 1999). Using these models, the proportion of microbial biomass in the mixed liquor volatile suspended solids (MLVSS) is predicted to be around 40–50% for a typical municipal activated sludge treatment plant. Considering a 40% reduction in sludge production, it appears that the ozone process would dramatically limit the growth of bacteria suggesting that process failures are likely, which is not observed in the field. A second hypothesis on the effect of ozone is that it transforms (not oxidizes) RAS solids COD into soluble and particulate biodegradable COD. COD solubilisation upon exposure to ozone is well known (Dytczak *et al.* 2007; Chu *et al.* 2009). In this hypothesis, the sludge production is reduced because, as the transformed COD is consumed by bacteria, a proportion “1–Yield” is lost. Furthermore, cryptic growth could be

accounted for by the transformation of biomass COD into substrate COD and its subsequent re-consumption by the biomass. A third hypothesis on the effect of ozone on the RAS solids is that it could inactivate (kill) bacterial biomass as it does for disinfection processes (Kobayashi *et al.* 2001; Lee *et al.* 2005). Note that this hypothesis is not independent from the second hypothesis.

The current work aims at investigating which one or which combination of hypotheses has the highest explanatory power. It also aims at clarifying if all COD fractions of the solids are affected similarly by the ozone. To this end, the analysis of results from a pilot-plant study is presented. In addition to the common measurements of the reduction of excess sludge production and of the concentrations of soluble COD and BOD₅ in the effluent, three other sludge properties were measured: maximum nitrification rate, ATP content (related to the level of active biomass) and anaerobic digestibility of the excess sludge. As the various modeling hypotheses have specific predictions on the biomass content of the sludge, the latter three

measurements provided important information for their accurate differentiation.

MODEL PRESENTATION

The three model configurations that were tested are simple extensions of the IWA-ASM3 model (Gujer *et al.* 1999). They are presented in Gujer Matrix format in Table 1; note that natural decay is the same as in IWA-ASM3 and is presented only for reference purposes. The notation follows the same one used in the initial presentation of ASM3 (Gujer *et al.* 1999). The model extensions can be referred to as (1) *Transformation of Biomass Only* (i.e., cryptic growth), (2) *Equal Transformation of All Solids COD Pools* (this extension is a simplified version of the model of Manterola *et al.* (2001)) (3) *Inactivation of Biomass and Transformation of Non-Biomass Pools* ($X_S + X_I$). The transformation processes describe the conversion of the solids COD into fractions of substrate COD (both soluble [f_{SS}] and particulate [$1 - (f_{SI} + f_{SS} + f_{OX})$]) and

Table 1 | Gujer matrix of the tested model extensions to the IWA-ASM3 model to describe the effect of ozone contactors

Process	COD Pools					Rates				Tested Models		
	X_H	X_{STO}	X_A	X_I	X_S	S_I	S_S	SO_2	SO_3	1	2	3
Natural decay*												
1) Heterotrophs	-1			f_d				$-(1-f_d)$	$b X_H$	■ [†]	■	■
2) Storage		-1						-1	$b X_{STO}$	■	■	■
3) Nitrifiers			-1	f_d				$-(1-f_d)$	$b X_A$	■	■	■
Transformation due to ozone												
4) Heterotrophs	-1				$1 - (f_{SI} + f_{SS} + f_{OX})$	f_{SI}	f_{SS}		$-f_{OX}$	$ko_3^{\ddagger} X_H$	■	■
5) Storage		-1			$1 - (f_{SS} + f_{OX})$		f_{SS}		$-f_{OX}$	$ko_3 X_{STO}$	■	■
6) Nitrifiers			-1		$1 - (f_{SI} + f_{SS} + f_{OX})$	f_{SI}	f_{SS}		$-f_{OX}$	$ko_3 X_A$	■	■
7) Inert Solids				-1	$1 - (f_{SI} + f_{SS} + f_{OX})$	f_{SI}	f_{SS}		$-f_{OX}$	$ko_3^{\ddagger, \bullet} X_I$	■	■
8) Substrate Solids					$-(f_{SI} + f_{SS} + f_{OX})$	f_{SI}	f_{SS}		$-f_{OX}$	$ko_3 X_S$	■	■
Inactivation due to ozone												
9) Heterotrophs	-1			$f_d - f_{SI}$	$1 - (f_d + f_{SS} + f_{OX})$	f_{SI}	f_{SS}		$-f_{OX}$	$bo_3^{\bullet} X_H$		■
10) Storage		-1			$1 - (f_{SS} + f_{OX})$		f_{SS}		$-f_{OX}$	$bo_3 X_{STO}$		■
11) Nitrifiers			-1	$f_d - f_{SI}$	$1 - (f_d + f_{SS} + f_{OX})$	f_{SI}	f_{SS}		$-f_{OX}$	$bo_3 X_A$		■

*Decay is presented simply as reference

† ■: Means that the process was included in the model extension.

‡ Definition of rate constants in Model Extensions 1 and 2: $ko_3 = ko_{3, solubilisation} / (f_{SI} + f_{SS})$; $ko_{3, solubilisation}$ = measured.

• Definition of rate constants in Models Extensions 3: $ko_3 = [ko_{3, solubilisation} / (f_{SI} + f_{SS}) - bo_3 \times f_{biomass}] / [1 - f_{biomass}]$; $ko_{3, solubilisation}$ = measured; $f_{biomass} = (X_H + X_A + X_{STO}) / (X_H + X_A + X_{STO} + X_S + X_I)$; $bo_3 = k_{treated} \times f_{inactivation}$; $k_{treated}$ = measured.

non-biodegradable soluble COD (f_{SI}) with a small fraction (f_{OX}) of the COD being oxidized. For simplicity and because it is not possible to really differentiate them at this time, the fractions (f_{SI} , f_{SS} , and f_{OX}) considered in every transformation processes are the same (Table 1). The transformation of slowly biodegradable substrates (X_S) was described differently than the transformation of the other fractions since it was assumed that a fraction “ $1 - (f_{SI} + f_{SS} + f_{OX})$ ” would be generated for each unit (-1) of X_S transformed (Table 1). For the inactivation of the biomass of both the heterotrophs (X_H) and nitrifiers (X_A), it was assumed that the same non-biodegradable particulate fraction of biomass formed during decay (f_d) was also generated. It was further assumed that the non-biodegradable soluble fraction (f_{SI}) was generated from the non-biodegradable fraction of biomass. Note that for transformation and inactivation, the storage pool (X_{STO}) was assumed not to generate non-biodegradable fractions.

Two new rate constants are introduced with the proposed modeling framework: the transformation rate constant (ko_3) and the inactivation rate constant (bo_3). The two rate constants could not be measured directly in our experiments, but they are related to two specific rates readily available from the pilot-reactor operation data: the fraction of the aerated COD solids solubilised per day ($ko_3, solubilisation$) and the fraction of aerated COD solids inventory treated by ozone per day ($k_{treated}$). Assuming (1) that the rate of solubilisation is a good measure of the extent of the ozone reaction, and (2) that it is proportional to the fraction of biomass inactivated ($f_{inactivation}$) upon exposure to ozone, we get the following relationships:

$$\begin{aligned} \text{Extensions 1 \& 2: } ko_3 &= ko_3, solubilisation / (f_{SI} + f_{SS}) \\ \text{Extension 3: } ko_3 &= [ko_3, solubilisation / (f_{SI} + f_{SS}) \\ &- bo_3 \times f_{biomass}] / [1 - f_{biomass}] ; bo_3 = k_{treated} \times f_{inactivation} \\ \text{where } f_{biomass} &= (X_H + X_A + X_{STO}) / (X_H + X_A + X_{STO} \\ &+ X_S + X_I) \end{aligned}$$

EXPERIMENTAL APPROACH

Pilot-scale study

The accuracy of the model extensions at describing the experimental data was tested by analysing the results of a pilot-scale study conducted at the Régie d'Assainissement des Eaux du Bassin LaPrairie (RAEBL) wastewater treatment plant near Montréal (Canada) during Fall 2009. This municipal activated sludge plant has the particularity of receiving approximately 50% of its COD load from two industrial

sources: a paper products plant and a food processing plant with a roughing biological treatment. The RAEBL plant produces 1.25 g-sludge/g-BOD₅ at a volatile fraction (%VSS) of 79%, which is much higher than the typical 0.3–0.5 g-sludge/g-BOD₅ often observed at municipal treatment plants (Tchobanoglous *et al.* 2003). This is due to the high influent fraction of non-biodegradable particulate COD which was found during this study using the IWA-ASM3 model to be 46% (typical $f_{Xi} \approx 10\text{--}20\%$) (Gujer *et al.* 1999; Koch *et al.* 2001). Finally, the high inert particulate COD fraction could be due in part to the lack of primary clarifier.

Two parallel pilot-scale reactors (ozonated test and control) were built outfitted with a 1 m³ aeration tank (split in two by a baffle with a 1/5 area opening) and a 0.7 m³ settling tank. The influent wastewater was pumped from after the full-scale aerated grit chamber to an overflowing distribution tank with a few minutes hydraulic residence time (HRT), and then it passed through a 5-mm screen in line to the reactors. The average aerated HRTs of the units were 13.7 h and the average aerated solids residence time (SRT) of the control reactor was 5.9 days (including effluent solids). The ozonated test system was equipped with an ozone contactor from Air Liquide Canada, comprised of an ozone dissolution loop with a venturi and a vertical gas separator tube. A constant portion of the RAS flow was treated in the ozone contactor, but the ozone dosage in the contactor was varied to change the COD solubilisation rate. The dosage was measured using ozone gas analyzers located on the O₃-generator exit gas stream and O₃-contactor exit gas stream. Finally, the test reactor was operated such that the MLVSS was kept the same in both reactors.

Analyses

The analysis of (volatile) suspended solids ([V]SS), COD, nitrogen species and BOD₅ were performed on 24-hour composite samples of the influent stream and effluent streams by using the methodology described in standard methods (APHA 1995). Ammonia (NH₄⁺), nitrite (NO₂⁻), and nitrate (NO₃⁻) were measured in microplate-scale version of colorimetric methods by the 2-phenylphenol-nitroprusside, N-(1-naphthyl)-ethylenediamin dihydrochloride and salicylic acid methods, respectively (Vendrell & Zupancic 1990; Rhine *et al.* 1998). The low food-to-microorganism (F/M) bioassay described by Water Environment Research Foundation (WERF) was used to measure the nitrifier's activity (Melcer *et al.* 2003). In this procedure approximately 2000 mg/L of MLSS was incubated with NH₄⁺ at constant pH, and the NO₂⁻ and NO₃⁻ production was monitored for a few

hours. Adenosine triphosphate (ATP) was quantified by QuenchGone 21 Wastewater ATP bioluminescence kit from LuminUltra (Fredericton, New Brunswick, Canada) according to the manufacturer's protocol. Anaerobic digestion was performed by mixing a portion of mixed liquor with anaerobic biomass and monitoring for three weeks the production of CH_4 by pressure difference and gas chromatography.

Model simulations and calibration

Model simulations were performed using the solver Aquasim ver 2.0 (Reichert *et al.* 1998). The influent COD fractionation was determined by considering the measured total COD and ammonia in the influent and fitting the influent soluble COD, influent VSS, effluent soluble COD, effluent soluble BOD_5 , effluent nitrate and sludge production of the control reactor. The kinetics parameters used were the same as the ones published with the IWA-ASM3 except for the half-saturation constants for the heterotrophic organic substrate (K_S) and for the nitrifiers' ammonia substrate ($K_{A, NH}$), which were respectively calibrated to approximate the observed effluent soluble BOD_5 and ammonia concentrations in the control reactor.

Once the basic model was calibrated (COD fractionation, K_S and $K_{A, NH}$), the parameters related to the model extensions were adjusted to fit as closely as possible the observations from the ozonated test reactor. The COD solubilisation rate ($k_{O_3, \text{solubilisation}}$) and the solids ozonation rate (k_{treated}) were measured during the experiment and the fraction of COD oxidation (f_{OX}) was determined from the amount of ozone utilized and the theoretical stoichiometry. Once these parameters were set, it was observed that the remaining parameters could be adjusted sequentially as follows: (I) (f_{SI}/f_{SS}) was adjusted to fit the effluent inert soluble COD, (IIa-only for Model Extension 3) $f_{\text{inactivation}}$ was adjusted to the maximum level that could maintain good nitrification, and (IIb-for all extensions) f_{SS} was adjusted to match the level of sludge reduction observed.

RESULTS AND DISCUSSION

Initial model simulations

An extensive simulations study found that, for operations at constant MLVSS, the reduction in sludge production is linear with respect to the solids inventory solubilisation rate constant ($k_{O_3, \text{solubilisation}}$) for Model Extension 1 and 2 (data not presented). The same applies to Model Extension 3 if it is assumed that the inactivation rate constant (b_{O_3}) is propor-

tional to the solubilisation rate constant. Previous studies by others have reported a decrease in sludge production linearly related to the ozone utilized when the MLVSS was kept constant (Dytczak *et al.* 2007; Paul & Debellefontaine 2007; Manterola *et al.* 2008). In these studies, it was observed that the amount of COD solubilised was linearly related to the dose of ozone contacted with the solids, even if a non-productive (no solubilisation) low dose is sometimes observed (i.e., it is linear after the non-productive dose is met) (Paul & Debellefontaine 2007; Manterola *et al.* 2008). In our pilot-scale experiment, we also observed a linear solubilisation of COD with respect to the ozone dose (data not shown). Consequently, this modeling exercise shows that the linear reduction of sludge production in function of the ozone dose is an intrinsic property of the system. Furthermore, it suggests that the efficiency of sludge reduction (mass of sludge production reduced per mass of ozone utilized) is as much a function of the efficiency of the ozone reactions as it is a function of the activated sludge and biomass characteristics. For example, conventional and nitrification/denitrification activated sludge systems have different sludge yields, which leads to a higher efficiency for the system with the lower yield (e.g., Dytczak *et al.* 2007). A complete discussion of this point is beyond the scope of this paper.

Pilot-scale study

The pilot-study was conducted in Fall 2009. Four levels of ozone treatment (including no ozone treatment) were tested during the study, each with a test period of 3–4 weeks (Phase I to IV in increasing ozone dosage). The average SRT for the control reactor remained around 5.9 days while it reached approximately 12 days for the ozonated reactor during the last test phase with the highest ozone treatment level (Figure 1a). The aerated MLSS concentrations in both reactors were kept at similar concentrations throughout the study (Figure 1b). The %VSS remained similar in both reactors throughout the study at 82% with a slight decrease to 78% in the ozonated reactor towards the end of the last test period (Figure 1b). Finally, the temperature of the process went from 18°C to 15°C during the study period, which probably had minimal effects on the activated sludge operation and performance.

The sludge production was evaluated by monitoring the amount of solids wasted through either the WAS or effluent flows and by calculating differences in total solids inventory (aeration tank and clarifier). The reduction in sludge production was evaluated by averaging the sludge production per COD loaded in each phase and calculating the ratio between the two reactors (production in ozonated reactor/production

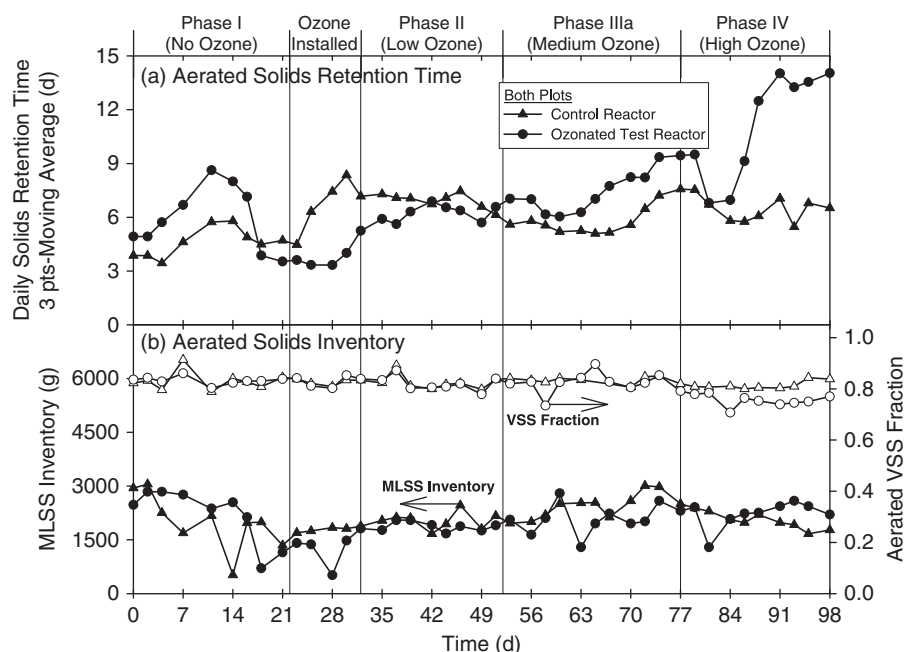


Figure 1 | Aerated solids retention time (a) and mixed liquor suspended solids (b) during the course of the pilot experiment.

in control reactor; hereafter the sludge production ratio, SPR). SPR reduction with respect to the sludge solubilisation rate was observed to fall close to the linear relationship suggested by the initial modeling exercise with a slope for the SPR_{VSS} and SPR_{SS} of -7.35 days and -6.48 days, respectively (Figure 2). In the last test period, when the highest level of ozone treatment was applied, the SPR reached 0.54 as evaluated for VSS production and 0.56 for total solids production, which was slightly higher than what

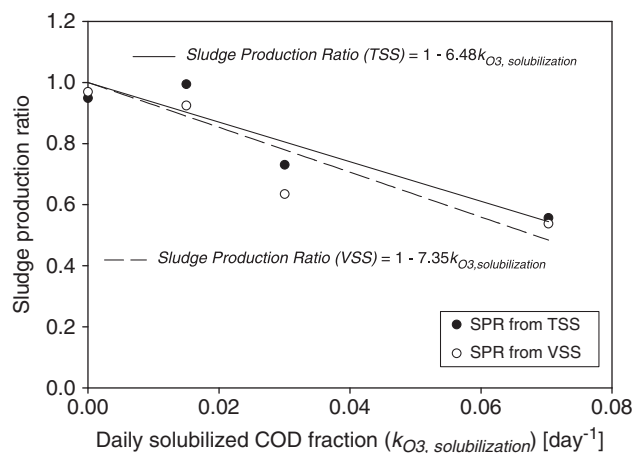


Figure 2 | Sludge production ratio (SPR; Production in Ozonated Test Reactor/Production in Control Reactor) as function of COD solubilisation rate (fraction of aerated solids COD inventory solubilised in a day). Lines are ordinary least square regressions forced through 1.

was evaluated by linear regression. Finally, during the experiment, approximately 0.17 g- O_3 per day was necessary to reduce excess sludge production per day by 1 g-TS. This is slightly higher than previously observed ratios (Chu *et al.* 2009) probably due to the difficulty of thickening the solids at the bottom of the clarifier (RAS suspended solids remained on average around 3300 mg/L, 1.66 times the average MLSS concentration).

The effluents of both reactors were also monitored during the study period. We report here only the average soluble COD, soluble BOD_5 and nitrate concentrations observed during the last experimental phase (Phase IV; Table 2). Ammonia nitrogen was never detected for both reactors, suggesting a complete nitrification. However, the nitrate concentration in the effluent of the ozonated reactor was higher than in the control reactor by 58% (Table 2), suggesting that more ammonia was available for nitrification. The effluent soluble COD and soluble BOD_5 were higher in the ozonated reactor by 106% and 161%, respectively (Table 2). A higher soluble COD was probably due to the formation of non-biodegradable soluble COD during sludge solubilisation by ozone. However, a higher soluble BOD_5 is indicative of an increased effective decay rate of the heterotrophic biomass in the ozonated reactor or the formation of slowly degradable soluble microbial product (Grady *et al.* 1999).

In addition to typical operation parameters, a number of sludge characteristics were determined for this study

Table 2 | Comparison between observed experimental data from pilot-scale reactors during the last experimental phase and the best simulations from the calibrated model extensions to the WWA-ASM3

	Experimental Observations		Control Reactor ASM3 Model†		Model Extension 1		Model Extension 2		Model Extension 3	
	Control	ozonated	Change	Value	Change	Value	Change	Value	Change	Value
Specific Parameters										
f_{SI}/f_{SS}	NA†	NA†		0.45		0.45		0.45		0.48
f_{SS}	NA	NA		0.125		0.432		0.382		0.382
$f_{inactivation}$	NA	NA		NA		NA		0.41		0.41
k_{O_3} (day ⁻¹)	NA	NA		0.388		0.112		0.100		0.100
b_{O_3} (day ⁻¹)	NA	NA		NA		NA		0.248		0.248
Sludge Reduction										
SPR _{VSS} vs k_{O_3} , <i>solutibilisation</i> (day)		-7.35 ± 1.19‡		NA		-1.72		-7.35		-7.35
Effluent Qualities										
Soluble COD (mg/L)	30 ± 3	63 ± 3	+111%	32	+269%	118	+88%	66	+106%	66
Soluble BOD ₅ (mg/L)	2.7 ± 0.8	7.0 ± 1.8	+163%	3.0 ●	+1716%	54.8	+16%	7.0	+132%	7.0
Effluent NO ₃ ⁻ (mg-N/L)	7.6 ± 1.1	12.0 ± 0.8	+58%	8.0	-100%	0.0	+14%	9.1	-18%	6.6
Sludge Qualities										
ATP Level (mg ATP/g VSS)										
Heterotroph/VSS	1062 ± 105	766 ± 189	-28%	12.8	-73%	3.7	+48%	18.7	+2%	12.9
Nitrifiers/VSS										
Max. NO ₃ ⁻ Prod. (g N/kg VSS/day)										
Nitrifiers/VSS (10 ⁻² g X _A VSS/g VSS)										
Nitrifiers/VSS	46.2 ± 2.8	37.0 ± 2.8	-20%	0.806	-100%	0.000	+15%	0.924	-37%	0.506
Anaerobic Digestion										
Anaerobic Digestion (% mineralized)										
Anaerobic Digestion	43.2 ± 2.4	45.8 ± 0.2	+6%	43.2***	-15%	36.5	+55%	49.7	+6%	46.0
Model: 1-X _I /VSS		NA		17.2	-56%	7.5	+24%	26.7		21.3

†Default model parameters were used except for $K_{NH_4} = 0.05$ and $K_S = 10.31$; influent total COD: 405 mg/L, and influent NH₄⁺-N: 14.2 mg/L

‡NA: not applicable

§ ± standard error

● Model simulations were performed in COD units and 0.6 g BOD₅/g COD was assumed (Grady *et al.*, 1999).

***To fit the observed mineralization rate, 31.4% of X_I was assumed to be anaerobically biodegradable.

(Table 2). The maximum nitrification rate as defined by the maximum nitrate production rate following an ammonia spike in a lab-scale assay was determined five times in a period spanning the second half of Phase III and the first half of Phase IV. The average nitrification rate was reduced by 20% in the ozonated test reactor compared to the control reactor (Table 2), which indicates that the level of nitrifiers in the MLVSS was lower in the test reactor. During Phase IV, the average level of ATP per VSS was 28% lower in the test reactor, suggesting that also the overall biomass level in the ozonated test reactor was lower. Finally, the anaerobic biodegradability of the sludge was maximally 6% higher in the test reactor (Table 2).

Evaluation of model extensions

The influent COD fractionations and two half-saturation constants (K_S and $K_{A, NH}$) was determined by considering the measured total COD and NH_4^+ in the influent and fitting the influent soluble COD, influent VSS, effluent soluble COD, effluent soluble BOD_5 , effluent NH_4^+ , effluent NO_3^- and sludge production of the control reactor. The influent COD fractions were found to be: soluble readily substrate (S_s) 44%, particulate slowly degradable substrate (X_s) 6%, inert soluble COD (S_I) 3%, and inert particulate COD (X_I) 46%. The high level of inert particulate COD (X_I) in the influent of the RAEBL wastewater treatment plant is helpful in evaluating the model extensions because the level of active biomass in the sludge as calculated by the model is pretty low, which amplifies the biomass dynamics. Consequently, the three model extensions can produce clearly different simulation predictions for the effluent and biomass levels in the MLVSS between the ozonated test and the control reactors. These differences provide crucial evidence for the evaluation of the model extensions' accuracy.

Simulations of Model Extension 1 *Transformation of Biomass Only* found the steepest possible slope of SPR_{VSS} vs specific ozone solubilisation rate to be -1.72 days, a much lower efficiency than the slope of -7.35 days observed in the pilot-scale experiment (Table 2). Other trends in effluent and sludge characteristics were also not met (Table 2). These results suggest that the cryptic growth hypothesis alone is insufficient to predict the performance of RAS-ozonation processes for sludge reduction.

Simulations of Model Extension 2 *Equal Transformation of All Solids COD Pools* reproduced the performance of reduction in SPR_{VSS} vs specific ozone solubilisation rate (slope -7.35 days) observed experimentally (Table 2). However, this model extension predicts an increase in hetero-

trophic and nitrifying biomass levels. Since the level of ATP/MLVSS and the specific nitrification rate (NO_3^- formation rate/MLVSS) decreased, this model extension is less likely. Furthermore, this model extension failed to predict the increase in soluble BOD_5 in the effluent. Therefore, it seems that Model Extension 2 has deficiencies that would be difficult to fix within this modeling framework.

Model Extension 3 *Inactivation of Biomass and Transformation of Non-Biomass Pools* was able to produce model simulations that reproduced the increase in effluent soluble BOD_5 and the reduction in specific maximum nitrification rate (observed in the model as a reduction in the nitrifying biomass; Table 2). The simulation result reported in Table 2 was the one with the highest biomass inactivation factor ($f_{inactivation} = 0.41$) that would maintain a reasonable nitrification activity. The level of heterotrophic biomass from this simulation is the same as in the control, which is an improvement over Model Extension 2 (i.e., lower biomass level than for Model 2 simulation). The accuracy of the simulation would be improved by lowering the heterotrophic biomass level without lowering further the nitrifying biomass. This simulation result could be achieved either (i) by assuming lower inactivation rates for nitrifiers than for ordinary heterotrophs as suggested before (Boehler & Siegrist 2004), (ii) by assuming that the nitrifying biomass metabolically adapts to the ozone treatment which changes model parameters such as maximum specific growth rate, or (iii) by assuming both. At this point, the data are insufficient to make any recommendations on how to increase the accuracy of the predictions.

In the simulation of Model Extension 3, the concentration of nitrate in the effluent is slightly reduced due to a reduction in the extent of nitrification. This deficiency would be solved (i) by including the improvements suggested above, and (ii) by considering that the inert particulate COD from the influent contains a certain quantity of ammonia nitrogen released upon the action of ozone. At the time of writing this paper, the level of nitrogen in the influent solids remained to be determined. Yet, it appears that the inaccuracies of Model Extension 3 could be solved while the ones of the other model extensions could not. In conclusion, it seems that Model Extension 3 is the most promising to predict the performance of the activated sludge treatment system and the changes in sludge characteristics.

CONCLUSION

- We tested the accuracy of three model extensions to IWA-ASM3 to predict the performance of an activated sludge

system submitted to RAS-ozonation for the reduction of sludge production.

- A pilot-scale study was used as the case study for model evaluation.
- The model extension which assumed that the biomass was inactivated at a higher rate than the rest of the solids fraction were transformed by ozone showed the best performance at fitting the data from a pilot-scale study.
- Although the trends were more correctly predicted by this model extension, some inaccuracies in model predictions of the nitrification rates were observed, suggesting that further model improvement are necessary to describe the nitrifiers.

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