Modeling waste stabilization ponds with an extended version of ASM 3

T. Gehring[1], J. D. Silva[3], O. Kehl[2], A. B. Castilhos Jr.[4], R. H. R. Costa[4], F. Uhlenhut[5], J. Alex[6], H. Horn[2], M. Wichern[1]

[1]Institute of Environmental Engineering, Ruhr-Universität Bochum, 44780, Bochum, Germany (E-mail: siwawi@rub.de)
[2]Institute of Water Quality Control, Technische Universität München, Am Coulombwall, 85748, Garching, Germany (E-mail: wga@bv.tum.de)
[3]Programa de Pós-Graduação em Engenharia Ambiental, Universidade Regional de Blumenau, 89030-000, Blumenau, Brazil (E-mail: dias_joel@hotmail.com)
[4]Departamento de Engenharia Sanitária e Ambiental, Universidade Federal de Santa Catarina, Campus Univesitário, Trindade, 88010-970, Florianópolis, Brazil (E-mail: borges@ens.ufsc.br, rejane@ufsc.ens.br)
[5]FH Oldenburg/Ostfriesland/Wilhelmshaven, FB Technik; Institut für Umwelttechnik – EUTEC, 26723 Emden, Germany (E-mail: uhlenhut@fho-emden.de)
[6]ifak - Institut für Automation und Kommunikation e.V. Magdeburg, 39106, Magdeburg , Germany (E-mail: jens.alex@ifak.eu)

Abstract
In this paper an extended version of IWA's Activated Sludge Model No 3 (ASM3) was developed to simulate processes in waste stabilization ponds (WSP). The model modifications included the integration of algae biomass and gas transfer processes for oxygen, carbon dioxide and ammonia depending on wind velocity and an simple ionic equilibrium. The model was applied to a pilot-scale WSP system operated in the city of Florianópolis (Brazil). The system was used to treat leachate from a municipal waste landfill. Mean influent concentrations to the facultative pond of 1456 g_{COD}/m^3 and 505$g_{NH4\text{-}N}/m^3$ were measured. Experimental results indicated an ammonia nitrogen removal of 89.5% with negligible rates of nitrification but intensive ammonia stripping to the atmosphere. Measured data was used in the simulations to consider the impact of wind velocity on oxygen input of 11.1 to 14.4 $g_{O2}/(m^2{*}d)$ and sun radiation on photosynthesis. Good results for pH and ammonia removal were achieved with mean stripping rates of 18.2 and 4.5 $g_N/(m^2{*}d)$ for the facultative and maturation pond respectively. Based on measured chlorophyll *a* concentrations of up to 1205 μgL^{-1} and depending on light intensity and TSS concentration it was possible to model algae concentrations with good quality in different depths of the tanks.

Keywords
Mathematical Modeling; Waste Stabilization Ponds; Ammonia Stripping; ASM; Landfill Leachate;

INTRODUCTION

Waste stabilization ponds are still one common option for treatment of different effluents, due to their lower costs and operational simplicity. But this simplicity doesn't reflect in a complete understanding of processes that occur in wastewater treatment ponds. Although biochemical processes like nitrification or denitrification can be expected to be the same as in activated sludge systems other highly important factors are often not considered in detail. Environmental impacts like sun radiation, wind, stripping of ammonia and carbon dioxide, biological processes and also hydrodynamics are still not completely understood or are difficult to validate with experimental data. Only few mathematical models have been publicized to simulate such complex systems (e.g. in Buhr and Miller, 1983). Juspin et al. (2003) developed an adaptation from the River Water Quality Model No.1 (Reichert et al., 2001) to simulate high rate algae ponds including the influence of daily light variance.

Solar radiation is the principal factor influencing algae growth that occurs mostly in the superficial layer of the pond. In depths less than 50 cm high oxygen production can be expected (von

Sperling and Chernicharo, 2005). Although algae photosynthesis is identified as one of the main sources of oxygen input in facultative WSP's, superficial oxygen transfer has been found to be a relevant factor under conditions with high wind velocities (Ro et al., 2007). In the river water quality model QUAL2Kw (Pelletier and Chapra, 2008) wind influence is considered as one oxygen source. Anyway, up to now wind impact is rarely quantified and included into mathematical models for ponds.

Ammonia nitrogen removal in waste stabilization ponds can be result of both, nitrification followed by denitrification (Hurse and Connor, 1999; Zimmo et al., 2003; Camargo Valero and Mara, 2007), or free ammonia release to the atmosphere (Pano and Middlebrooks et al., 1982; Smith and Arab, 1988; Shilton, 1996). It can be observed that in rich ammonia environments the importance of ammonia release may increase, considering that the mass transfer of the free ammonia is directly proportional to its concentration in the liquid phase.

For landfill leachate ponds desorption from the superficial surface was observed as the main pathway to ammonia stripping, if compared with the release through air bubbles. Turbulence in the liquid-atmosphere interface is a determinant factor (Smith and Arab, 1988). Ni (1999) analyzed 30 mechanistic models of ammonia release of manure. He revealed the importance of air velocity to determine the mass transfer coefficient and the necessity to consider carbon dioxide release depending on dynamic pH calculation. Wett and Rauch (2003) described the importance of inorganic carbon balance for the nitrification process in high concentrate ammonia wastewaters. They found it necessary to include the stripping of carbon dioxide in the ASM models. Husted et al. (1991) described the buffer capacity of ammonia, bicarbonate and a solid phase carbonate as most important components for pH and ammonia volatilization.

In this paper the influence of sun and wind as well as stripping of ammonia and carbon dioxide is modeled in combination with degradation processes describing carbon removal, nitrification and denitrification. Simulation results were validated with detailed measurement data of a pilot scale pond system in the south of Brazil.

MATERIAL AND METHODS

Pilot ponds description

The experiments were conducted in the city of Florianópolis in the south of Brazil (Santa Catarina) by Silva (2007). A pilot-scale system with three ponds in line (anaerobic-facultative-maturation) was used to treat leachate from an approximately 15 years old municipal waste landfill. This paper will refer just to the facultative (Fac) and maturation (Mat) pond. Each pond has a superficial area of 1.2 m² and a depth of 1.0 m resulting in a hydraulic retention time of 18 days. The main characteristics of the influent and effluent of both ponds are summarized in Table 1. Parameters like chlorophyll *a*, oxygen, pH and temperature were measured in three different depths: 0.2 m, 0.5 m and 0.8 m, that will be related here as top, middle and bottom layer. Wind was measured at an automatic hydrological station. Sun radiation measurements were collected in a solar station from the project BSRN – Baseline Surface Radiation Network / WMO – World Meteorological Organization (27°38'S, 48°30'O). The measurement period lasted from July 2005 to February 2006, in total 243 days.

Mathematical Model

ASM 3 (Gujer et al. 1999) was used with some extensions that will be described here. The model was implemented in the SIMBA 4.2 (Alex, 2005).

Hydraulic Concepts

Due to the reduced scale of the pilot ponds and strong wind influence it was assumed that each pond can be modeled by three completely mixed reactors. These CSTR's represented the depth of the system, with stratified layers: bottom, middle and top. The flow was constant with 60 L/d. Sedimentation was neglected in the model.

Table 1: Mean influent and effluent parameters of the system

Parameter	Unit	Affluent Fac	Effluent Fac			Effluent Mat		
COD	g/m^3	1456	1233			743		
COD filtered	g/m^3	n.m.	1065			504		
COD/BOD_5	-	6.8	7.3			7.3		
Total suspended solids	g/m^3	301	239			146		
Total Ammonia nitrogen	g/m^3	505	208			53		
Nitrate nitrogen	g/m^3	8	6			4		
Depth of measurements			*Top*	*Middle*	*Bottom*	*Top*	*Middle*	*Bottom*
Chlorophyll *a*	µg /L	n.m.	79	61	57	163	110	101
Dissolved Oxygen	g/m^3	n.m.	3.9	3.7	3.4	4.5	4.2	3.9
PH	-	8.73	8.8	8.8	8.8	9.1	8.9	8.7
Temperature	^oC	25.9	26.9	25.1	24.8	25.7	23.4	23.0

n.m. = parameter not measured

Algae Processes

According to experiments each g_{COD}/m^3 of algae had a concentration of 12 µg/L chlorophyll *a*. Two different growth processes, based on ammonia and nitrate, and an endogenous respiration process were included in ASM 3 analogue to RWQM No. 1 (Reichert et al., 2001). Below the rates [g/(m^3*d)] for algae growth on ammonia and nitrate as well as for algae respiration are given.

$$rate_{alg\_growth\_snh} = k_{m\_alg} \times \frac{S_{NO}+S_{NH}}{K_{N,ALG}+S_{NO}+S_{NH}} \times \frac{S_{NH}}{K_{NH_4,ALG}+S_{NH}} \times \frac{I_{AV}}{K_I} \times \exp^{(1-\frac{1}{K_I})} \times X_{ALG} \quad (1)$$

$$rate_{alg\_growth\_sno} = k_{m\_alg} \times \frac{S_{NO}+S_{NH}}{K_{N,ALG}+S_{NO}+S_{NH}} \times \frac{K_{NH_4,ALG}}{K_{NH_4,ALG}+S_{NH}} \times \frac{I_{AV}}{K_I} \times \exp^{(1-\frac{1}{K_I})} \times X_{ALG} \quad (2)$$

$$rate_{alg\_resp} = k_{resp\_alg} \times \frac{S_O}{S_O+K_{O,ALG}} \times X_{ALG} \quad (3)$$

where k_{m_alg} is the algae growth rate [d^{-1}], k_{resp_alg} the algae endogenous respiration rate [d^{-1}], $K_{N,ALG}$ saturation constant to ammonia and nitrate [g/m^3], K_{NH4_ALG} the ammonia inhibition constant [g/m^3], I_{AV} the available light radiation [W/m^2], K_I the light limitation and saturation coefficient [W/m^2], S_{NO} the nitrate concentration [g/m^3], S_{NH} the ammonia concentration [g/m^3], S_O the oxygen concentration [g/m^3] and X_{ALG} the algae biomass concentration [g_{COD}/m^3].

Light attenuation through depth is usually described by the Beer's Law, where the attenuation coefficient is considered by the absorption properties of water. That means light absorption is mainly determined by gilvin (dissolved yellow matter), algae and tripton (inanimate particulate matter) concentrations (Curtis et al., 1994). Heaven et al. (2005) observed that only few light attenuation coefficients for WSP's have been published until yet and highlighted their importance to mathematical description of algae processes. Photosynthetic Available Radiation - PAR – was assumed to be 0.47 from total light radiation. Attenuation across the water depth was calculated from the Beer-Lambert Equation to determine the available light radiation I_{AV}. The light attenuation parameter K_d [m^{-1}] was defined as function of the mixed liquor suspended solid in the tank:

$$I_{av} = 0.47 \times I \times e^{-K_d \times H} \quad (4) \qquad K_d = \alpha_1 + \alpha_2 \times X_{TSS} \quad (5)$$

where I is the light radiation [W/m^2], H is the depth [m], α_1 is the light attenuation constant from water colour and turbidity [m^{-1}], α_2 is the light attenuation constant factor from suspended solids [m^3/(g*m)] and X_{TSS} the total mixed liquor solids concentration [g/m^3].

Ionic Equilibrium Processes

Free ammonia concentration S_{NH3} [g/m^3] and carbon dioxide S_{CO2} [mol$_C$/m^3] were added into the model as new state variables. The ionized ammonium concentration S_{NH4} [g/m^3] resulted from the difference of S_{NH} (total ammonia) and S_{NH3}. The concentrations of free ammonia and carbon dioxide in the liquid phase were defined through two equilibrium processes, considering the acid/base pairs: S_{CO2}/S_{ALK} and S_{NH4}/S_{NH3}. Both equilibrium equations and the equilibrium constants were set according to Reichert et al. (2001). To allow for dynamic pH calculation a charge balance was solved. The influence of ionized ammonia, nitrate and alkalinity was considered. Equivalent charges added with the dissociation products of water, H^+ and OH^-, resulted to zero.

Gas transfer Processes

Three different gas transfer processes between the top layer and atmosphere were considered to determine the mass transfer of oxygen, free ammonia and carbon dioxide. Determination of the gas transfer rate J_{gas_i} [g/(m³*d)] was done as follows:

$$J_{gas\_i} = kla_{gas\_i} \times \frac{A}{V} \times (S_{sat\_gas\_i} - S_i) \qquad (6)$$

where kla_{gas_i} is the mass transfer coefficient of the gas [m/d], A the surface area available for gas exchange [m^2], V the layer volume [m^3], S_{gas_i} the gas saturation or critical concentration between liquid and gas phases [g/m^3] and S_i the concentration in the liquid phase [g/ m^3]. This process depends on concentration gradient between atmosphere and liquid. Positive values of J_{gas_i} represent gas absorption and negative ones gas release from the liquid. The oxygen transfer coefficient, kla_{O2} [m/d], was calculated according to Ro and Hunt (2006), which developed an empirical equation based on 297 data points from transfer coefficients published in the last 50 years. This equation was recommended by the authors to be applied to WSP.

$$kla_{O2} = 0.24 \times 170.6 \times Sc^{-\frac{1}{2}} \times U_{10}^{\ 1.81} \times \left(\frac{\rho_a}{\rho_w}\right)^{\frac{1}{2}} \qquad (7)$$

Sc is the dimensionless Schmidt number, U_{10} is the velocity 10 meters over the ground [m/s], p_a and p_w are the atmosphere and water densities respectively [kg/m^3]. Saturation of oxygen was defined through an empirical equation and carbon dioxide and free ammonia saturation as function of the Henry's constant and the atmosphere pressure of the gas:

$$S_{sat\_O2} = 13.89 - 0.3825T + 0.007311T^2 - 0.00006588T^3 \quad (8) \qquad S_{sat\_gas\_i} = \frac{p_{atm\_gas\_i}}{K_H} \quad (9)$$

where T is the temperature [^{o}C], K_H the Henry constant [atm*m^3/mol] and $p_{atm\_gas\_}$ the partial pressure in the atmosphere [atm]. Wind measurements were corrected to 10 meters height with the seventh-root profile and the mass transfer coefficients from carbon dioxide and ammonia were normalized to the oxygen transfer coefficient considering the surface renewal theory (in Ro and Hunt, 2006):

$$\frac{U_z}{U_{10}} = \left(\frac{z}{10}\right)^{\frac{1}{7}} \quad (10) \qquad \frac{K_{la\_1}}{K_{la\_2}} = \left(\frac{Sc_1}{Sc_2}\right)^{-\frac{1}{2}} \quad (11)$$

U_z is the wind velocity [m/s] in the measured height z [m].

RESULTS AND DISCUSSION

Pilot ponds results

COD in the inflow to the facultative pond was fractioned as follows: Inert fraction S_I = 45 % of total COD, readily and slowly biodegradable substrate fractions were S_S=35% and X_S=20% respectively. There was no measured data available for alkalinity that was calculated from ammonia, nitrate and pH values. The measured environmental data, sun radiation and wind velocity, was applied in the simulations with mean values per hour and temperature (25°C).

COD and TSS

Results of COD removal are depicted in Figures 1c and 1d. Total suspended solids are shown in Figure 1f. In the extended ASM 3 the heterotrophic growth rate was calibrated to 0.52 d^{-1}, almost four times smaller than found for activated sludge (Gujer et al., 1999). The reduced rate could be explained by the high concentration of ammonia in the leachate (Li and Zhao, 2001). This was also reported by other authors (Yang et al., 2004; Lee et al., 2000), who published the effect of free ammonia in the liquid phase on heterotrophic growth and oxygen consumption rates. Mean concentrations of free ammonia obtained from model calculations in the facultative and maturation ponds were 11.4 and 1.5 [g/m^3] respectively. Both concentrations could explain the impact on heterotrophic growth rate.

Algae parameters

Algae concentrations in the maturation pond were higher than in the facultative pond. Mean values were 93 and 63 μgL^{-1} chlorophyll *a* respectively. These concentrations were relatively low compared with reported data in facultative ponds of 500 – 2000 μgL^{-1} (Mara et al., 1992). One reason for low chlorophyll *a* could also be free ammonia concentrations (Azov and Goldman, 1982). For model calibration only the light inhibition/saturation constant K_I was calibrated to fit algae concentration. The adopted value was 1200 W/m^2. Other algae parameters from equations (1), (2) and (3) were maintained as suggested by Reichert et al. (2001). In the absence of measurements for the light attenuation coefficients α_1 and α_2, they were adopted as 0.3 and 0.032 (Juspin et al., 2003) respectively. Simulated and measured algae concentrations are shown in Figure 1a and 1b. Despite the strong influence of depth on light availability to photosynthesis, good mixing of the three pond layers equalized algae concentrations. The increase of algae after day 150 in both ponds reflected the higher light intensities in the summer. Algae concentrations in the maturation pond were higher than in the facultative pond. The model is not able to explain concentration peaks of algae on day 235. Concentrations reached a maximum of 1205 μgL^{-1} chlorophyll *a* in the maturation pond.

Dissolved oxygen production

In the model oxygen input was considered from two different sources: algae growth and wind aeration. Low values of chlorophyll *a* suggest that wind is most important here. Model calculations showed a mean oxygen input from wind in the facultative pond of 14.4 and in the maturation pond of 11.1 $gO_2/(m^2*d)$.

pH

Simulations revealed the strong influence of the ammonia dissociation constant on pH results. For animal manure Ni (1999) found that the dissociation constant ranged from one fifth to one sixth of that for ammonia in pure water. In our model the ammonia dissociation constant was calibrated to one fifth of the value of pure water (Reichert et al., 2001), $7.75*10^{-8}$ g_H/m^3 at 20 °C. Figure 1e displays simulation and experimental results of pH in the facultative pond. The loss of ammonia and inorganic carbon through gas stripping in the facultative pond could explain slightly lower simulation values of pH in the maturation pond. Due to ammonia release pH variations in the top layer are bigger than in the other two layers.

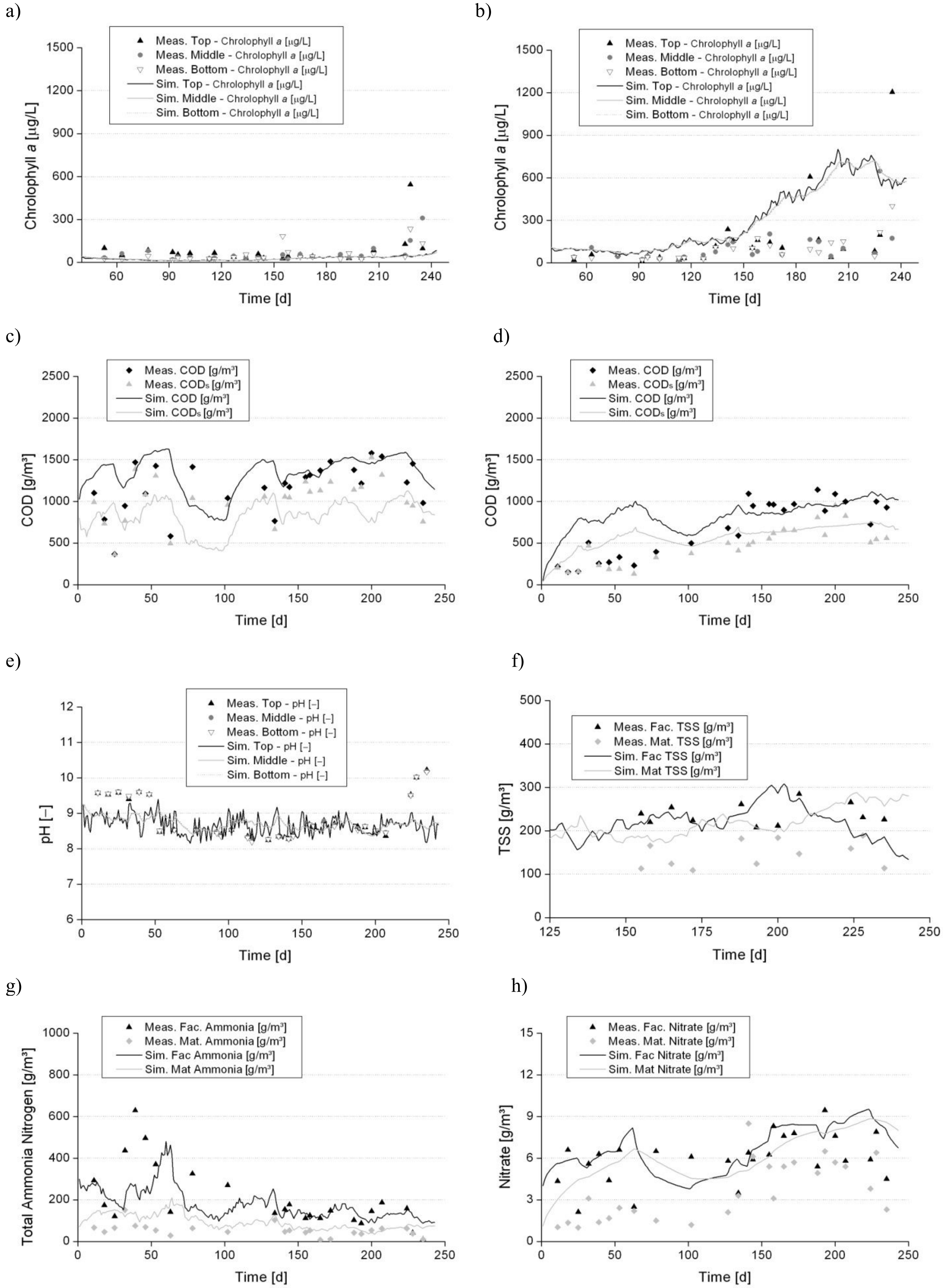

Figure 1: Measured and simulated results, chlorophyll *a* concentration Fac (a) and Mat (b), COD effluent concentrations Fac (c) and Mat (d), pH Fac (e), TSS effluent concentrations (f), total ammonia nitrogen concentrations (g) and nitrate effluent concentration (h).

Nitrification and denitrification
In both ponds nitrate concentrations showed low variations. Mean effluent values of 6 $g_{NO3\text{-}N}/m^3$ in the outflow of the facultative pond and 4 $g_{NO3\text{-}N}/m^3$ after the maturation pond were reached. Considering that according to measurement data in both ponds oxygen concentrations never fell below 1 mg/L, no denitrification occurred. Diverse factors can be related to the absence/inhibition of the nitrification process: inhibitions from sun light (Lipschultz et al., 1985; Abeliovich and Vonshak, 1993), free ammonia (Anthonisen et al., 1976, Lee et al., 2000; Wett and Rauch, 2003; Yang et al., 2004; Vadivelu et al., 2007) or inhibition of nitrification by the leachate. Therefore according to measurement data no nitrification and denitrification was considered in the simulations. Measured and simulated nitrate concentrations are depicted in Figure 1h.

Ammonia release
Simulated concentrations of ammonia effluent values showed good results for both ponds (Figure 1g). Modeling was based on a mechanistic mass transfer concept with an empirical equation for the gas transfer. Mean ammonia stripping rates were 18.2 and 4.5 $g_N/(m^2{*}d)$ in the facultative and maturation pond respectively. Unfortunately there are not many published data available to compare. Smith and Arab (1988) and Cheung et al. (1997) through experiments with landfill leachate related high ammonia stripping rates in free tanks (without aeration), that could be calculated from the published data as 45 to 142 $g_N/(m^2{*}d)$. Although initial ammonia concentrations in both studies were very similar to the present ponds (575 to 705 g/m^3), pH was adjusted above 11, what complicates to establish parallels with the results presented. The same applies to other published stripping rates from different sources and also to pH and nitrogen concentrations. Rumburg et al. (2008) reported fluxes from a plant scale anaerobic dairy waste lagoon of 2.6 to 13.0 $g_N/(m^2{*}d)$ determined through tracer experiments. And experimental stripping rates measured for domestic WSP lay very below from these values (Zimmo et al., 2003; Camargo Valero and Mara, 2007).

CONCLUSIONS
In this paper measured data and simulated results for a three stage pond system at pilot scale were presented. The system operated in the south of Brazil in the city of Florianópolis treating leachate from a waste stabilization facility. In this paper we focused on the treatment efficiency of the second stage, a facultative pond and a third stage maturation pond. Both ponds treated landfill leachate with a COD removal of 49% and an ammonia removal of 89.5%. For modeling ASM 3 was extended with additional processes to consider sun radiation, light attenuation in dependence of TSS, algae photosynthesis, algae endogenous respiration, wind velocity as source for oxygen input, gas transfer of carbon dioxide and ammonia, ionic equilibrium processes and pH calculation. Measured data and simulation results showed a) COD, TSS and nitrogen removal can be modeled well with the extended ASM3; b) pH could be well simulated with calibration of the ammonia dissociation constant; c) light attenuation can be well modeled by the Beer-Lambert Equation. However better determination of related parameters is recommended in further studies; d) mean oxygen input through wind into the two ponds is in between 11.1 and 14.4 $g_{O2}/(m^2{*}d)$; e) free ammonia concentrations seem to inhibit nitrification completely and heterotrophic biomass partly, resulting in a maximum heterotrophic growth rate of 0.5 d^{-1}; algae growth also seems to be inhibited by free ammonia concentrations; f) high ammonia stripping rates of 18.2 and 4.5 $g_N/(m^2{*}d)$ in the facultative and maturation pond were reached.

ACKNOWLEDGEMENTS
The authors would like to thank the German Research Ministry (BMBF, grants 02WA0577 and 02WA0575) and the Brazilian National Counsel of Technological and Scientific Development (CNPQ) who financed this study. And the Solar Energy Laboratory from Federal University of Santa Catarina (LABSOLAR, UFSC) that gently provided solar radiation data.

REFERENCES

Alex, J. (2005). SIMBA 4.2, software for the simulation of biological wastewater treatment. Institute for Automation and Communication e.V. (ifak), Magdeburg.

Abeliovich A., Vonshak, A. (1993). Factors inhibiting nitrification of ammonia in deep wastewater reservoirs. Wat. Res., **27**(10), 1585-1590.

Anthonisen, A.C., Loehr R.C., Prakasam T.B.S., Srinath E.G. (1976). Inhibition of nitrification by ammonia and nitrous acid. JWPCF., **48**(5),835 – 52.

Azov, Y., Goldman, J. C. (1982). Free Ammonia Inhibition of Algal Photosynthesis in Intensive Cultures. Applied and Environmental Microbiology, **43**(4), 735 – 739.

Buhr, H.O., Miller, S.B. (1983). A dynamic model of high rate algal bacterial wastewater treatment ponds. Wat. Res., **17**, 29 – 37.

Camargo Valero, M.A., Mara D.D. (2007). Nitrogen removal via ammonia volatilization in maturation ponds. Wat. Sci. Tech. **55**(11), 87 – 92.

Cheung, K. C., Chu, L. M. and Wong, M. H. (1997). Ammonia stripping as a pretreatment for landfill leachate. Water, Air, and Soil Pollution, **94,** 209 – 221.

Curtis, T.P., Mara, D.D., Dixo, N.G.H., Silva, S.A. (1994). Light penetration in waste stabilization ponds. Wat. Res., **28**(5), 1031 – 1038.

Gujer, W., Henze, M., Mino, T., van Loosdrecht, M. (1999). Activated Sludge Model No. 3, Wat. Sci. Tech., **39**(1), 183 – 193.

Heaven S., Banks C.J., Zotova E.A. (2005). Light attenuation parameters for waste stabilisation ponds. Wat. Sci. Tech., **51** (12), 143 – 152.

Husted, S., Jensen, L. S., Jorgensen, S. S. (1991). Reducing ammonia loss from cattle slurry by the use of acidifying additives the role of the buffer system. Journal of the Science of Food and Agriculture, **57**(3), 335 – 350.

Jupsin, H., Praet E., Vasel, J.-L. (2003). Dynamic mathematical model of high rate algal ponds (HRAP). Wat. Sci. Tech., 48(2), 197 – 204.

Lee, S.-M.; Jung, J.-Y.; Chung, Y.-C. (2000). Measurement of ammonia inhibition of microbial activity in biological wastewater treatment process using dehydrogenase assay. Biotechnol. Lett., **22**, 991 – 994.

Li X. Z., Zhao, Q. L. (2001). Efficiency of biological treatment affected by high strength of ammonium-nitrogen in leachate and chemical precipitation of ammonium-nitrogen as pretreatment. Chemosphere **44**, 37 – 43.

Lipschultz, F., Wofsy, S. C., Fox, L. E. (1985). The effects of light and nutrients on rates of ammonium transformation in a eutrophic river. Marine Chemistry, **16**, 341 – 329.

Mara, D.D, Alabaster, G. P., Pearson, H. W., Mills, S. W. (1992). Waste stabilization ponds. A design manual for Eastern Africa. Laggon Technology International Ltd. 121 pp.

Ni, J.Q. (1999). Mechanistic models of ammonia release from liquid manure: a review, Journal of Agricultural Engineering Research 72, 1 – 17.

Pano, A., Middlebrooks, E. J. (1982). Ammonia Nitrogen Removal in Facultative Ponds. Journal of the Water Pollution Control Federation, **4** (54), 344 – 351.

Pelletier, G.J., Chapra, S.C. (2008). QUAL2Kw theory and documentation (version 5.1), A Modeling Framework for Simulating River and Stream Water Quality. Accessed October 2008 from: http://www.ecy.wa.gov/programs/eap/models/.

Ro, K. S., Hunt, P. G, Poach, M. E. (2006). Wind-Driven Surficial Oxygen Transfer and Dinitrogen Gas Emission from Treatment Lagoons, Journal of Environmental Science and Health, Part A,**41**(8), 1627 – 1638.

Ro, K.S., and Hunt, P.G. (2006). A new unified equation for wind-driven surficial oxygen transfer into stationary water bodies. Trans. ASAE, **49**(5), 1 – 8.

Reichert, P., Borchardt, D., Henze, M., Rauch, W., Shanahan, P., Somlyody, L. and Vanrolleghem, P. (2001). River Water Quality Model No 1 (RWQM1) II. Biochemical Process Equations. Wat. Sci. Tech., **43**(5), 11 – 30.

Rumburg, B., Mount, G. H., Yonge, D., Lamb, B., Westberg, H., Neger, M., Filipy, J., Kincaid R., Johnson, K. (2008). Measurements and modelling of atmospheric flux of ammonia from an anaerobic dairy waste lagoon. Atmospheric Environment, **42**, 3380 – 3393.

Silva, J. D. (2007). Treatment of landfill leachate by stabilization ponds in line: pilot scale studies. PhD Thesis. Federal University of Santa Catarina 199 pp.

Shilton, A. (1996). Ammonia volatilisation from a piggery pond. Wat. Sci. Tech. **33**(7), 183 – 189.

Smith, P. G. and Arab, F. K. (1988). The role of air bubbles in the desorption of ammonia from landfill leachates in high pH aerated lagoon. Water, Air, and Soil Pollution, **38**, 333 – 343.

Timothy J. H., Connor M. A. (1999). Nitrogen removal from wastewater treatment lagoons, Wat. Sci. Tech., **39**(6), 191 – 198.

Vadivelu, V. M., Keller J., Yuan, Z. (2007). Effect of free ammonia on the respiration and growth processes of an enriched Nitrobacter culture. Wat. Res., **41**, 826 – 834.

von Sperling M., Chernicharo, C. A. L. (2005). Biological wastewater treatment in warm climate regions. IWA Publishing. 835 pp.

Wett, B., Rauch W. (2003). The role of inorganic carbon limitation in biological nitrogen removal of extremely ammonia concentrated wastewater. Wat. Res., **37**, 1100 – 1110.

Yang, S.-F., Tay, J.-H., Liu, Y. (2004). Inhibition of free ammonia to the formation of aerobic granules. Biochemical Eng. J., **17,** 41 – 48.

Zimmo, O.R., van der Stehen, N.P. and Gijzen, H.J. (2003). Comparison of ammonia volatilisation rates in algae and duckweed-based waste stabilisation ponds treating domestic wastewater. Wat. Res., **37**, 4587 – 4594.