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Review paper

Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives

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Abstract

The technology of anaerobic digestion of organic solid wastes is, in many aspects, mature. Topics such as fundamentals (kinetics, modelling, etc.), process aspects (performance, two- and single-phase systems, wet and dry technologies), digestion enhancement (several pre-treatments), co-digestion with other substrates and its relation to composting technology are examined in this review. Special attention is paid to the advantages of anaerobic digestion in limiting the emission of greenhouse gases. An overview of industrial achievements and future developments is given. © 2000 Elsevier Science Ltd. All rights reserved.

Keywords: Biogas; Biomethanization; Biogasification; Hydrolysis; Co-digestion; Industrial; Digestion enhancement; Pre-treatment

1. Introduction

A few months ago, in June 1999, the second International Symposium on Anaerobic Digestion of Solid Waste (II-ISAD-SW) was held in Barcelona. Seven years after the first Symposium, the meeting attracted more than 350 people from 47 countries, which points to the relevance of this biotechnology topic. In consequence, a substantial amount of the research and developments presented at this symposium will be used as a source of data for this review. It has to be stated that this review is focused on the latest publications on the topic of solid organic wastes, but some of the issues discussed may have been under investigation for some years in a number of centres.

'Solid' organic waste is not a very precise term. Normally it is understood as organic-biodegradable-waste with a moisture content below 85–90%. There are many agricultural and industrial wastes meeting this criterion, but the most relevant is the organic fraction of Municipal Solid Waste (MSW), with a daily production in Europe of about 400,000 tons. Due to the large environmental impact of landfills, many of them are due to close in Europe because of the evolution of legislation. In fact, a European directive is being prepared to restrict this practice. Concomitantly, separate collection of fractions of MSW has increased significantly. Biological treatments are the clearest alternative for the putrescent fraction collected separately. These technologies can maximise recycling and recovery of waste components. Among biological treatments, anaerobic digestion is frequently the most cost-effective, due to the high energy recovery linked to the process and its limited environmental impact. Biogas production throughout Europe, could reach over 15 million m^3/d of methane (Tilche and Malaspina, 1998).

Another important waste is municipal sewage sludge. In fact, more than 36,000 anaerobic digesters are today in operation in Europe, treating around 40–50% of the sludges generated (Tilche and Malaspina, 1998). Dewatered sludge can be considered a solid waste and, as such, is included here.

The diversity of papers in the literature dealing with anaerobic digestion of solid wastes reflects the large number of topics within this area. From fundamental aspects, including kinetics and modelling, co-digestion with other organic wastes, pre-treatments to enhance the performance of the digesters, up to the practical full-scale application of this technology, a wide range of topics are relevant and will be discussed in this review.

2. Fundamentals of solid waste digestion

Varied studies of the fundamentals of solid waste digestion have been undertaken. One area is the

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modelling of the process. The availability of a robust anaerobic digestion model, allowing the best operating parameters for optimal control to be defined, would be invaluable. Because of this, many studies have been devoted to this aim. However, there are many difficulties in the modelling of the anaerobic digestion of the organic fraction of municipal solid waste (OFMSW). There are many steps and types of micro-organisms involved and the substrate is a complex one. Most of the models reported in the literature discuss the kinetics of soluble substances and so only consider the fermentative, acetogenic and methanogenic steps (Mata-Alvarez and Cecchi, 1990; Costello et al., 1991). However, when considering solid wastes, as for instance in the model of Kiely et al. (1997) simulating the co-digestion of OFMSW and primary sewage sludge, hydrolysis of complex polymeric substances constitutes the rate-limiting step and must be included in the model (Pavlosthathis and Giraldo-Gomez, 1991; Vavilin et al., 1997, 1999). Thus, this step has engaged most attention from researchers into this particular aspect of anaerobic digestion of solid wastes. For instance, the model of Siegrist et al. (1993) for sewage sludge digestion was used to simulate the hydrolysis of solid wastes, allowing the constants for the hydrolysis of lipids, proteins and carbohydrates to be determined (Christ et al., 1999). These constants are pH-dependent (Zeeman et al., 1999) and even, in a continuous process, slightly dependent on the hydraulic retention time (HRT). This is the case of the dynamic model of Zeeman et al. (1999). In this model, based on a two-bacterial community (hydrolytic-fermentative and aceto-methanogenic), the enzymatic reactions involving enzyme inactivation imply the dependence of the hydrolysis rate constant on both the pH and HRT. Table 1 shows the values of the hydrolysis first-order constant, k, for this model and others in the literature. In the application of another model called METHANE (Vavilin et al., 1997, 1999) to the anaerobic digestion of solid poultry slaughterhouse waste, it was shown that the limiting step was hydrolysis and that it was inhibited by high propionate concentrations (Salminen et al., 1999). Similar results were also found during sheep tallow anaerobic digestion (Broughton et al., 1998).

In fact, due to its relevance within overall biodegradation kinetics, hydrolysis has been widely studied. For instance, Veeken and Hammelers (1999a) determined the rates of hydrolysis for six components of biowaste (whole wheat bread, leaves, bark, straw, orange peelings and grass). The first-order hydrolysis kinetic constants ranged from 0.003–0.15 d^{-1} at 20°C to 0.24–0.47 d^{-1} at 40°C, values which are consistent with those reported for carbohydrates and food waste mixtures (Christ et al., 1999; Vavilin et al., 1999). However, biodegradability of biowaste components ranged from 5% to 90% without dependence on temperature. The authors argue that the increase in hydrolysis rate at increasing biodegradability suggests that the rate of hydrolysis of particulate organic matter is determined by the adsorption of hydrolytic enzymes to the biodegradable surface sites (Veeken and Hammelers, 1999a,b). This concept is supported by the Arrhenius-type behaviour of the hydrolysis rate and the calculated activation energy of 64 ± 14 kJ/mol. Comparison of the hydrolysis rates with data on the performances of batch, dry biowaste digesters showed that the digesters were not running optimally. Authors proposed that the reduction in conversion efficiencies was related to VFA inhibition of hydrolysis due to limited transport of VFA in the biowaste bed.

Sanders et al. (1999) have presented a new mathematical description of surface-related hydrolysis kinetics, appropriate for particulate substrates. They used spherical particles in a batch digestion and particulate starch as substrate. They proposed for this substrate a surface-dependent constant of 4 mg starch/μm^2/h, showing that the surface of the particulate substrate is a key factor in the hydrolysis process. However, in this model authors do not consider the possibility of particles being broken down into smaller pieces.

Other fundamental studies look at different aspects of the anaerobic digestion of solid waste. For instance, Okamoto et al. (1999) discuss the hydrogen potential of several components of solid wastes. They found that carbohydrates were a better precursor than lipids or proteins for hydrogen coming from biological fermentation. Specifically they found the following hydrogen production potentials: cabbage, 26.3–61.7 ml/g VS;

Table 1
First order kinetic constant values for hydrolysis of different materials

Component	Hydrolysis constants	
	k values (d^{-1})	
Lipids	0.005–0.010 (Christ et al., 1999)	
Proteins	0.015–0.075 (Christ et al., 1999)	0.081–0.177 (Zeeman et al., 1999) (value dependent on pH)
Carbohydrates	0.025–0.200 (Christ et al., 1999)	
Food wastes (mixt.)	0.4 (Vavilin et al., 1999)	
Solid wastes (mixt.)	0.012pH–0.042 (Kalyuzhnyi et al., 1999)	
Biowaste components	0.03–0.15 (20°C) (Veeken and Hamelers, 1999a)	
	0.24–0.47 (40°C)	

carrot 44.9–70.7 ml/g VS and rice 19.3–96.0 ml/g VS. The same group studied the potential for hydrogen production using the response surface methodology with central composite designs and found experimentally that the hydrogen composition of the biogas was greater than 60% except for initial incubation, with no significant methane being found throughout the study. Further experiments confirmed that the OFMSW had a considerable effect on biological hydrogen production (Lay et al., 1999).

Other important papers are devoted to analytical questions. Thus, some contributions to a better understanding of the biological mechanisms responsible for an inefficient anaerobic process were made by Ahring and Angelidaki (1997a). These authors focused on the composition and dynamics of the microbial community. Along similar lines, and as part of a larger project for evaluating the possibilities of using a set of electronic gas sensors and near-infrared spectroscopy for on-line monitoring and control of the biogas process, are the studies of Nordberg et al. (1999) and Sundh and Carlsson (1999). The electronic gas sensors for volatile compound mapping, acting as a kind of "electronic nose" and infrared spectroscopy, was aimed at liquid samples. As an application, the authors studied the effects of disturbance caused by glucose overloading on the size and structure of the microbial community. This overloading led to an increased total microbial biomass and a change in the phospholipid components, which implied that different microbial groups reacted differently to the substrate addition (namely some fatty acids doubled in concentration, whereas some diether lipids remained nearly unchanged).

A good example of how fundamental analysis leads to quite practical conclusions is reported in a detailed microbiological study of the well-known leach-bed process (Silvey et al., 1999). Results showed that a new batch could be started in 18–38 d, rather than 60–90 d. Thus, the analytical methods enabled the ecological system to be followed, with control of when the Archaeal numbers were high and the microbial community in the leachate was at its best point for metabolising soluble substrates. Starting another round of the sequencing system at an earlier stage would not only result in a faster turnaround in the reactor, but would lead to a higher quality and quantity of biogas in the first month. In addition, biomass estimates derived from total microbial phospholipid-fatty acid and phospholipid-ether lipid cell wall analysis indicated when conditions were most suited to the respective bacterial and Archaeal populations and the new batch could be started. This, together with denaturing gradient gel electrophoresis analysis, has provided important information on microbial ecology during anaerobic digestion, which helped considerably in the optimisation and better management of MSW treatment using this system.

3. Process aspects: digester performance, ammonia inhibition, one vs. two-phase systems

Many papers have been published dealing with the performance of different reactor configurations digesting organic solid wastes. Most of them focus on aspects of the anaerobic biodegradation of the putrescent fraction of MSW. Despite the increasing number of full-scale plants, research activity continues, especially at universities. For instance, Pavan et al. (1999) studied, at pilot-scale, the application of the semi-dry single-phase thermophilic anaerobic digestion process of two different kinds of substrates: the mechanically sorted and source-sorted OFMSW, the two with very different biodegradabilities. The digester was a CSTR type and was fed with different mixtures of both substrates. To ensure the complete stability of the operation, it was necessary to reduce the organic load when the contents of source-sorted OFMSW increased. In the case of feeding exclusively source-sorted OFMSW, or fruit and vegetable wastes, or, in general, highly biodegradable wastes, it is advisable to use a two-phase anaerobic digestion process, which permits much higher loads in the digester. In fact, many of the studies of reactor performance consider the possibility of a two-phase configuration. Thus, another study dealing with the anaerobic digestion of the "grey waste", i.e. the residual refuse after the OFMSW has been selected, showed how a simple two-stage concept at 65°C in the first stage ($HRT = 4.3$ d) and 55°C in the second stage ($HRT = 14.2$ d) achieved 80% degradation of the volatile solids (VS) (Scherer et al., 1999). These authors also showed that a distinct hydrolysis step could be reached only at hyperthermophilic conditions or at an HRT below 4 d. Degradation extents obtained from the biogas yield (up to 797 l/kg VS fed) revealed up to 98% of the theoretical possible yield. The remaining fraction was lignin-like components. As is known, the physical association between lignin and cellulose is in many cases the limiting factor for anaerobic degradation. This studied topic was also reviewed in the Barcelona conference, in a paper on the biomethanisation of newsprint (Clarkson and Xiao, 1999). These authors argued that alkali pre-treatment significantly improved newsprint biodegradability, but treatments for longer duration or at elevated temperatures did not improve bioconversion of newsprint to methane. A somewhat similar study, because of substrate characteristics, to the one by Scherer et al. mentioned above, was conducted by Ghosh et al. (1999) on treating refuse-derived fuel (RDF) pellets, obtained from unsorted MSW, with bench-scale digesters. They compared a conventional high-rate reactor with a two-phase configuration. With RDF in a conventional digester, they observed that thermophilic digestion at 55°C increased its methane yield by only 7% over digestion at 35°C. Decreasing the

RDF particle size – a classical variable in this type of study – from 2.2 to 1.1 mm had no beneficial effects at mesophilic temperatures, but methane yield was increased by 14% when changing from meso to thermophilic conditions. They also found a 35% increase of methane yield with an optimal dosage of NaOH of 0.5 g per 100 g of RDF – VS. All these yields were higher – around 20% – using a two-phase configuration. As can be seen, considerable efforts have been made to improve digester performance by using some kind of pre-treatment. These aspects are further discussed in Section 4. Two-phase digestion was also considered the right option for treating high-solid wastes (Vieitez and Ghosh, 1999), and source-sorted OFMSW from fruit and vegetable markets (with very high biodegradability) (Pavan et al., 1999a). These latter authors, in a detailed study including kinetics, found optimal operating conditions for both hydrolytic (meso and thermophilic temperatures) and methanizer (thermophilic temperature) reactors. The overall HRT was around 12 d, with an optimal specific biogas production of around 0.6 m^3/kg VS. Raynal et al. (1998) chose the two-phase system to treat several vegetable substrates: potato peelings, green salad leaves, green beans mixed with carrots, apple pomace. The system involved several liquefaction laboratory-digesters, each of them treating one type of waste, linked to a central methane fixed-film reactor. The influences of pH, loading and hydraulic retention time on the process performances were studied at 35°C. On an average, except for apple pomace, hydrolysis yields were high (up to 80%) during the liquefaction step. Likewise, the acidogenic effluent was degraded in a methanation reactor by up to 80%. In a final run with average loading rates near 4 g COD l^{-1} d^{-1} and 17 d for hydraulic retention time, overall organic matter removal reached a value as high as 87%. Another study supporting the advantages of two-phase anaerobic reactors was conducted at mesophilic conditions by Poirrier et al. (1999) who examined the solid waste from the brewery industry. It should be added that on an industrial scale one-phase systems for OFMSW digestion are absolutely predominant (De Baere and Boelens, 1999), probably because they are cheaper (investment and maintenance).

Economically, the use of an anaerobic digester in small-scale sewage treatment plants is not always feasible. Fujishima et al. (1999) suggest a system in which the dewatered sludge discharged from small-scale plants is collected and sent to a plant with an anaerobic digester. In order to find the most appropiate solid contents of the dewatered sludge to be fed to the centralised digesters, they investigated the effect of moisture content on anaerobic digestion of dewatered sewage sludge under mesophilic conditions. They found that methane production decreased when the moisture content of sludge was lower than 91% and they observed that this was due to the non-acclimatisation to high ammonia concentrations of hydrogenotrophic methanogenic bacteria. In order to understand the effect of ammonia concentration on glucose degradation and on the acetoclastic methanogenic reaction by anaerobic microflora incubated at an ammonia-N concentration of 3100 mg N/l, batch experiments were performed at various ammonia-N concentrations. As the concentration of ammonia nitrogen increased from 740 to 3500 mg N/l, the glucose degradation rate significantly decreased. This result from batch experiments was fairly similar to the decrease in carbohydrate removal efficiencies in continuous experiments. Therefore, it seems that the accumulation of ammonia had an inhibitory effect on the glycolytic pathway via which glucose hydrolyzed from carbohydrates was degraded. Methane production rate from acetic acid was almost equivalent at the ammonia-N concentrations of 750, 3400 and 4400 mg N/l. It was clear that the acetoclastic methanogenic bacteria had resistance to the high ammonia concentration. This was the result of sufficient adaptation of the anaerobic microflora to the high ammonia-N concentration of 3100 mg N/l in the digester. In fact, the problem of ammonia inhibition has been addressed by many authors in the literature (see for instance some recent publications by Angelidaki and Ahring, 1993; Hansen et al., 1998 and, more practically, De Baere et al., 1984). As is known, relative concentrations of dissolved ammonia and ionised ammonium are dictated by the system pH. At high pH values unionized form of N-ammonia dominates, and this form is more inhibitory than the ion (see for instance, Hobson et al., 1981). However, dissolved ammonia and ionised ammonium are measured together and most of the papers report this total amount as the ammonia concentration responsible for inhibition.

It is not the aim of this review to cover these aspects in depth, as they have been discussed widely in the literature. Only a few papers directly related to OFMSW and dewatered SS will be mentioned here. More details can be found in a comprehensive literature search conducted by Kayhanian (1999) in order to evaluate ammonia inhibition in anaerobic digestion. The review included the fundamentals of biochemical pathways of N compounds during degradation, mechanisms of ammonia inhibition and the role of free ammonia. With reference to the digestion of OFMSW in dry systems at thermophilic temperature, long-term experimental studies at the pilot-scale revealed that ammonia inhibition occurred at an ammonia concentration of 1200 mg/l. To overcome this problem, two practical methods were tested successfully: (a) dilution of digester content with water; and (b) adjustment of feedstock C/N ratio. Both methods are also discussed by Kayhanian. An alternative way of overcoming the toxicity caused by high ammonia concentration, during dairy-waste anaerobic digestion, was described by Jewell et al. (1999) and consisted in the recirculation of ambient air-dried digested effluent. Total solid (TS) concentration was increased from 10% to 27%, in this way, diluting ammonia-N concentrations. Two reactor configurations were used, namely completely mixed and plug-flow reactors. Recirculated of the dried ("mature") material required less than two-thirds of the mass of the wet feed.

Poggi-Varaldo et al. (1997, 1998) focused their experimental work on the determination of the effects of ammonia-N concentration on the specific methanogenic activity of microbial consortia from a solid substrate anaerobic digestion of municipal and industrial wastes. Bench-scale digesters were operated at COD/N ratios of 90, 80, 65 and 50. At mesophilic temperatures, the process deteriorated with increasing dosages of ammonia-N, with a process cessation at the COD/N ratio of 50. In a similar study, the moisture content limit, at which the methanogenic activity dropped to zero, was investigated by Lay et al. (1997). The threshold limit was found to be 56.6% for the sludge cake, but greater than 80% for meat, carrot and cabbage. In the high-solids sludge digestion, methanogenic activity dropped from 100% to 53% when the moisture content decreased from 96% to 90%. Methane was produced at a good rate at moisture contents of 90–96% in a pH range between 6.6 and 7.8 (optimum at pH 6.8) with possible failures if the pH was lower than 6.1 or higher than 8.3. The authors also considered the dependence of methanogenic activity on NH_4^+ concentration and they found little influence of free ammonia. In the wide pH range of 6.5–8.5, methanogenic activity decreased with the increase in NH_4^+-N concentration, and dropped to 10% at the concentration of 1670–3720 mg NH_4^+-N/l, 50% at 4090–5550 mg NH_4^+-N/l, and dropped to zero at 5880–6600 mg NH_4^+-N/l. However, the lagphase time was dependent on the NH_3 level, but not on NH_4^+, and when NH_3-N was higher than 500 mg/l, a notable shock was observed. This confirms the known fact that the NH_3 level is a more sensitive factor than the NH_4^+ level for an unacclimatized bacterial system.

Finally, it is worth mentioning that in simulating the behaviour of the anaerobic co-digestion process, of OFMSW and primary sewage sludge, including ammonia inhibition, the mathematical model developed by Kyeli et al. (1997) successfully predicts the performance of methane production, and the evolution of pH and ammonia. Free ammonia pH, VFA and long-chain fatty acids constitute the primary modulating factors in the model simulating different co-digestion systems, which was developed by Angelidaki et al. (1999). The model included two enzymatic hydrolytic steps, eight bacterial steps and involved 19 compounds. The model was tested in laboratory-scale reactors co-digesting manure with glycerol trioleate or manure with gelatin. The model was validated using results from a full-scale biogas plant co-digesting manure together with proteinaceous wastewater and with bentonite-bound oil.

There are many references in the literature to successful operations in both mesophilic and thermophilic conditions. Thermophilic temperature was found optimal for digesting mechanically selected OFMSW by Cecchi et al. (1991) in a pilot-plant study. Its advantages were not the same when digesting source-sorted OFMSW (Bernal et al., 1992). Nimmrichter and Kübler (1999) in a laboratory test showed that for a hydraulic retention time of 7 d at 55°C there was greater process stability than at 37°C. With a HRT of 7–12 d, the methane yield of thermophilic digestion was less than 10% above the yield of mesophilic digestion. Against this surplus energy yield, thermophilic digestion involves greater energy demand for heating, which is in many cases approximately the same as the excess energy. Other examples can be given, but, although biogas production yields and bioreaction kinetics seem more favourable at thermophilic temperature, optimal conditions depend on the type of substrate (biodegradability) and type of system (one/two-phase) used.

Table 2 reports other studies of the performance of anaerobic digestion of solid wastes.

Table 2
Studies concerning the performance of digestion of solid wastes presented at the II International Symposium on Anaerobic Digestion of Solid Waste

Substrate	Scale/Reactor type/Temperature	Reference
Slaughterhouse and catering	Pilot/Mesophilic	Membrez et al. (1999)
Poultry mortalities	Lab/Two-phase (Leach bed + UASB) Mesophilic	Chen (1999)
OFMSW in Bamako (Mali)	Pilot (Leach bed + UASB) Psychrophilic	Ouedraogo (1999)
Sewage sludge	Lab/Two-phase/Mesophilic	García-Heras (1999)
Mycelium waste (India)	Non-stirred digester Psychrophilic	Yeole and Ranade (1999)
OFMSW	Lab/One and two stages Psychro and Mesophilic	Wang and Banks (1999)
Coffee pulp	Lab/Batch/Psychrophilic	Valdés et al. (1999)
Fish farming sludge	Lab/Batch/Mesophilic	Gebauer (1999)
OFMSW	Pilot /Two-phase/Thermophilic	Madokoro et al. (1999)
Food Wastes	Lab/Leach Bed/Mesophilic	Paik et al. (1999)
OFMSW	Lab/CSTR/Mesophilic	Houbron et al. (1999)
Coffee pulp	Pilot/Plug flow/Mesophilic	Farinet and Pommares (1999)
OFMSW/Coffee pulp	Pilot/Two-phase	Edelmann et al. (1999)

4. Anaerobic digestion enhancement

As discussed above, significant effort has been dedicated in recent years to find ways of improving the performance of digesters treating different wastes, especially solid wastes, because of the obvious link between successful pre-treatments and improved yields. These treatments can be biological, mechanical or physico-chemical. The economic aspects of digestion enhancement are very important to industry, a point not usually covered in the studies reported.

Among biological methods of improvement, Capela et al. (1999) report the influence of a pre-composting treatment on the start-up and performance of dry anaerobic digestion of pulp mill sludge. The effect was clearly visible through methane yields and consequently solids reduction which were greater than in the digestion of untreated sludge. In the same line of pre-treatment, Hasegawa and Katsura (1999) reported a 50% improvement in yields when sewage sludge was solubilised under slightly thermophilic aerobic conditions prior to anaerobic digestion. They suggest that thermophilic aerobic bacteria secrete external enzymes which dissolve sludge more actively than commercial proteinase. A similar study has also been carried out in a pilot plant in which there is an aerobic step before a leaching operation takes the lixiviates to an anaerobic reactor (Wellinger et al., 1999).

The classic addition of complexes of enzymes has been carried out recently by Rademacher et al. (1999) to improve the efficiency of anaerobic sewage sludge digestion, and by Scheidat et al. (1999) who added to thickened municipal primary sludge a mixture of peptidases, carbohydrolases and lipases (from 0% to 10% on TS) which significantly improved hydrolysis at 39°C and 51°C. However they did not study the technical and economic feasibility of this addition, which are key points in the application of these methods.

4.1. Mechanical pre-treatment

Size reduction of particles and the resulting increase in the specific surface available to the medium improves the biological process. Two effects have been reported: first, if the substrate has a high fibre content and low degradability, their comminution leads to improved gas production; and second, size reduction can lead to more rapid digestion (Palmowsky and Müller, 1999a,b).

Engelhart et al. (1999) studied the effects of mechanical disintegration (by a high-pressure homogenizer) on anaerobic biodegradability of sewage sludge. A 25% increase in volatile solids reduction was achieved. Investigations of degradation of soluble proteins and carbohydrates showed that a slowly degradable fraction of carbohydrates was released via disintegration.

In another study, Hartmann et al. (1999) found an increase of up to 25% in biogas from fibres in manure feedstock, after pre-treatment of the whole feed in a macerator before digestion. The authors recommend this method because of its low operational cost for a fuller degradation of particulate organic matter. Furthermore, looking at the size distribution, they found that the change in biogas potential did not correlate with a smaller size of fibre. Results from the maceration indicate that the biodegradability of the fibres is rather enhanced by shearing, which is not necessarily reflected by a change in size distribution. Confirming these results, Angelidaki and Ahring (1999) found an average increase of 17% biogas potential after mechanical maceration of biofibres contained in manure. In general the smaller the fibres, the higher the biogas potential. The best results showed an increase of about 20% with fibres smaller than 0.35 mm. The chemical treatment of the fibres with NaOH, NH_4OH or a combination also led to increased methane potential. Combination of both treatments, chemical and mechanical, did not lead to any further increase. No significant difference in the biogas potential was found from fibres in the 5–20 mm range. They also studied the above-mentioned addition of hemicellulolytic or cellulolytic enzymes without any improvement in biogas potential. However, with the hemicellulose-degrading bacterium B4, a 30% increase was recorded.

4.2. Solubilization by other means

As stated, anaerobic digestion of solid wastes is rate-limited by the hydrolysis step, and so physico-chemical treatments are often used to promote solubilisation of organic matter. However, the substrate solubilisation step limited the anaerobic digestion of an industrial microbial biomass (Delgenes et al., 1999). A thermo-chemical pre-treatment based on sodium hydroxide addition was used to enhance COD solubilization at the following optimal conditions: $pH = 12$, $T = 140°C$ for 30 min; 70% solubilization was achieved. However, anaerobic biodegradability of the pre-treated substrate did not improve, remaining near 40%. The poor anaerobic biodegradability performances were attributed to the soluble molecules generated being refractory and/or inhibitory to anaerobic micro-organisms. Fractionation of the soluble pre-treated microbial biomass demonstrated that high molecular weight compounds were involved in the poor biodegradability and biotoxicity observed. Contrary to these findings, Schieder et al. (1999) stated that, with increasing pressure and temperature, the organic part of the waste is split up into short-chain fragments that are biologically well-suited to micro-organisms. After testing with food scraps and canteen waste in a pilot plant for 1800 tons raw material per year, they claim that the thermal hydrolysis process gives complete energy recovery, i.e. more energy is produced than is needed for running the plant.

5. Co-digestion

An interesting option for improving yields of anaerobic digestion of solid wastes is co-digestion. That is, the use of a co-substrate, that in most cases improves the biogas yields due to positive synergisms established in the digestion medium and the supply of missing nutrients by the co-substrates. In addition, economic advantages derived from the fact of sharing equipment are quite significant. Sometimes the use of a co-substrate can also help to establish the required moisture contents of the digester feed. Other advantages are the easier handling of mixed wastes, the use of common access facilities and the known effect of economy of scale. However, some drawbacks also exist, mainly due to slurry transport costs and the problems arising from the harmonisation of different policies of the waste-generators.

The advantages of co-digestion have been sung for some time. For instance, Mata-Alvarez and Cecchi (1990) referred to the co-digestion of OFMSW with sewage sludge in existing digesters. Both wastes are produced in large quantities and in many places, and much research has focused on this particular issue. Co-digestion of manure and industrial organic wastes has been widespread in Denmark – around 20 centralised co-digestion plants – since the late 1980s, with very interesting results (Danish Energy Agency, 1995). However, apart from this, few reports on industrial applications of this concept have been published. Most industrial co-digestion plants treat OFMSW plus (in a relatively small percentage) some other organic waste, such as sewage sludge (Rintala and Jarvinen, 1996). However, despite the positive results in the studies listed here (Demirekler and Anderson, 1998; Di Palma et al., 1999; Edelmann et al., 1999; Converti et al., 1997; Poggi-Varaldo et al., 1997a; Sundararajan et al., 1997; Griffin et al., 1998; Ahring and Angelidaki, 1997b, inter alia), the scarce industrial application of co-digestion is surprising. De Baere (1999) explains this by suggesting that commonly an organic solid co-substrate is added to manure digesters in small amounts, but often these co-substrates are high-energy yielding industrial sludges and only exceptionally is solid waste from households or market waste added. In fact, less than 7% of the overall anaerobic digestion of OFMSW capacity is at present co-digested. Nevertheless, there is ongoing research. For instance, Kübler et al. (1999), following an extensive study sponsored by the Bavarian Environmental Authority (Hoppenheidt et al., 1998), tested different co-substrates for OFMSW. The tests carried out in an industrial facility showed surplus energy production and no negative impact on digester performance. A full-scale simulation was staged to co-digest, thermophilically, wastes coming from kitchens, slaughterhouses and meat-processing industries (Brinkman, 1999). For instance, in North America, this option has been examined in a study evaluating the technical feasibility of the anaerobic co-digestion process for typical solid wastes (Hamzawi et al., 1998a,b). Using biological activity tests, an optimal mixture, for biogas production, was identified as 25% OFMSW and 75% sewage sludge (65% raw primary sludge, 35% thickened activated sludge (TWAS)). Also based on the rate of biogas production, the most anaerobically biodegradable components of the OFMSW were paper and grass. TWAS and newspaper were found to be the least biodegradable components. As stated before, lab-scale testing indicated that alkaline pre-treatment increased the biodegradability of the sewage sludge/OFMSW mixture over that of the untreated control. Thermochemically pre-treated feedstocks inhibited anaerobic biodegradability in comparison with the untreated control, whereas the anaerobic biodegradability of thermally pre-treated feed was found not to be significantly different from that of the control. The authors also developed empirical models based on alkaline dose, total solids concentration in feed and particle size, biogas production and removal of TS and VS. All five experimental factors were found to be significant with respect to the response variables studied. Co-digestion of OFMSW and primary sewage sludge (SS) has also been modelled under a wide range of operative conditions (Kyeli et al., 1997).

Also in North America, another comparative study of two digestion systems (wet and dry) was based on the co-digestion of tuna sludge and OFMSW. This study is currently scaled to pilot-plant after the good yields obtained (Rivard et al., 1998b). Details of the demonstration plant can be found in Rivard et al. (1998a).

In quite an interesting study of co-digestion, Hammes et al. (1999) considered the possibilities of treating black water (produced from recently developed vacuum/dry toilets) together with other types of human-generated solid wastes (biowastes/mixed wastes) in an anaerobic reactor system at thermophilic conditions. Among other conclusions these authors found that anaerobic digestion offers the possibility of integrating and simplifying domestic waste management while producing biogas and residues, which can either be used for agricultural purposes or be further treated through processes such as incineration (Hammes et al., 1999).

As a solution to the problem of ammonia inhibition during the anaerobic digestion of chicken manure (see for instance, Krylova et al., 1997), it has been proposed that co-digestion with cattle slurry could be a possible disposal route (Callaghan et al., 1999). In fact, these authors have tested several mixtures of cattle slurries with a range of different wastes, allowing them to digest in 1-l batch digesters. The criteria for judging the success of a co-digestion were volatile solids (VS) reduction, total methane production and methane yield. In terms of the VS reductions (%), there was little difference between the various digestions. In terms of the cumulative methane production, the co-digestions with fruit and vegetable waste, fish offal and dissolved air flotation sludge were more effective than the digestion with cattle slurry alone. In terms of the specific methane yield (m^3 CH_4/kg VS removed), the co-digestions containing fish offal and brewery sludge gave higher values than the control digestion with cattle slurry alone. Compared with their control (cattle slurry alone), both co-digestions with poultry manure (7.5% and 15% TS) gave higher cumulative productions of methane, and the system with the lower concentration of poultry manure gave a higher specific methane yield. However, there was some evidence of ammonia inhibition.

Di Palma et al. (1999) reported the co-digestion of OFMSW and sewage sludge in a laboratory experiment and Edelmann et al. (1999) did the same, but in an industrial trial in Switzerland: fruit and vegetable wastes were chopped and then reduced to a size of 1–2 mm in order to obtain a homogeneous suspension with the primary sludge. The results showed an acceleration of the digestion process as well as an increase in the degree of anaerobic digestion of the sludge. These findings corroborate those of Oleszkiewicz and Poggi-Varaldo (1997). In addition, the particle size of the organic waste has an influence on its dewaterability after co-digestion with sewage sludge. The presence of organic waste residues improves the dewaterability measured as specific resistance (Palmowsky and Müller, 1999a). With the same components, Demirekler and Anderson (1998) recommended a ratio for primary sewage sludge: OFMSW of 80:20, after testing ratios of 100:0, 80:20 and 60:40 on a TSs basis. The experimental work was carried out in three laboratory-scale semi-batch anaerobic digesters, operated at 35°C. Addition of SS significantly decreased the imbalances observed during the start-up and improved the process performance. In all cases inoculum from an operating anaerobic digester was used, enabling both a quick start-up and the capacity for handling high organic loading rates (OLRs) from the beginning of the experiment.

Working with unsorted MSW, Borghi et al. (1999) examined co-digestion with sewage sludge, and also the digestion enhancement experienced with different chemical, biological and thermal pre-treatments.

Other examples of co-digestion presented at the II-ISAD-SW are summarised in Table 3. Finally, co-digestion is used in many plants for starting up digesters. As an example, Griffin et al. (1998) showed how a mesophilic (35°C) anaerobic sewage sludge, together with cattle manure, was used successfully to start up a thermophilic (55°C) digestion of biosolids and simulated municipal solid waste.

6. Anaerobic digestion vs. composting

Anaerobic digestion effluents are not generally suitable for putting directly onto the land. They are too wet, contain a notable amount of volatile fatty acids which are somewhat phytotoxic and, if digestion has not occurred within the thermophilic range of temperatures, are not hygienised. Thus, it is generally accepted that post-treatment after anaerobic digestion is needed to obtain a high-quality, finished product (Poggi-Varaldo et al., 1999). A question is posed when comparing direct aerobic composting with the combination of anaerobic + aerobic treatments, in that, it appears that with anaerobic technology a variable amount of energy is recovered, whereas composting is a net energy consumer. However, it is also true that anaerobic technology requires larger investment and that the overall process is more complex. Many comparisons have been carried out in the past, but with different results, many of which depended on energy costs. However, with the development of holistic tools such as Life Cycle Analysis (LCA), a more ecological comparison can be made. In this sense Edelmann et al. (1999) compared, in both ecological and economical terms, different processes for treating biogenic wastes in plants with a treatment

Table 3
Examples of codigestion presented at the II-ISAD-SW

Co-substrates	Comments	Reference
Olive mill effluents (OME) with pig manure (PM) and sewage sludge (SS)	Ratios used: OME/SS: 1/5 OME/PM: 1/1 With PM, a COD reduction of up to 75% was achieved	Schmidt et al. (1999)
Organic wastes and agricultural manures	Discussion of technical and quality requirements for co-digestion	Amon and Boxberger (1999)
Landfill leachate and septage	Overall COD removal of around 71%	Lin et al. (1999)
PM and organic wastes from food industry	Mesophilic results were better than thermophilic ones. In both T's biogas yields of PM were improved	Campos et al. (1999)
PM and other organic wastes	Agricultural cooperative-project partially using laboratory results	Pouech and Castaing (1999)
Solid manure and OFMSW	Pilot plant with agronomic tests	Membrez and Glauser (1997)

capacity of 10,000 tons/yr of organic household wastes. After a series of measurements at compost plants they found that the methane emissions were greater than they had assumed and showed, with LCA tools, that anaerobic digestion had the advantage over composting, incineration or combination of digestion and composting, mainly because of its improved energy balance. They concluded that anaerobic processes will become much more important in the future for ecological reasons. In fact, the future of anaerobic digestion should be sought in the context of an overall sustainable waste-management perspective. Aerobic treatment produces large and uncontrolled emissions of volatile compounds, such as ketones, aldehydes, ammonia and methane. Interest in emission into the atmosphere of gases from organic solid-waste treatment has increased in recent years (Bjorkqvist et al., 1998). Table 4, extracted from De Baere (1999), shows the different emissions of volatile compounds during aerobic composting and during maturation after anaerobic digestion. Quite explicit in this sense is a study in which two different biowaste composting techniques were compared for their overall emission of volatile organic compounds (VOC) during the composting period (Smet et al., 1999). In the aerobic composting process, the biowaste was aerated for 12 weeks, while the combined anaerobic/aerobic composting process consisted of a 3-week anaerobic digestion period (phase I) and a 2-week aeration period (phase II). While the emission of volatiles during phase I of the combined anaerobic/aerobic composting process was measured in a full-scale composting plant, the aerobic stages of both composting techniques were performed in pilot-scale composting bins. Predominance of alcohols (38% wt/wt of the cumulative emission) was observed in the exhaust air of the aerobic composting process, while predominance of terpenes (87%) and ammonia (93%) was observed in phases I and II of the combined anaerobic/aerobic composting process, respectively. In the aerobic composting process, 2-propanol, ethanol, acetone, limonene and ethyl acetate made up about 82% of the total VOC emission. As well as this, the gas analysis during the aerobic composting process revealed a strong difference in emission profile as a function of time between different groups of volatiles. The total emissions of VOC, NH_3 and H_2S during the aerobic composting process was 742 g/tons biowaste, while the total emission during phases I and II of the combined anaerobic/aerobic composting process were 236 and 44 g/tons biowaste, respectively. The 99% removal efficiency of volatiles upon combustion of the biogas in phase I in the electricity generator made the combined anaerobic/aerobic composting process an attractive alternative to aerobic biowaste composting. Its emission of volatiles was 17 times lower than aerobic composting.

In terms of global warming, which is often used as a reference value for ecological balance, anaerobic digestion scores much better than other options, as can be seen in Table 5, extracted from Baldasano and Soriano (1999). This table corroborates another study carried out by Pier and Kelly (1997) dealing with sawdust waste. More specifically, Kübler and Rumphorst (1999) report that a 25–67% reduction in CO_2 emission (depending on the use of the exhaust heat) is achieved with a combined plant using anaerobic digestion and aerobic post-composting. In addition, the total produced electrical energy exceeds the amount of energy used for erection and operation of the plant. In fact, for a plant treating 15,000 tons/yr of OFMSW, around 0.75 million kWh/yr are needed, whereas for anaerobic digestion the net production is ca. 2.40 million kWh/yr.

7. Industrial perspective

In industrial terms, anaerobic digestion of solid waste can be seen as a mature technology (Riggle, 1998). Over the past 10 years and for the treatment of OFMSW, it has evolved positively from an overall capacity of

Table 4
Emissions of volatile compounds during aerobic composting and during maturation after anaerobic digestion, expressed as grams per ton of biowaste (extracted from De Baere (1999))

Compounds	Aerobic	Maturation after anaerobic	Ratio aerobic/anaerobic
Alcohols	283.6	0.033	8593.9
Ketones	150.4	0.466	322.7
Terpenes	82.4	2.2	37.5
Esters	52.7	0.003	17566.7
Organic sulphides	9.3	0.202	46.0
Aldehydes	7.5	0.086	87.2
Ethers	2.6	0.027	96.3
Total VOC	588.5	3.017	195.1
NH_3	158.9	97.6	1.6
Total	747.4	100.617	7.4

Table 5
Emission factors for different MSW management systems

Treatment	Emission factor (tons eq. CO_2/tons of MSW)
Landfill	1.97
Incineration	1.67
Sorting + Composting + Landfill	1.61
Sorting + Composting + Incineration	1.41
Sorting + Dry biomethaniza-tion + Landfill	1.42
Sorting + Wet biomethaniza-tion + Incineration + Landfill	1.19

Table 6
Biogas production from different sources in Europe[a]

Source	Today estimate (10^6 m^3/d CH_4)	Estimated potential (10^6 m^3/d CH_4)
SS	1.7	4
OFMSW	4.5[a]	15[a]
Industrial wastewater	0.8	3
Animal wastes	0.5[a]	10[a]
Total	7.5	32

[a] Estimates for today and potential (Tilche and Malaspina, 1998).
[a] Net methane emission reduction.

122,000 tons/yr in 1990 to more than 1,000,000 in the year 2000, according to De Baere (1999). Compared with the installed capacity for composting plants this is not a large figure, but it has to be taken into account that most aerobic plants were constructed before anaerobic digestion was considered a fully established technology. De Baere's study identified a total of 53 plants with a capacity larger than 3000 tons/yr. Around 60% of the plants operate at the mesophilic range (40% thermophilic). Capacity has increased at a rate of around 30,000 tons/yr during the period 1990–1995 and around 150,000 during the last five years. An increase of around 200,000 tons/yr by the year 2001 is envisaged.

Yields from the biomethanization process are very much dependent on the particular situation of each plant. Of course the main factor affecting this yield is the kind of substrate used. In the case of the OFMSW it is interesting to see how different values are obtained during a long period, using the same process but different sorting procedures (Saint-Joly et al., 1999), but there are also some other influences such as the desired end destination of the product and the type of process selected.

In the last few years a remarkable interest in digesting "grey wastes" or "residual refuse", i.e. what remains after source separation, has arisen. Options for this fraction are landfilling or incineration. However, anaerobic digestion offers a number of advantages such as: (a) greater flexibility, (b) the possibility of additional material recovery (up to 25%) and (c) a more efficient and ecological energy recovery: the low-calorific organic fraction is digested, the high-calorific fraction is treated thermally and the non-energy fractions can be recovered and reused (De Baere and Boelens, 1999). It is expected that this residual refuse will open up a new "market" for anaerobic digestion.

8. Final considerations

A very high growth potential is envisaged for the anaerobic digestion of OFMSW. Today around 50% of MSW is landfilled, with a content of around 30% of organic fraction (without considering paper and cardboard). Following in potential is the digestion of sewage sludges. Table 6, extracted from Tilche and Malaspina (1998) shows an estimate of the biogas potential of several wastes, among them solid ones. As can be seen, the growth potential for this technology is very important, especially because of the important factor of the greenhouse gases emission reduction agreed at the Kyoto Summit. This aspect gets reinforced considering that composting, the current biotechnology for OFMSW recycling, is more problematic, as discussed in this paper. For instance, looking at Table 6 and considering data from Tilche and Malaspina (1998), a daily reduction of 180,000 tons of CO_2 equivalents can be estimated, that is ca. 30% of the global emission reductions agreed in Kyoto.

Another factor that in the near future will contribute to the consolidation of anaerobic digestion as a mainstream technology for the OFMSW is the fact that the digested residue can be considered quite stable organic matter with a very slow turnover of several decades given adequate soil conditions. In this way the natural imbalance in CO_2 can be adjusted by restoring or creating organic rich soil (Verstraete et al., 1999). The removal of CO_2 constitutes an extra benefit that could help place AD among the most relevant technologies in this field.

Aspects such as the degradation of chlorinated compounds need to be examined in greater depth, as anaerobic treatment offers high potential in this area (Christiansen et al., 1999; Verstraete et al., 1999). The same can be said about the treatments for digestion enhancement described in this paper.

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