ARTICLE IN PRESS

Waste Management xxx (2015) xxx–xxx

Contents lists available at ScienceDirect

Waste Management

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

Environmental assessment of digestate treatment technologies using LCA methodology

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ARTICLE INFO

Article history:
Received 15 December 2014
Accepted 6 May 2015
Available online xxxx

Keywords:
Ammonia
Digestate spreading
Digestate treatment
Dried digestate
Flanders
Life Cycle Assessment

ABSTRACT

The production of biogas from energy crops, organic waste and manure has augmented considerably the amounts of digestate available in Flanders. This has pushed authorities to steadily introduce legislative changes to promote its use as a fertilising agent. There is limited arable land in Flanders, which entails that digestate has to compete with animal manure to be spread. This forces many anaerobic digestion plants to further treat digestate in such a way that it can either be exported or the nitrogen be removed. Nevertheless, the environmental impact of these treatment options is still widely unknown, as well as the influence of these impacts on the sustainability of Flemish anaerobic digestion plants in comparison to other regions where spreading of raw digestate is allowed. Despite important economic aspects that must be considered, the use of Life Cycle Assessment (LCA) is suggested in this study to identify the environmental impacts of spreading digestate directly as compared to four different treatment technologies. Results suggest relevant environmental gains when the digestate mix is treated using the examined conversion technologies prior to spreading, although important trade-offs between impact categories were observed and discussed. The promising results of digestate conversion technologies suggest that further LCA analyses should be performed to delve into, for instance, the appropriateness to shift to nutrient recovery technologies rather than digestate conversion treatments.

1. Introduction

Biogas production across Europe has experienced a rapid growth throughout many nations (Hamelin et al., 2011). The annual primary biogas production in the European Union (EU) has increased by 106% from 4899 ktoe in 2006 to 10,085 ktoe in 2011 (EUROBSERV'ER, 2012). This proliferation has become an extended practice in many farms in countries like Germany, Italy or Belgium (BMU, 2009; Fabbri et al., 2010), who have benefited from a wide range of agricultural and livestock substrates (e.g. manure, agricultural waste, energy crops, etc.) to produce energy. The main reasons for this growth are linked to a wide range of environmental benefits (Rehl and Müller, 2011). For instance, the high moisture content of fresh wastes inhibits the use of alternative conversion processes, such as thermochemical processes. In addition, from a waste treatment perspective, microbial action allows a substantial reduction of pathogens in the waste stream, as well as an important decrease in the chemical oxygen demand (COD), nitrates or organic nitrogen among other relevant parameters (Lauwers et al., 2013).

In addition, the production of biogas from livestock slurry has shown to be a technique that has a great potential for the reduction of greenhouse gas (GHG) emissions (Hamelin et al., 2011). This characteristic, added to its versatile application possibilities including direct substitution of natural gas, its use as CHP or as a liquid fuel in the transportation sector, makes biogas a key energy carrier source in the renewable energy strategy in many European countries, such as Belgium (Astrup et al., 2011). However, in a similar way to most novel energy production technologies, biogas production is not exempt of certain drawbacks, which range from operational issues to waste management (Berglund and Borjesson,

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http://dx.doi.org/10.1016/j.wasman.2015.05.007
0956-053X/

2006; Mezzullo et al., 2013). The concentration of biogas plants digesting organic waste, energy crops and/or livestock manure in regions with intensive livestock farming has led to a local surplus of one of its by-products: digestate. Consequently, these areas are prone to experience an excess of nutrients on the fields if the surplus is not managed correctly (Prapasponga et al., 2010; Rehl and Müller, 2011). This scenario has been reported in some agricultural regions in Europe, such as Belgium, the Netherlands or certain regions of Italy or Germany (Brouwer et al., 1999).

In Flanders (Belgium) high levels of nitrate in water bodies were already identified two decades ago. At the time this problem was mainly attributed to local manure overproduction associated with the intensive livestock activities on limited arable land (Lebuf et al., 2012). The introduction of Nitrogen Vulnerable Zones in the framework of the European Nitrate Directive (European Council, 1991), limiting the allowed N spreading concentrations, only partially solved the problem. The parallel development of biogas plants, not only digesting manure but also organic waste and energy plants, additionally increased the available digestate amounts (and organic N amounts). In 2007, Flemish authorities enforced compulsory treatment of manure and digestate (Flemish Manure Decree, 2007) and subsequent export of the treated products (or N mineralisation) to prevent further nitrogen concentration in the soils and aquifers. Consequently, anaerobic digestion plants in Flanders are currently focusing their efforts on providing adequate technologies to process digestate, thereby avoiding the direct spreading of raw digestate.

To reach this goal, research and implementation of digestate treatment technologies, which reduce the water content and nutrient leakage, has gained importance in Flanders in order to produce easily transportable mineral concentrates (Forbes et al., 2005; Rehl and Müller, 2011). This transition allows in some cases to forward surplus minerals to regions with nutrient shortages (Forbes et al., 2005; Holm-Nielsen et al., 2009) or, in others, the use of products with low ammonia content in Flemish fields (Decree BS13.05.2011, 2011). Given the strategic importance of these conversion technologies to reduce the environmental risks linked to the excess of nutrients in this region, the use of environmental management tools appears as an appropriate mechanism to evaluate the suitability of these technologies.

In fact, the environmental profile of these digestate treatment technologies have been previously analysed using Life Cycle Assessment – LCA (Rehl and Müller, 2011; Laurent et al., 2014), an environmental management tool that consists of the evaluation of the potential environmental impacts that are generated by a product or process during its life-cycle (ISO, 2006). Hence, the use of LCA in this specific context arises as a useful decision tool for policy makers and agricultural and farming industry stakeholders when assessing these technologies. The main aim of the study is to assist in selecting environmentally sustainable treatment technologies to deal with digestate. More specifically, this manuscript focuses on a comparative environmental assessment of a series of digestate treatment systems. These comparisons are intended to be of support in public policies in order to steer digestate management towards desirable environmental targets, as well as aid the sector with identifying key environmental indicators. For the purpose of the study, several operational full-scale treatment plants in Flanders (Belgium) have been analysed, as well as one future plant scenario, using the ammonia stripping technology. The environmental analysis was based on the technology systems but also involves a certain level of assumptions and theoretical scenarios (e.g. for transport, spreading, input characteristics, energy supply) to reach a higher level of transparency and transferability of the results to other regions in North-West Europe (NWE).

2. Materials and methods

2.1. Goal and scope definition

The main goal of this study is to analyse the environmental impacts linked to a set of digestate treatment technologies and the subsequent use of the output products as compared to the direct spreading of digestate on Flemish agricultural fields. Consequently, the results expected will allow a thorough evaluation of the environmental benefits and drawbacks linked to treating digestate, as well as a discussion on how different treatment options and digestate characteristics may influence the final outcomes of the study. Given the steady-state conditions in which the different treatment plants were examined, without delving into the consequences that these systems may have on other production systems, a retrospective (i.e. attributional) LCA approach was selected for the evaluation, in accordance with situation C2[1] in the decision support framework suggested by the ILCD guidelines (ILCD, 2010).

The main function of the system is to analyse the environmental impacts occurring during the digestate treatment process and subsequent spreading on fields of the final products in Flanders, or abroad when appropriate. The retrospective nature of the study allowed the modelling of comparable functions between the different digestate treatment scenarios. Therefore, the selected functional unit (FU) was 1 tonne of digestate product entering the plant ready to be processed. This FU selection is in accordance with previous studies analysing the environmental profile of products resulting from digestate treatment (Rehl and Müller, 2011; Golkowska et al., 2012). In addition, this choice was considered based on an environmental improvement perspective to identify the potential benefits of substituting direct spreading of digestate by the conversion processes described in Section 2.2. Nevertheless, it should be noted, as discussed in Section 4.2, that the direct comparability of the different treatment plants was constrained by the variable characteristics of the incoming digestate used in each plant.

2.2. Definition of treatment systems

A total of five different digestate treatment systems were assessed. A brief description of each of the different treatment technologies is provided below. Additionally, a baseline scenario (BAS), which depicts the direct spreading of the digestate on Flemish agricultural fields if legislation would allow this practice (see Fig. 1), was created. Given the different dry matter (DM) contents of the digestate used in the different treatment plants, the BAS were modelled as follows: (i) BAS-1, in which a mix of raw digestate and dried digestate with a DM content of 55.6% is assumed as an input, for direct comparability with the scenario described in Section 2.2.1; (ii) BAS-2, with the input indicating a DM content of 25.5% (i.e. solid fraction – SF), to allow comparability with the composting scenario (Section 2.2.2); and (iii) BAS-3 (DM = 8.9%; raw digestate), to compare with the final two scenarios (Sections 2.2.3 and 2.2.4).

2.2.1. Digestate drying and pelletizing (D&P)

The digestate drying and pelletizing (D&P) plant treats an annual amount of 99,000 t year^{-1} of digestate. This incoming product is composed of a mixture of 50% digestate SF and 50% of dried

[1] Situation C2, following the ILCD Handbook for LCA, includes studies that according to their goal definition do not include any interaction with other systems. In other words, substitution processes that may occur, or the consequences of a shock on a specific process, which is a function that is foreseen in consequential LCA, are not included in this study.

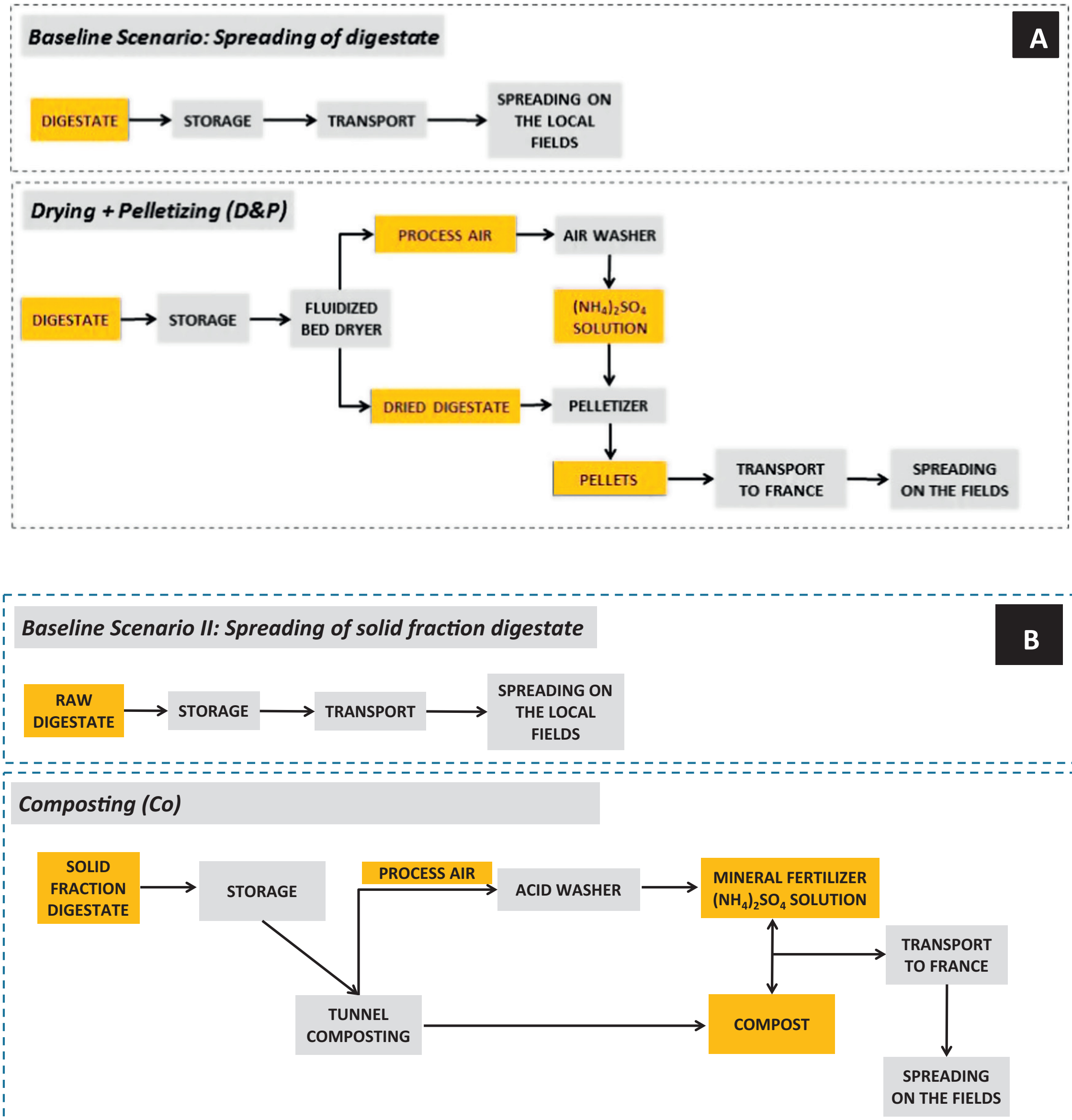

Fig. 1. Schematic representation of the conversion systems modelled. Baseline scenario I – BAS-1, and drying and pelletizing – D&P (a); baseline scenario II – BAS-2 compared with composting – Co (b); and baseline scenario III – BAS-3 compared with biological treatment, reverse osmosis and drying – RO&D, and ammonia stripping and drying – ASD (c).

digestate. The average DM content of the product entering the plant is 56% (see Table 1). After a short storage period (3–4 days), the input stream goes through a fluidized bed dryer. The evaporated water is passed through the acid washer, generating annually 35,700 t of vapour and 2000 t of a $(NH_4)_2SO_4$ solution. Thereafter, the solid output from the drying stage and the $(NH_4)_2SO_4$ solution are mixed and pelletized. Finally, the *ca.* 60,000 t year^{-1} of pellets are cooled in a ventilation system, stored and transported for use as an organic fertilizer abroad.

2.2.2. Digestate composting (Co)

The digestate composting (Co) scenario is fed with an incoming product that is composed of SF exclusively. A total of approximately 112,000 t year^{-1} of SF enters the plant and automatically undergoes a composting process in six parallel composting tunnels. The composting time is strongly dependent on the season. In the summertime, when demand for this product is high, the composting process lasts for three days, whereas in the winter this process can be extended to six weeks. The variable processing time of the compost also has an influence on the aeration time, which is one hour during the summer months and one week in winter. The aeration process is in compliance with the international standards for hygienisation (Petterson and Ashbolt, 2003). Hence, the end-product is free of pathogens. From the composting process a total of nearly 50,000 t year^{-1} of output is produced, as well as an air flow that is treated in a combined acid air washer and biofilter. The air treatment produces an $(NH_4)_2SO_4$ solution that is finally mixed with the compost and exported to France for spreading on agricultural fields. The final annual output adds up to a total of 50,000 t year^{-1}.

2.2.3. Biological treatment, reverse osmosis and drying (RO&D)

The fourth plant analysed in this study constitutes a complex system in which 55,000 t of raw digestate are treated on an annual basis in a three-step biological treatment, reverse osmosis and drying process (RO&D). After a very short storage period the raw

Please cite this article in press as: Vázquez-Rowe, I., et al. Environmental assessment of digestate treatment technologies using LCA methodology. Waste Management (2015), http://dx.doi.org/10.1016/j.wasman.2015.05.007

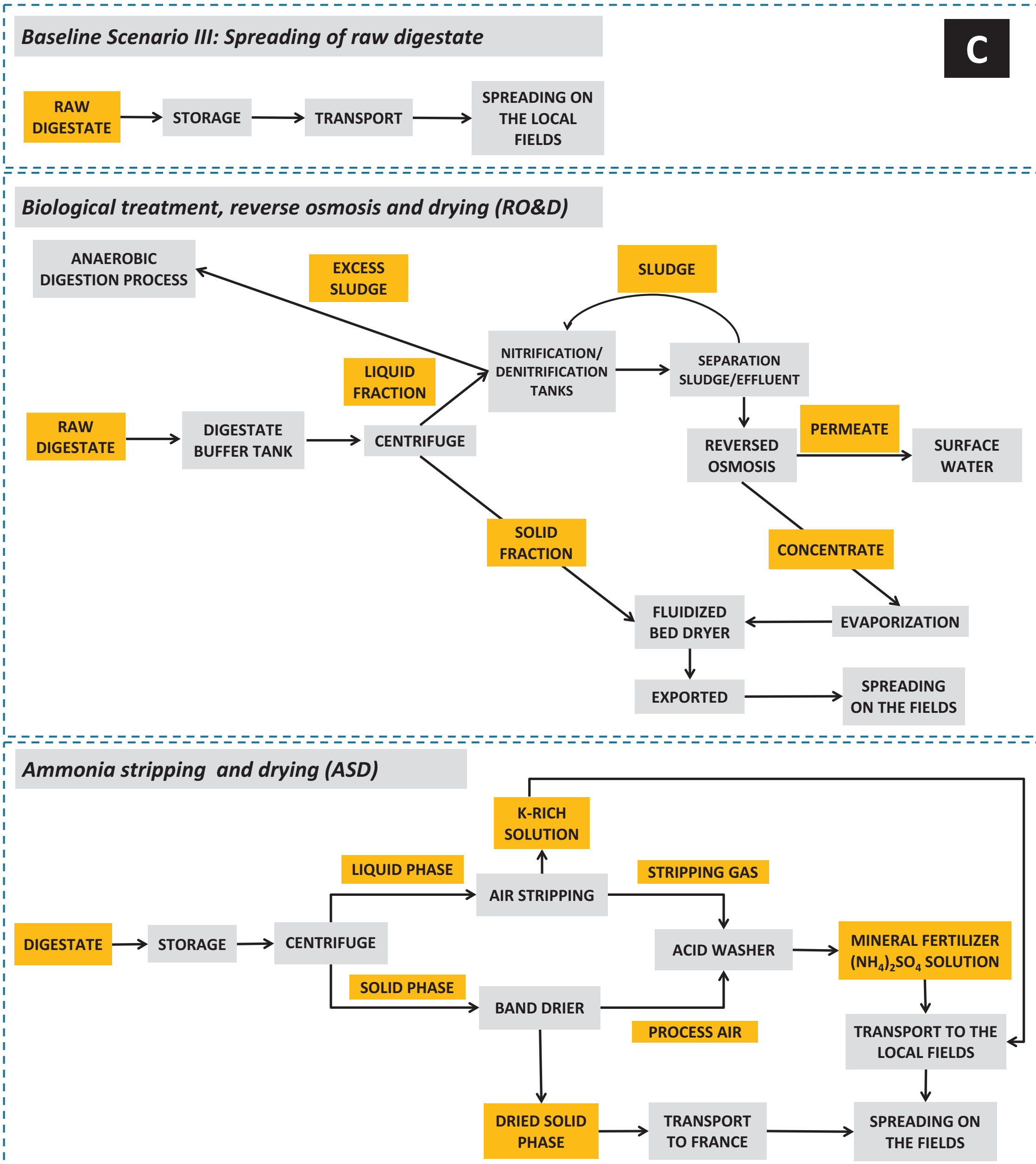

Fig. 1 (*continued*)

Table 1
Physico-chemical characterisation of the input digestate products per treatment system.

	Units	BAS-1	BAS-2	BAS-3	Drying and pelletizing	Composting	Biological treatment, reverse osmosis and drying	Ammonia stripping and drying
Dry matter (DM)	%	55.6	25.5	8.9	56.1	25.5	8.9	11.0
N_{tot}	kg/tonne	15.8	6.00	4.33	14.90	6.00	4.33	6.80
$N\text{-}NH_4^+$	kg/tonne	1.18	0.77	2.24	1.37	0.77	2.24	5.50
K_2O	kg/tonne	16.20	10.20	3.50	14.20	4.20	3.50	3.80
P_2O_5	kg/tonne	21.30	4.20	2.79	22.60	10.20	3.79	2.40

BAS = baseline scenario.

digestate enters a centrifuge separating it into 20% SF and 80% LF. The SF is transported to the fluidised bed dryer where a 90% DM final product is produced. Thereafter, the air generated in the drying phase is treated in a multistep process, while the produced ammonia water is recirculated to the biological treatment stage. In parallel, the LF is initially treated in an open air nitrification/denitrification basin. The effluent generated in this bioreactor is then sent to a reverse osmosis unit, engendering two co-products, a permeate and a concentrate. The permeate is directly discharged to surface water, whereas, the concentrate is evaporated and mixed with the dry product.

2.2.4. Ammonia stripping and drying (ASD)

Ammonia stripping and drying (ASD) is the only scenario included in this study that is currently not being commercially applied for treatment of digestate in Flanders. This new conversion method may potentially offer certain advantages, such as reduced ammonia emissions or the production of easily transportable ammonium sulphate (Bakx et al., 2009; VLM, 2012). The inclusion of ASD in this study is intended to determine whether this technology may imply environmental gains as compared to the previously described treatment technologies.

Therefore, a medium-scale treatment plant, with a similar capacity to those described above (i.e. roughly 60,000 t of raw digestate per year), was modelled. All the data to model the inputs and outputs of the plant were retrieved mainly from bibliographical data and, to a lesser extent, from expert opinions (Bakx et al., 2009). Once the stored digestate enters the treatment plant, the separation into a liquid and solid fraction is performed through centrifugation (Lootsma and Raussen, 2008). Subsequently, with the aim of increasing the pH for treatment, the LF is processed with NaOH (Bakx et al., 2009). The main product of the ammonia stripping in the LF is the stripped air, which undergoes a final acid washer step in which an $(NH_4)_2SO_4$ mineral concentrate is produced. The additional effluent produced during ammonia stripping has a high content of K, but a low N concentration. Both products (i.e. the effluent rich in K and the mineral concentrate) are subsequently applied on the fields in West Flanders (see Fig. 1) as fertilizer. The SF coming from centrifuge is dried on site, allowing the evaporation of approximately 70% of its total volume. The evaporated process air is exposed to an acid washing procedure which generates a $(NH_4)_2SO_4$ mineral concentrate that is thereafter applied locally. The remaining dried SF is transported for spreading on fields in neighbouring regions of France.

2.2.5. Storage

Short-term storage of raw digestate (5–7 days) was considered for all five treatment processes, based on the descriptions provided by the plants. However, some differences in how the digestate is stored were observed between plants. In contrast, long-term storage time of three months was assumed for raw digestate spreading, i.e. the baseline scenarios, based on common practices in The Netherlands (De Vries, personal communication, May 2012).[2] The emission factors associated with the storage of intermediate products (LF and SF) were modelled based on data retrieved from Oenema et al. (2000) and IPCC (2006), as depicted in Table 2.

2.2.6. Spreading of final outputs

Most of the final products are eventually spread in Northern regions of France (*ca.* 250 km), although some final co-products were spread on local fields in Flanders (*ca.* 100 km). One of the digestate treatment plants also had an export line to some African and Asian countries. However, for the sake of comparability, exports for this plant were also assumed to be done to Northern regions of France. All co-products in the different case studies were assumed to be used for winter wheat cultivations with secondary crops and sandy loam texture, for which a maximum spread of 170 kg/ha and 75 kg P/ha are allowed (VLM, 2012). In accordance with Flemish legislation different nitrogen uptake efficiency (NUE) rates were assumed (BS13.05.2011, 2011). Therefore, the NUE for raw digestate and the LF was 60%, whereas the NUE for the SF, dried digestate or compost was 30%. Finally, in the ASD case study a total spread of 40 kg/ha was considered for the K fertiliser produced when treating the LF, which is the average K fertiliser use for Belgium according to FERTISTAT (FAO, 2012).

The emissions to the atmosphere, to water bodies and to the soil caused by the spreading of the different products, as shown in Table 2, were calculated based on the ecoinvent® guidelines (Nemecek et al., 2007).

2.3. Data acquisition and quality

Primary data for the different treatment options were obtained from a wide range of sources. In the first place, designated plant operators completed a detailed questionnaire linked to the main input and output flows in the plants. In addition, they provided a series of data related to operational inputs, including the source and amount of energy inputs, the input of raw digestate, the use of chemicals or the final fate of output products. These questionnaires, once returned to the Flemish Coordination Centre for Manure Processing (VCM) were improved through a series of telephone interviews.

Secondly, data regarding feedstock composition for anaerobic digestion, as well as digestate characteristics were provided by Ghent University and completed with internal data from VCM. In the case of the BAS, a substrate with similar characteristics to those available for the treatment plants was modelled to allow direct comparability. Therefore, three different digestate compositions were assumed for BAS (enclosed as the BAS-I, BAS-II and BAS-III scenarios), as described in Section 2.2.

Finally, data quality requirements were met through the definition of a series of parameters that were considered common to the technologies analysed, as defined in the technical specification ISO/TS 14048/2002 (ISO, 2002). First, time-related, geographical and technology coverage were handled by fixing a single year of assessment (year 2012), a specific geographical area in West Flanders and the technology production processes described in Section 2.2. Secondly, the precision of the inventory data, understood as the degree of variability of data values, was not possible to quantify since only one set of data per technology was provided. Nevertheless, the lack of precision was counterbalanced with sensitivity analysis modelling as described in Section 2.6. The inventories of the conversion technologies analysed in this study contain accurate primary data guaranteeing completeness. In contrast, consistency was ensured by applying the same assumptions for all the conversion technologies. Finally, reproducibility was attained by providing a detailed description of the Life Cycle Inventory – LCI (see Section 2.4) and the methods used (see Section 2.5).

2.4. Life Cycle Inventory

A summarized LCI, including the main energy inputs and product outputs, is shown in Table 3, with all values allocated to the selected FU. Background processes for the different inputs and outputs considered in the systems were obtained from the ecoinvent® database (Frischknecht et al., 2007). An exemplary detailed LCI can also be consulted for the D&P scenario in Table S1 of the Supplementary Material (SM).

2.4.1. Emissions

Despite the relatively good quality of the inventory data regarding operational inputs and flow streams throughout the different

[2] Individual digestate treatment plants, as well as individual farmers consulted in the frame of the project reported variable storage times. However, it was decided to use an average storage time for all the different plants, based on the high variability during the year of operation also. Therefore, based on common practices reported by specialists in The Netherlands, a fixed value was assumed for both treatment technologies and the baseline scenarios.

Table 2
Estimated emissions for different stages of the analysed production systems reported in kg/functional unit (FU = tonne of incoming digestate).

	Storage				Treatment				Spreading of final products			
	CH_4	N_2O	NH_3	NO	CH_4	N_2O	NH_3	NO	NH_3	N_2O	NO_x	NO_3^-
BAS-1	1.33	–	4.95E−1	–	N/Ap	N/Ap	N/Ap	N/Ap	2.88	2.48E−1	4.90E−2	5.68
BAS-2	0.56	–	1.12E−1	–	N/Ap	N/Ap	N/Ap	N/Ap	4.67E−1	1.14E−1	2.40E−2	2.44
BAS-3	1.34	–	4.56E−1	–	N/Ap	N/Ap	N/Ap	N/Ap	2.59	1.97E−1	4.13E−2	2.90
D&P	1.47E−2	9.00E−2	–	–	–	–	N/Av	–	1.39E−1	9.48E−2	1.98E−2	2.28
Co	2.78E−1	–	1.45E−1	–	6.57E−1	6.29E−2	5.31E−1	–	2.91E−2	3.38E−2	7.18E−3	7.07E−1
RO&D	–	–	–	–	–	1.11E−2	1.11E−3	2.22E−3	1.23E−2	2.42E−2	5.09E−1	5.81E−1
ASD	2.90E−1	–	4.95E−1	–	N/Av	N/Av	N/Av	N/Av	2.81E−1	1.01E−2	2.30E−2	4.77E−1

BAS = baseline scenario; D&P = drying and pelletizing; Co = composting; RO&D = biological treatment, reverse osmosis and drying; ASD = ammonia stripping and drying.

Table 3
Summarized inventory data for the digestate treatment scenarios. Data reported per FU tonne of incoming digestate.

		D&P	Co	RO&D	ASD
Inputs					
Energy					
Electricity	kWh	26.13	53.23	42.04	17.40
Heat	kWh	141.9	49.85	299.52	222.22
Diesel	kg	6.38	–	–	–
Natural gas	kWh	3.79	–	–	–
Materials and chemicals					
Sulphuric acid	kg	–	3.20	0.28	2.40
NaOH	L	–	–	0.28	21.80
Powder polymer	kg	–	–	0.14	–
Outputs					
Co-products					
Dried digestate pellets[a]	kg	606.1	–	–	–
Dried digestate	kg	–	–	64.54	33.60
Compost	kg	–	800.0	–	
Mineral concentrate	kg	–	–	–	18.20
K-fertilizer	kg	–	–	–	855.0

D&P = drying and pelletizing; Co = composting; RO&D = biological treatment, reverse osmosis and drying; ASD = ammonia stripping and drying.
[a] The pellets are a mix of dried digestate and mineral concentrate – $(NH_4)_2SO_4$. The content of mineral concentrate in the final product per FU is 20.20 kg.

treatment plants, no primary data were available for the emissions to air, soil and water of the different processes, as well as the spreading of the final products on agricultural land. Therefore, storage emissions data were retrieved from available literature (De Mol and Hilhorst, 2003; IPCC, 2006; De Vries et al., 2012). Emissions to the air and water occurring in the treatment processes were obtained from a wide range of bibliographical data (Smet et al., 2003; De Vries et al., 2012). Finally, emissions to air, water and soil related to the spreading on fields of the final products were calculated based on the ecoinvent® guidelines (Nemecek et al., 2007).

2.4.2. Conversion technologies

The main inputs linked to the conversion technologies were linked to the energy sources used to power the plants. For instance, the Belgian electricity production inventory available in ecoinvent® was modified based on the current electricity mix in 2012 (Groupe Gemix, 2009). In addition, other energy sources, such as the use of heat from the CHP in all scenarios or diesel and natural gas for the D&P scenario were modelled based on current practices in Belgium regarding these carriers (Dones et al., 2007). Moreover, the use of chemicals, such as NaOH for the RO&D and ASD scenarios or sulphuric acid in the Co, RO&D and ASD scenarios were included in the inventories to account for their production processes. For these materials the average European production processes were assumed based on the ecoinvent® processes and, therefore, the average European energy and material flows were taken into account (Althaus et al., 2007).

2.4.3. Transportation and spreading

Transportation of final products to agricultural sites for spreading was modelled based on current practices. All outputs were assumed to be transported by trucks with a 32 t capacity following the EURO 5 emissions directive. Similarly, the emissions linked to fuel combustion in the transport phase was modelled based on data available in the ecoinvent® datasets (Spielmann et al., 2007). Regarding the spreading techniques described in Table 4, the ecoinvent® database was also utilized to model these processes for digestate spreading (Nemecek et al., 2007).

2.5. Life cycle impact assessment

The selected treatment options, as well as the BAS were computed following the ReCiPe assessment method (Goedkoop et al., 2009). In particular, the midpoint approach was chosen to evaluate each impact category individually and identify the key inputs and emissions responsible for environmental burdens. Furthermore, the endpoint hierarchist method was used in order to provide a weighted single score environmental value for each case study, thereby identifying their overall environmental profile. This hierarchist perspective was selected due to the fact that it takes into consideration the main policy approaches linked to time horizons (e.g. 100-year horizon for climate change – CC). Consequently, it is based on consensus, and is foreseen in many environmental standards (e.g. ISO 14040). Nevertheless, comparability with the other two perspectives (i.e. egalitarian and individualist) is provided in Section 4.1. On the one hand, egalitarian perspective was discarded from the main objectives of the study due to the extended time horizons, which make it a more conservative calculation, and the computation of impacts that are yet to be fully standardised in life cycle thinking (Goedkoop et al., 2009). On the other hand, the individualist perspective only reflects a 20 year time frame, considering that human kind fully adapts to changing environmental conditions, such as rising global temperatures. In fact, in this specific case study this choice would pose important methodological issues in terms of the inclusion of methane (CH_4) in the final results (Goedkoop et al., 2009).

Currently, the selection of ReCiPe as the preferred assessment method to compute the Life Cycle Impact Assessment (LCIA) results represents the highest level of convergence with the ILCD recommendations (ILCD, 2010, 2011). Finally, regarding land use impacts, which are monitored in ReCiPe through three different impact categories (i.e. natural land transformation, agricultural land occupation and urban land occupation), we decided to obviate the recommendation of ILCD to use the soil organic matter (SOM) impact category (ILCD, 2011). The rationale behind this decision is linked to the lack of data relating to specific spreading areas for the digestate outputs examined in this study. In fact, the use of the

Table 4
Additional scenarios to monitor the environmental impacts of fluctuating parameters.

	Baseline scenario I	Drying and pelletizing	Baseline scenario II	Composting	Baseline scenario III	Biological treatment, reverse osmosis and drying	Ammonia stripping and drying
Digestate spreading technique							
Surface incorporation (harrowing)	BAS-1-SI(H)	D&P-SI(H)	BAS-2-SI(H)	Co-SI(H)	BAS-3-SI(H)	RO&D-SI(H)	ASD-SI(H)
Surface incorporation (ploughing)	BAS-1-SI(P)	D&P-SI(P)	BAS-2-SI(P)	Co-SI(P)	BAS-3-SI(P)	RO&D-SI(P)	ASD-SI(P)
Energy carrier							
Biogas heat	BAS-1-E	D&P-E	BAS-2-E	Co-E	BAS-3-E	RO&D-E	ASD-E
Transport							
Transport (+25% distance)	BAS-1-T1	D&P-T1	BAS-2-T1	Co-T1	BAS-3-T2	RO&D-T1	ASD-T1
Transport (+50% distance)	BAS-1-T2	D&P-T2	BAS-2-T2	Co-T2	BAS-3-T2	RO&D-T2	ASD-T2

digestate outputs analysed in this case study may imply changes in the organic carbon content of the soil as compared to other fertilising agents. However, this particular aspect was considered beyond the scope of the present study.

2.6. Definition of alternative scenarios. Sensitivity analysis

In order to monitor the changes in environmental impacts expected due to variations in the production system a series of alternative scenarios were modelled. To this end, the alternative scenarios were divided into three main groups. Firstly, as seen in Table 4, two additional scenarios were modelled to evaluate the effect of different spreading techniques for digestate. While the main results that are presented in Section 3 consider the surface spreading of the digestate products evaluated in this study, two specific spreading techniques that have been shown to mitigate ammonia emissions on fields were modelled (Carozzi et al., 2013). Secondly, a set of scenarios were modelled to identify the variation in environmental impacts based on feasible changes in the energy source of the different plants. Finally, a set of changes in transport distances from the gate of the plants to the agricultural fields was considered. It should be noted that the range of scenarios that can be modelled could be extended to other dimensions, such as the characteristics of the digestate or specific changes throughout the treatment of the digestate. However, the alternative scenarios in the present study were selected based on specific issues that were considered of interest for stakeholders in the farming sector in Flanders, when the preliminary results were presented to them at a meeting coordinated by VCM.

3. Results

3.1. Environmental impacts for the baseline scenario (BAS)

The overall environmental profile of the three different BAS scenarios using ReCiPe endpoint (hierarchist perspective) showed differing results depending on the nature of the raw product. For instance, BAS-1 and BAS-3 show a much higher environmental impact (see Fig. 2) than BAS-2 (61% and 51% higher, respectively). This increased environmental impact in BAS-1 and BAS-3 is linked to the higher emissions in terms of particulate matter formation (PMF) and, to a lesser extent, of climate change (CC). This picture is related ultimately to the fact that the digestate in BAS-2 corresponds exclusively to the solid fraction of digestate and, therefore, has partially undergone previous pre-treatment procedures before arriving to the plant premises, and indicates a lower concentration of N-NH_4. Consequently, the relative contribution of the PMF category to the endpoint single score results in BAS-2 was 20%, while 39% was reported for BAS-1 and 38% for BAS-3. In contrast, despite the lower absolute values for CC as compared to BAS-1 and BAS-3, BAS-2 shows higher relative contributions to the endpoint damages of CC on human health (40%) than BAS-1 (27%) and BAS-3 (26%).

The difference in endpoint environmental impacts between BAS-1 and BAS-3 was fairly low, with impacts slightly higher to the former (11.52 Pt and 10.81 Pt, respectively, 7% higher). In addition, when the absolute and relative contributions of each impact category to the total impact are analysed, there are minimal differences between the two scenarios. Finally, it should be noted that in all three scenarios four impact categories (i.e. CC damage on human health, CC damage on ecosystems, PMF and fossil depletion – FD), accounted for at least 95% of the total environmental impact. The reason for the predominance of these categories is mainly due to the fact that the modelled production processes have a limited number of operational inputs, as quantified below, which linked most burdens to air emissions and transportation.

When the midpoint results are analysed individually for each impact category (see Tables 5–7), the relative contributions of the different operational inputs included in the LCI are fairly analogous for BAS-1 and BAS-3. Nevertheless, for the sake of simplicity, only the three categories that contributed to 95% of the endpoint single score values are discussed, while the complete set of categories is shown in Fig. 3. Firstly, field emissions linked to digestate spreading accounted for most of the environmental impacts in terms of PMF (81% for BAS-1 and 79% for BAS-3). These impacts are attributable mainly to the high content of N-NH_4 in the digestates. In fact, in BAS-2, in which the content of N-NH_4 is much lower, the relative contribution of field emissions was 41%, slightly below the emissions estimated for the storage phase (42%). Finally, it should be noted that ammonia emissions also account for important contributions to the terrestrial acidification (TA) and marine eutrophication (ME) categories, even though the weight of these impacts in the overall endpoint single score results is minimal.

CC impacts were dominated by field emissions in all three scenarios, ranging from 50% (BAS-1) to 31% (BAS-2). The main pollutant contributing to these emissions was N_2O. Nevertheless, it should be highlighted that once again the emissions on fields were lower for BAS-2, in which the digestate product used has been subject to prior pre-treatment. In fact, for BAS-2 the emissions of mainly CH_4, and to a lesser extent N_2O, occurring in the storage stage account for 30% of the total impact, while in the remaining scenarios it only accounts for 20–25% of the impact. Interestingly, fossil fuel emissions from the several transportation stages only account for roughly 15–17% of the final impacts.

The transport of the digestate products to the fields showed to have an important contribution throughout most of the remaining categories, especially for the resource and toxicity related indicators (see Fig. 3). While the relative weight of most of these categories in terms of endpoints is below 1%, this is not the case for FD. In fact, transport of digestate to the fields represented between 67% (BAS-2 and BAS-3) and 73% of the midpoint impacts for BAS-1.

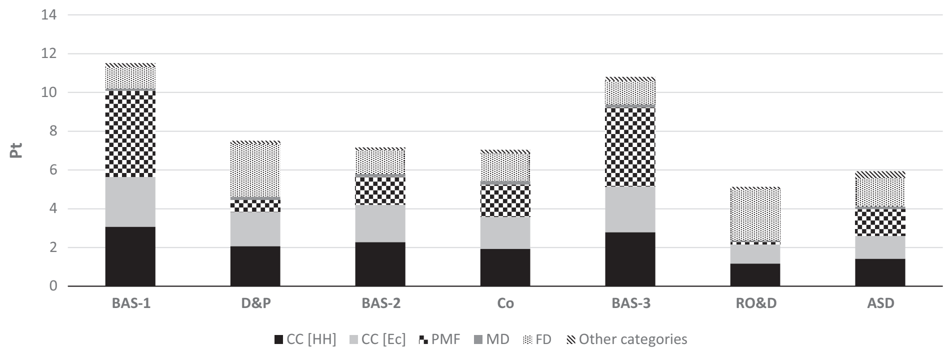

Fig. 2. Endpoint single score environmental impact results for the selected scenarios (Results reported per FU = 1 tonne of digestate entering the treatment plant). NOTE: CC [HH] = climate change–human health; CC [Ec] = climate change – ecosystems; TA = terrestrial acidification; HT = human toxicity; PMF = particulate matter formation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; MD = metal depletion; FD = fossil depletion; Other categories = ozone depletion, freshwater eutrophication, photochemical oxidant formation, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity and ionizing radiation; BAS-1 = baseline scenario (1); BAS-2 = baseline scenario (2); BAS-3 = baseline scenario (3); D&P = drying and pelletizing scenario; Co = composting scenario; RO&D = biological treatment, reverse osmosis and drying scenario; ASD = ammonia stripping and drying scenario.

Table 5
Environmental impacts for the mixed digestate treatment scenarios selected using the ReCiPe midpoint assessment method (data reported per FU, 1 tonne of incoming mixed digestate).

Impact category	Unit	BAS-1	Drying with pelletizing (D&P)
CC	kg CO_2 eq	147.77	99.98
OD	kg CFC-11	2.97E−6	6.40E−6
TA	kg SO_2 eq	8.42	6.21E−1
FE	kg P eq	2.64E−3	6.39E−3
ME	kg N eq	0.32	3.48E−2
HT	kg 1,4-DB eq	3.46	6.99
POF	kg NMVOC	0.25	4.32−1
PMF	kg PM10 eq	1.15	1.63E−1
TET	kg 1,4-DB eq	2.46E−3	4.21E−3
FET	kg 1,4-DB eq	7.01E−2	1.32E-1
MET	kg 1,4-DB eq	7.43E−2	1.42E−1
IR	kg U235 eq	2.24	20.57
ALO	m^2a	0.89	6.29E−1
ULO	m^2a	0.34	3.83E−1
NLT	m^2	6.73E−3	1.55E−2
WD	m^3	8.29E−2	2.47E−1
MD	kg Fe eq	1.78	2.17
FD	kg oil eq	6.84	17.00

CC = climate change; OD = ozone depletion; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; IR = ionizing radiation; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = metal depletion; FD = fossil depletion.

In addition, site-specific transport on fields (i.e. by truck or tractor) accounted for an important portion of the remaining impacts in all three scenarios (approximately 20%).

3.2. Environmental impacts for the drying and pelletizing (D&P) scenario

The total final endpoint value for this scenario was 7.52 Pt, 35% lower than for the BAS-1 scenario. In this treatment plant the main impact category contributing to this weighted single score (see Fig. 2) was found to be FD (37%), followed by CC – human health (28%), CC – ecosystems (5%) and PMF (8%). The higher importance of FD as compared to CC is linked to the higher reliance on fossil fuels-based energy for drying. In addition, the low relative impacts of PMF demonstrate the adequacy of this conversion technology to reduce ammonia emissions. Nevertheless, the increased distances related to the transport of final products for their use on fields in France implied considerable impacts throughout all the different categories.

From a midpoint perspective (see Fig. 4) the CC emissions from the spreading of the final products represented 32% of the total impacts, followed by the operational activities in the treatment plant (i.e. drying, screw conveyor, acid air washing and pelletizing), which represented 27% of the global warming impacts, and the emissions from the incoming product in the storage silo (26%). For FD the main contributors were the operational activities in the plant (60%), of which the most representative activity was the processing in the fluidised bed dryer (47%), followed by the transport subsystem (28%). Finally, regarding the remaining impact categories, these were mainly dominated by impacts linked to the transport subsystem, except for PMF, TA and ME, where spreading emissions linked to ammonia and nitrate were still the main source of environmental impact. Nevertheless, as can be seen in Fig. 2 and Table 5, the reduction in environmental impacts as

Table 6
Environmental impacts for the solid fraction treatment scenarios selected using the ReCiPe midpoint assessment method (data reported per FU, 1 tonne of incoming solid fraction digestate).

Impact category	Unit	BAS-2	Composting (Co)
CC	kg CO_2 eq	109.83	93.44
OD	kg CFC-11	3.22E−6	3.19E−6
TA	kg SO_2 eq	2.50	2.93
FE	kg P eq	3.06E−3	6.55E−3
ME	kg N eq	9.72E−2	1.11E−1
HT	kg 1,4-DB eq	3.94	7.61
POF	kg NMVOC	2.44E−1	2.34E−1
PMF	kg PM10 eq	3.76E−1	4.32E−1
TET	kg 1,4-DB eq	2.64E−3	2.88E−3
FET	kg 1,4-DB eq	8.19E−2	1.46E−1
MET	kg 1,4-DB eq	8.68E−2	1.50E−1
IR	kg U235 eq	2.59	18.53
ALO	m^2a	1.05	6.62E−1
ULO	m^2a	3.81E−1	4.89E−1
NLT	m^2	7.33E−3	7.99E−3
WD	m^3	9.08E−2	4.20E−1
MD	kg Fe eq	2.28	2.80
FD	kg oil eq	7.49	8.76

CC = climate change; OD = ozone depletion; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; IR = ionizing radiation; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = metal depletion; FD = fossil depletion.

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Table 7
Comparative environmental impacts for the raw digestate treatment scenarios selected using the ReCiPe midpoint assessment method (data reported per FU, 1 tonne of incoming raw digestate).

Impact category	Unit	BAS-3	Biological treatment, reverse osmosis and drying (RO&D)	Ammonia stripping and drying (ASD)
CC	kg CO_2 eq	134.39	56.58	68.41
OD	kg CFC-11	3.22E−6	6.30E−6	2.27E−6
TA	kg SO_2 eq	7.08	1.25E−1	2.51
FE	kg P eq	3.06E−3	6.84E−3	2.61E−2
ME	kg N eq	2.93E−1	4.01E−1	9.83E−2
HT	kg 1,4-DB eq	3.94	6.20	26.51
POF	kg NMVOC	2.61E−1	9.04E−2	5.11E−2
PMF	kg PM10 eq	1.06	3.58E−2	3.52E−1
TET	kg 1,4-DB eq	2.64E−3	2.12E−3	4.32E−3
FET	kg 1,4-DB eq	8.19E−2	1.12E−1	4.00E−1
MET	kg 1,4-DB eq	8.68E−2	1.26E−1	4.12E−1
IR	kg U235 eq	2.59	27.48	28.25
ALO	m^2a	1.05	3.29E−1	7.66E−1
ULO	m^2a	3.81E−1	1.06E−1	1.77E−1
NLT	m^2	7.33E−3	1.05E−2	4.91E−3
WD	m^3	9.08E−2	2.49E−1	0.52
MD	kg Fe eq	2.28	8.57E−1	2.18
FD	kg oil eq	7.49	16.49	8.98

CC = climate change; OD = ozone depletion; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; IR = ionizing radiation; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = metal depletion; FD = fossil depletion.

compared to the BAS-1 in those categories to which ammonia is contributing is considerable.

3.3. Environmental impacts for the composting (Co) scenario

The Co scenario presented an overall weighted value of 7.04 Pt (see Fig. 2), which was very similar to the overall impact received in BAS-2 (7.17 Pt). The Single score impacts were dominated mainly by four impact categories in a similar range of relative weights. The two CC categories represented 28% (human health) and 23% (ecosystems) of the environmental impacts, PMF accounted for 24% of these impacts and FD for 20.0%.

When analysing the midpoint impacts (see Table 6 and Fig. 5), 44% of the environmental impacts linked to CC were attributable to the operation of the composting plant, namely the composting itself (37%) and to a lesser extent the acid washer and biofilter stage (7%). In terms of PMF, 41% of the total impact was linked to the storage of the input digestate and another 39% was related to the composting treatment itself. Finally, for the FD category most impacts were concentrated in the transportation of the compost (55%) and, to a lesser extent, to the acid washer stage of the treatment process (20%).

3.4. Environmental impacts for the biological treatment, reverse osmosis and drying (RO&D) scenario

The final weighted endpoint values for the RO&D scenario was 5.13 Pt, 53% lower than BAS-3 (see Fig. 2), and 14% lower than the ammonia stripping scenario (i.e. ASD). The FD impact category was predominant in terms of overall contribution (52%). The remaining impacts were linked to the two climate change categories (23% for CC – human health and 19% for CC – ecosystems). PMF only represented 2.7% of the total impacts, while the remaining 3.3% was divided up between the other categories computed.

Midpoint impacts shown in Table 7 and Fig. 6 highlight the importance of the evaporation step in the overall environmental profile of this scenario, since it represents 70% of the impacts related to FD and 58% for CC. For the latter category, the nitrification–denitrification basin (14%) and the spreading of the final product on fields (13%) also had important contributions. An important observation in the PMF category is the fact that besides its low relative contribution to the endpoint impacts, the contribution of on-field emissions only represented 16%, whereas different treatment steps in the digestate plant had more important contributions, such as the evaporation stage (28%) or the nitrification–denitrification basin (20%). Finally, it should be mentioned that for ME the main contributor to environmental impacts was the effluent (65%) and on-field spreading was the main impact in terms of agricultural land occupation.

3.5. Environmental impacts for the ammonia stripping and drying (ASD) scenario

The overall endpoint environmental impact for this scenario was 5.94 Pt, 45% lower than for BAS-3. A vast proportion of this final value was evenly distributed between the following endpoint impact categories: CC – human health (24%), FD (24%), PMF (23%) and CC – ecosystems (20%). This distribution, however, differs substantially from the one observed for RO&D, since the biological treatment in the latter is capable of reducing PMF impacts to extremely low levels. In contrast, despite the lower efficacy of ASD (as compared to RO&D) in terms of reducing the final environmental impacts due to ammonia, its dependence on fossil fuels is much lower, which allows obtaining competitive reductions of ammonia emissions, while controlling the expenditure of resources and derived GHG emissions.

The midpoint values for the highest contributing categories in this scenario suggest that the stripping of the liquid fraction is the main process responsible for environmental impact, since it represents 37% of the impacts for CC and 76% for FD. This is mainly due to the energy requirements needed to power a plant of these characteristics. In addition, emissions in the storage phase of raw digestate would also account for an outstanding environmental contribution: 45% for PMF and 11% for CC. Finally, field activities of spreading one of the final products (i.e. mineral fertilizer in the form of ammonium sulphate) would add up to 17% of the environmental impacts for the CC category and 13% for PMF. These impacts were mainly linked to the air emissions engendered during the surface spreading of the mineral fertilizer (i.e. dinitrogen monoxide in the case of CC and ammonia in PMF).

Finally, as pointed out for the remaining scenarios and treatment processes, the application of this technology implies important reductions in terms of ammonia emissions as compared to the baseline scenario (BAS-3) and, therefore, substantial benefits in the categories influenced by these emissions at the sacrifice of higher impacts in terms of energy use and climate change emissions.

3.6. Environmental impacts for the sensitivity analysis scenarios

The selection of the techniques for spreading digestate on the fields has shown to have important impacts on the mitigation of ammonia emissions (Carozzi et al., 2013). Fig. 7 shows the influence of different spreading techniques on the final results. Environmental reductions for BAS when spreading digestate using the surface incorporation technique were strongly dependent on the application of this technique during ploughing or harrowing. In fact, environmental gains during harrowing ranged from 12% (BAS-1) to 3% (BAS-2) for the untreated products, whereas these gains augmented to 27% (BAS-1) and 7% (BAS-2) if the digestate

Fig. 3. Detailed midpoint characterisation relative values per operational activity for the baseline scenarios. NOTE: CC = climate change; OD = ozone depletion; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; IR = ionizing radiation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = mineral depletion; FD = fossil depletion; BAS-1 = baseline scenario 1 (mix of raw digestate and dried digestate); BAS-2 = baseline scenario 2 (solid fraction digestate); BAS-3 = baseline scenario 3 (raw digestate).

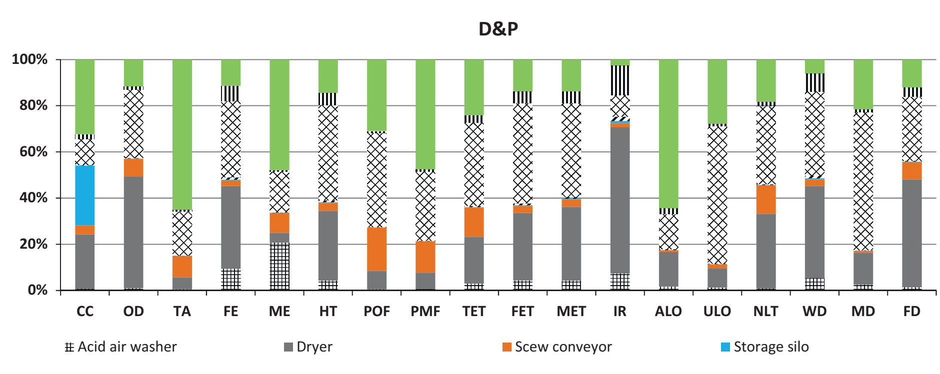

Fig. 4. Detailed midpoint characterisation relative values per operational activity for the treatment of mixed digestate plants. NOTE: CC = climate change; OD = ozone depletion; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; IR = ionizing radiation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = mineral depletion; FD = fossil depletion; D&W = drying with acid washer; D&P = drying and pelletizing.

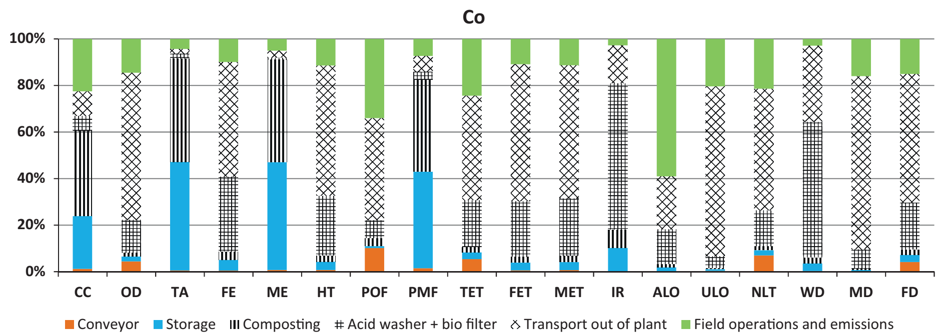

Fig. 5. Detailed midpoint characterisation relative values per operational activity for the composting plant (treatment of solid fraction digestate). NOTE: CC = climate change; OD = ozone depletion; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; IR = ionizing radiation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = mineral depletion; FD = fossil depletion; Co = composting.

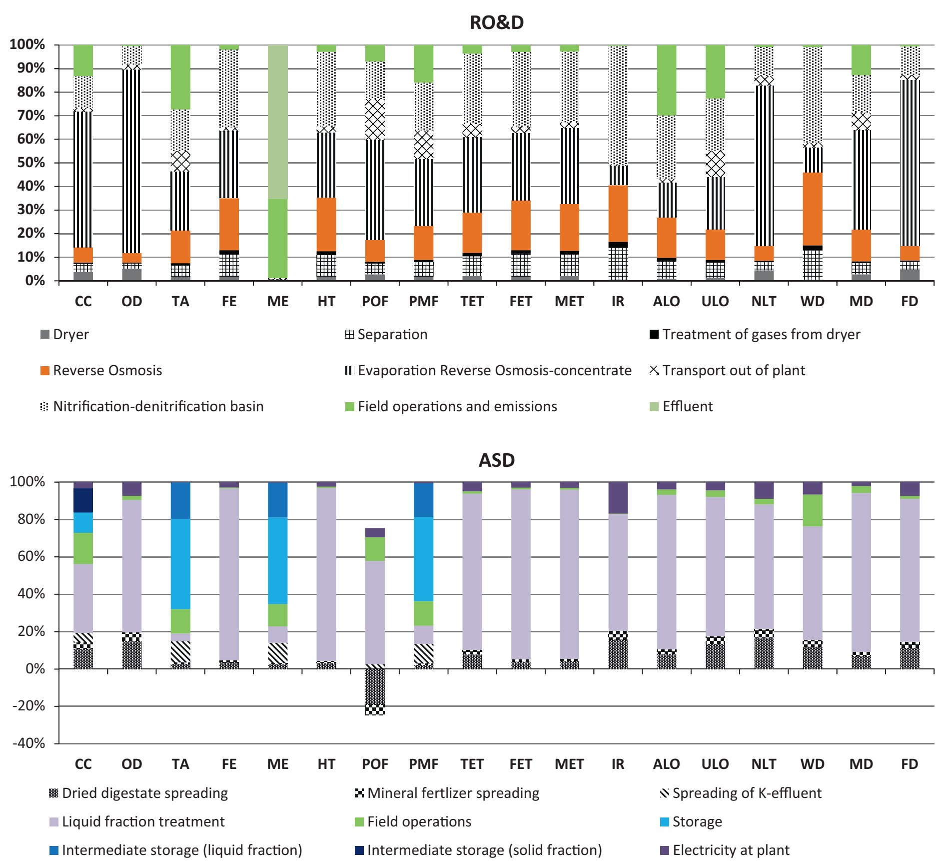

Fig. 6. Detailed midpoint characterisation relative values per operational activity for the treatment of raw digestate. NOTE: CC = climate change; OD = ozone depletion; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; TET = terrestrial eco-toxicity; FET = freshwater eco-toxicity; MET = marine eco-toxicity; IR = ionizing radiation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; WD = water depletion; MD = mineral depletion; FD = fossil depletion; RO&D = biological treatment, reverse osmosis and drying; ASD = ammonia stripping and drying.

is spread while ploughing. However, for the spreading of all treated products the diversification of spreading techniques was found to have marginal influence on the total environmental impact.

The alternative scenarios for transport of the final co-products showed to have a minimal impact on the final environmental profile in the different treatment processes evaluated. More specifically, increases of up to 4% in the final impact were observed for the RO&D case study, while in the remaining treatment technologies these were below 2%.

Finally, the change in the energy source to the use of biogas heat from the anaerobic digester implied environmental improvements that ranged from 52% for the RO&D case study to less than 10% for D&P. In fact, except for RO&D all case studies showed low improvements by shifting their energy profile to biogas heat from the digester.

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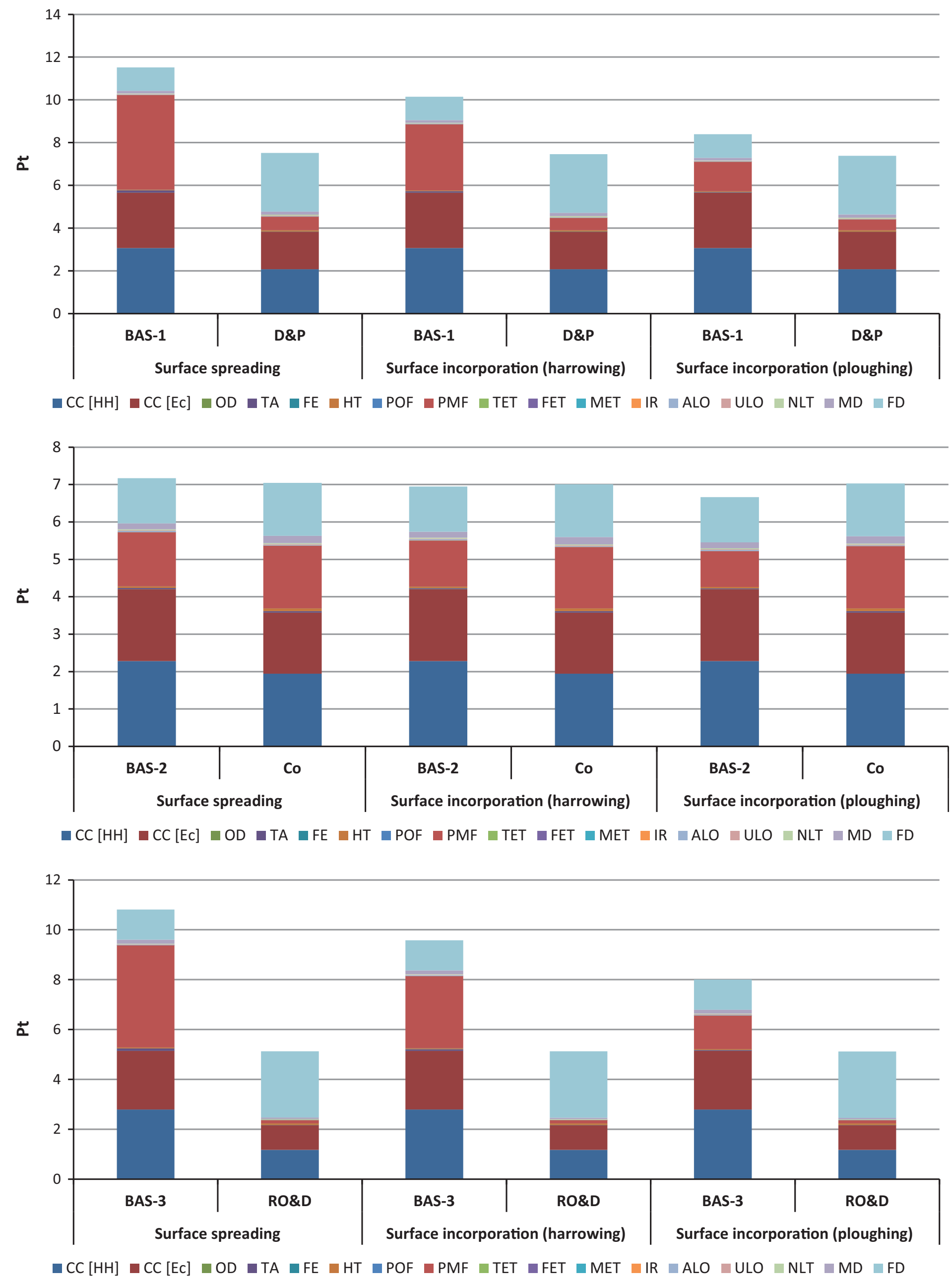

Fig. 7. Endpoint single score environmental impact results for the selected scenarios considering different spreading techniques (Results reported per FU = 1 tonne of digestate entering the treatment plant). NOTE: CC [HH] = climate change – human health; CC [Ec] = climate change – ecosystems; TA = terrestrial acidification; HT = human toxicity; PMF = particulate matter formation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; MD = metal depletion; FD = fossil depletion; Other categories = ozone depletion, freshwater eutrophication, photochemical oxidant formation, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity and ionizing radiation; BAS-1 = baseline scenario (1); BAS-2 = baseline scenario (2); BAS-3 = baseline scenario (3); D&P = drying and pelletizing scenario; Co = composting scenario; RO&D = biological treatment, reverse osmosis and drying scenario.

4. Discussion

4.1. The importance of holistic environmental perspectives in digestate analysis

Digestate treatment across different systems shows an interesting performance in terms of environmental impacts as compared to the vast majority of production systems that have been examined in LCA studies to date. Despite the higher impact in terms of resource depletion and emissions linked to climate change, the overall environmental burdens related to digestate treatment in all the systems except for composting were substantially lower than for direct spreading of the input digestate, independently of whether the raw digestate or the mixed input (see. input description in Chapter 2.2) were considered. While it may be argued that the endpoint perspective adopted (i.e. hierarchist) may skew the results in a certain direction, when all three approaches are confronted overall impacts are still considerably lower when digestate is treated. For instance, Fig. 8 shows the effect of using different endpoint normalisation weighting perspectives on the final results for the three baseline scenarios. Further discussion on weighting scenarios is included in Section 4.2.

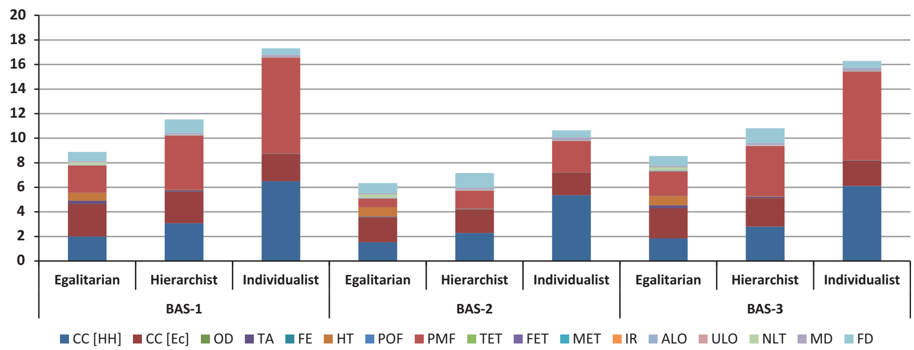

Fig. 8. Endpoint single score environmental impact results for the selected baseline scenarios: comparison across endpoint single score perspectives (Results reported per FU = 1 tonne of digestate entering the treatment plant). NOTE: CC [HH] = climate change – human health; CC [Ec] = climate change – ecosystems; TA = terrestrial acidification; HT = human toxicity; PMF = particulate matter formation; ALO = agricultural land occupation; ULO = urban land occupation; NLT = natural land transformation; MD = metal depletion; FD = fossil depletion; Other categories = ozone depletion, freshwater eutrophication, photochemical oxidant formation, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity and ionizing radiation; BAS-1 = baseline scenario (1); BAS-2 = baseline scenario (2); BAS-3 = baseline scenario (3).

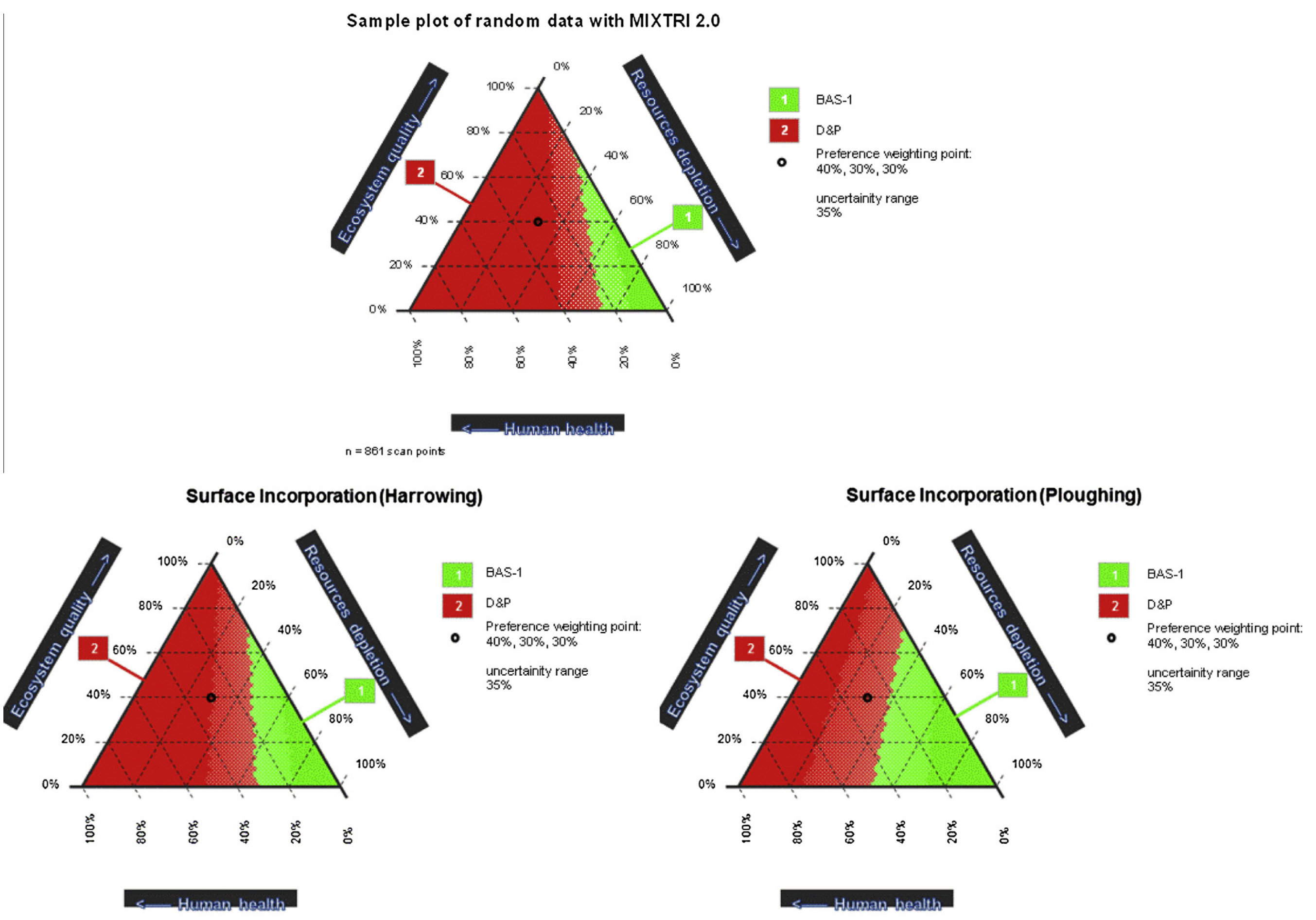

Fig. 9. Weighting triangle matrix for different spreading techniques in Scenario A. BAS-1 = baseline scenario (1); D&P = drying and pelletizing scenario.

Nevertheless, the results obtained demonstrate that commonly used single issue perspectives, such as carbon footprint (CFP), may provide a highly misleading view of which scenarios show an improved environmental profile (Laurent et al., 2012). In fact, while CFP, which is the single issue equivalent to the CC impact category in the ReCiPe method, has shown to be a reasonable indicator category for many processes, as described by Weidema et al. (2008), it also shows a myopic vision in systems, such as digestate, whose environmental rationale is linked to the minimisation of non-carbon oriented issues (e.g. ammonia reduction).

4.2. Sensitivity analysis and uncertainties

The three different incoming digestate products analysed in this research paper were evaluated considering digestate conversion technologies currently implemented in Flanders. These treatment processes were compared to the direct spreading of digestate products on agricultural fields, in order to understand the trade-offs in terms of environmental impact between the different possibilities permitted by Flemish legislation, even if digestate (raw or processed) can only sparingly be delivered to agricultural land

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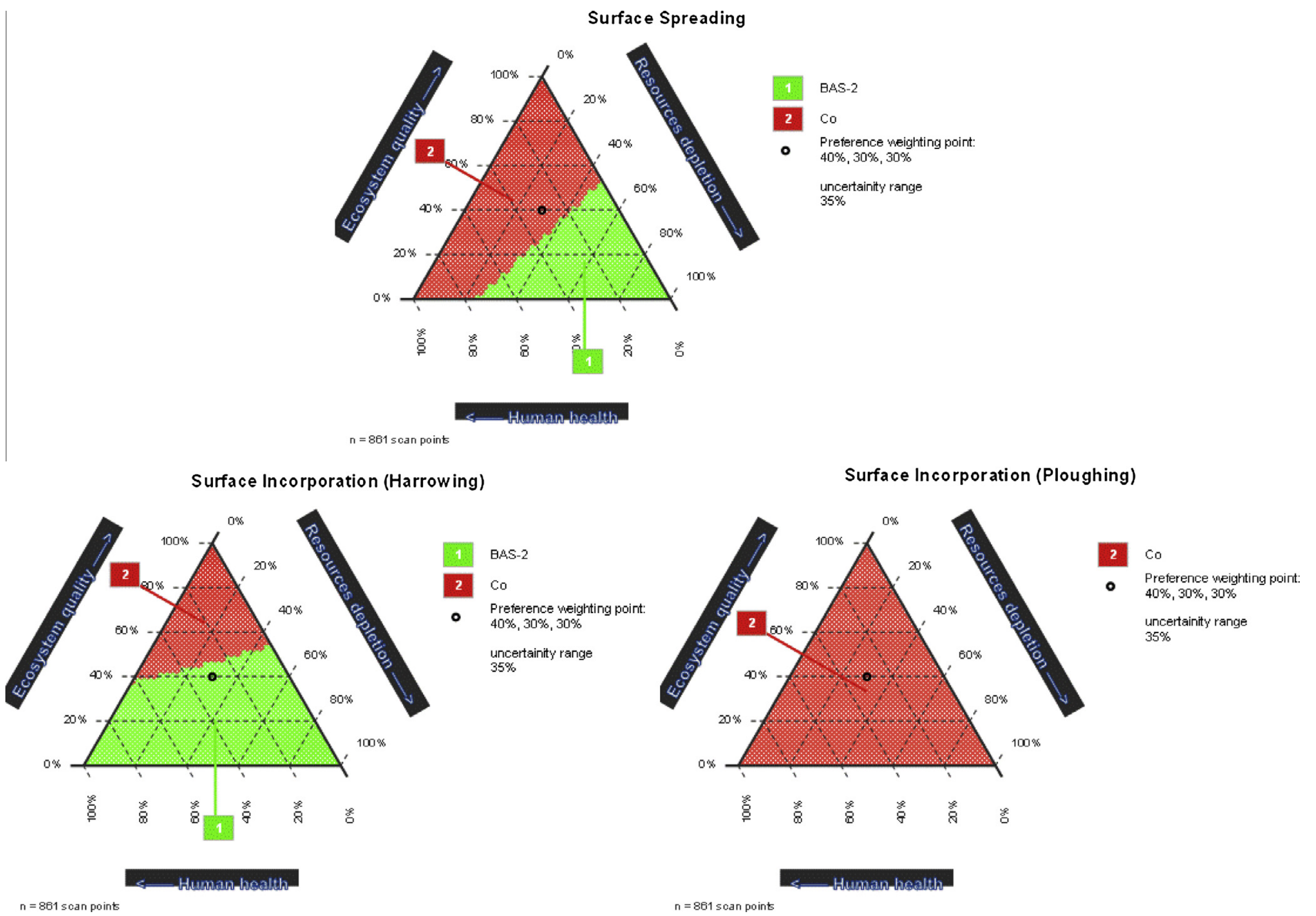

Fig. 10. Weighting triangle matrix for different spreading techniques in Scenario B. BAS-2 = baseline scenario (2); Co = composting scenario.

(Vaneeckhaute et al., 2013a). Nevertheless, given the complexity of the analysed treatment, this direct comparability between crude and treated digestate remained insufficient to deliver a thorough analysis to stakeholders regarding the convenience of the different scenarios. Therefore, as aforementioned in Section 2.4 and illustrated in Table 4, a series of alternative scenarios were considered.

However, it should be noted that variable digestate characteristics, which are prone to imply important changes in environmental impact, especially in terms of certain digestate parameters, such as dry matter content or N_{tot}, were not considered in this study due to the way in which the inventory data for the different treatment plants were reported. Hence, given the annual data that were collected, it was not feasible to model disaggregated mass loadings for the digestate for highly specific time frames due to the lack of granularity in the annual inventories (Levasseur et al., 2010). In other words, it would not be feasible to consider that a period with a low N content (or any other change in digestate characteristics) in the incoming digestate should have the same use of chemicals and other operational inputs as a period in which the N content is above average. Consequently, the use of a steady-state model depicted throughout the assessment, in spite of its important limitations regarding accuracy, was the only feasible temporal perspective that could be developed (Reap et al., 2008; Levasseur et al., 2010).

For an in-depth analysis of which scenarios are more environmentally-friendly based on the selection of impact and damage categories, the mixing triangle approach, as suggested by Hofstetter et al. (1999), was selected using the MIXTRI 2.0 model developed by Doka (2011). The advantages of applying this model are linked to the consideration of all weighting possibilities between damage categories in the form of a graphical representation. In other words, the aggregated environmental effects that are measured in the single score endpoint indicator are based on a relative (and subjective) weighting of the three types of damage categories. The mixing triangle avoids this subjective weighting by representing all possible weighting schemes (Doka, 2011). This leads to comparing areas of preference between alternative scenarios. In other words, the different subareas linked to treatment scenarios differentiated in the triangle represent the zones in which the given technology performs environmentally better than the others. Furthermore, an uncertainty range was fixed in order to identify areas of uncertainty in which the dominance of a particular alternative is not significant.[3] More specifically, the uncertainty for the scenarios assessed was fixed at 35% due to the numerous assumptions that were taken into account as described in Section 2.

In Scenario A three spreading techniques showed an overwhelming dominance of the D&P digestate treatment process as being the most beneficial (see Fig. 9). Finally, direct spreading of digestate showed dominance at very high weighting for the resource damage category. The default weighting point is located in all cases in the uncertainty zone of D&P dominance. Based on these results, it appears that the D&P treatment plant examined presents the most favourable values for balanced weighting scenarios, although high uncertainties remain.

[3] It should be noted that the mixing tringles are based on an initial endpoint weighting of 30% of the environmental impacts for human health (HH), 40% for ecosystems (Ec) and 30% for resources (Re). This particular point is marked in Figs. 9–12.

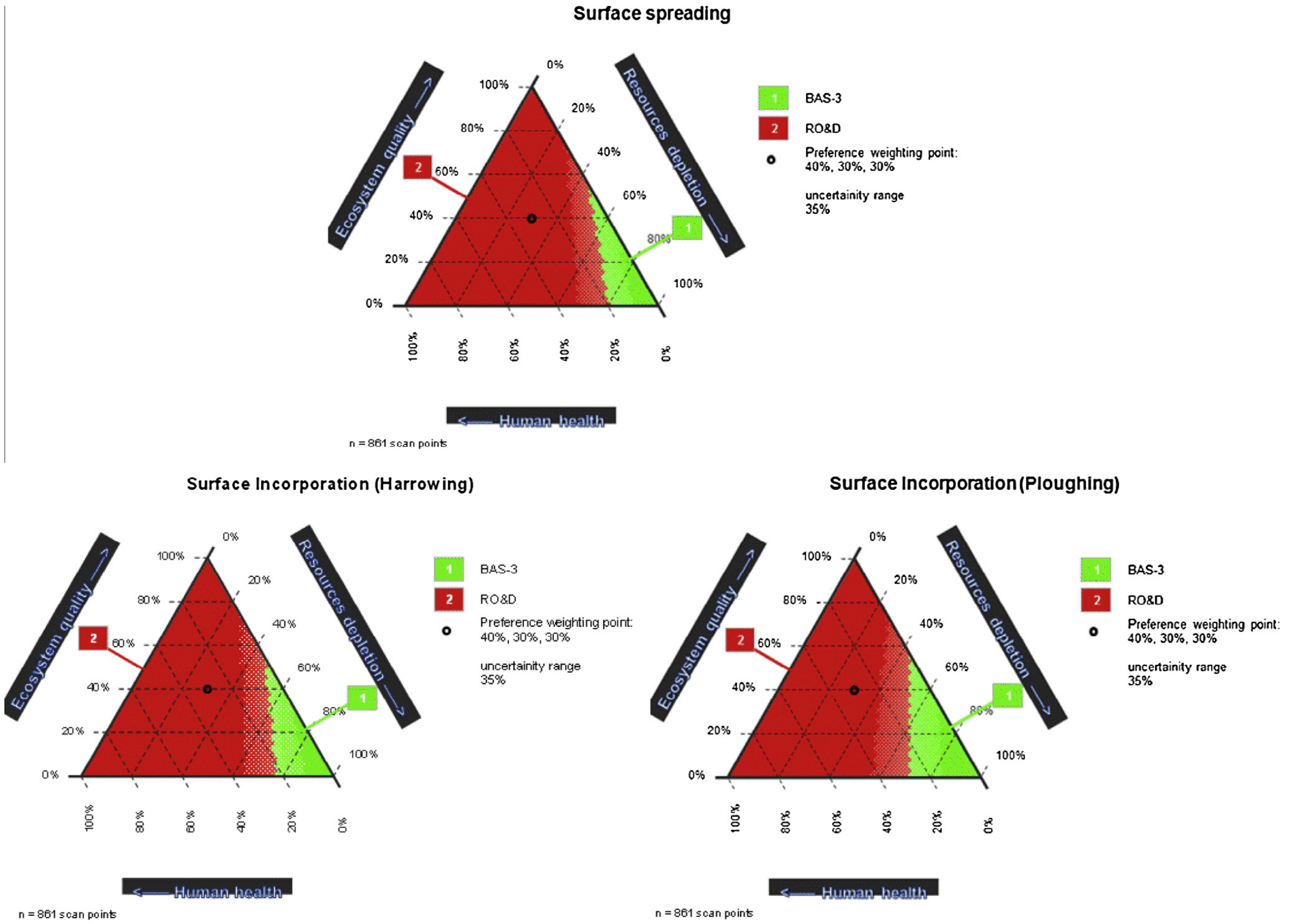

Fig. 11. Weighting triangle matrix for different spreading techniques in Scenario C. BAS-3 = baseline scenario (3); RO&D = biological treatment, reverse osmosis and drying scenario.

For Scenario B the results that are presented were found to be more straightforward (see Fig. 10). Consequently, as long as some type of ammonia emission reduction technique is applied on fields, the spreading of dried digestate appears to be more favourable than the composting of this product. For instance, very high weightings for ecosystem quality, in most cases above 60–65% would be needed to defend the use of composting treatment on this already pre-treated product.

Scenario C presents the clearest results for analysis, since the default weighting point is situated within the RO&D treatment technology in all three spreading techniques.[4] In fact, the dominance goes beyond the dotted area of uncertainty, as shown in Fig. 11, indicating that this dominance is statistically significant. Direct spreading of raw digestate would only show improved environmental sustainability as compared to its treatment if very high resources and very low human health damage impacts are computed, demonstrating that the direct spreading of raw digestate is not recommended in Flanders for average digestate characterisation values.

Based on the joint interpretation of the weighting triangles for all three scenarios, the results suggest that the treatment of raw digestate shows an overall environmental benefit when compared to the direct spreading of the digestate provided that a balanced weighting of the damage categories is considered. This statement is valid across all digestate spreading techniques. However, in Scenarios A and B the interpretation is not as clear, since high levels of pre-treatment (not considered in this study) imply that further digestate treatment is not useful in most spreading technique scenarios. Therefore, composting of the solid fraction does not provide environmental gains with respect to direct spreading of this product, unless undesired spreading techniques (i.e. surface spreading are applied). Mixed raw and dried digestate examined in Scenario A leads to the most complex interpretation, given the presence of several best-performing techniques in the triangle matrix. In any case, it appears as if digestate treatment of this product is desirable regardless of the spreading technique for the final products.

When energy scenarios are compared (see Fig. 12), the shift to using biogas heat as the exclusive energy source for the treatment processes shows important improvements in the benefits of treating digestate rather than direct spreading. In fact, the use of the D&P technology (i.e. D&P-E) in Scenario A and RO&D in Scenario C present overwhelming dominance over alternative practices. Moreover, despite high uncertainties, the composting of the solid fraction in Scenario B would also entail an environmental improvement as compared to direct spreading.

In the same way as most LCA studies, there are important sources of uncertainties linked to different stages of the LCA that should be analysed with care. Some of these have been analysed already through the modelling of alternative scenarios, as discussed in the previous section, as well as through providing

[4] The ASD scenario was not included in the mixing triangle due to the fact that it is not based on primary data, but only on bibliographical data.

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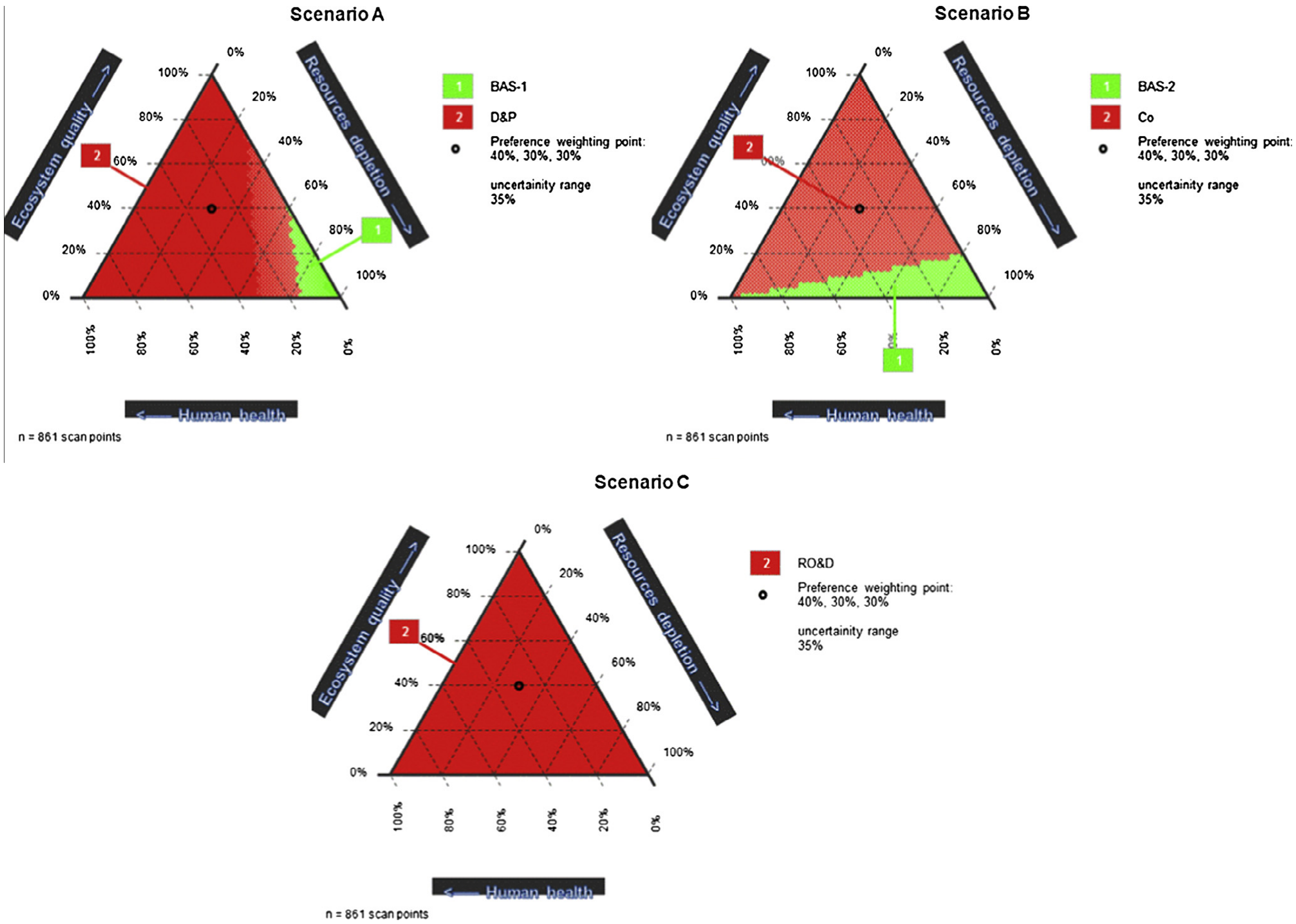

Fig. 12. Weighting triangle matrix for the different scenarios considering biogas heat as the sole energy carrier in the treatment of digestate. BAS-1 = baseline scenario (1); BAS-2 = baseline scenario (2); BAS-3 = baseline scenario (3); D&P = drying and pelletizing scenario; Co = composting scenario; RO&D = biological treatment, reverse osmosis and drying scenario.

discussion on the different endpoint single score perspective that can be modelled (see Section 4.1). Nevertheless, uncertainty should not only be viewed in terms of adding variable scenarios to the system analysed, but also through understanding underlying sources of uncertainty linked to the quality of the LCI data, as well as the inherent uncertainties associated with the mathematical computation behind the impact categories selected. For instance, CC computation has shown to have substantial limitations related to the treatment of indirect effects and feedbacks which can lead to uncertainties as high as ±40% (IPCC, 2013). In fact, ReCiPe does not provide any uncertainty factors for this specific impact category (ILCD, 2011).

The ILCD recommendations for the selection of impact categories provide a series of guidance that allow getting a general idea of the main weaknesses and strengths of these categories in terms of uncertainty (ILCD, 2011). Hence, certain impact categories within the ReCiPe LCIA method have been rated positively, given the consideration of several uncertainty factors, such as toxicity categories, PMF, OD or eutrophication and resource categories, whereas others showed certain limitations that should be considered in the interpretation of the results, such as acidification (due to lack of uncertainties provided) or land use (due to the lack of an environmental characterisation model).

5. Conclusions and future outlook

The main aim of this study was to analyse the environmental feasibility and risks of using different conversion technologies to treat digestate products in Flanders. The results presented in this study prove the convenience of applying conversion technologies prior to digestate spreading on fields for fertilisation rather than directly spreading the raw product, provided that the incoming digestate product has suffered limited pre-treatment processes. Despite a substantial increase in impacts associated with global warming and energy and mineral use, most analysed scenarios imply considerable environmental gains as compared to direct spreading, due to the important reductions in air emissions, namely ammonia. Consequently, this study proves the suitability of assessing a wide range of impact categories, as a way to understand the trade-offs that may occur between different substitutable technologies. Moreover, the increase in energy intensity when introducing conversion technologies appears to be, despite the discussed uncertainties, marginal as compared to the environmental benefits in other environmental dimensions. Nevertheless, energy environmental impacts may be reduced substantially through changes in the energy source. In a similar way, the use of spreading techniques aimed at minimising the impact of ammonia emissions prove to have an important impact on the final interpretation of the results.

The use of the Mixing Triangle approach was implemented as a way to deliver a clear message to the stakeholders when it comes to decision making on which are the most appropriate technologies to select based on the average characteristics of digestate in Flanders. In addition, this system allows constructing cooperate decisions through the prioritisation of the different impact and damage categories that are considered in the assessment.

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Nonetheless, it should be noted that further research should be undergone to attain new insights regarding the appropriateness of further developing these techniques. At first, the development of dynamic models in order to understand the time dependent behaviour of the processes included in the life cycle structure (Levasseur et al., 2010, 2012) should be addressed. For instance, the amounts of raw materials used within the treatment processes are linked to a crucial temporal aspect of the functioning of the system: the characteristics of the incoming digestate. Moreover, associated emissions during the storage, processing and spreading of the products will also suffer important variations that are worth assessing through time.

Secondly, a further study in which the shift from the current state-of-the-art regarding digestate treatment technology in Flanders to nutrient recovery is recommended, in order to identify the benefits of exploiting intermediate products of digestate treatment rather than the final mixed products assessed in the current publication. It is expected that such an analysis, which is justified by the increasing awareness concerning the depletion of phosphorus and potassium in the mining sector, should allow assessing from an environmental perspective the advantages and drawbacks of closing the cycle for the most relevant agricultural nutrients (Vaneeckhaute et al., 2013b).

Finally, a third issue that should be considered in future studies is an analysis linked to the environmental consequences (consequential LCA) of an increase in available fertilising agents obtained through digestate treatment, which can potentially provide a substitute for chemical mineral fertilisers, with the aim of understanding how physical flows and their associated environmental impacts can vary in response to changes in market-driven implications beyond the foreground system under analysis (Golkowska et al., 2014; Vázquez-Rowe et al., 2014).

Acknowledgements

This research project was financially supported by the European Commission within the frame of the INTERREG IVb NWE Project "Accelarating Renewable Energies through valorization of Biogenic Organic Raw Material (ARBOR)". The authors wish to express their gratitude to J. de Vries and P. Hoekma (Wageningen University), B. Ryckaert (Inagro) and T. Rehl and J. Müller (University of Hohenheim) for valuable scientific exchange. Dr. Ian Vázquez-Rowe would like to thank the Galician Government for financial support (I2C postdoctoral student grants program). Authors with affiliation to the University of Santiago de Compostela (Spain) belong to the Galician Competitive Research Group GRC 2013-032.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2015.05.007.

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