Solved scaling problems for implementing deammonification of rejection water

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Abstract So far very 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 years lasting start-up period when the reactor size was gradually up-scaled in 3 steps. STRASS rejection water treatment system has reached design capacity eliminating about 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.
Key words rejection water treatment, ammonia, pH-control, STRASS, anammox

INTRODUCTION

Commonly advances in wastewater treatment start from observations of biological processes which then will 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 following process: $NH_4^+ + NO_2^- \rightarrow N_2 + 2\ H_2O$ …(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 following equation which considers generation of cell mass and reducing equivalents in form of nitrate necessary for the reduction of CO_2.

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

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 (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 built-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 about 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

Since 1996 at the WWTP Strass (200000 pe) in western Austria rejection water produced from dewatering digested sludge is treated separately in a full-scale single sludge system. The STRASS process is characterised by a 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. Provided process conditions – explicitly low DO values – successfully repress the second oxidation step from nitrite to nitrate despite high sludge retention times of about 30 days.

Now the aim of an additional process optimisation was 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 was 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. In order to shorten the transition period in the full-scale reactor a stepwise strategy was applied. Starting from 4 litres of inoculum taken from the pilot-plant operated by the EAWAG in Zuerich (Fux et al., 2002) a 300 l reactor was seeded. Then 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 (fig.1). At each scaling 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 total start-up period took almost 2.5 years until the end of 2004, when the colour of sludge granules changed from brownish to the characteristic red (Strous et. al., 1997).

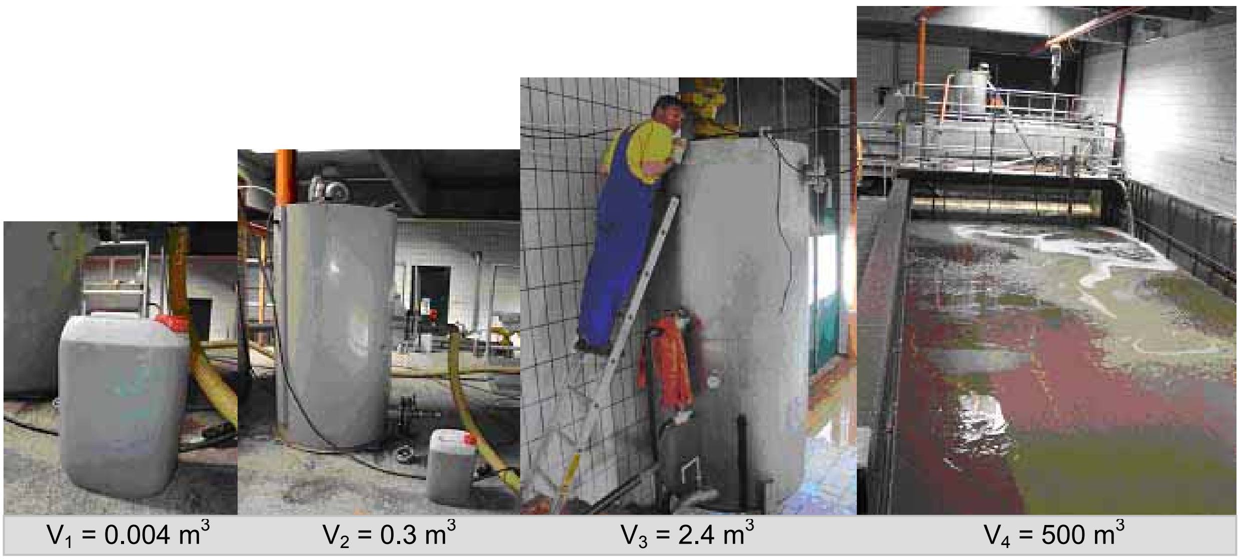

Fig.1 Gradual reactor size scale-up for enrichment of anaerobic ammonia oxidising biomass starting from 4 litres of inoculum to the final reactor volume of 500 m^3

No molecular biological tools have been applied for species identification or activity measurements so far. Mass balances have been used to investigate process conversion 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).

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 (fig.2). The potential for energy saving is even higher when the 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 rates at lower solids concentration TSS in the reactor.

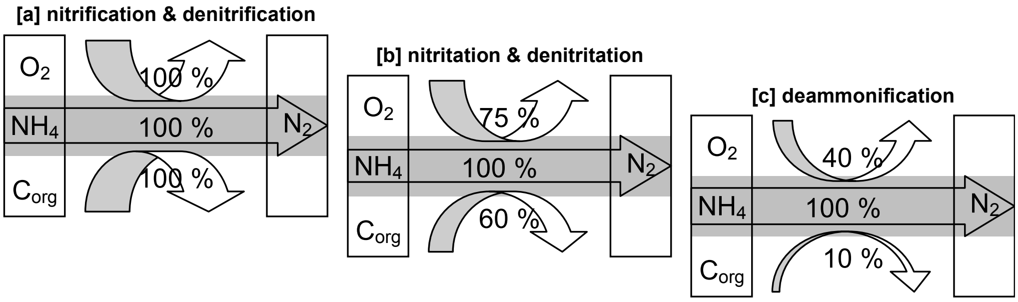

Fig.2 Relative demand for ressources at 3 different metabolic routes ([a] via nitrite and nitrate, [b] exclusively via nitrite or [c] partially via nitrite)

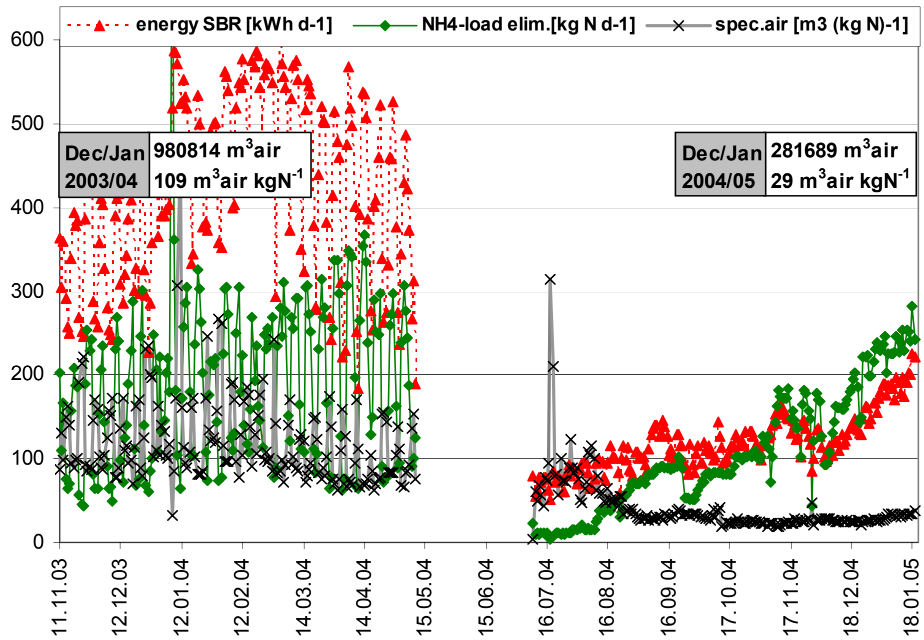

Fig.3 Daily total energy demand (for aeration, stirring and pumping), specific air supply and eliminated nitrogen load of the STRASS rejection water treatment system before and after inoculation (indication of 50 days' mean values in boxes)

The monitored energy demand up to 600 kWh per day (fig.3) 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 at the 8th of July the specific energy demand decreases from initial peaks to a stable minimum of 0.79 kWh per kg N eliminated (fig.4).

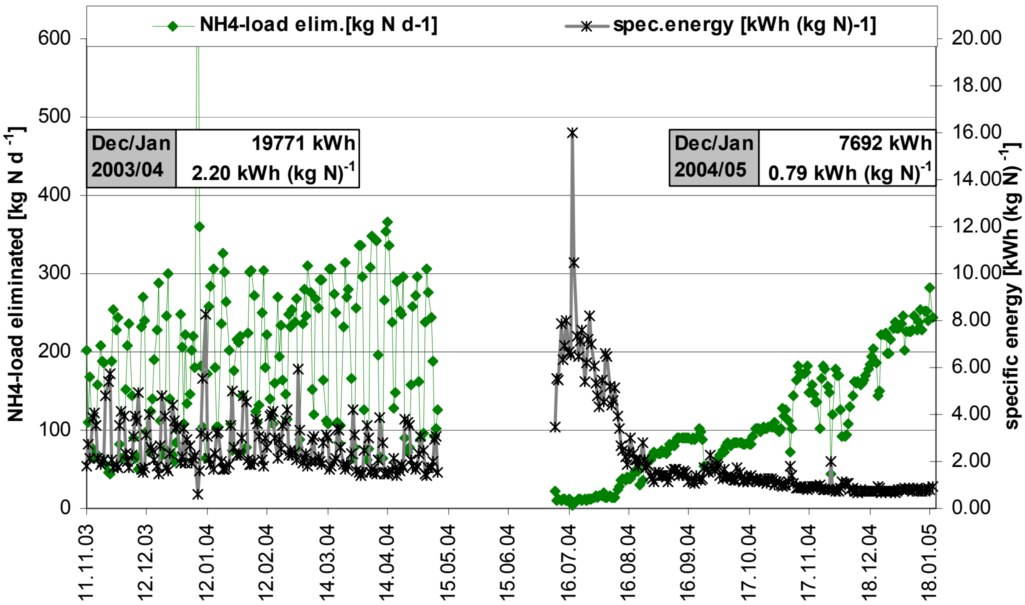

Fig.4 Specific energy demand (for aeration, stirring and pumping) related to the eliminated nitrogen load of the STRASS treatment system before and after inoculation (indication of 50 days' mean values in boxes)

SCALING PROBLEMS

Temperature conditions

Both smaller vessels used for biomass enrichment had been supplied by external heat in order to maintain a temperature of about 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 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 °C 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 2 minimums of 25 °C at the beginning of October and December.

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 10000 m^3 pressurised air per day. Insufficient air supply results in an accumulation of bicarbonate blocking the pH-control and oxygen overload causes 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 fig.3). 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 affects especially low-load operation.

Foaming

Generally the activated sludge produced in this rejection water treatment system shows satisfying settling properties (SVI=116 ± 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 days' monitoring period amounted to 89.3 % ± 1.2 % (tab.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 (tab.2).

Tab.1: Sludge properties, influent and effluent characteristics of STRASS rejection water treatment system (50 days' mean values and standard deviation measured between 2004-12-01 and 2005-01-19)

	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	1832 ± 40	0	0	435 ± 176	233 ± 119
effluent	196 ± 22	6.2 ± 4.5	93.7 ± 12.0	268 ± 61	311 ± 185

Tab.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 =	**+10.7**
aerobic autotrophic growth	NH_4 (1832 - 196)	* 0.17	* 119	$NO_2/NH_{4\ elim.}$ * 1.32/2.32 =	**+18.8**
anaerobic autotrophic growth	NH_4 (1832 - 196)	* 0.12	* 119	$NH_4/NH_{4\ elim.}$ * 1.00/2.32 =	**+10.1**
influent solids	$COD_{particulate}$ 233		* 119	$VSS/COD_{part.}$ * 0.67 =	**+18.6**
effluent solids	$COD_{particulate}$ 311		* 119	$VSS/COD_{part.}$ * 0.67 =	**-24.8**
accumulation	VSS_{start} - VSS_{end} (1485 - 2074)		$V_{reactor}$/days * 500/50	* 1 =	**-5.9**
1st flush, foaming, stabilisation					**-27.5**

The loss of solids cannot be attributed to these 3 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 days, i.e. about 25 days ($SRT=VSS_{reactor}*V_{reactor}/WOS$).

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 rate was significantly lower (2004-07-08 to 2004-10-07: Q_{mean}= 29 m^3 d^{-1}) and the volatile suspended solids concentration higher (VSS_{mean}= 2.86 kg m^{-3}) comparing 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. about 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 days' 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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