Evaluation of the reject waters from co-digestion of solid wastes from agro-industries in a municipal WWTP

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Abstract

Overcapacities of anaerobic digesters at municipal WWTPs are frequently used for the treatment of organic wastes in order to increase the biogas production. However, “co-digestion” of organic wastes leads to additional nitrogen loading and to additional loads of non-biodegradable COD. The effects of (co-)digestion of organic wastes from agro-industries (slaughterhouses, dairies and leather industry) on the wastewater cycle have been evaluated in full-scale investigations at Leoben WWTP with a capacity of 90,000 pe where the methane production was increased from 700 to more than 1,700 Nm³ CH₄ per day. For this evaluation, mass balances for COD and nitrogen have been applied to estimate the fluxes of these substances. Application of this method is described in detail. As the additional loadings, it was found that related to methane production less nitrogen is released from the organic wastes than from the waste sludge. While the ammonia nitrogen load in the effluent from sludge digestion was about 100 g NH₄-N per Nm³ of CH₄ produced, in the effluent from the digestion of organic wastes only 70 g NH₄-N/Nm³ CH₄ were found. The decrease in the COD removal efficiency after digestion of the organic wastes started was not regarded as significant enough to be seen as a consequence of the treatment of external substrate.

Keywords Anaerobic digestion; co-digestion; mass balances; municipal WWTP; organic wastes

Introduction

In Austria, as in most countries of Central Europe, municipal WWTPs with a design capacity of more than 20,000–50,000 pe (= population equivalents) are mostly equipped with mesophilic anaerobic digestion. In general, the energy from the digester gas (“biogas”) is used in combined heat and power (CHP) generation applications. In Austria, about 80 CHP plants with an installed capacity of approximately 17.5 MWel and an electricity generation of 100 GWh/a are currently in operation (Austrian Energy Agency, 2005). Although CHPs enable the simultaneous production of thermal energy and electricity with the focus on maximising the overall efficiency, it has to be taken into consideration that combustion-based CHP units need a lot of maintenance and are therefore often not economic at smaller plants, the more so as CHPs below the power range of 100 kW are not very energy efficient. The net electrical efficiency of modern “biogas-CHPs” within an installed capacity of about 250 kW is defined in the range of 37%, whereas small CHPs below an electric capacity of 100 kW generate a net electrical efficiency lower than 30% (ASUE, 2005). At medium-sized WWTPs, below approximately 50,000 pe, the most effective way to use the energy of this biogas would be heat generation only. However, to enable an economic plant operation this requires a continuous and rather constant heat demand in the vicinity of the WWTP, e.g. in green houses, spas, but also in public and private houses for heating and warm water. An alternative to CHPs at medium-sized WWTPs are small gas turbines, so called micro-gas turbines in the power range of 30–500 kW, which are favourable in terms of low emissions and maintenance costs but are affected by a lower energy efficiency in comparison with internal...
combustion engines of the same power size (Invernizzi et al., 2006). Another possibility for the efficient use of the digester gas could be the infeed into public gas grids. However, this solution requires an integrated analysis of the economic efficiency, including regional and local circumstances, e.g. load characteristics of the gas grid, quality specifications for the infeed-gas as well as the cleaning and conditioning of the biogas (Theissing, 2006).

The anaerobic digesters of WWTPs in Austria do often have overcapacities. Partly, they have been overdimensioned and partly the overcapacities are due to seasonal peak loads, primarily from winter (or summer) tourism, for which the treatment plants have been dimensioned. In order to use these overcapacities, about 10 to 20 years ago, mainly in tourist areas with a lot of hotels and restaurants, the operators of WWTPs started to add organic wastes (mostly fat, oil and kitchen waste) to their digesters. This led to a significant increase in the gas production at these plants, and in the case of fat and oil there is no additional equipment required and the amount of sludge produced is virtually not increased. Meanwhile, some municipal WWTPs with free capacities in their digesters started to use these also for organic wastes from the food processing industry.

The WWTP of Leoben, situated in central Austria, with a design capacity of 90,000 pe is equipped with two anaerobic digesters. As the mean loading of the plant is only about half of the design capacity, one of the two digester is enough for sludge treatment. Starting in December 2004, the other digester was used for the fermentation of organic wastes, so far only from agro-industries (slaughterhouses, dairies and leather production). As a result, the gas production exceeds the capacity of the existing CHP unit which now produces twice the electrical power the WWTP needs for its operation. The “excess power” is fed into the power grid. In the future, also biowastes from households should be used as substrate. To prepare these biowastes for the fermentation process, special mechanical devices (screen and pulper) are necessary, that are presently installed. Moreover, the management of the WWTP plans to invest in one or two new digesters of the same size.

For the use of the energy from the digester gas in the future, three different ways are considered:

- Installation of CHP units with usage of the produced heat in a spa.
- Cleaning of the biogas (removal of CO₂, H₂S, etc.), so that the remaining methane gas can be fed into the regional gas grid or used as fuel for the municipal car park.
- Direct use of the biogas in a steel works situated about 7 km from the WWTP.

Starting in June 2005 a comprehensive study has been carried out to investigate the fermentation process (quality of the biogas and of the residues as a function of the mixture of the substrates by means of mass balances of the digesters) and the optimisation cleaning of the biogas. However, (co-)digestion of organic wastes may lead to additional pollution loads, mainly of (ammonia) nitrogen, in the reject water from the dewatering of the residues of the fermentation process. Therefore, not only does the digestion process have to be investigated in detail, but also the pollution loads in the reject water from the dewatering of the residues of the digestion process.

Hence, the investigations presented here have been focused on:

- The methodology of the application of mass balances to the system “treatment of sludge and the organic wastes” and to the total WWTP:
  - in order to estimate the mass fluxes of COD and N in the whole system;
  - for evaluating the pollutions loads in the reject waters.
- The assessment of the nitrogen removal capacity of the plant with respect to the additional nitrogen load from the dewatering of the residues of co-digestion.
- Evaluating the nitrogen loads in the reject water relative to the type of organic waste.
- A possible increase in the COD in the effluent from the WWTP.
Materials and methods
Site description
The WWTP of the city of Leoben is an activated sludge plant with primary sedimentation and mesophilic anaerobic sludge digestion. The influent load to the plant is about 5,300 kg COD/d on average and the influent load to the aeration tank is around 3,400 kg COD/d. The N-to-COD ratio of the influent is about 0.085 g N/g COD. The aeration tank (9,100 m³) aerated with fine bubble aeration is operated with nitrification as well as a combination of pre- and simultaneous denitrification. On average, the sludge age (SRT) is about 25 days. The WWTP of Leoben is equipped with two anaerobic digesters with a volume of 2,500 m³ each. With two digesters in operation, the retention time of the waste sludge is in the range of 50–60 days. Two gravity thickeners exist for pre-thickening but they are used only as storage tanks for the primary sludge. The excess sludge is thickened mechanically. The digested sludge is dewatered by a belt filter press and the dewatered sludge is composted together with wood chips. The produced compost is used for gardening and landscaping. The reject water from sludge dewatering is equalised and stored in one of the two existing primary settling tanks.

Operation modes during full-scale investigations
The intensive investigations started in June 2005, about 6 months after the beginning of the digestion of organic wastes, and lasted until the end of November. Until the end of July, one of the two digesters was fed only with waste sludge (primary and excess sludge) and the other one only with organic wastes as external substrate. For operational reasons, the external substrate was mixed with some reject water to elevate the water content. In both of the digesters, the retention time was about 30 days. For the rest of the investigation period (August until November 2005), both digesters were fed with a mixture of sludge from wastewater treatment and of “co-substrate”. The fact that during the months of June and July the two digesters were operated separately (one with the sludge from the WWTP and the other with external substrate) gave the opportunity to correlate the ammonia loads in the reject water to the respective sources. Until the end of these investigations, the organic substrate was not treated by a pulper, but was simply mixed with some reject water before it was pumped into the digesters.

Composition of organic wastes
During these investigations the external substrate fed into the anaerobic digesters consisted of organic wastes from slaughterhouses (flotation residues; about 45% of dry matter), dairies (mainly whey; about 35% of dry matter), leather industry (residues from hide cleaning and cutting; about 15% of dry matter), and some green clippings.

Mass balances of the wastewater treatment system
For evaluating the mass balances of COD and of nitrogen a static model utilising the operating plant data has been applied to the wastewater and sludge treatment system in order to validate the influent loads, denitrification capacity, gas production, nitrogen load in the digested sludge, etc. For carrying out these mass balances, the system boundaries had to be defined. Mass balances of COD and nitrogen have been conducted of the entire wastewater and sludge treatment system of the plant, excluding the digester for the fermentation of the organic wastes. Therefore, the mass fluxes in the reject water related to organic waste fermentation have to be regarded as input parameters.

COD mass balance. The fluxes for the COD balance of the activated sludge plant used for these evaluations are shown in Figure 1. Most of the input and output parameters can
be evaluated from data recorded during normal plant operation. The COD of the digested waste sludge (COD\textsubscript{XWD}) was estimated from the load of VSS (volatile suspended solids) in the digested sludge presuming a value of 1.4 g COD/g VSS. The COD of the methane gas from the digester for the waste sludge (COD\textsubscript{CH4}) was estimated by the gas production and the CO\textsubscript{2} content of the digester gas, for 0.35 norm-litre of methane gas correspond to 1 g of COD. The COD introduced by the nitrifying bacteria (COD\textsubscript{XA}) was deduced from the oxidised nitrogen (N\textsubscript{ox}) from the nitrogen mass balance (Nowak et al., 1999). What cannot be measured or calculated at all – in particular in a full-scale plant with simultaneous nitrification-denitrification – is the oxygen consumption for the degradation of carbonaceous matter (OU\textsubscript{C}).

However, OU\textsubscript{C} can be estimated by the COD balance:

\[
OU\textsubscript{C} = \text{COD}\textsubscript{i} + \text{COD}\textsubscript{XA} + \text{COD}\textsubscript{RoW} - \text{COD}\textsubscript{XWD} - \text{COD}\textsubscript{CH4} - \text{COD}\textsubscript{e} \quad \text{[kg O}_2\text{ or COD/d]} 
\]  

Figure 1 COD mass balance of the activated sludge plant (wastewater and sludge treatment, excluding the digester for fermentation of organic waste)

Figure 2 Nitrogen mass balance of the activated sludge plant (wastewater and sludge treatment, excluding the digester for fermentation of organic waste)
Nitrogen mass balance. The nitrogen mass balance used for evaluating the nitrogen fluxes in this activated sludge plant with digestion of organic waste is shown in Figure 2.

The nitrogen mass balance has to remain “open”, as denitrified nitrogen (N_{DN}) cannot be measured during the operation of the treatment plant. However, N_{DN} can be estimated by the nitrogen mass balance:

\[
N_{DN} = N_i + N_{REW} - N_{XWD} - N_e \ [\text{kg N/d}]
\]  

Validation of operating data of the WWTP. For validating the plant data, such as influent loads, gas production, etc., by means off the mass balances specific representative parameters have been utilised. If these “representative parameters” are found to be within a certain range, the whole set of operating data can be regarded as validated (Melcher, 2002).

The representative parameters utilised are VSS in the digested sludge as g/pe/d on the basis of 110 g COD/pe/d, O_P – the ratio of oxygen transfer to energy consumed for aeration [kg O_2/kWh] and the ratio O_{UC,ND}/O_{UC} where O_{UC,ND} is the oxygen consumption for the degradation of carbonaceous matter with nitrogen as electron acceptor (“nitrate respiration”) [kg O_2/d].

The values for O_{UC} and O_{UC,ND} are derived from the mass balances of COD and nitrogen, respectively. Nitrate respiration can be calculated from the load of denitrified nitrogen (N_{DN}) by

\[
O_{UC,ND} = 2.86 \times N_{DN} \ [\text{kg O}_2/\text{d}]
\]  

Accordingly, both parameters (O_{UC} and O_{UC,ND}) are the results from two different mass balances (Equations 1 and 2) and, therefore, a good validity check for the whole data set. As the degradation of organic carbon with nitrate is slower than that with oxygen as electron acceptor and as there is some oxic volume required in the aeration tank for nitrification, the ratio O_{UC,ND}/O_{UC} cannot exceed a value of about 0.65. As regards the other two parameters, experience shows that the specific VSS production should be in the range of 17–21 g VSS/pe/d in the stabilised sludge from municipal WWTPs (Nowak et al., 1996; Parravicini et al., 2006), and that the O_P-value varies between about 1.2–2.1 kg O_2/kWh under operating conditions for fine bubble aeration or for surface aerators.

Analysis and balances of the reject water

From grab samples taken from the storage tank of the reject water, i.e. the effluent from the belt filter press, the following parameters were analysed during the investigation period: three times a week ammonia nitrogen (NH_4-N), about once a week COD, BOD_5 and total nitrogen (TN) and occasionally total phosphorus (TP). In addition, every two weeks the NH_4-N concentration in filtered grab samples from the effluent from both digesters was analysed. All relevant flow rates, such as the effluents from the digesters and the effluent from the belt filter press, were continuously measured. Also, the flow of freshwater required for washing the belt filter press, which is about twice the flow from the digesters to the press was continually recorded. Thus the mass fluxes of ammonia nitrogen could be validated through the different measurements.

Results and discussion

Mass balances of the wastewater treatment system and validation of operating data

The results of the COD and nitrogen balance of the wastewater treatment system for the first period of the full-scale investigations (6 June to 31 July 2005) are presented in Figure 3.
During this first period the methane production from primary and excess sludge was approximately 700 Nm$^3$ CH$_4$/d corresponding to about 2000 kg COD/d (Figure 3). The release of nitrogen during anaerobic digestion of the organic wastes led to an increase in the N-to-COD ratio of the influent from about 0.085 to 0.108 g N/g COD. Nevertheless, during the summer season, there was still enough COD available for denitrification in the activated sludge system to achieve a nitrogen removal efficiency of 86% related to the nitrogen in the influent. This high extent of nitrogen removal could be attained only because of the high SRT of about 25 days in combination with a comparable high temperature in the aeration tank of about 20$^\circ$C. According to this, only a small section of the aeration tank had to be kept under aerobic conditions to achieve full nitrification. This is confirmed by the value of 0.62 for the ratio $O_{UC,N/D}/O_{UC}$ (Table 1). This means that almost 80% of the aeration tank volume as kept under anoxic conditions on average during this period. This is plausible, as under these conditions a fraction of 20% of the whole aeration tank volume is certainly enough for nitrification.

The value for VSS in the digested sludge is relatively low, but within the usual range (Table 1), and that for $O_p$ (oxygen produced to energy consumed for aeration) is plausible as well. Hence, the operational plant data of this period (June/July 2005) can be regarded as validated. Results from the other periods (until November 2005) confirmed the plausibility of the operating data. During the winter season, however, when the temperature in the aeration tank is much lower (in the range of 10–12$^\circ$C), significantly more volume of the aeration tank is required for nitrification leading to a decrease in the denitrification capacity. Consequently, the nitrogen removal was only about 70% during the winter period 2005/06.

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**Figure 3** Mass balances of COD (left side) and nitrogen (right side) of the wastewater system of the WWTP of Leoben (first investigation period: 6 June until 31 July 2005) – values in kg COD/d and kg N/d, respectively

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**Table 1** Representative parameters for the validation of the operating data by means of mass balances of the WWTP of Leoben (1st investigation period: 6th June until 31st July 2005)

<table>
<thead>
<tr>
<th>Representative parameter</th>
<th>(Unit)</th>
<th>First period</th>
<th>Usual range</th>
</tr>
</thead>
<tbody>
<tr>
<td>$O_{UC,N/D}/O_{UC}$</td>
<td>[-]</td>
<td>0.62</td>
<td>&lt; 0.65</td>
</tr>
<tr>
<td>VSS in digested sludge</td>
<td>[g/pe/d]</td>
<td>17.4</td>
<td>17 to 21</td>
</tr>
<tr>
<td>$O_p$ – oxygen transfer to energy</td>
<td>[kgO$_2$/kWh]</td>
<td>1.86</td>
<td>1.2 to 2.1</td>
</tr>
</tbody>
</table>
Nitrogen release in relation to methane production and to the origin of the organic wastes

In order to prove the results of the analysis of the soluble fractions in the digesters’ effluent and in the reject water, a mass balance of ammonia nitrogen of the dewatering of digested sludges and wastes was conducted (Figure 4).

Especially in the first investigation period, the sum of ammonia nitrogen in the effluent from the digesters (225 kg NH₄-N/d) fitted very well to the ammonia load in the reject water (204 kg NH₄-N/d) considering the fraction of ammonia in the dewatered residues (21 kg NH₄-N/d) that are conveyed to the composting site (Figure 4). Total nitrogen concentrations analysed from the reject water were only slightly higher, but less reliable than the NH₄-N concentrations, and were therefore not used for further evaluations.

CH₄ production from the external substrate (organic wastes) was approximately 1,750 Nm³/d, and therefore more than twice as high as that from the “internal substrate” (primary and excess sludge). The ammonia nitrogen load in the effluent from the sludge digester was about 100 g NH₄-N per Nm³ CH₄, whereas in the effluent from the digestion of organic wastes only 70 g NH₄-N/Nm³ CH₄ were found (Table 2). This means that in relation to the CH₄ produced, the release of nitrogen from the external substrate was somewhat lower than the release from the sludge.

As for the different co-substrates, it was estimated that the comparatively lowest release of nitrogen – again related to the CH₄ produced – occurred from the flotation residues due to the higher fat content, than from the dairy wastes and that the relatively highest nitrogen release comes from the residues from the leather industry. Table 2 shows the results from analyses of the different organic wastes compared to the release of ammonia related to the COD of the methane gas.

Table 2 NH₄-N in digester effluent related to CH₄ produced and to the COD of the produced CH₄ (upper table) N-to-COD ratio of the organic wastes (lower table)

<table>
<thead>
<tr>
<th>Release from:</th>
<th>NH₄-N/CH₄ [g/m³]</th>
<th>NH₄-N/COD-CH₄ [g/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic wastes</td>
<td>70</td>
<td>24</td>
</tr>
<tr>
<td>Waste sludge</td>
<td>99</td>
<td>34</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Organic wastes from:</th>
<th>N/COD [g/kg]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Slaughterhouses</td>
<td>21</td>
</tr>
<tr>
<td>Dairies</td>
<td>27</td>
</tr>
<tr>
<td>Leather industry</td>
<td>33</td>
</tr>
</tbody>
</table>
The effect of the digestion of organic wastes on the COD removal efficiency of the WWTP

The COD and, in particular, the phosphorus concentrations in the reject water were comparably low throughout these investigations. The COD load was mostly lower than the NH$_4$-N load (Figure 3). A comparison of the COD removal efficiency of two long-term periods before and after the start-up of the digestion of organic wastes revealed that the COD removal decreased from 93.8% (January to November 2004) to 93.1% (January to November 2005). Although this means a measured increase in the effluent load of about 10%, this increase was not regarded as significant enough to be seen as a consequence of the treatment of external substrate in this municipal WWTP.

Conclusion

Co-digestion of organic wastes at municipal WWTPs can not only increase biogas production, but also the nitrogen loading of the plant considerably. However, at least with the substrates investigated in this study (residues from different agro-industries), the ratio of nitrogen in the reject water to methane produced is lower than with waste sludge. Nevertheless, the nitrogen-to-COD ratio of possible organic wastes should be analysed before they are treated in the anaerobic digester of a municipal WWTP. In addition, the denitrification capacity of the activated sludge stage of the plant has to be examined in detail to find out to what extent organic wastes can be applied to the digesters without the installation of an additional treatment step for nitrogen removal from the reject waters. Moreover, the effect of the treatment of additional organic wastes on the effluent COD has to be investigated for every single case.

At the WWTP of Leoben, presently one of the two primary clarifiers is used as a storage tank for the reject waters. In future, one section of this storage tank (about half of the volume) will be equipped with an aeration system for “nitritation” (= oxidation of ammonia to nitrite), the other section will be used for nitrogen removal.

The usage of overcapacities in digester volume at municipal WWTPs for co-digestion of organic wastes can be a good solution for waste reduction mainly from agro-industries – with the side-effect of energy production. However, the local conditions have to be considered and, moreover, the organic wastes in question should be tested in pilot or full-scale investigations for each particular application.

References


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