Saved documents

Tuesday, May 14, 2019 at 3:56 a.m.

1 document

By BUTC-SCD - Bienvenue sur Europresse

Summary

Saved documents • 1 document

Environmental Reviews

Source name
Environmental Reviews
Source type
Press • Online Press
Periodicity
Bimonthly or Quarterly
Geographical coverage
International
Origin
Ottawa, Ontario, Canada

Thursday, December 1, 2016

Environmental Reviews • no. Vol. 24, no 4 • p. 426(34) • 42772 words

Anaerobic digestion for the stabilization of the organic fraction of municipal solid waste: A review.

dx.doi.org/10.1139/er-2016-0077

Abstract: Anaerobic digestion (AD) of the organic fraction of municipal solid waste (OFMSW) has gained great interest in the last few decades. Presently, among the various municipal solid waste management (MSWM) techniques, such as sanitary land-filling, aerobic or anaerobic composting, and thermal incineration, AD is being considered as the most techno-economically viable method owing to its many advantages. AD not only serves to efficiently manage and treat an enormous quantity of OFMSW but also acts as a convenient source of non-conventional energy. The methane generated via AD of OFMSW serves as a potent substitute for fossil-based fuels. Given the current global energy crisis, this technology may be a welcome boost to the global energy demand. In this review an attempt has been made to provide a comprehensive understanding of: (i) the origin and scope of AD of OFMSW; (ii) the potential of AD for OFMSW stabilization; (iii) various pilot and bench-scale studies conducted hitherto; (iv) the process design aspects of AD of OFMSW; (v) the potential of energy recovery from AD of OFMSW; and (vi) the past experiences of AD of OFMSW. This review also delves into the critical issues that govern the process of AD in stabilizing OFMSW. In addition, the compatibility of AD for MSWM in the Indian scenario compared to other classical methodologies, such as landfilling, composting, thermal incineration, and pyrolysis or gasification, is highlighted. An overview of the overall future prospect of AD of OFMSW is discussed.

Key words: anaerobic digestion, OFMSW, co-digestion, process design, potential, critical issues in AD.

Resume : La digestion anaerobie (DA) de la fraction organique des dechets solides urbains (FODSU) a suscite beaucoup d'interet au cours des dernieres decennies. Actuellement, parmi les diverses techniques de la gestion des dechets solides urbains (GDSU), telles que l'enfouissement sanitaire, le compostage aerobie et anaerobie et l'incineration thermique, la DA est consideree comme la methode la plus techno-economiquement viable en raison de ses nombreux avantages. La DA ne sert pas seulement a gerer efficacement et a traiter une enorme quantite de FODSU, mais elle est une source avantageuse d'energie non conventionnelle. Le methane produit par la DA de la FODSU sert comme un important produit de substitution aux combustibles fossiles. etant donne la crise energetique mondiale actuelle, cette technologie peut apporter une contribution opportune a resoudre le probleme de la demande energetique mondiale. Dans le cadre de cette revue, on a tente d'apporter une comprehension

complete : (i) de l'origine et de l'envergure de la DA de la FODSU; (ii) du role potentiel de la DA dans la stabilisation de la FODSU; (iii) des diverses etudes pilotes et a l'echelle de banc d'essai menees jusqu'ici; (iv) des aspects de la conception de procede de la DA de la FODSU; (v) du potentiel de recuperation d'energie decoulant de la DA de la FODSU; (vi) et des experiences passees de la DA de la FODSU. Cette revue explore aussi les enjeux critiques qui dominent le procede de la DA dans la stabilisation de la FODSU. De plus, on fait ressortir la compatibilite de la DA en matiere de la GDSU pour le scenario indien comparee a d'autres methodologies traditionnelles telles que l'enfouissement, le compostage, l'incineration thermique et la pyrolyse ou gazeification. On considere une vue d'ensemble de la perspective future de la DA de la FODSU. [Traduit par la Redaction]

Mots-cles : digestion anaerobie (DA), fraction organique des dechets solides urbains (FODSU), codigestion, conception de procede, potentiel, enjeux critiques en matiere da la DA

1. Introduction

Over the past few decades, many developing countries all over the world have experienced a sharp increase in population. This uncontrolled population growth coupled with rapid urbanization and gradual rise in the per capita income have led to an unprecedented increase in the annual generation of municipal solid waste (MSW). MSW necessarily comprises market refuse, food waste (FW), kitchen waste, garden waste, dead animals, etc. It was reported by Kawai and Tasaki (2016) that the average MSW generation rate in the Organization for Economic Cooperation and Development (OECD), the European Union (EU), and the United States of America (USA) was 1.324 kg/person per day. In seven South Asian countries (Maldives, Sri-Lanka, Pakistan, Bhutan, Bangladesh, India, and Nepal) the average MSW generation rate has been found to be 0.853 kg/person per day with the highest generation of 2.8 kg/person per day in Maldives (Peterson 2013). Similarly, in 15 countries of East Asia and the Pacific Islands (Solomon Islands, Tonga, Mongolia, Myanmar, China, Philippines, Vietnam, Indonesia, Lao PDR, Thailand, Malaysia, Singapore, Brunei Darussalam, Fiji, and Vanuatu) the average per capita solid waste generation rate has been found to be 1.288 kg/person per day with the highest generation of 4.30 kg/person per day in Solomon Islands (Hoornweg and Bhada-Tata 2012). The present quantity of solid waste generation worldwide is 2 billion tons per year and is predicted to increase to 3 billion tons by 2025 (Key Note Publications Ltd. 2007). By 2100 the MSW generation rate is expected to exceed 11 million tons per day (World Bank 2013). Because of lack of infrastructure, suitable strategies, man power, and economic constraints the municipal bodies in most cities of developing countries have been unable to come up with an effective and sustainable methodology for collection and disposal of MSW. As a result, these solid wastes are either dumped out into open spaces, in low lying areas, causing all sorts of nuisance and health hazards, or are disposed of into primitive sanitary landfills in the outskirts of cities. In a few cases, the generated MSW is also subjected to composting or thermal incineration.

The landfilling operation of organic fraction of municipal solid wastes (OFMSW) is a common and widely practiced mode of solid waste disposal worldwide. But, with the decreasing availability of land, this method is now under a great challenge to manage the requisite space. It is also worthy to mention that a poorly managed sanitary landfill for solid waste may lead to many problems, such as uncontrolled emission of greenhouse gases (GHG) into the atmosphere evolving a catastrophic scenario, generation of leachate contaminating the ground water and generation of disease vectors adversely affecting the public health and hygiene. Composting, on the other hand, is a process that also requires considerable land akin to sanitary landfilling. But the composting facilitates the treatment of only a few types of the organic waste generated on a daily basis. Applications of composting include: (i) yard waste, (ii) separated MSW, (iii) commingled MSW, etc. (Tchobanoglous et al. 1993). The product obtained after composting of the OFMSW is generally used up as a soil conditioner, whereas landfilling operations occasionally generate biogas and also enhance soil fertility. Hence, the scope of marketing the products after the solid organic waste is subjected to composting is very much limited as it is a localized process. However, both composting and sanitary landfilling are not able to strike a balance between the generated and stabilized quanta of waste. Thermal incineration, which involves considerable volume reduction of the generated MSW, has various drawbacks, such as the requirement of an external supply of oxygen, a well-equipped burning system, and other appropriate pollution control accessories. Thus it can be inferred that despite all the benefits obtained from the aforesaid processes, the involvement of an exorbitant cost paradigm and an imbalance in the generation and stabilization rate of the MSW has necessitated the municipalities to look for other techno-economically viable solutions. Anaerobic digestion (AD) of OFMSW has proved to be the most effective and sound methodology for replacement in this regard (Mata-Alvarez et al. 1990, 2000).

AD of OFMSW is a biochemical process where solid organic matter is converted to biogas and highly concentrated sludge through a series of reactions mediated by several micro-organisms. AD of OFMSW involves the stabilization of solid organic matter via three major phases, namely, the semi-anaerobic hydrolysis phase, the semi-anaerobic acidogenesis phase, and the strictly anaerobic methanogenesis phase (Gerardi 2003) to produce C[H.sub.4] and C[O.sub.2] along with a small fraction of other gases (such as [H.sub.2], N[H.sub.3], [H.sub.2]S, [N.sub.2], etc.). AD mainly focuses on the volumetric reduction of the large initial amount of the solid organic fraction to liquid state (via hydrolysis). Thereafter, different bacteria convert that liquid into volatile fatty acids (VFAs) and alcohols (via acidogenesis) and in the final phase, those acids and alcohols get converted to gases, primarily C[O.sub.2] and C[H.sub.4] (via methanogenesis).

However, there is an additional and equally important acetogenesis phase (Ljungdahl and Wood 1969; Deublein and Steinhauser 2008; Khanal 2008) preceding the methanogenesis phase, which also plays a crucial role in the AD. The fermentative bacteria firstly degrade polysaccharides to monosaccharides or glucose via hydrolysis, which is then converted into fatty acids, alcohols, and [H.sub.2] plus C[O.sub.2] via acidogenesis. The acetogens, however, convert the higher fatty acids into acetate and [H.sub.2] plus C[O.sub.2]. Homoacetogens, on the other hand, convert [H.sub.2] plus C[O.sub.2] into acetate. Thus the fermentative bacteria and the syntrophic acetogen produce [H.sub.2], C[O.sub.2], acetate, and other products, while the homoacetogen utilize the [H.sub.2] plus C[O.sub.2] to produce acetate (Nie et al. 2008). In an earlier study by the same team (Nie et al. 2007) it was established that the property of homoacetogens to convert [H.sub.2] plus C[O.sub.2] into acetate was a viable strategy to enhance the production of acetate by collecting and converting [H.sub.2] plus C[O.sub.2] into acetate via homoace-togenesis.

These homoacetogenic bacteria, which are often known as acetogenic bacteria, are anaerobic microbes, most of whom catalyze the formation of acetate from [H.sub.2] plus C[O.sub.2] in their energy metabolism (Diekert and Wohlfarth 1994). These anaerobes can use the Wood-Ljungdahl pathway to: (i) synthesize acetyl-CoA by the reduction of CO or C[O.sub.2] plus [H.sub.2]; (ii) conserve energy; and (iii) fix (assimilate) C[O.sub.2] for the synthesis of cell carbon (Drake et al. 2008). The first isolated acetogen was Clostridium aceticum, which was spore-forming and mesophilic in nature (Wieringa 1939). Later on, the bacterium could not be isolated anymore and further investigations led to the isolation of the second acetogen, which was Clostridium ihermoaceticum (spore-forming, thermophilic) (Fontaine et al. 1942). This bacterium, later re-classified as Moorella thermoacetica (Collins et al. 1994), was the model organism for the elucidation of the biochemical and enzymological features of the acetyl-CoA pathway by Harland G. Wood and Lars G. Ljungdahl, which was dubbed the Wood-Ljungdahl pathway (Ljungdahl and Wood 1969). The acetogens are able to grow chemolithoautotrophically under anaerobic conditions converting CO or [H.sub.2] plus C[O.sub.2] as sole carbon sources primarily into acetate. CO and (or) C[O.sub.2] are the substrates for the two branches, namely, the methyl and the carbonyl branches, of the Wood-Ljungdahl pathway. Acetyl-CoA is the main intermediate of the pathway and it serves as a precursor for the anabolism and catabolism of the respective microbes (Schiel-Bengelsdorf and Durre 2012).

Unlike other types of conventional waste matter, the OFMSW need to be shredded and homogenized into fine particles before treatment. If the initial amount of moisture present in the waste, which usually depends on the nature of the waste, is not substantial, water should be externally added to the shredded and homogenized material to promote hydrolysis and the subsequent biochemical reactions. The biochemical pathway for degradation of the solid organic wastes via AD is shown in Fig. 1. The important factors influencing this pathway are temperature, pressure, particle size, moisture content, total solid (TS) content, etc.

The several benefits that can be obtained through AD include reduction of a significant amount of OFMSW that is generated daily in a balancing manner, generation of substantial amounts of biogas and the generation of bio-sludge, which can be an effective soil conditioner. The generated biogas primarily comprises carbon dioxide (30%-40%) and methane (60%-70%) along with a small fraction of other gases, of which the produced methane gas can be utilized as a source of heat for electricity generation and as fuel for automobiles. Other advantages of AD include requirement of small space for treatment, prevention of spreading of disease vectors and minimization of ground water contamination, which is a problem especially associated with sanitary landfilling and composting. The AD of OFMSW produces biogas, which has a comparatively lower percentage of carbon dioxide (30%-40%) and higher percentage of methane (60%-70%). The methane produced can be utilized in various ways and thus significantly contributes to the global scheme of renewable energy; whereas the carbon dioxide produced does not have such wide applications. Moreover, carbon dioxide is the principal contributor to the greenhouse effect. As a result, AD of OFMSW, unlike classical MSW management practices, does not contribute to the greenhouse effect. In this review the authors primarily focus on the: (i) importance and potential of AD of OFMSW, (ii) various operational aspects of AD of OFMSW, (iii) various case studies pertaining to pilot and field-scale application and also full-scale implementation in large towns and cities, (iii) kinetic models that have been developed hitherto, (v) the different approaches for process design, and (vi) different modalities for utilization of the produced energy at the community and industrial level.

2. Origin of AD of OFMSW

Interestingly, OFMSW was first observed to undergo AD in sanitary landfills and this was detected essentially at a very high solids concentration (Farquhar and Rovers 1973; Rees 1980). This result prompted many scientists and researchers to come up with the idea of an effective methodology, which would not only serve as a potent substitute for landfilling operations, but would also be economical with regard to capital investment and generated sludge. As such, AD of OFMSW came into the thrust area of research and gradually it was developed. The development was brought about by various experimental studies on AD of OFMSW. These were conducted over the last three decades and led to the invention of a wide range of modifications on AD of OFMSW.

After the discovery of AD of OFMSW in sanitary landfills, in the early 1970s, very little development was observed in the next four to five years. However, since the early 1980s, the AD of OFMSW received greater attention with the introduction of both pilot and commercial AD plant designs. In the past decade, AD of OFMSW has found a wide application across Europe with the development of many full-scale plants (Mata-Alvarez et al. 2000). The present statistics indicate that 1.3 billion metric tons of MSW are generated annually in the world, which is expected to rise to about 2.2 billion tons by 2025 (Hoornweg and Bhada-Tata 2012). Earlier, it was reported by Bolzonella et al. (2003) that nearly 1 million tons of organic wastes (wet weight) were being digested throughout the world. AD of the solid organic wastes not only ensures stabilization of these enormous quantities of waste, but also converts them into biogas (Verstraete et al. 2000). Moreover, AD when coupled with post-composting (Kubier and Rumphorst 1999; Verstraete et al. 2000) also minimizes the emission of carbon dioxide into the atmosphere thereby playing a significant role in the abatement of global warming.

3. Innovative developments on AD of OFMSW

Chynoweth et al. (1992) and Chugh et al. (1995) proposed a leachate management strategy that ensured: (i) the microorganisms that were involved in the AD were able to get sufficient moisture and nutrients required for rapid conversion of the OFMSW; and (ii) the removal of inhibitory fermentative products during startup. The reason behind developing this leachate management strategy was that there was a lack of stability in reactors that were loaded with high solids-content during long start-up periods. The multistage leachate recycle configuration proposed by them overcame this problem. In a separate study, Nopharatana et al. (1998) designed an arrangement consisting of two-stage digesters where recirculation of leachate into the fresh organic matter was done. It was observed that formate degradation activity followed the methane production rate with both reaching a maximum value simultaneously at a certain time. Hence, it was observed that the recirculation of leachate offered better optimization of the AD of OFMSW, with regards to the energy yield and process economics (Di Maria et al. 2012).

Since the late 20th century, AD of OFMSW has improved by leaps and bounds with commercial and pilot AD plants coming into the fore. As such, Rodriguez-Iglesias et al. (1998) studied the pilot scale AD of OFMSW and achieved a maximum methane yield of 66% of the total biogas produced. Earlier, Kayhanian and Rich (1995) conducted a pilot scale study on high solids thermophilic AD of OFMSW with an emphasis on nutrient requirements and concluded that addition of micro- and macronutrients to OFMSW prior to digestion resulted in stable operation and elevated gas production. The robustness of the methanogens depends largely on a variety of mineral nutrients because of their diverse population dynamics. Various activities of the methanogens, such as protein synthesis, nucleic acid synthesis, increasing cell wall permeability, and enhanced metabolism, depend on the presence of various macro- and micronutrients, such as carbon, nitrogen, phosphorus, potassium, sulfur, and iron. The heterogeneous nature of the biodegradable organic fraction of municipal solid waste (BOFMSW) makes it imperative for nutrient supplementation during the addition of the synthetic feedstock so that there is an increased stabilization of the digester performance treating BOFMSW in a high solids AD.

Dry digestion or high solids AD systems are more robust and flexible as opposed to wet digestion systems because of their higher biomass concentration and controlled feeding. It was observed that, compared to food waste (FW) and shredded organic fraction of municipal solid waste (SH-OFMSW), OFMSW had the highest methane yield per gram of VS degraded, which was also a maximum along with dissolved organic carbon (DOC) removal. Forster-Carneiro et al. (2008a) stressed the importance of the dry digestion of FW and conducted AD for the stabilization of three types of organic substrates--FW, SH-OFMSW, and OFMSW. Hence, it was once again inferred that the nature of the organic substrate had an important role to play on the biodegradation process and methane yield. A kinetic model was developed by Bollon et al. (2011), which specifically assessed the degradation of the OFMSW in the dry AD.

The phenomena of inhibition arising out of high solids anaerobic digestion (HSAD) of OFMSW were addressed by Schievano et al. (2010) when the same process was applied to the highly putrescent OFMSW. Later on, Yu et al. (2012) established a comprehensive model to configure a new two-stage HSAD system designed for highly degradatile OFMSW. Earlier in another study, Fdez.-Guelfo et al. (2010) focused on the start-up and stabilization of an AD system at thermophilic temperature range (55[degrees]C) and dry conditions (30% TS) and semi-continuously fed for the treatment of OFMSW. The focusing on the start-up and stabilization phase can be linked with the adaption of the inoculum employed, which in the aforesaid study was a mixture of leachate and sewage sludge in 1:1 v/v.

Temperature, in both the mesophilic (30-38 [degrees]C) and the thermophilic (49-57 [degrees]C) range, plays a crucial role in the AD of OFMSW with regards to high methane production. Earlier AD of OFMSW was done at mesophilic temperatures (Cecchi et al. 1993), but after successful demonstrations at thermophilic temperatures, both in laboratory-scale (Wellinger et al. 1992) and full-scale levels (Cozzolino et al. 1992), the thermophilic anaerobic process gained significant interest because of its higher loading rate and greater volume of gas production (Cecchi et al. 1991). The sequential development of AD of OFMSW, considering the operating temperature, is shown in Fig. 2. The selection of dry conditions at thermophilic temperature range results in the process being much faster, with a much cleaner product obtained, than when mesophilic or wet conditions are used. Besides, the hydrolysis stage on the complex organic or biological material is better and the C[H.sub.4] generation is higher in the case of the thermophilic temperature regime than it is under mesophilic and wet conditions. The effect of change in temperature regime from mesophilic to thermophilic on the start-up phase of AD of OFMSW has also been well demonstrated.

In a study by Sasaki et al. (2011), the methanogenic pathway and the microbial community in a thermophilic AD of organic solid waste in a continuous-flow stirred-tank reactor using artificial garbage slurry as a feedstock was investigated. The study implied that the microbial community in the thermophilic degrading process consisted exclusively of unidentified bacteria, which efficiently removed acetate through a non-acetoclastic oxidative pathway thereby increasing the methane yield. In another study, a comparison among the most common types of anaerobic digesters, such as that of the single-stage, the two-stage, and the batch systems, was made by Vandevivere et al. (2002). The starting stages of AD are significantly influenced by two most important factors--inocula (Castillo et al. 2006) and TS content (Li et al. 2009); obviously pressure also plays a crucial role in the generation of biogas and stabilization. In high-altitude areas, where the atmospheric pressure is relatively low, the AD process would have to proceed under relatively low partial pressure of C[O.sub.2] resulting in a higher pH environment. This would enable the AD system to resist acidification and achieve a higher organic loading rate (OLR) (Zhang et al. 2005). Jiang et al. (2010) examined the influence of atmospheric pressure on the performance of AD of OFMSW, using a self-designed AD experimental system that simulated different pressures.

Hitherto, the enzyme activities in AD of real organic wastes have been paid very little attention because the enzymes are too scarce to determine selectively and the enzymatic reactions are too specific and diverse to cover various enzymes involved in overall pathways. However, Kim et al. (2012) enhanced the understanding of the enzyme activities of anaerobic microorganisms in an anaerobic digester treating real OFMSW by quantitatively determining the representative enzyme activities of extracellular cell-free liquids.

Mixing has a tremendous impact on the start-up of mesophilic co-digestion systems and thermophilic digesters fed with acetate. This was brought to light by overcoming the operational difficulties and instability problems encountered during thermophilic AD of OFMSW by Angelidaki et al. (2006). Besides, Ghanimeh et al. (2012) assessed the effect of mixing on the performance of thermophilic AD treating source-sorted organic fraction of municipal solid waste (SS-OFMSW) during the start-up phase, using cattle manure as a seed source. Recently, small-scale decentralized anaerobic systems have become attractive and easy to manipulate with the feedstock characteristics. Zeshan et al. (2012) studied the effect of the carbon to nitrogen (C/N) ratio and associated ammonia-nitrogen accumulation in a pilot-scale dry thermophilic AD system designed for decentralized applications. It was concluded that the problem associated with ammonia-nitrogen accumulation can be easily overcome by adjusting the feedstock C/N ratio in case of a decentralized small-scale AD system as opposed to a large-scale centralized system.

It is a well-known fact that incineration of MSW has the potential to produce steam and electricity besides producing residues, such as fly ash (FA) and bottom ash (BA). Lo et al. (2012) investigated the addition of various MSW incineration ashes to the AD of OFMSW. It was observed from the study that, an solids retention time (SRT) of 20 days resulted in an increased biogas production rate compared to an SRT of 5 and 10 days for a given organic load of both FA and BA. However, it was also seen that at an SRT of 20 days the reactor dosed with BA showed better biogas production rate compared to the reactor dosed with FA. The significance of a higher SRT was emphasized along with the increased efficiency of BA compared to FA. The heterogeneous nature of OFMSW necessitates the requirement of specific studies for optimizing the dry mesophilic AD of various OFMSW. The heterogeneous nature of OFMSW warrants optimization of the AD process so that the different types of OFMSW may be treated in a specific manner with regards to certain critical parameters, such as SRT and OLR, which are two of the most important operational parameters in AD. Rodriguez et al. (2012) determined the optimum SRT and the relevant OLR for the mesophilic AD of OFMSW. It was observed that, at an SRT of 20 days the methane productivity rate and the organic matter removal rate were higher than those observed at SRTs of 15 and 30 days.

So far as energy is concerned, hydrogen can also be considered as one of the clean alternative fuels produced from AD of OFMSW, along with methane. Hydrogen is one of the most promising energy sources for the future because it is more environmental friendly than methane and has the ability to be used in fuel cells in the transportation infrastructure. As such, when blended with compressed natural gas as a fuel for vehicles hydrogen has been found to reduce [NO.sub.x] emission by as much as 50% (DiStefano and Palomar 2010). Also this fuel may be assigned the role of replacing the fossil fuels in the near future thereby highlighting the need for "hydrogen economy" (Das and Veziroglu 2001). Romero Aguilar et al. (2013) analyzed the effect of hydraulic retention time (HRT) on the hydrogen production from the OFMSW coming from a full-scale mechanical-biological-treatment (MBT) plant using an anaerobic continuous stirred tank reactor (CSTR) operated at thermophilic-dry conditions (55 [degrees]C and 20% in TS concentration, respectively). It was observed that the maximum hydrogen and specific hydrogen were produced at an HRT of 1.9 days with a feeding regime of twice a day. In another study, conducted by Kim and Kim (2013), a three-stage fermentation system was developed that achieved stabilization of FW to [H.sub.2] and C[H.sub.4]. The system especially emphasized achieving higher [H.sub.2] yield than the mere 20% recovery of [H.sub.2] in most AD reactors stabilizing OFMSW. Fig. 3 represents a typical flow sheet of AD of OFMSW up to biogas generation and sludge recovery.

4. Role of various factors influencing AD of OFMSW

The performance of AD in stabilizing OFMSW depends on a wide variety of factors, of which some important ones are listed in the following subsections.

4.1. Role of pH

pH has always been a crucial parameter

in the generation of the biogas by AD of OFMSW. Results have shown that a pH range of 6.8-7.3 is the most favorable for methane-producing bacteria. This can be attributed to the fact that on either side of the mentioned pH range methane production ceases, as acid-forming bacteria are dominating at lower pH (4-5), whereas at a higher pH (>7.3) the production of the toxic form of ammonia (N[H.sub.3]) is favored (El-Fadel et al. 2013). Addition of inoculum and dry digestion at higher seed or substrate solids ratios, which is essential at times, result in these varying unfavorable ranges of pH. During instances when the pH is not favoring the methane-producing bacteria, several pH control chemicals (buffers) can be added to the AD system to restore pH neutrality. Brummeler and Koster (1989) studied the effect of several pH control chemicals on the dry batch digestion of the OFMSW and detected higher stabilization in terms of methane generation compared to that in sanitary landfills. It was also concluded that a start-up phase of approximately 6 months yielded 80 L C[H.sub.4]/kg of organic fraction. Mata-Alvarez et al. (1990) made a variety of comparisons in digester performances treating OFMSW pertaining to the biodegradation achieved, process kinetics, and biodegradability of the substrate. All these factors are related directly to the nature of the feed provided, which in turn depends largely on the way the waste is sorted.

As such, the biodegradation achieved in the case of the SS-OFMSW is greater than the mechanically sorted organic fraction of municipal solid waste (MS-OFMSW). MS-OFMSW contains a higher fraction of non-biodegradable solids, such as plastic pieces, wood, paper, etc., resulting in a lower percentage of soluble VS with respect to total volatile solids (TVS), when compared to SS-OFMSW. This in turn causes low biogas production rate. Consequently, it was observed by Mata-Alvarez et al. (1990) that the biodegradation rate, estimated through the first-order kinetic constant k, was almost 10 times greater when the feed was SS-OFMSW. Later, in a separate experimental study where the AD of food market wastes (Mata-Alvarez et al. 1992) was carried out, a methane yield of 0.478 [m.sup.3]/kg of added VS was reported. In addition, the kinetic analysis on the designed reactor using a first-order model resulted in a kinetic constant of 3.1 [day.sup.-1] and an ultimate methane yield of 0.489 C[H.sub.4]/kg VS. Not only that, the leachate produced during the process was suitably utilized too.

4.2. Role of co-digestion

One of the most classical ways of increasing the efficiency of an anaerobic digester treating OFMSW is by co-digestion. Co-digestion of OFMSW and bio-solids provides an attractive alternative for the management of two separate waste streams that are produced in every community (Cecchi et al. 1988). As such, the concept of co-digestion of OFMSW with manure and other industrial organic wastes was successfully materialized by Callaghan et al. (1999). However, in a separate bench-scale study, primary sewage sludge was co-digested with OFMSW instead of manure for substantial biogas production (Kiely et al. 1997). Similar co-digestion of semisolid organic waste with sewage sludge was also investigated by Sharma et al. (2000) on a newly designed reactor, which showed greater biogas production and stabilization. On two separate occasions, McMahon et al. (2001) and Stroot et al. (2001) studied the digester performance and microbial population dynamics while conducting anaerobic co-digestion of MSW and bio-solids under various mixing conditions. It was inferred from both the experiments that continuous mixing may cause an inhibitory effect on the archael population dynamics during higher OLRs. That was primarily attributed to the fact that continuous mixing at a high OLR inhibited syntrophic oxidation of VFAs thereby resulting in an abrupt reduction of the methanogenic abundance. Consequently an increased stability in the digester performance was noticed by reducing the level of mixing.

A study conducted by Schmit and Ellis (2001) showed the comparison between temperature-phased and two-phased anaerobic co-digestion of primary sludge and OFMSW, as substrate, in terms of all the key phases (hydrolysis, acidogenesis, and methanogenesis). On a separate occasion, Hartmann and Ahring (2005) reported a study on co-digestion of OFMSW with animal manure, which showed stable performance with a significant amount of methane and very low VFA levels. It is obvious that the economic feasibility of an AD plant treating OFMSW increases with higher biogas yield. In the case of an AD plant solely depending on OFMSW, as its influent substrate, an increase in biogas yield can be achieved by co-digesting OFMSW with animal manure, because of the high biogas potential of the animal manure. In a separate experimental study, Fernandez et al. (2005) evaluated the potential of mesophilic AD for the treatment of fats of different origin through co-digestion with the OFMSW. Fats are generally characterized by the presence of lipids, which are considered to be important biochemical components for high theoretical methane production because of their reduced nature (Pereira et al. 2003). Thus it can be seen that the benefits of co-digestion of OFMSW with another residue are well described in several studies (Sosnowski et al. 2003; Dohanyos et al. 2004; Bouallagui et al. 2004). However, the application of co-digestion with industrial sludge is very much limited. Capela et al. (2008) evaluated the technical feasibility of anaerobic co-digestion of three organic solid wastes under mesophilic conditions: OFMSW, industrial sludge, and cattle manure.

Co-digestion of wasted sewage sludge (WSS) (WSS can be defined as the sludge wasted during secondary treatment either from the main aeration unit or the secondary clarifier of an activated sludge process system) with OFMSW has become a popular mode of improving the biogas yield and stability of anaerobic digesters (Hartmann and Ahring 2006; De Baere 2006; Bouallagui et al. 2009). Similarly, slaughterhouse wastes can also be used as an efficient co-substrate in the AD of OFMSW. Anaerobic co-digestion of the slaughterhouse waste not only provides an option for the reduction of such harmful wastes, but also provides an alternative pathway of extracting the potential benefits in terms of biogas generation. Hence, it can be said that AD of OFMSW using certain co-substrates not only serves to enhance the rate of methane production and operational stability, but also contributes to a low energy requirement for operation, a low initial investment cost, and a low sludge production (Nguyena et al. 2007). In addition, the final digested material from co-digestion is more stabilized and hence serves better as a soil conditioner. Alvarez and Liden (2008) experimentally evaluated the potential of semi-continuous mesophilic AD for the treatment of solid slaughterhouse waste, fruit and vegetable wastes (FVWs), and manure in a co-digestion process. It was observed that the digestion of the mixed substrates yielded better VS reduction and steady-state biogas production than with the digestion of the pure substrates, with the exception of solid cattle-swine slaughterhouse waste with FVW mixture. This finding emphasizes the importance of combined treatment for certain wastes that cannot be successfully treated individually. In another study, Cuetos et al. (2008) carried out the AD of both slaughterhouse waste and mixtures of solid slaughterhouse waste with OFMSW in a mesophilic semi-continuously fed digester. It was observed that in a medium with high fat and ammonia content (such as slaughterhouse waste) the sludge needed to be acclimated properly for a certain period of time (initial HRT 50 days) for the AD to proceed steadily at a subsequently shorter HRT and progressively increasing OLR. This resulted in a successful co-digestion with a better yield and VS removal than for co-digestion carried out at an initial HRT of 25 days, which led to digester failure. A comparative study was carried out by Zhang et al. (2008) demonstrating the performance difference between sole digestion of bio-solids and co-digestion of bio-solids and OFMSW. It was observed that co-digestion of bio-solids with OFMSW resulted in increased biogas yield and efficient removal of VS and TS, along with significant reduction in the volume of bio-solids and OFMSW.

Nowadays, it can be said that AD is the most acceptable technology for the treatment of OFMSW, because it is not only self-sustaining, but there is a provision of energy recovery in the form of methane gas (Lv et al. 2010). It also has a tremendous potential to stabilize a huge quantum of the generated MSW (Ge et al. 2010). Co-digestion provides a scope for easy stabilization of complex and diverse substrates with OFMSW. Moreover, the nutritional requirement of the inoculum employed is met by means of co-digesting OFMSW with various other substrates. Ponsa et al. (2011) studied the feasibility of co-digesting OFMSW with different kinds of pure organic co-substrates, such as vegetable oil, animal fat, cellulose, and peptone (protein). It improved the functioning of the anaerobic digesters by increasing the biogas yield. Waste-activated sludge has been reported to offer better biogas yield and increased process stability when co-digested with OFMSW. It is also imperative to consider the influence of OLRs on the system stability when OFMSW is co-digested with waste-activated sludge. Consequently, Liu et al. (2012b) investigated the effects of OLR on the performance and stability of anaerobic co-digestion of municipal biomass waste and waste-activated sludge on a pilot-scale reactor.

4.3. Role of sorting techniques and pretreatments

The substrate concentration is another important consideration, as Fernandez et al. (2008) analyzed the effect of substrate concentration (based on the TS contents in the reactor) on the mesophilic AD during the start-up phase. The higher the initial organic substrate concentration (DOC or TS concentration) the longer the start-up phase, because of prolonged hydrolysis and acidogenesis phases, and the lower the methane yield. Forster-Carneiro et al. (2008b) analyzed the performance of two laboratory-scale reactors treating two types of SS-OFMSW obtained from a university restaurant and MS-OFMSW obtained from a municipal treatment plant located in Cadiz, Spain. SS-OFMSW was characterized by high start-up phase because it followed the classical waste decomposition pattern (i.e., a first start-up phase followed by the acclimation phase, and finally the stabilization phase). On the contrary, the MS-OFMSW exhibited a methanogenic pattern throughout the entire experimental period thereby showing no visible difference between the acidogenic, acetogenic, and methanogenic phases. In the case of the MS-OFMSW a lower C/N ratio, which is indispensable for biomass growth and diversification, was marked. This resulted in a stable pH for optimal biological activity until the stabilization period. Also, the biodegradability of the MS-OFMSW was higher because of lower DOC. With the gradual advancement in the various design features of the AD, it is essential to set a benchmark against which the different designs can be compared for their benefits, constraints, kinetics, and pollution potential of the OFMSW.

It is appropriate to say that AD of OFMSW has been considerably boosted by various worldwide developments in the last 15 years. Dong et al. (2010) developed one innovative methodology, where the feasibility of methane generation from semi-dry mesophilic AD of water-sorted organic fraction of municipal solid waste (WS-OFMSW) was worked out. WS-OFMSW is obtained by separating completely mixed MSW into different classes of fractions, such as the biodegradable fraction, metals, heavy materials, plastics, and the combustible fraction by the buoyant and stink force of water with various machines assisting. The biodegradable fraction obtained is the WS-OFMSW comprising kitchen waste, FVW, garden waste, paper waste, etc. The biodegradability of the WS-OFMSW is evidently greater than the MS-OFMSW and hence semi-dry AD of WS-OFMSW under mesophilic conditions resulted in a satisfactory biogas yield with as high as 66% methane content. Pognani et al. (2012) assessed the performance of a combined anaerobic-aerobic full-scale plant designed for the treatment of the SS-OFMSW and it was observed from the study that during the pretreatment step about 32% of the initial waste matter was rejected without any treatment, decreasing the overall biodegradability of the organic matter. However, the final compost was found to contain 50% of the initial nitrogen and 86.4% of the initial phosphorus, resulting in a high level of stabilization. Novarino and Zanetti (2012) studied the effect of "pressure extrusion", a mechanical pre-treatment process that haas seen recent application in treatment plants on AD of OFMSW across European countries. This was done to separate the undesired fractions and to reduce the final OFMSW content into a homogenous jam. The addition of this extruded OFMSW was done with activated sludge, which acted as a diluting agent. The result showed that with the stepwise increase in the TS content rate, the average specific bio-gas production (SBP) increased significantly per kilogram of VS degraded and the percentage of methane, by volume, also increased considerably.

Of late, the application of AD for the treatment of the industrial contribution of OFMSW has been of special interest. Biotransformation has been regarded as the main barrier in the treatment of those wastes. These industrial contributions of MSW along with certain other procured MSW sometimes have complex organic fractions, which cannot be easily broken down via microbial activity. This is especially the problem if the digester is dedicated to the anaerobic stabilization of such complex OFMSW. In such cases various pretreatments, such as biological, thermal, chemical, and combinations of all three may be adopted prior to the main AD process (Fdez.-Guelfo et al. 2011a, 2011b). The biodegradability of the complex substrates can be improved by the various aforesaid pretreatments. For instance, in the case of cellulose fermentation, the hydrolytic phase, which is always the rate-limiting step, can be enhanced by addition of certain extracellular enzymes called hydrolases, which catalyze the reactions.

The pretreatment causes a deep modification by weakening the molecular bonds in the structure of the complex substrates thereby breaking the polymers and increasing the superficial area of particulate wastes. Biological pretreatment ensures breaking of complex molecules into simple monomers so that the solubilization of the organic material is increased and subsequently the efficiency of the AD process is bettered, resulting in enhanced biogas and methane production (Fdez.-Guelfo et al. 2011c). Other conventional types of biological pretreatment that can be applied to OFMSW prior to the main AD include aerobic pretreatments, such as composting or micro-aeration. These methods obtain better hydrolysis of complex substrates because of the higher production of hydrolytic enzymes. Zhou et al. (2013) studied the effect of thermal hydrolysis, also a pretreatment, on the physical and the chemical properties of OFMSW while co-digesting WSS, restaurant kitchen waste, and FVW in a pilot plant. The results indicated that thermal hydrolysis increased digestibility by as much as 115% by dissolving 38.3% of the volatile suspended solids (VSS). It was also concluded from the study that the digestion rate doubled as a result of performing the thermal pre-treatment, with less accumulation of VFAs, compared to an AD process without the thermal pretreatment.

From the perspective of an increased rate of degradation of the OFMSW, enhanced gas production, cost efficiency, energy balance, and process sustainability (Ariunbaatar et al. 2014) various pre-treatment techniques, such as that of mechanical pre-treatment, thermal pre-treatment, chemical pre-treatment, and biological pre-treatment, play a pivotal role. For instance, different mechanical pretreatment techniques, such as rotary drum, screw press, disc screen shredder, FW disposer, piston press, and eletroporation, have different impacts on chemical oxygen demand (COD) solubilization, biogas production, and high methane yield per unit weight of VS degraded. However, the selection of a particular technology depends largely on the nature of the OFMSW to be treated. This is because each technology has an unique impact on the various factors, such as particle size, VFA accumulation, and quantity as well as quality of the sorted OFMSW, that enhance the COD solubilization rate, biogas production rate, and the percentage of methane yield. Earlier, it was established that the organic fraction obtained from the waste separation was particularly more useful for AD than that obtained from mechanical separation because of the presence of higher TVS content (Bolzonella et al. 2001). However, Fantozzi and Buratti (2011) evaluated the performance of AD on different portions of OFMSW by squeezing in the OFMSW with or without inoculum via mechanical treatment.

Similarly, in the case of thermal pre-treatment, such as thermal hydrolysis, the disintegration of cell membranes of the complex substrates results in the solubilization of the organic compounds thereby enhancing the COD solubilization rate. Not only that, thermal pretreatment of OFMSW results in pathogen removal, improvement in dewatering performance, and viscosity reduction of the digestate. Studies have shown that thermal pretreatments of FW and FVW at temperatures between 70 and 90 [degrees]C for 60 min led to a significant increase in the biogas production rate with high methane yield. However, to economize the thermal pretreatment technology, the duration of the pretreatment has been shortened considerably to 30-40 min by elevating the temperature to 120 from 90 [degrees]C, which also resulted in a similar outcome, in terms of biogas production rate and methane yield. The application of chemical pre-treatment on OFMSW has been mostly by alkaline pretreatment. Studies have shown that alkaline pretreatment of OFMSW resulted in higher methane production. Apart from alkaline pre-treatment, ozonation pretreatment (Cesaro and Belgiorno, 2013) of SS-OFMSW has also resulted in a 37% higher cumulative methane production.

5. Potential of AD of OFMSW

AD of OFMSW primarily involves a series of metabolic reactions (hydrolysis, acidogenesis, acetogenesis, and methanogenesis) controlled by diverse microbial populations to convert or degrade the organic waste into biogas and other energy-rich organic compounds as end products (Charles et al. 2009). In other words, the process of anaerobic decomposition has the potential to provide useful products, such as methane and bio-hydrogen, and the sludge generated is also used as an organic conditioner for soil. Besides these, the process of AD does not require external supply of oxygen (Guermoud et al. 2009; Chanakya et al. 2007) and there is no dependency on fossil fuel for energy production (Jingura and Matengaifa 2009). AD thus represents an opportunity to decrease environmental pollution as well as producing biogas and organic fertilizer or carrier material for bio-fertilizers (Khalid et al. 2011). The two major benefits of this process that reduce environmental pollution are (i) prevention of methane exiting into the atmosphere because AD occurs under closed conditions; and (ii) carbon-neutral carbon dioxide is released into the atmosphere during the burning of methane (Ward et al. 2008). Compared to other MSW management techniques, such as sanitary landfilling, thermal incineration, pyrolysis or gasification, and aerobic composting, the numerous potentials of AD in treating OFMSW include rapid digestion of solid organic waste constituents, huge volume reduction of organic matter and energy recovery with high-grade soil conditioner. In addition, the enclosed system also prevents the spread of diseases vectors (rodent, fly menace) and bad odor.

It is a well-known fact that AD is preferred over aerobic conversion because of the lower volume of final solid sludge produced. In the last few decades, the per-capita solid waste generation rate has been increasing mostly in an uncontrolled manner, and therefore AD has gained much importance. The rapid rate of digestion of these huge quanta of generated waste is making the process of AD not only economically viable but also environmentally indispensable. Economic viability is owed to the fact that in the treatment of the solid organic waste, virtually no power is required except for maintaining the operational temperature, which again can be sustained by the usage of the generated biogas. Thus in a way the AD of OFMSW is a self-sustaining process that may be needful to prevent the gradual exhaustion of fossil fuels. The ability of AD to stabilize enormous quanta of solid organic waste within a short period facilitates substantial reduction of waste volume on a daily basis thereby ensuring that comparatively smaller areas are required forthe final disposal of solid organic waste before these are subjected to AD. This allows the possibility to eliminate unhygienic conditions in cities, especially during periods when there is a sudden increase in the generation of the total organic solid waste load.

Nowadays, the availability of space for sustaining any environmental friendly technology is a major issue and the unprecedented urbanization coupled with the mushroom-like development of suburbs hinders this target. This particular challenge has also led municipalities to look for some pragmatic approaches to solid waste management, such as that of AD, where the land required for constructing a full-scale plant is smaller than for other MSW management techniques. Anaerobic treatment units require modular construction and the area required basically depends on the type of digester unit to be used. AD of OFMSW also has the potential to treat other types of wastes in conjunction with the solid organic fraction. It follows that other kinds of waste like sewage sludge, poultry residue, slaughterhouse waste, etc., can be mixed with the OFMSW to increase the rate of digestion. This process, better known as co-digestion or co-fermentation, not only ensures a faster rate of degradation of the primary substrate (OFMSW) but also of the co-substrate. Common benefits of co-digestion include easier handling of mixed waste (Li et al. 2009), adjustment of the C/N ratio (Xie et al. 2011), easy dilution of potentially toxic compounds, improved nutritional balance, increased loads of decomposable organic matter (Gannoun et al. 2007; Bouallagui et al. 2009), and increased gas yield (Macias-Corral et al. 2008). The toxic elements present in the OFMSW may disrupt the AD process. In this regard, co-digestion is the most efficient option to overcome the problems associated with toxicity of certain elements in the OFMSW. Apart from that, other potential benefits of co-digestion include stabilization of an increased load of organic matter, improved nutrient balance, better biogas yield, and synergistic effect of microbes (Khalid et al. 2011). The nutrient requirement gets complemented by co-digestion of an organic waste that provides excess nutrients thereby improving the C/N ratio, accelerating biodegradation of the solid organic fraction, and increasing the digestion and stabilization rate. For instance, the use of a co-substrate with a low nitrogen and lipid content increases the production of biogas because of complementary characteristics of both types of waste. This reduces problems associated with the accumulation of various intermediate volatile compounds and high ammonia concentrations (Castillo et al. 2006). Moreover, co-digestion with sewage sludge enables an integrated approach allowing the wastewater treatment plant to be independent as far as energy is concerned (Mata-Alvarez and Cecchi 1989). Figure 4 represents the various co-substrates that can be anaerobically co-digested with OFMSW for the production of biogas and sludge.

The potential of AD of OFMSW is also envisaged in the domain of energy production. The biogas generated from this process may greatly complement a society's need for energy. Today, the energy crisis has affected most parts of the world, and the generation of such a renewable source of energy is a welcome boost to the energy needs of civilizations worldwide. The energy produced via AD of OFMSW has a wide range of applications in both domestic and industrial domains. The biogas produced may be used in electricity generation, heating of furnaces, boilers, and fuel for vehicles. Even the sludge produced has a nutritional value, making it an effective soil conditioner. Instead of costly fertilizers, farmers can look forward to using this inexpensive organic soil conditioner thereby preventing damage to the parent soil via the use of certain chemical fertilizers. The abundant use of chemical fertilizers in modern agricultural practices, initiates imbalance in soil nutrients and their wash-out via surface runoff contaminates aquatic life. Hence, it is advisable to use this naturally procured soil conditioner as an organic amendment for the betterment of and to safeguard the overall environment.

6. Critical issues in AD of OFMSW

The stability of an anaerobic digester treating OFMSW depends on a number of critical issues, which if left unaddressed, would possibly result in the failure of the digester. The aforementioned potentials of the AD of OFMSW can be correlated with the smooth and hassle-free operation of an anaerobic digester treating OFMSW, which is a major challenge considering the number of critical operational issues that need to be carefully monitored. Among the number of critical operational issues linked with the AD of OFMSW, the following can be considered most important: (i) nature of the seed and inoculum source; (ii) operational temperature; (iii) OLR; (iv) high solids/low solids digestion; (v) mixing condition; (vi) nutrient requirements; (vii) pH control; (viii) C/N ratio; (ix) retention times; and (x) sorting techniques employed for selecting the OFMSW. Depending on these critical issues, the reactor configuration, pretreatments, recirculation strategies, start-up strategies, and adoption of co-digestion may be adjusted accordingly. In general, the performance of an anaerobic digester treating OFMSW largely depends on the previously mentioned critical issues.

6.1. Start-up strategies for AD of OFMSW

The start-up of an anaerobic digester holds the key to its overall performance and stability. Once the start-up has been made successfully, an anaerobic digester is capable of running with minimal attention, as long as the steady-state operating conditions are not altered significantly (Hobson and Wheatley 1993). Even though the most crucial issue that dictates the start-up of an AD reactor is the nature of the seed source, there are other important factors, such as the mode of giving the feed, the type of the reactor, the nature of the feed or substrate, the volume of the inoculum added, the adaptation of the inoculum and the process acclimatization from the mesophilic to the thermophilic temperature regime.

6.1.1. Fundamental requirements

The start-up of an anaerobic digester is mainly dictated by the nature of the inoculum chosen. Sometimes the anaerobic digesters treating OFMSW may have to be started up using inoculum that is not acclimated to digesting such complex feedstock, resembling OFMSW. Under such circumstances, a combination of inocula from two different anaerobic environments may be used as a seed. Anaerobic digesters operated at high substrate input are very much prone to imbalance in the activities of the hydrofytic-fermentative bacteria, the proton-reducing acetogenic bacteria, and the methanogens (Schink 1988). This can be overcome by the use of anaerobic sludge from a stable sewage sludge digester, which provides a balanced combination of fermenters, acetogens, and methanogens. Similarly, for successful start-up of a thermophilic AD reactor, non-availability of a thermophilic inoculum may be overcome by the use of two different mesophilic inocula as seed sludge (Griffin et al. 1998).

In the case of start-up of a pilot-scale CSTR treating a mixture of both SS-OFMSW and MS-OFMSW, and operated under a thermophilic temperature regime, the best possible result, in terms of specific methane production (SMP), was obtained when the start-up of the reactor was made at mesophilic temperatures and then gradually shifted to the thermophilic regime (Bolzonella et al. 2003). The inoculum type was a combination of primary and secondary sludge from a wastewater treatment plant. The initiation of the feed was done only after there was an appreciable drop in the initial VFA level (<200 mg/L) and the inoculum had been acclimatized at the mesophilic temperature regime. Significant specific gas production was noticed at an OLR of 1 kg TVS/[m.sup.3] of reactor per day with an HRT of 15 days. The changing from mesophilic to thermophilic conditions contributed to the degradation rate of the organic matter and proteins, causing an increase in the buffer capacity (Poggi-Varaldo et al. 1997a). In another study conducted by Angelidaki et al. (2006) it was concluded that the start-up of a thermophilic CSTR treating SS-OFMSW and other high-strength substrates can be successfully made by the selection of a thermophilic inoculum and the progressive increase in the inoculum loading rate. The importance of obtaining inoculum adapted to the same temperature regime or within [+ or -]10 [degrees]C of the intended start-up process temperature was inferred from the study. The main reason for procuring inoculum adapted to [+ or -]10 [degrees]C of the intended start-up process temperature is the development and retention of high concentrations of active biomass content inside the reactor. This is especially important for thermophilic processes because mesophilic sludge has been reported to contain only 9% thermophiles and 1% obligate-thermophiles (Chen 1983). The progressive increase in the inoculum loading rate and the subsequent ratio of low substrate concentration to constant inoculum volume can be attributed to the fact that the complete anaerobic degradation of the organic solids with high specific methane yield can be achieved without the accumulation of inhibiting levels of intermediate anaerobic degradation products. In other words, the initial low-feeding strategy ensures a gradual balance between the various anaerobic degradation steps, which is necessary to develop sufficient microbial degradation capacity, thereby avoiding any possible

VFA accumulation. Another reason for optimizing the low initial amount of fresh organic feed relative to the inoculum amount during start-up is due to the nature of the conventional digester type like CSTR, where the acidogenesis and methanogenesis phases occur together.

In the absence of classical inoculum, the start-up of a mesophilic anaerobic digester, treating SS-OFMSW, can be successfully carried out by the right selection of one or two different inocula and subjecting them to chemical and microbiological characterization. The importance of subjecting the inoculum to chemical and microbiological characterization is related to the need to determine the presence of microflora in the inoculum and assess their methanogenic activity before they are used. As such, it was observed from the study conducted by Maroun and El Fadel (2007) that during the start-up of an anaerobic digester (CSTR) treating SS-OFMSW, high initial levels of ammonia and VFA were exhibited at inhibitory levels, which restricted the increase of the OLR. However, this was overcome by the dilution of the reactor content with mineral water and re-inoculation with enriched cattle manure (representing an appropriate source of methanogens), which resulted in the increase of the OLR by 116%.

6.1.2. Classical issues related with start-up

Even though it is stated in the literature that a gradual increase in the OLR results in a high methane production rate, the overall TS content of the feed plays a very important role in the start-up of an anaerobic digester and the rate of methane production. Based on the TS content of solid organic waste, there are three types of AD technology: wet process ([less than or equal to]10% TS), semi-dry process (10%-20% TS), and dry process ([greater than or equal to]20% TS). It has been observed from various studies that, with an increase in TS content above 30%, the methane production rate decreases significantly when compared to the semi-dry AD process (Fernandez et al. 2008; Forster-Carneiro et al. 2008c). This is mainly due to reduced substrate degradation rate and accumulation of intermediate products (VFA mainly). However, this can be overcome by significantly increasing the moisture content of the feed by: (i) increasing the moisture content from 65% to 82% corresponding to a feed with TS 35%, leading to a linear increase in the specific methanogenic activity by a factor of 3.5 (Le Hyaric et al. 2011); (ii) gradually acclimatizing the methanogens to a feed with high TS content (in excess of 30%), following start-up with an inocula from an anaerobic digester, which has reached a steady state condition even after being subjected to shock-loadings.

It was inferred from a study conducted by Abbassi-Guendouz et al. (2012) that with the increase in TS content, the first-order hydrolysis rate decreases. The first-order rate of hydrolysis, a limiting step in an AD process, was required to be reduced to explain the cumulative methane production rate corresponding to a few experiments, which were uninhibited by high VFA concentrations and low pH at 30% TS.

The mass-transfer rate, which depends on the texture of the substrate-biomass mixture and the solid-liquid or -gas interface, decreases with an increase in the TS content. This can be attributed to the fact that with the increase in the TS content there is an increase in the viscosity of the substrate-biomass mixture, which causes a pasty texture and a decrease in the porosity (water content) of the solid-liquid or -gas interface, leading to low biogas bubble generation. Thus the volumetric liquid or gas mass-transfer coefficient, [k.sub.I] (which was equivalent to [k.sub.L] a in the study), depends on [k.sub.L] (the mass transfer coefficient) and a (the specific surface area). It was also determined from a study conducted by Abbassi-Guendouz et al. (2012) that the methane production rate was not influenced by the overall mass transfer above a critical [k.sub.T] value and that the cumulative methane production remained almost constant independent of the [k.sub.T]. value. However, below the critical [k.sub.T] value, limited overall mass transfer considerably lowered the cumulative methane production rate. At high TS (30%-35%) content, the overall mass transfer limitation was due to the accumulation of inorganic carbon (C[O.sub.2]), dissolved methane, and dissolved hydrogen. The inorganic carbon produced during methanogenes got entrapped in the matrix at high TS content, due to the low release of dissolved compounds in the digestate. As a result, the C[O.sub.2] accumulation in the matrix increased, leading to local acidification and subsequent inhibition of methanogenesis from the start-up phase implicating VFA accumulation. The accumulation of dissolved hydrogen also caused inhibition of valerate, butyrate, and propionate degradation, resulting in overall VFA accumulation (Lovely et al. 1982; Ahring and Westermann 1988; Fukuzaki et al. 1990). Hence, the produced hydrogen was also required to be consumed locally or removed from the liquid phase through liquid or gas mass transfer (Vavilin et al. 1995).

6.1.3. Selection criteria for seed source

As highlighted previously, the start-up of an anaerobic digester treating OFMSW depends on numerous factors. However, it can be inferred from the preceding discussions that the seed source continues to be the most crucial aspect in the start-up of an anaerobic digester treating OFMSW, because the correct composition of bacteria in the seed-source is mainly essential for the stability of the reactor following start-up. In fact, the seed source must contain a well-balanced supply of the following classes of microorganisms:

1. Fermentative group of bacteria, which are capable of carrying out the process of hydrolysis by degrading the macromolecular products (lipids, carbohydrates/polysaccharides, proteins, nucleic acids) to soluble or simple monomers (fatty acids, sugars/monosaccharides, amino acids/long-chain fatty acids, purines and pyrimidines) (Metcalf and Eddy Inc. 2003).

2. A group of fermentative bacteria, which are responsible for carrying out the process of acidogenesis, whereby the soluble or simple monomers (amino acids, sugars, some fatty acids), produced via hydrolysis, are fermented further to [H.sub.2], C[O.sub.2], VFAs, formate, propionate, butyrate, pyruvate, alcohols, ketones, lactic acid, etc. (Metcalf and Eddy Inc. 2003).

3. Acetogens, which are responsible for converting the VFAs, alcohols, formate, propionate, butyrate, etc., to hydrogen, C[O.sub.2], and acetate. These acetogens are known as obligate [H.sub.2]producing acetogens.

4. Homoacetogens, which are capable of carrying out homo acetogenesis by reduction of C[O.sub.2] with [H.sub.2] to acetate via the acetyl-CoA pathway (2C[O.sub.2] + 4[H.sub.2] [right arrow] C[H.sub.3]COOH + 2[H.sub.2]O) (Diekert and Wohlfarth 1994; Drake et al. 2008).

5. Methanogens, which are acetoclastic methanogens, which convert acetic acid to C[O.sub.2] and C[H.sub.4] and hydrogenotrophic methanogens, that utilize [H.sub.2] as the electron donor and C[O.sub.2] as the electron acceptor to produce C[H.sub.4] (Metcalf and Eddy Inc. 2003).

Apart from these classes of microorganisms, which can be considered as the most important and fundamental groups of bacteria necessary for the start-up of any anaerobic digester, the seed must also contain various syntrophs, such as nitrate and sulfate reducing bacteria, and propionate, acetate, lactate oxidizing bacteria.

Depending on the nature of the operational temperature regime opted for, the seed has to be procured. However, because the number of thermophilic digester systems under operation is much lower than the number of mesophilic digester systems, the start-up of a thermophilic anaerobic digester system can be made with mesophilic seeds (i.e., mesophilic anaerobic sludge). This is attributed to the fact that mesophilic anaerobic sludge hosts a large diversity of anaerobes that can be effective for start-up under thermophilic conditions as well (Fernandez et al. 2008; Gallert and Winter 1997).

Methanogens are obligate anaerobes that cannot tolerate [O.sub.2] because they are unable to form protective spores in unfavorable environments (Valdez-Vazquez et al. 2009). However, methanogens can be found in oxic conditions (Liu and Conrad 2010), such as those prevailing in landfill leachate, soils, manure (raw, dried, and aged), compost, primary and secondary sludge from activated sludge process systems, and waste-activated sludge. Hence, the preceding sources can be used for inoculating anaerobic digesters. Cattle manure, waste-activated sludge, sludge from ASP systems are found to contain acetotrophic methanogens, hydrogenotrophic methanogens, and syntrophs, which are essential for the start-up of an anaerobic digester system. In the case of compost, the process of composting at 60 [degrees]C and above enables the thermo-tolerant methanogens (present in the aerated compost) to carry out exothermal aerobic respiration, thereby increasing their numbers. They remain in anoxic niches during the thermophilic phase. Hence, during the start-up, the purpose of proper seeding can thus be satisfied by filling up the reactor using an equal volume of a mesophilic or thermophilic seed and either of the previously mentioned inocula.

6.2. Operational temperature selection

An AD system treating OFMSW can either be operated under a mesophilic temperature regime (30-38 [degrees]C) or a thermophilic temperature regime (49-57 [degrees]C), depending on the extent or the need for biogas production. It is well known that an AD system operating under a thermophilic temperature regime has a higher processing rate and superior biogas production rate, thereby enabling the use of smaller digestion units compared to an AD system operated under a mesophilic temperature regime, provided all the critical operational parameters are addressed properly. In addition, an AD system operating under a thermophilic temperature regime has several other advantages over a system operating under mesophilic temperature regime, such as provision for the selection of a high OLR, higher VS and COD removal rates, resistance to foaming, and superior destruction of pathogens (Mao et al. 2015). Moreover, thermophilic AD is characterized by a significantly higher hydrolysis rate (the rate limiting step in the AD process), which is especially notable when working with wastes with high solid content, such as OFMSW (Li et al. 2015). However, there are also several disadvantages, such as slow start-up and instability of the reactor system. The instability of the reactor system may arise out of lack of acclimated seeds, which is evident in the case of thermophilic anaerobic digester systems. Another reason for the reactor instability may be accumulation of various intermediate products (alcohols, VFAs), which necessitates long recovery periods (Rastogi et al. 2008), resulting in eventual system failure. Other problems associated with thermophilic AD in comparison to mesophilic AD include higher sensitivity to operational conditions, decreased stability due to ammonia and VFA (especially propionic) accumulation, poor quality of the supernatant, lower methane content in biogas, and higher net energy input (Song et al. 2004).

Another major problem associated with a thermophilic AD system treating OFMSW is the accumulation of propionic acid. This can be explained from the perspective of protein degradation, which is greater at thermophilic temperatures than mesophilic temperatures, leading to generation of higher ammonium ([NH.sub.4.sup.+]-N) concentration (Ariunbaatar et al. 2015). The ammonium ions (N[H.sub.4.sup.+]) and free unionized ammonia (N[H.sub.3]) ions in an anaerobic solution are present in chemical equilibrium, thereby forming total ammoniacal nitrogen (TAN). This equilibrium between ammonium and free ammonia is dictated by pH and temperature of the system. It was observed by several researchers that higher concentrations of ammonium ions in TAN reduce the activities of the propionic acid utilizing anaerobes leading to propionic acid accumulation (Angelidaki and Ahring 1993; Banks et al. 2012). TAN is required to be present in an adequate amount to provide buffer capacity and meet nutritional requirements for the methanogens. In a study conducted by El-Hadj et al. (2009) it was inferred that during AD of OFMSW the production of methane was inhibited by as much as 50% when the free ammonia (N[H.sub.3]) concentration was 215 mg/L under mesophilic conditions and 468 mg/L under thermophilic conditions. The concentration of ammonium ion (N[H.sub.4.sup.+]-N) was found to be 3860 mg/L under mesophilic conditions and 5600 mg/L under thermophilic conditions to cause similar inhibition of methane production. However, El-Hadj et al. (2009) also inferred that an optimum pH value of around 7 was found throughout the mesophilic condition independent of the TAN concentrations, whereas during the thermophilic condition, a TAN concentration of higher than 1331 mg/L resulted in an optimum pH of 7.5, but when the TAN concentration was below or equal to 1331 mg/L, the optimum pH was in the range 7-8.

The availability of a higher number of mesophilic digester systems substantiates the selection of a mesophilic temperature regime (Kim and Speece 2002; Suwannoppadol et al. 2011). However, if a thermophilic temperature regime is opted for, it has to be ensured that the operational temperature never exceeds 60 [degrees]C, as this may cause the inhibition of propionate degradation and the inhibition of hydrolysis. Also, if thermo-tolerant mesophiles are used for seeding there has to be a gradual increase in the temperature profile during the start-up phase.

Apart from the thermophilic and the mesophilic AD systems, another relatively new technology developed at Iowa State University (Schmit and Ellis 2001), is the temperature-phased anaerobic digestion (TPAD), which basically combines the advantages of thermophilic and mesophilic processes while managing to avoid the disadvantages of each one. This consists of a short (SRT of 1-3 days) thermophilic pre-treatment stage followed by a second mesophilic stage, operated with a longer retention time. The thermophilic stage enhances the hydrolysis and the acidogenesis rates, which are considered to be the rate-limiting steps in the AD process, whereas the mesophilic stage provides stable conditions for syntrophic acetogenesis and methanogenesis. In other words, the thermophilic stage in TPAD promotes hydrolysis and the mesophilic stage ensures the stability of the system by lessening the risk of inhibition due to the accumulation of ammonia and VFA. The thermophilic stage in TPAD can be operated at either acidic or neutral pH. At acidic pH, the hydrolysis and the acidogenesis stages are favored inhibiting the methanogenesis stage, whereas at neutral pH, a dynamic balance between hydrolysis or acidogenesis and methanogenesis is achieved. The subsequent mesophilic stage is thus used as a polishing stage eliminating the disadvantages of thermophilic digestion and facilitating the production of methane at both acidic and neutral pH stages (Lv et al. 2010, 2013). The concatenation of the two temperature systems allows a relatively faster process, resulting in enhanced degradation of the OFMSW, (feedstock) leading to a greater yield in biogas.

Even though the TPAD system has been mainly applied in the stabilization of municipal sewage sludge, there have been occasions where the technology has been found to be effective in treating FW (Chu et al. 2008; H.W. Kim et al. 2011) and OFMSW, as well (Schmit and Ellis 2001). In a study conducted by Borowski (2015) the TPAD system was applied in the treatment of a mixture of municipal sewage sludge and hydro-mechanically separated OFMSW, which resulted in a methane yield of 333 L C[H.sub.4]/kg VS and 52.1% VS reduction, compared to a single-stage mesophilic process treating the same waste, which showed a methane yield of 230 L C[H.sub.4]/kg VS and VS removal of 37.23%. As mentioned before, the thermophilic stage of the TPAD system was chosen to be operated at a shorter SRT of 1 day followed by the mesophilic stage at an SRT of 14 days. It was observed from the study that when the thermophilic stage of the TPAD system was operated at an SRT of 2 days instead of 1 day, the methane production in the subsequent mesophilic reactor was inhibited by high amounts of free ammonia, which were liberated in the supernatant because of greater intensity of protein degradation under prolonged exposure of the feedstock to the thermophilic temperature. It was also observed that high VFA concentrations, with predominant propionic acid, also contributed to the inhibition of methanogenesis, when the SRT was kept high during the thermophilic stage. The need for a longer SRT for the mesophilic stage in a TPAD system was asserted in another study carried out by Fernandez-Rodriguez et al. (2016). It was observed that under SRT of 3 and 6 days for thermophilic and mesophilic stages, respectively, the yield of methane was lower and the organic matter removal was also lower than when the TPAD was run at an SRT of 4 and 10 days for the same, during semi-continuous TPAD of OFMSW coming from a MBT plant. However, at both sets of SRT the TPAD system showed better performance than the single-reactor systems at similar SRTs, while treating OFMSW coming from an MBT.

It can be concluded from this discussion that a multi-stage (preferably a three-stage) anaerobic digester may be not only economical but also appropriate. It is well-known that in terms of the operational temperature, the methanogens favor higher temperatures, in the range of 30-38 [degrees]C (for mesophilic operations) and 49-57 [degrees]C (for thermophilic operations), for increased methane production. In a single-staged system with large digesters, the energy required for elevating temperatures, as mentioned, is often greater than the energy produced, thereby making it uneconomical.

6.3. OLR

The OLR depends on numerous factors, such as the nature of the substrate, type of digestion (wet or dry) and reactor configuration (single-stage or multi-stage). For instance, highly biodegradable substrates (such as FW) account for rapid degradation, thereby resulting in the production of VFA and other intermediate products within a very short time. This causes a decrease in pH below 6.8 resulting in reactor instability. Apart from that, foaming also occurs because of propionate accumulation adding to the reactor instability. A low OLR, during the start-up phase, ensures complete degradation of the accumulated products. The OLR should only be increased gradually after pH, alkalinity, and other important operational parameters are stabilized (such as VFA < 300 mg/L as C[H.sub.3]COOH, COD removal above 70% or when the C[H.sub.4] content in the biogas is 60%) (Poggi-Varaldo and Oleszkiewicz 1992; Poggi-Varaldo et al. 1997b). Similarly, if wet digestion (3%-6% TS content) is adopted, the OLR could be kept high and if dry digestion (TS content > 15% of the substrate input) is adopted, the OLR has to be kept low.

Reactor configuration also plays a vital role in determining the OLR. Single-stage reactor treating highly biodegradable wastes needs to be operated at low OLR. This is because the highly biodegradable wastes mostly comprise up to 75% sugars and hemicelluloses and 15% recalcitrant (lignin and cellulose). This composition results in the rapid production of VFAs, necessitating a low OLR to avoid reactor instability. The need to lower the OLR can again be attributed to the sharp drop in pH, arising from high VFA production. The rate of acidogenesis being greater than the rate of methanogenesis results in the destruction of the VFAs by the methanogens very slowly (Bouallagui et al. 2005; Ward et al. 2008). Thus it can be inferred that a high OLR can only be employed if the reactor configuration is multi-staged, so that the process of methanogenesis is not hampered by VFA accumulation.

6.4. Type of digestion

Generally there are two types of digestion that can be employed in an anaerobic digester system. They are dry digestion, also known as high-solids digestion, and wet digestion, also known as low-solids digestion. Wet digestion is defined as the presence of 3%-6% (or less than 15%) TS in the substrate input, whereas dry digestion is defined as the presence of more than 20% TS in the input substrate. However, the Theological behavior of solid organic wastes classifies AD into three types, with semi-dry AD being the intermediate one. Hence, in a broader perspective it can be said that wet AD is classified as [less than or equal to]10% TS content in the substrate, semi-dry AD is classified as 10%-20% TS in the substrate, and dry AD is classified as [greater than or equal to]20% TS content in the substrate (Abbassi-Guendouz et al. 2012). The selection of the digestion type depends on the nature of the substrate, operational temperature, and reactor configuration. For instance, highly biodegradable substrates (FWs) necessitate wet digestion, because the large fraction of carbohydrates results in the production of a high concentration of VFAs, which gets removed by the methanogens very slowly (i.e., the rate of VFA production is greater than the rate of VFA destruction). However, in the case of a multi-staged reactor system, where hydrolysis, acidogenesis, and methanogenesis occur in separate chambers, dry digestion can be adopted even if the substrate is highly biodegradable, because the accumulated VFAs never interfere with the methanogenesis.

The operational temperature regime also plays a role in determining the type of digestion. A system employing a thermophilic temperature regime has a high OLR, which means it can opt for high-solids digestion. However, systems employing a mesophilic temperature regime can also opt for high-solids digestion provided wet digestion is selected during the start-up phase and continued until the digester system is stabilized (i.e., the digestion system reaches a steady state condition) (Bolzonella et al. 2003).

6.5. Mixing

One of the most crucial operational factors that influences the performance of an AD system involving stabilization of OFMSW is mixing. The impact of mixing on the digester performance depends largely on the intensity and the mode of employing the mixing operation. Proper mixing is important for the homogenization of the waste, the interaction between the nutrients, and the viable bacteria population, uniform heat distribution, improved retention (SRT and HRT) times, the efficient dispersion of metabolic waste, the degradation of the accumulated propionate, and pathogen removal (El-Fadel et al. 2013). Apart from these, mixing also reduces the particle size by shear forces, thereby increasing the waste surface area. Hydrolysis, which is a rate-limiting step, gets enhanced as a result of increase in waste surface area, because the accessibility of the substrates inside the reactor increases (Halalsheh et al. 2011). Effective mixing also results in the inactivation of dead-zones, thereby minimizing channeling. In the absence of mixing, there is a reduction in the effective digester volume by up to 60%. The degradation of accumulated propionate results in improved reactor stability.

Even though it is seen that mixing is one of the critical issues in dictating the stability and consistency of an AD system, and that the performance of AD system treating OFMSW gets improved by employing the mixing operation, there are also considerable debates surrounding the various problems associated with the mixing operation. The absence of mixing during the starting up of a reactor shortens the start-up period. Interrupted mixing leads to formation of hydraulic dead-zones, thereby reducing the HRT and adversely affecting the reaction kinetics. The most common problem associated with mixing is that it increases the distance between syntrophic partners and destroys their associations (McMahon et al. 2001; Stroot et al. 2001). The degradation of organic acids, alcohols, propionate, butyrate, valerate, etc., is due to close microbial proximity of less than 10 [micro]m, and mixing increases the distance between syntrophic partners. For the production of [H.sub.2] and formate, the [H.sub.2] partial pressure should be less than [10.sup.-4] atm. The acidogenic bacteria functions at [H.sub.2] partial pressure of less than [10.sup.-4] atm, which is only possible if the hydrogenotrophic bacteria converts the [H.sub.2] and C[O.sub.2] to [CH.sub.4]. In the absence of mixing, the microbial proximity remains less than 10 [micro]m, thereby facilitating the transfer of [H.sub.2], C[O.sub.2], and acetate to their consumers (the hydrogenotrophic and the acetoclastic methanogens). This prevents the accumulation of propionate, which is degraded by the acidogens. The acidogens can survive at [H.sub.2] partial pressure of less than [10.sup.-4] atm (Batstone et al. 2004).

The strategy of recirculation over mixing also seems advantageous. Studies show that AD systems employing recirculation of slurry have better performances than those employing mechanical mixing by impellers (Karim et al. 2005), which are difficult to maintain. Recirculation has a negligible effect in systems operating at low solids content. Recirculation of leachate (El-Mashad et al. 2003) also enhances the performance of an AD system, because it enables the transportation of VFAs from acidogenic pockets (fresh substrates) to methanogenic pockets (seeds) (Veeken and Hamelers 2000).

6.6. Nutrient requirements

For the smooth functioning of an AD system, the microbes are needed to be supplemented with various macro- and micro-nutrients along with trace elements. This is because certain substrates lack the necessary nutrients required for the stable operation of the AD system. For instance, vegetable waste lacks sufficient cobalt concentration (Jiang et al. 2011); hence it has to be compensated by the external addition of cobalt. However, in certain cases even if the nutrients are present, they become inaccessible to the microbes because of various hindrances. For instance, sulfide precipitation reduces the availability of iron, cobalt, nickel, etc. (Barber and Stuckey 2000), which are essential nutrients for the methanogens. The digestion procedure (mono or co-digestion) employed, along with the nature of the substrate to be digested and the digester type, also dictates the extent of the trace element supplementation (Demirel and Scherer 2011). If the performance of an anaerobic digester is poor without any obvious reason, it should be checked for certain trace elements, the deficiency of which may be a major problem.

It was reported in a study by Lane (1984) that lab-scale mesophilic AD using a trace solution containing Fe, Zn, Mn, Cu, and Mo resulted in organic solids conversion ranging from 88% to 96% and a methane yield between 50% and 65%. More recently, a comprehensive study was carried out by Kayhanian and Rich (1995) exploring the addition of both micro- and macro-nutrients to OFMSW undergoing a thermophilic pilot-scale AD. It was observed that the addition of various macro-nutrients, such as Co, Cu, Fe, Mo, Ni, Se, W, and Zn, resulted in an elevated rate of gas production by 30%, along with increased digester stability. It was inferred that obtaining digester stability was primarily on account of supplementation of both macro- (N, K, P, and S) and micro-nutrients (trace metals). The addition of trace metals, such as Co, Ni, Mo, B, Se, and W was investigated by Feng et al. (2010) in a lab-scale AD of food industry residues. It was observed that the addition of high concentrations of Se or W and low level Co resulted in the highest production of methane. The concentration ranges of Se and W after addition varied from 0.008 to 0.8 mg/L and 0.018 to 1.80 mg/L, respectively, whereas the Co concentration ranged between 0.06 and 6 mg/L. In a separate study by Lo et al. (2012) the effects of eight different metals, namely, Ca, Cr, Ni, Co, Mo, W, Zn, and K were investigated on the AD of OFMSW under mesophilic conditions. Consequently, it was observed that there was enhanced biogas production due to the addition of Ca, Cr, Ni, Co, Mo, and W, with concentrations in the range of 728-1461 mg/L, 0.0022-0.0212 mg/L, 0.801-5.362 mg/L, 0.148-0.580 mg/L, 0.044-52.94 mg/L, and 0.658-40.39 mg/L, respectively. However, the addition of K and Zn did not show any enhancement in biogas production. On the other hand, it was also observed that, with the exception of Mo and W, inhibitory concentrations I[C.sub.50] of Ca, K, Cr, Ni, Zn, and Co were 3252, 2097, 0.124, 7.239, 0.482, and 8.625 mg/L, respectively. Above all, the study by Lo et al. (2012) inferred that the addition of the eight spiked metals resulted in their adsorption by the OFMSW to a different extent, thereby causing different liquid metal levels, potential stimulation, and inhibition on AD of OFMSW.

Climenhaga and Banks (2008a) inferred that during bench-scale AD of source separated food residues (FVW, cooked foods including meats and fried foods, peelings, bones, and fat trimmings) at a constant OLR and varied HRT levels of 25, 50, and 100 days, the regular addition of micro-nutrients provided stable digestion, which was otherwise absent. This study also concluded that the presence of heavy metals in the feedstock may not have been bioavailable for the methanogens for various reasons. As mentioned earlier, various heavy metals, despite being present in the feedstock, become inaccessible to the methanogens because they get precipitated by sulfides. In some cases, long-chain fatty acids may also bind with minerals, such as calcium, making them inaccessible to methanogens. In a separate study by Kumar et al. (2006) it was concluded that when certain heavy metals, such as Ni, Zn, and Cd, were added at concentrations of 2.5 mg/L to an AD reactor treating potato waste and cattle manure, there was enhanced biogas production. However, the biogas production was accessible for only a very short period because the pH of the substrate decreased to a toxic level after 2-3 days of AD after which it was not possible to recover it.

Mostly methanogenic bacteria present in pure cultures use [H.sub.2] and C[O.sub.2] as growth substrates. However, Methanosarcina barkeri has been reported to use methanol and acetate in addition to [H.sub.2] and C[O.sub.2]. Among the vast group of highly specialized methanogenic bacteria Methanosarcina barkeri seemed to be the most versatile with respect to the number of substrates utilized. This is attributed to the fact that it not only grows chemoautotrophically on [H.sub.2] plus C[O.sub.2], but also chemoorganotrophically on methanol, methylamine, dimethylamine, trimethylamine or acetate as carbon and energy sources (Schnellen 1947; Kluyver and Schnellen 1947; Hippe et al. 1979). It was reported in a study by Mountfort and Asher (1979) that M. barkeri has a specific physiological requirement for sulfide. Later on, Scherer and Sahm (1981) observed that M. barkeri was dependent on cobalt when grown in a defined medium with methanol, as energy and carbon source. In the absence of cobalt M. barkeri grew only very slowly compared with growth on the medium containing 1 [micro]mol/L Co[Cl.sub.2]. Moreover, Scherer and Sahm (1981) demonstrated that the addition of 0.5 [micro]mol/L molybdenum was stimulatory to growth under those conditions. They also observed that the growth of M. barken was significantly stimulated by the addition of nickel and selenium to the culture medium.

Another important methanogenic bacterium is Methanobacterium thermoautotrophicum. However, unlike M. barkeri, the growth of M. thermoautotrophicum depends on [H.sub.2] and C[O.sub.2] as the sole energy and carbon sources. It was observed by Zeikus and Wolfe (1972) that N[H.sub.3] and [H.sub.2]S were used by M. thermoautotrophicum as nitrogen and sulfur sources. In addition, Taylor and Pirt (1977) observed the growth of the bacterium to be dependent on iron. In a later study by Schonheit et al. (1979) it was observed that the formation of M. thermoautotrophicum was proportional to the addition of Ni, Co, and Mo into the medium. Interestingly, Schonheit et al. (1979) observed that the cell proliferation of the M. thermoautotrophicum was dependent on nickel, which is generally not an essential element for bacterial growth. The dependency on cobalt did not indicate the presence of corrinoids, which are synthesized by cobalt, in the bacterium. This was because the added transition metal was also a component of transcarboxylase (Northrop and Wood 1969).

The presence of trace metals as micronutrients is crucial for an anaerobic reactor stabilizing solid organic waste. In the absence of these elements, there may be poor process efficiency and eventually reactor failure. However, the presence of those elements above certain concentrations poses difficulties for the methanogenic bacteria to function properly. For instance, in a study by Ashley et al. (1982) it was shown that dissolved concentrations of nickel greater than 1 mg/L caused inhibition of methanogenesis during AD of sewage sludge. Bhattacharya et al. (1995) demonstrated that in a medium containing acetate-utilizing methanogenic enrichment culture, complete inhibition of methanogenesis was caused by the presence of 70 mg/L of free cobalt and 280 mg/L of total soluble cobalt, respectively. Also, free cobalt to VSS ratio exceeding 0.05 and total soluble cobalt to VSS ratio exceeding 0.16, caused complete failure of methanogenesis. Earlier, studies found that various heavy metals, such as zinc, nickel, and cadmium were much more toxic to methanogenesis from acetate compared to nickel (Lin 1992; Leslie 1991; Akanbi 1990).

It can be said that the source of the MSW plays a key role in determining the heavy metal content of the OFMSW undergoing AD. As such, the total absence of heavy metals can prove to be detrimental to the growth of microbes involved in AD (Abbasi and Ramasami 1999). Hence, the addition of micro- and macronutrients in the form of certain trace elements and heavy metal supplementation is essential during the start-up of an AD process, dedicated to stabilizing OFMSW. But, adequate care has to be taken so that the addition of those elements does not exceed inhibitory levels, which may cause unrecoverable cessation of the anaerobic digester.

6.7. pH Control

The stability and consistency of an AD system greatly depends on the pH of the system. The fermentative and the methanogenic group of bacteria are all pH-sensitive and operate at different pH-ranges. The slightest change in the desirable pH results in the cessation of the reactor performance. The favorable pH range for the growth of methanogenic bacteria is 6.8-7.3 (Brummeler and Koster 1989). The optimum pH for hydrolysis and acidogenesis is between 5.5 and 6.5 (Kim et al. 2003; Yu and Fang 2002), whereas the growth rate of methanogens is greatly reduced below pH 6.6 (Mosey and Fernandes 1989). At an excessively alkaline pH, disintegration of microbial granules occurs, which subsequently causes process failure (Sandberg and Ahring 1992). The anaerobic acid-forming bacteria have a lower pH range of less than 6.8

for growth (e.g., 5.0-6.0). Undissociated volatile acids, which are present below pH 6.8, are harmful to methanogens, because they can penetrate the bacterial cells without any resistance. The pH inside the methanogens is higher than the pH of the external medium, which results in the release of protons from the undissociated volatile acids causing inhibition in the formation of ATP and other processes inside the cytoplasm of the methanogens. This in turn inhibits methane formation (Anderson and Yang 1992; Mounfort 1978). Also, atpHabove 7.3, N[H.sub.3], a toxic form of N[H.sub.4]-N, formation occurs resulting in suppression of the methanogenic activity (El-Fadel et al. 2013).

It can be garnered from the preceding discussion that a multi-staged reactor (preferably a three-stage reactor system) is more advantageous in terms of stability and operational consistency than a single-stage reactor. For instance, the working pH of the hydrolytic and the acidogenic bacteria lies in the range of 5.5-6.5, whereas the methanogens have an optimum pH of 7.8-8.2 (Khanal 2008). A single-stage reactor system thus makes it difficult for the methanogens to operate because they are not only highly pH sensitive, but the organic acids produced inhibit their growth and metabolic activity. In the case of a single-stage reactor, the drop in pH may be countered by the use buffering agents (e.g., lime) so that a suitable pH in the range of 7.8-8.5 is obtained for the operation of the methanogens. But, when and if the buffering agents are added, the pH adjustments should be so estimated and worked out that the conditions in the reactor are equally suitable for hydrolysis and acidogenesis to occur.

Alkalinity plays a vital role in controlling the pH of an AD system. The total alkalinity of an AD system should generally lie between 7400 and 27000 mg/ L as CaC[O.sub.3] (Cecchi et al. 2003; Banks et al. 2008). The alkalinity should be properly maintained, because a drop in alkalinity indicates a subsequent drop in pH level. On the other hand, the bicarbonate alkalinity of an AD system should be in the range of 2400-5400 mg/ L (Anderson and Yang 1992; Ferrer et al. 2010). The bicarbonate alkalinity is the main carbon source for the autotrophic methanogens and represents the true buffering capacity of the system. In an AD system, the buffering capacity is often referred to as alkalinity and it is responsible for the equilibrium of carbon dioxide and bicarbonate ions. It provides resistance to significant and rapid changes in pH and therefore is proportional to the concentration of bicarbonate. The digester imbalance can be measured more reliably using buffering capacity than by pH measurement, which is a direct approach, because accumulation of VFAs reduces the buffering capacity significantly before the decrease in pH (Ward et al. 2008). The increase in a low buffering capacity can be brought about by reduction in the OLR. However, the addition of strong bases or carbonate salts, which results in the removal of C[O.sub.2] from the gas space by converting it into bicarbonate, is a more rapid approach to increase the buffering capacity (Guwy et al. 1997). The latter process of increasing buffering capacity is more accurate, because the conversion from C[O.sub.2] to bicarbonate requires a time lag for gas equilibrium to occur, which may result in over-dosing. Other approaches include modification of the inoculum-to-feed ratio, which helps to maintain a constant pH (Gunaseelan 1995), and co-digestion with another substrate, which has a higher alkalinity.

The VFA to alkalinity ratio ([alpha]) is a parameter that indicates the balance between the potential pH drop (VFA accumulation) and the buffering capacity (alkalinity) of the system (Poggi-Varaldo and Oleszkiewicz 1992). The alkalinity ratio ([alpha]) can be represented by,

[alpha] = [Acetic acid equivalents] / [Calcium carbonate equivalents]

The desired and optimal value of [alpha] is less than 0.3, even though the values of [alpha] between 0.3 and 0.4 are acceptable. If the value of [alpha] exceeds 0.4 then the digester is at risk because of potential VFA accumulation (Schoen et al. 2009). The accumulation of VFA can be prevented by suspended feeding, until the excess VFA is consumed, or by bio-augmentation using pure hydrogenotrophic culture or hydrogenotroph-rich seeds (Angenent et al. 2004).

MSW can be deemed as the most variable feedstock, especially given the fact that the yield of methane largely depends not only on the sorting technique, but also on the location of the source of feedstock and the time of collection. For instance, during the summer season the MSW will have a larger portion of garden waste in its organic fraction eventually leading to lower ultimate methane yield. Similarly, locations also influence the type of FW produced and the organic fraction generated by lifestyle and cultural differences (Ward et al. 2008). As such, the pH of the substrate or feedstock is also greatly influenced by the different composition and nature of the FW, kitchen waste, market waste, restaurant waste, etc., based on the season and the source where from they are being collected. If the substrate has organic fractions enriched with VS (FVWs) then there is rapid degradation in the anaerobic digester. This leads to rapid hydrolysis that may lead to acidification of the digester and the consequent inhibition of methanogenesis. Hence, feedstock or substrates, which are rapidly hydrolyzed, are capable of inhibiting methane formation by acidification of the digester. On the contrary, if the substrate has organic fractions, which are poorly degradable (yard waste, paper waste), the hydrolysis phase becomes the rate-limiting step.

In a study conducted by Forster-Carneiro et al. (2008a), the feedstock characteristics of three different organic wastes, FW, OFMSW, and SH-OFMSW, were carried out. It was observed from the study that the initial mean pH of the three substrates was FW-7.0, SH-OFMSW-7.6, OFMSW-7.3; and the initial mean alkalinity was FW-200 mg/L as CaC[O.sub.3], SH-OFMSW-300 mg/L as CaC[O.sub.3], OFMSW-510 mg/L as CaC[O.sub.3], emphasizing the aforementioned perspective of change in the pH and alkalinity. On the contrary, during thermophilic AD of the previously mentioned feedstock the pH values remained around 8.1 for the FW reactor; 7.8 for the SH-OFMSW reactor; and 7.9 for the OFMSW reactor. The mean alkalinity values were 900 mg/L as CaC[O.sub.3] for the FW reactor; 1400 mg/L as CaC[O.sub.3], for the SH-OFMSW reactor; and 1650 mg/L as CaC[O.sub.3], for the OFMSW reactor. It was concluded that the nature of organic substrate has an important bearing on the overall biodegradation process and methane yield.

VFA is a key intermediate in the process of AD, which is capable of inhibiting methanogenesis at high concentrations. It is a crucial parameter that influences the pH of an AD system. It is evident that the higher the VFA concentration in a system the lower the pH and the greater the environment is favoring the growth of acid-forming bacteria. In other words, the environment becomes increasingly unfavorable for the methanogens to survive. Normally the VFA inhibiting range is 1000-3000 mg/L as C[H.sub.3]COOH (Angelidaki et al. 2005). The wide range is due to various microbial structures having different buffering capacities because of their varying toleration and acclimation capacities. For instance, the methanosarcinaceae is found to be more tolerant to VFA than the methanosaetaceae (Karakashev et al. 2005). In a study carried out by Siegert and Banks (2005) it was observed that fermentation of glucose was inhibited at total VFA concentrations above 4000 mg/L as C[H.sub.3]COOH. Although acetic acid is usually present in higher concentrations during AD than other fatty acids, the activities of the methanogens are more inhibited by propionic and butyric acids (Wang et al. 1999). Even though Boone and Xun (1987) reported that propionic acid concentrations above 3000 mg/L caused digester failure Pullammanappallil et al. (2001) concluded that the propionic acid was the effect of inhibition in an AD system rather than the cause. An increase in VFA concentration can be due to overload of OLR, hence monitoring of VFAs, particularly butyrate and isobutyrate, has been demonstrated to indicate process stability (Ahring et al. 1995). This may be attributed to the fact that the methanogens present in an AD system are not able to metabolize the acetate produced unless their numbers are sufficiently high. As mentioned before, in the case of organic feedstock, which has high VS, the degree of hydrolysis is rapid and may especially contribute to this problem of VFA accumulation. Because methanogenesis inhibitors, such as excessive fatty acids, [H.sub.2]S, and N[H.sub.3], are mainly toxic in their unionized forms, it is imperative to control their concentrations so that there is a balance in the overall pH. This is because the relative proportion of the ionized and unionized forms is pH-dependent. For instance, ammonia is toxic above pH 7 whereas, VFA and hydrogen sulfide are toxic below pH 7.

As mentioned before, the composition of OFMSW varies widely, ranging from FW (vegetable waste or fruit peels) to yard waste (leaves or grasses). The strategy of the waste collection greatly affects the characteristics of OFMSW, hence the yield of biogas and disposal (agricultural fields, landfill, or incineration) of the solid-sludge or digestate, following the AD process. The SBP and SMP was observed to be 200 [m.sup.3]/ton of waste treated and 0.4 [m.sup.3]/kg of VS (fed to the reactor), respectively, in case of SS-OFMSW. However, for a mixture of the grey fraction of MSW, MS-OFMSW, and sludge, the SBP and SMP were recorded as 60 [m.sup.3]/ton of waste treated and 0.13 [m.sup.3]/kg of VS, respectively (Bolzonella et al. 2006). Thus it was inferred that higher biogas yields were generally obtained with SS-OFMSW than from MS-OFMSW. Also, the digestate of MS-OFMSW was difficult to handle and had to be disposed of in landfills, or incinerated. However, the heterogeneous nature of OFMSW in SS-AD system may create a multiplicity of ideal micro-environments for the growth of each of the microbial families required to complete the AD process. This in turn may result in the fermentative processes of hydrolysis and acidogenesis proceeding differently than in conventional submerged culture (Shankaranand et al. 1992).

Fermentative microorganisms, especially hydrolytic acidogens, require large surface areas to colonize and hydrolyze feedstock, which are insoluble. They operate actively at relatively low pH (5.5-6.0) (Chyi and Dague 1994). On the other hand, syntrophic acetogens and methanogens require stable neutral pH for functioning and are very sensitive to even low concentrations of inhibitors (e.g., N[H.sub.3], [H.sub.2]S, and short chain fatty acid). Not only that, syntrophic acetogens also need close proximity with methanogens for efficient interspecies hydrogen transfer. In the case of feedstock, which are rich in lignocellulose, the enhancement in hydrolysis is brought about by high temperatures, at which the loosening of the structure of the lignocellulose starts, thereby making them more accessible to colonization by cellulolytic and xylanolytic microbes, and their glycosyl hydrolases, eventually accelerating the enzymatic hydrolytic reaction (Lv et al. 2010). However, high-temperature based thermophilic anaerobic reactors suffer from poor stability, because of accumulation of short chain fatty acids, especially propionate; reduced methane production; and increased carbon dioxide content (Speece et al. 2006; Iranpour 2006). During the early stages of the thermophilic AD process, it is possible to reduce methanogenic inhibition due to excessive VFA accumulation and the subsequent drop in pH by co-digesting carbohydrate-rich feedstock with other feedstock, or by using a two-phase digestion system, such as TPAD (Li et al. 2011). Another way of preventing the methanogenic inhibition is by leachate recycling, which enables the colonization of the bacteria throughout the digester by providing a passive transport mechanism for microbial communities. This also considerably reduces the need for the large amount of digestate material required to inoculate the fresh feedstock before loading the digester. Even though this serves to be beneficial, in the case of AD system employing SS-OFMSW, the requirement of the solid-sludge or digestate to inoculate the fresh feedstock is also one of the major limitations of SS-AD (Li et al. 2011).

6.8. C/N ratio

The C/N ratio is an important parameter indicative of the nature of the waste and its potential biodegradability. The stability of an anaerobic digester can be ascertained by selecting the C/N ratio as the variable parameter. This can be attributed to the fact that the C/N ratio determines both the N[H.sub.3]-N and VFA concentrations observed in the anaerobic digester. With the increase in N[H.sub.3]-N, the pH is raised, whereas the VFAs, which neutralize the HC[O.sub.3.sup.-], C[O.sub.3.sup.2.-], and acetate ions, govern the decrease in pH (Shanmugam and Horan 2009). It was reported by Callaghan et al. (2000) and Salminen and Rintala (2002) that in case of AD systems acclimatized to treat high protein wastes the tolerance level of N[H.sub.3] were 11600 and 6000 mg/L, respectively. Normally, FWs have an optimum C/N ratio in the range of 30:1-40:1 (total carbon) and 25:1-30:1 (biodegradable carbon) (Kayhanian and Tchobanoglous 1993). However, in some instances the C/N ratio of FW may be below 20. In such cases, rich carbon sources (such as glucose) (Hills 1979), cellulosic waste (such as grass, cereals, straws, and other crop residues) (Fraser 1977), or mixed paper (C/N ratio of 100:1-173:1) (Yen and Brune 2007) may be added to achieve optimum conditions for the AD process. However, the addition of paper has to be made very carefully taking into consideration the manufacturing procedure of the paper to be added, because there are certain types of papers that are treated with additives for improving the bleaching process and the addition of these types of papers may lead to the release of certain chlorinated compounds, which are one of the principal inhibitors of microbial activity. The optimum COD/N range for efficient microbial activity, involving AD of OFMSW, is from 20:1 to 36.36:1, with 25:1 being considered as the optimum (Hills 1979; Kivaisi and Mtila 1998; Rao and Singh 2004; Boualllagui et al. 2005; Yen and Brune 2007; Guermoud et al. 2009; Khalid et al. 2011; El-Fadel et al. 2013). One of the problems associated with high C/N ratio is that the availability of nutrients will be lower, because nitrogen is regarded as a crucial nutrient for the methanogens. On the other hand, a low C/N ratio would result in increased ammonia production, thereby causing ammonia inhibition.

In an anaerobic digester, ammonia can be found in two forms, namely, unionized ammonia (N[H.sub.3]) or free ammonia and ionized ammonia or ammonium ion (N[H.sub.4.sup.+]). Between these two, the N[H.sub.4.sup.+] ion may inhibit the methane production directly. In general, it was established that TAN concentrations of around 1700-1800 mg/L caused anaerobic digester failure (Albertson 1961; Melbinger and Donnellon 1971). This was emphasized by a batch study conducted in a high-solids AD process, where ammonium-nitrogen concentration was a more significant factor than free ammonia in affecting the methanogenic activity (Lay et al. 1998). The methanogenic activity was observed to decrease with an increase in N[H.sub.4.sup.+]-N over a wide pH of 6.5-8.5. The methane production dropped by up to 10%, at an ammonium-nitrogen concentration of 1670-3710 mg/L; 50% at 4090-5550 mg/L; and to zero at 5880-6000 mg/L. In addition, it was observed that the lag-phase time in the batch experiment was dependent on the ammonia level, but not on the ammonium. It was further noticed that, when the free ammonia-nitrogen concentration was higher than 500 mg/L, a considerable shock appeared, suggesting the free ammonia level to be a more sensitive factor than the ammonium level in a non-acclimatized bacterial system.

The performance of an AD reactor is associated with the structure of the microbial community present within the reactor (Yenigun and Demirel 2013). It was reported in a study by Hobson and Shaw (1976) that the growth of Methanobacterium formicicum was partly inhibited at a TAN concentration of 3000 mg/L, and at a pH of 7.1, and it was completely inhibited at 4000 mg TAN/L. A more recent study by Schnurrer and Nordberg (2008) reported a gradual increase in ammonia concentration beyond 3000 mg TAN/L, resulting in a shift from the acetoclastic methanogenesis to syntrophic acetate oxidation during lab-scale AD of SS-OFMSW, at an HRT of 30 days. During hydrolysis, the first stage of the AD process, the frmentative group of hydrolyzing bacteria deaminate nitrogenous compounds, proteins mainly, present in the organic fraction to produce ammonia. As such, a large amount of ammonia is produced during the hydrolysis stage, in accordance with the stoichiometric relationship (Kayhanian 1999). It has also been established that the effect of ammonia production is greater on the activity of the methanogens, rather than the acetogens or acidogens. Even though nitrogen is an essential nutrient for microbes performing AD of organic wastes, and it is available from ammonia (McCarty and Mckinney 1961a; McCarty and Mckinney 1961b), the excess of ammonia may inhibit the methanogenic activity. The inhibition is primarily due to TAN, which is a combination of free ammonia nitrogen (N[H.sub.3]) and ionized ammonium nitrogen (N[H.sub.4.sup.+]).

The TAN affects the methanogenic activity in the following three ways:

1. The N[H.sub.4.sup.+] ions directly inhibit the methane synthesizing enzyme.

2. The N[H.sub.3] molecules passively diffuse into the cells of the methanogens, owing to their hydrophobic nature, causing proton imbalance and potassium deficiency (Sprott and Patel 1986). This is because the difference in intracellular pH causes the N[H.sub.3] molecules to convert into N[H.sub.4.sup.+] ions by absorbing protons from the cell itself. In turn, this triggers balancing of protons in the cell by expending energy using potassium antiporter (Sprott et al. 1984).

3. The growth rates of hydrogen-utilizing methanogens are severely affected by high concentration of TAN (Wiegant and Zeeman 1986). An increase in TAN concentration results in the fast production of acetate.

In the absence of [H.sub.2]-consuming methanogens, intermediate compounds, propionate and (or) [H.sub.2], accumulate, which affects the production of C[H.sub.4] from the acetate. As the methanogenic activity is inhibited by TAN concentration, the fatty acids produced by the acidogens continue to accumulate resulting in a drop in pH, thereby eventually leading to digester failure. The effect of ammonia molecules into the cellular structure of methanogens was supported by many studies. In a separate experiment by Sprott et al. (1985), the transport of [K.sup.+] ions in M. hungatei pretreated with ammonia was studied and no methanogenic activity was observed. However, the addition of [Ca.sup.2+] or [Mg.sup.2+] once again activated the methanogenic cells, enabling them to make ATP and transport [K.sup.+] ions.

It was proposed by McCarty and McKinney (1961a, 1961b) that the equilibrium concentrations of free ammonia and ammonium ions are dependent on pH and temperature of the solution. Moreover, it was also inferred that the inhibition was mainly due to free ammonia in solution rather ammonium ions. However, in a study showing the relationship between OLR and ammonia inhibition by Melbinger and Donnellon (1971) it was concluded that the methanogenic bacteria could be acclimated to ammonium ions if the feed was gradually increased. As such, in those digesters, TAN concentrations of up to 2700 mg/L may exist without any failure. In a separate study by Gallert and Winter (1997) it was observed that methane production was inhibited by 50% at 220-280 mg N[H.sub.3]-N/L and 680-690 mg N[H.sub.3]-N/L, during AD of glucose, under mesophilic and thermophilic conditions, respectively. It was concluded from the study that the thermophilic flora was able to tolerate ammonia toxicity twice as great as the mesophilic flora. In a later study, it was pointed out that the temperature increase in the thermophilic range of 40-64 [degrees]C had a negative effect on the AD of cattle manure from 700 mg N[H.sub.3]/L onwards, inhibiting methane production (Angelidaki and Ahring 1994). Sung and Liu (2003) observed that ammonium nitrogen (N[H.sub.4.sup.+]-N) concentration in the range of 8000-13 000 mg/L, inhibited 100% methane production, under thermophilic conditions. Thus, it was asserted that the AD of OFMSW at a certain pH and temperature was influenced by ammonia, especially free ammonia, because the amount of N[H.sub.3] is a function of TAN, as per the following equation proposed by Anthonisen et al. (1976):

N[H.sub.3]-N= [N[H.sub.4.sup.+] -N x [10.sup.pH]] / [[e.sup.6344/ 273+T]+[10.sup.pH]]

It can be inferred from this equation that with an increase in temperature and pH, a higher amount of free ammonia is released. Hence, under thermophilic AD it is best to operate at the lowest possible pH, and under mesophilic AD the increase in pH may be afforded. In a study carried out by El-Hadj et al. (2009) it was concluded that during high solids AD of OFMSW ammonia and ammonium could be released at high concentrations. It was also seen that there was 50% inhibition in methane production at 215 and 468 mg/L of N[H.sub.3]-N, under mesophilic and thermophilic conditions. Moreover, it was further observed that the presence of N[H.sub.4.sup.+]-N ions at 3860 and 5600 mg/L, under mesophilic and thermophilic conditions, respectively, inhibited methane production by 50%. Under mesophilic conditions the pH increase was found to negatively affect the biogas generation. Cuetos et al. (2008) also observed that during mesophilic (34 [degrees]C) anaerobic co-digestion of slaughterhouse waste and OFMSW, in a CSTR, at an OLR of 3.7 kg VS/[m.sup.3] per day, the inhibition of methane production started to occur at a TAN concentration of 4100 mg/L and a FAN concentration of 337 mg/L, under a pH of 7.5. Similar results were also observed earlier, when OFMSW was digested in a CSTR at mesophilic and thermophilic temperatures, with ammonia un-adapted inoculum (Gallert and Winter 1997; Gallert et al. 1998). It was noted that, at a pH of 7.5, 220 mg/L of FAN caused 50% inhibition in methane production, under mesophilic conditions, whereas under thermophilic conditions, 690 mg/L of FAN caused 50% inhibition in methane production.

However, the decrease in methane production caused by ammonia inhibition due to high TAN concentration can be removed by lowering the pH and a subsequent decrease of free ammonia concentration. It was reported by Kayhanian (1999) that the inhibition at TAN concentrations of 1200 mg/L during pilot-scale thermophilic AD of OFMSW was mitigated by dilution of the digester content with water, or by adjusting the feedstock C/N ratio, in between 30 and 40. In another study by Shanmugam and Horan (2009) an adjustment of the C/N ratio at 15 and the pH at 6.5, maintained the TAN and FAN concentrations under control, thereby giving the best results in terms of biogas production.

6.9. Retention times

Retention times play a crucial role in dictating the consistency and the efficiency of an anaerobic digester system, treating OFMSW. There are two concepts of retention times in an AD system. These are SRT and HRT. The SRT is the average time that the biomass (solids) remains in the digester. The HRT is the average time that the waste is retained in the digester. Studies have shown that the biogas yield of a digester, treating highly biodegradable organic fraction, such as FW, with longer SRT than HRT, is significantly greater than the biogas yield of a digester, having SRT almost equal to the HRT. This can be attributed to the fact that a longer SRT, in the case of digesters treating FW, results in the regeneration of alkalinity and flushing of excess VFA (Climenhaga and Banks 2008b). Another reason, which asserts for longer SRT than HRT, is that the generation time, which is the time required for a population of bacteria to double in size, of methanogens is relatively long (Gerardi 2003). Systems with high SRT are characterized by biofilm or granule formation inside the reactor, or biomass settling outside the reactor (Angenent et al. 2004). Such systems with high SRT maximize removal capacity, reduce reactor volume, and provide buffering capacity for protection against the effects of shock loadings and toxic compounds present in the OFMSW. Longer SRTs can be achieved by biomass immobilization and retention technique, multistage (two- or three-stage) AD systems, long stagnant periods, followed by liquid wastage for CSTR systems, and by collection of the effluent from the top, with minor removal of solids from the bottom.

6.10. Sorting techniques

Even though the major fraction of the generated MSW comprises organic matter, there are other recyclable fractions, such as plastics, mixed paper, cardboard, iron and aluminum scrap, glass and inert, textiles, etc., which necessitate the application of various separation techniques. This is because the recyclable fractions, other than the organic matter, are not only valuable for sustaining the recycling industries, but their proper separation from the commingled MSW ensures better biodegradability of the OFMSW and also helps to economize the overall separation process. In Europe, with the introduction of S and MS techniques, the AD of OFMSW has received a considerable boost in the last two decades (Dong et al. 2010). The SS technique, as the name suggests, can be defined as a mechanism of procuring the biodegradable fraction of the MSW directly from the source, whereas the mechanical sorting technique relies on the assistance of certain machines to procure the organic feedstock from the MSW.

The biodegradability of OFMSW is one of the crucial components in controlling the AD rate and the application of SS (Angelidaki et al. 2006; Davidsson et al. 2007; Forster-Carneiro et al. 2007, 2008a; Hartmann and Ahring 2005; Ghanimeh et al. 2012; Pognani et al. 2012) and MS (Bolzonella et al. 2003, 2006; Forster-Carneiro et al. 2008b; Fantozzi and Buratti 2011) techniques, while procuring OFMSW have massively contributed to the cause of high biodegradability. In a study by Davidsson et al. (2007) the contribution of source-sorting in the procurement of highly BOFMSW was emphasized. It was observed from the study that the thermophilic pilot scale digestion of 17 types of domestically SS-OFMSW originating from seven full-scale sorting systems had given a VS-degradation rate of around 80%, with a methane yield of 300-400 N[m.sup.3] C[H.sub.4]/ton [VS.sub.in] achieved, for a retention time of 15 days. Similarly, Forster-Carneiro et al. (2008b) analyzed the performances of two lab-scale (5.0 L) reactors anaerobically treating SS-OFMSW (obtained from a university restaurant) and MS-OFMSW (obtained from a municipal treatment plant located in Cadiz, Spain). It was observed that, under thermophilic conditions (55 [degrees]C) and at TS content of 20%, the SS-OFMSW exhibited the classical waste decomposition pattern with a fast start-up phase beginning within 0-5 days, an acclimation stage between days 5 and 30, and a subsequent stabilization phase. In case of the MS-OFMSW, a methanogenic pattern throughout the whole experimental period was observed. The VS removal in the case of the SS-OFMSW was 45% with a cumulative gas production of 120.0 L, whereas in the case of the MS-OFMSW, a higher VS removal rate of 56% was observed, with cumulative biogas production of 82.0 L.

Apart from the SS and MS techniques, there is also another sorting technique that can be applied while procuring organic fractions of MSW and that is the water-sorting (WS) technique (Dong et al. 2010). The WS technique can be defined as a sorting technique that relies on the buoyant and sink forces of water for the separation of the biodegradable fractions of the MSW. The commingled MSW comprises biodegradable fractions, metals, heavy metals, plastics, and combustible fractions. The WS-OFMSW is the biodegradable fraction, which is a mixture of kitchen waste, FVW, garden waste, waste paper, etc. In a study conducted by Dong et al. (2010), it was observed that the biodegradability of the WS-OFMSW, with VS/TS of 61.6%, was superior to the MS-OFMSW but poorer than the SS-OFMSW. This can be attributed to the fact that a major fraction of the WS-OFMSW comprises carbohydrates followed by proteins and lipids, even though the C/N ratio of the WS-OFMSW feedstock is similar to MS-OFMSW. The superiority of the SS-OFMSW is attributed to its high average organic content of 77.7% (in terms of VS) in comparison to MS-OFMSW, which has an average organic content of 43.0% (in terms of VS). SS-OFMSW mostly comprises FVW, whereas MS-OFMSW comprises not only organic matter but a large amount of inorganic material, mainly from soil or sand and small inorganic material (Forster-Carneiro et al. 2008b). This was further asserted in a study by Bolzonella et al. (2006) where it was shown that AD of SS-OFMSW gave a SBP of up to 200 [m.sup.3]/ton of treated waste, with a SMP of some 0.4 [m.sup.3] C[H.sub.4]/kg [VS.sub.feed]. On the contrary, the AD of a combination of MS-OFMSW (35%), grey fraction of MSW (50%), and sewage sludge (15%) gave a SBP of up to 60 [m.sup.3]/ton of waste treated, with a SMP of some 0.13 [m.sup.3] C[H.sub.4]/kg [VS.sub.feed]. However, the VS removal was superior in the latter case (40%-45%) compared to the AD of SS-OFMSW, where the VS removal was 35%-40%.

7. Process design aspect of AD of OFMSW

The rapid development in the various facets of AD, involving stabilization of OFMSW, saw many scientists and researchers come up with several kinetic expressions. Those kinetic formulations were obtained upon conducting various pilot-scale and lab-scale studies.

Mata-Alvarez et al. (1990, 1992) upon conducting two experimental studies established that a first-order kinetic model was relevant in determining the methane yield pertaining to AD of OFMSW. In 1990, the first experimental study dealing with the performance evaluation of anaerobic digesters treating differently sorted OFMSW showed a kinetic constant (k) of 2.95 [day.sup.-1], for SS-OFMSW. On the other hand, a kinetic constant (k) of 0.410 [day.sup.-1] was observed for MS-OFMSW. The second experiment in 1992 resulted in a k-value of 3.1 [day.sup.-1], for SS-OFMSW. A bench-scale study was performed on AD of OFMSW under co-digestion with primary sewage sludge and consequently, a two-stage mathematical model of acidogenesis and methanogenesis was developed and validated. It was observed that anaerobic co-digestion of OFMSW with dewatered sewage sludge, primary sewage sludge and several other pure organic co-substrates, such as vegetable oil, animal fat, cellulose and protein, had a significant impact on the rate of methane production (Kiely et al. 1997). The said model was coded to simulate observed pH, N[H.sub.3], and C[H.sub.4] production also from the experimental work.

In a separate study, Mora-Naranjo et al. (2004) characterized a structured model based on the considerations of coupled reaction processes and interactions between relevant microorganism populations. Liu et al. (2008) predicted the methane yield at an optimum pH, and later on developed a mathematical model that simulated the optimal pH of an anaerobic batch digestion process, treating OFMSW, under varying temperatures. The model was developed on the basis of the microbial growth kinetics and was divided into three types of kinetics, pertaining to hydrolysis, acidification, and methanogenesis, respectively. Monod-type kinetics was used to develop the growth of anaerobic microbes, whereas the consumption of substrates and acids as well as the biomass decay were described by first-order reactions. In another instance, the maximum specific growth rate, [[micro].sub.max], of the anaerobic mixed culture was estimated with the help of an autocatalytic kinetic model developed by Fernandez et al. (2010). They conducted AD of OFMSW at mesophilic temperatures, under different initial substrate concentrations. A kinetic model was developed from this study, which established that the reactor with low TS concentration showed a greater [[micro].sub.max] for the microbes responsible for the biodegradation process, and also the yield coefficient for methane generation was higher. In fact, the experimental results, demonstrated by Rao and Singh (2004) were very much akin to that shown by Fernandez et al. (2010) in terms of a high TS concentration. In the latter experiment, anaerobic batch digestion of municipal garbage at different temperatures and OLRs was conducted, and the process kinetics was developed using a first-order model, based on the limiting substrate condition.

Fdez.-Guelfo et al. (2011d) conducted an experimental study on dry-thermophilic AD and used the substrate consumption model as proposed by Romero Garcia (1991) for determining respective kinetic parameters. Out of various key operational parameters, SRT was varied in the range of 8-40 days, to determine its effect on methane yield. Consequently, the process kinetics was studied and the maximum specific growth rate of the micro-organisms was obtained as 0.580 [day.sup.-1]. The degradation of the VS by AD was observed to follow a first-order reaction kinetics. In another instance, Bollon et al. (2011) developed a kinetic model resembling dry AD of OFMSW and assessed the degradation of acetate. The acetate uptake rate and the half-saturation constant were found to be satisfactorily high at a moisture content of 65%, compared to that of 82%. In fact, the methane production was evidently high at high solid and low moisture content in congruence with the validated model. Particle size and organic content of OFMSW also had a pivotal impact in the overall kinetics of dry AD of OFMSW, at 30% TS concentration and under thermophilic conditions, as reported by Fdez.-Guelfo et al. (2012a). A modified mathematical expression based on the widely validated product-generation kinetic model (Romero Garcia 1991) was proposed while conducting the aforesaid experiment, in a semi-continuous stirred tank reactor. The different kinetic models, as proposed by various researchers, available for AD of OFMSW are shown in Fig. 5.

Increased dependencies on fossil-derived fuels and energy insecurity, due to political instability in major oil exporting countries, have led to growing global environmental concerns. The burning of fossil-fuel leads to a harmful effect on the atmosphere and global climate by the emission of pollutants, such as [CO.sub.x], [NO.sub.x], [SO.sub.x], [C.sub.x][H.sub.x], soot, ash, droplets of tars, and other organic compounds. Under these circumstances, there has been a renewed interest in biological hydrogen production, because it produces water when combusted as a fuel or converted to electricity. Hydrogen, as a sustainable and clean energy source, is a promising alternative to fossil fuel with minimal or zero use of hydrocarbons. It makes up about three-quarters of all organic matter. Different substrates, including OFMSW, have been reported to favor hydrogen production (Okamoto et al. 2000). The evolution of photosynthetic hydrogen from MSW was worked out by Fascetti and Todini (1995). It was observed from the batch and continuous experiments that the acidic aqueous stream obtained from such refuse was a good substrate for the growth of Khodobacter spaeroides RV. It was also observed that substrate from the acidogenesis of FVW gave higher hydrogen evolution rate, as compared to synthetic medium (100 versus 35 mL [H.sub.2]/g dry weight h). According to Hallenbeck and Benemann (2002) there are predominantly three microbial groups that are responsible for producing hydrogen. These are cyanobacteria, which are autotrophs and directly decompose water into hydrogen and oxygen in the presence of light energy by photosynthesis, and heterotrophs, which use organic substrates for hydrogen production. The heterotrophic microbes are responsible for producing hydrogen, under anaerobic conditions, either in the presence or in the absence of light energy. Accordingly, the anaerobic process of hydrogen production may be of two types: (i) photo-fermentation, and (ii) dark-fermentation. Photo-fermentation can be defined as the production of hydrogen carried out in presence of light by photo-synthetic purple non-sulfur bacteria, whereas dark-fermentation is the production of hydrogen carried out in the absence of light by fermentative bacteria, primarily Clostridia.

Chen et al. (2006) studied the growth kinetics of hydrogen in dark fermentation using various substrates including OFMSW, through a series of batch experiments. The modified Gompertz equation was applied to determine the hydrogen production potential (H), hydrogen production rate (R), and lag phase (A). As such, the hydrogen production was well correlated to the modified Gompertz equation ([R.sup.2] > 0.98). The hydrogen yield and the specific hydrogen production rates were calculated from H and added substrate, as well as from R and biomass concentration, respectively. The bio-hydrogen production efficiency was found to be highly related to the optimal control of substrate to biomass (S/X) ratio. It was observed that at a substrate (FW as OFMSW) concentration of 4600 mg COD/L, the hydrogen yield was maximized with 101 mL/g COD. The corresponding hydrogen production potential (H), hydrogen production rate (R), lag phase ([lambda]), and specific hydrogen production rate were 69 mL, 12.1 mL/h, 4 h, and 139 mL/g VSS h, respectively, with [R.sup.2] being 0.9991. Similarly, the maximum specific hydrogen production rate was reached (286 mL/g VSS h) at a substrate concentration of 32300 mg COD/L with H, R, [lambda], and hydrogen yield being 178 mL, 25.0 mL/h, 3 h, and 37 mL/g COD, respectively, with [R.sup.2] being 0.9981. The growth kinetics of hydrogen-producing bacteria for OFMSW (FW) was also studied using the Michaelis-Menten equation, where the dependence of hydrogen production rate (R) on substrate concentration was shown. The half-saturation constant ([K.sub.S]) and the [R.sub.max] obtained were 8700 mg COD/L and 29.9 mL/h, respectively. However, the specific growth rate ([micro]), being 0.568 [h.sup.-1], was observed to be maximized in the case of glucose as test substrate, at a temperature of 37 [degrees]C with Enterobacter cloacae (Kumar et al. 2000).

The production of fermentative hydrogen from organic substrates proceeds simultaneously with the growth of hydrogen producing bacteria (HPB) and certain soluble metabolites. Various kinetic models, such as the modified Gompertz model, have been proposed to describe the substrate degradation, HPB growth, hydrogen production, and some soluble metabolite formation in a batch fermentative hydrogen production process. These models allow the prediction of substrate degradation, HPB growth, hydrogen production, and some soluble metabolite formation at a given time in a batch fermentative hydrogen production process. As such, Zwietering et al. (1990) developed the modified Gompertz model, which has been widely used to describe the progress of substrate degradation, HPB growth, hydrogen production, and some soluble metabolite production in a batch fermentative hydrogen production process. It was observed that the cumulative substrate degradation (H) increased very slowly with increasing cultivation time from 0 to [lambda] (lag time) and then rapidly increased at the rate of [R.sub.max] (maximum rate), finally reaching an asymptotic value [H.sub.max] (maximum cumulative value) with further increase in the cultivation time. In a separate study by Wang and Wan (2008b) the progress of hydrogen production in batch tests using glucose as substrate was determined using the modified logistic model, which is very similar to the modified Gompertz model. A similar model was used by Nathoa et al. (2014) to describe the cumulative hydrogen production, the maximum production rate, and the lag phase duration during two-phase anaerobic fermentation of banana peels. This modified logistic model was previously used by Mu et al. (2006, 2007) to describe the progress of HPB growth in batch tests. Furthermore, it was also inferred by the same team (Mu et al. 2007) that the modified Gompertz model was the most suitable model, after comparing the ability of the modified Gompertz model, modified logistic model, and modified Richards model to describe the progress of HPB growth in batch tests. It was inferred from these studies that the modified Gompertz model was a near omnipotent model, in terms of describing the progress of substrate degradation, HPB growth, hydrogen production, and some soluble metabolite production in a batch fermentative hydrogen production process. It was also concluded that the modified logistic model, which has a similar property as the modified Gompertz model, can also be used to describe the progress of a batch fermentative hydrogen production process (Wang and Wan 2009b).

For the production of fermentative hydrogen any substrate source containing carbohydrate is considered very important, because carbohydrates provide carbon and energy sources for HPB. In a study by Lay et al. (2003) it was concluded that HPB were more effective in producing hydrogen from carbohydrate-rich substrates. Various kinetic models have been proposed to describe the effects of substrate concentrations on the substrate degradation rate, HPB growth, and hydrogen production. The classical Monod model (or the Michaelis-Menten model) is one of them, which shows that the rate (R) increases with the increase in substrate concentration, finally reaching an asymptotic value [R.sub.max]. In addition, it also suggests that at lower substrate concentration (relative to the half-saturation constant, [K.sub.s]), R is approximately proportional to the substrate concentration (first order in substrate concentration), while at higher substrate concentration, R is independent of the substrate concentration (zero order in substrate concentration). However, in the case of inhibition of the fermentative hydrogen produced from the substrate at much higher concentrations, the classical Monod model becomes unsatisfactory and the modified Monod model, such as the Andrew model, becomes appropriate to describe the effects of substrate concentrations on the hydrogen production rate and specific HPB growth rate (Wang and Wan 2009a).

Kumar et al. (2000) used a modified Andrew model to describe the effects of substrate concentrations on specific E. cloacae IIT-BT 08 growth rate, in batch tests. Aguilar et al. (2013) applied a model based on acidogenic substrate as carbon, to study the effect of HRT on hydrogen production and organic matter solubilization in acidogenic AD of OFMSW. Acidogenic substrate as carbon can be defined as the fraction of solubilized organic matter that has not been transformed into VFA and, therefore, may be used to study the behavior of the hydrolytic and acidogenesis phases (Fdez.-Guelfo et al. 2012b). It was inferred that high acidogenic substrate as carbon concentration results in the transformation of high VFA from the higher amounts of solubilized organic matter during the acidogenesis step, eventually leading to higher [H.sub.2] production. The model was developed using classical parameters, such as organic carbon concentration measured by total organic carbon analyzer, each individual VFA concentration measured by gas chromatography, number of carbons of each VFA and the molecular weight of each VFA.

In addition to the models discussed earlier, there are other kinetic models that describe the effect of inhibitor concentrations on hydrogen production, the effect of temperatures on hydrogen production, the effects of pH on fermentative hydrogen production, the effect of dilution rates on hydrogen production, and the relationship among the substrate degradation rate, HPB growth, and product formation. For instance, the modified Han-Levenspiel model has been widely used to describe the effect of inhibitor concentrations on hydrogen production (Wang et al. 2008; Liu et al. 2006; van Niel et al. 2003). Similarly, the Arrhenius model and the Ratkowsky model best describe the effect of temperatures on hydrogen production (Wang and Wan 2008a, 2009b). The Andrew model (Wang and Wan 2009a; Fabiano and Perego 2002) and the Ratkowsky model best describe the effects of pH on the rates of fermentative hydrogen production, substrate degradation, HPB growth, and some soluble metabolite production. The effects of the dilution rate on hydrogen production are best described by the single-substrate model without biomass decay, the single-substrate model with biomass decay, and the dual-substrate model with biomass decay (Chen et al. 2001; Whang et al. 2006). The Luedeking-Piret model and its modified form have been used to describe the relationship between the growth rate of various HPB cultures, such as C. saccharolyticus and Thermotoga elfii (van Niel et al. 2002), and product formation rate. In addition, it has also been used by Mu et al. (2006) to describe the relationship between the rate of substrate degradation and the rates of hydrogen production, acetate production, and butyrate production.

8. Past experiences of AD of OFMSW

With the rapid development of societies, the solid waste generation rate per capita is also increasing and consequently becoming a matter of universal concern to all developing countries. From disruption of environmental sustainability to uncontrolled litter, which is a cause of visual resentment among society dwellers, this uncontrolled increase in solid waste generation rate has led to the exacerbation of various ecological problems. The rapid enhancement in the process of AD of OFMSW is mainly attributed to several successful lab-scale studies conducted by various researchers over the last two decades. Kayhanian and Rich (1995) emphasized the requirement of organic-rich nutrients, such as wastewater treatment plant sludges, dairy manure, and synthetic chemical solutions, in AD of OFMSW. Upon conducting a high-solids lab-scale AD of OFMSW combined with the aforesaid nutrients in four different digesters, under mesophilic temperature range, at a mean OLR of 7.8 g/kg active reactor mass/day and with a mass retention time (mass retention time can be defined as the active reactor mass divided by the total wet mass that is fed each day) of 25 days, stable performance was reported along with improved gas production. However, it was observed that the reactor treating OFMSW with wastewater treatment plant sludge and dairy manure, as a nutrient supplement, exhibited the most effective results.

Rao and Singh (2004) conducted batch digestions of municipal garbage for 100 days at room temperature (26 [+ or -] 4 [degrees]C; average temperature 25 [degrees]C) and at ambient temperature (32 [+ or -] 10 [degrees]C; average temperature 29 [degrees]C) conditions, at TS concentrations varying from 45 to 135 g/L to study the effect of organic solids concentration and digestion time on biogas yield. The said team also observed the net bio-energy yield from municipal garbage and corresponding bioprocess conversion efficiency, over the span of the digestion time as 12 528 kJ/kg VS and 84.51%, respectively. It was concluded that the digesters should preferably be run at 67.5 g TS/L under ambient temperature, with a digestion time close to 60 days for optimum energy yield. Hartmann and Ahring (2005) investigated AD of OFMSW using two 4.5 L lab-scale reactors, reactor 1 and reactor 2, with an active volume of 3.0 L, operated under thermophilic conditions (55 [degrees]C). Initially OFMSW was co-digested with manure, with a successively increasing concentration of OFMSW, at a HRT of 14-18 days, with an OLR of 3.3-4.0 g VS/L per day until the adaptation of the co-digestion process was established. Gradually, over a period of 8 weeks, the ratio of OFMSW to manure was slowly increased to 100% in reactor 1, in addition to recirculating the process liquid. In reactor 2 the co-digestion ratio was maintained as 50% (V/V). Both the reactors showed stable operation with significant gas production.

Forster-Carneiro et al. (2007) investigated the effect of inoculum source on thermophilic AD of separately collected OFMSW, by analyzing the performance of laboratory-scale reactors (volume of 1.1 L), using six different inoculums sources: (i) corn silage; (ii) restaurant waste digested mixed with rice hulls; (iii) cattle excrement; (iv) swine excrement (SWINE); (v) digested sludge (SLUDGE); and (vi) SWINE mixed with SLUDGE (1:1) (SWINE/ SLUDGE). These wastes were subjected to AD with the operating conditions, such as 25% inoculum, 30% TS, and 55 [degrees]C temperature. The SLUDGE was reported to be the best inoculum for thermophilic AD of OFMSW, in respect of methane yield (L C[H.sub.4]/g VS), followed by SWINE/SLUDGE and SWINE, against an operating period of 60 days. In a separate study, Forster-Carneiro et al. (2008b) again highlighted the influence of sorting technique during AD of OFMSW, operated under batch mode in thermophilic range and dry (20% TS) condition. Two lab-scale reactors (5.0 L each) were used for treating the SS-OFMSW, obtained from a university restaurant and MS-OFMSW, obtained from a municipal treatment plant located in Cadiz, Spain. The results showed that the reactor treating SS-OFMSW had a cumulative biogas production of 120 L, in approximately 60 days and the other reactor treating MS-OFMSW had a comparatively lower biogas yield of 82 L, in 60 days, thereby inferring the superiority of the source sorting technique in terms of biogas yield.

Fernandez et al. (2008) investigated the influence of TS contents during mesophilic AD of OFMSW, using lab-scale batch reactors of 1.7 L capacity, during a period of 85-95 days. The two reactors were operated under two different OLR of 931.1 mg DOC/L (20% TS) and 1423.4 mg DOC/L (30% TS) and the observed results indicated that the reactor with low OLR had a superior methane production at the end of methanogenic phase. Alvarez and Liden (2008) made a lab-scale study of the potential of semi-continuous mesophilic AD for the treatment of solid slaughterhouse waste, FVWs, and manure in a co-digestion process. The OLR was examined initially, and it was found that OLR in the range of 0.3-1.3 kgVS[m.sup.-3][day.sup.-1] resulted in a stable performance with a methane yield of 0.3 [m.sup.3][kg.sup.-1] VS added and methane content of 54%-56% in the biogas. Subsequently, with the gradual increase in the OLR the methane productivity decreased, indicating organic overload or insufficient buffering capacity in the digester. In another study, Forster-Carneiro et al. (2008b) analyzed the bio-methanization process of FW from a university campus restaurant, in six reactors, with three different TS percentages (20%, 25%, and 30% TS) and two different inoculum percentages (20%-30% of mesophilic sludge), initially using a lab-reactor with volume 1100 mL. After validation of the operational parameters, another lab-scale batch reactor, with volume 5 L, was employed. The observed results showed best performance with respect to methane generation for the reactor with 20% of TS and 30% of inoculum under an acclimation period between 20 and 60 days for acidogenic/acetogenic activity. It was inferred that an enhancement of the start-up phase for dry thermophilic anaerobic digester treating organic FW was essential.

Cuetos et al. (2008) also investigated anaerobic co-digestion of OFMSW with slaughterhouse waste in a laboratory plant, operated under semi-continuous mode and mesophilic temperatures. The study showed that AD was a suitable technology for efficiently treating lipid and protein waste. An HRT of 50 days and an OLR of 0.9 kgVS[m.sup.-3][day.sup.-1] was found essential for the efficient start-up and sludge acclimation. Consequently, the HRT and OLR were adjusted to 25 days and 1.70 and 3.70 kgVS[m.sup.-3][day.sup.-1], respectively, in steps, and stable operation with 83% fat removal was reported, in addition to undetectable traces of VFAs and long chain fatty acids. A comparative study for finding out the effect of atmospheric pressure on the performance of AD of OFMSW was made by Jiang et al. (2010) using two lab-scale reactors, treating OFMSW under two different pressures: 101 and 65.8 kPa. The process performance was evaluated on the basis of gas production, gas composition, VS degradation, pH, alkalinity, VFAs, and ammonia concentration, under varying OLR. It was observed that pressure had an obvious effect on the gas production rate, because the high pressurized system produced more biogas than the lower one. However, at a low HRT, pressure had no obvious effect on the gas production rate. Thus, the study revealed the feasibility of AD of OFMSW in high altitude areas, where there is a comparatively low atmospheric pressure. Fdez.-Guelfo et al. (2011c) carried out a labscale study of the stabilization of industrial OFMSW using semi-continuous completely mixed reactor and inferred that the biogas and methane production considerably increased, in the case of biological pretreatment with mature compost.

Apart from lab-scale studies, several pilot-scale studies were also carried out for the development of AD of OFMSW. Bolzonella et al. (2003) conducted a pilot-scale study on the simulation of the start-up phase of the thermophilic semidry AD of OFMSW, to aid and shorten the start-up phase. MS-OFMSW enriched with the putrescent fraction from the SS-OFMSW was used in this experiment, with a view to simulate the substrate, which was dealt with in the original plant. The results showed a specific gas production of 0.23 [m.sup.3]/kg[TVS.sub.feed], and a gas production rate of 2.1 [m.sup.3]/[m.sup.3]d, when operated at a specific OLR of 0.135 kg [TVS.sub.feed]/kg [TVS.sub.reactor] day. Fernandez et al. (2005) studied anaerobic co-digestion of fats of different origin (animal fat, vegetable fat) with OFMSW in a pilot plant, operated under semi-continuous mode, in the mesophilic range (37 [degrees]C) and with an HRT of 17 days. They observed that the biogas and methane yield after substituting the simulated OFMSW, which was used for starting up of the digester, with animal fats and subsequently with vegetable fats were appreciably similar to those of the simulated OFMSW. This was attributed to the fact that both the animal fat and the vegetable fat were similar to that of the simulated OFMSW, in relation to the long-chain fatty acid profile. Davidsson et al. (2007) studied the effect of sorting and collection systems on the amount of methane produced, by conducting thermophilic pilot-scale AD of 17 types of domestically SS-OFMSW. Each type of waste was identified by its origin and by pre-sorting, collection, and pre-treatment methods. The results reported a VS degradation rate of around 80%, and a methane yield of 300-400 N[m.sup.3] C[H.sub.4]/ton [VS.sub.in], with a retention time of 15 days.

Subsequently, Nopharatana et al. (2007) conducted a series of batch experiments on AD, whereby the soluble and insoluble fractions, and unwashed MSW were separately digested in a 220 L stirred stainless steel vessel, at a pH of 7.2, under mesophilic temperature regime (38 [degrees]C). It was found that 7% of the total COD arising out of MSW was readily soluble, of which 80% was converted to biogas, and 50% of the insoluble fraction was solubilized. Kim et al. (2008) established the practical feasibility of a three-stage fermentation system, on a commercial level to treat OFMSW, thereby contributing to the cause of effective disposal and volumetric stabilization of the same. A volumetric scale-up of this system was designed and built on a pilot-scale, providing a total digester volume of 10 [m.sup.3]. The results from the pilot-scale experiments were compared with those from the bench-scale study, utilizing a total digester volume of 0.4 [m.sup.3], prior to field scale-up. The total COD reduction was reported to be 90.6% and the maximum methane content in the produced biogas was reported to be an impressive 72%, for the bench-scale system. On the other hand, the field pilot-scale system showed COD reduction and methane content as 90.1% and 68%, respectively. Fdez.-Guelfo et al. (2010) also evaluated the start-up and stabilization of AD of OFMSW, using a bench-scale configuration that consisted of a CSTR, semi-continuously fed with OFMSW, at thermophilic temperature range and dry conditions (30% TS). The operating conditions (thermophilic and dry) and the type of waste were adapted with the help of an inoculum from a modified SEBAC (sequential batch anaerobic composting) system. The process was started up directly at thermophilic temperature regime and a significant reduction in the start-up time and stabilization was achieved within 110 days, in comparison to 250 days for the processes reported elsewhere for the same type of waste and digester.

In a bench-scale study by Fantozzi and Buratti (2011), AD and anaerobic biogasification potential were tested on three different samples of OFMSW from waste separation, one as received in original, and two obtained after mechanical treatment (squeezing): OFMSW slurry (liquid fraction) and OFMSW (residual solid fraction). The effects of inoculum and pH were investigated, and it was inferred that the OFMSW slurry must be diluted with inoculation, and pH control in the start-up phase must be ensured to have significant biogas production. The effect of AD while treating extruded OFMSW was also investigated by Novarino and Zanetti (2012) by carrying out pilot-scale tests in semi-continuous conditions with a stepwise progressive increase of the TS content in the input material from 3% TS w/w to 10% TS w/w, using activated sludge as a diluting agent. The reactor with 10% TS content produced higher specific biogas and superior VS removal, inferring strong degradation. Zeshan et al. (2012) studied the effect of ammonia-N accumulation in dry AD of OFMSW, using a pilot-scale thermophilic reactor. Biodegradable feedstock, such as FW, FVW, green waste, and paper waste, were used to prepare two simulations of C/N ratio: 27 and 32. The process performance, in terms of pH, VFA, alkalinity, ammonia-N, and biogas yield was noted and the results showed that the simulation with C/N ratio 32 had about 30% less ammonia in the digestate, compared to that with C/N ratio 27. The system was observed to perform well up to OLR 7-10 kg VS/[m.sup.3]d, against a retention time up to 19 days, with surplus energy production of 50%-73%. Thereafter, Ganesh et al. (2013) studied the impact of stepwise increase in OLR (up to 7.5 kg VS/[m.sup.3]d) on the methane production, reactor performance, and solubilized organic matter production, using high-loaded bench-scale reactors. Specific methane yield was higher at the lowest OLR and vice versa.

Recently, in in small localities, municipal towns, and cities there have been various full-scale applications of AD for stabilization or volume reduction of the large quantum of generated solid waste, and energy recovery (in the form of biogas, C[H.sub.4], [H.sub.2]) from it. In developing countries, especially in the European Union, many anaerobic treatment plants, treating OFMSW, have been developed in the last decade. In an overview of the state-of-the-art development of AD of OFMSW, De Baere and Mattheeuws (2008) outlined various sets of criteria to be followed by European treatment plants. It was mentioned in the overview that at least 10% of organic solid waste from household origin was required to be treated in the plant, with a minimum capacity of 3000 tons per year. They also reviewed a variety of considerations regarding the smooth and sustainable operation of the treatment plants. It was concluded that substantial improvements were to be made in terms optimization and pre- and post-treatment of the end products. The anaerobic systems used in those plants were predominantly batch reactors and one-, two-, or multi-stage continuously fed systems, but also included plug-flow reactors and three-stage continuous flow reactors (Khalid et al. 2011). Those advanced bio-reactor systems have significantly improved the rate of reaction for the treatment of the OFMSW (Bouallagui et al. 2003; Mumme et al. 2010; Xing et al. 2010). In addition, "co-fermentation" or "co-digestion" is preferred for the numerous benefits pertaining to the yields of AD of OFMSW (Khalid et al. 2011). The various wastes that are mixed or co-digested with the municipal organic solid waste include fat, oil, and grease waste from sewage treatment plants (Martin-Gonzalez et al. 2010), FA (Lo et al. 2012), abattoir wastewater (Bouallagui et al. 2009), sewage sludge (Sosnowski et al. 2003), and slaughterhouse waste (Cuetos et al. 2008). In the City of Karlsruhe, Germany, AD of 7200 tons per year of separately collected bio-waste at an OLR of up to 8.5 kg COD [m.sup.-3][d.sup.-1], proved feasible for co-digestion. Gallert et al. (2003) examined the effectiveness of AD, in terms of volume reduction and process stability, by further increasing the bio-waste to 12000 tons per year at a higher OLR of 15 kgCOD [m.sup.-3][d.sup.-1]. In accordance with the lab-scale results, the OLR in the full-scale plant was increased to 15 kgCOD [m.sup.-3][d.sup.-1], and even after 7 months of operation, the performance of the full-scale digestion plant was consistent with the previously obtained laboratory results. Based on the preceding literature, the process conditions of a few important pilot-scale operations of AD of OFMSW are summarized in Table 1. In addition, the effects of sorting technique on gas production and volume reduction, in AD of OFMSW are presented in Table 2.

AD of OFMSW produces [H.sub.2] in the acidogenesis step. Under anaerobic dark conditions using a conventional AD process, the production of [H.sub.2] is carried out by the fermentative bacteria to dispose of excess reducing equivalents contained in the organics. During anaerobic light conditions, photosynthetic fermentative bacteria produce [H.sub.2] as a by-product of activity related to nitrogenase, which is a common enzyme that is necessary for [N.sub.2] fixation (Kim and Kim 2011). The main advantages of dark fermentation include a faster [H.sub.2] production rate than photo-fermentation and the direct usage of complex and problematic organic wastes as a substrate, resulting in enhanced economic viability (Kim et al. 2009). However, the disadvantage is that it includes the incomplete decomposition of substrates leading to an accumulation of organic acids. This, in turn, limits the [H.sub.2] yield to less than 20% of the theoretical maximum value of 12 mol [H.sub.2]/mol glucose (D.H. Kim et al. 2011). In contrast, photo-fermentation allows for near-stoichiometric conversion of organic acids to [H.sub.2]. Thus, the overall increase in the [H.sub.2] yield is possible through a sequential fermentation system consisting of both dark- and photo-fermentation. A recent attempt of this process by Liu et al. (2010) and Su et al. (2009) resulted in a significant improvement of the [H.sub.2] yield, approaching 4-6 mol [H.sub.2]/mol glucose. However, it was observed from these studies that the yield still represented less than 50% of the theoretical maximum value, which may be attributed to the fact that the main organic acids, acetate and butyrate, produced by dark-fermentation, were not favorable substrates for the subsequent photo-fermentative [H.sub.2] production. This problem was eliminated by a three-stage (lactate- + photo-[H.sub.2] + C[H.sub.4]) fermentation system developed by Kim and Kim (2013), which converted FW to C[H.sub.4] and [H.sub.2], with an added emphasis on achieving high [H.sub.2] yield. The system consisted of three-stages, starting with the fermentation of FW to lactate, using indigenous lactic acid bacteria, followed by the centrifugation of the lactate fermentation effluent. In the second stage, the supernatant obtained after centrifugation was used for [H.sub.2] production by photo-fermentation, and in the third stage the residue from the centrifugation was used for C[H.sub.4] production by AD. It was observed that 41% and 37% of the energy content in the FW was converted to [H.sub.2] and C[H.sub.4], respectively. This was corresponding to the electrical energy yield of 1146 MJ/ton FW, which was 1.4 times higher than that of previous two-stage dark ([H.sub.2] + C[H.sub.4]) fermentation system.

9. Scope of energy yield from AD of OFMSW

Conserving natural resources is becoming imperative with the progress of time. The dependency on fossil fuels for various industrial and commercial activities needs to be curtailed as a part of implementation of the conservation programme. This will not only help in minimizing the depletion of exhaustible resources, thereby fostering the usage of various inexhaustible resources (e.g., wind energy, solar energy, tidal energy, geothermal energy, hydro-electricity, etc.) at a large scale, but will also mitigate the various climatic changes caused by the emission of GHG. According to the second phase of the Kyoto Protocol (2012), the industrial countries should ensure reduction of GHG emissions by at least 18% below the levels found in 1990, during the 8 year period from 2013 to 2020. This objective can be achieved by using technologies that are non-polluting, inexpensive, and are based on the previously mentioned inexhaustible sources of energy (Saxena et al. 2009). AD of OFMSW can be considered as one of the most efficient modes of renewable energy generation in this regard. Since its inception, it has become a widespread and reliable technology, not only in terms of its adeptness in striking a balance between generation and stabilization of MSW, but also in terms of biogas production. Moreover, the sludge produced from AD can be used as a quality fertilizer and soil conditioner. Because there is no dearth of MSWs, as those are generated in huge proportions daily across the world, sustaining this technology would not be an issue if its various operational stages are managed properly.

As the process of AD of OFMSW is becoming increasingly popular worldwide, different modifications are being incorporated to enhance the extent of energy recovery. It has been observed that the important process parameters influencing the rate of energy generation are OLR, pH, alkalinity, particle size, total (TS), temperature, pressure, moisture content, VFA production, VS added, and nature of the co-substrate or inoculum in the case of co-digestion. Sensitive methane-producing bacteria depend heavily on all of the parameters as well as a few others, as reported by scientists and researchers since the inception of AD of OFMSW. The TS concentration also greatly influences the methane yield.

Brummeler and Koster (1989) conducted dry AD of OFMSW at a low TS concentration and a seed/substrate solids ratio of 0.08. They reported an abrupt increase in the yield of methane per kilogram of organic fraction when several pH control chemicals were added at a certain buffer/substrate solids ratio. Mata-Alvarez et al. (1992) performed an experimental study on AD of market organic wastes in central Barcelona and confirmed that the production of methane is higher from bio-methanization of the organic fraction of the FWs from a vegetable market, compared to other MSWs. The FWs obtained from the marketplace comprised a higher percentage of biodegradable material in comparison to other domestic MSW, which necessarily had a significant percentage of non-biodegradable components (peels, bones, etc.). As mentioned earlier, the nature of the organic substrate has an important influence on the VS removal and the DOC removal along with the methane yield. Kayhanian and Rich (1995), while conducting a pilot-scale study on high solids thermophilic AD of OFMSW, emphasized the need for macro- and micro-nutrients requirement for robust and stable digestion, along with significantly high gas production. The nutritional deficiencies were supplemented by the addition of various nutrient-rich organic wastes (co-substrates), such as wastewater treatment plant sludge, dairy manure, and synthetic chemical solutions, to the substrate (OFMSW), that enhanced process stability and biogas yield. Stroot et al. (2001) and McMahon et al. (2001) studied the influence of mixing on the high solids AD of OFMSW with pre-thickened sewage sludge from wastewater treatment plant, and both studies observed that biogas production was enhanced under reduced mixing condition and high OLR.

It is a well-established fact that anaerobic co-digestion of OFMSW with various co-substrates has the potential to significantly contribute to biogas production and operational stability. For example, experimental studies by Lastella et al. (2002) observed enhanced biogas production and its methane content when co-digesting semi-solid ortho-fruit waste with the recycled digested sludge, under anaerobic conditions and a maximum retention period. Rao and Singh (2004) observed that a digestion time of 60 days and a TS concentration of 67.5 mg/L was essential for optimum biogas yield, while conducting AD of municipal garbage at ambient temperatures (32 [+ or -] 10 [degrees]C; average temperature 29 [degrees]C) by Rao and Singh (2004). Methane productivity per kilogram of COD input was also found to increase by increasing the OLR during the acidification step, while conducting two-phase AD of a mixture of FVW in two coupled anaerobic sequencing batch reactors, operated at mesophilic temperature (Bouallagui et al. 2004). Furthermore, Hartmann and Ahring (2005) also investigated anaerobic co-digestion of OFMSW with manure and 100% AD of OFMSW, under mesophilic conditions and reported that recirculation of the generated leachate from one reactor to another resulted in stable reactor performance with significant VS reduction (while treating 100% OFMSW) as well as considerable biogas yield per ton of waste. Thereafter, Fernandez et al. (2005) conducted pilot-scale studies of anaerobic co-digestion of simulated OFMSW with fats of animal and vegetable origin (high lipid content) and concluded that after a particular adaptation period with simulated OFMSW, the co-digestion with fat increases the amount of biogas produced per kilogram of TVS degraded. Similar results were obtained by Cuetos et al. (2008) when anaerobic co-digestion of slaughterhouse waste and OFMSW were conducted. However, they concluded that for anaerobic co-digestion to happen successfully at a low HRT and high OLR, the inoculum needed to be acclimated to a high fat and high ammonia content medium for a longer HRT and low OLR at the start-up phase, and then switching to low HRT and high OLR should be followed once the digester was stabilized.

Angelidaki et al. (2006) found that using a low fresh TS content resulted in the maximum methane yield and best degradation rates of SS-OFMSW, after carrying out anaerobic batch digestion of SS-OFMSW under thermophilic conditions. Experimental studies on three-stage methane fermentation of OFMSW by Kim et al. (2006) revealed that biogas and methane production rates in thermophilic digesters were higher than those in mesophilic digesters, irrespective of HRT, even though the results indicated that AD of FW yielded appreciable methane under a thermophilic temperature range compared to a mesophilic temperature range, with a 10-day HRT. It was also observed from the study that liquor FW showed higher removal of soluble COD under thermophilic conditions with the highest removal achieved at 50 [degrees]C, thereby indicating the influence of temperature on the soluble COD removal. Nopharatana et al. (2007) conducted an experimental study pertaining to batch-slurry AD, where the soluble and the insoluble fractions along with the unwashed MSW were digested separately. This study observed that the particle sizes between 2 and 50 mm did not affect the rate of digestion and methane yield. In addition, it was concluded that the rapidity of digestion of the insoluble fractions can be increased by acclimatizing the culture with soluble fractions having higher biodegradable potentials followed by digestion of the insoluble fractions. Capela et al. (2008) confirmed that there was a considerable increase in the methane production and VS removal when the proportion of OFMSW (substrate) was increased during anaerobic co-digestion with industrial sludge and cattle manure, under mesophilic conditions. Forster-Carneiro et al. (2008b) carried out thermophilic AD of different organic matter like FW, SS-OFMSW, and OFMSW and observed that the methane yield was higher for FW. They also deemed that SS-OFMSW was superior, in the production of biogas than MS-OFMSW by analyzing the performance of two lab-scale reactors treating SS-OFMSW and MS-OFMSW under thermophilic temperatures and at a constant TS concentration.

In a study by Fdez.-Guelfo et al. (2010) it was observed that the thermophilic dry AD (at 30% TS) of OFMSW resulted in greater methane production than wet mesophilic AD of OFMSW, at 5%-10% TS conditions. Dong et al. (2010) observed an impaired methane yield once again with low TS concentration while conducting semi-dry mesophilic AD of WS-OFMSW. It was concluded that, in terms of methane yield, WS-OFMSW remained between MS-OFMSW and SS-OFMSW. At the same time, Schievano et al. (2010) showed that HSAD of OFMSW resulted in a better yield of biogas per kilogram of VS, only when the putrescibility ([O.sub.2] consumption in 20 h biodegradation, [OD.sub.20]) of the organic fraction was under a certain value. Not only the nature of the waste, but also pretreatment of the waste, is vital in the context of greater biogas yield. Fdez.-Guelfo et al. (2011c) revealed that biological pretreatment of industrial OFMSW with mature composts, followed by dry thermophilic AD, yielded the highest cumulative biogas and methane production. Later on, Fantozzi and Buratti (2011) conducted AD of MS-OFMSW and showed that biogas production was improved when the OFMSW slurry was diluted and inoculated. Novarino and Zanetti (2012) performed the pilot-scale AD of extruded OFMSW (treatment of OFMSW by exerting pressure so that the undesired fractions of waste get separated from the organic waste) and concluded that the biogas-specific production tended to enhance by increasing the TS content of the input material. Anaerobic co-digestion of FW, FVW, and dewatered sewage sludge was conducted by Liu et al. (2012a) and it was reported that a high biogas production rate of 4.25 [m.sup.3][([m.sup.3]d).sup.-1] was obtained, at an OLR of 6.0 kg VS [([m.sup.3]d).sup.-1]. It was also observed that a total of 16.5% of lipids content was beneficial to the biogas production of the feedstock without causing inhibition to the AD. Zhang and Banks (2012) co-digested mechanically recovered OFMSW with slaughterhouse wastes and observed that it was useful in achieving a volumetric methane productivity of 1.25 STP [m.sup.3][m.sup.-3][d.sup.-1], at a loading rate of 4 kg VS [m.sup.-3][d.sup.-1].

Zhang and Banks (2013) studied the impact of different particle size distributions on AD of OFMSW and it was concluded that differences in the particle size distribution did not change the specific biogas yield; instead the digester performance was affected. It was also found that the SMP in the "dry" digesters was slightly lower than in the "wet" digesters when processing the same material. Zeshan et al. (2012) studied the effect of C/N ratio and ammonia-N accumulation in a pilot-scale inclined thermophilic dry anaerobic digester. It was observed that the inclined thermophilic dry anaerobic digester performed better with the various OLRs and produced 200-300 L C[H.sub.4]/kg

VS. However, the free ammonia concentration was found to inhibit the steady-state methane production in inclined thermophilic dry anaerobic digester. Ganesh et al. (2013) investigated the impact of stepwise increase in OLR, up to 7.5 kg VS/[m.sup.3]d, on methane production in a high-loading reactor. It was observed that the volumetric biogas and methane production increased significantly with the increase in the OLR.

10. Applicability of AD for management of MSW under Indian scenario

The gradual improvement in the economic growth coupled with the rapid urbanization has led to a high standard of living, even in a developing country like India. This has invariably resulted in an increased rate of annual MSW generation. Table 3, (status of MSW generation, collection, treatment, and disposal in class-I cities, CPCB 2000b) representing the MSW generation rates in different states in India, gives a clear idea regarding the need for an efficient MSW management strategy. In this context, AD can not only strike a balance between the generation and stabilization rate of the MSW but can also cater to the energy needs of a society, especially in a developing country like India. It is evident that in the developed countries, such as those within the European Union, where the per-capita generation rate of MSW is much higher (in the range of 1.5-2.0 kg/day), AD is a must for the stabilization of the generated MSW. In a developing country like India, AD is also equally necessary because the population is much higher and the land needed for other classical management strategies is decreasing day-by-day. Apart from that, the effectiveness of AD largely depends on the physical, chemical, and biological characterization of the generated MSW. The selection of AD over composting, sanitary landfilling, and thermal incineration is dictated mainly by the organic nature of the MSW. The physical and chemical characterizations of MSW in various Indian cities are shown in Figs. 6 and 7, respectively (NEERI Report (1996) strategy paper on SWM in India, 1995). Table 4 (Status of solid waste generation, collection, treatment and disposal in metro cities, CPCB 2000a) highlights the differences in MSW characteristics, with the effect of urbanization and development in various Indian metrocities. The major fraction of MSW in urban areas is compostable matter (40%-60%) and inert materials (30%-50%). However, there is a gradual increase in the relative percentage of organic matter present in the MSW, with a decrease in socio-economic status (Sharholy et al. 2008). Apart from that, in the various developed cities and towns, the rapid increase in the number of marketplaces (fish market, fruit and vegetables market, etc.) is also massively contributing to the quantity of organic fraction in the generated MSW, on a daily basis. AD is mainly concerned with the presence of organic material in the MSW to be treated. Hence, it can be a suitable methodology for treatment of the OFMSW in various suburban, as well as urban, areas of India.

In many metropolitan cities and towns of India, there are numerous scenarios of open and uncontrolled dumping of MSW. More than 90% of the generated MSW is directly disposed of on land in an unsatisfactory manner (Das et al. 1998). The shortage of land in the large and congested towns or cities, like Delhi, also contributes to unsatisfactory landfilling activities (Mor et al. 2006; Siddiqui et al. 2006; Sharholy et al. 2006; Gupta et al. 1998; Kansal et al. 1998). Selection of landfill sites far away from these cities may upset the viability of the operation owing to significant increase in transportation costs (Singh et al. 2011). The principles of sanitary landfilling are seldom followed as the generated MSW is disposed of by filling in low-lying areas (Singh et al. 2011; Sharholy et al. 2008). Unlike the developed countries, in developing countries, such as India, the system of landfilling is neither properly managed nor is it techno-economically feasible. The important practices pertaining to landfilling operation, such as lining of suitable membrane liners along the excavated portions, compaction and levelling of waste, final covering by earth material, etc., are rarely observed. Absence of suitable leachate collection system leads to contamination of ground and surface water (Bhide and Shekdar 1998; Gupta et al. 1998). In coastal towns, improper and unsatisfactory landfilling leads to rapid leaching of heavy metals and toxic substances into the coastal waters (Singh et al. 2011; Sharholy et al. 2008). Also, the absence of proper landfill gas monitoring and collection equipment may sometimes lead to spontaneous ignition or explosions, due to high concentrations methane buildup in the atmosphere (Singh et al. 2011). Apart from that, the uncontrolled emission of the GHG into the atmosphere significantly contributes to global warming. In some instances, landfill sites meant for domestic waste are also found to be filled with industrial waste (Datta 1997). In India, landfilling is a widely practiced mode of MSW disposal. Hence, the previously mentioned problems project landfilling as a not-so-efficient technology in the management of MSW. Therefore, a suitable methodology, such as that of AD, is the most obvious and effective substitute for landfilling, particularly for future scenarios in India.

Also, other processes of MSW management, such as composting, thermal incineration, and pyrolysis or gasification, have not been able to completely solve the problems associated with the comprehensive management and handling of MSW. In the case of pyrolysis or gasification, which necessarily takes place under sub-stoichiometric condition (gasification) or in the complete absence of oxygen (pyrolysis or destructive distillation), the production of the gaseous product largely depends on the nature of the MSW to be treated. MSW has been reported to constitute 15%-63% of FW worldwide (Kim et al. 2009). FWs are characterized by moisture content as high as 75%-93%. Excessive moisture adversely affects the net energy recovery. As such, the OFMSW usually possesses a high quantity of moisture, which is why pyrolysis or gasification cannot be looked upon as a superior method for the stabilization of the OFMSW. Moreover, the generated pyrolysis oil, by virtue of its high viscosity, often poses different problems during its transportation and burning (Singh et al. 2011). In contrast to pyrolysis or gasification, thermal incineration requires auxiliary fuel support to sustain the process of combustion. The thermal energy produced can only be utilized in heating boilers. Moreover, there is an involvement of high capital and operation and maintenance cost, in addition to the requirement of skilled personnel. This is because the control of the release of harmful gases and particulate matter into the atmosphere requires well-equipped incinerators and other appropriate pollution control accessories. In addition, the overall efficiency in the case of small power stations is extremely low. Unlike thermal incineration, the process of AD does not require any external supply of oxygen, nor is there any chance of noxious gas release into the atmosphere and the scope of marketing the end products is also positive. Hence, compared to sanitary landfilling and thermal incineration, the techno-economic viability of AD can be regarded as the best option of MSWM in a developing country like India.

Fig. 6. Physical characterizations of MSWs in Indian cities withpopulation range above 5 million.paper 4.71%rubber, leather and synthetics 0.71%glass 0.46%metals 0.49%total compostable matter 38.95%inert 9.95%others 44.73%Note: Table made from pie chart.Fig. 7. Chemical characterizations of MSWs in Indian cities withpopulation range above 5 million.moisture 38.72%zorganic matter 39.07%nitrogen as total nitrogen 0.56%phosphorous as 0.52%phosphorous pentoxidepotassium as potassiumoxide 0.52%others 20.61%Note: Table made from pie chart.

Since the 1960s the process of composting has gained considerable importance in India. This practice was encouraged in the early initiatives of the Government of India in the management of MSW in urban areas (Sharholy et al. 2008). Composting can be defined as the biological decomposition of the biodegradable OFMSW under controlled conditions to a state sufficiently stable for nuisance-free storage and handling, and for safe use in land applications. Composting has the ability to reduce waste volume by 50%-85% (Sharholy et al. 2008). Even though a considerable number of treatment plants have been built in various key places in India, such as Delhi, Bangalore, Ahmedabad, Hyderabad, Bhopal, Lucknow, Gwalior, Indore, Baroda, Mumbai, Kolkata, Jaipur, and Kanpur, the long-term acceptability of the process is on a gradual decline because of several inherent problems. This is attributed to the facts that a large area is required for the construction of a composting plant, external supply of oxygen is required in the case of aerobic composting, and leachate management is required in some cases. Apart from that, other important issues associated with composting include improper control, in the case of windrow composting, leading to spreading of biological odors and improper management, leading to the potential existence of pathogens in the marketed products. In this context, involvement of shredders may lead to heavy metal toxicity in the marketed products. Unlike AD, composting does not facilitate the production of methane and hydrogen. It is evident from the literature that composting in India constitutes only 9% of the MSW treatment. Hence, there is a definite possibility of AD being perceived as a better and more efficient technology, thereby substituting the process of composting.

11. Overall future prospect of AD of OFMSW

As discussed earlier, since the discovery of MSW undergoing AD in sanitary landfills in the early 1970s, the process witnessed several improvements in the next decade and a half (i.e., between mid-1970s and 1990). This invariably resulted in the development of different systems, such as "Valorga" (de Laclos et al. 1997), "Kompogas" (Wellinger et al. 1993), "Dranco" (Six and De Baere 1992), "BRV" (Lissens et al. 2001), and "BTA" (Verma 2002; Rahn and Gandolfi 2007), capable of treating the solid organic waste in an anaerobic environment. Further, pilot- and lab-scale studies based on those processes eventually led to the construction of numerous commercial plants across Europe (Netherlands, Switzerland, Belgium, Italy, Spain, Germany, UK, France, etc.) and North America (USA, Canada). The crucial modifications, which essentially proved to be imperative in the development of this process, include shifting to the thermophilic temperature regime from the mesophilic temperature regime, improvement of the startup phase by employment of certain inoculum, application of several types of enzymes for increasing the rate of degradation of the OFMSW during the hydrolysis stage, subjecting the OFMSW to different types of pre-treatment prior to the main digestion process, and adoption of several types of sorting techniques (SS, MS, WS) in segregating or separating the OFMSW prior to treatment. Another major modification of AD for the stabilization of OFMSW was co-digestion. At the beginning, when OFMSW was digested alone, there was a noticeable failure regarding the gas production. The importance of co-digestion was not discovered until the studies conducted by Golueke and McGadhey (1971) and Pfeffer and Liebmann (1976) demonstrated the advantages of mixing MSW and sewage sludge, with regard to improving the process efficiency and enhancing the gas production (Cecchi and Traverso 1986). The importance of co-digestion or mixing of the OFMSW (substrate) with another type co-substrate is essentially attributed to the fact that the consortium of bacteria responsible for the degradation and eventual conversion of the waste into biogas become rapidly acclimated. Also, in some cases when there is not enough initial moisture available in the MSW to be treated, co-digestion helps in the compensation of the moisture quantity up to the desirable level, by the addition of another moisture-rich substrate, thereby emphasizing the importance of hydrolysis, as the rate-limiting step. Compensation of the essential nutrients required for the operational stability of the overall process is also another facet of co-digestion, which can be considered as vital. Apart from enhancing the gas production and rendering operational stability, co-digestion has also proved to be economically advantageous because it has provided the scope for treating two or more types of waste in a single facility by sharing the same equipment, thereby enabling easier handling of mixed waste and increasing the economy of sale (Mata-Alvarez et al. 2000).

The ability of AD to treat a large amount of municipal solid organic waste within a small space also provides significant leeway in the efficient handling of whatever quantity of waste is generated on a daily basis in a particular town or city. The mushroom-like development of cities and towns are creating a "space-crunch" for sustaining classical methods of disposal (like landfilling, composting). Furthermore, the management and operational problems associated with sanitary landfilling, pyrolysis or gasification, thermal incineration, and composting have led to the increased acceptability of AD. AD is superior to pyrolysis or gasification, in treating organic waste with low calorific-value (generated in large quantities in a developing country like India). Also, unlike thermal incineration it does not require an external supply of air for sustaining the operation. Apart from that, the closed or sealed operational environment prevents the escape of leachate and GHG, unlike composting and thermal incineration. Thus it can be concluded that techno-economic viability of AD is greater than other conventional MSWM techniques. As mentioned before, AD requires a smaller space in comparison to other processes for treatment of an enormous quantity of waste, thereby reducing the carbon footprint.

The biogas produced from AD can be used for generating electricity with the help of high quality, costly fuel cells. It accounts for 0.5% of the electricity generation in the UK and 1% of the electricity output in the USA (Xuereb 1997). As well, the produced biogas can be directly used up to generate electricity and steam. However, the economic viability of generating electricity through this produced biogas is still under question, owing to the involvement of the costly fuel cells. Apart from generating electricity, the biogas is also viewed as an environmentally attractive alternative to diesel and gasoline for operating public transit systems, and in the operation of internal combustion engines. It is used in the same manner as compressed natural gas for fueling light- and heavy-duty vehicles. It has been reported that usage of biogas as a substitute for diesel and gasoline has greatly reduced the emission of fumes and [NO.sub.x] from automobile exhaust. In addition, the sound generated from methane-powered engines is generally much lower than diesel-operated engines, thereby controlling noise pollution. The hydrogen produced has a wide spectrum of applications as an energy source at both the industrial and commercial levels. The aforesaid form of non-conventional energy plays a crucial role in mitigating the present and future energy crisis. The sludge produced also can be used as an effective soil conditioner. Instead of costly chemical fertilizers, farmers can look forward to using this inexpensive soil conditioner, thereby preventing degradation of the soil quality. Today it is clear that overdoses of chemical or inorganic fertilizers jeopardize soil quality and endanger crop life, due to the increased toxicity levels of certain chemicals. In addition, surface runoff can adversely affect aquatic life. Hence, it is advisable to use this naturally procured soil conditioner as an organic amendment for the betterment or safeguard of the environment. The production of a number of commercial and full-scale plants based on the AD of OFMSW across the globe is a certain indication of the credibility of the process to be an indispensable form of MSWM technique as well as an energy-generating option for the future.

12. Concluding remarks

The present review clearly reveals the importance of AD of OFMSW in terms of its compatibility in stabilizing OFMSW. The various advantages of AD, such as low requirement of space and driving energy, zero emission of GHG, and the ability to treat huge quantities of waste, makes it an obvious choice over other conventional MSWM techniques. It can be concluded that with the gradual development of this process many new modifications, such as co-digestion with several types of other wastes and adequate mixing and pre-treatment of the OFMSW prior to AD, have resulted in a wide range of environmental footprints. Improvement of the startup phase by employing certain inoculum, adopting various sorting techniques, and selection of thermophilic rather than mesophilic temperature regimes has ultimately resulted in a substantial enhancement of biogas production and a volumetric reduction of the generated OFMSW. As far as the energy utilization is concerned, from electricity generation to powering of automobiles, all these can be supported by AD of OFMSW. All of these essentialities coupled with many other factors make AD of OFMSW a basic tool in terms of comprehensive MSWM. Nevertheless, the process has not yet been implemented on a large scale in developing countries, due to a relatively high initial cost and sophisticated operation and maintenance. Other reasons, which have possibly prevented the large-scale implementation of AD of OFMSW, are the sensitivity of the reactors to pH and temperature and the storage of the produced gas, all of which require efficient monitoring. If all of these factors are provided for, AD may prove to be an invaluable technology for the treatment of the MSW in both developing and developed countries all over the world.

Acknowledgements

The Indian Institute of Engineering Science and Technology, Shibpur (IIESTS) and Ministry of Human Resource Development, Government of India, supported and funded for this review.

References

Abbassi-Guendouz, A., Brockmann, D., Trably, E., Dumas, C., Delgenes, J.P., Steyer, J., and Escudie, R. 2012. Total solids content drives high solid anaerobic digestion via mass transfer limitation. Bioresour. Technol. 111: 55-61. doi:10.1016/j.biortech.2012.01.174. PMID:22386469.

Abbasi, S.A., and Ramasami, E. 1999. Biowaste treatment with anaerobic digestion-the dual rate of heavy metals. Biotechnological methods of pollution control. University Press (India) Ltd.

Aguilar, MAR., Fdez.-Guelfo, LA., Alvarez-Gallego, C.J., and Garcia, L.I.R. 2013. Effect of HRT on hydrogen production and organic matter solubilization in acidogenic anaerobic digestion of OFMSW. Chem. Eng. J. 219: 443-449. doi: 10.1016/j.cej.2012.12.090.

Ahring, B.K., and Westermann, P. 1988. Product inhibition of butyrate metabolism by acetate and hydrogen in a thermophilic coculture. Appl. Environ. Microbiol. 54: 2393-2397. PMID:16347751.

Ahring, B.K., Sandberg, M., and Angelidaki, I. 1995. Volatile fatty acids as indicators of process imbalance in anaerobic digesters. Appl. Microbiol. Biotechnol. 43: 559-565. doi:10.1007/BF00218466.

Akanbi, F.A. 1990. Fate and effects of zinc and cadmium on acetate enrichment systems. Master's Thesis, University of Cincinnati, Cincinnati, OH.

Albertson, O.E. 1961. Ammonia nitrogen and the anaerobic environment. J. Water Pollut. Control Fed. 33: 978-995.

Alvarez, R., and Liden, G 2008. Semi-continuous co-digestion of solid slaughterhouse waste, manure, and fruit and vegetable waste. Renew. Energy, 33: 726-734. doi:10.1016/

j.renene.2007.05.001.

Anderson, G.K., and Yang, G. 1992. Determination of bicarbonate and total volatile acid concentration in anaerobic digesters using a simple titration. Water Environ. Res. 64: 53-59. doi:10.2175/WER.64.1.8.

Angelidaki, I., and Ahring, B.K. 1993. Thermophilic anaerobic digestion of livestock waste: the effect of ammonia. Appl. Microbiol. Biotechnol. 38:560-564. doi:10.1007/BF00242955.

Angelidaki, I., and Ahring, B.K. 1994. Anaerobic thermophilic digestion of manure at different ammonia loads: effect of temperature. Water Res. 28: 727-731. doi:10.1016/0043-1354(94)90153-8.

Angelidaki, I., Boe, K., and Ellegaard, L. 2005. Effect of operating conditions and reactor configuration on efficiency of full-scale biogas plants. Water Sci. Technol. 52:189-194. PMID:16180427.

Angelidaki, I., Chen, X., Cui, J., Kaparaju, P., and Ellegaard, L. 2006. Thermophilic anaerobic digestion of source-sorted organic fraction of household municipal solid waste: Start-up procedure for continuously stirred tank reactor. Water Res. 40(14): 2631-2628. doi:10.1016/j.watres.2006.05.015.

Angenent, L.T., Sung, S., and Raskin, L. 2004. Formation of granules and methanosaeta fibers in an anaerobic migrating blanket reactor (AMBR). Environ. Microbiol. 6: 315-322. doi:10.1111/j.1462-2920.2004.00597.x. PMID:15008810.

Anthonisen, A.C., Loehr, R.C., Prakasam, T.B.S., and Srinath, E.G. 1976. Inhibition of nitrification by ammonia and nitrous acid. J. Water Pollut. Control Fed. 48: 835-852. PMID:948105.

Ariunbaatar, J., Panico, A., Esposito, G, Pirozzi, F., Lens, N.L., and Piet. 2014. Pretreatment methods to enhance anaerobic digestion of organic solid waste. Appl. Energy, 123: 143-156. doi:10.1016/j.apenergy.2014.02.035.

Ariunbaatar, J., Di-Perta, E.S., Panico, A., Frunzo, L., Esposito, G., Lens, P.N.L., and Pirozzi, F. 2015. Effect of ammoniacal nitrogen on one-stage and two-stage anaerobic digestion of food waste. Waste Manage. 38: 388-398. doi:10.1016/j.wasman.2014.12.001.

Ashley, N.V., Davies, M., and Hurst, T. 1982. The effect of increased nickel ion concentrations on microbial populations in the anaerobic digestion of sewage sludge. Water Res. 16: 963-971. doi:10.1016/0043-1354(82)90030-6.

Banks, C.J., Chesshire, M., and Stringfellow, A. 2008. A pilot-scale comparison of mesophilic and thermophilic digestion of source segregated domestic food waste. Water Sci. Technol. 58: 1475-1481. doi:10.2166/wst.2008.513. PMID: 18957762.

Banks, C.J., Zhang, Y., Jiang, Y., and Heaven, S. 2012. Trace element requirements for stable food waste digestion at elevated ammonia concentrations. Bioresour. Technol. 104:127-135. doi:10.1016/j.biortech.201110.068. PMID:22100238.

Barber, W., and Stuckey, D.C. 2000. Metal bioavailability and trivalent chromium removal in ABR. J. Environ. Eng. 126: 649-665. doi:10.1061/(ASCE)0733-9372(2000)126:7(649).

Batstone, D.J., Keller, J., and Blackall, L.L. 2004. The influence of substrate kinetics on the microbial community structure in granular anaerobic biomass. Water Res. 38:1390-1404. doi:10.1016/j.watres.2003.12.003.

Bhattacharya, S.K., Uberoi, V., Madura, R.L., and Haghighi-Podeh, M.R. 1995. Effect of cobalt on methanogenesis. Environ Technol. 16:271-278. doi:10.1080/09593331608616269.

Bhide, A.D., and Shekdar, A.V. 1998. Solid waste management in Indian urban centers. Int. Solid Waste Assoc. Times, 1: 26-28.

Billington, R.S. 1988. A review of the kinetics of the methanogenic fermentation of lignocellulosic wastes. J. Agr. Eng. Res. 39: 71-84. doi:10.1016/0021-8634(88)90131-X.

Bollon, J., Le-hyaric, R., Benbelkacem, H., and Buffiere, P. 2011. Development of a kinetic model for anaerobic dry digestion processes: Focus on acetate degradation and moisture content. Biochem. Eng. J. 56:212-218. doi:10.1016/j.bej.2011.06.011.

Bolzonella, D., Innocenti, L., Pavan, P., and Cecchi, F. 2001. Denitrification potential enhancement by addition of the anaerobic fermented of the organic fraction of municipal solid waste. Water Science Technol. 44:187-194. PMID:11496672.

Bolzonella, D., Innocenti, L., Pavan, P., Traverso, P., and Cecchi, F. 2003. Semi-dry thermophilic anaerobic digestion of the organic fraction of municipal solid waste: focusing on the start-up phase. Bioresour. Technol. 86:123-129. doi: 10.1016/S0960-8524(02)00161-X. PMID:12653276.

Bolzonella, D., Pavan, P., Mace, S., and Cecchi, F. 2006. Dry anaerobic digestion of differently sorted organic municipal solid waste: a full-scale experience. Water Sci. Technol. 53(8): 23-32. doi:10.2166/wst.2006.232. PMID:16784186.

Boone, D.R., and Xun, L.Y. 1987. Effects of pH, temperature, and nutrients on propionate degradation by a methanogenic enrichment culture. Appl. Environ. Microbiol. 53:1589-1592. PMID:16347387.

Borowski, S. 2015. Temperature-phased anaerobic digestion of the hydromechanically separated organic fraction of municipal solid waste with sewage sludge. International Biodeterior. Biodegrad. 105:106-113. doi:10.1016/j.ibiod.2015.08.022.

Bouallagui, H., Cheikh, R.B., Marouani, L., and Hamdi, M. 2003. Mesophilic biogas production from fruit and vegetable waste in a tubular digester. Bioresour. Technol. 86: 85-89. doi:10.1016/S0960-8524(02)00097-4. PMID:12421014.

Bouallagui, H., Torrijos, M., Godon, J.J., Moietta, R., Cheikh, R.B., Touhami, Y., Delgenes, J.P., and Hamdi, M. 2004. Two-phases anaerobic digestion of fruit and vegetable wastes: bioreactors performance. Biochem. Eng. J. 21:193-197. doi:10.1016/j.bej.2004.05.001.

Bouallagui, H., Touhami, Y., BenCheikh, R., and Hamdi, M. 2005. Bioreactor performance in anaerobic digestion of fruit and vegetable wastes. Process Biochem. 40: 989-995. doi:10.1016/j.procbio.2004.03.007.

Bouallagui, H., Lahdheb, H., Romdan, E.B., Rachdi, B., and Hamdi, M. 2009. Improvement of fruit and vegetable waste anaerobic digestion performance and stability with co-substrates addition. J. Environ. Manage. 90:1844-1849. doi:10.1016/j.jenvman.2008.12.002. PMID:19117663.

Brummeier, E.T., and Koster, I.W. 1989. The Effect of Several pH Control chemicals on the Dry Batch Digestion of the Organic Fraction of Municipal Solid Waste. Resour. Conserv. Recycling, 3:19-32. doi:10.1016/0921-3449(89)90011-6.

Callaghan, F.J., Wase, D.A.J., Thayanithy, K., and Forster, C.F. 1999. Co-digestion of waste organic solids: batch studies. Bioresour. Technol. 67:117-122. doi:10.1016/S0960-8524(98)00108-4.

Callaghan, F.J., Wase, D.A.J., Thayanithy, K., and Forster, C.F. 2000. Continuous co digestion of cattle slurry with fruit and vegetable wastes and chicken manure. Biomass Bioenerg. 27: 71-77. doi:10.1016/S0961-9534(01)00057-5.

Capela, I., Rodrigues, A., Silva, F., Nadais, H., and Arroja, L. 2008. Impact of industrial sludge and cattle manure on anaerobic digestion of the OFMSW under mesophilic conditions. Biomass Bioenergy, 32: 245-251. doi:10.1016/j.biombioe.2007.09.004.

Castillo, E.F.M., Cristancho, D.E., and Arellano, VA. 2006. Study of the operational conditions for anaerobic digestion of urban solid wastes. Waste Manage. 26: 546-556. doi:10.1016/j.wasman.2005.06.003.

Cecchi, F., and Traverso, P.G. 1986. Biogas from the organic fraction of the municipal solid waste and primary sludge. Part II. Chim. Ind. Quad. Ing. Chim. Ital. 22(7-9): 9-13.

Cecchi, F., Traverso, P.G., Mata-Alvarez, J., Clancy, J., and Zaror, C. 1988. State of the art of R & D in the anaerobic digestion process of municipal solid waste in Europe. Biomass, 16: 257-284. doi:10.1016/0144-4565(88)90031-5.

Cecchi, F., Traverso, P.G., Medici, F., Fazzini, G., and Mata-Alvarez, J. 1989. Comparison between the literature and a new approach to the kinetics of anaerobic degradation of organic fraction of MSW. Biomass, 16: 257-284.

Cecchi, F., Pavan, P., Mata-Alvarez, J., Bassetti, A., and Cozzolino, C 1991. Anaerobic-digestion of municipal solid-waste--thermophilic vs. mesophilic performance at high solids. Waste Manage. Res. 9(4): 305-315. doi:10.1016/0734-242X(91)90020-8.

Cecchi, F., Pavan, P., Mata-Alvarez, J., Musacco, A., and Vallini, G. 1993. Digesting the organic fraction of municipal solid waste. Movingfrom mesophilic (37oC) to thermophilic (55oC) conditions. Waste Manage. Res. 11: 433-444. doi:10.1006/wmre.1993.1042.

Cecchi, F., Traverso, P., Pavan, P., Bolzonella, D., and Innocenti, L. 2003. Characteristics of the OFMSW and behavior of the anaerobic digestion process. In Biomethanization of the organic fraction of municipal solid waste. Edited by J. Mata-Alvarez. IWA: UK. pp. 141-179.

Cesaro, A., and Belgiorno, V. 2013. Sonolysis and ozonation as pretreatment for anaerobic digestion of solid organic waste. Ultrason. Sonochem. 20:931-936. doi:10.1016/j.ultsonch.2012.10.017. PMID:23231941.

Chanakya, H.N., Ramachandra, T.V., and Vijayachamundeeswari, M. 2007. Resource recovery potential from secondary components of segregated municipal solid wastes. Environ. Monit. Assess. 135:119-127. doi:10.1007/s10661-007-9712-4. PMID:17503209.

Charles, W., Walker, L., and Cord-Ruwisch, R. 2009. Effect of pre-aeration and inoculum on the start-up of batch thermophilic anaerobic digestion of municipal solid waste. Bioresour. Technol.100:2329-2335. doi:10.1016/j.biortech. 2008.11.051. PMID:19128961.

Chen, C.C., Lin, C.Y., and Chang, J.S. 2001. Kinetics of hydrogen production with continuous anaerobic cultures utilizing sucrose as the limiting substrate. Appl. Microbiol. Biotechnol. 57(1-2): 56-64. doi:10.1007/s002530100747. PMID:11693934.

Chen, M. 1983. Adaptation of mesophilic anaerobic sewage fermentor population to thermophilic temperatures. Appl. Enviorn. Microbiol. 45(4): 1271-1276. PMID:6859847.

Chen, T.H., and Hashimoto, A.G. 1996. Effects of pH and substrate: inoculum ratio on batch methane fermentation. Bioresour. Technol. 56:179-186. doi:10.1016/0960-8524(96)00016-8.

Chen, W.H., Chen, S.Y., Khanal, S.K., and Sung, S. 2006. Kinetic study of biological hydrogen production by anaerobic fermentation. Int. J. Hydrogen. Energy, 31: 2170-2178. doi:10.1016/j.ijhydene.2006.02.020.

Chen, Y., and Hashimoto, A. 1978. Kinetics of methane fermentation. Biotechnol. Bioeng. Symp. 8: 269-282.

Chu, CF., Li, Y.Y., Xu, K.Q., Ebie, Y., Inamori, Y., and Kong, H.N. 2008. A pH- and temperature-phased two-stage process for hydrogen and methane production from food waste. Int. J. Hydrogen Energy, 33: 4739-4746. doi:10.1016/j.ijhydene.2008.06.060.

Chugh, S., Clarke, W., Nopharatana, A., Pullammanappallil, P., and Rudolph, V. 1995. Degradation of unsorted MSWby sequential batch anaerobic reactor. In Sardinia 95, Fifth International Landfill Symposium, Cagliari, Italy, pp. 67-77.

Chyi, Y.T., and Dague, R.R. 1994. Effects of particulate size in anaerobic acidogenesis using cellulose as a sole carbon source. Water Environ. Res. 66: 670-678. doi:10.2175/WER.66.5.2.

Chynoweth, D.P., Owens, J., O'Keefe, D., Earle, J.F.K., Bosch, G., and Legrand, R. 1992. Sequential batch anaerobic composting of the organic fraction of municipal solid waste. Water Sci. Technol. 25: 327-339.

Climenhaga, M.A., and Banks, C.J. 2008a. Anaerobic digestion of catering wastes: effect of micronutrients and retention time. Water Sci. Technol. 57:687-692. doi:10.2166/wst.2008.092. PMID:18401139.

Climenhaga, M.A., and Banks, C.J. 2008b. Uncoupling of liquid and solid retention times in anaerobic digestion of catering wastes. Water Sci. Technol. 58: 1581-1587. doi:10.2166/wst.2008.497. PMID:19001711.

Collins, M.D., Lawson, P.A., Willems, A., Cordoba, J.J., Fernandez-Garayzabal, J., Garcia, P., Cai, J., Hippe, H., and Farrow, JA.E. 1994. The phylogeny of the genus Clostridium: proposal of five new genera and eleven new species combinations. Int. J. Syst. Bacteriol. 44: 812-826. doi:10.1099/00207713-44-4-812. PMID:7981107.

Costello, D.J., Greenfield, P.F., and Lee, P.L. 1991. Dynamic modelling of a singlestage high-rate anaerobic reactor - I Model derivation. Water Res. 25: 847-858. doi:10.1016/0043-1354(91)90166-N.

Cozzolino, C., Bassetti, A., and Rondelli, R. 1992. Industrial application of semidry digestion process of organic solid waste. In Proceedings of the International Symposium on Anaerobic digestion of Solid Waste. Edited by F. Cecchi, J. Mata-Alvarez, and F.G. Pohland. Venice: pp. 551-555.

CPCB. 2000a. Status of Solid Waste Generation, Collection, Treatment and Disposal in Metro-cities. Series: CUPS/46/1999-2000.

CPCB. 2000b. Status of Municipal Solid waste Generation, Collection, Treatment and Disposal in Class I Cities. Series: ADSORBS/31/1999-2000.

Cuetos, M.J., Gomez, X., Otero, M., and Moran, A. 2008. Anaerobic digestion of solid slaughterhouse waste (SHW) at laboratory scale: Influence of co-digestion with the organic fraction of municipal solid waste (OFMSW). Biochem. Eng. J. 40: 99-106. doi:10.1016/j.bej.2007.11.019.

Das, D., and Veziroglu, T. 2001. Hydrogen production by biological processes: a survey of literature. Int. J. Hydrogen Energy, 26: 13-28. doi:10.1016/S0360-3199(00)00058-6.

Das, D., Srinivasu, M., and Bandyopadhyay, M. 1998. Solid state acidification of vegetable waste. Indian J. Environ. Health, 40(4): 333-342.

Datta, M. 1997. Waste Disposal in Engineered Landfills. Narosa publishing house, New Delhi, India.

Davidsson, A., Gruvberger, C., Christensen, T.H., Hansen, T.L., and Jansen, J.L.C. 2007. Methane yield in source-sorted organic fraction of municipal solid waste. Waste Manage. 27: 406-414. doi:10.1016/j.wasman.2006.02.013.

De Baere, L. 2006. Will anaerobic digestion of solid waste survive in the future? Water Sci. Technol. 53:187-194. doi:10.2166/wst.2006.249. PMID:16784203.

De Baere, L. and Mattheeuws, B. 2008. State-of-the-art 2008-Anaerobic digestion of solid waste. Waste Management World.

De Laclos, H.F., Desbois, S., and Joly, CS. 1997. Anaerobic digestion of municipal solid organic waste: Valorga full-scale plant in Tilburg, The Netherlands. Water Sci. Tech. 36(6-7): 457-462. doi:10.1016/S0273-1223(97)00555-6.

Demirel, B., and Scherer, P. 2011. Trace element requirements of agricultural biogas digesters during biological conversion of renewable biomass to methane. Biomass Bioenergy, 35: 992-998. doi:10.1016/j.biombioe.2010.12.022.

Deublein, D., and Steinhauser, A. 2008. Biogas From Waste and Renewable Resources. Weinheim: Wiley-VCH. 443.

Di Maria, F., Sordi, A., and Micale, C. 2012. Optimization of Solid State Anaerobic digestion by inoculum recirculation. The case of an existing Mechanical Biological Treatment plant. Appl. Energy, 97: 462-469. doi:10.1016/j.apenergy. 2011.12.093.

Diekert, G., and Wohlfarth, G. 1994. Metabolism of homoacetogens. Antonie van Leeuwenhoek, 66: 209-221. doi:10.1007/BF00871640. PMID:7747932.

DiStefano, T.D., and Palomar, A. 2010. Effect of anaerobic reactor process configuration on useful energy production. Water Res. 44:2583-2591. doi:10.1016/j.watres.2010.01.010. PMID:20181375.

Dohanyos, M., Zabranska, J., Kutil, J., and Jenicek, P. 2004. Improvement of anaerobic digestion of sludge. Water Sci. Technol. 49(10): 97-104.

Dong, L., Zhenhong, Y., and Yongming, S. 2010. Semi-dry mesophilic anaerobic digestion of water sorted organic fraction of municipal solid waste (WS-OFMSW). Bioresour. Technol. 101: 2722-2728. doi:10.1016/j.biortech.2009.12.007. PMID:20042331.

Drake, H.L., Go[beta]ner, AS., and Daniel, S.L. 2008. Old acetogens, new light. Ann. N. Y. Acad. Sci. 1125:100-128. doi:10.1196/annals.1419.016. PMID:18378590.

El-Fadel, M., Saikaly, P., and Ghanimeh, S. 2013. Startup and Stability of Thermophilic Anaerobic digestion of OFMSW. Crit. Rev. Environ. Sci. Technol. 43: 2685-2721. doi:10.1080/10643389.2012.694333.

El-Hadj, T.B., Astals, S., Gali, A, Mace, S., and Mata-Alvarez, J. 2009. Ammonia influence in anaerobic digestion of OFMSW. Water Sci. Technol. 59(6): 1153-1158. doi:10.2166/wst.2009.100. PMID:19342811.

El-Mashad, H.M., Zeeman, G., van Loon, W.K.P., Bot, G.P.A., and Lettinga, G. 2003. Anaerobic digestion of solid animal waste in an accumulation system at mesophilic and thermophilic conditions, start up. Water Sci. Technol. 48: 217-220. PMID:14531445.

Fabiano, B., and Perego, P. 2002. Thermodynamic study and optimization of hydrogen production by Enterobacter aerogenes. Int. J. Hydrogen Energy, 27(2): 149-156. doi:10.1016/S0360-3199(01)00102-1.

Fantozzi, F., and Buratti, C. 2011. Anaerobic digestion of mechanically treated OFMSW: Experimental data on biogas/methane production and residues characterization. Bioresour. Technol. 102: 8885-8892. doi:10.1016/j.biortech.2011.06.077. PMID:21763131.

Farquhar, G.J., and Rovers, F.A. 1973. Gas production during refuse decomposition. Water Air Soil Pollut. 2: 483-495. doi:10.1007/BF00585092.

Fascetti, E., and Todini, O. 1995. Rhodobacter sphaeroides RV cultivation and hydrogen production in a one and two-stage chemostat. Appl. Microbiol. Biotechnol. 44(3-4): 300-305. doi:10.1007/BF00169920.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C., Marquez, D.S., Romero, Garcia, L.I. 2010. Start-up of thermophilic-dry anaerobic digestion of OFMSW using adapted modified SEBAC inoculum. Bioresour. Technol. 101: 9031-9039. doi:10.1016/j.biortech.2010.07.021. PMID:20659796.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C., Sales, D., and Romero, L.I. 2011a. The use of thermochemical and biological pretreatments to enhance organic matter hydrolysis and solubilization from organic fraction of municipal solid waste (OFMSW). Chem. Eng. J. 168(1): 249-254. doi:10.1016/j.cej.2010.12.074.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C., Marquez, D.S., Romero, Garcia, L.I. 2011b. The effect of different pretreatments on biomethanation kinetics of industrial Organic Fraction of Municipal Solid Wastes (OFMSW). Chem. Eng. J. 171(2): 411-417. doi:10.1016/j.cej.2011.03.095.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C., Marquez, D.S., and Romero, L.I. 2011c. Biological pre-treatment applied to industrial organic fraction of municipal solid wastes (OFMSW): Effect on anaerobic digestion. Chem. Eng. J. 172: 321-325. doi:10.1016/j.cej.2011.06.010.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C., Marquez, D.S., and Garcia, L.R. 2011d. Dry-thermophilic anaerobic digestion of simulated organic fraction of municipal solid waste: process modeling. Biores. Tech. 102(2): 606-611.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C., Marquez, D.S., and Romero, L.I. 2012a. Dry-thermophilic anaerobic digestion of organic fraction of municipal solid waste: Methane production modelling. Waste Manage. 32: 382-388. doi:10.1016/j.wasman.2011.11.002.

Fdez.-Guelfo, L.A., Alvarez-Gallego, C.J., Marquez, D.S., and Romero, L.I. 2012b. New indirect parameters for interpreting a destabilization episode in an anaerobic reactor. Chem. Eng. J. 180: 32-38. doi:10.1016/j.cej.2011.10.091.

Feng, X.M., Karlsson, A., Svensson, B.H., and Bertilsson, S. 2010. Impact of trace element addition on biogas production from food industrial waste-linking process to microbial communities. FEMS Microbiol. Ecol. 74: 226-240. doi:10.1111/j.1574-6941.2010.00932.x. PMID:20633047.

Fernandez, A., Sanchez, A., and Font, X. 2005. Anaerobic co-digestion of a simulated organic fraction of municipal solid wastes and fats of animal and vegetable origin. Biochem. Eng. J. 26: 22-28. doi:10.1016/j.bej.2005.02.018.

Fernandez, J., Perez, M., and Romero, L.I. 2008. Effect of substrate concentration on dry mesophilic anaerobic digestion of organic fraction of municipal solid waste (OFMSW). Bioresour. Technol. 99(14): 6075-6080. doi:10.1016/j.biortech.2007.12.048. PMID:18255282.

Fernandez, J., Perez, M., and Romero, L.I. 2010. Kinetics of mesophilic anaerobic digestion of the organic fraction of municipal solid waste: Influence of initial total solid concentration. Bioresour. Technol. 101: 6322-6328. doi:10.1016/j.biortech.2010.03.046. PMID:20362435.

Fernandez-Rodriguez, J., Perez, M., and Romero, L.I. 2016. Semi-continuous Temperature-Phased Anaerobic digestion (TPAD) of Organic Fraction of Municipal Solid Waste (OFMSW). Comparison with single-stage processes. Chem. Eng. J. 285: 409-416.

Ferrer, I., Vazquez, F., and Font, X. 2010. Long term operation of a thermophilic anaerobic reactor: process stability and efficiency at decreasing sludge retention time. Bioresour. Technol. 101: 2972-2980. doi:10.1016/j.biortech.2009.12.006. PMID:20056412.

Fontaine, F.E., Peterson, W.H., McCoy, E., Johnson, M.J., and Ritter, G.J. 1942. A new type of glucose fermentation by Clostridium thermoaceticum. J. Bacteriol. 43: 701-715. PMID:16560531.

Forster-Carneiro, T., Perez, M., Romero, L.I., and Sales, D. 2007. Dry-thermophilic anaerobic digestion of organic fraction of the municipal solid waste: Focusing on the inoculum sources. Bioresour. Technol. 98: 3195-3203. doi:10.1016/j.biortech.2006.07.008. PMID:16919940.

Forster-Carneiro, T., Perez, M., and Romero, L.I. 2008a. Thermophilic anaerobic digestion of source-sorted organic fraction of municipal solid waste. Bioresour. Technol. 99: 6763-6770. dor10.1016/j.biortech.2008.01.052. PMID:18343105.

Forster-Carneiro, T., Perez, M., and Romero, L.I. 2008b. Anaerobic digestion of municipal solid wastes: Dry thermophilic performance. Bioresour. Technol. 99: 8180-8184. doi:10.1016/j.biortech.2008.03.021. PMID:18434139.

Forster-Carneiro, J., Perez, M., and Romero, L.I. 2008c. Influence of total solid and inoculum contents on performance of anaerobic reactors treating food waste. Bioresour. Technol. 99(15): 6994-7002. doi:10.1016/j.biortech.2008.01.025.

Fraser, M.D. 1977. The economics of SNG production by anaerobic digester of specially grown plant matter. In Proceedings of Symposium Fuels from Biogas and Wastes, Institute of Gas Technology, Chicago.

Fukuzaki, S., Nishio, N., Shobayashi, M., and Nagai, S. 1990. Inhibition of the Fermentation of Propionate to Methane by Hydrogen, Acetate and Propionate. Appl. Environ. Microbiol. 56(3): 719-723. PMID:16348146.

Gallert, C., and Winter, J. 1997. Mesophilic and thermophilic anaerobic of source-sorted organic wastes: effect of ammonia on glucose degradation and methane production. Appl. Microbiol. Biotechnol. 48:405-410. doi:10.1007/S002530051071.

Gallert, C., Bauer, S., and Winter, J. 1998. Effect of ammonia on the anaerobic degradation of protein by a mesophilic and thermophilic biowaste population. Appl. Microbiol. Biotechnol. 50(4): 495-501.

Gallert, C., Henning, A., and Winter, J. 2003. Scale-up of anaerobic digestion of the biowaste fraction from domestic wastes. Water Res. 37:1433-1441. doi:10.1016/S0043-1354(02)00537-7. PMID:12598207.

Ganesh, R., Torrijos, M., Sousbie, P., Steyer, J.P., Lugardon, A., and Delgenes, J.P. 2013. Anaerobic co-digestion of solid waste: Effect of increasing organic loading rates and characterization of the solubilised organic matter. Bioresour. Technol. 130: 559-569. doi:10.1016/j.biortech.2012.12.119. PMID:23334011.

Gannoun, H., Othman, N.B., Bouallagui, H., and Hamdi, M. 2007. Mesophilic and thermophilic anaerobic co-digestion of olive mill wastewaters and abattoir wastewaters in an upflow anaerobic Alter. Ind. Eng. Chem. Res. 46(21): 6737-6743. doi:10.1021/ie061676r.

Ge, H.Q., Jensen, P.D., and Batstone, D.J. 2010. Pre-treatment mechanisms during thermophilic-mesophilic temperature phased anaerobic digestion of primary sludge. Water Res. 44(1): 123-130. doi:10.1016/j.watres.2009.09.005.PMID:19800093.

Gerardi, M.H. 2003. The Microbiology of Anaerobic Digesters. Hoboken, NJ: Wiley-Interscience. pp. 87.

Ghanimeh, S., El Fadel, M., and Saikaly, P. 2012. Mixing effect on thermophilic anaerobic digestion of source-sorted organic fraction of municipal solid waste. Bioresour. Technol. 117: 63-71. doi:10.1016/j.biortech.2012.02.125.PMID:22609715.

Ghosh, S., and Lall, U. 1988. Kinetics of anaerobic digestion of solid waste. In Proceedings of the 5th International Symposium on Anaerobic Digestion, Bologna, Italy. Edited by E.R. Hall and P.N. Hobson. Advances in Water Pollution Control Series (I.A.W.P.R.C.), Pergamon Press.

Golueke, G.C., and McGadhey, P.H. 1971. Comprehensive studies of solid waste management. In 3rd Annual report SERL, University of California, Berkeley.

Griffin, M.E., McMahon, K.D., Mackie, R.I., and Raskin, L. 1998. Methanogenic Population Dynamics during Start-Up of Anaerobic Digesters Treating Municipal Solid Waste and Biosolids. Biotechnol. Bioeng. 57(3): 342-355. doi:10.1002/(SICI)1097-0290(19980205)57:3<342::AID-BIT11>3.0.CO;2-I. PMID:10099211.

Guermoud, N., Ouadjnia, F., Avdelmalek, F., Taleb, F., and Addou, A. 2009. Municipal solid waste in Mostaganem city (Western Algeria). Waste Manage. 29: 896-902. doi:10.1016/j.wasman.2008.03.027.

Gunaseelan, V.N. 1995. Effect of inoculum/substrate ratio and pretreatments on methane yield from Parthenium. Biomass Bioenergy, 8: 39-44. doi:10.1016/0961-9534(94)00086-9.

Gupta, S., Mohan, K., Prasad, R.K., Gupta, S., and Kansal, A. 1998. Solid waste management in India: options and opportunities. Resour. Conserv. Recycl. 24:137-154. doi:10.1016/S0921-3449(98)00033-0.

Guwy, A.J., Hawkes, F.R., Wilcox, S.J., and Hawkes, D.L. 1997. Neural network and on-off control of bicarbonate alkalinity in a fluidised-bed anaerobic digester. Water Res. 31: 2019-2025. doi:10.1016/S0043-1354(97)00016-X.

Haarstrick, A., Hempel, D.C., Ostermann, L., Ahrens, H., and Dinkier, D. 2001. Modelling of the biodegradation of organic matter in municipal landfills. Waste Manage. Res. 19: 320-331. doi:10.1177/0734242X0101900409.

Halalsheh, M., Kassab, G., Yazajeen, H., Qumsieh, S., and Field, J. 2011. Effect of increasing the surface area of primary sludge on anaerobic digestion at low temperature. Bioresour. Technol. 102: 748-752. doi:10.1016/j.biortech.2010.08.075. PMID:20863692.

Hallenbeck, P.C., and Benemann, J.R. 2002. Biological hydrogen production; fundamentals and limiting processes. Int. J. Hydrogen En. 27(11): 1185-1193.

Hartmann, H., and Ahring, B.K. 2005. Anaerobic digestion of the organic fraction of municipal solid waste: Influence of co-digestion with manure. Water Res.

39:1543-1552. doi:10.1016/j.watres.2005.02.001. PMID:15878026.

Hartmann, H., and Ahring, B.K. 2006. Strategies for the anaerobic digestion of the organic fraction of municipal solid waste: an overview. Water Sci. Technol. 53: 7-22. doi:10.2166/wst.2006.231. PMID:16784185.

Hashimoto, A.G. 1989. Effect of inoculum/substrate ratio on methane yield and production rate. Biol. Wastes, 28: 247-255. doi:10.1016/0269-7483(89)90108-0.

Hills, D.J. 1979. Effects of Carbon:Nitrogen Ratio on Anaerobic Digestion of Dairy Manure. Applied Science Publishers Ltd, England, pp. 267-278.

Hippe, H., Caspari, D., Fiebig, K., and Gottschalk, G. 1979. Utilization of trimethylamine and other N-methyl compounds for growth and methane formation by Methanosarcina barkeri. Proc. Natl. Acad. Sci. U.S.A. 76: 494-498. doi:10.1073/pnas.76.1.494. PMID:284366.

Hobson, P.N., and Shaw, B.G. 1976. Inhibition of methane production by Methanobacterium formicicum. Water Res. 10: 849-852. doi:10.1016/0043-1354(76)90018-X.

Hobson, P.N., and Wheatley, A.D. 1993. Anaerobic digestion: modern theory and practice. Elsevier Science Publishers LTD, Essex, U.K., p. 269.

Hoornweg, D., and Bhada-Tata, P. 2012. What a waste: a global review of solid waste management. World Bank, Washington, DC.

Iranpour, R. 2006. Retrospective and perspectives of thermophilic anaerobic digestion: part I. Water Environ. Res. 78: 99. PMID:16566516.

Jiang, J., Du, X., Ng, S., and Zhang, C. 2010. Comparison of atmospheric pressure effects on the anaerobic digestion of municipal solid waste. Bioresour. Technol. 101: 6361-6367. doi:10.1016/j.biortech.2010.03.068. PMID:20362438.

Jiang, Y., Heaven, S., and Banks, C.J. 2011. Strategies for performance and stability improvement in mesophilic anaerobic digestion of vegetable waste. Paper presented at the International IWA-Symposium on Anaerobic digestion of Solid Waste and Energy Crops, Vienna, Austria, 28 Aug -1 Sep.

Jingura, R.M., and Matengaifa, R. 2009. Optimization of biogas production by anaerobic digestion for sustainable energy development in Zimbabwe. Renew. Sustain. Energy Rev. 13:1116-1120. doi:10.1016/j.rser.2007.06.015.

Kansal, A., Prasad, R.K., and Gupta, S. 1998. Delhi municipal solid waste and environment - an appraisal. Indian J. Environ. Prot. 18(2): 123-128.

Karakashev, D., Batstone, D.J., and Angelidaki, I. 2005. Influence of environmental conditions on methanogenic compositions in anaerobic biogas reactors. Appl. Environ. Microbiol. 71: 331-338. doi:10.1128/AEM.71.1.331-338.2005. PMID:15640206.

Karim, K., Hoffmann, R., Klasson, T., and Al-Dahlan, M.H. 2005. Anaerobic digestion of animal waste: Waste strength versus impact of mixing. Bioresour. Technol. 96:1771-1781. doi:10.1016/j.biortech.2005.01.020. PMID:16051083.

Kawai, K., and Tasaki, T. 2016. Revisiting estimates of municipal solid waste generation per capita and their reliability. J. Mater. Cycles Waste Manag. 18:1-13. doi:10.1007/s10163-015-0355-1.

Kayhanian, M. 1999. Ammonia Inhibition in High-Solids Biogasification: An Overview and Practical Solutions. Environ. Technol. 20: 355-365. doi:10.1080/09593332008616828.

Kayhanian, M., and Rich, D. 1995. Pilot-scale high solids thermophilic anaerobic digestion of municipal solid waste with an emphasis on nutrient requirements. Biomass Bioenergy, 8(6): 433-444. doi:10.1016/0961-9534(95)00043-7.

Kayhanian, M., and Tchobanoglous, G. 1993. Computation of C/N ratios for various organic fractions. Biocycle, 33: 58-60.

Keshtkar, A., Ghaforian, H., Abolhand, G, and Meyssami, B. 2001. Dynamic simulation of cyclic batch anaerobic digestion of cattle manure. Bioresour. Technol. 80: 9-17. doi:10.1016/S0960-8524(01)00071-2. PMID:11554607.

Key Note Publications Ltd. 2007. Global Waste Management Market Assessment 2007. 1 March 2007. Pub ID: KEYL1470786.

Khalid, A., Arshad, M., Anjum, M., Mahmood, T., and Dawson, L. 2011. The anaerobic digestion of solid organic waste. Waste Manage. 31: 1737-1744. doi:10.1016/j.wasman.2011.03.021.

Khanal, S.K. 2008. Anaerobic Biotechnology for Bioenergy Production. Ames, IA: Wiley-Blackwell. pp. 301.

Kiely, G., Tayfur, G., Dolan, C., and Tanji, K. 1997. Physical and mathematical modelling of anaerobic digestion of organic wastes. Water Res. 31(3): 534-540. doi:10.1016/S0043-1354(96)00175-3.

Kim, D.H., and Kim, M.S. 2011. Hydrogenases for biological hydrogen production. Bioresour. Technol. 102: 8423-8431. doi:10.1016/j.biortech.2011.02.113. PMID:21435869.

Kim, D.H., and Kim, M.S. 2013. Development of a novel three-stage fermentation system converting food waste to hydrogen and methane. Bioresour. Technol. 127: 267-274. doi:10.1016/j.biortech.2012.09.088. PMID:23131651.

Kim, D.H., and Oh, S.E. 2011. Continuous high-solids anaerobic co-digestion of organic solid wastes under mesophilic conditions. Waste Manage. 31:1943-1948. doi:10.1016/j.wasman.2011.05.007.

Kim, D.H., Kim, S.H., and Shin, H.S. 2009. Hydrogen fermentation of food waste without inoculum addition. Enzyme Microb. Technol. 45:181-187. doi:10.1016/j.enzmictec.2009.06.013.

Kim, D.H., Kim, S.H., Kim, H.W., Kim, M.S., and Shin, H.S. 2011. Sewage sludge addition to food waste synergistically enhances hydrogen fermentation performance. Bioresour. Technol. 102: 8501-8506. doi:10.1016/j.biortech.2011.04. 089. PMID:21571523.

Kim, H.W., Nam, J.Y., and Shin, H.S. 2011. A comparison study on the high-rate co-digestion of sewage sludge and food waste using a temperature-phased anaerobic sequencing batch reactor system. Bioresour. Technol. 102: 7272-7279. doi:10.1016/j.biortech.2011.04.088. PMID:21600764.

Kim, H.W., Nam, J.Y., Kang, S.T., Kim, D.H., Jung, K.W., and Shin, H.S. 2012. Hydrolytic activities of extracellular enzymes in thermophilic and mesophilic anaerobic sequencing-batch reactors treating organic fractions of municipal solid wastes. Bioresour. Technol. 110: 130-134. doi:10.1016/j.biortech.2012.01.146. PMID:22342040.

Kim, J.K., Cho, J.H., Lee, J.S., Hahm, K.S., Park, D.H., and Kim, S.W. 2002. Mass Production of Methane from Food Wastes with Concomitant Wastewater Treatment. Appl. Biochem. Biotechnol. 98-100: 753-764. doi:10.1385/ABAB:98-100:1-9:753. PMID:12018299.

Kim, J.K., Oh, B.R., Chun, Y.N., and Kim, S.W. 2006. Effects of Temperature and Hydraulic Retention Time on Anaerobic digestion of Food Waste. J. Biosci. Bioeng. 102(4): 328-332. doi:10.1263/jbb.102.328. PMID:17116580.

Kim, J.K., Han, G.H., Oh, B.R., Chun, Y.N., Eom, C.-Y., and Kim, S.W. 2008. Volumetric scale-up of a three stage fermentation system for food waste treatment. Bioresour. Technol. 99: 4394-4399. doi:10.1016/j.biortech.2007.08.031.PMID:17911015.

Kim, M., and Speece, R.E. 2002. Aerobic waste activated sludge (WAS) for start-up seed of mesophilic and thermophilic anaerobic digestion. Water Res. 36: 3860-3866. doi:10.1016/S0043-1354(02)00080-5. PMID:12369531.

Kim, M., Gomec, C.Y., Ahn, Y., and Speece, R.E. 2003. Hydrolysis and acidogenesis of particulate organic material in mesophilic and thermophilic anaerobic digestion. Environ. Technol. 24: 1183-1190. doi:10.1080/09593330309385659. PMID:14599152.

Kim, S.W., Park, J.Y., Kim, J.K., Cho, J.H., Chun, Y.N., Lee, I.H., Lee, J.S., Park, J.S., and Park, D.H. 2000. Development of a Modified Three-Stage Methane Production Process Using Food Wastes. Appl. Biochem. Biotechnol. 84-86: 731-742. doi:10.1385/ABAB:84-86:1-9:731. PMID:10849831.

Kivaisi, A.K., and Mtila, M. 1998. Production of biogas from water hyacinth in a two-stage bioreactor. World J. Microb. Biot. 14: 125-31. doi:10.1023/A:1008845005155.

Kluyver, A.J., and Schnellen, C.G.T.P. 1947. On the fermentation of carbon monoxide by pure cultures of methanogenic bacteria. Arch. Biochem. 14: 57.

Kubier, H., and Rumphorst, M. 1999. Evaluation of processes for treatment of biowaste under the aspects of energy balance and C[O.sub.2] -emission. In Proc. 2nd ISAD-SW, vol. I, Barcelona, 15-17 June 1999, pp. 405-410.

Kumar, A., Miglani, P., Gupta, R.K., and Bhattacharya, T.K. 2006. Impact of Ni, Zn and Co on biogasiflcation of potato waste. J Environ. Biol. 27: 61-66. PMID:16850877.

Kumar, N., Monga, P.S., Biswas, A.K., and Das, D. 2000. Modeling and simulation of clean fuel production by Enterobacter cloacae IIT-BT 08. Int. J. Hydrogen Energy, 25(10): 945-952. doi:10.1016/S0360-3199(00)00017-3.

Kyoto Protocol. 2012. 18th COP. Doha, 2012.

Lane, A.G. 1984. Laboratory scale anaerobic digestion of fruit and vegetable solid waste. Biomass, 5: 245-259. doi:10.1016/0144-4565(84)90072-6.

Lastella, G., Testa, C., Cornacchia, G, Notornicola, M., Voltasio, F., and Sharma, V.K. 2002. Anaerobic digestion of semi-solid organic waste: biogas production and its purification. Energy Conv. Manage. 43:63-75. doi:10.1016/S0196-8904(01)00011-5.

Lay, J.J., Li, Y.Y., and Noike, T. 1998. The influence of pH and ammonia concentration on the methane production in high-solids digestion processes. Water Environ. Res. 70(5): 1075-1082. doi:10.2175/106143098X123426.

Lay, J.J., Fan, K.S., Chang, J., and Ku, C.H. 2003. Influence of chemical nature of organic wastes on their conversion to hydrogen by heat-shock digested sludge. Int. J. Hydrogen Energy, 28:1361-1367. doi:10.1016/S0360-3199(03)00027-2.

Le Hyaric, R., Chardin, C., Benbelkacem, H., Bollon, J., Bayard, R., Escudie, R., and Bufflere, P. 2011. Influence of substrate concentration and moisture content on the specific methanogenic activity of dry mesophilic municipal solid waste digestate spiked with propionate. Bioresour. Technol. 102(2): 822-827. doi:10.1016/j.biortech.2010.08.124. PMID:20863691.

Leslie, R.E. 1991. Effects of Bioavailable Cadmium on Acetate Enrichment Systems. Master's Thesis, Tulane University, New Orleans, LA.

Li, D., Sun, Y.M., and Yuan, Z.H. 2009. Effect of solid content on start-up of mesophilic middle solid anaerobic digestion for water-sorted organic fraction of municipal solid waste. Chin. J. Process Eng. 9: 987-992.

Li, Q., Qiao, W., Wang, X.C., Takayanagi, K., Shofle, M., and Li, Y. 2015. Kinetic characterization of thermophilic and mesophilic anaerobic digestion for coffee grounds and waste activated sludge. Waste Manage. 36:77-85. doi:10.1016/j.wasman.2014.11.016.

Li, Y., Park, S.Y., and Zhu, J. 2011. Solid-state anaerobic digestion for methane production from organic waste. Renew. Sust. Energy Rev. 15: 821-826. doi:10.1016/j.rser.2010.07.042.

Lin, C.Y. 1992. Effect of heavy metals on volatile fatty acid degradation in anaerobic digestion. Water Res. 26:177-183. doi:10.1016/0043-1354(92)90217-R.

Lissens, G., Vandevivere, P., De Baere, L., Biey, E.M., and Verstrae, W. 2001. Solid waste digesters: process performance and practice for municipal solid waste digestion. Water Sci. Technol. 44(8): 91-102. PMID:11730142.

Liu, B.F., Ren, N.Q., Xie, G.J., Ding, J., Guo, W.Q., and Xing, D.F. 2010. Enhanced bio-hydrogen production by the combination of dark- and photofermentation in batch culture. Bioresour. Technol. 101: 5325-5329. doi:10.1016/j.biortech.2010.02.024. PMID:20202826.

Liu, C.F., Yuan, X.Z., Zeng, G.M., Li, W.W., and Li, J. 2008. Prediction of methane yield at optimum pH for anaerobic digestion of organic fraction of municipal solid waste. Bioresour. Technol. 99: 882-888. doi:10.1016/j.biortech.2007.01.013. PMID:17369040.

Liu, F., and Conrad, R. 2010. Thermoanaerobacteriaceae oxidize acetate in methanogenic rice field soil at 50oC. Environ. Microbiol. 12: 2341-2354.PMID:21966924.

Liu, X.G., Zhu, Y., and Yang, S.T. 2006. Construction and characterization of ack deleted mutant of Clostridium tyrobutyricum for enhanced butyric acid and hydrogen production. Biotechnol. Prog. 22(5): 1265-1275. PMID:17022663.

Liu, X., Gao, X., Wang, W., Zheng, L., Zhou, Y., and Sun, Y. 2012a. Pilot-scale anaerobic co-digestion of municipal biomass waste: Focusing on biogas production and GHGreduction. Renew. Energy, 44:463-468. doi:10.1016/j.renene.2012.01.092.

Liu, X., Wang, W., Shi, Y., Zheng, L., Gao, X., Qiao, W., and Zhou, Y. 2012b. Pilot-scale anaerobic co-digestion of municipal biomass waste and waste activated sludge in China: Effect of organic loading rate. Waste Manage. 32: 2056-2060. doi:10.1016/j.wasman.2012.03.003.

Ljungdahl, L.G., and Wood, H.G 1969. Total Synthesis of Acetate from C[O.sub.2] by Heterotrophic Bacteria. Annu. Rev. Microbiol. 23: 515-538. doi:10.1146/annurev.mi.23.100169.002503. PMID:4899080.

Llabres-Luengo, P., and Mata-Alvarez, J. 1987. Kinetic study of the anaerobic digestion of straw-pig manure mixtures. Biomass, 14: 129-142. doi:10.1016/0144-4565(87)90015-1.

Lo, H.M., Chiang, C.F., Tsao, H.C., Pai, T.Y., Liu, M.H., Kurniawan, T.A., Chao, K.P., Liou, C.T., Lin, K.C., Chang, C.Y., et al. 2012. Effect of spiked metals on the MSW anaerobic digestion. Waste Manag. Res. 30(1): 32-48. PMID:20880938.

Lo, H.M., Kurniawan, T.A., Sillanpaa, M.E.T., Pai, T.Y., Chiang, C.F., Chao, K.P., Liu, M.H., Chuang, S.H., Banks, C.J., Wang, S.C., et al. 2010. Modelling biogas production from organic fraction of MSW co-digested with MSWI ashes in anaerobic bioreactors. Bioresour. Technol. 101: 6329-6335. doi:10.1016/j.biortech.2010.03.048. PMID:20400299.

Lo, H.M., Chiu, H.Y., Lo, S.W., and Lo, F.C. 2012. Effects of different SRT on anaerobic digestion of MSW dosed with various MSWI ashes. Bioresour. Technol. 125: 233-238. doi:10.1016/j.biortech.2012.08.084. PMID:23026339.

Lovely, D.R., Dwyer, D.F., and Klug, M.J. 1982. Kinetic Analysis of Competition Between Sulfate Reducers and Methanogens for Hydrogen in Sediments. Appl. Environ. Microbiol. 43(6): 1373-1379. PMID:16346033.

Lv, W., Schanbacher, F.L., and Yu, Z.T. 2010. Putting microbes to work in sequence: recent advances in temperature-phased anaerobic digestion processes. Bioresour. Technol. 101(24): 9409-9414. doi:10.1016/j.biortech.2010.07.100. PMID:20709534.

Lv, W., Zhang, W., and Yu, Z. 2013. Evaluation of system performances and microbial communities of two temperature-phased anaerobic digestion systems treating dairy manure. Bioresour. Technol. 143: 431-438. doi:10.1016/j.biortech.2013.06.013. PMID:23819980.

Macias-Corral, M., Samani, Z., Hanson, A., Smith, G., Funk, P., Yu, H., and Longworth, J. 2008. Anaerobic digestion of municipal solid waste and agricultural waste and the effect of co-digestion with dairy cow manure. Bioresour. Technol. 99: 8288-8293. doi:10.1016/j.biortech.2008.03.057. PMID:18482835.

Mao, C., Feng, Y., Wang, X., and Ren, G 2015. Review on research achievements of biogas from anaerobic digestion. Renew. Sust. Energy Rev. 45: 540-555. doi:10.1016/j.rser.2015.02.032.

Maroun, R., and El Fadel, M. 2007. Start-up of Anaerobic digestion of Source-Sorted Organic Municipal Solid Waste in the Absence of Classical Inocula. Environ. Sci. Technol. 41(19): 6808-6814. doi:10.1021/es070686w. PMID:17969699.

Martin-Gonzalez, L., Colturato, L.F., Font, X., and Vicent, T. 2010. Anaerobic co-digestion of the organic fraction of municipal solid waste with FOG waste from a sewage treatment plant: recovering a wasted methane potential and enhancing the biogas yield. Waste Manage. 30: 1854-1859. doi:10.1016/j.wasman.2010.03.029.

Mata-Alvarez, J., and Cecchi, F. 1989. A review of kinetic models applied to the anaerobic biodegradation of complex organic matter. Kinetics of the biomethanization of the organic fraction of municipal solid waste. In Proceedings of NATO International Workshop on Biotechnology and Biodegradation, Vale de Lobos, Portugal. Edited by D. Kamely, A. Chackrabardy, and G.S. Omenn. Portfolio Publishing Co., Woodland, TX, pp. 27-54.

Mata-Alvarez, J., Mace, S., and Llabres, P. 1990. The Performances of Digesters Treating the Organic Fraction of Municipal Solid Wastes Differently Sorted. Biol. Wastes, 33:181-199. doi:10.1016/0269-7483(90)90004-C

Mata-Alvarez, J., Llabres, P., Cecchi, F., and Pavan, P. 1992. Anaerobic digestion of the Barcelona Central Food Market Organic Wastes: Experimental Study. Bioresour. Technol. 39: 39-48. doi:10.1016/0960-8524(92)90054-2.

Mata-Alvarez, J., Mtz-Viturtia, A., Llabres-Luengo, P., and Cecchi, F. 1993. Kinetic and performance study of a batch two-phase anaerobic digestion of fruit and vegetable wastes. Biomass Bioenergy, 5(6): 481-488. doi:10.1016/0961-9534(93)90043-4.

Mata-Alvarez, J., Mace, S., and Llabres, P. 2000. Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives. Bioresour. Technol. 74: 3-16. doi:10.1016/S0960-8524(00)00023-7.

McCarty, P.L., and McKinney, R.E. 1961a. Volatile acid toxicity in anaerobic digestion. J. Water Pollut. Control Fed. 33: 223-232.

McCarty, P.L., and McKinney, R.E. 1961b. Salt toxicity in anaerobic digestion. J. Water Pollut. Control Fed. 33: 399-415.

McMahon, K.D., Stroot, P.G., Mackie, R.I., and Raskin, L. 2001. Anaerobic co-digestion of municipal solid waste and biosolids under various mixing conditions-II: microbial population dynamics. Water Resour. 35(7): 1817-1827. PMID:11329684.

Melbinger, N.R., and Donnellon, J. 1971. Toxic effects of ammonia nitrogen in high-rate digestion. J. Water Pollut. Control Fed. 43:1658-1670.

Metcalf and Eddy, Inc. 2003. Wastewater Engineering: Treatment and Reuse. Revised by Tchobanoglous, G., Burton, F.L. and Stensel, H.D. Tata McGraw Hill Education Pvt. Ltd. pp. 630-631.

Moietta, R., Verrier, D., and Albagnac, G. 1986. Dynamic modelling of anaerobic digestion. Water Res. 20(4): 427-434. doi:10.1016/0043-1354(86)90189-2.

Monod, J. 1949. The growth of bacterial cultures. Ann. Rev. Microbiol. 3: 371. doi:10.1146/annurev.mi.03.100149.002103.

Mor, S., Ravindra, K., Visscher, A.D., Dahiya, R.P., and Chandra, A. 2006. Municipal solid waste characterization and its assessment for potential methane generation: a case study. J. Sci. Total Environ. 371(1): 1-10. doi:10.1016/j.scitotenv.2006.04.014.

Mora-Naranjo, N., Meima, J.A., Haarstrick, A., and Hempel, D.C. 2004. Modelling and experimental investigation of environmental influences on the acetate and methane formation in solid waste. Waste Manage. 24: 763-773. doi:10.1016/j.wasman.2004.04.006.

Mosey, F.E., and Fernandes, X.A. 1988. Patterns of hydrogen in biogas from the anaerobic-digestion of milk-sugars. Water Sci. Technol. 21: 187-196. doi:10.1016/B978-1-4832-8439-2.50022-5.

Mounfort, D.O. 1978. Evidence for ATP synthesis driven by a proton gradient in Methanosarcine barkeri. Biochem. Biophys. Res. Commun. 85: 1346-1351. PMID:33677.

Mountfort, D.O., and Asher, R.A. 1979. Effect of inorganic sulfide on the growth and metabolism of Methanosarcina barkeri strain DM. Appl. Environ. Microbiol. 37(4): 670-675. PMID:572199.

Mu, Y., Wang, G., and Yu, H.Q. 2006. Kinetic modeling of batch hydrogen production process by mixed anaerobic cultures. Bioresour. Technol. 97(11): 1302-1307. doi:10.1016/j.biortech.2005.05.014. PMID:16055330.

Mu, Y., Yu, H.Q., and Wang, G. 2007. A kinetic approach to anaerobic hydrogen-producing process. Water Res. 41(5): 1152-1160. doi:10.1016/j.watres.2006.11.047. PMID:17267006.

Mumme, J., Linke, B., and Tolle, R. 2010. Novel upflow anaerobic solid-state (UASS) reactor. Bioresour. Technol. 101: 592-599. doi:10.1016/j.biortech.2009.08.073. PMID:19748268.

Nathoa, C., Sirisukpoca, U., and Pisutpaisal, N. 2014. Production of Hydrogen and Methane from Banana Peel by Two-Phase Anaerobic Fermentation. Energy Procedia, 50: 702-710. doi:10.1016/j.egypro.2014.06.086.

NEERI Report. 1996. Strategy Paper on Solid Waste Management in India. 1996: 1-7.

Nguyena, P.H.L., Kuruparan, P., and Visvanathan, C. 2007. Anaerobic digestion of municipal solid waste as a treatment prior to landfill. Bioresour. Technol. 98(2): 380-387. doi:10.1016/j.biortech.2005.12.018. PMID:16480863.

Nie, Y.Q., Liu, H., Du, G.C., and Chen, J. 2007. Enhancement of acetate production by a novel coupled syntrophic acetogenesis with homoacetogenesis process. Process Biochem. 42: 599-605. doi:10.1016/j.procbio.2006.11.007.

Nie, Y.Q., Liu, H., Du, G.C., and Chen, J. 2008. Acetate yield increased by gas circulation and fed-batch fermentation in a novel syntrophic acetogenesis and homoacetogenesis coupling system. Bioresour. Technol. 99: 2989-2995. doi:10.1016/j.biortech.2007.06.018. PMID:17681788.

Nopharatana, A., Clarke, W.P., Pullammanappallil, P.C., Silvey, P., and Chynoweth, D.P. 1998. Evaluation of methanogenic activities during anaerobic digestion of municipal solid waste. Bioresour. Technol. 64:169-174. doi:10.1016/S0960-8524(97)00190-9.

Nopharatana, A., Pullammanappallil, P.C., and Clarke, W.P. 2007. Kinetics and dynamic modelling of batch anaerobic digestion of municipal solid waste in a stirred reactor. Waste Manage. 27: 595-603. doi:10.1016/j.wasman.2006.04.010.

Northrop, D.B., and Wood, H.G. 1969. Transcarboxylase V. The presence of bound zinc and cobalt. J. Biol. Chem. 244: 5801-5807. PMID:5350936.

Novarino, D., and Zanetti, M.C. 2012. Anaerobic digestion of extruded OFMSW. Bioresour. Technol. 104: 44-50.

doi:10.1016/j.biortech.2011.10.001. PMID:22074901.

Okamoto, M., Miyahara, T., Mizuno, O., and Noike, T. 2000. Biological hydrogen potential of materials characteristic of the organic fraction of municipal solid wastes. Water Sci. Technol. 41: 25-32. PMID:11381999.

Pereira, M.A., Cavaleiro, A.J., Mota, M., and Alves, M.M. 2003. Accumulation of long-chain fatty acids onto anaerobic sludge under steady state and shock loading conditions: effect on acetogenic and methanogenic activity. Water Sci. Technol. 48: 33-40. PMID:14640197.

Peterson, C. 2013. Assessment of solid waste management practices and its vulnerability to climate risks in Maldives Tourism Sector. http://tourism.gov.mv/downloads/tap/2014/Solid_Waste.pdf.

Pfeffer, J.T. 1974. Temperature effects on anaerobic fermentation of domestic refuse. Biotechnol. Bioeng. 16: 771-787. doi:10.1002/bit.260160607.

Pfeffer, J.T., and Liebmann, J.C. 1976. Energy from refuse by bioconversion, fermentation and residue disposal processes. Resour. Recovery Conserv. 1: 295-313.

Poggi-Varaldo, H.M., and Oleszkiewicz, J.A. 1992. Anaerobic co-composting of municipal solid waste and waste sludge at high total solids levels. Environ. Technol. 13: 409-421. doi:10.1080/09593339209385169.

Poggi-Varaldo, H.M., Rodriguez-Vazquez, R., Fernandez-Villagomez, G., and Esparza-Garcia, F. 1997a. Inhibition of mesophilic solid-substrate anaerobic digestion by ammonia nitrogen. Appl. Microbiol. Biotechnol. 47: 284-291. doi:10.1007/s002530050928.

Poggi-Varaldo, H.M., Valdes, L., Esparza-Garcia, F., and Fernandez-Villagomez, G 1997b. Solid substrate anaerobic co-digeston of paper mill sludge, biosolids, and municipal solid waste. Water Sci. Technol. 35:197-204. doi:10.1016/S0273-1223(96)00957-2.

Pognani, M., Barrena, R., Font, X., and Sanchez, A. 2012. A complete mass balance of a complex combined anaerobic/aerobic municipal source-separated waste treatment plant. Waste Manage. 32: 799-805. doi:10.1016/j.wasman.2011.12.018.

Ponsa, S., Gea, T., and Sanchez, A. 2011. Anaerobic co-digestion of the organic fraction of municipal solid waste with several pure organic co-substrates. Biosyst. Eng. 108: 352-360. doi:10.1016/j.biosystemseng.2011.01.007.

Pullammanappallil, P.C., Chynoweth, D.P., Lyberatos, G., and Svoronos, S.A. 2001. Stable performance of anaerobic digestion in the presence of a high concentration of propionic acid. Bioresour. Technol. 78:165-169. doi:10.1016/S0960-8524(00)00187-5. PMID:11333036.

Rahn, T., and Gandolfi, P.B. 2007. Modification and Optimisation of Existing MBT Plants using BTA Technology. In 2nd International Symposium MBT, Hanover, Germany, May, pp. 177-186.

Rao, M.S., and Singh, S.P. 2004. Bioenergy conversion studies of organic fraction of MSW: kinetic studies and gas yield - organic loading relationships for process optimization. Bioresour. Technol. 95:173-185. doi:10.1016/j.biortech.2004.02.013. PMID:15246442.

Rastogi, G., Ranade, D.R., Yeole, T.Y., Patole, M.S., and Shouche, Y.S. 2008. Investigation of methanogen population structure in biogas reactor by molecular characterization of methyl-coenzyme M reductase A (mcrA) genes. Bioresour. Technol. 99: 5317-5326. doi:10.1016/j.biortech.2007.11.024. PMID:18155901.

Rees, J.F. 1980. The fate of carbon compounds in the landfill disposal of organic matter. J. Chem. Technol. Biotechnol. 30:161-175. doi:10.1002/jctb.503300121.

Rodriguez, J.F., Perez, M., and Romero, L.I. 2012. Mesophilic anaerobic digestion of the organic fraction of municipal solid waste: Optimisation of the semicontinuous process. Chem. Eng. J. 193-194: 10-15. doi:10.1016/j.cej.2012.04.018.

Rodriguez-Iglesias, J., Castrillon, L., Maranon, E., and Sastre, H. 1998. Solid-state Anaerobic digestion of Unsorted Municipal Solid Waste in a Pilot-plant Scale Digester. Bioresour. Technol. 63(1): 29-35. doi:10.1016/S0960-8524(97)00102-8.

Romero Aguilar, M.A., Fdez-Guelfo, L.A., Alvarez-Gallego, C.J., and Romero Garcia, L.I. 2013. Effect of HRT on hydrogen production and organic matter solubilization in acidogenic anaerobic digestion of OFMSW. Chem. Eng. J. 219: 443-449. doi:10.1016/j.cej.2012.12.090.

Romero Garcia, L.I. 1991. Desarrollo de un modelo matematico general para los procesos fermentativos: cinetica de la degradation anaerobia. Doctoral Thesis. University of Cadiz.

Salminen, E.A., and Rintala, J.A. 2002. Semi-continuous anaerobic digestion of solid poultry slaughterhouse waste: effect of hydraulic retention time and loading. Water Res. 36: 3175-3182. doi:10.1016/S0043-1354(02)00010-6. PMID:12188113.

Sanchez, E., Borja, R., and Lopez, M. 1996. Determination of the kinetic constants of anaerobic digestion of sugar-mill-mud waste (SMMW). Bioresour. Technol. 56: 245-249. doi:10.1016/0960-8524(96)00037-5.

Sandberg, M., and Ahring, B.K. 1992. Anaerobic treatment of fish-meal process waste-water in a UASB reactor at high pH. Appl. Microbiol. Biotechnol. 36: 800-804. doi:10.1007/BF00172198.

Sasaki, D., Hori, T., Haruta, S., Ueno, Y., Ishii, M., and Igarashi, Y. 2011. Methanogenic pathway and community structure in a thermophilic anaerobic digestion process of organic solid waste. J. Biosci. Bioeng. 111(1): 41-46. doi:10.1016/j.jbiosc.2010.08.011. PMID:20851673.

Saxena, R.C., Adhikari, D.K., and Goyal, H.B. 2009. Biomass-based energy fuel through biochemical routes. Renew. Sustain. Energy Rev. 13(1): 167-178. doi: 10.1016/j.rser.2007.07.011.

Scherer, P., and Sahm, H. 1981. Effect of trace elements and vitamins on the growth of Methanosarcina barkeri. Acta Biotechnol. 1(1): 57-65. doi:10.1002/abio.370010108.

Schiel-Bengelsdorf, B., and Durre, P. 2012. Pathway engineering and synthetic biology using acetogens. FEBS Lett. 586: 2191-2198. doi:10.1016/j.febslet.2012.04.043. PMID:22710156.

Schievano, A., DTmporzano, G., Malagutti, L., Fragali, E., Ruboni, G., and Adani, F. 2010. Evaluating inhibition conditions in high-solids anaerobic digestion of organic fraction of municipal solid waste. Bioresour. Technol. 101: 5728-5732. doi:10.1016/j.biortech.2010.02.032. PMID:20206503.

Schink, B. 1988. Principles and limits of anaerobic degradation environmental and technological aspects. In Biology of anaerobic microorganisms. Edited by A.J.B. Zehnder. Wiley, New York, pp. 771-846.

Schmit, K.H., and Ellis, T.G. 2001. Comparison of temperature-phased and Two-Phase Anaerobic Co-Digestion of Primary Sludge and municipal solid waste. Water Environ. Res. 73: 314-321. doi:10.2175/106143001X139335. PMID:11561591.

Schnellen, C.G.T.P. 1947. Onderzoekingen over de methaangisting. Thesis Technical, University, Delft. Rotterdam: De Maastad, publisher.

Schnurrer, A., and Nordberg, A. 2008. Ammonia, a selective agent for methane production by syntrophic acetate oxidation at mesophilic temperature. Water Sci Technol. 57: 735-740. doi:10.2166/wst.2008.097. PMID:18401146.

Schoen, M., Speri, D., Gadermaier, M., Goberna, M., Franke-Whittle, I., Insam, H., Ablinger, J., and Wett, B. 2009. Population dynamics at digester overload conditions. Bioresour. Technol. 100: 5648-5655. doi:10.1016/j.biortech.2009.06.033. PMID:19586768.

Schonheit, P., Moll, J., and Thauer, R.K. 1979. Nickel, Cobalt, and Molybdenum Requirement for Growth of Methanobacterium thermoautotrophicum. Arch. Microbiol. 123:105-107. doi:10.1007/BF00403508. PMID:120728.

Shankaranand, V.S., Ramesh, M.V., and Lonsane, B.K. 1992. Idiosyncrasies of solid-state fermentation systems in the biosynthesis of metabolites by some bacterial and fungal cultures. Process Biochem. 27(1): 33-36. doi:10.1016/0032-9592(92)80006-O.

Shanmugam, P., and Horan, N.J. 2009. Simple and rapid methods to evaluate methane potential and biomass yield for a range of mixed solid wastes. Bioresour. Technol. 100: 471-474. doi:10.1016/j.biortech.2008.06.027. PMID:18694638.

Sharholy, M., Ahmad, K., Mahmood, G., and Trivedi, R.C. 2006. Development of prediction models for municipal solid waste generation for Delhi city. In Proceedings of National Conference of Advanced in Mechanical Engineering (AIME-2006), Jamia Millia Islamia, New Delhi, India, pp. 1176-1186.

Sharholy, M., Ahmad, K., Mahmood, G., and Trivedi, R.C. 2008. Municipal solid waste management in Indian cities - A review. Waste Manage. 28: 459-467. doi:10.1016/j.wasman.2007.02.008.

Sharma, V.K., Testa, C., Lastella, G., Cornacchia, G., and Comparato, M.P. 2000. Inclined-plug-flow type reactor for anaerobic digestion of semi-solid waste. Appl. Energy, 65(1-4): 173-185. doi:10.1016/S0306-2619(99)00084-7.

Siddiqui, T.Z., Siddiqui, F.Z., and Khan, E. 2006. Sustainable development through integrated municipal solid waste management (MSWM) approach

-a case study of Aligarh District. In Proceedings of National Conference of Advanced in Mechanical Engineering (AIME-2006), Jamia Millia Islamia, New Delhi, India, pp. 1168-1175.

Siegert, I., and Banks, C. 2005. The effect of volatile fatty acid additions on the anaerobic digestion of cellulose and glucose in batch reactors. Process Biochem. 40: 3412-3418. doi:10.1016/j.procbio.2005.01.025.

Singh, R.P., Tyagi, V.V., Allen, T., Ibrahim, H.M., and Kothari, R. 2011. An overview for exploring the possibilities of energy generation from municipal solid waste (MSW) in Indian scenario. Renew. Sustain. Energy Rev. 15:4797-4808. doi:10.1016/j.rser.2011.07.071.

Six, W., and De Baere, L. 1992. Dry Anaerobic Conversion of Municipal Solid Waste by Means of the Dranco Process. Water Sci. Tech. 25(7): 295-300.

Song, Y.C., Kwon, S.J., and Woo, J.H. 2004. Mesophilic and thermophilic temperature co-phase anaerobic digestion compared with single-stage mesophilic and thermophilic digestion of sewage sludge. Water Res. 38:1653-1662. doi: 10.1016/j.watres.2003.12.019. PMID:15026219.

Sosnowski, P., Wieczorek, A., and Ledakowicz, S. 2003. Anaerobic co-digestion of sewage sludge and organic fraction of municipal solid wastes. Adv. Environ. Res. 7: 609-616. doi:10.1016/S1093-0191(02)00049-7.

Speece, R.E., Boonyakitsombut, S., Kim, M., Azbar, N., and Ursillo, P. 2006. Overview of anaerobic treatment: thermophilic and propionate implications. Water Environ. Res. 78: 460-473. doi:10.2175/106143006X95492. PMID:16752608.

Sprott, G.D., and Patel, GB. 1986. Ammonia toxicity in pure culture of methanogenic bacteria. System. Appl. Microbiol. 7: 358-363. doi:10.1016/S0723-2020(86)80034-0.

Sprott, G.D., Shaw, K.M., and Jarell, K.F. 1984. Ammonium/potassium exchange in methanogenic bacteria. J. Biol. Chem. 259:12602-12608. PMID:6490632.

Sprott, G.D., Shaw, K.M., and Jarrell, K.F. 1985. Methanogenesis and the K+ transport systemare activated by divalent cations in ammonia-treated cells of Methanospiril-lum hungatei. J. Biol. Chem. 260: 9244-9250. PMID:4019470.

Stroot, P.G., McMahon, K.D., Mackie, R.I., and Raskin, L. 2001. Anaerobic codigestion of municipal solid waste and biosolids under various mixing conditions-I. Digester performance. Water Res. 35(7): 1804-1816. PMID:11329683.

Su, H., Cheng, J., Zhou, J., Song, W., and Cen, K. 2009. Combination of dark- and photo-fermentation to enhance hydrogen production and energy conversion efficiency. Int. J. Hydrogen Energy, 34:8846-8853. doi:10.1016/j.ijhydene.2009.09.001.

Sung, S., and Liu, T. 2003. Ammonia inhibition on thermophilic anaerobic digestion. Chemosphere, 53: 43-52. doi:10.1016/S0045-6535(03)00434-X. PMID:12892665.

Suwannoppadol, S., Ho, G., and Cord-Ruwisch, R. 2011. Rapid start-up of thermophilic anaerobic digestion with the turf fraction of MSW as inoculums. Biores. Technol. 102: 7762-7767. doi:10.1016/j.biortech.2011.06.008.

Taylor, G.T., and Pirt, S.J. 1977. Nutrition and factors limiting the growth of a methanogenic bacterium (Methanobacterium thermoautotrophieum). Arch. Microbiol. 113:17-22. doi:10.1007/BF00428574. PMID:889384.

Tchobanoglous, G., Theisen, H., and Vigil, S. 1993. Integrated Solid Waste Management: engineering principles and management issues. McGraw-Hill Inc., New York, pp. 684-685.

Trzcinski, A.P., and Stuckey, D.C. 2011. Parameters affecting the stability of the digestate from a two-stage anaerobic process treating the organic fraction of municipal solid waste. Waste Manage. 31:1480-1487. doi:10.1016/j.wasman.2011.02.015.

Turick, C.E., Peck, M.W., Chynoweth, D.P., Jerger, D.E., White, E.H., Zsuffa, L., and Kenney, A.W. 1991. Methane fermentation of woody biomass. Bioresour. Technol. 37:141-147. doi:10.1016/0960-8524(91)90202-U.

Valdez-Vazquez, I., Ponce-Noyola, M.T., and Poggi-Varaldo, H.M. 2009. Nutrients related to spore germination improve H2 production from heat-shock-treated consortia. Int. J. Hydrogen Energy, 34: 4291-4295. doi:10.1016/j.ijhydene.2008.09.074.

van Niel, E.W.J., Budde, M.A.W., de Haas, G.G., van der Wal, F.J., Ciaassen, P.A.M., and Stams, A.J.M. 2002. Distinctive properties of high hydrogen producing extreme thermophiles, Caldicellulosiruptor saccharolyticus and Thermotoga elfii. Int. J. Hydrogen Energy, 27(11-12): 1391-1398. doi:10.1016/S0360-3199(02)00115-5.

van Niel, E.W.J., Claassen, P.A.M., and Stams, A.J.M. 2003. Substrate and product inhibition of hydrogen production by the extreme thermophile, Caldicellulosiruptor saccharolyticus. Biotechnol. Bioenergy, 81(3): 255-262. doi:10.1002/bit.10463.

Vandevivere, P., De Baere, L., and Verstraete, W. 2002. Types of anaerobic digester for solid wastes. In Biomethanization of the Organic Fraction of Municipal Solid Wastes. Edited by J. Mata-Alvarez. IWA Publishing Company, pp. 111-140.

Vavilin, V.A., Rytow, S.V., and Lokshina, L.Y. 1995. Modelling Hydrogen Partial Pressure change as a result of competition between the Butyric and Propionic Groups of Acidogenic Bacteria. Bioresour. Technol. 54: 171-177. doi:10.1016/0960-8524(95)00127-1.

Veeken, A., and Hamelers, B. 1999. Effect of temperature on hydrolysis rates of selected biowaste components. Bioresour. Technol. 69: 249-254. doi:10.1016/S0960-8524(98)00188-6.

Veeken, A.H.M., and Hamelers, B.V.M. 2000. Effect of substrate-seed mixing and leachate recirculation on solid state digestion of biowaste. Water Sci. Technol. 41: 255-262. PMID:11382000.

Verma, Sh. 2002. Anaerobic Digestion of Biodegradable Organics in Municipal Solid Wastes. Department of Earth and Environmental Engineering, Fu Foundation School of Engineering and Applied Science, Columbia University.

Verstraete, W., Van Lier, J., Pohland, F., Tilche, A., Mata-Alvarez, J., Ahring, B., Hawkes, D., Cecchi, F., Moietta, R., and Noike, T. 2000. Developments at the Second International Symposium on An aerobic Digestion of Solid Waste (Barcelona, 15-19 June 1999). Bioresour. Technol. 73: 287-289.

Wang, J.L., and Wan, W. 2008a. Effect of temperature on fermentative hydrogen production by mixed cultures. Int. J. Hydrogen Energy, 33(20): 5392-5397. doi:10.1016/j.ijhydene.2008.07.010.

Wang, J.L., and Wan, W. 2008b. The effect of substrate concentration on biohy-drogen production by using kinetic models. Sci. China Ser. B: Chem. 51(11): 1110-1117. doi:10.1007/s11426-008-0104-6.

Wang, J.L., and Wan, W. 2009a. Application of desirability function based on neural network for optimizing biohydrogen production process. Int. J. Hydrogen Energy, 34(3): 1253-1259. doi:10.1016/j.ijhydene.2008.11.055.

Wang, J.L., and Wan, W. 2009b. Kinetic models for fermentative hydrogen production: A review. Int. J. Hydrogen Energy, 34: 3313-3323. doi:10.1016/j.ijhydene.2009.02.031.

Wang, Q.H., Kuninobu, M., Ogawa, H.I., and Kato, Y. 1999. Degradation of volatile fatty acids in highly efficient anaerobic digestion. Biomass Bioenergy, 16: 407-416. doi:10.1016/S0961-9534(99)00016-l.

Wang, Y., Zhao, Q.B., Mu, Y., Yu, H.Q., Harada, H., and Li, Y.Y. 2008. Biohydrogen production with mixed anaerobic cultures in the presence of

high-concentration acetate. Int. J. Hydrogen Energy, 33(4): 1164-1171. doi:10.1016/j.ijhydene.2007.12.018.

Ward, A.J., Hobbs, P.J., Holliman, P.J., and Jones, D.L. 2008. Optimisation of the anaerobic digestion of agricultural resources. Bioresour. Technol. 99: 7928-7940. doi:10.1016/j.biortech.2008.02.044. PMID:18406612.

Wellinger, A., Baserga, U., and Egger, K. 1992. New systems for the digestion of solid-wastes. Water Sci. Technol. 25(7): 319-326.

Wellinger, A., Wyder, K., and Metzler, A.E. 1993. Kompogas -A New System for the Anaerobic Treatment of Source Separated Waste. Water Sci. Tech. 27(2): 153-158.

Whang, L.M., Hsiao, C.J., and Cheng, S.S. 2006. A dual-substrate steady-state model for biological hydrogen production in an anaerobic hydrogen fermentation process. Biotechnol. Bioeng. 95(3): 492-500. doi:10.1002/bit.21041. PMID:16823879.

Wiegant, W.M., and Zeeman, G. 1986. The mechanism of ammonia inhibition in the thermophilic digestion of livestock wastes. Agric. Wastes, 16: 243-253. doi:10.1016/0141-4607(86)90056-9.

Wieringa, K.T. 1939. The formation of acetic acid from carbon dioxide and hydrogen by anaerobic spore-forming bacteria. Antonie Van Leeuwenhoek, 6: 251-262. doi:10.1007/BF02146190.

World Bank. 2013. The World Bank Annual Report 2013. Washington, DC. [C] World Bank. https://openknowledge.worldbank.org/handle/10986/16091. License: CC BY 3.0 IGO.

Xie, S., Lawlor, P.G., Frost, J.P., Hu, Z., and Zhan, X. 2011. Effect of pig manure to grass silage ratio on methane production in batch anaerobic co-digestion of concentrated pig manure and grass silage. Bioresour. Technol. 102: 5728-5733. doi:10.1016/j.biortech.2011.03.009. PMID:21444203.

Xing, W., Ngo, H.H., Guo, W., Wu, Z., Nguyen, T.T., Cullum, P., Listowski, A, and Yang, N. 2010. Enhancement of the performance of anaerobic fluidized bed bioreactors (AFBBRs) by a new starch based flocculant. Sep. Purif. Technol. 72: 140-146. doi:10.1016/j.seppur.2010.01.015.

Xuereb, P. 1997. Biogas - A Fuel produced from waste. Newsl. Michigan Inst. Waste Manage. 1(1): 1-3.

Yen, H.W., and Brune, D.E. 2007. Anaerobic co-digestion of algal sludge and waste paper to produce methane. Bioresour. Technol. 98: 130-134. doi:10.1016/j.biortech.2005.11.010. PMID:16386894.

Yenigun, O., and Demirel, B. 2013. Ammonia inhibition in anaerobic digestion: A review. Process Biochem. 48(5-6): 901-911. doi:10.1016/j.procbio.2013.04. 012.

Yu, H.Q., and Fang, H.H.P. 2002. Acidogenesis of dairy wastewater at various pH levels. Water Sci. Technol. 45: 201-206. PMID:12188545.

Yu, L., Zhao, Q., Ma, J., Frear, C., and Chen, S. 2012. Experimental and modelling study of a two-stage pilot scale high solid anaerobic digester system. Bioresour. Technol. 124: 8-17. doi:10.1016/j.biortech.2012.08.088. PMID:22989632.

Zeikus, J.G., and Wolfe, R.S. 1972. Methanobacterium thermoautotrophicum sp. n., an anaerobic, autotrophic, extreme thermophile. J. Bacteriol. 109: 707-715. PMID:4550816.

Zeshan, Karthikeyan, O.P., and Visvanathan, C. 2012. Effect of C/N ratio and ammonia-N accumulation in a pilot-scale thermophilic dry anaerobic digester. Bioresour. Technol. 113: 294-302. doi:10.1016/j.biortech.2012.02.028. PMID:22386200.

Zhang, B., He, Z.G., Zhang, L.L., Xu, J.B., Shi, H.Z., and Cai, W.M. 2005. Anaerobic digestion of kitchen wastes in a single-phased anaerobic sequencing batch reactor (ASBR) with gas-phased absorb of CO2. J. Environ. Sci. 17: 249-255. PMID:16295899.

Zhang, P., Zeng, G., Zhang, G., Li, Y., Zhang, B., and Fan, M. 2008. Anaerobic co-digestion of biosolids and organic fraction of municipal solid waste by sequencing batch process. Fuel Process. Technol. 89(4): 485-489. doi:10.1016/j.fuproc.2007.11.013.

Zhang, Y., and Banks, C.J. 2012. Co-digestion of the mechanically recovered organic fraction of municipal solid waste with slaughterhouse wastes. Biochem. Eng. J. 68:129-137. doi:10.1016/j.bej.2012.07.017.

Zhang, Y., and Banks, C.J. 2013. Impact of different particle size distributions on anaerobic digestion of the organic fraction of municipal solid waste. Waste Manage. 33(2): 297-307. doi:10.1016/j.wasman.2012.09.024.

Zhou, Y., Takaoka, M., Wang, W., Liu, X., and Oshita, K. 2013. Effect of thermal hydrolysis pre-treatment on anaerobic digestion of municipal biowaste: A pilot scale study in China. J. Biosci. Bioeng. 116(1): 101-105. doi:10.1016/j.jbiosc.2013.01.014. PMID:23419457.

Zwietering, M.H., Jongenburger, I., Rombouts, F.M., and van't Riet, K. 1990. Modeling of the bacterial growth curve. 1990. Appl. Environ. Microbiol. 56(6): 1875-1881. PMID:16348228.

Biswabandhu Chatterjee and Debabrata Mazumder

Received 29 October 2015. Accepted 12 May 2016.

B. Chatterjee and D. Mazumder. Indian Institute of Engineering Science and Technology, Shibpur, Howrah - 711103, India.

Corresponding author: Biswabandhu Chatterjee (email: cbiswabandhu@yahoo.com).

Table 1. The details of various process conditions of a few importantpilot-scale anaerobic digestions of organic fraction of municipalsolid waste (OFMSW).Substrate Co-substrate Biogas yieldSimulated Animal and vegetable 0.8[m.sup.3]/kg TVSsynthetic fat degradedOFMSWSHW OFMSW 0.8[+ or -]0.1 [m.sup.3]/kg VS feedOFMSW Manure 0.69-0.711 [m.sup.3]/kg VS feedSHW Manure and FVW 0.55[m.sup.3]/kgVS

addedOFMSW IS and CM --OFMSW Waste water treatment 0.75375 [m.sup.3]/kgVS plant sludge and dairy (biodegradable) manure addedOFMSW PSS 2.38 [m.sup.3]/[m.sup.3] reactor per day-OFMSW Cattle slurries; FVW, fish -- offal, brewery sludge; dissolved air floatation sludge; poultry manure-FW FVW and DSS 0.66-0.72 [m.sup.3]/kg VS addedOFMSW Sheep blood; mixture of -- pig intestines with flotation fat Methane OperationalSubstrate production temperatureSimulated 0.5 [m.sup.3]/kg TVS Mesophilicsynthetic degraded (37 [degrees]C)OFMSWSHW 0.5[+ or -]0.1 [m.sup.3]/kg Mesophilic VS feed (34 [degrees]C)OFMSW -- Thermophilic (55 [degrees]C)SHW 0.3[m.sup.3]/kgVS Mesophilic added (35 [degrees]C)OFMSW 0.225 [m.sup.3]/kg TVS Mesophilic added (with IS (35 [degrees]C) as co-substrate)OFMSW 0.375 [m.sup.3]/kgVS Thermophilic (biodegradable) (53-60 [degrees]C) addedOFMSW -- Mesophilic (36 [degrees]C)OFMSW 0.13-0.37 [m.sup.3]/kg Mesophilic VS removed (35 [degrees]C) (with different co-substrates)FW 45.4-51.4 [m.sup.3] per Mesophilic ton feedstock (35[+ or -]2 [degrees]C)OFMSW 0.313 [m.sup.3]/kg VS Mesophilic added (36[+ or -]1 [degrees]C) OrganicSubstrate VS reduction loading rateSimulated 73% 0.97 kg VS/[m.sup.3]synthetic per dayOFMSWSHW 82.6%[+ or -]0.6% 1.7-3.7 kg VS/[m.sup.3] per dayOFMSW -- 3.3 to 4.0 kg VS/[m.sup.3] per daySHW 50%-67% 1.3 kg VS/[m.sup.3] per dayOFMSW 65% (with IS as -- co-substrate)OFMSW 85% 7.8 g/kg active reactor mass per day-OFMSW -- 3.8 g VS/kg reactorOFMSW 31.1%-81% (using 8 kg VS/[m.sup.3] per day different combinations of mentioned co-substrates)FW 64.9% 6 kg VS/[m.sup.3] per dayOFMSW -- 4 kg VS/[m.sup.3] per day Retention times Reactor orSubstrate (HRT/SRT) (days) digester typeSimulated 17 (HRT) Semi-continuoussynthetic type reactorOFMSWSHW 25 (run-time) Batch type reactorOFMSW 14 to 18 (run-time) Batch type reactorSHW 30 (HRT) Semi-continuously stirred type reactorOFMSW 65 (run-time) Batch-type reactorOFMSW 25 (SRT) High solid anaerobic digesterOFMSW 115 (run-time) CSTROFMSW 140-175 (run-time) Batch type reactorFW 20 (HRT) CSTROFMSW 30 (SRT) CSTRSubstrate ReferencesSimulated Fernandez et al.synthetic 2005OFMSWSHW Cuetos et al. 2008OFMSW Hartmann and Ahring 2005SHW Alvarez and Liden 2008OFMSW Capela et al. 2008OFMSW Kayhanian and Rich 1995OFMSW Kiely et al. 1997OFMSW Callaghan et al. 1999FW Liu et al. 2012aOFMSW Zhang and Banks 2012Note: VS, volatile solids; HRT, hydraulic retention time; SRT, solidsretention time; TVS, total volatile solids; SHW, slaughterhouse waste;FVW, fruit and vegetable waste; IS, industrial sludge; CM, cattlemanure; PSS, primary sewage sludge; CSTR, continuous stirred typereactor; FW, food waste; DSS, dewatered sewage sludge.Table 2. The effects of sorting technique on gas production and volumereduction in Anaerobic Digestion of OFMSW.Sorting technique adoptedand type of waste Biogas or C[H.sub.4] yieldSS-OFMSW 0.394

[m.sup.3] C[H.sub.4]/kg TVS addedMS-OFMSW 0.140 [m.sup.3] C[H.sub.4]/kg TVS addedFW 0.18 [m.sup.3] C[H.sub.4]/kg VS added-Shredded OFMSW 0.05 [m.sup.3] C[H.sub.4]/kg VS addedSS-OFMSW 0.08 [m.sup.3] C[H.sub.4]/kg VS addedSS-OFMSW 3.0 L/dayMS-OFMSW 2.0 L/dayMS-OFMSW+SS-OFMSW 0.23 [m.sup.3]/kg TVS addedSS-OFMSW 0.43 [m.sup.3] C[H.sub.4]/kg VS addedSS-OFMSW 300-400 N[m.sup.3] C[H.sub.4]/ton VS addedWS-OFMSW 0.314 [m.sup.3] C[H.sub.4]/kg VS addedMS-OFMSW Biogas, 0.258 N[m.sup.3]/kgVS added; C[H.sub.4], 0.035 [m.sup.3]/kg VS addedSS-OFMSW 0.36 [m.sup.3] C[H.sub.4]/kg VS addedSS-OFMSW Biogas produced, 5200 N[m.sup.3]/daySorting technique adopted VSand type of waste reduction Organic loading rateSS-OFMSW 68% 3.1 kg TVS/[m.sup.3] per dayMS-OFMSW 30% 4.8 kg TVS/[m.sup.3] per dayFW 32.4% --Shredded OFMSW 73.3% --SS-OFMSW 79.5% --SS-OFMSW 45% --MS-OFMSW 56% --MS-OFMSW+SS-OFMSW -- 9.2 kg TVS/[m.sup.3] per daySS-OFMSW --SS-OFMSW -- 80% 2.8 kg TVS/[m.sup.3] per dayWS-OFMSW 41.8% --MS-OFMSW 55% --SS-OFMSW -- 0.84 kg TVS/[m.sup.3] per daySS-OFMSW 36.17% --Sorting technique adopted Operational HRTand type of waste temperature (days)SS-OFMSW -- 18.8MS-OFMSW -- 21.3FW 55 [degrees]C --Shredded OFMSW -- --SS-OFMSW -- --SS-OFMSW 55 [degrees]C 60MS-OFMSW -- --MS-OFMSW+SS-OFMSW 54.9 [degrees]C 13.5SS-OFMSW 55 [degrees]C 40SS-OFMSW 55 [degrees]C 15WS-OFMSW 30[+ or -]2 [degrees]C 60MS-OFMSW 55 [degrees]C 40SS-OFMSW 55-60 [degrees]C 130SS-OFMSW 50-55 [degrees]C 22Sorting technique adoptedand type of waste ReferencesSS-OFMSW Mata-Alvarez et al. 1990MS-OFMSW --FW Forster-Carneiro et al. 2008aShredded OFMSW --SS-OFMSW --SS-OFMSW Forster-Carneiro et al. 2008bMS-OFMSW --MS-OFMSW+SS-OFMSW Bolzonella et al. 2003SS-OFMSW Angelidaki et al. 2006SS-OFMSW Davidsson et al. 2007WS-OFMSW Dong et al. 2010MS-OFMSW Fantozzi and Buratti 2011SS-OFMSW Ghanimeh et al. 2012SS-OFMSW Pognani et al. 2012Note: SS-OFMSW, source-sorted organic fraction of municipal solid waste;MS-OFMSW, mechanically sorted organic fraction of municipal solid waste;WS-OFMSW, water-sorted organic fraction of municipal solid waste.Table 3. Municipal solid waste generation rates in different states ofIndia. Municipal Per capita No. Municipal solid waste generatedState cities population (t/day) (kg/day)Andhra Pradesh 32 10 845 907 3943 0.364Assam 4 878 310 196 0.223Bihar 17 5 278 361 1479 0.280Gujarat 21 8 443 962 3805 0.451Haryana 12 2 254 353 623 0.276Himachal Pradesh 1 82 054 35 0.427Karnataka 21 8 283 498 3118 0.376Kerala 146 3 107 358 1220 0.393Madhya Pradesh 23 7 225 833 2286 0.316Maharashtra 27 22 727 186 8589 0.378Manipur 1 198 535 40 0.201Meghalaya 1 223 366 35 0.157Mizoram 1 155 240 46 0.296Orissa 7 1 766 021 646 0.366Punjab 10 3 209 903 1001 0.312Rajasthan 14 4 979 301 1768 0.355Tamil Nadu 25 10 745 773 5021 0.467Tripura 1 157 358 33 0.210Uttar Pradesh 41 14 480 479 5515 0.381West Bengal 23 13 943 445 4475 0.321Chandigarh 1 504 094 200 0.397Delhi 1 8 419 084 4000 0.475Pondicherry 1 203 065 60 0.295Overall 299 128 113 865 48 134 0.376Table 4. Physical characteristics of MSW in different Indian cities(characteristics: percentage by weight).Metro city Paper Textile Leather Plastic Metals GlassAhmedabad 6.0 1.0 -- 3.0 -- --Bangalore 8.0 5.0 -- 6.0 3.0 6.0Bhopal 10.0 5.0 2.0 2.0 -- 1.0Mumbai 10.0 3.6 0.2 2.0 -- 0.2Kolkata 10.0 3.0 1.0 8.0 -- 3.0Coimbatore 5.0 9.0 -- 1.0 -- --Delhi 6.6 4.0 0.6 1.5 2.5 1.2Hyderabad 7.0 1.7 -- 1.3 -- --Indore 5.0 2.0 -- 1.0 -- -- Jaipur 6.0 2.0 -- 1.0 -- 2.0Kanpur 5.0 1.0 5.0 1.5 -- -- Kochi 4.9 -- -- 1.1 -- --Lucknow 4.0 2.0 -- 4.0 1.0 -- Ludhiana 3.0 5.0 -- 3.0 -- --Madras 10.0 5.0 5.0 3.0 -- -- Madurai 5.0 1.0 -- 3.0 -- --Nagpur 4.5 7.0 1.9 1.25 0.35 1.2Patna 4.0 5.0 2.0 6.0 1.0 2.0Pune 5.0 -- -- 5.0 -- 10.0Surat 4.0 5.0 -- 3.0 -- 3.0Vadodara 4.0 -- -- 7.0 -- --Varanasi 3.0 4.0 -- 10.0 -- -- Visakhapatnam 3.0 2.0 -- 5.0 -- 5.0Average 5.7 3.5 0.8 3.9 1.9 2.1 Ash, fine earth, CompostableMetro city and others matter-Ahmedabad 50.0 40.00Bangalore 27.0 45.00Bhopal 35.0 45.00Mumbai 44.0 40.00Kolkata 35.0 40.00Coimbatore 50.0 35.00Delhi 51.5 31.78Hyderabad 50.0 40.00Indore 49.0 43.00Jaipur 47.0 42.00Kanpur 52.5 40.00Kochi 36.0 58.00Lucknow 49.0 40.00Ludhiana 30.0 40.00Madras 33.0 44.00Madurai 46.0 45.00Nagpur 53.4 30.40Patna 35.0 45.00Pune 15.0 55.00Surat 45.0 40.00Vadodara 49.0 40.00Varanasi 35.0 48.00Visakhapatnam 50.0 35.00Average 40.3 41.80

Please note: Some tables or figures were omitted from this article.

Note(s):

ISSN: 1181-8700

dx.doi.org/10.1139/er-2016-0077