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Assessing the environmental performance for more local and more circular biowaste management options at city-region level

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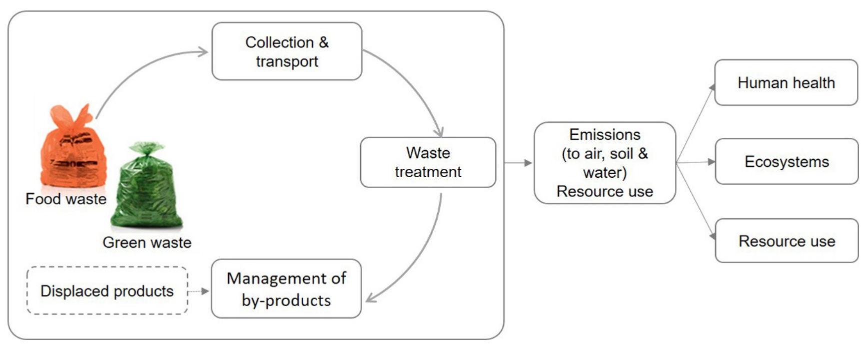
HIGHLIGHTS

- Unclear environmental performance of ongoing transitions to circular economy (CE)
- Novel combination of local data, databases and models applied to a case study.
- Life cycle assessment with damage assessment applied to an urban biowaste system.
- Combined green and food waste management in AD showed benefits compared to baseline.
- Local CE biowaste systems show environmental benefits under certain conditions.

GRAPHICAL ABSTRACT



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ABSTRACT

Biomass, biobased materials and food waste are considered priority areas for Europe's transition towards a circular economy (CE). Waste management is a central activity for this transition and offers multiple CE implementation options which should be evaluated from environmental perspective.

The purpose of this work was to analyze the environmental consequences when redirecting biowaste flows from conventional to more circular management systems and to identify the CE option with the best environmental performance. We were particularly interested in studying the combined management of green and food waste, analyzing the challenges when introducing separate collection and different treatment processes, and evaluating the substitution potential for by-products. To determine environmental impacts, we performed a life cycle assessment (LCA) based on local data. Following the purpose analyzing a change in the system, we applied a consequential LCA and compared impacts from processes that are replaced with impacts from alternative management options such as co-composting, anaerobic digestion (AD) and decentralized composting.

The LCA results show clear advantages for impacts on ecosystems and resource use for the local AD system with separate combined collection. The decentralized system shows reductions in resource use, whereas the industrial co-composting system has higher or similar impacts than the baseline scenario. We conclude that local systems with combined food and green waste management can show benefits if process emissions are properly managed and if by-products are used in applications with high substitution potentials. However, a change towards a CE does not necessarily result in environmental benefits.

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Our research highlights the complexity of biowaste systems and proposes a novel combination of local data, databases and models to handle this issue. With this research we are further contributing to the understanding of the combined management of food and green waste, which is a relevant, but so far under-researched, management option for cities.

1. Introduction

Within the Circular Economy Action Plan (EC, 2015), biomass, biobased materials and food waste are considered priority areas for Europe's transition towards a circular economy (CE). To implement a CE, a wide range of measures is suggested, from material management to waste prevention. However, the central activity to achieve circularity for bioresources is waste management because this activity determines whether the cycles of organic matter can be closed and whether nutrients and energy can be recovered.

Cities play an important role in a CE because, due to the high population densities, they are the main producers of solid waste, which contains between 20 and 40% of organic content in Europe (Di Maria et al., 2016). Currently, the collection rates and recovery schemes vary greatly between cities (BiPRO/CRI, 2015), and the potential for the recovery of nutrients and energy has not been fully exploited yet. To improve local performances, many cities are turning towards CE concepts. A current review of CE initiatives around the globe identified 83 cities that promote CE, but with different targets and interests (Petit-Boix and Leipold, 2018). Brussels, for example, has since 2016 a Regional Program for a Circular Economy (PREC, 2016) which includes transversal, sectorial, territorial and governance measures to support the city's CE transition.

To assess the circularity performance of a city or a region, circularity indicators (CI) have been proposed and discussed in the literature (see for example Corona et al. (2019) and Saidani et al. (2019)). The material circularity indicator (MCI) proposed by the Ellen MacArthur foundation (EMF, 2015) assesses the circularity performance of a product or an organization. For waste management systems, circularity indicators such as recycling/reuse rates are often used to measure the degree of circularity of an economy. For example, in the CE monitoring framework (European Commission, 2018) the overall recycling rates and the recycling rates for specific waste streams (such as municipal biowaste) are proposed to measure the circularity performance of the waste management sector. Haupt et al. (2017) distinguish closed- and open-loop collection and recycling rate to assess circularity. Since these material flow indicators are hardly applicable to organic waste, more specific ones have been developed for organic waste management systems. Cobo et al. (2018) propose a nutrient circularity indicator that 'accounts for the extended service of the components recovered from waste'. But even with such an approach that goes as far as to the nutrient uptake in crops, CI remain at the level of flow analysis and do not evaluate environmental impacts. This is why supplementary indicators such as energy use, CO_2 emissions, water, toxicity and resource scarcity have been proposed as complementary risk and impact indicators, for example in EMF (2015). Also other authors suggest going beyond circularity metrics (Geyer et al., 2016; Haupt et al., 2017).

Life cycle assessment (LCA) is a method to quantitatively assess environmental impacts of goods and services from 'cradle to grave'. In waste management studies, such as this one, the typical system boundary is from 'bin to grave' (Hauschild and Barlaz, 2010). An LCA expands the scope of analysis beyond the waste management system by including (i) the environmental impacts caused by surrounding systems and (ii) the potential environmental benefits created through by-products. Such environmental benefits occur for a variety of waste management processes, for example, when energy, materials or nutrients are recovered (Ekvall et al., 2007). Through its holistic perspective, LCA is particularly suited to support decision-making in waste management (Hellweg and Canals, 2014) and it is also required in the waste framework directive (WFD) to justify possible deviations from the waste hierarchy (EU Directive 2008/98/EC). LCA studies use data from the anthroposphere (transportation, land use, process in- and outputs, waste data, etc.), emissions to air (atmosphere), soil (lithosphere) and water (hydrosphere) and are based on for example atmospheric models to assess global warming potential (GWP), or species abundance models (biosphere). Thus, they are multi-impact studies and cover several spheres of the total environment.

LCA has been extensively used to study solid waste management (Laurent et al., 2014) and, more recently, to study CE options. Some LCAs demonstrated that the most circular solution is not necessarily the most environmentally preferable option (Haupt and Zschokke, 2017). For biowaste management systems, research is carried out to identify high value pathways for a circular biowaste valorization. For example, food waste is studied for the production of specialty chemicals, biofuel-precursors and biodegradable polymers. For heterogeneous organic waste, 1st generation technologies such as composting and anaerobic digestion (AD) are still the most feasible pathways (Lin et al., 2013). Thus, most of the current LCA studies investigate different composting and AD concepts compared to landfill or incineration of biowaste.

In addition to waste treatment/valorization, the waste collection scheme under evaluations plays a vital role for environmental impacts as waste collection trucks consume fuel, materials and emit emissions. Therefore, a collection system change, which impacts the distance travelled by waste collection trucks, will have a significant impact on the total environment. As stated by Brambilla Pisoni et al. (2009), 'neglecting the effects of collection and transport might result in a severe underestimation of the environmental impacts of a waste management system […]'. However, waste-related LCA studies differ vastly in the methodology on how collection distances are taken into account, ranging from completely ignoring the waste collection distance (see for instance (Thomsen et al., 2017)), to a detailed optimization of the transportation problem using operations research techniques (e.g. (Mora et al., 2014)). Most LCA studies calculate the distance travelled by waste collection trucks by making a distinction between the collection and the non-collection distance. Examples of such studies are (Teixeira et al., 2014; Brambilla Pisoni et al., 2009; Iriarte et al., 2009; Ripa et al., 2017; Merrild et al., 2012; Aranda Usón et al., 2013). Changes in the amount of waste collected per waste stream effect the environmental impact of waste collection trucks. Merrild et al. (2012) propose to use a fuel amount per ton of waste ratio introduced by Larsen et al. (2009). The fuel consumption for changes in the collected waste amount is calculated by extrapolation. The authors however ignore that collection rounds most often need to be executed, regardless of the amount of waste each household generates.

A complete LC-based assessment for biowaste is carried out in Jensen et al. (2016) who compared management concepts of different regions. Their case study showed for most impact categories a better performance of the region where biowaste is treated in incineration compared to a region with a more circular bioresource management with combined AD and composting, and mechanical and biological treatment. In a hypothetical case study representing Denmark and framework conditions representative for the EU, Naroznova et al. (2016) found that wet biowaste such as animal food waste, kitchen tissue and vegetation waste have a better GWP in AD compared to incineration, unless compared to a highly efficient incinerator. Other multi-impact LCAs investigated different scenarios for a region, e.g. Thomsen et al. (2017) who studied an increased circular bioresource management system obtained by diverting organic waste from combustion to AD co-digestion in sludge and manure-based biogas plants. For this case study in an agricultural setting they found a significant improvement of the efficiency in use of resources, but also environmental trade-offs. Colón et al. (2015) studied biowaste scenarios for Catalonia and found mostly advantages for the new scenarios that included increased AD treatment. Also Cobo et al. (2018) investigated scenarios for the management of organic waste in a Spanish region (Cantabria) with a focus on the recovery of nutrients in an agricultural application. They found that an improved nutrient circularity increased eutrophication.

The environmental performance of biowaste management systems has also been studied in an urban context. For example, Bernstad and la Cour Jansen (2011) compared different scenarios for biowaste management system in a residential area in Malmö, Sweden. Ahamed et al. (2016) compared a scenario with centralized AD, waste-to-energy biodiesel and incineration of food waste in a densely populated urban city (Singapore).

For a medium sized, densely populated city such as Brussels that aims to change its current systems to a local and more circular biowaste management system, the environmental consequences of such changes have not yet been studied. Previous publications analyzed water, energy and material and pollution flows (Athanassiadis et al., 2017) as well as waste flows and their potential for CE (Zeller et al., 2017, 2019). These publications provided a diagnosis of the current state of flow management in Brussels. The subsequent step to define different CE scenarios and assess their environmental performance has not been carried out so far. Concretely, the consequences of implementing a local and circular biowaste management on the urban waste collection and transport system, the integrated effects on the existing system and the environmental performance of the new system including different management options for by-products have not yet been studied from an LC perspective. Although the transition towards more local and circular systems is suggested in CE plans, the environmental implications of such changes are rarely quantified. How do more circular and local biowaste management systems perform in an urban setting?

Regarding the evaluation of transport, to the best of our knowledge, no proper strategy has been proposed for estimating the impact of changes in collected waste amounts on waste collection transportation distances without resorting to more elaborate operations research techniques. This paper therefore proposes a novel strategy which is simple, yet effective for estimating the impact of such changes.

Cities such as Brussels have to deal with significant amounts of garden and park waste as well as food and kitchen waste from households and economic activities. In Brussels, food waste is mainly managed as part of the residual municipal solid waste (MSW) stream and green waste is managed separately. Until now, little is known about the environmental performance of the combined management of food and green waste in an urban setting. Thus, we focus on a combined management of these fractions and compare different stand-alone composting systems with AD coupled with post-composting. Their performance is evaluated against the conventional waste management system in which green and food waste flows are handled in different systems. Accordingly, this study adds to the existing literature a joint analysis of the by-products electricity and compost that can be used in different applications such as nutrient supply in agriculture, soil amendments and growth media substitution in parks or gardens. For these systems and applications relevant for cities, we identify the biowaste management system with the best environmental performances and study the role of processes and substitutions.

The study is structured as follows: Section 2 provides a description of data and method that are used in the study. It includes the description of the case study (Section 2.1), the study design (Section 2.2), the general approach (Section 2.3) and the components of the LC model (Section 2.4) as well as the substitution approach (Section 2.5). Furthermore, we present the chosen impact assessment method in Section 2.6 and the uncertainty and sensitivity analysis in Section 2.7. The results and discussion section starts with LCA results for the individual processes (Section 3.1) and continues with the scenario comparison in Section 3.2. After the presentation of the results of the sensitivity analysis (Section 3.3) we further discuss the limitations of this study in Section 3.4. Section 4 presents the conclusions that can be drawn from this study.

2. Data and method

2.1. Case study description

The case study is conducted in Brussels, Belgium, a densely populated European city (7384 inhab./km^2) with around 1.2 million inhabitants. The waste management system in Brussels and the potential of waste flows for CE are analyzed in Zeller et al. (2019) for all types of solid waste. In the present study, we focus on '**biowaste**' defined as 'biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants' in the WFD. Thus, the two principal components of biowaste in this definition are (i) garden and park waste, which is summarized and named as 'green waste' in this study and (ii) 'food and kitchen' waste, summarized as food waste.

In the current waste management system in Brussels, the main part of the total generated food waste (around 160,000 Mg*yr^{-1}) is managed as part of the residual MSW stream. The latter is the MSW fraction that is supposed to be not recyclable and corresponds to around 500,000 Mg generated per year. The residual MSW is mainly collected by a public agency (with 70% bags collection) and treated in the local waste to energy facility (WtE). Since 2018 food waste is also collected separately in all municipalities of Brussels. Thus, the separate collection is only recently introduced and not obligatory which explains that only small amounts are currently collected (500 Mg in 2014, 4300 Mg in 2017). Due to the absence of a treatment facility for food waste in Brussels, the separately collected food waste is exported to an AD facility located 130 km from of Brussels.

Green waste generated by households is separately collected (bags collection) since 2002. In 2018 around 12,000 Mg were collected by the public service and sent to the green waste composting facility in Brussels (capacity: 20,000 Mg*yr^{-1}). Green waste is also collected by private professional gardening and landscaping companies, sent to the local green waste composting facility or exported to composting and AD facilities outside of Brussels.

2.2. Study design

2.2.1. Modelling approach

Two types of modelling are distinguished in LCA: attributional (a-LCA) and consequential LCA (c-LCA). The first models environmental interventions of an existing product system, the second focusses on changes resulting from an action taken place in the system (Rebitzer et al., 2004). C-LCA is defined as a 'system modelling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit.' (UNEP, 2011). In this study we evaluated the environmental consequences of changes in a local biowaste system, so this study is a **consequential LCA**. The change can be described as a transition towards a more circular and local management of biowaste and includes changes in the existing collection and treatment modes and in the management of the by-products of the biowaste system.

Potential changes in the waste management system have been discussed intensively over the last years in Brussels. In this context, biowaste scenarios have been developed by an inter-project collaboration between different research teams (Bortolotti et al., 2019): a baseline scenario that extrapolates current trends in urban biowaste management until 2025, a CE scenario that foresees investment in regional industrial infrastructures and a CE scenario with larger implication of local, decentralized initiatives. The CE scenarios assume that 50,000 Mg of green and food waste will be collected separately by 2025 and that new treatment facilities, either industrial ones (co-composting and AD) or decentralized systems will be operated in Brussels. The estimated amount of 50,000 Mg corresponds to 31% of the currently managed biowaste in Brussels. This share is considered to be realistically implementable for the time horizon 2025.

This estimation and the developed scenarios are used as basis for the c-LCA where only the part of an overall system is studied that is going to be changed. Thus, we study the management of 50,000 Mg of biowaste that is assumed to be separately collected and compare the impacts from the new systems that are installed (i.e. the CE scenarios) with the system that is replaced (i.e. the baseline scenario). We model the replacement of an existing systems with facilities that run on lower capacity or have to close. According to the ambitious political CE targets as described in the PREC (2016), we do not expect that capacity levels of the (publicly managed) facilities are maintained with additional waste imports.

2.2.2. Scenario description with circularity indicator

The following scenarios are analyzed in the study:

- Baseline 2025 that considers the current biowaste management system (state 2018), applied to the quantities managed in 2025
- Scenario 0 (S0) that extrapolates trends for the export of food waste to 2025
- Scenario 1 (S1) that considers the installation of a local co-composting facility in 2025
- Scenario 2 (S2) that considers the installation of a local AD facility in 2025
- Scenario 3 (S3) that considers a larger implication of local, decentralized initiatives (home & neighborhood composting, a small scale composting type called 'in-situ' composting) in 2025.

Fig. 1 illustrates the study design.

In addition, Table 1 shows the quantitative waste flows and characterizes the circularity performance of each scenario with a circularity indicator (CI). As CI we use the recycling rate, measured as input into a recycling system. In accordance with official reporting systems (European Commission, 2018), composting and AD are considered as recycling systems, while the share sent to incineration with energy recovery is not considered in the recycling rate. We further distinguish into a general and a local recycling rate. The CIs show for the baseline scenario that it is already partly circular (recycling rates of 0.5 and 0.4, for the general and local recycling rate, respectively).

2.2.3. Functional unit and system boundary

The goal of this LCA is to identify the best environmental option for the management of biowaste in Brussels. Therefore, the ***functional unit*** (FU) is the treatment of biowaste, more precisely, the treatment of separately collected biowaste in Brussels in 2025 with a reference flow of 50,000 Mg. The exact waste composition is defined later (Section 2.4.1). Like most waste treatment systems, the biowaste system is a multifunctional one, providing not only the function of waste treatment, but also by-products such as compost and electricity. In c-LCA these by-products are addressed with the substitution approach (Schrijvers et al., 2016) in which avoided environmental impacts from the production of displaced products are subtracted from the waste treatment system which produced these by-products. This principle of granting credits for avoided or displaced products is applied in this study, and illustrated in Fig. 2 (dashed boxes).

As shown in Fig. 2, the ***system boundary*** of this LCA is a bin to cradle boundary, starting from waste generation until the final treatment of residuals. The main LC stages are waste collection, transport to the waste treatment facility, the waste treatment including use on land processes (if relevant), the final treatment of residual (such as fly ashes from incineration) and the production of displaced products.

2.3. General approach

To estimate LC-based environmental impacts from changes in Brussels' biowaste system, it is necessary to compile an inventory covering all relevant emissions and resource uses from the different LC phases. In the following sections we describe the model behind this inventory, the so-called biowaste **LC model**. The detailed description of each LC phase of the model follows in the next section (Section 2.4).

As illustrated in Fig. 3, the **LC model** covers waste generation, collection and transport, treatment, and management of the final residuals as well as displaced products from the by-products of the waste management system. To feed the LC model, we used different data sources and sub-models such as (i) local data and data from databases, (ii) a material flow model and (iii) a substitution model.

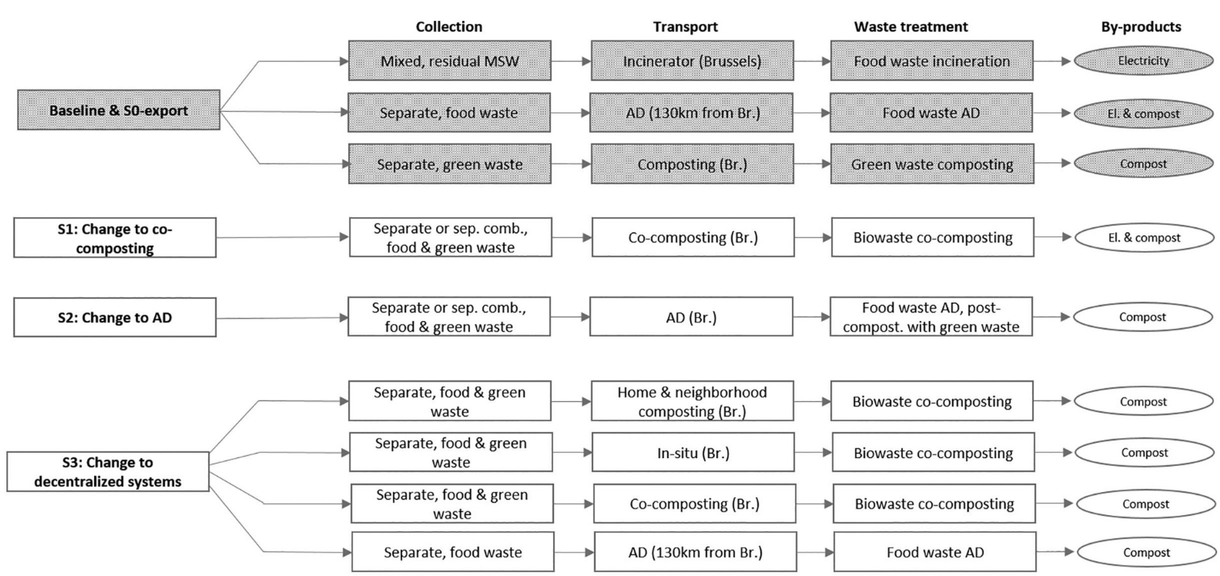

Fig. 1. Illustration of scenarios. The figure shows the type of collection, transport, waste treatment and byproducts in the different scenarios (AD.: Anaerobic digestion; sep. comb.: separate combined collection; Br.: Brussels).

Table 1
Waste flows treated in each scenario and circularity performance of scenarios.

	Flows to treatment							Total flows			Circularity indicator	
	FW Inc.	GW Comp.	FW AD-exp.	BioW Co-comp.	BioW AD-Br.	BioW Home comp.	BioW In-situ comp.	Green waste	Food waste	Total flow	Rec. rate	Local rec. rate
	in Mg^*yr^{-1}											
Baseline	25,000	20,000	5000					20,000	30,000	50,000	0.5	0.4
S0	13,000	20,000	17,000					20,000	30,000	50,000	0.74	0.4
S1				50,000				20,000	30,000	50,000	1	1
S2					50,000			20,000	30,000	50,000	1	1
S3			3100	17,000		22,900	7000	20,000	30,000	50,000	1	0.94

FW: Food waste; GW: Green waste; BioW: Biowaste; Inc.: Incineration; Comp.: Composting; AD.: Anaerobic digestion; Br.: Brussels; Rec.: Recycling.

As pointed out by Laurent et al. (2014) LCAs on waste management should be based on local data to capture local specificities of waste management systems. For this research, we studied the local sorting and collection system (bags, bins, collection fleet, locations, etc.) and collected to the most possible extend **local data** such as 'real life' transport data from transport authorities and site-specific process data from waste treatment facilities. Some of these datasets (e.g. process emissions) can be directly used in the LC model. Other datasets are used to feed additional models such as the integrated transport model which calculates collection and transport distances for the new collection systems that are studied. Most datasets were then combined with an **LC database** (ecoinvent) to estimate for example the CO_2 emissions from transport. As in most data collections for local waste management systems, we were confronted with data limitations, for example, the absence of emission data from decentralized biowaste systems. Also, the use of generic waste treatment datasets from LC databases has limitations if different biowaste compositions in different treatments options will be compared.

To avoid these limitations, we worked with a material flow model for the assessment of environmental technologies (EASETECH). This model characterizes each waste flow as a mix of waste fractions with specific properties and elementary composition, so that substances can be traced

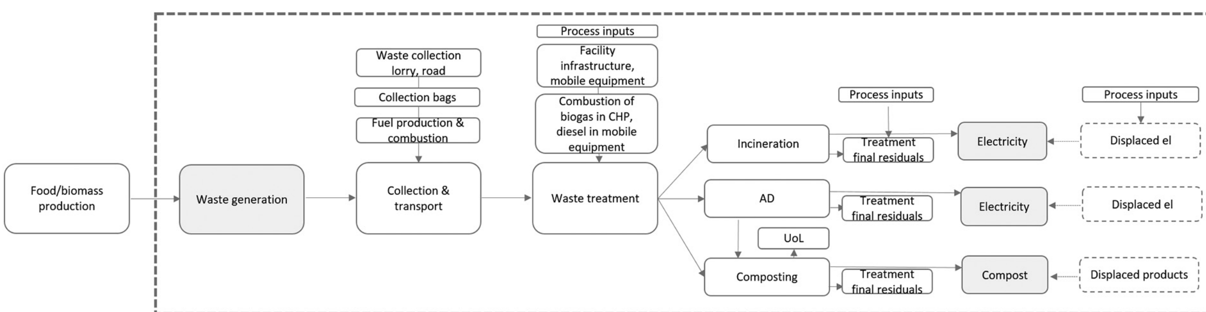

Fig. 2. System boundary for the LCA. The figure illustrates which processes are included in the LCA (AD: Anaerobic digestion, Uol: Use on land, CHP: combined heat and power, el: electricity).

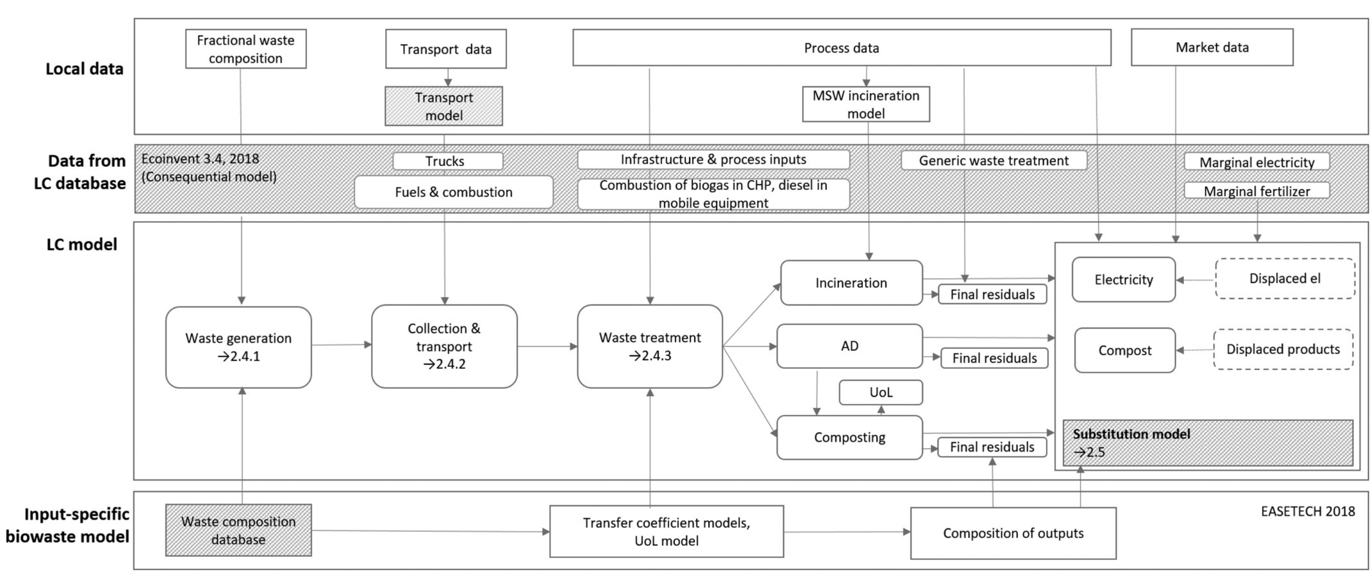

Fig. 3. General approach: Data flow and combination of databases to develop the LC model (UoL: Use on land; AD: Anaerobic digestion). Dashed boxes: additional information is available in the supplementary material.

throughout the different stages of the waste management chain (Clavreul et al., 2014). As illustrated in Fig. 3, the main model components are a waste composition database, transfer coefficient models and a use on land (UoL) model. We applied this model to the biowaste management system in Brussels to determine emissions from the different waste treatments and from the application of compost. Furthermore, it was used to determine intermediate parameters such as the nutrient composition of the compost, which are needed to analyze substitution effects. The calculated emission data and composition of by-products consider the specific composition of the different biowaste flows, so we call it the **input-specific biowaste model**.

The third component of the LC model is the **substitution model**. Previous studies have demonstrated the importance of substitution effects for studies on waste management (Laurent et al., 2014). In this study we used the framework developed by Vadenbo et al. (2017) which is specific for substitution effects in waste management systems. Local information on the current use of by-products and market requirements as well as data from LC databases (consequential datasets in ecoinvent) supported the calculation of the substitution potential for by-products from the biowaste management system in Brussels.

The presented specific combination of local data, databases and models is relevant for other waste treatment studies that aim to develop (i) a local LC model, but facing data gaps such as the lack of physico-chemical composition data and local emission measurements, (ii) a comparative model that takes input-specific variations into account and (iii) a consequential LCA model.

2.4. Components of the LC model

2.4.1. Waste generation

The starting point of the LC model is the generation of biowaste in households and/or economic activities. Based on the definitions in 2.1, we consider the two principal fractions 'food and green waste' generated 'at source' and seven mixes of biowaste fractions 'at treatment', i.e. when entering the different waste treatment facilities that are studied. These mixes depend on the waste composition 'at source', the sorting and collection system and the specific handling of waste in the waste treatment facility.

Local data on the fractional composition of waste was obtained from composition analyses conducted by the authority in charge of the public collection system. Data is available for mixed residual bags that are sent to incineration. For the other treatment facilities, local information on sorting requirements and recommendations on compositions was used to estimate the *fractional compositions* indicated in Table 2. Since most waste in Brussels is collected in bags (e.g. 70% of residual waste), the waste mix entering a treatment facility can also include a plastic fraction (HDPE or biodegradable plastic). For green waste composting (already collected in biodegradable bags), co-composting and AD we assume the use of biodegradable bags by 2025. Based on site-specific data and results from a feasibility study (Bortolotti et al., 2018b), we estimated the share of 'other fractions' which represent process losses.

The **input-specific biowaste model** was used to determine the *physico-chemical waste* composition. The waste composition database in EASETECH (Clavreul et al., 2014; DTU, 2018) provides such physico-chemical data per waste fraction. Thus, by combining this data with the fractional composition, we calculated the physico-chemical composition for the local food and green waste mixes. The fraction 'other' consisting of stones, branches or plastic the physico-chemical composition could not be quantitatively defined. Therefore, the composition is shown without this fraction. The complete composition of the studied biowaste mixes is given in SM1-Table A1.

2.4.2. Waste collection and transport

When studying the impact of waste management scenarios in a setting with bin-to-cradle system boundaries, proper estimations of the transportation requirements of each scenario are vital. The introduction of an additional waste fraction to be collected separately will create additional transportation and therefore both additional costs and negative externalities. Our estimations are based on **local data**, more specifically, transport data provided by the responsible authority in the Brussels Capital Region (BCR) for the door-to-door waste collection. The data provides information on how much waste was collected in which areas of the BCR during 5 months in 2018 for the different municipal

Table 2
Fractional and physico-chemical composition.

	Waste composition at source			Waste composition at treatment						
	Food waste mix	Green waste mix	Plastic bags	Food waste mix (Inc.)	Green waste mix (Comp.)	Food waste mix (AD-exp.)	Biowaste mix (AD-Brussels)	Biowaste mix (Co-comp.)	Biowaste mix (HC)	Biowaste mix (in-situ)
Fractional composition										
Vegetable waste	70.0%			69.8%		67.1%	37.3%	39.5%	50.0%	52.0%
Animal based	30.0%			29.9%		28.8%	16.0%	16.9%		22.3%
Plants		31.0%			30.6%		11.0%	11.7%	16.7%	
Grass and leaves		35.0%			34.5%		12.4%	13.2%	16.7%	
Branches		17.0%			16.8%		6.0%	6.4%	16.7%	24.8%
Tree		17.0%			16.8%		6.0%	6.4%		
Plastic bag				0.3%	0.2%	0.1%	0.2%	0.2%		
Other fractions					1.2%	4.0%	10.9%	5.7%		1.0%
Physico-chemical composition										
Total Wet Weight (kg)	1000.00	1000.00	1000.00	1000.00	1000.00	1000.00	1000.00	1000.00	1000.00	1000.00
Water (kg)	710.30	530.20	71.00	708.36	530.20	710.30	638.26	638.26	655.00	650.23
Total solids (kg)	289.70	469.80	929.00	291.64	469.80	289.70	361.74	361.74	345.00	349.78
Volatile solids (kg)	270.13	297.60	877.91	271.98	297.60	270.13	281.12	281.12	251.50	325.82
Ash (kg)	19.57	172.20	51.10	19.66	172.20	19.57	80.62	80.62	93.50	23.95
Energy (MJ)	6105.89	5488.97	29,690.84	6177.54	5488.97	6105.89	5859.12	5859.12	4757.50	6945.86
C bio (kg)	147.77	121.76	3.30	147.34	121.76	147.77	137.37	137.37	113.19	167.83
C fossil (kg)	1.84	1.22	655.87	3.83	1.22	1.84	1.59	1.59	0.89	1.38
H (kg)	20.79	19.18	90.11	21.00	19.18	20.79	20.15	20.15	16.78	23.41
O (kg)	87.02	121.73	103.12	87.07	121.73	87.02	100.90	100.90	102.77	114.69
N (kg)	12.07	3.71	4.65	12.05	3.71	12.07	8.72	8.72	4.36	10.38
S (kg)	0.78	0.35	0.48	0.78	0.35	0.78	0.61	0.61	0.41	0.68
P (kg)	1.65	0.54	5.21	1.66	0.54	1.65	1.21	1.21	0.57	1.40

Inc. = Incineration, Comp = Composting, AD-exp. = AD export, HC = home composting, including neighborhood composting waste streams collected separately. A summary of the 2018 data can be found in SM1 (Table A 2). Note that we only look into the door-to-door collection provided by the public service in the BCR. Part of the green waste is transported by private actors and part is collected in civic amenity sites where residents can drop off all sorts of waste in dedicated containers.

The transportation distances were calculated for the baseline scenario and the scenarios 0, 1 & 2 (export, co-composting and AD) presented earlier. For scenarios 1 and 2 the same type of waste collection is required. We therefore discuss them together. Two options are available for collecting food waste:

- option 1: food waste is taken out of the residual waste fraction and collected together with green waste (called: **separate combined collection**). The two fractions can be collected in the same bag or in two different bags depending on whether the treatment facility needs to be able to create an optimal green/food waste mix. This choice however does not impact the distance travelled by waste collection trucks;
- option 2: food waste is taken out of the residual waste fraction and collected separately from green waste (called: **separate collection**);

The distance driven for a newly separately collected waste stream depends on the area serviced (e.g. green waste is only collected in some areas of the BCR) and on how often trucks have to drive from the area being serviced to a treatment facility. The latter is largely determined by the amount of waste to be collected. To estimate the transportation distance in each scenario, we make a distinction between the collection distance and the non-collection distance. The former comprises of the distance travelled during the actual collection, i.e. while bags and bin contents are deposited in the collection truck. The latter contains the distance travelled from the truck depot to the service area, between service areas, from the service area to the treatment facility, from the treatment facility to the service area and from the treatment facility back to the depot. For the estimation of the collection and non-collection distances for each waste stream in each scenario we refer to the supplementary material (SM1).

Combining the collection and non-collection distances and the waste quantities per waste stream enables us to calculate a km*Mg^{-1} ratio which will be used in the LC model. Table 3 presents the total transportation distance, the collected weight and the km*Mg^{-1} per waste stream in each scenario. The last column in Table 3 clearly shows that the three scenarios S1-S3 bring about a reduction in transport compared to the baseline and export scenario (S0). For scenario 1 and 2 this is mainly due to the elimination of the transportation to the external AD facility. Separate combined collection of food and green waste as opposed to separate collection further reduces the transportation distance with 150,000 km. In scenario 3, some food waste is still sent to the external AD facility located 130 km from Brussels. Therefore, only option 2 is feasible as food waste must be kept separately. The reduction in transportation distance in this scenario is mainly due to higher levels of home composting and a low transportation distance for the in-situ collection.

Emissions from the collection of waste are modelled based on a representative collection and hydraulic compression vehicle for MSW collection as inventoried in ecoinvent 3.4 (21 ton lorry, gross load capacity 8.2 ton, load factor 50%). Included activities are diesel fuel consumption (0.4 kg/tkm driven), air emissions from fuel combustion for stop and go drying, abrasion (tire, brake lining, road), the vehicle and road construction.

2.4.3. Biowaste treatment- Incineration

Brussels' incineration plant is a WtE facility for the treatment of residual MSW. The facility produces steam which is used in the neighbor power plant to generate electricity. In 2018, 490.000 Mg of MSW were incinerated to produce 280 GWh electricity. The combustion technology is a grate-based incineration. The facility is equipped with an air pollution prevention system (electrofilter and wet scrubber) and a DeNOx unit.

Local data was collected including material and energy flows, process inputs, data on the treatment of final residuals as well as emission data. The local data used to feed the LC model are process inputs (natural gas, caustic soda, activated carbon etc.), process emissions and residuals treatment (type and transport distances).

Process emissions (such as NOx, SO_2, HCl, etc.) are emissions that are mainly determined by process conditions (e.g. temperature, type of installed APC system). Input-specific emissions are mainly determined by the composition of the waste input (e.g. CO_2 and heavy metals) (Damgaard et al., 2010). The collected process emission data (as well as process inputs) refer to the incineration of MSW and not specifically to the food waste fraction of MSW. In order to create such a specific dataset from this **multi-input dataset**, we distributed process emissions and inputs over the multiple waste fractions proportional to their wet weight. Thus, food waste received, for example, 34% of the

Table 3
Yearly collected weight, transportation distance and km/ton for each waste stream under the baseline case and the two transportation scenarios.

	Weight per waste stream (Mg)	Total distance per waste stream (km)	Distance per waste stream (km*Mg^{-1})	Total distance per scenario (km)
Baseline & S0				
Baseline (5000 Mg)				
Residual waste	340,007	2,034,880	5.98	2,675,941
Food waste	5000	419,433	83.89	
Green waste	14,500	221,629	15.28	
S0 (17,000 Mg)				
Residual waste	328,007	1,982,470	6.04	2,970,281
Food waste	17,000	766,182	45.07	
Green waste	14,500	221,629	15.28	
Scenario 1 & 2				
Option 1				
Residual waste	315,007	1,925,693	6.11	2395,000
Food + Green waste	44,500	469,307	10.55	
Option 2				
Residual waste	315,007	1,925,693	6.11	2,553,300
Green waste	14,500	221,629	15.28	
Food waste	30,000	405,978	13.53	
Scenario 3				
Option 2				
Residual waste	315,007	1,925,693	6.11	2,667,908
Green waste (co-composting)	6800	149,610	22.00	
Food waste (co-composting + AD)	13,300	504,965	37.97	
Food + green waste (in situ)	7000	87,640	12.52	

ammonia input used in the DeNOx process and 34% of NOx emissions. This decision is justified by the fact, that process emissions are driven by the process conditions and not by the type of waste input.

Data on electricity generation and use was also provided by the facility. As explained in Section 2.1, we use the substitution method to handle by-products such as electricity and need to determine the amount of electricity that can displace electricity from marginal electricity production. Other waste-type specific incineration models (Thomsen et al., 2017; Doka, 2013) calculate the amount of electricity that can be achieved from a specific waste fraction based on its energy content. This seems a correct approach under the assumption that the relative composition of the mix entering the facility remains stable. However, if a specific fraction is diverted from the incinerator, MSW composition will change and the remaining MSW will have a different average heating value. In our model, we consider this effect and calculate how the energy production will be affected if 25,000 Mg food waste (or 13,000 Mg in baseline 2025) is redirected from the incinerator. The calculation (see SM2-A) is based on plant-specific information on heating values, food waste content and electricity output and results in an electricity surplus of 0.14 $kWh^{*}kg^{-1}_{food\ waste}$.

Local data on final residual treatment was also provided by the facility: Fly ash from this facility is transported by lorry to Germany where it is disposed in salt mines. Bottom ash is transported by boat to the Netherlands and used in road constructions. Environmental burdens from transport are modelled with ecoinvent datasets. For the final deposit of fly ash in salt mines we assume that no environmental impact occurs. For the application of bottom ash in road construction we include leaching of heavy metals according to Allegrini et al. (2015) and give a credit for the substitution of gravel production. The type and quantities of process inputs and process emissions as well as chosen ecoinvent models and references are documented in SM2-A.

The **input-specific biowaste model** was used to determine the input-specific emissions and the amount of residuals from the incineration of food waste. The input-specific emissions are calculated based on the physico-chemical composition of the food waste mix entering the incinerator (see SM1-Table A1) and based on the transfer coefficients specified in EASETECH's incineration model (Riber et al., 2008; DTU, 2018). For example, based on the amount of C_{bio} and C_{fossil} (Table 2) and the transfer coefficient for carbon (99.9 to air and 0.1 to bottom ash) the CO_2 emissions are calculated. These CO_2 emissions are also measured at the incineration facility, but can not be linked with the input 'food waste'. Based on the transfer coefficients, the amount of bottom and fly ash was calculated, resulting in 134 kg of bottom ash, 1.5 kg of fly ash$^{*}Mg^{-1}$ $_{food\ waste}$. Emission data from the input-specific biowaste model are available in SM2-A for the incineration process.

2.4.4. Biowaste treatment- Anaerobic digestion

Two biogas facilities are evaluated in this study: the first, **AD-export**, is located approximately 130 km from Brussels. The amounts of food waste from Brussels treated in the facility are small, but increasing: 500 Mg in 2014, 4300 Mg in 2017, 17,000 Mg expected in 2025. The AD process is a wet process that uses BTA® process for mechanical biological waste treatment. The input waste undergoes a dry (drum screen to remove impurities) and a wet pretreatment (pulping, separation of heavy and light fractions). The pulp is sent to the digester (two stage process). The dewatered digestate is mixed with chopped green waste. The raw compost stays 2 weeks in tunnels and 8 weeks in windrow area. With an input capacity of 50,000 Mg per year the facility treats a mix of vegetable, fruit and garden waste from households (so called VFG waste, 49%), solid (6%) and liquid (15%) organic biological waste from professional activities, as well as green waste (30%). The facility provides electricity (for internal and external use), heat (for internal use) and compost.

For the second facility (**AD-Brussels**), possible locations in Brussels and plant designs have been studied in a feasibility assessment (Bortolotti et al., 2018b). The proposed technology is a dry AD process in combination with post-composting of the digestate together with the green waste. The food waste is pretreated (sieving, chopping, metal separation) and then sent to the digester (AD stage of 3 weeks). The produced biogas can be used for electricity generation or upgraded to biomethane. The digestate is mixed and composted with green waste which was previously chopped and sieved. The composting process takes place in a closed hall which is equipped with a ventilation system and biofilter. After 2 weeks of composting, the precompost is sieved and the small fractions are sent to maturation in the maturation hall (2 weeks). The input capacity is expected to be 50,000 Mg biowaste, composed of 60% food and 40% green waste. It is planned that the facility provides electricity (for internal and external use), heat (for internal use) and compost. The main process characteristics of the two facilities are given in Table 4.

The existing AD facility is a multi-input process treating multiple feedstock, not only food waste. Therefore, not all data measured in the facility (e.g. biogas and electricity yields) could be used for this model and we developed an AD model that considers the process conditions of the facility (in terms of electricity and heat demand, process inputs and efficiencies of the CHP modules), but studies the digestion of food waste, only. Therefore,

Table 4
Process characteristics- AD.

		AD-export	AD-Brussels
AD process		Wet process, BTA process for mechanical biological waste treatment	Dry process
		Two stage digestion	One stage
		Mesophilic	Mesophilic/thermophilic
Retention time		14 days	21 days
		De-watering and post composting	Post composting
		Stationary CHP modules	
Stationary engines			
Efficiency (el)	%	32	39
Efficiency (th)	%	40	40
El & heat use			
El, internal use	% of generated el	44	44
El, to public grid	% of generated el	56	56
Heat, internal use	% of generated heat	28	6
Heat, external use	% of generated heat	0	0
Composting process			
Technology		Closed-building tunnel composting	
Composting duration	Weeks	10	4 (2 composting, 2 maturation)
Compost yield	$Mg^{*}Mg^{-1}$ $_{biowaste\ in\ composting}$	0.35	0.35
Biofilter		Present	Present

CHP = Combined heat and power.

the estimated shares of electricity and heat use (in Table 4) and the biomethane yield differ from what is measured in the facility.

Local data was collected from the existing biogas plant (AD-export) including data on material and energy flows, process inputs and treatment of final residuals. Regarding emission data, only NH_3 emissions are measured in this facility. For the future facility (AD-Brussels) material and energy balances as well as process inputs specified in a feasibility study (Bortolotti et al., 2018b) are used here. The local data to feed the LC model consists of process inputs such as diesel for the mobile equipment (for example for the pretreatment), tap water or sulfuric acid for the waste water and air treatment. These process inputs are distributed over the different waste fractions of this multi-input process (VFG, liquid and solid fraction) according to their mass. We also used the efficiencies of the stationary CHP modules and the internal heat and electricity demand specified for the two facilities to feed the LC model. Emissions from the combustion of biogas in the CHP modules, from the combustion of diesel in the mobile equipment as well as impacts from the production of the different process inputs and infrastructure are modelled with ecoinvent data. The type and quantities of process inputs, chosen ecoinvent models and references are documented in SM2-B for the two AD processes.

The **input-specific biowaste model** was used to determine the biogas yields, the fugitive CH_4 emissions from the AD process, emissions from the composting process and the composition of the produced compost. Following the same approach as later applied for composting (see Section 2.4.5), we estimated emissions from the AD processes with post-composting with a model that calculates C-containing emissions as a function of the degradation of C-containing compounds in the biowaste (Boldrin et al., 2011). The starting point for the modelling of emissions from the AD process is the potentially anaerobically digestible organic carbon, expressed in kg $C_{bio\ and}$. The calculated $C_{bio\ and}$ content for the food waste mix in Brussels is 102 kg^*Mg^{-1} which corresponds to a theoretical biomethane potential of $120m^{3}{}^*Mg^{-1}_{food\ waste}$. From this theoretical potential, we defined the gas yield (as proportion of $C_{bio\ and}$) that can be achieved in the facilities: 50% for the wet (AD-export) and 60% for the dry process (AD-Brussels). The latter yield corresponds to the yields estimated in the feasibility study. The yield for AD-export is assumed to be lower due to the shorter retention time. The final biomethane yields are around $42m^{3}{}^*Mg^{-1}_{biowaste}$ for both facilities which corresponds to $60m^{3}{}^*Mg^{-1}_{food\ waste}$ for AD-export and $71m^{3}{}^*Mg^{-1}_{food\ waste}$ for AD-Brussels. Following the default value in EASETECH (DTU, 2018), we estimate that 2% of the generated methane are fugitive emissions, which corresponds to 0.85 $kg^*\ Mg^{-1}_{biowaste}$.

To model the post-composting process, we use a combined technology model that estimates the physico-chemical composition of the material entering the composting stage (i.e. the digestate output) after biodegradation in the reactor. Thus, the composition of the digestate corresponds to the biowaste input, minus the fraction that goes to the gas phase. The model does not take into account potential losses in the dewatering phase of the wet process (AD-Brussels), but considers the degradation and losses in the subsequent composting process. The post-composting process of the (dewatered) digestate takes place (for both processes) in a closed building tunnel composting with the same characteristics as the co-composting process indicated in Table 5. Due to the absence of specific degradation values and emission coefficient for the digestate, we take directly the values indicated for the co-composting process.

2.4.5. Biowaste treatment- Composting

Four composting systems are evaluated in this study: (i) home and neighborhood composting systems, (ii) an industrial green waste composting facility, (iii) an industrial co-composting system and (iv) and a small scale food composting system (in-situ composting). The main process characteristics are summarized in Table 5.

Home and neighborhood composting is a decentralized waste treatment option that is used for the treatment of household food and green waste. In Brussels, 150 neighborhood composts existed in 2015 that treated around 400 Mg of biowaste. They are expected to increase to around 1100 Mg in 2025. The number of composting units and amount of biowaste treated in home composting are not monitored. A survey indicated that 30% of Brussels' residents composted at home their green waste and 14% composted kitchen waste in 2014 (IPSOS, 2014). The produced compost from these composting systems is mainly used in community or private gardens.

The **green waste composting facility** in Brussels is an open windrow composting for green waste collected from gardens and parks by the public service, municipalities and professional garden enterprises. In 2018, 14,800 tons of green waste were treated and around of 7400 tons of compost were produced. The produced compost is mainly sold unpacked to professional enterprises and private clients. In the first two weeks of the process, the green waste is placed under the dome where the air is aspirated and passes a biofilter. The process steps are chopping, composting under the dome, maturation of the compost (outside in compost heaps), sieving and separation of plastic waste with a windsifter.

Possible designs and locations of a future **industrial co-composting** facility in Brussels have been studied in a feasibility analysis (Bortolotti et al., 2018b). The proposed technology is a closed-building tunnel composting facility for green and food waste. The process steps are chopping, sieving and separation of the biowaste, composting in the tunnel (2 weeks with automatic aeration and hydration), maturation of the compost (4 weeks in the maturation zone in the building) and final sieving. The air of the complete building is planned to be aspirated and to pass a biofilter.

Decentralized, small to medium scale composting systems is another option discussed for Brussels. Different systems (heaps or chalets) have been proposed in a scenario assessment for Brussels (Bortolotti et al., 2019). For this study, we selected an **'in-situ'** wood chalet system as a representative system. It handles between 25 and 200 $Mg_{food\ waste}{}^*yr^{-1}$. Food waste is collected from restaurants, canteens and retailers and transported in boxes to closed-by composting stations where it is composted with wood chips (from green waste chipped in parks). In order to achieve hygienisation of the food waste, a temperature level of at least 55 °C must be reached for 14 days.

Local data was collected from the existing industrial green waste facility including data on material and energy flows, process inputs (electricity and diesel), and treatment of final residuals. Emissions are not measured in this facility. For the neighborhood composting systems, basic input-output flows are monitored and descriptions of the systems are available, such as locations and the types of composting system. These datasets have been used to specify an average composting unit. For the industrial co-composting facility, mass flows and process inputs have been studied in a feasibility analysis (Bortolotti et al., 2018b). For the in-situ composting, local data is not available, but data on material flows, techniques, machinery use and transport is available from a case study in France where these systems are already in place. The local data used to feed the LC model includes process inputs such as diesel for mobile equipment, electricity for the management of the facility as well as the compost yields from the different systems. Emissions from the combustion of diesel in the mobile equipment as well as impacts from the production of the different process inputs and infrastructure are modelled based on ecoinvent data. The type and quantities of process inputs and chosen ecoinvent models are documented in SM2-C.

The **input-specific biowaste model** was used to determine the emissions from the composting process and the composition of the produced compost. Emissions from the composting process were modelled with EASETECH (Boldrin et al., 2011) due to its ability to take a specific biowaste composition into account. The composting model estimates the amount of C-containing (CO_2, CH_4, CO) and N-containing gaseous emissions (NH_3, N_2O and N_2) as a function of the degradation of C- and N-containing compounds in the biowaste. Table 6 shows the degradation values and conversion ratios to gaseous emissions that are used in this study.

For the two facilities that use a biofilter, we use a removal efficiency of 99% for ammonia and 95% for methane as specified in EASETECH for a biofilter in a closed tunnel facility (DTU, 2018).

Table 5
Process characteristics- Composting.

		Home & neighborhood composting	Green waste composting	Co-composting (industrial)	In-situ composting
Technology		Home composting	Open windrow composting	Closed-building tunnel composting	Open chalet composting
Duration	Weeks	26–39	22–26	6	26–35
Biofilter		Absent	Present	Present	Absent
Mass flows					
Total	$Mg^{*}yr^{-1}$	435	17,000	50,000	6890
Capacity per unit	$Mg^{*}yr^{-1}$	3	17,000	50,000	78
Green waste	%	50	100	40	25
Food waste	%	50	0	60	75
Compost yield	$Mg_{out}{}^{*}Mg_{biowaste}^{-1}$	0.3	0.5	0.31	0.33
Compost density	$kg^{*}m^{-3}$	705	410	600	716

For the green waste composting facility, we assume that 60% of emissions passes the biofilter during the 2 weeks composting process under the dome according to measurements of volatile solid degradation in a closed tunnel facility (DTU, 2018; Boldrin et al., 2009). For home composting systems, leaching (emission to groundwater) is included, based on the measurements for home composting systems (Andersen et al., 2011).

2.4.6. Application of compost on soils

Environmental impacts from the application of compost (and other organic fertilizer) on soils depend on the type and composition of the compost, environmental conditions such as climate and soil type and, if applied on agricultural soils, on the agricultural practice (e.g. crop rotations), thus on 'complex and interacting processes largely depending on local conditions' (Hansen et al., 2006). To model these impacts, we use the 'use on land' model in EASETECH which is part of the **input-specific biowaste model** (DTU, 2018). It describes emissions to air, surface water, groundwater and soil accumulation from land application of compost on different soil types. In this model, C and N emissions from the application of compost have been modelled with the agroecosystem model DAISY which includes a hydrological model, a crop model, a mineral nitrogen model, and a soil organic matter model. The degradation values and emissions factors for heavy clay soils (see Table 7) have been chosen which is one of the most dominant soil types in Belgium. Due to the absence of emission coefficients for soils in garden or parks, we apply the same emission coefficients as for agricultural soils. C-sequestration and NH_3 emissions are in the same order of magnitude as found in other studies (2–16% for C-sequestration for a 100-year period (Martínez-Blanco et al., 2013); default volatilization coefficients of 15% for NH_3 (Hansen et al., 2006)).

Leaching of other elements to groundwater and soil is modelled based on measurements from leaching tests as specified in the LCA inventory for green waste and kitchen waste compost (Boldrin et al., 2010). Depending on the fractional composition, leaching profiles have been calculated for each compost type.

2.5. Substitution

An important aspect of c-LCA is the modelling of substitution effects from the by-products of the product system under study. The chosen substitution framework developed by Vadenbo et al. (2017) provides calculation steps and a reporting system to determine the substitution potential of a by-product from a waste management system. The substitution potential (γ) is defined as 'a measure of the end-use–specific change in consumption of the directly affected products resulting from supplying a co-product, for example, a recovered secondary resource, to a particular end use or market' (Vadenbo et al., 2017). It is a function of four determining factors:

$$\gamma = \mathrm{Urec} * \eta * \alpha * \pi, \tag{1}$$

where (Urec) is the physical resource potential, (η) is the resource recovery efficiency, (α) the substitutability and the (π) the market response. For example, Urec can be the NPK content or the biomethane potential in the initial biowaste. Substitutability (α) is defined as the ratio of a recovered resource (φ^{rec}) over the functionality of the substituted alternative product (φ^{dis}) $\alpha = \varphi^{rec}/\varphi^{dis}$. Substitutability

Table 6
Degradation values and emission coefficients for the different composting types.

Degradation values and emission coefficients		Home & neighborhood composting	Green waste composting	Co-composting	In-situ composting
		Average values for HC for organic waste (Andersen et al., 2011)	Values for open-air windrow composting, garden waste (Andersen et al., 2010a; Andersen et al., 2010b)	Values for closed tunnel composting, garden & kitchen waste, values from EASETECH (Boldrin et al., 2009; DTU, 2018)	Values for decentralized composting (food waste and wood chips) (Bernstad and la Cour Jansen, 2011)
Degradation of input N	Ratio	0.595	0.080	0.710	0.330
Conversion to N_2	Ratio	0.948	0.020	0.001	0.032
Conversion to NH_3	Ratio	0.000	0.830	0.985	0.960
Conversion to N_2O	Ratio	0.048	0.150	0.014	0.008
		Remaining to leaching			
Degradation of input C	Ratio	0.700		0.620	0.700
Degradation of input C (food waste)	Ratio			0.740	
Degradation of input C (green waste)	Ratio		0.556	0.540	
Conversion to CO_2	Ratio	0.800	0.976	0.998	0.800
Conversion to CH_4	Ratio	0.018	0.021	0.002	0.018

HC = home composting.

Table 7
Degradation values and emission coefficients for the application of compost on soils (DTU, 2018).

Degradation values and emission coefficients		
Degradation of input N (related to total N-input)	%	18.15
Conversion of degraded N to N_2	%	71.79
Conversion of degraded N to NH_3	%	19.34
Conversion of degraded N to N_2O (related to degraded N)	%	8.87
N (NO_3) Leaching to groundwater (related to total N-input)	%	7.54
N (NO_3) Leaching to surface water (related to total N-input)	%	19.37
N plant uptake (related to total N-input)	%	24.76
Degradation of input C	%	89.14
C-sequestration	%	10.86
Conversion of degraded C to CO_2	%	99.99
Conversion of degraded C to CH_4	%	0.01
P (PO_3) Leaching to groundwater (related to total P-input)	%	0.47
P (PO_3) Leaching to surface water (related to total P-input)	%	0.47
P plant uptake (related to total P-input)	%	84.10

and market response are analyzed in a step-by-step procedure taking systematically constraints into account.

In c-LCA, the market response parameter (π) refers to marginal markets, in contrast to the average market mix used in a-LCA. The marginal technology is the technology actually affected by a small change in demand, usually from a long term perspective. It represents the unconstrained most or least competitive technology and can be determined with a step-wise procedure illustrated in Weidema et al. (1999). In this study, we use the marginal technologies from ecoinvent's consequential system model (Weidema et al., 2013) to determine the marginal fertilizer, peat and electricity market. In the following, we provide a brief description on how the substitution potential was determined in this study. The complete documentation of parameters from the framework and the calculation steps are given in SM2-D.

2.5.1. Substitution potential of compost

In order to calculate the substitution potential for each of the studied compost types, it is necessary to determine (i) the application area of the specific compost (e.g. in agriculture, professional landscaping, or private gardens), (ii) the functionality of compost within its specific application (e.g. as fertilizer in agriculture, as growth media in gardens) and (iii) the substitution potential per functionality (e.g. the potential of compost to substitute mineral fertilizer).

The **application areas** (i) per compost type are shown in Table 8. For the existing composting systems, the application area corresponds to the current use and has been determined by the treatment facilities. For the future facilities scenarios have been created in line with the initial biowaste management scenarios (Bortolotti et al., 2019). These scenarios consider the city's political ambitions (support of food production and agricultural applications) and experiences from decentralized management systems.

The **functionalities of compost** within an application area are given in part (ii) of Table 8. We used the results from a survey of Danish hobby gardeners to determine the compost use in Brussels' private and

Table 8
Substitution potential for the different compost types.

		Home & neighborhood composting	Green waste composting	Co-composting (industrial)	In-situ composting	Post composting (AD-export)	Post composting (AD-Br)
(i) Application area							
Agriculture	Fertilizer & soil conditioner			95%	65%	20%	100%
Parks and gardens (prof.)	Soil conditioner & growth media		95%			60%	
Private & com. gardens	Soil conditioner & growth media	100%	5%	5%	35%	20%	
(ii) Functionality							
Agriculture	Fertilizer (100%)			95%	65%	20%	100%
	Soil conditioner						
Parks & gardens (prof.)	Soil conditioner (77%)		73%			46%	
	Growth media (23%)		22%			14%	
Private & com. gardens	Soil conditioner (77%)	77%	4%	4%	27%	15%	
	Growth media (23%)	23%	1%	1%	8%	5%	
(iii) Technical substitution potential per functionality							
Tech. sub. potential ($\gamma^{\text{fertilizer}}$)	Min. N ($kg{*}Mg^{-1}{}_{\text{biowaste}}$)	0.43	0.83	0.59	1.70	0.60	0.56
	Min. P_2O_5 ($kg{*}Mg^{-1}_{\text{biowaste}}$)	0.94	1.02	2.19	2.67	2.23	2.07
	Min. K_2O ($kg{*}Mg^{-1}_{\text{biowaste}}$)	4.30	4.67	3.64	3.31	3.71	3.45
Tech. sub. potential peat ($\gamma^{\text{soil}}{}_{\text{conditioner}}$)	$kg_{\text{peat}}{*}Mg^{-1}_{\text{biowaste}}$	222.88	354.94	301.03	330.48	228.11	216.40
Tech. sub. potential straw ($\gamma^{\text{soil}}{}_{\text{conditioner}}$)	$kg_{\text{straw}}{*}Mg^{-1}_{\text{biowaste}}$	206.17	328.32	278.46	305.70	211.00	200.17
Tech. sub. potential peat ($\gamma^{\text{growth}}{}_{\text{media}}$)	$m^3_{\text{peat}}{*}Mg^{-1}_{\text{biowaste}}$	0.43	1.22	0.52	0.46	0.58	0.58
Tech. sub. potential peat ($\gamma^{\text{growth}}{}_{\text{media}}$)	$kg_{\text{peat}}{*}Mg^{-1}_{\text{biowaste}}$	85.11	243.90	103.33	92.24	116.67	116.67
(iv) Technical substitution potential per compost type							
Min. N fertilizer	$kg{*}Mg^{-1}{}_{\text{biowaste}}$	0.00	0.00	0.60	1.12	0.13	0.63
Min. P_2O_5 fertilizer	$kg{*}Mg^{-1}{}_{\text{biowaste}}$	0.00	0.00	2.21	1.75	0.47	2.33
Min. K_2O fertilizer	$kg{*}Mg^{-1}{}_{\text{biowaste}}$	0.00	0.00	3.67	2.18	0.77	3.87
Peat	$kg{*}Mg^{-1}{}_{\text{biowaste}}$	191.19	329.40	12.78	96.49	161.98	0.00
(v) User specific substitution potential per compost type							
Peat	$kg{*}Mg^{-1}{}_{\text{biowaste}}$	38.24	184.55	2.42	19.12	73.10	0.00

Prof.: professional, sub.: substitution.

community gardens as well as in parks. These indicated that 77% of compost was used as soil improver, 23% as growth media (Andersen et al., 2010c). Regarding the use of compost in agriculture we study only fertilizer use, because all types of produced compost would be considered as a fertilizer according to fertilization legislation. However, compost is applied due to its fertilizing function and soil improvement quality (Viaene et al., 2016).

The technical **substitution potential per functionality** is given in part (iii) of Table 8. In order to determine the substitution potential for compost used as a **fertilizer**, we use the mineral fertilizer equivalent approach (MFE) which is the most widely used in LCA to quantify fertilizing effects. A MFE determines the share of nutrients in the organic fertilizer that has the same fertilizing effect as a mineral fertilizer, i.e. the share of plant available nutrients in the organic fertilizer (Hanserud et al., 2018). We first determined the NPK content of the recovered compost which was then multiplied by the MFE for N, P and K: 0.248 for N, 0.841 for P (as specified with the land use model, see Table 7) and 1 for K as specified for example in Boldrin et al. (2010) and Jensen et al. (2016). The MFE coefficients can be directly used as substitutability factor α. The market response parameter (π) refers to the marginal markets for N, P and K fertilizer as specified in ecoinvent (ecoinvent, 2017b, 2017c, 2017d). The composition of these marginal markets is given in SM2-E. The substitution potential for compost as fertilizer is the amount of substituted marginal NPK fertilizer (in $kg^*Mg^{-1}_{biowaste}$).

In order to determine the substitution potential for compost used as a **soil conditioner**, we use the 'humus equivalent' (HE) approach which determines the capacity of an organic fertilizer to build up humus. HEs depict the amount of organic carbon, which would lead to a buildup of humus (Dinkel et al., 2012). Based on the HEs per type of organic soil conditioner such as compost, straw, peat (Reinhard and Mueller in Dinkel et al., 2012) and their specific C_{bio} content, we calculated the humus-C content per type of soil conditioner ($kg^*Mg^{-1}_{soil\ conditioner}$). The substitutability factor α is the ratio of humus-C content of compost over humus-C of the alternative soil conditioners (such as peat). Depending on the C_{bio}-content and HE, α is between 0.58 and 0.99 for peat. The substitution potential is then calculated based on the amount of recovered compost ($kg^*Mg^{-1}_{biowaste}$), the substitutability α and the market response (π) which refers to marginal peat production. It is expressed as the amount of displaced peat ($kg^*Mg^{-1}_{biowaste}$).

In order to determine the substitution potential if compost is used as **growth media**, we apply a volume based substitution. The amount of recovered compost per FU is simply converted to its equivalent volume using the densities indicated in Table 5. The substitutability α is 1, indicating that the same volume of an alternative growth media is replaced. The substitution potential is calculated based on the amount of recovered compost ($m^{3*}\ Mg^{-1}_{biowaste}$), α and π which refer to marginal peat. It is expressed as the volume of displaced marginal peat ($m^{3*}\ Mg^{-1}_{biowaste}$).

In the next step of the calculation, we used the specified functionality (ii) and the technical substitution potential per functionality (iii) to calculate the **technical substitution potential per compost type (iv)**. As indicated in Table 8, green waste compost has the highest technical substitution potential for peat while compost from the two industrial facilities (co-composting and post composting/AD) shows the highest technical substitution potential for NPK fertilizer.

Vadenbo et al. (2017) highlight the importance to integrate user behavior in substitution models. The survey by Andersen et al. (2010c) indicated that private compost user substitute only in 20% of cases an equivalent product such as peat. For the application in a professional context, for which no surveys could be found, we assume a more rational use of compost and assume a user-specific factor of 0.5 for the substitution of peat as soil conditioner and 1 for the substitution as growth media. Applying these user-specific factors on the technical substitution potential gives **(v), the user-based substitution** for peat.

The values for the user-based substitution potential per compost type are used in the LC inventory. For example, the inventory for home and neighborhood composting systems includes the avoided production of 38.24 kg peat*Mg^{-1} biowaste treated. For fertilizer substitution, we included the avoided production of the fertilizer, and the avoided emissions from field application of mineral fertilizer. Field emissions are calculated based on emissions factors from Nemecek et al. (2016) and from the use on land model in Easetech (DTU, 2018), documented in SM2.

2.5.2. Substitution potential of electricity

For the waste treatment systems that have **electricity as by-product** their substitution potential needs to be determined. For AD, the resource potential ($U^{rec,tech}$) corresponds to the theoretical biomethane potential ($m^{3*}Mg^{-1}_{biowaste}$, documented in 2.4.4). The recovery efficiency (η) considers several factors, such as the biogas yields achieved in the two facilities (50 and 60%), the loss of methane as fugitive emissions (2%), the efficiency of the stationary CHP engines and the share of electricity and heat for external use (see Table 4). The amount of recovered electricity, calculated as $U^{rec,tech} * \eta$, is 95kWh*$Mg^{-1}_{biowaste}$ for AD export and 119kWh*$Mg^{-1}_{biowaste}$ for AD-Brussels. Since the recovered gas amounts are equal, the difference in electricity output is due to the higher efficiency that is specified for the CHP module in AD-Brussels. The substitutability factor α is 1, indicating that 1 kWh of electricity replaces 1kWh electricity from the marginal market. The market response parameter (π) refers to the marginal electricity mix for Belgium, taken from the consequential model in ecoinvent (ecoinvent, 2017a). It is mainly composed of electricity from natural gas (combined cycle power plant, 55.7%) and wind energy (41.9%) and has a GWP of 275 kg CO_2 eq.*kWh^{-1}. The substitution potential for heat is zero in AD-export since the current facility uses heat internally only and the same concept is planned for AD-Brussels.

As described in Section 2.4.3, an energy gain occurs and therefore a substitution effect if food waste is not incinerated. The theoretical resource potential ($U^{rec,tech}$) for electricity from not incinerating corresponds to the energy content in waste (based on the lower heating value). The recovery efficiency (η) considers the electricity efficiency of the facility and the share of electricity that is provided to the grid. The substitutability factor (α) and market response parameter (π) is the same as for electricity from AD. Thus, the substitution potential for not incinerating 1 Mg of food waste is 141.40 kWh electricity from the marginal electricity market.

2.6. Impact assessment method

For the impact assessment, we apply the state-of the art impact assessment method ReCiPe2016 that converts the substances of the life cycle inventory into 17 midpoint and 3 endpoint impact categories (Huijbregts et al., 2017). The endpoint results indicate potential environmental impacts on human health, on ecosystems and on resources. Impacts on human health are expressed in DALY which stands for disability adjusted life years and represents 'the years that are lost or that a person is disabled due to a disease or accident'. Damages on ecosystems are expressed as potentially disappeared fraction of species·m2·year or potentially disappeared fraction of species · m3 · year. This damage category describes the 'local relative species loss in terrestrial, freshwater and marine ecosystems, respectively, integrated over space and time'. Impacts on the availability of resources are measured in US dollars ($), which represents the extra costs involved for future mineral and fossil resource extraction. This impact category aggregates mineral and fossil resource scarcity.

From the three sets of midpoint and endpoint characterization factors, we chose the hierarchist scenario. It refers to a set of values that consider a 100-year time horizon and integrates effects accepted by international bodies such as the World Health Organization.

For the processes that are evaluated in this study, the counting of biogenic carbon is of particular importance. For example, the main gaseous emissions from incineration and composting is biogenic CO_2, the main emission from AD is biogenic CH_4. In the chosen impact assessment method for global warming (that refers the IPCC 2013 method), biogenic CO_2 is accounted as neutral (i.e. the GWP is zero), biogenic methane has a characterization factor of 34 kg CO_2 eq.*kg^{-1}.

2.7. Uncertainty and sensitivity analysis

A data quality assessment and uncertainty analysis was carried out to analyze uncertainties related to the quality of data that we used in this study. We applied the data quality indicator approach developed by Weidema and Wesnæs (1996) in which the quality of input data such as products, infrastructure, emissions, is estimated based on a pedigree matrix. This matrix considers a basic uncertainty, reliability, completeness, temporal, geographical and technological correlation of the data. It allows to calculate uncertainty ranges, expressed as the squared geometric standard deviation (variance, σ^2) of a lognormal distributed population, for the different input parameters. After the description of the data quality, we performed a Monte Carlo analysis, in which a random variable is taken for each input data for which distribution and variance was specified. The statistical parameters of this analysis and the 95% confidence interval of the LCA results (called uncertainty range) are presented in S1 in Table A10 and Figs. A1–A5 for the different LCA results. The Monte Carlo analysis is also used to describe the uncertainty of the scenario comparison. For each scenario the sampling is repeated 1000 times and for each run the difference between two scenarios (x-y) is calculated. Based on this comparison a share can be calculated how often scenario x < or > y. A probability in the range of 95%–100% or, inversely, 0–5% that one of the values is higher or lower is interpreted as a significant difference.

In addition to the uncertainty related to data, other sources of uncertainty can be identified for LCA studies, for example, uncertainty related to the use of models. In this study, the substitution model, more specifically, the scenarios on the future application of compost are particularly uncertain. Therefore, we conduct a sensitivity analyses for this parameter and study the effect of alternative substitution scenario on the LCA results (see Section 3.3).

In the sensitivity analysis, we performed two sensitivity tests in which we varied the parameters that have shown a high contribution on results and are based on an uncertain assumption/scenario.

In the first **sensitivity test (A)** we changed the scenarios on the future application of compost which are highly uncertain. We created an alternative scenario for all treatment facilities, except for home composting for which another application than the use in gardens is unlikely. The new scenario assumes that 33.3% of produced compost are used as fertilizer in agriculture, 33.3% is used by professionals in parks and gardens and 33.3% is used in private and community gardens. We maintain the repartition of functionalities (77% soil conditioner and 23% growth media) and the user-specific factors (0.2 for private use, 0.5 for professional use). The calculation of the new substitution potential for compost from each facility is given in SM2-D.

In the second **sensitivity test (B)** we used the characterization factor of hard coal as a proxy for peat as proposed in the egalitarian version of the ReCiPe model (Huijbregts et al., 2016). The results are presented in Section 3.3.

3. Results and discussion

3.1. LCA results for individual processes

Figs. 4–6 show the LCA results for the different management options related to the treatment of 1 Mg food, green or biowaste. The endpoint results indicate environmental impacts on human health (HH) in DALY, ecosystems (ES) in potentially disappeared species per year and resources (R) in USD. The figures show the contribution of processes to the total impact, such as the contribution of collection/transport, infrastructure, process inputs, and direct emissions from the waste treatment process. The figures show positive values indicating environmental impacts, negative values indicating environmental credits and the net balance which is the sum of impacts and credits. The absolute results for the different waste treatment processes cannot be compared directly, because they refer to different waste fractions with different compositions.

Fig. 4 shows the impacts from the **incineration of food waste**. Impacts on HH and ES are mainly dominated by process inputs, for example by chemicals used in air pollution control (APC) such as sodium hydroxide which has a contribution to HH and ES of 39 and 31%, respectively. Impacts on resource uses are mainly caused by the use of natural gas in the incineration process (41%). In terms of credits, the results show only a small credit for the substitution of gravel by bottom ash. As we study a change in incineration, i.e. the redirection of food waste from the incinerator, there is no energy credit for the incineration facility. In the scenario comparison (Fig. 7) we attribute the energy gain from the redirection of food waste to the alternative scenarios. The net balance shows impacts for the three endpoint categories.

Environmental impacts from the treatment of **food waste with AD** are shown in Fig. 5. For both AD options, impacts on HH and ES are mainly, or to a high share, driven by direct process emissions from the digestion process and the post-composting process. The contribution of direct emissions to HH and ES is between 35 and 42% for AD-export, respectively, and between 41 and 50% for AD-Brussels, respectively. In both AD systems, N_2O emissions, followed by methane and ammonia emissions are the most dominating emissions contributing via GWP to impacts on ES. However, for AD export, fine particulate matter formation from waste collection has the highest contribution to HH impacts, while it is process emissions via GWP that have the highest contribution to HH impacts for AD Brussels.

Resource use is mainly due to fuel consumption during waste collection, with a contribution of 72% for AD-export and between 48 and 56% for AD-Brussels. In all three endpoint categories, credits occur for the avoided production of peat, fertilizer and electricity. The net balance, however, shows only for AD-Brussels a net credit for resource use. This is due to the higher electricity output achieved in this facility compared to AD-export.

The results for the **composting processes** are shown in Fig. 6. They show a significant contribution of direct emissions from the composting

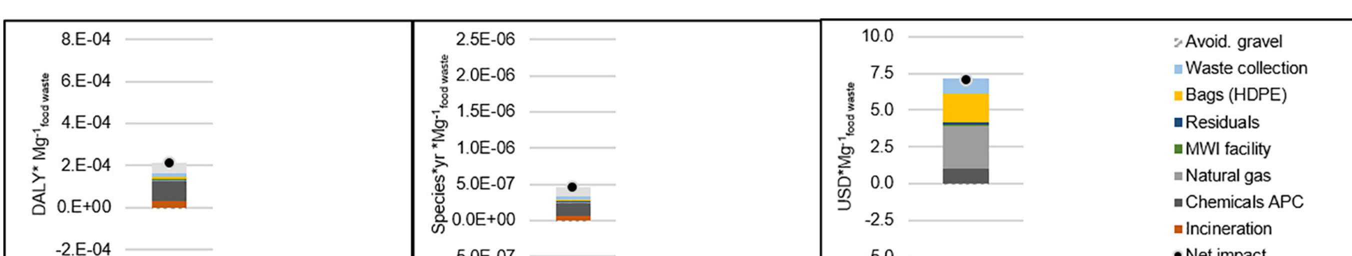

Fig. 4. Environmental impacts from the incineration of food waste. The figure shows the process contribution to the impact categories human health (in DALY), ecosystems (in species.yr) and resource use (in USD) related to the incineration of 1 Mg food waste. The dot represents the net impact.

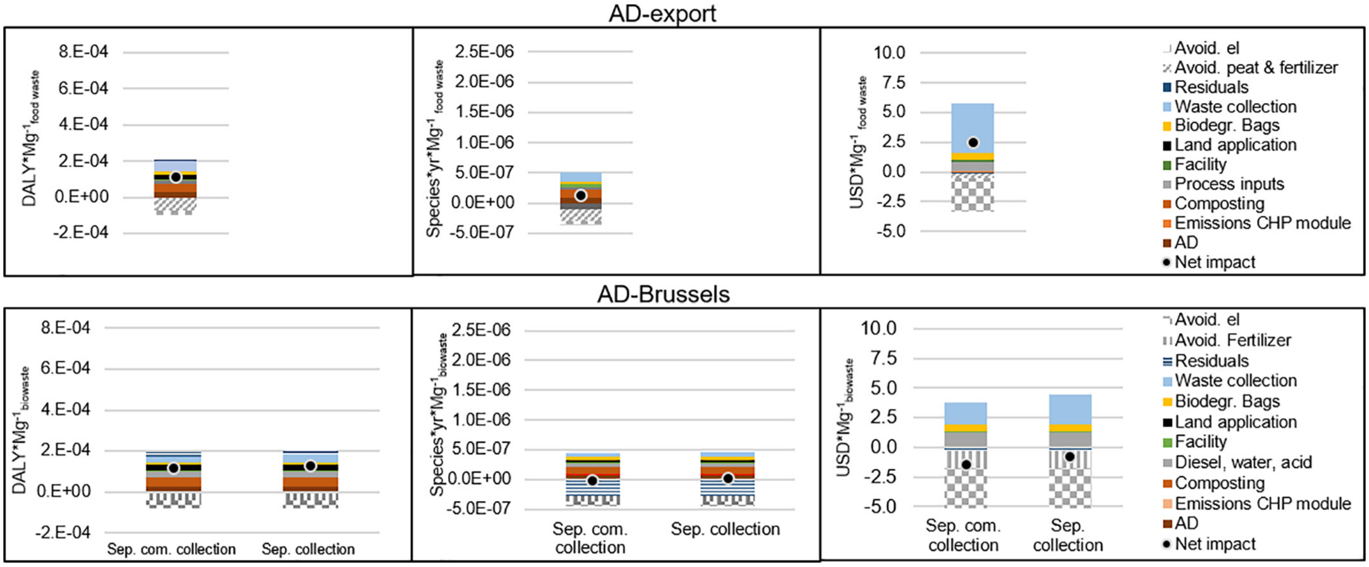

Fig. 5. Environmental impacts from AD. The figure shows the process contribution to the impact categories human health (in DALY), ecosystems (in species.yr) and resource use (in USD) related to the digestion of 1 Mg food waste. The dot represents the net impact.

process for the impact categories HH and ES. However, the contribution can be highly variable depending on the waste input composition, the type of composting system and the presence of a biofilter. For example, the closed tunnel composting system (co-composting) equipped with a biofilter shows a contribution of direct process emissions between 25 and 35%, respectively, while it is 81 to 87% for the home composting system. Furthermore, not only the relative contribution of process emissions is variable, but also the composition of emissions and accordingly the environmental impacts that lead to damages on HH and ES: In the in-situ composting system NH_3 emissions are the most dominating emissions contributing via particulate matter formation to impacts on HH and via terrestrial acidification to impacts on ES. In the other composting systems methane is the most important process emission which contributes via GWP to impacts on HH and ES.

In terms of resource use, the industrial systems show high contributions from the consumption of fossil fuels: a contribution of 47% from waste collection in the green waste composting system and 64% for diesel and electricity use in the industrial co-composting facility. The decentralized composting systems have low to zero fossil fuel inputs and accordingly low contributions.

Environmental credits occur for the avoided production of peat and fertilizer. The high substitution potential of compost from green waste composting in the impact category HH and ES is due to the comparatively high compost yield and compost use in applications that lead to avoided CO_2 emissions from the degradation of peat. Peat substitution does not lead to high credits in the category 'resource use', because peat is not included in the endpoint modelling of resource use in ReCiPe (hierarchist). Thus, only the compost with fertilizer application shows credits in this category. The net balance shows for all endpoints net impacts, but it may be close to zero, for example, for resource use in the decentralized systems.

Summarizing the results for the existing biowaste management systems (incineration, AD-export, green waste composting), we find net impacts for all three endpoints. We also find significant contributions of direct emissions (HH & ES) and of waste collection (R) for the biological treatments (green waste composting & AD-export). The future biowaste management systems which are all biological treatments, represent diverse composting types and systems, from small to large scale. We find for all of them net impacts, except for AD-Brussels that shows a net credit in the category resource use. For small scale systems (home composting and in-situ composting) HH and ES impacts are mainly driven by direct process emissions due to the absence of biofilters. The industrial systems have high contributions from the high demand of process inputs (energy & chemicals).

3.2. Scenario comparison

3.2.1. Overview

Fig. 7 shows the results of the scenario comparison for the three endpoint impacts HH, ES and resource use. The results represent the impacts per FU, so they refer to the total amounts of waste treated in the scenarios. They consider the total amounts of biowaste per waste treatment option and the impact intensities for each waste treatment option as shown in Section 3.1. The amount of totally managed biowaste is balanced in the scenarios (total: 50,000 Mg biowaste, composed of 30,000 Mg of food and 20,000 Mg of green waste), so that the absolute values can be compared.

Regarding **human health impacts**, the results for the local CE scenarios are higher (or similar high) than the baseline and the export scenario (S0). However, only the differences between S3-Baseline, S3-S0 and S1-S0 are significant. Regarding the impacts on **ES**, scenario S2 (AD-Brussels) shows significantly less impacts than the baseline and the S0 scenario. S3 scores significantly higher and S1 is situated between baseline and S0, not indicating a significant difference. The results for **resource use**, however, show a different picture: The decentralized systems (S3) show significantly less impacts than the baseline and the export scenario S0 (around 58% less compared to baseline). Scenario 2 shows even negative results, i.e. a net resource credit. S1 shows results situated between the two baseline scenarios with a significant difference only to the baseline. In summary, scenario S2 shows the best environmental performance of the three CE alternatives, but it does not show a significant difference regarding impacts on HH. Scenario 3 has advantages compared to baseline in terms of resource use, but shows significantly higher impacts in the other categories. Taking the data uncertainty into account, we cannot determine significant differences in environmental impacts of S1 compared to the baseline scenario. The quantitative results of the uncertainty analysis are given in section 5 of the supplementary material. The differences between the S1/S2/S3 are all significant.

With this scenario comparison we add to the existing literature a case study in an urban context that evaluates the environmental consequences of a change towards more local and circular biowaste management. The scenarios represent biowaste management options that can be realistically implemented in the short term. This is why they

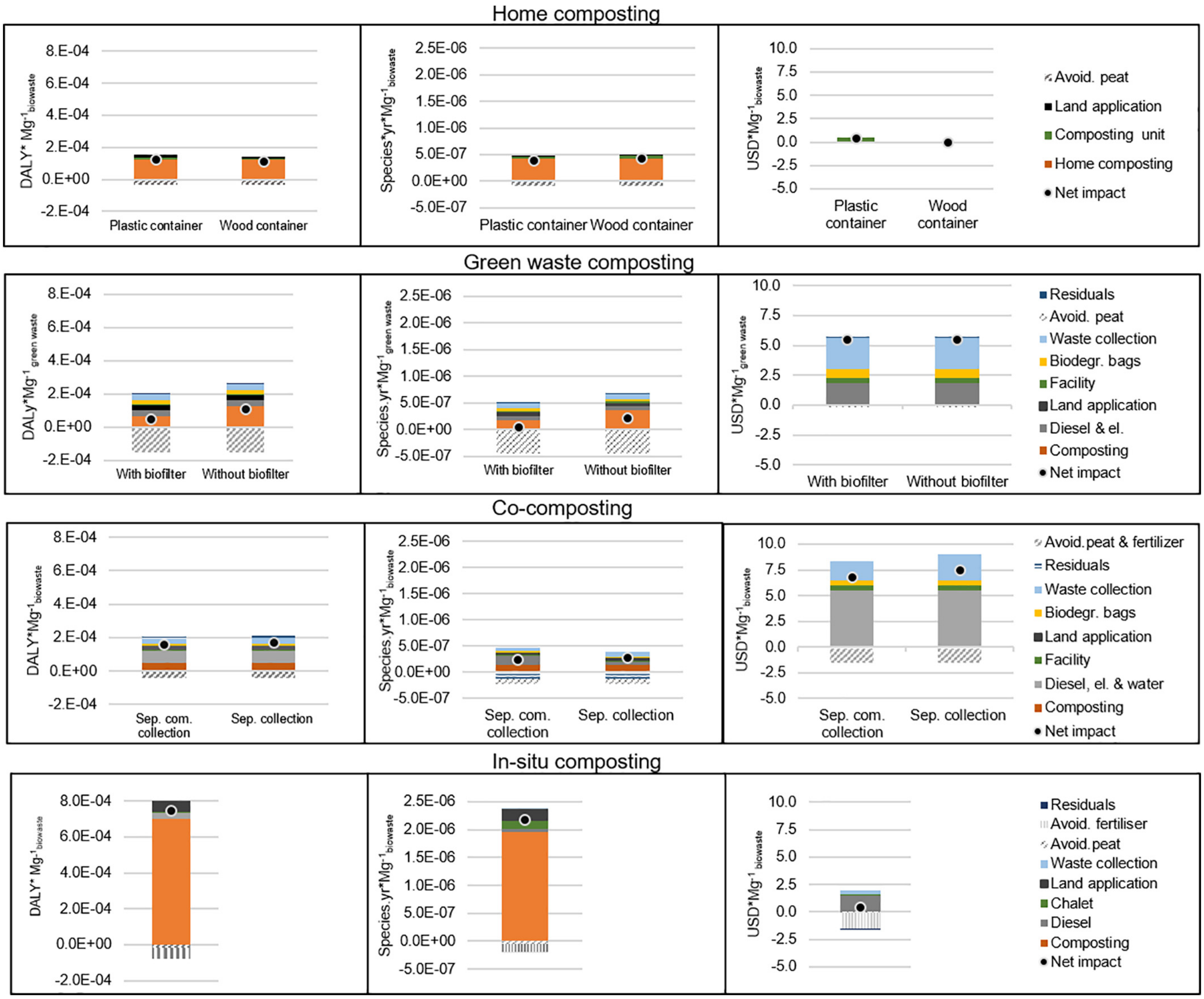

Fig. 6. Environmental impacts from biowaste composting. The figure shows the process contribution to the impact categories human health (in DALY), ecosystems (in species.yr) and resource use (in USD) related to the composting of 1 Mg biowaste. The dot represents the net impact.

represent a moderate change in flows and improvement of the CE performance (max. 30% of biowaste diverted to CE options). Expressed as CI and related to the share of flows that is studied here, it represents an improvement from a local recycling rate of 0.4 in the baseline and S0 to 0.94 in S3 and 1 in S1 and S2.

The CE scenarios represent the combined management of green and food waste fractions which is a relevant, but so far under-researched, management option for cities that want to improve CE performances. Thus, with this analysis we add a relevant comparison between different stand-alone composting systems (in S1 and S3), AD coupled with post-composting (S2) and less circular waste management systems in which green and food waste flows are handled in different systems and food waste is partially incinerated.

A direct comparison of the absolute values from this damage assessment with other study results is difficult due to the different impact assessment method that are applied and biowaste fractions that are studied. Regarding the principal ranking of options, the present study is in line with results in Bernstad and la Cour Jansen (2011) and Lombardi et al. (2015) that showed better performances of AD compared to composting. Bernstad and la Cour Jansen (2011) found advantages for biological treatments in terms of GWP, but disadvantages due to higher NH_3 emissions (acidification) and nitrate run-off from clay soils (nutrient enrichment). Although we also used the land on use model for clay soils and identified relatively high NH_3 levels for some of the biological treatments, acidification and eutrophication did not turn out to be the most contributing impacts regarding its damage potential on ecosystems. Based on the ReCiPe damage model, the ecosystem impacts for S1 and S3 are driven by GWP, fine particulate matter and water consumption. Regarding the role of incineration, the present study results are consistent with Thomsen et al. (2017) and Naroznova et al. (2016) that showed an overall energy gain when biowaste is diverted from the existing systems towards AD.

Regarding the performances of decentralized versus centralized management options, different conclusions can be found in the existing LCA literature. This study is in line with the results found in Bernstad and la Cour Jansen (2011), and Martínez-Blanco et al. (2010) that showed disadvantages for decentralized systems, more precisely impacts related to ammonia, methane and nitrous oxide released from home composting systems.

The scenario comparison also adds to the existing literature a quantitative comparison between a local- (S1-S3) and an export-oriented system (S0). The overall performances of these systems depend on the type of biowaste treatment and the collection and transport requirements. The

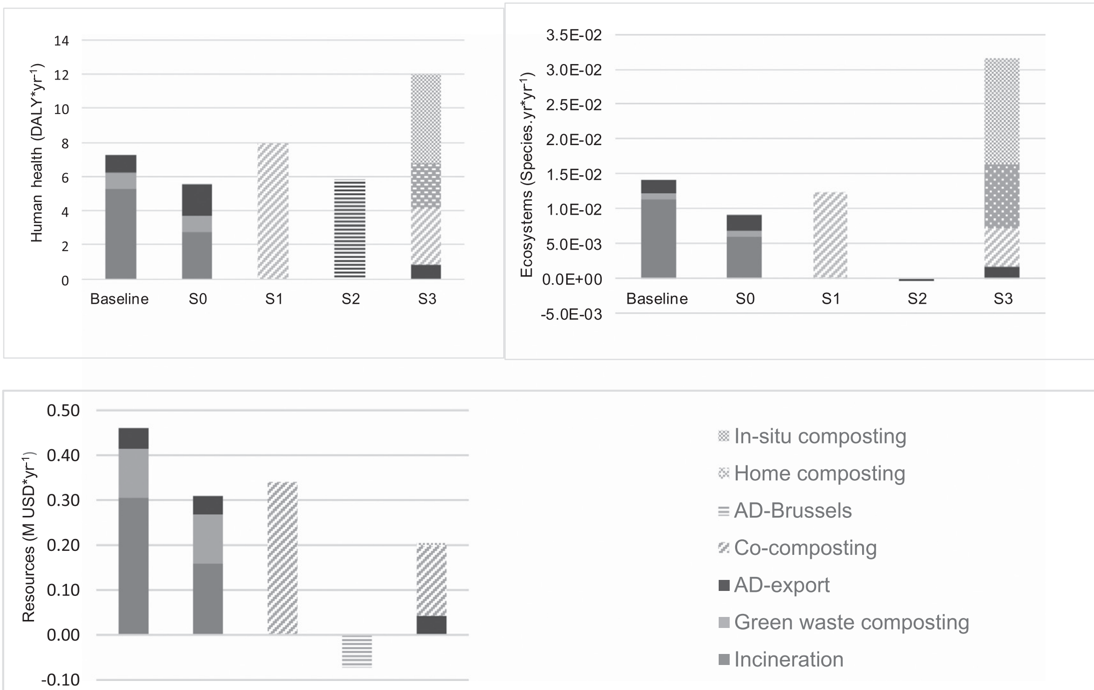

Fig. 7. Comparative LCA results. The figure shows the comparison of the scenarios in terms of human health (in DALY), ecosystems (in species.yr) and resource use (in USD). These net impacts are related to the management of 50,000 Mg of biowaste.

latter was analyzed in this study based on real life data and in an integrated and dynamic way, i.e. considering the effects on the MSW stream when a certain amount is redirected to a new collection and treatment system. Although the local scenarios have better performances for this LC stage, the improvement in transport cannot offset possible trade-offs from the waste treatment. Therefore, the local management systems do not show a systematic advantage in terms of resource use.

3.2.2. Results per LC phase

In order to analyze trends between the different scenarios and to understand their environmental implications, the results are further subdivided in Tables 9 and 10 and discussed in this section. We present as examples the detailed results for resource use and GWP. GWP was selected because it the most contributing impact category to HH and ES (for most composting systems and for AD-Brussels). Furthermore, it is the only impact category that can be easily compared with other studies. The additional results for HH and ES are given in SM1 (Tables A8, A9). We will discuss three major trends: (i) the change in the collection system including the change from export to local management for AD, (ii) the change in the treatment system and (iii) the change in the management of by-products.

Regarding the **waste collection systems**, we analyzed a trend towards more separate collection and more local management in the CE scenarios (S1–3). The results confirm that this change has

Table 9
Composition of the impact 'resource use' (in 1000 USD*yr^{-1}) in the different scenarios. The highest impacts (or lowest credits, respectively) are marked in grey. The lowest impacts (or highest credits, respectively) are marked in green.

		Collection		Treatment	Management of by-products					
		Collection	Bags		Peat	Fert.	El.	Res. Treat.	Other	Total
Baseline	Incineration	26	49	229	0	0	0	-3	3	305
	Green waste comp.	53	14	46	-4	0	0	0	0	109
	AD-export	55	3	5	0	-1	-14	-1	0	46
	Total	134	66	280	-5	-1	-14	-3	4	460
S0	Incineration	14	26	119	0	0	0	-1	2	159
	Green waste comp.	53	14	46	-4	0	0	0	0	109
	AD-export	71	10	17	-1	-5	-49	-2	0	41
	Total	138	50	182	-6	-5	-49	-3	2	309
S1	Co-compost. T1.	92	26	299	0	-69	0	-7	0	340
	Co-compost., T2	124	26	299	0	-69	0	-7	0	372
S2	AD, T1	92	30	69	0	-69	-177	-17	0	-72
	AD, T2	124	30	69	0	-69	-177	-17	0	-40
S3	Home composting	0	0	6	-1	0	0	0	0	5
	In-situ composting	2	0	11	0	-10	0	0	0	3
	Co-compost., T1	71	9	102	0	-24	0	-3	0	155
	AD-export	48	2	3	0	-1	-9	0	0	42
	Total	120	11	122	-1	-35	-9	-3	0	205

Table 10
Composition of the impact 'GWP' (in Mg CO_2-eq.*yr^{-1}) in the different scenarios. The highest impacts (or lowest credits, respectively) are marked in grey. The lowest impacts (or highest credits, respectively) are marked in green.

		Collection		Treatment	Credits					
		Collection	Bags		Peat	Fert.	El.	Res. Treat.	Other	Total
Baseline	Incineration	184	164	2,043	0	0	0	-24	28	2,395
	Green waste comp.	376	97	1,273	-3,100	0	0	0	85	-1,268
	AD-export	387	20	383	-307	-22	-108	-13	20	359
	Total	947	282	3,699	-3,407	-22	-108	-38	132	1,485
S0	Incineration	97	85	1,062	0	0	0	-13	14	1,246
	Green waste comp.	376	97	1,273	-3,100	0	0	0	85	-1,268
	AD-export	504	69	1,301	-1,044	-75	-366	-45	66	409
	Total	976	251	3,636	-4,144	-75	-366	-58	166	387
S1	Co-compost. T1.	649	178	4,119	-102	-1,025	0	-176	44	3,687
	Co-compost., T2	876	178	4,119	-102	-1,025	0	-176	44	3,914
S2	AD, T1	649	202	4,005	0	-1,024	-1,333	-383	227	2,342
	AD, T2	876	202	4,005	0	-1,024	-1,333	-383	227	2,569
S3	Home composting	0	0	3,121	-735	0	0	0	0	2,386
	In-situ composting	16	0	815	-112	-227	0	-5	229	715
	Co-compost., T1	499	60	1,401	-35	-349	0	-60	15	1,532
	AD-export	338	13	237	-190	-14	-67	-8	12	320
	Total	852	73	5,574	-1,073	-589	-67	-73	256	4,953

Fert. = Fertilizer; El. = Electricity; Res. = Residual treatment; Other = land application and residual treatment; T1 = separate combined collection, T2 = separate collection.

environmental benefits in terms of resource use and GWP: The highest impacts for this LC stage occur for the export scenario S0. This is due to the high share of separately collected food waste transported long distances to facilities outside of Brussels (AD-export). The baseline scenario shows less impacts due to a higher share of joint collection of food and MSW (sent to incineration) which is, in terms of transport requirements, an efficient system. Compared to the export scenario, the new collection systems with more local management (in S1 and S2) cause less impacts. However, it is necessary to switch to a combined separate collection where food waste and green waste are collected in the same trucks to achieve a reduction of 34%. The lowest impacts for transport are achieved in the decentralized scenario in which transport by truck can be avoided completely for some treatments (home composting) combined with local waste treatment.

Regarding the **treatment of biowaste**, we studied a trend towards more diverse biological treatments and a reduction of food waste incineration in the CE scenarios. When comparing the results only for this LC phase, we do not observe clear environmental benefits related to this change: We find highest impacts for S1 in terms of resource use due to the high process inputs and for S3 in terms of GWP due to direct process emissions. This LC stage includes direct process emissions, process inputs and infrastructure related to the treatment of biowaste and does not consider potential credits from the by-products created during waste treatment.

Regarding the **management of by-products**, we analyzed a trend from the current, market-driven sales of compost towards a more circular management where compost from the industrial systems is brought back to agriculture to close nutrient cycles and to improve soil quality. However, this trend did not show advantages in terms of GWP (and neither in terms of HH and ES). Our results indicate that more environmental credits could be achieved when compost is used in applications that substitute peat, as it is the case in the baseline and the S0 scenarios. In addition to compost, electricity is the other important by-product that can strongly influence the results, in this case on resource use. Our results show the best performance for the application with maximal electricity generation (AD, S2). The electricity output depends on the efficiency of the systems and the achieved biomethane yields. In this study we calculated 60 and 71m^3 biomethane * $Mg^{-1}_{food\ waste}$. This value lies in the upper range of values from comparable studies (29–74 m^3 CH_4* Mg^{-1}_{waste}, mean: 50 m^3 CH_4* Mg^{-1}_{waste} (Colón et al., 2015; Jensen et al., 2016; Ardolino et al., 2018; Jensen et al., 2017).

This study confirms the special importance of the modelling of substitutions in waste management studies as analyzed in Bernstad Saraiva Schott et al. (2016) and Vadenbo et al. (2017). Through the scope of this study on combined management systems and AD options with post-composting, we analyzed a situation where electricity, fertilizer and peat substitutions occur simultaneously. From the detailed results in Tables 9 and 10 (as well as A8 and A9) we find that peat substitution is more important than the often discussed fertilizer and energy substitution, at least regarding the contributing to GWP and HH impacts. This is in line with the results in Boldrin et al. (2009) who found that the use of compost achieves the highest savings for GWP as a substitute for peat in the production of growth media. Regarding its potential for 'resource use', the damage assessment in ReCiPe (hierarchist) cannot provide complete results, since peat is not included in this category. In the sensitivity analysis (see Section 3.3) we provide a comparison with a proxy for peat as applied in the egalitarian version of the ReCiPe model to study effects on resource use.

When discussing the **net results per scenario**, we find for resource use the best performance for the option with separate combined collection and local AD treatment, due to lowest resource use during waste treatment and highest credits through electricity provision (AD, S2). Surprisingly, in terms of GWP, the export scenario with separate green and food waste collection, partially local and exterior treatment and market-oriented use of compost and electricity provision shows the best results. This is mainly due to the highest substitution potential that occurs in this scenario.

3.3. Sensitivity

The purpose of this sensitivity analysis was to test the robustness of the scenario comparison and to analyze whether the main trends are maintained if sensitive parameters are changed. However, not all of them could be studied in a quantitative way and are therefore discussed qualitatively in the limitations (Section 3.4).

The comparison of the results from the sensitivity and the initial calculation is shown in Table 11. The first **sensitivity test (A)** studied the effect of an alternative and more uniform substitution scenario for compost. For this scenario we observe an increase of HH and ES impacts in the baseline and S0 scenario (mean of +23% and +41%, respectively) and a decrease of impacts from the CE scenarios in HH and ES (mean of −8% and − 40%, respectively). This is due to the fact that peat substitution is strongly reduced in the baseline and S0, while it is increased in the CE scenarios.

In **sensitivity test (B)** a proxy characterization factor is used to evaluate peat as a resource and to study the effect of a new substitution

Table 11
Comparison of the results from the sensitivity with the original results.

		Baseline	S0	S1	S2	S3
HH	DALY*yr^{-1}	7.281	5.572	7.927	5.832	11.993
HH (A: Substitution scenario)	DALY*yr^{-1}	8.664	7.120	6.896	5.276	11.623
ES	Species.yr*yr^{-1}	0.014	0.009	0.012	−0.001	0.032
ES (A: Substitution scenario)	Species.yr*yr^{-1}	0.018	0.014	0.008	−0.003	0.030
Res.	M US$*yr^{-1}	0.460	0.309	0.340	−0.072	0.205
Res. (A: Substitution scenario)	M US$*yr^{-1}	0.450	0.297	0.382	−0.029	0.223
Res. (B: Proxy peat)	M US$*yr^{-1}	0.331	0.144	0.319	−0.113	0.155
Res. (A& B, Substitution scenario & proxy peat)	M US$*yr^{-1}	0.393	0.213	0.262	−0.150	0.131

scenario in terms of resource use. Thus, the comparison from sensitivity test (B) is the most relevant for resource use (comparison between 'Res. (A& B, Substitution scenario & proxy peat)' to 'Res. (B: Proxy peat)'). We observe an increase of resource use in the baseline and S0 scenario (+19%, +48%) and a decrease of impacts from the CE scenarios (between −15 and − 33%).

Thus, the results show a general improvement for the CE scenarios when based on more uniform assumptions regarding the substitution scenario (sensitivity A) and when using an impact assessment method that integrates a proxy characterization factor for peat. Regarding the ranking of options we find that the initial ranking (as shown in Fig. 7) is maintained for most of the analyzed options. Only the position of S1 has improved for HH and ES where it shows now a better performance than the baseline scenario. Thus, with an optimized scenario for the use of compost, also this CE option could be beneficial from environmental point of view. Although improvements for S3 can be reached, the general ranking has not changed.

3.4. Limitations

For the modelling of the substitution potential it was necessary to create a scenario on the future application of by-products, to determine the repartition of functionalities within a certain application and to determine the potential user behavior. The repartition of functionalities and the factors for the potential user behavior used in this study are based on literature values and estimations and have not been empirically assessed in the context of Brussels. Furthermore, the substitution approach for fertilizer based on MFEs has been criticized since it can lead to an overestimation of credits (Hanserud et al., 2018).

The substitution approach considers only the functions of by-products that can be quantified and evaluated in the impact assessment phase. However, as pointed out by Martínez-Blanco et al. (2013), for the 'full assessment of the benefits, apart from nutrient supply and carbon sequestration; additional impact categories—dealing with phosphorus resources, biodiversity, soil losses, and water depletion—may be needed for a comprehensive assessment of compost application'. In addition, a particular limitation for studies that use the ReCiPe damage assessment, is that resource use is incompletely assessed due to the lack of a characterization factor for peat in the damage assessment. This is explained by the lack of production and cost data for peat (Huijbregts et al., 2016).

Regarding the calculation of direct process emissions from waste treatment, the initial physico-chemical waste composition as well as fractional composition and the transfer coefficients are the most determining factors. Due to the consistency and completeness of the database in EASETECH in terms of physico-chemical characterization, we used this database in this study. It is based on the results from Danish residual household waste as published in Götze et al. (2016). The physico-chemical waste composition data of the organic waste fractions that we used in this study refer to the average results measured in this study. As discussed in Götze et al. (2016), the individual samples differed significantly regarding the water content, heating value and nutrient content. Furthermore, for example, the animal-derived food waste showed comparatively high N and P contents. In order to assess, whether the physico-chemical composition for animal-derived food waste is representative for Brussels, more detailed fractional analysis that reflect the consumers' disposal behavior would be necessary. Thus, a limitation of the study is that, for example, N-emissions could be overestimated for the situation in Brussels. However, this limitation does not prevent a fair comparison between the different scenarios, because green and food waste amounts are balanced between the scenarios. The same is true regarding the use on land model that refers to a specific soil type and typical Danish agricultural conditions. Due to differences in environmental conditions and agricultural practices, it is possible that the general emission level is under- or overestimated for Belgium. The ranking of results between the biological treatments is still valid since we use the same use on land model.

Another limitation occurred in the AD model (wet model). Emissions from the treatment of waste water in this facility are modelled with an average waste water treatment process from ecoinvent. Thus it does not consider the specific processes within the facility and neither, the treatment of the salt slurry that is generated. Thus, the environmental impacts (HH and ES) could be underestimated in the baseline scenario.

Also regarding the modelling of the decentralized scenario and processes, some limitations need to be discussed. First, in reality, much more processes are part of the scenario, such as vermicomposting, anaerobic digestion, animal valorization, dehydration or mulching (Bortolotti et al., 2018a). The current scenario considers only home/neighborhood composting and in-situ composting. Both of them are modelled as an 'average' treatment, although compost management could vary greatly in practice. However, emission models that consider such variations could not be found.

4. Conclusions

This research showed the complexity of studying a 'simplified' biowaste management system at city-region level and of determining environmental consequences from changes in the system. With a novel combination of local data, databases and models, we offered an approach to handle this issue. This approach is also relevant for other comparative waste treatment studies that want to take input-specific variations into account. Furthermore, individual model components, such as the substitution model, can be adapted to other biowaste compositions or application scenarios.

To situate properly the lessons learnt from this research, we want to recall that the type of research is a *real case study* to evaluate the environmental impacts from waste management/CE *applications* that can be implemented from a *short term perspective* in an urban context. For this type of research, it was highly beneficial to cooperate with all actors of the waste management chain (operators, authorities and politicians) who are typically involved in such studies. Due to the importance and challenges to design and parametrise substitution scenarios for the by-products, future studies should involve from the beginning additional actors such as the intermediate user (landscaping companies, soil amendment industries) and the end-user of the compost (private garden user or agricultural user).

Another recommendation is to consider the dynamics of policy and to choose an appropriate study design that could involve i) a *modular study design* to calculate new scenario combinations and include prevention and ii) by defining an extended study scope and a corresponding functional unit that covers the entire MSW system.

In line with our study purpose we collected real life data on waste collection and proposed a new strategy to exploit and to embed them in an LCA in order to integrate scale effects and indirect effects on the residual MSW collection. This strategy is relevant for all studies that analyze the redirection of waste fractions from mixed collections and for studies that evaluate changes of the total waste amounts collected and treated, for example, through prevention or changes in consumption behavior.

We also provide new datasets for the management of biowaste in an urban context, covering the combined management of food and green waste in decentral and industrial processes such as AD with co-composting, in-situ composting and home composting. These datasets were exploited and linked with an input specific modelling approach and a detailed substitution model. With this novel combination of local data, databases and input specific modelling, we offer an approach to deal with frequent gaps in case studies (lack of detailed composition and measurements of direct emissions) without being obliged to apply general emission factors and uniform compositions. It allows the researcher to trace elements through all stages of the waste management chain and to develop biowaste specific datasets from multi-input datasets, thus to provide environmental impact profiles for a specific waste fraction. The detailed substitution model increases the accuracy of the modelling by distinguishing the technical substitution potential (application area, functionality within its specific application, substitution potential per functionality) and end-user behavior.

Thus, we move LCA applied to waste management system further to c-LCA with input-specific modelling and contribute to increase the accuracy of results through the detailed substitution model. The high variations in results confirm the necessity to rely on local data and to use detailed substitution approaches, at least in the context of biological treatments and applications where the by-products are assumed to be (re)used on soils. However, we also emphasize the high complexity (and high time requirements to set-up/use such models) and the uncertainty to define scenarios when empirical data on some parameters are lacking. Thus, for studies that have more homogenous biowaste compositions and focus only on the management of separately collected fractions, model simplifications might be possible.

The consideration of by-products and the most accurate and feasible substitution approach is a current discussion subject within the LCA research community with a clear focus on energy and fertilizer substitution. This research contributes to this discussion with a case study that emphasize the importance of the modelling of peat replacements. Also for future applications in the context of circular and bioeconomy, more diverse material applications could become more relevant than energy applications.

With the results from the developed LC biowaste model, we are further contributing to the understanding of the combined management of food and green waste in cities. This option is a relevant, but so far under-researched, management option for cities.

The results have shown that the change towards a more circular or a more local biowaste management does not necessarily result in a better environmental performance, but it can under certain conditions. We found that the industrial co-composting system (with high input requirements which uses compost in agriculture) is not a CE option leading to overall environmental benefits. The decentralized option offers advantages in terms of resource use, but shows the risk of increasing direct process emissions and related impacts. Only the AD scenario provides benefits in two impact categories (ES and resources) and similar results for HH compared to baseline. Thus, we conclude that local systems and a combined treatment of food and green waste can have environmental benefits if process emissions are properly managed, i.e. with closed systems with biofilters, and if by-products with high substitution potentials for electricity, peat and fertilizer are used. In addition, resource use and GWP can be moderately reduced with more efficient collection systems, with the separate combined collection being the most efficient.

The results indicated for GWP and for three endpoint categories that the systematic redirection of compost to agriculture, as part of the CE concept, is less favorable than when used as a replacement for peat in landscaping or in private gardens. Thus, the use of compost in this way should be encouraged, but only if soils can be sustainably managed with alternative organic fertilizers, such as straw or manure.

Finally, we want to highlight that although LCA includes a multi-impact assessment method covering all spheres of the total environment, not all aspects pertinent to an environmental evaluation of biowaste management have been considered. Especially, the benefits of compost application on soils cannot yet be properly assessed.

Thus, additional research is needed to improve existing impact assessment methods, to provide quantitative data on the functionalities of compost in different application and on real life substitution behavior of different user groups. Further research demand exists to cover the variety of decentralized biowaste treatment processes and systems.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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