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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 system. 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.
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.

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Cover Letter

Dear editors and reviewers,

We are very pleased to submit our article '**Assessing the environmental performance of circular economy options for biowaste management at city-region level'** to Science of the Total Environment. We believe that our article fits well with the aims and scope of the journal, because we present an evaluation of impacts from a waste management system on the total environment including potential impacts on human health, ecosystems and resource use.

In our research we performed a life cycle assessment, an approach that includes a multi-criteria impact assessment and covers several spheres of the total environment. In our study, we collected data from the anthroposphere, such as transportation data, process in- and outputs, waste data, etc. We also collected or modelled emission and resource use data including emissions to air (atmosphere), soil (lithosphere) and water (hydrosphere). In the impact assessment phase of our LCA study, we used the inventory of emission and resources to model 'intermediate' environmental impacts such as the global warming potential and evaluated the final damage of these impacts on human health, ecosystems and resources. The applied impact assessment method ReCiPe combines different models such as the IPCC model for global warming potential (atmosphere), or species abundance models to estimate the loss of biodiversity (biosphere).

Our article fits also with the scope of the Special Issue on 'Circular economy and environment with emphasis on waste management & resource valorization'. Based on a full scale case study, we analysed different circular economy options and their potential impact on the environment. Are more circular solutions necessarily the most environmentally preferable options? And which option shows the best environmental performance? We focused on the management of biowaste in cities, where high waste volumes are generated and the potential for recovery of nutrients and energy has not been fully exploited yet.

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.

*Title Page

Assessing the environmental performance of circular economy options for biowaste management at city-region level

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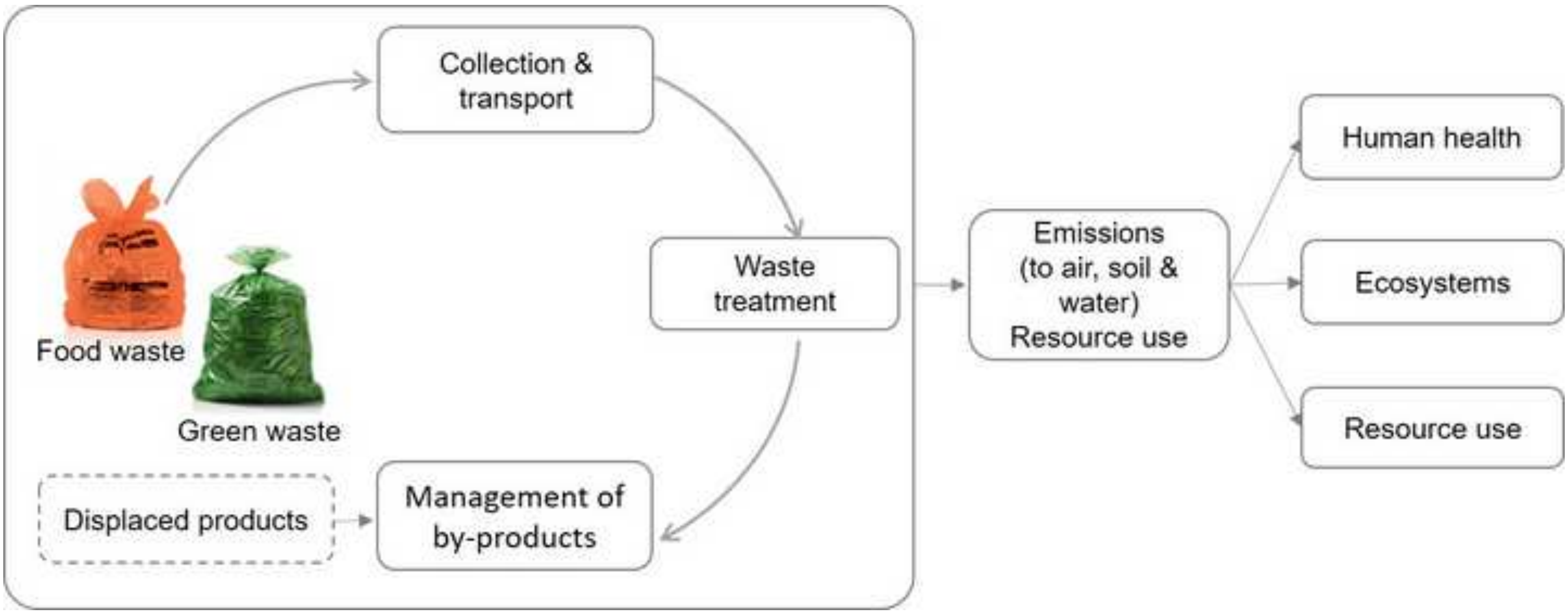
*Graphical Abstract



***Highlights (for review : 3 to 5 bullet points (maximum 85 characters including spaces per bullet point)**

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

Acronyms[1]

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 20-40% 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, for example organized in the circular cities network. 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 a Regional Program for a Circular Economy (PREC 2016) since 2016. It includes a set of transversal, sectorial, territorial and governance measures to support the city's CE transition. In this context, researchers, policy-makers and citizens discuss the following issues: How much waste is exploitable in the future, which type of collection should be introduced, which type of waste treatment facility should be installed and which management system should be prioritized (decentralized/centralized system)? With the installation of new treatment facilities within a city also the by-products such as digestate or compost need to be managed. Following the CE concept for biowaste these organic fertilizers should be used in agriculture to close agricultural nutrient cycles (ISWA 2015). However, in addition to practical barriers that may occur for the use of compost in agriculture (Viaene et al. 2016), also additional transport is required to bring compost to agricultural areas.

Thus, transitions in the biowaste management system require changes in sorting, waste collection and treatment as well as regarding the management of by-products. All these aspects need to be included in an environmental assessment in order to verify whether a certain CE option such as a biological waste treatment with separate collection is actually beneficial compared to a reference system.

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' (Laurent, Bakas, et al. 2014). 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). Also the waste framework directive (WFD) requires LCA to justify possible deviations from the waste hierarchy (EU Directive 2008/98/EC).

In the data collection phase of an LCA, most of the collected data is from the anthroposphere, such as transportation data, land use, process in- and outputs, waste data, etc. Furthermore, emissions and resource use data is collected or modelled, including emissions to air (atmosphere), soil (lithosphere) and water (hydrosphere). For example, to determine emissions to air, soil and water from the application of organic fertilizers on land, hydrological, crop, nitrogen model, and soil organic matter models have been applied (Hansen et al. 2006).

In the impact assessment phase of an LCA study, the inventory of emission and resource use is then used to model environmental impacts at midpoint level, such as global warming, or/and at endpoint level, to evaluate the final damage on human health, ecosystems and resources (Hauschild et al. 2012). Impact assessment methods combine different models such as the IPCC model for global warming potential (atmosphere), or

[1] AD: anaerobic digestion
CE: circular economy
CHP: combined heat and power
c-LCA: consequential life cycle assessment
ES: ecosystem
El: electricity
FU: functional unit
GWP: global warming potential
HC: home composting
HH: human health
LC: life cycle
LCA: life cycle assessment
MFE: mineral fertilizer equivalent
MSW: municipal solid waste
NPK: nitrogen, phosphor, potassium
R: resources
UoL: use on land
WFD: Waste Framework Directive

species abundance models to estimate impacts on ecosystems (biosphere). Thus, by definition, LCA studies are multi-impact studies and cover several spheres of the total environment.

LCA has been extensively used to study solid waste management (Laurent, Bakas, 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). Jensen, Møller, and Scheutz (2016) confirmed this for biowaste management systems. Their case study showed a better performance of incineration in most impact categories, compared to a more circular bioresource management system with combined anaerobic digestion (AD) and composting, and mechanical and biological treatment. Naroznova, Møller, and Scheutz (2016) found that wet biowaste such as animal food waste, kitchen tissue, vegetation waste and dirty paper have a better global warming potential in AD compared to incineration, unless compared to a highly efficient incinerator. Other multi-impact comparative LCAs (Bernstad and la Cour Jansen 2011; Thomsen et al. 2017; Colón et al. 2015) found more favorable environmental performances for circular bioresource systems. The comparison between the biological treatment options (AD and composting) shows often advantages for AD due to less direct emissions and additional energy recovery (Bernstad and la Cour Jansen 2011; Lombardi, Carnevale, and Corti 2015). Studies that analyzed combined AD with composting found better performances than the stand alone technologies (Di Maria and Micale 2015; Jensen, Møller, and Scheutz 2016; Lombardi, Carnevale, and Corti 2015). Regarding the performances of decentralized versus centralized management options, different conclusions can be found. For example, decentralized composting showed higher impacts than centralized AD in the study by Bernstad and la Cour Jansen (2011), whereas the scenario on decentralized AD plants combined with composting plant showed the lowest impacts in Lombardi, Carnevale, and Corti (2015).

Julia Martínez-Blanco et al. (2010) performed a comparative LCA between composting of biowaste at home and a full scale industrial composting facility located in the Barcelona province (Spain). They found that ammonia, methane and nitrous oxide released from home composting (HC) were more than five times higher than those of industrial composting, but the latter involved within 2 and 53 times more inputs for the treatment process and transport. They concluded that HC may be an interesting alternative in low density areas of population.

While the range of industrial biowaste technologies is generally well covered in LCA studies and combinations of organic household waste and agricultural waste (sludge and manure) have been studied, little is known about the performance of the combined management of biowaste fractions that occur in cities, i.e. food or kitchen waste from households, but also from economic activities and the biodegradable waste that occurs in garden and parks. How do more circular and local management systems of these biowaste fractions perform? What is the performance of HC in more densely populated areas?

Thus, the objective of this research is to study the biowaste fractions that are particularly interesting for cities and to evaluate whether a more circular management has actually environmental benefits compared to a reference situation. More specifically, we aim to study different types of separate collection and different types of treatments green waste from urban gardens or parks, and food waste from households and from professional activities. Following the idea of circular management of bioresources, we also focus on the use of by-products such as compost in agriculture.

2. Data and method

2.1. Case study description

The case study is conducted in Brussels, Belgium, a densely populated European city (7,384 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. Here, 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 definitions are (i) garden and park waste, which is summarized and named as 'green waste' in this study and (ii) 'food and kitchen', 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 municipal solid waste (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, 4,300 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

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 models environmental interventions due to a change 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 the biowaste system of Brussels, 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 (Andrea 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 correspond 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. In c-LCA 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). More precisely, the following scenarios are included:

- Baseline 2018 that represents the biowaste management in 2018
- Baseline 2025 that extrapolates current trends biowaste management until 2025
- Scenario 1 (S1) that considers the installation of a co-composting facility in Brussels
- Scenario 2 (S2) that considers the installation of an AD facility in Brussels
- Scenario 3 (S3) that considers a larger implication of local, decentralized initiatives (home & neighborhood composting, a small scale composting type called 'in-situ' composting).

Figure 1 illustrates the study design.

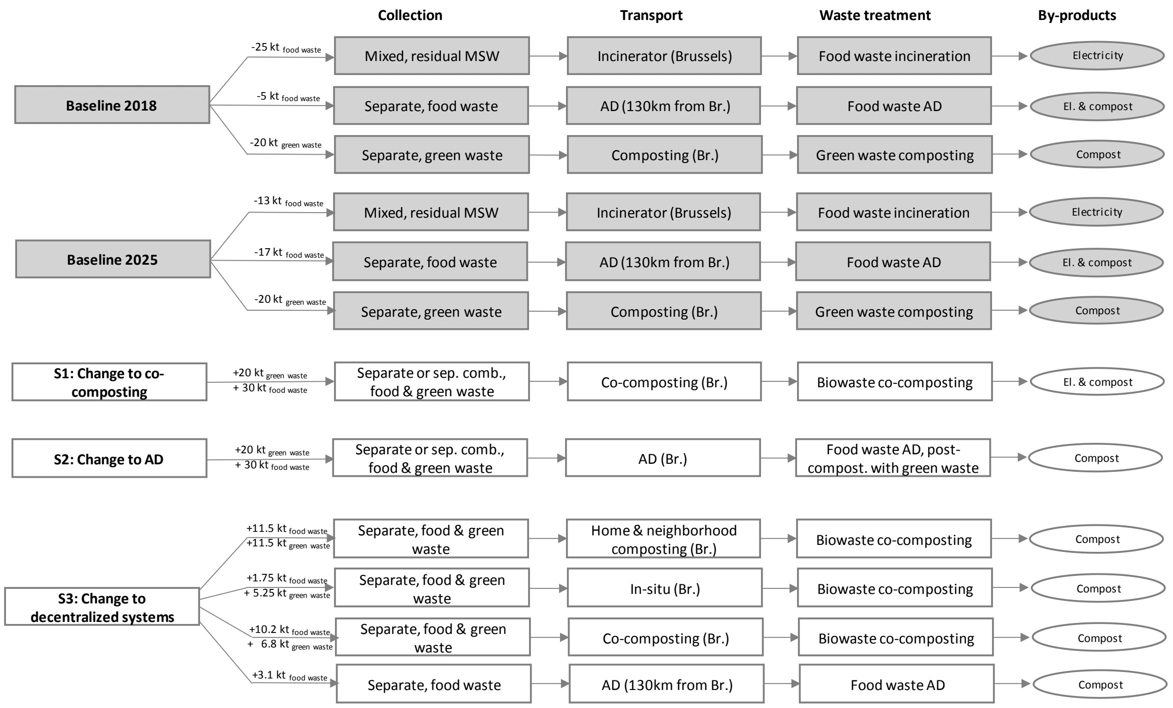


Figure 1: Consequential study design: Flows and treatment indicated with a negative sign and marked in grey illustrate the system that is going to be replaced. (AD= Anaerobic digestion; sep. comb.= separate combined collection; Br= Brussels)

The goal of this study 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 (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 fertilizer and electricity. In c-LCA these by-products are addressed with the substitution approach (Schrijvers, Loubet, and Sonnemann 2016) in which avoided environmental impacts from the production of displaced products are subtracted from the waste treatment system which produced these products as by-products. This principle of granting credits for avoided or displaced products is applied in this study, and illustrated in Figure 2 (dashed boxes).

As shown in Figure 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.

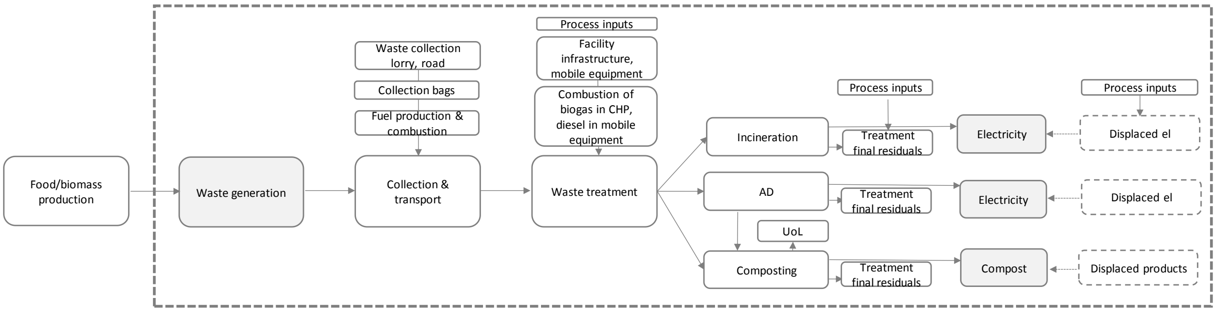


Figure 2: System boundary for the LCA. The figure illustrates which processes are included in the LCA (AD: Anaerobic digestion, Uol: Use on land, CHP: combined heat and power, el: electricity).

2.3. General approach

To estimate LC-based impacts on human health, ecosystems and resources 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 (2.4).

As illustrated in Figure 3, the **LC model** covers waste generation, waste collection and transport, waste treatment, treatment of the final residuals and 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.

LCAs on waste management should be based on local data to capture local specificities of waste management systems (Laurent, Clavreul, et al. 2014). 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 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. In practice, local data collection for waste management systems shows always limitations. In our case study, for example, emission data from decentralized biowaste systems was not available. 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 material flow model characterizes each waste flow as a mix of waste fractions with specific properties and elementary composition, so that substances can be traced throughout the different stages of the waste management chain (Clavreul et al. 2014). As illustrated in Figure 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, Bakas, et al. 2014). In this study we used the framework developed by (Vadenbo, Hellweg, and Astrup 2017) which is specific for substitution effects in waste management systems. Local information on the current use of by-products and market requirements (market data) 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.

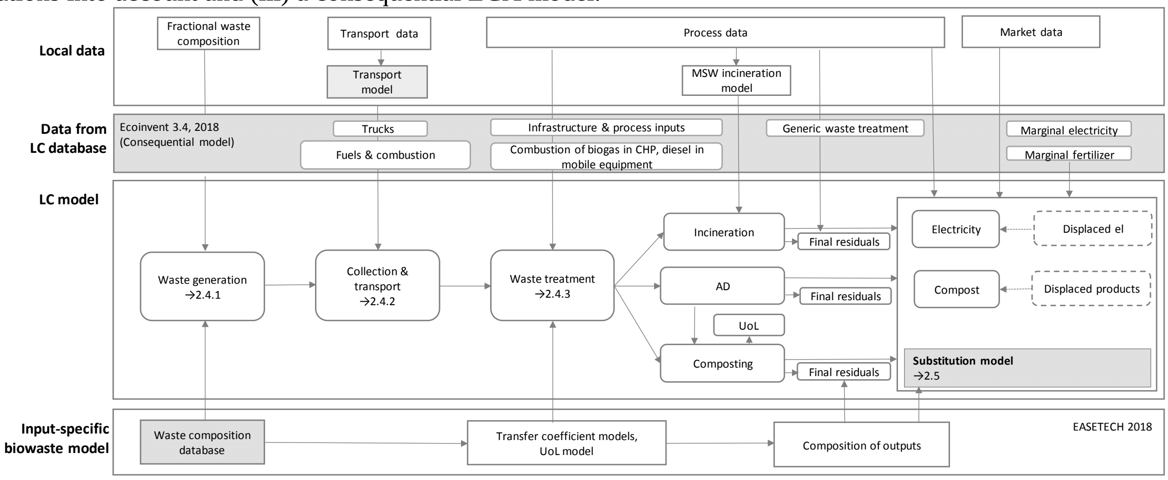


Figure 3: General approach: Data flow and combination of databases to develop the LC model (UoL: Use on land; AD: Anaerobic digestion). Dashed boxes: additional information is available in the supplementary material

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 1. 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 (A. Bortolotti et al. 2018), the share of 'other fractions' was determined 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 Brussels' food and green waste mixes. The fraction 'other' consisting of stones, branches or plastic could not be quantitatively defined. Therefore, the composition is shown without this fraction. The complete physico-chemical composition of the studied biowaste mixes is given in SM1-Table A1.

Table 1: 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	29690.84	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

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 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 scenarios and the scenarios 1 & 2 (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 2 presents the total transportation distance, the collected weight and the km*Mg^{-1} per waste stream in each scenario. The last column in Table 2 clearly shows that all three scenarios bring about a reduction in transport compared to the 2025 baseline scenario. 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.

Table 2: Yearly collected weight, transportation distance and km/ton for each waste stream under the baseline case and the two transportation scenarios

Baseline	Weight per waste stream (Mg)	Total distance per waste stream (km)	Distance per waste stream (km*Mg^{-1})	Total distance per scenario (km)
Baseline 2018 (5,000 Mg)				
Residual waste	340,007	2,034,880	5.98	
Food waste	5,000	419,433	83.89	2,675,941
Green waste	14,500	221,629	15.28	
Baseline 2025 (17,000 Mg)				
Residual waste	328,007	1,982,470	6.04	
Food waste	17,000	766,182	45.07	2,970,281
Green waste	14,500	221,629	15.28	
Scenario 1 & 2				
Option 1				
Residual waste	315,007	1,925,693	6.11	2,395,000
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)	6,800	149,610	22.00	
Food waste (co-composting + AD)	13,300	504,965	37.97	
Food + green waste (in situ)	7,000	87,640	12.52	

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 from the incinerator in Brussels 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, ammonia, 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 emissions that 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 ammonia input used in the DeNOx process and 34% of NOx emissions. This decision is justified by the fact, that these process emissions are driven by the conditions of the process 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 integrated 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, Bhander, and Christensen 2008),(DTU 2018). For example, based on the amount of C_{bio} and C_{fossil} (Table 1) 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 is not possible to link them 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, 4,300 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. After the digestion, the digestate is dewatered and composted with green waste. 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 (A. Bortolotti et al. 2018). The proposed technology is a dry AD process in combination with post-composting of the digestate together with the green waste. 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 3.

Table 3: 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 engines		Stationary CHP modules	
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 existing AD facility is a multi-input process treating multiple feedstock, not only food waste. Therefore, it is not possible to use all data measured in the facility (e.g. biogas and electricity yields). 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, the estimated shares of electricity and heat use (in Table 3) 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 are specified in a feasibility study (A. Bortolotti et al. 2018) and used in this study. The local data to feed the LC model consists of process inputs such as diesel for the mobile equipment, 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 based on 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 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 120m³*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 and lower T. The final biomethane yields are around 42m³*Mg$^{-1}_{biowaste}$ for both facilities which corresponds to 60m^3*Mg$^{-1}_{food\ waste}$ for AD-export and 71m³*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 0.85kg* 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 4. 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 4.

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 exist that treated around 400 Mg of biowaste in 2015 and are expected to increase to around 1,100 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 7,400 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 (A. Bortolotti et al. 2018). 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 (Andrea Bortolotti et al. 2019). For this study, we selected an **'in-situ'** wood chalet system as a representative system. It handles between 25-200 Mg$_{food\ waste}$*yr^{-1}. The food waste is collected from restaurants, canteens and retailers and transported in boxes to the closed-by composting station where it is composted together with wood chips from the 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.

Table 4: 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	6,890
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$^{-1}_{biowaste}$	0.3	0.5	0.31	0.33
Compost density	kg*m^{-3}	705	410	600	716

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 (A. Bortolotti et al. 2018). For the in-situ composting, local data is not available, but basic 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. To model emissions from the composting process the model for biological treatment of organic municipal waste in EASETECH (Boldrin et al. 2011) was used 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 5 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). 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 (J. K. Andersen et al. 2011).

Table 5: Degradation values and emission coefficients for the different composting types

		Home & neighborhood composting	Green waste composting	Co-composting	In-situ composting
Degradation values and emission coefficients					
		Average values for HC for organic waste (J. K. Andersen et al. 2011)	Values for open-air windrow composting, garden waste (Jacob K Andersen et al. 2010; Jacob K. Andersen et al. 2010)	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 Remaining to leaching	0.150	0.014	0.008
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

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 6) 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 (J. 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.

Table 6: 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 GW (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

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, Hellweg, and Astrup 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, Hellweg, and Astrup 2017). It is a function of four determining factors:

$$\gamma = \mathrm{Urec} * \eta * \alpha * \pi, \qquad \text{Equation 1}$$

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 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 Bo P. Weidema, Frees, and Nielsen (1999). In this study, we use the marginal technologies from ecoinvent's consequential system model (B.P. 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 types are shown in Table 7. For the existing composting systems, the application area corresponds to the current use, determined by the facilities. For the future facilities/systems scenarios have been created in line with the initial biowaste management scenarios (Andrea 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 7. We used the results from a survey of Danish hobby gardeners to determine the compost use in Brussels' private and community gardens as well as in parks. These indicated that 77% of compost was used as soil improver and 23% as growth media (Jacob K. Andersen, Christensen, and Scheutz 2010). Regarding the use of compost in agriculture we study only fertilizer use, because all types of produced compost would fall under fertilization legislation, although compost application is considered is applied due to its fertilizing function and soil improvement effects (Viaene et al. 2016) in Belgian

The technical **substitution potential per functionality** is given in part (iii) of Table 7. 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 6) and 1 for K as specified for example in (Boldrin et al. 2010; Jensen, Møller, and Scheutz 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 are 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, Zschokke, and Schleiss 2012). Based on the HEs per type of organic soil conditioner such as compost, straw, peat (Reinhard and Mueller in Dinkel 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 4. 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 refers 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 7, green waste compost has the highest technical substitution potential for peat while compost from the two industrial facilities (co-composting and post composting/ AD) show the highest technical substitution potential for NPK fertilizer.

Vadenbo, Hellweg, and Astrup (2017) highlight the importance to integrate user behavior in substitution models. The survey by Jacob K. Andersen, Christensen, and Scheutz (2010) 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, Schnetzer, and Reinhard (2016) and from the use on land model in Easetech (DTU 2018), documented in SM2.

Table 7: 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%) Soil conditioner			95%	65%	20%	100%
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 ($\gamma^{fertilizer}$)	Min. N ($kg*Mg^{-1}_{biowaste}$)	0.43	0.83	0.59	1.70	0.60	0.56
	Min. P_2O_5 ($kg*Mg^{-1}_{biowaste}$)	0.94	1.02	2.19	2.67	2.23	2.07
	Min. K_2O ($kg*Mg^{-1}_{biowaste}$)	4.30	4.67	3.64	3.31	3.71	3.45
Tech. sub. potential peat ($\gamma^{soil}_{conditioner}$)	$kg_{peat}*Mg^{-1}_{biowaste}$	222.88	354.94	301.03	330.48	228.11	216.40
Tech. sub. potential straw ($\gamma^{soil}_{conditioner}$)	$kg_{straw}*Mg^{-1}_{biowaste}$	206.17	328.32	278.46	305.70	211.00	200.17
Tech. sub. potential peat (γ^{growth}_{media})	$m^3_{peat}*Mg^{-1}_{biowaste}$	0.43	1.22	0.52	0.46	0.58	0.58
Tech. sub. potential peat (γ^{growth}_{media})	$kg_{peat}*Mg^{-1}_{biowaste}$	85.11	243.90	103.33	92.24	116.67	116.67
(iv) Technical substitution potential per compost type							
Min. N fertilizer	$kg*Mg^{-1}_{biowaste}$	0.00	0.00	0.60	1.12	0.13	0.63
Min. P_2O_5 fertilizer	$kg*Mg^{-1}_{biowaste}$	0.00	0.00	2.21	1.75	0.47	2.33
Min. K_2O fertilizer	$kg*Mg^{-1}_{biowaste}$	0.00	0.00	3.67	2.18	0.77	3.87
Peat	$kg*Mg^{-1}_{biowaste}$	191.19	329.40	12.78	96.49	161.98	0.00
(v) User specific substitution potential per compost type							
Peat	$kg*Mg^{-1}_{biowaste}$	38.24	184.55	2.42	19.12	73.10	0.00

Prof.: professional, sub.: substitution

2.5.2 Substitution potential of electricity

For the waste treatment systems that have **electricity as by-product**, the substitution potential for electricity will 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 3). 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 system 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 global warming potential 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 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}.

3. Results and discussion

3.1. LCA results for individual processes

Figure 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.

Figure 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 potential loss of electricity through incineration of food waste in the MSW mix (42%) and the use of natural gas in the incineration process (24%). In terms of credits, the results show only a small credit for the substitution of gravel by bottom ash. Thus, the net balance shows impacts for the three endpoint categories.

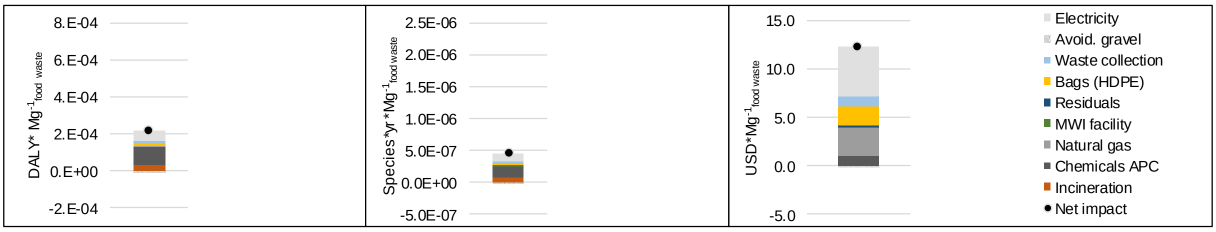


Figure 4: Environmental impacts from the incineration of 1Mg food waste for the impact categories human health (in DALY), ecosystems (in species.yr) and resource use (in USD)

Environmental impacts from the treatment of **food waste with AD** are shown in Figure 5. For both AD options, impacts on HH and ES are mainly driven by direct process emissions such as CH_4 emissions from AD and N_2O, CH_4 and NH_3 emissions from 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. 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. For the latter, this is due to the higher electricity output achieved in this facility compared to AD-export.

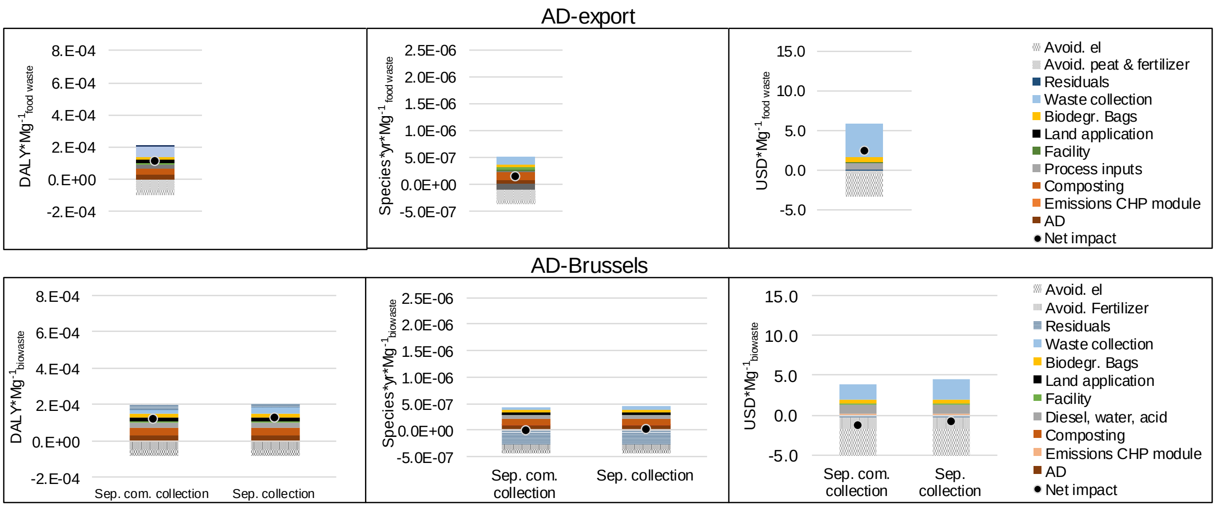


Figure 5: Impacts from AD of 1Mg food waste for the impact categories human health (in DALY), ecosystems (in species.yr) and resource use (in USD)

The results for the **composting processes** are shown in Figure 6. They show a significant contribution of direct emissions from the composting 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 to 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 pacification to impacts on ES. In the other composting systems methane is the most important process emission which contributes via global warming 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. 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.

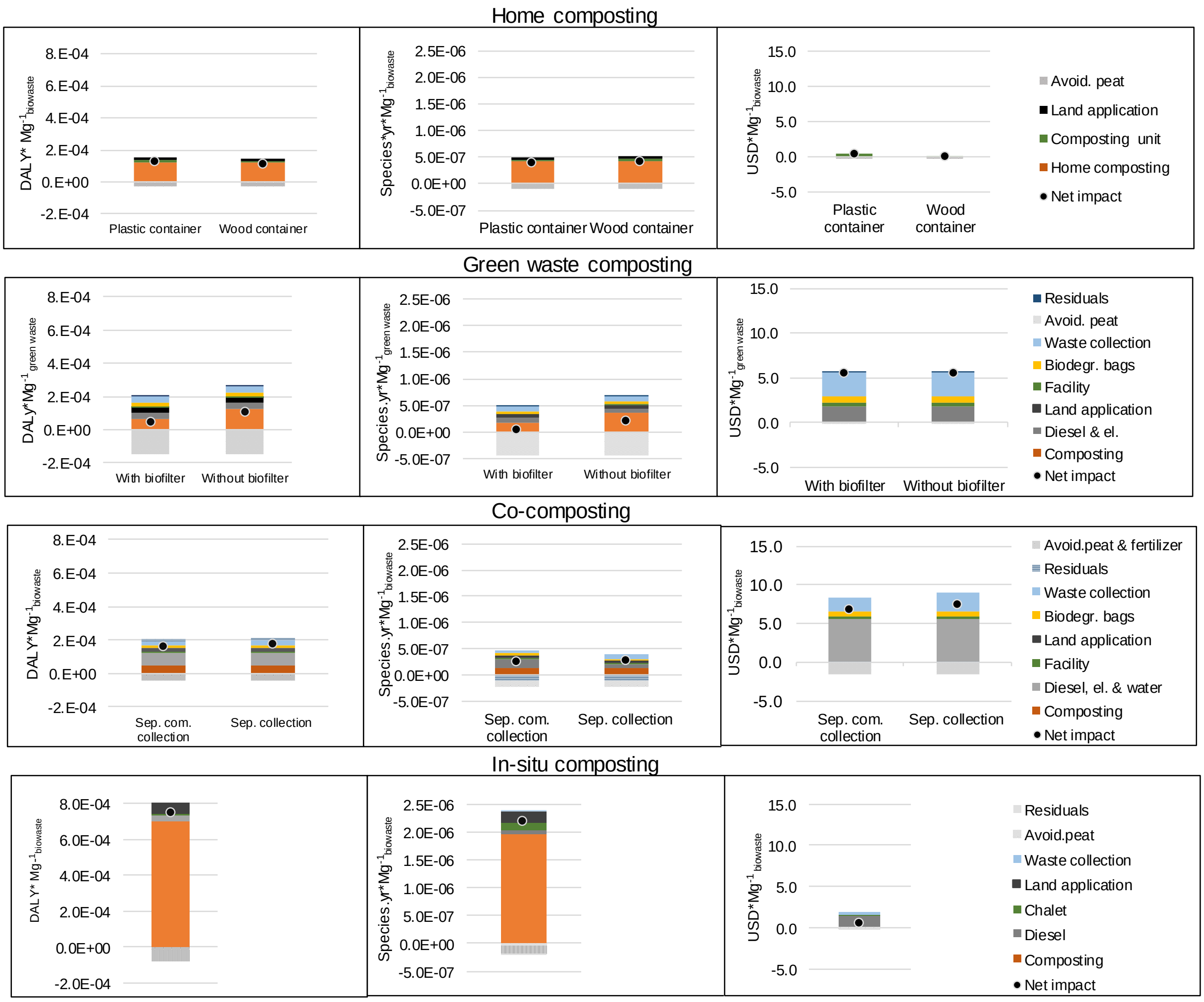


Figure 6: Impacts from the composting of 1Mg biowaste for the impact categories human health (in DALY), ecosystems (in species.yr) and resource use (in USD)

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

3.2 Scenario comparison

3.2.1 Overview

Figure 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 and consider the total amounts of biowaste per waste treatment option and the impact intensities for each waste treatment option. 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.

The results for HH are higher (or similar high) for the CE scenarios than the baseline scenarios. For ES, scenario S2 (AD-Brussels) shows less impacts than the baseline scenarios. S3 scores higher and S1 is situated between baseline 2018 and 2025. The results for resource use, however, show a different picture: The decentralized systems (S3) show significant less impacts than the baseline scenarios (around 58% compared to baseline 2025). Scenario 2 shows even negative results, i.e. a net resource credit. S1 shows results situated between the two baseline scenarios. In summary, scenario S2 shows the best environmental performance from the three CE alternatives, but it does not show benefits for the impacts on HH. Scenario 3 has advantages compared to baseline in terms of resource use, but shows higher impacts in the other categories. Scenario 1 shows higher or similar environmental impacts compared to the baseline scenarios.

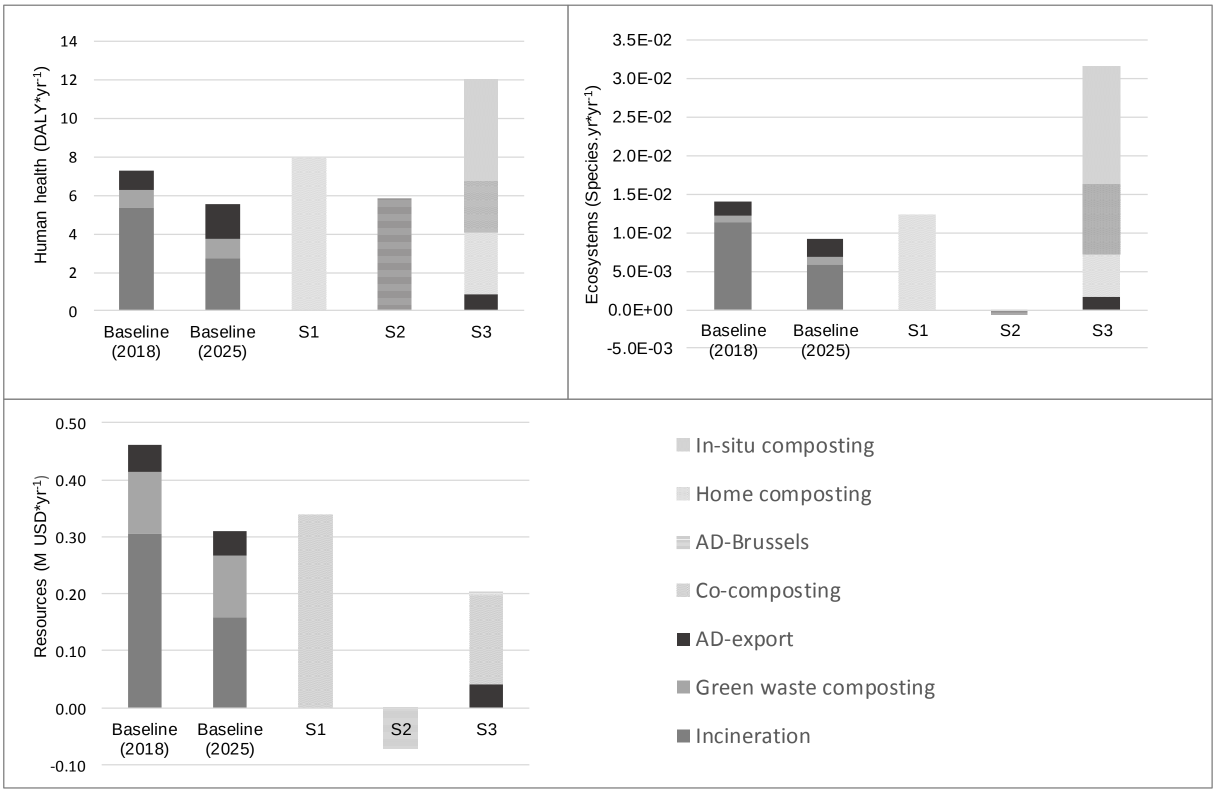


Figure 7: LCA results of the scenario comparison for the three endpoint impacts HH, ES and resource use

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 decomposed in Table 8 and Table 9 and discussed in this section. We present as examples the detailed results for resource use and GWP. GWP was selected because it is important for the impacts for HH and ES and it is the only impact categories that can be easily compared with other studies. The additional results for HH and ES are given in SM1 (Table A8, A9). We will discuss three major trends: (i) the change in the collection system and from export to local management, (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 environmental benefits in terms of resource use and GWP: The highest impacts occur for the baseline scenario 2025. This is due to the high share of separately collected food waste transported long distances to facilities outside of Brussels (AD-export). Baseline 2018 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 baseline 2025, 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 and for S3 in terms of GWP. 'Waste treatment' includes direct process emissions, process inputs and infrastructure and does not consider potential credits from this LC phase. S1 scores high in terms of resource use due to the high process inputs, while S3 scores high in the category GWP due to direct process emissions. However, for the complete performance the by-products of the waste treatment need to be considered.

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 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 scenarios.

Electricity is the other important by-product that can achieve high credits and strongly influence the results on resource use. Our results show the best performance for the application that maximizes 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 71m³ biomethane * $Mg^{-1}_{food\ waste}$. This value lies in the upper range of values from comparable studies (29-74 m³ CH_4* Mg^{-1}_{waste}, mean: 50 m³ CH_4* Mg^{-1}_{waste} (Colón et al. 2015; Jensen, Møller, and Scheutz 2016; Ardolino, Parrillo, and Arena 2018; Jensen et al. 2017).

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 baseline system (2025) 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.

Table 8: Composition of the impact 'resource use' (in 1000 USDyr^{-1}) in the different scenarios. The highest impacts (or lowest credits, respectively) are marked in grey. The lowest impacts (or highest credits, respectively) are market in green.*

		Collection		Treatment	Management of by-products					
		Collection	Bags		Peat	Fert.	El.	Res. Treat.	Other	Total
Baseline 2018	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
Baseline 2025	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 9: Composition of the impact 'GWP' (in Mg CO_2-eq.yr^{-1}) in the different scenario*

		Collection		Treatment	Credits					
		Collection	Bags		Peat	Fert.	El.	Res. Treat.	Other	Total
Baseline 2018	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
Baseline 2025	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

3.3 Limitations

For the modeling 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 overestimation (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, 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 (J. Martínez-Blanco et al. 2013).

For the calculation of direct process emissions from waste treatment, the initial composition and the transfer coefficients are the most determining factors. For some biowaste mixes, the fractional composition is based on literature results or estimations and has not been measured. Also, the fraction 'other' which are process residues (losses) could not be characterized by the facilities or in the feasibility study. Since the losses for AD-Brussels and co-composting are comparably high this adds uncertainties regarding the impacts from the final treatment of residues from these facilities.

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) in the baseline scenario could be underestimated.

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 (Andrea Bortolotti, Kampelmann, and De Muynck 2018). 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.

3.4 Sensitivity

The purpose of this sensitivity analysis is to test whether the main trends from the scenario comparison are maintained if sensitive parameters are changed. In this sensitivity analyses we will vary the parameters that have shown a high contribution and uncertainty. However, not all of the parameters discussed in the limitations can be varied due to data limitations (for example, due to absence of alternative degradation values).

Since the scenarios on the future application of compost are uncertain, we create an alternative scenario for all treatment facilities, except for home composting. The 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 of 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.

The comparison of the results from the sensitivity and the initial calculation is shown in **Error! Reference source not found.**. We observe an increase of impacts from the baseline scenario in HH and ES (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 while it is increased in the CE scenarios. We also observe that resource use has slightly decreased (around -2%) in the baseline while it has increased in the CE scenarios (+20% in average). Resource use is reduced in the baseline scenarios, because fertilizer substitution has increased and is increased in the CE scenarios because fertilizer substitution decreased.

Regarding the ranking of options we find that the initial ranking (as shown in Figure 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 scenarios. In resource use, the position is maintained. 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.

Table 10: Comparison of results from the sensitivity analysis with original results

		Baseline (2018)	Baseline (2025)	S1	S2	S3
HH	DALY*yr^{-1}	7.281	5.572	7.927	5.832	11.993
HH (Sensitivity)	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 (Sensitivity)	Species.yr*yr^{-1}	0.018	0.014	0.008	-0.003	0.030
Res.	M US$*yr^{-1}	0.460	0.309	0.340	-0.072	0.205
Res. (Sensitivity)	M US$*yr^{-1}	0.450	0.297	0.382	-0.029	0.223

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. 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 from the LC biowaste model and scenario analyses are specific to local conditions, but the identified trends per LC stage are valuable in other contexts as well. Furthermore, individual model components, such as the substitution model, can be adapted to other biowaste compositions or application scenarios.

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 an 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.

5. Acknowledgements

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***conflict of Interest Statement**

Declaration of interests

☒ 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.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: