

Modelling waste stabilisation ponds with an extended version of ASM3

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ABSTRACT

In this paper an extended version of IWA's Activated Sludge Model No 3 (ASM3) was developed to simulate processes in waste stabilisation 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 a 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 $1,456 \text{ g}_{\text{COD}}/\text{m}^3$ and $505 \text{ g}_{\text{NH}_4\text{-N}}/\text{m}^3$ 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 $14.4 \text{ g}_{\text{O}_2}/(\text{m}^2 \text{ d})$ and sun radiation on photosynthesis. Good results for pH and ammonia removal were achieved with mean stripping rates of 18.2 and $4.5 \text{ g}_{\text{N}}/(\text{m}^2 \text{ d})$ for the facultative and maturation pond respectively. Based on measured chlorophyll *a* concentrations and depending on light intensity and TSS concentration it was possible to model algae concentrations.

Key words | ammonia stripping, ASM, landfill leachate, mathematical modelling, waste stabilisation ponds

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INTRODUCTION

Waste stabilisation 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 publicised to simulate such complex systems (e.g. in Buhr & Miller 1983). Jupsin *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 & Chernicharo 2005). Although algae photosynthesis is identified as one of the main sources of oxygen input in facultative WSPs, superficial oxygen transfer has been found to be a relevant factor under conditions with high wind velocities (Ro *et al.* 2006). In the river water quality model QUAL2Kw (Pelletier & 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 stabilisation ponds can be a result of nitrification followed by denitrification (Hurse & Connor 1999; Zimmo *et al.* 2003; Camargo Valero & Mara 2007), or of free ammonia release to the atmosphere (Pano & Middlebrooks 1982; Smith & 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 & Arab 1988). Ni (1999) analysed 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 & 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 volatilisation.

In this paper the influence of sun and wind as well as stripping of ammonia and carbon dioxide is modelled

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) ponds. 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 summarised in Table 1. Parameters like chlorophyll *a*, oxygen, pH and temperature were measured at 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'W). The measurement period lasted from July 2005 to February 2006, in total 243 days.

Mathematical model

ASM3 (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 modelled by three completely mixed reactors. These CSTRs 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		
			Top	Middle	Bottom	Top	Middle	Bottom
COD	g/m ³	1,456				743		
COD filtered	g/m ³	NM				504		
COD/BOD ₅	–	6.8				7.3		
Total suspended solids	g/m ³	301				146		
Total ammonia nitrogen	g/m ³	505				53		
Nitrate nitrogen	g/m ³	8				4		
<i>Depth of measurements</i>			<i>Top</i>	<i>Middle</i>	<i>Bottom</i>	<i>Top</i>	<i>Middle</i>	<i>Bottom</i>
Chlorophyll <i>a</i>	µg/L	NM	79	61	57	163	110	101
Dissolved Oxygen	g/m ³	NM	3.9	3.7	3.4	4.5	4.2	3.9
pH	–	8.7	8.8	8.8	8.8	9.1	8.9	8.7
Temperature	°C	25.9	26.9	25.1	24.8	25.7	23.4	23.0

NM = Parameter not measured.

Algae processes

According to experiments each g_{COD}/m³ 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 ASM3 analogue to RWQM No. 1 (Reichert *et al.* 2001). Below the rates [g/(m³d)] for algae growth on ammonia and nitrate as well as for algae respiration are given.

$$\begin{aligned} \text{rate}_{\text{alg_growth_snh}} &= k_{m_alg} \times \frac{S_{NO} + S_{NH}}{K_{N,ALG} + S_{NO} + S_{NH}} \\ &\times \frac{S_{NH}}{K_{NH_4,ALG} + S_{NH}} \times \frac{I_{AV}}{K_I} \times \exp\left(1 - \frac{I}{K_I}\right) \\ &\times X_{ALG} \end{aligned} \quad (1)$$

$$\begin{aligned} \text{rate}_{\text{alg_growth_sno}} &= k_{m_alg} \times \frac{S_{NO} + S_{NH}}{K_{N,ALG} + S_{NO} + S_{NH}} \\ &\times \frac{K_{NH_4,ALG}}{K_{NH_4,ALG} + S_{NH}} \times \frac{I_{AV}}{K_I} \times \exp\left(1 - \frac{I}{K_I}\right) \\ &\times X_{ALG} \end{aligned} \quad (2)$$

$$\text{rate}_{\text{alg_resp}} = k_{\text{resp_alg}} \times \frac{S_O}{S_O + K_{O,ALG}} \times X_{ALG} \quad (3)$$

where k_{m_alg} is the algae growth rate [d⁻¹], $k_{\text{resp_alg}}$ the algae endogenous respiration rate [d⁻¹], $K_{N,ALG}$ saturation constant to ammonia and nitrate [g/m³], $K_{NH_4,ALG}$ the ammonia inhibition constant [g/m³], I_{AV} the available light

radiation [W/m²], K_I the light limitation and saturation coefficient [W/m²], S_{NO} the nitrate concentration [g/m³], S_{NH} the ammonia concentration [g/m³], S_O the oxygen concentration [g/m³] and X_{ALG} the algae biomass concentration [g_{COD}/m³].

Light attenuation through depth is usually described by the Beer's Law, where the attenuation coefficient is determined 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 WSPs 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:

$$I_{av} = 0.47 \times I \times e^{-K_d \times H} \quad (4)$$

$$K_d = \alpha_1 + \alpha_2 \times X_{TSS} \quad (5)$$

where I is the light radiation [W/m²], 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 [$\text{m}^3/(\text{g}\cdot\text{m})$] and X_{TSS} the total mixed liquor solids concentration [g/m^3].

Ionic equilibrium processes

Free ammonia concentration S_{NH_3} [g/m^3] and carbon dioxide S_{CO_2} [$\text{mol}_\text{C}/\text{m}^3$] were added into the model as new state variables. The ionised ammonium concentration S_{NH_4} [g/m^3] resulted from the difference of S_{NH} (total ammonia) and S_{NH_3} . The concentrations of free ammonia and carbon dioxide in the liquid phase were defined through two equilibrium processes, considering the acid/base pairs: $S_{\text{CO}_2}/S_{\text{ALK}}$ and $S_{\text{NH}_4}/S_{\text{NH}_3}$. 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 ionised 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} [$\text{g}/(\text{m}^3 \cdot \text{d})$] was done as follows:

$$J_{\text{gas}_i} = k l a_{\text{gas}_i} \times \frac{A}{V} \times (S_{\text{sat}_{\text{gas}_i}} - S_i) \quad (6)$$

where $k l a_{\text{gas}_i}$ 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, $k l a_{\text{O}_2}$ [m/d], was calculated according to Ro & Hunt (2006), who 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.

$$k l a_{\text{O}_2} = 0.24 \times 170.6 \times \text{Sc}^{-(1/2)} \times U_{10}^{1.81} \times \left(\frac{\rho_a}{\rho_w}\right)^{1/2} \quad (7)$$

Sc is the dimensionless Schmidt number, U_{10} is the velocity 10 metres over the ground [m/s], ρ_a and ρ_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:

$$S_{\text{sat}_{\text{O}_2}} = 13.89 - 0.3825T + 0.007311T^2 - 0.00006588T^3 \quad (8)$$

$$S_{\text{sat}_{\text{gas}_i}} = \frac{p_{\text{atm}_{\text{gas}_i}}}{K_H} \quad (9)$$

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

$$\frac{U_z}{U_{10}} = \left(\frac{z}{10}\right)^{(1/7)} \quad (10)$$

$$\frac{K_{\text{la}_1}}{K_{\text{la}_2}} = \left(\frac{\text{Sc}_1}{\text{Sc}_2}\right)^{-(1/2)} \quad (11)$$

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 Figure 1c and d. Total suspended solids are shown in Figure 1f. In the extended ASM3 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 & Zhao 2001). This was also

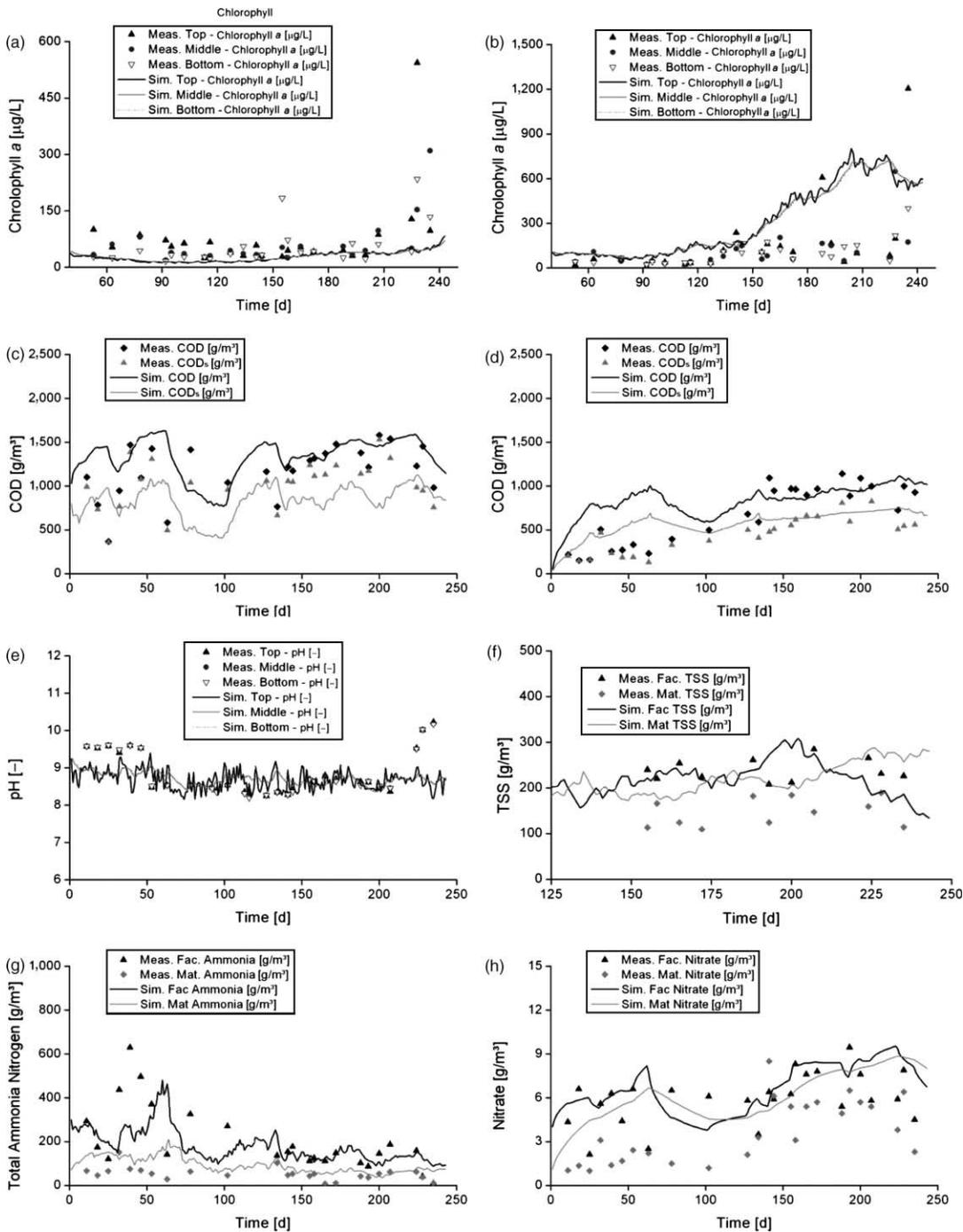


Figure 1 | Measured and simulated results, chlorophyll a concentration Fac (a) and Mat (b), COD effluent concentrations Fac (c) and Mat (d), pH Fac (e), TSS effluent concentrations (f), total ammonia nitrogen concentrations (g) and nitrate effluent concentration (h).

reported by other authors (Lee *et al.* 2000; Yang *et al.* 2004), 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/m³] 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 µg/L chlorophyll *a* respectively. These concentrations were relatively low compared with reported data in facultative ponds of 500–2,000 µg/L (Mara *et al.* 1992). One reason for low chlorophyll *a* could also be free ammonia concentrations (Azov & Goldman 1982). For model calibration only the light inhibition/saturation constant K_I was calibrated to fit algae concentration. The adopted value was 1,200 W/m². 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 (Jupsin *et al.* 2003) respectively. Simulated and measured algae concentrations are shown in Figure 1a and b. Despite the strong influence of depth on light availability to photosynthesis, good mixing of the three pond layers equalised algae concentrations. The increase of algae after day 150 in both ponds reflected the higher light intensities in the summer. The model is not able to explain concentration peaks of algae on day 235. Concentrations reached a maximum of 1,205 µg/L 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 gO₂/(m² 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/m³ 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.

Nitrification and denitrification

In both ponds nitrate concentrations showed low variations. Mean effluent values of 6 g_{NO₃-N}/m³ in the outflow of the facultative pond and 4 g_{NO₃-N}/m³ 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 & Vonshak 1993), free ammonia (Anthonisen *et al.* 1976; Lee *et al.* 2000; Wett & 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). Modelling 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/(m² d) in the facultative and maturation pond respectively. Unfortunately there are not many published data available to compare. Smith & 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/(m² d). Although initial ammonia concentrations in both studies were very similar to the present ponds (575 to 705 g/m³), pH was adjusted above 11, which complicates establishing 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/(m² d) determined through tracer experiments. And experimental stripping rates measured for domestic WSP lay very far below these values (Zimmo *et al.* 2003; Camargo Valero & Mara 2007).

CONCLUSION AND RECOMMENDATIONS

In this paper measured data and simulated results for a three stage pond system treating landfill leachate in Brazil at pilot scale were presented. Measured data and simulation results showed (a) COD, TSS and nitrogen removal can be modelled well with the extended ASM3; (b) pH could be well simulated with calibration of the ammonia dissociation constant; (c) light attenuation can be well modelled 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}/(m² 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; algae growth also seems to be inhibited by free ammonia concentrations; (f) high ammonia stripping rates of 18.2 and 4.5 g_N/(m² d) in the facultative and maturation pond were reached.

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