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Research Paper

Comprehensive assessment of environmental and economic impacts of the entire EU waste management system

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A R T I C L E I N F O

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A B S T R A C T

This study presents a comprehensive model that quantifies environmental and economic impacts across the entire EU27 waste management system using Life Cycle Assessment and Life Cycle Costing. Addressing a crucial gap in official statistics, this model evaluates waste from both collected and generated perspectives, providing insights into the significant portion of waste (approximately 171 Mt annually) classified as mixed waste (also commonly referred to as "mixed residual waste" or "residual waste"). Results demonstrate that assessing waste from both perspectives effectively identifies inefficiencies in collection and treatment systems, revealing critical waste groups and high-impact processes. Although the waste management system as a whole shows net climate benefits, these are mainly related to metals, and the overall modest savings underscore the need for targeted improvements. Our findings highlight that net impacts on Climate Change are driven mainly by management of mineral waste and biowaste, and the incineration of plastic and textiles misallocated to mixed waste, while internal and external costs are highest for mineral waste and biowaste.

While current EU policies focus prevalently on plastic and textiles, our findings indicate that biowaste, mineral waste and sludge require renewed attention, with special efforts needed to reduce misallocations of recyclable waste to mixed waste. Improving reporting standards, especially to monitor mixed waste more accurately, could help identify misallocation patterns. This model offers policymakers a valuable tool for assessing scenarios, investment decisions, and advancing the EU's circular economy objectives.

1. Introduction

In the European Union (EU), waste management policy requires Member States (MSs) to align their national practices with EU-wide circular economy objectives. This is primarily achieved through EU directives, which set legally binding targets that each country must achieve. While the EU defines these targets, MSs can decide how to implement them in their national laws and systems. However, effective policies require comprehensive data and a clear understanding of waste flows, including both environmental and economic impacts, to properly prioritize high-impact waste streams and balance economic efficiency with environmental protection (De Laurentiis et al., 2024; Martinez-Sanchez et al., 2017). Albizzati et al. (2024) and Haupt et al. (2017) identify substantial gaps in current EU assessments, noting that most monitoring mechanisms rely on mass-based indicators (e.g., total waste generated per capita, recycling rate by weight, landfill diversion rate, and tonnes of waste collected), which do not capture the full range of impacts. This underscores the need for tools that go beyond mass flows and incorporate life-cycle-based assessments to support more effective prioritization.

At the MS level, waste prioritization is often based on qualitative assessments and limited or inconsistent quantitative metrics. For example, in Germany, the Environmental Agency (UBA, 2022) relied heavily on qualitative data when recently identifying critical waste streams for policy intervention (e.g., bureaucratic complexity, legal framework requirements, and recycling potential). Similarly, waste prioritization in France and Italy largely emphasize recycling and reuse targets or landfill reduction targets, applying limited quantitative environmental impact assessments across waste streams (European Environmental Agency (EEA), 2023a; 2023b). Despite efforts to prioritize waste, reliance on basic indicators limits the ability to fully address environmental and economic implications.

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Several studies have offered valuable insights into the environmental and economic performance of waste management in Europe, especially for well-documented streams like municipal waste (MW). However, earlier research has often assessed environmental and economic impacts separately, focusing only on a few waste types, or been limited to single MSs or regions. There is now a growing emphasis on more holistic assessments that integrate both types of impacts (European Commission, 2021), yet a comprehensive EU-level assessment covering all waste types is still lacking.

Andreasi Bassi et al. (2017) offer a broad environmental assessment using 11 impact indicators based on ILCD v1.0.6 characterization factors. While comprehensive, their study is limited to household waste in seven EU countries and excludes economic impacts. Bijleveld et al. (2022), on the other hand, assess ten waste streams across the EU but focus solely on greenhouse gas (GHG) emissions and do not disclose the underlying data. This lack of transparency hinders replicability and comparison, given that LCA results are highly sensitive to methodology and data choices (Allesch & Brunner, 2014; Kulczycka et al., 2016; Laurent et al., 2014; Merrild et al., 2008).

Albizzati et al. (2024) provide a more comprehensive assessment by presenting a model that evaluates both environmental and economic impacts of MW management at the EU level. The model includes 16 environmental impact categories, as well as financial costs, external costs, and generation of employment. However, it focuses exclusively on MW, which represents only about 9 % of total EU waste (Eurostat, 2025). To gain a more holistic picture of EU waste management, other major waste types, such as industrial and construction and demolition waste, must also be included.

Recent studies have addressed this gap by assessing the environmental and economic impacts of these underrepresented streams. For example, Caro et al. (2024) and Cristóbal et al. (2024), evaluated construction and demolition waste (CDW), and excavated soil and dredging spoil, respectively, which are streams that represent a significant share of total EU waste generation. Solis et al. (2024), and Garcia-Gutierrez et al. (2023) focus on textile and plastic waste, which are increasingly addressed through dedicated EU strategies such as the Circular Economy Action Plan and the EU Strategy for Sustainable and Circular Textiles. These studies offer more detailed, waste-type-specific analyses that complement system-wide approaches.

With no studies assessing all EU waste streams, it remains difficult for policymakers to secure an effective prioritization of high-impact waste types when designing circular economy policies. Moreover, Albizzati et al. (2024) highlight the importance of assessing waste from both collected and generated perspectives. The distinction between *collected* and *generated* waste is key to understanding actual waste flows (see Fig. 1). Collected waste refers to the material processed through formal systems, including both correctly sorted (target) materials and mis-sorted materials (impurities). In contrast, generated waste reflects the total amount of material discarded, regardless of sorting accuracy. Since collected waste data includes impurities and excludes misallocated or uncollected waste, it can significantly misrepresent actual waste generation. Understanding generated waste is therefore crucial for obtaining a more accurate representation of actual waste production.

Nevertheless, both collected and generated data are essential for a complete understanding of EU waste management. While collected waste data provides insights into the performance of waste treatment systems, generated waste data reveals inefficiencies of collection and sorting. Although the distinction between collected and generated waste is critical for accurate assessment, it is often unclear whether data are being reported on a collected or generated basis, as the terms are frequently used interchangeably. Albizzati et al. (2024) address this gap by proposing a methodology to estimate generated municipal waste (MW) based on collected amounts.

Building on the approach of Albizzati et al. (2024), this study presents a model designed to assess the environmental and economic impacts of the entire EU27 waste management system. The model is applied here to illustrate the impacts of waste management in the EU27 as of 2020. Specifically, this study (i) presents a comprehensive analysis of the environmental and economic impacts of the entire EU27 waste management system, for both collected and generated waste, highlighting the distinct insights gained from assessing waste from both perspectives; (ii) identifies the most critical waste streams in terms of Climate Change impacts and associated internal and external costs; and (iii) provides a detailed description of the methodology used to estimate generated waste amounts from officially reported collected data, along with all underlying data used in the system-wide model in the Supporting Information (SI). Insights from the results support efficient resource allocation toward high-impact areas, foster an economically sound approach to achieving EU-level sustainability goals, and highlight areas where further research is needed. The model is not intended to evaluate the performance of specific waste management technologies or the systems of individual MSs.

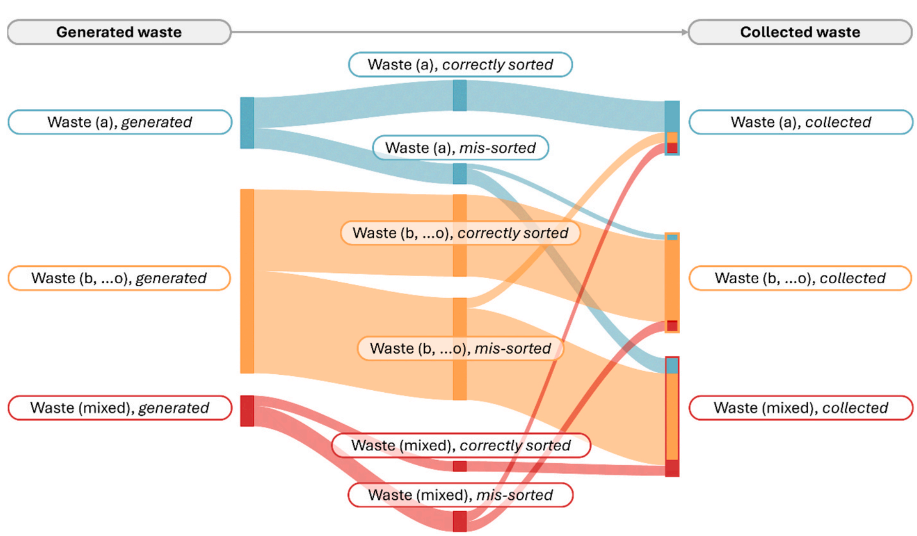

Fig. 1. Conceptual representation of the distinction between generated (left) and collected (right) waste. Waste (a) represents a specific non-mixed waste flow, while waste (b, …o) represents all other non-mixed waste flows, and waste (mixed) represents mixed waste (also commonly referred to as “mixed residual waste” or “residual waste”).

2. Materials and methods

2.1. Quantifying waste flows: Data sources and handling

2.1.1. Estimating generated amounts from collected amounts

The methodology presented in this study builds upon Albizzati et al. (2024) but diverges in underlying data sources and the approach for estimating generated waste amounts. Unlike Albizzati et al. (2024), which focuses on MW, this study encompasses all waste types in the EU27 using Eurostat's "Generation of Waste by Waste Category, Hazardousness, and NACE Rev. 2 Activity [env_wasgen]" dataset. Although labeled as "generation of waste," this dataset actually presents collected amounts (inclusive of target materials and impurities), as there is no standardized method for predicting generated waste. The [env_wasgen] dataset provides biennial data on 51 waste categories, which classify all waste collected across EU27 countries at both the EU and MS level, while distinguishing between hazardous and non-hazardous waste.

To estimate generated waste amounts, this study follows a six-step approach (see Fig. 2, or **Section S2**, SI, for a detailed elaboration), expanding upon the methodology introduced in Albizzati et al. (2024). Step 1 consolidates Eurostat's 51 waste categories into 16 broader "*waste groups*" based on material similarity, streamlining the dataset for impact assessment. For example, categories associated with metals (e.g., steel, aluminum, mixed metals) were consolidated under a waste group titled "metal waste," while categories linked to electronics (e.g., batteries, accumulators, discarded equipment) were grouped into "electronic waste." This resulted in the following 16 waste groups: glass waste, metal waste, paper and cardboard waste, plastic, textiles, wood, electronics, biowaste, mixed waste, discarded vehicles, soil, sludge, mineral waste, combustion residues, non-hazardous chemicals, and other hazardous waste.

Building on step 3 of Albizzati et al. (2024), which defines target material and impurity shares, this study further refines this process in steps 2–4. It uses the [env_wasgen] dataset along with Eurostat's "Packaging Waste by Waste Management Operations [env_waspac]" dataset, which provides data on the amounts of packaging waste collected for paper and cardboard, plastic, wood, metal, and glass. This allows for distinguishing between packaging and non-packaging target materials, and ultimately defining the shares of non-packaging target materials, packaging target materials and impurities (referred to in this paper as *"sub-groups"*).

Step 5, corresponding to step 4 in Albizzati et al. (2024), defines the material fractional compositions of each sub-group. Fig. 2, presents the literature sources used to obtain the sub-group shares and their material compositions. Steps 2–5 were modified for the mixed waste group to achieve a more detailed breakdown of sub-groups. Other waste groups subjected to modifications during these steps included the discarded vehicles and mineral waste (see **Section S2.4.1**, SI).

The final step (step 6) calculates the generated waste. While Albizzati et al. (2024) estimate generated waste as the ratio of collected waste and collection rates, this study instead estimates generated waste by tracking and reassigning impurities to their respective target waste groups. This deviation was necessary due to the lack of comprehensive collection rate data covering both municipal and non-municipal waste. Consequently, the collection rates obtained in this study are derived based on Eurostat values for collected waste and the estimated generated waste amounts (see **Section S2.5**, SI).

Lastly, calculating total generated biowaste differs from other waste groups due to the inclusion of biowaste treated through home composting, an informal practice typically not accounted for in collected waste data from municipalities or national authorities. To improve accuracy, home-composted biowaste is included in the final estimation (see **Section S2.1.1,** SI). Andersen et al. (2011) note that home composting can significantly contribute to waste diversion rates when widely adopted at the municipal level.

2.1.2. Waste treatment

Once collected and generated amounts are harmonized, one needs to track what treatment the collected waste undergoes, using the dataset titled "Treatment of Waste by Waste Category, Hazardousness, and Waste Management Operations [env_wastrt]." This dataset provides waste treatment data across three disposal types (landfilling, incineration, other disposal) and three recovery methods (energy recovery, recycling, backfilling). For simplification, this study consolidates these into four categories: recycling, backfilling, incineration (with energy recovery), and landfilling. Note that energy recovery and incineration are grouped together under the assumption that all waste directed to these methods undergoes incineration with energy recovery. Similarly, landfill and other disposal methods are combined, with the assumption that all waste categorized under these methods is sent to landfill, see **Section S2.1** (SI).

A comparison of the collected amount per waste group (e.g., glass collected), as reported in the [env_wasgen] dataset, and the corresponding waste treated (e.g., glass treated), as reported in the [env_wastrt] dataset, reveals a discrepancy: the amount of waste treated per waste group is consistently lower than the amount of the corresponding waste collected for the given year (i.e., 2020). This discrepancy may arise due to factors such as inconsistent data collection, temporary storage, waste exports, or variations in waste classification. Eurostat does not provide datasets on waste storage or methods to address data inconsistencies but provides data on the amount of waste exported by the EU27 each year (under the dataset titled, "Trade in Waste by Type of Material and Partner [env_wastrdmp]"). Therefore, this study makes the following assumption to correct for the inconsistencies:

$$W_{Col,i} = W_{Treat,i} + W_{Exp,i} \tag{1}$$

where $W_{Col,i}$ is the amount of waste collected of waste group i (where $i = [1 \ldots 16]$) (obtained from the [env_wasgen] dataset), $W_{Treat,i}$ is the treated amount of waste group i (where $i = [1 \ldots 16]$) (obtained from the [env_wastrt] dataset), and $W_{Exp,i}$ is the exported amount of waste group i (where $i = [1 \ldots 16]$) (obtained from the [env_wastrdmp] dataset). See **Section S3.1** (SI) for an overview of the exported waste categories and how they are integrated into the [env_wastrt] dataset. Finally, since this study focuses on waste generated and collected within the EU27, imported waste is excluded from the analysis.

2.2. Life Cycle Assessment and Life Cycle Costing

This study applies Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) to quantify the environmental and economic impacts of EU waste management.

The LCA follows ISO 14040/14044 standards (ISO, 2006a; 2006b) and uses the Environmental Footprint Life Cycle Impact Assessment method (EF, v3.0) (EC-JRC, 2012), covering 16 impact categories, including Climate Change, Ozone Depletion, Human Toxicity (cancer and non-cancer), Particulate Matter, Ionizing Radiation, Photochemical Ozone Formation, Acidification, Eutrophication (terrestrial, freshwater, and marine), Ecotoxicity, Land Use, Water Use, and Resource Use (fossil, minerals, and metals). The uptake/release of biogenic CO_2 was assigned a characterization factor equal to zero. The sequestered biogenic CO_2 in soils within the 100-year time horizon considered was assigned a factor equal to −1 for Climate Change, following Christensen et al. (2009).

The LCC aligns with the LCA in goal and scope, functional unit, and system boundaries, and quantifies the economic impacts following the methodologies for waste management economics described by Hunkeler et al. (2008) and Martinez-Sanchez et al. (2015). The LCC includes both an Environmental LCC (eLCC), which accounts for internal financial costs (notably operational and capital expenditures) and externalities that have been internalized (such as landfill and incineration taxes), and a Full Environmental LCC (feLCC), which incorporates both internal costs and monetized environmental externalities (also known as external

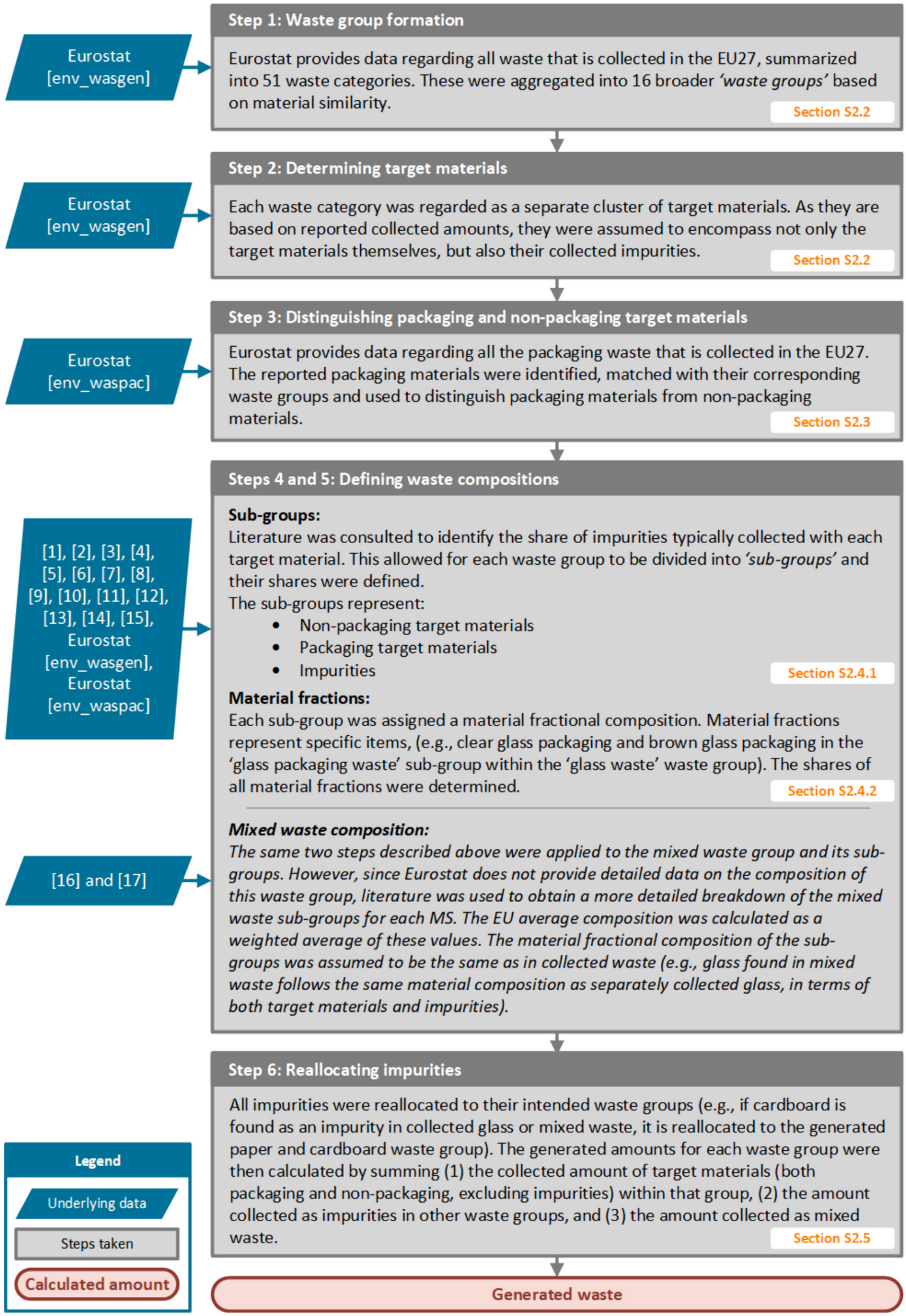

Fig. 2. Flowchart illustrating the methodology for estimating generated waste amounts, showing the underlying data sources (parallelograms) used to carry out each of the procedural steps (rectangles) necessary for calculating the generated amounts across all waste group (ovals). Orange text indicates where further details regarding each step can be found in the SI. References: [1] Accardo et al. (2023), [2] Basel Convention Technical Working Group (2000), [3] Biganzoli et al. (2015), [4] Boldrin (2009), [5] Boldrin & Christensen (2010), [6] Caro et al. (2024), [7] Edjabou et al. (2016), [8] Eriksen & Astrup (2019), [9] Faraca et al. (2019), [10] Götze et al. (2016), [11] Madsen (2021), [12] the Danish National Waste Database from the Danish Environmental Protection Agency, Miljøstyrelsen (2018), [13] Pivnenko et al. (2016), [14] Riber et al. (2009), [15] confidential datasets from an anonymous glass recycling facility (unpublished) (2022), and [16] Albizzati et al. (2024) and [17] Edjabou et al. (2021). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

costs) that are not yet reflected in market prices (Hoogmartens et al., 2014).

The inclusion of feLCC provides a more holistic assessment of waste management costs, aligning with the European Commission's Better Regulation Guidelines and Toolbox. Specifically, Tool #63 (Cost-Benefit Analysis) recommends monetizing externalities to ensure that policy decisions reflect the full societal costs and benefits of waste management strategies (European Commission, 2021; 2023a).

External costs are applied at the LCI (emission) level to avoid double counting, following the approach of De Bruyn et al. (2018). They cover air, water, and soil emissions but exclude social costs like time spent segregating waste, disamenities, and accidents. They are assigned to both foreground and background emissions, meaning they apply to total emissions (e.g., total CO_2), regardless of where they occur. Since EU-based cost factors are used, emissions occurring outside the EU may be overestimated. However, as the model focuses on the EU waste management system, this simplification is expected to have only a limited effect on overall results.

Both the environmental and economic assessments are performed employing the LCA-software EASETECH for its ability to track material, substance and energy balances (Clavreul et al., 2014). All economic data was adjusted for inflation to EUR_{2020} using the Harmonized Indices of Consumer Prices (Eurostat, 2024).

2.3. Case study

This study examines the environmental and economic impacts of 16 consolidated waste groups, which together represent the entire waste production in the EU27, addressing two functional units. First the life cycle impacts of 1 tonne (wet weight) of the 16 waste groups is quantified as *collected*, and then, as *generated*. Collected amounts refer to the quantity of waste collected in 2020, including both target materials and

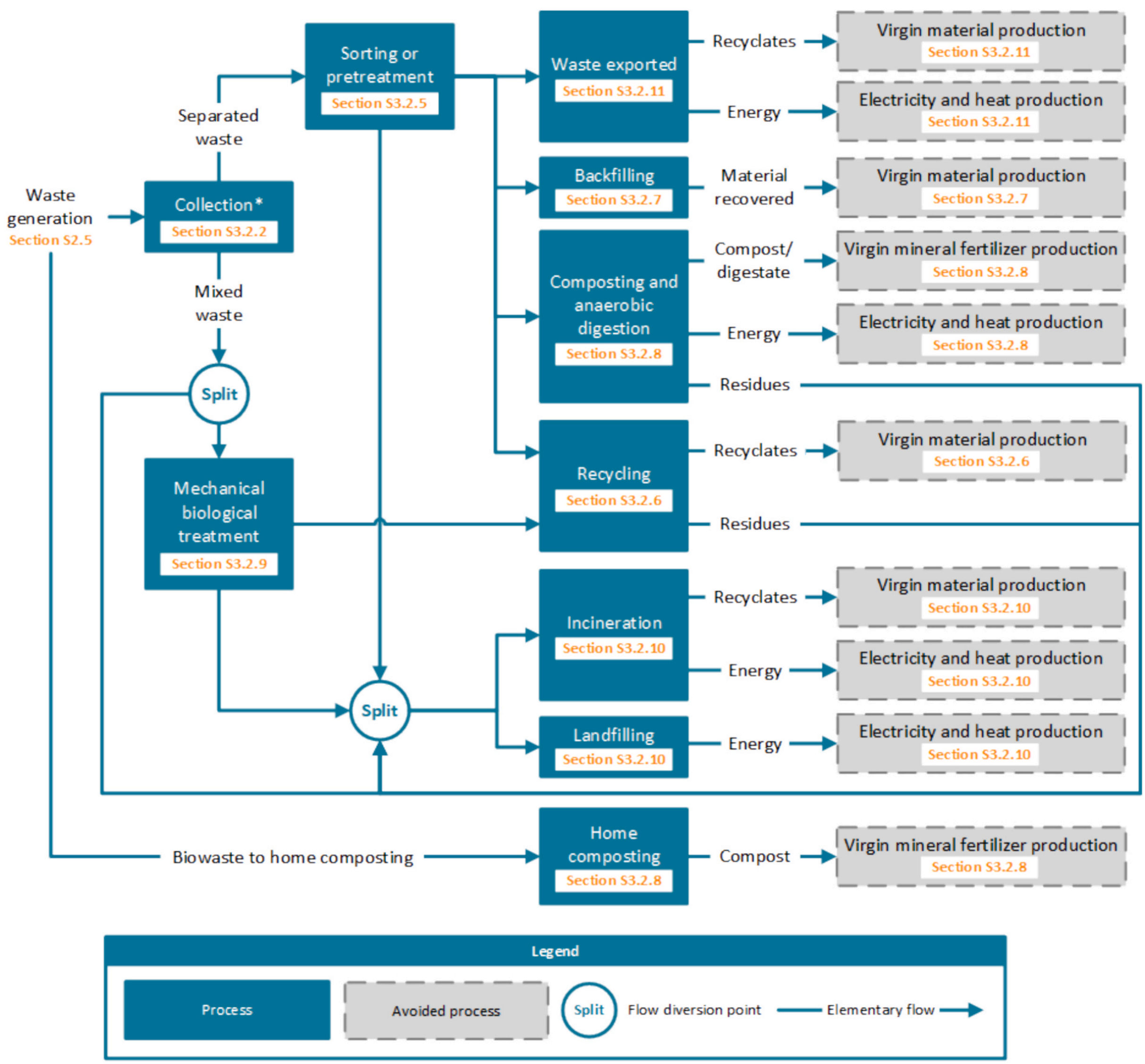

Fig. 3. Generic depiction of the system boundaries for the waste management system in the EU27. Green continuous boxes represent induced processes, while grey dashed boxes represent avoided processes (e.g. substitution of material or energy via waste-derived products). Green arrows represent the flow of materials, compost, digestate and energy. Splits represent flow diversion points where materials are merely redirected without inclusion of any fuel or material consumption. Orange text indicates where detailed information can be found in the SI regarding the modeling and underlying inventory for each process. *Some waste groups are collected with collection vehicles and transported to sorting facilities, while others are collected on-site, where they are processed through pretreatment activities such as demolition, excavation, dredging or dewatering before being transported for further treatment (e.g., recycling, backfilling, incineration, etc.). A detailed description of these processes and the specific waste groups they apply to can be found in **Sections S3.2.2 and S3.2.3** in the SI. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

impurities, as reported in the [env_wasgen] dataset. In contrast, generated waste refers to the quantity of target material discarded in total, which is estimated by applying the methodology explained in **Section 2.1.1**.

The system boundaries cover all activities involved in the lifecycle of waste after generation, including collection, transport, sorting, recycling, backfilling (considered as a form of recovery), mechanical biological treatment (MBT), incineration, and landfilling, including household processing (home composting) (see Fig. 3). To reflect the complete lifecycle, and environmental impacts of waste generated and collected within EU27, exported waste impacts are included, covering all processing and transport activities to non-EU27 destinations, while imported waste is excluded. Following the zero-burden approach (Ekvall et al., 2007), waste is considered to carry no prior environmental burdens.

Waste management systems are multifunctional, producing secondary raw materials (e.g., recyclates) and energy (e.g., electricity) while managing waste. These outputs are accounted for using system expansion, where co-products are credited by assuming they replace equivalent market products represented by their market supply processes. To reflect the situation as of 2020, the substituted processes are modeled as average market processes using an attributional approach. Substitution factors, which define the proportion of primary material that is effectively replaced by secondary materials, are determined as the ratio between the respective qualities. This study applies substitution factors based on literature values (see **Sections S3.2.6–3.2.8,** SI).

2.4. Modeling foreground and background systems

The model represents the average EU27 waste management system for 2020 (Fig. 3), designed to assess both collected and generated waste amounts without accounting for variations across individual MSs. Initially, most waste groups are collected, except for inert materials, wood, sludge, non-hazardous chemicals, and other hazardous wastes, which undergo alternative processes like demolition or excavation to prepare them for treatment or transport. Once collected, the waste undergoes sorting or pretreatment, where it is directed toward recycling, recovery (e.g., backfilling), biological treatment (i.e., composting or anaerobic digestion), incineration (assumed with energy recovery), landfill (assumed with gas capture), or exported outside the EU27.

Mixed waste is generally sent to incineration or landfill; however, a share of biological waste within the mixed waste is separated at MBT facilities, which divert recyclable or recoverable materials to the appropriate treatment pathways.

In the case of biowaste, an additional generated portion is assumed to avoid collection entirely and undergo home composting, reflecting informal waste management practices in the EU27. Exported waste follows similar treatment processes as within the EU but with a different energy mix. Incineration is assumed to involve a combination of incineration without energy recovery and open burning, while landfill practices range from controlled landfills to open dumps. Inventories for each modeled process are provided in **Section S3.2** (SI), allowing for a comprehensive overview of the treatment and recovery practices used across the different waste groups.

The background system (e.g., production and supply of energy and materials needed to manage waste), is modeled using data from Ecoinvent 3.8 database (Wernet et al., 2016), applying the Allocation at the Point of Substitution (APOS) system model, though the model can also accommodate alternative datasets. For this assessment, an average electricity and space heat mix for EU27, calculated for the year 2020 from the GECO (Keramidas et al., 2023), is applied (see **Section S2.2.1,** SI). The electricity mix for exported waste treated outside the EU27 was calculated using 2020 data from the International Energy Agency (IEA) (2020). A weighted average electricity mix was determined based on the top five EU27 export destinations (Turkey, India, Egypt, Switzerland, and the United Kingdom). For the space heating mix, an average of natural gas, oil, and coal was assumed (see **Section S2.2.11,** SI).

2.5. Sensitivity analysis

The sensitivity analysis focused on the primary unknown variable: the composition of mixed waste, as it can significantly affect the distribution of the generated waste groups, when converting from collected to generated amounts. Furthermore, as highlighted by Bisinella et al. (2017), waste composition plays a central role in shaping the environmental impacts of treatment, recycling, and disposal processes. As described in **Section 2.1.1**, the composition of mixed waste was estimated based on the results of Albizzati et al. (2024), as no data was available for 2020 (i.e., the “default” case). However, the “Early Warning Assessments” published by the EEA (2022) provide data on municipal mixed waste compositions for all MSs, except Romania and Poland. These compositions are based on assessments conducted by individual MSs between 2010 and 2019, though they were all reported in the 2022 EEA reports. Assuming these compositions are representative of 2020 waste generation, this data, provided a second avenue for estimating mixed waste composition.

To derive a new composition, MS-reported data was compiled, and a weighted average distribution between the different groups of waste found in the mixed composition waste was calculated based on 2020 collection volumes. The reported groups of waste included paper and cardboard, metals, glass, plastic, biowaste, textiles, wood, and other waste. Some MSs further differentiated between ferrous metals and aluminum within the metals category, as well as food waste and garden waste under biowaste. Additionally, while some MSs reported composite packaging and electronics separately, these were assumed to be part of the “other waste” category due to inconsistencies in reporting.

For the material fractional breakdown of each group of waste, the same distributions defined for the original mixed waste composition were retained (as detailed in **Section S2.4.2, SI**). Consequently, while the overall share of each waste group within the mixed waste differed between the default- and EEA-based compositions, the internal composition within each group remained consistent.

Using the EEA data, an alternative mixed waste composition was derived and used to re-estimate total waste generation for 2020, following the methodology outlined in **Section 2.1.1**. Finally, this alternative waste composition was applied to recalculate the LCA results and assess their sensitivity to changes in the mixed waste composition.

3. Results and discussion

This section presents the estimated composition of waste generated in the EU27 for 2020, detailing the mass balance of both collected and generated waste amounts, as well as the environmental and economic impacts across selected impact categories. Results are shown for both collected and generated amounts per tonne for each waste group and per tonne of total waste, with each group weighted according to its contribution to the EU27′s total waste in 2020. The weighting procedure adjusts the amounts of each waste group to reflect their actual share of the total waste generated or collected. This means that non-weighted results present impacts per tonne of each waste group individually, while weighted results represent impacts per tonne of all waste groups combined, ensuring that each group contributes proportionally to the total waste generated or collected in 2020.

The analysis uses average waste compositions, treatment distributions, and efficiencies, without factoring in variations across MSs, meaning all results apply at the EU27 level. Based on the findings, critical waste streams are identified, followed by a discussion of policy implications, areas for future research, and the study’s overall limitations.

3.1. Addressing the data gap concerning waste generated amounts

Fig. 4**a** presents the waste composition as a percentage of both total collected (2008 Mt) and generated waste (2014 Mt). To facilitate direct comparison, both totals have been scaled proportionally so that their sum is 2000 Mt. The minor difference arises from generated biowaste treated through home composting, which is not included in the collected waste.

The EU27 waste management data for 2020 highlights notable differences between collected and generated waste for several groups. Higher generated amounts relative to collected amounts indicate inefficiencies in capturing target materials, whereas higher collected than generated amounts suggest contamination by impurities.

Mixed waste particularly illustrates inefficiencies: although 171 Mt are collected, only 10.1 Mt likely represent actual mixed waste, indicating significant misallocation. Textile waste shows a large gap, with approximately 10.3 Mt generated versus only 1.95 Mt collected, and polymeric waste shows 38.2 Mt generated compared to 22.0 Mt collected, underscoring critical management issues for these materials.

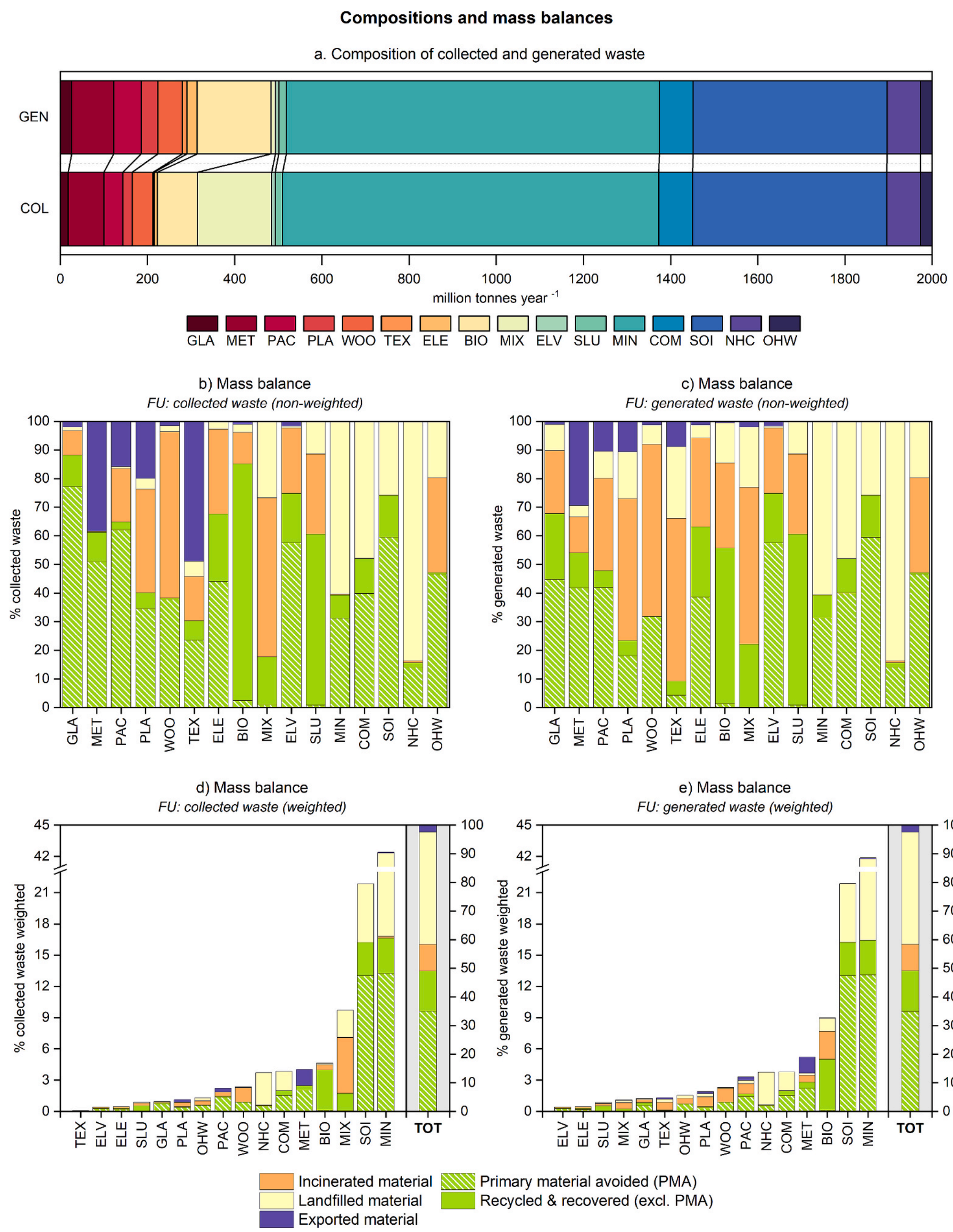

Fig. 4. Composition of waste reported as a percent of the total amount of waste collected and generated in the EU27 in 2020 (3a). Exact values for collected and generated amounts are available in **Tables S1 and S6** (SI), respectively. Mass balances per tonne of individual waste groups are shown as a percent of collected waste (3b) and generated waste (3c), and as a percent of collected waste, weighted (3d), and generated waste, weighted (3e). The waste groups are depicted with the following abbreviations: biowaste (BIO), combustion waste (COM), electronic waste (ELE), discarded vehicles (ELV), glass waste (GLA), metal waste (MET), mineral waste (MIN), non-hazardous chemical waste (NHC), other hazardous waste (OHW), paper and cardboard waste (PAC), polymeric waste (PLA), mixed waste (MIX), sludge (SLU), soil (SOI), textile waste (TEX) and wood waste (WOO). Other abbreviations include collected waste (COL), functional unit (FU), generated waste (GEN) and total waste (TOT).

3.2. Distribution of treatments: Rates of recycling and primary material avoided

Fig. 4**b-e** illustrates the treatment distribution for each waste group, showing the shares of waste recycled, recovered, incinerated, landfilled, and exported, both per tonne of each waste group (**4b** for collected, **4c** for generated) and weighted by total waste (**4d** for collected, **4e** for generated). 'Recycled & recovered' and 'Primary material avoided' (PMA) together represent the total amount of material that has undergone recycling and has been recovered. Within this total, PMA refers to the portion that actually substitutes primary materials (e.g., recycled steel replacing virgin steel in manufacturing), while 'Recycled & recovered' represents the remaining share that has been recycled or recovered but does not replace primary material production due to lower quality.

For collected waste (Fig. 4**b and d**), higher recycling rates occur than for generated as they ignore the material that was misallocated to other waste groups during initial segregation at household-level and collection, resulting in inflated recycling rates. This trend is particularly evident for glass waste, which shows a very high recovery and PMA rate, indicating potential for recovering material lost due to initial segregation inefficiencies.

Polymeric and textile waste also show large discrepancies between collected and generated recovery rates. However, unlike glass waste, their recycling rates in the collected waste stream are not as high, suggesting that while there is a need to improve the initial segregation at household level to increase collection rates, there is also a need for advancements in sorting and recycling technologies for treating these waste groups. Collected textile waste has the highest export rate, suggesting that an increase in textile collection volumes could lead to more export, if appropriate technologies are not in place in the EU27 to process them, including an available market for secondary material. Both waste groups face challenges due to their complex compositions, often involving mixtures of polymers or fibers that are difficult to recycle. Their heavy dependence on external markets also raises concerns about the long-term sustainability of their management systems, particularly as international markets become increasingly restrictive. Moreover, the reliance on exports complicates the EU's ability to monitor the actual fate of these waste groups, potentially leading to improper management.

Biowaste has the lowest PMA as most of the collected biowaste is transformed into CO_2 (directly or via energy production), limiting the substitution of fertilizing and soil-amending materials via compost and digestate. However, the substitution of primary energy carriers is not accounted for in Fig. 4 (also applying to incineration), but is accounted for as environmental savings in the LCA and economic revenues in the LCC, in **Sections 3.3** and **3.4**, respectively.

For electronics and discarded vehicles, high reported recycling rates (100% and 97%, respectively; see Table S8, SI) face uncertainties due to informal handling and illegal exports (European Commission, 2023a; Huisman et al., 2015). Gaining insight into the estimated amounts that are mismanaged would provide a more accurate depiction of the actual recycling rates for these waste groups.

Although most mineral waste is collected, over half (61%) is landfilled, reflecting limited recycling opportunities. Various studies (notably Caro et al. (2024) and EEA (2020)) suggest that recycling of mineral waste, particularly from construction and demolition, is hindered by high treatment costs relative to landfilling, as well as an underdeveloped market for secondary materials. This issue is compounded by a lack of harmonized EU regulations, such as inconsistent end-of-waste criteria across MSs and varied landfill tax policies, with much reliance on voluntary guidelines rather than binding legislation.

Weighted results in Fig. 4**d** and **e** highlight how soil and mineral waste dominate waste volumes, underscoring the importance of diverting them from landfills. Across all waste groups, the EU27 recycling rate for generated waste for 2020 (Fig. 4**e**, 'TOT') is 49% (989 Mt), with a PMA rate of about 35% (702 Mt). Landfilling and incineration rates are 39% and 9.3%, with 2.4% for exports.

3.3. Environmental impacts

This section presents the results for Climate Change (see Fig. 5). The results for the remaining 15 impact categories can be consulted in **Section S4.1** (SI).

3.3.1. Climate Change

Fig. 5**a** shows that for collected waste, plastic, mixed waste, sludge, hazardous waste, non-hazardous chemicals, biowaste, mineral waste, combustion residues and soil result in net Climate Change impacts (13 to 386 kg CO_2-eq t^{-1}), with plastic having the highest impact due to incineration. Conversely, metals, textiles, electronics, discarded vehicles, glass, paper/cardboard, and wood yield net savings (-1374 to -171 kg CO_2-eq t^{-1}), with metals achieving the highest savings due to minimal treatment burdens and substantial material recovery benefits.

When weighted (Fig. 5**c**), mixed and mineral waste incur the highest Climate Change contributions. Mixed waste is particularly affected by misallocated materials sent to incineration or landfill, highlighting potential areas for collection improvement. Emissions could be further reduced by increasing energy recovery from incineration or implementing abatement techniques for incineration and landfill.

Fig. 5**b** shows that for generated waste, textiles incur net Climate Change impacts (352 kg CO_2-eq t^{-1}), while plastic and biowaste reach 775 and 83 kg CO_2-eq t^{-1}, respectively. These waste groups are major contributors to collected mixed waste impacts. The impacts are primarily driven by fossil CO_2 emissions from incinerating plastics and synthetic textiles, as well as methane emissions from landfills (textiles, biowaste) and biogas plants (biowaste). Glass and paper/cardboard also exhibit higher impacts, as they frequently end up in mixed waste, leading to increased GHG emissions from incineration or landfill instead of recovery. Mixed waste remains a significant contributor due to its composition of diverse, non-recyclable combustibles (e.g., cigarette butts, sanitary products and non-recyclable plastics; see Table S4, SI) with a high fossil carbon content (approximately 240 kg C t^{-1} dry weight), making incineration and landfill currently unavoidable. When weighted (Fig. 5**d**), plastic, biowaste, and mineral waste have the highest impacts. Mineral waste's impact stems mainly from transport (assumed as 25 km t^{-1}) and its weight (approximately 43%), as well as limited savings from material recovery, as substitution of primary materials (mainly natural aggregates) offers low GHG savings (Caro et al., 2024). Plastic, textiles, biowaste, mixed waste and sludge see most impacts from incineration and landfill emissions, due to collection and sorting inefficiencies causing missed recovery opportunities.

The total Climate Change saving for generated and collected waste is -17 kg CO_2-eq t^{-1} (Fig. 5**d**, 'TOT'), and -21 kg CO_2-eq t^{-1} (Fig. 5**c**; 'TOT'), respectively, indicating that GHG burdens from treatments are outweighed by emissions avoided through material and energy recovery. The slight difference occurs as there is not full linearity when shifting the model from collection to generation.

Results for non-hazardous chemicals and other hazardous waste should be interpreted with caution due to high uncertainty linked to assumptions made regarding their material composition. This highlights a potential hotspot within these groups, warranting further investigation as more data becomes available. Key contributors to GHG savings include electronics, discarded vehicles, glass, paper/cardboard, wood, and metals, with metals accounting for about 83% of total savings. Material recovery is the primary driver of climate savings by offsetting emissions from virgin material production. While energy recovery from incineration helps reduce climate impacts, it does not fully offset corresponding burdens.

The results for electronics are also subject to uncertainty, as current statistics only report amounts 'sent for recycling' (97%), and the estimates in this study are based on limited literature and selected electronic categories (heaters/refrigerators, small and large household appliances,

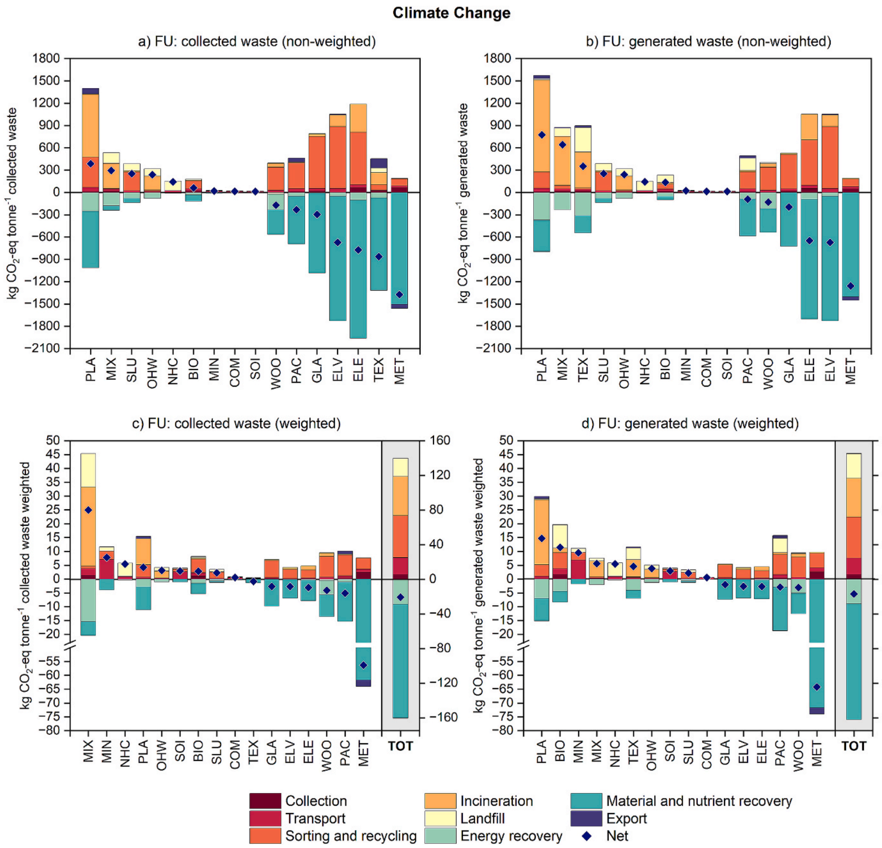

Fig. 5. Results obtained for the Climate Change impact category per tonne of individual waste groups in EU27 in 2020 reported per tonne of collected waste (4a) and generated waste (4b) and per tonne of collected waste, weighted (4c) and generated waste, weighted (4d). The waste groups are depicted with the following abbreviations: biowaste (BIO), combustion waste (COM), electronic waste (ELE), discarded vehicles (ELV), glass waste (GLA), metal waste (MET), mineral waste (MIN), non-hazardous chemical waste (NHC), other hazardous waste (OHW), paper and cardboard waste (PAC), polymeric waste (PLA), mixed waste (MIX), sludge (SLU), soil (SOI), textile waste (TEX), and wood waste (WOO). Total waste is abbreviated as "TOT" and functional unit as "FU".

TV/monitors, and lighting equipment). Additionally, this study does not account for informal management/mismanagement, which are known issues in electronic waste handling (Huisman et al., 2015). Despite these limitations, the estimated savings for electronic waste align well with Biganzoli et al. (2015), who reported a range of –737 to –2187 kg CO_2-eq per tonne for various e-waste categories. Similar concerns regarding informal management apply to illegally exported end-of-life vehicles (European Commission, 2023b). Nonetheless, the results obtained are broadly consistent with Accardo et al. (2023), although this study may underestimate avoided production savings. This is likely due to the exclusion of reuse modeling for specific automotive components (e.g., engines or spare parts), although reuse of more general material fractions, such as textiles, is included elsewhere in the system. Notably, the estimated savings for textiles in this study are substantially higher than those reported by Bijleveld et al. (2022). This is likely because their analysis does not appear to include reuse, which is identified here as the primary driver of climate benefits for this waste group.

Overall, the findings somewhat align with the 2018 baseline results of Bijleveld et al. (2022), highlighting metal recycling as a key driver of GHG savings and mixed waste as a key contributor to emissions. The substantial climate benefits of metal recycling are further supported by Andreasi Bassi et al. (2017) and Caro et al. (2024), who likewise report significant savings from both ferrous and non-ferrous metal recovery. Similarly, the net total climate impact in this study aligns with Albizzati et al. (2024), who found average savings of –49 kg CO_2-eq tonne^{-1} across MW fractions.

One notable difference from Bijleveld et al. (2022) concerns the paper and cardboard waste group. While they identified it as the largest GHG contributor, this study finds it yields net climate savings, consistent with Andreasi Bassi et al. (2017). This divergence stems from differing landfill assumptions, with Bijleveld et al. (2022) assuming high landfill rates (and methane emissions), while this study assumes limited landfill disposal, based on Eurostat data and recycling yields obtained from Haupt et al. (2018). Differences also arise regarding the burden of plastic and mixed waste, which Bijleveld et al. (2022) find to be substantially lower for plastic and higher for mixed waste, despite assuming a lower recycling rate for plastics and having a comparable distribution between landfill and incineration for treatment of mixed waste. However, Bijleveld et al. (2022) claim to greatly underestimate the impacts from plastics due to uncertainty regarding their treatment. For mixed waste, the difference reflects compositional differences, such as a higher organic or fossil content in their model, though further investigation is needed to confirm this.

3.3.2. Impacts on the remaining environmental categories

Section S4.1 (SI) provides an overview of impacts across all environmental categories that, like Climate Change, show slight net savings, except for Human Toxicity (non-carcinogenic) and Eutrophication (terrestrial and marine) (see **Sections S4.1.3, S4.1.8** and **S4.10,** SI), and for Ozone Depletion when assessed as collected (see **Section S4.1.1**, SI). Human Toxicity impacts are primarily driven by sorting and recycling, especially for metals, biowaste and sludge (because of use-on-land), and discarded vehicles.. Incineration of electronics further contributes to these impacts. For Eutrophication, the main impacts are due to use-on-land (displayed under sorting and recycling) of biowaste and sludge, and on-site fuel combustion for mineral waste recycling operations (a result of the large quantities of mineral waste). For Ozone Depletion, impacts are mainly driven by the landfilling of textiles that have been misallocated to the mixed waste.

3.4. Economic impacts

This section presents the eLCC and feLCC results (see Fig. 6 and Fig. 7, respectively).

3.4.1. Environmental Life Cycle costs

Fig. 6 illustrates that revenues from material recovery do not offset internal costs incurred for most waste groups, either as collected or generated, with metals demonstrating the lowest costs (7.7 EUR t^{-1}) due to high material recovery revenues (Fig. 6**a**). For the remaining collected waste groups, the net eLCC ranges from 16 to 540 EUR t^{-1} for soil and textiles, respectively. The main processes contributing to internal costs are sorting and recycling of electronics, textiles, plastics and discarded vehicles; however, collection also exhibits high costs for several waste groups. Material recovery serves as the primary driver of revenues, particularly for textiles, electronics and metals. When weighted (Fig. 6**c**), collected mineral and mixed waste exhibit the highest eLCC, primarily due to incineration and landfill costs, whereas metals continue to exhibit the lowest management cost per unit collected.

For generated waste (Fig. 6**b**), management costs are significantly lower for textiles (180 EUR t^{-1}) and plastics (220 EUR t^{-1}) due to lower sorting and recycling costs as significant amounts are misallocated to mixed waste (approximately 81% of textiles and 54% of plastics are collected as mixed waste). Moreover, incinerated plastics benefit mainly from energy recovery revenues, highlighting the lack of economic incentives to divert plastics and textiles from incineration to recycling. A similar issue exists for discarded vehicles, which face high sorting and recycling costs. When weighted (Fig. 6**d**), mineral waste still exhibits the highest eLCC as generated, now followed by biowaste, which management costs are mainly driven by collection and incineration. Metals lose net revenue due to approximately 8.0% misallocation to mixed waste, resulting in lost recovery potential.

The total eLCC for generated waste (Fig. 6**d**, 'TOT', 77 EUR t^{-1}) is primarily driven by collection, sorting and recycling, followed by landfill, transport, and incineration costs. Collected waste exhibits similar costs (Fig. 6**c**, 'TOT'), which aligns well with Albizzati et al. (2024). Revenues mainly come from material recovery (approximately 60%) and energy (40%), though the latter may be slightly overestimated.

3.4.2. Full environmental life cycle costs

Fig. 7 illustrates the feLCC, encompassing both internal and external costs (see **Section S4.2.1** (SI) for a detailed overview of external costs alone). Fig. 7**a** shows that collected waste groups exhibit net costs of 2.7 to 400 EUR t^{-1}, as revenues and external cost savings fail to cover internal costs incurred. Metals is thus the only collected waste group to distribute savings (−230 EUR t^{-1}). Fig. 7**a** shows that only textiles, electronics, metals, discarded vehicles, paper/cardboard, wood, plastics, and glass have external cost savings, but only substantial enough for metals to compensate for internal costs, hence incurring a net saving (internal costs for metals are already close to zero, see Fig. 6**a, c**). In contrast, biowaste, sludge, other hazardous waste, mixed waste, non-hazardous chemicals, mineral waste, combustion residues, and soil all display net external costs, meaning the monetised environmental

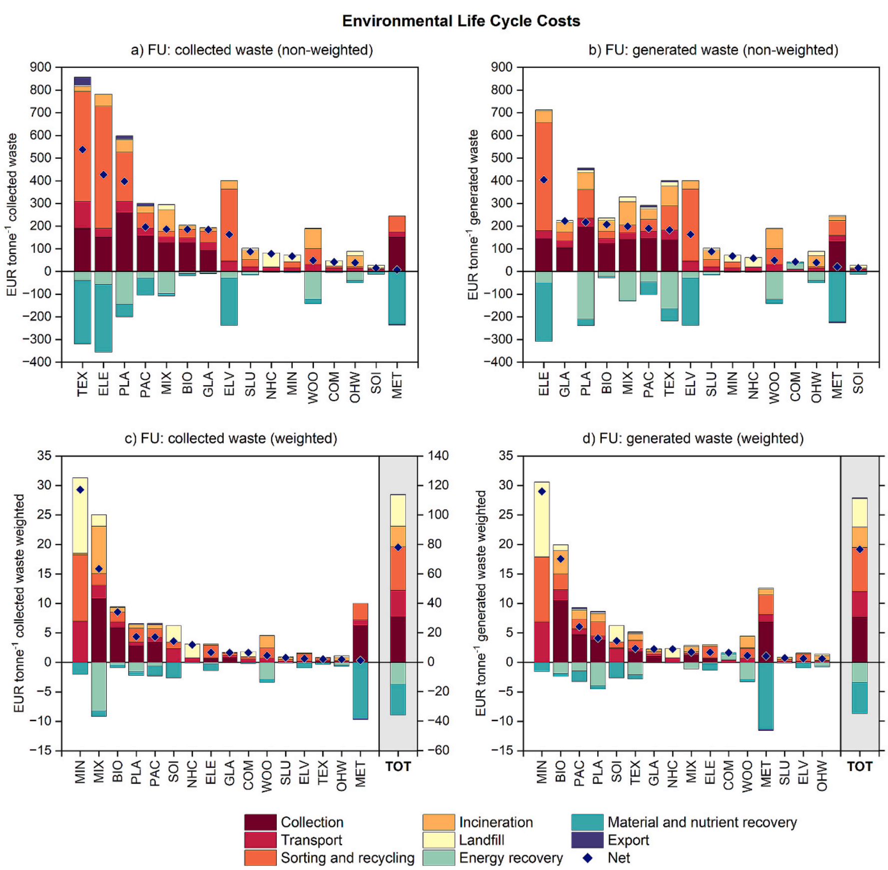

Fig. 6. Results obtained for the Environmental Life Cycle Cost impact category per tonne of individual waste groups in EU27 in 2020 reported per tonne of collected waste (5a) and generated waste (5b) and per tonne of collected waste, weighted (5c) and generated waste, weighted (5d). The waste groups are depicted with the following abbreviations: biowaste (BIO), combustion waste (COM), electronic waste (ELE), discarded vehicles (ELV), glass waste (GLA), metal waste (MET), mineral waste (MIN), non-hazardous chemical waste (NHC), other hazardous waste (OHW), paper and cardboard waste (PAC), polymeric waste (PLA), mixed waste (MIX), sludge (SLU), soil (SOI), textile waste (TEX) and wood waste (WOO). Total waste is abbreviated as "TOT" and functional unit as "FU".

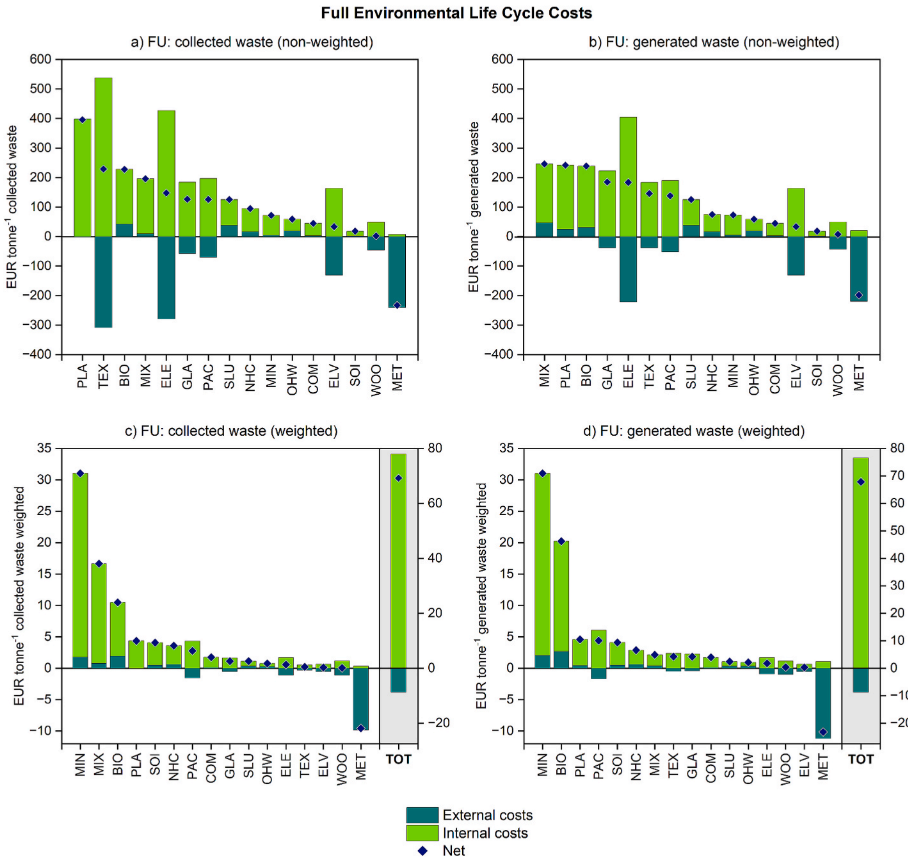

Fig. 7. Results obtained for the Full Environmental Life Cycle Cost impact category per tonne of individual waste groups in EU27 in 2020 reported per tonne of collected waste (7a) and generated waste (7b) and per tonne of collected waste, weighted (7c) and generated waste, weighted (7d). The waste groups are depicted with the following abbreviations: biowaste (BIO), combustion waste (COM), electronic waste (ELE), discarded vehicles (ELV), glass waste (GLA), metal waste (MET), mineral waste (MIN), non-hazardous chemical waste (NHC), other hazardous waste (OHW), paper and cardboard waste (PAC), polymeric waste (PLA), mixed waste (MIX), sludge (SLU), soil (SOI), textile waste (TEX) and wood waste (WOO). Total waste is abbreviated as "TOT" and functional unit as "FU".

burdens associated with their management outweigh any potential monetised environmental savings. These external costs further exacerbate their overall financial burdens, underscoring the need for improved sorting and recycling processes for these waste groups. Weighted costs (Fig. 7**c**), reflect the same trend as in the eLCC (Fig. 6**c**) where mineral waste, mixed waste, and biowaste have the highest costs, and metal waste the largest savings.

From the generated waste (Fig. 7**b**), two key messages emerge: i) once again, only metal exhibits net savings (−200 EUR t^{-1}) and ii) external costs savings are reduced for all other waste groups relative to the collection-based results, suggesting poor collection performance. Notably, a shift is observed for plastics, which now exhibit a much lower net cost relative to the collection-based results (240 EUR t^{-1}), driven by lower internal costs. This is expected because the plastic waste misallocated to the mixed residual waste incurs lower internal costs (incineration or landfill are cheaper than recycling). When weighted (Fig. 7**d**), results reflect those for collected waste (Fig. 7**c**), with mineral waste showing the highest net costs and metal waste the highest net savings. The results align closely with trends observed in the corresponding eLCC (Fig. 6**d**), with minor exceptions. Fig. 7**d**, (' TOT') quantifies the total costs of EU27 waste management in 2020 for generated waste (68 EUR t^{-1}), similar to the collected waste (Fig. 7**c**, 69 EUR t^{-1}), suggesting that the EU27 waste management system is far from achieving a net cost benefit.

3.5. *Sensitivity of mixed waste composition*

Fig. 8 presents results obtained from the sensitivity analysis. Fig. 8**a** illustrates the generated amounts of the waste groups mostly affected by changes in mixed waste composition (glass, metals, paper and cardboard, plastic, wood, textiles, electronics and biowaste). Percentage differences between the default and EEA composition ranges from 1.6% for plastic to 13% for textiles, while electronics stand out with a 63% change making it an outlier. The large discrepancy for electronics was expected, as MSs often categorize them under "other waste" (see **Section 2.5**). Observing the composition of the total waste generated (Fig. 8**b**), the main differences were found for electronics (1.9% for the EEA and 1.2% for the default), mixed waste (0.8% and 0.5%) and textiles (0.4% and 0.5%).

Waste groups most affected by mixed waste composition also exhibit the largest Climate Change impact differences, although the relationship between amount and impact is not linear. For example, electronics show the largest difference in generated amounts and also the greatest difference in non-weighted Climate Change net impacts (Fig. 8**c**), yet it is followed by wood, having a 17% increase in savings, although exhibiting only 12.5% difference in amount. Despite discrepancies observed for mixed waste and electronics, the overall ranking of waste groups remains largely consistent across both compositions, as shown in Fig. 8**d** (displaying weighted net Climate Change impacts), while also resulting in similar net total savings (i.e., approximately −17 kg CO_2-eq t^{-1} for both; Fig. 8**d** 'TOT'). Results indicate that the method used to estimate the generated amounts in this study provides a solid foundation for modeling waste composition and its associated impacts.

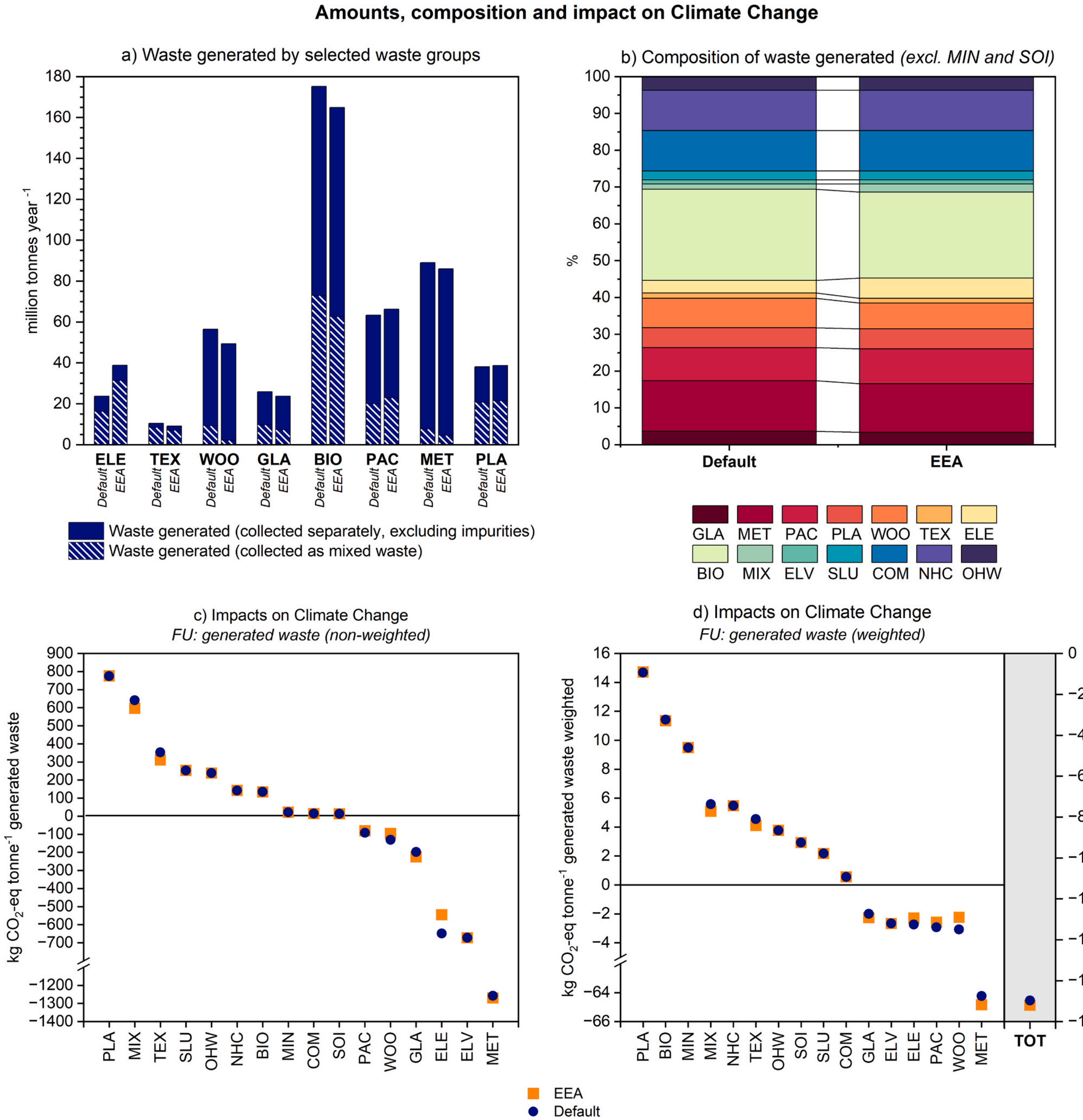

Fig. 8. Comparison of the total amount of waste generated in million tonnes for selected waste groups in EU27 in 2020, when assuming either the default mixed waste composition based on Albizzati et al. (2024) or the composition based data obtained from the EEA (7a), of the overall composition as percent of the total waste generated (excluding soil and mineral waste) (7b), and of the results obtained for the Climate Change impact category per tonne of individual waste groups reported per tonne of generated waste (7c) and per tonne of generated waste, weighted (7d). The waste groups are depicted with the following abbreviations: biowaste (BIO), combustion waste (COM), electronic waste (ELE), discarded vehicles (ELV), glass waste (GLA), metal waste (MET), mineral waste (MIN), non-hazardous chemical waste (NHC), other hazardous waste (OHW), paper and cardboard waste (PAC), polymeric waste (PLA), mixed waste (MIX), sludge (SLU), soil (SOI), textile waste (TEX) and wood waste (WOO). Total waste is abbreviated as "TOT" and functional unit as "FU".

3.6. Critical waste groups

Waste groups with net burdens on Climate Change and external costs highlight key areas for improvement. While waste management in EU27 overall provides a net Climate Change benefit, an in-depth analysis reveals modest savings, driven primarily by metals, indicating the need for targeted improvements. Increasing recycling rates while reducing incineration and landfill reliance is crucial, as these are the primary drivers of current climate burdens. Special attention should be given to the collected mixed waste, as it contains high amounts of misallocated waste that could otherwise be recycled, especially for textiles and polymers. A crucial first step involves enhancing monitoring of the collected mixed waste compositions across EU27 (e.g., through standardized compositional analyses), which are currently infrequent and inconsistent. Fostering centralized sorting of mixed waste could serve as a logical next step.

Textiles stand out due to their high climate and economic impacts, with economic disincentives leading to incineration over recycling, and large volumes exported due to limited EU recycling capacity, aligning with the findings of Solis et al. (2024). The high internal costs associated with sorting and recycling, combined with relatively low recycling rates, suggest the need for policy interventions. Given the considerable environmental and economic burdens associated with textiles, developing more efficient recycling technologies and creating incentives for textile recovery and reuse should be high on the agenda.

Polymers also stand out due to substantial discrepancies between generated and collected amounts. The low recycling rates, combined with high climate and economic impacts, point to the need for advancements in both collection schemes and recycling technologies (also for non-packaging). Addressing challenges posed by complex polymer design-compositions, which often involve mixtures that are difficult to recycle, is essential.

Biowaste also emerges as a critical stream due to its low collection rate and significant impacts from landfilling and fugitive emissions in biological processes. Increasing collection rates and capacity for biogas production could significantly increase environmental benefits (Li et al., 2023). Further benefits could be obtained by fostering advanced technologies (Albizzati et al., 2021a, 2021b) for the extraction of specific chemicals. Similarly for sludge, fostering phosphorous recovery could reduce EU dependency on external sources (Tonini et al., 2019).

Another challenge is the high volume of mineral waste sent to landfill. Given it represents the largest share of generated waste, this practice poses substantial economic and environmental burdens. Redirecting it to high-quality recycling could greatly reduce impacts and enhance circularity (Caro et al., 2024). Management of non-hazardous chemicals and other hazardous waste also produces net impacts and could be improved. Lastly, the impact of the residually generated 'mixed waste' can be reduced with design-for-recycling and selective elimination or substitution of hard-to-recycle products.

3.7. Limitations

This model provides a broad overview of the entire EU27 waste management system, enabling the identification of focus areas and prioritization of different waste groups. Due to the large variety of waste streams and treatment options, each with varying data availability, a balance had to be struck between detail and feasibility. As such, the model represents an average EU27 waste management system and does not account for variations across individual MSs. Important national and regional differences in collection systems, treatment technologies, energy mixes, and waste composition are therefore not be reflected in the results. This approach has led to several limitations, which are further discussed in this section.

Waste compositions rely on literature and assumptions that do not capture country-specific differences, which according to Bisinella et al. (2017), can have a fundamental influence on the environmental emissions associated with waste treatment, recycling and disposal. While the sensitivity analysis confirmed the model's robustness to general changes in waste composition (e.g., adjusting the proportion of glass in the mixed waste), it does not account for more detailed shifts, such as the specific compositional breakdown of individual glass sub-groups within the mixed waste. Moreover, the level of material similarity among subgroups varied greatly across the 16 waste groups. For example, the metal waste group consists of relatively homogeneous sub-groups, composed primarily of ferrous and non-ferrous packaging and non-packaging materials. In contrast, other groups, such as other hazardous waste, are much more heterogenous and include sub-groups such as spent solvents, waste containing PCB and dredging spoils. These differences in internal composition likely influence results, especially given that all waste groups are treated using average processes in the model. For example, all metals are sent to an average sorting facility prior to recycling, while all other hazardous waste is sent to a single average disinfection process prior to further treatment, even though different types of hazardous waste may require varying levels or disinfection and treatment.

The use of average processes under typical conditions, coupled with generic compositions of waste groups, inevitably lowers the accuracy of the results. For example, Manfredi et al. (2010) discuss the importance of understanding the specific composition of waste streams when assessing the environmental impacts of landfilling. This particularly affects the results for non-hazardous chemicals, other hazardous waste, and electronic waste, which generate pollutants different from those of MW. The lack of detailed composition data prevents these differences from being reflected in the life cycle inventory, leading to greater uncertainty for heterogeneous or data-poor streams compared to well-documented streams like metals, glass and paper/cardboard.

Nevertheless, the results indicate that the environmental impacts associated with these streams may be non-negligible albeit their low share of total EU waste. This points to a potential policy gap in the current EU *acquis,* which does not set material-specific circularity or recycling targets for these waste groups. For example, while the WEEE Directive defines overall recovery and recycling targets by weight, it does not include targets for the separation or recovery of specific materials such as plastics or critical raw materials. Similarly, other complex waste streams, such as non-hazardous chemical waste and other hazardous waste, are not covered by detailed recovery targets. Instead, current, EU legislation refers to the use of Best Available Techniques Reference Documents (BREFs) to guide their treatment and permitting, leaving gaps in the promotion of high-quality recycling and material circularity.

All treatment processes are linked to the EU average electricity and heat mix for 2020, including substitution related to differences in the energy systems of the MSs where specific treatment processes take place is not accounted for. For example, MW incineration with energy recovery is not uniformly distributed across the EU, and countries with a higher share of such facilities would ideally have a greater influence on the electricity mix used for substitution. However, assessing such variation would require detailed data on the geographical distribution of all waste treatment processes.

While a sensitivity analysis was conducted for one of the most influential parameters, namely the composition of mixed waste, a global sensitivity analysis was not performed due to the lack of consistent, data-backed uncertainty ranges across all model parameters. For the same reason, an uncertainty analysis was not carried out. These analyses should be prioritized in future studies to better assess the robustness and variability of the results. Despite these limitations, the model offers a valuable system-level overview of the EU27 waste management system and supports the evaluation of policy scenarios, (e.g., raising recycling targets, reducing landfill use, or limiting exports) enabling policymakers to explore impacts of various interventions. Future work could enhance model accuracy by integrating region-specific data, incorporating emerging treatment technologies such as chemical recycling, and including social indicators to further expand the sustainability perspective.

4. Conclusion

This study proposes a comprehensive model for evaluating environmental and economic impacts across the entire EU waste management system, including a methodology to estimate waste generated based on official statistics. This approach is essential, as these sources report 171 Mt of collected waste as mixed waste.

By quantifying impacts for both collected and generated waste, this model underscores the unique insights each approach provides. The generation-based perspective reveals inefficiencies in collection and treatment, accounting for waste that is collected separately, misallocated as mixed waste or classified as impurities. Meanwhile, the collection-based perspective shows system performance under optimal sorting conditions. Together, these perspectives present a balanced picture of the waste management system.

Key findings reveal that the EU27 waste management system falls short of making a substantial contribution to climate change mitigation, environmental protection, and cost reduction. Special attention should be devoted to reducing misallocation in mixed waste, with a focus on improving the collection and sorting of plastic, textiles and biowaste. Mineral waste and sludge also require targeted attention. Hitherto overlooked waste streams (non-hazardous chemicals and other hazardous waste) warrant further investigation.

Disclaimer

The views expressed in the article are the sole responsibility of the authors and in no way represent the view of the European Commission and its services.

CRediT authorship contribution statement

J.H. Sund: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation. **P.F. Albizzati:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Investigation, Conceptualization. **C. Scheutz:** Writing – review & editing, Funding acquisition. **D. Tonini:** Writing – review & editing, Writing – original draft, Validation, Supervision, Project administration, Methodology, Investigation, 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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2025.114910.

Data availability

Data will be made available on request.

References

Accardo, A., Dotelli, G., Miretti, F., Spessa, E., 2023. End-of-life impact on the cradle-to-grave LCA of light-duty commercial vehicles in Europe. *Applied Sciences (switzerland)* 13 (3). https://doi.org/10.3390/app13031494.

Albizzati, P.F., Foster, G., Gaudillat, P., Manfredi, S., Tonini, D., 2024. A model to assess the environmental and economic impacts of municipal waste management in Europe. *Waste Manag.* 174, 605–617. https://doi.org/10.1016/j.wasman.2023.12.029.

Albizzati, P.F., Tonini, D., Astrup, T.F., 2021a. A Quantitative Sustainability Assessment of Food Waste Management in the European Union. *Environ. Sci. Tech.* 55 (23), 16099–16109. https://doi.org/10.1021/acs.est.1c03940.

Albizzati, P.F., Tonini, D., Astrup, T.F., 2021b. High-value products from food waste: An environmental and socio-economic assessment. *Sci. Total Environ.* 755, 142466. https://doi.org/10.1016/j.scitotenv.2020.142466.

Allesch, A., Brunner, P.H., 2014. Assessment methods for solid waste management: A literature review. *Waste Manag. Res.* 32 (6), 461–473. https://doi.org/10.1177/0734242X14535653.

Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2011. Mass balances and life cycle inventory of home composting of organic waste. *Waste Manag.* 31 (9–10), 1934–1942. https://doi.org/10.1016/j.wasman.2011.05.004.

Andreasi Bassi, S., Christensen, T.H., Damgaard, A., 2017. Environmental performance of household waste management in Europe - An example of 7 countries. *Waste Manag.* 69, 545–557. https://doi.org/10.1016/j.wasman.2017.07.042.

Basel Convention Technical Working Group, 2000. Technical Guidelines on the Identification and Management of Used Tyres. In *Basel Convention on the Control of Transboundary Movement of Hazardous Wastes and Their Disposal* (issue 10) docs/tech-usedtyres.pdf.

Biganzoli, L., Falbo, A., Forte, F., Grosso, M., Rigamonti, L., 2015. Mass balance and life cycle assessment of the waste electrical and electronic equipment management system implemented in Lombardia Region (Italy). *Sci. Total Environ.* 524–525, 361–375. https://doi.org/10.1016/j.scitotenv.2015.04.041.

Bijleveld, M. ., Birnstengel, B., Simpson, R., & Kölmel, R. (2022). *CO2 Reduction Potential in European Waste Management*. https://cedelft.eu/publications/co2-reduction-potential-in-european-waste-management/.

Bisinella, V., Götze, R., Conradsen, K., Damgaard, A., Christensen, T.H., Astrup, T.F., 2017. Importance of waste composition for Life Cycle Assessment of waste management solutions. *J. Clean. Prod.* 164, 1180–1191. https://doi.org/10.1016/j.jclepro.2017.07.013.

Boldrin, A. (2009). Environmental Assessment of Garden Waste Management. In *PhD thesis* (Issue September).

Boldrin, A., Christensen, T.H., 2010. Seasonal generation and composition of garden waste in Aarhus (Denmark). *Waste Manag.* 30 (4), 551–557. https://doi.org/10.1016/j.wasman.2009.11.031.

Caro, D., Lodato, C., Damgaard, A., Cristóbal, J., Foster, G., Flachenecker, F., & Tonini, D. (2024). Environmental and socio-economic effects of construction and demolition waste recycling in the European Union. *Science of the Total Environment*, *908*(October 2023). doi: 10.1016/j.scitotenv.2023.168295.

Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P., Hauschild, M., 2009. C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems. *Waste Manag. Res.* 27 (8), 707–715. https://doi.org/10.1177/0734242X08096304.

Clavreul, J., Baumeister, H., Christensen, T.H., Damgaard, A., 2014. An environmental assessment system for environmental technologies. *Environ. Model. Softw.* 60, 18–30. https://doi.org/10.1016/j.envsoft.2014.06.007.

Cristóbal, J., Foster, G., Caro, D., Yunta, F., Manfredi, S., Tonini, D., 2024. Management of excavated soil and dredging spoil waste from construction and demolition within the EU: Practices, impacts and perspectives. *Sci. Total Environ.* 944 (May). https://doi.org/10.1016/j.scitotenv.2024.173859.

De Bruyn, S., Bijleveld, M., de Graaff, L., Schep, E., Schroten, A., Vergeer, R., & Ahdour, S. (2018). Environmental Prices Handbook 2017. *CE Delft*, 175.

De Laurentiis, V., Caldeira, C., Sala, S., & Tonini, D. (2024). Life cycle thinking for the assessment of waste and circular economy policy: status and perspectives from the EU example. *Waste Management*, *179*(September 2023), 205–215. doi: 10.1016/j.wasman.2024.02.037.

Ec-jrc., 2012. Product Environmental Footprint (PEF) Guide. methodology final draft.pdf *Eropean Commision Joint Research Centre* 154. http://ec.europa.eu/environment/eussd/pdf/footprint/PEF.

Edjabou, M.E., Boldrin, A., Scheutz, C., Astrup, T.F., 2016. Generation of organic waste from institutions in Denmark: case study of the Technical University of Denmark. *10th International Conference on "circular Economy and Organic Waste."*.

Edjabou, M.E., Takou, V., Boldrin, A., Petersen, C., Astrup, T.F., 2021. The influence of recycling schemes on the composition and generation of municipal solid waste. *J. Clean. Prod.* 295. https://doi.org/10.1016/j.jclepro.2021.126439.

Ekvall, T., Assefa, G., Björklund, A., Eriksson, O., Finnveden, G., 2007. What life-cycle assessment does and does not do in assessments of waste management. *Waste Manag.* 27 (8), 989–996. https://doi.org/10.1016/j.wasman.2007.02.015.

Eriksen, M.K., Astrup, T.F., 2019. Characterisation of source-separated, rigid plastic waste and evaluation of recycling initiatives: Effects of product design and source-separation system. *Waste Manag.* 87, 161–172. https://doi.org/10.1016/j.wasman.2019.02.006.

European Commission. (2021). *Better Regulation Guidelines*. https://commission.europa.eu/law/law-making-process/better-regulation/better-regulation-guidelines-and-toolbox_en.

European Commission. (2023). *Better Regulation Toolbox*. https://commission.europa.eu/law/law-making-process/better-regulation/better-regulation-guidelines-and-toolbox_en.

European Commission (EC). (2023). *Questions and Answers : End-of-Life vehicles*. *July*, 1–3.

European Environment Agency (EEA). (2020). *Construction and demolition waste: challenges and opportunities in a circular economy*. https://www.eea.europa.eu/publications/construction-and-demolition-waste-challenges.

European Environment Agency (EEA). (2022). *Early warning assessment related to the 2025 targets for municipal waste and packaging waste*. https://www.eea.europa.eu/publications/many-eu-member-states/early-warning-assessment-related-to.

European Environment Agency (EEA). (2023a). *Waste prevention country profile, France. April*.

European Environment Agency (EEA). (2023b). *Waste prevention country profile, Italy. April*.

Eurostat. (2024). *HICP - inflation rate*. https://ec.europa.eu/eurostat/databrowser/view/tec00118/default/table?lang=en.

Eurostat. (2025). *Municipal Waste Statistics*. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Municipal_waste_statistics.
Faraca, G., Boldrin, A., Astrup, T., 2019. Resource quality of wood waste: The importance of physical and chemical impurities in wood waste for recycling. *Waste Manag.* 87, 135–147. https://doi.org/10.1016/j.wasman.2019.02.005.
Garcia-Gutierrez, P., Amadei, A. M., Klenert, D., Nessi, S., Tonini, D., Tosches, D., Ardente, F., & Saveyn, H. (2023). *Environmental and economic assessment of plastic waste recycling* (Issue September). doi: 10.2760/0472.
German Environment Agency (UBA). (2022). *Evaluation of the collection and recovery of selected waste streams for the further development of circular economy*. https://www.umweltbundesamt.de/en/publikationen/evaluation-of-the-collection-recovery-of-selected.
Götze, R., Pivnenko, K., Boldrin, A., Scheutz, C., Astrup, T.F., 2016. Physico-chemical characterisation of material fractions in residual and source-segregated household waste in Denmark. *Waste Manag.* 54, 13–26. https://doi.org/10.1016/j.wasman.2016.05.009.
Haupt, M., Kägi, T., Hellweg, S., 2018. Life cycle inventories of waste management processes. *Data Brief* 19, 1441–1457. https://doi.org/10.1016/j.dib.2018.05.067.
Haupt, M., Vadenbo, C., Hellweg, S., 2017. Do We Have the Right Performance Indicators for the Circular Economy?: Insight into the Swiss Waste Management System. *J. Ind. Ecol.* 21 (3), 615–627. https://doi.org/10.1111/jiec.12506.
Hoogmartens, R., Van Passel, S., Van Acker, K., Dubois, M., 2014. Bridging the gap between LCA, LCC and CBA as sustainability assessment tools. *Environ. Impact Assess. Rev.* 48, 27–33. https://doi.org/10.1016/j.eiar.2014.05.001.
Huisman, J., Botezatu, I., Herreras, L., Liddane, M., Hintsa, J., Luda di Cortemiglia, V., Leroy, P., Vermeersch, E., Mohanty, S., van den Brink, S., Ghenciu, B., Dimitrova, D., Nash, E., Shryane, T., Wieting, M., Kehoe, J., Baldé, C. P., Magalini, F., Zanasi, A., … Bonzio, A. (2015). *Countering WEEE Illegal Trade Summary Report*.
Hunkeler, D., Lichtenvort, K., Rebitzer, G., 2008. *Environmental Life Cycle Costing*. CRC Press.
International Energy Agency (IEA). (2020). *Countries and regions*. https://www.iea.org/countries.
ISO. (2006a). *Environmental management - Life cycle assessment - Requirements and guidelines - Amendment 1 (ISO 14044:2006/Amd 1:2017)*. *2*, 106.
ISO. (2006b). Environmental Management – Life Cycle Assessment – Principles and Framework Secon Edit. 20.
Keramidas, K., Fosse, F., Diaz, R. A., & Dowling, P. (2023). *Global Energy and Climate Outlook 2023*. doi: 10.2760/836798.
Kulczycka, J., Kowalski, Z., Smol, M., Wirth, H., 2016. Evaluation of the recovery of Rare Earth Elements (REE) from phosphogypsum waste - Case study of the WIZÓW Chemical Plant (Poland). *J. Clean. Prod.* 113, 345–354. https://doi.org/10.1016/j.jclepro.2015.11.039.
Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T.H., 2014. Review of LCA studies of solid waste management systems - Part I: Lessons learned and perspectives. *Waste Manag.* 34 (3), 573–588. https://doi.org/10.1016/j.wasman.2013.10.045.
Li, Y., Meenatchisundaram, K., Rajendran, K., Gohil, N., Kumar, V., Singh, V., Solanki, M. K., Harirchi, S., Zhang, Z., Sindhu, R., Taherzadeh, M.J., Awasthi, M.K., 2023. Sustainable Conversion of Biowaste to Energy to Tackle the Emerging Pollutants: A Review. *Curr. Pollut. Rep.* 9 (4), 660–679. https://doi.org/10.1007/s40726-023-00281-8.
Madsen, M. H. (2021). *Assessment of potential end of life scenarios for textile waste in Denmark Bachelor Project*.
Manfredi, S., Tonini, D., Christensen, T.H., 2010. Contribution of individual waste fractions to the environmental impacts from landfilling of municipal solid waste. *Waste Manag.* 30 (3), 433–440. https://doi.org/10.1016/j.wasman.2009.09.017.
Martinez-Sanchez, V., Kromann, M.A., Astrup, T.F., 2015. Life cycle costing of waste management systems: Overview, calculation principles and case studies. *Waste Manag.* 36, 343–355. https://doi.org/10.1016/j.wasman.2014.10.033.
Martinez-Sanchez, V., Levis, J.W., Damgaard, A., DeCarolis, J.F., Barlaz, M.A., Astrup, T. F., 2017. Evaluation of externality costs in life-cycle optimization of municipal solid waste management systems. *Environ. Sci. Tech.* 51 (6), 3119–3127. https://doi.org/10.1021/acs.est.6b06125.
Merrild, H., Damgaard, A., Christensen, T.H., 2008. Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration. *Resour. Conserv. Recycl.* 52 (12), 1391–1398. https://doi.org/10.1016/j.resconrec.2008.08.004.
Miljøstyrelsen. (2018). *Affaldsdatasystemet (ADS)*. https://www.ads.mst.dk/.
Pivnenko, K., Olsson, M.E., Götze, R., Eriksson, E., Astrup, T.F., 2016. Quantification of chemical contaminants in the paper and board fractions of municipal solid waste. *Waste Manag.* 51, 43–54. https://doi.org/10.1016/j.wasman.2016.03.008.
Riber, C., Petersen, C., Christensen, T.H., 2009. Chemical composition of material fractions in Danish household waste. *Waste Manag.* 29 (4), 1251–1257. https://doi.org/10.1016/j.wasman.2008.09.013.
Solis, M., Huygens, D., Tonini, D., & Fruergaard Astrup, T. (2024). Management of textile waste in Europe: An environmental and a socio-economic assessment of current and future scenarios. *Resources, Conservation and Recycling, 207*(May). doi: 10.1016/j.resconrec.2024.107693.
Tonini, D., Saveyn, H.G.M., Huygens, D., 2019. Environmental and health co-benefits for advanced phosphorus recovery. *Nat. Sustainability* 2 (11), 1051–1061. https://doi.org/10.1038/s41893-019-0416-x.
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. *Int. J. Life Cycle Assess.* 21 (9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8.