Solved upscaling problems for implementing deammonification of rejection water

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Abstract So far, extremely efficient metabolic pathways for nitrogen removal exclusively by autotrophic organisms are well established in scientific literature but not in practice. This paper presents results from the successful implementation of rejection water deammonification in a full-scale single sludge system at the WWTP Strass, Austria. Anaerobic ammonia oxidising biomass has been accumulated during a 2.5 year start-up period when the reactor size was gradually scaled up in the steps. The pH-controlled deammonification system (DEMON) has reached a design capacity of eliminating approximately 300 kg of nitrogen per day. Energy savings outperform expectations, decreasing the mean specific demand for compressed air from 109 $m^3(kg\,N)^{-1}$ to 29 $m^3(kg\,N)^{-1}$. Dominance of autotrophic metabolism is confirmed by organic effluent loads topping influent loads.

Keywords Ammonia; anammox; DEMON; pH-control; rejection water; Strass

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Introduction

Commonly, advances in wastewater treatment start from observations of biological processes which will then be transferred to technologically optimised environments. Sometimes, the idea comes first and the concept is developed at the desk. This happened when the Austrian chemist Broda (1977) wondered about possible ancestors of nitrifying organisms. Based on considerations about the evolution of bioenergetic processes, he predicted a "missing lithotroph" which might have existed or still exists and can catalyse the following process:

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \qquad (1)$$

Thermodynamic calculations promised a free enthalpy change of $-360\,kJ\,mol^{-1}$ from this reaction. Later, nitrogen mass balances at treatment facilities for food processing wastewater (Mulder *et al.*, 1995) and for landfill leachate (Hippen *et al.*, 1997; Siegrist *et al.*, 1998) supported the postulate of feasible chemolithotrophic nitrogen removal. Again, the question arose "what's eating the free lunch?" (Olsen, 1999). Finally, Strous *et al.* (1999) managed to identify the missing lithotroph as a new planctomycete. Stoichiometric studies (Strous *et al.*, 1998), focusing on an enriched community of anaerobic ammonia oxidising organisms, confirmed the following equation which considers generation of cell mass and reducing equivalents in form of nitrate necessary for the reduction of CO_2:

$$NH_4^+ + 1.32\,NO_2^- + 0.066\,HCO_3^- + 0.13\,H^+ \rightarrow 0.26\,NO_3^- + 1.02\,N_2 + 0.066\,CH_2O_{0.5}N_{0.15} + 2.03\,H_2O \qquad (2)$$

Following reaction (2), the anaerobic autotrophic ammonia oxidation with nitrite requires a preceding aerobic autotrophic oxidation of at least 57% of total ammonia to be eliminated

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(1.32 mol NO_2^- per 2.32 mol NH_4^+). These two reaction steps can be conducted in two individual units, providing different sludge retention times and conditions where nitrite produced in the aerobic reactor and residual or by-passed ammonia is fed to the anaerobic reactor (Van Dongen *et al.*, 2001). In an alternative approach, both process steps are operated in a single-sludge system (Sliekers *et al.*, 2002). This process requires an aerated system and appropriate process control to prevent build-up of toxic nitrite concentrations due to oxygen excess. The concept has not been "purposefully tested on pilot or full scale, but is known to occur accidentally in sub-optimally functioning full-scale nitrification systems" (Schmidt *et al.*, 2003). Also, Nielsen *et al.* (2005) point out "the continuous threat of reactor failure due to oxygen overloading and subsequent nitrite poisoning of the anammox biomass" and the missing information regarding the NO_2^- status in the reactor (reliable on-line nitrite monitoring) is still a difficult task.

This paper will present full-scale experiences with a pH-based control system which determines the length of aeration intervals depending on the current production of H^+ ions or nitrite, respectively.

Methods

Nitritation/denitritation system

Since 1996, at the WWTP Strass (200,000 pe) in western Austria, rejection water produced from dewatering digested sludge is treated separately in a full-scale single sludge system. The process is characterised by an SBR with an intermittent aeration system controlled by the pH signal (Wett *et al.*, 1998). The pH-value in the reactor is driven downwards during aeration intervals due to H^+ production from nitrification and upwards again during anoxic periods due to continuous dosage of rejection water and denitrification. A simple on-off controller benefits from the opposed pH-effect of both involved process steps and the appropriate selection of the pH-setpoints determines conversion rates. Process conditions – explicitly low DO values – successfully repress the second oxidation step from nitrite to nitrate despite high sludge retention times of approximately 30 d.

The aim of a new process development is to decrease the demand for resources without endangering the proven robustness of the treatment system, i.e. the single sludge concept and the pH-controlled intermittent aeration system. The main challenge is the shift from a functioning nitritation/denitritation system – that is, a system substantially supplied with external organic carbon (primary sludge addition) and enriched with heterotrophic biomass – to a dominantly slowly growing autotrophic biomass.

Enrichment of anaerobic ammonia oxidising biomass

In order to shorten the transition period in the full-scale reactor, a stepwise strategy for enrichment of the slowly growing biomass was applied. Starting from 4 L of inoculum taken from the pilot-plant operated by the EAWAG in Zuerich (Fux *et al.*, 2002), a 300 L reactor was seeded. Then, the reactor size was gradually increased at steps of 1 and 2 orders of magnitude to a volume of 2.4 m^3 and finally to 500 m^3 (Figure 1). At each upscaling stage biomass enrichment was a vulnerable process until robustness due to a critical mass was achieved, which then served as an inoculum for the next reactor. The enrichment period took 2 years and the actual start-up of the full-scale reactor another 6 months until the end of 2004, when the colour of sludge granules changed from brownish to the characteristic red (Strous *et al.*, 1998).

Development of a pH-controlled deammonification system (DEMON)

To date, no molecular biological tools have been applied for species identification or activity measurements. Mass balances have been used to investigate process conversion

Figure 1 Gradual reactor size scale-up for enrichment of anaerobic ammonia oxidising biomass starting from 4 L of inoculum to the final reactor volume of 500 m^3

rates and stoichiometry. The aeration control system has been developed and optimised by means of numerical modelling for the identification of relevant inhibition and limitation kinetics (Wett and Rauch, 2003).

The set-point of dissolved oxygen (DO) control was specified at a low range close to 0.3 mg/L in order to prevent rapid nitrite accumulation and to maintain a continuous repression of the second oxidation step of nitrite to nitrate. During the aeration period of approximately 6 h of an SBR cycle (Figure 2), both deammonification processes – partial nitritation and anaerobic ammonia oxidation – are operated. These two processes show converse impacts on the pH-value.

The aeration system is activated only within a very tight pH-control interval of 0.01. Due to oxygen input, nitritation runs at a higher rate than anaerobic ammonia oxidation and H^+ production drives the pH-value to the lower set-point and aeration stops. While dissolved oxygen is depleted, all the nitrite that has been accumulated during the aeration interval is used for oxidising ammonia. In the course of this biochemical process some alkalinity recovers and, additionally, alkaline rejection water is fed continuously to the reactor until the pH-value reaches the upper set-point and aeration is switched on again.

Results and discussion

Energy saving

The expected stoichiometric benefit in the oxygen demand yields almost half the value when the predominant metabolic route is shifted from nitritation/denitritation to deammonification (Figure 3). The potential for energy saving is even higher when heterotrophic respiration is considered. An external carbon source for denitritation (60% of the carbon demand comparing to denitrification) causes additional oxygen requirements for the heterotrophic biomass.

Nitrate produced by deammonification shows a stoichiometric carbon demand of only 10% of the base value, which is usually covered by the organic content of rejection water. Energy savings do not only derive from less heterotrophic respiration but also from improved oxygen transfer efficiency at lower solids concentration TSS in the reactor.

The monitored energy demand up to 600 kWh per day (Figure 4) shows significant fluctuations between workdays and weekends in correspondence with the operation periods of the sludge dewatering units. After inoculation and start-up of the SBR on

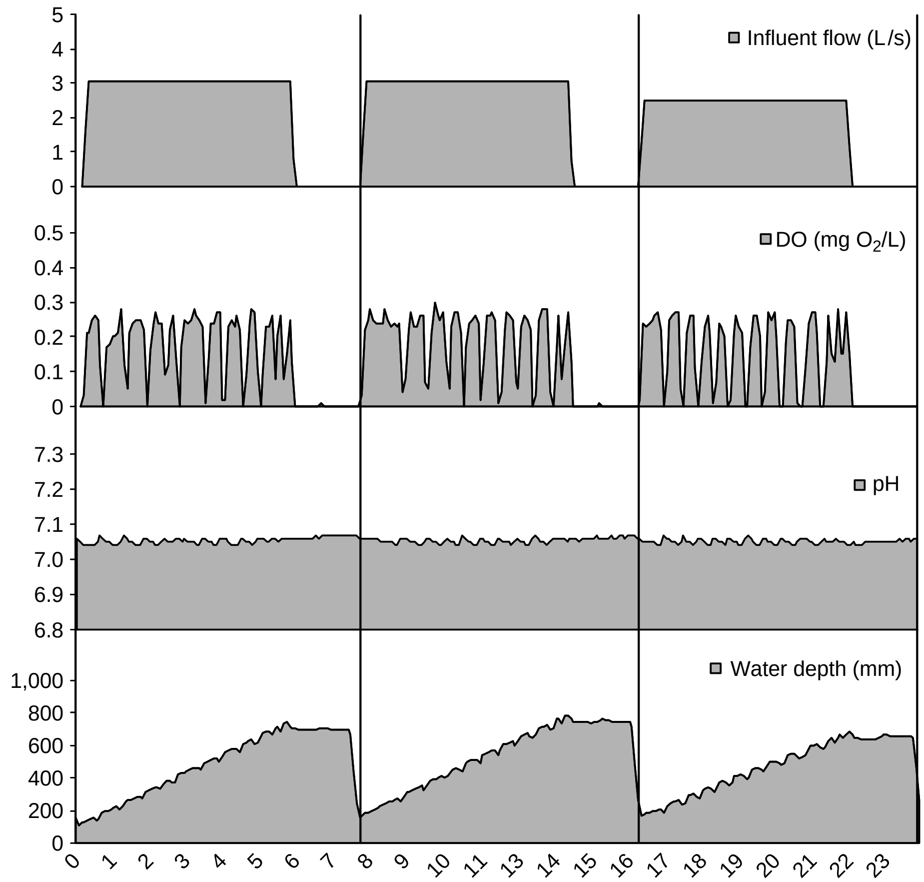

Figure 2 Profiles of process variables (flowrate, DO, pH and water table) displaying the control of intermittent aeration by a tight interval of pH-setpoints

the 8th of July, the specific energy demand decreases from initial peaks to a stable minimum of 0.79 kWh per kg N eliminated (Figure 5).

Upscaling problems

Temperature conditions. Both smaller vessels used for biomass enrichment had been supplied by external heat in order to maintain a temperature of approximately 30°C. The large reactor, with a volume of 500 m^3, was not equipped with heating facilities. Therefore, the reactor was inoculated at the beginning of summer 2004 at high ambient

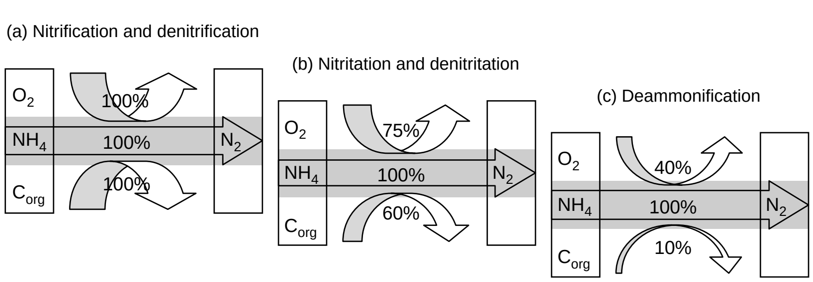

Figure 3 Relative demand for ressources at three different metabolic routes: (a) via nitrite and nitrate, (b) exclusively via nitrite or (c) partially via nitrite

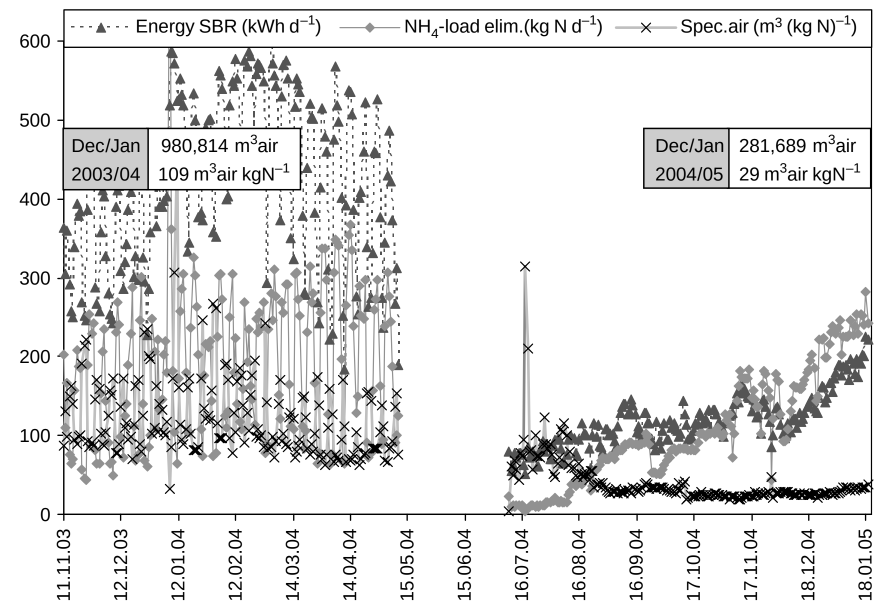

Figure 4 Daily total energy demand (for aeration, stirring and pumping), specific air supply and eliminated nitrogen load of the rejection water treatment system in Strass before and after inoculation (indication of 50 d mean values in boxes)

temperatures. A time window of half a year was available until temperature conditions deteriorate. During this start-up period the temperature in the reactor increased from 22 to 30 °C due to growing feed rates of mesophilic digested sludge liquors and exothermic processes in the reactor. Draw-backs in process performance induced by solids wash-out can be tracked back in temperature profiles showing two minima of 25 °C at the beginning of October and December.

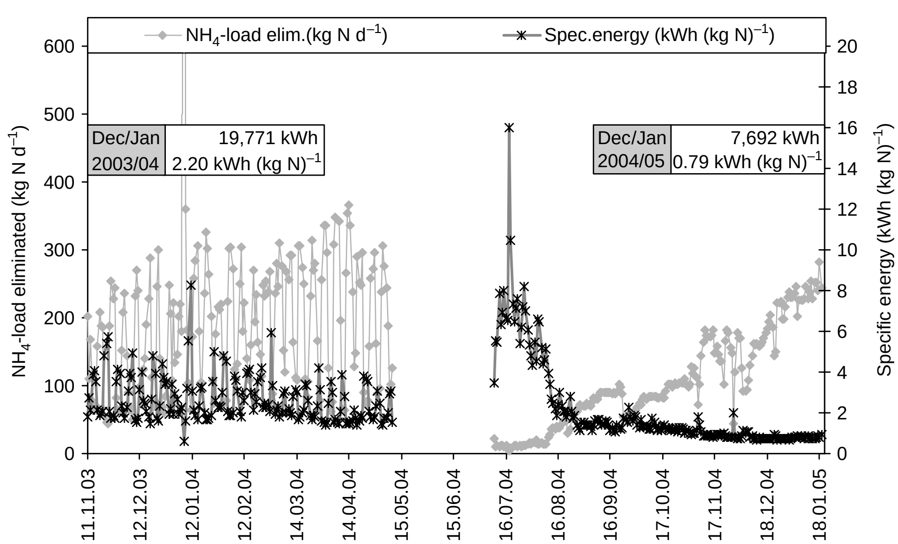

Figure 5 Specific energy demand (for aeration, stirring and pumping) related to the eliminated nitrogen load of the treatment system before and after inoculation (indication of 50 d mean values in boxes)

Flexible aeration. Like temperature development, the oxygen demand is closely connected to the process rates in the up-scaled reactor. Feed variations from, initially, 6 to 175 m^3 per day mean a required flexibility of the aeration system in the range between 500 and almost 10,000 m^3 pressurised air per day. Insufficient air supply results in an accumulation of bicarbonate blocking the pH-control and oxygen overload causing continuous switching of the aeration system. Obviously, the frequency transformer of the blower was overextended during the start-up period and mechanical adoption of transmission and tuning of the blower frequency was necessary.

Solids wash-out. After inoculation the supernatant water was discharged by submerged pumps. When the amount of treated rejection water increased, a fixed discharge device was put into operation causing a significant decrease in the solids concentration TSS in the reactor (drop-off in treatment capacity at the beginning of October indicated in Figure 5). A flush installation which re-suspended deposited solids in the discharge device improved the situation. Still, during the very initial period of the discharge phase, increased TSS concentrations are measured in the effluent which especially affects low-load operation.

Foaming. Generally, the activated sludge produced in this rejection water treatment system shows satisfying settling properties ($SVI = 116 \pm 19\,mL\,g^{-1}$). Foaming phenomena occur during aeration periods and of course depend on aeration intensity and TSS concentration (foaming is favoured at low sludge concentrations). Excessive foaming caused a severe loss of solids at the beginning of December. The problem was solved by the installation of a circular pipe along the reactor walls spraying the influent flow well distributed on the foam layer.

Elimination efficiency

The mean ammonia elimination efficiency calculated from daily measurement values within a 50 d monitoring period amounted to 89.3 ± 1.2% (Table 1). The total nitrogen removal rate was only slightly less (83.9 ± 1.8%) because the nitrate produced in the process was denitrified by the heterotrophic biomass grown on the organic carbon content of the rejection water (Table 2).

The loss of solids cannot be attributed to these three exits – flush, foam and stabilisation – individually and therefore the current sludge retention time (SRT) is difficult to estimate. Wash-out of solids (WOS) sums up continuous discharge of solids ($-24.8\,kg\,d^{-1}$) and uncontrolled discontinuous loss (0 to $-27.5\,kg\,d^{-1}$). Depending on the effectiveness of stabilisation processes, the SRT calculated from solids wash-out amounts to 16.3 to 34.5 d, i.e. approximately 25 d ($SRT = VSS_{reactor}{}^{*}V_{reactor}/WOS$).

The resulting SRT is surprisingly low considering the low growth rate of anaerobic autotrophic biomass. After inoculation and start-up of the treatment system the loading

Table 1 Sludge properties, influent and effluent characteristics of STRASS rejection water treatment system (50 d mean values and standard deviation measured between 1st December 2004 and 19th January 2005)

	TSS $g\,m^{-3}$	VSS $g\,m^{-3}$	SVI $mL\,g^{-1}$	Temp. °C	Flowrate $m^3\,d^{-1}$	NH_4-removal %	N-removal %
Reactor	2.93 ± 0.33	1.71 ± 0.14	116 ± 19	27.8 ± 1.7	119 ± 29	89.3 ± 1.2	83.9 ± 1.8
	NH_4 $g\,N\,m^{-3}$	NO_2 $g\,N\,m^{-3}$	NO_3 $g\,N\,m^{-3}$	$COD_{soluble}$ $g\,COD\,m^{-3}$	$COD_{particulate}$ $g\,COD\,m^{-3}$		
Influent	1,832 ± 40	0	0	435 ± 176	233 ± 119		
Effluent	196 ± 22	6.2 ± 4.5	93.7 ± 12.0	268 ± 61	311 ± 185		

Table 2 Loss of solids due to initial discharge flush, foaming and stabilisation estimated by a VSS balance of the STRASS rejection water system (autotrophic yields according to Strous *et al.*, 1998)

	Concentration g m^{-3}	Yield	Flow $m^3\,d^{-1}$	Stoich. factor	Sludge production kg VSS d^{-1}
Heterotrophic growth	$COD_{soluble}$ (435 − 268)	0.54	119	1 = $NO_2/NH_{4\ elim.}$	+10.7
Aerobic autotrophic growth	NH_4 (1,832 − 196)	0.17	119	1.32/2.32 = NH_4 /$NH_{4\ elim.}$	+18.8
Anaerobic autotrophic growth	NH_4 (1,832 − 196)	0.12	119	1.00/2.32 = VSS /$COD_{part.}$	+10.1
Influent solids	$COD_{particulate}$ 233		119	0.67 = VSS/$COD_{part.}$	+18.6
Effluent solids	$COD_{particulate}$ 311		119	0.67 =	−24.8
Accumulation	$VSS_{start} - VSS_{end}$ (1,485 − 2,074)		$V_{reactor}$/days 500/50	1 =	−5.9
1st flush, foaming, stabilisation					−27.5

rate was significantly lower (8th July 2004 to 7th October 2004: $Q_{mean} = 29\,m^3\,d^{-1}$) and the volatile suspended solids concentration was higher ($VSS_{mean} = 2.86\,kg\,m^{-3}$) compared to the final period. Under the assumption of the same flow-specific wash-out of solids as discussed above, the SRT during the initial 3 months of the SBR operation period was 6.8 times higher, i.e. approximately 170 days.

Conclusions

The development of an alternative metabolic nitrogen removal route shows a diversified 30 years history ranging from thermodynamic process prediction, via accidental detection of anaerobic ammonia oxidising biomass at landfills, via systematic biotechnological identification of species and feasible process schemes and now to successful full-scale implementation. Presented experiences from the rejection water treatment system STRASS prove the robustness of the deammonification process once it is established. The 50 d specific energy demand (including aeration, stirring and pumping; no heating-requirement) of 0.79 kWh per kg eliminated nitrogen demonstrates the attraction of separate biological treatment of sludge liquors due to significant resource savings.

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