Energy 137 (2017) 1130–1143

Contents lists available at ScienceDirect

Energy

journal homepage: www.elsevier.com/locate/energy

The environmental effect of substituting energy crops for food waste as feedstock for biogas production

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ARTICLE INFO

Article history:
Received 17 November 2016
Received in revised form
27 March 2017
Accepted 24 April 2017
Available online 28 April 2017

Keywords:
Anaerobic biogas potential
Biogas
Digestate composition
Life cycle assessment
Organic waste
Renewable energy

ABSTRACT

Assuring environmental sustainable bioenergy production is an international priority nowadays. The objective of this study was to identify the environmental consequences of feedstock selection in biogas production. Two real biogas plants were assessed and compared from a life cycle perspective. Plant A performs the co-digestion of energy crops (78%) and animal waste (22%) while Plant B consumes energy crops (4%), food waste (29%) and animal manure (67%). According to the results, electricity production from biogas implied lower impacts in climate change compared to the existing electric mix. Maize silage (650 Nm^3/TVS_{fed}) and food waste (660 Nm^3/TVS_{fed}) appeared as an interesting source of bioenergy. However, the cultivation of energy crops was identified as the main *hotspot* in Plant A. Finally, the use of organic substrates with lower energy potential and high nutrients concentration such as animal manure (450 Nm^3/TVS_{fed}) produced higher amounts of digestate, producing impacts in acidification and eutrophication categories. In order to improve the environmental sustainability of bioenergy, specific guideless should be established to achieve harmonised life cycle studies. In addition, environmental policies should promote the use of waste streams and prevent the use of energy crops as well as include goals related with acidification and eutrophication impacts.

1. Introduction

The use of renewable resources for bioenergy production has been supported by the European policy, including Directive 2001/77/EC and Directive 2009/28/EC, where it was established that each Member State should reach a 20% share of energy from renewable sources by 2020 [1,2]. Governments of many European countries (e.g. Germany, Italy, UK) encouraged the spread of biogas production through important economic incentives [3,4]. Germany, which is the largest European biogas producer, generated more than 6215 thousand tonnes of oil equivalent (ktoe) from decentralised agricultural plants, municipal solid waste methanisation plants and centralised co-digestion plants in 2013, accounting for 67% of the biogas produced in Europe [4]. Italy is the second European producer of agricultural biogas, with an extensive promotion of electricity production from biogas through the Ministerial Decree of 24 October 2005 [5]. However, the regulatory scheme changed its incentive system according to the Ministerial Decree of 6 July 2012, which encourages the development of small plants (up to 300 kW), giving preference to the use of farming waste over energy crops.

Despite its consideration as an environmental friendly technology, several studies have reported that the adoption of anaerobic digestion may not necessarily lead to sustainable practices [6]. Thus, the environmental sustainability of biogas systems should be properly evaluated by means of scientific and standardised methods such as Life Cycle Assessment (LCA), which enables assessing complete bioenergy production chains through all the stages of its life cycle. In this sense, Whiting and Azapagic [3] reported that co-generating electricity and heat from the anaerobic digestion of agricultural waste can lead to significant reduction in most impact categories compared to fossil-fuel alternatives. However, these authors found that the acidification and eutrophication potentials were 25 and 12 times higher, respectively. In line with this, Venkatesh and Elmi [7] criticised the focus on climate change and pointed out the importance of avoiding problem shifting; i.e., reducing the environmental impacts produced in climate change by increasing them to other impact categories. Nevertheless, generally

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http://dx.doi.org/10.1016/j.energy.2017.04.137
0360-5442/

Abbreviations

AGP	anaerobic gas potential
CC	climate change
CHP	co-generation heat and power
CSRT	continuous stirred tank reactor
FD	fossil depletion
FE	freshwater eutrophication
FU	functional unit
GHG	greenhouse gas
HRT	hydraulic retention time
iLUC	indirect land use change
ktoe	thousand tonnes of oil equivalent
LCA	life cycle assessment
ME	marine eutrophication
OD	ozone depletion
OFMSW	organic fraction of municipal solid waste
OLR	organic loading rate
POF	photochemical oxidants formation
TA	terrestrial acidification
TAN	total ammonia nitrogen
TN	total nitrogen
TP	total phosphorus
TS	total solids
TVS	total volatile solids

speaking, the outcomes from the available LCA reports and studies are difficult to compare due to the use of different input data, functional units, allocation methods, reference systems, characterisation methods and other assumptions [8]. Bacenetti et al. [9] performed an extensive review of LCA studies focused on agricultural biogas production and emphasised the importance of increasing their level of transparency and harmonisation in order to improve the comparability among results. Regarding the identification of the main contributing processes to the overall environmental impacts, common conclusions are shared among the different studies such as the predominant influence of feedstock composition, energy efficiency and digestate management [10–13].

Energy crops are a common substrate for bioenergy production due to their high biogas potential; however, plantations of short rotation may cause important environmental burdens due to the requirement of intensive agricultural activities and fertilisers, with negative impacts on soils and water [14]. Among them, maize is the most widely used in Europe [14]; though, the rising demand for maize can entail a change in the use of soil, increasing the pressure to convert grass- and peatlands in areas for maize cultivation [15]. Regarding this issue, Jacobs et al. [16] proposed sugar beet as an alternative to maize in order to diversify biomass for bioenergy production. In fact, the results proved that, in terms of climate change, sugar beet cultivated in crop rotation entailed lower environmental impacts compared with the cultivation of maize. In this context, alternative biomass sources are beginning to be considered for energy generation that are both economic and environmental sustainable. Different studies have been published trying to find alternative substrates for biogas production. Pardo et al. [17] informed that the use of maize improved the performance of the anaerobic digestion process, but its cultivation involved important environmental burdens related to energy and fertilisers consumption as well as changes in indirect land use changes (iLUC). The use of agri-food waste as an alternative co-substrate for biogas production was recommended, since no emissions are usually assigned to the production of waste streams, but special attention should be paid on possible environmental impacts linked with the shifting from its current use as animal feed. Additionally, Ertem et al. [18] proposed the use of marine macroalgal feedstock for the substitution of energy crops since it solves the dilemma between bioenergy and food production. The study proved that environmental sustainable energy production is achievable when co-digestion animal waste and macroalgae; however, it highly depends on transport distances from the coastal area to the digester. In this sense, the use of local available organic waste, such as agricultural waste and food waste, to recover bioenergy would not only help to improve energy scarcity and security, but also fulfils other desirable objectives such as sustainable management of waste streams [7]. Moreover, the proper management of organic waste is considered a key factor for moving away from our current linear economy to the circular one [19], since it is aimed to convert waste streams into valuable products that can be introduced again into the economy. In this regard, Evangelisti et al. [13] and Walker et al. [20] reported satisfactory technical and environmental results regarding the anaerobic digestion of food waste from different sources.

The main objective of this study was to identify the link between feedstock composition and environmental sustainability in biogas production, in terms of both biogas production potential and the amount and quality of the produced digestate. With this purpose, this paper analyses and compares, from a life cycle perspective, the environmental performance of two real biogas plants operating in Northern Italy. More deeply, Plant A digests a high ratio of energy crops mixed with animal manure while Plant B uses a high ratio of food waste with animal waste and a minor fraction of energy crops. Therefore, the environmental performance of biogas systems predominantly based on food waste instead of energy crops was also investigated.

2. Goal and scope definition

2.1. Objectives of the study

As aforementioned, the aim of this study was to evaluate the environmental profiles of two different biogas systems that produce electricity from biogas by means of LCA methodology. This approach will also allow the further comparison of the environmental performance of bioenergy production with fossil-based systems [21].

Specific objectives of the study included the identification of the most critical stages, named as environmental *hotspots*, in order to identify opportunities to gain environmental benefits. To do so, two real Italian biogas plants (Plants A and B) located in the Po Valley (Northern Italy) were assessed and compared according to ISO standards [22]. The daily composition of the feedstock depends on its seasonal availability. In this study, average data from the operation of year 2012 was managed. Table 1 depicts data of average composition of the feedstock, transport distances as well as the main outputs.

As shown, Plant A performs the anaerobic co-digestion of maize, triticale, pig slurry and chicken manure at different ratios whereas Plant B co-digests maize, pig slurry, OFMSW and food waste from retailers and supermarkets. In terms of percentages, energy crops account for 78% of the mass fed to the digester in Plant A, which

Table 1
Summary of the average daily inputs and outputs of the anaerobic digestion process for each biogas plant.

Feedstock	Unit	Plant A	Plant B
Maize silage	t/d	43.9	9.86
	km	2	6
Triticale silage	t/d	4.98	
	km	3	
Pig slurry	t/d	12.6	178
	km	1.5	[a]
Chicken manure	t/d	1.35	
	km	3	
OFMSW	t/d		65.7
	km		20
Food waste	t/d		11.0
	km		20–70
Biogas	m^3/d	12,746	17,423
	% CH_4	44.6	60
Raw digestate	t/d	48.05	246.90
Electricity	kWh/d	22,759	39,820
Heat	kWh/d	22,579	42,475

[a] In Plant B, pig slurry is pumped to the plant instead of transported by road.

represents 94% of the TVS; while in Plant B, food waste amounts to 41% of the mass input to the anaerobic digestion process and 75% of the TVS. In both systems, biogas is burnt in a co-generation heat and power (CHP) unit to produce electricity and heat. The electricity produced is injected into the national grid whereas heat is partially recirculated to fulfil the requirements of the digester and its surplus is wasted to the atmosphere. Therefore, it was considered that the main function of the studied system is the supply of electricity to the national grid. Accordingly, the functional unit (FU) selected in this study was 1 MWh of electricity produced.

2.2. *Description of the systems under assessment*

The system boundaries included the production of energy crops, transport, storage and pre-treatment of all input materials, the anaerobic digestion process, the use of biogas in the CHP unit and the management of the digestate. Since electricity is exported to the national grid and surplus heat is not subsequently exploited, all the impacts were allocated to electricity production. In parallel to biogas, digestate is also produced and it can be used as an organic fertiliser in the cultivation of the energy crops used as feedstock. However, Plant B produces more digestate that the required for maize cultivation. In this case, it has been considered that the surplus digestate could be used in other agricultural systems, reducing the use of mineral fertilisers by including the avoided products perspective. In this case, only avoided ammonium nitrate was considered as environmental credits because the agricultural soil in the Po region presents high contents of phosphorus and potassium, which makes their addition as mineral fertilisers unnecessary.

Regarding the quantification of emissions in the LCA study, carbon dioxide from organic sources was excluded from the system boundaries because biogenic carbon is usually considered as climate-neutral. Within the system boundaries, the transport and spread of digestate, the production of the machinery required as well as diesel combustion emissions and field emissions from the application of digestate were also considered.

Fig. 1 outlines the main processes considered within each biogas system. All the processes involved in both systems were aggregated into different main subsystems depending on the biogas plant: maize and triticale cultivation, waste production (outside the system boundaries), feedstock transport, bioenergy production, digestate management and use of surplus digestate.

2.3. *Waste production*

Pig slurry and chicken manure are the main waste streams of pig and chicken farms and their production is unaffected by its further valorisation by anaerobic digestion. In the same way, OFMSW and food waste from supermarkets are generated in other product systems (i.e. food supply chain). Therefore, their production was excluded from the system boundaries of Plants A and B.

2.4. *Maize and triticale cultivation*

This subsystem includes all the agricultural operations involved in the maize and triticale production, considering that these crops are exclusively dedicated to the production of biomass for energy purposes. The agricultural schemes can be based either on a double crop system, where the winter crop (triticale) is cultivated from September to May and the summer crop (maize, class 500) is cultivated in the same agricultural land from May to September; or on a single crop system, where only the summer crop (maize, class 700) is cultivated during summer. Tables 2 and 3 display the main agricultural operations in triticale and maize cultivation, respectively [23,24].

The subsystem boundaries include all the inputs such as the production and use of agricultural machinery (tractors and implements), diesel, fertilisers (urea and ammonium nitrate), herbicides (Terbutilazine, Alachlor and Lumax) and seeds as well as the emissions derived.

2.5. *Feedstock supply*

The supply of all waste streams comprises their transport to the biogas plant from farms, households and supermarkets. Transport distances for each feedstock can be found in Table 1. In addition, it has been considered that animal waste is stored for two days in the biogas plant facilities; therefore, associated emissions were computed. Moreover, OFMSW and food waste are firstly screened and shredded in order to reduce particle size to 12 mm. Concerning maize and triticale, chopped straw is transported to the biogas plant where it is ensiled and stored, considering 10% of losses. The production of the machinery and diesel required for the aforementioned operations as well as the combustion emissions derived from diesel consumption were taken into account.

2.6. *Bioenergy production*

This subsystem includes all the inputs and outputs required for the production of biogas by anaerobic digestion and its conversion in the CHP unit. In both plants, anaerobic digestion takes place in a continuous stirred tank reactor (CSTR), operated at mesophilic temperature by means of the circulation of hot water. Biogas is stored in a gasholder dome placed at the top of the digesters. In both plants, the biogas produced is filtered and dehumidified before being burned. Biogas from Plant A is desulphurised by adding a solution of ferric chloride, whereas biological desulphurisation is performed in Plant B.

The production of infrastructure, diesel, chemicals, lubricant oil and electricity from the grid as well as emissions from the CHP and biogas losses were also included within the system boundaries. A summary of the main parameters of the anaerobic digestion and CHP units can be found in Table 4, including the hydraulic retention time (HRT) and the organic loading rate (OLR), expressed in terms of total volatile solids (TVS).

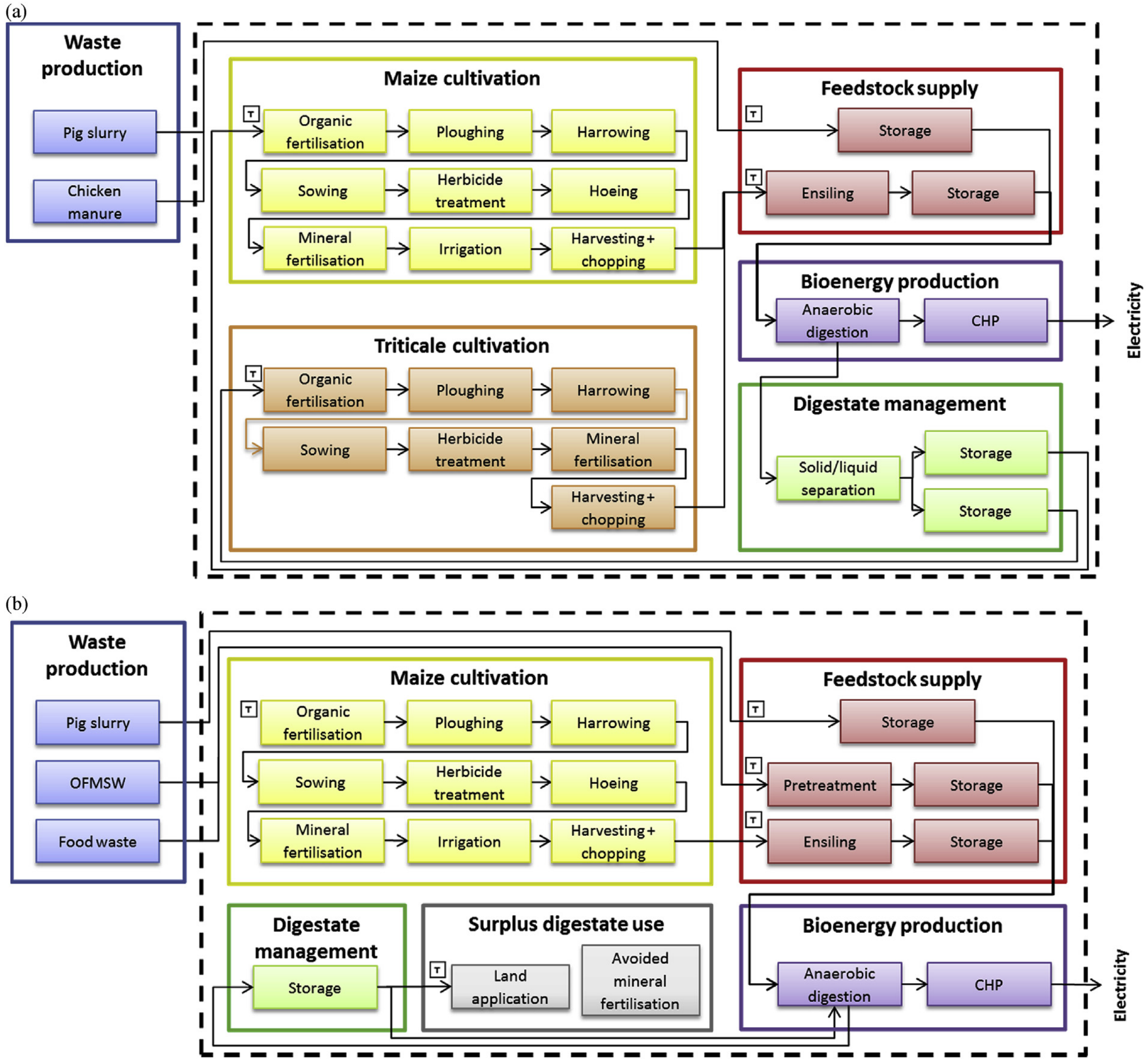

Fig. 1. Flowchart and system boundaries of (a) Plant A and (b) Plant B.

Table 2
Field operations per hectare of triticale cultivated.

Operation	Time	Tractor and implement		Inputs/outputs
		Effective work capacity (ha/h)	Diesel (kg/ha)	
Organic fertilisation	September	2.5	4.71	Digestate 160 kg N/ha
Ploughing	September	0.8	22.6	–
Harrowing	September	0.5	24.2	–
Sowing	October	0.7	8.53	200 kg/ha
Herbicide treatment	October	3.0	3.32	Terbutilazine + Alachlor 5 kg/ha
Mineral fertilisation	November	3.0	3.17	Ammonium nitrate 60 kg/ha
Mineral fertilisation	February	3.0	3.17	Urea 60 kg/ha
Harvesting	May	2.5	27.2	37 t fresh matter/ha

2.7. *Digestate management*

In Plant A, digestate is firstly separated into its solid and liquid fractions by means of a screw press. Then, both fractions are stored before being applied as organic fertilisers in the cultivation of maize and triticale. The solid fraction is stored in piles while the liquid fraction is stored in a closed tank. However, the produced digestate in the plant is lower than the required for both crops. Therefore, it was considered that digestate from a nearby biogas plant is also transported to the field.

Conversely, digestate is not separated in Plant B. Instead, raw digestate is stored in a closed tank. As mentioned, it is partially used in the cultivation of maize; however, more digestate than required is produced. Therefore, it can be used in another agricultural land,

Table 3
Field operations per hectare of maize cultivated.

Operation	Time	Tractor and implement		Inputs/outputs
		Effective work capacity (ha/h)	Diesel (kg/ha)	
Organic fertilisation	May	0.3	27.2	Digestate 340 kg N/ha[a] 180 kg N/ha[b]
Ploughing	May	0.9	22.6	–
Harrowing	May	0.8	23.7	–
Sowing	May	1.0	13.4	19 kg/ha
Herbicide treatment	June	3.0	3.5	Lumax 5 kg/ha
Mineral fertilisation	June	8.0	6.1	Urea 60 kg/ha
Irrigation (5 times)	July–August	0.83	22.4	3600 m^3/ha
Harvesting	September	1.0	36.0	65.1 t fresh matter/ha[a] 48.8 t fresh matter/ha[b]

[a] Maize FAO class 700.
[b] Maize FAO class 500.

replacing the production and use of nitrogen-based mineral fertilisers (environmental credits).

3. Life cycle inventory

Primary data related to the cultivation of energy crops regarding tractors and implements, labour hours and input rates were obtained from interviews and surveys with growers (Tables 2 and 3). Other inputs such as feedstocks, electricity, chemicals and lubricant oil as well as outputs such as biogas, digestate, heat and electricity were obtained from record data of the biogas plants.

In order to validate the primary data collected as well as to calculate other inventory data required in the study, mass balances were developed for each biogas plant regarding total solids (TS), total volatile solids (TVS), total nitrogen (TN), total ammonia nitrogen (TAN) and total phosphorus (TP). To do so, the quantity of water and ammonia contained in the produced biogas were considered negligible [6]. In addition, the solid/liquid separation was modelled according to the separation efficiencies presented in Bauer et al. [25]. The mass balances calculated per FU (1 MWh electricity produced) regarding the main inputs and outputs of both biogas plants are presented in Fig. 2.

Fig. 2 depicts detailed data of the different streams for both plants. More deeply, for the production of 1 MWh of electricity, Plant B requires the digestion of nearly 2.5-times more feedstock (6.65 t/FU) than Plant A (2.76 t/FU) for similar values of TVS. Moreover, the potential biogas production of Plant A (352 m^3 biogas/t TVS_{fed}) is also higher than the one of Plant B (322 m^3 biogas/TVS_{fed}). These differences can be explained taking into account the variability of the feedstock digested. Plant A digests a high ratio of energy crops with high content of TVS while Plant B processes more pig slurry, which provides low amount of TVS. In addition, this substrate also provides nutrients: 2.43 kg TN/t pig slurry (1.83 kg/t as TAN) and 2.1 kg TP/t pig slurry. As a result of this, Plant B ends up with the production of a higher amount of digestate per FU (6.20 t digestate/FU) than Plant A (2.11 t digestate/FU).

Table 4
Main data of the anaerobic digesters and co-generation engines of Plant A and B.

Parameter	Unit	Plant A	Plant B
Temperature	°C	44	40
Technology		Two stages	Single stage
OLR	t TVS/$m^3 \cdot$d	2.48	4.50
HRT	d	40–55	20–30
CHP power	kW	999	1664
Electric efficiency	%	41	39
Thermal efficiency	%	44	45

In LCA biogas studies, direct emissions are typically estimated by using available models or provided emission rates [6]. In this study, the emissions derived from digestate storage (raw digestate or liquid and solid fraction) were estimated according to the emission factors of methane, ammonia, nitrous oxide, nitrogen gas and nitric oxide provided by De Vries et al. [26]. In addition, it was considered that the closed storage of digestate emitted 80% lower emissions that the open one [12]. Field emissions derived from the application of organic and mineral fertilisers on land were also taken into account following the methodology described by Brentrup et al. [27] regarding nitrogen-based compounds such as ammonia, nitrous oxide, nitrogen gas and nitrate and Rossier and Charles [28] for phosphate-based compounds. Specifically, these authors proposed a methodology to estimate emissions derived from the volatilisation of ammonia from the application of organic fertilisers depending on many factors such as average air temperature, infiltration rate, elapsed time between application and rainfall or incorporation into the soil. Regarding the specific area under study, the climatic conditions in this area correspond to a transition between Mediterranean and Central European climate, with rainfall mainly concentrated in fall and spring. Average values of temperature and rainfall are 12 °C and 745 mm, respectively. Regarding the soil characteristics, it is mainly 52% sand, 30% silt and 17% clay [23]. On the contrary, ammonia volatilisation due to the application of mineral fertilisers depends on the type of fertiliser. Regarding the remaining emissions for organic and mineral fertilisers, nitrogen lost as nitrous oxide is estimated as 1.25% of the total nitrogen applied on land, while 9% is considered lost as nitrogen gas. Leachates of nitrate were estimated as the balance of nitrogen inputs and outputs to the agricultural system [27]. It is important to consider that the fertilising value of organic fertilisers is lower compared with mineral fertilisers since organic ones contain mineral and organic forms of nitrogen [29]. Accordingly, a nitrogen replacement value of 65% was considered for the digestate in comparison with ammonium nitrate [26]. The summary of the main inventory data of Plants A and B can be found in Tables 5–8.

Finally, background data regarding the production of all the required inputs such as diesel fuel, sodium hydroxide, lubricant oil and electricity, as well as avoided fertilisers, were taken from ecoinvent® database version 3.1 [30–34]. The Italian electricity profile was updated using the data for the average electricity production and import/export data for Italy in 2013 [35].

4. Life cycle impact assessment

Life Cycle Impact Assessment was conducted using updated characterisation factors from ReCiPe Midpoint methodology [36]

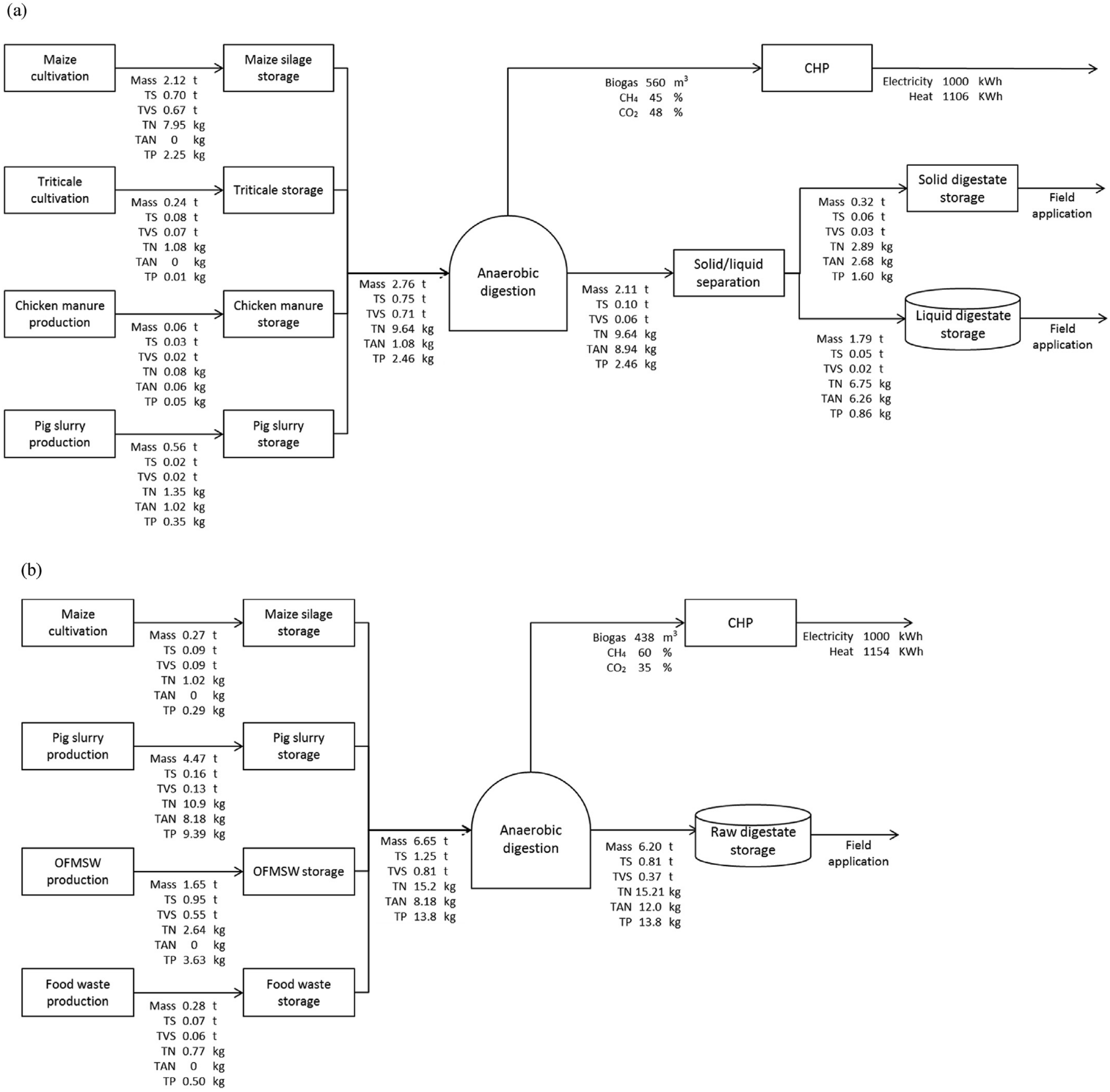

Fig. 2. Mass balances of Plant A (a) and Plant B (b) per FU (i.e. 1 MWh electricity produced).

for the following impact categories: climate change (CC), ozone depletion (OD), terrestrial acidification (TA), freshwater eutrophication (FE), marine eutrophication (ME), photochemical oxidant formation (POF) and fossil depletion (FD). It is important to note that positive values in the figures are indicative of environmental burdens, whereas negative values are indicative of environmental credits/benefits derived from avoided processes.

4.1. *Detailed assessment of Plant A*

The characterisation results per MWh of electricity produced in Plant A can be found in Table 9. In addition, Fig. 3 shows the relative contributions of the different subsystems to the environmental profile of Plant A.

Crop cultivation was identified as the most important *hotspot*, producing between 52% and 98% of the total environmental impacts. Maize production was by far the main contributor due to the highest mass input. In more detail, the environmental impacts within the cultivation step are related with the agricultural activities performed, specially regarding CC, OD, POF and FD, ranging from 44% to 93% in the case of maize and between 29% and 79% for triticale. The reason for these results is related with the consumption of diesel in the agricultural machinery and its derived combustion emissions. Regarding TA, FE and ME, the major

Table 5
Inventory data regarding energy corps production per MWh of electricity produced.

	Plant A		Plant B
	Maize	Triticale	Maize
Materials and fuels			
Digestate	6.04 t	0.28 t	0.59 t
Seeds	0.62 kg	0.01 kg	0.08 kg
Pesticides	0.20 kg	0.03 kg	0.03 kg
Urea	1.96 kg	0.39 kg	0.25 kg
Ammonium nitrate		0.39 kg	
Diesel	46.1 kg	1.01 kg	0.8 kg
Tractor	5.54 kg	0.09 kg	0.1 kg
Agricultural tillage	12.4 kg	0.09 kg	0.1 kg
Transport			
Tractor and trailer	343 t km	25.5 t km	3.5 t km
Resources			
Water	117 m^3		15 m^3
Products			
Straw	2.12 t	0.24 t	0.27 t
Emissions to air			
Ammonia	2.41 kg	0.22 kg	0.33 kg
Nitrous oxide	0.18 kg	0.02 kg	0.02 kg
Nitrogen	0.64 kg	0.21 kg	0.11 kg
Emissions to water			
Nitrate	4.36 kg	0.13 kg	0.66 kg
Phosphate	0.08 kg	0.01 kg	0.04 kg

contributors were the emissions associated to ammonia emissions (in TA), phosphate leaching (in FE) and nitrate leaching (in ME) from the application of organic and mineral fertilisers.

Feedstock supply also had a minor contribution, with shares around 5% for OD, POF and FD impact categories, mainly due to machinery and diesel consumption required in ensiling operations and transport. Regarding the bioenergy production subsystem, significant environmental impacts in CC (27%), OD (9%), POF (23%) and FD (10%) were observed, attributed to fugitive emissions of methane, diesel consumption and production and use of machinery and infrastructure.

Regarding the digestate management subsystem, it only comprises the storage of the digestate, since it is totally spreaded out in the cultivation of the required energy crops. However, digestate storage also produced important environmental burdens, including direct methane and nitrous oxide emissions that highly affected CC (19%). Since liquid fraction of the digestate is stored in a covered tank, the storage of the solid fraction of the digestate is the main source of these greenhouse gas (GHG) emissions. Finally, the electricity consumed in the biogas plant, that is taken from the national grid, represented 17% and 18% of the impacts produced in OD and FD, respectively.

Table 6
Inventory data regarding feedstock supply per MWh of electricity produced.

	Plant A	Plant B
Materials and fuels		
Straw	2.35 t	0.27 t
Animal waste	0.61 t	178 t
Food waste		76.7 t
Diesel	1.04 kg	0.12 kg
Tractor	0.06 kg	0.01 kg
Agricultural tillage	0.06 kg	0.01 kg
Transport		
Tractor and trailer	6.27 t km	46.8 t km
Products		
Silage	2.15 t	0.25 t
Animal waste	0.61 t	178 t
Food waste		76.7 t

4.2. Detailed assessment of Plant B

The characterisation results per MWh of electricity produced in Plant B are presented in Table 10 and Fig. 4 displays the relative contributions of the different subsystems.

The environmental profile obtained in Plant B was different compared with the one of Plant A. In this case, the contribution of maize cultivation subsystem was much lower than in Plant A due to the lower input, contributing between 11% and 28% of the total environmental impacts. Feedstock supply had a similar contribution than in Plant A, being responsible of 14% and 16% in OD and FD, mainly due to the transport of OFMSW and food waste to the biogas plant (see Table 1 for transport distances). Regarding CC and POF, bioenergy production also entailed remarkable environmental impacts (23% and 20%, respectively) due to fugitive methane emissions. The consumption of electricity from the national grid also showed important environmental burdens regarding energy-related impact categories such as CC, OD, POF and FD (ranging from 23% up to 52%).

The surplus digestate use in Plant B was identified as the main *hotspot* in this biogas system, especially regarding TA (76%), FE (69%) and ME (87%). Field emissions derived from the application of the digestate were the main contributors to these impact categories, specifically due to ammonia, phosphate and nitrate emissions, respectively. However, a positive effect was identified in this subsystem, since it has been considered that the use of digestate as an organic fertiliser helped to reduce the production and use of mineral fertilisers and their derived emissions. These environmental credits partially offset the environmental impacts produced by the application of digestate and impact categories such as CC, OD, POF and FD were specially favoured because of the avoided production of ammonium nitrate.

4.3. Comparative assessment

The European Commission published a working document on the sustainability of solid and gaseous biomass used for bioenergy production in 2014. Regarding biogas, the report highlighted the environmental concerns of the use of energy crops and encouraged

Table 7
Inventory data regarding bioenergy production per MWh of electricity produced.

	Plant A	Plant B
Materials and fuels		
Silage	2.15 t	0.25 t
Animal waste	0.61 t	178.08 t
Food waste		76.7 t
Diesel	0.33 kg	38.5 g
Tractor	0.04 kg	4.97 g
Agricultural implement	0.03 kg	3.64 g
Lubricant oil	0.16 kg	0.02 kg
Anaerobic digestion plant	$1.4\cdot10^{-4}$ p	$1.3\cdot10^{-4}$ p
Co-generation unit	$4.0\cdot10^{-5}$ p	$4.0\cdot10^{-5}$ p
Energy		
Electricity	50.2 kWh	235 kWh
Products		
Electricity	1000 kWh	1000 kWh
Heat	1106 kWh	1154 kWh
Digestate	2.11 t	6.20 t
Emissions to air		
Carbon dioxide	32.7 kg	18.7 kg
Methane	2.48 kg	2.60 kg
Carbon monoxide	14.2 g	8.11 g
Nitrogen oxides	4.44 g	2.54 g
NMVOC	0.59 g	0.34 g
Nitrous oxide	0.74 g	0.42 g
Sulphur dioxide	6.21 g	3.55 g

Table 8
Inventory data regarding digestate management and surplus digestate use per MWh of electricity produced.

	Plant A	Plant B
Digestate management		
Materials and fuels		
Raw Digestate	2.11 t	6.20 t
Products		
Digestate liquid fraction (own crops)	1.76 t	
Digestate solid fraction (own crops)	0.32 t	
Raw digestate for own crops		0.59 t
Raw surplus digestate		5.61 t
Emissions to air		
Ammonia	0.19 kg	0.12 kg
Nitrous oxide	0.09 kg	0.005 kg
Nitrogen	0.36 kg	0.15 kg
Nitric oxide	0.13 kg	0.01 kg
Methane	0.03 kg	0.21 kg
Carbon dioxide	0.04 kg	0.31 kg
Surplus digestate use		
Materials and fuels		
Digestate		5.61 t
Diesel		1.21 kg
Tractor		0.15 kg
Agricultural tillage		0.32 kg
Transport		
Tractor and trailer		44.9 t km
Avoided mineral fertilisers production		
Ammonium nitrate		8.89 kg
Emissions to air		
Ammonia		2.95 kg
Nitrous oxide		0.22 kg
Nitrogen		1.02 kg
Emissions to water		
Nitrate		5.89 kg
Phosphate		0.38 kg
Avoided emissions to air and water		
Ammonia		0.22 kg
Nitrous oxide		0.17 kg
Nitrogen		0.79 kg

the use of organic wastes. The report also established that existing bioenergy installations should achieve GHG savings of at least 70% compared to a fossil fuels reference system, defined as the benchmarking value (fossil fuel comparator) [14]. The estimation equates to life cycle emissions lower than 201 kg CO_2 eq per MWh of electricity produced from biomass.

Fig. 5 depicts a comparison of Plants A and B regarding the carbon footprint (CC), as well as the two reference systems: i) the fossil fuel comparator [14] and ii) the Italian electric profile for the year 2012 [35]. It is important to note that the methodological assumptions made in LCA studies affect the outcomes of the analysis. Therefore, the fossil fuel reference system and the results of the present study should be compared carefully, since it is possible that the system boundaries of both studies could be different.

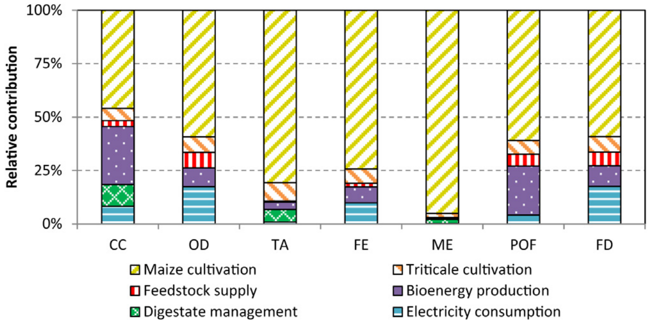

Fig. 3. Relative contributions from the processes considered in Plant A.

Both biogas systems achieved better environmental results compared with the environmental profile of the Italian electric grid. However, Plant B achieved results comparable to those proposed to consider bioenergy from biomass as environmental sustainable (i.e. fossil fuel reference). GHG emissions associated to feedstock supply and electricity consumption are higher in Plant B than in Plant A. The higher values of feedstock supply imply higher transport distances for the collection and delivery of OFMSW and food waste. In addition, once these waste streams are in the biogas plant, they need to be pretreated, which entails electricity requirements of 25% of the electricity produced in Plant B whereas it is only 7% in Plant A. However, as abovementioned, the cultivation of energy crops in Plant A is an important source of GHG emissions due to diesel consumption and direct emissions from the application of fertilisers (mineral and organic); therefore, the high ratio of energy crops digested in Plant A results in greater environmental impacts in CC compared to Plant B. Carbon footprint is the most widely used environmental indicator; however, addressing this indicator alone offers a very limited version of the overall environmental performance. The contribution of other relevant impact categories is shown in Fig. 6.

The comparison of the environmental profile of both biogas systems showed important differences for the different impact categories. While POF followed a similar behaviour to CC; in the case of OD and FD, the environmental impacts of Plant B were higher than in Plant A. The main reasons behind these results were the higher electricity consumption required in Plant B for the pretreatment of food waste and the transport distances.

Under a closer perspective, there are other factors that affect the performance of both biogas plants. One is the methane content of the biogas and the electric efficiency of the CHP. That is, for the production of 1 MWh of electricity, 560 m^3 of biogas are required in Plant A (45% CH_4) and 438 m^3 in Plant B (60% CH_4). Another issue is the feedstock, not only due to the different production and transport schemes but it will also influence the biogas yield and the composition of the digestate. As aforementioned, the production of energy crops was identified as the most important *hotspot* in Plant A, accounting for the major part of the environmental impacts.

Table 9
Characterisation results per MWh of electricity produced in Plant A.

Impact category	Total	Maize cultivation	Triticale cultivation	Feedstock supply	Bioenergy production	Digestate management	Electricity consumption
CC (kg CO_2 eq)	286	132	16.3	7.85	77.6	29.1	23.4
OD (kg CFC-11 eq)	$1.7 \cdot 10^{-5}$	$10.0 \cdot 10^{-6}$	$1.2 \cdot 10^{-6}$	$1.2 \cdot 10^{-6}$	$1.5 \cdot 10^{-6}$	0	$2.9 \cdot 10^{-6}$
TA (kg SO_2 eq)	8.76	7.29	0.774	0.138	0.297	0.174	0.084
FE (kg P eq)	0.059	0.044	0.004	0.001	0.004	0	0.006
ME (kg N eq)	1.33	1.27	0.027	0.002	0.008	0.018	0.003
POF (kg NMVOC)	1.20	0.727	0.077	0.066	0.272	0.009	0.049
FD (kg oil eq)	39.3	23.3	2.85	2.54	3.78	0	6.88

Table 10
Characterisation results per MWh of electricity produced in Plant B.

Impact category	Total	Maize cultivation	Feedstock supply	Bioenergy production	Digestate management	Electricity consumption	Electricity consumption
CC (kg CO_2 eq)	194	39.7	24.8	72.6	6.12	−59.0	110
OD (kg CFC-11 eq)	$1.8 \cdot 10^{-5}$	$3.9 \cdot 10^{-6}$	$4.2 \cdot 10^{-6}$	$4.8 \cdot 10^{-7}$	0	$-4.1 \cdot 10^{-6}$	$1.4 \cdot 10^{-5}$
TA (kg SO_2 eq)	8.36	1.02	0.079	0.226	0.286	6.38	0.363
FE (kg P eq)	0.174	0.021	0.003	0.002	0	0.121	0.027
ME (kg N eq)	1.82	0.194	0.003	0.005	0.011	1.59	0.014
POF (kg NMVOC)	0.633	0.281	0.092	0.200	0.002	−0.173	0.230
FD (kg oil eq)	42.2	9.62	8.63	1.49	0	−9.77	32.3

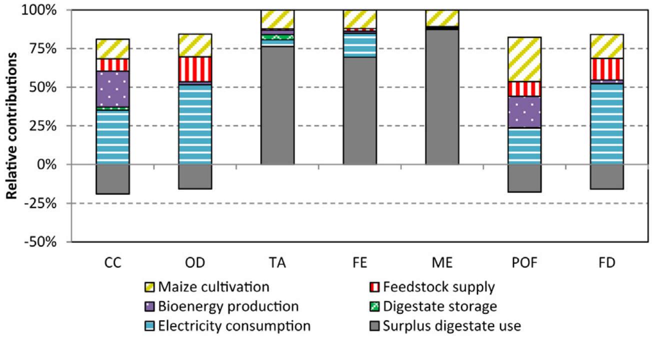

Fig. 4. Relative contributions from the processes considered in Plant B.

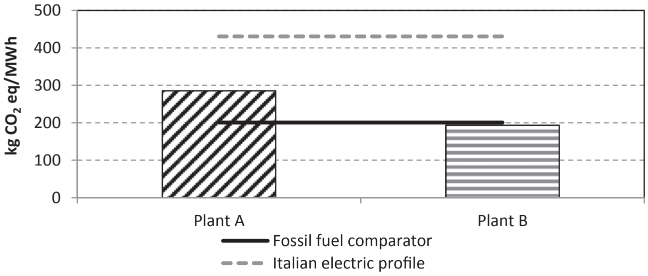

Fig. 5. Characterisation results of Plant A and B regarding CC.

However, it is important to note that they render into high biogas yield and it has influence in the overall environmental profile. More deeply, the anaerobic biogas potentials (AGP) of maize and triticale used as feedstock in these plants were 620 and 550 Nm^3 biogas/t TVS, respectively. Regarding Plant B, the AGP of OFMSW is reasonable (375 Nm^3 biogas/t TVS), but significantly higher for food waste (660 Nm^3 biogas/t TVS). Although the environmental burdens of its production are not considered as they are considered a waste, it is important to take into account the important energy requirements in the pre-treatment, which ended up into important impacts in energy-related categories such as OD and FD.

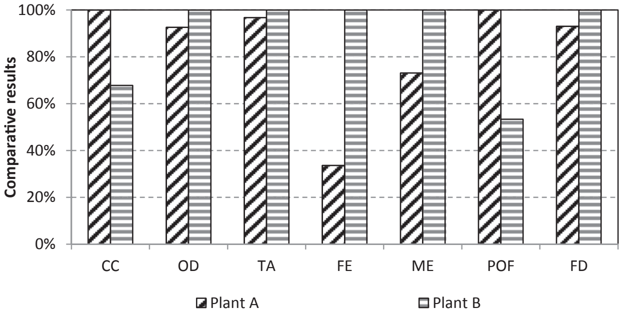

Fig. 6. Comparative environmental profile of each biogas plant.

Regarding TA, FE and ME, Plant B also achieved worse environmental results than Plant A. It is important to notice that more feedstock is fed to the digester, mainly associated to the ratio of pig slurry digested, which implies low biogas yield, higher digestate production and derived emissions from its storage and application on land. Moreover, TA and ME are mainly influenced by emissions of nitrogen-based compounds (i.e. ammonia and nitrate, respectively). Ammonia emissions arise from the storage and application of digestate while nitrate leaching only occurs in the agricultural land. In the case of FE, it is highly affected by phosphate leaching from the application of digestate as fertiliser.

As mentioned before, the LCI was built using average data from the operation of both biogas plants. However, background data was mostly taken from ecoinvent® v3.1. An uncertainty analysis was performed using Monte Carlo analysis in order to assess to what extent the uncertainties of the background data used in the study can influence the environmental results. The results obtained for a confidence internal of 95% and 5000 iterations [6] are depicted in Fig. 7.

According to the results, OD and FE present the highest data variability; while TA and ME displayed the lowest. This uncertainty comes from uncertainty values in some background processes taken from ecoinvent® that were expanded until the final results. In light of these results, a comparative uncertainty analysis was also performed in order to analyse which uncertainties affect the comparative assessment performed between both plants.

In order to interpret the results, 90% certainty has been established as a minimum to consider the difference significant. As shown in Fig. 8, regarding CC and POF Plant A showed higher impact than Plant B. Nevertheless, with 100% certainty Plant B produced higher impact than Plant A in terms of TA, FE and ME and with 95% in the case of FD. However, in terms of OD, Plant B produced higher impacts than A in 87% of the runs performed. These outcomes proved that the environmental comparative results obtained for the two plants under study were almost certain.

4.4. Sensitivity analysis

As demonstrated in the results obtained in this study, emissions from the storage of the digestate as well as from the application of digestate and mineral fertilisers such as ammonium nitrate and urea played a key role on the environmental profile, specifically in terms of impact categories such as TA, FE and ME. In related LCA studies available in literature [6,10,12], these emissions are usually estimated using different methodologies, since their measurement is difficult and not usually feasible to carry out. In this sense, although there is not general scientific consensus, there are different methodologies available that allow estimating these emissions [6].

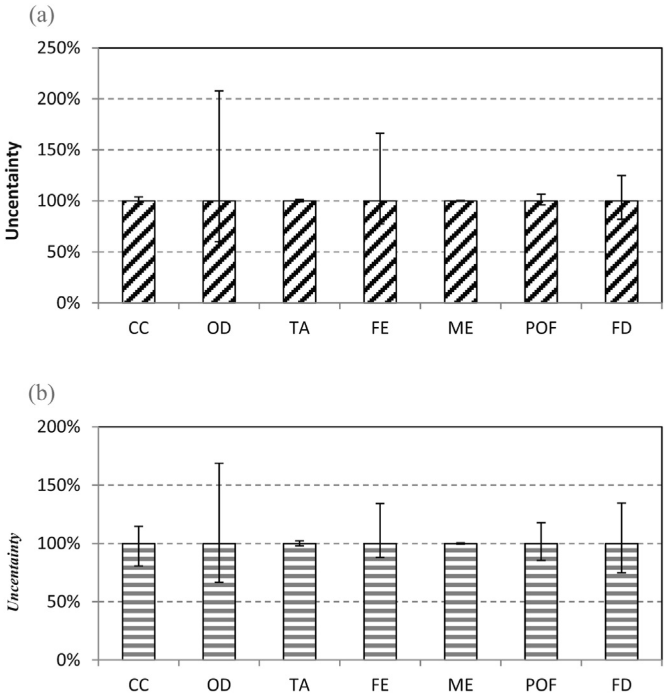

Fig. 7. Uncertainty of each impact category for (a) Plant A and (b) Plant B.

In this sensitivity analysis, the environmental results obtained in the base case study were compared with the results that would be obtained if two internationally accepted methodologies were considered: the IPCC and the EMEP/EEA methods.

- Option 1. In the base case, a combination between two methods were adopted. Storage emissions were calculated according to emission factors provided by De Vries et al. [26]. This methodology provides detailed emission factors for ammonia (0.04 kg NH_3-N/kg $TAN_{applied}$), nitrogen gas (0.01 kg N_2-N/kg $N_{applied}$) and nitric oxide (0.001 kg NO-N/kg $N_{applied}$). In addition, it allows differentiating methane and nitrous oxide emissions from the liquid and solid storage of organic substrates (0.001 kg N_2O-N/kg N and 0.17 kg CH_4/t for liquid storage and 0.02 kg N_2O-N/kg N and 0.004 kg CH_4/t for solid storage). Moreover, field emissions were computed according to Brentrup et al. [27]. This methodology takes into account different parameters that influence ammonia emissions such as average air temperature, infiltration rate and time between application and precipitation or incorporation into the soil. Additionally, emissions factors are defined for nitrous oxide (0.0125 N_2O-N/kg $N_{applied}$) and nitrogen gas (0.09 kg N_2-N/kg $N_{applied}$). Finally, nitrate is calculated by the balance between nitrogen entering the system (nitrogen from fertilisers and from atmospheric deposition) and leaving the system (nitrogen present in the crop as well as nitrogen as ammonia, nitrous oxide and nitrogen gas).
- Option 2. The methodology proposed in "IPCC Guidelines for National Greenhouse Gas Inventories" [37] was used as an alternative method. More deeply, Chapter 10 allows the calculation of methane and nitrous oxide emissions from manure storage. Indirectly, ammonia and nitrogen oxides can be also estimated. Specifically, using Tier 2 of the methodology, methane can be calculated considering temperature, the content in TVS, maximum methane producing capacity of the substrate and the type of storage (solid or liquid). Regarding the computation of nitrous oxide, Tier 2 provides emission factors for direct emissions depending on the type of storage (0.005 kg N_2O-N/kg N for solid storage and 0 kg N_2O-N/kg N for liquid storage). In order to calculate emissions of indirect nitrous oxide, this methodology also provides the percentage of nitrogen lost in the form of ammonia and nitrogen oxides (48% for liquid storage and 45% for solid storage); however, it does not allow differentiating between them. It has been considered that 90% of the nitrogen is lost in the form of ammonia, since it has been determined as the main emitted compound [38]. Chapter 11 of this report offers a methodology for the calculation of nitrogen-based emissions from the application of organic and inorganic substrates on land. In the same way, Tier 2 offers an emission factor for the calculation of direct nitrous oxide emissions (0.01 kg N_2O-N/kg $N_{applied}$); and through the calculation of indirect nitrous oxide emissions, ammonia, nitrogen oxides and nitrate can be also estimated (0.20 kg NH_3-N + NO_x-N/kg $N_{applied}$ for organic fertilisers, 0.10 kg NH_3-N + NO_x-N/kg N for mineral ones and 0.30 kg NO_3^--N/kg $N_{applied}$ for nitrate leaching). In this matter, in order to calculate the indirect nitrous oxide, the default emissions factors are 0.01 kg N_2O-N/kg NH_3-N + NO_x-N and 0.0075 kg N_2O-N/kg NO_3^--N).
- Option 3. The model presented in the EMEP/EEA Air Pollutant Emissions Inventory Guidebook 2009 enables the calculation of more accurate ammonia emissions [39]. Tier 2 suggests an emission factor of 0.0266 kg NH_3-N/kg $N_{applied}$ for raw digestate storage, 0.0116 kg NH_3-N/kg $N_{applied}$ for liquid digestate storage and 0.0150 kg NH_3-N/kg $N_{applied}$ for solid digestate storage. The methodology also points out that other compounds should be quantified such as total suspended particles, particulate matter and non-methane volatile organic compounds, but that at present, there are no methods to calculate them. Therefore, other emissions from the storage of digestate such as methane and nitrous oxide has been completed [37]. Regarding the application of fertilisers on land, this methodology also gives emission factors for the calculation of ammonia and nitric oxide emissions. In more detail, emission factors for the application of organic wastes are 0.08 kg NH_3/kg $N_{applied}$ and 0.04 kg NO/kg $N_{applied}$; while emissions from the application of ammonium nitrate are 0.016 kg NH_3/kg $N_{applied}$ and from the application of urea are 0.159 kg NH_3/kg $N_{applied}$.

Fig. 9 shows the comparative environmental profiles of Plant A and B when applying the different options for the calculation of direct emissions from storage and fertilisers application.

As expected, important differences were identified in the environmental impacts produced in TA, ME and POF. As aforementioned, the emission factors for ammonia are very different among methodologies. The variability observed in TA is caused

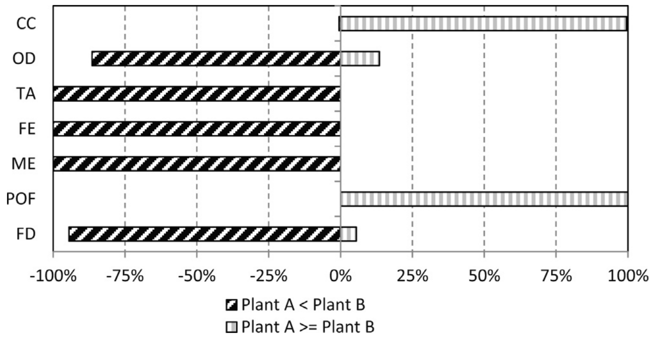

Fig. 8. Comparative uncertainty analysis of Plant A and B.

by differences in ammonia emissions. In more detail, the IPCC Guidelines for National Greenhouse Gas Inventories does not differentiate emissions from ammonia and nitrogen oxides and they are calculated from the TN of the substrate; while other methodologies such as the proposed in Option 1 estimates ammonia emissions according to its TAN content. This fact has been considered especially important since the ammonification process produced during anaerobic digestion is one of the main drawbacks of the process [6]. Moreover, Option 1 takes into account specific characteristics of the climate and agricultural field where fertilisers are applied. Moreover, differences observed in ME are the result of different nitrate leaching. Option 1 also estimates these emissions considering specific characteristics of the crop under study, including the yield and composition of the crop, other emissions and atmospheric nitrogen deposition, while Options 2 and 3 apply a direct emission factor. Finally, the variability in POF is motivated by different considerations of the methodologies. While Option 1 considers emissions of nitric oxide from the storage of substrates, Options 2 and 3 consider nitrogen oxides as NO_X. According to the LCA methodology applied, nitric oxide emissions produce impact only in ME (0.06 kg N eq/kg $NO_{emitted}$), while nitrogen oxides emissions impact on TA (0.56 kg SO_2 eq/kg $NO_{X\ emitted}$), ME (0.039 kg N eq/kg $NO_{X\ emitted}$) and POF (1 kg NMVOC/kg $NO_{X\ emitted}$).

5. Discussion

5.1. Potential methane production of substrates

As shown in the previous section, the variety of substrates used entailed different environmental performance due to a number of factors such as different composition (entailing different digestate characteristics) and methane yield as well as the production and transport schemes. The results reported here regarding the potential biogas production of the feedstock are specific for this case study. As shown in Table 11, other studies report different biogas productivities for the substrates under study.

For example, Gissén et al. [40] studied the potential biogas production from different energy crops in Sweden, ranging from 237 Nm^3 CH_4/t TVS for hemp to 408 Nm^3 CH_4/t TVS for beet root. They reported significantly higher methane yield for maize and triticale than the values reported here (Table 11). With these results, it would suppose 14% and 2% higher methane and thus, electricity production in Plant A and B, respectively. Although a reduction of the environmental impacts would be expected, the influence of agricultural practices would still be predominant on the environmental impact. González-García et al. [23] reported the environmental consequences of the cultivation of different energy crops in the same area of study, including wheat, triticale and different classes of maize. The authors concluded that the cultivation of different energy crops produced different environmental impacts due to different productivity yields, field requirements and direct emissions. Moreover, Dressler et al. [11] studied maize cultivation in three different areas of Germany and they pointed out that the environmental impacts varied according to regional farming procedures and specific characteristics of the area such as soil type and climate conditions.

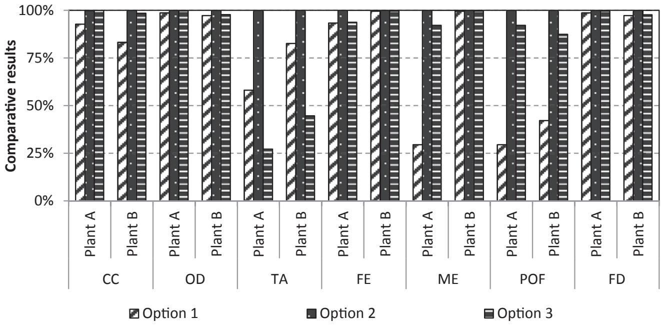

Fig. 9. Comparative environmental profiles of Plant A and Plant B considering different options for the estimation of storage and field emissions.

Besides, Møller et al. [41] also provided higher methane production factors for pig slurry compared with this study (see Table 11). This higher methane production would mean an increase in methane and electricity production of 1% and 8% in Plant A and B, respectively. From this minor contribution, it is important to highlight that the nitrogen and phosphorus content of the substrates has a strong influence on the environmental profile, especially in impact categories such as ME and FE due to nutrient leaching.

Finally, the methane potential of food waste depends on its composition which highly differs among regions and collection schemes. For example, Schott et al. [42] considered 443 Nm^3 CH_4/t TVS as the theoretical methane potential for separate collection of household food waste in Sweden. On the other hand, Evangelisti et al. [13] took into account for their calculations a methane production of 378 Nm^3 CH_4/t TVS regarding OFMSW in UK in accordance with Møller et al. [43].

5.2. Environmental sustainability of biogas

Similarly to other European countries, the public subsidy framework has led to the spread of biogas production in Italy [44]. In more detail, around 1300 biogas plants are nowadays operating in Italy, of which 56% are located in the Po Valley. Mainly due to the lack of a clear legislative framework, nearly all Italian biogas plants use biogas to co-generate electricity and heat. Heat produced is usually recycled to cover the heat demand of the plant [45] and electricity is injected in the national grid; however, biogas is a versatile biofuel that can be used for other purposes including its use as vehicle gas [46]. Interest has been paid to the use of organic waste from farm origin for biogas production [47]. In more detail, Carrosio [48] reported that in this area out of each five biogas plants, one uses animal manure as substrate, one digests energy crops, and the other three perform the co-digestion of both substrates. More specifically, processing manure to biogas through anaerobic digestion recovers the energy contained in the substrate and reduces the risk from pathogens during land spreading [47]. In fact, Tonini et al. [49] studied the most suitable pathway of converting biomass in bioenergy. The study proved that, for energy conversion of manure, anaerobic digestion appeared to be the most promising technology. Moreover, they concluded that unlocking the energy potential of manure and straw represented the greatest opportunity for GHG emission mitigation in both energy and agricultural sectors. However, the treatment of manure through anaerobic digestion depends on the economic viability of biogas plants installed in areas of livestock production, in which incentive policies play a key role. Smaller and dispersed installations allow a reduction of emissions associated with both manure transport and digestate management, while better supporting local farmer's income [50]. The study performed by De Vries et al. [10] presents the environmental consequences of the anaerobic mono-digestion of pig slurry and its co-digestion with different substrates. In order to boost agricultural biogas production, maize is nowadays the most used energy crop for biogas production in Europe due to its high productivity and potential of methane production [10]. The concerns related with the use of

Table 11
Methane yields for the substrates under study reported by different studies.

	Methane yield ($Nm^3_{methane}/t_{TVS}$)				
	Present study	Gissén et al. (2014)	Møller et al. (2004)	Schott et al. (2013)	Evangelisti et al. (2014)
Maize	322	361	–	–	–
Triticale	286	397	–	–	–
Pig slurry	252	–	356[a] 275[b]	–	–
Chicken manure	137	–	–	–	–
OFMSW	191	–	–	443	378
Food waste	350	–	–	–	–

[a] Fatteners.
[b] Sows.

cereals for energy purposes are not only related with the environmental impacts of its production. According to Mela and Canali [5], more than 10% of the available agricultural area in the Po Valley was occupied by energy crops, especially maize, and it tends to increase. Since different energy crops renders into different energy yields per hectare, it is essential to increase the efficiency of agricultural land use [40]. It is even possible that in the future, the agricultural land used for energy purposes may be limited by European regulations [40]. In this sense, Bartoli et al. [51] studied the effects of two alternative energy policies schemes for biogas subsidisation on maize price in Lombardy (Italy). They concluded that the change in policy measures implemented in 2013, which reduced the average subsidy per kWh of produced electricity, contributed to encourage the use of other substrates decreasing the price in maize due to lower the competition between the energy and food sectors. In this context, it is important to consider food waste as a possible and available co-substrate since it is generated in a daily basis worldwide and it needs to be managed in some way. Among the different valorisation technologies available, anaerobic digestion is regarded as one of the most efficient one since it allows reducing environmental impacts of other food waste treatment options such as landfill [52].

Some of the criteria needed to meet the requirements of sustainable biogas production includes secure energy supply, avoid competence with food production, socio-economic development including creation of local employment and reduced environmental impacts [53]. Moreover, according to the European Commission [15], the environmental sustainability of biomass use for bioenergy production may be reduced by several factors such as unsustainable agricultural practices, land use changes, direct and indirect emissions and inefficient bioenergy generation. Considering the results of the present study, some suggestions and further improvements can be made in order to improve the sustainability of biogas production.

- Since climate change is an important concern for European environmental policy, LCA studies are becoming increasingly relevant for policy decision-making. Therefore, for the proper development of LCA studies, a higher level of harmonisation in the methodology application is required to make the studies comparable and transparent [9]. Therefore, as already stated by Bacenetti et al. [9], there is a need for common and very specific guidelines for LCA studies to assess and communicate the environmental performance.
- It is also important to consider that, as shown in this study, climate change should not be the only focus of environmental concerns of policies regarding bioenergy production. Acidification and eutrophication impacts mainly linked to energy crops cultivation and digestate management should be integrated in the environmental policies.
- In addition, as proved by Dressler et al. [11], local factors and regional parameters have a strong effect in LCA results. Therefore, it is necessary to consider regional parameters (e.g., transport distances, agricultural area for biomass production and digestate spreading, competition for cereal silage between biogas production and livestock activity) with the aim of performing a representative LCA study. Only if regional variations are considered, the results of environmental indexes will be representative, as the results could vary from one region to another.
- Regarding the selection of the feedstock, as suggested by Whiting and Azapagic [3], economic incentives should include further requirements on feedstock type to promote the use of different types of wastes and to prevent the use of energy crops that may compete with other uses.

6. Conclusions

This research work demonstrates that renewable energies can achieve GHG emission savings when compared to conventional fossil reference systems, being able to improve the environmental profile of the national electric profile. However, the environmental results obtained were strongly dependent of the specific substrate selected and the digestate management. In Plant A, which uses high ratio of energy crops, the cultivation step was identified as the main environmental *hotspot*. Specifically, the consumption of diesel within the agricultural machinery required in the field activities produced impacts in energy-related categories; while the derived emissions from the application of mineral fertilisers and digestate contributed with important impacts in acidifying and eutrophying categories. The characterisation of the OFMSW and food waste used Plant B demonstrated that these substrates can be an alternative co-substrate able to improve the environmental profile of biogas production. More deeply, they have higher energy potential than pig slurry and no environmental burdens from their production are allocated to the biogas system. Nevertheless, electricity consumption in the biogas plant is increased due to the pre-treatment requirements of this type of waste. Moreover, the use of substrates such as pig slurry with lower biogas potential than energy crops resulted in a higher amount of digestate per unit of electricity produced in Plant B. In this sense, the spread of the produced digestate in agricultural land produced important environmental impacts of acidifying and eutrophying substances. In this regard, a sensitivity analysis demonstrated how the environmental profile of both biogas plants would change considering different accepted methodologies for the calculation of emissions derived from the storage and application of digestate.

Acknowledgements

This research was supported by the BBVA programme “2015 edition of the BBVA Foundation Grants for Researchers and Cultural Creators” (2015-PO027), by the UE projects LIVE-WASTE (LIFE 12 ENV/CY/000544) and ManureEcoMine (project funded by the European Community's Seventh Framework Programme under Grant

Agreement nº 603744) as well as by the Galician Government through the call “Proyectos de Excelencia – Modalidad D” (project reference ED431F 2016/001). Dr. S. González-Garcia would like to express her gratitude to the Spanish Ministry of Economy and Competitivity for financial support (Grant reference RYC-2014-14984). The authors (L. Lijó, S. González-García and M.T. Moreira) belong to the Galician Competitive Research Group GRC2013-032 and to the CRETUS Strategic Partnership (AGRUP2015/02). All these programmes are co-funded by FEDER (UE).

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