Going for mainstream deammonification from bench- to full-scale for maximized resource efficiency

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Short title: Going for mainstream deammonification

ABSTRACT

A three-pronged coordinated research effort was undertaken by cooperating utilities at three different experimental scales investigating bioaugmentation, enrichment and performance of anammox organisms in mainstream treatment. Two major technological components were applied: density-based sludge wasting by a selective cyclone to retain anammox granules and intermittent aeration to repress nitrite oxidizers. This paper evaluates process conditions and operation modes to direct more nitrogen to the resource-saving metabolic route of deammonification.

Key words: anammox, deammonification, Demon, energy efficiency, nitrogen removal

INTRODUCTION

“Deammonification” is a two-step process where ammonia-oxidizing bacteria (AOB) aerobically convert half of the ammonia to nitrite and anammox bacteria anaerobically oxidize the residual ammonia using nitrite to produce nitrogen gas without the organic carbon substrate required for conventional heterotrophic denitrification. Deammonification is successfully used to treat ammonia-rich waste-streams such as dewatering sidestreams from anaerobically digested sludge. Since 2004 when the first full-scale deammonification plant was successfully implemented at Strass WWTP (Wett, 2007), about 30 DEMON®-plants are operational, under construction or under design, mainly in Europe and the United States.

This research looks further into the application of this emerging technology in cold and dilute municipal wastewater streams. Full-plant, or mainstream deammonification (MSD), is an innovative technology that can be compatible with existing wastewater infrastructure, often with minimal modifications. Process flowsheets using mainstream deammonification maximize energy recovery by diverting more particulate organic carbon away from the nitrogen removal process and directing it toward anaerobic treatment where energy can be recovered through anaerobic digestion. Additional benefits for the energy balance of wastewater treatment plants are expected from the lower oxygen demand for the metabolic N-conversion by deammonification versus the conventional nitrification/denitrification route. The share of potential stoichiometric oxygen savings of 60% that can be actually harvested depends on the flux of nitrogen channeled to the deammonification pathway by an optimized process scheme.

The proposed process scheme considers four main components:

- Enrichment of anammox (AMX) biomass by installation of cyclones in the wastage (WAS) of the mainstream system (Wett et al., 2012), similar to the sidestream DEMON approach (Figure 1)
- Bioaugmentation of AOB via the cyclone overflow of the DEMON-sidestream reactor to the main liquid process train
- Bioaugmentation of AMX by transferring mixed liquor from the DEMON-sidestream reactor to the main liquid process train
- Intermittent aeration regime in the mainstream aeration tanks in order to repress nitrite-oxidizing bacteria (NOB) by transient anoxia

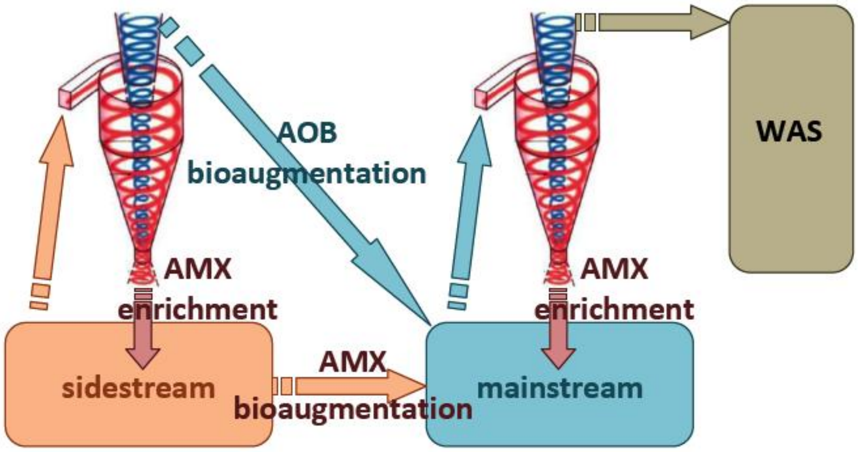


Figure 1 Biomass selection and transfer scheme to enhance full-plant deammonification.

METHODS

An ongoing three-pronged coordinated research effort is undertaken by cooperating utilities at three different experimental scales investigating bioaugmentation, enrichment and performance of anammox organisms in the mainstream treatment. Quantitative molecular techniques are used to track augmentation routes and monitor the population dynamics in the mainstream bioreactor. Activity measurements and other kinetic test results are translated into a dynamic model, which helps to develop efficient process schemes.

DC Water

Bench-scale sequencing batch reactor (SBR) systems with a volume of 10 L are operated at different operation modes (intermittent versus continuous aeration), temperatures (15°C versus 25°C) and different dissolved oxygen (DO) levels down to 0.05 mg DO/L (Omari et al., 2012). Different experimental protocols have been developed to monitor DO half-saturation values of AOB versus NOB.

Hampton Roads Sanitation District (HRSD)

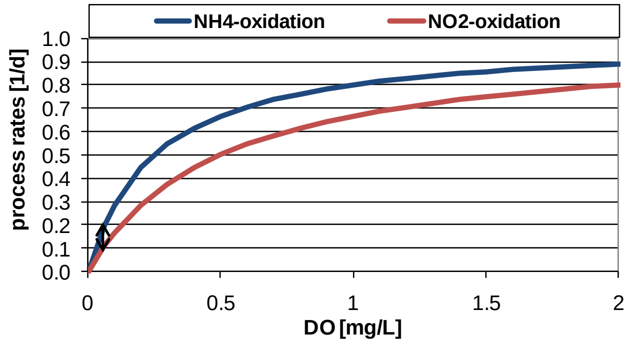
A pilot plant designed to accommodate a flow rate of about 0.1 L/s is operated to remove carbon in a high-rate stage and nitrogen in a low-rate stage (A/B process). The flexible set-up is developed to investigate both process options – separate process steps for nitrite production and consumption and simultaneous N-conversion.

Strass WWTP

The Strass plant is known as a net energy positive plant providing mainstream treatment by an A/B process and sidestream treatment by the DEMON process. Both simultaneous nitrification/denitrification (SND)-type of operation and modified Ludzack-Ettinger (MLE)-type of operation mode in the B-stage at different DO and on-line ammonia setpoints have been investigated. Anammox granules produced from sludge liquor treatment are seeded to the mainstream and retained and enriched by a hydro-cyclone classifier selecting for the high-density sludge fraction from the waste stream (Figure 1).

RESULTS AND DISCUSSION

The repression of nitrite oxidation is a precondition for all desired shortcuts in metabolic routes for nitrogen removal – for both the nitrite-shunt and deammonification. Within the nutrient removal research community it is a well-established theory that low oxygen concentrations promote the nitrite route (e.g. Wiesmann, 1994). A systematic literature review of oxygen affinity parameters (K_O) yielded a huge variation for AOBs from 0.1 to 1.45 mg DO/L and for NOBs from 0.3 to 1.1 mg DO/L (Sin et al., 2008). The average K_O-ratio of eight different data sources amounts to 1.64 indicating higher DO half-saturation for NOBs. One parameter recommendation of these is shown in Figure 2, resulting in significantly higher growth-rates for AOB especially at low DO-levels (arrow indicates 81% higher AOB-rates at a DO-level of 0.06 mg/L).



	AOB-growth Monod	NOB-growth Monod
μ_{max} [1/d]	0.9	0.7
Arrhenius	1.07	1.06
k_O [mg DO/L]	0.25	0.50

Figure 2 Frequently used parameter set (BioWin default values) for maximum AOB- and NOB-growth rates and oxygen affinity (K_O).

Following the theory of selective pressure on NOBs at low DO levels the operating DO level was gradually decreased during the long-term bench-scale experiment. Nitrification could be maintained at an SRT of 30 days and a significant nitrifier adaptation to low DO-levels down to 0.06 mg/L could be observed. However, most of the oxidized ammonia was converted to nitrate (Figure 3, left) and K_O tests revealed adaptation to lower values for NOB. Obviously the very limited nitrite availability was the process bottleneck. Anaerobic tests in the same reactor confirm anammox activity at 15°C (Figure 3, right) once nitrite has been spiked.

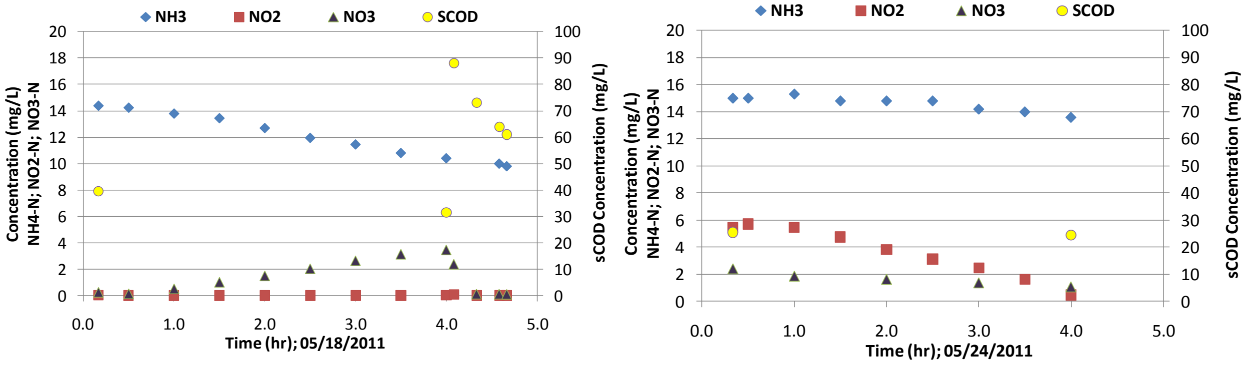
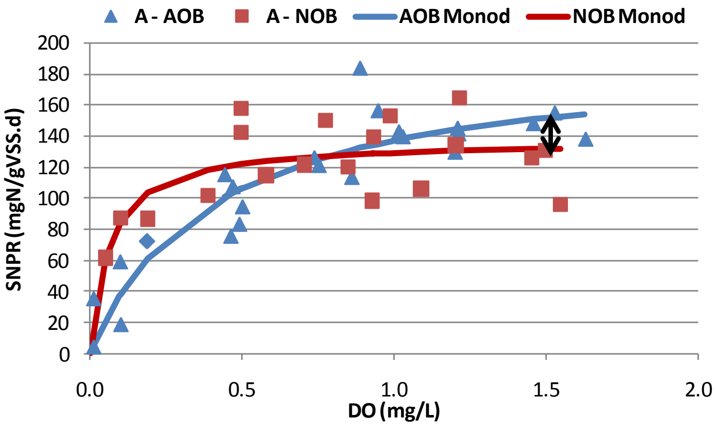


Figure 3 In-situ profiling of ammonia, nitrite, nitrate and soluble COD during an SBR-cycle of the intermittently operated reactor A at target DO of 0.06 mg DO/L and temperature of 15°C (left) and corresponding anammox activity profiles after spiking with 5 mg/L nitrite (right).



	AOB-growth Monod	NOB-growth Monod
$\mu_{max} * X_a / Y_a$	193	137
k_O [mg DO/L]	0.40	0.06

Figure 4 Specific nitrogen process rates in terms of ammonia removal per g VSS and day depending on the DO-setpoint of the intermittent aeration of the batch reactors. Measured data fitted to Monod expressions by applying least square error minimization (arrow indicates 15% higher N-processing rates of AOB at a DO-level of 1.5 mg/L).

Low food/mass ratio (F/M) bioassay-type tests were conducted using the bench-scale SBRs' sludge at Blue Plains to evaluate the AOB and NOB activities at various DO levels. These tests were conducted in a batch mode where a 1-L sample from the SBRs was spiked with ammonia when measuring AOB activity or nitrite when measuring NOB activity. The goal was to maintain a non-limiting level of substrate during the reaction time in the test. The test was carried out at various DO concentrations where the DO was held constant at a certain setpoint and the ammonia (or nitrite) reduction was determined by taking samples every 10–15 minutes. Then the DO level was adjusted to a new constant level and the sampling was repeated to determine the ammonia (or nitrite) reduction slopes at that DO level. Long-term data from bench-scale deammonification pilot summarized in Figure 4 consistently shows higher ammonia processing rates versus nitrite processing rates at low DO-concentrations (results in line with nitrification tests by Daebel et al., 2007). Monod, as the most commonly used kinetic approach to describe DO-dependent autotrophic growth (Figure4), seems appropriate to match characteristics of both NOB-performance with a moderate but continuous increase in rates (K_O = 0.40) and AOB-performance with a steep increase followed by a broad plateau (K_O= 0.06).

Looking at the chronological development of K_O at the bench-scale batch system, the NOB adapted well to the low DO operating range at setpoints between 0.06 and 0.3 mg/L (decreasing K_O in Figure 5, left) while the K_O for total nitrifiers remained rather stable. The full-scale results at Strass WWTP confirmed K_O values for total nitrifiers being more than double compared to NOB at operated DO set-points of 0.5 to 0.9 mg/L (Figure5, right). However, NOB-repression was not successfully achieved – neither at bench-scale nor at full-scale – as long as low DO-operation was applied (operation target indicated by the arrow in Figure 2). Once operation was switched to a higher DO-level at 1.5 mg/L (target indicated by the arrow in Figure 4) for the same process scheme, NOB-repression took effect.

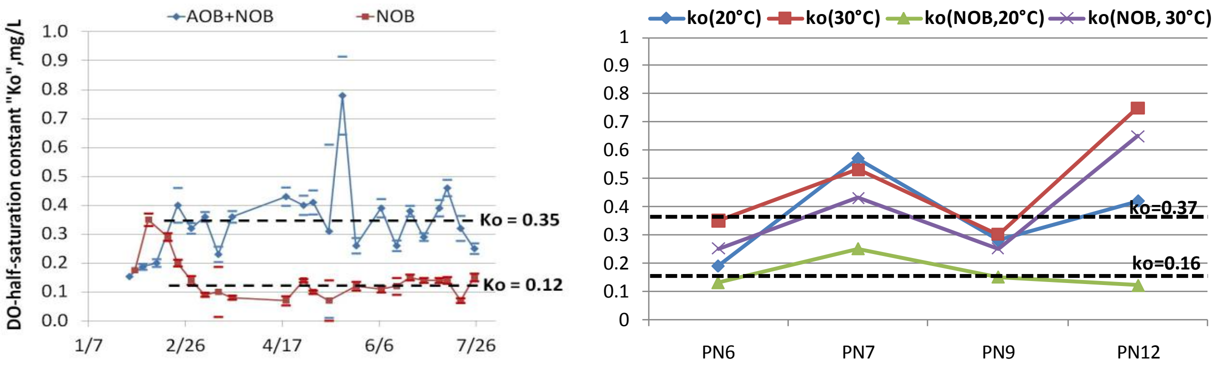


Figure 5 Comparison of K_O-values of total nitrifiers (AOB+NOB) and NOB only in bench-scale batch-reactors at Blue Plains WWTP (left) and in full-scale at Strass WWTP (right).

For the Strass plant, continuous seeding of biomass from the sidestream process and selective sludge wasting via the mainstream cyclones led to a visible enrichment of anammox granules in the aeration tanks and the enrichment development is still ongoing. In December 2011 the nitrite effluent concentration started to increase and when the main skiing tourism season started at Christmas (load increase from ca. 100 000 PE to more than 250 000 PE) the nitrite peaked to higher values than nitrate indicating enhanced NOB-repression (Figure 6).

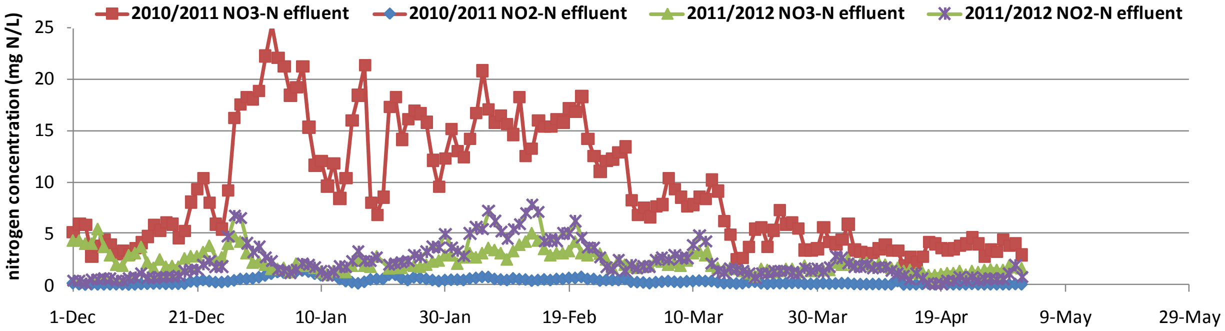


Figure 6 Comparison of this year's and last year's operational data of the full-scale pilot Strass indicating advanced NOB-repression (typically high nitrate level during main skiing tourism season; similar load and temperature conditions for both years).

The interaction of microbial players was more systematically investigated by aerobic and anaerobic activity tests. In 1-h aerobic ex-situ activity tests at 20°C incubation temperature increasing ammonia processing rates up to 7 mg N/g TSS/h have been observed throughout the experimental period with the measured NO_2/NO_3 ratio indicating NOB-repression up to 75% (Figure 7). After 2 h of anaerobic activity measurements at 20°C net removal of ammonia was finally achieved. Net ammonia removal includes both ammonia oxidation by AMX as well as ammonia release from the organic solids. The same activity test yielded relatively high nitrite reduction rates up to 1.3 mg N/g TSS/h compared to tests in earlier phases of AMX-enrichment (Figure 8).

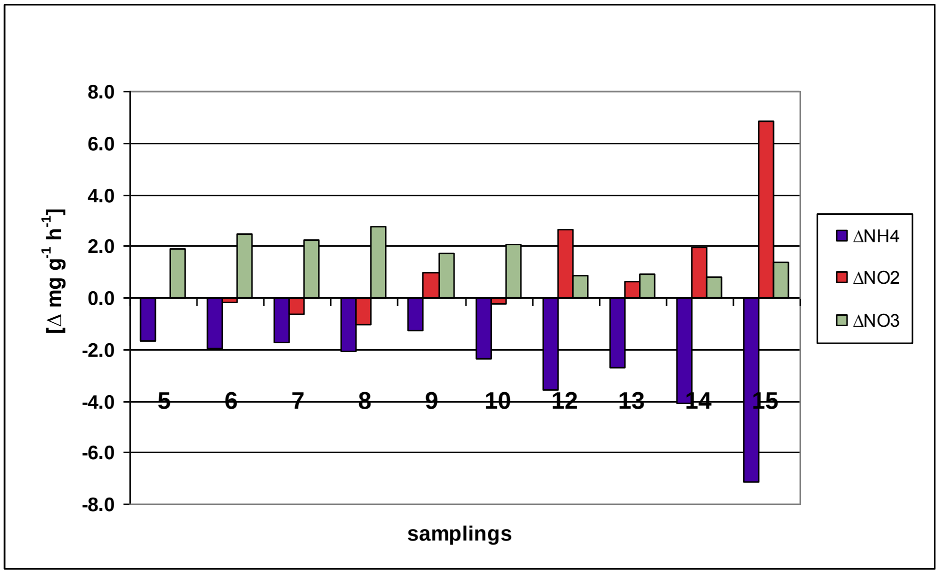


Figure 7 Results of ex-situ activity tests for aerobic ammonia oxidizers at 20°C (only ca. 25% of produced nitrite gets converted to nitrate at samplings 12 and 15).

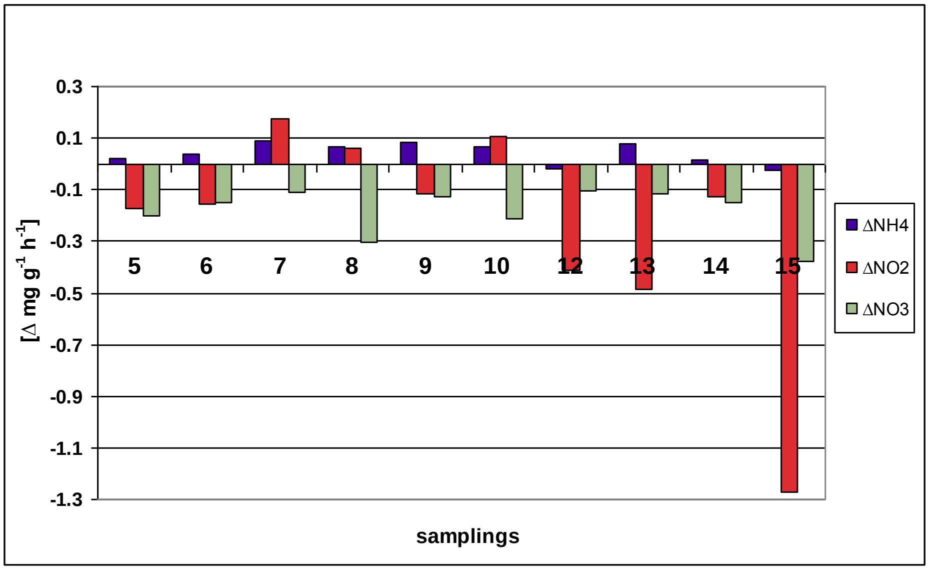


Figure 8 Results of activity tests for anaerobic ammonia oxidizers at 20°C (net ammonia removal at sampling 12 and 15 corresponds to the operation period displayed in Figure 6).

Operational data from high-DO intermittent aeration at bench scale and full scale demonstrate the feasibility of stable NOB-repression – but the initially targeted K_O-impact showed less effect than expected. So what is the crucial factor for NOB repression? For the evaluation of individual parameter sensitivities a numerical full-plant model was employed. In order to allow a steady state solution of simulated process scenarios all intermittent actions have been translated into continuous operations: e.g. intermittent aeration of the DEMON is mimicked by a continuous process using very high internal recycling rates between aerobic and anoxic compartments and periodical bioaugmentation flows are represented by continuous seed-fluxes (Figure 9). Default parameters of the BioWin simulator were used with the exception of K_O(AMX) = 0.4, K_O(AOB) = 0.37, K_O(NOB) = 0.16 (cf. Figure 5) and then the maximum growth rate for NOB was varied starting from the default value of 0.7/d down to 0.35/d. Simulated nitrogen profiles show a significant drop in nitrate concentrations at NOB growth rates below 0.5 and at 0.4 nitrate formation is almost completely stopped and nitrite is the major product (Figure 10).

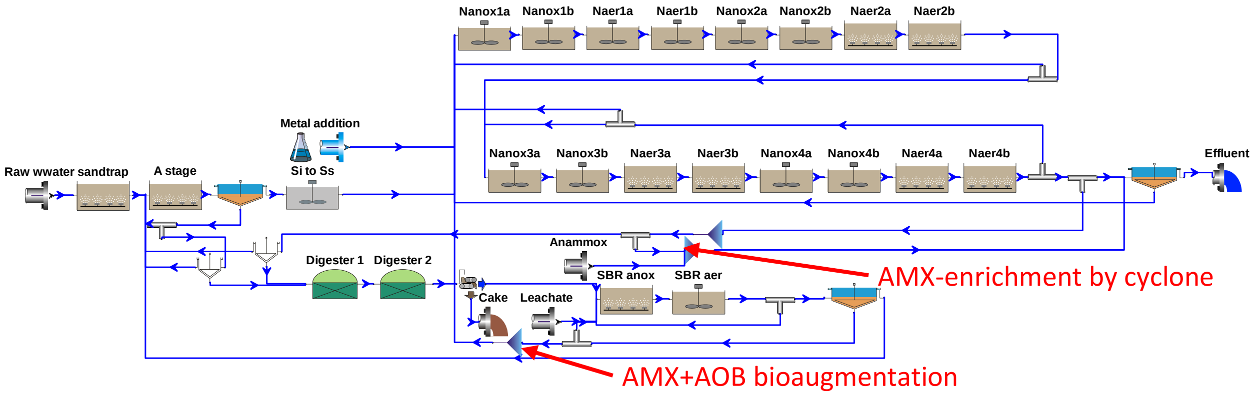


Figure 9 Full-plant model configuration of the Strass plant describing solids transfers for bioaugmentation between side- and mainstream and mimicking selective sludge wasting in the mainstream treatment lane by an anammox-enriched recycle flux based on retention efficiency of 75%.

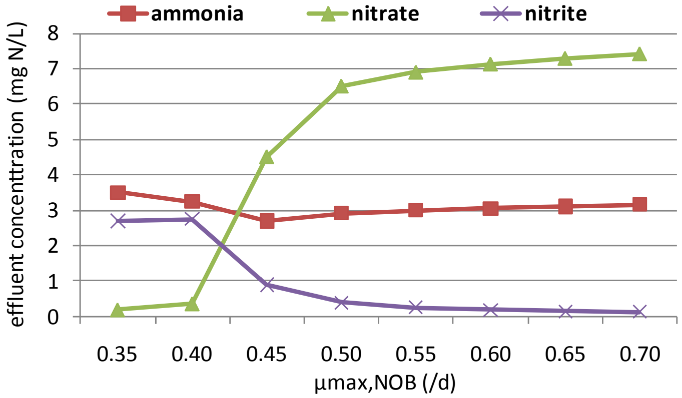


Figure 10 Simulated increase in nitrite versus decrease in nitrate concentrations at different lumped parameter maximum growth rates of NOB (cf. operational data in Figure 6).

Simulation results clearly demonstrate that NOB can hardly be repressed at typical maximum NOB growth, even at high bioaugmentation rates for AOB and AMX, the latter representing a competitor for nitrite. Obviously the maximum NOB growth-rate needs to be reduced by specific process conditions as they occur at rapid transitions from high to low DO levels. This slow-down impact on NOB growth has been lumped into the μ_{max}-parameter value in want of a more detailed model description. Simulated population dynamics draw a similar picture with an almost complete extinction of NOB undershooting default growth by 40% (Figure 11, left). The benefit from NOB-repression under the presented conditions is a reduction of 11% in oxygen uptake in the aerated zones (Figure 11, right). The anammox-concentration is calculated to be in the range between 30 and 35 mg COD/L, which is in the same order of magnitude as the red granule volume in mixed liquor estimated by image analyses tools (Wett et al., 2012).

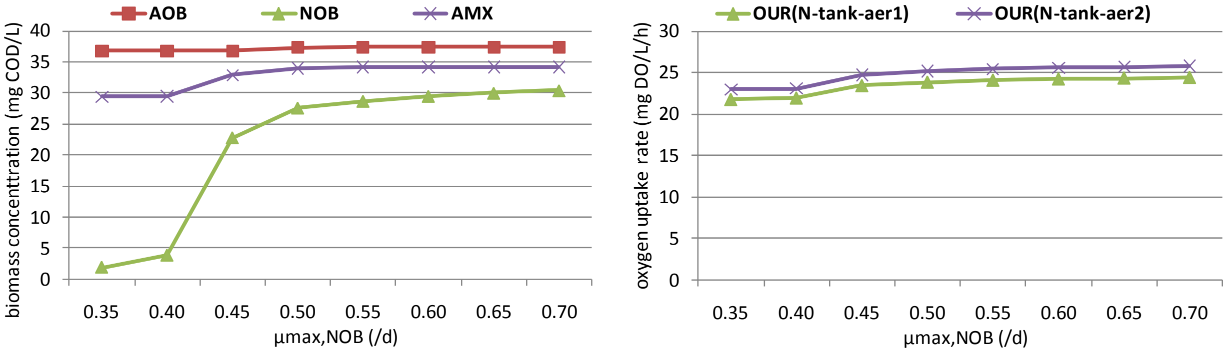


Figure 11 Simulated autotrophic biomass composition (left) and impact on oxygen uptake (right).

CONCLUSIONS

The success of mainstream deammonification depends, to a large extent, on the control of two crucial kinetic parameters for NOB repression and prevention of nitrate formation:

- competition between AOB and NOB for oxygen expressed by K_O (DO half saturation) and
- competition between AMX and NOB for nitrite determined by K_{NO} (nitrite affinity).

Transient anoxia turned out to be crucial process conditions to repress NOB growth. There are two potential explanations for the observed effect:

- a lag-phase in enzymatic activity when aeration is turned on and
- intermittent aeration interrupting metabolic conversions causing the formation of inhibitory intermediate products, e.g. nitric oxide.

A detailed analysis of the most relevant process mechanism and the optimization of control strategies is needed in future research work trying to direct more nitrogen to the resource saving metabolic route of deammonification.

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REFERENCES

Al-Omari, A., Wett, B., Han, H., Hell, M., Bott, C. & Murthy, S. 2012 Full-plant deammonification based on NOB-repression, AOB seeding, anammox seeding and successful retention. *Proc. IWA Conf. On Nutrient Removal, Harbin.*

Daebel, H., Manser, R. & Gujer, W. 2007 Exploring temporal variations of oxygen saturation constants of nitrifying bacteria. *Water Research* **41**, 1094–1102.

Sin, G., Kaelin, D., Kampschreur, M. J., Takács, I., Wett, B., Gernaey, K. V., Rieger, L., Siegrist, H. & van Loosdrecht, M. C. M. 2008 Modelling nitrite in wastewater treatment systems: a discussion of different modelling concepts. *Water Science & Technology* **58**(6), 1155–1171.

Wett, B. 2007 Development and implementation of a robust deammonification process. *Water Science & Technology*, **56**(7), 81–88.

Wett, B., Podmirseg, S. M., Hell, M., Nyhuis, G., Bott C. & Murthy, S. 2012 Expanding DEMON sidestream deammonification technology towards mainstream application. *Proc. IWA Conf. on Autotrophic Nitrogen Removal (SIDISA), Milan.*

Wiesmann U. 1994 Biological removal from wastewater. *Advances in Biochemical Engineering* **51**, 113–154

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