Journal of Material Cycles and Waste Management (2023) 25:3065–3080
https://doi.org/10.1007/s10163-023-01740-6

ORIGINAL ARTICLE

Estimating uncertainties in the life cycle assessment of composting household biowaste and urban green waste in Germany

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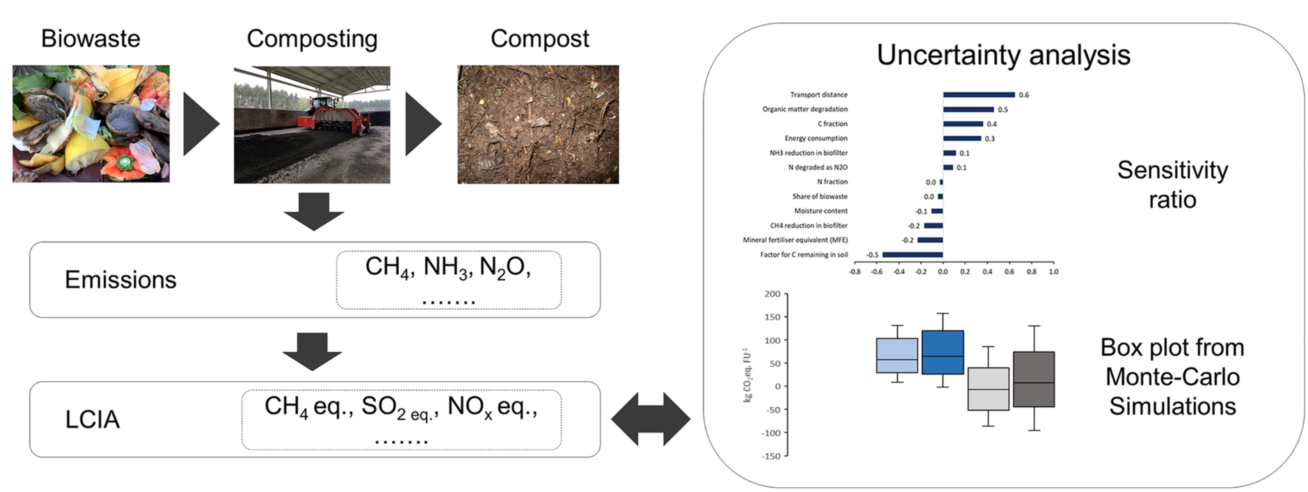
Received: 23 August 2022 / Accepted: 23 June 2023 / Published online: 29 June 2023

Abstract

Life cycle assessment (LCA) of waste treatment processes is often associated with considerable uncertainties. The aim of this study is to estimate the total uncertainty in the modelled composting system and the influence of material and process parameters on the uncertainty. Four composting combinations with fresh (FC) and mature substrate compost (MSC) from partially enclosed (PEC) and open composting (OC) were investigated. Perturbation analysis was used to determine the effect of parameters on the result and Monte Carlo simulation was used to estimate the total uncertainty. This study showed that the production of MSC using PEC had the lowest overall impacts across all impact categories except ozone depletion. Results of the Monte Carlo simulation showed that comparing composting options was challenging. The sensitivity ratios obtained from the perturbation analysis showed that the process parameter percentage of carbon fraction degraded was the most influential for FC. In MSC, the moisture content in the input material and the substitution factor used for peat were the most influential. Monte Carlo simulations demonstrated the overall uncertainty of the model and its relevance when comparing results between combinations. The perturbation analysis identified the parameters that required more accurate data to reduce the uncertainty in the model.

Graphical abstract



Keywords LCA · Monte-Carlo · Perturbation analysis · Uncertainty assessment · Compost

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Introduction

In Germany, composting is a widely used option for treating biowaste from households and municipal green waste [1]. The composition of these waste streams varies across the country by region and is partly influenced by the collection system and seasonal changes [2]. During composting, organic waste is decomposed by microorganisms; the process is influenced by the composition of waste as well as process parameters such as temperature and aeration [3]. It is known that the parameters of the composting process are dynamic and vary with the type of composting, for example, enclosed composting allows better temperature control and aeration than open composting [4]. The composting process and the composition of the input material have an influence on the quantity and composition of the final compost as well as on the emissions produced during composting [5].

Emissions from composting are calculated under defined conditions using models. Due to the complex dynamics of composting, parameters affecting composting need to be understood in detail. Parameters that influence the composting process can be roughly classified into process-related and material-related parameters. Material-related parameters are influenced by the properties of either the input waste material or the compost product obtained, while process-related parameters are influenced by the management practices in the composting process.

The moisture content of the input material is a key material-related parameter affecting the kinetics of biodegradation through oxygen diffusion, microbial growth rates and water activity during composting. Optimal moisture content is required for biodegradation and there is a significant reduction in biodegradation if this parameter is not maintained [6]. For an optimal composting process, the moisture content is around 65% at the beginning of the composting process and is reduced to around 40% at the end [3].

The chemical composition of the input material in terms of C and N concentration plays an important role in the composting process, and the C/N ratio is an indicator to assess its influence on the composting process. Biowaste typically has a lower ratio of almost 20:1 than green-cut waste, which has a ratio of almost 40:1 [1]. However, a C/N ratio between 25 and 30 is desirable to achieve ideal composting conditions. A higher ratio hinders the available N pool to sustain the decomposition process and a lower ratio results in an excess of N required for the microorganisms, which leads to emissions of NH_3 and N_2O [7, 8]. Therefore, biowaste and green-cut waste are usually mixed to obtain a material composition within the ideal limits of the C/N ratio.

In contrast to material-related parameters, process-related parameters can be controlled by management practices. Partially enclosed composting (PEC) and open composting (OC) are the two most common composting technologies [4]. Material degradation during composting occurs through the breakdown of organic material, mostly in the form of water vapor and CO_2. The degradation rate depends on several factors and is logarithmically related to time and is influenced by process parameters such as temperature, aeration rate, and turning intervals [9]. During the decomposition phase, the C present in the input material is released as CO_2 and CH_4 and N as NH_3, N_2O, NO_x, and N_2 [10–12]. The type and rate of C- and N-emissions differ depending on the process parameters. Enhancing the turning frequency leads to a higher quantity of available O_2, lowering the CH_4 release. However, NH_3 and N_2O-emissions slightly increase with a higher turning frequency [7]. A composting temperature above 60 °C in the thermophilic phase leads to an increased formation of CH_4 and NH_3, the formation of the latter is also positively influenced by the pH, while the formation of N_2O takes place at lower temperatures [8]. PEC begins with a phase of intensive decomposition, that usually takes place in an enclosed environment with the exhaust gases treated using a biofilter.

Biofilters are used in enclosed composting systems to reduce NH_3, N_2O, NO_x and CH_4 emissions. Usually, a reduction of 60 to 80% in the concentration of NH_3 is reported [13, 14], and a higher reduction of up to 90% is seen with frequent replacement of filter material [15]. It is assumed that 26% of the reduced NH_3 is converted into N_2O and 74% into NO_x. [16], leading to an increase in N_2O and NO_x levels post biofiltration. The reduction of the CH_4 content in biofilters is only moderate and ranges from 15% according to Amlinger et al. [7] up to 25% according to Trimborn et al. [17].

The complex processes and relationships between the parameters are simplified in LCA models and can therefore be subjected to uncertainties [18]. Uncertainties also arise when there is inherent variability or lack of data accuracy for the parameter values, these types of uncertainties can be included in the analysis by using an uncertainty distribution for the input values. The uncertainties in the input values propagate through the system, and small input uncertainties can lead to large variability in the results [19].

Monte Carlo simulation is an approach that randomly combines values from the input distribution and propagates them through the system using multiple iterations. The simulation results are presented as a distribution and contain information about the total uncertainty in the result of the modelled system [20]. In addition to estimating the total uncertainties from the parameter values, the extent of the influence of the parameters on the uncertainty in the results also had to be examined. Perturbation analysis can be used to estimate the magnitude of the influence of parameters representing input material properties and process parameters on the result [21]. The overall uncertainty in the system can be reduced by reducing the uncertainty of parameters with higher influence [22].

A review by Laurent et al. published in 2014 [23] revealed that less than 50% of the LCA studies performed sensitivity analysis and only 7% assessed the propagation of uncertainty. Uncertainties were mostly based on fixed percentages rather than using distributions to represent the uncertainties of input values [24].

Recognising the need to concentrate further on the uncertainty components of life cycle assessment, the aim of this study is (1) to estimate the overall uncertainty in the modelled composting system using Monte-Carlo simulations, (2) to identify parameters in the model that influence the uncertainty (3) to evaluate the magnitude of influence of the parameters affecting the uncertainty and the change in magnitude according to composting conditions.

Methodology

Investigated composting systems

In Germany, kitchen and garden waste is usually disposed of in the same organic bins and collected weekly by municipal refuse vehicles in urban areas. The containers are emptied at weekly intervals in densely populated areas and at bi-weekly intervals in less populated areas [25]. Whereas green-cut waste from public such as parks and roadsides and larger amounts of private garden waste are delivered separately to the composting facility. An average transport distance of 21.9 km was assumed for the transportation of wastes during the collection and delivery to the composting facility [26].

Upon arrival at the plant, the input material goes through the pre-processing steps of magnetic separation, screening and crushing. The pre-treated waste is mixed together to achieve a textural consistency for an ideal composting process, which requires a moisture content in the 45–65% range [3]. A default mixing ratio of 70% of biowaste and 30% of green-cut by weight was assumed in the modelled system.

Partially enclosed composting (PEC) and open composting (OC) are the two composting technologies analysed in this study. For PEC, the composting process begins with a phase of intensive decomposition that is assumed to take place in an enclosed environment with exhaust gases treated using a biofilter. The input material is laid in windrows and with the help of a turning device. For intensive composting, the material is turned on a weekly basis with an overall composting time of three weeks [12]. The energy consumption for intensive composting is assumed to be 20, 19.3 and 10 kWh/t of input material for forced aeration, exhaust gas treatment and turning mechanism, respectively [13]. After the intensive composting stage, the product is referred to as fresh compost which corresponds to a decomposition degree of 2 or 3 according to the German compost classification [27]. The fresh compost obtained is not completely composted and still contains degradable organic matter. The maturation stage takes place in an open environment over a period of 10 weeks. The turning frequency is every two weeks and is carried out with diesel-powered turning vehicles with an assumed fuel consumption of 0.76 l/t of waste handled. The compost produced after the maturation stage can be classified as mature compost with a decomposition degree of 4 or 5 [27].

For OC, the intensive composting and the maturation stage are assumed to take place in an open environment without exhaust gas treatment. The total decomposition period for the OC system is assumed to be almost 24 weeks, which is longer compared to PEC.

The technology used for the composting process affects the emissions during decomposition and also the final compost products [10]. PEC and OC were the two composting systems investigated. The compost output was also classified into fresh compost (FC) and mature substrate compost (MSC). Therefore, in this study, the environmental impact and the uncertainty of the results for four different technology and product combinations, hereinafter referred to as combinations, were examined:

- FC_PEC: Fresh compost with partially enclosed composting
- FC_OC: Fresh compost with open composting
- MSC_PEC: Mature substrate compost with partially enclosed composting
- MSC_OC: Mature substrate compost with open-enclosed composting

Compost contains plant-available N, P and K nutrients. The use of compost in agriculture and/or horticulture leads to a reduced demand for conventional synthetic fertilisers. Similarly, mature substrate compost can be used as a growing media compound, replacing synthetic fertilisers and peat. The substitution of conventional synthetic fertilisers and peat is carried out using Mineral fertiliser equivalent (MFE) and peat substitution factor, respectively. Therefore, these can be considered material-related parameters that are affected by the characteristics of the compost product.

MFE for N fertilizers typically varies between 0.2 and 0.4 depending on plant availability [28]. While in most studies the MFE for P and K is almost 1, this means that both compost and mineral fertilizer released the same amount of plant-available P and K over the same period [29, 30]. For peat substitution, the bulk density of compost and peat is used to estimate the mass of material needed to fill a specific volume. The densities of peat vary sharply according to the type of peat selected, white peat and black peat are 180 kg/

m^3 and 400 kg/m^3, respectively [31], whereas unspecified peat has a density of 200 kg/m^3 [30]. Likewise, the bulk density is in the range of 600 kg/m^3 for substrate compost based on kitchen waste and 550 kg/m^3 for compost based on garden waste [32].

Life cycle assessment (LCA)

The system is modelled with OpenLCA v.1.10.3 and its embedded features are used to conduct the uncertainty analysis. Datasets for the background processes in the modelled system were from Ecoinvent 3.7. Hotspots in the system were identified through quantitative analysis of the life cycle stages that contributed most to environmental impacts or credits in the respective impact categories.

Goal and scope of LCA

The goal of the life cycle assessment was to estimate the potential environmental impacts of converting biowaste and green-cut waste using the composting process to produce fresh and mature compost. In this study, the focus was on the uncertainties arising from the modelled composting system. The uncertainties were estimated using methods commonly used to handle uncertainties in waste management systems. The results would enable to identify the parameters in the composting process that contribute towards the uncertainty and also estimate the overall uncertainty arising from the parameters as well as the modelling choices. This study is aimed at LCA practitioners to gain a better understanding of the uncertainties to consider when modelling composting systems and when interpreting the results.

Kitchen and garden waste originating from households and green-cut waste are the two input materials used in this study. The system under study (Fig. 1) started with the transport of biowaste from households and green waste collected in compost bins, which were delivered to composting plants. Emissions from the transport of waste have been included, but emissions resulting from the decomposition of material during storage prior to transport have been excluded. The emissions from the storage of compost after the composting process have been included.

A process-based LCA approach is used in this study, which implies that the functional unit is based on the input material rather than the final compost products [33]. The input and output flows in the system were based on 1000 kg of input material handled in the composting facility.

Life cycle inventory modelling and data collection

The life cycle inventory consists of a compilation of data to quantify the use of resources. The emissions are modelled for each process as described below. The model for both open and partially closed composting was based on literature data on the input properties of the waste material, decomposition and associated emissions during composting. The data collected consisted of values for the respective parameters and the minimum and maximum uncertainty ranges (Tables S1-S4).

The mass balance of the composting process was carried out using transfer coefficients from literature data [3]. The input and output flows and the transfer coefficient used are listed in Table S5. With the mass balance modelling software STAN (Version 2.6), a graphic representation of the material flows was created as Sankey diagrams (Figure S1-S4), and the uncertainties in the modelled mass flow were included in the diagram as standard deviation. In addition, material flows from C and N was created as Sankey diagrams (Figure S1-S4). The transfer of P, K and heavy metals between the sub-processes during composting and the losses to the environment were assumed to be negligible. Data used for estimating the mass balance is shown in Table S6.

Emissions occur at different stages of the composting process; however, the intensity of these emissions varies depending on the composition of the input material and the process parameters at each stage. In this study CH_4, N_2O and

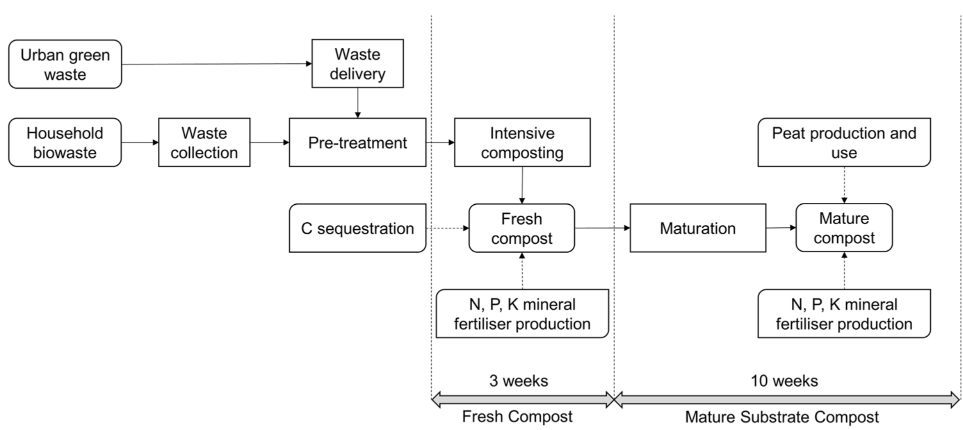

Fig. 1 Schematic representation of the assessed system, the doted lines denote the connection with the materials that are substituted and sources for environmental credits

NH_3 emissions were quantified separately for the composting stages and, due to the limited data availability, emissions of non-methane volatile organic compounds (NMVOC) for the overall composting process was quantified [12]. To estimate the emissions from the composting process, emission factors based on the type of input material (biowaste and green cut waste) and the type of composting process (OC and PEC) were used. The emission factors used for this study are from Cuhls et al. [12]. In addition, there are emissions from the operational activities during the composting process, which are mainly caused by upstream emissions from the energy production processes.

Based on the plant availability of the nutrients, an MFE of 0.2, 1 and 1 for N, P and K was assumed for the substitution of conventional fertilizer [24, 34]. Applying compost to the soil sequesters the carbon present in the compost and provides additional ecosystem benefits such as water retention and reducing soil erosion [35]. In this study, a factor for C sequestration was used based on the % C in the compost applied to the soil, taking into account that C is still bound in the soil after 100 years [34]. C sequestration was considered only for fresh compost because for mature compost the substitution of peat avoids the C released during peat degradation [36].

The inventory for peat [36] was modelled considering the mining and transportation Boldrin et al. [30]. The substitution of peat was done on a 1:1 volumetric basis, bulk densities of the compost (590 kg/m^3) and white peat (200 kg/m^3) were used to estimate the volume.

Impact assessment

In this analysis, the impact categories according to ReCiPe2016 midpoint (H) were used [37]. ReCiPe2016 was used because it is one of the newest LCIA methods that allows for both mid-point and end-point evaluation. Global Warming Potential (GWP), Acidification Potential (AP), Stratospheric Ozone Depletion Potential (ODP) and Ozone Creation Potential (OFP) were selected as relevant impact categories for the composting system in this study. These impact categories were directly affected by the emissions of CH_4, N_2O, NH_3, NO_x and NMVOC from foreground processes during the different stages of composting.

Uncertainty analysis

The uncertainty was analysed using the framework for handling uncertainties in waste management systems proposed by Clavreul et al. [22]. Under the broad term uncertainty analysis, several methods were used to estimate both the sensitivity and the uncertainty in the analysed system. The sensitivity analysis focused on the effect of changing input materials [38] and the uncertainty propagation analysis focused on uncertainty due to the inherent uncertainties in the input. According to Huijbregts [39] uncertainties in LCA can be classified into model uncertainties arising from mathematical relations used to model the system, scenario uncertainties arise as consequences of modelling choices [40] and parameter uncertainties arise due to the inherent uncertainties in each process parameter. The uncertainty values for the material and process parameters used in this study are listed in Table 1. The selection criteria of material and process parameters were mainly based on the relevance of the individual parameter for the C and N balance of the system.

Perturbation analysis

The influence of parameter uncertainties was evaluated using perturbation analysis by determining the effect of a fixed change in the input parameter value on the result [21]. Sensitivity coefficient (SC) and sensitivity ratios (SR) were used to determine the variation in result by incrementally varying the parameter values [22]. The sensitivity co-efficient is the ratio between absolute changes of input values and the result, whereas the sensitivity ratio is the ratio between the relative changes. A sensitivity ratio of 1.5 indicates that a 10% variation in the input values would cause the result to vary by 15%. SR provided an overview of the influence of each parameter on the environmental impact results. Identifying the impact of each parameter can help prioritize the parameters for which additional data needs to be collected.

Uncertainty propagation analysis

At the inventory level, the uncertainty of input values was integrated into the model using appropriate probability distributions [39]. The input values along with their respective distributions were included for computing the results using Monte-Carlo simulations. The result was computed by repeating the calculations 10,000 times using random variables in the distribution. Triangle distribution is used in this study to represent the uncertainty as the values are given as minimum and maximum. A frequency histogram with a probabilistic distribution of the result was thus obtained. There are associations between the system parameters, but these interactions were not estimated for this study. The exclusion of interactions between the parameters for the uncertainty analysis was seen in other LCA studies on waste management [22, 41].

Results

Mass balance

Microorganisms decompose organic waste during the composting process, retaining only $42 \pm 10\%$ of the fresh matter

Table 1 Parameters of the composting process considered for the uncertainty analysis. The uncertainty values are given in parentheses as (min–max)

Parameter		Unit	Value (Min–Max)	References
Moisture content		g/kg	613 (564–674)	[1–4]
Mineral fertiliser equivalent (MFE)		%	0.2 (0.2–0.4)	[5–8]
NH_3 reduction in biofilter		%	80 (60–90)	[1, 9]
CH_4 reduction in biofilter		%	16 (15–25)	[10, 11]
Organic matter degradation		%	57(50–70)	[12]
Peat substitution factor		kg/m^3	565 (508–620)	[13]
C Fraction		g/kg	137 (98–166)	[1–4]
N Fraction		g/kg	4.7 (4–7.2)	[1–4]
C degraded as CH4				[14]
FC_PEC		g/Mg	922 (606–3560)	
FC_OC		g/Mg	1476 (593–4519)	
MSC_PEC		g/Mg	1200 (830–4800)	
MSC_OC		g/Mg	1800 (730–5500)	
N degraded as N2O				[14]
FC_PEC		g/Mg	11 (0.4–57)	
FC_OC		g/Mg	62 (43–150)	
MSC_PEC		g/Mg	53 (2–270)	
MSC_OC		g/Mg	24 (15–57)	
Transport distance		km	21.9 (19,71–24,09)	[15]
Share of biowaste			70:30 (60:40–90:10)	
Energy consumption		MJ	220 (188–233)	[1]
Decomposition emissions				[14]
FC_PEC	NH_3	g/Mg	10 (6–25)	
	NO_x	g/Mg	51 (30–139)	
FC_OC	NH_3	g/Mg	278 (9–1057)	
	NO_x	g/Mg	18 (9–144)	
MSC_PEC	NH_3	g/Mg	23 (16–61)	
	NO_x	g/Mg	66 (45–280)	
	NMVOC	g/Mg	140 (100–750)	
MSC_OC	NH_3	g/Mg	370 (12–1400)	
	NO_x	g/Mg	85 (34–173)	
	NMVOC	g/Mg	370 (190–690)	

(FM) in FC_PEC and 45 ± 11% in FC_OC. Moisture content made up nearly 43% of the FM for FC and 33% for MSC. The remaining material was released into the atmosphere as gaseous C- and N-emissions as well as water vapor. However, for MSC, only 30 ± 7% of FM remained in the compost for both OC and PEC. PEC and OC underwent different degrees of decomposition during intensive composting and maturation, which was evident in the higher FM loss of 48 ± 12% for PEC compared to 44 ± 12% for OC.

43 ± 8% of C in the input material for OC and PEC was released into the exhaust gases during the composting process. For PEC, 3 ± 4% of the initial nitrogen was released in the exhaust gases, but for OC, 9 ± 14% was released. It is evident that OC causes higher reactive N-emissions than PEC. More than 90% of nitrogen and 56 ± 8% of carbon were still present in the compost, and only very small amounts of nitrogen and carbon were lost in the leachate.

Hotspot analysis/contribution analysis

The results of the life cycle impact assessment for the combinations examined in this study are displayed in Table 2. MSC PEC was found to have the lowest overall effect across all impact categories except ODP. PEC had lower GWP and AP than OC. Among the impact categories evaluated in this study, the results for GWP are examined in detail.

Three sources accounted for the majority of GWP emissions in composting systems as shown in Fig. 2. The composting process was the main source of emissions, accounting for 60% and 70% of all emissions for FC and MSC, respectively. Looking more closely at MSC with two composting stages, intensive composting had accounted for almost two-thirds of the emissions from the composting process. The collection and distribution of bio-waste and green-cut waste was the second largest source of emissions,

Table 2 Overview of the environmental impacts for each process step in the composting process for the impact categories GWP, AP, OFP and ODP. The total value is the combination of both emissions and credits of the respective combinations

	GWP (kg CO_2 eq.)				AP (kg SO_2 eq.)			
	FC_PEC	FC_OC	MSC_PEC	MSC_OC	FC_PEC	FC_OC	MSC_PEC	MSC_OC
Emissions								
Collection	22.4	22.4	22.4	22.4	0.07	0.07	0.07	0.07
Pre-treatment	2.7	2.3	2.7	2.3	0.01	0.01	0.01	0.01
Intensive composting	43.3	52.5	43.0	52.5	0.06	0.56	0.06	0.56
Maturation	0.0	0.0	20.9	23.2	0.00	0.00	0.04	0.19
Storage	1.9	2.6	2.9	3.9	0.00	0.01	0.00	0.04
Credits								
N-fertiliser replacement	− 8.1	− 7.7	− 8.0	− 7.5	0.03	0.03	0.03	0.03
P-fertiliser replacement	− 3.1	− 3.1	− 3.1	− 3.1	− 0.03	− 0.03	− 0.03	− 0.03
K-fertiliser replacement	− 5.6	− 5.6	− 5.6	− 5.6	0.02	0.02	0.02	0.02
C sequestration	− 18.9	− 21.5			0.00	0.00		
Peat replacement			− 112.5	− 115.2			-0.03	-0.03
Total	34.5	42.0	− 37.3	− 27.0	0.16	0.67	0.16	0.86
	OFP (kg NO_x eq.)				ODP (kg CFC11 eq.)			
	FC_PEC	FC_OC	MSC_PEC	MSC_OC	FC_PEC	FC_OC	MSC_PEC	MSC_OC
Emissions								
Collection	0.15	0.15	0.15	0.15	2E−05	2E−05	2E−05	2E−05
Pre-treatment	0.00	0.01	0.00	0.01	2E−06	3E−06	2E−06	3E−06
Intensive composting	0.09	0.03	0.09	0.03	3E−04	1E−04	3E−04	1E−04
Maturation	0.00	0.00	0.02	0.10	0E+00	0E+00	4E−04	4E−04
Storage	0.00	0.00	0.01	0.01	1E−05	6E−06	3E−05	3E−05
Credits								
N-fertiliser replacement	− 0.02	− 0.02	− 0.02	− 0.02	−2E−04	−2E−04	−2E−04	−2E−04
P-fertiliser replacement	− 0.01	− 0.01	− 0.01	− 0.01	−1E−06	−1E−06	−1E−06	−1E−06
K-fertiliser replacement	− 0.01	− 0.01	− 0.01	− 0.01	−8E−05	−8E−05	−8E−05	−8E−05
C sequestration	0.00	0.00			0E+00	0E+00		
Peat replacement			− 0.08	− 0.08			−2E−05	−2E−05
Total	0.20	0.15	0.15	0.18	7E−06	−1E−04	4E−04	3E−04

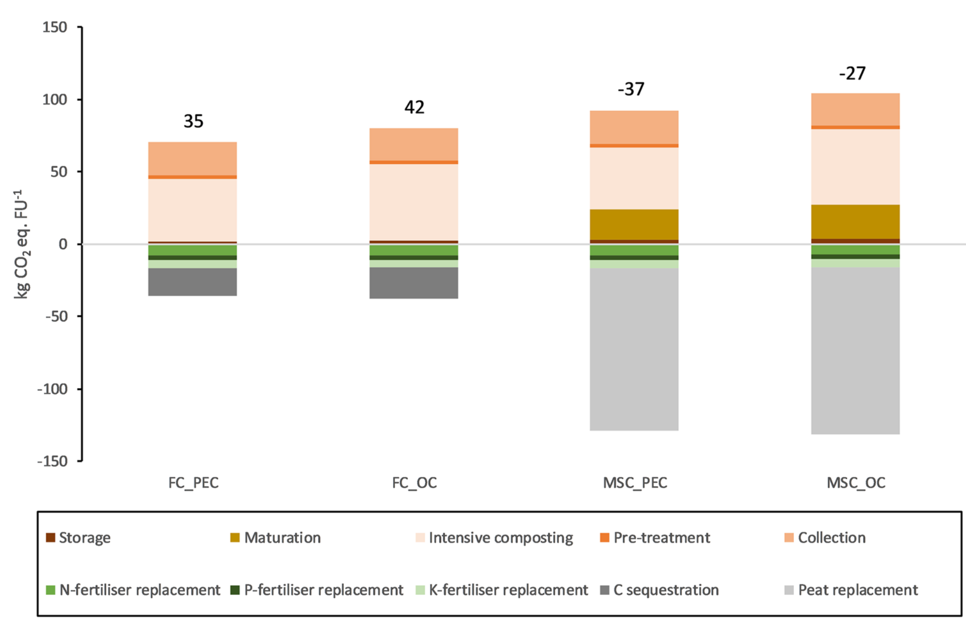

Fig. 2 Contribution analysis of GWP results for four combinations in the study, the vertical bars represent the contribution of each process. The numbers above the bar denote the net GWP impact of the combination

accounting for almost 20–30% of all emissions. Pre-treatment and storage emissions accounted for only 2–4% of all emissions, making them the least significant of the three.

The emissions offset by the substitute products are marked with a negative sign (Table 2). For MSC, peat substitution had the highest emissions offset, accounting for about 80% of the offset emissions. Similarly, for FC, the largest emissions offset came from C sequestration during usage, which accounted for over 55% of offset emissions. The replacement of nutrients in the compost with mineral fertilizers was responsible for the remaining credits.

For AP, emissions from intensive composting accounted for almost 87% and 65% of total emissions for FC_OC and MSC_OC, respectively (Table 2). For PEC, emissions from the intensive composting process were considerably less; they made up 44% for FC_PEC and 34% for MSC_PEC.

For the OFP impact category, emissions from collection and transport accounted for between 50 and 80% of all emissions expressed as NO_x eq., while emissions from the composting process came second (Table 2). When the composting process was examined closely, the intensive composting stage exhibited higher emissions compared to maturation. Looking at ODP, it was found that PEC had higher emissions than OC for both MSC and FC (Table 2). The maturation stage was the highest contributor with around 57% and 72% of the total emissions for MSC_PEC and MSC_OC respectively.

Uncertainty analysis

Perturbation analysis

The amount of each parameter's influence, expressed by sensitivity ratios (SRs), varied between the four combinations. The results in Fig. 3 represent the SR for each parameter in terms of the impact category GWP. For MSC, mineral fertiliser equivalent (MFE) had the highest negative influence with an SR of − 0.2 and − 0.3 for PEC and OC respectively. Similarly, MFE had the highest negative influence for FC_PEC with an SR of − 0.2. Whereas for FC_OC, the moisture content in the input material had the highest negative influence with an SR of -0.6. Looking at the SRs with a positive influence on the parameters, the substitution factor for peat had the strongest influence amongst the parameters for MSC_OC with an SR of 3.9, whereas, for MSC_PEC, this parameter was the second most influential with an SR 2.7. For both FC_OC and FC_PEC, C related emissions had the highest influence, with the parameter C degraded as CH_4 having the strongest positive influence in the range of 1.0. Similarly, C fraction in the input material also had a positive

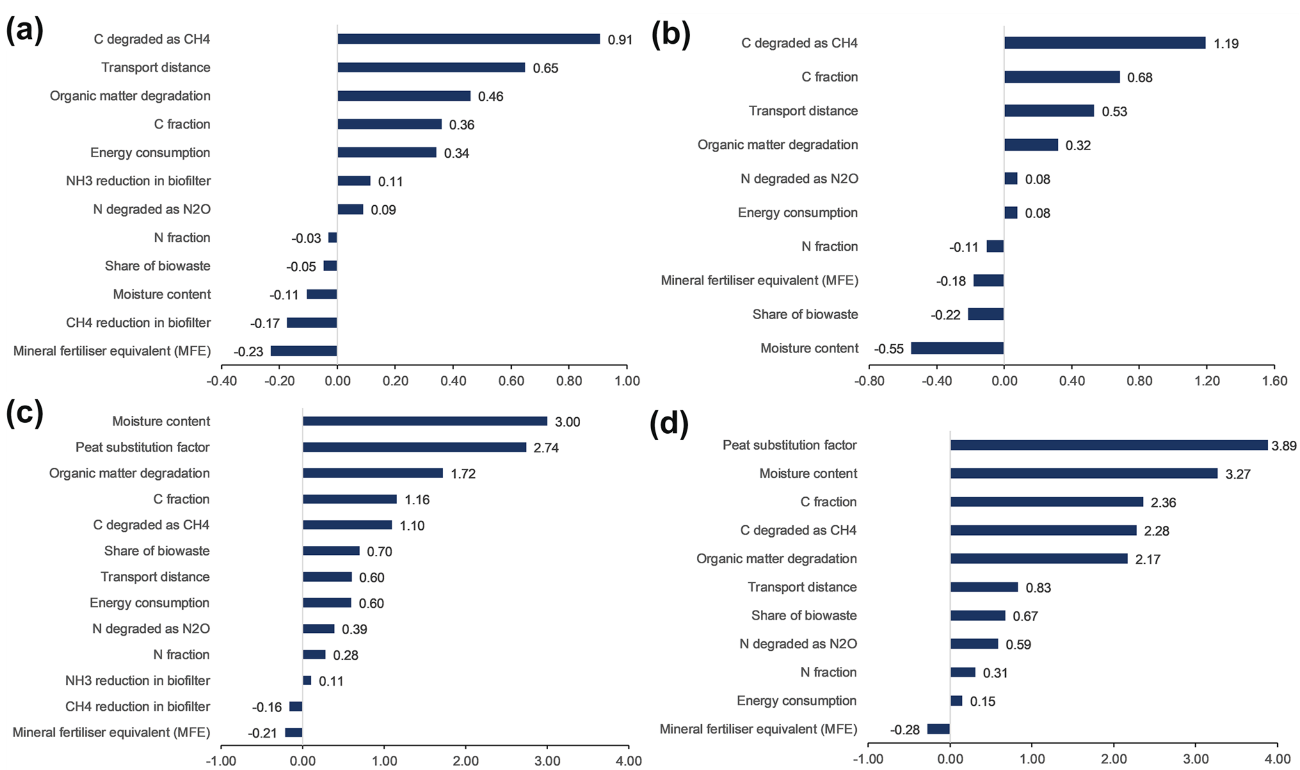

Fig. 3 Sensitivity ratio (SR) estimated using GWP for the assessed parameters in each combination. **a** FC_PEC, **b** FC_OC, **c** MSC_PEC, **d** MSC_OC

influence, and for MSC this influence was stronger compared to other C-related parameters.

The influence of N-related parameters on GWP was lower than C-related parameters. When the effects of the N-related factors were examined in further depth, it became clear that the N fraction in the input material had a negative impact on FC that ranged from − 0.03 to − 0.1. However, N degraded as N_2O had a slight positive influence ranging between 0.1 and 0.4, with a relatively higher positive influence for MSCs compared to FCs. The share of biowaste in the input material was a parameter with a relatively slight influence in comparison to other parameters. The parameter had a positive influence for MSCs with SRs of 0.7, whereas there was a negative influence for FC with an SR of almost -0.2.

The moisture content of the input material had a varying influence on MSC and FC, for MSCs a strong positive influence was seen with an SR 3.0 for PEC and 3.3 for OC. Whereas, for FCs, moisture content had the strongest negative influence of − 0.6 for OC and a slight negative influence of − 0.1 for PEC. The degradation of organic matter was a parameter that had a positive influence across all combinations. Organic matter degradation had a higher influence on PEC compared to OC.

The influence of transportation distance was the second and third highest for FC_PEC and FC_OC, respectively, the influence of this parameter was lower for the MSC compared to FC. The reduction of CH_4 and NH_3 emissions in the biofilters pertained only to PEC, CH_4 reduction had a negative influence in both FC and MSC with an SR in the range of -0.1. However, the reduction of NH_3 in the biofilter had a positive influence on PEC_FC and PEC_MSC. Energy consumption was a slightly influential parameter for PEC, while had little to no influence for OC. Overall, the SRs were higher for the MSC compared to FC. In terms of the positive and negative direction of SR, there was a variation in the parameters between the 4 combinations.

Uncertainty propagation

The magnitude of uncertainty in the results varies between the combinations for all impact categories (Fig. 4). According to the results of the Monte-Carlo simulation, the median value of GWP was up to 50–100% higher than the findings of the contribution analysis. Although comparisons between the combinations could be done using the net values for GWP from the contribution analysis, these comparisons were challenging due to the overlap in boxplots derived from the Monte-Carlo simulation. Compared to OC, the difference between 5 and 95th percentile values was lower for PEC. Likewise, FC showed less difference in percentile scores compared to MSC. From this it can be concluded that FC_PEC had the smallest variation in the result of all four combinations. The results also showed a moderate skewness

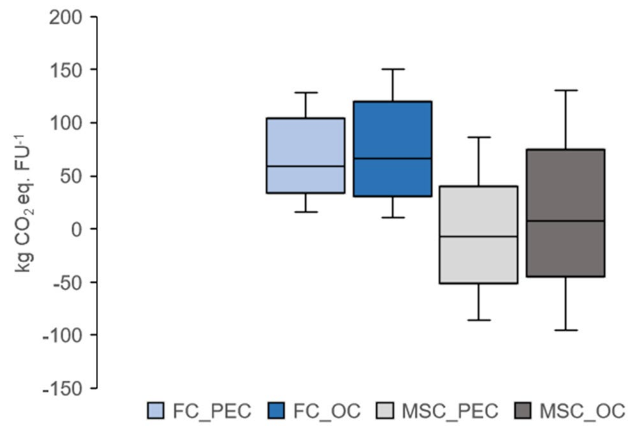

Fig. 4 Boxplots derived from the frequency distribution of Monte-Carlo simulation, the whiskers represent the minimum and maximum values. The boxes represent the 5th and 95th percentile values and the line inside the box represents the median

towards the lower values for FC, this was more evident for PEC.

Monte-Carlo simulation of AP (Figure S8) revealed that the uncertainty was significantly lower for PEC compared to OC, with a standard deviation of almost 0.02 and 0.37 to 0.47 respectively (Table S5). In contrast to the results from the contribution analysis for AP, the results of the Monte-Carlo simulations revealed higher uncertainties. The Monte-Carlo analysis results estimated a maximum value of AP of 2 kg $SO_{2eq.}$ per FU (Figure S8) for OC compared to the results from the contribution analysis of 0.86 kg $SO_{2eq.}$ However, for OFP the results of the Monte-Carlo simulation revealed that the uncertainty was similar for all combinations except FC_OC (Figure S9). The median values for all the combinations were similar to that of the values from the contribution analysis for OFP. The uncertainties in FC were up to 2.5 times lower than MSC for the impact category ODP (Figure S10).

Discussion

Mass balance

The rate of organic matter decomposition in the composting process is influenced by microbial activity, which depends on the process conditions and input material properties such as temperature, oxygen content, moisture, C/N ratio and pore space [4]. For MSC the organic substance degradation was assumed to be 60% for both OC and PEC [12]. However, for PEC the degradation during the intensive phase was higher compared to OC due to forced aeration and higher turning frequency. Intensive composting accounts for nearly 80% of all organic matter degradation using PEC. The amount of compost produced by PEC was almost 9% lower compared to OC due to roughly 16% greater OM-degradation.

During composting almost 96% of C degraded is released as CO_2. The estimation of C degraded was based on the organic matter degradation, which was lower for OC than for PEC [42]. As a result of a lower rate of degradation, FC_OC had a C content that was about 11% higher than FC_PEC. Only between 1–2% of C degraded is as CH_4 and NMVOC. The same amount is lost as dissolved organic carbon (DOC) and end up in the leachate.

The uncertainty associated with the N-transformation was higher compared to C-degradation. Almost 90% of the N-emissions was in the form of NH_3 according to Tiqua et al. [43]. It was found that the NH_3 degradation rate mainly depends on the C/N ratio of the input material. Hellebrand [44] found a degradation rate of up to 8% for materials with a low C/N ratio such as biowaste, while a degradation rate of only about 1% was achieved for garden waste due to the higher C/N ratio. Increased aeration in combination with increasing pH also leads to higher NH_3-emissions[7].

Hotspot analysis/contribution analysis

The proportion of gaseous CO_2, CH_4 and N_2O emissions contributing towards GWP varied between the combinations. The GWP emissions from PECs consisted of 54% CH_4, 24% N_2O and 12% CO_2, while the emissions from OCs consisted of 68% CH_4, 17% N_2O and 14% CO_2. Higher turning frequency and forced aeration used in PEC lead to lower CH_4 emissions compared to that of OC. The biofilter used in PEC further reduces CH_4 emissions by up to 20%; this agrees with Cuhls et al., [12]. Total CH4 emissions from PEC were lower at 1.2 kg/FU than OC at 1.8 kg/FU. However, the overall N_2O emissions were 0.062 kg/FU for PEC compared to 0.053 kg/FU for OC. The reason for higher N_2O emissions for PEC was due to the conversion of NH_3 to N_2O in the biofilter [15].

The contribution of biowaste collection and transport to the total emissions varied from insignificant up to 10% in most LCA studies [35, 45], these values were lower than the contribution estimated in this study. The emissions in this phase were mainly caused by the combustion of diesel and were influenced by the assumed transport distance for collection and delivery. For the stop and go collection vehicles an emission of almost 1.2 kg CO_2 eq. per t.km was estimated according to Doka [46], whereas for the green waste delivery vehicles it was 0.5 kg CO_2 eq. per t.km. The variation in results compared to other studies can be attributed mainly to the transport distance assumed and also the type of transportation vehicle used for collection. The use of a specific data set for stop-and-go vehicles in this study results in more than twice the emissions compared to a normal truck due to the energy-intensive driving technique [47].

Substituting compost with equivalent products reduced the net emissions associated with handling the input material. The negative emissions from peat substitution were due to savings in emissions from the extraction, transportation, and degradation of peat. Almost 85% of the CO_2 emissions from peat occur from the degradation of peat. CO_2 emissions from peat are considered to be fossil CO_2 because carbon accumulation as peat occurs over a long period of time [30]. In comparison, CO_2 released during the usage of compost is considered biogenic and it is not included in the GWP estimation. According to Kranert et al., peat use causes emissions of 621–1197 kg $CO_{2eq.}$ per ton [48].

In this study calcium ammonium nitrate was chosen as a substitute for mineral N-fertiliser; it has a GWP of 8.76 kg $CO_{2eq.}$ per kg N, in comparison urea ammonium nitrate had a GWP of 6.08 kg CO_2eq., hence selecting the latter would have lowered the negative emissions. In the case of N fertiliser, the MFE chosen also played a role in determining the N availability which influenced the amount of negative emissions. Differentiation was also present between the two compost types, mature compost (with low C/N ratio) had a slightly higher plant N availability, whereas for fresh compost (with high C/N ratio) there was higher N immobilization [49, 50].

For the impact category AP, the presence of biofilter was beneficial as there was a reduction in the amount of NH_3 emissions for PEC. In the biofilter, NH_3 was reduced by almost 80%. Of this, almost 75% was converted to NO_x, which also contributed to AP. The lower characterisation factor for NO_x compared to NH_3 was responsible for the lower AP results for the PEC scenarios. Transportation-related NO_x accounted for nearly half of AP emissions for PEC, whereas for OC it accounted for only 10%. For the Ozone formation impact category in addition to NO_x emissions NMVOC also played a role, the main sources for NO_x emissions were waste collection and also the emissions during the composting process. An NMVOC of 140 mg/ tonne and 370 mg/ tonne of NMVOC was estimated by Cuhls et al., [12] for PEC and OC, respectively.

Uncertainty analysis

Sensitivity analysis

In this section, the parameters considered for the sensitivity analysis and the reasons for the respective SR obtained are discussed. In compost, N is organically bound and the amount of N immediately available to plants is limited. The N availability of organic fertilizers compared to mineral fertilizers can be represented by MFE, which is assumed to be 20% for organic compost [51]. A higher MFE leads to a higher replacement quantity of mineral fertilizer and thus to a higher credit for the avoided emissions. The level of emissions avoided also influences the value of the SR; in this study, calcium ammonium nitrate has a GWP of almost 9 kg $CO_{2\text{-eq.}}$ per kg N and was used as a substitute mineral fertilizer. Hence, in this case, the mineral fertiliser chosen for the replacement of N is also a determinative factor for SR. The substitution factor for N present in compost varied sharply based on the time scale chosen for the study. For the short term, the substitution factor for N content in compost was found to be 5–15% effective compared to mineral fertiliser according to Stadtmüller [52] and Amlinger [53], 20–30% according to Hansen et al. [28] and 10% according to EPEA [54].

The amount of C from the compost bound in the soil depends on the climatic and topographical conditions of the area under consideration. For fresh composting scenarios, the amount of C bound in the soil is the most influential parameter in the negative direction. The selection of a factor to represent the C bound to the soil is hence a potential source of uncertainty, a general factor of 7% was used in this study. Percentages of C bound to soil in the long-term ranged from 2 to 10% according to Fisher [55], 9–14% according to Bruun et al. [56] and 11% according to Diacono and Montemurro [57].

The substitution factor for peat was estimated using the bulk density of both materials and it was found that to fill a defined volume, 2.9 times the quantity of mature compost was needed compared to peat. Peat substitution offsets the highest emissions for mature compost scenarios due to the emissions associated with peat production. According to Boldrin et al. [30] this was estimated to be 986 kg $CO_{2\text{eq.}}$ per tonne of peat produced. Since the substitution factor for peat is affected by a variation in the bulk density of the compost produced, the bulk density contributes significantly to influence the sensitivity of the modelled system.

During the composting process, more than 50% of the C present in the input material was released into the atmosphere mainly as CO_2, whereas C degraded into CH_4 accounted for 1–3%. Despite having a lower proportion of degraded carbon than CO_2, CH_4 had a considerable impact on the GWP emissions. CH_4 was responsible for almost 94% of GWP emissions from the intensive composting and 50% of GWP emissions from the maturation process for OC. For PEC the share of CH_4 in intensive composting emissions was 80% and the same as OC for maturation. Based on two parameters; the percentage of C degraded into CH_4 and the proportion of C in the input material, CH_4 emissions were calculated. An increase in either of the two parameters had an influence on the emission of CH_4. Hence, from SR it can be seen that the influence of C fraction is comparatively higher than C degraded as CH_4 for mature compost.

In a modelled system, the uncertainty involving the fraction of C degraded as CH_4 is difficult to estimate since several factors influence this parameter [3]. The main parameters influencing CH_4 formation were aeration and turning frequency, for well-managed systems C degraded as CH_4 was reported to be low [58]. However, the large particle size of the materials causes anaerobic pockets to form in the early stages of composting. Therefore, a significant portion of CH_4 emissions, ranging from 75 to 90% of the total CH_4 emissions during composting, often occur in the first few weeks [7]. The formation of CH_4 due to the anaerobic pockets can be mitigated with a higher turning frequency and forced aeration as they contribute to the breakdown of these pockets and the degradation of C mainly takes place as CO_2. Lower CH_4 emission during PEC compared to OC can be attributed to the higher turning frequency and the presence of forced aeration systems.

The relationship between the N fraction in input material and N released as N_2O is similar to the previously mentioned relation between the C fraction and C degraded as CH_4. Looking closely at the parameter N degraded as N_2O, we can see that it predominantly influenced N_2O emission during composting but had a limited influence on N remaining in the compost. The limited influence was mainly because the share of N degraded as N_2O from the total N degraded was less than 10%. In the modelled system, the higher N proportion in the input material causes an increase in N in the compost product in addition to having an impact on the N_2O emissions. The higher N content in compost resulted in higher N_2O emissions, which were counterbalanced by the substitution of additional mineral fertilizers. Because there were fewer N_2O emissions during the intense composting stage, the effect of N fertiliser substitution on SR in FC is greater than the effect of N_2O emissions. As a result, the SR for the parameter N fraction was negative for FC. Since about 75% of the total N_2O emissions occur during the maturation stage, the impact of N_2O emissions on SR for MSC was considerably greater [12].

The reason for lower N_2O emissions in the intensive composting stages can be attributed to the higher temperatures compared to the remaining composting stages. Usually for a temperature above 40 °C nitrifiers cease to exist, hence hindering the formation of N_2O. Furthermore, a high aeration rate and effective stripping of NH_3 during the intensive composting were also found to hinder N_2O formation [7].

In this study, the dry matter content in the waste material was used to determine the C and N content for biowaste and garden waste with values from literature [13]. The influence of the parameter moisture content on MSC_PEC is the greatest compared to other parameters. The reason for the strong increase in overall impacts despite a reduction in C and N in the input material can be attributed to the lower quantity of end compost produced. The amount of compost produced was reduced by nearly 16% due to a 10% increase in moisture content. Because there was less compost, less peat could be substituted, which reduced the emissions offset from peat substitution by nearly 16%. As previously stated, peat substitution is one of the most influential parameters for mature composting; thus, changes that influence peat substitution have an impact on the SR. The OC had higher emissions from the composting process than the PEC, so the impact of the emissions was greater than the savings from offset emissions. As a result, the SR for moisture content was less influential than the parameters involving the C fraction in OC. According to Clavreul et al. [22], the influence of moisture content on SRs was found to be negative for waste management systems such as incineration and anaerobic digestion. This implied that higher water content resulted in lower GWP impacts; this would have been the case in this study as well if the emissions offset was of lower magnitude or if the offset emissions were not considered.

Lower offset emission influence is visible in the FC, where the SR for moisture content is comparatively low and even negative for PEC and OC. Biowaste and green waste are mixed together in the composting plant to achieve the ideal material consistency for composting. The composition of moisture content, C, and N in the input material mixture changes as the proportion of biowaste or green waste changes. A greater proportion of biowaste increases the moisture and nitrogen content of the input mixture while decreasing the C [13]. Across all combinations, the direction of influence of the parameter share of biowaste was oriented with the parameter N fraction. This implies that, similar to the parameter N fraction, lower N_2O emissions during the intensive composting stage influenced SR for the share of biowaste to remain negative for FC.

It was seen the sensitivity can vary depending on the transport distance since a greater distance covered would lead to higher total emissions, which means that the proportion of transport-related emissions increases. Therefore, when interpreting the results, the influence of the transport distance on the value of SR should be taken into account. The influence of the energy consumption parameter is relatively higher for PEC since PEC is more energy-intensive than OC.

In this study, a CH_4 reduction potential in the biofilter of almost 15% was assumed according to Amlinger et al. [7]. Other studies confirm the high uncertainty of the CH_4-formation in the biofilter; a range between 7 and 27% is mentioned [59]. The oxidation of CH_4 to CO_2 in the biofilter takes place with the help of methanotrophic bacteria in the filter material. In some cases, it has been reported that there is a potential for further formation of CH_4 in the biofilter due to anaerobic pockets and the exact dynamics involved in the effect of NH_3 requires further investigation. Hence, the parameter CH_4 reduction involves uncertainty and this uncertainty should be considered during the interpretation of CH_4-sensitive results.

Uncertainty propagation

The boxplot obtained from the Monte-Carlo simulation reveals the range of uncertainty arising from the input values for each parameter. Although the results for SC and SR were used to interpret the Monte Carlo simulation result, they only provided information on the influence of each parameter on the uncertainty. In order to interpret the result, it was necessary to have information about the total uncertainty in the input values for each parameter. It should also be noted that the result's high minimum and maximum values could be due to a random combination of input values that may not be plausible in a real composting process. The goal of the Monte-Carlo simulation, on the other hand, was to understand the overall uncertainties in the model and to take these uncertainties into account when comparing the results for the combinations.

For MSC, the most influential parameters were found to be the peat substitution factor and moisture content in the input material. However, taking the variability of input values into consideration, it was seen that the relative standard deviation (RSD) for peat and the moisture was 12% and 6% respectively, compared to this the RSD for the parameter C fraction degraded as CH_4 was almost 30%. The uncertainty of C-fraction degraded as CH_4 was estimated according to the CH_4 emission values based on UBA [12]. For PEC, the minimum and maximum values for CH_4 emitted per tonne input material ranged from 830 to 4800 g, with a median value of 1200 g, and for OC, the values ranged from 730 to 5500 g, with a median value of 1800 g. As a result, the Monte-Carlo analysis results show a combination of moderately high SRs of 1.1 and 2.3 for PEC and OC, respectively, as well as an RSD of 30% in the input values for C degraded as CH_4.

C degraded as CH_4 was the most influential parameter for FC (Fig. 3) with an SR ranging from 0.9 to 1.2. This, together with a relatively high RSD of nearly 30%, had a significant impact on the overall uncertainty of FC. The influence of C degraded as CH_4 on FC was clear when comparing the skewness in the input values for C degraded as CH_4 and the output results used for the box-plot. For FC, the input skewness was 0.58, which was close to the output skewness of 0.53, but for MSC, the input and output skewness were 0.45 and 0.25, respectively. The smaller difference in skewness between the input and output parameters can explain why C degraded as CH_4 has a greater influence on FC. The influence of high maximum values for C degraded as CH_4, as well as its relatively higher SR, can explain the increase in the median values for the GWP balance in Monte-Carlo simulation.

The uncertainty of AP values in OC was caused by NH_3 emissions ranging from 14 to 1183 g NH_3 for the MSC and 9 to 1057 g NH_3 for the FC. However, the range of NH_3 emissions for PEC was significantly lower than for OC, so the uncertainty was lower in this case. The main source of uncertainty in the results for the impact category OFP was NO_x emissions from the waste collection process. Because the value of NO_x varied with transport distances, the uncertainty in the result for this impact category is highly dependent on the assumed distances. The results for the impact categories ODP were estimated using N_2O emissions and the lower values for OC can be attributed to the lower N_2O emissions due to the lack of a biofilter.

Comparison of LCA results with literature data

LCA studies on composting of biowastes and green cut material mainly focus on the associated GHG-emissions, seldomly measured and more frequently modelled. Therefore, the GWP is used for comparing the results of this study with results from other peer-reviewed publications. The studies compared in this section have varying input material, and composting parameters and produce different compost qualities, a reasonable comparison of the GWP results is consequently limited. When considering only GHG-emissions and excluding credits, then the GWP ranges from 50 to 100 kg $CO_{2eq.}$per t input material in this study. Considering similar LCA studies, i.e. similar input material and composting conditions, the GWP reported by Kim and Kim (2010) for producing compost from 1t food waste was found to be almost 123 kg $CO_{2eq.}$ [60]. A GWP of up to 81 kg $CO_{2eq.}$ from composting of 1t household biowaste was reported by Boldrin et al., (2009) [61]. A GWP of 218 kg $CO_{2eq.}$ per tonne of organic fraction of municipal solid waste was estimated by Weligama Thuppahige et al., (2022) [62]. Similarly, Eriksson et al., (2015) estimated a GWP of 43 kg $CO_{2ep.}$from composting 1t food waste. The results calculated in this study show a similar range to the presented studies with somehow comparable input materials and the presented results are therefore plausible. Composting of agricultural wastes was found to cause almost 200–250 kg $CO_{2eq.}$ per ton of waste handled [63], hence, different input materials and process conditions can also cause a higher GWP. Similar to the findings of this study, the other studies also identified the direct emissions due to the decomposition of organic as the main contributor to GHG-emission of composting [35, 64]. Overall it can be seen that the range of GWP associated with composting of biowaste ranges between almost 40 and 250 kg $CO_{2eq.}$. This wide range can be attributed to multiple factors, however, the emission factor for methane formation used to estimate direct CH_4-emissions from the composting process is the most relevant for GWP. The turning frequency, type of composting process and input material influence the GHG-emissions, respectively the associated emission factors [16].

Limitations of the study

One of the major limitations of accurately assessing the uncertainties in the environmental impacts of the composting process is the influence of spatial and temporal variation of the input material composition. Since biowaste and green cut waste vary between seasons and geographical locations, the composition also varies, this limits the reproducibility of the results to a certain extent. Throughout the year, the variation in waste composition is mainly due to the increased share of garden waste in the summer and autumn months in rural areas in Germany [65]. Similarly, biowaste originating from rural areas also varied compared to wastes from urban areas in terms of foreign material and contamination [66]. Interactions between process parameters, e.g. moisture content and CH_4-emissions, are not directly considered due to the complexity of the composting process. Moreover, the presence of specific pollutants and micro-plastics in the biowaste and the associated environmental emissions are not considered in this study.

Conclusion

The environmental impacts of composting systems were presented, paying special attention to the uncertainties. The environmental impacts assessed using LCA showed that PEC had a 17–38% lower net GWP than OC. However, this result did not take into account the modelled system uncertainties. When the uncertainties for the GWP estimated using Monte-Carlo simulations were considered, the boxplots overlapped, indicating that the net value for GWP obtained from the model could vary, making the previous comparison between PEC and OC ambiguous. Similarly, the boxplots for the OFP and ODF impact categories also overlapped across all combinations, making the comparison difficult. However, there was a distinction between the OC and PEC combinations only for the impact category AP. The sensitivity ratios obtained from the perturbation analysis revealed the input and process parameters that contributed the most towards uncertainty in the result. For FC, the parameter with the greatest influence on GWP emissions was C degraded as CH_4 during composting. This parameter, which is primarily influenced by aeration and turning frequency, can be controlled to some extent using best management practices. The C-fraction in the input material also had a significant influence, but this parameter is difficult to control because it is dependent on the characteristics of the input material. Similarly, the mineral fertiliser equivalent and peat substitution factor were two influential parameters effected by the model assumptions. However, the perturbation analysis results alone could not explain the uncertainty in the Monte-Carlo simulation results; uncertainty in the input values of the respective parameters also played a role. The results of the Monte-Carlo simulation were affected by parameters with a high influence as well as a high variability. The Monte-Carlo simulation results revealed the model's overall uncertainty and its relevance when comparing results between the combinations. The results of the perturbation analysis, on the other hand, identified the parameters that required more precise data to reduce the uncertainty. Uncertainties in future LCA studies pertaining to composting can be reduced by focusing on data for the influential parameters.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10163-023-01740-6.

Acknowledgements We would like to thank the two anonymous reviewers for their comments, which helped to amend the manuscript. The authors acknowledge the financial support by the Federal Ministry of Education and Research of Germany (BMBF) in the framework of the project RUN–"Rural Urban Nutrient Partnership".

Funding Open Access funding enabled and organized by Projekt DEAL.

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