ASSESSMENT OF ECONOMICAL BIOGAS PRODUCTION FROM CHILEAN MUNICIPAL SOLID WASTE IN A DECENTRALIZED OFF-GRID STRATEGY

MUÑOZ, P.[1,2*] – MUÑOZ, L.[2] – CORDERO, C.[3] – BIBIRE, L.[4] – MORALES, M. P.[2]

[1]*Universidad Autónoma de Chile, 5 Poniente, 1670, Talca, Chile*

[2]*Universidad Internacional de La Rioja, Av. de la Paz, 137, 26006 Logroño, La Rioja, Spain*

[3]*Centro de Estudios en Alimentos Procesados CEAP, Casilla 1007, Talca, Chile*

[4]*University "Vasile Alecsandri" of Bacau, Calea Mărășești 158, 600115 Bacău, Romania*

**Corresponding author*
e-mail: pmunozv@uautonoma.cl

(Received 10th Jan 2018; accepted 26th Mar 2018)

Abstract. The management of municipal solid waste (MSW) is one of biggest challenges that developing countries are currently facing. Despite several environmental friendly alternatives being available, in Chile 90 % of MSW is disposed of in landfills, which produces large amounts of greenhouse gases (GHG) by the uncontrolled anaerobic digestion of its organic fraction. In addition, the way in which MSW is collected and transported increases the negative impact to the environment, especially when wastes classification is not properly performed at the generation point. On the other hand, the use of resulting biogas, in a decentralized off-grid form, at the generation point, could directly reward to waste producers with economic saving. This strategy will reinforce population motivation and an important reduction of the GHG emissions can be expected. Thus, this paper explores the biogas potential of food wastes by using reactors with no heating and stirring devices, with the aim of assessing the minimum expectable energy and $CO_{2,eq}$ savings. Based on laboratory tests at batch and semi-continuous mode and Chilean historical data, it is concluded that more than 1,704 Gg of $CO_{2,eq}$ can be avoided per year and an average of 425 GWh primary energy may be saved by Chilean householders.

Keywords: *CO_2 emission, psychrophilic, food waste, household scale, waste management*

Introduction

Municipal solid waste (MSW) is currently a great concern for modern societies. A sustainable coping strategy is required in order to protect not only environment but human health as well. The increase in the urban population growth rate has led to increased issues related to MSW management, especially in developing countries where traditional landfill disposal is the most common method of MSW management (Guerrero et al., 2013). It is obvious, in accordance with the residues management hierarchy that the most favorable option to deal with MSW is prevention. However, the reduction of the current levels of MSW must be accompanied by better management of the waste since and it may be highlighted that some parts are inedible and will unavoidably become a waste stream.

There are countless alternatives to manage the organic fraction of MSW which is mainly composed by food waste (FW) and garden waste. However, anaerobic digestion (AD) has been highlighted as the most reliable and efficient technology for minimizing the environmental impact of FW waste disposal in terms of greenhouse gas (GHG) emissions, which are generated during landfilling. In addition, AD of FW offers an

http://www.aloki.hu ● ISSN 1589 1623 (Print) ● ISSN 1785 0037 (Online)
DOI: http://dx.doi.org/10.15666/aeer/1603_24232439

opportunity to produce a large energy yield of biogas and it has been demonstrated rapid public health improvements with important reductions in pathogens (Remais et al., 2009).

The AD process is well known, and related technology has already been successfully tested although MSW collection and separation still remain as key unknown factors for determining the feasibility of AD plants (Olsson and Fallde, 2015). Different systems for FW separation and collection at the generation point are successfully implemented in developed countries, in decentralized or centralized form. However, all these alternatives were supported by several subsystems, which are not yet extended in Chile, such as: the existence of a formal recycling industry, optimal distribution of curbside services and the development of policies for raising awareness, in terms of wastes classification, etc. Several countries have lately developed micro-scale applications to reduce transport requirements by means of creating a "biorefinery" that would dispose of local waste and produce energy (Walker et al., 2017). These micro-scale applications were developed by specific consortium of companies but general public was not included as an active stakeholder. In other words, the main goal was the enhancement of some industrial process rather than the minimization of urban waste management issue. These examples incorporated several complex devices such as; mixers, filters, feeding pumps, water heat exchangers, etc. that increase the installation and operation costs which have been highlighted as the most critical barrier for the popularization of biogas technology (Walekhwa et al., 2009).

On the contrary, small fixed dome digesters that work without stirring or heating have been successfully operated on the long term, mainly in developing countries with installation costs that range from 80 to 150 USD/m^3 (Buysman and Mol, 2013). These AD reactors show very low energy performance since the more complex the reactor configuration is, the more biogas will be produced and fewer impurities will be contained. Moreover, these simple AD plants are designed without cleaning and upgrading biogas stages and consequently power generation or methane grid injection are rarely considered. In spite of such limitations, household scale biogas reactors contribute to reduce $CO_{2,eq}$ emissions, soil and water pollution as well as improve sanitation by reducing pathogenic content of substrate materials. For instance, it has been showed that AD at 20°C reduced indigenous populations of total coliforms by 97.9 to 100 %, indigenous populations of E. coli by 99.6 to 100 %, and provided undetectable levels of indigenous strains of Salmonella, Cryptosporidium, and Giardia (Massé et al., 2010). Besides, the use of biogas represents an opportunity for lowering family expenditures cooking and/or heating fuels which ensure the participation, ownership and responsibility of the final users (Bond and Templeton, 2011), otherwise the lack of proper awareness among user (i.e. householders) inhibits the adaptation of technology.

The proposal

MSW management is nowadays especially worldwide discussed but specially into the Chilean context due to the reviewed national strategies regarding the environmental sustainability politics.

The first step for designing a properly strategy must be always the collection of characteristic data. However, the first national report was published in 2010 and till nowadays no further official update has been provided. Furthermore, Chilean statistics, regarding MSW management, show important differences, depending on whether data are provided by public services (CONAMA, 2009; OECD, 2005) or by previous research (Perez, 2010). Besides, some data regarding the organic fraction amount of MSW are missed (see *Fig. 1*). Nevertheless, by accepting data provided by this first national report, Chile is ranked as the major MSW producer in Latin America, with an annual average of 382.4 kg per capita and the organic fraction of Chilean MSW accounts for up to 60 % (90 % FW and 10 % garden wastes) (CONAMA, 2010a).

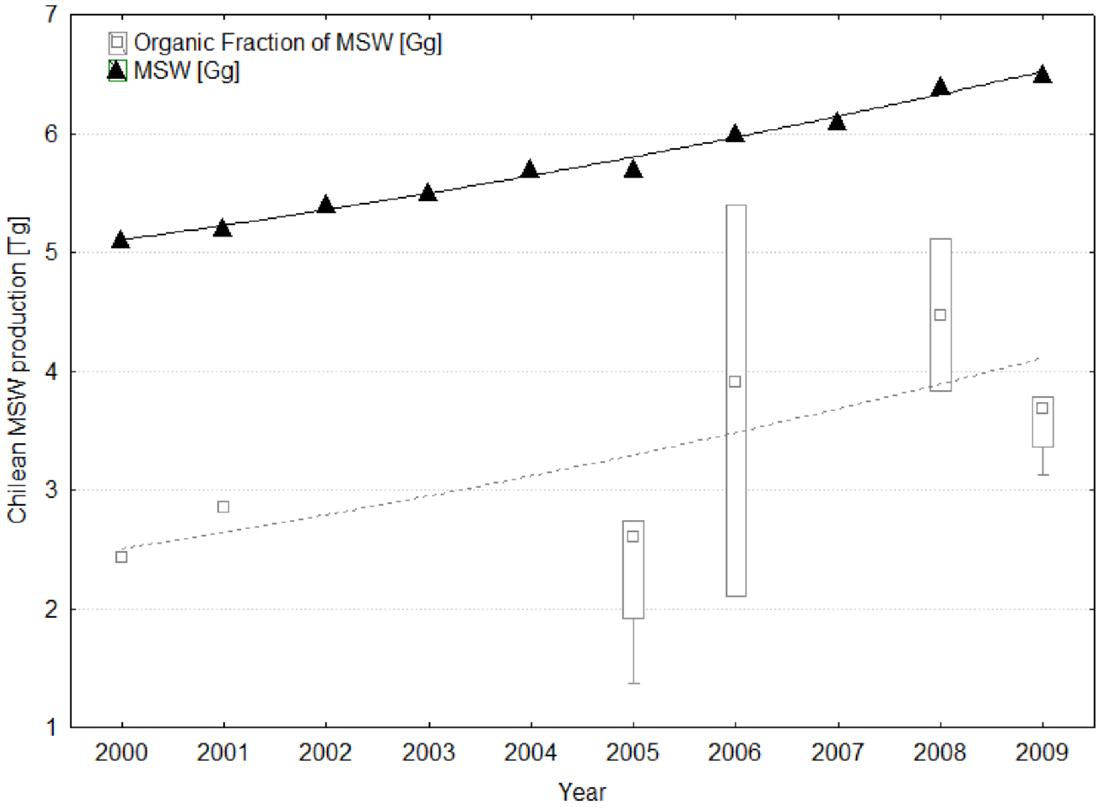

Figure 1. *Statistics of the Chilean MSW production and the amount of organic matter contained by the MSW*

Regarding the available MSW treatment facilities, Chile owns approximately 120 recycling facilities focused on plastics, cartons, oils and metals. These installations are able to recycle approximately 61 % of metals (including aluminium), 55 % of oils, 41 % of papers and cartons, 29 % of glass and 4 % of plastics. However, only 10 % of the organic fraction is currently treated. This organic matter is composed of 90 % food waste (FW) and 10 % gardening waste. Hence, there are 13 installations designed for composting and four that are based on co-incineration. The remaining organic waste is disposed of in 25 sanitary landfills and 139 uncontrolled waste dumping sites which receive MSW from approx. 239 cities. Distances between cities and these places range between 5 and 30 km (CONAMA, 2010b). Despite annual FW generation exhibiting increasing trends, there are no planned future investments, in the short-term, in terms of revalorization or recycling facilities.

MSW is not only an economic and social concern but also an environmental issue, since it is one of the major carbon emission sources from city operations (Galina, 2008).

According to the Chilean Ministry of Environment (2011), landfills gas emissions account for approximately 2,300 Gg of $CO_{2,eq}$ and these emissions are mainly produced by the uncontrolled AD of the organic matter. From this point of view, it is clear that a reduction of organic matter content leads to a decrease in GHG emissions. Since there is no gas collection and flaring systems in most of Chilean landfills, at present, landfills are the highest contributors to GHG emissions. In addition, MSW introduces another source of environmental impact, the so-called horizontal $CO_{2,eq}$ flows. These refer to the cumulative gases emission produced by the collection (i.e. from single households or other waste generation units) and transportation of collected MSW to landfills. In Chile, collection and transportation is performed by collection trucks that are fuelled by fossil fuels (i.e. diesel).

On the other hand, fossil fuels consumption highly impacts on the national economy since Chile does not own fossil fuels. In spite of Chile lacks of fossil fuel resources, more than 60 % of the energy, in terms of primary energy, is provided by imported fossil fuel. As result, the highest percentage of Chilean household expenses corresponds to energy supplies and this cost is increased year by year (e.g. from 2008 to 2012 natural and liquefied gas consumption was increased by 720 %) according to the Ministry of Economy, Promotion and Tourism (SERNAC, 2015).

As it has been previously stated, AD may address all these aspects by reducing the amount $CO_{2,eq}$ emissions and the household expenses related to the energy consumption. However, the key factor for a successful strategy, regarding MSW, is always related to the waste generation point (i.e. householders). Hence, it has been demonstrated that any strategy for MSW management should be well supported by citizen participation and therefore participation and motivation of population is always required to ensure the success of any waste management policy (Wen et al., 2016). At this point, any proposal must consider the waste producer (i.e. the householder) as the most important player.

It is well understood, that a decentralized strategy highly reduces the transportation of waste materials, which represents the main source of pollution regarding the MSW management chain (Alfonso et al., 2009). Besides, AD of FW contributes to reduce energy demand, and the aforementioned MSW impacts. However, one of the main difficulties of sustaining decentralized facilities is the organic waste availability for feeding the anaerobic digester (Righi et al., 2013) (i.e. the cooperation of waste producers). Therefore, in accordance with similar experiences in other countries (Reddy, 2004), this paper assesses the use of FW, generated by households, for biogas production by using household scale fixed dome reactors which will be daily loaded with this FW. However, the proposal cannot be based on expensive units but on simple small reactors working at psychrophilic range by means of simple fixed dome reactors.

The proposal considers that each building would incorporate a single stage reactor and biogas would be directly used on site by means of boilers or cooking stoves. With the aim of lowering the costs the proposal does not considered the use of biogas for electrical power generation or its injection to the natural gas grid. Consequently, complex cleaning biogas systems are not required and the corresponding installation costs are highly reduced. The FW generated would be disposed of in this AD unit and the effluent would be direct discharged as wastewater in sewers (see *Fig. 2*).

Despite previous authors have estimated a decade ago that biogas might contribute up to 45 % to reduce natural gas consumption in Chile (Seiffert et al., 2009; Bidart et al., 2013), currently AD provides approx. 0.03 % out of total primary energy consumption. This unexpected result is explained by two main reasons. On the one hand, authors assumed the expansion of the existing gas pipeline, but no relevant changes have been performed during last decade and this highly impacts economic feasibility. On the other hand, calculations were mainly based on residues provided by forestry industry, wood processing and energy crops which are already being used by other markets (Ignaciuk et al., 2006). In addition, it must be mentioned that co-digestion of FW with other household waste streams has not been considered. In spite of literature has demonstrated that specific biogas yield (SBY) is certainly improved by co-digestion (Iqbal et al., 2014) the potential inhibition of AD (e.g. detergents or papers towels) would make the proposal less reliable (Chen et al., 2008).

The proposal was assessed by determining the technical feasibility of the AD at psychrophilic temperature and by estimating the impact in terms of $CO_{2,eq}$. AD tests were performed in two phases: firstly batch and then semi-continuous operation mode, both at laboratory scale. This, since the batch test is of necessity performed with different substrate mass fractions than the semi-continuous test, and the substrate/inoculum ratio affects the kinetics. Finally, the obtained results were used for calculating the amount of $CO_{2,eq}$ that might be avoided.

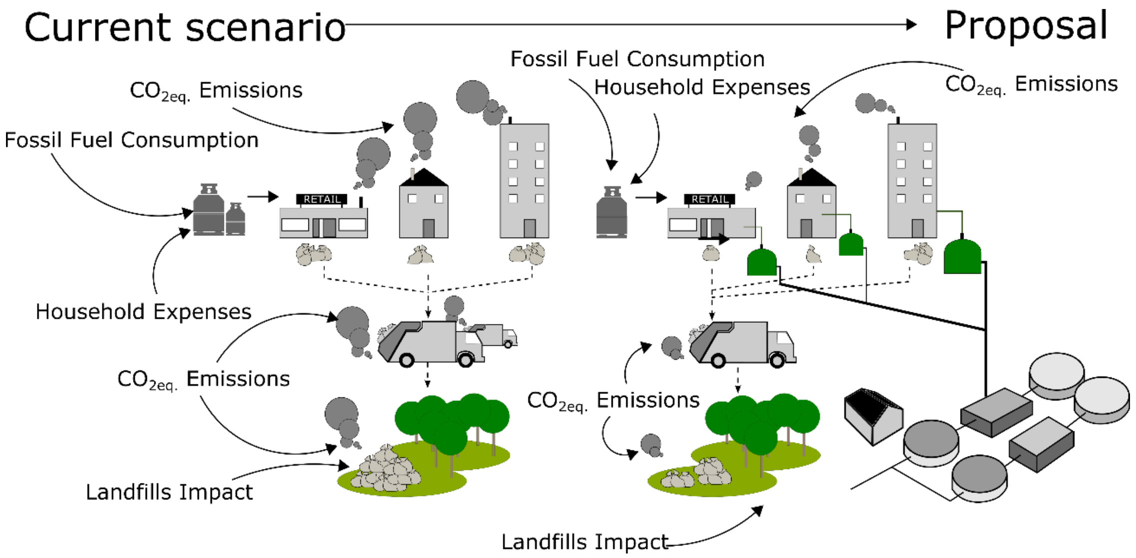

Figure 2. *Diagram of the decentralized AD approach proposed for reducing* CO_{2eq} *emissions by saving fossil fuels and fugitive emissions*

Materials and methods

AD tests

The AD test was divided into three stages. In the first step, FW and inoculum were characterized. Then, these values were used for setting the biochemical methane potential (BMP) tests. Finally, a semi-continuous experiment with lab-scale digester was carried out, along 90 days, maintaining a fixed organic load rate. These last two stages were performed in accordance with VDI 4630:2016-11.

The canteen, at the Autonomous University of Chile, provided FW on a weekly basis. This included animal and fish bones, fruits and vegetable peelings, seeds and any other organic matter. FW was blended with an industrial blender in order to permit comparison between the results of different fermentation tests (particle sizes of less than

10 mm) (Omniblend TM767, Chile) and homogenized. Then, 250 g of FW (approx. 250 g) was used for determining FW characteristics and 250 g of FW was frozen (-20 °C) till being used for loading reactors. An industrial partner, who operates a nearby AD plant, provided the inoculum. This AD plant works on mono-digestion, based on pig manure, at psychrophilic range. Inoculum was sieved throughout a 5 mm mesh for removing the death matter.

Regarding the FW and inoculum characteristics, total solids, (TS) and volatile solids (VS) were determined using an electric furnace set at 105 °C and 600 °C, respectively. Total soluble solids (TSS) and Total Dissolved Solids (TDS) were determined according to 2450-D and 2450-C methodology of the Standard Methods (APHA, AWWA, and WEF, 2005). Soluble nitrogen was measured using HANNA nitrogen determination (chromotropic acid method; range: 10 to 150 mg/L N at 420 nm; accuracy: ± 3 mg/L or ± 4 % of reading at 25 °C). The pH and Oxide Reduction Potential (ORP) were determined using a WTW pH 3110-2. Chemical Oxygen Demand (COD) was analyzed by COD Cell Test (Spectroquant®, Merck KGaA, 64271 Darmstadt, Germany) and the applied method corresponds to DIN ISO 15705, which is analogous to EPA 410.4, APHA 5220 D and ASTM D1252-06B. C:N ratio was determined following the standard methodology proposed by The U.S. Department of Agriculture and the U.S. Composting Council (TMECC, 2002). Total carbohydrates were quantified by following the visible range method using phenol-sulfuric. Total oils and fats were determined in accordance with the method extraction Soxhlet (APHA, AWWA, and WEF, 2005) and protein quantification was determined by means of the Pierce BCA Protein Assay Kit (Thermo scientific).

BMP was determined in triplicate (Ra, Rb, Rc) at laboratory scale (5 L batch reactors) (see *Fig. 3*). In spite of containers used for the fermentation test use to be 0.5 L, 1 L or 2 L flasks, in case of inhomogeneous substrate standards recommend to have larger fermentation volumes as this makes it easier to obtain representative samples. These reactors were equally filled with 150 g. of FW (i.e. 7.5 g VS/L), 1000 mL of inoculum and 3000 mL of fresh water. Hence, headspace volume was set at approx. 1000 mL. It must be highlighted that the organic total matter added from inoculum is quite lower than the recommended by standards and therefore the ratio substrate to inoculum rate (SIR) is higher. The main reason for using a high SIR is explained by two considerations. On the one hand, the results, provided by the semi-continuous test, in the long-term are not influenced by the inoculum used in the batch tests. On the other hand, further assays will be focused on the analysis of SIR in order to assess FW buffering capacity, which are not included within the present paper.

Reactors were kept at room temperature without stirring in order to simulate the conditions of the proposed household scale reactors by following VDI 4630:2016-11. The substrate temperature has been continuously measured and recorded by using a stainless steel temperature probe and data logger (Testo, Germany).

In spite of the initial pH adjustment is considered in different way, depending on the followed standards, previous authors have demonstrated that below an initial pH of 6.0, AD of FW is inhibited (Zhai et al., 2015). Thus, the pH was adjusted to 7.0, by means of NaOH, in all batches. Reactors were sealed and remaining air was removed by nitrogen gas flow.

According to VDI 4630:2016-11 guidelines when the new biogas production was found below 0.5 % after three days, the AD batch was ended and the semi-continuous phase took place.

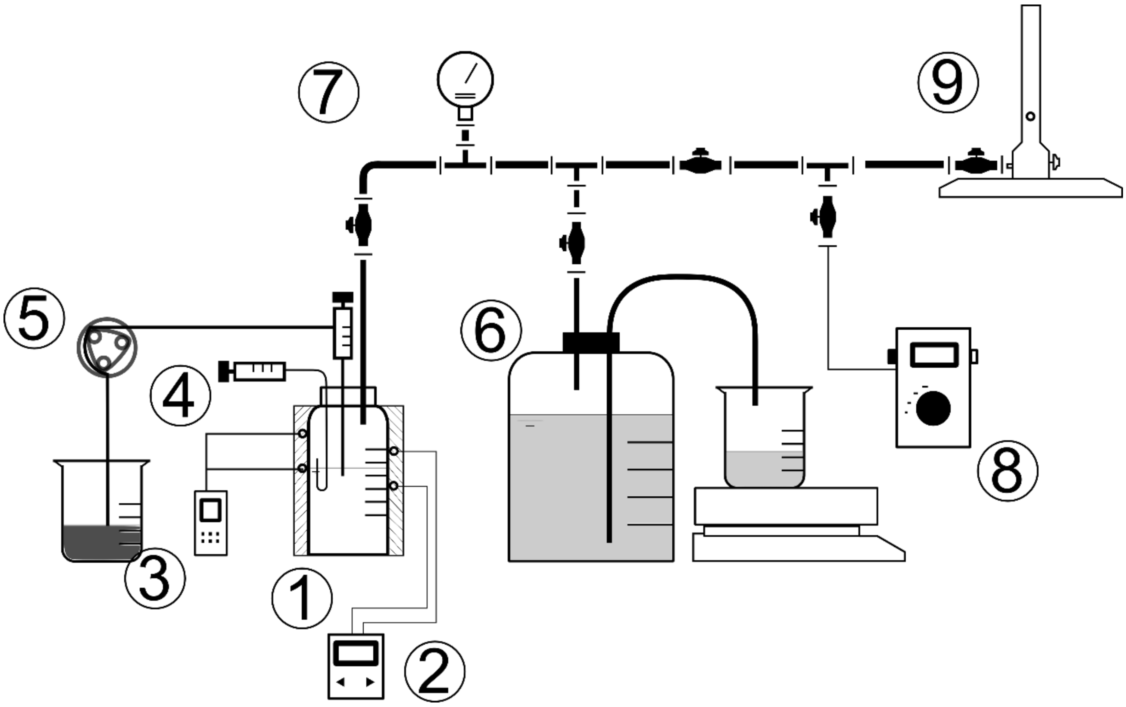

Figure 3. *The schematic representation of experimental laboratory scale biogas test showing (1) 5 L reactors; (2) Temperature probes and data logger; (3) pH and ORP probes; (4) overflow collection; (5) peristaltic pump for loading FW; (6) gas meter; (7) manometer; (8) gas analyzer; (9) burner*

Same reactors were used at semi-continuous stage and running without stirring and without temperature control. AD of FW is prone to instability at high organic load rate and because of that OLR was set at 1 g-VS/L day (i.e. 1-day-portions of 20 g FW) with a volumetric FW to water dilution of 1:2. During the loading operation, reactors were not de-areated. Feeding and digestate removal ware carried out automatically once every day by means of a peristaltic pump, during 90 days and removed digestate included solids and liquid.

The overflowed digestate was analyzed with the aim of determining whether this effluent meet with the requirements set by the national policies and guidelines on wastewater disposal systems. Therefore, parameters such as salinity, electrolytic conductivity (EC), ORP and TDS were measured by using a HI9828 probe made by Hanna.

The measured wet biogas and methane volumes were measured by water displacement and adjusted to the volumes at standard temperature (0 ºC) and pressure (1 atm). The effect of the vapor pressure of water was corrected by using the Antoine equation and the average methane content during the semi-continuous phase was showed (Angelidaki et al., 2009).

Calculations

Energy calculations have been calculated based on the primary energy carried by methane. Since statistics are developed at a regional scale, energy production is calculated as the sum of every regional contribution, in accordance with similar previous studies (Moreda, 2016; *Eq. 1*):

$$E = VS \cdot Q_{CH_4} \cdot H_{CH_4} \cdot \sum_{i}^{n} q_i \cdot f_i \qquad \text{(Eq.1)}$$

where VS is the average volatile solids content of MSW (g VS/Mg); Q_{CH4} is specific biomethane yield (m^3/g_VS); H_{CH4} is volumetric heating value (kWh/m^3) of biomethane at standard conditions; q is regional MSW and fi is the percentage of FW within MSW (%).

Regarding the $CO_{2,eq}$, calculations have taken into account two main aspects: On the one hand, the $CO_{2,eq}$ emissions avoided by both the reduction of organic waste disposed of at landfills and the corresponding horizontal flows, (i.e. those produced by collection and transportation). On the other hand, the impact of the substitution of fossil fuels (i.e. natural and liquefied gas) by biogas has been quantified.

Despite AD mainly produces CO_2 and CH_4, only CH_4 emissions have been accounted. This because of CO_2 release is only a partial “return” to the atmosphere of the same gas used by photosynthetic organisms to build their biomass (Calabrò, 2009). In addition, for the purpose of estimation, it is assumed that there is no methane oxidation in landfills cover soil and 100 % is released into the atmosphere. Hence, it is possible to estimate the avoided equivalent carbon emissions, based on the conversion factor of methane, in accordance with previous authors (Uusitalo et al., 2014; Friedrich and Toris, 2016; Liamsanguan and Gheewala, 2008). The system boundaries also include collection and transportation of FW from its source to landfills, the so-called horizontal $CO_{2,eq}$ flows. These refer to the cumulative effort of loading waste from single households or other waste generation units from the first stop to the last stop on the collection route and include driving the full truck to the point of unloading. In the estimation, only the energy used for operating the collection trucks is considered. GHG emissions created in the production of collection bags, bins and containers or the manufacture and maintenance of collection vehicles themselves are not considered (Eisted et al., 2009). Due to the on-site AD operation, the overall MSW weight requiring transportation is reduced which highly impacts on $CO_{2,eq}$ emissions produced by collection system (Lee et al., 2007; Lundie and Peters, 2005). In Chile, MSW transportation is performed by diesel-motor trucks. Hence, its contribution may be calculated by the following expression (*Eq. 2;* Zhou et al., 2014):

$$Q = M \cdot P \cdot \mathrm{C} \cdot \mathrm{D} \cdot \rho \qquad \text{(Eq.2)}$$

where M is the mass of FW per annum (Mg/y); P: is the average diesel consumption for transportation of FW (0.022 L/(Mg km)); C is the carbon content of the diesel (86.86 %); D is the average transportation distance of the MSW from city to landfill (approx. 15 km) and ρ is the density of the diesel (0.85 kg/L).

It can be noted that these estimations account only for the collection and transportation of the FW fraction since the proposal does not avoid the use of trucks for collecting the rest of waste produced by households. Therefore, values provided by other authors (Larsen et al., 2009) that include diesel consumption for driving empty truck from garage to collection area and from point of unloading to collection area or garage have been not considered. Furthermore, produced biogas is substituted for fossil fuels and consequently another reduction of GHG emissions must be considered. The avoided impacts due to the use of AD effluent as fertilizer have been excluded since compost may not comply with specific legislative requirements. Despite the remaining digestate may substitute nitrogen, phosphorous and potassium fertilizers with an efficiency of 34.5 %, 46 % and 60 %, respectively (Salemdeeb et al., 2017), effluent should be dewatered and refined, which leads to a more complex design. On the contrary, if the effluent is directly used as fertilizer, without further treatments, some problems such as pathogens, antibiotics and/or heavy metals, among others, might be expected in the long term.

Results and discussion

AD tests

It must be highlighted that it is particularly difficult to obtain representative FW samples and, therefore, an approach appropriate based in statistical methods is required. Despite the randomly strategy for collecting samples and the long term at it was performed, the results might show some scattering when they are comparing, mainly, due to the different eating habits. However, results (summarized in *Table 1*) were found within the range showed by previous authors (Lie et al., 2009; Yirong et al., 2015; Li et al., 2013). Only, carbohydrate and proteins contents were highlighted lower than average values 55.2 %, 15 % and 23.9 %, respectively in dry basis (Zhang et al., 2014).

Table 1. *Food waste (FW) and inoculum characteristics*

Characteristics	Units	FW	Inoculum
Moisture Content[1]	%	78.6 (3.8)	98.81 (0.1)
TS[1]	%	21.4 (3.8)	1.19 (0.8)
VS[1]	%	19.8 (3.7)	0.48 (0.2)
TSS[1]	g/L	13.8 (0.3)	2.02 (0.1)
pH[1]	-	4.42 (0.4)	6.97 (0.2)
ORP[1]	mV	192.5 (35.8)	-219 (17.2)
Total carbohydrates[2]	%	27.8 (4.8)	-
Proteins[2]	%	6.05 (0.64)	-
Oil and grease[2]	%	22.84 (0.1)	-
Soluble Nitrogen[1]	mgN/L	86.2 (26.0)	-
Ash[2]	%	5.48 (0.8)	-
C:N	-	21.49 (4.21)	-
COD[1]	mg O_2/L	33,488 (575)	6,213 (0.1)

[1]Wet basis; [2]dry basis; (-) values between parenthesis show standard deviation

These differences in FW composition highly influence biogas production since substrates poor in easily degradable carbohydrates yield the lowest methane potential. For instance, the aforementioned average values (Zhang et al., 2014) might produce a theoretical SBY of approx. 864 mL/g-VS while the theoretical biogas in case of used FW is approx. 573 mL/g-VS (Drosg et al., 2013). However, there are some other parameters that highly influence the biogas production, such as temperature and waste to inoculum ratio, among others (Liu et al., 2009; Deepanraj et al., 2015).

Batch test clearly shows three different steps (see *Fig. 4a*) and the shape of the obtained curve corresponds to the so-called retarded degradation. The first phase is provided by the substrate itself and the inoculum as the carrier of the methanogenic bacteria consortium. Then, there is an intermediate lag phase, in which hydrolysis is developed and pH drops from 7.00 to 6.00 because of the formed acids. This stage is followed by an exponential growth in biogas production that ends when most of the organic matter is transformed (Córdoba et al., 2016). At this point, the average biogas production was 150±13 mL/g-VS and at the end of the test pH value is set above 6.8 which indicates that the fatty acid concentration does not exceed 200 mg/L and therefore the measurement is accepted. Nevertheless, SBY is lower than values provided by previous BMP tests which are commonly performed at higher temperatures. Temperature plays an important role in terms of SBY regarding the anaerobic biodegradation of the complex organic matter. Thus, several authors have showed a reduction on biogas yield that varies from 31 to 60 % when AD temperature decreases from 30 °C to 15 °C and 55 °C to 20 °C, respectively (Kumar et al., 2016; Bouallagui et al., 2004).

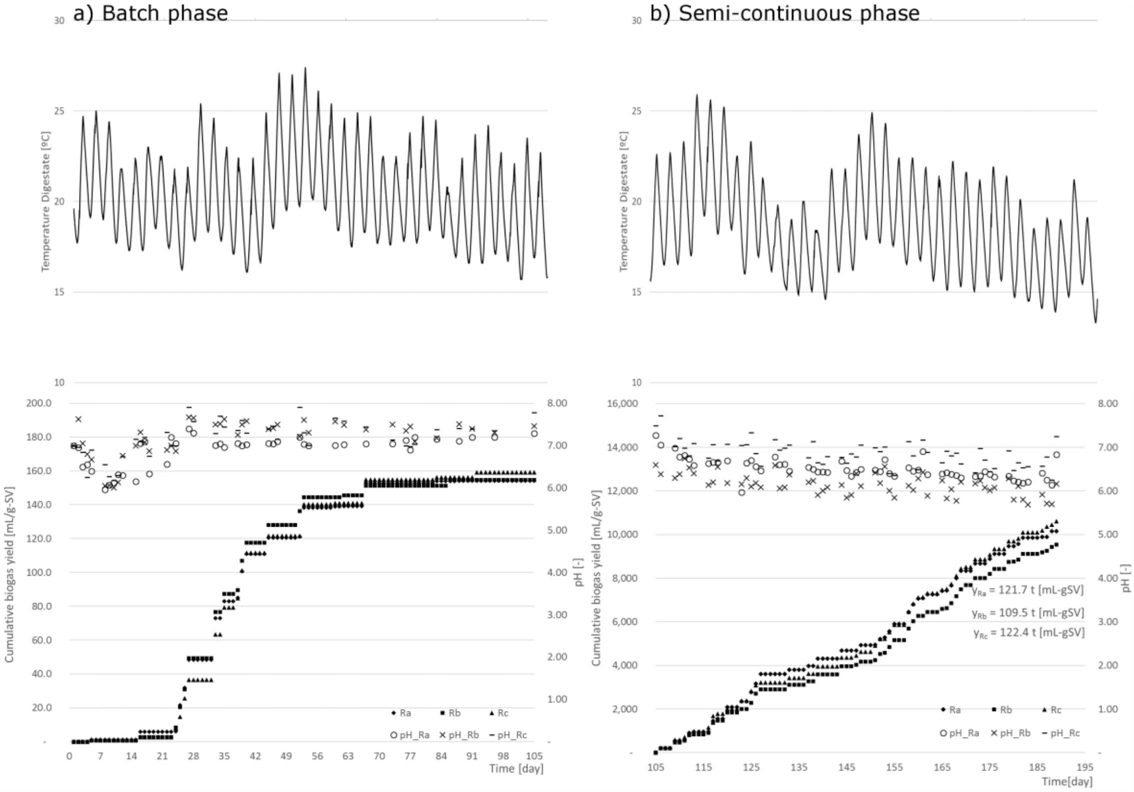

Figure 4. *SBY, temperature and pH during a) batch assays and b) semi-continuous assays*

Although latest reviews have demonstrated the lack of enough researches, regarding AD of FW at psychrophilic temperature, for comparing results (Komilis et al., 2017; Leung and Wang, 2016) the obtained values are still lower than the ones reported by some authors (Bernstad et al., 2013). Comparing SBY can prove difficult as different methods and protocols can all differ from one test to another (Angelidaki, 2009). For instance, the lower SBY, obtained in this research, may be explained due to the SIR. This ratio highly affects AD kinetics and SBY. Thus, SBY may be reduced up to 55 % when SIR is increased from 1.6 to 5.0 (Liu et al., 2009). Other authors, showed even higher variations (approx. 80 %) when SIR is increased from 1.3 to 4.0 (Kumar et al., 2016).

Regarding the semi-continuous phase (see *Fig. 4b*), the temperature lowered, compared to the batch assays from an average of 20.26 ± 2.33 °C to 18.66 ± 2.53 °C. In addition, pH slightly decreased. Hence, batch assays finished with a pH of 7.51 while the semi-continuous operation mode showed a pH of 6.75 after 90 days. During three months, the SBY was approx. constant with an average slope of 118 ± 7 mL/g-VS day. Related to the semi-continuous operation mode there are rare reports about successfully operating single-stage anaerobic digestion of FW at psychrophilic temperature. Besides, to the best of our knowledge, AD of FW without stirring has not been addressed. Stirrer devices are commonly used, at any scale, for avoiding foaming and floating layers. An appropriate agitation technology is essential since stirring has a crucial influence on the microbial community structure and the SBY. Thus, it has been highlighted by previous authors a SBY reduction between 30 % to 40 % when the digestate is unmixed (Rusin et al., 2017; Rojas et al., 2010).

From this point of view, it may be argued whether the effectiveness of a more complex reactor since it has been demonstrated the more complex the reactor, the higher the SBY (Rajagopal et al., 2017). However, complex reactors lead to increase the installation costs and the economic issue surely reduce the number of installed reactors and consequently the available amount of FW treated might be highly reduced. Nevertheless, when similar reactors were operated at similar operation conditions, (i.e. HRT, OLR) other authors found similar biogas production and methane content (Khalil et al., 2016; Lou et al., 2013).

Biogas contained an average of 57.7 % (3 %) of CH_4. H_2S was not detected and a minimum amount of CO (approx. 4 ppm) was measured. Based on SBY and its composition it can be concluded that the expectable amount of methane, in the semi-continuous running mode is approx. 14 L of methane per kg of FW. The biogas composition has been kept quite similar which is a good indicator of the AD stability. Despite volatile organic acids and total inorganic carbonate have not been measured, the pH has not set below 6.8 and the methane content has not decreased. These two latest indicators suggest there was not an ongoing rise in acid concentration.

It is clear that the biogas production is actually low but also clear that the system is very cheap to install and operate. Regarding the organic load rate (OLR) that should be considered for the proposed reactor design, it must be noted that mono-digestion of FW might lead to acidification of the fermentation process via an increasing of OLR (Kastner et al., 2012).

For this reason, it is important to consider the size of the proposed small household reactors according to the estimated daily FW produced by a typical Chilean household (approx. 1.5 kg of FW per day). Thus, cases reported by literature (Khalil et al., 2016; Lou et al., 2013) showed a continuous operation for basic small reactors when OLR varied between 0.6 and 1.0 kg VS/(m^3 day). This suggests a reactor volume from 500 m^3 up to 2000 m^3, depending on the building size as long as the HRT which should be set at approx. 60 days.

COD removal is also a key factor for promoting this decentralized off-grid system, since effluents must abide by Chilean laws (NCh 1333) prior to being discarded to the drainage lines. After the laboratory tests, the substrate showed a final COD value of 269 mg O_2/L. It can be observed that COD was reduced by 97 %, which is in concordance

DOI: http://dx.doi.org/10.15666/aeer/1603_24232439

with the ranges shown in the literature of 80 to 95 % (De Mes et al., 2003). The final obtained COD is even lower than the concentrations of most typical sanitary effluents from human manure, which range between 260 and 480 mg O_2/L (Suarez and Puertas, 2005). In addition, other values related to this effluent quality (see *Table 2*) also meet regulatory standards for being discharged into the municipal sewer system. Hence, it is concluded that the effluent can be directly emptied into the general drainage line without further treatment.

Table 2. *Effluent characteristics*

Characteristics	Value
ORP [mV]	-25.6
pH [-]	9.7
Salinity [%]	0.94
EC [uS/cm]	1,840.0
TDS [ppm]	919

Energy savings and avoided $CO_{2,eq}$

Chile produces yearly more than 3.5 Gg of FW suitable for AD. Assuming that the difference between the FW production from single- and multi-family households is not statistically significant (Schott et al., 2013), the total average amount of FW per household generated in Chile per year can be estimated at 510 kg per household, while in the European Union or India only 190 kg and 319 kg of FW are produced per annum and household, respectively (Bernstad et al., 2012; Rocky et al., 2014).

Results, categorized by region, are summarized in *Table 3*. Despite the metropolitan region (RM) is the largest contributor with more than 1.5 Gg, regions of Arica-Parinacota (XV), Araucania (IX) and Aysen (XI) showed the highest potential methane production due the amount of FW produced per household. In addition, Arica and Parinacota (XV) is one of the smallest regions, in terms of households, which means that could be considered for testing the proposal. On the contrary, the regions of B. O'Higgins (VI), Atacama (III) and Antofogasta (II) show the lowest potential savings, as less FW is produced and more energy is demanded per household each year. It must be noted that the energy consumed by each household, used for estimating potential savings, is based on the substitution of biogas for natural and/or liquefied gas.

Despite the theoretical and economic potential, differences of up to 30 % may be observed due to the reliable amount of FW that can actually be used, in case of decentralized off-grid system, both values may be assumed to be similar (Rios et al., 2016).

Hence, the overall Chilean potential primary energy generation from semi-continuous operation accounts for nearly 425 GWh per year, which represents approximately 0.2 % of the total primary energy consumption (approx. 419.8 TWh). Comparing these results with other values from similar research studies, similar outcomes were found in terms of the percentage of domestic energy consumption: Sweden (0.2 %) (Schott et al., 2013), Mexico (0.8 %) (Rios et al., 2016), Brazil (0.26 %) (Salomon and Lora, 2009) and Uruguay (0.2 %) (Moreda, 2016). From this point of view, AD of FW at a decentralized scale does not seem to be the best option for adding renewable energy to the Chilean energy supply.

Table 3. *Summary of FW produced, energy consumption and potential energy generation by AD of FW categorized by household and region*

Region	Total FW produced [t]	FW produced per household [kg]	Potential energy from AD of FW per household [kWh]
I	105,266.4	1,141.5	195.9
II	108,861.9	685.7	117.7
III	57,363.9	684.1	117.4
IV	122,489.0	541.3	92.9
V	325,883.0	585.5	100.5
VI	133,011.4	484.0	83.1
VII	199,579.5	633.2	108.7
VIII	358,202.3	581.8	99.8
IX	235,839.8	795.5	136.5
X	205,160.4	802.4	137.7
XI	24,911.5	751.4	128.9
XII	35,785.0	721.1	123.7
XIV	81,838.4	697.9	119.8
XV	63,495.6	1,200.2	206.0
RM	1,556,899.0	738.6	126.8

However, households may achieve an average saving of 1.5 % in terms of primary energy consumption, which reinforces the separation of FW at the generation point and the corresponding reduction of organic matter disposal. In spite of relatively low potential energy savings, there is an important impact in terms of avoided GHG emissions. These can be divided into two main categories: horizontal flows, produced by the MSW transport systems and the direct GHG emissions from landfills and fossil fuels replacement. Then, approximately 713 Gg $CO_{2,eq}$ can be avoided yearly from landfills emissions and 878.6 Gg $CO_{2,eq}$ from the reduction of fuel consumption in MSW transportation services. In addition, taking into account that biogas has a neutral carbon cycle and considering EPA emissions factors, which report an average of 2.15 kg $CO_{2,eq}$ per cubic meter of natural gas (measured at normal conditions), it can be estimated that around 112.8 Gg $CO_{2,eq}$ might be avoided when this fuel is replaced by methane.

Conclusions

The results suggest that AD of FW in Chile might incorporate approx. 425 GWh per annum (in terms of primary energy) by using household scale reactors in a decentralized off-grid system. Hence, firewood and natural or liquefied gas households can be partially replaced which impacts on the reduction of household expenses.

Despite the energy provided is quite lower than other alternatives based on high rate anaerobic reactors at psychrophilic temperature, depending on the corresponding installation costs of, it may be desirable to operate economical and simpler versions while these have shown a positive impact in terms of environmental protection.

Hence, due to the reduction of the amount of MSW that requires collection, transportation and disposal and the partial substitution of natural gas, it is estimated that 1,704 Gg $CO_{2,eq}$ emissions can be avoided annually.

This positive effect on environment may be used by Chilean authorities in order to support the required financial policies which have been highlighted as an encouraging key factor in other countries (mostly in China, India and EU). Results must be considered as a basic framework that may orient the decision-making process for the implementation of short-term environmental policies. At this point, XV region is suggested for testing this proposal at pilot scale due to the high estimated savings. Further investigations should be developed regarding the operational aspects with the aim of improving the technical design of reactors or the maintenance operations, among others.

Acknowledgements. Special thanks go out to the Chilean Ministry of Education, in particular to CONICYT which has been supported this research thorough the FONDECYT project number 11.140.728.

REFERENCES

[1] Alfonso, D., Perpiña, C., Pérez-Navarro, A., Peñalvo, E., Vargas, C., Cárdenas, R. (2009): Methodology for optimization of distributed biomass resources evaluation, management and final energy use. – Biomass Bioenerg. 33: 1070-1079.

[2] Angelidaki, I., Alves, M., Bolzonella, D., Borzacconi, L., Campos, J. L., Guwy, A. J., Kalyuzhnyi, S., Jenicek, P., Van Lier, J. B. (2009): Defining the biomethane potential (BMP) of solid organic wastes and energy crops: A proposed protocol for batch assays. – Water Sci. Technol. 59(5): 927-934.

[3] APHA, AWWA, and WEF (2005): Standard Methods for the Examination of Water and Wastewater, 21st ed. – American Public Health Association, New York.

[4] Bernstad, A., Jansen, J. (2012): Separate collection of household food waste for anaerobic degradation. Comparison of different techniques from a systems perspective. – Waste Manage. 32: 806-815.

[5] Bernstad, A., Davidsson, A., Tsai, J., Persson, E., Bissmont, M., Jansen J. C. (2013): Tank-connected food waste disposer systems. Current status and potential improvements. – Waste Manage. 33: 193-203.

[6] Bidart, C., Fröhling, M., Schultmann, F. (2013): Municipal solid waste and production of substitute natural gas and electricity alternatives. – Appl. Therm. Eng. 51: 1107-1115.

[7] Bond, T., Templeton, M. R. (2011): History and future of domestic biogas plants in the developing world. – Energy Sustainable Dev. 15(4): 347-354.

[8] Bouallagui, H., Haouari, O., Touhami, Y., Ben Cheikh, R., Marouani, L., Hamdi, M. (2004): Effect of temperature on the performance of an anaerobic tubular reactor treating fruit and vegetable waste. – Process Biochem. 9(12): 2143-2148.

[9] Buysman, E., Mol, A. P. J. (2013): Market-based biogas sector development in least developed countries. The case of Cambodia. – Energ. Policy 63: 44-51.

[10] Calabrò, P. S. (2009): Greenhouse gases emission from municipal waste management: The role of separate collection. – Waste Manage. 29: 2178-2187.

[11] Chen, Y., Cheng, J. J., Creamer, K. S. (2008): Inhibition of anaerobic digestion process: A review. – Bioresour. Technol. 99(10): 4044-4064.

[12] Chilean Ministry of Environment (2011): Second National Communication for the United Nations Framework. Convention on Climate Change (UNFCCC). – http://www.mma.gob.cl (accessed January 2018; Spanish).

[13] CONAMA (2009): National Inventory of Dioxins and Furans emissions. – National Council of the Environment, Santiago, Chile (Spanish).

[14] CONAMA (2010a): First report about solid residues management in Chile. Gobierno de Chile. – http://www.sinia.cl/1292/articles-49564_informe_final.pdf (accessed November 2017; Spanish).

[15] CONAMA (2010b): Data Acquisition, Analysis, Treatment and Publication of National Information about Solid Residues of Chile. – National Council of the Environment, Santiago, Chile, Spanish.

[16] Córdoba, V., Fernández, M., Santalla, E. (2016): The effect of different inoculums on anaerobic digestion of swine wastewater. – J. Environ. Chem. Eng. 4(1): 115-122.

[17] De Mes, T. Z. D., Stams, A. J. M., Reith, J. H., Zeeman, G. (2003): Methane Production by Anaerobic Digestion of Wastewater and Solid Wastes. – In: Reith, J. H., Wijffels, R. H., Barten, H. (eds.) Bio-Methane & Bio-Hydrogen: Status and Perspectives of Biological Methane and Hydrogen Production. Chapter 4, pp. 58-102. Dutch Biological Hydrogen Foundation, The Hague.

[18] Deepanraj, B., Sivasubramanian, V., Jayaraj, S. (2015): Kinetic study on the effect of temperature on biogas production using a lab scale batch reactor. – Ecotoxicol. Environ. Saf. 121: 100-104.

[19] Drosg, B., Braun, R., Bochmann, G., Al Saedi, T. (2013): Analysis and Characterisation of Biogas Feedstocks. – Wellinger A., Murphy J., Baxter. D. (eds.) The Biogas Handbook. Science, Production and Applications, pp. 52-84. Woodhead Publishing, Cambridge, UK.

[20] Eisted, R., Larsen, A. W., Christensen, T. H. (2009): Collection, transfer and transport of waste: accounting of greenhouse gases and global warming contribution. – Waste Manag. Res. 27: 738-745.

[21] Friedrich, E., Toris, C. (2016): Current and future greenhouse gas (GHG) emissions from the management of municipal solid waste in the eThekwini municipality South Africa. – J. Clean. Prod. 112: 4071-4083.

[22] Galina, C. (2008): Modeling the carbon cycle of urban systems. – Ecol. Model. 216(2): 107-113.

[23] Guerrero, L. A., Maas, G., Hogland, W. (2013): Solid waste management challenges for cities in developing countries. – Waste Manage. 33(1): 220-232.

[24] Ignaciuk, A., Vöhringer, F., Ruijs, A., van Ierland, E. C. (2006): Competition between biomass and food production in the presence of energy policies: A partial equilibrium analysis. – Energy Policy 34(10): 1127-1138.

[25] Iqbal, S. A., Rahaman, S., Rahman, M., Yousuf, A. (2014): Anaerobic digestion of kitchen waste to produce biogas. – 10th International Conference on Mechanical Engineering, ICME 2014 Procedia Engineering 90: 657-662.

[26] Kastner, V., Somitsch, W., Schnitzhofer, W. (2012): The anaerobic fermentation of food waste: a comparison of two bioreactor systems. – J. Clean. Prod. 34: 82-90.

[27] Khalil, C. A., Ibrahim, L., Ibrahim, E., Ghanimeh, S. (2016): Clean energy generation through psychrophilic anaerobic digestion of food and landscape wastes. – 3rd International Conference on Renewable Energies for Developing Countries, REDEC 2016, art. no. 7577568.

[28] Komilis, D., Barrena, R., Grando, R. L., Vogiatzi, V., Sánchez, A., Font, X. (2017): A state of the art literature review on anaerobic digestion of food waste: influential operating parameters on methane yield. – Rev. Environ. Sci. Biotechnol. 16(2): 347-360.

[29] Kumar, P., Hussain, A., Dubey, S. K. (2016): Methane formation from food waste by anaerobic digestion. – Biomass Convers. Biorefinery 6(3): 271-280.

[30] Larsen, A. W., Vrgoc, M., Lieberknecht, P., Christensen, T. H. (2009): Diesel consumption in waste collection and transport and its environmental significance. – Waste Manag. Res. 27: 652-659.

[31] Lee, S., Choi, K., Osako, M., Dong, J. (2007): Evaluation of environmental burdens caused by changes of food management systems in Seoul, Korea. – Science of the Total Environment 387: 42-53.

[32] Leung, D. Y. C., Wang, J. (2016): An overview on biogas generation from anaerobic digestion of food waste. – Int. J. Green Energy 13(2): 119-131.

[33] Li, R., Chen, S., Li, X. (2009): Anaerobic co-digestion of kitchen waste and cattle manure for methane production. – Energy Sources, Part A: Recovery, Utilization, and Environmental Effects 31(20): 1848-1856.

[34] Li, Z., Lu, H., Ren, L., He, L. (2013): Experimental and modeling approaches for food waste composting: A review. – Chemosphere 93: 1247-1257.

[35] Liamsanguan, C., Gheewala, S. H. (2008): The holistic impact of integrated solid waste management on greenhouse gas emissions in Phuket. – J. Clean. Prod. 16: 1865-1871.

[36] Liu, G., Zhang, R., El-Mashad, H. M., Dong, R. (2009): Effect of feed to inoculum ratios on biogas yields of food and green wastes. – Bioresour. Technol. 100(21): 5103-5108.

[37] Lou, X. F., Nair, J., Ho, G. (2013): Potential for energy generation from anaerobic digestion of food waste in Australia. – Waste Management and Research. 31(3): 283-294.

[38] Lundie, S, Peters, G. (2005): Life cycle assessment of food waste management options. – J. Clean. Prod. 13: 275-286.

[39] Massé, D. I., Masse, L., Xia, Y., Gilbert, Y. (2010): Potential of low-temperature anaerobic digestion to address current environmental concerns on swine production. – J. Anim. Sci. 88(13): 112-120.

[40] Moreda, I. L. (2016): The potential of biogas production in Uruguay. – Renewable Sustainable Energy Rev. 54: 1580-1591.

[41] OECD (2005): Environmental Performance Review: Chile. – OECD Publishing, Paris.

[42] Olsson, L., Fallde, M. (2015): Waste potential: a socio-technical analysis of biogas production and use in Sweden. – J. Clean. Prod. 98: 107-115.

[43] Perez, C. (2010): Census and Analysis of Residue Generation in Chile. Technical Report. – Technological Development Unit and National Environment Commission, Santiago, Chile (Spanish).

[44] Rajagopal, R., Bellavance, D., Rahaman, M. S. (2017): Psychrophilic anaerobic digestion of semi-dry mixed municipal food waste: For North American context. – Process Saf. Environ. Prot. 105: 101-108.

[45] Reddy, A. K. N. (2004): Lessons from the Pura community biogas project. – Energy Sustainable Dev. 8: 68-73.

[46] Remais, J., Chen, L., Seto, E. (2009): Leveraging rural energy investment for parasitic disease control: Schistosome ova inactivation and energy co-benefits of anaerobic digesters in rural China. – PLoS ONE 4(3): e4856.

[47] Righi, S., Oliviero, L., Pedrini, M., Buscaroli, A., Casa, C. D. (2013): Life Cycle Assessment of management systems for sewage sludge and food waste: centralized and decentralized approaches. – J. Clean. Prod. 44: 8-17.

[48] Rios, M., Kaltschmitt, M. (2016): Electricity generation potential from biogas produced from organic waste in Mexico. – Renew. Sust. Energ. Rev. 54: 384-395.

[49] Rocky, T. H., Islam, M., Saha, U. K. (2014): Resource recovery potential from kitchen waste. – 2nd International Conference on Green Energy and Technology, ICGET 2014, art. no. 6966654: 18-21.

[50] Rojas, C., Fang, S., Uhlenhut, F., Borchert, A., Stein, I., Schlaak, M. (2010): Stirring and biomass starter influences the anaerobic digestion of different substrates for biogas production. – Eng. Life Sci. 10(4): 339-347.

[51] Rusin, J., Chamradova, K., Grycova, B. (2017): The influence of biomass agitation on biogas and methane production using the high-solids thermophilic anaerobic digestion. – Green Process. Synth. 6(3): 273-279.

[52] Salemdeeb, R., Zu Ermgassen, E. K. H. J., Kim, M. H., Balmford, A., Al-Tabbaa, A. (2017): Environmental and health impacts of using food waste as animal feed: a comparative analysis of food waste management options. – J. Clean. Prod. 140: 871-880.

[53] Salomon, K. R., Lora, E. E. S. (2009): Estimate of the electric energy generating potential for different sources of biogas in Brazil. – Biomass Bioenerg. 33: 1101-1107.

[54] Schott, A. B., Vukicevic, S., Bohn, I., Andersson, T. (2013): Potentials for food waste minimization and effects on potential biogas production trough anaerobic digestion. – Waste Manage. Res. 31(8): 811-819.

[55] Seiffert, M., Kaltschmitt, M., Miranda, J. A. (2009): The biomethane potential in Chile. – Biomass Bioenerg. 33: 564-572.

[56] SERNAC (2015): Website of the Ministry of Economy, Promotion and Tourism. – www.sernac.cl (accessed January 2018; Spanish).

[57] Suarez, J., Puertas, J. (2005): Determination of COD, BOD, and suspended solids loads during combined sewer overflow (CSO) events in some combined catchments in Spain. – Ecol. Eng. 24: 201-219.

[58] Uusitalo, V., Havukainen, J., Manninen, K., Höhn, J., Lehtonen, E., Rasi, S., Soukka, R., Horttanainen, M. (2014): Carbon footprint of selected biomass to biogas production chains and GHG reduction potential in transportation use. – Renew. Energ. 66: 90-98.

[59] Walekhwa, P. N., Mugisha, J., Drake, L. (2009) Biogas energy from family-sized digesters in Uganda: Critical factors and policy implication. – Energ. Policy 37(7): 2754-2762.

[60] Walker, M., Theaker, H., Yaman, R., Poggio, D., Nimmo, W., Bywater, A., Blanch, G., Pourkashanian, M. (2017): Assessment of micro-scale anaerobic digestion for management of urban organic waste: A case study in London, UK. – Waste Manage. 61: 258-268.

[61] Wen, Z., Wang, Y., De Clercq, D. (2016): What is the true value of food waste? A case study of technology integration in urban food waste treatment in Suzhou City, China. – J. Clean. Prod. 118: 88-96.

[62] Yirong, Y., Heaven, S., Banks, C. J. (2015): Effect of a trace element addition strategy on volatile fatty acid accumulation in thermophilic anaerobic digestion of food waste. – Waste Biomass Valor 6: 1-12.

[63] Zhai, N., Zhang, T., Yin, D., Yang, G., Wang, X., Ren, G., Feng, Y. (2015): Effect of initial pH on anaerobic co-digestion of kitchen waste and cow manure. – Waste Manage. 38(1): 126-131.

[64] Zhang, C., Su, H., Baeyens, J., Tan, T. (2014): Reviewing the anaerobic digestion of food waste for biogas production. – Renew. Sust. Energ. Rev. 38: 383-392.

[65] Zhou, C., Huang, H., Cao, A., Xu, W. (2014): Modeling the carbon cycle of the municipal solid waste management system for urban metabolism. – Ecol. Model. 318: 150-156.