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Assessment of biogas production in Argentina from co-digestion of sludge and municipal solid waste

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ABSTRACT

In Argentina, there is an important potential to utilize organic waste to generate bioenergy. This work analyzes the environmental impacts and the energetic and economic requirements of the biogas produced by digesting the sewage sludge (SS) produced in a wastewater treatment plant in a medium city in Argentina. The SS is co-digested with the organic fraction of municipal solid waste (OFMSW), and the basis of this study is the life cycle assessment (LCA). The LCA is performed according to ISO 14040-44 using the SimaPro simulator. First, the transport of the raw materials to the biogas plant was defined. Then, the co-digestion and the biogas treatment for final use were evaluated. The co-digestion was improved with glycerol, and the generation of biogas was estimated using the GPS-X software. Two alternatives for the end use of biogas were considered: combined heat and power (CHP) and biomethane generation. For the first, H_2S and water vapor were removed from the raw biogas stream, and for the second, also CO_2 was removed. The H_2S removal process was simulated in the SuperPro software by anaerobic biofiltration. The same software was used to simulate the removal of CO_2 absorption-desorption with water as solvent. Finally, the environmental impacts related to the end use of biogas (CHP and biomethane) were evaluated. The environmental, energetic and economic analyses showed that the co-digestion of SS and OFMSW has great potential for reducing the environmental impacts and increasing the economic and energetic value of the substances via the production of biomethane, electricity and, potentially, fertilizer.

1. Introduction

Biogas is a renewable energy that is produced from different types of biomass, including energy crops, municipal solid waste, sewage and waste from agriculture, livestock and some industrial activities. A complete discussion on production, conditioning and utilization of biogas can be found in Budzianowski (2016). That work presents many potential process innovations from most recent patent and academic literature. Currently, considering the great potential that exists in Argentina, there is growing interest in the use of biomass for energy purposes to diversify its energy matrix. Municipal solid waste and sewage are the main organic wastes generated in urban centers, and treatment in Argentina is beginning to be required by local laws. The use of organic waste to generate energy is a very interesting alternative because it allows energy generation to be further decentralized. Biogas technical capacity is fairly well developed in Argentina. An inventory of biogas plants in Argentina list 105 anaerobic digesters in 16 provinces belonging to the public sector, the private sector, production cooperatives and non-governmental organizations (Goicoa, 2016). A survey of 61 of those biogas plants indicates that a large number of plants belong to the private sector (53.1%) with the objective to treat effluents and only a small portion (6%) to get energy. The biogas plants in the public sector are mainly used to treat effluents and a 33% are used for research and teaching. Several plants belonging to municipalities present operational and management problems. There are important differences between public and private. The private sector has larger plants built in rural areas using USAB, cover lagoons and continuous-flow stirred tank reactors with imported technology and materials. A common ground in both sectors is the lack of heating and mixing of the digesters indicating that the plant is not working in the optimum biological condition. The more common substrates are industrial and agro wastes (86%). The rest is organic municipal wastes with a small contribution from crops. The 42.6% of the plants do not use the biogas, 44.3% use the biogas for heating and only 12% have some sort

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of electrical use. Besides, Argentina has programs (EMGIRSU and PROBIOMAS) and two institution (INTA and INTI) targeting the development of the biogas technology. Biogas was included with a small participation (15 MW over 1000 MW) in the recent program RenovAr (2016) which offered a favorable tariff for renewable energy based electricity production.

To date, a wide range of studies have evaluated the impacts associated with biogas generation systems using the life-cycle assessment (LCA) (Borjesson and Berglund, 2006; Jury et al., 2010; Dressler et al., 2012; Lijó et al., 2014a; Morero et al., 2015b). These studies analyze a large variety of raw materials and end uses of biogas. Bacenetti et al. (2016) reviewed the LCA studies carried out in different countries focusing on agricultural AD plants. The review shows that the goal, scope, life cycle impact assessment methodology and feedstock vary widely making it difficult to compare the different LCA studies and to obtain common conclusions on the environmental impact of biogas production. Bacenetti et al. (2016), also, analyzed four plants and pointed out that energy crops production, anaerobic digester operation, and digestate emission from open tanks are the main contributors to environmental impacts. At the same time, papers have been published associated with the co-digestion of biogas, most of which focused on the use of crops as co-substrates (De Vries et al., 2012; Lansche and Müller, 2012; Poeschl et al., 2012a; Bacenetti et al., 2013; Lijó et al., 2014b). The advantage to this solution is that plants can achieve higher yields compared to the use of a single raw material. Additionally, the quality of the biogas is improved. Recent studies evaluating the co-digestion of livestock manure from organic waste (Poeschl et al., 2012b; De Vries et al., 2012; Rodriguez-Verde et al., 2014; Lijó et al., 2015) concluded that the use of organic waste as a co-substrate generates less of an environmental impact than does the use of energy crops. In addition, these studies demonstrate the environmental benefits of using the waste, resulting from the absence of impacts in its production, as opposed to the high environmental impacts of energy crop production. These works conclude that to maximize the environmental benefit, a higher proportion of agricultural or organic waste should be included in co-digestion. Therefore, an interesting alternative is to evaluate the co-digestion of the most abundant waste generated in a city.

This paper, carried out for first time in Argentina, analyzes the environmental impacts and the economic and energetic requirements of biogas production by digesting the sewage sludge (SS) produced in the wastewater treatment plant in a medium city. The SS is co-digested with the organic fraction of municipal solid waste (OFMSW), and the basis of the study is the life cycle assessment (LCA). The LCA was performed according to ISO 14040-44 (ISO 14044: 2006) using the SimaPro simulator (Pré Consultants, 2015).

2. Materials and methods

In Argentina, the largest amount of waste is disposed in sanitary landfills or in informal dumps without proper treatment. Only the 60% of the population reaches an adequate treatment of waste (SAyDS, 2015). In localities that have built landfills, there is no separation of organic waste generally, and the organic matter has the same disposal as non-recyclable waste. Therefore, it is interesting to propose alternatives to reduce both the environmental impacts and the costs of waste treatment to generate an attractive project.

The bioenergy processes included the collection and transportation of wastes, anaerobic digestion, biogas treatment (desulphurization, upgrading) and digestate management according to its use (organic fertilizer or incineration). The study was performed according to the daily production of OFMSW (17.5 t) and SS (24.4 m^3) in a medium-sized city (34,000 inhabitants) in Argentina.

2.1. LCA methodology

The LCA was conducted according to ISO 14040-44 (ISO 14040: 2006; ISO 14044: 2006) in four phases: the goal and scope definition phase, the inventory analysis phase, the impact assessment phase and the interpretation phase.

2.1.1. Goal and scope definition phase

The objectives and scope of the LCA were to determine the environmental impacts associated with each Stage of biogas production from the co-digestion of SS and OFMSW and its subsequent use as biomethane or electricity. In addition, we evaluated the environmental impacts associated with the management of digestate.

2.1.2. Functional unit

The functional unit chosen was the amount of organic waste (feedstock) that was annually digested (21,000 t). This amount corresponded to the OFMSW and SS generated in the town with the addition of glycerol, which was added to improve biogas production.

2.1.3. System boundaries

The boundaries of the system include the transport of the different feedstocks to the biogas plant and the materials and energy used. The materials and energy included those used in anaerobic digestion, in the desulphurization plant, in the upgrading biogas plant and in the conditioning of biogas for final use either as biomethane or as CHP. In addition, two digestate end-use options (organic fertilizer or incineration) were evaluated. Finally, the total process was compared in environmental and economic terms with a current waste treatment practice (landfill).

The system boundaries of the system under assessment was divided into four stages: feedstock transport (Stage 1), biogas production plant (Stage 2), digestate management (Stage 3) and biogas treatment plant (Stage 4). In Fig. 1 the system boundaries of each Stage are shown.

2.1.4. Inventory data

For this study the inventory data was collected from different sources, simulations and procedures. Sections 2.1.4.1–2.1.4.4 describe how the inventory of each Stage was made. Data regarding the production of mineral fertilizers, electricity, heat, diesel and sanitary landfill infrastructure were obtained from the ecoinvent database version 3 (Weidema et al., 2013).

2.1.4.1. Stage 1: feedstock transport. In Stage 1, the inputs and outputs were considered. These were associated with the transport of the feedstocks through the anaerobic digestion plant and the energy used in the municipal solid waste plant. It was assumed that the anaerobic digestion plant was located on the same site as the sewage treatment plant, so there would be no transport of SS. Data from a real plant located in a medium-sized city of Argentina (Groppelli et al., 2008) were used to calculate the electricity consumption of the municipal solid waste plant (related to the sorting, crushing and pressing of the OFMSW). Moreover, the transportation distance of OFMSW from the landfill to the anaerobic digestion plant was considered to be 9 km. The biodiesel plant (which produces glycerol as a by-product) was located 50 km from the biogas plant. Table 1 summarizes the annual inventory for the first Stage.

2.1.4.2. Stage 2: biogas production plant. The anaerobic digestion scenario was simulated using GPS-X v6.0.2 (Hydromantis, 2010).

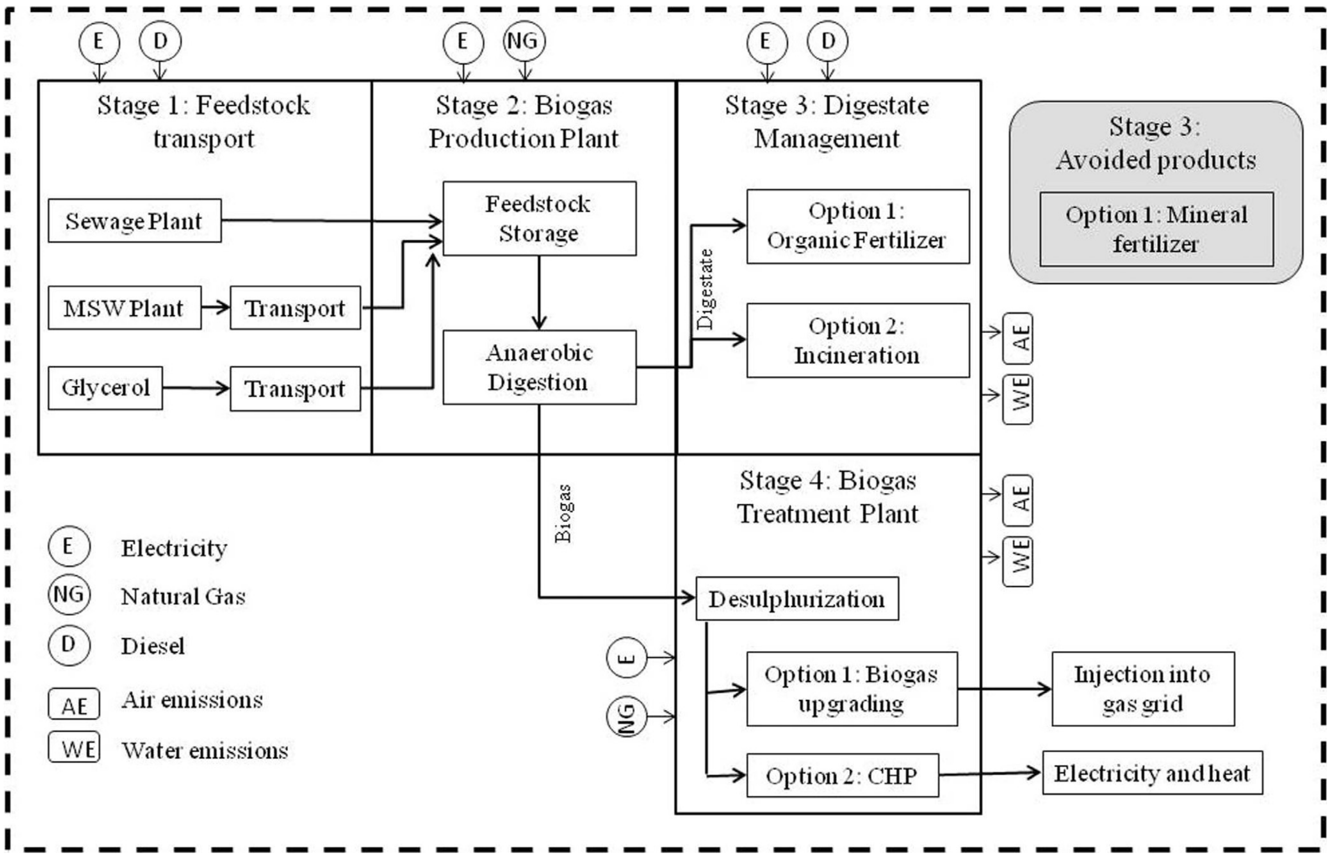

Fig. 1. Flow chart and system limits for bioenergy system being studied at different Stages.

Table 1
Annual inventory for Stage 1 (feedstock transport).

Input		Output	
Electricity (MSW plant)	82,474 kWh	Feedstock	21,072 t
Diesel (OFMSW transport)	2350 lt		
Diesel (Glycerol transport)	1826 lt		

Co-digestion substrates were characterized according to the literature (Forster-Carneiro et al., 2008; Alvarez et al., 2010). Suitable mixtures and total amount calculations were defined based on previous experiences with co-digestion problems, such as digester overload and acid inhibition (pH under 6.5). Anaerobic digestive simulations were performed at mesophilic conditions (35 °C) in a continuous-flow stirred tank reactor without recycling, with a hydraulic retention time of 20 days. The reactor's energy consumption (for heating and stirring) were estimated using SuperPro Designer (SuperPro v9.5). The heat produced by the CHP was used to heat the digester. In biomethane process the digester heating was performed with natural gas. Fig. 2 diagrams the process flow for the generation of biomethane (Fig. 2a) and CHP (Fig. 2b). Table 2 describes the annual inventory for Stage 2.

2.1.4.3. Stage 3: digestate management. Option 1 (*organic fertilizer*): The main advantage of dewatered biosolids is that farmers can apply them to the land by using conventional manure spreaders. The recommended solids concentrations range from 15% to 35% (Turovskiy and Mathai, 2006).

The dehydration process was simulated in the SuperPro Designer software, where a belt filter was used to obtain 30% of Total Solid (TS). It was assumed that the fertilizer was used in eucalyptus plantations near the town (70 km) because there is strict legislation in Argentina regarding the agricultural use of SS as fertilizer (Resolution 97/2001). In the case of forest plantations, nutrient spreading is annual (McFarland, 2001), so the digestate will be stored during that time. According to Poeschl et al., 2012b, the biogas production during digestate storage is close to 10%, so these emissions are accounted for in the inventory. The emissions generated by the spreading of digestate on the plantations were estimated using the 2006 IPCC Guidelines (see Supplementary Material), whereas the emissions to water were obtained from the simulation in the GPS-X software (see Table 3).

Products to avoid that were considered in this Stage were related to the amount of mineral fertilizer used in eucalyptus plantations. According to the literature (Carpineti et al., 1995), for plantations in this region, the necessary amount of fertilizer is 50 kg urea/ha and 110 kg of triple superphosphate/ha. The nitrogen fertilizer replacement values (NFRV) were used to calculate the avoided mineral N fertilizer from digestate. The NFRV were considered to be 65% for digestate (De Vries et al., 2012). Therefore, given the composition of digestate (obtained from the simulation in the GPS-X, see Supplementary Material), it would be possible to apply the required dose of N over 730 ha while avoiding 36 tons urea/year and 3.5 t triple superphosphate/year (in this case, it would not be possible to meet the total P dose, so an additional mineral fertilizer would be needed for that nutrient).

Option 2 (*Incineration*): The main parameter for digestate incineration is moisture. Digestates with a solids concentration of 30–50% (50–70% moisture) are autogenous; i.e., they can be burned without auxiliary fuel (Turovskiy and Mathai, 2006). Therefore, prior to incineration, the digestate moisture content must be reduced. To avoid the use of auxiliary fuel, this option, similar to the previous one, was simulated in the SuperPro software by using

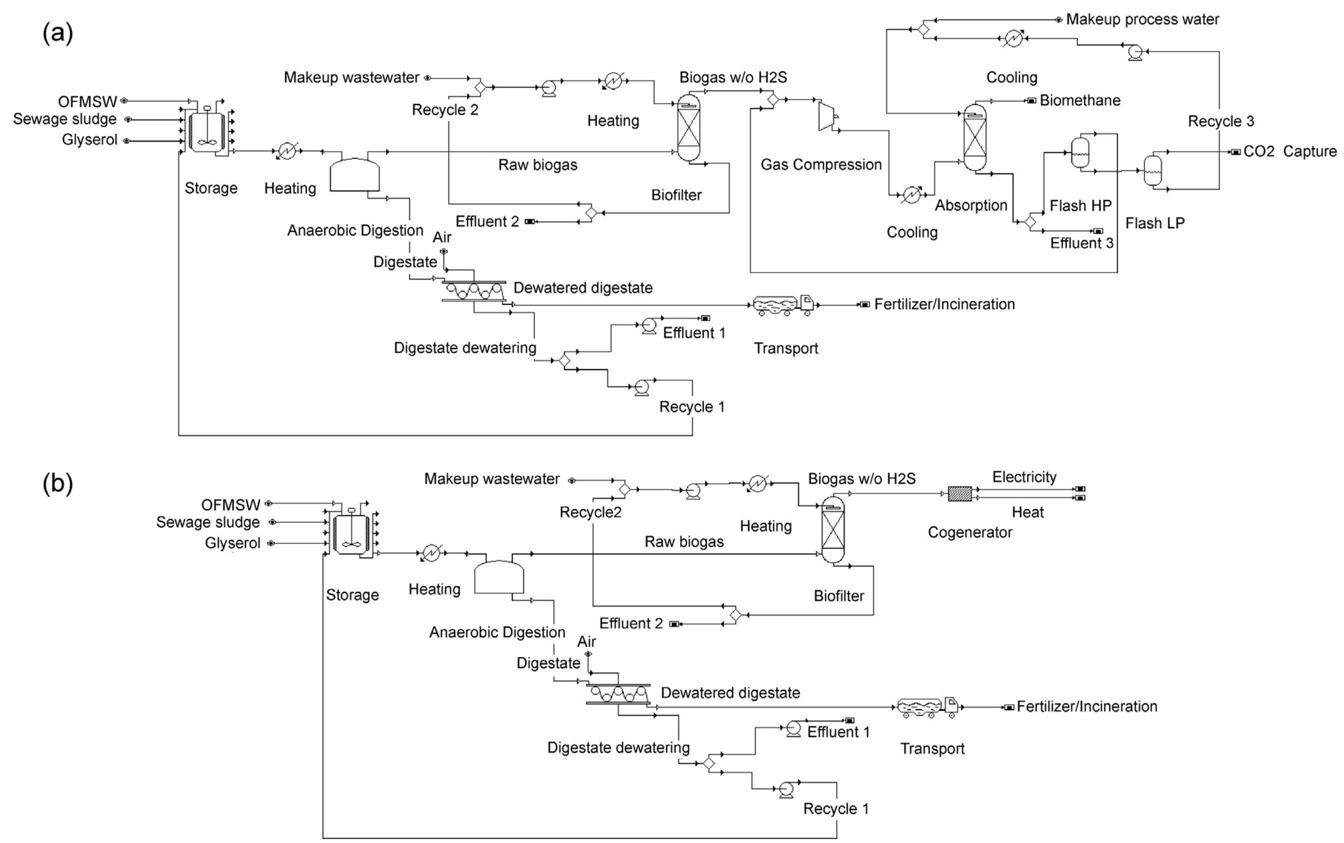

Fig. 2. Process flow diagram for generating (a) biomethane and (b) CHP.

Table 2
Annual inventory for Stage 2 (Biogas production plant).

Input		Output	
Feedstock	21,072 t	Digestate	21,707 t
Recycle (water)	1650 t	Raw biogas	1015 t
Electricity	97,267 kWh		
Natural Gas	54,324 m^3		

a belt filter to obtain 50% of TS. Thus, the transportation was reduced because, as in the previous option, a distance of 70 km to the incinerator was assumed. Emissions to air generated during incineration were calculated using the 2006 IPCC Guidelines (see Supplementary Material), and the amount of ash generated during the process was estimated from the literature (Turovskiy and Mathai, 2006). Emissions to water were obtained from the simulation in the GPS-X software.

2.1.4.4. Stage 4: biogas treatment plant. Two alternatives for the end use of biogas were considered: generation of biomethane to inject into the natural gas grid and generation of electricity. In both cases, it was first necessary to remove the H_2S from the biogas. That process required a biofilter working under anoxic conditions with recirculation of an aqueous stream rich in nitrate from the sewage treatment plant. The simulation of the biofilter was performed in SuperPro assuming an H_2S removal efficiency of 99% (Soreanou et al., 2008). In Option 1 (biomethane generation), an upgraded biogas plant was simulated in SuperPro using water as a solvent because previous studies showed that water was the most economical alternative and had the smallest environmental impact (Morero et al., 2015a). The upgrading process was simulated assuming a CO_2 removal of 98%, a remaining H_2S removal of 45%, and a methane loss of 3%. In Option 2 (electricity generation), the calculations were performed assuming an engine with a power conversion efficiency of 39% and a thermal conversion efficiency of 45%. Emissions related to heat and power cogeneration (CHP)

Table 3
Annual inventory for Stage 3 (Digestate Management).

Input		Output	
Option 1: Organic fertilizer			
Digestate	21,707 t	Organic fertilizer	6927 t
Electricity	668 kWh	Recycle (water)	1650 t
Diesel (Fertilizer transport)	21,923 lt	*Air emissions*	
Diesel (Fertilizer spreading)	4110 lt	Nitrous oxide	0.6 t
Avoided products		CH_4	35.53 t
Triple superphosphate	3.57 t	CO_2	65.98 t
Urea	36.18 t	*Water emissions*	
Air emissions avoided		Effluent 1 (Fig. 2a)	13,130 t
CO_2	40.81 t	COD	−1180 t
Nitrous oxide	0.56 t	Ammonia	1 t
Water emissions avoided		PO_4-P	0.59 t
Phosphate	0.02 t	Alkalinity	−12.12 t
Nitrous oxide	0.31 t	Nitrous oxide	0.31 t
		Phosphate	0.02 t
Option 2: Incineration			
Digestate	21,707 t	Digestate to incineration	4158 t
Electricity	794 kWh	Recycle (water)	1650 t
Diesel (Dry digestate transport)	13,161 lt	*Air emissions*	
		Nitrous oxide	4.12 t
		CH_4	3812 t
		Final Disposal	
		Ash	1248 t
		Water emissions	
		Effluent 1 (Fig. 2b)	15,899 t
		COD	−1131 t
		Ammonia	2.14 t
		Soluble PO_4-P	0.72 t
		Alkalinity	−9.07 t

were obtained from the information provided by the equipment manufacturer (see Table 4).

2.1.5. Impact assessment

The LCA was performed using the SimaPro 8 software with ecoinvent database version 3 (Weidema et al., 2013). The ReCiPe midpoint methodology (Goedkoop et al., 2009) was used with eight impact categories: Climate change (GWP) [kg CO_2 eq], Terrestrial acidification (TA) [kg SO_2 eq], Freshwater eutrophication (FE) [kg P eq], Human toxicity (HT) [kg 1,4-DB eq], Photochemical oxidant formation (POF) [kg NMVOC], Particulate matter formation (PMF) [kg PM10 eq], Urban land occupation (ULO) [m2a] and Fossil depletion (FD) [kg oil eq].

2.2. Energy analysis

An energy analysis was carried out based on the cumulative energy demand (CED) to investigate the energy use throughout the life cycle. This includes the direct uses as well as the indirect consumption of energy due to the use (Hischier et al., 2010). The analysis incorporated the non-renewable fossil and nuclear energy demand and the renewable water energy demand. We selected these categories because Argentina electricity mix is highly compose of fossil fuels (61%), nuclear (3.5%) and hydro power (35.5%) (SEN, 2015).

2.3. Economic analysis

The SuperPro simulator was used to simulate the different Stages of biomethane generation and CHP, where the mass and energy balances that are required for further economic evaluation were determined rigorously. For economic analysis, the Total Plant Cost (TPC) was calculated by adding the Total Plant Direct Cost (TPDC) to the Total Plant Indirect Cost (TPIC). The TPDC included the equipment purchase cost, installation, process piping, instrumentation, insulation, electrical, buildings, yard improvement and auxiliary facilities. The TPIC included the engineering and construction costs, which were calculated as 25% and 35% of the TPDC, respectively. The Direct Fixed Capital Cost (DFCC) was calculated as the sum of TPC, Contractor's Fee and Contingency (CFC). The CFC were calculated as 5% and 10% of the TPC, respectively.

In addition, it was assumed that the plant had a lifetime of 20 years and was operating 330 days a year. The construction time was assumed as 12 months and the startup period as 4 months. The Annual Operating Cost (AOC) was calculated as the sum of the utilities, transportation and labor. The year of analysis was 2015.

The Certified Emission Reductions (CERs) were considered equal to 8.5 US$/ton for 2015 (average price from January 1 to December 31, 2015, EEX, 2015). For the calculation associated with the CERs, the CO_2 was considered to be mitigated by the implementation of both technologies (biomethane and CHP), and the amount of CO_2 was considered to be captured by the upgrading of biogas. A conversion factor was used equal to 22.3 kg CO_2 eq/kg of biogenic CH_4 (Goedkoop et al., 2009).

The sales price of biomethane was considered to be equal to the price of natural gas injected into the domestic market, 7.5 US$/MMBtu (Resolution 1/2013). The sales price of the electricity was considered to be equal to 26.3 US$/MWh in April 2016 (EPE SF, 2016). This price was assumed due to the increase that electricity suffered in Argentina in the early months of 2016 as a result of the elimination of subsidies. We believe that this price provides a better point for analysis than does the subsidized price of 2015. The sales price of the heat produced in the cogeneration process was estimated to be equal to the price of hot water, which was set by default in the SuperPro software. Finally, the sales price of digestate as fertilizer was obtained from information available in the literature (McFarland, 2001) and was updated to 2015 from the United States Consumer Price Index (CPI, 2015), resulting in 79 US$/t.

Moreover, the costs associated with the disposal of all organic waste (OFMSW, SS, GLY) in a landfill were estimated for comparison with the above processes. The costs were obtained from theoretical calculations of integrated solid waste management in Argentina (SAyDS, 2005). These costs from February 2005 were actualized to 2015 using the building cost index published by the Argentina Government (INDEC, 2015).

The landfill costs include the direct costs for the construction of each cell, the construction of leachate treatment facilities, and the direct costs of constructing the common infrastructure (e.g., land acquisition, offices, services, roads). The AOC was calculated as the sum of the utilities, transportation and labor.

Table 4
Annual inventory for Stage 4 (*Biogas Treatment Plant*).

Input		Output	
Option 1: Biomethane			
Raw biogas	1015 t	Biomethane	397.5 t
Electricity	616,002 kWh	CO_2 captured	592.7 t
Natural Gas	21,892 m^3	***Water emissions***	
Make up wastewater	790 t	Effluent 2–3 (Fig. 2a)	1847 t
Make up process water	1057 t	NO_2	7.39 t
		N_2	4.5 t
		Sulfate	18.5 t
		Sulfur	6.2 t
		Air emissions	
		CH_4	11.7 t
		H_2S	0.003 t
Option 2: CHP			
Raw biogas	1015 t	Electricity	2,404,718 kWh
Electricity	351 kWh		
		Water emissions	
Make up wastewater	790 t	Effluent 2 (Fig. 2b)	790 t
		NO_2	7.39 t
		N_2	4.5 t
		Sulfate	18.5 t
		Sulfur	6.2 t
		Air emissions	
		CO_2	1684 t
		SO_2	0.01 t
		CO	2.92 t
		NO_x	1.46 t

3. Results

3.1. LCA results

Table 5 shows the results of the LCA for the four Stages studied, considering the annual feedstock generation (21,072 tons) of organic waste (MSW, SS and glycerol) as a functional unit. Table 5 summarizes the environmental impacts associated with the end use of biogas as biomethane (Stage 4, Option 1) or as CHP (Stage 4, Option 2) and the use of digestate as fertilizer (Stage 3, Option 1) or in incineration (Stage 3, Option 2). Table 5 also shows the impact related with anaerobic digestion (Stage 2) considering that the digester is heated with natural gas (Option 1) and with the CHP heat (Option 2). The impacts associated with incineration are higher than the impacts associated with the use of the digestate as organic fertilizer. The generation of CHP results in a lower impacts than does the generation of biomethane. However, the impact on the category GWP is negative in the generation of biomethane due to the capture of CO_2 in the biogas upgrading process.

Table 5
Environmental impacts for each Stages of the process.

Category	Unit	Stage 1	Stage 2		Stage 3		Stage 4	
			Option 1	Option 2	Option 1	Option 2	Option 1	Option 2
GWP	kg CO_2 eq	55,794	74,502	47,239	729,363	3,861,990	−21,385	6.8
TA	kg SO_2 eq	277.3	644.2	228.2	−582.4	2573	1614	830.0
FE	kg P eq	2.3	3.1	1.5	176.6	241.0	10.0	0.0
HT	kg 1,4-DB eq	9588	12,481	3668	−7082	20,445	26,845	0.5
POF	kg NMVOC	214.5	246.8	106.4	1001	4513	849.5	1594
PMF	kg PM10 eq	94.0	151.9	58.8	−43.0	1.046	410.1	323.7
ULO	m2a	870,1	123.1	50.9	2983	2607	352.7	0.007
FD	kg oil eq	17,923	65,409	14,449	−21,503	17,993	112,121	2.1

3.1.1. Stage 1: feedstock transport

The impacts associated with transport of feedstock are shown in Fig. 3. As shown in Fig. 3, the impacts associated with the transport of feedstock to the biogas production plant primarily influence the HT, POF and ULO categories. Due to the high dependence on fossil fuels for electricity generation in Argentina, the electricity consumption of municipal solid waste influenced the GWP, TA, FE, PMF and FD categories. Fig. 3 shows that source separation allows the better use of the organic fraction and reduces the impacts associated with the use of conventional energy in their mechanical separation.

3.1.2. Stage 2: biogas production plant

Fig. 4 compares the impact related with anaerobic digestion considering that the digester was heated with natural gas (Option 1, biomethane) and with the heat produced by the CHP (Option 2, CHP). Results show a greater environmental impact when natural gas is used to heat the digester during winter. These impacts were reduced when the energy generated in the CHP process was used.

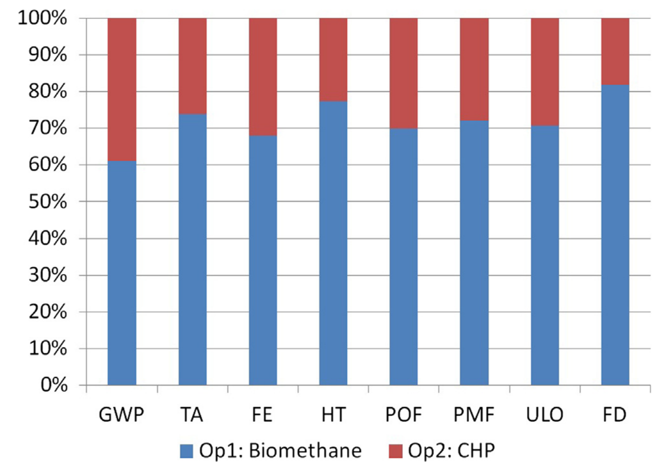

Fig. 4. Comparison of the impacts associated with anaerobic digestion of organic waste (for nomenclature, see Fig. 3).

3.1.3. Stage 3: digestate management

The two options of digestate management are shown in Fig. 5. As shown in Table 5, both options have significant impacts on the overall process. When analyzing the impacts associated with the use of digestate as fertilizer (Fig. 5a), it is observed that the greatest contribution is related to the transport of digestate. The emissions generated during digestate storage and spreading contribute mainly to the GWP and FE categories. The emissions associated with open digestate storage (primarily CH_4 and CO_2) could be nearly 10% of the biogas yield (Poeschl et al., 2012b). Those emissions could be reduced if CH_4 were more effectively captured during the production of biogas. Due to the replacement of mineral fertilizers by the digestate, a significant reduction in impacts in all categories was observed. Those reductions were mainly associated with the production of urea and, to a lesser extent, with the production of triple superphosphate. The amount of nitrogen present in the digestate was considerably greater than the amount of phosphorus.

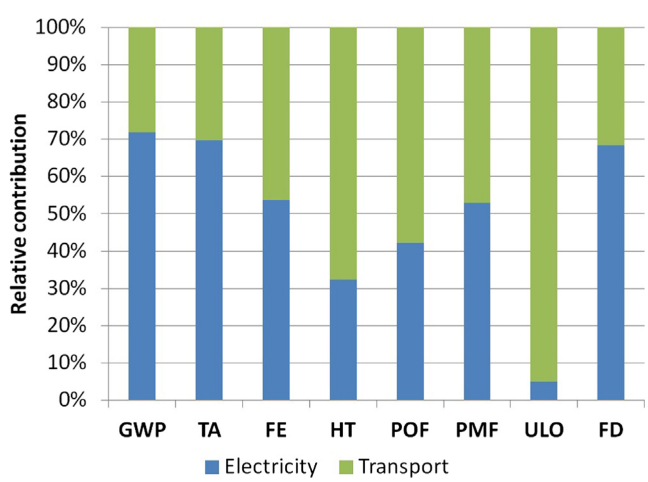

Fig. 3. Comparison of the impacts associated with the sorting and transport of feedstock. *Abbreviations*: GWP, climate change; TA, Terrestrial acidification; FE, Freshwater eutrophication; HT, Human toxicity; POF, Photochemical oxidant formation; PMF, Particulate matter formation; ULO, Urban land occupation; FD, Fossil depletion.

The impacts associated with incineration of digestate (Fig. 5b) were related to the emissions during incineration (GWP, TA, FE, POF and PMF) and to the transport of digestate to the incineration plant (HT, ULO and FD). The contributions associated with the use of electricity for the dehydration of digestate had very little impact on both options (organic fertilizer and incineration), so increasing the dehydration would reduce the impact associated with transportation.

Fig. 5c shows a comparison between the options (organic fertilizer and incineration). It can be observed that the impacts of incineration were considerably higher in all categories evaluated, except in the ULO category, because the fertilizer is less dehydrated (and thus has a greater volume) than is the digestate transferred to the incineration plant.

3.1.4. Stage 4: biogas treatment plant

Fig. 6 shows the impacts associated with the treatment of biogas considering two end uses: biomethane (Fig. 6-a) and CHP (Fig. 6b). With respect to the biomethane generation process, Fig. 6a shows that electricity related to the high demand of energy in the biogas upgrading process was responsible for the greatest impacts. The use of natural gas to heat the biofilter and the anaerobic digester had a low influence in all categories. Additionally, a significant reduction was observed in the category GWP due to the CO_2 capture in the process.

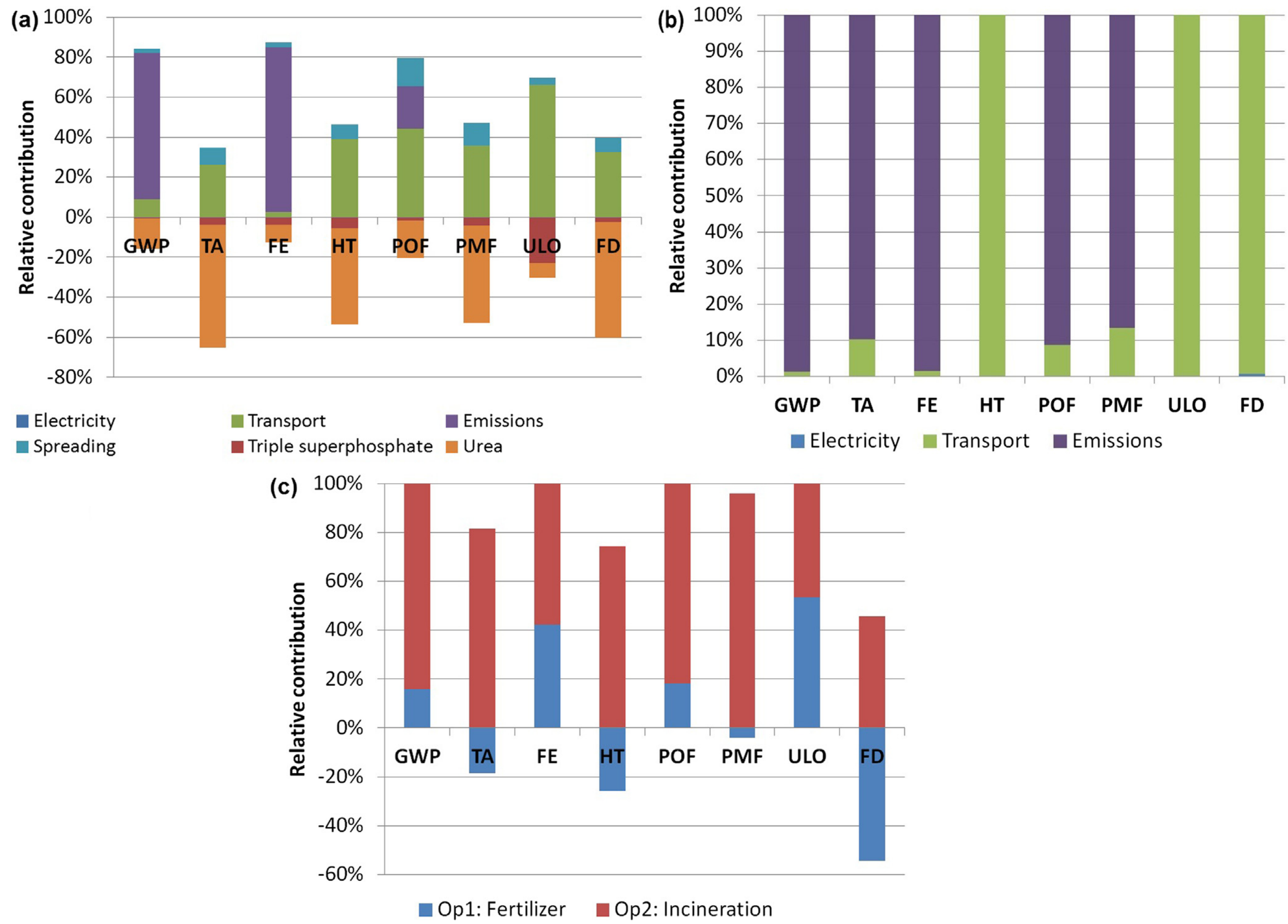

Fig. 5. Comparison of the impacts associated with digestate management: (a) organic fertilizer, (b) incineration and (c) comparison between both processes (for nomenclature, see Fig. 3).

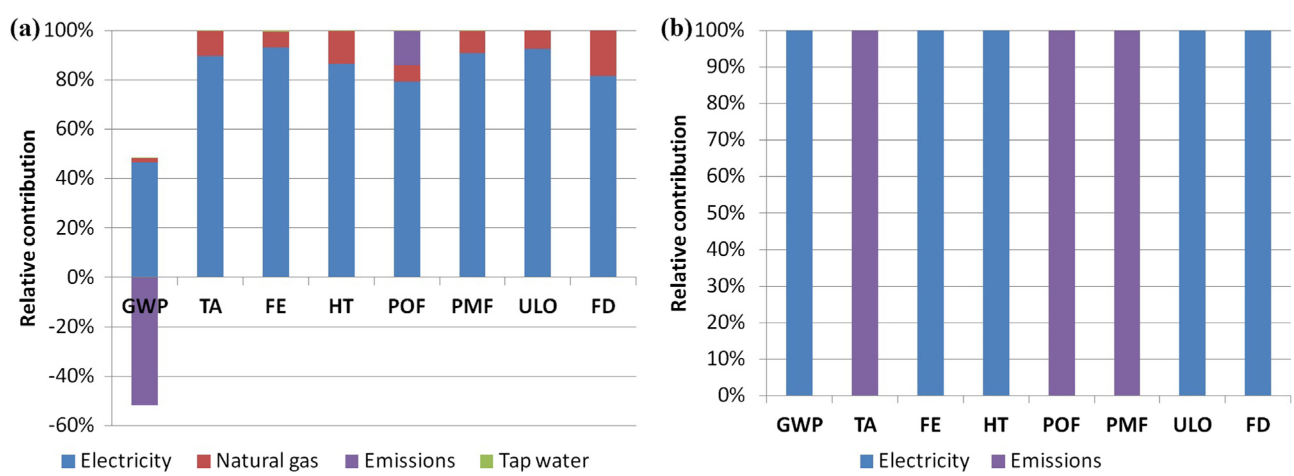

Fig. 6. Comparison of the impacts associated with the biogas treatment: (a) biomethane and (b) CHP (for nomenclature, see Fig. 3).

Fig. 6b shows that the greatest impacts of CHP are related to the use of electricity in the categories GWP, FE, HT ULO and FD. Additionally, an important contribution was observed in the TA, POF and PMF categories due to the emissions of the engine during the transformation of biogas into electricity. Those contributions are also displayed in Table 5.

3.1.5. Comparison of waste treatment alternatives

Fig. 7 shows the comparison between current practice of waste disposal (landfill) and the generation of bioenergy: biomethane and CHP. Both processes included the four Stages (Stage 1: Feedstock transport, Stage 2: Biogas production plant, Stage 3: Digestate Management and Stage 4: Biogas Treatment Plant). Fig. 7

shows that CHP's end use was the most environmentally friendly option. Additionally, the impacts of the landfill were considerably higher in 5 of the 8 categories analyzed (GWP, TA, POF, PMF, ULO). The high impacts of the first four categories were related to the emissions generated in a landfill, a situation that is common in most landfills currently operating in Argentina. Conversely, the high demand of land needed to apply landfill contributed to the ULO category.

Fig. 7 also shows that both processes (biomethane and CHP) have the same impact in the category FE because the emissions generated during the storage and spreading of digestate (such as shown in Fig. 5) are the same. These types of emissions do not occur if the waste is disposed of in a landfill. The high impacts observed in the biomethane process in the HT and FD categories are related to the high amount of electricity that is consumed in biogas upgrading.

3.2. Energy analysis

The results obtained of CED analysis are shown in Fig. 8. Fig. 8a compares the energy demanded of the waste treatment alternatives (biomethane, CHP and landfill). Fig. 8b includes the avoided energy generated by the biomethane and CHP processes (natural gas and electricity, respectively).

Fig. 8a shows that the highest energy demand occurred in the biomethane process because the biogas treatment plant consumed a great deal of electricity (Argentina electricity mix is highly composed of non-renewable fuels), and the anaerobic digester was heated with natural gas. In the CHP process the CED was not so higher because the upgrading of biogas was not necessary, only the H_2S removal was required, and the heat produced by the CHP was used to heat the digester. The use of digestate as fertilizer has a significant saving in the CED (112 MJ eq per ton of feedstock digested) associated with the mineral fertilizer avoided. In the CHP process the CED save in non-renewable sources was 36 MJ eq per ton of feedstock digested.

Fig. 8b shows a highly save of CED when energy avoided (natural gas and electricity) was included in the analysis. It is observed that CED save in non-renewable fossil for biomethane process was 959 MJ eq per ton of feedstock digested and 774 MJ eq per ton of feedstock digested in CHP process. Fig. 8b also shows a save in non-renewable nuclear and renewable water for CHP process, related with the Argentina electricity mix composition (fossil, nuclear and hydro).

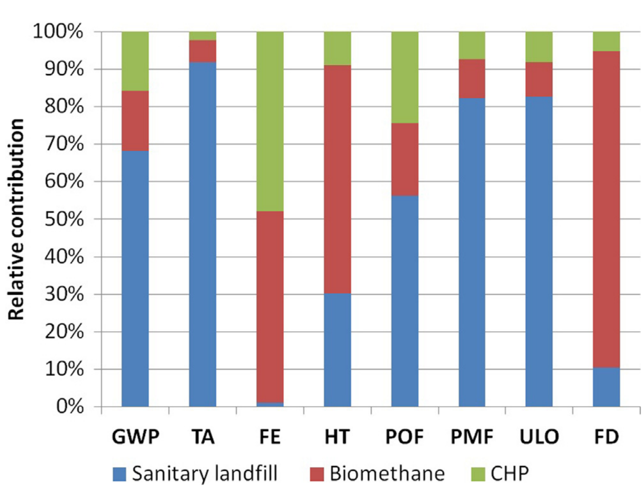

Fig. 7. Comparison of the impacts associated with the overall process of biomethane, CHP and landfill (for nomenclature, see Fig. 3).

3.3. Economic analysis

Table 6 summarizes the costs of the biomethane and the CHP processes showing the DFCC (TPDC, TPIC and CFC). Also included in the table are the AOC and the total revenue for each process.

Table 7 compares the two options for bioenergy generation (biomethane and CHP) with current practice of waste disposal in a landfill and summarizes the annual investment costs, annual operating costs and annual revenues. The annual investment cost was calculated by dividing the DFCC by the useful life of the plant, which was estimated at 20 years for the three alternatives (biomethane, CHP and landfill; in the last, each cell had a useful life of 4 years). Table 7 also includes the difference between the annual costs and revenues to estimate the annual cost associated with the implementation of each technology.

4. Discussion

4.1. Environmental analysis

The impacts associated with the production of biomethane and CHP depend on both the process and the feedstock used in the digestion. A paper published by Lijó et al. (2014b) summarized the differences in the environmental impacts of biogas processes, and the impacts of the different functional units selected for analysis have been analyzed in several publications (De Vries et al., 2012; Borjesson and Berglund, 2006; Poeschl et al., 2012b; Dressler et al., 2012). Borjesson and Berglund (2006) reported the environmental impacts by MJ of bioenergy produced (electricity and heat), and Dressler et al. (2012) reported the results by kW of electricity generated. Moreover, Poeschl et al. (2012b) and De Vries et al. (2012) expressed the results by tons of feedstock digested. These authors showed that environmental impacts associated with bioenergy production from biogas depend on the feedstock, the limits of the system selected, the avoided products and the end use of biogas. However, an important difference between the authors is the assessment methodologies selected to characterize the impacts. Whereas Lijó et al. (2014b) and Dressler et al. (2012) used the CML methodology (2000), Poeschl et al. (2012b) and De Vries et al. (2012) used the ReCiPe Midpoint. Moreover, the environmental impacts reported by Borjesson and Berglund (2006) are based on the methodology developed by Baumann and Tillman (2004). Table 8 shows the environmental impacts' results considering the different assessment methodologies in the different functional units (MJ of bioenergy, $kWh_{(el)}$ and t of feedstock). Only the impacts that can be compared using the same units in both methodologies are shown in Table 8.

As shown in Table 8, the values obtained in the TA and FE categories differ considerably according to the methodology used, with the CML values being larger than ReCiPe values. In the case of the FE category, the main difference between the methodologies is that CML uses a characterization factor for aquatic eutrophication that was first proposed by Heijungs et al. (1992) without further differentiation between the initial emission compartments and regions involved. Therefore, this approach can be considered a worst case. On the FE ReCiPe, Midpoint (Goedkoop et al., 2009) divides the eutrophication in freshwaters and marine water and assumes that the limiting nutrient is N in all coastal waters and P in all freshwaters (Goedkoop et al., 2009).

Table 8 shows that when energy crops are used in the process, the environmental impacts are higher. De Vries et al. (2012) obtained values ranging from −16 kg CO_2 eq per ton of feedstock digested (when only the mono-digestion of organic waste was evaluated) to 105 CO_2 eq per ton of feedstock digested (when the co-digestion of energy crops was included). Similarly, in the TA

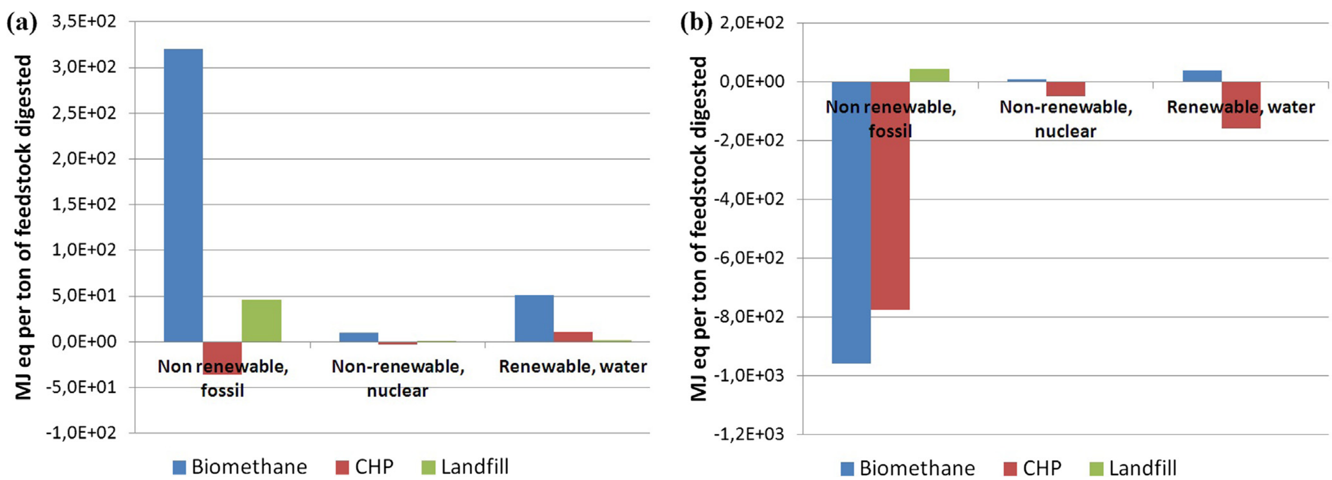

Fig. 8. CED results: (a) comparison of alternatives and (b) included avoided energy from the biomethane and CHP processes.

Table 6
Annual costs and revenues of the biomethane and CHP processes, in US dollars.

	Biomethane	CHP
Direct Fixed Capital Cost (DFCC)		
Total Plant Direct Cost (TPDC)	299,550	284,100
Total Plant Indirect Cost (TPIC)	179,700	170,450
Contractor's Fee & Contingency (CFC)	71,900	68,200
Total DFCC	551,150	522,750
Annual Operating Cost (AOC)		
Labor-dependent	347,000	459,000
Utilities	94,000	2562
Transportation	274,000	274,000
Total AOC	715,000	749,000
Revenues		
CERs	76,281	66,054
Biomethane	137,130	0
Organic fertilizer	497,436	497,436
Electric Energy	0	57,098
Heat	0	13,697
Total Revenues	710,847	634,285
Difference	555,303	624,027

Table 7
Comparison of annual costs associated with the implementation of the different technologies for waste treatment, in US dollars.

	Landfill	Biomethane	CHP
Investment Cost	494,864	551,150	522,750
Operating Cost	402,046	715,000	749,000
Revenue	$ 0	710,847	634,285
Annual Cost	896,909	555,303	624,027

and FE categories, the authors obtained values ranging from -0.33 kg SO_2 eq and -0.012 kg P-eq per ton of feedstock digested (when using organic waste for co-digestion) to 1.61 kg SO_2 eq and 0.252 kg P-eq per ton of feedstock digested (when using energy crops for co-digestion).

Table 8 shows that the impacts of the processes under study were on the order of similar processes available in the literature. Therefore, the co-digestion of SS and OFMSW solved the problem of the disposal of organic waste and produce energy with lower environmental impacts.

4.2. Energy analysis

The use of organic waste to produce energy was positive from an energetic point of view. An energetic balance demonstrate that the energy demands of the overall process was approximately 39% of the energy produced when the final use was biomethane and 30% when the final use was CHP. This difference was associated with the power generated by the motor generator, which has an electrical conversion efficiency of 39% and a thermal conversion of 45%, resulting in a total process efficiency of 84%, whereas the energy available in the biomethane process was 100%. The CED analysis also shows energy saving when bioenergy processes were compared with current practice of waste disposal (landfill). In addition, the use of waste contributed to diversifying the energy matrix and decentralizing the energy production.

4.3. Sensitivity analysis

4.3.1. Biogas production efficiency

The design parameters used to characterize the co-digestion substrates were obtained from the literature (Forster-Carneiro et al., 2008; Alvarez et al., 2010). As results are very dependent of these parameters, we assess the influence of biogas production efficiency on the impact assessment. The efficiency was varied plus and minus 20%. Results show that the impacts decrease with an increase of biogas production efficiency and increase with a decrease of efficiency. The changes affect manly the impact categories GWP, FE and ULO in the biomethane process. In the base scenario, around 1.44 kg CO_2 eq per m^3 of biomethane were emitted. When efficiency is 20% higher, the emissions were reduced to 1.22 kg CO_2 eq per m^3 of biomethane (a reduction of 16%). When the efficiency was 20% lower, the emissions increase 29%. Results also showed that FE and ULO impacts decreased 12% with a 20% increase of the biogas production efficiency and increased 23% with a decreased of efficiency.

For the CHP process, the impact categorized GWP, TA and PMF presented important effects. The emissions in the base scenario were 0.34 kg CO_2 eq per kWe and the variation was between a decrease of 83% and a increase of 125% when biogas production efficiency increase and decrease 20%, respectively. An important reduction was observed in TA (108%) because the emissions avoided using the organic fertilizer were higher than the emitted in the CHP process. In the same way, PMF shows a reduction of 85%. Both impact categories (TA and PMF) increase less than 5% when biogas production efficiency decreases 20%.

Table 8
Comparison of environmental impacts with different assessment methodologies and functional units.

	Units	This work Recipe	Lijó et al. (2014b) CML	Borjesson and Berglund (2006)	Dressler et al. (2012) CML	De Vries et al. (2012) Recipe	Poeschl et al. (2012b) Recipe
GWP	g CO_2 eq/$MJ_{bioenergy}$	39–45	30	15.5–35			(−119) to 163
	kg CO_2 eq/$kWh_{(el)}$	0.34	0.24		0.058–0.18		
	kg CO_2 eq/ton feedstock	39–40	14			(−16) to 105	
TA	g SO_2 eq/$MJ_{bioenergy}$	0.04–0.09	0.80	0.065–0.175			
	kg SO_2 eq/$kWh_{(el)}$	0.31	6.60		1.62–1.94		
	kg SO_2 eq/ton feedstock	0.04–0.09	0.37			(−0.33) to 1.61	(−0.05) to 2.90
FE	g P eq/$MJ_{bioenergy}$	0.009–0.01	0.15	0.009–0.65			
	kg P eq/$kWh_{(el)}$	0.07	1.24		0.33–0.40		
	kg P eq/ton feedstock	0.008–0.009				(−0.01) to 0.252	(−0.01) to 0.09

4.3.2. NFRV of mineral fertilizer

The NFRV of mineral fertilizer vary considerably depending on some factors: soil type, method of application, weather conditions. To assess the influence of a change in the NFRV on the impact categories, the NFRV was modified plus and minus 20% (De Vries et al., 2012). Results showed that the impacts decreased with an increase of the NFRV and vice versa. These changes were mainly due to the amount of urea that was substituted. The main impact categories modified were TA (76–123%), HT (34–165%) and PMF (81–118%).

Results of environmental impacts' (taking as functional unit t of feedstock) considering in Table 8, GWP, TA and FE were: 43.17–44.23 kg CO_2 eq/t; 0.089–0.126 kg SO_2 eq/t and 0.0090–0.0095 kg P eq/t, respectively, when NFRV was 45%. For NFRV of 85% the results for GWP, TA and FE were: 38.46–39.53 kg CO_2 eq/t, 0.055–0.091 kg SO_2 eq/t and 0.008–0.009 kg P eq/t, respectively.

4.3.3. Energy cost

The analysis of the profitability of using organic waste for bioenergy production was greatly affected by the cost of energy (very low in December 2015). A thorough economic analysis must account for several factors that increase the profitability of the project: (a) integration of wastewater treatment with municipal solid waste projects; (b) scale technology of the biogas plant; and (c) regulations and laws concerning tariffs, the disposal of solid, liquid and gaseous effluents, and the addition of gas and electricity to distribution grids.

The Argentine Government promulgated the last year the Law 27,191 which encourages the use of renewable energy sources for the production of electricity, to make a 20% transition by 2025. The price of that energy is US$ 113/MWh, through public bidding process. Recently in the country were tendered 1000 MW (RenovAr, 2016). Therefore, in the sensitivity analysis the price of energy was evaluated. The subsidy electricity price, USD 113/MWh representing an increase of 400% over the current price (26.3 US$/MWh), was evaluated. In the same way, an increase of 400% of natural gas price was considered (30 US$/MMBtu instead of 7.5 US$/MMBtu). We presume that future policies will also encourage the generation of other renewable energy sources such as biomethane.

Results showed a reduction of 34% annual in the CHP process (from US$ 624,027 to US$ 422,834). In the biomethane plant, annual costs were reduced from US$ 555,303 to US$ 143,913, equivalent to an annual saving of 74%. The biggest saving in the biomethane process was related to the efficiency of both processes, as mentioned in Section 4.2.

The economic sensitivity analysis shows that in order to encourage alternative waste treatment plants and avoided the landfill, a serious and long-term subsidy policy is needed. The subsidies are necessary to reduce costs in waste treatment, but also to promote the diversification of the energy matrix.

5. Conclusions

Argentina has a great potential to exploit organic waste as an energy source. This work shows that the generation of bioenergy reduces the environmental impacts, improves energy efficiency and is economically attractive. The environmental analysis showed that the use of organic waste as an energy source solved problems related to waste disposal and generated fewer environmental impacts and lower costs than does current practice of waste disposal in a landfill. The feed to the digester needed to be sufficiently large for an economically attractive biogas production, and disposal of organic waste (OFMSW + SS) in a medium-size city should be complemented by waste from nearby towns or agricultural/livestock waste. The energy balance was highly positive, with the use of energy for the overall process (Feedstock collection and transport, Biogas Production Plant, Digestate Management and Biogas Treatment Plant) between 30 and 39% of the energy generated.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2016.11.033.

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