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Research article

Consequential environmental life cycle assessment of a farm-scale biogas plant

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ABSTRACT

Producing biogas via anaerobic digestion is a promising technology for meeting European and regional goals on energy production from renewable sources. It offers interesting opportunities for the agricultural sector, allowing waste and by-products to be converted into bioenergy and bio-based materials. A consequential life cycle assessment (cLCA) was conducted to examine the consequences of the installation of a farm-scale biogas plant, taking account of assumptions about processes displaced by biogas plant co-products (power, heat and digestate) and the uses of the biogas plant feedstock prior to plant installation.

Inventory data were collected on an existing farm-scale biogas plant. The plant inputs are maize cultivated for energy, solid cattle manure and various by-products from surrounding agro-food industries. Based on hypotheses about displaced electricity production (oil or gas) and the initial uses of the plant feedstock (animal feed, compost or incineration), six scenarios were analyzed and compared. Digested feedstock previously used in animal feed was replaced with other feed ingredients in equivalent feed diets, designed to take account of various nutritional parameters for bovine feeding. The displaced production of mineral fertilizers and field emissions due to the use of digestate as organic fertilizer was balanced against the avoided use of manure and compost.

For all of the envisaged scenarios, the installation of the biogas plant led to reduced impacts on water depletion and aquatic ecotoxicity (thanks mainly to the displaced mineral fertilizer production). However, with the additional animal feed ingredients required to replace digested feedstock in the bovine diets, extra agricultural land was needed in all scenarios. Field emissions from the digestate used as organic fertilizer also had a significant impact on acidification and eutrophication.

The choice of displaced marginal technologies has a huge influence on the results, as have the assumptions about the previous uses of the biogas plant inputs. The main finding emerging from this study was that the biogas plant should not use feedstock that is intended for animal feed because their replacement in animal diets involves additional impacts mostly in terms of extra agricultural land. cLCA appears to be a useful instrument for giving decision-makers information on the consequences of introducing new multifunctional systems such as farm-scale biogas plants, provided that the study uses specific local data and identifies displaced reference systems on a case-by-case basis.

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1. Introduction

The massive development of the biofuel industry has been accompanied by several controversial issues, including the food *versus* fuel debate. The use of raw food materials for bioenergy production has diverted some resources (including agricultural products and land) from their initial use. For example, in Wallonia (the southern part of Belgium) in 2012, more than 25% of the wheat produced was transformed into bioethanol (Delcour et al., 2014).

The issue of the optimum management of any product, co-

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product, by-product or waste has become very important. Regarding waste management in particular, the European Commission issued Directive 2008/98/EC (European Commission, 2008), which proposed the following hierarchy for dealing with waste: (a) prevention, (b) preparation for re-use, (c) recycling, (d) other recovery, *e.g.*, energy recovery, and (e) disposal. As one of the three Belgian Regions, Wallonia translated this Directive into a Walloon Decree (Walloon Government, 2012), but this decree does not go as far as its Flemish counterpart in northern Belgium, which set up a more detailed hierarchy for food waste management: (a) prevention, (b) use for human nutrition, (c) conversion for human nutrition, (d) use for animal feed, (e) use as raw materials in industry (in a bio-based economy), (f) processing into fertilizer by anaerobic digestion or composting, (g) use as renewable energy, (h) incineration and (i) landfill (Roels and Van Gijseghem, 2011).

The European Renewable Energy Directive 2009/28/EC (European Commission, 2009) set a restrictive goal whereby Belgium, overall, should obtain 13% of its energy from renewable sources by 2020. With regard to electricity, the goal in Wallonia is to produce a little more than 25% of its estimated final electricity consumption from renewable sources by 2020 (CWaPE, 2012). Among the available sources, biogas production via anaerobic digestion is a promising technology that could contribute significantly to these goals. Biogas plants can be fed with numerous raw materials, including agricultural, industrial and domestic by-products and waste, and can deliver various types of energy, such as electricity, heat, steam, combined heat and power (CHP), and gas that can be supplied to the natural gas grid or used as transportation fuel (Holm-Nielsen et al., 2009).

Supported by the Green Certificates mechanism (CWaPE, 2012; Van Stappen et al., 2007) and other investment aid schemes, an increasing amount of electricity in Wallonia has been generated from biogas-fueled CHP plants in recent years; between 2002 and 2012, this amount increased by 20%, and in 2012, it represented approximately 3% of total electricity consumption in the region (Simus, 2014). There are 37 biogas plants in Wallonia, 10 of which are fed with agricultural raw materials; between them, these 10 plants have an installed power capacity of 9.2 MW_{el} (EBA, 2012).

The environmental impacts of biogas production from farm-scale plants vary considerably, depending on regional parameters such as raw material availability for digestion, the energy service provided, soil, climate and the reference systems affected by the use of the co-products (Dressler et al., 2012). The influence of farming practices has also been highlighted (Alig, 2012; Börjesson and Berglund, 2007; Jury et al., 2010; Stucki et al., 2011). Optimizing the potential benefits of biogas plants calls for systems designed and located wisely (Börjesson and Berglund, 2007), as well as for environmental assessment studies such as life cycle assessments (LCA) that take account of local conditions (Dressler et al., 2012).

Using LCA fed with local data collected on-site, this study aimed at evaluating the environmental consequences of the installation of a farm-scale biogas plant producing electricity, heat and organic fertilizer. Plant feedstock was silage maize and farmyard manure, as well as by-products from surrounding agro-food industries (sugar beet tails, downgraded potatoes, cereal middlings, mown lawn grass and starch from potato fry cleaning). This study sought to explore the sensitivity of the results to assumptions on (i) the reference systems displaced by the use of the co-products and (ii) the uses of the feedstock prior to the biogas plant being installed.

2. Methods

The study followed ISO standards for LCA guidelines and requirements (ISO, 2006a, b).

2.1. Consequential LCA

There are two types of LCA, depending on the goal of the study: attributional LCA (aLCA) and consequential LCA (cLCA). aLCA describes the relevant physical input and output flows entering and exiting from a product system, whereas cLCA defines how these flows might be modified in response to a decision or a change (Finnveden et al., 2009). aLCA is useful for identifying systems with important impacts, whereas cLCA is useful for evaluating the consequences of individual decisions. The complementary goals of aLCA and cLCA make them both valid for decision-making (Ekvall et al., 2005): cLCA is more complete but less certain while aLCA is more certain but implies blind spots related to deficient consideration of secondary effects, such as affected processes and technologies outside aLCA system boundary (Schmidt, 2008). cLCA, however, seems more appropriate in regard to informing decision-makers about the environmental impact of installing a new multifunctional technology that increases the amount of products on the market (Jury et al., 2010; Rehl et al., 2012). Approaches for conducting a cLCA can utilize economic data to measure physical flows of indirectly affected processes (Earles and Halog, 2011) or include economic concepts such as marginal production costs, elasticity of supply and demand (Finnveden et al., 2009). An alternative to economic models is based on the qualitative identification of the most likely processes marginally affected by a change in the main production system (Vázquez-Rowe et al., 2013). This approach uses market information and identifies the scale and time horizon of the potential change studied (Schmidt, 2008; Weidema et al., 2009). Processes affected by diverted inputs required by the system and products provided by the system are called *displaced technologies*. They are *short-term marginal technologies* (*i.e.*, existing technologies whose output changes due to small changes in demand in the market). They need to be *unconstrained* so that they can adjust their capacity in response to changes in demand. *Short-term* implies that the changes take place within the existing production capacity and are not expected to affect capital investment (Weidema et al., 1999).

2.2. Goal and scope of the cLCA

The biogas plant under study produces three co-products: power, heat and digestate (the sludge-like material remaining after anaerobic digestion). All three co-products are used, replacing processes that delivered the same service pre-installation. The electricity produced is used for the plant and the farm, with the excess being sold to the grid. Heat is used partly for digester heating and partly sold to neighborhood houses *via* a 440 m district heating network. Excess heat (*i.e.*, surplus after the needs of the biogas plant and the houses are met) is dissipated (although in the future, the plan is for this excess heat to be used for drying wood chips). The digestate is stored in an open tank before being used as organic fertilizer.

In the joint production of power, heat and digestate by the biogas plant, power is identified as the *determining* product, *i.e*. the co-product for which a change in demand will affect the production volume of the co-producing unit process (Weidema et al., 2009). Indeed the economic viability of a biogas plant in Wallonia is closely linked to electricity sales and subsidies for green electricity production *via* the Green Certificates mechanism (Heneffe, 2014). In order to evaluate the consequences of the installation of the biogas plant, the following functional unit was used: 1 additional MJ of electricity supplied to the grid by the biogas plant.

The analysis was based on the framework proposed by Weidema et al. (2009); Schmidt (2008) and included the impacts of (1) the biogas plant operation (*i.e*., energy crop [silage maize] production, feedstock transport to the plant, digester feeding and operation, CHP plant operation, heat and power distribution and digestate storage and use), (2) the processes displaced by co-product (power, heat and digestate) use, and (3) the processes displaced by digester feedstock initial use (*i.e.*, pre-installation use). Indeed the plant inputs (silage maize, farmyard manure, sugar beet tails, downgraded potatoes, cereal middlings, mown lawn grass and starch from potato fry cleaning) had other uses (or treatments) prior to plant installation and are now replaced by other products or processes delivering equivalent services. Therefore, the system boundaries were expanded to include (Fig. 1):

1. Electricity replaced by the electricity produced by the biogas plant and supplied to the grid or used on the farm (displaced electricity)
2. Heat replaced by the heat produced by the biogas plant and sold to neighboring houses (displaced heat)
3. Mineral fertilizers (production and use) replaced by digestate as organic fertilizer (displaced fertilizers)
4. Processes replacing digester feedstock according to their previous uses or treatments

2.2.1. *Electricity supplied to the grid and used on the farm*

Electricity production from a biogas-fueled CHP plant is, in principle, stable and continuous, apart from interruptions for maintenance or because of an unstable equilibrium in the digester. It would therefore replace a dynamic electricity production, that is to say, able to adjust its operation according to demand on an hour-by-hour basis (Mathiesen et al., 2009).

In the present study, to identify this marginal technology, the production, demand and capacity statistics of the Belgian electricity system were analyzed. In Belgium, electricity is produced mainly from nuclear power plants (47.5%). The contribution of thermal power plants (natural gas, oil and coal) is on the decline (FEBEG, 2015), with a share of 31.91%, 1.54%, 3.24% for natural gas, oil and coal, respectively (ELIA, 2016). Electricity from nuclear power plants is constrained, at least in the short-term, and could not, in the context of this case study, be considered the marginal technology, whereas power production from thermal plants is fairly flexible (they can be operated during the day only, when the electricity price is higher, as is the profit margin). Electricity from oil power plants could be seen as the marginal technology because of its volatile pricing, linked to oil prices, and its decreasing share in Belgian electricity production, whereas electricity from biomass is continuously increasing (Delporte, 2011; Massant, 2010). Electricity production from natural gas power plants has also been decreasing since 2010 and gas-fueled power plants are often used for secondary production back-up. The profit margin for electricity production from gas is, however, almost always lower than that for a coal-fired power plant, and in 2011 and 2012, the cost of electricity from gas-powered plants was greater than the price at which electricity was sold (CREG, 2013).

Considering these elements, our analysis of the Belgian electricity market showed that electricity produced by thermal power plants fueled with natural gas, oil or coal could be affected when additional electricity from a biogas plant was fed into the grid. The last coal-fired power plant in Wallonia, however, was shut down in 2010, and this technology is now present only in Flanders (northern region of Belgium); consequently this marginal technology was explored in the sensitivity analysis only. Thermal power plants fueled by either oil or gas were identified as marginal technologies for the electricity supplied by the biogas plant to the grid; these two technologies were distinguished in scenario analysis.

The farm requirements for animal husbandry and the farmhouse are met by the electricity produced by the plant. This electricity was initially supplied by the grid. The Belgian electricity mix was therefore the marginal technology displaced by the electricity used on the farm.

2.2.2. *Heat supply to neighboring houses*

The heat produced by the biogas plant is fed into a district

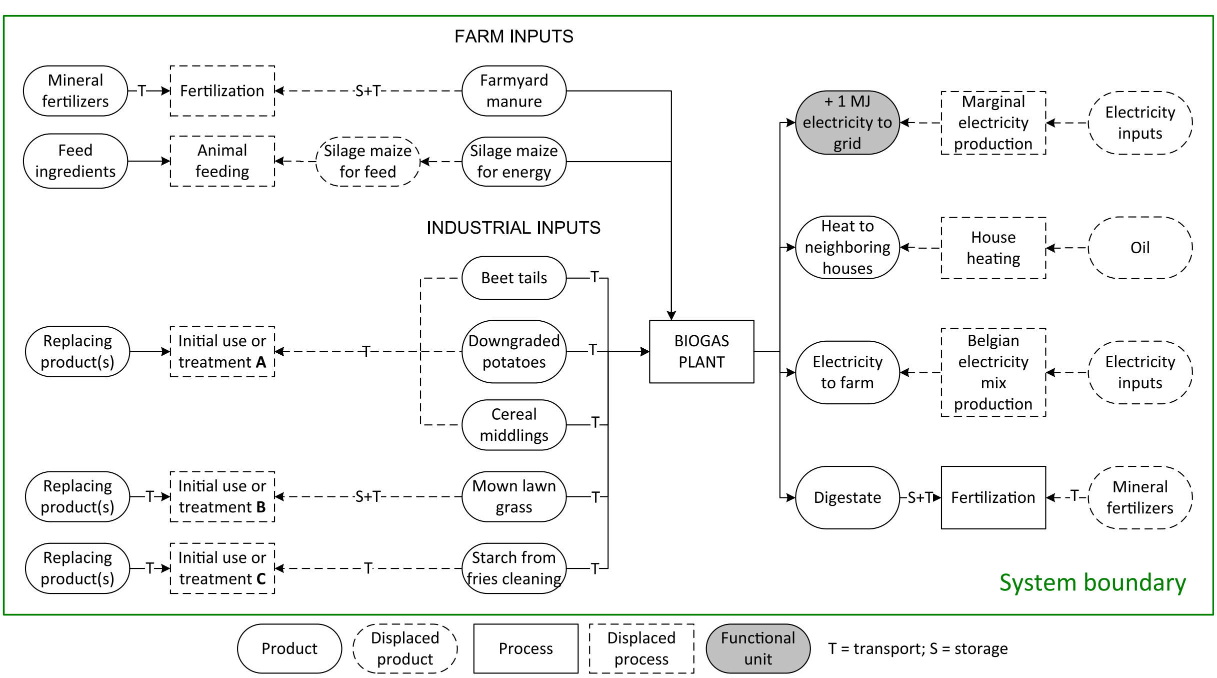


Fig. 1. System boundary; in the FEED-scenarios: initial use or treatment A is animal feeding, B is composting, use as fertilizer, C is starch production; in the COMP- scenarios: initial use or treatment A, B, C are composting, use as fertilizer; and in the INCI- scenarios: initial use or treatment A, B, C are incineration.

heating network connected to neighboring houses. These include eight single-family houses and nine apartments that, in total, previously consumed 40,000 l of light fuel oil per year. Heat from oil-fueled domestic boilers was identified as the marginal technology.

2.2.3. Digestate use as organic fertilizer

On-farm data collection showed that the annual production of digestate (3500 t) is used to fertilize 10 ha of maize silage, 25 ha of cereals and 40 ha of meadow annually at rates of 30, 30 and 60 t fresh matter (FM)/ha year, respectively. Using the digestate as an organic fertilizer displaces the mineral fertilizers previously used to fertilize these crops. However, according to our methodology (Fig. 1), the farmyard manure and compost previously used as organic fertilizers and now fed into the digester need to be replaced by mineral fertilizers. Calculations related to mineral equivalences of organic fertilizers are detailed in the supplementary material (section S1).

The marginal technologies for mineral fertilizer production are triple superphosphate and potassium chloride, the most commonly used single nutrient mineral fertilizers in Wallonia for P and K nutrition, respectively (Van Stappen et al., under review to the International Journal of Life Cycle Assessment). As for N fertilization, calcium ammonium nitrate (CAN) and urea ammonium nitrate (UAN) fertilizers are used almost equally in Wallonia, and both were investigated. The influence of the displaced mineral fertilizer (CAN or UAN) was tested through a sensitivity analysis.

In addition to the impact on the production of displaced fertilizers, another important aspect is field emissions resulting from their application (Van Stappen et al., under review to the International Journal of Life Cycle Assessment). Although there is much uncertainty about field emissions related to mineral and organic fertilizer application (Van Stappen et al., under review to the International Journal of Life Cycle Assessment), a simple approach (Tier1-Tier2) was used to determine the importance of these emissions. Emissions from digestate use were accounted for, as well as displaced emissions from the manure, compost and mineral fertilizers previously applied to 10 ha of silage maize, 25 ha of cereals (winter wheat) and 40 ha of meadow. The crop yield amounts were provided by the farm's accounting data (SPW/DAEA, 2015). Field emissions were calculated using the emission models described in the supplementary material (section S6). A field emissions balance (FEB) was calculated accounting for the addition of emissions from digestate application and the subtraction of emissions from the solid manure, compost and mineral fertilizers applied pre-installation (see calculations in the supplementary material, section S2).

Previously stored manure was considered to now be fed directly into the digester. Displaced emissions (CH_4, N_2O and NH_3) from the avoided need to store manure were calculated (see supplementary material, section S3).

2.2.4. Pre-installation use or treatment of the digester feedstock

Since no reliable information could be collected regarding the use or treatment of the industrial feedstock prior to plant installation, scenarios were designed to take account of their realistic pre-installation uses. These by-products from surrounding agro-food industries would have been sold to the animal feed industry or put into a composting facility or an incineration plant, as reported by Börjesson and Berglund (2007).

If animal feed was assumed to be the initial use, then silage maize and some of the digested agro-industrial by-products (sugar beet tails, downgraded potatoes and cereal middlings) had previously been used as animal feed, whereas mown lawn grass had previously been composted and the starch from potato fry cleaning sold to the starch industry market. After the installation of the biogas plant, silage maize, beet tails, potatoes and cereal middlings were replaced by other feed ingredients in equivalent animal diets. Dairy cows and fattening bulls were selected for the present case study because they are the most important livestock in Wallonia (as opposed to pigs) (DGSIE, 2014). A single feed ingredient cannot usually simply be replaced by another single ingredient, given the numerous parameters to be taken into account in designing a balanced animal diet (*e.g.*, energy, protein, digestibility, fibers). Using diet calculation sheets developed by Sekul (2011) and adapted to Belgian feed requirements according to CVB (2000) and Subel (2008), balanced diets for fattening bulls and dairy cows were designed. All diets were considered representative of the Walloon feed market and used equally (the choice depending on ongoing prices, a parameter not accounted for here). The fattening bull diets were balanced in terms of animal energy and protein supply requirements for rearing 500 kg live-weight bulls with an average daily weight gain of 1.3 kg/day. The dairy cow diets were balanced to meet daily requirements of energy, protein, acid detergent fiber (ADF), neutral detergent fiber (NDF) and a milk production of 22.5 kg/day.

Some of the diets used digested feedstock (silage maize, beet tails, potatoes and cereal middlings), but others did not. A *feed ingredient balance* was calculated, assuming that feed diets including digested feedstock (pre-installation) were replaced by equivalent balanced feed diets that did not contain digested feedstock (current situation, post-installation). These balances between equivalent diets enabled us to evaluate which feed ingredient in which amount would be added to or subtracted from the system for each kg of digested feedstock used in the biogas plant. Calculations related to these feed ingredients balances are explained in the supplementary material (section S4).

In this study, digested raw materials were considered to be equally distributed between the fattening bull and dairy cow diets. The influence of the type of animal (cow or bull) whose diet was displaced was explored through a sensitivity analysis.

If compost is assumed to be the pre-installation use of industrial digester feedstock, then all digester feedstocks from neighboring agro-industries (sugar beet tails, downgraded potatoes, cereal middlings, mown lawn grass and starch from potato fry cleaning) were composted up to the time that the biogas plant was installed.

If incineration is assumed to be the treatment pre-installation, then all digester feedstocks from neighboring agro-industries were incinerated up to the time that the biogas plant was installed.

Wherever possible, local data were used for the inventories of displaced processes. If data were missing, LCI data were extracted from ecoinvent v3.1, with the system model using allocation at the point of substitution (Weidema et al., 2013). If these data were not available, the system model using cut-off allocation was used. Data process names and sources are listed in the supplementary material (section S5).

2.3. Life cycle inventory data collection for the plant

2.3.1. Description of the biogas plant

The biogas plant is on a dairy farm in Famenne, an area of Wallonia dedicated to forage crop production (grass, maize) and cattle breeding. It is 215 m.a.s.l. and is located at 50.2°N and 4.7°E, with a mean temperature of 9.6 °C and an annual precipitation of 940 mm. The biogas plant is fed with silage maize and solid cattle manure, produced on the farm and on two neighboring farms, as well as by-products from surrounding agro-food industries. The raw materials fed to the digester are listed in Table 1. The biogas produced by the plant, 45% carbon dioxide (CO_2) and 55% methane (CH_4), is burnt in a CHP plant (Table 1).

Table 1
Main characteristics of the biogas combined heat and power (CHP) plant (source: on-farm data collection).

Parameter	Unit	Value
Feedstock (total)	tFM/year	4159
Silage maize	tFM/year	387
Solid cattle manure	tFM/year	410
Sugar beet tails	tFM/year	2006
	km	77
Mown lawn grass	tFM/year	788
	km	10
Downgraded potatoes	tFM/year	300
	km	15
Cereal middlings	tFM/year	152
	km	40–150
Starch from potato fry cleaning	tFM/year	116
	km	15
Technology	One-step continuously stirred tank reactor (CSTR)	
Digester capacity	m^3	1500
Hydraulic residence time (HRT)	Days	40–50
Working temperature	°C	35–38
Digestate production	t/year	3500
Biogas production	m^3/year	525,881
CHP plant operation time	h/year	7680
Installed electrical power	kW_{el}	104
Electrical efficiency	%	37.6
Electricity sold to network	kWh_{el}/year	535,393
Electricity used on farm	kWh_{el}/year	31,107
Installed thermal power	kW_{th}	138
Thermal efficiency	%	50.74
Heat production	kWh_{th}/year	1,059,840
Digester heating	kWh_{th}/year	260,000
To district network	kWh_{th}/year	402,810
Excess heat (dissipated)	kWh_{th}/year	397,030
Emissions		
Digester degassing in case of overpressure:		
CO_2, biogenic	kg/year	4425
CH_4, biogenic	kg/year	1940
Digestate storage emissions:		
CO_2, biogenic	kg/year	13,768
CH_4, biogenic	kg/year	7409
CHP plant emissions:		
Particulates	kg/year	152
CO, biogenic	kg/year	2534
NO_x	kg/year	1352
NMVOC	kg/year	11
N_2O	kg/year	14
SO_2	kg/year	120
Platinum	kg/year	3.98E^{-5}

FM: fresh matter.

2.3.2. Silage maize production

Since the installation of the biogas plant, silage maize produced for energy purposes ('energy maize'), yielding 39,166 kg of fresh matter (FM) per ha and harvested at 33% dry matter (DM), has been cropped on land previously used to produce silage maize for feed purposes ('feed maize'). The energy maize is fertilized with digestate from the biogas plant. Silage maize conventionally produced for animal feed had a yield of 51,243 kgFM/ha at 33% DM (SPW/DAEA, 2015). The yields differ because the use of digestate as the only fertilizer provides the maize with fewer nutrients than the mineral fertilizers used pre-installation. The amount of energy maize displacing feed maize was calculated using a ratio based on the corresponding yields (*i.e.*, 1 kg FM of energy maize replaces 1.31 kg FM of feed maize [51,243/39,166]). Details on inventory data collection for silage maize production are given in supplementary material (section S6).

2.4. Life cycle impact assessment

In order to fit to the goal of the study, life cycle impact assessment (LCIA) is achieved at the midpoint level. Even though more understandable for decision support, LCIA at endpoint level, *i.e.* expressing results as damages to human health, ecosystems and resource availability, is not considered desirable in this study, due to its increased uncertainty (Bare et al., 2000). LCIA is based on a composite method using midpoint LCIA methods recommended by the International Reference Life Cycle Data System (ILCD) Handbook (European Commission, 2010), later updated by Hauschild et al. (2013). The LCA software used is SimaPro 8.0.4.30 (PRé, 2015). Studied impact categories are selected according to their relevance for agricultural LCAs (Haas et al., 2000; Audsley et al., 1997). The selected impact categories, related indicator units and reference methods are listed in Table 2.

2.5. Scenario analysis

The combination of the assumptions about the use of the biogas plant co-products (sections 2.2.1, 2.2.2 and 2.2.3) and the pre-installation uses of the agro-industrial feedstock (section 2.2.4) led us to define six scenarios to be analyzed and compared in this study. These scenarios, summarized in Table 3, are 1) FEED-OIL, 2) FEED-NGAS, 3) COMP-OIL, 4) COMP-NGAS, 5) INCI-OIL and 6) INCI-NGAS. The -OIL and -NGAS scenarios relate to the displaced electricity originally produced from oil or natural gas. The FEED-, COMP- and INCI- scenarios relate to the pre-installation uses of the agro-industrial feedstock as animal feed, compost and incinerated matter, respectively. They also influence the impact of the displaced mineral fertilizers and field emissions (see supplementary material, sections S1 and S2).

2.6. Sensitivity analyses

Sensitivity analyses were conducted to establish the extent to which the final results of the study depend on a given choice or assumption (Wegener Sleeswijk et al., 1996). The tested hypotheses were expressed as the percentage change between the results of the sensitivity analysis and the original results using the following calculation:

$$\%\ change = \frac{(sensitivity - original)}{original} * 100 \tag{1}$$

An assumption was considered sensitive if the percentage of change between the sensitivity test and the original results reached a given minimum difference, depending on the impact category. These minimum differences were based on thresholds established by expert judgment and proposed by Jolliet et al. (2010). This approach accounts for individual impact characterization factors uncertainty as well as variation between substances characterization factors. Jolliet et al. (2010) recommend minimum 10% difference in order to declare results are significantly different in GWP and RED categories. For TAP and EUP, this minimum difference is 30%. Regarding HTP and AEP, one or even two orders of magnitude are necessary to declare two contributions are significantly different, considering impact characterization factors high uncertainty and the huge number of substances playing a role in toxicity impacts. As no threshold is proposed for ALO, POF and WDP, we used a 10% minimum difference.

Apart from the scenario analyses, five additional sensitivity analyses were conducted, using the FEED-OIL scenario as the baseline for comparison.

Coal-fired power plants could also be seen as a marginal

Table 2
Selected impact categories, related indicator units and reference methods used for LCIA.

Abbreviation	Impact category	Indicator unit	Reference method
GWP	Global warming potential with a timeframe of 100 years	kg CO_2 eq.	(IPCC, 2013)
TAP	Terrestrial acidification potential based on Accumulated Exceedance (AE)	10^{-3} AE eq./kg	(Posch et al., 2008)
EUP	Eutrophication potential	10^{-3} kg PO_4^{3-} eq.	CML-IA baseline v4.2 (Guinée et al., 2002)
ALO	Agricultural land occupation	m^2y (m^2 year)	ReCiPe v1.11 (Goedkoop et al., 2009)
WDP	Water depletion potential	10^{-3} m^3	Water Scarcity (Pfister et al., 2009)
POF	Photochemical oxidant formation	10^{-3} kg NMVOC eq.	ReCiPe v1.11 (Goedkoop et al., 2009)
RED	Mineral, fossil and renewable resource depletion	kg Sb eq.	ILCD 2011 Midpoint (Van Oers et al., 2002)
HTP	Human toxicity potential	10^{-6} CTUh (Comparative Toxic Units)	USEtox (Rosenbaum et al., 2008)
AEP	Aquatic ecotoxicity potential	CTUe (Comparative Toxic Units)	USEtox (Rosenbaum et al., 2008)

Table 3
Tested scenarios, combining assumptions regarding the use of the biogas plant co-products, the initial use of the digester feedstock and related displaced processes.

Displaced processes	Scenarios					
	FEED-OIL	FEED-NGAS	COMP-OIL	COMP-NGAS	INCI-OIL	INCI-NGAS
Displaced electricity to grid	Oil	Natural gas	Oil	Natural gas	Oil	Natural gas
Displaced electricity to farm	Belgian mix		Belgian mix		Belgian mix	
Displaced heat to houses	Oil		Oil		Oil	
Initial uses of the digester feedstock:						
Silage maize for energy	Silage maize for animal feed		Silage maize for animal feed		Silage maize for animal feed	
Solid cattle manure	Organic fertilizer		Organic fertilizer		Organic fertilizer	
Agro-food industry by-products (beet tails, downgraded potatoes and cereal middlings)	Animal feed		Compost		Incineration	
Mown lawn grass	Compost		Compost		Incineration	
Waste starch from potato fry cleaning	Potato starch market		Compost		Incineration	
Balance for displaced mineral fertilizers and field emissions	=Digestate − (farmyard manure + composted grass)		=Digestate − (farmyard manure + composted beet tails, potatoes, cereal middlings, grass and starch)		=Digestate − farmyard manure	

technology for electricity production, as investigated in the *first* sensitivity analysis.

At the time of data collection on the biogas plant (2012), roughly half of the heat produced by the CHP plant was unused and dissipated. Regulations in Wallonia now encourage the optimal use of heat produced in CHP plants, and the plant owners planned to use the excess heat for drying wood chips. The *second* sensitivity analysis explored this possibility to evaluate the additional benefits of avoiding the use of conventional means (oil) to dry wood chips.

The *third* sensitivity analysis related to the choice of N mineral fertilizer displaced by digestate use and the avoided use of farmyard manure and compost. In the scenarios, CAN and UAN were used in equal proportions. The sensitivity of the results was analyzed when either 100% CAN or 100% UAN were used as displaced N mineral fertilizers.

The FEED-scenarios considered that half the digested materials were displaced in the fattening bull diets and half in the dairy cow diets. The *fourth* sensitivity analysis related to the type of animals whose diets were displaced (either only fattening bulls or only dairy cows).

In the *fifth* sensitivity analysis, one digested substrate was displaced by only one feed ingredient on the basis of their energy content for fattening bull and dairy cow feed, reflecting the approach used by van Zanten et al. (2014). Silage maize was replaced by grass silage, and beet tails, potatoes and cereal middlings were replaced by barley (calculations are available in the supplementary material, section S7). This enabled us to test the soundness of our in-depth approach accounting for all nutritional factors in a balanced diet (*e.g.*, energy, protein, digestibility, fibers) to evaluate the consequences of subtracting any feed ingredient.

2.7. Uncertainty analyses

Uncertainty analyses were run for the scenario and sensitivity analyses using Monte-Carlo simulations (MCS) implemented with SimaPro 8.0.4.30 (Hedemann and König, 2003). MSC calculate the difference between compared processes (subtracting results from A from those of B). The procedure for propagating dispersions in data uses dependent sampling yielding relative results (Henriksson et al., 2015). Standard deviations were attached to each input parameter in the process inventories. When the standard deviations could not be calculated from primary data, data quality indicators in pedigree matrixes were used, enabling us to consider the data reliability, completeness and temporal, geographical and technological correlations (Weidema and Wesnaes, 1996). MCS (1000 runs, 95% confidence interval) were performed to determine (i) the error range in the LCIA results of the scenarios and (ii) if the results differed significantly in the scenario and sensitivity analyses.

MCS performed with SimaPro accounted for input and output inventory data uncertainty but not for uncertainties linked to impact characterization factors. It was therefore proposed to mitigate the uncertainty results with semi-quantitative analyses based on expert judgment, as recommended by Jolliet et al. (2010). To account for this uncertainty mitigation in the results, significant differences were invalidated when the above-mentioned minimum difference between compared results (section 2.6) was not reached.

Besides MCS for the water depletion potential (WDP) gave insignificant differences even when the results differed considerably. This can be seen as a consequence of the updated methodology in ecoinvent v3.1 for water balance; most processes are now water balanced (*i.e.*, the amount of water entering an activity is equal to the amount leaving the activity) (Moreno Ruiz et al., 2014).

Input water flows are given a positive characterization factor (consumed water) in all water footprint methods implemented in SimaPro 8.0.4, whereas output water flows released to the water are given a negative characterization factor (water returning to water). It seems that MCS are unable to take into account the potential correlations between inputs and outputs (*e.g.*, if irrigation increases, water flows released to rivers also increase, and *vice versa*, thus correlating these input and output flows). Given the standard deviations attached to the inventory data, this implies that MCS can pick up a high value for an input and a low value for an output, although they could be correlated in terms of water balance. Considering the many processes involving water flows, MCS with water balance as modeled in ecoinvent v3.1 cannot produce significant differences. As upstream ecoinvent processes were also used in our data, it was decided not to take MCS results into consideration for WDP.

3. Results and discussion

The LCIA results for 1 additional MJ of electricity produced by the biogas plant in the different scenarios are presented in Table 4. The details and error ranges of the results in relation to the processes involved are shown in Fig. 2.

3.1. Biogas plant operation

The operational impact of the biogas plant (energy crop production, feedstock transport, digester feeding and emissions, digestate storage emissions, CHP plant functioning and emissions) represented less than 10% of the absolute total impact (*i.e.*, adding positive impacts to the absolute value of negative impacts) for the WDP and AEP impact categories. This figure, however, rose to between 27% and 34% for the GWP category, 43–75% for TAP, 44–66% for EUP, 6–17% for ALO, 36–50% for POF, 11–28% for RED, and 8–55% for HTP. The remaining impact values were attributable to displaced processes, producing either a positive or negative impact.

Digestate storage in an open tank led to residual CO_2 and CH_4 losses and, to a lesser extent, N_2O emissions, contributing to 12–16% of the GWP impact. Other LCA studies showed that biogas collection during digestate storage is an important factor for GHG emissions reduction (Hijazi et al., 2016); covering the digestate storage tank would be an efficient way to improve the plant GHG balance.

CHP plant emissions represented 22–31% of the POF impact due to NO_x, carbon monoxide (CO), non-methane volatile organic compound (NMVOC) and sulfur dioxide (SO_2) emissions.

Digester feedstock transport from the surrounding industries had a small impact (representing maximum 3% of the total impact) in all impact categories, except for RED, where it contributed to 9–22% of the total impact.

3.2. Impact of displaced processes and products

The results depicted in Fig. 2 show the importance of the impact of displaced processes and products, which was often higher than the impact of the plant itself. This corroborates findings reported by Börjesson and Berglund (2007) and Reinhard and Zah (2009), who observed that the results of a cLCA depended on the environmental scores of the marginal replacement products on the market rather than on local production factors. In other words, the marginal products assumed to be affected are the most important factor in the results.

3.2.1. Scenario analysis

All of the scenarios showed that, compared with the pre-installation situation, there was a reduced impact for AEP and WDP and additional impact for ALO. In the other impact categories, the results were divided, depending on the scenario (Table 4).

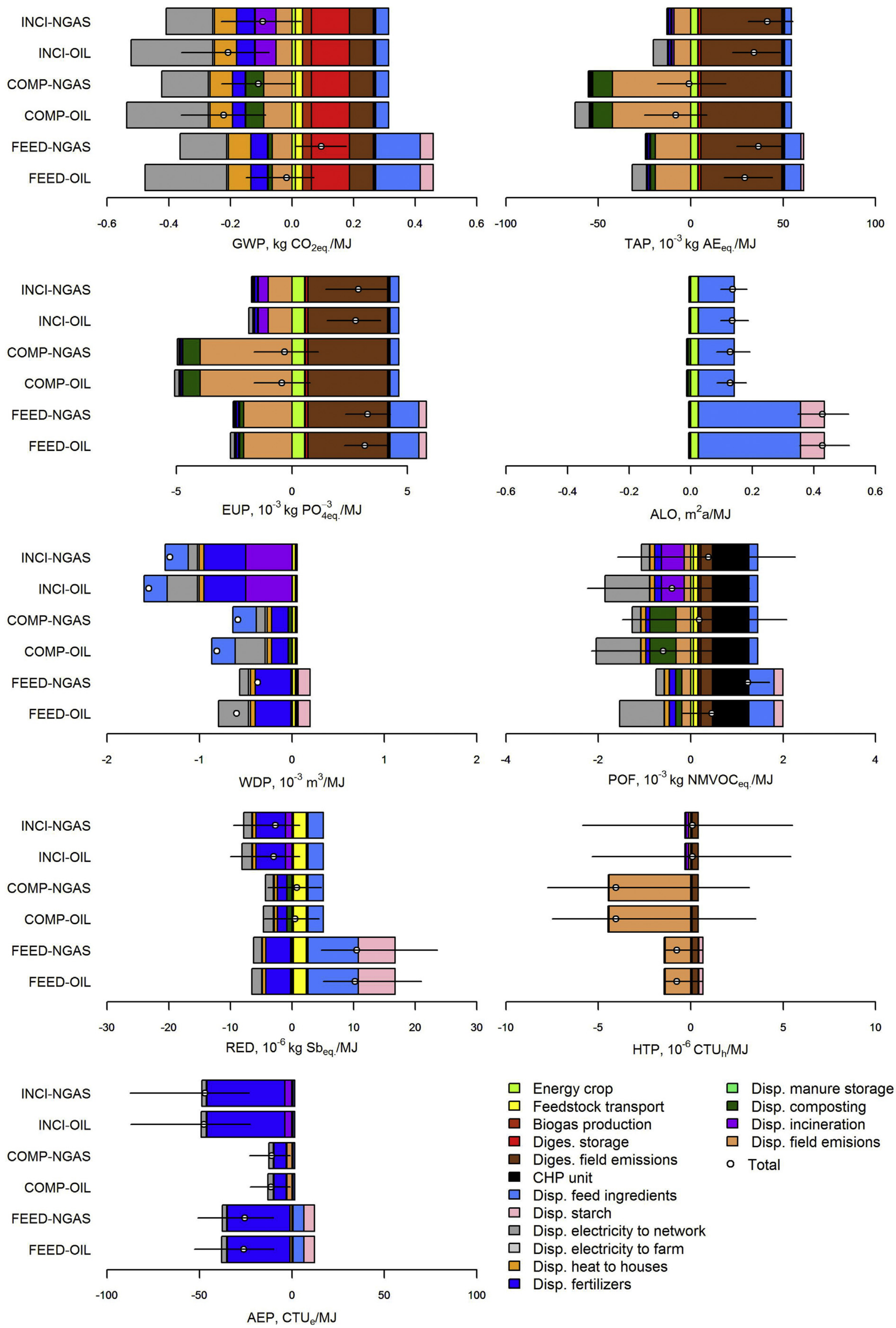
The impact of the FEED-OIL scenario was similar to the pre-installation situation with regard to GHG emissions. Where natural gas was the marginal technology for electricity production (FEED-NGAS), however, the impact for GWP was significantly greater than in the pre-installation situation. In the INCI-and COMP- scenarios, the GHG balance was in favor of the biogas plant, significantly if oil was the marginal technology for electricity production but not significantly for natural gas (Fig. 2).

For TAP and EUP, compared with the pre-installation situation, the impact of the COMP- scenarios was similar, but in the FEED- and

Table 4
LCIA results and Monte-Carlo simulations (MCS) per impact category for each scenario.

Results	Unit	FEED-OIL	FEED-NGAS	COMP-OIL	COMP-NGAS	INCI-OIL	INCI-NGAS
GWP	kg CO_2 eq.	−0.018	0.096	−0.222	−0.050	−0.208	−0.094
MCS[a]		–	*	***	–	***	–
TAP	10^{-3} AE eq.	29.37	36.64	34.15	9.81	34.25	41.52
MCS[a]		–	(***)	(***)	–	0.06	–
EUP	10^{-3} kg PO_4^{3-} eq.	3.16	3.28	3.53	0.43	2.76	2.88
MCS[a]		–	(***)	(***)	–	0.22	–
ALO	m^2y (m^2 year)	0.43	0.43	0.13	0.14	0.14	0.14
MCS[a]		–	0.45	***	–	***	–
WDP	10^{-3} m^3	−0.60	−0.37	−0.81	−0.55	−1.55	−1.32
MCS[a,b]		–	N/A	N/A	–	N/A	–
POF	10^{-3} kg NMVOC eq.	0.46	1.24	−0.28	0.75	−0.40	0.38
MCS[a]		–	***	***	–	0.13	–
RED	10^{-6} kg Sb eq.	10.23	10.53	0.50	0.80	−2.99	−2.69
MCS[a]		–	0.29	***	–	***	–
HTP	10^{-6} CTUh	−0.76	−0.75	−4.06	−4.05	0.092	0.098
MCS[a]		–	0.31	0.12	–	0.34	–
AEP	CTUe	−26.02	−25.54	−11.42	−10.94	−47.56	−47.08
MCS[a]		–	0.29	(*)	–	(***)	–

Shaded areas highlight reduced impact after installation of the biogas plant.
GWP: global warming potential; TAP: terrestrial acidification potential; EUP: eutrophication potential; ALO: agricultural land occupation; WDP: water depletion potential; POF: photochemical oxidant formation; RED: mineral, fossil and renewable resource depletion; HTP: human toxicity potential; AEP: aquatic ecotoxicity potential.
[a] Comparisons with the FEED-OIL scenario; *: significant differences ($p < 0.05$); **: highly significant differences ($p < 0.01$); ***: very highly significant differences ($p < 0.001$); N/S: *p* values are indicated for non-significant differences ($p \geq 0.05$); MCS results between brackets are invalidated (see Section 2.7).
[b] No MCS run for WDP (see Section 2.7).


INCI-NGAS
INCI-OIL
COMP-NGAS
COMP-OIL
FEED-NGAS
FEED-OIL
-0.6 -0.4 -0.2 0.0 0.2 0.4 0.6
GWP, kg CO_{2eq}/MJ
-100 -50 0 50 100
TAP, 10^{-3} kg AE_{eq}/MJ
-5 0 5
EUP, 10^{-3} kg PO_{4eq}^{3-}/MJ
-0.6 -0.4 -0.2 0.0 0.2 0.4 0.6
ALO, m^2a/MJ
-2 -1 0 1 2
WDP, 10^{-3} m^3/MJ
-4 -2 0 2 4
POF, 10^{-3} kg $NMVOC_{eq}$/MJ
-30 -20 -10 0 10 20 30
RED, 10^{-6} kg Sb_{eq}/MJ
-10 -5 0 5 10
HTP, 10^{-6} CTU_h/MJ
-100 -50 0 50 100
AEP, CTU_e/MJ
Energy crop
Feedstock transport
Biogas production
Diges. storage
Diges. field emissions
CHP unit
Disp. feed ingredients
Disp. starch
Disp. electricity to network
Disp. electricity to farm
Disp. heat to houses
Disp. fertilizers
Disp. manure storage
Disp. composting
Disp. incineration
Disp. field emisions
Total

INCI- scenarios, it was significantly greater.

For POF, the FEED-NGAS scenario had a significant additional impact. The FEED-OIL scenario, as well as the COMP- and INCI- scenarios, was similar to the pre-installation situation, with again a small advantage when oil was the marginal technology used for electricity production. The error range in the COMP- and INCI- scenarios for POF, however, was considerable (Fig. 2), making the results highly uncertain. This is related to CHP emissions which were extrapolated from other plants due to lack of on-site emission measurements.

Displaced field emissions led to avoided impacts, though highly uncertain, for HTP in the FEED-scenarios and even more notably in the COMP- scenarios.

For RED, FEED-scenarios added a significant impact to the pre-installation situation, due to displaced feed ingredients and starch production. COMP- scenarios were neutral while INCI- scenarios were slightly in favor of the biogas plant. Large uncertainty is connected with this impact category.

For all of the impact categories, the FEED-scenarios had a greater impact than the COMP- and INCI- scenarios, primarily because of the greater feed balance in the FEED-scenarios.

The influence of the marginal technology chosen for electricity production was also crucial. The -NGAS scenarios had a greater impact than the -OIL scenarios in all of the impact categories (except ALO, where the impact was not influenced by the marginal electricity technology). These differences were significant, however, only for GWP and POF (Table 4); the MCS showed significant differences for TAP and EUP as well, but these differences were invalidated, according to Section 2.7.

3.2.2. *Electricity supplied to the grid and used on the farm*

The displaced electricity production contributed substantially to the avoided impact in the GWP (17–29% of the absolute total, depending on the scenario), AEP (5–20%), WDP (7–35%), POF (7–29%) and RED (6–17%) impact categories. The impact reduction varied considerably, however, with the marginal technology chosen. For GWP, for instance, displacing the electricity production from oil (OIL-scenarios) led to a reduction of 0.265 kg CO_2eq/MJ. Displacing the electricity production from natural gas, a cleaner technology (NGAS- scenarios), led to a reduction of 0.152 kg CO_2eq/MJ. In some impact categories, such as GWP and POF, when the displaced electricity was produced from gas-fired power plants, the installation of the biogas plant created an extra burden compared with the pre-installation situation. When this technology was oil-fueled power plants, however, there were more potential benefits.

As tested in sensitivity analysis, electricity production from coal is the technology with the greatest impact: its displacement led to a reduction of 0.317 kg CO_2eq/MJ. Replacing electricity from oil-fired power plants with electricity from coal-fired power plants therefore reduced the impact. The switch between oil and coal, however, was sensitive only for GWP (−298%), WDP (−16%) and POF (+86%), and these differences were not significant in terms of the MCS.

It also seems likely that not just one but several electricity production technologies could be affected when additional electricity is supplied to the grid. Identifying one single marginal technology is therefore not only difficult but also probably inaccurate because these technologies operate and interact in a complex energy system (Mathiesen et al., 2009). Considering the long-term, the consequences of installing a biogas plant should also take into account, in addition to flexible thermal power plants, other technology displacements for electricity production, such as a reduction in imported electricity or the planned shutdown of some of the nuclear power plants (Dufresne et al., 2009).

Electricity used on the farm, representing 6% of the amount of electricity sold to the grid (Table 1) and displacing the Belgian electricity mix, had negligible effects on the results.

3.2.3. *Heat supply to neighboring houses*

With the displaced heat production from oil-fueled domestic boilers, there was a visible impact reduction in GWP (7–9%), leading to savings of 0.073 kg CO_2eq for each MJ of electricity produced by the plant. Furthermore, as tested in sensitivity analysis, using the excess heat produced for drying wood chips affected the results by displacing industrial heat produced from oil. If this drying was included in the FEED-OIL scenario, a further decrease of 0.070 kg CO_2eq./MJ could be achieved for GWP. This would roughly double the benefits already observed for heat supplied to the neighboring houses. This highlights the soundness of optimizing the use of heat produced by CHP plants, as similarly observed by De Meester et al. (2012).

3.2.4. *Digestate use as organic fertilizer*

The mineral fertilizer balance reduced the impact in AEP (−6.69 to −42.10 CTUe/MJ), representing 46–84% of the absolute total in this category, and to a lesser extent, in GWP (−0.04 to −0.06 kg CO_2 eq./MJ), WDP ($-1.81E^{-4}$ to $-4.52E^{-4}$ m^3/MJ) and RED (−1.57 to $-4.82E^{-6}$ kg Sb eq./MJ), representing 5–8%, 20–51% and 16–37% of the absolute totals in these categories, respectively. The displaced production of mineral fertilizers enables fairly reducing the impact of the biogas plant, as similarly demonstrated by Schaubroeck et al. (2015) and Alanya et al. (2015). As tested in sensitivity analysis, the choice of N mineral fertilizer (CAN or UAN) in the fertilizer balance did not prove sensitive, except for GWP: the production of CAN has more impact than UAN, resulting in a greater impact for GWP (+13%).

Field emissions account for a substantial share of the GWP, TAP, EUP, HTP and POF impact categories. Additional emissions from digestate use, however, were to some extent compensated for by avoided emissions from the replaced manure and mineral fertilizer, especially in the COMP- scenarios.

In the FEED-scenarios, the field emissions balance (*i.e.* emissions from digestate use less emissions from displaced manure, compost and mineral fertilizers use) had a notable additional impact for TAP ($+2.47E^{-2}$ AE eq./MJ) and EUP ($+1.36E^{-3}$ kg PO_4^{3-} eq./MJ), representing 28% and 16% of the absolute totals in these categories, respectively. In the INCI- scenarios, the field emissions balance led to a substantial additional impact for TAP ($+3.48E^{-2}$ AE eq./MJ), EUP ($+2.42E^{-3}$ kg PO_4^{3-} eq./MJ) and HTP ($+2.28E^{-7}$ CTUh/MJ), representing 49%, 38% and 30% of the absolute totals in these categories, respectively. The field emissions balance, however, led to reduced impacts in HTP in the FEED- ($-1.05E^{-6}$ CTUh/MJ) and the COMP- scenarios ($-4.09E^{-6}$ CTUh/MJ), representing 50% and 84% of the absolute totals in these categories, respectively, and in AEP in the COMP- scenarios (−2.41 CTUe/MJ), representing 17% of the absolute total. These avoided impacts in HTP contrast with findings from Schaubroeck et al. (2015) who estimated the impact caused by digestate application to be much higher than the impact avoided due to displaced usage of conventional fertilizers. This is explained

Fig. 2. Life cycle impact assessment for the production of 1 MJ of electricity by the biogas plant according to each scenario and for each impact category. GWP: global warming potential; TAP: terrestrial acidification potential; EUP: eutrophication potential; ALO: agricultural land occupation; WDP: water depletion potential; POF: photochemical oxidant formation; RED: mineral, fossil and renewable resource depletion; HTP: human toxicity potential; AEP: aquatic ecotoxicity potential; Diges.: digestate; Disp.: displaced; whiskers represent the confidence interval of the population at $p = 0.05$.

by the fact that the present study also considers displaced use of manure and compost, both organic fertilizers with high trace metal contents (Piazzalunga et al., 2012) contributing to HTP impacts.

Results show the significant contribution of field emissions to the impact of the biogas plant, especially for TAP and EUP. For these impact categories, using digestate as an organic fertilizer increased the acidification and eutrophication potentials compared with using the conventional combination of mineral fertilizers and farmyard manure. There is high uncertainty, however, about the generic emission models for fertilizer application impact (Van Stappen et al., under review to the International Journal of Life Cycle Assessment), and there are no specific emission factors for the digestate in the models used (De Vries et al., 2012; EMEP/EEA, 2013). Furthermore, with regard to using digestate as fertilizer, there are contradictions in the literature relating to pollution risks. Walsh et al. (2012) reported that digestate use reduces the risk of NH_3 volatilization because it is more rapidly absorbed by the soil than the thicker undigested slurry, which remains longer on the surface. Börjesson and Berglund (2007), however, stated that the higher content of ammonium in the digestate, which can be converted into NH_3, leads to increased emissions of NH_3 (on average, 250–310 g NH_3/t of digested manure). Amon et al. (2006) and De Meester et al. (2012) share this opinion. Juarez-Rodriguez et al. (2012) considered that the high soluble C and N contents of digestate could lead to significant N_2O emissions, especially if applied to moderately wet soils. Petersen (1999), however, reported larger N_2O emissions from undigested manure than from anaerobically digested manure due to the less easily decomposed organic matter in the latter. In practice risks of pollution from digestate application can be reduced by digestate injection, reducing risks of NH_3 volatilization, and an appropriate timing of application, reducing risks of NO_3^- leaching (Lukehurst et al., 2010). The importance of field emissions combined with the lack of consensus in the scientific community reinforces the need for specific local data and emissions measurements when considering mineral and organic fertilizer replacement by digestate from biogas production.

3.2.5. Pre-installation use or treatment of the digester feedstock

The pre-installation use or treatment of the digester feedstock had a substantial influence on the results.

The feed ingredient balance (*i.e.*, feed ingredients replacing digested feedstock in equivalent bovine diets) required the occupation of additional agricultural land: 0.33 m^2y/MJ in the FEED-scenarios and 0.12 m^2y/MJ in the COMP- and INCI- scenarios, representing 75–79% of the absolute total in the ALO impact category. Based on an input of 1,927,415 MJ/year (535,393 kWh_{el}/year) to the grid, this would correspond to an extra land occupation of 64 ha for the FEED-scenarios and 22 ha for the other scenarios for each year of plant operation.

A similar study to ours reported that a reduction of 154 m^2 could be achieved for each ton of sugar beet tails diverted from a biogas production system towards dairy cattle feed (van Zanten et al., 2014). The authors found that, as in the present case, the additional land occupation was mainly due to additional grain production (barley in their case, barley and wheat in our case) required to replace the industrial by-products in animal feed.

The impact from direct and indirect land-use changes in terms of, *inter alia*, GHG emissions and biodiversity is very difficult to estimate and uncertain (Tonini et al., 2012; Van Stappen et al., 2011), and it was not accounted for in this study. Although this impact was not quantified, results showed indisputable land-use changes as a result of the additional agricultural land required with the installation of the biogas plant, whatever the situation with regard to the pre-installation uses of the digester feedstock or the displaced electricity production technology.

In the FEED-scenarios, the feed ingredient balance also had an impact on GWP (14–16% of the total impact), TAP (10–11%), EUP (15%), AEP (12%), POF (15–20%), RED (35–36%). In the digester feedstock that, pre-installation, had been used for animal feed, cereal middlings were the ingredients whose replacement had the greatest impact. They were replaced mainly by feed ingredients whose production, mostly or entirely intended for animal feed (*e.g.*, barley, wheat or soybean meal), has a high impact. In the COMP- and INCI- scenarios, where only silage maize needed to be replaced, the impact of the feed balance was less visible, except for WDP (15–36% of the total impact), where there was a negative impact because rape meal production was subtracted from the system.

Where sugar beet tails, downgraded potatoes and cereal middlings had been used in animal feed, the installation of the biogas plant had an additional environmental impact in most of the impact categories. Taking these by-products out of the animal feed market added to the environmental impact of the biogas plant. Using such by-products as animal feed instead of biofuel is therefore more environmentally friendly, as similarly reported in other studies (Alig, 2012; Vandermeersch et al., 2014; van Zanten et al., 2014). It also supports the Flemish waste hierarchy (Roels and Van Gijseghem, 2011), whereby use for animal feed should be given priority over processing into fertilizer through anaerobic digestion or composting and over use as renewable energy.

As noted in sensitivity analysis, whether the diets of dairy cows or fattening bulls were displaced was not a sensitive parameter, except for WDP and RED: feeding dairy cows instead of fattening bulls reduces impacts by 24% and 30%, respectively. Performing the same exercise with pig diets, though, would probably have produced different results, as reported by van Zanten et al. (2014), who observed a decrease in GHG emissions per ton of wheat middlings when these were fed to dairy cattle rather than to pigs. Indeed the diets of monogastric animals, such as pigs, include a higher share of grains and cereal by-products (60%) compared to the ruminant diets (20%) (Van Stappen et al., 2014).

When compared to the present approach to designing balanced diets to replace other balanced diets, taking account of all nutritional factors, the sensitivity analysis considering a single feed ingredient that replaced the digested feedstock based solely on its energy content, gave sensitive differences for the GWP (+254%), WDP (−35%), POF (−41%) and RED (−33%) categories. This means that the present in-depth approach, taking account of all of the nutritional requirements of animal feed, led to a better understanding of the consequences of replacing feed ingredients in animal diets. It is worth mentioning, however, that equivalent diets used in this study were based on nutrient values only and not on economic considerations. In practice, farmers switch from one equivalent ingredient to another, depending on price, availability and supply mode. These diets were seen as being equally used, but some might have been preferred over others for economic and market availability reasons. In this respect, combining a socio-economic LCA (UNEP/SETAC, 2009) with environmental LCA would be appropriate to fully understand all of the consequences of installing a biogas plant, taking account of such factors as the input prices (feed, fertilizers), working conditions and added value (also for local communities) of the considered system and displaced reference systems (Delcour et al., 2015).

Animal husbandry was seen as unchanged by the installation of the biogas plant (Rehl et al., 2012) and was therefore excluded from the system boundary. However, potential changes in enteric methane emissions from cattle due to the changes in their diets could have been considered because these emissions are linked to feed intake and digestibility (Doreau et al., 2011; Mills et al., 2001).

The impact of additional starch from the market in the FEED-scenarios for GWP was +0.04 kg CO_2 eq./MJ, for ALO +0.08 m^2y/

MJ, for HTP $+2.27E^{-7}$ CTUh/MJ, for AEP +5.63 CTUe/MJ, for POF $+1.90E^{-4}$ kg NMVOC eq./MJ and for RED $+5.95E^{-6}$ kg Sb eq./MJ, representing from 4 to 36% of the total impact in these categories.

In the COMP-scenarios, the displaced composting operation (not the use of the compost) reduced the impact for POF ($-5.67E^{-4}$ kg NMVOC/MJ, representing 16–21% of the absolute total) and, to a lesser extent, GWP (−0.06 kg CO_2eq/MJ), TAP ($-1.07E^{-2}$ kg SO_2eq/MJ), EUP ($-7.54E^{-4}$ kg PO_4^{3-} eq/MJ) and RED ($-8.02E^{-7}$ kg Sb eq./MJ), representing 6–10% of the absolute total in these categories.

In the INCI- scenarios, the impact of the displaced incineration of biowaste for WDP was $-5.01E^{-4}$ m^3/MJ, representing 30–35% of the absolute total, for POF $-4.89E^{-4}$ kg NMVOC/MJ, representing 15–19% of the absolute total, for HTP $-1.57E^{-7}$ CTUh/MJ, representing 21% of the absolute total, and, to a lesser extent, for GWP −0.07 kg CO_2eq/MJ and RED $-1.04E^{-6}$ kg Sb eq./MJ, representing 6–9% of the absolute totals in these two categories.

4. Conclusions, limitations and recommendations

This case study used a cLCA approach to investigate the impact of installing a farm-scale biogas plant. Various scenarios with regard to the displaced technology for electricity production and the pre-installation uses of the digester feedstock were compared. Furthermore, scientific novelty was brought by the elaborated methodology to calculate the consequences of digestate use on displaced mineral fertilizers and field emissions. An original, in-depth approach was also applied to evaluate the consequences of the replacement of digester feedstock previously used in animal diets.

The environmental impact of the farm-scale biogas plant depended mainly on the processes displaced by the use of the plant's co-products and the pre-installation uses of its feedstock. Displaced electricity and mineral fertilizers and, to a lesser extent, displaced heat enabled this technology to show environmental benefits. These benefits were offset, however, if the digester materials had previously been used in animal feed, resulting in an additional impact in most of the impact categories compared to the pre-installation situation.

The marginal technology chosen for electricity production has a great influence on the results and, in some impact categories, led to opposing results (*i.e.*, the biogas plant had a total positive or negative impact, depending on the chosen marginal technology). Identifying the marginal technology for electricity production displacement on the grid is therefore a sensitive factor, and an in-depth study on displaced reference processes needs to be conducted to objectively assess the environmental impact of a farm-scale biogas system.

The impact of feeding the biogas plant with agro-food by-products previously used in animal feed and replacing these ingredients in equivalent animal diets can outweigh the environmental benefits of producing electricity, heat and organic fertilizer from this plant. The additional land occupation was the most important factor in the installation of the biogas plant. Direct and indirect impacts of this additional agricultural land should be accounted for in future work to assess the extent to which this detracts from the benefits of the biogas system.

In conclusion, this study showed that using specific local data when conducting a cLCA should be preferred in regard to informing decision-makers about the contribution of biogas technologies towards meeting regional and European renewable energy targets. The study showed that installing a biogas plant can have environmental benefits if the raw materials have not previously been used in animal feed. The additional impact in terms of the extra agricultural land required because of the displaced feed ingredients reinforces the need to make it a priority to feed biogas plants with non-edible raw materials.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2016.03.020.

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