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*Review*

# Research advances in dry anaerobic digestion process of solid organic wastes

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**The dry anaerobic digestion process is an innovative waste-recycling method to treat high-solid-content bio-wastes. This can be done without dilution with water by microbial consortia in an oxygen-free environment to recover potential renewable energy and nutrient-rich fertilizer for sustainable solid waste management. It generally takes place at solid concentrations higher than 10% and enables a higher volumetric organic loading rate, minimal material handling, lower energy requirements for heating, limited environmental consequences and energetically effective performance. The long retention time, poor startup performance, incomplete mixing and the accumulation of volatile fatty acids (VFAs) are considered as the main disadvantages for the solid-state fermentation process. In order to develop feasible dry anaerobic digestion processes, it is important to review the optimization techniques and suggest possible areas where improvements could be made. These include reactor configuration, mixing, solid retention time, feedstocks, organic loading rate, inoculation, co-digestion, pretreatment, percolation, additives and environmental conditions within the digester such as temperature, pH, buffering capacity and VFAs concentration.**

**Key words:** Solid organic wastes, dry anaerobic digestion process, biogas, optimization.

## INTRODUCTION

Waste generation is a natural consequence of human life, and is increasing along with population growth, urbanization and industrialization. The quantity of waste generation is mostly associated with the economic status of the society, and the proper management of that waste is consistent with an improved quality of life. Continued open dumping and unsophisticated land filling of solid waste in major cities of developing world will result in significant health and environmental consequences (Lou and Nair, 2009) because the uncontrolled decomposition of waste could lead to epidemic diseases, proliferation of foul odors and climate change (Ghosh et al., 1997). Incineration for energy recovery requires costly capital investment and poses potential societal and environmental health risks (Oliveira and Rosa, 2003).

Though high quality compost can be obtained, composting emits uncontrolled leachate, methane, and is a net energy consumer (Walker et al., 2009). Anaerobic treatment is a cost effective, efficient and feasible process to solve multifaceted waste problems. It has reduced environmental impact, especially with respect to the greenhouse effect and global warming (Braber, 1995; Edelmann, 2003; Mbuligwe and Kassenga, 2004). Anaerobic digestion (AD) is the complex decomposition process of organic matter by microbial consortia in an oxygen-free environment to produce renewable energy, such as methane and hydrogen, and reclaim nutrient rich fertilizer (Angelidaki and Sanders, 2004; Chen et al., 2008; Radwan et al., 1993; Tafdrup, 1995). Besides generating biogas for energy use, the process also destroys pathogens and produces stabilized material to be used as organic compost for land application (Asia et al., 2006). Thus, anaerobic treatment provides a method of reducing pollution from agricultural, municipal and industrial operations while offsetting the operations' usage of fossil fuels.

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The dry anaerobic digestion process has been regarded as an innovative waste recycling approach to treat high-solid-content bio-wastes (>10%) in its produced form (De Baere, 2000; Kottner, 2005; Schafer et al., 2006). The performance of dry digestion process is very robust as it allows very high production rates (Gunaseelan, 1997). It can be applied to digest a wide range of solid feed stocks, including animal wastes, agricultural residues, municipal wastes, industrial solid effluents, energy crops and waste water sludge, during which the chemical oxygen demand (COD), biochemical oxygen demand (BOD) and volume of bio-wastes can be reduced with the recovery of renewable energy (Bolzonella et al., 2000; Li et al., 2011b; Radwan et al., 1993).

Compared with wet anaerobic digestion, dry anaerobic treatment is beneficial due to its compact digester with higher volumetric organic loading rate, lower energy requirements for heating, no process energy for stirring, reduced nutrient run off during storage and distribution of residues, limited environmental consequences and its energetically effective performance as it requires less pre-treatment and added water (De Baere, 2000; Kuroshima et al., 2001; Pavan et al., 2000). This process also results in a lower production of leachate and easy handling of digested residues that can be further treated by aerobic composting processes or used as organic fertilizer (Brummeler, 2000). As the biodegradation of undiluted and diluted manures do not vary considerably (Bhattacharya and Mishra, 2003), the quality of biogas and the specific gas production rates are practically identical in both solid and liquid anaerobic digestion processes (Jha et al., 2010a; Luning et al., 2003). The average methane content in biogas was about 66% in dry mesophilic anaerobic digestion of water sorted organic fraction of municipal solid waste (Li et al., 2010).

Though the dry anaerobic digestion process has attracted increased attention all around the world because of its reduced cost in digesters and slurry handling, the process sometimes suffers from inhibition problems (Liu et al., 2006) and is harder to control. First, the solid-state anaerobic digestion requires a larger amount of inocula and much longer retention time (Li et al., 2010). The retention time of dry digestion for farm wastes is approximately three times longer than wet digestion (Schafer et al., 2006). Second, the accumulation of volatile fatty acids (VFAs) restricts the biogas yield. Third, the medium (solid wastes) is complex and heterogeneous in the terms of structure, composition and size (Buffière et al., 2006). The digester behave as a viscoelastic material with yield stresses that increase with both solid concentration and the larger size of the aggregates, ranging between 200 and 800 Pa (Garcia-Bernet et al., 2010). Thus, the complete mixing is hard to achieve. Therefore, this technology needs enhanced reliability of operation to become more sustainable (De Baere, 2006). In order to develop feasible dry fermentation processes for potential energy recovery and sustainable waste management, it is important to review the optimization techniques and suggest possible areas where improvements could be made, including the reactor configuration, mixing, feedstocks, organic loading rate, inoculation, retention time, co-digestion, pretreatment, percolation and additives. Optimization of environmental conditions within the digester such as temperature, pH, buffering capacity and volatile fatty acid concentrations should also be considered for maximizing the biogas production in a shorter retention time.

## BASICS OF ANAEROBIC DIGESTION PROCESS

The anaerobic digestion process is complex with a number of sequential and parallel steps that are carried out by different types of microbes (Balstone et al., 2002; Pavlostathis and Giraldogomez, 1991; Rittmann and McCarty, 2001). In the dry anaerobic digestion process, the undiluted substrate is pretreated and fed into an airtight digester under strict anaerobic conditions. In the absence of oxygen, the multifaceted coordinated activities of the anaerobic bacteria decompose biodegradable matter into methane, carbon dioxide and other gases. The relative abundance of Archaea such as methanogens in the anaerobic reactor is directly correlated with organic loading rate, volatile solids removals and methane production (Montero et al., 2008). The main reactions in the dry anaerobic digestion process are explained in Figure 1. The methane fermentation process of solid organic material basically includes hydrolysis, acidogenesis, acetogenesis and methanogenesis in sequence (Veeken et al., 2000). Anaerobic digestion begins with bacteria that hydrolyze complex organic polymers, such as carbohydrates, proteins, lipids and fats, into simple monomeric carbohydrates, amino acids, sugars and long chain fatty acids (LCFA) by extra cellular enzymes. Dry anaerobic digestion generally exhibited a poor startup performance as hydrolysis is a rate-limiting step (Ahn and Smith, 2008; Veeken and Hamelers, 1999; Veeken et al., 2000). The monomeric compounds are then converted by fermentative anaerobic bacteria into a mixture of VFAs and other minor products such as alcohol, carbon dioxide and hydrogen. Guendouz et al. (2010) reported that hydrolysis is not the only rate-limiting step during dry anaerobic digestion of the solid wastes. The mechanisms associated with VFAs uptake might play an important role in the process. Acetogenic bacteria further convert the organic acids to acetate, carbon dioxide, and hydrogen, which are the direct substrates for methane production (Gerardi, 2003). The final stage of methane fermentation process is methanogenesis by two groups of methanogens. Acetotrophic methanogens split acetate into methane and carbon dioxide (approximately 70%) while hydrogenotrophic methanogens use hydrogen and

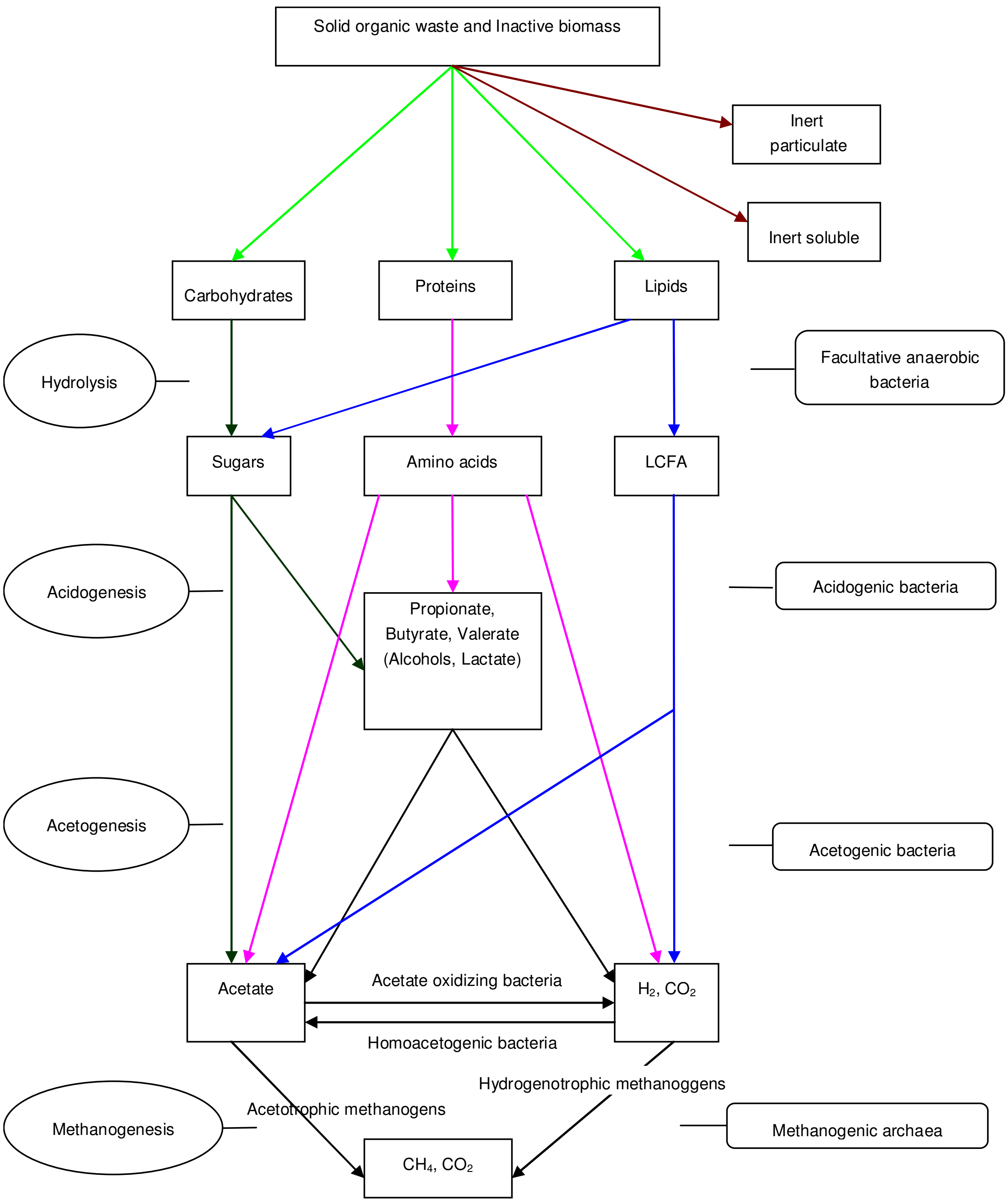


**Figure 1.** Main steps and pathways of dry anaerobic digestion process (modified from Batstone et al., 2002).

carbon dioxide to produce methane (approximately 30%) (Appels et al., 2008; Gerardi, 2003). During methanogenesis, hydrogen is used as an electron donor, with carbon dioxide as an electron acceptor to form methane, while acetate is cleaved to form methane from the methyl group and carbon dioxide from the carboxyl group in a fermentation reaction (Rittmann and McCarty, 2001). As intermediate products, VFAs have been treated as an indicator of the digestion efficiency, but high concentrations of VFAs will result in a decrease of pH, inhibition of acidification, destruction of methogenesis, ultimately leading to performance failure of the digester

**Table 1.** Operational characteristics of dry anaerobic digesters.

| Substrate | Reactor type | Temperature (℃) | Retention time (day) | Optimal gas yield (L $CH_4$/kg VS) | Configuration | VS reduction (%) | Reference |
|---|---|---|---|---|---|---|---|
| Horse dung with straw | Batch with percolation | Mesophilic | 42 | 170 | Single-phase | 44 -49 | Sigrid et al., 2008 |
| Corn stover | Batch | 37 | 40 | 372.4 | Single-phase | 44.4 | Zhu et al., 2010 |
| Municipal solid waste | Batch | 30 | 60 | 314 | Single-phase | 41.8 | Li et al., 2010 |
| Municipal solid waste | Continuous, plug flow | 55 | 25 | 278.4 | Single -phase | 59.21 | Chaudhary, 2008 |
| Municipal solid waste | Batch | 55 | 28 | 320 | Single -phase | 86 | Juanga, 2005 |
| Municipal solid waste | Batch with paddle mixer | Mesophilic | 30- 40 | 200 | Single-phase | | Guendouz et al., 2010 |
| Food wastes | Batch | 37 | 120 | 373 | Two-phase | 90 | Cho et al, 1995 |

(Gerardi, 2003).

The kinetic modeling of the anaerobic degradation of solid organic wastes is increasingly needed for a better understanding of the performance of these systems. It is essential for the rational design and operation of biological waste-treatment systems to predict the system stability, effluent quality, and waste stabilization. Kalyuzhnyi et al. (2000) has developed a structured mathematical model of anaerobic solid-state fermentation (ASSF), including multiple-reaction stoichiometry, microbial growth kinetics, material balances, liquid-gas interactions and liquid phase equilibrium chemistry. The theoretical model agrees on the qualitative level with existing experimental studies of ASSF. Based on computer simulations that model influence of biodegradability and mass transfer intensity on the fermentation process stability, possible measures were proposed to prevent accumulation of VFAs inside the "seed" particles beyond their assimilative methanogenic capacity. Viéitez et al. (2000) developed a Monod-type product-formation model that was used to predict methane formation and to determine kinetic parameters for the methanogenic processes in a simulated landfill or methane reactors. They found that landfill solids hydrolysis was even slower than the inhibited methanogenesis rate. 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 depend on pH and retention time (Zeeman et al., 1999).

## HIGH SOLID DIGESTER

### Reactor configuration

The performance of a reactor to produce the maximum volume of methane depends upon various factors including its configuration, substrate characteristics, organic loading rate, and retention time. In dry digestion process, batch and plug flow digesters (continuous process) are generally used (Table 1). A single-stage process is traditional and a two-stage process has been developed based on the separation of acidogenic phase and methanogenic phases in the digestion process. Both single- and two-stage processes can be conducted in batch fermentation or in continuous flow fermentation. The batch digestion process is a simple, cost-effective and economically viable means for the conversion of organic wastes to useful energy (Brummeler and Koster, 1989b). A multi-stage system can improve both stability of the process and performance, but is more expensive and complicated in construction and operation. It was found that the two-stage digester had a 6 to 8% higher specific methane yield and a 9% more effective volatile solids removal than the single-stage digester when thermophilically treating cattle manure (Nielsen et al., 2004). A 21% increase in methane yield was observed when a two-stage digester was used to ferment municipal solid waste instead of single-stage reactor (Liu et al., 2006). The VFAs produced at the initial stage of solid-state fermentation process for Korean food wastes containing 15 to 30% total solids (TS) was largely controlled using the two-phase digestion system (Cho et al., 1995). The two-stage fermentation process is also useful in the anaerobic treatment of nitrogen-rich wastes such as excess waste sludge, cow feces, chicken feces, and food waste without the dilution of ammonia produced either by water or carbon-rich wastes (Naomichi and Yutaka, 2007). The biphasic process, which consisted of solid-state, acidogenic fermentation of the organic fraction of municipal solid wastes (MSW) followed by biomethanation of acidic hydrolysates in a separate methane fermenter, resulted in a carbohydrate, lipid, and protein conversion efficiency of 90, 49, and 37%, respectively (Viéitez et al., 2000). The phase-separated anaerobic reactor stabilizing *Colocasia esculenta* (Taro) processing waste showed a better performance in the terms of biogas yield (Bindu and Ramasamy, 2005). Sharma et al. (2000) has reported that for the purpose of energy recovery, the plug-flow type reactor can be operated with shorter retention times, thus reducing its overall dimensions and subsequently the overall cost, significantly. The feasibility of tumble-mix fermentation of dry beef cattle manure was established by Schulte and Kottwitz (1982) and it was found that at steady state, using a 23% influent volatile solid content, 54% destruction of the organic matter was achieved with a volumetric biogas production rate of 1.4 $m^3 \cdot m^{-3} \cdot d^{-1}$ at an organic loading rate of 4.7 $kg \cdot m^{-3} \cdot d^{-1}$ in terms of volatile solid (VS). The mathematical model of a continuous steady state plug-flow reactor for VFAs production during the acidogenic process at different retention times (between 2 and 6 days) at 37±2°C was developed with an inhibition effect of the fermentation product on microorganisms' growth was taken into account (Sans et al., 1994).

## Mixing

Adequate mixing can enhance biogas production (Kalia and Singh, 2001) due to the distribution of substrates, enzymes and microorganisms throughout the digester. Mixing creates a homogeneous substrate preventing stratification and formation of a surface crust, and ensures solids remain in suspension. Mixing also promotes heat transfer, particle size reduction as digestion progresses and release of produced gas from the digester contents (Prasad et al., 2008). Mixing can not only ensure well organized transfer of organic material to the active microbial biomass, but also discharges gas bubbles trapped in the medium and helps avoid sedimentation of denser particulate matter (Ward et al., 2008). The feedstocks for the most of the anaerobic digesters are slowly mixed to attain optimal digestion, as excessive mixing will reduce biogas production (Prasad et al., 2008). Prasad et al. (2008) also observed that in comparison to continuous mixing, intermittent and minimal mixing strategies improved methane productions by 1.3 and 12.5%, respectively. Karim et al. (2005) suggested various mixing methods such as mechanical agitation, recirculation of biogas through the bottom of the reactor, or hydraulic mixing by recirculation of the fermented liquid with a pump. In the dry anaerobic digestion process, mixing is difficult and comparatively expensive (Guendouz et al., 2010) as dry reactors have smaller volume than wet reactors for a comparable organic loading rate (Ward et al., 2008).

## Solid retention time

Solid retention time (SRT) is well known as one of the most important factors that influence the performance of anaerobic digestion systems. The optimum SRT can vary depending on factors like waste characteristics, type and design of reactor, environmental conditions within the digester, and microorganisms involved in the process. Nges and Liu (2010) observed that shortening the SRT led to increases in gas production rate and volumetric methane productivity from dewatered sewage_sludge and a decrease in VS destruction efficiency. Nges and Liu (2010) also found that the thermophilic conditions exhibited superior VS-reduction at shorter SRTs (15 to 5 days) while mesophilic digestion yielded the highest VS-reduction at longer SRT (35 to 20 days). Thermophilic digestion has been proven to accelerate reaction rates and reduce SRT. Indeed, lower retention times are required in digesters operated in the thermophilc range. The performance of the dry (with a total solid (TS) of 30% ) thermophilic (55°C) anaerobic digestion of the organic fraction of municipal solid wastes (OFMSW) operating in a semi-continuous regime of feeding showed that 15 days was the optimum SRT for this process (Fdez.-Güelfo et al., 2011).

## Organic loading rate

Organic loading rate (OLR) is a measure of the biological conversion capacity of the anaerobic digestion system (Chaudhary, 2008; Trans, 2009; Verma, 2002). OLR evaluates the efficiency of a digester, required food-to-microbes (F/M) ratio, and indeed the overall process-performance. Dry digesters can tolerate much higher OLR than the wet anaerobic digestion process. A single-stage batch system was simple to operate but had low OLR and fluctuating gas production, while a higher OLR and more stable reaction was observed in a two-stage continuous system (Liu et al., 2007). The increase in OLR by shortening SRT enhanced biogas production (Nges and Liu , 2010) but the percentage degradation in TS, VS, COD and BOD was comparatively low. Chaudhary (2008) has reported that the dry continuous anaerobic digestion reactor stabilizing source-sorted OFMSW showed stable performance with highest biogas yield (278.4 $LCH_4$/kgVS) and VS reduction of around 59.21% during loading rate 2.5 kg VS/$m^3$.d in thermophilic condition among the three different OLRs of 2.5, 3.3 and 3.9 VS/$m^3$.d for constant retention time of 25 days. This happened due to more accumulation of VFAs and decrease in pH during the loading rates 3.3 and 3.9 VS/$m^3$.d.

# ENVIRONMENTAL CONDITIONS WITHIN DIGESTER

## Temperature

Anaerobic digestion can take place at psychrophilic temperatures below 20°C, but most reactors operate at either mesophilic or thermophilic temperatures with optima at 35 and 55°C respectively (Bhattacharya and Mishra, 2003; Chynoweth et al., 2000; Liu et al., 2006) because the biomass activities and anaerobic treatment capacities have been significantly reduced at lower temperature. Compared with mesophilic digestion, thermophilic digestion is a more feasible process for achieving a better performance, especially during the start-up period of a dry anaerobic digestion system (Lu et al., 2007). During batch digestion of vegetable waste, and wood chips, more rapid degradation of fatty acids was found at 55°C than at 38°C (Hegde and Pullammanappallil, 2007). Moreover, the biodegradability and methane yield were greater at 55°C than 35°C (Jha et al., 2010a). The gradual decrease in temperature from 55°C to under 35°C resulted in a decrease in volumetric gas production rate from 2.53 to 0.7 $m^3$ $d^{-1}$ in a 80 $m^3$ batch fermenter treating a mixture (C:N ratio of 16:1) of 25% beef cattle manure and 75% corn stalks with high solids concentration (approximately 27%) for 30 days (Molnar and Bartha, 1988). However, thermophilic condition performed well over mesophilic condition with higher rate of waste conversion (Juanga, 2005) and shorter retention time (Amani et al., 2011); it should be noted that an increase in methane yield from the thermophilic process has to be balanced against the increased energy requirement for maintaining the reactor at the higher temperature. Increased temperatures within a certain range can enhance hydrolysis and accelerate the digestion process, but make the fermentation system less stable in performance while more heat input is needed. Thermophilic bacteria are very sensitive to small temperature-changes and so most of the digesters operate at mesophilic temperatures. Brummeler et al. (1992) has reported that at the start-up, reactor temperature of 20°C was gradually increased to 35°C; it resulted in a prolonged digestion time, while a temperature of 43°C at start-up with a gradual decrease to 30°C gave a similar digestion time as start-up at 35°C.

**pH and buffering capacity**

The pH is the most important and principle operational parameter of anaerobic digestion processes. Variation in pH affects the anaerobic digestion because the hydrogen ion concentration has direct influence on microbial growth. The ideal pH for methanogens ranges from 6.8 to 7.6, and their growth rate will be greatly reduced below pH 6.6 (Mosey and Fernandes, 1989). A pH less than 6.1 or more than 8.3 will cause bad performance, even failure of a digester (Lay et al., 1997). The optimum pH for hydrolysis and acidogenesis is between 5.5 and 6.5 (Arshad et al., 2011). During dry anaerobic digestion of organic solid wastes, Lu et al. (2007) had found that the pH decreased from the original 6.7 to 4.2 under mesophilic conditions and to 5.8 under thermophilic conditions in the first two weeks due to the lower pH buffering alkalinity (3000 mg/l as $CaCO_3$) in the original mixtures. Subsequently, the pH gradually increased to 7.6 after 4 weeks in the 35°C digester and to 8.0 after 3 weeks in the 55°C digester. An excessively alkaline pH is also not favorable for the digestion process because it can lead to crumbling microbial granules and consequent failure of the process (Sandberg and Ahring, 1992). Inhibitors of methanogenesis such as excessive fatty acids, hydrogen sulphide, and ammonia are toxic only in their non-ionized forms (Lay et al., 1997). The relative proportion of the ionized and non-ionized forms and toxicity are pH-dependant. For example, ammonia is toxic when the pH is above 7, while VFAs and hydrogen sulfide are more toxic below pH 7 (Ward et al., 2008). If the ammonia produced can be effectively removed, the dry digestion of sludge can be feasible with a shorter retention time (Naomichi and Yutaka, 2007). Fatma et al. (2009) has shown that chicken manure could be used as a substrate for methane fermentation in a dry state under mesophilic conditions, generating 4.4 L methane gas per kg of chicken manure, despite ammonia at a high level ranging from 8 to 14 g-N·$kg^{-1}$ chicken manure. This clearly demonstrates that spontaneous acclimation of the methanogenic consortia to high levels of ammonia could occur and result in the production of methane even under a high percentage of total solids (25%) and a high level of ammonia (Fatma et al., 2009).

**Volatile fatty acids**

The VFAs uptake might play a crucial role in the whole degradation kinetics of solid organic waste digestion, as the accumulation of the intermediate products, VFAs, is the rate-limiting step (Guendouz et al., 2010). High concentrations VFAs in the digester would lower the pH, inhibit methanogenic activity and cause possible failure of the anaerobic digestion process. Viéitez et al. (2000) demonstrated that fermentative reactions stopped at a VFAs concentration of 13 g/l accompanied by a low pH of 5. The limiting step in anaerobic digestion is hydrolysis, which is usually inhibited by high propionate concentrations (Juanga, 2005). The presence of acetic acid in higher concentrations is not generally treated as inhibitory, while propionic acid is believed to be the most toxic volatile fatty acid appearing in anaerobic digestion, and its oxidation to acetic acid is the slowest among all volatile organic acid transformations (Amani et al., 2011; Hanaki et al., 1994). The development of anaerobic digestion systems for high solids digestion is impeded by the accumulation of VFAs during start-up phase of the process which results in a pH drop in the system. Thermophilic digestion was found to be a faster process as less butyric and propionate acids were accumulated in comparison to a mesophilic process (Lu et al., 2007). As the degradation of propionate and butyrate are regarded as the rate limiting steps, their accumulation can be mitigated using optimum ratios of propionate-degrading bacteria to butyrate-degrading bacteria among the bulk acetogenic microorganisms (Amani et al., 2011).

## FEEDSTOCKS

A direct comparison of potential biogas yields from different substrates is difficult as performance data for specific types are often produced under a wide variety of experimental conditions (for example, mixing regime, temperature, total solids, volatile solids, and retention time) (Ward et al., 2008). Water content in the substrates is essential for the activities of the anaerobes because the methanogenic activity would decrease with a decrease in the moisture content (Lay et al., 1997). To balance the nutrition, the C/N ratio should range up to 20 to 30:1 in the raw material where carbon constitutes the energy source for the microorganisms and nitrogen serves as a critical nutrient for microbial growth. If the amount of nitrogen is inadequate, microbial populations will remain small and it will take longer to decay the available carbon. Surplus nitrogen, beyond the microbial necessity, is often lost from the process as ammonia gas. A dry digestion plant in Japan has maintained stable operations with a mixture of garbage and leftovers from hotels, yard waste, and used paper to control C/N ratio, generating biogas at a rate of about 820 $m^3$ per ton of VS (Naomichi and Yutaka, 2007). During dry (15% TS) anaerobic digestion of an animal manure-switch grass mixture at 55°C, the swine manure had the highest biogas production potential (0.229 $LCH_4$/gVS) with 58% VS removal compared to the dairy (0.09 $LCH_4$/gVS) with 24% VS removal and poultry manure (0.02 $LCH_4$/gVS) with 31% VS removal (Ahn and Smith, 2008). Each gram of dry organic waste from sludge cake, meat, carrot, rice, potato or cabbage had a methane production potential of 450, 424, 269, 214, 203 and 96 ml, respectively (Lay et al., 1997). The methane yields of cooked meat, cellulose, boiled rice, fresh cabbage and mixed food waste, with the same TS of 15 to 30%, were 482, 356, 294, 277 and 472 $mlCH_4$/gVS added respectively, and anaerobic biodegradability based on the stoichiometric methane yield were 0.82, 0.92, 0.72, 0.73 and 0.86, respectively (Cho et al., 1995). Relatively-small particle sizes enlarge the total surface area of the particles and so improve microbial activities. The initial substrate concentration influenced the biogas production as more methane yield was observed in the substrate of 20% TS than 30% TS on dry mesophilic anaerobic digestion of municipal waste (Fernández et al., 2008).

## OPTIMIZATION TECHNIQUES

### Inoculation

Fresh feedstock is inoculated using digestate or leachate to speed up the reaction processes. The anaerobic digestion of MSW will require inoculation of solid waste to speed up the process instead of allowing the process to depend on self-generation and subsequent regeneration (Nwabanne et al., 2009). Inoculation will also enhance reactor digestion efficiency and diminish solid retention time. The addition of an appropriate inoculum ratio is favourable during start-up (Brummeler and Koster, 1989a). Sans et al. (1994) found that the effective percentage of inoculation for the fermentation of organic urban wastes in a plug-flow reactor was approximately 30% (w/w). Based on the dry anaerobic digestion of the separated organic fraction from municipal solid waste in pilot-plant-scale reactors, Brummeler et al. (1992) reported the optimum ratio of inoculum solid and the initial total solid at start-up was 0.5 to 0.6 with a SRT of 30 days. Longer retention times were observed at lower inoculum ratio due to a suboptimal leachate recycle flow rate. When the inoculum ratio decreased to 0.4 or lower, the SRT increased to 50 days or longer due to the relatively long period of suboptimal conditions, such as low pH and high organic acid concentrations. With the dry fermentation of rice straw, additions of inoculum and fresh pig manure can result in relatively high total biogas yield, biogas production rate, and degradation rate of raw materials (Sun et al., 1987). As inoculum sources, corn silage, restaurant waste mixed with rice hulls, cattle excrement, swine excrement, digested sludge and swine excrement mixed with sludge (1:1) were tested and evaluated for their effect on anaerobic thermophilic digestion of the separately collected organic fraction of municipal solid wastes at dry conditions (30% TS) (Forster-Carneiro et al., 2007). The results indicate that digested sludge was the best inoculum source and, with an inoculum of 25%, 44% COD removal and 43% VS removal was achieved in the digester over 60 days operating period. In the stabilization phase, the sludge-inoculated reactor gave a methane yield of 0.53 $LCH_4$/gVS. Swine waste mixed with digested sludge was also good inocula at these experimental conditions.

### Co-digestion

Co-digestion involves mixing of different types of feed stocks before digestion to control the C/N ratio. The process benefits of co-digestion are an improved nutrient balance, decreased effect of toxic compounds on the digestion process, or improved rheological qualities of the substrate. Co-digestion improves the methane yield because of the bacterial diversities in different wastes, the positive synergisms established in the digestion medium, and the supply of missing nutrients by the co-substrates. The use of a co-substrate can also help to establish the required moisture contents of the digester feed. A dry anaerobic digestion plant in Japan treating a mixture of garbage and leftovers from hotels, yard waste, and used paper is an example of co-digestion (Naomichi and Yutaka, 2007). Li et al. (2011a) observed that the mixing of cow manure with solid sludge increases the biogas production by 3 to 14%. The co-digestion of the organic fraction of municipal solid waste and cotton gin waste with cow manure enhances the digestion of fiber in cotton gin and the municipal waste (Maritza et al., 2008). The use of poultry manure at high-solid-content for anaerobic fermentation is logical, but the high nitrogen content of the manure can caused ammonia toxicity in the treatment process, resulting in a reduced co-digestion of the manure when mixed with corn stover (Jantrania and White, 1985). According to Liu et al. (2009), increasing the manure-crop ratio during dry anaerobic co-digestion, only promoted digestion efficiency by improving the digestion kinetics, but not the biogas production potential.

**Pretreatment, percolation and additives**

One of the main objectives of pretreatment methods is to increase solubility of the substrate and to accelerate the hydrolysis process. Pretreatment normally includes (1) physical separation of the organic fraction from inorganic materials; (2) reduction of particle size; (3) the addition of inoculants, leachates or additives into the feedstock; (4) treatment of the substrates with acid, alkali, ultrasonic or thermal energy or their combination before digestion. The methane yield and solid reduction were found greater in pre-composting of pulp-mill sludge than in the digestion of the untreated sludge (Capela et al., 1999). Juannga (2005) has reported that reduced substrate particle size of 30 mm offer degradation benefits over 60 mm. Yan et al. (2010) observed that the mean particle size of waste activated sludge deceased from 25.0 to 3.2 μm and its disintegration degree increased to 60.8% in the ultrasonic specific energy range of 0 to 90000 kJ/kg DS. The pretreatment with NaOH can enhance the biodegradability of rice straw because of degradation of cellulose, hemicellulose and lignin, resulting in an increase in biogas yield by 27.3 to 64.5% (He et al., 2008). When pretreated with 5.0% NaOH, the solid state anaerobic digestion of corn stover can also produce 37% more biogas than that of the untreated (Zhu et al., 2010). Pretreatment of sorted substrates with an optimal ratio (20%) of inoculants or leachate can increase biogas yield and volatile solids reduction (Sigrid et al., 2008). Bolzonella et al. (2000) presented a comparison of two dry anaerobic digestion reactors fed with differently sorted municipal organic solid wastes at full scale. Utilizing the same process (Valorga) and operational conditions, one reactor was fed with source sorted organic wastes while the other reactor was fed with mixed organic wastes consisting of grey wastes, mechanically selected municipal solid wastes and sludge. The results indicate that the reactor treating the source sorted organic waste and the reactor treating the mixed organic wastes generated some 200 and 60 $m^3$ of biogas per ton of treated waste, respectively while the specific methane production was some 0.40 and 0.13 $m^3$ $CH_4$/kgTVS, respectively. The biodegradability of water sorted organic fraction of municipal solid waste (OFMSW) with a VS/TS ratio of 61.6% was better than that of mechanically sorted OFMSW, but still poorer than that of source sorted OFMSW (Li et al., 2010). Forster-Carneiro et al. (2008) reported that the source sorted OFMSW exhibited the classical waste decomposition pattern with a fast start up phase and a subsequent stabilization phase, while the mechanically sorted OFMSW showed a methanogenic pattern throughout the whole experimental period (60 days) and this gave higher levels of organic biodegradation (56% VS) and biogas production. The degradation process can be enhanced using aerobic pretreatment. Brummeler and Koster (1989b) found that it was in balance with the methane formation from the organic fraction of municipal solid waste of TS 35% when 19.5% of the VS were converted during the aerobic composting period before acid formation in the digestion, but 40% loss in potential biogas occurred. The loss of a part of the biogas is a major drawback of partial composting as a method for enhancing the start-up of the dry anaerobic digestion. A shorter composting period which is combined with another start-up method might be a feasible method to decrease the energy input of the dry digestion process.

In percolation process, liquid is re-circulated and sprinkled over the stacked material in order to initiate biogas production and encourage bacteriological activity in the decomposing biomass throughout the process. Leachate recirculation for enhancing biogas production seems quite viable since stirring of the digester's contents in dry reactors is difficult. Percolation systems can be utilized to increase the digestion rate (Brummeler et al., 1992); reduce amount of inoculum needed due to reintroduction of the washed-away microbes back into the reactor (Yadvika et al., 2004) and to enable the colonization of the bacteria throughout the digester by providing an active transport mechanism for microbial communities (Li et al., 2010). Chugh et al. (1999) studied the anaerobic digestion of municipal solid waste and showed that very fast digestion rates can be achieved with proper leachate circulation. Leachate recycling has been found favourable for anaerobic decomposition (Chan et al., 2002; Juanga, 2005) as the addition of water or leachate recycling can encourage bacteriological activity, leading to faster degradation kinetics. Kanwar and Guleri (1994) found that about 60 to 65% more biogas production can be obtained by simply recycling the digested slurry in 1 $m^3$ plug flow type pilot plants. Similarly, solid-state fermentation of the MSW with effluent recirculation resulted in rapid hydrolysis, acidification and denitrification, with 70 to 85% methane-content in the biogas, yielded from the methanogenic reactor (Viéitez et al., 2000). As liquid can be extracted from the effluent and recycled to the reactor, considerable heating energy may be saved, which is crucial to the efficient biogasification of dry substrates (Legrand and Jewell, 1987).

Nutritional deficiencies may result in an incomplete unstable bioconversion of the organic substrates and may ultimately cause digester failure. Proper additives and micronutrients can enhance the production rate of biogas and performance of the reactor, and increase the speed of start up by stimulating the microbial activities in the digester (Radwan et al., 1993). Yadvika et al. (2004) have emphasized the use of inorganic and organic additives for enhancing biogas production while Uemura (2010) observed that the addition of minerals (Ni, Co, and Fe) in the batch mesophilic reactor treating organic fraction of municipal solid waste improved the digester performance and biogas yield. Increase in biogas production by biological additives, including different plants, weeds, crop residues and microbial cultures appears to be due to adsorption of the substrate on the surface of the additives, which can lead to high-localized substrate concentration and a more favourable environment for growth of microbes. The additives also help to maintain favourable conditions such as pH, inhibition/promotion of acetogenesis and methanogenesis inside the digester for rapid gas production. However, their additional cost must always be balanced against resultant improvements in efficiency.

## DISCUSSION

### Feasibility of dry anaerobic digestion process

The organic wastes is required to be managed in a sustainable way to avoid depletion of natural resources, minimize risk to human health, reduce environmental burdens and maintain an overall balance in the ecosystem. Anaerobic digestion is one of the most effective biological processes to treat a wide variety of solid organic wastes. Anaerobic digestion scores much better than other waste treatment options in the terms of energy, cost and ecological balance (Mata-Alvarez et al., 2000). It provides both green energy (biogas) and bio-fertilizer. The bio-fertilizer enriches soil with no detrimental effects on the environment (Iyagba et al., 2009; Uzodinma et al., 2008). Furthermore, during the anaerobic digestion process, the worms initially in the raw droppings died off and are undetected in the treated slurry (Yongabi et al., 2009). Thus, the increasing volumes of solid organic waste become a valuable commodity when being viewed as an energy resource that is managed properly through anaerobic digestion (Jha et al., 2010b).

During the 1990s, dry digestion prevailed over wet digestion, and several commercialized dry digestion systems, for example, DRANCO (Six and Debare, 1992), KOMPOGAS (Willinger et al., 1993), and VALORGA (Laclos et al., 1997), were developed to treat solid organic wastes. Karagiannidis and Perkoulidis (2009) evaluated five different anaerobic digestion technologies in the terms of energy yield, material recovery, operating cost and $CO_2$ emissions and found that DRANCO ranked the best position due to low cost and high energy recovery. The DRANCO was followed by WASSA, VALORGA, KOMPOGAS, and BTA. The BTA, which is a low solid system, ranked at the worst position due to its high cost and lower organic loading rate.

### Perspectives and conclusions

The future of dry anaerobic digestion process should be sought in the context of energy recovery and sustainable waste-management perspective. This technology has tremendous application in the future for sustainability of both environment and agriculture because it represents a feasible and effective waste-stabilization method to convert the undiluted solid bio-waste into renewable energy with nutrient rich organic fertilizer. Although, it is seldom used in practice, it would be a promising excellent sustainable alternative to the conventional waste treatment processes to solve today's energy and environmental challenges.

Batch and plug flow digesters have been proven to be feasible for anaerobic solid-state fermentation and two stage digesters have better performance efficiency than single phase reactors. Thermophilic anaerobic digestion can yield more biogas in short retention times and has better start-up performance than mesophilic digestion, but the increase in methane yield has to be balanced against the increased energy requirement for maintaining the reactor at the higher temperature. Co-digestion, pretreatment, percolation and additives are recognized ways to improve biogas yield. As mechanical mixing is hard to implement in dry anaerobic digestion systems, recirculation of leachate and/ or biogas may be one of the solution. It is important to monitor parameters like pH, temperature, buffering capacity or fatty acid levels to ensure optimal efficiency and maximize gas yield in short retention time. The commercial dry anaerobic systems (DRANCO, VALORGA, or KOMPOGAS) assist to prove the capability of this process to effectively convert waste material into energy.

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