



Waste LCA and the future

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ABSTRACT

Life cycle assessment (LCA) models quantifying the environmental aspects of waste management have become an integral part of waste management decision-making over the last two decades and have provided ample knowledge on both environmental benefits and drawbacks in the way we handle waste. Waste management and LCA modelling of waste management systems will soon be challenged by profound changes necessary in our societies and sectors to meet sustainable development goals. Foreseen changes in energy, material, and nutrient provision will directly and indirectly affect waste management in terms of its operation and goals. This study reflects on anticipated changes in society and industrial sectors and how these changes may affect waste management and LCA modelling of waste management systems in terms of waste input, the modelling of technologies and systems and exchanges of energy, materials, and nutrients, as well as how it may affect impact assessment and the interpretation of results. The study provides practical recommendations for LCA modelling of future waste management systems, which will hopefully lead to robust assessments that can support decision-making in an evolving society subject to great changes.

1. Introduction

Life Cycle Assessment (LCA) has provided a holistic and systematic quantification of environmental and resource-related issues of waste management for over two decades (Clift et al., 2000; Finnveden, 1999; Finnveden et al., 1995; Christensen et al., 2020; Jeswani et al., 2021; Laurent et al., 2014). Among the many issues addressed, LCAs have quantified the environmental loads and benefits of individual waste management technologies as well as complex waste systems. In particular, LCA has played – and is still expected to play – a crucial role in six application areas: (i) Understanding waste management systems; (ii) Improving existing waste management systems; (iii) Comparing alternative technologies or technology performances; (iv) Technology development; (v) Policy development; (vi) Reporting (Christensen et al., 2020).

LCA modelling of the waste management sector will soon have to evolve to address the needed fundamental and transformative changes that will involve all sectors in our societies, for example to achieve climate goals (e.g. climate neutrality in the European Union by 2050, European Environment Agency, 2020), and to defossilise energy and material production (Riahi et al., 2017), towards a post-fossil society

(Haas et al., 2022; Hajer and Versteeg, 2018; Pelzer and Versteeg, 2019). New societal priorities and needs may also transform the waste management sector. Key challenges will be waste prevention, increasing resource efficiency and recovery and improving emission control. The criteria for optimal waste management may also change. For example, need for resource recovery may increase over volume reduction; discovery of novel exposure pathways to chemicals of concern may need specific treatment prior to material recycling. Overall, these new needs will require the transformation of technologies, systems and objectives associated with existing waste management systems.

Accordingly, LCA modelling of the waste management sector will be challenged, for example in terms of different waste compositions and characteristics, different exchanges with the energy and material production sectors and, potentially, the introduction of new technologies. Moreover, having a long-term perspective is necessary for the waste management sector, because waste management systems last for decades after they have been developed and implemented (e.g., Brogaard et al., 2013). With large investments in facilities and long-term operational lifetimes, an LCA needs to be able to identify environmentally optimal solutions that will also serve future societal needs – even expanding into a post-fossil society.

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Addressing the future of LCA modelling is not a new issue. Most LCAs are implicitly “future LCAs,” supporting decisions stretching decades ahead. A systematic review of over 500 journal articles across all sectors highlighted that coupling future scenario methods and LCAs could provide early sustainability assessments of novel products, technologies and systems (Bisinella et al., 2021a). Practical challenges across sectors involve devising precise terminology and definition of goal and scope, the explicit use of foresight methodology in foreground and background systems (e.g., Börjeson et al., 2006), integration with external models, such as future energy system models, and a thorough interpretation of results (Bisinella et al., 2021a). The review identified 56 articles relating to waste management, with ten prominent examples (e.g., Arushanyan et al., 2017; Eriksson et al., 2013; Meylan et al., 2014). LCAs of future waste management systems generally pay more attention to developments in the foreground system (waste input, waste management technologies and system) than to the interactions with and the development of the background system. Only 40 % of the reviewed LCAs of future waste systems mentioned changes in processes and systems providing or reusing materials, energy and nutrients in which waste management systems operate (Bisinella et al., 2021a). An example of considering such changes is given, for example, by Levis et al. (2014). The review highlighted several articles providing recommendations for specific sectors. “Ex-ante LCA” and “prospective LCA” are terms

currently used to denote LCAs of emerging technologies, products and systems (Arvidsson et al., 2018; Cucurachi et al., 2018; Thonemann et al., 2020; Villares et al., 2017). However, to date, no article has provided an overview of the challenges facing LCAs of waste management systems and recommendations for the future.

This paper aims to provide a systematic overview of the future changes that may challenge waste LCA modelling and to provide practical recommendations. Starting with the characteristics and challenges of traditional waste LCAs (Section 2), and relevant future societal challenges (Section 3), to the extent possible, the paper provides practical approaches and recommendations on how to deal with these issues (Section 4). The recommendations, provided as a checklist for consideration, do not presume to carry out the impossible task of predicting the future. By identifying potential issues, and by providing transparent approaches and recommendations, the scientific community can begin to formulate a common approach to addressing these uncertainties in future LCAs. This will hopefully lead to better decision-making when employing LCA in planning future waste management systems embedded in an evolving society subject to great changes.

2. Characteristics and challenges of traditional waste LCA

LCA modelling, a standardised product-oriented method (ISO,

Table 1

Waste LCA characteristics, current challenges, societal changes and waste LCA future challenges in relation to phases of LCA. Characteristics, challenges and future changes are selected according to the challenges addressed in this paper.

LCA phases (ISO, 2006a, 2006b)	Waste LCA characteristics	Waste LCA: current challenges (Christensen et al., 2020; Laurent et al., 2014a, 2014b)	Societal changes (Section 3)	Waste LCA: future challenges addressed in this paper (Section 4)
Goal and scope definition	<ul style="list-style-type: none"> Purpose of the specific study (Consoli et al., 1993; Finnveden et al., 2009) System boundaries (geography, temporal scope, technology, etc.) Functional unit and reference flow, defining the function of both the system and the waste handled (type, amount, composition, time horizon, geography, etc.) Inventory modelling approach adopted for additional functionalities (material and energy recovery) 	<ul style="list-style-type: none"> The temporal scope of the study is not specified System boundaries are not well defined Unclear distinction and incomplete definition of the functional unit and reference flow 	Societal needs and aims, production and consumption patterns	<ul style="list-style-type: none"> Waste management system (Section 4.1)
Life Cycle Inventory (LCI)	<p>Foreground system</p> <ul style="list-style-type: none"> Detailed waste characterisation, representative material fractions and composition Technology modelling: material flows, rejects, emissions, consumption and recovery <p>Background system</p> <ul style="list-style-type: none"> Upstream and downstream activities linked to waste management (e.g., auxiliary energy and materials, recovered energy and materials) 	<ul style="list-style-type: none"> Data not representative of the goal and scope of the study (e.g., waste composition, water content) Case-specific data often missing Challenging and time consuming Use of single values Insufficient linking of inputs to outputs No attention to the quality of recovered materials Superficial use of external databases No attention to representativeness and compliance with goal and scope 	<p>Societal needs and aims, production and consumption patterns</p> <p>Energy sources and system; products, materials and markets</p>	<ul style="list-style-type: none"> Waste input (Section 4.2) Waste technologies (Section 4.3) Energy exchanges (Section 4.4) Material exchanges (Section 4.5) Nutrient exchanges (Section 4.6) Impact assessment (Section 4.7)
Life Cycle Impact Assessment (LCIA)	<ul style="list-style-type: none"> Classification: LCI associated with substances emitted to different compartments Characterised impacts (midpoint, endpoint). Classified substances associated with a characterisation factor for specific impact assessment models Normalised impacts. Characterised impacts are compared to reference impacts (e.g., Benini et al., 2014; Sala et al., 2018, 2017). Weighting, associates normalised impacts to a defined value system (e.g., Sala et al., 2018) 	<ul style="list-style-type: none"> Not specifically developed and not always sufficient for waste management Issues between the temporal scope of the assessment and the time horizon of characterisation methods Characterisation factors for biogenic carbon (e.g., stored carbon) unspecified 	Substances of concern, exposure pathways, environmental challenges	
Interpretation	<ul style="list-style-type: none"> Evaluation of the results to the goal and scope Discussion of assumptions and limitations Sensitivity and uncertainty analysis (Bisinella et al., 2016; Clavreul et al., 2012) Conclusions and recommendations 	<ul style="list-style-type: none"> Relying only on single values Missing sensitivity and uncertainty analysis Missing identification of critical parameters 	Societal needs and aims	<ul style="list-style-type: none"> Interpretation (Section 4.8)

2006a, 2006b), has been used for over two decades to quantify the use of resources and the environmental benefits and burdens of waste management (Finnveden et al., 1995). The first prominent studies addressing methodological aspects in this regard were published by Finnveden (1999) and Clift et al. (2000). More recently, Laurent et al., (2014a, 2014b) reviewed 222 scientific papers published before 2013, and Khandelwal et al. (2019) reviewed 153 papers published between 2013 and 2019, all on LCA of waste management. Recently, Mulya et al. (2022) reviewed 240 papers, ranging from 2009 to 2020, concerning methodological trends in waste management LCAs.

With over 500 peer-reviewed scientific articles on waste management LCA modelling, we can clearly define existing characteristics and challenges that differentiate waste LCAs from LCAs of products and systems (Table 1). Table 1 provides key waste LCA characteristics based on Christensen et al. (2020), associated with a summary of current challenges based on the review by Laurent et al., (2014a, 2014b), for each standard phase of an LCA study (goal and scope, life cycle inventory (LCI), life cycle impact assessment (LCIA) and interpretation (European Commission, 2010; Finnveden et al., 2009).

We acknowledge that many published studies fully meet the criteria for a well-structured and documented waste LCA, but many have the limitations listed in Table 1, as highlighted in the reviews of Laurent et al. (2014) and Bisinella et al., (2021a). Such limitations must be acknowledged to understand the need for a new approach focused on future challenges. For example, a missing or unclear definition of the time frame of a study may lead to unclear temporal validity in relation to the assumptions and temporal representativeness of data, thereby potentially leading to results only relevant for a period much shorter than what is relevant for the decision-support situation. Unclear modelling of additional functionalities, on top of the primary function of treating waste, can limit the long-term societal relevance of the system. The time- and data-demanding task of compiling a detailed LCI often leads to the use of old and “borrowed” data, with little representativeness given to the goal and scope of the study (e.g., temporal and geographical validity). Data on waste technologies are also in relatively short supply, and thus they are often used with little consideration as to their representativeness. Technology parameters are usually single values, and distributions are rarely used to express parameter ranges. Background data, e.g., representing materials and energy used in operations or recovered materials and energy, are obtained from external databases, for example Ecoinvent (Wernet et al., 2016). While this is typical for any LCA, LCAs of waste management systems typically focus on the foreground system and do not employ an equally critical approach to the relevance of the background data, e.g. regarding the quality of recovered materials substituting for virgin materials, or the validity of the data for the goal and scope of the LCA (e.g., temporal and geographical, assumptions and modelling approach) (Bisinella et al., 2021a). LCIA methods are general and not specifically developed for waste management systems. Challenges may arise concerning the time horizon of the impacts (Hellweg et al., 2005) and the characterization factors for the emission and sequestration of biogenic carbon (Christensen et al., 2009). Finally, the interpretation phase is still too rarely complemented with sensitivity and uncertainty analysis (Bisinella et al., 2021a). Focusing on single-value results with little quantification of the uncertainty related to values and parameters governing the results limits a study’s usefulness in complex decision-making.

Overall, waste LCAs often ignore how limited the results may be in terms of the scope and time relative to the decisions and investments they represent. These limitations are severe for waste LCAs performed today, because the decisions they support involve technical investments that will be in operation for decades to come. Therefore, waste LCA approaches must evolve in order to be compatible with the fundamental and transformative changes that are advocated for our societies and sectors. Specific challenges and suggestions for dealing with them are provided in Section 4.

3. Societal changes relevant to waste LCA

Societies around the world will change in the coming decades, beyond general economic and political development: climate change and the transition to renewable energy may top the list of priorities (Kim et al., 2023; Svensson and Wahlström, 2023). How and when countries will mitigate and adapt to climate change cannot be accurately predicted, but many have already set ambitious targets for reducing greenhouse gas emissions, e.g., climate neutrality by 2050 – as dictated by the European Green Deal (European Commission, 2021). The geopolitical situation and the urgent nature of the energy crisis may also push even further toward a green transition (e.g., International Energy Agency, 2022).

Internationally, researchers and authorities have formulated models aiming to provide insights into sustainable development scenarios, such as integrated assessment models (IAMs) and energy-economy-environment (E3) models (O’Neill et al., 2014; van Vuuren et al., 2011). For example, shared socioeconomic pathways (SSPs, i.e., projections of socioeconomic global changes) are combined with representative concentration pathways (RCPs, namely potential trajectories for atmospheric radiative forcing) to derive greenhouse gas emission scenarios according to different climate policies (Riahi et al., 2017). Some approaches exist to link these large-scale scenarios to product and system LCAs (Mendoza Beltran et al., 2020; Sacchi et al., 2022; Steubing and de Koning, 2021). The Python package PREMISE (Sacchi et al., 2022), for instance, transforms the Ecoinvent database (Wernet et al., 2016) in line with the integrated assessment models IMAGE (Stehfest et al., 2014) and REMIND (Baumstark et al., 2021) under future policy scenarios (SSPs and RCPs) for any year between 2005 and 2100. Using these models allows changing the background database used for the LCA model, which changes burdens and benefits associated to energy, materials and transport used in the foreground system model. Another approach is scenario analysis, which involves choosing a set of potential alternative futures for the system at hand (Bisinella et al., 2021a). In this case, changes in the background datasets need to be done manually. Langkau et al. (2023) suggest using PESTEL (political, economic, sociological, technological, environmental and legal) analysis to study key external factors influencing the assessed system.

Regardless of the specific scenarios and models used, the advocated phasing out of fossil inputs into the energy system and industry (e.g., Svensson and Wahlström, 2023) will necessarily lead to profound changes in many areas, some of which we probably cannot anticipate now, and some of which we already face. For example, we will need a new fleet of non-fossil energy sources to supply for a growing energy demand (e.g., Olkkonen et al., 2023). Phasing out fossil fuels directly affects feedstock availability for the materials sector by limiting the availability of fossil building blocks for plastic polymers, which will have to be recycled from existing polymers or provided by biological pathways (Bauer et al., 2022). While this is only an example, it illustrates that profound transitions do not only lead to significant technological changes, but will also ultimately affect waste management practices, for example in terms of energy and feedstock recovery from the waste, but also in terms of evolving composition of waste materials.

The following implications will be particularly relevant for the background system interacting with waste management systems, as it will, in turn, affect all waste LCA phases, for example in terms of societal needs and aims, the function of waste management systems, environmental impacts of concern, etc.

(i) Energy. The production of electricity, heat and fuel will need to implement profound changes to service a post-fossil society (Zhao et al., 2021; Korkmaz et al., 2020). Electricity consumption is foreseen to increase many times over, not only because electrification will be employed as a way to reduce the use of fossil energy (Olkkonen et al., 2023), but also because of the need for power to produce, for instance, hydrogen for storable fuels for anything that cannot be electrified (Daiyan et al., 2020). The intensified use of non-fossil energy sources

will change the environmental profile of electricity, heat and fuels – and thus the environmental loads related to using them and the environmental savings in recovering them.

(ii) Materials. The global feedstock provision, for example for polymers, will need to change to adapt to a progressive defossilisation of society, while needing to supply for an ever-increasing demand (Hundertmark et al., 2018; Levi and Cullen, 2018). This may lead to an intensified use of secondary and bio-based feedstock materials (Kähler et al., 2021; Skoczinski et al., 2021), while optimising production processes in terms of emissions, resource use and energy efficiency (e.g., Rivas-Interian et al., 2023). Together, these transformations are intended to result in a drastic decrease in the impacts of material production, which will be reflected in waste LCAs by reduced credits for recycled materials.

(iii) Nutrients. The need for land will change, partly because of the needs described above, and partly because of population growth and increasing middle-class prosperity (Byrnes and Bumb, 1998; Penuelas et al., 2023). This may affect the need for fertilisers (Ahvo et al., 2022; Sandström et al., 2023), and, in addition, nitrogen fertiliser (ammonia, urea) production will also change, as it is currently produced primarily on natural gas (Jensen and Hauggaard-Nielsen, 2003; Peoples et al., 2019). Today, nutrient recovery from waste only plays a small local role, but this could potentially be different in the future and subsequently increase the focus on nutrients, particularly phosphorus, recovered from organic waste and ashes.

The societal changes sketched above are already emerging today in some parts of the world and will, to varying degrees, form the societal background for many of the investments in waste management that we make today and in the near future.

4. Challenges in future waste LCA

Table 1 summarises the characteristics of – and current challenges facing – waste LCAs. Future societal changes will affect waste management systems and technologies and, in turn, waste LCAs. Future waste LCAs will therefore also have to address current issues in light of future societal changes (Section 3).

In the following sections, we describe challenges and developments and provide recommendations in selected areas (see Table 1): modelling of the waste management system (Section 4.1), in terms of identifying the main function (goal), system boundaries (scope). Additionally, how to take into account additional functionalities (the inventory modelling approach and, consequently, database choices) will be challenged in terms of, among others, performance criteria, temporal scope and functional units. In the foreground system, waste input will change in relation to composition, type and amount (Section 4.2). Likewise, technologies may change to meet quality criteria valid within the societal framework (Section 4.3). In the background system, the use and recovery of energy (Section 4.3), materials (Section 4.4) and nutrients (Section 4.5) will have different environmental implications. Material recovery may, in the long term, also include carbon. The LCIA phase may require defining the relevant time horizon critical for characterisation factors, including potential changes in the future, and be adjusted to cover relevant aspects of waste management (Section 4.6). The interpretation phase will need to identify environmentally robust and flexible solutions that respect criteria valid today as well as those applying to a post-fossil society. Furthermore, results will identify a potential range of performances, using uncertainty analysis rather than providing accurate quantitative results (Section 4.7). Individual issues are discussed in more detail below.

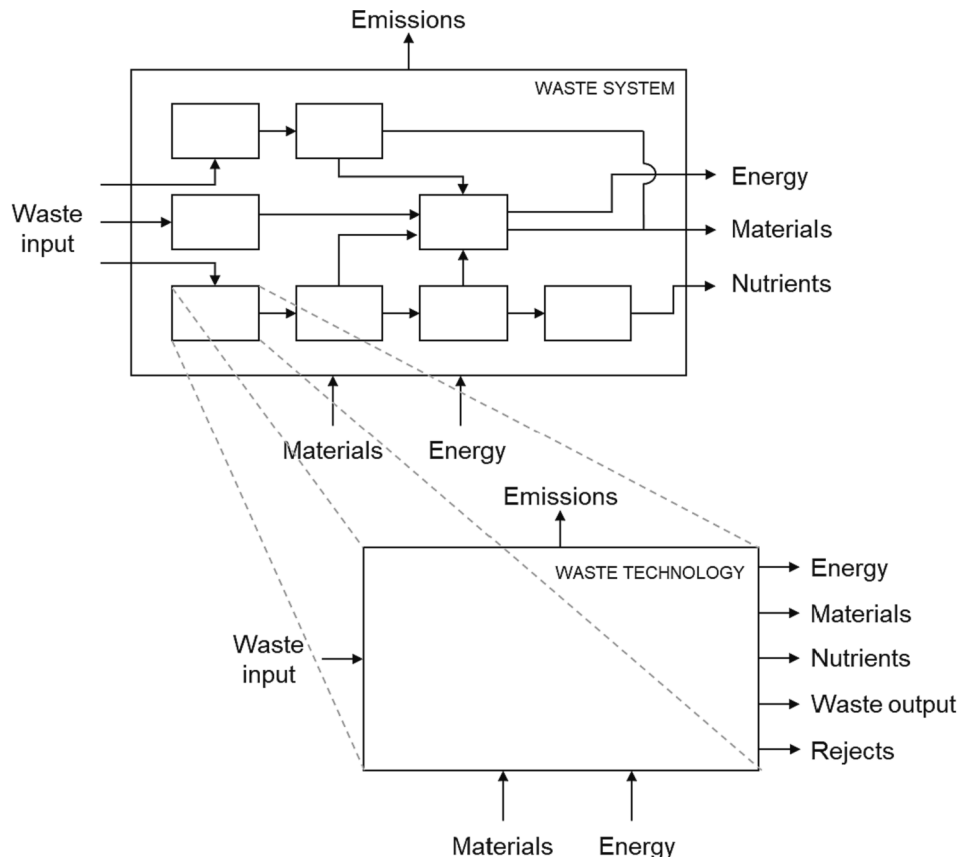


Fig. 1. Illustration of a waste management system, including an illustration of a zoomed-out technology.

4.1. Waste system modelling

A waste system is the web of flows and technologies handling waste, including collection, transport, treatment and the recovery of energy, materials and nutrients (Fig. 1) (Christensen and Fruergaard, 2010). It uses materials and energy for its construction and operation, and it causes emissions from waste and energy used during operation and treatment. Recovered energy, materials and nutrients are used in society and contribute to reducing the need for virgin materials and energy production. Data describing waste management systems (foreground and background systems) are the sum of data taken from each of the individual technologies. The difference in the data for individual technologies and the system is that individual technologies most often have an output of waste for further treatment, as well as rejects or side streams, while on a system level, material flows are followed until final disposal or until the flow becomes a secondary raw material. Data on waste systems represent the physical performance of the system and thus constitute the LCI of an LCA model.

The modelling and conceptualisation of a waste management system is the first and most important step in any waste LCA. Data and material flows depend not only on the physical representation of the waste management system, but also on the use of subsequent results, i.e., the function to be addressed. This is reflected in the functional unit of the study and the reference flow dealt with (type and amount of waste considered). These aspects are defined in the goal and scope phase of the LCA, which also specifies system boundaries (time horizon of the study, use of energy and materials) and how to take into account additional functionalities other than treating waste.

Additional functionalities in an LCA are commonly accounted for by substitution, i.e. by crediting the waste management system with an avoided production involving primary energy sources and virgin materials. Another way to account for additional functionalities is system expansion (Heijungs et al., 2021). Additional functionalities such as energy recovery, material recovery and carbon sequestration are very important for waste management system LCAs because they constitute the environmental benefits that could be obtained by treating waste. In this study, we refer to energy, material and nutrient use and recovery by waste management technologies as “energy exchanges” (e.g., Zhao et al., 2022) (Section 4.4), “material exchanges” (Section 4.5) and “nutrient exchanges” (Section 4.6). The modelling of such exchanges depends on – and should be compliant with – the chosen modelling framework for the LCA (e.g., attributional, consequential), which determines the type of processes and data used (average or marginal) (cf. Ekvall, 2019). The temporal scope of the LCA is very important as well, in that short-term assessments are usually based on production data (e.g., energy sources for electricity production for the geographical scope of the study), while long-term assessments are based on foreseen increased capacity (e.g., scenarios for planned capacity increase in order to meet future electricity demand). Depending on the chosen approach, the exchange is referred to as an “average mix”, a “single-marginal” or a “marginal mix” (Fruergaard et al., 2009; Weidema et al., 1999). Once such exchanges have been identified, the challenge resides in identifying an appropriate dataset to represent the environmental impacts connected to the provision of such resources. In this regard, important factors to consider are the temporal and geographical representativeness of such datasets.

Future LCAs must be based on a stringent and transparent representation of waste management systems in order to provide results accommodating the main future developments. This representation depends on the goal and scope of the study. In particular, previous waste LCAs have focused, to a large degree, on comparing waste technologies and have often used “treatment of 1 ton of waste” as both the functional unit and the reference flow, as observed by Galvez-Martos and Schoenberger (2014). This approach is useful, albeit only in a static situation, and it does not account for the complexity of the future, for example, including capacities and lifetimes of waste management systems, “treatment of waste in a specific country in a specific time frame”.

Cimpan et al. (2015) offer an example of investigating capacities, albeit not including this in the functional unit. Examples of more comprehensive functional units are provided in LCAs of products, for example the packaging sector (Frigerio et al., 2023).

We suggest that future waste LCAs should consider:

- A clear definition of the goal, scope and function of the system related to societal change to come. This involves waste composition, the time horizon of the study, technologies considered, impacts in focus and geographic location. The LCA should contribute to finding a reasonable development plan, thus not only ensuring consistent, environmentally sound waste management, but also allowing for introducing changes as needed. Considering the numerous and profound changes that would be required for a sustainable transition, it may be useful to model the system for selected time intervals or transition stages, each representing an important development in the foreground or background system in the transition to a post-fossil society. This will help identify critical transitions, i.e., “stages of the future,” before they happen, for example conditions for which one waste management technology or solution might be preferable over another in a specific stage of the sustainable transition, or the technology or solution providing a good environmental performance in a range of potential future developments.
- Identification of existing waste management facilities and capacity available for each development phase, facilitating a smooth and environmentally sound transition. Only a few models are currently capable of doing so (Levis et al., 2014).
- Systematic inclusion of capital goods. Earlier studies have shown that capital goods, primarily in terms of steel and concrete, could be significant contributors to the overall environmental impact (up to 25 %) (Brogaard and Christensen, 2016). The cement and steel industries are heavily dependent on fossil fuels, and since they work at high temperatures, alternative fuels are less obvious. This suggests that the use of these materials in capital goods may still constitute a significant environmental load in many years ahead.
- Physical-based system modelling of future waste LCAs. Physical-based system modelling (or input-specific modelling) helps connect differences in input waste to each technology and to system outputs. This includes information on the characteristics of waste input (see Section 4.2), as well as relationships between technological inputs and outputs (see Section 4.3). The physical-based and input-specific modelling of the system is crucial for ensuring compatibility between the characteristics of waste input and the amount and quality of recovered outputs, for example energy (Section 4.4), materials (Section 4.5), and nutrients (Section 4.6).

4.2. Waste input

Waste is a left-over, discarded material (Christensen and Fruergaard, 2010) constituting the input to the waste management system. Waste input is characterised by waste from different sources in society (e.g., households, services and industries) with varying fractional (different materials, e.g., plastic, paper; Edjabou et al., 2015; Riber et al., 2009) and physicochemical composition (e.g., water, carbon, and energy content; Götze et al., 2016; Larsen et al., 2013). Moreover, waste input is an important factor when planning, understanding and improving waste management technologies and systems. For example, the type of waste input affects the choice of treatment (Montejo et al., 2013) as well as the capacities of treatment plants (Cimpan et al., 2015). Waste material causes direct emissions into the environment during treatment and disposal (e.g., incineration and landfilling, Aurell et al., 2019), but it also contains valuable resources, such as carbon, energy and nutrients, all of which should be recovered and recycled to the greatest extent possible (European Commission, 2015). Waste input is highly context-specific and may vary greatly over time and location due to societal, geographical and cultural patterns (Christensen and Fruergaard, 2010).

Variations affect waste sources, waste fractions (e.g., due to consumer behaviour or collection strategies; [Edjabou et al., 2019](#)) and physico-chemical composition (e.g., large data ranges observed by [Götze et al., 2016](#)). In addition to this heterogeneity, waste input data can be subjected to significant uncertainty due to differences in sampling and characterisation procedures ([Lagerkvist et al., 2010](#)).

Understanding the link between context-specific waste input, waste technologies and emissions into the environment is essential when seeking to quantify the efficiency and environmental performance of a waste management system ([Laurent et al., 2014a](#)). Nevertheless, [Laurent et al., \(2014a\)](#) found that waste input was poorly documented and not sufficiently described in most peer-reviewed waste-LCA studies. However, when included, waste composition was a very sensitive parameter for the outcomes of LCA studies. [Bisinella et al. \(2017\)](#) showed that the choice of composition and uncertainty surrounding waste input in input-specific LCA models had a fundamental influence on environmental emissions associated with waste treatment, recycling and disposal. Applied to the incineration of waste, LCA results are directly influenced by the carbon, energy and trace elements contained in the waste input ([Astrup et al., 2011](#)). Carbon, nutrient and trace element contents affect the LCA performance of waste refineries ([Tonini et al., 2013](#)), the use on land of residues from biological waste treatment ([Bruun et al., 2006](#); [Hansen et al., 2006](#)) and the storage of biogenic carbon in landfills ([Christensen et al., 2009](#)).

The amount, types, fractions and composition of input waste will likely change in the future. Additionally, the amount of generated waste is expected to increase, due to increasing population and prosperity. Based on SSP scenarios from the IPCC ([Riahi et al., 2017](#)), higher waste generation rates are connected to higher income, with an expected peak in global waste generation foreseen to happen in this century ([Hoornweg et al., 2015](#)). Specific studies have been carried out for waste types of concern, for example, global plastic waste ([Borrelle et al., 2020](#); [Lebreton and Andrady, 2019](#)) and electronic waste ([Parajuly et al., 2019](#)). In Europe, despite improvements in waste management, driven by various EU waste policies and targets ([European Commission, 2020](#)), the amount of neither total nor residual (non-recycled) municipal waste generated has decreased since 2015 ([European Environment Agency, 2023](#)). Variations in waste amounts mean that studies will need to take into account the capacities and lifetimes of facilities and waste management networks, for example locally and nationally ([Cimpan et al., 2015](#); [Pizarro-Alonso et al., 2018](#)).

Consumption patterns, waste policies and practices are foreseen to change the composition of waste input into waste management systems, in terms of waste type, fractions and composition. Furthermore, waste trading policies in Europe can affect the composition of waste received by existing waste treatment facilities ([European Commission, 2021](#)). [Arushanyan et al. \(2017\)](#) and [Thomsen et al. \(2017\)](#) used scenario analysis to assess changes in waste flows and collected fractions as well as changes in the overall waste composition. Other relevant examples where authors have used scenarios to assess the evolution of waste amounts and waste fractions for waste management system LCAs are [De Moraes Lima et al. \(2019\)](#), [Deus et al. \(2017\)](#), [Irfan et al. \(2020\)](#), [Mastrucci et al. \(2017\)](#), [Meylan et al. \(2018\)](#), [Pizarro-Alonso et al. \(2018\)](#) and [Song et al. \(2018\)](#). The geographical location of waste generation and policies with respect to waste trade can also affect the composition of locally handled waste.

The physicochemical composition of future waste fractions is also foreseen to change due to different materials used in society. For example, the use of bio-based polymers will increase, in turn affecting the share of biogenic carbon in plastic waste, with unclear end-of-life effects ([Rosenboom et al., 2022](#)). In this respect, scenario analysis and MFAs can be used as foresight tools, since products and materials in use today will become waste in the future. This approach is used, for example, by [Mastrucci et al. \(2017\)](#) for the building sector and by [Pratt et al. \(2016\)](#) for regional circularity initiatives. Waste generation can be estimated based on historic use patterns, which has been done with

respect to recovery potentials (for instance by [Buchner et al. \(2017\)](#) regarding aluminum recycling potentials in Austria) as well as the potential contamination of recycling materials (for instance by [Pivnenko et al., 2016](#), regarding chemicals in paper cycles). New products and materials in society may also require taking novel attention to chemicals and substances of concern, such as persistent chemicals (e.g., PFAS, BPS and BPA in paper products, [Pivnenko et al., 2018](#)) and nano-enabled materials and waste ([Hartmann et al., 2019](#); [Heggelund et al., 2016](#)). In turn, these challenges may require unique pollution control objectives.

To date, energy and material recovery has governed the results of most studies in the literature ([Laurent et al., 2014b](#)), almost always resulting in net environmental benefits despite burdens incurred by recovery processes ([Christensen et al., 2020](#)). However, another anticipated change is that waste composition will become one of the most sensitive – if not *the* most sensitive – aspects in waste management system LCAs ([Bisinella et al., 2017](#)). For example, to date, climate change impacts connected to CO₂ emitted from waste incineration have been compensated by climate change savings in energy recovery ([Fruegaard et al., 2009](#)). However, with energy systems transitioning to lower carbon intensity, energy recovered from waste incineration may substitute for its production from renewable sources with a low carbon footprint, thereby making the incinerator a net climate change burden ([Bisinella et al., 2021b](#)). This was also observed, for example, in a Chinese case study ([Zhao et al., 2022](#)). Another example is changed benefits from material recovery due to expected electricity de-carbonization ([Šimaitis et al., 2023](#)). In these cases, the representativeness of waste composition data used for a study is fundamental in obtaining reliable results. Other authors have also identified outdated waste composition data as limiting their studies ([Chen et al., 2019](#); [Friedrich and Trois, 2016](#)).

We thus suggest that future waste LCAs should consider:

- Amounts and compositions of all relevant waste types for the study. Composition is a detailed list of material fractions and the physico-chemical content of each material fraction. Waste amounts and compositions are crucial to providing a relevant input-specific model that helps connect waste material fractions and compositions to waste management system outputs. For example, this is necessary to model detailed source separation schemes and to assess the quality of recyclables.
- The best available data on amounts, waste types and composition (waste fractions and physico-chemical characteristics), in that the quality of waste input data used for the study should be discussed in terms of representativeness in relation to the goal and scope of the LCA (i.e. the data used should be representative for the specific study and the time horizon it aims to cover). Ideally, waste sampling campaigns should be carried out for each study, as the season, waste collection scheme and geography are known to influence the waste composition data ([Edjabou et al., 2021](#)). When this is not possible, practitioners should look for the most recent and geographically-accurate waste data available.
- Encouraging local authorities and stakeholders to carry out frequent sampling and characterisation of waste. This will guarantee the presence of waste data; the frequency of waste characterization will allow achieving good knowledge of the waste management system at hand. In particular, comprehensive analyses should improve our knowledge about carbon (fossil and biogenic carbon), energy and moisture content.
- The potential evolutions of waste input (amounts and compositions) within the LCA's time horizon. In this respect, we can utilise scenario analysis to assess the sensitivity of our assumptions (population, consumption patterns, waste amounts, composition, etc.). An MFA can be used for specific issues to help determine waste generation from long-lived products (construction and demolition waste, textile

waste, etc.). A solution could be defining waste input with reasonable time steps, for example 5 years, representing likely developments.

4.3. Waste technologies

Waste treatment technologies employ mechanical, biological, chemical and thermal processes to recover materials, energy and nutrients from waste and to dispose of residues. Examples of common waste treatment technologies include separate collection, mechanical sorting, anaerobic digestion, composting, incineration and landfilling. These technologies are applied in different combinations and to different waste streams, ideally to conserve natural resources and minimise emissions from waste management (cf. article 1 of the European Waste Framework Directive, European Commission, 2008). In addition, logistics is a central element of waste management, because the collection and transportation of waste are needed to concentrate flows and make them technically and economically suitable for treatment.

Treatment technologies are modelled as part of the foreground system in a waste LCA. This can be done in two ways. The first is “input-specific,” whereby waste material fractions in waste input (see Section 4.2; e.g., paper, plastic, ferrous metal) and their physical–chemical compositions (e.g., fossil C, biogenic C, calorific value, water content) are related mathematically to emissions, residues and product yields from waste treatment processes. The second uses generic LCIs, which are based on pre-defined reference flows to represent average performances of a waste treatment technology. In fact, waste treatment technologies are often modelled as a combination of input-specific modelling and generic LCIs, as illustrated in Fig. 2. Generic LCIs may be simply linked to the amount of waste treated or to the composition of treated waste. Input-specific LCAs may utilise MFA principles to track material and substance flows throughout the modelled waste management system (e.g., Clavreuil et al., 2014), thereby linking resource recovery (materials,

energy, nutrients) and short- and long-term emissions of waste technologies to input composition.

Data on waste treatment technologies are often collected specifically for major processes in the foreground system and extracted from secondary sources, such as databases or literature, for processes in the background system (Mulya et al., 2022). In general, waste treatment technologies are therefore represented by data related to a certain scale and/or a certain plant corresponding to a specific development stage. Technology implementation (even at the same stage) may vary from one plant to another, which implies uncertainty with respect to technology inventories on a more aggregated level, which is typically needed for assessments from a waste system perspective with a broad geographical scope. Therefore, modular LCA approaches have been proposed (cf. Steubing et al., 2016) and linked to material flow analysis of waste management systems (Haupt et al., 2018; Schmidt and Laner, in press). With respect to the maturity of technologies, inventories (including uncertainty ranges) need to be generated in consideration of technology readiness levels (TRLs) and the availability of plant operation data at various scales (lab-, pilot-, full-scale). Different data estimation methods are used to generate technology inventory data, which comprise process simulation, manual calculations, structural models, and use of proxies (cf. Tsoy et al., 2020). The uncertainties associated with technology inventories depend on the data estimation method and the quality of underlying information. Typically, the uncertainty associated with technology inventories increases with lower TRL and smaller scale of implementation. Nevertheless, early technology assessment has great potential to identify environmental issues before technologies achieve broad application (Villares et al., 2017).

Landfilling constitutes a special challenge regarding technology inventories, because emissions occur over extensive time periods, resulting in the so-called ‘flux-pulse problem’ (Laner, 2009). While emissions differ between mineral landfills and those containing organic waste, the

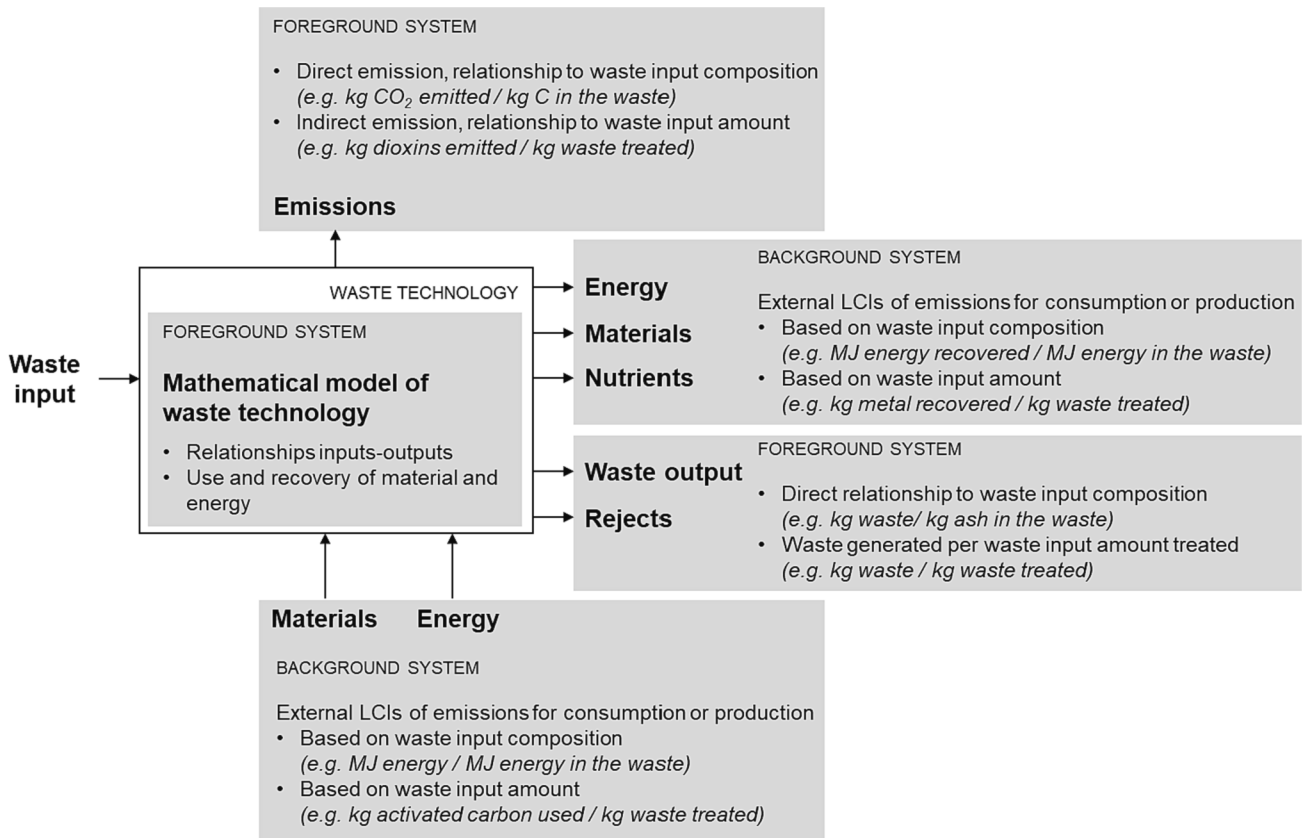


Fig. 2. Illustrative structure of input-specific waste technology modelling in an LCA. Waste input data can have a direct (input-specific) and indirect (external LCIs, process-specific per property of reference flow) effect on results.

principal issue of emission periods extending beyond the temporal scope of the LCA is common to both types of landfills. For mineral landfills that have received, for instance, fly ashes from incineration, contaminants could be leached for many thousands of years (Hellweg et al., 2005). Initially, emissions will be captured and treated (and can be considered via wastewater treatment inventories), but pollution potentials will persist for millennia and need to be considered even if they extend far beyond the temporal scope of the LCA. For instance, by distinguishing different emission time frames (cut-off vs. indefinite) in the life cycle inventory and evaluating the effects on the LCA results (e.g., Huber et al., 2018; Maresca et al., 2021). This issue of stored toxicity is addressed later herein in the section on impact assessment. For landfills containing biodegradable organic matter, landfill gas mainly containing CH₄ and CO₂ is produced under anaerobic conditions. Typically, some of it is collected and utilised, some is oxidised in the top cover and some will be released into the atmosphere. Gas generation is typically estimated based on a first-order decay function (e.g., Barlaz et al., 1990), with parameters based on laboratory-scale investigations and measurements of the stability of materials removed from old landfills (e.g., Cruz and Barlaz, 2010; Niu et al., 2023). The generation and emission of landfill gas into the atmosphere varies from one site to another (subject to waste composition, collection and treatment, landfill engineering and operation, climate, etc.) and over time (e.g., operation and degradation phases), which is why inventory data are associated with significant uncertainty (e.g. Frank et al., 2017). Furthermore, biogenic waste fractions, which are not – or hardly – degradable under anaerobic conditions (such as wood or certain textiles), should be accounted for as stored biogenic carbon (Manfredi et al., 2009). Therefore, estimates on methane emitted into the atmosphere, as well as stored carbon, are dependent on many site-specific factors and uncertain model parameters, which is why inventories on landfilling need to be consistent regarding waste input and technology implementation as well as explicit with respect to the temporal scope of the inventory data.

A central characteristic of waste treatment technologies is that there is a constant need for adaptation and development because of not only quantitative and qualitative changes in waste input over time, but also research opening up new technological options to recover material, energy and nutrients more efficiently or via different process routes.

As an example, plastic materials and their waste management have been under constant pressure in recent years to create more sustainable plastic handling systems (e.g., plastic strategy, European Commission, 2018). Rapidly growing waste amounts (Geyer et al., 2017), the large diversity of plastic types and compounds (Wiesinger et al., 2021) and the need to transform plastic use from a highly linear to a more circular system require rapid systemic and technological developments. Furthermore, because source separation of plastic waste at the household level is not expected to be able to take full account of the diversity of plastic types used in different products, mechanical sorting in the future may also further gain importance in plastic waste management systems. Apart from mechanical sorting as a post-treatment solution for residual household waste (Cimpan et al., 2015), technological developments could involve tracer-based sorting. Experiments on the lab to pilot scale have shown good sorting performances for plastic packaging waste (Woidasky et al., 2021), but technology implementations on the full scale do not yet exist. Chemical recycling, on the other hand, has been discussed as an emerging treatment technology for a variety of mixed plastic waste streams that are not suitable for mechanical recycling. This could be particularly attractive for composite materials, multi-layer foils or residues from plastic waste sorting, where environmental benefits can be achieved compared to incineration with energy recovery (Civancik-Uslu et al., 2021; Volk et al., 2021). Despite some initial LCA studies, technology performances in relation to chemical plastic recycling are currently difficult to assess due to the low number of technologies in operation and the lack of data on their operating conditions (Quicker et al., 2022). Another emerging technology for plastic recycling could be enzymatic or microbial recycling, whereby

specific enzymes break down polymer chains into oligomers and monomers, which can then be re-polymerised into high-quality plastics. Though potentially attractive, particularly for composite materials, applications in this field are still on very low TRLs (up to 4, i.e., lab-scale experiments) and very little is known to date about the enzymatic degradation of the majority of polymers by enzymes (Chow et al., 2022). Hence, several challenges exist with regard to full-scale implementation and the availability of operational data from industrial-size plants. In order to allow for meaningful comparisons between these technology options for plastic waste management, technology performance needs to be modelled mechanistically on a system level and be input-specific.

Variations in waste inputs, auxiliary materials and energy demand, as well as in emissions, residues and product outputs, need to be consistently accounted for allowing fair technology comparisons. In general, environmental assessments of waste technologies need to consider technological maturity via the explicit consideration of TRLs. For existing (high TRL) technologies, different types of implementation need to be accounted for, and potential effects of specific process datasets and choices should be communicated by uncertainty and sensitivity analysis. Furthermore, the role of capital goods and treatment capacities associated with existing technologies needs to be considered with respect to technology transformation pathways. For emerging (or low TRL) technologies, model results should be assessed from an ex-ante LCA perspective (Cucurachi et al., 2018; Villares et al., 2017) and used to identify critical factors and suitable system settings for technology implementation (Laner et al., 2016a). With respect to prospective performance and changes over time, it should be considered not only that simpler technologies have shorter development times and may reach market maturity more quickly, but also the effect of their implementation needs to be assessed from a system perspective. For more complex technologies, longer development times are needed until they can be put to market, which implies that prospective system configurations (foreground and background) should be used to assess the potential effects of their implementation (see Klotz et al. (2023) for an example related to plastic waste management). LCA modelling of emerging technologies is characterised by uncertain conceptual modelling of flows, material and energy use, as well as emissions. These data are most often lab- and pilot-scale and require estimates for scaling up to system integration (Tsoy et al., 2020).

We thus suggest that future waste LCAs should consider:

- Modelling of technologies with a physical approach representing waste material fractions and composition, thereby linking technology performance and outputs to the input in respect to overall mass balances. Only in this way can technologies reflect changes in input and provide information that allows for the quantification of all residues and the quality of recyclables.
- Technology data as case-specific as possible. Even within the same technology, performance varies greatly, and technology data should always be carefully assessed in view of the scope of the study.
- Within known technologies, landfilling constitutes a specific challenge because of highly variable emissions and long-term perspectives. Therefore, considered time frames and modelling approaches should be made explicit and critically discussed with respect to their effect on the environmental technology performance, in particular, regarding climate change aspects and the degradation of organic matter in landfills.
- Data completeness and comparability between technologies of different TRLs. This issue is relevant in many areas, but it is particularly urgent regarding technologies treating plastic waste, since many different approaches are emerging. For early-stage technologies, estimates will be necessary for scaling-up to system integration, for example as outlined in the step-wise approach provided by Tsoy et al. (2020).
- Technology forecasts within relevant areas may identify upcoming technologies useful within waste management. Complex

technologies may have long development horizons, and only those potentially reaching sufficient TRL within the scope of the study should be considered. Technology simulations can be used to sketch the environmental performance of potential new technologies.

4.4. Energy exchanges

Waste management systems interact significantly with energy systems in terms of the use of electricity, heat and fuels to run the system, as well as in terms of recovered energy delivered to the surrounding society.

All waste management technologies use energy, i.e., transportation, equipment, heating and in control and regulation. The energy content of waste can be recovered through different treatment processes, such as thermal (e.g., incineration, pyrolysis and gasification) and biological (e.g., anaerobic digestion and landfill). This recovered energy can be in the form of electricity, heat, biogas and transportation fuels potentially used in the surrounding society, thereby reducing its need for primary energy sources. Energy recovery efficiency (net from energy used and produced) depends on the case-specific waste treatment process and conditions.

When modelled in an LCA, energy recovery from waste treatment represents an additional functionality to the primary function of treating waste (see Section 4.1). Taking account of this additional function has been described thoroughly, for example, by Fruergaard et al. (2009). The type of energy exchange and its efficiency significantly affects the environmental impacts of a waste management system and is often one of the most sensitive parameters in waste LCA studies (Bisinella et al., 2021b; Christensen et al., 2010).

The modelling of the energy exchange in the specific waste LCAs depends on – and should be compliant with – the chosen modelling framework for the LCA (e.g., attributional, consequential, see Section 4.1). Irrespective of the modelling framework, however, the challenge for the practitioner is to identify energy sources potentially changing in production or the capacity share in the temporal scope of the study, as well as the datasets representing these energy sources. Identifying energy sources characterising energy exchange can be based on external databases (e.g., Ecoinvent), scenario analysis (Christensen and Bisinella, 2021) or more complex modelling, such as integrated assessment models (IAMs) or general- or partial-equilibrium models (PEMs) (Eriksson et al., 2007; Garcia-Gusano et al., 2017; Gibon et al., 2015; Münster et al., 2010). When using external models for energy systems, it is important to verify alignment between the model and the case-specific LCA in terms of goal and scope (De Camillis et al., 2013; Mendoza Beltran et al., 2020). Other important factors to consider, for example, are whether the energy exchange is storable (e.g., gas versus heat) and if an utilisation network is available (e.g., electricity, district heating or natural gas network). Some energy exchanges require additional considerations, for example firewood and related land use changes (Faraca et al., 2019).

We expect that in the future, energy exchanges will be affected by changes in waste input, influencing the overall energy content of waste, but mostly by changes in the surrounding energy system interacting with the waste management system. Changes in consumption patterns and product policies will influence the amount of energy in waste, by affecting material fractions and their overall physicochemical characteristics (e.g., Bisinella et al., 2022). Changes in waste energy content will also influence the throughput of material in thermal treatment plants, which are designed for specific materials and thermal inputs. More importantly, the environmental relevance of energy exchange per unit of energy recovered depends on the energy systems surrounding waste management systems. Any change in an energy system is a crucial issue, and it is important to make consistent and robust choices regarding the energy exchange used in the study at hand. The temporal (short- or long-term) and geographical scope (local or national network) and the chosen modelling framework for the specific LCA study (attributional, consequential) will also influence affected energy sources in

terms of energy exchange. Moreover, while energy demand will continue to grow, current political targets focus on decreasing the carbon intensity of energy systems.

Precisely determining capacity changes in an energy system over the time horizon of an assessment is a complex task. While electricity markets are national and connected to neighbouring countries, heat cannot be transported far and made available nationally, whilst the determination of heat needs to comply with local conditions and local foreseen changes. To solve this issue, a scenario-based approach can be used, an example of which is given by the energy exchanges scenario analysis approach in Bisinella et al., (2021b) (Table 2). The purpose of the scenarios obtained was to cover potential alternative options for exchanges with the energy system (and their environmental impacts), in order to test the robustness of LCA results. The 12 energy scenarios listed in Table 2 were chosen in order to range from fossil fuel-based energy sources to non-fossil-based energy sources. The highest impacts arose from 1.2 kg CO₂-eq/kWh electricity based on oil; 0.132 kg CO₂-eq/MJ heat based on hard coal, to non-fossil-based energy systems, e.g., the “greenest” scenario presents 0.01 kg CO₂-eq/kWh electricity based on off-shore wind power and 0.009 kg CO₂-eq/MJ heat based on electricity-driven heat pumps. The learning from this scenario analysis, in relation to the climate change impacts of energy exchanges, was that the climate change impact of electricity provision can be expected to decrease by two orders of magnitude, while the change in the climate change impact of heat provision is a decrease of one order of magnitude. In either case, moving towards less carbon-intensive futures dramatically lowered the climate change impacts of energy recovery and their relative importance with respect to LCA results. The environmental impact of energy exchanges in other impact categories will be difficult to estimate a priori, since energy sources with lower carbon intensity do not necessarily translate into lower environmental impacts for other impact categories. For example, electricity generated from solar and wind energy is associated with higher impacts in terms of mineral resource depletion than for electricity from hard coal (based on Ecoinvent v3.8 (cut-off), Wernet et al., 2016).

Other methods for identifying future energy exchanges involve using global scenarios or energy system modelling tools (see Section 3). For example, following SSP scenarios, and according to IMAGE modelling (Stehfest et al., 2014), under the assumption of a limitation of global warming to 1.8–2 °C until 2100 (SSP2-RCP2.6), the inventory data of the market for 1 kWh electricity in 2050 can be transformed by using, for example, PREMISE (Sacchi et al., 2022) to quantify the expected decrease in greenhouse gas emissions. The extent of the expected decrease in greenhouse gas emissions associated with the supply of 1 kWh electricity differs depending on the initial energy system. For example, greenhouse gas emissions from the “market for electricity, medium voltage” in Denmark and Germany are expected to decrease from, respectively, 0.270 kg CO₂-eq and 0.544 kg CO₂-eq to 0.132 kg CO₂-eq (Cf. Section S1 in the Supplementary Material). In this context, it must be mentioned that the geographical resolution of IAMs is lower than the typical geographical scope of a waste LCA (for example, IMAGE distinguishes 26 regions globally and allocates Denmark and Germany to Western Europe), and thus it should be decided on a case-by-case basis whether or not the IAM region appropriately represents the region under study. Political targets may also influence energy exchanges depending on unforeseen events, for example willingness to use carbon-intensive energy sources as a consequence of geopolitical instability (e.g., the use of natural gas for heat), or accelerated defossilisation to decrease dependency on foreign energy sources Lambert et al., 2022).

An additional issue to consider could be the speed of defossilisation of the energy sector; for example, electricity and heat transition to renewable energy may be faster than the defossilisation of the transport sector. Nevertheless, the large-scale decrease in the fossil carbon intensity of the energy system will affect the energy used and recovered by not only waste management systems, but also the auxiliary material supply chain necessary for treating waste.

Table 2

Energy exchange scenarios and corresponding climate change impacts per unit energy delivered for electricity and heat, adapted from Bisinella et al., (2021b). Climate change impacts for single energy technologies are based on Ecoinvent 3.6, substitution, long-term. * Includes land use change. ** Follows the electricity marginal in the corresponding scenario.

Scenarios		1	2	3	4	5	6	7	8	9	10	11	12
Electricity, % share of technologies	Oil	100											
	Hard coal		100			17							
	Natural gas conventional			100	100	51	23		17				
	Biomass*							100					
	Solar panels						45		29	100			
	Wind onshore					23	19		15		100		
Electricity, kg CO ₂ -eq/kWh	Wind offshore					9	13		39			100	100
	Heat, % share of technologies												
	Hard coal	1.20	1.04	0.70	0.70	0.54	0.21	0.21	0.16	0.09	0.02	0.02	0.02
	Oil		100										
	Natural gas			100									
	Biomass*				100	100	54	100		100	100	100	
Heat, kg CO ₂ -eq/MJ	Electricity**						46						100
		0.13	0.09	0.07	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.01

The predicted changes will affect waste management system LCAs in terms of data collection, energy system modelling and the relative importance of energy efficiencies and parameters for LCA results.

We thus suggest that future waste LCAs should consider:

- Addressing energy exchanges in the goal and scope of the study, to make consistent and robust choices regarding the energy exchange used in the assessment. In order to do so, both scenarios and energy system modelling are relevant approaches. It may be advisable to work with different energy exchanges over the time horizon of the study. The precise future energy exchange source and environmental impact may be hard to identify, but the exercise in itself will be extremely relevant in order to assess the robustness of the results in view of potentially different conditions surrounding waste management systems.
- Systematic assessment and interpretation of the energy exchange effect on results, as well as how this changes over the time horizon of the study as energy exchanges develop.
- Verifying that assumptions relating to energy exchanges with slower penetration of renewables are justifiable for the case study at hand (for example, energy exchanges slower to defossilise, such as fuels and biogas, will have higher relative importance for climate change).
- Linking the energy recovered to foreseen changes in the energy content of waste input (see Section 4.1).
- Changing the energy system consistently throughout the model (foreground and background), irrespective of whether we select energy exchanges with scenario analysis or other methods.

4.5. Material exchanges

Material exchanges are mass-related, physical exchanges between the foreground and background systems of an LCA model. Common examples of material exchanges in waste LCAs include the consumption of auxiliaries or the avoided production of material resulting from reuse or recycling. Usually, burdens associated with auxiliary material consumption are considerably lower than the benefits related to recovered materials.

Credits associated with recycled materials in the foreground system depend on the quantity and quality of recyclables and the potential ways of using secondary material. For example, with regard to plastic waste management in Europe, Lazarevic et al. (2010) found that virgin material substitution and assumed substitution factors substantially affect the environmental performance of mechanical recycling. In the majority of waste LCAs, it is assumed that recovered material substitutes for primary material in a 1:1 mass ratio (Viau et al., 2020), which is justified if primary and secondary materials can be used in the same manner. Potential justifications for deviating from assumptions concerning

substitution ratios are a loss of quality, differences in properties, price differences between primary and secondary materials, a limited number of recycling cycles or the mineral fertiliser equivalent in the specific case of compost and digestate (Viau et al., 2020).

A second challenge, besides determining substitution factors, is the choice of suitable datasets to reflect the substituted material. Often, several datasets exist, thus reflecting, for example, different production technologies for a specific material. The selection of the dataset can be crucial for results. Using paper as a case study, Man et al. (2020) showed that the greenhouse gas emissions associated with the production of 1 t of paper ranged from 1.96 t CO₂ eq to 6.55 t CO₂ eq depending on the feedstock material, the supply chain, and the paper product. This underlines the importance of conscious decisions regarding the selection of specific datasets for material substitution, as well as with regard to background datasets in general. Material production and market datasets, employed to model environmental impacts associated with material use, typically rely on external databases. Although data quality standards exist (e.g., Ecoinvent Data Quality Guideline, Weidema et al., 2013), the data quality of specific datasets within databases may fluctuate, because they are partly outdated or reflect a specific process that does not match exactly with the system under study. In selected cases, mono-material substitution does not meet the requirements required to reflect the actual practice of secondary material use, for instance when recycled materials are used in applications with lower quality requirements (e.g., quality deterioration of recycled plastics, which leads to the restricted applicability of recycled plastics in the plastic sector and a shift to other material systems, cf. Demets et al., 2021; Klotz et al., 2022). In this case, reasonable assumptions must be made and documented, and the potential effect of this assumption on the results has to be discussed.

In 2015, material production contributed to 23 % of global greenhouse gas emissions (Hertwich, 2021). To manage shrinking carbon budgets while simultaneously increasing material consumption, drastic transformations of various material sectors are needed. A transition of the energy system contributes to a reduction of material system greenhouse gas emissions. However, to comply with climate change mitigation targets, further measures are needed. The assessment of potential future trends in material use, and the identification of corresponding measures for different material systems, is the subject of various studies (cf. Table 3). Anticipated transformations of material systems comprise the intensified use of secondary and bio-based feedstock materials and the optimisation of production processes in terms of emissions, resource use and energy efficiency. Together, these transformations are intended to result in a drastic decrease in the climate change impacts of material production, which will be reflected in waste LCAs by reduced credits for recycled materials. In LCA studies, it is a common practice to put the focus of future scenarios on the transformation of the foreground system,

Table 3

Anticipated future development of selected materials relevant within material recovery. (This table provides an overview of expected transformations in various material sectors based on a variety of studies. The focus is on material sectors which are relevant for waste LCAs because of secondary material provision. As each of the studies is based on different assumptions, inconsistencies between them cannot be avoided).

Material	Anticipated Transformation	Reference
Aluminum	<ul style="list-style-type: none"> The smelting of aluminum causes 8 % of global industrial electricity use. The environmental impacts of virgin aluminum production strongly depend on the electricity mix consumed by the smelting process. Hence, the anticipated transformation of energy systems is expected to have considerable effects on the environmental impacts of virgin aluminum. Improvements in smelting technology (reduced energy intensity and the adoption of the inert anode process) are expected to reduce further the environmental footprint of virgin aluminum. 	Pedneault et al. (2021)
Copper	<ul style="list-style-type: none"> Under a business-as-usual scenario, the global final demand for copper is anticipated to increase by a factor of 2.5, from 2015 to 2050, driven by an increase in renewable energy-based power plants and electric vehicles. Energy efficiency improvements, electrification and recycling have the potential to half the climate change impacts in the copper life cycle, but they are not sufficient to comply with climate mitigation targets. Technology options to reduce the carbon intensity of primary copper production include the electrification of high-temperature processes, the use of biobased and synthetic fuels and the implementation of carbon capture, utilisation and storage. 	Watari et al. (2022)
Iron	<ul style="list-style-type: none"> The largest energy user in the iron sector is the blast furnace. A significant proportion of process energy losses can be reduced by implementing heat recovery. The reduction of carbon emissions in the iron industry depends on a small number of key technologies (energy efficiency techniques, fuel switching towards bioenergy, carbon capture and storage, decarbonisation of the electricity supply). Further improvement potential lies in the implementation of the best available technologies for hot-rolling. In the longer term, technological innovations, such as electrowinning or the HISARNA process, may contribute to further reductions in carbon emissions. 	Griffin & Hammond (2019)
Steel	<ul style="list-style-type: none"> By 2050, global steel consumption is expected to increase, with an annual growth rate of 1.5 % to 2,300 Mt / year. In 2019, the steel industry was responsible for around 10 % of annual global CO₂ emissions, 54 % of which were attributed to China. Major economies across the globe have formulated strategies to reduce greenhouse gas emissions in the steel sector. For instance, in China, by 2025, a production capacity equivalent to 530 Mt shall be equipped with ultra-low emission technologies, and the clean transformation of coal-fired boilers shall take place. A scrap-based production route of steel via electric arc furnaces is estimated to result in a reduction in carbon emissions of 85 % compared to the coke-based blast furnace-basic oxygen furnace route, which uses iron ore and coke for the reaction and as a heat source. 	Kermeli et al. (2022) Kang et al. (2022)
Glass	<ul style="list-style-type: none"> Glass production is energy intensive, as the melting of raw materials requires temperatures between 1000 and 1600 °C. The dominant energy source in the glass industry is natural gas. Direct electrification is not always feasible. Hydrogen-fired furnaces are an alternative to reduce climate change impacts of the glass sector, but they are expected to result in trade-offs in the impact categories metal depletion, acidification and particulate matter. 	Westbroek et al. (2021)Wulf et al. (2022)
Paper & Cardboard	<ul style="list-style-type: none"> Greenhouse gas emissions from paper and cardboard production strongly depend on the paper/cardboard type and the feedstock material. Around 75 % of the total greenhouse gas emissions from paper and cardboard production are related to energy consumption. Hence, the anticipated transformation of energy systems is expected to considerably reduce the environmental impacts of this sector. 	Man et al. (2020)
Textiles	<ul style="list-style-type: none"> Using natural pulp instead of bleached pulp can reduce greenhouse gas emissions by up to 15 %. Global clothing consumption increased by factor of 4 in the first two decades of the 21st century. Production growth is driven by the fluid fashion industry. Annual global textile waste quantities are expected to increase to 134 million tons per year by 2030. By 2050, the textile industry will use up to 25 % of the remaining global carbon budget, if no measures are implemented. Already today, synthetic fibres (69 %) dominate the textile industry. In the future, the share of synthetic fibres is expected to increase to up to 98 %. On average, energy consumption and CO₂ emissions from synthetic fibre production are higher than for natural fibres. General climate protection measures (renewable power, energy conservation, carbon pricing) are likely to show stronger effects on greenhouse gas emissions from the textile industry than specific measures in the textile sector itself (enhanced circular economy, bio-fuel replacement, low-carbon agriculture, preference for organic feedstock materials). 	Chen et al. (2021)Peng et al. (2022)
Plastics	<ul style="list-style-type: none"> Global plastic demand is expected to double by 2050, without any policies on consumption. In 2015, plastics were already responsible for 4.5 % of global greenhouse gas emissions. In 2050, they are expected to demand 15 % of the remaining fossil carbon budget when limiting global warming to 1.5 °C. Taking into account all human activities and applying economic downscaling principles, the ecological budget of plastics in 2030 is limited to 1 % of global emissions. To achieve sustainability in the plastic sector, comprehensive actions are needed. According to Stegmann (2022) a circular bioeconomy approach, combining recycling with higher biomass use, could reduce resource consumption in the plastic sector and transform it into a net carbon sink by the end of the century. However, to achieve carbon negativity, a certain set of conditions (implementing a globally uniform carbon-pricing scheme, offering subsidies for biomass use in plastic production and increasing recycling yields) is required, which strongly depends on socio-economic and political transformations (Suh and Bardow, 2022). Bachmann et al. (2023) describe a scenario in which plastics comply with their assigned safe operating space in 2030, based on improving recycling technologies and recycling rates up to at least 75 % in combination with biomass and CO₂ utilisation in plastic production. The closing of material loops as a precondition for a sustainable plastic sector requires considerable effort regarding waste treatment and management (fostering separate collection, optimising the sorting of mixed waste streams and implementing high-quality recycling technologies) and highlights the redefined role of waste management in a post-fossil circular economy uniting waste treatment and material supply. 	Bachmann et al. (2023)Stegmann et al. (2022)Cabernad et al. (2021) Suh and Bardow (2022)

whereas modifications to the background system are often neglected (Bisinella et al., 2021a). Such a simplified approach does not account for the anticipated future and may lead to a misinterpretation of results, in particular regarding the benefits obtained from recyclables, for example as highlighted by Simaitis et al. (2023).

Process-based IAMs (see Section 3) can provide first insights into the potential quantitative effects of anticipated system transformations on the environmental footprint of materials, for example by using the PREMISE tool (Sacchi et al., 2022). In this context, it is important to note that most IAMs have limited resolution in terms of future material recycling and circularity (Keppo et al., 2021; Pauliuk et al., 2017). Database transformations of material systems with PREMISE are limited to the cement and steel sectors and focus mainly on the implementation of carbon capture and storage and efficiency adjustments, whereas the emergence of potential new technologies is not represented. Under the assumption of a limitation of global warming to 1.8–2 °C by 2100, greenhouse gases emitted during the production of steel are expected to decrease by 34–40 % by 2050 (cf. Fig. 3). However, in six to eight of the other impact categories, an increase in impacts is expected. In other material sectors, only indirect effects resulting from the transformation of the energy sector (electricity and fuels), the transport sector, the cement sector and the steel sector are considered by database transformations. Nevertheless, in all material sectors, a reduction in greenhouse gas emissions from material production by up to 70 % is observed. Effects on impacts in other impact categories and potential trade-offs differ between material sectors. Particularly frequently affected by trade-offs are toxicity-related impact categories, ionising radiation, land use and the depletion of mineral resources. The results presented in Fig. 3 may serve as a first approximation of the future environmental impacts of material production, considering that they are linked to high levels of uncertainty because future developments are unknown and assumptions on material system representation in IAMs are simplified.

The considerable changes anticipated in the environmental impacts of material production imply that material system transformations cannot be neglected in future LCAs, especially in waste LCAs, in which material exchanges dominate environmental performances. In this case, materials making a high contribution to environmental impacts shall be identified, whilst future developments in their production – and thus benefits when recycled – must be assessed.

We thus suggest that future waste LCAs should consider:

- The notion that the provision of auxiliary materials used in waste management will result in lower environmental burdens, in particular regarding climate change. Likewise, recyclables will have less substitutional value, because sectors using recyclables will convert to renewable energy and use less fossil-based materials.
- Careful selection and transparent documentation of reasonable substitution ratios, especially when increasing recycling rates (cf. Vadenbo et al. 2017; Viau et al., 2020; Schaubroeck et al. 2021).
- Selection of appropriate datasets in view of the future availability of materials, evolving markets and supply chains and anticipated technological developments. We suggest the following options:
 - o Link an LCA model's background system to the output results of an IAM addressing future developments. An advantage of this approach is that users do not have to make their own informed estimates on future development, and database transformations can be easily reproduced by others. Whether or not the database transformation based on IAMs can sufficiently represent the anticipated changes in the material system under study has to be determined on a case-by-case basis. Currently, such an implementation requires basic programming skills, and generated modified database versions are compatible only with a limited number of LCA tools (Brightway2, Activity Browser and Simapro).
 - o Develop scenarios describing the potential transformation of the material sectors receiving recyclables by using informed estimates based on relevant literature (cf. Table 3 for a first orientation). This

option may result in the most realistic transformation scenarios but is associated with a large effort in terms of literature research, modelling and documentation.

- o Simulate future developments in the form of a scenario-oriented sensitivity analysis and communicate potential ranges of resulting environmental impacts. If the results deviate to any degree in terms of affecting the overall conclusions, further analysis must be introduced.

4.6. Nutrient exchanges

The main nutrients required for plant growth are nitrogen (N), phosphorous (P) and potassium (K). Micronutrients also play a role in soil management for plant growth, and some soils are amended, for example, with copper (Cu) and zinc (Zn) in order to optimise soil fertility. Soil amendments based on waste are seldom low in micronutrients, and it may even be an issue to stay below current guidelines and limitations for microelements (Holm et al., 2010). Nutrients can be recovered and applied to land as stabilised organic waste after anaerobic digestion and/or composting, as well as extracted from ashes.

Two approaches have been used in crediting waste systems for recovered nutrients: considering the nutrients individually as a product that can substitute on a weight basis for commercial fertiliser (e.g., Blengini, 2008) and, in addition to avoiding the production of commercial fertilisers, considering the differences in plant availability and leaching between waste-based fertiliser and commercial fertiliser (e.g., Yoshida et al., 2018). The latter addresses changes in N dynamics in soils, affecting both N₂O released into the air and the leaching of NO₃⁻ into groundwater and surface water. The latter is due to the continuous mineralisation of N after plants have been harvested (e.g., Nielsen et al., 2019).

Nutrient recovery may play a different role in the future for two reasons. First, as impacts related to fossil-based products and energy decrease, nutrient application and substitution impacts may increase relatively. Second, the substitutional values of nutrients may change over time, as detailed below.

Commercial N-fertilisers such as ammonium and urea are currently primarily based on natural gas (Jensen and Hauggaard-Nielsen, 2003; Peoples et al., 2019). In a post-fossil society, other production technologies will have to be in place, and this could involve renewable energy, H₂ and captured CO₂ (Aravindan et al., 2023; Cho et al., 2023). Therefore, from a post-fossil perspective, the climate change load from the production of commercial N-fertiliser may decrease. However, other impacts may increase due to expected growth in the use of electrodes and catalysts in H₂ production and fertiliser synthesis (Gerloff, 2021).

Commercial P-fertilisers are mineral-based and, in most cases, subject to industrial extraction and cleaning (e.g., removing cadmium (Cd)). Natural deposits of P-rich minerals are located in a few countries and are potentially subject to export–import restrictions (Scholz et al., 2013). A potential future shortage of P-fertiliser has thus been debated (Alewell et al., 2020; Neset and Cordell, 2012; Scholz et al., 2013). Moreover, the need for waste-based fertiliser may increase, depending on supply and demand issues.

Commercial K-fertilisers are mineral-based and extracted as chlorides or sulphates from underground deposits, brines or salt lakes. The main deposits are found in Canada (Jasinski, 2021), with no foreseeable limitation in relation to supply.

Soil fertility is also affected by soil carbon content in terms of its importance for soil structure and water-holding capacity. Thus, organic waste products may be able to amend the carbon content of the soil; however, due to the relative scale of this issue, it may only have local importance. The C in peat used in soil manufacturing for horticulture, landscaping and gardening is considered fossil C (stable in water-locked moors) and is readily mineralised and emitted as CO₂ upon use in soil manufacturing. Here, organic waste products could substitute for peat and thus reduce the climate change impact of soil manufacturing

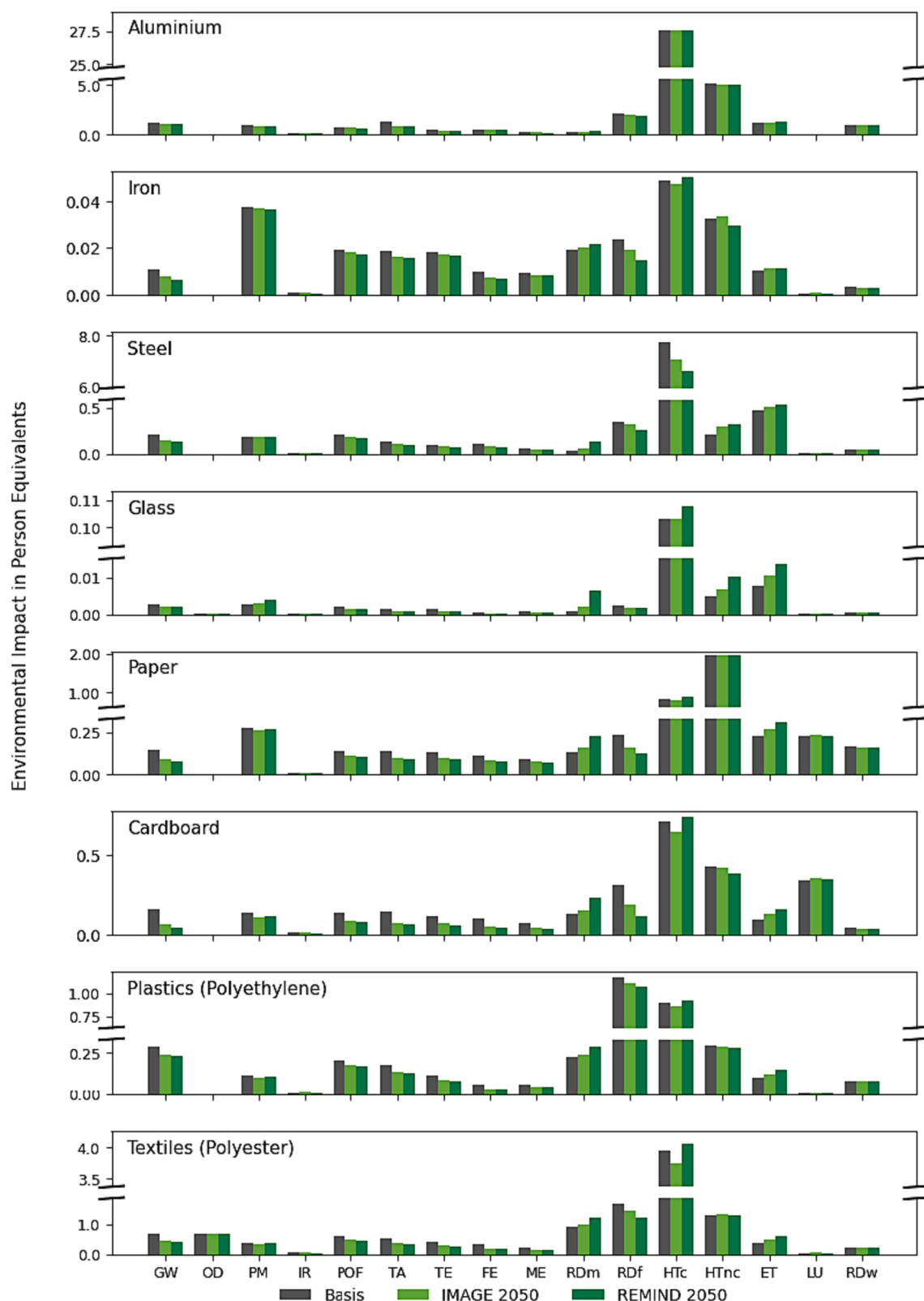


Fig. 3. Estimation of the future environmental impacts of material production based on database transformations with premise (Functional Unit: 1 metric ton of material; System Boundaries: Cradle-to-Market; Databases: The Ecoinvent database (v3.8, cut-off; Basis) was transformed in line with the output results of the integrated assessment models IMAGE and REMIND for the year 2050 in correspondence with a limitation of global warming to 1.8 – 2 °C; Acronyms: GW - Global warming; OD - Ozone depletion; PM - particulate matter; IR - Ionising radiation (human health); POF - Photochemical ozone formation; TA - Terrestrial acidification; TE - Terrestrial eutrophication; FE - Freshwater eutrophication; ME - Marine eutrophication; RDm - Resource depletion (minerals and metals); RDf - Resource depletion (fossil); HTc - Human toxicity (cancer effects); HTnc - human toxicity (non-cancer effects); ET - Ecotoxicity; LU - Land use; RDw - Resource depletion (water)). Underlying methods and data are described in section S2 of the Supplementary Material.

(Boldrin et al., 2010).

From a long-term perspective, it is impossible to predict the availability and demand for fertilisers, as geopolitical issues may play a role (Ahvo et al., 2022; Sandström et al., 2023) – as may various trends in food consumption. The increasing world population and the continued growth of the middle class suggest an increasing demand for food and hence also for fertiliser. On the other hand, an increasing focus on the environmental footprint of food production, particularly meat production, may affect food consumption patterns and, potentially, fertiliser demand (Metson et al., 2012). In addition to the above considerations, developing organic and ecological farming practices may increase the demand for organic fertilisers – and thus the interest in recovering nutrients from waste (Penuelas et al., 2023). We believe, however, that in the post-fossil society, there will also be a significant demand for fertiliser, perhaps even an increasing demand for nutrients from organic, high-quality waste that meet the standards required by alternative farming.

Although nutrients within waste management primarily originate from organic waste, their relative presence in waste-based products may not match the relative demands of a fertilisation scheme. Supplementing with other sources of nutrients may therefore be needed, in order to prepare attractive fertiliser products.

Future LCAs may not change substantially in addressing the issue of nutrients, but we suggest that future waste LCAs should consider:

- Mass-based modelling of the flows of a waste management system, thus keeping track of all fractions containing N, P and K in order to quantify fertiliser value and quality, for instance heavy metal content.
- Base the modelling of waste technologies, such as anaerobic digestion, digestate separation, composting, nutrient extraction and storage, on consistent mass balances, since all nutrients are subject to losses during processing. Some of these losses are also significant emission sources, a classic example of which is ammonia lost into the air during composting (Mo et al., 2023).
- The substitutional values of recovered nutrients are hard to quantify and highly dependent on the actual case:
 - o Private gardeners' use of waste-based fertilisers may provide little credit from what it substitutes. If used instead of commercial fertilisers, the production of the avoided commercial fertiliser would appear as a credit, while changing the nutrient dynamics of soil may be impossible to quantify, because the uses are so manifold and ways of applying unspecified. However, the main concern is that waste-based fertilisers do not foster an equivalent reduction in the use of commercial fertilisers. This aspect was addressed by Andersen et al. (2010).
 - o Professional gardeners and landscapers may use organic waste products in a way that actually provides savings in terms of using fewer peat and commercial fertiliser products. However, the differences between waste-based fertiliser and commercial fertiliser in terms of availability, effects on N dynamics in soil and leaching are also difficult to quantify here. Where waste-based compost substitutes for peat in soil manufacturing, significant savings in climate change impacts can be quantified due to the avoided use of peat (Boldrin et al., 2010).
 - o Agricultural use of waste-based fertiliser must address both avoided commercial fertiliser use and long-term changes in soil dynamics. This is about changes in the release of N₂O from soil and the leaching of nutrients, particularly NO₃⁻; see, for example Nielsen et al. (2019). Considering only the avoided production of commercial fertiliser, however, is insufficient, as the environmental load from changing soil dynamics may easily outbalance the environmental benefits from fertiliser substitution (ten Hoeve et al., 2019; Yoshida et al., 2018).
- The commercial N-fertiliser production inventory may change substantially in a post-fossil society, for example constituting a smaller

climate change load. The inventory for P- and K-fertilisers may change less, except that the contribution from energy use will change.

4.7. Impact assessment

The impact assessment of an LCA – the LCIA phase – involves an LCI being translated into potential impacts on the environment, human health and resources, based on characterisation factors (JRC, 2010; Andreasi Bassi et al., 2023). The characterised results may also be normalised and weighted to help compare and prioritise impact categories.

Many characterisation methods exist. Waste management LCAs often use CML and ReCiPe, although many studies do not disclose this choice, and more than 80 % of reported studies used mid-point characterisation methods (Mulya et al., 2022). Impact methods have varying categories, often assessing between 13 and 16 midpoint impact categories, but not all are necessarily employed, due to relevance or lack of data. Characterization methods are often grouped in method collections, such as the EF 3.1 (Andreasi Bassi et al., 2023). Due to great political attention, climate change impacts are often in focus, but other impact categories should also be considered in order to avoid burden shifting. According to Mulya et al. (2022), climate change impacts are included in more than 95 % of reported studies, while resource-related impacts are included in fewer than 30 %. The scientific robustness behind the impact methods varies substantially. For example, toxicity categories are associated with substantial uncertainty, and they are all recommended but in need of improvements (Fazio et al., 2018; Andreasi Bassi et al., 2023) and are thus often given lower priority during interpretation. Impact assessment methods should be the same across all sectors, although some impact categories may be more relevant than others in specific sectors.

The LCA practitioner may choose to aggregate the midpoint impacts categories into three endpoint categories: protection of human health, the environment and natural resources, in order to aid communication and interpretation (JRC, 2010). Midpoint assessment yields results in many impact categories, which are relatively precise, but also difficult to communicate. The endpoint assessment on the other hand loses some precision, but is more easily interpreted (Yi et al., 2011). According to Mulya et al. (2022) only 15 % of the assessed waste LCA papers cover endpoint categories, so the waste LCA community apparently prefers midpoint assessment.

Impact assessments in waste management should follow the general course of action. However, five issues should require particular focus in an LCIA regarding waste management, as described below.

Climate change accounting – biogenic CO₂: Concerning carbon, the critical issue is how emissions of biogenic CO₂ into the atmosphere are accounted for. Most studies assume that biogenic CO₂ emitted is climate-neutral, while some count biogenic CO₂ as part of the climate impact. Both approaches can be justified; however, the characterisation factors of the other carbon flows must be assigned consistently (Christensen et al., 2009). The IPCC (IPCC, 2021a) does not provide explicit recommendations for characterising biogenic C in waste management. When the emission of biogenic CO₂ into the air is considered neutral, mathematical stringency demands that stored biogenic C is ascribed a negative characterisation factor (-3.66 kg CO₂-eq/kg biogenic C stored). This issue is relevant in landfills receiving organic C, when applying compost to land and when carbon capture and storage (CCS) is considered. In the latter case, CO₂ is geologically stored for hundreds of years, while in the two other cases, it is related to the period within which emissions are considered. In terms of landfilling, a 100-year period is often used as a cut-off approach (Manfredi and Christensen, 2009), because after 100 years, a significant amount of biogenic carbon will still be present in the landfill, which constitutes a saving for climate change. This may correspond to as much as 200–300 kg CO₂-eq. per ton waste landfilled (Zhao et al., 2022). Another issue arises when carbon removal technologies, such as CCS, are modelled. When applying system expansion, negative emissions from carbon removal are modelled identically to

avoided emissions, such as electricity co-generation. However, these are two distinct processes, and only the first results in net negative emissions (Terlouw et al., 2021).

Climate change accounting – methane: Methane emissions are significant in landfills and at anaerobic digestion plants, but they may also appear in composting processes. CH₄ is a potent greenhouse gas, and the characterisation factor has developed significantly over the years due to better knowledge and methods for estimating its climate change impact. Values range from 21 in the IPCC's first assessment report (IPCC, 1990) to 27.9 kg CO₂-eq/kg CH₄ in 2021, according to the recent IPPC report (IPCC, 2021b). First, this suggests that it is essential to state the characterisation factor used explicitly. However, it has also fostered the idea that a separate account of CH₄ emissions and their climate change impacts should be maintained, in order to ease comparison with studies using different characterisation factors.

Groundwater pollution: Although groundwater pollution is a major concern in landfills and where ashes and other waste-based products are used on land, traditional impact assessment methods do not have an impact category that accommodates this issue. This has led in turn to the suggestion of introducing a local resource-related category called “spoiled groundwater”, representing the amount of groundwater that could be rendered unsafe for drinking (Bjørn et al., 2014; Manfredi and Christensen, 2009). The unit would be m³ water, calculated as the amount needed to dilute the contaminants' flow to a drinking water level. However, the proposed new impact category has not gained acceptance, and thus one of the main concerns regarding landfills is that it is still not well represented in impact assessments.

Stored toxicity: Stored toxicity has been suggested as an impact category representing toxic elements, heavy metals and persistent organic chemicals still present in a landfill, a construction material with reuse materials and in soils receiving waste-based products beyond the period for which the emissions have been quantified. For example, in ash disposal sites, leaching for 100 years may release less than 1 % of toxic elements in the ash (e.g., Hyks et al., 2009). This impact category, which conceptually is analogous to stored C, could be used not to dismiss what is present beyond the usual period considered. However, this impact category is still not generally accepted (Hauschild et al., 2008).

Quality of recyclables: While environmental impacts often decrease in line with increasing circularity, recycling may also re-introduce contaminants used in historic products and which meanwhile have been banned in new products due to their hazardous nature. Such legacy contaminants are particularly relevant for long-lived products, which are used for many years before they turn into waste and are potentially recycled. Prominent examples are cadmium or lead in recycled PVC window frames (Borgmann et al., 2019; Ooms and Cuperus, 2013) or brominated flame retardants found in plastic recycling from waste electric and electronic equipment (Turner, 2017). Chemical cycles have also been investigated for paper products, and it has been shown that higher exposure to hazardous substances because of material recycling should be accounted for in the environmental assessment of waste and resource systems (cf. Pivnenko et al., 2016). Nevertheless, to date, product-related exposure has hardly been considered in LCIA, particularly when it comes to secondary production (cf. Fantke et al., 2016). In the future, given increasing levels of circularity, information on relevant substance and material flows in products should be traced and potentially considered in impact assessments, for instance via relatively simple concepts such as the product intake fraction (Jolliet et al., 2015), which provides a clear interface between the life cycle inventory and impact assessment phases to calculate overall exposure to chemicals in consumer products.

Normalisation is done by dividing the results in each impact category with the average contribution made by one person per year from all activities, including housing, food, transportation, etc. Normalisation references are provided, for instance, by Sala et al. (2017) and Andreasi Bassi et al. (2023). After normalisation, all impacts are assigned a person-equivalent unit. Normalisation is often used in reporting, since it

is frequently difficult to relate to different impact category units and it allows for some comparison across impact categories.

Weighting is done by multiplying normalised results with politically determined factors expressing the relative importance of the impact category (weighting factors are proposed, for example, by Sala et al., 2018). Weighting is rarely done in waste LCAs, however, as commonly accepted weighting factors do not exist.

Basic LCIA concepts in waste management will mostly stay the same in the coming years and should always remain equilibrated with impact assessment methods used in other sectors. However, some changes are imminent, as described in the following paragraphs.

The time horizon for climate change characterisation factors: Traditionally, characterisation factors for the main air emissions contributing to climate change (fossil-CO₂, CH₄ and N₂O) are based on a 100-year average. Considering the urgency of many political initiatives to combat climate change, using shorter time horizons may be more appropriate, as it will dramatically increase the impact of emitted CH₄, as shown in Table 4. Please note that using future scenario tools, such as PREMISE, will require adjusting climate change characterisation factors (Simaitis et al., 2023).

New impact categories: As our understanding of environmental impacts increases, additional impact categories may likely be introduced, covering areas such as biodiversity and marine plastic pollution (Dorber et al., 2020; Woods et al., 2021). Furthermore, the characterisation factor relating to known impact categories may be adjusted due to better understanding and knowledge (Rosenbaum et al., 2008).

Normalisation references will change as we move towards a post-fossil society. Moreover, the normalisation factor for climate change will go down dramatically, while factors for some toxicity-related impacts may go up as we increase our consumption of rare elements needed in batteries, catalysts and fuel cells. This constitutes a challenge in comparing normalised results outside a specific study.

LCIAs in waste management are not expected to change conceptually in the coming years. However, we suggest that future waste LCAs should consider:

- Transparent reporting on climate change characterisation factors, particularly biogenic CO₂, stored biogenic C, CH₄ and N₂O.
- A careful determination of the time horizon for characterising CH₄. While a 100-year period may be relevant for comparison with earlier studies using this time frame, it is recommended also to include 20-year horizons. This will bring the LCA results closer to the time horizons used in current climate policies and will improve estimates of the climate change impact and the focus on reducing CH₄ emissions.
- Transparent reporting and justification about normalisation factors used. Climate change impacts should be reported firstly in kg CO₂-equivalents to allow for broader comparison.
- Impact categories in focus, by clearly stating:
 - o Which impact categories have the highest priority in assessments (climate change should always have a high priority) and which ones are given a lower priority.
 - o Which environmental aspects are not well represented by the impact method used in the study, including new impact categories of potential relevance.
 - o Which impact categories are not included in an assessment.

4.8. Interpretation

Interpretation is the phase in an LCA where model results are assessed in view of the choices and data used in other phases in relation to completeness, consistency, sensitivity and attention to the goal and scope of the study. Interpretation is iterative and serves to steer the work toward improving the LCI model to meet the needs derived from the study goal (European Commission, 2010). Once the model is finalised, the outcome of the interpretation phase should be conclusions and recommendations for the goal and scope of the study, together with their

robustness and potential weaknesses in light of any identified study limitations (Hauschild et al., 2017).

Interpretation is necessary and important because LCA models can only approximate real waste management systems, and there is always inherent uncertainty in model choices and data used – and thus in LCA results. Uncertainties stem from various sources: data variability (e.g., natural variability of, for example, waste input composition and uncertain technology parameters), data quality (e.g., representativeness in regard to the goal and scope of the study), epistemic (or scenario) uncertainty (e.g., assumptions relating to context, estimates, lack of knowledge) and model choices (e.g., model configuration, system boundaries) (Bisinella et al., 2016; Clavreul et al., 2012; Groen, 2016; Walker et al., 2003).

While some uncertainties can be addressed qualitatively (e.g., completeness, consistency, limitations), quantitative methods such as sensitivity and uncertainty analyses help assess the effect of data and assumptions on model results. An LCA's standard methodology defines sensitivity and uncertainty analyses and states that an analysis of the sensitivity and uncertainty of LCA results should be used for the interpretation phase, especially for comparative LCAs (Finkbeiner et al., 2006). However, they are still rarely used, as pointed out in the reviews by Laurent et al., (2014a, 2014b) and Bisinella et al., (2021a). The lack of these analyses indicates that the interpretation phase of an LCA may not be as thorough as it should be. LCAs should provide not only quantitative results, but also an analysis of the aspects of the model governing such results, their uncertainty and representativeness of and limitations to goals and scopes as a basis on which to derive robust conclusions. Quantifying uncertainty in an LCA represents a challenge due to the lack of harmonised procedures and guidelines and because there is no “silver bullet” capable of analysing all sources of uncertainty at the same time. For this reason, step-wise approaches or checklists have been suggested. Laurent et al. (2020), for instance, provided a methodological review and detailed guidance for the interpretation phase, with step-wise checks for completeness, consistency, sensitivity and the identification of significant issues and limitations. Step-wise procedures for sensitivity and uncertainty analyses applied to waste LCAs already exist (Bisinella et al., 2016; Clavreul et al., 2012).

Uncertainties, assumptions and limitations arise in all phases of an LCA and are even greater when the practitioner aims to assess a future system, for example modelling LCA technologies and systems that do not yet exist on a full scale. The predicted changes and challenges illustrated in the previous sections (4.1 – 4.7; waste input, technologies, exchanges and impact assessment, as well as the conceptual modelling of waste management systems per se) result in uncertainty in relation to data used, assumptions and modelling choices in all phases of a waste LCA (see Table 1). We recommend that any LCA, and especially future incarnations, include a thorough interpretation phase and sensitivity analysis to identify aspects of the model governing the results. This is especially important with processes, technologies and systems in the early development phase, where the LCA works as a screening tool for potential bottlenecks and helps set data priority needs.

In general, the uncertainty of the model results can be analysed quantitatively, for example by using parameters and distributions as model input values, propagating uncertainty and evaluating results through statistical methods (Groen et al., 2014; Groen, 2016; Heijungs, 2021; Mendoza Beltran et al., 2018). Uncertainty analysis such as Monte Carlo simulation is included in most LCA models, and it can be carried

out by providing distributions to LCA input values. Usually, the bottleneck of such analysis is finding uncertainty data for values in the model. For the foreground system, such distributions can simply be ranges of expected variations, such as using a data quality approach (Henriksen et al., 2021; Laner et al., 2016b) or uniform distributions, representing ranges of expected values (Bisinella et al., 2016). For background data (exchanges), parameters can be represented by distributions, and choices (e.g., of materials) can be analysed following a scenario approach, such as using SSPs (e.g., Mendoza Beltran et al., 2020; Sacchi et al., 2022). Model result uncertainty can also be assessed analytically (Saltelli et al., 2006; Bisinella et al., 2016).

Uncertainty deriving from scenarios and discrete choices can be analysed using scenario analysis (Hellweg et al., 2005; Laner and Mol, 2022; Varling et al., 2023), which is a sensitivity analysis of the assumptions of a model and is the very basis of using future scenarios in an LCA (Bisinella et al., 2021a; Pesonen et al., 2000; Spielmann et al., 2005; Weidema et al., 2004). Future scenarios (also known as “foresight” or “future scenario” analysis) offer a methodological approach that can offer a structured framework for addressing the epistemic uncertainty of LCAs examining future products and systems (Mendoza Beltran et al., 2020). The study by Spielmann et al. (2005) is an early and prominent example of the application of future scenarios in an LCA, in order to address numerous alternatives, uncertain conditions and unpredictable systemic future developments. Börjeson et al. (2006) introduced well-known definitions of scenario types: (i) predictive (probable future, what will happen?), (ii) explorative (possible future, what can happen?) and (iii) normative (preferable future, how can a specific target be reached?). We suggest referring to the systematic review of Bisinella et al., (2021b) on future scenario use in LCAs for definitions, applications and recommendations.

It is very important to keep in mind that uncertainty analysis and scenario analysis answer two different interpretation questions. While uncertainty analysis can help understand how uncertain the results are for data variability, scenario analysis can be carried out to quantify how different discrete assumptions or choices affect results (Bisinella et al., 2021a; Esguerra et al., 2021). The two methods can also be combined, thereby also answering different interpretation questions, depending on how practitioners treat data. For example, when choices are represented as a distribution span, the results of the uncertainty analysis represent how variable the results can be due to data and choices, representing the “span” or “potential range” in which we can expect results. Laner et al., (2016a) used an exploratory scenario analysis approach in combination with a probabilistic approach to identify critical factors and model parameters for the climate impacts of landfill mining. In addition, Blanco et al. (2020) used a probabilistic approach to represent uncertainty emanating from possible emerging technology development pathways, whilst Gregory et al. (2016) proposed using a probabilistic approach to cover the uncertainty engendered by different scenarios. On the other hand, scenario analysis, represented by strictly discrete and different scenarios, allows for maintaining a link between individual choices and scenario outcomes. Examples are provided in the study by Esguerra et al. (2021) and via the approach to discrete exchanges with the background system proposed by, for example, Bisinella et al., (2021b) and Varling et al. (2023).

Once result uncertainty has been estimated, another important question to answer is what can be concluded. This is particularly important if the LCA study aims to support decisions and be used to compare waste management options (a comparative LCA). Step-wise uncertainty statistics methods are employed to study uncertainty around results, which is particularly necessary in cases when distributions of comparative LCA results overlap, as demonstrated, for instance, in Heijungs (2021), Mendoza Beltran et al. (2018) and Prado et al. (2017). Whether comparative or not, interpreting uncertainty analysis results should include the identification of important aspects in the case-specific model influencing the LCA results. Important aspects in this regard are both choices made and data used, and importance is a global

Table 4

Characterisation factors (kg CO₂-eq/kg emission) for main climate change air emissions for different time horizons from Chapter 7 of the IPCC Assessment Report 6 (Forster et al., 2021).

Impact horizon	CO ₂ -fossil	CH ₄ -fossil	CH ₄ -non fossil	N ₂ O
20 years	1	82.5	79.7	273
100 years	1	29.8	27.0	273

concept, defined by how the interaction of sensitivity and uncertainty influences a model (Heijungs, 1996; Saltelli et al., 2006). Global sensitivity analysis (GSA) takes account of both factors (Saltelli et al., 2006). An illustrative example of the relationship between sensitivity and uncertainty in a model is shown for fictitious parameters in Fig. 4. The axes represent the sensitivity and input uncertainty associated with parameters in an LCA model, while the dots in the plot represent the portion of the overall model uncertainty associated with each input parameter. If no input uncertainty is assigned to data in the model, it is not possible to calculate any overall uncertainty but only to calculate sensitivity associated with the used data. If the same input uncertainty is assigned to all data (e.g., 10 % variation around their mean), the output uncertainty associated with each model parameter will mirror the results of the sensitivity analysis. Data used in the model may have high sensitivity but low input uncertainty, or vice versa. For this reason, a systematic quantification of importance should take into account both aspects. Important model aspects are those that, because of the interaction between model choices (sensitivity) and their inherent uncertainty (input uncertainty), make a large contribution to the uncertainty of the overall LCA results.

Important aspects in the LCA model, as well as factors influencing its results, thus cannot be defined a priori and need to be determined contextually (Bisinella, 2017). GSA offers the opportunity to identify important features which can then be discussed in view of predicted changes and developments over the time frame of the study. This is in line with principles relating to future scenarios, which are designed by starting with possible variations of key aspects (Ringland and Schwartz, 1998). The fact that the importance of aspects and parameters should be determined contextually also highlights that what is an important parameter today may not be important in 30 years. Many recommendations in the previous sections have advised using scenario analysis to test the robustness of data and choices, which will likely vary over the time of the study. However, it is important to carry out a follow-up

sensitivity analysis after the scenarios have been implemented, to assess which new factors are contextually important and whether those initially governing the results have changed their sensitivity.

The societal changes envisaged for the next decades call for the increased combination of future societal scenarios and LCAs in order to obtain a more flexible and foresight-oriented interpretation of LCA results. As pointed out by Villares et al. (2017), due to temporal uncertainties, we should not see results as absolute but rather as serving the purpose of identifying potential environmental hotspots, advising on directions for sustainable development and raising questions on environmental features and alternative perspectives. In general, we are aware that the precise estimation of LCA results for future waste systems is most probably not achievable. LCAs on future waste systems are more exploratory in nature and should be interpreted accordingly. A thorough interpretation of LCA modelling provides an accurate picture of the magnitude of potential environmental impacts connected with future system developments and an assessment of the aspects governing results. With the development of and variations in waste input, technology performance, energy systems and material and nutrient exchanges, we will end up with many scenarios and widely distributed results. Moreover, in the future, we will not look for the best solution but rather identify good development paths and exclude solutions that do not look attractive over the whole planning timeline. We should be looking for maximally robust solutions that perform well under a variety of background conditions and across a broad range of model parameter realisations. Therefore, given a number of significant uncertainties and unknowns, not identifying the absolute best performing system configuration but a system configuration which performs well over a wide range of potential futures or scenarios in general, will become a priority.

We thus suggest that future waste LCAs should consider:

- Utilising the interpretation phase iteratively to address uncertainty, assumptions and limitations. Sensitivity and uncertainty analysis

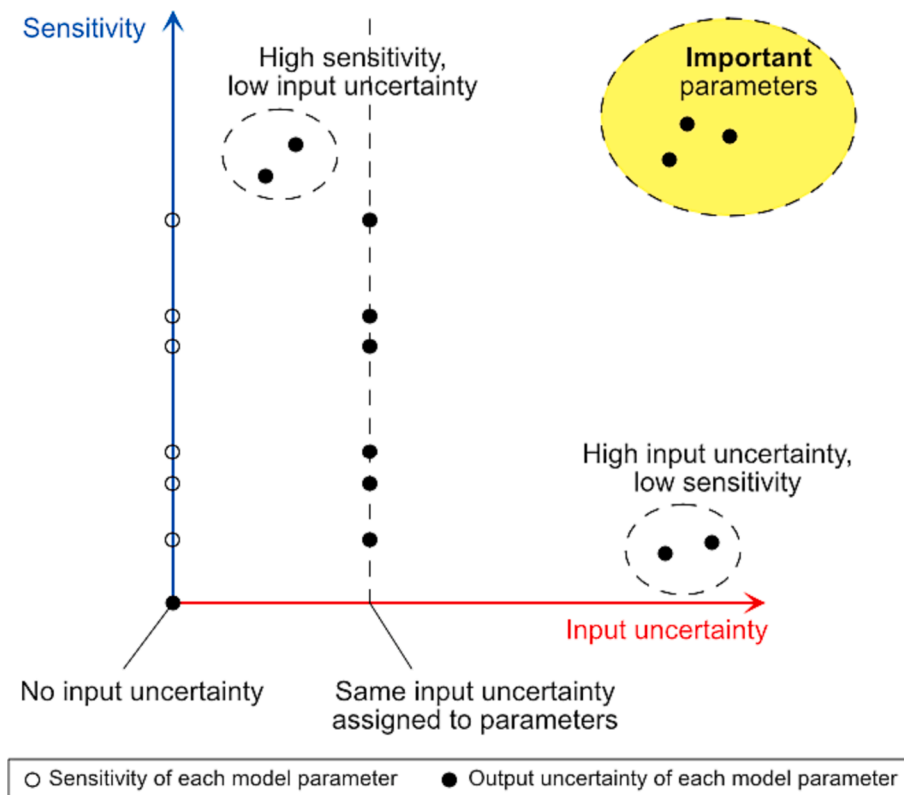


Fig. 4. Illustration of the interaction between sensitivity, input uncertainty and share of model uncertainty (output uncertainty) associated with fictitious LCA model parameters. Important parameters here make a large contribution to the uncertainty of the LCA results. Modified from Bisinella (2017).

should be carried out to identify aspects governing the results. Existing step-wise procedures can guide interpretation, sensitivity and uncertainty analyses.

- Not only identifying important factors governing the results, but also verifying whether they are still important in the identified development routes, as new and important features may arise. Understanding the system at hand equips us for understanding how changes outside a waste management system will or will not affect it. Robust waste management solutions will be the most attractive in this regard.
- Adopting an exploratory approach when modelling future waste management systems with LCAs. Focus not on single-value results but rather on obtaining a good understanding of the system at hand, in order to identify possible development routes.

5. Conclusions

LCA has been used for waste management decision-making processes for over two decades, involving technological investments stretching 30 years or more into the future. So far, most LCAs have only implicitly addressed the long term uncertainty of such choices. In the light of fundamental and transformative changes in our societies and sectors to achieve our sustainability goals, LCA practitioners will have to explicitly address such uncertainties and whether decisions made today will be environmentally sound choices in the long term.

The uncertainty about which and when changes will take place will force us to consider many different developments both within waste management and in the sectors with which it interacts. In particular, specific issues will characterize the modelling of the waste material composition, the physico-chemical relationships between waste material input and treatment technologies, both existing and novel, the recovered resources in terms of materials, energy, and nutrients, and the quantification and interpretation of the potential environmental impacts.

Waste must be thoroughly characterised in order to track materials and contaminants throughout the waste system. In particular, where different source separation schemes are under consideration, careful waste characterisation representing actual waste is mandatory. In addition, the environmental aspects of waste itself will play a relatively more important role in the future, as fossil components of both the energy system and waste products will be phased out.

Waste management modelling must be physically based, meaning that outputs at both the technological and the system level must depend on specific inputs. Otherwise, we cannot quantify the amount and quality of rejects, side streams and recyclables. Knowing the quality of recyclables and assigning appropriate substitution factors is key to assigning the right credit. Physical-based modelling should use distributed parameters instead of single-value parameters, thereby allowing for the proper quantification of uncertainties.

Technologies in a waste management system should be represented in the LCI at a compatible and comparable level. This is a challenge when innovative and emerging technologies are compared to well-established solutions. Technologies at a low TRL tend to have less data and thus may seem to produce fewer emissions contributing to environmental impacts, while well-established technologies may have been, over time, subject to optimisation and thus in some regards be advantageous to technologies with low TRLs. If not adjusted properly, this may lead to unfair comparisons.

Landfilling involving organic waste is difficult to describe accurately at the LCI level because emissions occur over an extended time (>100 years) and LCI data are based on modelling supported by a small amount of experimental data. Although associated with substantial uncertainty, biogenic carbon stored in a landfill for an extended time may be significant and should be consequently accounted for as a climate change saving.

As the energy system becomes increasingly de-fossilised, the climate

change burden and the benefits of energy exchanges in waste management will diminish. This is probably the single-most important future change affecting waste LCAs from a climate change perspective. We recommend that a range of energy exchanges are considered and that the assumed transition to the post-fossil society is made explicit.

The fossil aspect of materials will decrease as the fossil component of energy used in production will decrease and as fossil-based materials will be fewer. This will decrease the climate change benefits of recycling, while other impacts are likely to change less. Approaches to address this are emerging, but care must be taken when assigning credits for recyclables during the transition to a post-fossil society.

Nutrients may have more focus in the future, but assigning credits to nutrient recycling, such as compost and digestate used on soil, requires that products are used correctly. This means that they do substitute for other soil amendment products and fertilisers. Additional focus should also be on long-term impacts on agricultural soils.

Among environmental impacts, climate change impact may have the highest priority in the decades to come, due to its political focus. The time horizon for the impact of greenhouse gases in the future should be 20 years, in addition to the 100 years usually used, in order to accord with the urgency of climate change issues. Normalisation should be avoided for climate change, as the normalisation reference is expected to decrease during the transition to a post-fossil society.

Modelling a large number of development scenarios in LCA of waste management systems, aligned with a proper representation of potential changes in society allows us to prepare for the future. These novel LCAs of waste management systems will generate many results, which can be assessed from an exploratory perspective and converted into quantitative knowledge useful in the decision-making process. The credibility of transforming complex model results into technical options demands that the modelling is transparent, stringent and documented, the liability of the impact assessment method should be made clear and the robustness of the results is explained. Assessing the robustness of the results in terms of uncertainties, and identifying governing parameters, should be done systematically to operationalise the results into decision options. It is important to understand that what governs the results today may not be the same in a post-fossil society. Utilizing this explorative approach, we will move from identifying the single best environmental solution to identifying robust and environmentally sound development pathways while isolating less suitable solutions of value only within a narrow range of development scenarios or only valid over a certain amount of time. Attractive solutions will be environmentally sound over the time horizon and allow us to adjust our waste management systems as conditions change.

The suggestions we have made herein will hopefully inspire good practice and contribute to environmentally sound and robust decision support for the design of waste management systems on the way to a post-fossil society.

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.

Data availability

No data was used for the research described in the article.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.wasman.2023.11.021>.

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