Simple improvement of digested sludge quality: is postaeration the key?

M. Vojtiskova, B. Satkova, J. Bindzar and P. Jenicek

ABSTRACT

Postaeration, where digested sludge is introduced into aerobic conditions, is a technology that could improve the quality of sludge and sludge liquor in many ways. Although it is a fairly simple process, only few data about the effect of postaeration have been published. In this study, batch experiments have been performed first, indicating that postaeration affects the total ammonia nitrogen (TAN) and sludge dewaterability. In the removal of TAN, both stripping and biological oxidation can play an important role depending on specific condition. Then the postaeration was investigated in a semicontinuous batch reactor. In addition, the effect of postaeration on the concentration of selected micropollutants such as pharmaceuticals, EOX and AOX was studied. The hydraulic retention time (HRT) of 8, 6, 4 and 2 days and different aeration intensities were tested. The TAN removal efficiency achieved was about 40–60%, sludge dewaterability expressed by sludge cake total solids (TS) concentration after dewatering improved relatively by 5–30%. In addition, TS degradation is also taking place and therefore the reduction of the amount of final sludge to be disposed could be even higher. The biggest changes in observed parameters were recorded at the longest HRT.

Key words | anaerobic digestion, dewaterability, postaeration, sludge stabilization, TAN removal

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INTRODUCTION

Sludge is an inevitable by-product of the wastewater treatment process. While it represents only 1–2% of the volume of treated wastewater, sludge accumulates a major portion of pollutants, whose treatment and disposal consequently typically account for up to 50–60% of the operating costs of the wastewater treatment plant (WWTP). The main task of sludge management is sludge stabilization, which involves converting sludge into a stable state for which minimal further sludge degradation is characteristic. Moreover, well-stabilized sludge should be to a high extent odourless, easily dewaterable and hygienic. More recently, recognition of the potential for recovering resources and energy from sludge, together with the implementation of stricter regulatory legislation related to sludge utilization, has seen effective sludge stabilization become an extremely important issue. Therefore, innovative technologies for the improvement of sludge properties are being sought.

While it is relatively simple to manage and produce efficient aerobic stabilization, it is characterized by high energy consumption. Conversely, anaerobic stabilization, which is generally applied in large WWTPs, enables energy to be recovered from the sludge in the form of biogas, but is more complex and requires a skilled operator. The postaeration of anaerobically digested sludge, however, is a promising process that combines the advantages and limits the disadvantages of both aerobic and anaerobic stabilization.

During postaeration, anaerobically digested sludge is introduced to aerobic conditions. Under these conditions, the decomposition of organic matter, which is hard to degrade in a single anaerobic stage, is intensified (Novak *et al.* 2011; Tomei *et al.* 2011). This process is believed to have other benefits too. It is possible to remove ammonia nitrogen with great efficiency, thereby reducing the amount of nitrogen recycled throughout the entire wastewater treatment process (Parravicini *et al.* 2008a, 2008b). And improved sludge dewaterability and odour minimization are other potential effects of postaeration (Kumar *et al.* 2006; Kevbrina *et al.* 2011; Tomei *et al.* 2011). Despite this, much more data are needed about the effects of postaeration on the quality and amount of anaerobically digested sludge produced.

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In terms of digested sludge quality, the dewaterability starts to be a key parameter, which is generally determined by the capillary suction time (CST) method. Depending on the experimental setup and conditions applied, the CST decrease reported has varied from 25% (Braguglia *et al.* 2014) up to 50% (Tomei *et al.* 2016). Although widely used, this method is not really suitable for WWTPs where the final sludge cake concentration needs to be evaluated. For this reason, it is desirable to find a method that complements or even replaces the CST method. Another required sludge quality improvement which is often discussed is related to the presence of micropollutants such as PPCPs (pharmaceuticals and personal care products) or halogenated organic compounds (AOX, EOX – adsorbable organically bound halogens, extractable organically bound halogens). It is important to know if the postaeration can change the sludge quality also in this respect. There is still a gap of knowledge in this field. As regards sludge liquor quality, the main actual operational problem is recirculation of high amounts of TAN back to the main wastewater treatment line and postaeration is a promising solution in this respect (Parravicini *et al.* 2008a, 2008b).

In this study, we conduct batch experiments under different conditions (type of sludge, temperature, aeration mode etc.) to analyze the impact of the postaeration process on the following parameters of digested sludge – total ammonia nitrogen concentration (TAN), sludge dewaterability, and the final amount of sludge produced. Subsequently also semicontinuous batch reactor experiments were performed with the aim of evaluating the effect of main technological parameters such as hydraulic retention time (HRT) and aeration intensity on the TAN removal efficiency, dewaterability and the removal of selected micropollutants. Until now, all presented studies have reported quite high HRT (10–15 days), but our aim was to evaluate the potential of the postaeration process at much shorter HRT. To better understand the impact on dewaterability, we aimed to develop a new method for the determination of sludge dewaterability that produces results that are more reliable for operational purposes.

MATERIAL AND METHODS

Sludge source

Samples of digested sludge were collected at four WWTPs in the Czech Republic. A short description is given in Table 1.

Table 1 | General specifications of the WWTPs (PE – population equivalent)

WWTP	Type of WWTP	Sludge stabilization
A	<100,000 PE, mechanical-biological, activated sludge process (R-D-N)[a]	Anaerobic mesophilic (40–42 °C)
B	>100,000 PE, mechanical-chemical-biological, activated sludge process (R-D-N)[a]	Anaerobic thermophilic (55 °C)
C	>100,000 PE, mechanical-biological, activated sludge process (R-An-D-N)[b]	Anaerobic thermophilic (52–56 °C)
D	>100,000 PE, mechanical-biological, activated sludge process (R-D-N)[a]	Anaerobic mesophilic (35–37 °C)

[a] R-D-N – compartmentalized activated sludge process with regeneration, denitrification and nitrification zones.

[b] R-An-D-N – compartmentalized activated sludge process with regeneration, anaerobic, denitrification and nitrification zones (Kos *et al.* 1992).

Both mesophilic and thermophilic digested sludges were used for the experiments to obtain more general data. The main difference in the sludge composition was found in the TAN concentration (398–1,954 mg/L), on the other hand the sludges were quite uniform in the volatile solids/total solids ratio (VS/TS) (0.57–0.61).

Postaeration reactor

For batch experiments, a special glass cylinder of 4 L and 10 cm in diameter was used in which postaeration of 1.5 L of digested sludge was performed (air flow 90 $L \cdot h^{-1}$) by coarse-bubble aeration.

In the long-term experiments, a glass reactor of 8 L was used. The contents of the reactor (3.2 L of digested sludge) were continuously mixed by magnetic stirrer and aerated. As the air source, several compressors with an air flow of 300 and 400 $L \cdot h^{-1}$ or their combination were used. During these experiments, the amount of sludge withdrawn and supplied daily depended on the tested HRT (e.g. 400 mL for an HRT of 8 days).

Analytical methods

During both batch and semicontinuous experiments, the following parameters were determined: pH value, inorganic nitrogen concentration (nitrite, nitrate, total ammonia), COD (chemical oxygen demand), TOC (total organic carbon), total and organic dry mass and CST. Samples for determination of the concentration of dissolved COD, TOC and inorganic nitrogen were centrifuged (14,926 g for 10 min) and filtered through a glass fiber filter (0.45 μm). Analyses were performed in triplicates and according to standard methods (APHA 1998) or Czech standards (modified CSN EN 75 7452 (nitrite nitrogen), CSN 75 7455 (nitrate nitrogen), CSN ISO 15705 (COD). The other methods used are described below.

CST

CST is a simple parameter and a method used for the characterization of sludge dewaterability. The CST is the time required to collect a unit volume of filtrate of a sludge undergoing filtration in a standard-sized CST funnel, when placed upon a standard grade of chromatography paper. The original circular setup introduced by Baskerville & Gale (1968) and Whatman-17 filter paper were used to conduct the measurement. During this measurement, no polymer was added.

Centrifuge dewatering method

The modified method for determining sludge dewaterability is based on measuring the quantity of water removed by centrifugation and the quantity of sludge cake remaining at the bottom of the centrifuge tube. A similar method was used by To *et al.* (2016). For the centrifuge dewatering method, about 25 mL of the sludge was poured into a pre-weighed centrifuge tube and then weighed again. The sample was centrifuged (14,926 g for 10 min) and the sludge liquor was decanted for further analysis. The quantity of removed water was calculated as the difference between the weight of the sludge and the weight of the sludge cake. The sludge cake concentration was calculated from the weight of the sludge cake and its concentration.

AOX (adsorbable organically bound halogens)

AOX were analyzed according to the Czech standard CSN EN 16166. This determination is based on isolation of organic halogenated compounds on activated carbon (AC). The AC is then washed with a mixture of $NaNO_3$ and HNO_3 to remove the inorganic halides and burned in an oxygen atmosphere. The organically bound halides then form the corresponding hydrogen halides and are determined by microcoulometric argentometric titration. The result is expressed in weight of chlorides (Bindzar & Jenicek 2012). This parameter was determined only in sludge samples from WWTP D.

EOX (extractable organically bound halogens)

EOX were analyzed according to U.S. EPA 9023 (1996). This determination is based on extracting the sample with a suitable organic solvent and subsequently analyzing the extract. Differences may apply in the extraction agent used, the extraction method and the analytical termination. In this study, the solvents applied were ethylacetate and n-hexane (Bindzar & Jenicek 2012). This parameter was determined only in sludge samples from WWTP D.

Micropollutants

The concentration of selected micropollutants (pharmaceuticals) was determined using ultra-high performance liquid chromatography coupled to tandem mass spectrometry (UHPLC-MS/MS). Sludge samples were firstly dried and then the optimized liquid-liquid extraction (LLE) was applied.

The sample preparation was as follows: 2 ± 0.1 g of a solid matrix was weighed into a 50 mL centrifuge tube and spiked with surrogate standards at a concentration of $5\ \mu g \cdot kg^{-1}$ and at two spiking concentrations of target analytes. Samples were vortexed for 5 s after the addition of the standards. Then, 10 mL of the extraction solvent MeOH: ACN (1:1 v/v) +100 μL of FA (formic acid) was added to the glass tube and samples were homogenized by the multi-tube vortexer for 60 s, which is a sufficient time for proper homogenization. For extraction, samples were shaken for 30 min by the head-over-head shaker (at mid-speed) and subsequently centrifuged for 5 min at 2,296 **g**.

After centrifugation, an aliquot of 1 mL was pipetted into a glass tube and diluted 10 times with 9 mL of 0.1% FA, yielding a theoretical concentration of $0.1\ ng \cdot mL^{-1}$ of the surrogate standards. The full volume of the tube was transferred into a 10 mL syringe and filtered through cellulose filters – 5 mL to waste and the rest to an amber vial. An aliquot of 250 μL was taken from the vial into another amber vial and diluted four times with 750 μL of 0.1% FA, yielding theoretical concentrations of $0.025\ ng \cdot mL^{-1}$ and $0.05\ ng \cdot mL^{-1}$ (for demanded LOQ) of the target analytes, respectively. For the surrogate standards, the theoretical concentration was $0.025\ ng \cdot mL^{-1}$.

Determination of pharmaceuticals and biocides was done by ACQUITY UPLC® I-Class liquid chromatograph/ Xevo® TQ-S with multi-mode mass spectrometer (Waters). Separation was achieved using 0.01% FA: ACN – 98:2 (v/v) on ACQUITY BEH Phenyl column (50 × 2.1 mm, 1.7 μm particle size, Waters); the MS was operated simultaneously in both positive and negative ionization modes using ESI mode. The injection volume was 50 μL and the column temperature was set to 40 °C. The limits of quantification ranged from 1.4 and 118.8 $\mu g\ kg^{-1}$ (Holicek 2017). Micropollutants were analysed only in sludge samples in phase 1B of the first semicontinuous experiment.

RESULTS AND DISCUSSION

Batch experiments

Information about all batch experiments, tested sludge and conditions applied are given in Table 2. In experiment No. 5 the sludge was aerated intermittently from the beginning (45 min ON, 15 min OFF each hour); in parallel experiment No. 6 the sludge was aerated continuously from the beginning and then from day 10 intermittently (60 min ON, 60 min OFF). In experiments No. 7 and 8 the sludge was aerated intermittently from day 7 (120 min ON, 120 min OFF).

Nitrogen removal

TAN was significantly removed in all batch experiments – see Figure 1(a). It must be pointed out that especially at the beginning of each experiment TAN was stripped out rather than biologically removed due to the high pH value of the sludge. Figure 1(c) is showing a typical relation between the pH value and the TAN stripping efficiency. At the start of the test, the pH increased due to CO_2 release, which stimulated the TAN stripping. Simultaneously with the TAN stripping, the pH decreased and the stripping slowed down. Afterwards the biological oxidation of the TAN began, causing a further pH decrease. Nitritation took place in five experiments and in the same number of experiments nitrates, formed by subsequent nitratation, appeared in relatively low concentrations. The contribution of stripping and biochemical oxidation to TAN removal, calculated on the basis of concentrations, is shown in Figure 1(b). While in experiment No. 3 TAN was removed mainly biochemically, in experiments No. 5 and 7 biochemical oxidation did not take place and TAN was completely stripped out. It is, therefore, apparent that the configuration of the postaeration process could influence the contribution of biochemical oxidation on the whole TAN removal.

The extent of biological oxidation was lower in experiments with intermittent aeration (5–8), indicating that continuous aeration is required to initiate nitritation activity. In addition, the intermittent aeration in the second phase of experiment 6 did not bring the expected reduction of nitrites, probably due to the lack of organic substrate for efficient denitritation. Another explanation for the lower efficiency of biological oxidation in experiments (5–8) is the use of thermophilic sludge in which ammonia oxidizing bacteria could be more suppressed. However, the following continuous experiments, achieving high nitritation efficiency with thermophilic sludge, did not confirm this hypothesis.

Dewaterability change

To determine the dewaterability, the standard CST method and the modified centrifuge dewatering method were used and the results were compared. Although the CST method is widely used, it turns out not to be quite suitable for assessing dewaterability because it characterizes mainly the rate of water release. Therefore, a method that would be able to complement or even replace the CST method was sought. While the parameter CST_S decreased only slightly, remained unchanged or even increased during the experiments, the sludge cake dry mass determined by the

Table 2 | Parameters of batch experiments

Experiment number	Temperature [°C]	WWTP (source of the sludge)	Type of the sludge	Experiment duration [d]	Aeration regime
1	25	A	Mesophilic	12	Continuous
2	36	A	Mesophilic	12	Continuous
3	25	A	Mesophilic	16	Continuous
4	36	A	Mesophilic	16	Continuous
5	36	B	Thermophilic	15	Intermittent
6	36	B	Thermophilic	15	Intermittent
7	36	B	Thermophilic	15	Intermittent
8	36	C	Thermophilic	15	Intermittent

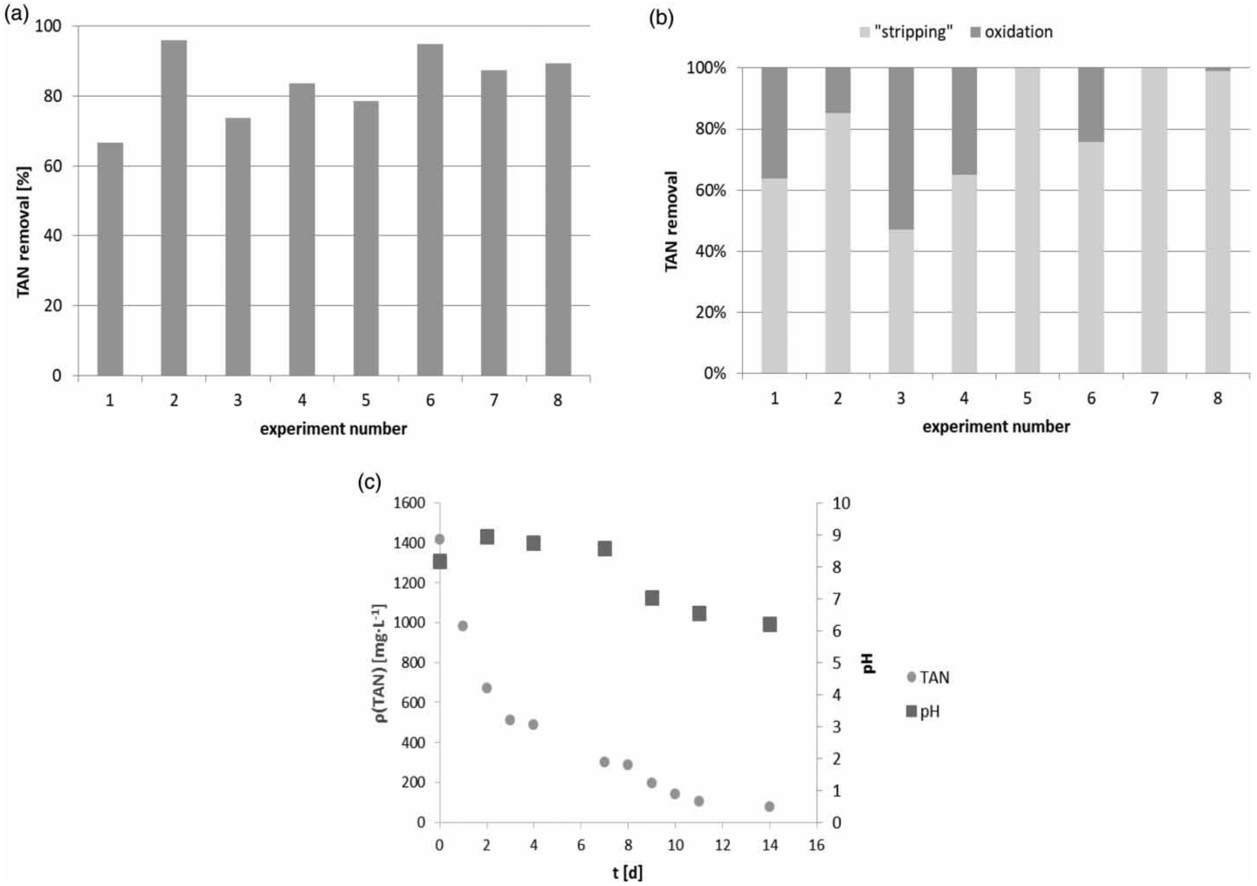

Figure 1 | (a) TAN removal in each batch experiment; (b) TAN removal through biochemical oxidation and stripping; (c) typical TAN and pH value during experiment No. 6.

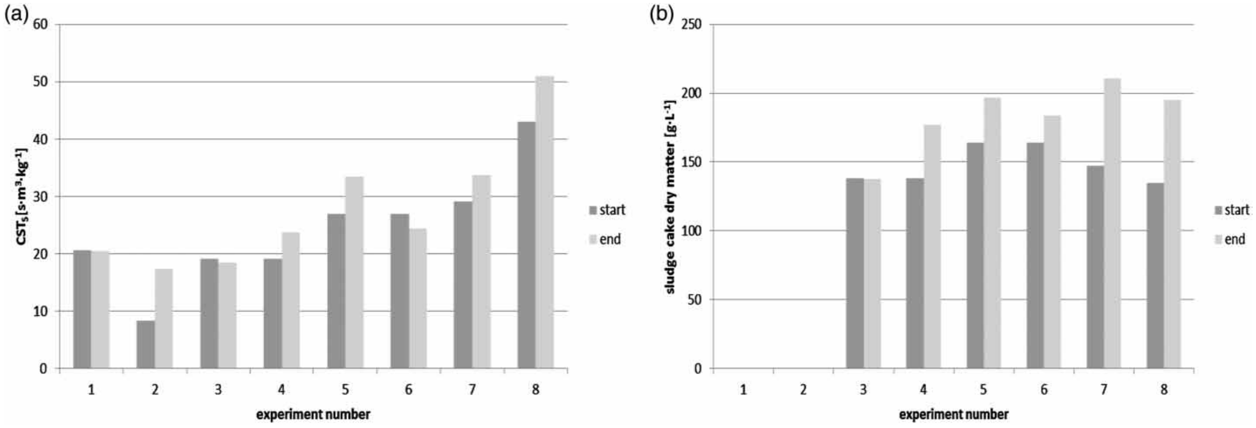

Figure 2 | (a) Initial and final specific CST values in each batch experiment; (b) Initial and final sludge cake dry mass determined by centrifugation dewatering method in each batch experiment. In experiments No. 1 and 2 the centrifugation method was not used.

centrifugation method increased in all cases and these results make the comparison of dewatering results very interesting (Figure 2).

In both the CST and centrifugation methods, the non-conditioned sludge; that is, sludge without coagulant or flocculant added, was tested. Many authors recommend the use of conditioned sludge for dewaterability testing to be closer to full scale conditions. The CST method was primarily developed to provide a wealth of data on the effect of chemical conditioners on sludge dewaterability (Vesilind

1988). However, our aim was to compare the change in sludge quality, and we feared that the flocculant selection and flocculant dose optimization could interfere with the sludge quality assessment.

Dry matter change

During the experiments, sludge dry matter was also analyzed as the concentration of total suspended solids (TSS). Except experiments No. 5 and 6, in which the concentration of the measured suspended solids slightly increased probably due to insufficient homogeneity of the samples, the sludge was degraded in all experiments, in one case even by 39% (Figure 3). It was proved that in some cases the reduction of the sludge amount could be really significant.

AOX and EOX removal

To study the effect of postaeration on the concentration of AOX and EOX, an extra batch experiment was performed using the sludge from WWTP D, which is characterized by high concentrations of these substances. The conditions were identical to those of experiment 8. During the experiment, the AOX concentration slightly decreased (Table 3) with a removal efficiency of 11%. The results for EOX removal depended on the extraction agent used. In the case of ethylacetate extraction, the EOX were eliminated with a removal efficiency of 27%. When extraction into hexane was applied, the removal efficiency was even higher at −79%. In general, hexane gives smaller yields since only hydrophobic substances are extracted within. Although a slight decrease of monitored parameters was observed during storage of the sludge as well, these results suggest another positive effect of postaeration.

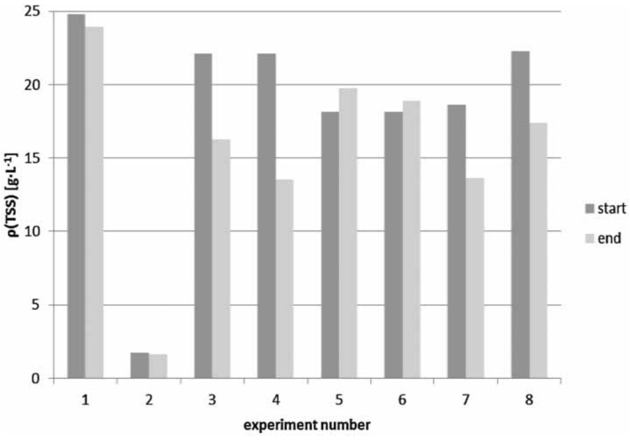

Figure 3 | Initial and final TSS concentration in each batch experiment.

Table 3 | Results for AOX and EOX removal

Compound	Initial concentration [mg · kg^{-1}]	Final concentration [mg · kg^{-1}]
AOX	950 ± 70	840 ± 60
EOX (ethylacetate extraction)	330 ± 30	240 ± 20
EOX (hexane extraction)	48 ± 4	9.9 ± 2

Semicontinuous experiments

Based on the batch test results, three long-term experiments were conducted to investigate postaeration in a semicontinuous batch reactor system. Characteristics of each experiment are given in Table 4. Sludge B (thermophilic) was tested in semicontinuous experiments. Results were evaluated for each HRT separately.

Nitrogen removal

TAN was efficiently removed at all tested HRTs (Figure 4). In the first and second experiments, a decrease of pH value associated with a significant increase in nitrite nitrogen concentration during biochemical oxidation of TAN was observed after the start-up phase. Nitrite nitrogen was a significant product in most experiments (Figure 5). The highest nitrite nitrogen concentrations achieved in experiments 1, 2 and 3 were 393, 465 and 6 mg · L^{-1} respectively. Fluctuation of nitrite concentration was very high because of changes in pH and the inhibitory effect of free nitrous acid or free ammonia (Jenicek *et al.* 2004).

The values of the related selected parameters measured over the same days are listed in Table 5. Low nitrite nitrogen concentrations detected in the third experiment indicate that TAN was removed mainly by stripping. The HRT of

Table 4 | Characteristics of individual long-term experiments

Experiment number	Experiment duration [d]	Designation	HRT [d]	Air flow [m^3 · h^{-1}]	Aeration intensity [m^3 · m^{-2} · h^{-1}]
1	235	1A	8	0.3	11.8
		1B	6	0.3	11.8
		1C	4	0.3	11.8
2	114	2A	4	0.6	23.5
		2B	2	0.9	35.3
3	71	3A	2	0.6	23.5
		3B	2	1.2	47.1
		3C	2	2.0	78.4

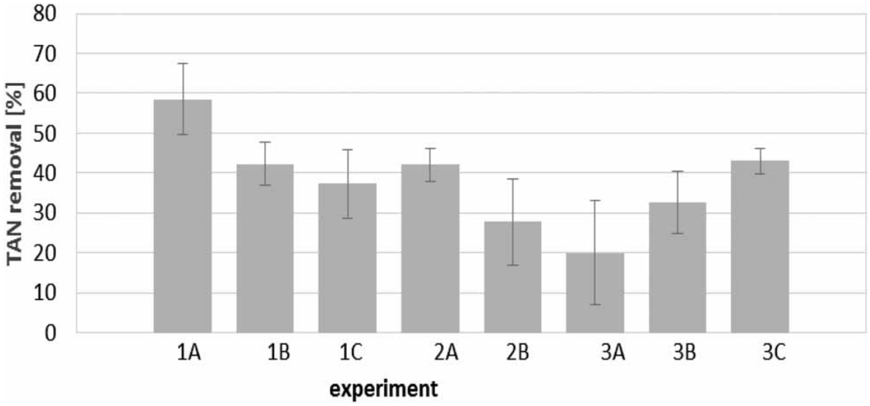

Figure 4 | TAN removal in individual experiments.

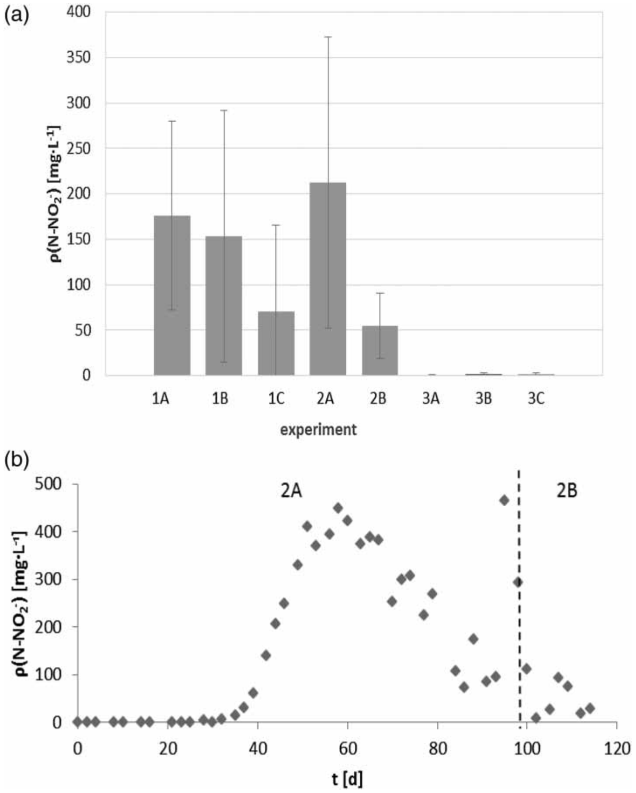

Figure 5 | (a) Mean values of nitrite nitrogen concentrations in the effluent. (b) Nitrite nitrogen concentrations in the effluent during the second experiment.

2 days applied in this experiment is probably too short for the formation of nitrites even at increased aeration intensity. This statement was later confirmed by fluorescence *in situ* hybridization (FISH), by which ammonia oxidizing bacteria (AOB) from the *Beta proteobacteria* group were detected in samples taken in the second experiment at HRT 4 days (Figure 6). After reducing the HRT to 2 days the AOB were no longer observed.

Table 5 | Comparison of the contribution of nitrogen oxidation to the removal of TAN

Experiment designation	pH value [–]	T [°C]	ρ(N-NO$_2^-$) [mg · L^{-1}]	ρ(TAN) in the influent [mg · L^{-1}]	ρ(TAN) in the effluent [mg · L^{-1}]
1A	6.25	20.5	393	1,284	331
2A	7.60	24.2	449	1,131	687
3B	8.77	19.5	6	1,287	767

The disadvantage of biological nitrogen removal by nitritation is the relatively long HRT in this case. Therefore, it is possible to focus alternatively only on nitrogen stripping, which can be realized with significantly shorter HRTs. So far, many authors have tried to avoid stripping in their experiments by keeping the reactor at approximately neutral pH and using limited aeration intensity. Conversely, by adjusting the pH of the sludge in the reactor to higher values, it

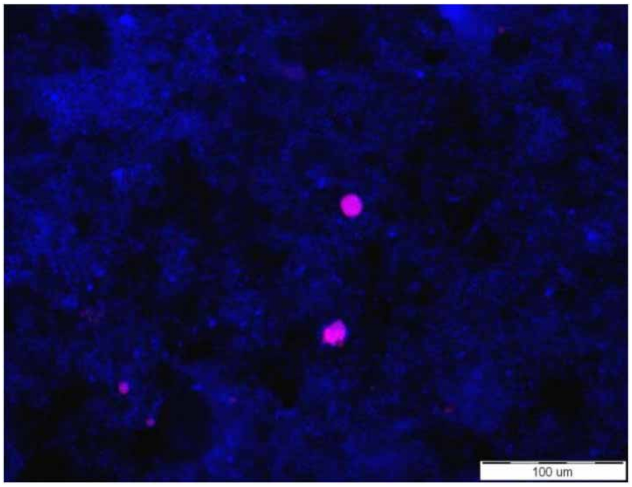

Figure 6 | Confirmation of the presence of AOB by FISH in the sludge after postaeration.

would be possible, especially at the beginning of the experiment, to release as much ammonia as possible and to collect it as the ammonia nitrogen into the acid to form ammonium salts useful as fertilizer. It is advantageous to utilize simultaneous carbon dioxide stripping to manipulate the pH. De Vrieze *et al.* (2016) indicate that ammonium sulfate formed during the stripping of ammonia at temperatures of 60 to 80 °C achieves a market price of €1 per kg of nitrogen comparable to the market price for fertilizers.

Such nitrogen recycling can be feasible also from a quantitative point of view. For example, let us take a model case of a WWTP with a capacity of 100,000 PE with $12\ \mathrm{g \cdot (PE \cdot d)^{-1}}$ of total nitrogen in the influent. If we assume that 10 to 20% of the initial nitrogen ends up in the sludge liquor after anaerobic stabilization and 50% will be then extracted, the WWTP would be capable of producing the following amount of nitrogen: 0.6 to $1.2\ \mathrm{g \cdot (PE \cdot d)^{-1}}$, i.e. 60 to 120 kg per day, respectively 20 to 40 t annually.

Dewaterability change

In all experiments, influent and effluent sludge dewaterability was determined. For the purpose of evaluating the change in dewaterability, two parameters were selected: specific CST and sludge cake dry mass after centrifugation. While the sludge cake dry mass increased in all tested HRT, up to 34% in the final phase of the first experiment (Figure 7), an increase instead of decrease of specific CST did not show improved dewaterability. This increase could be caused by change in extracellular polymers at high pH, by the need for dosing an anti-foaming agent due to excessive foaming, and last but not least by the fluctuating quality of digested sludge (particularly the total solids concentration). Our experiences indicate CST as a good parameter for assessment of the rate of water removal but not for the efficiency of water removal. However, the dewaterability tests also need to be done with the conditioned sludge in the next experiments to verify possible improvement in the informative value of the data.

Dry matter change

The concentration of TSS in the effluent decreased with longer HRT, thus reducing the amount of sludge. At shorter HRTs, there was no significant difference in TSS concentrations; in some phases, an increase was even observed (Figure 8). This increase was probably caused by the evaporation at high aeration intensities and the growth of aerobic biomass cannot be excluded either.

Effect on micropollutants

During the first experiment's phase 1B, micropollutants, particularly pharmaceuticals, were analyzed in the sludge samples before and after postaeration. The results are shown in Table 6. Higher concentrations of some pharmaceuticals in the effluent were probably due to the matrix effect. Partial removal has been observed with metoprolole, tramadole and valsartan (30–50%). Diclofenac and furosemide have even been completely eliminated. Although only spot samples were analyzed and repeated measurements would be necessary, these results could confirm the statements of some authors (Braguglia *et al.* 2015) that some micropollutants can be removed or at least influenced by postaeration of the digested sludge. However, the mechanism of micropollutants removal needs to be examined in more detail. It does not have to be related only to aeration, but also sorption and desorption processes, influenced by changes in pH and other parameters during postaeration, are important (Ivanova *et al.* 2018).

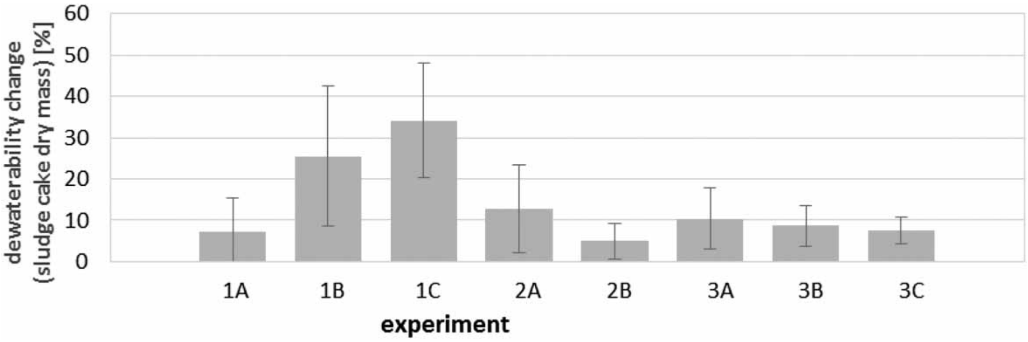

Figure 7 | Changes of sludge cake dry mass in individual experiments.

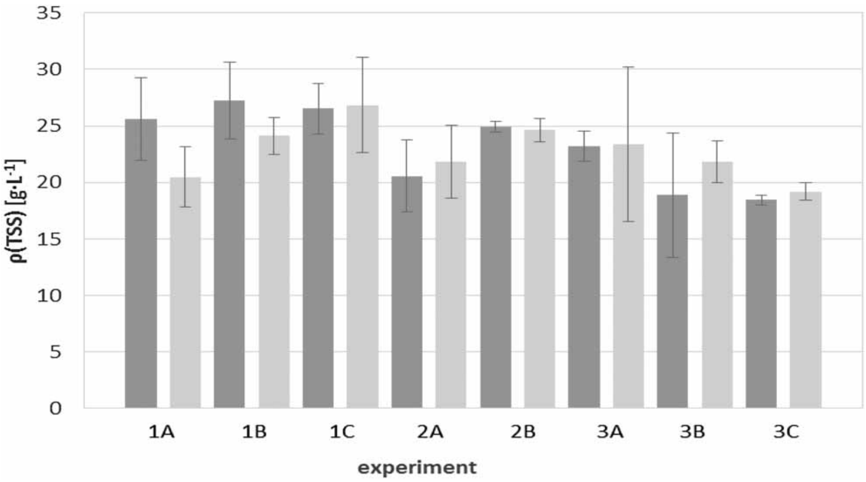

Figure 8 | TSS concentration in the influent (red) and effluent (blue). Please refer to the online version of this paper to see this figure in colour: http://dx.doi.org/10.2166/wst.2019.409.

Table 6 | Results of micropollutants determination using UHPLC-MS/MS (LOD – limit of detection)

Compound	Influent concentration [μg·kg^{-1}]	Effluent concentration [μg·kg^{-1}]
Caffeine	under LOD (2.8)	99
Carbamazepine	473	397
Citalopram	427	509
Diclofenac	679	Under LOD (2.3)
Fluoxetine	Under LOD (1.5)	53
Furosemide	1,025	Under LOD (3.6)
Metoprolole	255	172
Sertraline	1,193	1,158
Tramadole	143	97
Valsartan	531	268

Other potential benefits and drawbacks of postaeration

While the postaeration process may seem to bring benefits mostly, certain drawbacks and uncertainties cannot be overlooked. Firstly the long retention time needed for biological nitrogen oxidation would result in relatively large volume and thus investment costs required for the construction of the postaeration reactor. On the other hand the postaeration can be potentially applied in existing sludge storage tanks with the HRT considerably shorter. Then only part of benefits will be implemented however, it will be achieved as low cost solution.

Another economic aspect is related to energy consumption. Postaeration has the potential to remove part of TAN out of the wastewater treatment system and thus reduce the nitrogen load of activated sludge technology and to improve sludge dewaterability. This could have a positive impact on the energy balance of WWTPs, because the energy consumption of the post-treatment could be sufficiently compensated by the reduced energy consumption for nitrogen removal in wastewater treatment and lower energy consumption in case of sludge drying or incineration.

One of the operational risks is the foaming that can occur if the sludge is not sufficiently stabilized or contains filamentous foaming bacteria. The resulting foam can then affect the postaeration process operation and needs to be controlled (Vojtískova *et al.* 2019).

CONCLUSIONS

Presented experiments have shown that the postaeration decreases TAN concentration, the amount of sludge and also improves sludge dewaterability. Although the HRT applied in experiments was in some cases high for economically feasible application on the WWTP, the obtained results significantly increased our knowledge of the process.

The results of the semicontinuous experiments demonstrated that the effect of postaeration is strongly dependent on HRT and aeration intensity. In terms of nitrogen removal, the longest HRT was the most efficient. During this period, slight dewaterability improvement and reduction of both total and organic dry matter were also observed. In shorter HRT only TAN removal and increase in sludge cake dry mass occurred. HRT of 2 days does not guarantee conditions suitable for biochemical nitrogen oxidation.

Thus the postaeration can be applied in two variants:

A) Short retention time (about 2 days) – low cost solution which can bring sludge dewaterability improvement and limited TAN removal.

B) Long retention time (4–8 days) which can bring sludge dewaterability improvement and biological TAN oxidation, resulting in higher TAN removal efficiency and a decrease of sludge amount.

Exploration and mainly the optimization of the postaeration process will continue at bigger scale, with the help of pilot unit operated on selected WWTP.

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