Waste Management 190 (2024) 578–592

Contents lists available at ScienceDirect

Waste Management

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

Research Paper

Biodegradable plastics – Where to throw? A life cycle assessment of waste collection and management pathways in Austria

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ARTICLE INFO

Keywords:
Biodegradable plastic waste management
Waste collection & sorting
Life cycle assessment
Recycling
Packaging waste

ABSTRACT

The current waste management systems are struggling to optimally handle biodegradable plastics (BDPs) and are facing numerous challenges; one of which is the consumer confusion about how to best source-segregate BDPs. Based on an environmental life-cycle assessment, this study investigated the consequences of collecting BDPs in one of three waste streams (packaging waste, biowaste, and residual waste) in Austria. Collecting BDPs as (i) packaging waste resulted in incineration (SP1) or mechanical recycling (SP2), (ii) biowaste resulted in composting (SB1) or anaerobic digestion (AD) (SB2), and (iii) residual waste in incineration (SR1). SP2 performed best in most of the 16 investigated impact categories (ICs). Three scenario analyses demonstrated that (i) utilisation of BDPs as an alternative fuel for process heat substitution yielded more environmental benefits than incineration in SP1 and SP2, (ii) adding a material recovery facility (MRF) with AD increased the environmental load for SB2, while (iii) the energy scenario with zero electricity imports plus heat from biomass performed best for most alternative pathways across the 16 ICs. Eight technology parameters (out of 97) were identified as most relevant for the results based on data quality, sensitivity ratio, and analytical uncertainty; they were related to waste incineration, MRF, recycling facility, compost- and AD processes. Overall, mechanical recycling emerged as the most favourable option which is aligned with the waste-hierarchy mentioned in the European Union Waste Framework Directive. However, effective mechanical recycling of BDPs requires (i) a 'sufficient' waste amount, (ii) a market for recyclates, and (iii) relevant mechanical recycling infrastructure.

1. Introduction

With the current fight against growing mismanaged global plastic waste (~460 million tonnes) (Hannah Ritchie et al., 2023), bioplastics are often suggested as an environmentally-friendly solution (Castro-Aguirre et al., 2016; Cucina et al., 2021; Thakur et al., 2018). Bioplastics, either biobased[a] or biodegradable[a] or both (European Bioplastics, 2018), currently amount to 0.5 % (2,182 million tonnes in 2023) of the total plastic production and are forecasted to grow to 7,432 million tonnes in 2028 (European Bioplastics e.V., 2024). Biodegradable plastics (both biobased and fossil-based) have a 52.1 % market share of bioplastics produced in 2023 (European Bioplastics e.V., 2024) and their ability to biodegrade is observed to be their significant attraction (Babaremu et al., 2023; Calabrò & Grosso, 2018; Meeks et al., 2015). However, these plastics have several challenges related to waste management (WM), e.g., contamination of conventional plastic recyclates, consumer confusion about proper disposal options, and insufficient volume of secondary material for economically feasible recycling (Dilkes-Hoffman et al., 2019; Feghali et al., 2020; Rujnić-Sokele & Pilipović, 2017).

Nevertheless, consumer confusion about where to throw these plastics remains one of the main challenges, potentially leading to the mismanagement of biodegradable plastic (BDP) waste (Dilkes-Hoffman et al., 2019). Many qualitative studies (Lynch et al., 2017; Marchi et al., 2020; Patrício Silva, 2021) and waste characterisation studies involving

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[a] Biobased plastics are partly or fully derived from plants, whereas biodegradable plastics can be converted into natural substances (water, CO_2, biomass) by microorganisms (European Bioplastics (2018)).

https://doi.org/10.1016/j.wasman.2024.10.018
Received 11 April 2024; Received in revised form 14 September 2024; Accepted 17 October 2024

List of Abbreviations

A	Acidification impact category
AD	Anaerobic digestion
BDP	Biodegradable plastic
CC	Climate change impact category
CPP	Compostable plastic packaging
DFS	Direct fuel for process-heat production substitution
DQ	Data quality
DQR	Data quality ratio
DTU	Technical University of Denmark
E-F	Eutrophication – freshwater impact category
E-T	Eutrophication – terrestrial impact category
EF	Ecotoxicity freshwater impact category
EU	European Union
HT-C	Human toxicity – cancer impact category
HT-NC	Human toxicity – non-carcinogenic impact category
IC	Impact category
IoR	Ionising radiation impact category
LCA	Life cycle assessment
LDPE	Low-density polyethylene
LU	Land use impact category
MBT	Mechanical biological treatment
MRF	Material recovery facility
NIR	Near-infrared
NSR	Normalised sensitivity ratio
OD	Ozone depletion impact category
PE	Polyethylene
PLA	Polylactic acid
PM	Particulate matter impact category
POF	Photochemical ozone formation impact category
RDF	Refuse-derived fuel
RU-EC	Resource use – energy carrier impact category
RU-MM	Resource use – minerals and metals impact category
SB1	Alternative pathway with biodegradable plastics collected with biowaste and composted
SB2	Alternative pathway with biodegradable plastics collected with biowaste and anaerobically digested
SB2a	Scenario analysis of alternative pathway with biodegradable plastics collected with biowaste and anaerobically digested with material recovery facility
SP1	Alternative pathway with biodegradable plastics collected with packaging waste and incinerated
SP1a	Scenario analysis of alternative pathway with biodegradable plastics collected with packaging waste and used as direct fuel for process-heat substitution
SP2	Alternative pathway with biodegradable plastics collected with packaging waste and mechanically recycled
SP2a	Scenario analysis of alternative pathway with biodegradable plastics collected with packaging waste and mechanically recycled plus used as direct fuel for process-heat substitution
SR1	Alternative pathway with biodegradable plastics collected with residual waste and incinerated
TPS	Thermoplastic starch
WAM	Reduced electricity imports plus heat from only natural gas energy scenario (abbreviation for 'with additional measures')
WAM +	Zero electricity imports plus heat from only biomass energy scenario (abbreviation for 'with additional measures – plus')
WEM	Base energy scenario (abbreviation for 'with existing measures')
WM	Waste management
WU	Water use impact category

BDPs (Mhaddolkar et al., 2024), found that consumers are often unsure about the appropriate waste collection option for BDP. In Austria, similar BDP products were found in either of three waste streams: packaging waste, biowaste, and residual waste (Mhaddolkar et al., 2024), even though the sorting guidelines indicate their collection as packaging waste (Holding Graz, 2023a, 2023b). This ambiguity directly affects their environmental performance in the WM system. To consider BDPs as the environmentally friendly solution to conventional plastics, clarity on the environmental consequences of these plastics in the different waste collection and management routes is critical.

Life cycle assessment (LCA) has been applied within WM for holistic quantification of environmental consequences in numerous studies (Bisinella et al., 2024; Christensen et al., 2020). As such, waste LCA assists in comparing the environmental consequences of choosing one waste treatment method over others (Arena et al., 2004; Cherubini et al., 2009; Mazhandu et al., 2023). Most of the available LCA studies on BDP WM compared the environmental performance of using BDP instead of conventional plastic (Bohlmann, 2004; Fieschi & Pretato, 2018; Moretti et al., 2021; Mori et al., 2013). Some are comparative LCAs studying BDP versus paper (Dolci et al., 2021; Stafford et al., 2022; Zhu et al., 2023) and BDP versus metal (Changwichan & Gheewala, 2020; Desole et al., 2024; Tamburini et al., 2021; Wei et al., 2022). While some studies discussed their end-of-life options, they mainly focussed on polylactic acid (PLA) (Bishop et al., 2021; Maga et al., 2019; Park et al., 2024; van der Harst et al., 2014) and one on cellulose-based-plastic WM (Gadaleta et al., 2023). Very few studies discussed starch-based polymers and their blends (also BDP), which had a 6.4 % share of the global bioplastic production capacity in 2023 (European Bioplastics e.V., 2024). Of the few available studies, Hottle et al. (2017) compared the environmental performance of cradle-to-grave LCA of 1 kg thermoplastic starch (TPS); they found that composting TPS had certain benefits over landfilling. Whereas Hermann et al. (2011) concluded that anaerobic digestion (AD) offered lower environmental loads than composting (home and industrial) and incineration. However, both these studies omitted mechanical recycling. Conversely, Piemonte (2011) and Rossi et al. (2015) included mechanical recycling while comparing the environmental performance of different TPS waste treatment options and found that mechanical recycling had the lowest environmental impact. Cristóbal et al. (2023) conducted an LCA of compostable plastic packaging (of which TPS was a small share), where mechanical recyling demonstrated the lowest environmental impacts. Yet these studies did not consider inseparable food waste with BDPs, which are often used as biowaste-collection-aids (Bátori et al., 2018; Kakadellis & Harris, 2020) and the presence of inseparable food waste could affect the overall environmental performance results (Rossi et al., 2015; United Nations Environment Programme, 2022).

While existing LCA studies involving BDPs cover a range of different WM routes, it is difficult to compare and derive clear conclusions across different LCA studies with different scope definitions, data collection approaches, and impact assessment methods (van Eygen et al., 2018). Two main challenges associated with BDP LCA studies are relatively few data sources and few experimental studies investigating their WM (Rossi et al., 2015). Consequently, including systematic sensitivity and uncertainty analysis as part of LCA is important for providing a robust conclusion (Bisinella et al., 2016). However, only one of the above-mentioned LCA studies on TPS-blend BDPs included comprehensive uncertainty analysis (Cristóbal et al., 2023), but none discussed data quality (DQ). To improve the basis for managing BDP waste and guiding consumers regarding source segregation, DQ and uncertainties should be included as part of LCA for a systematic comparison of all relevant waste collection and management options.

This study addressed the 'where to throw BDPs' dilemma of consumers by conducting an LCA to identify the consequences of collecting BDPs with one of the three waste streams: (i) packaging waste, (ii) biowaste, and (iii) residual waste, and the resulting treatments through a wide range of waste technologies, namely, mechanical sorting and recycling, composting, anaerobic digestion with and without mechanical pre-sorting, and incineration. The focus was to contribute to the following three missing aspects of the LCA of BDPs. First, the analysis of BDP waste management was focused on Austria, which has not been looked into before; thus, the inventory building and marginal energy calculation were based on the Austrian situation. Second, the existing LCAs focused on clean BDP products, but they overlooked the point that BDPs often come in contact with food waste and its presence could influence the LCA results; therefore, this study included a fraction of inseparable food waste with BDP in the functional unit. Third, to ascertain the robustness of conclusions, extensive sensitivity and uncertainty analysis was conducted using 97 parameters which affect the material and elemental flow within the wide range of technologies considered. As the knowledge of the most relevant parameters is important to indicate the significant processes in the model (Hauschild et al., 2018), this study used two methods by considering data quality ratio with normalised sensitivity ratio and analytical uncertainty to obtain two sets of the most relevant parameters.

2. Methodology

2.1. Scope and functional unit

The function of this study was to collect and treat BDP waste with a small fraction of inseparable food waste in different pathways. Although BDPs have a wide range of packaging applications, namely, single-use cutlery and food packaging (Briassoulis et al., 2019), TPS supermarket carrier bags were selected as representative BDP for the study as they were found present in three waste streams in an Austrian urban area (Mhaddolkar et al., 2024), and were promoted to be reused as biowaste-collection-aid (NATURABIOMAT GmbH, 2024). Although starch-based plastics occupied a 6.4 % (~0.179 million tonnes) share of total global bioplastics production (2.18 million tonnes) in 2023 (European Bioplastics e.V., 2024), they are widely used in Austria after the 2019 single-use plastic carrier bags ban (Federal Ministry Republic of Austria, 2020; NATURABIOMAT GmbH, 2024). A blend of TPS, polybutylene adipate terephthalate, and polycaprolactone (brand-name Mater-bi) was used because it is one of the most commercially available TPS-blend (Hejna et al., 2022; Russo & Stafford, 2023). Thus, the functional unit was to collect and treat **1.25 kg** of BDP (80 %wt.) and inseparable food waste (20 %wt.) in different waste treatment plants. The 20 % inseparable food waste was based on the average level of contamination observed in a field study characterising TPS-blend supermarket carrier bag samples from the packaging waste, biowaste, and residual waste in (Mhaddolkar et al., 2024). The systems boundary was from the household waste collection point to the waste treatment facility product (section 2.3); thus, waste collection and material- and energy recovery were included in the assessment, while capital goods were excluded. This study considered zero 'dragging effect', which is the additional material dragged with sorted fraction, e.g., biowaste dragged with the sorted plastic in an AD plant or conventional plastics with sorted BDP (Cristóbal et al., 2023). The geographical scope was Austria, and the temporal scope for the energy scenario was 2020–2030.

2.2. Alternate pathways and life cycle inventory

As consumers are unclear about the best source separation option for BDPs, these plastics could be found in either of three waste streams – packaging waste, biowaste, and residual waste (Mhaddolkar et al., 2024). Moreover, although the Austrian compost ordinance (Kompostverordnung, 2001) allowed EN 13432 certified compostable plastic bags (for biowaste-collection) to be accepted in compost plants, the source separation guidelines instruct that these plastics belong to packaging or residual waste (Altstoff Recycling Austria AG, 2022; Sametinger, 2017). Hence, the alternate pathways represent that BDPs may end up in either of these three waste streams including the relevant waste treatment pathways. Therefore, five alternate pathways were selected: (i) two pathways for BDPs collected with packaging waste: one pathway with BDPs landing in the reject fraction after sorting to be incinerated **(SP1)**, and another pathway with BDPs being recovered for recycling and substituting new BDPs **(SP2)**; (ii) two pathways for BDPs collected with biowaste: a pathway with BDPs being composted **(SB1)** and another with BDPs being anaerobically digested **(SB2)**, and (iii) one pathway with BDPs collected with residual waste and being incinerated **(SR1)**. The grouped processes for each alternate pathway are explained in Table 1 and henceforth denoted by square brackets '[]'.

2.2.1. SP1 – Packaging waste pathway plus incineration

BDPs were collected with packaging[b] waste and transported to a material recovery facility (MRF), where they passed through a near-infrared (NIR) sorter. It was assumed that 100 % of the BDPs and the inseparable food waste were sorted out in the reject fraction, which was transported to a waste incineration plant. The input to the incineration plant was first converted into refuse-derived fuel (RDF). A grate incinerator with wet air pollution control was modelled based on (van Eygen et al., 2018), and the produced energy substituted marginal energy (electricity and heat). Treatment and disposal of bottom ash were excluded from the model because their impacts are small as well as uncertain (Andreasi Bassi et al., 2017; Birgisdóttir et al., 2007; Clavreul et al., 2012). The detailed inventory is provided in **S1** (**Section 1.4**).

2.2.2. SP2 – Packaging waste pathway plus mechanical recycling

BDPs were collected with packaging[b] waste and transported to MRF, where the NIR machine sorted them out as recyclables. Sorting efficiencies for BDP and the attached food waste were given by parameters 'SortEff' and 'ImpLost', respectively (**Table S1.5**). The sorted-out BDPs were transported to a recycling facility, where they underwent washing, drying, and shredding; ultimately, substituting virgin BDP. The contaminants (including food waste) from the recycling facility and the reject fraction from MRF were transported separately to the incineration plant, modelled the same as for SP1 with the RDF plant. The recycling facility was modelled based on the LDPE[c] plant from (van Eygen et al., 2018), because of the unavailability of an industrial-scale plant for TPS-blend BDP; the detailed explanation and inventory are provided in **S1** (**Section 1.4**).

2.2.3. SB1 – Biowaste pathway plus composting

BDPs were collected with biowaste and transported to a compost plant. They should be compostable to be treated in compost facilities (Utilitalia, 2020). BDP and the attached food waste were provided with different biodegradation rates (**Table S1.10**), and the resulting compost was screened using a wind-sifter. The relatively clean compost was used for substituting the chemical fertilizer. The sifted-out impurities were incinerated (modelled as in SP1). **S1** (**Section 1.5**) includes the detailed inventory.

2.2.4. SB2 – Biowaste pathway plus anaerobic digestion

BDPs were collected in biowaste and transported to an AD plant. BDP and food waste were provided with different biogas yields

[b] Packaging waste in Austria contains plastic bottles, beverage cartons, plastic bags, yoghurt pots WMW by ISWA (2022).

[c] LDPE – low-density polyethylene.

Table 1
Description of aggregated processes and the groups are presented inside squared brackets '[]'. AD – anaerobic digestion; MRF – material recovery facility; MBT – mechanical biological treatment; RDF – refuse-derived fuel; SP1 – packaging waste pathway plus incineration; SP2 – packaging waste pathway plus mechanical recycling; SB1 – biowaste pathway plus composting; SB2 – biowaste pathway plus anaerobic digestion; SR1 – residual waste pathway plus incineration; SP1a – packaging waste pathway plus direct fuel for process-heat substitution (scenario analysis); SP2a – packaging waste pathway plus mechanical recycling and direct fuel for process-heat substitution (scenario analysis); SB2a – biowaste pathway plus anaerobic digestion with MRF (scenario analysis).

Grouped processes	Aggregated unit processes				
	SP1 (SP1a)[1]	SP2 (SP2a)[1]	SB1	SB2 (SB2a)[1]	SR1
[Transportation]	Transport from: Household to MRF, MRF to waste to energy plant.	Transport from: Household to MRF, MRF to recycling plant, recycling plant to waste to energy plant.	Transport from: Household to compost plant, composting facility to waste to energy plant, composting facility to farmland.	Transport from: Household to AD plant[2], AD plant to farmland.	Transport from: Household to MBT plant, MBT to waste to energy plant.
[Waste pre-treatment]	MRF plant, RDF plant.		–	MRF plant.	MBT plant.
[Waste to energy processes]	Waste to energy plant (Direct fuel for process-heat substitution)[1].		Waste to energy plant.	(Waste to energy plant)[1].	Waste to energy plant.
[Energy substitution]	Electricity and heat production from waste to energy process.			Energy production from biogas and own consumption[3] in AD plant (AND Electricity and heat production from waste to energy process)[1].	Electricity and heat production from waste to energy process.
[Recycling plant process]	–	Recycling facility.	–	–	–
[Material substitution]	–	Virgin biodegradable plastics substitution.	–	–	–
[Organic treatment process]	–	–	Composting facility.	AD plant (diesel consumption excluding own energy consumption[3]), digestate storage, and dewatering unit.	–
[Fertilizer substitution]			Chemical fertilizer substitution, application of fertilizer on farmland.		

[1] These unit processes are specific to alternate scenarios.
[2] The MRF is assumed to be located within the AD plant, hence the distance remains the same.
[3] The energy consumption of AD plant is supplied by the energy produced using biogas; therefore, it is included in the [Energy substitution] process.

(Table S1.11). Energy from the produced biogas substituted marginal energy. Digestate was stored, dewatered, and used to substitute chemical fertilizers. S1 (Section 1.5) includes the detailed inventory.

2.2.5. SR1 – Residual waste pathway plus incineration

BDPs were collected with residual waste and transported to a mechanical biological treatment (MBT) plant. It was assumed that the

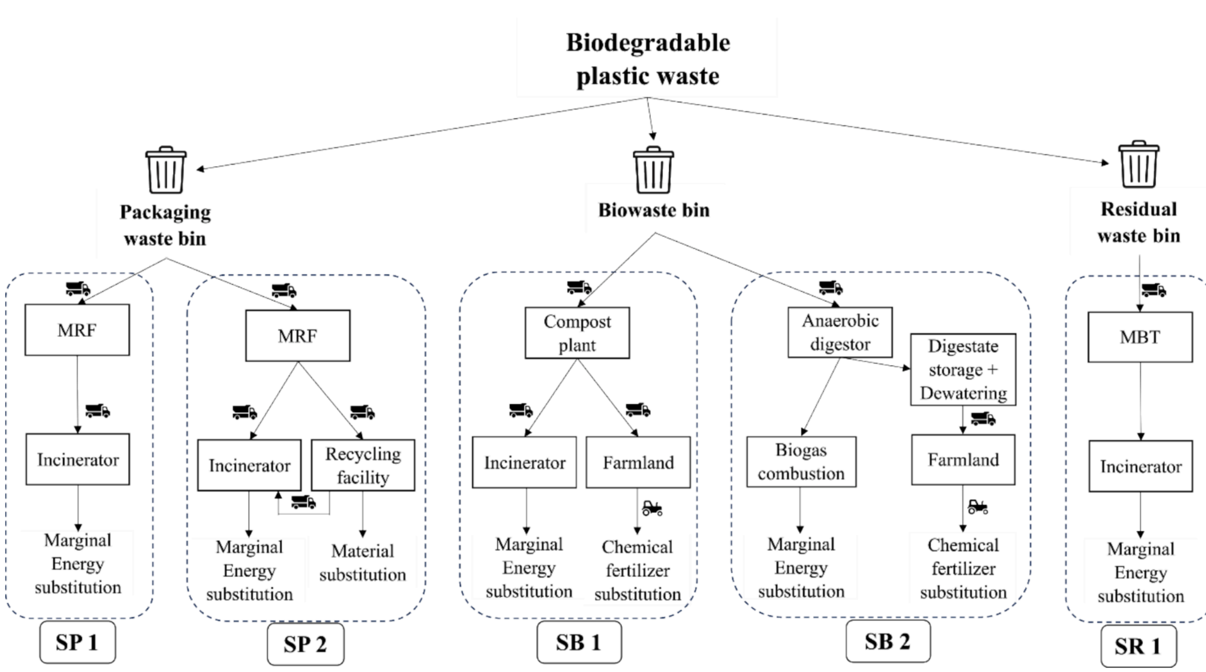


Fig. 1. Systems boundary (denoted by dashed lines) and 5 alternate pathways. MRF – material recovery facility; MBT – mechanical biological treatment; SP1 – packaging waste pathway plus incineration; SP2 – packaging waste pathway plus mechanical recycling; SB1 – biowaste pathway plus composting; SB2 – biowaste pathway plus anaerobic digestion; SR1 – residual waste pathway plus incineration.

attached food waste remains intact and sent for incineration with BDP. The MBT was considered to include its energy consumption in the calculations. The incinerator was modelled as in SP1, and **S1** (**Section 1.6**) includes the detailed inventory.

2.3. Systems boundary and LCA model

The system includes collection, sorting, treatment, and transportation of waste. Thus, the system boundaries (Fig. 1) start by transporting the collected waste to the waste pre-treatment (e.g. MRF) or waste treatment (e.g. AD) facility and end after substituting either energy (marginal electricity and heat) and/or material (virgin BDP or chemical fertilizer). For instance, the SP1 pathway ends by substituting marginal energy, whereas SB1 ends by substituting marginal energy and chemical fertilizers.

The LCA model was prepared using EASETECH software (v3.4.4) developed by DTU Environment, Technical University of Denmark (Clavreul et al., 2014; Faraca et al., 2019). A consequential modelling approach with system expansion via substitution was used. The life cycle impact assessment was conducted using the Environmental Footprint EF3.0 method (European Commission, 2021) without long-term calculations. The included impact categories (ICs) were: *Climate change (CC)*, *Ozone depletion (OD)*, *Human toxicity – cancer (HT-C)*, *Human toxicity – non-carcinogenic (HT-NC)*, *Particulate matter (PM)*, *Ionising radiation (IoR)*, *Photochemical ozone formation (POF)*, *Acidification (A)*, *Eutrophication – terrestrial (E-T)*, *Eutrophication – freshwater (E-F)*, *Eutrophication – marine (E-M)*, *Ecotoxicity freshwater (EF)*, *Land use (LU)*, *Water use (WU)*, *Resource use – minerals and metals (RU-MM)*, and *Resource use – energy carrier (RU-EC)*.

2.4. Sensitivity scenario analysis

Using scenario analysis (part of sensitivity analysis), the effects of modifying one modelling parameter over the results were identified (Andreasi Bassi et al., 2017; Bisinella et al., 2016). Three aspects were changed to observe the effects on the IC results.

2.4.1. Direct fuel for process-heat substitution (SP1a, SP2a)

The effect of using BDPs for direct fuel for process-heat production substitution (DFS) was studied because in Austria plastic rejects are commonly used as alternative fuels in the cement industry (Affenzeller, 2018). It was assumed that the presence of a small fraction of food waste would not have any adverse effects on the incineration. Similar to the LCA study by Rossi et al. (2015), the incineration process was modelled the same as for the alternate processes, while substituting only heat energy (supplied by coal). A detailed inventory is provided in **S1** (**Section 1.4.4**).

2.4.2. AD with MRF (SB2a)

An MRF unit was added to sort out conventional plastics before sending the biowaste for AD. The loss of BDPs and the inseparable food waste due to false sorting was considered, which prevented the falsely sorted material from being used for biogas and digestate production. Additionally, a sensitivity analysis was conducted with 100 % of TPS-blend BDP sorted out as reject.

2.4.3. Energy system

The marginal electricity and heat for Austria for 2020–2030 were calculated based on (Muñoz & Weidema, 2023; Weidema et al., 1999) and the calculations are explained in **S1** (**Section 1.8**). Marginal electricity and district heat for the three scenarios were modelled based on (Baumann & Kalt, 2015). The results for the five alternate pathways were compared based on three different energy scenarios, namely: baseline energy scenario, which was 'with existing measures' (WEM); reduced electricity imports and heat from natural gas energy scenario, which was 'with additional measures' (WAM); and zero electricity imports and heat from biomass energy scenario, which was 'with additional measures – plus' (WAM +). Table 2a and Table 2b show the calculated marginal electricity and district heat mix, respectively.

2.5. Uncertainty analysis and identification of the most relevant parameters

2.5.1. Uncertainty analysis

Modelling choices, assumptions, and inherent data limitations often introduce uncertainty in LCA (Saur et al., 2009), which are characterised by model-, scenario-, and parameter uncertainties (Heijungs et al., 2005; Huijbregts, 1998). Uncertainty analysis can be conducted by sensitivity analysis (evaluating the effect of input uncertainties on results) and uncertainty propagation (calculating uncertainties of results based on input uncertainties) (Clavreul et al., 2012).

The inventory data for almost every foreground data was added as a parameter with a probability distribution (Bisinella et al., 2016). The probability distributions were mostly added based on the available data. In cases where only the average value was available, normal distribution was assumed. Where information was unavailable, uncertainty ranges were selected using literature (Damgaard et al., 2022; Jacobsen, 2023; Lloyd & Ries, 2007; Weidema et al., 2013). The uncertainty propagation was calculated using Monte Carlo simulation for 10,000 cycles. The analytical uncertainty was calculated using EASETECH software, and the % contribution of each parameter to this analytical uncertainty was calculated (**Equation S3.3** in **S3**) for every alternate pathway and IC. The details are presented in **S1** (**Section 1.2**).

2.5.2. Data quality assessment

The knowledge of DQ gives information about the reliability of the selected data (Saur et al., 2009). DQ assessment was conducted by scoring the indicators presented by Weidema and Wesnæs (1996); namely, reliability, completeness, temporal correlation, geographical correlation, and further technological correlation, for the slected data. The data quality ratio (DQR) was calculated using these indicator scores and **Equation S3.1** (**S3**) (Andreasi Bassi et al., 2017; EC - JRC, 2010). Accordingly, the input DQ of the parameter was classified as 'high' (DQR $<$ 1.6), 'basic' (1.6 $<$ DQR $<$ 3), and 'estimate' (DQR $>$ 3), based on (EC - JRC, 2010). The DQR for each parameter was calculated during the life cycle inventory building stage. **S3** (**Section 3.1.1**) includes the details.

2.5.3. Identifying the most relevant parameters for environmental performance

The parameters' criticality (relevance) was assessed considering the

Table 2
Calculated marginal energy mix for the three energy system scenarios. (a) Marginal electricity mix and (b) Marginal district heat mix. WEM – energy scenario with existing measures, baseline energy scenario; WAM – energy scenario with additional measures, reduced electricity imports and heat from natural gas; WAM+ – energy scenario with additional measures plus, zero electricity imports and heat from biomass.

(a)

Source	Calculated marginal electricity mix (%)		
	WEM	WAM	WAM+
Hydro	4.60	4.85	6.65
Wind	6.73	29.24	37.00
Solar	39.02	41.11	56.35
Imports	49.64	24.81	0

(b)

Source	Calculated marginal district heat mix (%)		
	WEM	WAM	WAM+
Natural gas-based heating plants	10.74	100	0
Biomass-based heating plants (wood pellets)	89.26	0	100

results of the DQ assessment together with the calculated (i) normalised sensitivity (Method-1) and (ii) analytical uncertainty (Method-2). Using two methods provided different information about the identified most relevant parameters.

In Method-1 (**Section 3.1.2** in **S3**), the most relevant parameters were identified using the contribution of sensitivity ratio and DQ based on (Andreasi Bassi et al., 2017). The sensitivity ratio is the 'ratio between the relative change of the result and the relative change of the parameter' (Andreasi Bassi et al., 2017), calculated using perturbation analysis (10 %) in EASETECH software. The normalised sensitivity ratio (NSR) was calculated by normalising the sensitivity ratio values according to the highest value in the respective IC results (**Equation S3.2**). The parameters were designated as Low ($0.1 < NSR < 0.5$), Medium ($0.5 < NSR < 0.8$), and High (>0.8) sensitive. After considering both DQR and NSR, the parameters were categorized as very critical, critical, or less critical (Fig. 2). For example, a parameter with $DQR > 3$ and $NSR > 0.8$ was identified as a 'very critical' parameter, which meant it is one of the most relevant parameters.

In Method-2 (**Section 3.1.3** in **S3**), with the same ranges as for Method-1 (using NSR), after considering the contribution to analytical uncertainty and DQR, the parameters were identified as very critical, critical, or less critical (Fig. 2).

3. Results

3.1. Contribution analysis and hotspot identification

Fig. 3 shows the contribution analysis results of the five alternate pathways using a stacked bar chart. While comparing the environmental impacts (across 16 ICs) of collecting BDP (TPS-blend) in one of the three waste bins (packaging, bio, and residual), it was observed that SP2 (packaging waste plus mechanical recycling) had the highest net savings in 14 ICs, except for 2 ICs, where SR1 (*PM*) and SB2 (*WU*) had highest net savings (**Figure S2.1** to **Figure S2.5**). On the other hand, SB1 had maximum net load in seven ICs (*CC*, *HT-NC*, *POF*, *E-F*, *EF*, *LU*, and *RE-MM*), SB2 in five ICs (*PM*, *IoR*, *A*, *E-T*, and *E-M*), and both SP1 (*HT-C* and *WU*) and SR1 (*OD* and *RU-EC*) in two ICs. For ease of understanding, the results of 3 ICs (*CC*, *RU-EC*, and *HT-C*) are discussed in detail and those for the remaining 13 ICs are included in **S2** (**Section 2.1**).

For *CC* (Fig. 3a), SP1, SB1, and SR1 had net load, while SP2 and SB2 had net savings. [Energy substitution] was the hotspot process (savings) for SP1 (contributing 48 %[d] to the overall impacts) and SR1 (46 %), and in both these cases, the most contributing elemental flow to the process' impacts was 'avoided CO_2 emissions' from marginal-heat substitution. [Transportation] was the second-best hotspot process contributing to the environmental load of SP1, due to 'fossil CO_2 emissions' from diesel combustion. Contrarily, [Waste pre-treatment] contributed to the environmental load of SR1, due to 'fossil CO_2 emissions' from diesel combustion by wheel loader in MBT. [Energy substitution] was the hotspot process for SB2 (60 %), with 'avoided fossil CO_2 emissions' as the elemental flow contributing most to the overall impact; whereas [Organic treatment process] was the second-most-contributing hotspot process (load), with 'methane emissions' as the major elemental flow. The hotspot processes for SP2 were [Material substitution] contributing 77 % to savings from 'avoided fossil CO_2 emissions' from virgin BDP substitution. This was followed by [Recycling plant process] contributing to environmental load, due to 'fossil CO_2 emissions' from marginal process heat consumption. Lastly, SB1 had [Organic treatment process] contributing (32 %) to the environmental load, followed by [Fertilizer substitution]; in both cases, the main elemental flow as 'dinitrogen monoxide emissions' due to the elementary exchanges from wheel loader and tractor.

For *RU-EC* (Fig. 3c), SP1 and SR1 had a net load, while SP2, SB1, and SB2 had net environmental savings. Here, [Energy substitution] was again the hotspot process for SP1 (48 %), SB2 (83 %), and SR1 (47 %). The most contributing elemental flow for SP1 and SR1 was 'avoided natural gas resource consumption' from substituted marginal energy; whereas for SB2 was 'avoided coal consumption' from substituted marginal process heat for own consumption. [Transportation] was again the second hotspot process for SP1 and SB2, and [Waste pre-treatment] for SR1; in all three cases, the main elemental flow was 'consumption of oil' from diesel consumption. SP2 again had [Material substitution] as a hotspot process, contributing 82 % to savings and 'avoided natural gas consumption' as the major elemental flow; [Recycling plant process] contributed to environmental load due to 'consumption of oil' from diesel consumption. [Transportation] contributed to 48 % of the environmental load for SB1, due to 'consumed oil' during diesel consumption by transportation to the compost plant; whereas [Energy substitution] (39 %) and [Fertilizer substitution] (12 %) contributed to savings.

For *HT-C*, all five alternate pathways had net savings. SP1, SB2, and SR1 had [Energy substitution] as the hotspot process, with 71 %, 64 %, and 83 % respective contributions to environmental savings. In all these cases, 'avoided benzo(a)pyrene emissions to air' from marginal-energy production was the most contributing elemental flow. Benzo(a)pyrene, a toxic and carcinogenic polycyclic aromatic hydrocarbon, is often found in emissions from wood and coal combustion (Bieser et al., 2012). Contrarily, [Material substitution] contributed 70 % to savings in SP2 due to 'avoided chromium emissions to air' from virgin BDPs substitution and [Fertilizer substitution] contributed to 72 % savings in SB1 due to 'avoided chromium emissions to soil' from chemical fertilizer substitution.

Thus, in the three ICs, [Energy substitution], [Material substitution], and [Fertilizer substitution] were often identified as hotspot categories contributing to savings. Similarly, [Transportation] and [Waste pre-treatment] were identified as hotspot processes contributing to the environmental load. Moreover, [Energy substitution] and [Material substitution] processes contributed to savings across the remaining 13 ICs (**Section 2.1** in **S2**). Whereas [Transportation], [Waste pre-treatment], [Waste to energy], and [Recycling plant processes] had loads across 16 ICs. [Organic treatment process] contributed to environmental load for all ICs in SB1 and 12 ICs in SB2. Lastly, [Fertilizer substitution] contributed to savings in 10 ICs for SB1 and 11 for SB2 (**Section 2.1** in **S2**).

3.2. Most relevant parameters for environmental performance

The results from the DQ assessment, sensitivity ratio calculation, and analytical uncertainty analysis were together used to identify the most relevant parameters for every alternate pathway and 16 ICs. Most of the 97 parameters had a 'basic' DQ (**Table S3.1**).

Table 3 shows the identified most relevant parameters in the *CC* for the five alternate pathways, with their level of criticality obtained for both methods. Based on Method-1, SP1 had 'very critical' parameters from [Waste to energy] and [Energy substitution] processes and SP2 parameters from [Material substitution], [Recycling plant process], and [Waste pre-treatment]. For both SB1 and SB2, parameters related to [Organic treatment process] yielded 'very critical' and 'critical' results. In SR1, [Waste pre-treatment], [Energy substitution], and [Waste to energy] processes had 'very critical' parameters. Table 3 shows that the parameters identified in Method-2 were often also identified in Method-1, except for SP1. For instance, *MaterBiSR* (SP2), *MaterBiInComp* (SB1), *ADWaterContent* (SB2), and *DiesMBT* (SR1) were identified as most relevant in both methods.

After conducting a similar analysis for the remaining 15 ICs (**Section 3.1.4** in **S3**), the following eight parameters were identified either as 'critical' or 'very critical' (i.e., most relevant) in more than nine ICs for

[d] For calculating the % contribution of a grouped process to the overall impacts of an IC, the sum of absolute values of the result of each grouped process was divided by the absolute value of the grouped process in question.

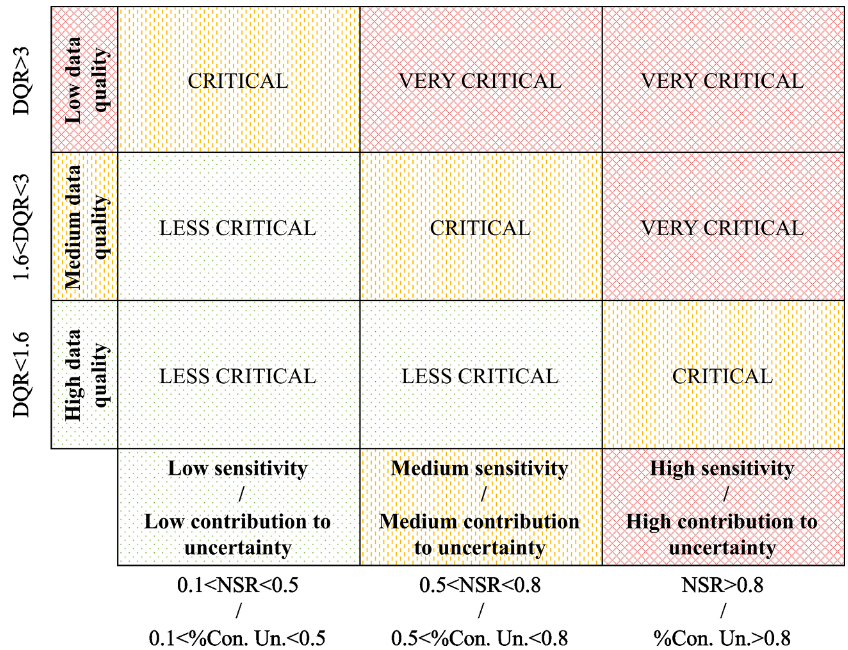


Fig. 2. Identifying the most relevant parameters by two methods. Method-1 – compared data quality ratio (DQR) with normalised sensitivity ratio (NSR), and Method-2 – compared DQR with the contribution of a parameter to overall uncertainty (%Con. Un.).

each alternate pathway using Method-1: (i) *HeatShare* – share of heat in 'Energy' produced during the waste-to-energy process (SP1 and SR1); (ii) *HeatSR* – substitution ratio of substituted heat from incineration (SP1 and SR1); (iii) *MaterBiSR* – substitution ratio of recycled BDPs (TPS-blend) (SP2); (iv) *RecShare* – share of BDP (TPS-blend) from the sorted output which is recycled (SP2); (v) *SortEff* – sorting efficiency of BDP (TPS-blend) in MRF (SP2); (vi) *MaterBiInComp* – fraction of BDP (TPS-blend) landing in compost (SB1); (vii) *VegWasteInCompost* – fraction of food waste landing in compost (SB1); and (viii) *ADWaterContent* – required water-content by AD plant (SB2). Similarly, for Method-2, *HeatShare* (SP1)*, MaterBiSR* (SP2)*, MaterBiInComp* (SB1)*,* and *ADWaterContent* (SB2) were identified as most relevant in more than nine ICs. **Table S3.2** includes the detailed results.

3.3. Uncertainty analysis

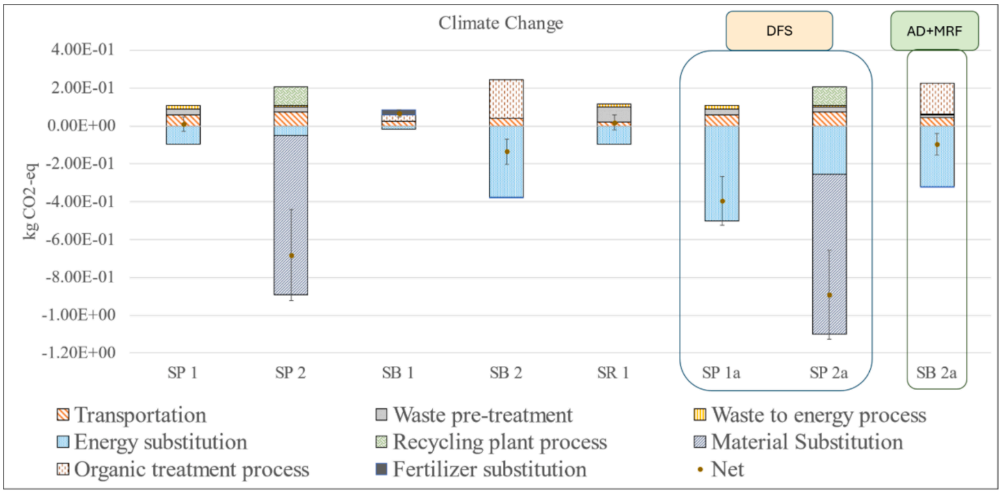
The results of uncertainty propagation were represented by error bars (Fig. 3), which highlight the variability of input data. Overall, for each alternate pathway, there was a grouped process which contributed to > 50 % of the overall uncertainty. For instance, SP1 had [Energy substitution] contributing to > 50 % overall uncertainty in 10 ICs, while for SP2 it was [Material substitution] for 15 ICs. Similarly, for SB1 and SB2 it was [Organic treatment process] for all ICs, and for SR1 there were two processes: [Waste pre-treatment] and [Energy substitution]. The detailed results are presented in **Figure S3.8**. The contribution of grouped processes to the overall uncertainties of an IC was calculated based on the analytical uncertainty calculations for identifying the most uncertain parameters.

This section discusses the processes contributing (and their parameters[e] from **Figure S3.6**) to > 80 % of the overall uncertainties in *CC*, *HT-C*, and *RU-EC* (Fig. 4); the detailed calculation for the remaining ICs is presented in **S3** (**Section 3.1.3**). For the three studied ICs, uncertainty was highest for SP2, followed by SB2, and least in SB1; while it was identical for SP1 and SR1. In all three ICs, SP1 had three processes contributing to > 80 % of overall uncertainty: [Energy substitution] (HeatSR)[e], [Transportation] (DistIncin)[e], and [Waste to energy] (HeatShare)[e]. Whereas in SP2 and SB2 it was [Material substitution] (MaterBiSR)[e] and [Organic treatment process] (ADWaterContent)[e], respectively. SB1 had [Organic treatment process] (MaterBiInComp)[e], [Transportation] (DistComp)[e], and [Energy substitution] (HeatSR)[e] processes contributing to > 80 % uncertainties in *CC* and *RU-EC*, and in *HT-C* they were [Organic treatment process] (MaterBiInComp)[e] and [Fertilizer substitution] (PChemFerSR)[e]. Lastly, for SR1 they were [Energy substitution] (HeatSR)[e], [Waste to Energy] (HeatShare)[e], and [Waste pre-treatment] (DiesMBT)[e] for *CC* and *RU-EC*, and [Energy substitution] (HeatSR)[e] and [Waste to Energy] (HeatShare, ElecShare)[e] for *HT-C*.

3.4. Sensitivity scenario analysis

This section presents the results of the DFS scenario (SP1a and SP2a) for *CC*, *HT-C*, and *RU-EC* (Fig. 3a-c), while the remaining results for 13 ICs are presented in **S2** (**Section 2.2.1**). In this scenario, the results of the [Energy substitution] process were drastically affected. In *CC*, for instance, the contribution of this process to overall environmental savings in SP1a and SP2a increased from 48 % to 82 % and 5 % to 20 %, respectively. This change is attributed to the shift from marginal heat substitution to the substitution of coal in the cement industry. In both cases, the elemental flow remained as 'avoided fossil CO_2 emissions' (as in SP1 and SP2). A similar trend was observed in *RU-EC*, where the savings increased from 48 % to 82 % and 3 % to 14 % in SP1a and SP2a, respectively. The most contributing elemental flow was 'avoided coal consumption'. Contradictorily, in *HT-C* the environmental savings decreased from 71 % to 55 % and 9 % to 5 % for SP1a and SP2a, respectively. This change was attributed to the reduction in 'avoided benzo(a)pyrene emissions', possibly due to the transfer coefficients adopted for substitution ratio calculation (**Section 1.4.4.2** in **S1**). Benzo (a)pyrene is a toxic and carcinogenic polycyclic aromatic hydrocarbon, which is often found in emissions from wood and coal combustion

[e] The parameter contributing most to the overall uncertainty of a grouped process is shown inside the round brackets () based on the result of analytical uncertainty analysis shown in Figure S3.6 in section 3.1.3 of the supplementary material. For example, 'HeatSR' is the parameter contributing most to the grouped process [Energy substitution] in case of SP1 for all three ICs. The description of the parameters is provided in Table 3, except for PChemFerSR, which is 'Phosphorous fertilizer substitution ratio'.



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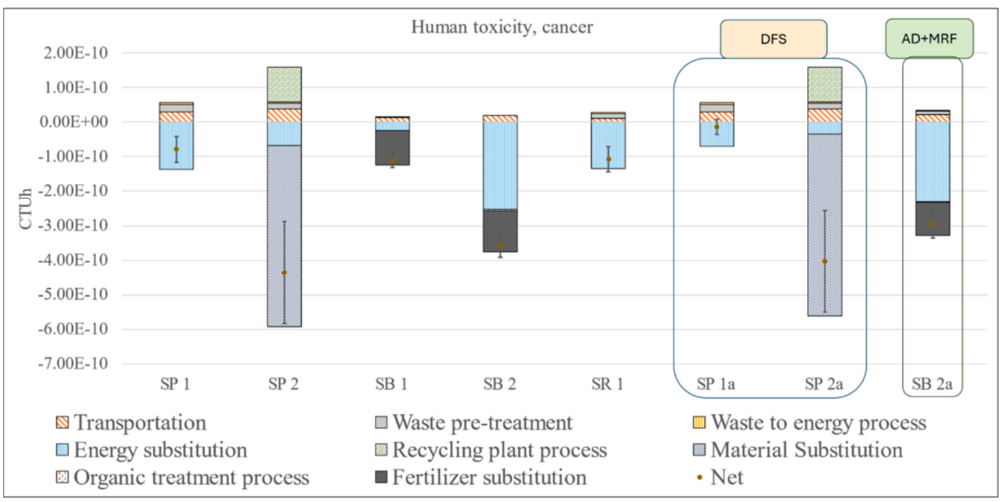

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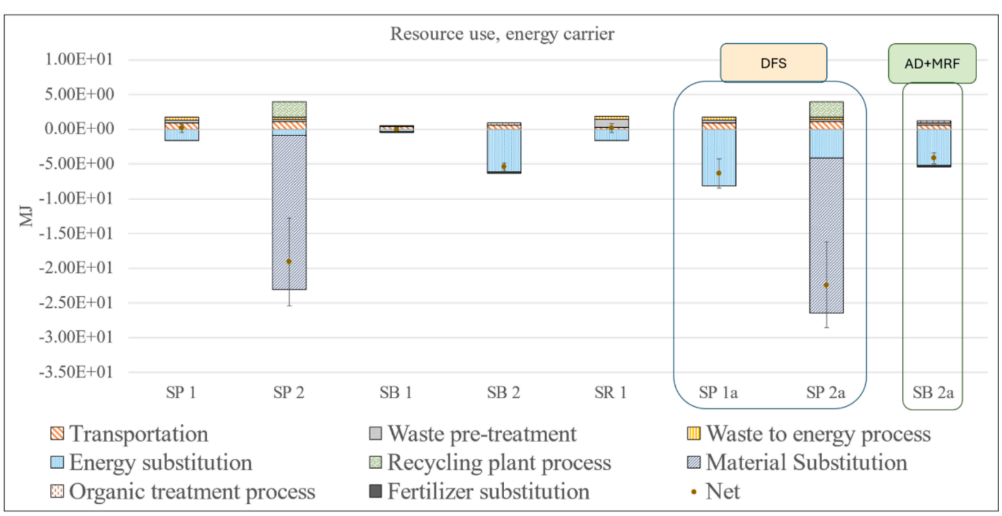

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Fig. 3. Results of characterisation for comparison of baseline scenario (SP1, SP2, SB1, SB2, SR1) with Direct fuel for process-heat substitution (DFS) scenario (SP1a, SP2a) and Anaerobic digestion with MRF (AD + MRF) scenario (SB2a) for three impact categories, namely (a) Climate Change; (b) Human toxicity – cancer; (c) Resource use – energy carrier. The results for the rest of the impact categories are discussed in the supplementary material (**Section 2.1** in **S2**). Each coloured stack in an individual bar represents the net value of relevant grouped processes and the net value of the bar is represented by a 'dot', where the negative values are termed as savings and the positive values as load. The results of uncertainty propagation were represented by error bars. SP1 – packaging waste pathway plus incineration; SP2 – packaging waste pathway plus mechanical recycling; SB1 – biowaste pathway plus composting; SB2 – biowaste pathway plus anaerobic digestion; SR1 – residual waste pathway plus incineration; SP1a – packaging waste pathway plus direct fuel for process-heat substitution (scenario analysis); SP2a – packaging waste pathway plus mechanical recycling and direct fuel for process-heat substitution (scenario analysis); SB2a – biowaste pathway plus anaerobic digestion with MRF (scenario analysis).

Table 3
The most relevant parameters for the five alternate pathways for the climate change impact category identified using (Method-1) DQR and NSR; and (Method-2) DQR and % contribution to analytical uncertainty. Parameters highlighted with bold text were identified as 'critical' or 'very critical' in one of the two methods, whereas parameters highlighted with bold and italicised text were identified as 'critical' or 'very critical' in both methods. AD – anaerobic digestion; DQR – data quality ratio; NSR – normalised sensitivity ratio; MBT – mechanical biological treatment; MRF – material recovery facility; RDF – refuse-derived fuel; SP1 – packaging waste pathway plus incineration; SP2 – packaging waste pathway plus mechanical recycling; SB1 – biowaste pathway plus composting; SB2 – biowaste pathway plus anaerobic digestion; SR1 – residual waste pathway plus incineration; TPS – thermoplastic starch.

Alternate pathway	Grouped process	Parameter	Description	Criticality level	
				Method-1	Method-2
SP1	[Waste pre-treatment]	DiesRDF	Diesel consumed by RDF production plant	Less Critical	–
SP1	[Waste pre-treatment]	DiesSort	Diesel consumption by MRF	Less Critical	–
SP1	[Transportation]	DistIncin	Distance from MRF to incineration plant	Less Critical	Less Critical
SP1	[Transportation]	DistMRF	Distance from household to MRF	Less Critical	–
SP1	[Waste to energy]	ElecShare	Share of electricity in energy part of material transfer in waste to energy plan unit process	Less Critical	–
SP1	[Energy substitution]	ElecSR	Substitution ratio of substituted electricity from incineration	Less Critical	–
SP1	[Waste to energy]	**HeatShare**	Share of heat in energy part of material transfer in waste to energy plan unit process	**Very Critical**	Less Critical
SP1	[Energy substitution]	**HeatSR**	Substitution ratio of substituted heat from incineration	**Very Critical**	Less Critical
SP2	[Material substitution]	***MaterBiSR***	Substitution ratio of recycled biodegradable plastic (TPS-blend)	**Very Critical**	**Very Critical**
SP2	[Recycling plant process]	**RecShare**	Share of biodegradable plastic (TPS-blend) from the sorted output which is recycled	**Very Critical**	–
SP2	[Waste pre-treatment]	**SortEff**	Sorting efficiency of biodegradable plastic (TPS-blend) in MRF	**Very Critical**	Less Critical
SB1	[Transportation]	DistComp	Distance from household to compost plant	Less Critical	–
SB1	[Organic treatment process]	HeatShare	Share of heat in energy part of material transfer in waste to energy plan unit process	Less Critical	–
SB1	[Energy substitution]	HeatSR	Substitution ratio of substituted heat from incineration – for waste incineration scenario.	Less Critical	–
SB1	[Organic treatment process]	MaterBiDegRateComp	Volatile solids & carbon degradation rate of biodegradable plastic (TPS-blend) in compost	Less Critical	–
SB1	[Organic treatment process]	***MaterBiInComp***	Fraction of biodegradable plastic (TPS-blend) landing in compost	**Very Critical**	**Critical**
SB1	[Fertilizer substitution]	NChemFerSR	Nitrogen fertilizer substitution	Less Critical	–
SB1	[Organic treatment process]	NDegRateComp	Nitrogen degradation rate of biodegradable plastic (TPS-blend) and food waste in compost	Less Critical	–
SB1	[Organic treatment process]	**VegWasteInCompost**	Fraction of food waste landing in compost	**Critical**	–
SB2	[Organic treatment process]	***ADWaterContent***	Required water content by AD plant	**Very Critical**	**Very Critical**
SB2	[Organic treatment process]	MethADStorage	CH_4 emissions during storage	–	Less Critical
SR1	[Waste pre-treatment]	***DiesMBT***	Diesel consumption by MBT plant	**Very Critical**	**Critical**
SR1	[Transportation]	DistResidual	Distance from household to residual waste treatment	Less Critical	–
SR1	[Waste to energy]	ElecShare	Share of electricity in energy part of material transfer in waste to energy plan unit process	Less Critical	–
SR1	[Energy substitution]	ElecSR	Substitution ratio of substituted electricity from incineration	Less Critical	–
SR1	[Waste to energy]	**HeatShare**	Share of electricity in energy part of material transfer in waste to energy plan unit process	**Very Critical**	Less Critical
SR1	[Energy substitution]	**HeatSR**	Substitution ratio of substituted heat from incineration	**Very Critical**	Less Critical

(Bieser et al., 2012).

Including MRF in an AD plant (SB2a) reduced the environmental savings in 11 ICs, except for *PM*, *A*, *E-T*, *E-M*, and *LU*, where the savings increased (**Figure S2.7**). For the three ICs (Fig. 3a-c), the change negatively affected all the processes' impacts, with additional environmental load from [Waste pre-treatment] and [Waste to energy] processes; except for [Organic treatment process], where the environmental load decreased by 11–12 % due to the reduced process emissions resulting from reduced material quantity to be treated (as some part of the input is sorted out). For the remaining processes, the decline in savings was due to incineration of the falsely sorted-out waste. The results for the remaining 13 ICs are presented in **S2** (**Section 2.2.2**). Additionally, assuming that 100 % of TPS-blend BDP would be sorted out to be incinerated, the overall environmental savings for all the ICs

Fig. 4. Uncertainty analysis – identified grouped processes contributing most to the overall uncertainties in the three impact categories: Climate change; Human toxicity – cancer; Resource use – energy carrier. The % contribution to uncertainties by the grouped process is computed from the parameter's analytical uncertainty, which is calculated using global sensitivity analysis. The results are presented for the five alternate pathways. (a) SP1 – Packaging waste pathway plus incineration; (b) SP2 – Packaging waste pathway plus mechanical recycling; (c) SB1 – Biowaste pathway plus composting; (d) SB2 – Biowaste pathway plus anaerobic digestion; (e) SR1 – Residual waste pathway plus incineration.

dropped (**Figure S2.8**).

The third scenario analysis assessed the environmental performance of the five alternative pathways in the three energy scenarios. For WAM and WAM + scenarios, SP2 demonstrated maximum environmental savings in 13 ICs with [Material substitution] as the hotspot process; except for *HT-C, RE-MM*, and *WU* for WAM, where SB2 had maximum savings and *PM* (instead of *HT-C*) for WAM +, where SR1 had maximum savings (*WU* and *RU-MM* same as WAM), with [Energy substitution] as the hotspot process. The detailed results for all the ICs are provided in **S2** (**Section 2.2.3**). The major observations for *CC*, *HT-C*, and *RU-EC* are discussed as follows (Fig. 5a-c). WAM scenario had net savings for SP1, SP2, SB2, and SR1 in the 3 ICs. SB1 had net savings for the *HT-C* and *RU-EC* and a net load for the *CC*. WAM + scenario had net savings for all alternate pathways in *HT-C* but had net savings for SP2 and SB2, and net load for SP1, SB1, and SR1 for *CC* and *RU-EC*. SP1 had [Energy substitution] as the hotspot process for the three ICs in all three energy scenarios and the same elemental flow for *CC* and *HT-C*. Whereas in *RU-EC* it was 'avoided oil consumption' (instead of natural gas in WEM and WAM) resulting from the changed energy mix. SP2 had [Material substitution] as the hotspot process in all three ICs for the three energy scenarios and the same elemental flows. SB1 had the same hotspot processes in WEM and WAM + scenarios, with [Organic treatment process] hotspot process for *CC*, [Fertilizer substitution] for *HT-C*, and [Transportation] for *RU-EC*. However, in the WAM scenario, the hotspot process changed to [Energy substitution] for *CC* and *RU-EC*, with the elemental flows as 'avoided fossil CO_2 emissions' and 'avoided natural gas resource consumption', respectively. SB2 had [Energy substitution] as the hotspot process in all three ICs for the three energy scenarios, with the relevant elemental flows of 'avoided CO_2 emissions' in *CC*, 'avoided benzo(a)pyrene emissions' for *HT-C*, and 'avoided coal consumption' for *RU-EC*. SR1 had [Energy substitution] as a hotspot process for three ICs in the WAM scenario and had the same elemental flows as in the WEM scenario. However, in the WAM + scenario, [Waste pre-treatment] was the hotspot process for *CC* and *RU-EC*, with 'CO_2 emissions contributing' to the environmental load in *CC* and 'consumption of oil' in *RU-EC*. [Energy substitution] remained the hotspot process for *HT-C*, with the major elemental flows as 'avoided benzo(a)pyrene emissions'.

General observation on the uncertainty propagation (Fig. 5) shows that SP2 had the highest uncertainty compared to the other four pathways for all three scenarios and ICs. For SP1, SB1, and SR1, the results for *CC* and *RU-EC* from the WAM scenario were more uncertain than WEM and WAM + scenarios; while *HT-C* results had the lowest uncertainty for WAM and higher for WEM and WAM + scenarios. Lastly, SB2 had a constant uncertainty range for three scenarios in the three ICs. The contributing parameters to these uncertainties were discussed in the previous section. The results for the rest of the ICs are listed in **S2** (**Section 2.2.3**).

4. Discussion

4.1. Comparison with other studies

Quantitative comparison of LCA studies related to WM is challenging owing to the different goals, scope, assumptions, and modelling choices (LCA software, data inventory, DQ, impact assessment method, and uncertainty) (van Eygen et al., 2018). Nevertheless, the results of the present study were compared with three previous LCA studies assessing different disposal options for BDPs (TPS-blend).

The first study by Piemonte (2011), conducted a cradle-to-gate LCA of TPS using SimaPro7.2 software. This study also compared the environmental performance of different TPS waste disposal methods, namely: open-loop mechanical recycling (recycled product used to manufacture different products); closed-loop recycling (recycled product used to manufacture the same product); composting; AD; and incineration. The closed loop mechanical recycling consistently had maximum savings, in global warming potential (IPCC GWP 100a) and three categories from Ecoindicator 99 methodology (*Human health*, *Ecosystem quality*, and *Resources*). This coincides with the findings of the current paper, where SP2 contributed to maximum savings in most ICs. However, no uncertainty analysis was conducted for their study.

The second study by Rossi et al. (2015) conducted LCA for TPS assessing the environmental impacts of six disposal options, namely: mechanical recycling; industrial composting; AD; direct fuel substitution in industrial facilities; incineration; and landfilling. They used the IMPACT 2002þ impact assessment method, with *global warming potential*, *water withdrawal*, *ecosystem quality*, and *human health* as the studied ICs. Again, mechanical recycling was found to contribute to environmental savings in most ICs, like SP2. Unlike the previous study, this study addressed uncertainty analysis by scenario analysis; however, they did not conduct uncertainty propagation. In contrast, the present study conducted a comparatively detailed uncertainty analysis, by addressing the DQ, and most relevant parameters plus modelling uncertainties.

The third study by Cristóbal et al. (2023) conducted LCA for a functional unit containing five types of compostable plastic packaging (CPP), including TPS, and a reference flow to consider the 'dragging effect' in packaging waste and biowaste[f]. This study also used EASETECH software and had three alternate pathways – CPP collected as (i) biowaste, pre-sorted before biological treatment (composting and AD); (ii) packaging waste, sorted and mechanically recycled; and (iii) packaging waste, sorted and underwent biological treatment. They used the product environmental footprint method with eight ICs and assessed economic impacts using lifecycle costing. Similar to the present study, they concluded that the mechanical recycling scenario had the highest environmental benefits. Conversely, their sensitivity analysis covered different topics than the ones discussed in the present paper. Although they included a parametric uncertainty assessment, the present paper conducted a more detailed sensitivity and uncertainty analysis by identifying the most relevant parameters (out of 97) by considering DQ with sensitivity ratio and analytical uncertainty.

Thus, although in the three existing studies, the scope, modelling choices, and the studied impact assessment methods with their quantitative results were different from those in the present paper, the common conclusion was that mechanical recycling demonstrated the highest environmental performance.

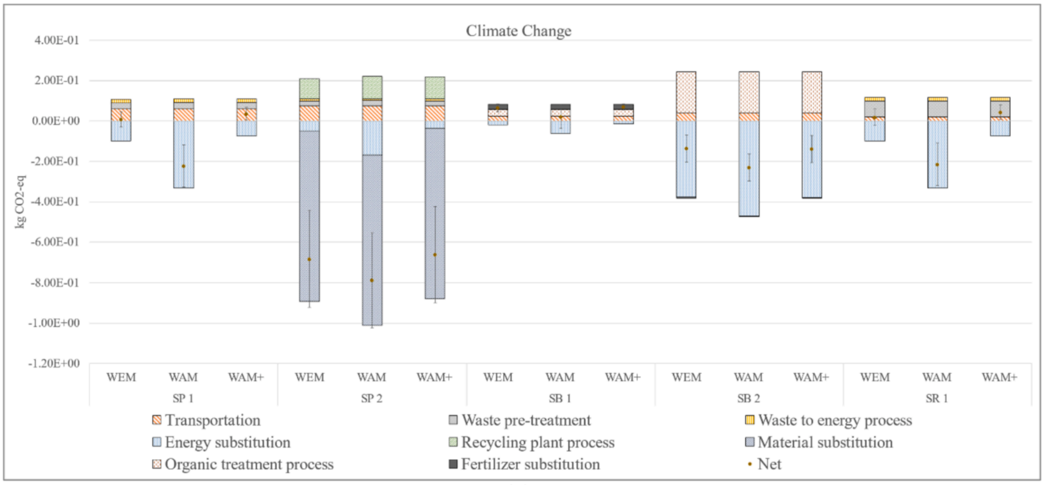
4.2. Technology aspects

Based on the results, SP2 demonstrated the largest savings for most ICs. However, for effective mechanical recycling, it is crucial to have the necessary infrastructure and enough waste volumes. For instance, installing an expensive NIR sorting machine (~200–400 EUR standard capital expenditure[g]) requires the availability of an adequate quantity of material-to-be-sorted in the waste stream for it to be profitable, which is quite challenging for BDPs owing to their lower volume and wide variety (Cristóbal et al., 2023; Siltaloppi & Jähi, 2021). Moreover, BDPs were often reported to contaminate the conventional plastic recyclate streams (Moshood et al., 2022; Nagy et al., 2018; Rujnić-Sokele & Pilipović, 2017), which necessitates the optimization of the NIR databases with the information of these plastics (Hasso von Pogrell, 2017; Mhaddolkar et al., 2024). Also, the industrial-level recycling facility for BDPs is nonexistent, which is necessary for the mechanical recycling of BDPs to be feasible (Maga et al., 2019). Lastly, the mechanically recycled BDPs have reduced mechanical properties and molecular weight after 4–6 cycles (Brüster et al., 2016; Ibáñez-García et al., 2021; Yarahmadi et al., 2016), and the inclusion of the washing step reportedly resulted in their degradation (Beltrán et al., 2018).

Moreover, while discussing the organic treatment of BDPs (with

[f] Refer to section 2.1 in this paper.

[g] STADLER Anlagenbau GmbH (personal communication, June 24, 2024).



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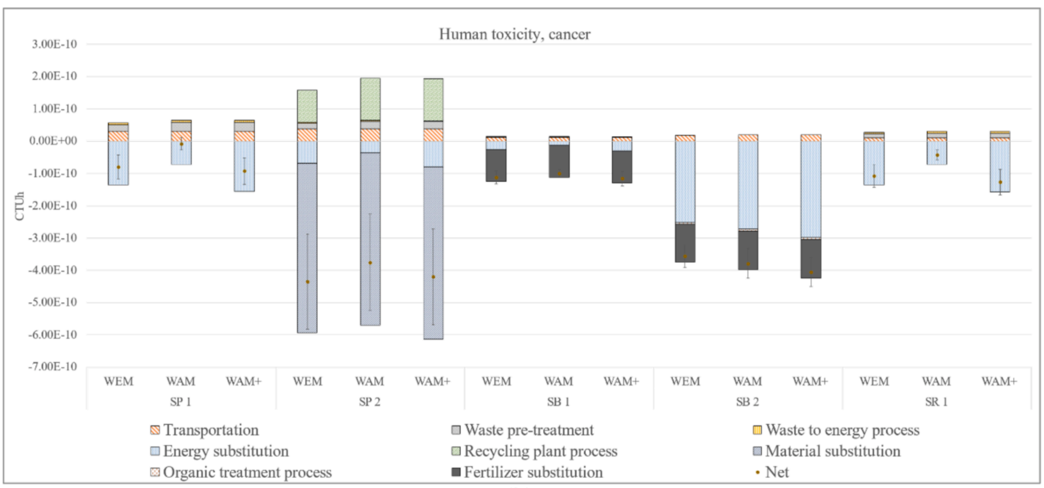

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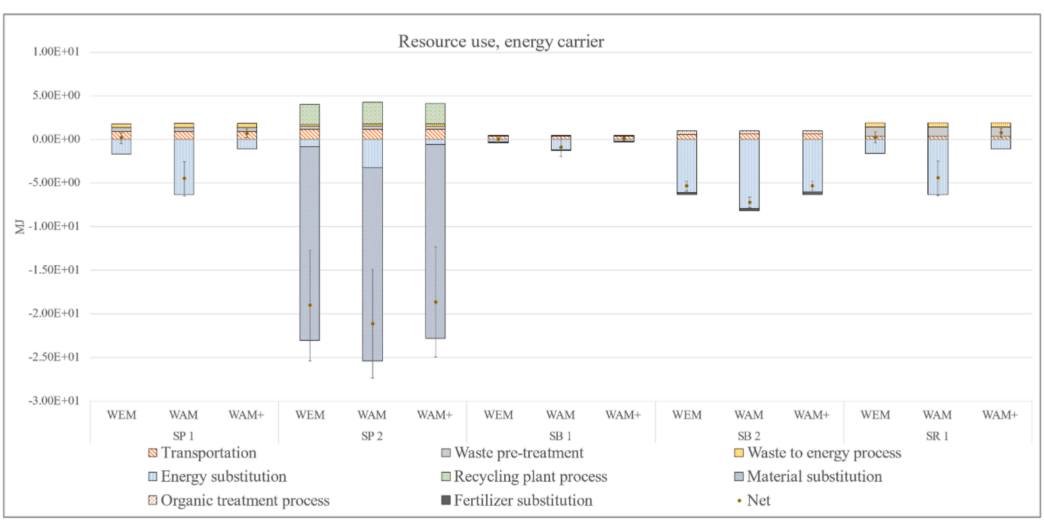

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Fig. 5. Results of characterisation for comparison of three energy scenarios (WEM, WAM, WAM +) for the following impact categories (a) Climate change; (b) Human toxicity – cancer; (c) Resource use – energy carrier. Throughout the manuscript, the WEM scenario is considered the default energy scenario. Each coloured stack in an individual bar represents the net value of relevant grouped processes and the net value of the bar is represented by a 'dot', where the negative values are termed as savings and the positive values as load. The results of uncertainty propagation were represented by error bars. SP1 – packaging waste pathway plus incineration; SP2 – packaging waste pathway plus mechanical recycling; SB1 – biowaste pathway plus composting; SB2 – biowaste pathway plus anaerobic digestion; SR1 – residual waste pathway plus incineration. Energy scenarios were based on (Baumann & Kalt, 2015), where, WEM – energy scenario with existing measures, baseline energy scenario; WAM – energy scenario with additional measures, reduced electricity imports and heat from natural gas; WAM + energy scenario with additional measures plus, zero electricity imports and heat from biomass.

composting and AD), the adverse effects of introducing BDPs from the compost/digestate to the soil were not considered; for instance, including the environmental impact of microplastics produced from BDPs might lead to different results (Accinelli et al., 2022; Fan et al., 2022). Moreover, the environmental impacts are considering the treatment of food waste with BDPs (except in BDP recycling facility and material substitution); thus, the environmental performance is mostly attributed to this combination of BDPs (TPS-blend) and food waste, and not solely BDPs. Additionally, the unacceptance of these BDPs by the biowaste treatment facilities ought to be addressed for this option to be feasible (Meeks et al., 2015), especially for the BDPs used as biowaste-collection-aids. Also, an important assumption for AD with MRF scenario was that the entire amount of BDP is not screened out, which is the status-quo (Utilitalia, 2020).

Lastly, DQ plays a vital role in LCA (Weidema & Wesnæs, 1996). Of the 97 parameters, input data of 75 parameters were classified as 'basic quality', eight as 'high quality', and 14 as 'data estimate'. Although the DQ was not very low, a parameter with basic-quality data combined with a medium/high sensitivity or uncertainty contribution was identified as critical (more relevant). For instance, in the case of 'MaterBiSR' (parameter for substitution of virgin TPS-blend BDP in SP2), the parameter had a basic DQ and high NSR plus a high contribution to the overall uncertainty for *CC*. Such a comparison indicates the robustness of the LCA results (Andreasi Bassi et al., 2017). Nevertheless, a higher DQ is always desirable in LCA (Clavreul et al., 2012; Weidema & Wesnæs, 1996).

4.3. Regulatory aspects

The results show that SP2 (collection in packaging waste for mechanical recycling) had maximum savings for most ICs, followed by SB2 (collection in biowaste for anaerobic digestion). These results coincide with the recommendations in the (EU policy framework on biobased, biodegradable and compostable plastics, 2022), where items other than tea bags, coffee capsules, fruit and vegetable stickers, and very lightweight carrier bags, were directed to be sent for material recovery (hence, to be collected with packaging waste). In the Austrian context, biodegradable supermarket carrier bags were excluded from the government-introduced plastic bag ban in 2020 (Federal Ministry Republic of Austria, 2020), and are often promoted to be reused as biowaste-collection-aid (NATURABIOMAT GmbH, 2024). However, both the Austrian national laws (Bundesministerium für Nachhaltigkeit und Tourismus, 2019) and the source separation guidelines (Holding Graz, 2023a, 2023b) instruct that BDP packaging should be collected with packaging waste. Although this is in line with the present results, the question remains: Is the system ready for handling this waste?

There are certain regulatory prerequisites for successful mechanical recycling of BDPs. For instance, labelling should be coordinated with the choice of waste treatment method; for instance, it is pointless to have a compostability label on the packaging to be mechanically recycled which might add to consumer confusion. Also, for mechanical recycling, BDP packaging needs to be covered under the extended producer responsibility scheme (Proposal for a Regulation on PPW Directive, 2022).

5. Conclusion

This study investigated potential environmental impacts related to confusion from consumers about how to appropriately source-segregate biodegradable plastic (BDP) waste. This was done based on a life cycle assessment of five pathways representing a range of waste collection and treatment technologies for starch-blend BDP (with a small fraction of inseparable food waste). Collecting these plastics as packaging waste for mechanical recycling (SP2) resulted in maximum environmental savings in most of the 16 impact categories (ICs); whereas collecting them with biowaste for anaerobic digestion (SB2) yielded the second-best net savings throughout all ICs. Results from scenario analysis show that their utilisation as an alternative fuel for process-heat production offered more net savings compared to incineration in SP1 (collection as packaging waste for incineration) and SP2, because of the increased savings from substituting coal consumption in the cement industry. Whereas including a mechanical sorting unit (MRF) in SB2, increased the environmental load, due to energy consumption by MRF. Lastly, the zero electricity imports plus heat from biomass energy scenario contributed to maximum savings for most alternative pathways in 16 ICs, because of the complete reliance on locally produced renewable electricity and 100 % district heat produced from biomass. In uncertainty analysis, the eight most relevant parameters were identified by comparing the data quality ratio, normalised sensitivity ratio, and contribution to analytical uncertainty. These identified parameters were marginal heat substitution ratio and share of heat in the produced energy in the waste-to-energy plant, substitution ratio and share of recycled BDPs from the sorted output in the recycling facility, sorting efficiency of BDPs in MRF, the fraction of BDPs and food waste in compost, and required water content in anaerobic digestor. While the results were aligned with the overall regulatory framework of waste management in the European Union (EU) (Waste Framework Directive, 2018), currently BDPs represent a very small share of the overall plastic flow in the EU. Moreover, their mechanical recycling performance requires sufficient sorting and recycling infrastructure adapted to BDPs as well as appropriate labelling and coverage under extender producer responsibility.

CRediT authorship contribution statement

Namrata Mhaddolkar: Writing – review & editing, Writing – original draft, Visualization, Resources, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Concetta Lodato:** Writing – review & editing, Validation, Supervision, Methodology, Conceptualization. **Alexia Tischberger-Aldrian:** Supervision, Conceptualization. **Daniel Vollprecht:** Writing – review & editing, Supervision. **Thomas Fruergaard Astrup:** Writing – review & editing, Supervision, Methodology, Conceptualization.

Declaration of competing interest

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

Acknowledgment

We are grateful to everyone who supported us in obtaining the data required for building the life cycle inventory.

The project C-PlaNeT has received funding from the European Union's Horizon 2020 research and innovation programme under the Marie Skłodowska-Curie grant agreement No. 859885.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2024.10.018.

Data availability

All the used data is included in the supplementary material.

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