Tank-connected food waste disposer systems – Current status and potential improvements

A. Bernstad a,⇑, Å. Davidsson a, J. Tsai a, E. Persson a, M. Bissmont b, J. la Cour Jansen a

a Water and Environmental Engineering, Department of Chemical Engineering, Lund University, Sweden
b VA SYD, Malmö Municipality, Sweden

ABSTRACT

An unconventional system for separate collection of food waste was investigated through evaluation of three full-scale systems in the city of Malmö, Sweden. Ground food waste is led to a separate settling tank where food waste sludge is collected regularly with a tank-vehicle. These tank-connected systems can be seen as a promising method for separate collection of food waste from both households and restaurants. Ground food waste collected from these systems is rich in fat and has a high methane potential when compared to food waste collected in conventional bag systems. The content of heavy metals is low. The concentrations of N-tot and P-tot in sludge collected from sedimentation tanks were on average 46.2 and 3.9 g/kg TS, equalling an estimated 0.48 and 0.05 kg N-tot and P-tot respectively per year and household connected to the food waste disposer system. Detergents in low concentrations can result in increased degradation rates and biogas production, while higher concentrations can result in temporary inhibition of methane production. Concentrations of COD and fat in effluent from full-scale tanks reached an average of 1068 mg/l and 149 mg/l respectively over the five month long evaluation period. Hydrolysis of the ground material is initiated between sludge collection occasions (30 days). Older food waste sludge increases the degradation rate and the risks of fugitive emissions of methane from tanks between collection occasions. Increased particle size decreases hydrolysis rate and could thus decrease losses of carbon and nutrients in the sewerage system, but further studies in full-scale systems are needed to confirm this.

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1. Background

Source sorting of organic household waste has been proposed within the framework of a new European Waste Framework Directive (European Parliament, 2008) and different systems for food waste collection are currently being established in various European countries. Use of food waste disposers (FWDs) has been suggested as a practical way to establish source-separation of food waste without increasing transport, avoiding problems related to odor and increased need for waste bins amongst others (Marashlian and El-Fadel, 2005). However, there are several questions regarding the effects of FWD connected to the conventional sewer system. Several potential adverse effects have previously been described. Bolzonella et al. (2003) stated that FWD can cause an increased organic load in the biological step at the wastewater treatment plant (WWTP) and thereby the energy demand for wastewater treatment. Others have raised problems with increased oil and grease load at WWTPs and risk of increased H₂S-production in sewerage systems, which could result in corrosion of cement pipes (Nilsson et al., 1990). The knowledge on the quantity and quality of output from FWD actually reaching WWTP is still incomplete. Considerable removal of dissolved organic matter and proteins in wastewater during sewage transport to WWTP has been seen in previous studies (Raunkjaer et al., 1995). Effects on WWTP processes and potential increased sludge generation due to FWDs will also to a large extent depend on the WWTP design and whether incoming particulate matter is separated with primary sludge or not. Bolzonella et al. (2003) state that an increased carbon concentration in incoming wastewater can improve the C/N and C/P-ratio in WWTPs and, depending on the process design, result in an improved nutrient removal and reduced requirement for external carbon sources. Battistoni et al. (2007) showed that a 41% market penetration of FWD in a small Italian municipality increased both COD and N-tot in incoming wastewater, but did not resulted in increased energy use at the local WWTP, where activated sludge processes were used. Evans et al. (2010) and Galil and Yaacov (2001) presented measurements of significant increases in biogas production in WWTP-sludge digestion when 50% of connected households introduced FWD. The net-effects on sewerage systems and WWTP-processes from FWD installation are still an area of further investigation. These earlier stated
problems or questions connected to waste grinders have led to bans on waste grinders for food waste in several areas, such as New York in combined sewerage systems in the 1970s, Italy in 1999 and Raleigh, North Carolina in 2008. These bans have in all cases been lifted after monitoring of the effects to the systems. However, reluctance concerning a more extensive use of food waste disposers is still seen in many countries.

1.1. Aim and scope

In order to exploit the advantages of FWD and at the same time avoid possible negative effects in sewerage systems and WWTPs, an unconventional disposer system was introduced in two residential areas in the city of Malmö, southern Sweden. Another system was connected to kitchen plumbing in a restaurant in the same city. In the present paper, these three full-scale FWD installations are investigated and potential improvements to the systems are presented.

1.2. Description of the systems

FWDs were installed in kitchen sinks in approximately 60 apartments in a residential area in Malmö (area A) in 2001. The kitchen sinks were connected to a pipe system separated from the other wastewater system in the building. This wastewater stream was led through an LPS-unit (Low Pressure Sewer) to a settling tank (volume 2.7 m$^3$) divided in sections from which supernatant is led to the WWTP and the settled waste is collected and transported for further anaerobic biological treatment. A cutting pump was used in the LPS-unit. The system in area B was installed in 2007 and was to a large extent a replica of the system in area A, but as the area consisted of 147 apartments in a high rise building (Turning Torso in Malmö), there was no need for the LPS and subsequently not for the cutting pump used in area A.

System C was installed in 2010 and was connected to the food waste disposers installed in two restaurants in the same city. Also in this case, no cutting pump was used. The principle of the systems is shown in Fig. 1, where the truck for transportation of the settled material is shown together with the sewer, and the wastewater treatment plant for handling of the non-settled material.

The system can be said to be self-regulating in relation to mis-sorting as the disposers in general are sensitive to other materials than bio-waste and normally stop if other materials are disposed. However, materials such as soft plastics and paper could probably be disposed together with food waste without causing severe problems in the disposer and thus enter the settling tanks. Sludge collection was performed once a month in system A and B and every second week from system C with a tank-vehicle (capacity 12 m$^3$) with a total energy use of 34.0 kW h natural gas/ton collected food waste sludge. Information regarding the energy use in LPS-system could not be gained from the owner of the system and was therefore not included in the analysis.

Detergents have earlier been seen disrupting the performance of anaerobic digesters (Tanaka and Ichikawa, 1993; Gavala et al., 2001) and it is known that surfactants can harm methane-forming bacteria through lysis due to the absence of protecting envelope around many methane-forming bacteria. In normal municipal wastewater, detergents used in households are mixed with a large amount of water through different flows; showers, toilets and kitchens etc. The tank-system could result in higher concentration of detergents in ground food waste sludge, as the system separates kitchen sink flows from other wastewater flows. Thus, the effect of detergents on biological processes in collected food waste sludge was of interest to investigate. The following aspects were investigated and used as a basis for the evaluation:

- Quantity and characteristics of collected food waste sludge – including potential methane production.
- Quantity and characteristics of effluent from collection tanks.
- Degradation of freshly ground food waste under different conditions.
- Potential fugitive methane production from ground food waste under different conditions.
- Potential inhibition of biological processes due to inhibitory substances (detergents).

2. Methods

The study combines data from analyses of samples from the full-scale installations as well as laboratory experiments with freshly ground food waste. In order to mimic the full-scale systems, a recipe for food waste ground for laboratory experiments were developed based on the ratio of fat, proteins, carbohydrates and VS in full-scale samples (Table 1). The amount of water used for laboratory grinding was chosen based on a literature review of previously assessed water use for food waste grinding, recalculated to the same unit (litre water/kg food waste) (Table 1).

Energy use for grinding of food waste with the equipment used in the full-scale areas was determined based on five measurements where 1 kg of fresh food waste was ground per occasion. Each grinding session was timed and the effect of the disposer was controlled. Energy-use in disposers was determined to 4.4 kW h/ton food waste. Samples (201 per occasion) were collected from the three full-scale suspension-tanks on several occasions during the period September 2009 – December 2011. Sampling was performed once a month, and always from full tanks. Effluent was also sampled during a period of five months. Samplings were made on six occasions from the outflow from the suspension tank. Five litres of effluent was collected per occasion. Samples were stored at 6 °C within three hours and analyses were made within 24 h. Samples of sludge from suspension-tanks and effluent were used to determine the following parameters with the following methods:

- TS (% of wet weight) and VS (% of TS) in sludge and TS, VS and SS (mg/l) in effluent was determined in triplicates using standard methods (APHA, 2005, 2540 G).
- pH was determined with an instrument from the manufacturer WTW, model 320.
Table 1
Water use in FWD-systems; literature values and used assumption.

<table>
<thead>
<tr>
<th>Value</th>
<th>Unit</th>
<th>Equal to (l/kg food waste)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>3</td>
<td>l/household, d</td>
<td>7.2</td>
<td>Kappaläfbundet and SORAB (2009)</td>
</tr>
<tr>
<td>6</td>
<td>l/household, d</td>
<td>14.5</td>
<td>Kappaläfbundet and SORAB (2009)</td>
</tr>
<tr>
<td>4.3</td>
<td>l/person, d</td>
<td>15.6</td>
<td>Marashlian and El-Fadel (2005)</td>
</tr>
<tr>
<td>4.5</td>
<td>l/household, d</td>
<td>10.9</td>
<td>Göteborgs Stad (2011)</td>
</tr>
<tr>
<td>15.3</td>
<td>l/kg food waste</td>
<td>19.3</td>
<td>Nilsson et al. (1990)</td>
</tr>
<tr>
<td>12.4</td>
<td>l/kg food waste</td>
<td>12.4</td>
<td>Wainberg et al. (2000)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>12.0</td>
<td>Used in the present study</td>
</tr>
</tbody>
</table>

- Particle size compositions in food waste sludge and effluent were found from sieving of the material (8, 4, 2, 1 and 0.15 mm perforations). In order to decrease the influence of surplus water sieves were left running for 10 min, after which the weight of each sieve was recorded. Measurements were performed in triplicate.
- Sedimentation was determined using standard methods (SS-EN 14702-1:2006). Recordings of settling were made after 1, 2.5, 5, 10, 20 and 30 min. Settling velocity was determined as the volume of settled material over 30 min. Determinations were made in triplicates.
- Volatile fatty acids (VFAs) (propionate and acetate) were measured in food waste sludge and effluent from tank systems as well as in hydrolized food waste samples and samples used for determination of spontaneous methane production. Food waste sludge samples were centrifuged for five minutes (1250 g, MSE Scientific Instruments, Model CR-8) and all samples (both effluent and sludge) were filtered (Munktell 1002, 110 mm). Before analysis, the sample (0.9 ml) was conserved with phosphoric acid (0.1 ml) and a sample volume of 0.2 ml was injected into the gas chromatograph (GC) (Agilent 6850 Series) equipped with a HP-FFAP column (30 m/0.53 mm/1 μm) at 80–130 °C (temperature inlet flow = 180 °C, oven temperature 260 °C).
- Heavy metals (Al, Cd, Cr, Fe, Pb and Zn) S, K and P were determined in hydrolized food waste sludge and effluent from WWTP (SP, 2010; REVAQ, 2011). The results are based on measurements of the water level after emptying of tanks with precise indications of the time between emptying and measurement. An average was based on six separate measurements, covering different hours of the day and days of the week.

3. Results

3.1. Characteristics of food waste sludge

TS in sludge collected from areas A and B are in general low and both TS and VS are higher from the restaurant (C) compared to the residential areas (Table 2). Although the variations are large between samplings, the differences between all areas (t-test (two tailed) p < 0.01). In the case of pH, significant differences were seen between all areas (t-test (two tailed) p < 0.01).

Heavy metals, C, N, K, P and S were analysed in FWD-sludge from areas A and B at two occasions (August and October 2010) and area C at one occasion (October 2010) (Table 3). The percentage of heavy metals on a DS basis increases after digestion, a comparison can be made between results and the SEPA guidance on heavy metals in sludge from WWTP, which is spread on farmland as well as the certification of digestate from AD-plants and sludge from WWTP (SP, 2010; REVAQ, 2011). The results are also compared to an analysis of food waste ground in laboratory...
with use of non-ionic water as carrier water (FW fresh, based on recipe presented in Davidsson et al., 2011).

3.2. Particle size and sedimentation qualities

The investigation of the particle size distribution shows a consistently higher fraction of larger particles (>4 mm) in sludge collected from area B compared to area A, where a cutting pump was used in the system \((p < 0.5, t\)-test, 2-tailed) while no significant differences were found between area C and other areas (Fig. 2). The particle size generated was seen to be below the 12 mm demanded by the hygienization standards for this type of biogas substrate according to the EU ABP-directive (European Parliament, 2009).

Settling tests with samples from areas A and B showed that an absolute majority of the particles in the samples sediments within 10 min (Fig. 3). No settling test could be done with samples from area C, due to the high TS in samples from this area.

The initial settling velocity during the first 10 min is calculated as:

\[
0.001 \times \frac{(1000 - V_s)}{(10/60)}
\]

with \(V_s\) = sludge volume (mm) after 10 min (Table 4).

It is seen that the majority of settling particles in the ground food waste will settle within 10 min, provided a surface load below 3–5 m/h. The effluent flow velocity was determined based on six separate measurements, covering different hours of the day and days of the week. As no measurements were done over-night, it was assumed that the measured flow was representative for 12 h of the day (07AM-7PM) and that the flow during the other 12 h was 5% of this measured flow. This gives a water use of 40.3 l/capita and day. Based on this assumption, the initial settling velocity could be calculated for the surface area of 2.9 m² in full-scale tanks, giving a surface load in the settling-tank of 0.11 m/h, thus well below the needed 3–5 m/h.

3.3. Degradation of ground food waste

Degradation tests with freshly ground food (alone as well as mixed with different types of other waste flows) show that biological hydrolysis is taking place in all samples. Increased concentrations of dissolved COD and VFA (acetate and propionate) and decreased TS and VS are seen in all cases, as degradation of VS to a large extent is mirrored in an increase of dissolved COD. These processes are all enhanced when food waste was mixed with wastewater (WW), black water (BW) and sludge from full-scale FWD-tanks (FWS) (Fig. 4). Mixing with these flows also increased the concentration of \(\text{NH}_4^+\), which was not seen when food waste was degraded separately (Table 5). A previous study of the same system has also shown that the degradation rate is enhanced by increasing temperatures (Davidsson et al., 2011).

3.4. Methane production – theoretical and experimental

In order to determine the theoretical methane production potential in the ground food waste, the content of proteins, carbohydrates and fat was measured. The calculation of the potential was based on data from Christensen et al. (2003) (Table 6). Higher fat and lower protein content is seen in samples from area C (restaurant). The theoretic methane production per VS is 16–35%
higher in ground food waste from households (area A and B) compared to previous assessments of household food waste collected in paper bags, due to a larger fraction of carbohydrates in the latter. The reason behind this difference in composition could only partly be explained by the influence of the collection bags used. Methane

![Diagram of settling tests for area A and B.](https://example.com/diagram)

**Fig. 3.** Results from settling tests with samples for area A and B. Average of triplicates.

<table>
<thead>
<tr>
<th>Table 4</th>
<th>Initial settling velocity in food waste sludge from full-scale tanks in areas A and B.</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Area A</td>
</tr>
<tr>
<td>Initial settling velocity (m/h)</td>
<td>5.2</td>
</tr>
<tr>
<td>SD (m/h)</td>
<td>0.4</td>
</tr>
</tbody>
</table>

![Graph showing VS, COD, and VFA in hydrolysis experiments.](https://example.com/graph)

**Fig. 4.** Development of VS, COD and VFA (acetate + propionate) in hydrolysis experiments with ground food waste (FW1, small particles and FW2, larger particles) and ground food waste mixed with wastewater (FW + WW), black water (FW + BW) and sludge from the settling full scale FWD tank (FW + FWS).

<table>
<thead>
<tr>
<th>Table 5</th>
<th>Total change in VS, COD (dissolved) and NH₄⁺ over the experiment in samples with ground food waste mixed with wastewater (FW + WW), black water (FW + BW) and sludge from the settling full scale FWD tank (FW + FWS). FW₁ small particles and FW₂ larger particles after 30 days.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wastewater combination</td>
<td>VS reduction (%)</td>
</tr>
<tr>
<td>FW + WW</td>
<td>54</td>
</tr>
<tr>
<td>FW + BW</td>
<td>48</td>
</tr>
<tr>
<td>FW + FWS</td>
<td>38</td>
</tr>
<tr>
<td>FW₁</td>
<td>27</td>
</tr>
<tr>
<td>FW₂</td>
<td>22</td>
</tr>
</tbody>
</table>
production potential from similar material has previously been assessed to 75 N dm\(^3\)/kg VS (Riber and Christensen, 2006). Previous studies have suggested an average use of paper bags equal to 11.7–19.5 kg/ton wet food waste or 3–5% of total TS (Bernstad et al., 2011) in a system similar to the ones described by Davidsson et al. (2007). A recalculation of above presented data from Davidsson et al. (2007) was made in order to exclude the impact of paper bags from the theoretical methane potential. However, also recalculated data for the theoretical methane potential show low values when compared to samples from full-scale areas A and B (Fig. 5).

Methane production was also determined experimentally through batch methane potential (BMP) analyses of freshly ground food waste samples as well as samples from hydrolysis experiments. The methane production per VS was similar in all cases (398–405 N dm\(^3\) CH\(_4\)/kg VS). However, when related to the initial VS content (prior to hydrolysis), results suggest a potential decrease of potential methane production of more than 25% in samples hydrolyzed during 4 weeks and 7–9% in samples hydrolyzed during 36 h. Results suggest a decreased methane production potential of 5–26% per ton ground food waste (Fig. 6). Thus, although the methane potential per VS in FWD sludge, the recovered methane is potentially decreased through the degradation taking place before the sludge is collected.

3.5. Influence of detergents

Addition of detergents to the ground food waste caused a rapid increase of dissolved COD, which could only partly be explained by the COD added through the detergents themselves (Table 7). Thus, the detergents resulted in an increased solubilisation of particulate organic matter. At the same time, addition of detergents in higher concentrations resulted in lower concentrations of both VFA and NH\(_4\)\(^+\) and a lesser decrease in pH compared to samples were detergents not had been added, which was interpreted as inhibition of biological hydrolysis (Table 7). Also the methane production per VS increased when lower concentrations of detergents had been added to the samples, while addition of detergents at higher concentrations resulted in signs of temporary inhibition of methane production (Fig. 7). After 12 days, a clear difference was seen in reactors without and with the lowest addition of detergent on the one hand, compared to reactors where detergent had been added in higher concentrations on the other (\(p<0.01\), t-test

### Table 6

<table>
<thead>
<tr>
<th>Fraction</th>
<th>Unit</th>
<th>Area A</th>
<th>Area B</th>
<th>Area C</th>
<th>Household(^b)</th>
<th>CH(_4) potential (N dm(^3)/kg VS)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Proteins(^a)</td>
<td>% of DS</td>
<td>22.4</td>
<td>29.3</td>
<td>9.8</td>
<td>10–18</td>
<td>496</td>
</tr>
<tr>
<td>Carbohydrates</td>
<td>% of DS</td>
<td>27.9</td>
<td>27.2</td>
<td>7.7</td>
<td>19–55</td>
<td>415</td>
</tr>
<tr>
<td>Fat</td>
<td>% of DS</td>
<td>42.3</td>
<td>37.5</td>
<td>81.2</td>
<td>10–18</td>
<td>1014</td>
</tr>
<tr>
<td>Ash</td>
<td>% of DS</td>
<td>7.4</td>
<td>6.0</td>
<td>1.3</td>
<td>8–19</td>
<td>0</td>
</tr>
<tr>
<td>Energy</td>
<td>kJ/100 g</td>
<td>2345</td>
<td>2285</td>
<td>3201</td>
<td>1900–2200</td>
<td></td>
</tr>
<tr>
<td>Theoretical CH(_4) potential</td>
<td>N dm(^3)/kg DS</td>
<td>656</td>
<td>638</td>
<td>904</td>
<td>267–500</td>
<td></td>
</tr>
<tr>
<td>Theoretical CH(_4) potential</td>
<td>N dm(^3)/kg VS</td>
<td>764</td>
<td>713</td>
<td>928</td>
<td>567–615</td>
<td></td>
</tr>
</tbody>
</table>

\(^a\) As Kjeldahl nitrogen.

\(^b\) Food waste from households with separate collection in paper bags (Davidsson et al., 2007).

![Fig. 5](image1.png)

**Fig. 5.** Theoretical methane potential based on content of fat, carbohydrates and proteins in areas A–C and paper bag collected food waste (PB1-2).

![Fig. 6](image2.png)

**Fig. 6.** Methane potential per kg VS and kg food waste as well as degradation ratio during hydrolysis according to degradation experiments.

Addition of detergents to the ground food waste caused a rapid increase of dissolved COD, which could only partly be explained by the COD added through the detergents themselves (Table 7). Thus, the detergents resulted in an increased solubilisation of particulate organic matter. At the same time, addition of detergents in higher concentrations resulted in lower concentrations of both VFA and NH\(_4\)\(^+\) and a lesser decrease in pH compared to samples were detergents not had been added, which was interpreted as inhibition of biological hydrolysis (Table 7). Also the methane production per VS increased when lower concentrations of detergents had been added to the samples, while addition of detergents at higher concentrations resulted in signs of temporary inhibition of methane production (Fig. 7). After 12 days, a clear difference was seen in reactors without and with the lowest addition of detergent on the one hand, compared to reactors where detergent had been added in higher concentrations on the other (\(p<0.01\), t-test

### Table 7

<table>
<thead>
<tr>
<th>Sample</th>
<th>NH(_4)(^+)</th>
<th>VFA</th>
<th>COD Water(^a)</th>
<th>Sample</th>
</tr>
</thead>
<tbody>
<tr>
<td>FW</td>
<td>15</td>
<td>500</td>
<td>1800</td>
<td></td>
</tr>
<tr>
<td>FW + 0.8 mg/l</td>
<td>30</td>
<td>290</td>
<td>670</td>
<td>500</td>
</tr>
<tr>
<td>FW + 3.2 mg/l</td>
<td>2</td>
<td>310</td>
<td>2700</td>
<td>11,000</td>
</tr>
<tr>
<td>FW+8.0 mg/l</td>
<td>2</td>
<td>310</td>
<td>6700</td>
<td>22,500</td>
</tr>
</tbody>
</table>

\(^a\) Potential contribution of COD from detergents alone in fresh water.
2-tailed). After 30 days, the highest production was seen in reactors to which the lowest concentration of detergents had been added. However, differences between samples were no longer statistically significant ($p > 0.1$, t-test, 2-tailed).

3.6. Potential fugitive methane emissions from settling tanks

Results from the degradation tests show that hydrolysis occurs in the tanks between sludge collection occasions. Thus there could also be a possibility of unwanted emissions of methane from the tank system between sludge collections. The risk for unwanted (spontaneous) methane production was assessed through simulations of the tank process in laboratory scale experiments under different conditions. Freshly ground food waste and freshly ground food waste together with sludge from area B in different ratios (10% and 15% of total VS respectively) was assessed (Fig. 8). The first measurement was made nine days after setup. Temperature measurements in settling tanks in full-scale systems A and B showed an average temperature of 17.0°C and 19.1°C in the two systems over the sampling period (SD = 4.4 and 2.3°C respectively).

3.7. Composition of effluent

The composition of effluent from the tank-system in area B is displayed in Table 8 as averages based on seven measurements over a period of five months. Both TS and VS were low. A large part of the solids are non-organic, which could be explained by solved salts. The conductivity in effluent was high in comparison to previously presented data for average wastewater and above local guidelines for industrial wastewater released to municipal treatment plants (Levlin and Hultman, 2008). pH was in all cases below neutral; 6.1–6.6. Analyses of effluent from full-scale installations were compared with average levels of N-tot, P-tot, COD$_{cr}$, SS, fat and conductivity in Swedish municipal wastewater and in the inflow to the WWTP at which the effluent was received. Also national and local regulations on these emissions in effluent from WWTP are presented in the comparison (Table 8).

The concentrations of the metals Al, Cu and Zn were determined to an average of 0.2, 0.01 and 0.02 mg/l respectively (SD = 0.033, 0.003 and 0.009 mg/l respectively). Concentrations of Cd and Pb were in all cases under detection limits.

Analyses of nutrients, carbon and metals in food waste sludge and effluent respectively, together with measurements of the flow in effluent from Area B can be used to estimate the mass-flow of these compounds over the system. However, as seen in Tables 2, 3 and 8, concentration in some cases varied largely between sampling occasions food waste sludge as well as of varied largely between sampling occasions. Using minimum and maximum values from analyses of both sludge and effluent and the flow assumed above, 40–55% of N-tot and C-tot as well as 38–62% of P-tot and 95–99% of K and S in ground food and kitchen waste are transported from the sedimentation tank with effluent. In the case of fat, between 30% and 57% is transported to the sewerage system with effluent.

4. Discussion

4.1. Losses of food waste to wastewater system

The FWDs used in the investigated full-scale systems were originally designed to transport ground waste in the sewerage system to a WWTP, as this would be the normal use of FWDs. Larger particles have been assumed to increase the risk for settling of particles in the sewerage system and subsequent clogging and there has therefore been a striving towards ever smaller particle sizes amongst FWD producers. This could be assumed to decrease the amount of settled material in tanks. However, both freshly ground food waste and sludge collected from full-scale systems were seen to have good settling-qualities and the low TS and high levels of dissolved COD in effluent indicate that losses of organic matter from the system occur through dissolved substances and non-settling fat. Hydrolysis tests show a slower biological hydrolysis rate in ground food waste when the particle size was increased. Thus, an increased particle size in ground food waste could potentially decrease losses of organic matter and nutrients from the
tanks system, mainly due to a slower hydrolysis rate rather than through improved settling qualities. Hydrolysis tests showed that degradation of VS and levels of dissolved COD also increased when food waste was mixed with other waste flows, including sludge from full-scale tank systems. Thus, it is not advisable to use the tank system for combined systems for food waste and wastewater/black water and there are reasons to try to diminish the amount of old food waste sludge left in the sedimentation tank when emptied through proper cleansing.

Accumulation of fat in municipal sewerage systems is an increasing problem in many Swedish municipalities (Blecken et al., 2010). Although experiences have suggested that in-sewer biological processes can acclimate to the change in wastewater composition caused by FWD installations and thereby reduce problems related to accumulation of organic matter in sewage pipes, there can still be reluctance towards FWD in areas already experiencing problems with fat-accumulation. This could also be seen as one of the main reasons behind the development of the tank-connected system. Limits for fat in wastewater from commercial entities to the municipal sewerage system varies between Swedish municipalities, but commonly range between 50 and 150 mg/l (Blecken et al., 2010). Analyses of effluent from full-scale tank systems investigated in the present study show that the average concentration of fat in effluent is 50% above municipal limits in the region of application (VA SYD, 2011) equal to 1.8 kg/person and year. Thus, the study indicates that effluent from non-commercial entities can result in considerable contribution of fat to the municipal sewerage system and that the tank-system in its current design not fully solves potential problems related to fat-accumulation.

The estimated mass-flow of carbon and nutrients over the system indicate that around 50% of C-tot, N-tot, P-tot and fat is transferred from the settling tank with effluent. In the case of K and S, the fraction seems to be even higher. As the major parts of nutrients can be assumed to be conserved through the anaerobic digestion process, it can be assumed to be available as fertilizers if digestate is used on farmland. Whether carbon and nutrients are recovered later in the system will to a large extent depend on the WWTP design and potential use of WWTP sludge as fertilizer. Independently of this, results show a potential decrease in problems related to for example fat accumulation in sewages when compared to systems where FWD are connected to sewerage systems directly, as a large part of the fat is collected with the sludge.

The level of biogas potentially non-realized due to losses of organic matter with effluent can be assessed using values for theoretical methane production per kg COD (0.35 N m³/kg COD) (Spinosa and Vesilind, 2001) in effluent. Assuming a realization of 75% of the theoretical methane production and a degradation ratio of 80% of COD would result in non-recovered methane potential of 0.26 N m³ CH₄/m³ effluent, or 2.25 N m³ CH₄ per year and person connected to the FWD tank-system. This can be compared to the methane potential of 3.7 N m³ per person from sludge collected from the systems over the same period, assuming an 80% degradation of VS. This should be seen as an estimate, as no measurements were made of dissolved methane in effluent, and as the calculation is based on a series of assumptions. However, it is relevant to acknowledge that the current design of the system does not result in an optimal collection of substrate for methane production from the tanks.

### 4.2. Fugitive emissions of methane from settling tanks

Fugitive emissions of methane from tank systems are most unwanted. Laboratory simulations of fugitive emissions show that the addition of older food waste sludge increased the fugitive methane emissions. Using values from measurements in 20 °C, i.e. slightly above the average temperature in full-scale systems settling tanks over the sampling period, state a potential production of 0.2–1.5 N ml CH₄/g VS between sludge collection occasions (35 days). Increasing the period between collections to 70 days could increase the production to 0.4–3.2 N ml CH₄/g VS. Thus, accumulated methane production generally increased with increasing time, addition of old food waste sludge and temperature. However, at higher temperatures, the methane production stopped after 35 days, probably related to an inhibition as the pH dropped radically. The potential relation between pH in the tank and methane production makes it important to highlight that the method used for estimating potential fugitive emissions of methane does not take the constant inflow of new organic matter and water with higher pH to the tank into consideration, which could underestimate the risks for fugitive methane emissions. Decreasing the current collection frequency could increase the risk for methane emissions from settling tanks and is therefore not advisable. Levels of potential fugitive emissions detected in laboratory experiments equals a production of 0.03–0.23% of the calculated total methane yield from food waste sludge and can thus be regarded as insignificant. However, if the tank not is emptied completely and rinsed, there could be remains of older food waste in the tank. This could be seen as an inoculum, increasing the degradation in tank, as seen also in hydrolysis experiments.

### 4.3. Characteristics of food waste sludge

Previous studies have shown a higher per VS methane production in batch test AD of food waste collected with food waste grinders compared to paper bags (Davidsson et al., 2007). This could be explained by household behavioural factors and the difficulties to collect liquid/semi-liquid food waste with high ratio of easily biodegradable carbon such as dairy products, and juice and a decreased ratio of non-anaerobically biodegradable carbon such as

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**Table 8**

Composition in effluent from full-scale tank system in area B (as averages and SD = standard deviation) together with national average, national guidelines and local conditions.

<table>
<thead>
<tr>
<th>Flow</th>
<th>P-tot (mg/l)</th>
<th>N-tot (mg/l)</th>
<th>K (mg/l)</th>
<th>S (mg/l)</th>
<th>TOC (mg/l)</th>
<th>COD (mg/l)</th>
<th>COD₄ (&lt;mg/l)</th>
<th>SS⁰ (mg/l)</th>
<th>Fat (mg/l)</th>
<th>Cond.⁺ (ms/m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wastewater</td>
<td>16</td>
<td>50</td>
<td></td>
<td></td>
<td>530</td>
<td>445</td>
<td>70</td>
<td>93</td>
<td></td>
<td></td>
</tr>
<tr>
<td>WWTP effluent</td>
<td>0.3</td>
<td>10</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>50–150</td>
<td>500</td>
</tr>
<tr>
<td>Inflow WWTP⁰</td>
<td>5.3</td>
<td>42</td>
<td>19.6</td>
<td>23.3</td>
<td>177.1</td>
<td>1101</td>
<td>575</td>
<td>540</td>
<td>149</td>
<td>772</td>
</tr>
<tr>
<td>Average effluent</td>
<td>2.2</td>
<td>16.1</td>
<td>1.3</td>
<td>36.4</td>
<td>86.7</td>
<td>354</td>
<td>150</td>
<td>120</td>
<td>65</td>
<td>317</td>
</tr>
<tr>
<td>SD effluent</td>
<td>0.8</td>
<td>6.4</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* Dissolved COD.  
* Suspended Solids.  
* Conductivity.  
* National and regional regulations (SFS, 1998; Blecken et al., 2010).  
* Average in inflow to local WWTP (Sjölunda, Malmö) (VA SYD, 2010).
plastics and lignin in food waste collected through FWG in comparison to bag-collection systems. According to previous studies in the UK, dairy products and drinks (not including tea and bottled water) contributes to 12% of the total household food waste generation (WRAP, 2008). The fat content in Swedish and Danish household food waste collected in paper bags was according to previous studies equal to 12.2–18.1% or 15% of TS (Hansen et al., 2007; Davidsson et al., 2009; respectively), i.e. low compared to the 37.5–42.3% of TS in sludge collected from FWD tank-systems according to the present study. As a consequence, the theoretical methane potential per ton VS is 36% higher in FWD-collected household food waste compared to bag collected. The gastronomic variations between households and areas will have a large impact on the waste composition and thereby also potentially on the methane production from the ground food waste. A high fat content could cause instability in the digestion process and decrease the methane yield. However, if ground food waste is mixed with other substrates the risk of such problems will decrease. Ground food waste could also be used as media for decreasing the DS in separately collected food waste from areas were paper or plastic bag collection is used (Truedsson, 2010).

The concentration of Cd varies between 0.1 and 0.7 mg/kg TS in full-scale samples of sludge from households FWD systems. This can be compared to results from analyses of food waste collected in paper or plastic bags from three Swedish plants for anaerobic digestion of food waste and other organic materials; 0.2–0.3 mg/kg TS (Truedsson, 2010; Malqvist, 2011). Due to varying levels of both Cd and P in analysed samples, the Cd/P ratio varies largely between different areas, but also between analysis occasions and values are in many cases higher than current limitations for spreading of WWTP-sludge on farmland (REVAQ, 2011). In the case of Zn and Cu, measurements show concentrations above current regulations. This could be related to the material composition in the plumbing systems, but further studies are needed to confirm this.

Thus, based on the present results, no conclusions can be made regarding the possibilities of reducing Cd concentrations in separately collected household food waste. According to previous studies, batteries, pigments in plastics and surface treatment products are responsible for the largest outflows of Cd in Sweden. Cd is also found in artist paint, food, drinking water and as a contamination in zinc products (Månsson and Bergbäck, 2007). The plausible reasons for the in some cases high levels of Cd in food waste sludge from tank systems are therefore many. Further investigations and more samples are needed for increased understanding of the sources to Cd in separately collected food waste from households.

4.4. Household behaviour

Previous studies on disposal behaviour in Swedish households state source-separation ratios of 22–45% in multi-family dwellings (Dahlén, 2008; LRV, 2009; Bernstad, 2010) and 72–79% in single-family households (Swedish Waste Management Association, 2011), in both cases using paper bags for separate collection of food waste. Lack of proper space in kitchen has been singled out as a factor which can decrease the willingness to participate in food waste recycling from the side of the households in areas with paper bag collection schemes (Bernstad, 2010). Such problems are to a large extent avoided with the use of FWD and the source-separation ratio could therefore be assumed to be higher in households with FWD compared to households with bag collection schemes. Due to uncertainties in relation to the actual amount of households using FWD, number of persons in households and losses of organic matter with effluent, the source-separation ratio in residential areas with FWD (A and B) could not be calculated. However, an estimation was made based on the following assumptions and the below equation (Eq. (2)). The number of households connected to system B was 147, but as a large part of these not were occupied during the major part of the sampling period, calculations were based on an estimation of 100 occupied households. TSf was determined based on average TS in collected sludge over the sampling period, while TS was based on data from Carlsson and Uldal (2009) on average TS in separately collected food waste (30%). The amount of collected sludge (\( V_s = 38.4 \text{ m}^3 / \text{year} \)) would thereby reply to 34.2 kg food waste/household/year. A generation of 100 kg food waste per person and year (\( FW_{tot} \)) was assumed (Konsumentföreningen Stockholm, 2009). The number of persons per household in area B was assumed to 1.5.

\[
\frac{(V_s + \delta) \times TS_f}{100 \times FW_{tot} \times n} = 100
\]

\( V_s \), is volume sludge (\( \text{m}^3/\text{year} \)), \( \delta \) is density in collected sludge (kg/m\(^3\)), \( n \) is the Number of connected households, TS is the dry substance in sludge (% of wet waste), TSf is the dry substance in food waste (% of wet waste), \( FW_{tot} \) is the total amount of food waste generated per person and year.

The source-separation ratio was determined to 23%. The relatively low source-separation ratio is in line with waste composition analyses from Swedish areas where FWDs were installed in 1997, where more food waste was found in residual waste from households with FWD compared with households using bag collection or home composting for selective disposal of food waste (Sura-hammar, 2011). More research is needed in relation to different food waste collection schemes and source-separation behaviour.

4.5. Detergents

An increased degradation of particulate organic matter in hydrolysis experiments where shown when detergents had been added to freshly ground food waste. As one of the primary missions of tensides in detergents is degradation of fat, it is probable that the increased levels of dissolved COD mainly where caused by degradation of fat. However, no analyses of hydrolyzed samples were made in order to confirm this.

In a study by Khalil et al. (1989), methane production was halved and the pH dropped from 7.4 to 6.0 over a period of 20 days when anionic detergent (sodium dodecyl-benzene sulphonate – SDBS) had been added at concentrations of 20–50 mg/l, while no inhibition was seen when non-ionic detergent (Tergitol – nonyl phenyl polyethylene glycol ether) or soap had been added at the same concentrations. In the present study, addition of the anaerobic surfactant Sodium dodecyl sulphate at a concentration of 8 mg/l to freshly ground food waste. As one of the primary missions for methane producing organisms without causing lysis of micro-organisms (Khalil et al., 1989).

Inhibition of methane production as a result of spiking with anionic detergents (Linear Alkylbenzene Sulphonate, LAS) has previously been related to the fraction of surfactant in the aqueous phase, rather than the total concentration of detergents (Garcia et al., 2006). The reason is according to Garcia et al. (2006) that a sorption of anionic detergents to suspended solids reduces the availability and thereby the toxicity on anaerobic microorganisms. As sorption is promoted by presence of cations, the concentration of calcium and magnesium extractable cations could be of importance to inhibitory effects from anionic detergents. Further studies
are thereby needed to explore the relation between surfactants and water hardness in relation to inhibitory effects on methanogenic microorganisms. Further studies are also needed to investigate typical concentrations of different types of tensides in household kitchen sink effluent.

4.6. Resource use

Previous studies have shown that the water use in FWD systems can vary largely (Table 1). Nilsson et al. (1990) saw a decrease of water use of 13 l/day in households after FWD had been installed while Lundie and Peters (2005) lift the extra use of water (assumed water use of 13 l/day in households after FWD had been installed in Australia. The data used by Lundie and Peters (2005) would reply to less than 2% of the average daily per capita consumption and be close to irrelevant in a Swedish context (Swedish Energy Authority, 2007). Thus, the feasibility of FWD use must be studies within the local context.

Lundie and Peters (2005) measured the energy need for grinding of food waste to 0.02 kW h/kg waste, while Evans et al. (2010) assume the yearly energy use to FWD to 2–3 kW h/household. The current collection scheme of ground food waste result in an energy use equal to 38 kW h/ton ground food waste, adding energy use in grinders and fuel use for collection and transportation. This can be compared to the energy use for collection of food waste separately collected in paper bags. Use of paper bags, heating/ventilation of facilities where food waste is disposed of by households and temporarily stored before collection (recycling buildings) as well as energy use for collection, transportation and physical pretreatment of food waste has previously been estimated to 56 kW h/ton selectively collected food waste (Bernstad and la Cour Jansen, 2012). Thus, although there are potentials to reduce the energy consumption in the investigated system radically, principally through improved collection technique and reduced transportation of water, the energy consumption in the current system is well below the energy use in more conventional systems for separately collected food waste. The energy input can also be compared to the potential energy recovery from produced food waste sludge through biogas production. The production equals 80–100 N m3 CH4/ton ground food waste, depending on the degradation in the tank before collection (assuming a degradation ratio of 80% of VS in biogas production facilities, based on Davidsson et al., 2007). Thus, the energy consumption in the system equals between 3% and 5% of potential energy recovery. This does not take potential energy savings related to substitution of chemical fertilizers with digestate on farmland into consideration.

4.7. Methods for evaluation

Tank connected FWD-systems were in the present study investigated through a combination of full-scale sampling and measurements and lab scale studies. Although laboratory studies were aimed to mimic full-scale systems, this might not have succeeded in all cases. As an example, potential fugitive emissions of methane between emptying of tanks assessed in laboratory did not include the effect of the constant inflow of new organic material and water to the tank, which might have decreased the risk for a pH drop and thus increased the risk for methane production in the tank.

5. Conclusions

The unconventional food waste disposal systems investigated in the present paper can be seen as a promising method for separate collection of food and kitchen waste from both households and restaurants. The sludge collected from the systems is rich in fat and has a high methane potential when compared to food waste collected in more conventional food waste collection systems. The content of heavy metals is low. However, the Cd/P quote varies largely and can exceed Swedish national regulations for use of bio-fertilizers from an anaerobic digestion on farmland. Detergents in low concentrations in food waste disposer sludge can result in increased degradations rates and biogas production, while higher concentrations can result in temporary inhibition of methane production from ground food waste. The concentrations of N-tot and P-tot in sludge collected from sedimentation tanks were on average 46.2 and 3.9 g/kg TS, equalling an estimated 0.48 and 0.05 kg N-tot and P-tot respectively per year and household connected to the food waste disposer system. The concentration of fat in collected food waste sludge was high, and the losses of fat to the sewerage system are thus vastly decreased through the use of settling tanks. The average concentration of COD in effluent from tanks exceeded 1000 mg/l, of which more than 50% was dissolved. Losses to effluent represent more than 50% of C-tot, N-tot and P-tot in ground food waste. Results indicate that a hydrolysis takes place in the tank, and that this is induced in the presence of older food waste sludge. Increased particle size decreases the hydrolysis rate and could thus decrease losses of carbon and nutrients to the sewerage system, but further studies in full-scale systems are needed to confirm this. Increased collection frequency could increase the amount of carbon and nutrients recovered through the system through sludge collection, but would also increase costs and energy use.

Acknowledgements

The authors would like to warmly thank the VA SYD Solid Waste Management Department, in particular Henrik Aspegren and Roland Nilsson, for all their support and help during the work presented in this paper. The author would also like to thank Hamse Kjerstadius for his help with collection of material used for parts of the laboratory work presented in the paper.

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