Optimizing sequencing batch reactor (SBR) reactor operation for treatment of dairy wastewater with aerobic granular sludge

M. Wichern, M. Lübken and H. Horn

ABSTRACT

The biological wastewater treatment using aerobic granular sludge is a new and very promising method, which is predominantly used in SBR reactors which have higher volumetric conversion rates than methods with flocculent sludge. With suitable reactor operation, flocculent biomass will accumulate into globular aggregates, due to the creation of increased substrate gradients and high shearing power degrees. In the research project described in this paper dairy wastewater with a high particle load was treated with aerobic granular sludge in an SBR reactor. A dynamic mathematical model was developed describing COD and nitrogen removal as well as typical biofilm processes such as diffusion or substrate limitation in greater detail. The calibrated model was excellently able to reproduce the measuring data despite of strongly varying wastewater composition. In this paper scenario calculations with a calibrated biokinetic model were executed to evaluate the effect of different operation strategies for the granular SBR. Modeling results showed that the granules with an average diameter of 2.5 mm had an aerobic layer in between 65–95 μm. Density of the granules was 40 kg_{VSS}/m^3. Results revealed amongst others optimal operation conditions for nitrogen removal with oxygen concentrations below 5 g_{O2}/m^3. Lower oxygen concentrations led to thinner aerobic but thicker anoxic granular layers with higher nitrate removal efficiencies. Total SBR-cycle times should be in between 360–480 minutes. Reduction of the cycle time from 480 to 360 minutes with a 50% higher throughput resulted in an increase of peak nitrogen effluent concentrations by 40%. Considering biochemical processes the volumetric loading rate for dairy wastewater should be higher than 4.5 $kg_{COD}/(m^3 * d)$. Higher COD input load with a COD-based volumetric loading rate of 9.0 $kg_{COD}/(m^3 * d)$ nearly led to complete nitrogen removal. Under different operational conditions average nitrification rates up to 5 $g_{NH}/(m^3 * h)$ and denitrification rates up to 3.7 $g_{NO}/(m^3 * h)$ were achieved.

Key words | aerobic granular sludge, biofilm, dairy wastewater, industrial wastewater, mathematical modelling, SBR optimization, simulation

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INTRODUCTION

Biological wastewater treatment using aerobic granular sludge is a new and very promising approach to overcome the principal limitations of the activated sludge process. Implementing specific reactor operating conditions (i.e. operating a sequencing batch reactor (SBR) for enhanced nutrient removal, applying high hydrodynamic shear forces and using selective sludge wasting), an originally flocculent activated sludge will grow into spherical sludge aggregates (granular sludge). The research project described in this paper was investigating the possibilities of treating dairy wastewater with a high content of particulate substrates in an aerobic granular sludge SBR. Up to now modelling

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information on the degradation of particle rich wastewaters can only very rarely be found in literature (De Kreuk *et al.* 2007 for municipal wastewater based on the work of Murnleitner *et al.* 1997; Lübken *et al.* 2004 for malting house wastewater based on ASM 3 (Gujer *et al.* 1999)). Results are mainly presented for soluble substrates (e.g. Beun *et al.* 2000). The dynamic model used here was excellently able to simulate the degradation of particulate wastewater substrates from a real wastewater in a granular activated sludge bed, describing typical biofilm related mass transport and conversion processes such as diffusion or substrate limitation in detail. Anyway, this paper is giving detailed information how to optimize SBR reactor operation. Literature values that are documenting optimal granular sludge reactor operation differ significantly. Sometimes data for optimal operation is difficult to interpret as different operation strategies are not tested, substrate composition is not completely analyzed, substrate limitations are not investigated, diameters of the granules are not measured and turnover rates are not compared with the active biomass.

Here, a calibrated mathematical model is used generate valuable information for better reactor operation and to reveal optimal reactor operation. The following parameters are investigated in detail in this research:

Granular sludge diameter

Granula sludge diameter is documented to be within the range of 1 to 5 mm (Beun *et al.* 1999: 3.3 mm, Dangcong *et al.* 1999: 0.3–0.5 mm, Liu & Tay 2004: 1.6–1.9 mm, Schwarzenbeck *et al.* 2005: 2.5 mm, Mosquera-Corral *et al.* 2005: 2 mm, Zheng *et al.* 2005: 0.5–1.2 mm, De Kreuk *et al.* 2005: 1.1–1.6 mm, Beun *et al.* 2002: 2.5 mm, Li *et al.* 2006: 3–5 mm). Although thicker granules may lead to substrate limitations within the granule the effect of granule density must be considered. Finding the optimal diameter can supported by mathematical simulation.

Oxygen concentration in the bulk

Also for oxygen a wide range of concentrations are given in literature resulting in values in between 2–8 g/m^3 (e.g. Liu & Tay 2004; Tsuneda *et al.* 2005: 4–6 g/m^3). Oxygen concentrations certainly have an influence on both, the granule structure and nitrogen removal.

Volumetric loading rate

Typical values are around 2–8 kg$_{COD}$/(m^3 * d), given in Beun *et al.* 1999: 7.5 kg$_{COD}$/(m^3 * d), Mulder *et al.* 2001: 6–7 kg$_{COD}$/(m^3 * d), Beun *et al.* 2000: 2.3 kg$_{COD}$/(m^3 * d), Tay *et al.* 2001: 6 kg$_{COD}$/(m^3 * d), Arrojo *et al.* 2004: 7 kg$_{COD}$/(m^3 * d), Schwarzenbeck *et al.* 2004: 3.2 kg$_{COD}$/(m^3 * d), Wang *et al.* 2005: 3.3 kg$_{COD}$/(m^3 * d), Zheng *et al.* 2006: 6 kg$_{COD}$/(m^3 * d) und Li *et al.* 2006: 4.5–5.9 kg$_{COD}$/(m^3 * d)). Moy *et al.* (2002) used high volumetric loading rates of up to 15 kg$_{COD}$/(m^3 * d).

SBR cycle time

Typical cycle times in granular SBR reactors are varying in between 6–12 h. Their effect on COD and nitrogen removal can be shown efficiently with a mathematical model.

Aerobic layer of the granule

Measured aerobic layers of granules are varying from 70 to 500μm (Tay *et al.* 2001: 70 μm, Beun *et al.* 2002: 80 μm, Tay *et al.* 2002*a*: 70–100 μm, Jang *et al.* 2003: >30 μm, Li *et al.* 2006: 80–140 μm, Wilen *et al.* 2004: with 8 mg$_{O2}$/L ~ 250 μm, with 20 mg$_{O2}$/L ~ 500 μm). The aerobic layer is affecting both, NH_4-N and NO_3-N removal. In general the structure of aerobic granules has still not been completely identified although anaerobic, anoxic and aerobic zones have been documented by Zhu *et al.* (2003), McSwain *et al.* (2005), Li *et al.* (2006) and Wang *et al.* (2006).

COD fractions

From the modeling point of view and also in practical experiences the composition of COD has a great influence on carbon and nitrogen removal but also on the biomass composition especially when establishing aerobic granules. For particle-rich wastewater Schwarzenbeck *et al.* (2004) describe the importance of protozoa especially during the growth of the granular sludge. The detailed modeling of protozoa would be an important task as Morgenroth *et al.*

(2002) investigated that protozoa are able to directly uptake particles if they are in the size of micrometers. The model developed here uses two external hydrolysis processes for soluble and particular substrates. COD is divided into seven fractions and a shift in between easy degradable and slowly degradable COD is part of this investigation.

MATERIAL AND METHODS

Laboratory plant

The SBR reactor was fed with dairy wastewater and run cyclically by Schwarzenbeck *et al.* (2005). The changes in wastewater composition were extremely strong. During the simulation periods, the COD concentrations ranged from 1,800 to 3,600 mg/l, N_{total} from 1 to 160 mg/l, and TSS from 80 to 900 mg/l. The MLSS concentrations in the reactor changed from 3.8 to 6.2 g/l, in the effluent even from 400 to 1,600 mg/l. The amounts of ammonium, nitrate, nitrite, total nitrogen, total phosphorous, and COD were determined spectro-photometrically with the Dr.-Lange-Device ISIS 6000. The sedimentation speeds after 2 h were measured in an Imhoff tank. MLSS and VSS were determined according to the German standards laid down by the Fachgruppe Wasserchemie 2003 (Section Hydro-Chemistry). Respiration measurements to determine the easily degradable COD ratio were run by Sozzi (2004) in temperature-controlled batch experiments using an oxygen electrode (WTW Oxi 340i, with WTW CellOx 325) and resulted in easy degradable COD concentrations of 20% COD_{hom}. The reactor was run at an average volumetric COD load of $5.9\,kg_{COD}/(m^3 * d)$ in the first period and $4.5\,kg_{COD}/(m^3 * d)$ in later periods. The sludge age was 3–7 days (Table 1).

Figure 1 shows the sludge morphology during the experiments.

Mathematical model

The model developed to simulate the treatment of dairy wastewater in an aerobic granular sludge SBR was implemented in the software AQUASIM (Reichert 1998). The model allows for considering the resulting cyclical water level in the SBR and the ensuing changing TSS contents during the cycle (Wichern *et al.* 2008). Aerobic and anoxic zones within the granules were simulated, diffusion of the soluble components O_2, NH_4-N, NO_3-N, HCO_{3-} and COD was considered by using diffusive links between compartments that represent the different conditions in the granules. Furthermore the model includes the heterotrophic X_H and autotrophic biomass X_N, inert material X_I, S_I and easy degradable COD S_S as well as hydrolysable particular and soluble products X_S and S_{XS}. It was built according to the PETERSON-Matrix. 11 processes were implemented including aerobic and anoxic heterotrophic growth, nitrification, hydrolysis of slowly degradable COD, aerobic and anoxic heterotrophic and autotrophic decay. Measured data and simulation results revealed that for heterotrophic biomass a maintenance process was necessary to model sludge production. Nitrate nitrogen could be incorporated into biomass if there was a lack of ammonia nitrogen. Good simulation results were achieved with a

Table 1 | Operation method of a granular SBR reactor

Operation/Period	Period 1 Weeks 1–9	Intermediate phase Week 10	Period 2 Weeks 11–20
Filling [min]	60	60	60
Anaerobic [min]	60	60	0
Aeration [min]	335	345	405
Sedimentation [min]	15	5	4
Decanting [min]	5	5	5
Idling [min]	5	5	5
Total cycle [min]	480	480	480
Exchange ratio [%]	75	50	50

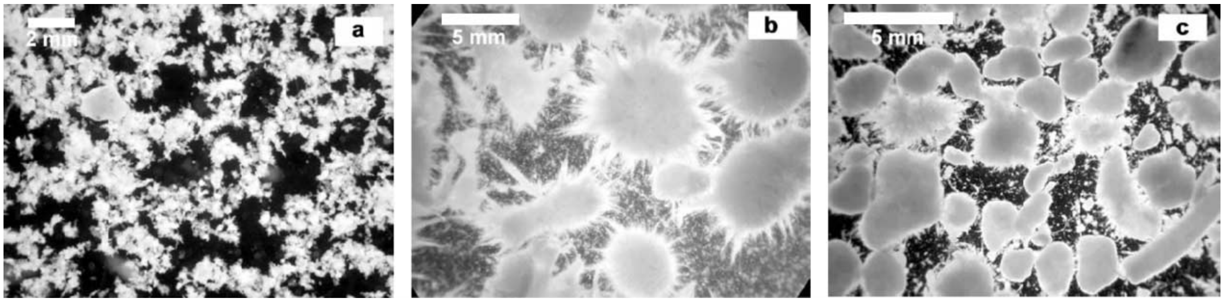

Figure 1 | Sludge morphology during the experiments (Schwarzenbeck *et al.* 2005).

fractionation related to homogenised COD of $S_I = 6\%$, $S_S = 20\%$, $S_{XS} = 59\%$, $X_H = 9\%$, $X_N = 1\%$ and $X_I = 5\%$. The fraction of easily degradable COD ($S_S = 20\%$) was confirmed in respiration measurements. The aerobic thickness of the granula was calculated according to Equation 1. Assuming that oxygen is the first substrate to be consumed, it is possible to approximately calculate the minimum penetration depth of oxygen into the granules.

$$x_{min} = \sqrt{\frac{2 \cdot D_{O_2} \cdot S_O}{r_{O_2,max}}} \ [m] \quad (1)$$

With D_{O_2} being the oxygen diffusion coefficient, S_O being the oxygen concentration in the liquid phase (Bulk) [g/m^3] and $r_{O_2,max}$ the maximum reaction rate [g/(m^3 * d)].

RESULTS AND DISCUSSION

Modeling results showed that the aerobic layer of the granules was between 65–95 μm depending on the changing oxygen concentrations in the SBR in between 0 to 9 g/m^3. The average total diameter of the granules was measured at 2.5 mm. Below the aerobic layer, anoxic

Table 2 | Average COD, nitrification and denitrification rates from scenario calculations, in brackets: maximum rates during one SBR cycle

	Nitrification rate [$g_{NH}/(m^3 * h)$]	Denitrification rate [$g_{NO}/(m^3 * h)$]	COD degradation rate [$g_{COD}/(m^3 * h)$]
$O_{2,SBR} = 5\,g/m^3$	4.9 (37.5)	3.7 (31.4)	113.9 (515.3)
$O_{2,SBR} = 7\,g/m^3$	5.0 (38.4)	3.6 (31.4)	114.1 (511.8)
$O_{2,SBR} = 9\,g/m^3$	5.0 (39.1)	3.5 (31.4)	114.2 (509.1)
T = 10°C	4.3 (28.3)	3.2 (23.4)	107.1 (497.9)
T = 15°C	4.7 (35.3)	3.4 (25.5)	111.7 (505.7)
T = 18°C	5.0 (39.1)	3.5 (31.4)	114.2 (509.1)
$S_S = 10\% * CSB_{hom}$	5.0 (38.6)	3.3 (31.4)	109.2 (448.6)
$S_S = 20\% * CSB_{hom}$	5.0 (39.1)	3.5 (31.4)	114.2 (509.1)
$S_S = 30\% * CSB_{hom}$	5.0 (39.5)	3.6 (40.9)	119.0 (707.9)
Aer.Layer = 40 μm	4.9 (37.7)	3.5 (31.4)	112.9 (503.1)
Aer.Layer = 80 μm	5.0 (39.1)	3.5 (31.4)	114.2 (509.1)
Aer.Layer = 160 μm	5.1 (38.9)	3.0 (31.4)	115.4 (507.4)
Diameter = 1.0 mm	4.9 (37.6)	3.5 (31.4)	112.7 (501.3)
Diameter = 2.5 mm	5.0 (39.1)	3.5 (31.4)	114.2 (509.1)
Diameter = 4.0 mm	5.1 (39.0)	3.2 (31.4)	115.1 (510.5)
$2.25\,kg_{COD}/(m^3 * d)$	3.4 (27.5)	2.5 (31.4)	50.2 (448.6)
$4.5\,kg_{COD}/(m^3 * d)$	5.0 (39.1)	3.5 (31.4)	114.2 (509.1)
$9.0\,kg_{COD}/(m^3 * d)$	4.9 (38.3)	3.6 (91.4)	242.0 (1059.5)

and/or anaerobic conditions were found. At a greater depth, inert material from cellular lysis was identified. The specific surface of the granules was approximately $175\,m^2/m^3$, where as the granular sludge density was nearly $40{,}000\,g_{oTS}/m^3$. Moreover, model calculations confirmed that the COD conversion of the heterotrophic biomass was reduced by 25–40% in the denitrification zone of aerobic granular sludge. For the diffusion coefficient D_{O_2} a value of $1.0 * 10^{-4}\,m^2/d$ according to results from Wanner & Reichert (1996) was used. According to literature nitrogen and COD diffusion coefficients were $D_{Ss} = 0.6 * 10^{-4}\,m^2/d$, $D_{NH} = 0.8 * 10^{-4}\,m^2/d$ and $D_{NO} = 0.8 * 10^{-4}\,m^2/d$.

Process optimization

In the following different scenarios were calculated concerning their effect on the system behavior. Table 2 summarizes the results for COD and nitrogen removal for different oxygen concentrations, temperatures, easy degradable COD concentrations and volumetric loading rates. Furthermore the effect of the granule diameter and the thickness of the aerobic layer were investigated. In general the effect of different operation conditions on nitrification and denitrification rates is lower than in municipal wastewater as both ammonia and nitrate concentrations via nitrification are lower than for typical municipal wastewater.

Effect of oxygen concentrations

Nevertheless it can be seen that for better nitrogen removal–if the metabolism of the biomass is modeled in detail–the oxygen concentrations should be lower than during the measurement phase ($9\,g_{O2}/m^3$). Simulation revealed ideal conditions for nitrate removal at oxygen concentrations of $5\,g_{O2}/m^3$. Oxygen is penetrating via diffusion to a minor degree into deeper zones of the granules. As a result anoxic zones increase. Figure 2(a + b) shows the effect of higher oxygen concentrations on the dynamic course of the reactor operation. Lower oxygen

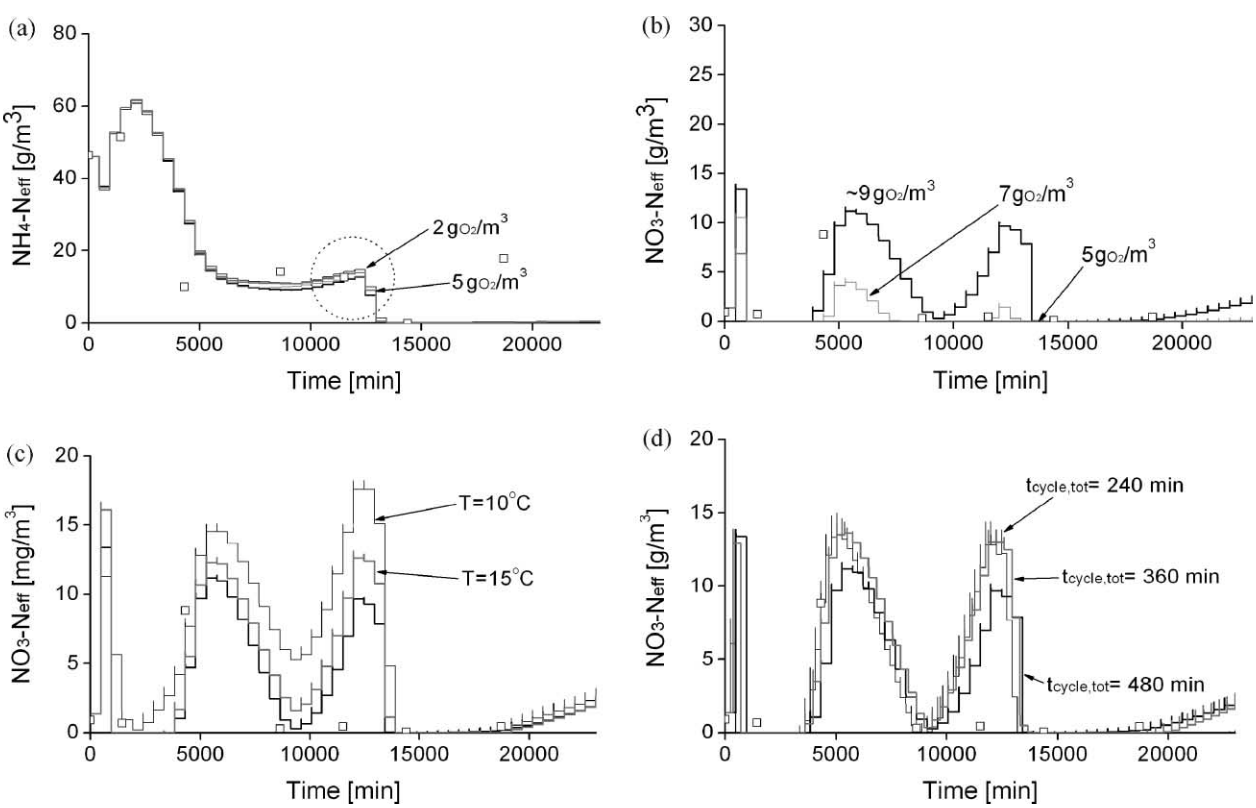

Figure 2 | Results from different types of SBR operation concerning oxygen concentration (a + b), temperature (c) and cycle time (d).

concentrations do not result in smaller nitrification efficiencies as elimination is only seriously effected at concentrations below 1 g_{O2}/m^3. Furthermore nitrate effluent concentrations improve to nearly zero during the investigated period of 16 days.

Effect of operation temperature

The decrease of temperature is affecting all biochemical processes. This means the reduction of 8°C leads to a decrease of nitrification capacity of nearly 20%. Under peak conditions nitrate effluent concentrations increase by 70% when the operation is decreased from originally 18°C to 10°C (see Figure 2 c).

Effect of cycle time

Furthermore, a reduced aerobic cycle time was investigated whilst the exchange ratio of the SBR was kept constant. The impact of the cycle time reduction can be seen in Figure 2 (d) for the COD and nitrate removal. Decreased cycle times of 240 min led to increased effluent concentrations, here to an increase of COD and nitrate nitrogen of 50 and 40%, respectively. The influence of a lower aerobic cycle time still was lower than expected, but only in that case when effluent TSS concentrations from the SBR were kept constant.

Effect of the easy degradable COD fraction and the COD volumetric loading rate

Both, higher quantities of easy degradable COD and higher volumetric loading rates result in better nitrogen removal. This can be seen in Table 2 for the average COD and denitrification rates as well as in Figure 3 (b + d). The change of the easy degradable COD fraction S_S has no effect on nitrification. If the easy degradable COD fraction is increased to 30% of total COD, nitrogen removal improves by 20–30% at peak loadings. A similar effect on nitrate removal as with the increase of the fraction of easy degradable COD can be found with higher loads of total

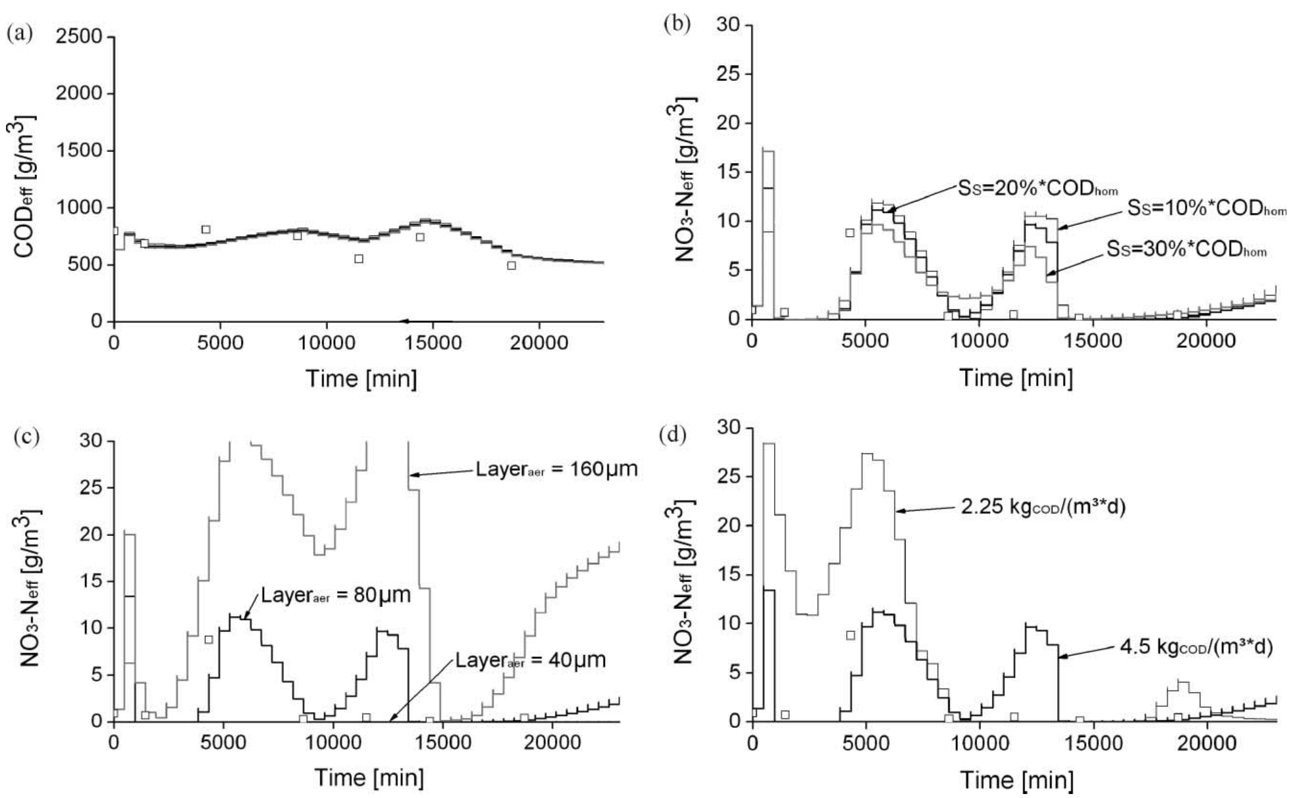

Figure 3 | Results from different types of SBR operation concerning the fraction of easy degradable COD (a + b), aerobic layer of the granules (c) and volumetric loading rate (d).

COD in the influent. As also slowly degradable COD is consumed by the biomass after hydrolysis the effect of higher COD loads on nitrate removal is even stronger. If the COD volumetric loading rate is doubled nearly no nitrate effluent concentrations can be found even at peak nitrogen loading (Figure 3d). If COD load is halved NO_3-N concentrations are meanwhile doubled.

Effect of the aerobic granular diameter and the aerobic granular layer

Two scenarios were calculated with the calibrated model regarding the aerobic layer of the granules and their diameter. We can expect bigger anoxic or anaerobic zones at higher granule diameters. But interestingly the diffusion of easy degradable COD into deeper zones and the availability after the aerobic degradation is limiting the nitrate removal. A similar effect can be seen if the aerobic layer of the granules is increased. Nitrate effluent concentrations rise by more than 200%, if peak concentrations are considered (see Figure 3c).

CONCLUSIONS

In the research project described in this paper dairy wastewater with a high particle load was treated with aerobic granular sludge in an SBR reactor. A dynamic mathematical model was developed describing COD and nitrogen removal as well as typical biofilm processes such as diffusion or substrate limitation. The calibrated model was excellently able to reproduce the measuring data despite of strongly varying wastewater composition. The following conclusions can be drawn from scenario calculations with the calibrated model: (a) Oxygen concentrations in the SBR should be below $5\,g_{O2}/m^3$. Lower oxygen concentrations led to thinner aerobic but thicker anoxic granular layers with higher NO_3-N load being eliminated. (b) Total SBR-cycle times should be in between 360–480 minutes, as the reduction of the cycle time to 360 minutes with a 50% higher throughput resulted in an increase of peak NO_3-N effluent concentrations by only 40%. (c) Regarding nitrogen removal the COD-based volumetric loading rate for dairy wastewater should be in between $4.5–9.0\,kg_{COD}/(m^3 * d)$. (d) In the scenario calculations average nitrification rates up to $5\,g_{NH}/(m^3 * h)$ and denitrification rates up to $3.7\,g_{NO}/(m^3 * h)$ could be achieved. Peak nitrification rates reached up to $39\,g_{NH}/(m^3 * h)$ and peak denitrification rates up to $91\,g_{NO}/(m^3 * h)$. (e) A reduction of temperature from 18 to 10°C resulted in an increase of peak nitrate effluent concentrations of 70%.

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