Aeration of anaerobically digested sewage sludge for COD and Nitrogen removal: optimization at large-scale

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Abstract: The paper will report about the experiences matured at an Austrian large wastewater treatment plant of 720,000 population equivalents, where anaerobically digested sewage sludge is further stabilised under aerobic conditions. Enhanced stabilisation of the anaerobically digested sludge was required at the plant in order to get a permit for landfill disposal of the dewatered stabilized sludge. By implementing a post-aeration treatment (SRT~6d; 36°C) after anaerobic digestion the organic content of the anaerobically well digested sludge can be decreased by 16%. Investigations on site showed that during digested sludge post-aeration anoxic phases for denitrification are needed to provide stable process conditions. In this way the pH value can be kept in a more favourable range for micro-organisms and concrete structures. Additionally, inhibition of the biological process due to nitrite accumulation can be avoided. By optimising the aeration/pause ratio ~45% of total nitrogen in digested sludge can be removed. This significantly improves nitrogen removal efficiency at the wastewater treatment plant. NH₄-removal occurs mainly through nitritation and denitritation with an efficiency of 98%. The costs/benefit analysis shows that post-aeration of digested sludge results in an increase of total annual costs for wastewater treatment of only 0.84%, corresponding to 0.19 Euro/pe/a. Result of molecular biological analyses (DGGE) indicate that all four ammonium-oxidizing bacteria species present in activated sludge can survive anaerobic digestion, but only two of them can adapt in the digested sludge post-aeration tanks. Additionally, in the post-aerated digested sludge a further ammonium-oxidizing bacteria species was identified.

Keywords: Enhanced sewage sludge stabilisation; digested sludge aeration; nitrogen removal in reject water; nitritation and denitritation, ammonia-oxidizing bacteria.

INTRODUCTION

The EU Landfill Directive 99/31/EC sets targets for the continuous reduction of biodegradable waste sent to landfills. The Austrian landfill regulation (BGBI, 1996) consistently with the EU directive does not allow the disposal of solid waste with an organic content (TOC) exceeding 5%, beyond the year 2004. However, for mechanical-biologically treated waste the less strict standard for the calorific value < 6,600 kJ/kg TS applies. At a large Austrian municipal wastewater treatment plant (WWTP) anaerobically digested (lime conditioned) dewatered sludge had been disposed at a mono-landfill site. In order to get a permit for the continuation of this economically favourable disposal option an aerobic biological post-stabilisation process was developed and implemented.

Lab-scale investigations (Parravicini et al., 2004) showed that the degree of stabilisation of anaerobically digested sludge can be enhanced by introducing a post-aeration treatment. For the sludge of the WWTP the investigations indicated that the organic solids in the anaerobically digested sludge (30 days solid retention time, 38°C) can be reduced by 20% through post-aeration (5 days SRT, 30°C). With this process it was possible to meet the standards for the calorific value of the lime-conditioned dewatered sludge.

Post-aeration of digested sludge enables further nitrogen removal by intermittent aeration through nitrification and denitrification. This additional nitrogen removal is of great importance at the WWTP in consideration, as the nitrogen removal capacity of the plant is limited by unfavourable TKN/COD ratio of the raw wastewater.
caused by a high portion of industrial effluents. Intermittent aeration also results in reduced energy requirements and is relevant for stable buffer capacity in the process and to avoid nitrite toxicity. The paper reports on the large-scale results after the implementation of the digested sludge post-aeration treatment.

Additionally, molecular biological analyses (DGGE) were conducted to compare the species of ammonium-oxidizing bacteria (AOB) present in post-aerated digested sludge with the ones found in the activated sludge tanks. Within the nitrification process AOB are responsible for the oxidation of ammonia into nitrite. The fundamental enzyme of AOB the ammonia monoxygenase (amoA) is the limiting step by the biochemical reaction. The gene encoding the active site of the amoA can be exploited as molecular marker for studying AOB diversity in the environment (Hornek et al., 2006). Besides the scientific interest of the results, molecular biological analyses might provide further information for process optimisation. Changes in the composition of the AOB biocenosis might be used as parameter for process optimization in the post-aeration plant.

METHODS

Post-aeration of anaerobically digested sewage sludge

WWTP is designed for 950,000 population equivalents (pe) and has an actual mean load of 720,000 peCOD. At this plant thickened primary and secondary sludge (sludge age: 20 d) are digested anaerobically under mesophilic conditions (37°C). Average SRT in the digesters is 30 days. Process conditions for anaerobic digestion (e.g. stable temperature and pH-value, high SRT, screw type mixers) are favourable for very good stabilisation result. Table 1 shows characteristic data of the anaerobically digested sludge.

Table 1 Characteristic data of anaerobically digested sludge.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Mean value</th>
<th>Parameter</th>
<th>Mean value</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD (mg/L)</td>
<td>29,500</td>
<td>VSS (%)</td>
<td>54.3</td>
</tr>
<tr>
<td>Acetic acid (mg/L)</td>
<td>12</td>
<td>TKN (mg/L)</td>
<td>2,400</td>
</tr>
<tr>
<td>TSS (g/L)</td>
<td>36.0</td>
<td>NH₄-N (mg/L)</td>
<td>950</td>
</tr>
</tbody>
</table>

Table Legend: TSS: Total Suspended Solids; VSS: Volatile Suspended Solids; TKN: Total Kjeldahl Nitrogen

The post-aeration plant for the digested sludge consists of 4 square concrete tanks each with a volume of 1,500 m³. They can be operated either in series or in parallel. Each tank is equipped with a cone-aerator (135 kW, ø 3m). Aeration is controlled by the oxygen concentration measured on-line in every single tank with a LDO probe (set value: 1 mgO₂/d). Temperature and pH-value are also monitored on-line. The tanks are covered to avoid odour problems. The exhaust air is treated in biofilters. The airflow in the tanks can be regulated to keep the process temperature by ~36°C. The effluent of the anaerobic mesophilic digesters reaches the aeration tanks by gravity flow. The effluent of the post-aeration plant is transferred to storage tanks before dewatering in chamber filter-presses with lime conditioning.

COD mass balances

Organic matter removal efficiency of digested sludge post-aeration was calculated from the difference between COD and VSS mass flows in influent and effluent over periods in steady state conditions. From first results it was concluded that at large-scale it is difficult to get representative samples. COD mass balances were therefore checked by oxygen consumption. Oxygen uptake rate (OUR) was measured on site using short term respirometry (Standard Methods, 1995). OUR was also derived from energy consumption for aeration (P) as described by Svardal et al. (1998). Under steady state conditions OUR can be calculated as follows if the characteristic data of the aeration equipment are available:

\[
\text{OUR} = \text{SAE} \times \frac{\text{P} \times (S_0^* - S_0^*)}{S_0^*} (\text{kgO}_2/\text{d})
\]

\[
\text{OUR} \quad \text{oxygen consumption due to bacterial respiration (kgO}_2/\text{d})
\]

\[
\text{SAE} \quad \text{standard aeration efficiency under process conditions (kgO}_2/\text{kWh})
\]
For COD mass balances oxygen uptake by nitrification (OUR\(\text{NO}_3\)) has to be detracted from total O₂-consumption:

\[
\text{OUR}_{\text{COD}} = \text{OUR} - \text{OUR}_{\text{NO}_3} \quad \text{(kg O₂/d)}
\]

\[
\text{OUR}_{\text{NO}_3} = (\text{TKN}_{\text{in}} - \text{TKN}_{\text{out}}) \times 4.33 \quad \text{(kg O₂/d)}
\]

TKN\(_{\text{in}}\)  Nitrogen influent load (kg N/d)

TKN\(_{\text{out}}\)  Nitrogen influent load (kg N/d)

Oxygen consumption by oxidation of ammonia to nitrate (kg O₂/kg NO\(_3\)-N produced)

In case ammonia oxidation proceeds only to nitrite (nitrification) the specific oxygen consumption decreases by 25%. The equivalent oxygen consumption for denitrification must also be considered in the COD mass balance: 2.86 kg O₂/kg NO\(_3\)-N removed.

The organic matter removal efficiency of sludge post-aeration can also be estimated using a heat balance. Removal of organic compounds by respiration results in heat production. Under aerobic conditions according to McCarty (1964) 14 to 15 MJ heat are set free per kg oxygen consumed. Nitrification results in lower heat production: 10.5 MJ per kg O₂ consumed (COD equivalents) for complete oxidation of ammonia to nitrate and 5.6 MJ/kg O₂ for partial oxidation of ammonia to nitrite. During the digested sludge post-aeration the released thermal energy results in a temperature increase in the tanks. The largest part of the heat however leaves the plant via the exhaust air (evaporation, temperature increase). The thermal energy transferred to the exhaust air can be estimated from the increase of the air enthalpy using e.g. a Mollier diagram for humid air. The heat loss through the concrete walls of the tanks turned out to be negligible.

**Analyses of sludge samples**

All measurements were carried out according to German standard methods for water, wastewater and sludge analysis (DIN). Sludge samples were analysed twice a week. Samples for dissolved COD, NH\(_4\), NO\(_3\), NO\(_2\), PO\(_4\) were centrifuged (1800 rpm, 15 min) and filtrated (0.45 m) immediately after sampling and stored at 4°C no longer than 24h until measuring.

**Molecular biological analyses of ammonia-oxidizing bacteria**

Several sludge samples of primary sludge, activated sludge, anaerobically digested sludge and post-aerated digested sludge were collected at the WWTP at regular time intervals and stored frozen at -20°C until analysis. DNA extraction, polymerase chain reaction (PCR) and denaturing gradient gel electrophoresis (DGGE) were performed according to Hornek et al. (2006). The DGGE gels were digitalised and cluster analyses were performed using Gelcompare Software (Version 3.0, Applied Maths, Belgium). Different similarity coefficients (Jaccard, Pearson) and algorithms (single linkage, complete linkage, UPGMA-Unweighted Pair Group Method with Arithmetic Mean) were applied for the statistical analysis of the DGGE results.

**RESULTS AND DISCUSSION**

**Post-aeration of anaerobically digested sludge**

Within the first 250 days of operation the 4 tanks of the sludge aeration plant were operated completely aerobically in 2 lines with 2 cascades each (phase 1). Nitrite was the main end product of ammonia oxidation under the process conditions applied: 36°C, 0.5 to 1 mg O₂/L (Figure 1). Between day 140 and 250
concentrations of free nitrous acid up to 0.4 mg HNO$_2$-N/L were measured in the tanks. The accumulation of nitrite from day 140 was caused by the increase of the set value for oxygen concentration to 1 mgO$_2$/L. Nitrifying bacteria and especially the ammonia oxidisers are known to be inhibited already at concentrations of unionized nitrous acid of 0.06 mg HNO$_2$-N/L (Anthonisen et al., 1976). In addition, the low pH value during phase 1 was detrimental for the process by shifting the dissociation equilibrium in favour of the un-ionized (inhibiting) acid form.

After day 250 the operation of the aeration devices was switched from continuous to intermittent in order to maintain the pH-value in a more favourable range both for micro-organisms and concrete structures as well as to prevent nitrite accumulation (phase 2). Between day 250 and 280 the aeration/pause ratio was 50%, with intervals lasting 1 hour respectively. From day 280 the duration of the aeration intervals was increased up to 70%, considering low nitrite and nitrate concentrations and residual ammonia in the effluent. Table 2 shows the average process parameters for different operation phases. Ammonia could be only partially oxidised under the process conditions applied. After removing nitrite inhibition through intermittent aeration (phase 2) effluent NH$_4$-concentration dropped from 330 to 150 mg NH$_4$-N/L in average.

Table 2 Process parameters during post-aeration of digested sludge (average values)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Phase 1 continuous aeration (120-249)*</th>
<th>Phase 2 intermittent aeration (250-340)</th>
<th>Phase 3 4 cascades, cont. aeration (400-460)</th>
<th>Phase 4 recirc: pump inter.aeration (495-540)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Influent flow (m$^3$/d)</td>
<td>930</td>
<td>1000</td>
<td>1300</td>
<td>1200</td>
</tr>
<tr>
<td>Influent COD load (t COD/d)</td>
<td>27.4</td>
<td>29.7</td>
<td>34.2</td>
<td>28.0</td>
</tr>
<tr>
<td>SRT (d)</td>
<td>~ 6.5</td>
<td>~ 6</td>
<td>~ 4.5</td>
<td>~ 5</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>36.4</td>
<td>36.2</td>
<td>38.3</td>
<td>39.5</td>
</tr>
<tr>
<td>Oxygen conc. (mgO$_2$/L)</td>
<td>0.9</td>
<td>0.5</td>
<td>0.2</td>
<td>0.6</td>
</tr>
<tr>
<td>pH value</td>
<td>6.4</td>
<td>6.9</td>
<td>7.6</td>
<td>6.9</td>
</tr>
<tr>
<td>NH$_4$-N conc. (mg NH$_4$-N/L)</td>
<td>330</td>
<td>150</td>
<td>350</td>
<td>23</td>
</tr>
<tr>
<td>NO$_2$-N conc. (mg NO$_2$-N/L)</td>
<td>350</td>
<td>60</td>
<td>35</td>
<td>15</td>
</tr>
<tr>
<td>NO$_3$-N conc. (mg NO$_3$-N/L)</td>
<td>80</td>
<td>25</td>
<td>&lt; 1</td>
<td>&lt; 1</td>
</tr>
</tbody>
</table>

* Time intervals in days

A further step to optimise the process was to shift sludge loading from 2 lines into a cascades mode (all 4 tanks in series) on day 340 (phase 3). During the previous operation mode it had been difficult to achieve an equal partition of the inlet load to the 2 lines. The disadvantage of having one under loaded line was that in these tanks the oxygen concentration stabilised at higher values (+ 0.3 mgO$_2$/L), thus promoting nitrite accumulation. A continuous manual adjustment of the aeration intervals was therefore necessary for stable process operation. The attempt to control aeration only via pH was not satisfying. Periodic analysis (twice a week) of ammonia, nitrite and nitrate were necessary for process control.

The optimisation of the plant operation was also hampered by the limited aeration capacity of the plant. The -value was overestimated during the design of the aeration capacity ( =0.9). From on site OUR measurements an -value of 0.55 could be derived under process conditions, which is much lower than the design value. Aeration efficiency under process conditions ( SAE) and referred to standard conditions (20°C, 1013 hPa) was estimated 1.1 kgO$_2$/kWh on average.

From day 340 to 400 the aeration intervals were extended stepwise to face the increased oxygen demand of the influent load (digested sludge). Between day 340 and 460 sludge production at the WWTP increased by about 30%, due to the higher flow of waste water to be treated. Although from day 400 all 4 tanks were continuously aerated, the oxygen concentration was below 0.25 mg O$_2$/L even in the last cascade. These process conditions clearly affected ammonia oxidation, as indicated by the higher concentrations measured in the aerated sludge (400 mgNH$_4$-N/L). From day 460 the organic load in digested sludge decreased to previous values. To avoid nitrite accumulation intermittent aeration (55% aeration/pause ratio) was re-established.
A decisive measure for process optimisation was met on day 495 (phase 4) with the introduction of a recirculation pump from the last in the first cascade (recycling ratio ~300%). The effect was to equalize the oxygen uptake rate gradient in the tanks, leading to an increased oxygen concentration in the first high loaded cascades. Additionally, the recycled sludge acted as continuous inoculation of the first cascade with nitrifying biomass. Under these conditions it was possible to enhance ammonium oxidation up to 98%, decreasing the effluent concentration to approx. 23 mg NH₄-N/L. From this time onwards also low effluent concentrations of nitrite and nitrate could be achieved.

**COD removal**

The estimation of the COD removal efficiency during post-aeration of digested sludge at large-scale is a challenging task. The main problem is that the removal efficiency is in the same range as the accuracy of the measured parameters. The heterogeneity, complexity and viscosity of the sludge matrix make sampling, sample treatment and analytics difficult. At the WWTP the quality of sampling was checked using a total phosphor mass balance. During phase 2 the four methods applied to estimate COD removal (analytics, respirometry, energy consumption and heat balance) resulted in values ranging from 13 to 23% (Figure 2). Results indicate that heat balances can be successfully applied to estimate COD-removal efficiency. Up to now it is not possible to rank the methods applied for their accuracy and reliability. Considering the relative narrow variation among the values obtained an average value of 16% COD removal can be assumed as a most probable result. This value is below the COD removal efficiency achieved in lab-scale (Parravicini et al., 2004). Presumably hydrolysis of organic solids during sludge aeration in lab-scale was favoured by the higher oxygen concentration applied (2 to 4 mgO₂/L). During phase 3 which was characterised by a higher influent organic load, COD removal increased to approx. 25% (results not shown). In all other operation phases COD-removal accounted 15% to 17%.

Figure 1 NH₄-N, NO₂-N and NO₃-N concentrations in aerobically post-treated sludge. The concentration of NH₄-N in digested sludge (influent) averaged 950 mg NH₄-N/L.
Figure 2 COD mass balance for the period from days 280 to 340. Balance recovery calculated as:
\[
\frac{(\text{COD}_{\text{effluent}} + \text{COD}_{\text{removed}})}{\text{COD}_{\text{influent}}}
\]

Nitrogen removal
Total nitrogen removal achieved by intermittent aeration averaged 45.6%, corresponding to 1.2 t N/d. Nitrogen mass balances for two different periods one before and one after the implementation of the post-aeration of digested sludge (Figures 3) show that through the sludge post-treatment nitrogen removal efficiency at the WWTP increased by 5 percentage points. The nitrogen load in reject water is decreased by 0.8 t N/d, which represents about 13% of the influent nitrogen load to the WWTP. Also the nitrogen load reaching the landfill is reduced by 0.4 tons of nitrogen per day, which corresponds to a reduction of 23.5%.

Cost/benefit analysis of post-aeration of digested sludge
Table 3 shows costs and benefits resulting from digested sludge post-aeration at the WWTP in consideration. Benefits result from the reduction of sludge to be dewatered and disposed (-0.29 Euro/pe/a). These benefits cover the costs for sludge aeration and maintenance of the plant (+ 0.26 Euro/pe/a).
Table 3 Costs and benefits at the WWTP resulting from digested sludge post-aeration

<table>
<thead>
<tr>
<th>Costs/Benefits</th>
<th>Sludge post-aeration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Investment costs (Euro/pe/a)</td>
<td>+ 0.28</td>
</tr>
<tr>
<td>(25 years payback time, 3% interest)</td>
<td></td>
</tr>
<tr>
<td>Costs for maintenance, manpower, air treatment and pumps (Euro/pe/a)</td>
<td>+ 0.017</td>
</tr>
<tr>
<td>Electric energy for sludge aeration (kWh/d)</td>
<td>8,640</td>
</tr>
<tr>
<td>Costs for sludge aeration (Euro/pe/a)</td>
<td>+ 0.26</td>
</tr>
<tr>
<td>Dispensed nitrification in activated sludge tank of NH₄-N in reject water (Euro/pe/a)</td>
<td>- 0.06</td>
</tr>
<tr>
<td>Dewatered sludge to dispose (t/d)</td>
<td>- 11.8</td>
</tr>
<tr>
<td>Saved disposal costs (Euro/pe/a)</td>
<td>- 0.20</td>
</tr>
<tr>
<td>Costs reduction for time and FeCl₆ (Euro/pe/a)</td>
<td>- 0.096</td>
</tr>
<tr>
<td>Electric energy costs for methane loss</td>
<td>-</td>
</tr>
<tr>
<td>Balance operational costs (Euro/pe/a)</td>
<td>- 0.68</td>
</tr>
<tr>
<td>Balance annual costs (Euro/pe/a)</td>
<td>+ 0.19</td>
</tr>
<tr>
<td><strong>Operational costs of WWTP</strong>: 10.8 Euro/pe/a</td>
<td></td>
</tr>
<tr>
<td><strong>Annual costs of WWTP</strong>: 22.9 Euro/pe/a</td>
<td>+ 0.84%</td>
</tr>
</tbody>
</table>

*Costs*: operational and annual costs of the WWTP are based on the results of a detailed process benchmarking (Kroiss & Lindner, 2003): electric energy: 0.66 Euro/kWh; sludge disposal: 34 Euro/t(digested sludge); conditioning chemicals: 16.07 Euro/t(digested sludge); Digestion sludge aeration: aerobic/anoxic ratio: 70%; VSS removal: 16%; N-removed by nitrification/denitrification: 1.2 t/N (incl. NH₄ from VSS degradation); 0.03 NH₄-N/CODrem.; aS/E=1 kgO₂/kWh.

Results of the costs/benefits analysis were compared with the operational and annual costs of the WWTP. The relative increase of operational costs is negligible. Annual costs increase by 0.84%, corresponding to 0.19 Euro/pe/a. If the actual landfill disposal had to be replaced by another form of disposal (e.g. incineration), disposal costs would rise from currently 34 Euro per ton of dewatered sludge (40% TS) to approx. 50 Euro/t. In this case annual costs for digested sludge aeration decreased to 0.1 Euro/pe/a.

**Molecular biological analyses**

Analyses were conducted on a total of 82 sludge samples collected at the WWTP at different time intervals. DNA extraction was successful in 83% of the samples. DNA isolation in anaerobic digested sludge turned out to be difficult due to its heterogeneous matrix. PCR and DGGE analyses were very well reproducible.

Results indicate that exclusively AOB species belonging to the *Nitrosomonas* cluster were present in the sludge samples analysed (primary sludge, activated excess sludge, anaerobically digested sludge, post-aerated digested sludge). Activated excess sludge samples comprise four AOB species. All these four AOB species could be found in anaerobic digested sludge, indicating that AOB can survive long term exposition to anaerobic milieu (SRT in digesters: 30d). During post-aeration of digested sludge AOB biocenosis changes under the process conditions applied e.g. ~38°C, ~5.5 SRT, higher ammonia concentration and low oxygen concentration. Only two of the AOB species present in activated sludge could adapt in the post-aeration stage. Additionally a third AOB species was identified in the post-aerated sludge. This third species occurs only in this type of sludge and can not be found either in primary sludge, or in activated sludge and anaerobically digested sludge.

According to DGGE analyses the different operational conditions applied in the post-aeration tanks did not influence the composition of the AOB biocenosis. The same AOB species were found e.g. in sludge samples collected during phase 1, characterized by high effluent concentrations of nitrite and ammonia, and in the ones belonging to the following phases under optimized process operation (intermitted aeration). Unfavourable process conditions for AOB (reflected by higher NH₄ effluent concentrations) apparently affected more AOB activity than their diversity. This result is somewhat surprising considering the relative high eco-physiological specificity of AOB. Considering that molecular biological analyses provide only qualitative information on AOB species it can not be excluded, that varying process conditions caused mainly concentration changes within the AOB biocenosis.

On the basis of the results obtained analysis of AOB species in post-aerated sludge can not be applied as supplemental parameter for the optimization of process operation.
CONCLUSIONS
At an Austrian WWTP post-aeration (SRT ~6d; 36°C) of anaerobically digested sludge (SRT ~30d; 38°C) resulted in an additional organic solids removal of approx. 16%. In large-scale operation a 45.6% nitrogen removal could be achieved by intermittent aeration, improving significantly nitrogen removal at the WWTP (+5.5%). NH4-N removal through nitrification/denitrification was complete (98%). A decisive measure for process optimisation was the introduction of a recirculation pump from the last in the first cascade of the sludge post-aeration plant. The costs/benefit analysis indicates that post-aeration of digested sludge leads to a very small increase of the actual total annual costs by +0.84% corresponding to 0.19 Euro/pe/a. Digested sludge aeration is therefore an interesting option for WWTP with an insufficient N-removal and high specific costs for disposal of dewatered sludge. Further advantages of the process are the compact and simple construction and the easy operation.

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