Swedish Environmental Research Institute

Comparing high-quality recycling and downcycling of plastics

Calculating carbon footprints using a basket of functions approach

Report number: B2498
In cooperation with: Svensk Plaståtervinning (SPÅ), Tomas Ekvall Research, Review & Assessment (TERRA)
Authors: Tomas Ekvall (TERRA), Linnea Granström (SPÅ), Rickard Jansson (SPÅ), Emma Moberg (IVL), Tomas Rydberg (IVL)

Funded by: SIVL and SPÅ

ISBN: 978-91-7883-653-6

Keywords: waste management, Extended Producer Responsibility, recycling, downcycling, carbon footprint, plastics, allocation, system expansion, basket-of-functions

Summary

This report was written within the project *Differentiated recycling criteria for plastics for a more sustainable recycling of plastics in Sweden (Differentierade plaståtervinningskriterier för en mer hållbar svensk plaståtervinning).* It is based on the observation that the Swedish Extended Producer Responsibility stipulates the share of plastic packaging waste that should be recycled but does not address the quality of the recycling. The existing recycling targets are most easily met by grinding mixed waste plastics together and putting a low-quality product on the market. This low-quality material can replace, for example, wood in construction products. To replace primary polymers, waste plastics need to be sorted into separate polymer fractions before recycling. This entails additional costs. However, such high-quality recycling can reduce both primary-plastics production and incineration of plastic waste. The choice between downcycling and high-quality recycling also affects the waste treatment of subsequent products and has indirect effects on the broader waste-management and energy systems.

The project aimed at contributing to the understanding of the climate aspects, in a broad systems perspective, of downcycling and high-quality recycling of packaging plastics from Swedish households. We did this by identifying a method for modelling recycling that accounts for important systems impacts, and by applying this method in a comparison between no recycling, downcycling, and advanced sorting that allows for substantial high-quality recycling.

Our assessment model accounts for the impacts of downcycling and high-quality recycling on the production of primary materials, on Swedish waste incineration, on waste imports, and on the waste management and energy systems in the rest of Europe. The system investigated was expanded into a basket of functions: the function provided by 1 tonne of primary-plastics packaging, 5 railway sleepers, the treatment of 1.02 tonnes of near-term and 0.50 tonnes of future European waste, the production and use of 5.1 GJ of near-term and 1.3 GJ of future gas, and the supply of 238 kWh of electricity. The assessment was comprehensive but includes substantial uncertainties.

Our results are rough estimates of actual systems effects. They indicate that downcycling is better for the climate than incineration of the packaging plastics, mainly because the incineration is postponed until after the end of the second use of the downcycled material. This allows for a greater near-term import of waste

that, in turn, reduces landfill disposal of mixed waste in other countries. The case with advanced sorting has the lowest climate impact, because it involves the least primary production of polymers and the least total incineration of waste plastics.

A plastics-recycling policy should provide incentives for high-quality recycling as this facilitates the substitution of primary plastics and a reduced incineration of plastic waste. The basket-of-functions approach allows for assessing the complex system impacts of recycling. The results include no avoided burdens, but only the emissions associated with generating the many functions of the system. The approach can be used to quantify the system impact of changing a single flow.

Table of contents

1 Introduction

1.1 Background

Material recycling reduces the need for primary production of materials and for waste-treatment methods such as incineration or landfill. This typically results in an environmental benefit (Björklund & Finnveden 2005; Villanueva & Wenzel 2007). Recent studies indicate that this also holds true for the recycling of plastics (Civancik-Uslu et al. 2021; Ekvall et al. 2021; Hermanns et al. 2023). To increase the recycling rate, Extended Producer Responsibility (EPR) systems have been introduced by policy makers in most developed countries and many emerging economies (OECD 2014; Gupt & Sahay 2015).

However, the environmental benefit of plastics recycling depends heavily on what material the recycled plastic substitutes (Björklund & Finnveden 2005). For example, Ekvall et al. (2021) and Hermanns et al. (2023) conclude that the environmental benefit is lower for chemical recycling compared to mechanical recycling, as mechanical recycling produces a recyclate that can substitute primary polymers, while chemical recycling displaces the feedstock for polymer production. In addition, chemical recycling requires more energy (Lange et al. 2024). On the other hand, polymers produced from chemically recycled plastics have the same properties as primary plastics.

To replace primary polymers through mechanical recycling, mixed plastic waste needs to be sorted into separate polymer fractions. We refer to the recycling of separate polymers as high-quality recycling, based on the work of Caro et al. (2023, p. 73). The polymer can be recycled multiple times, although part of the material collected for recycling is lost as residues in the sorting process (Antonopoulos et al. 2021). The quality of the polymer is degraded when the material is recycled (Oblak et al. 2015; Schweighuber et al. 2021). This is in part because the recycled polymer includes a mix of grades with varying molecular length and branching (Demets et al. 2021). Hence, a recycled polymer might not be used in all applications where the primary polymer is used (Raghuram et al. 2023) or may not replace the primary polymers to a 1:1 ratio (e.g., Demets et al. 2021).

If mixed plastics are mechanically recycled without prior sorting into separate polymers, even less of the primary quality is preserved in the recycled material (Ragaert et al. 2017; Karaagac et al. 2021). This low-quality secondary material is not suitable as a replacement for primary plastic in most plastic applications, and will often be used in other applications as a substitute for wood (Mølgaard 1995; Dias & de Alvarez 2017), for example in planks, pallets, railways sleepers. It can also be mixed with wood to produce a composite material (Najafi 2013) or used as aggregates in concrete (Jubinville et al. 2020). These mixed materials are difficult to recycle mechanically again after use. This can be regarded as an extension of the linear economy rather than as part of a circular economy. From now on, we use the term downcycling for the grinding and agglomeration of mixed plastic waste into a low-quality recycled material.

When downcycled plastics substitute wood or concrete aggregates in subsequent products, energy recovery and emissions from the waste management of these products can be strongly affected. This, in turn, can have a significant impact on other parts of the energy and waste-management systems (Hagberg et al. 2017). To take all these aspects into account in an environmental comparison between high-quality recycling and downcycling, the assessment must have a systems perspective broad enough to include the production of a variety of substituted materials, the waste management of the products where the recycled material is used, and other waste-management and energy-supply processes affected by this waste management.

Swedish Plastic Recycling recently started Site Zero, a plant for the advanced sorting of mixed plastic packaging waste to facilitate high-quality recycling. However, Swedish recycling requirements do not distinguish between high-quality recycling and downcycling. The Swedish EPR specifies that 90% of PET bottles and 50% of other plastic packaging should be recycled (Ministry of Environment 2018; Ministry of Climate and Economy 2022), i.e., should enter pelletizing, extrusion, or molding operations, or be used in a final product (Eurostat 2023b, p.26). PET bottles have a separate target, because they are collected for recycling through a separate deposit system. From the year 2030, the target for recycling of plastic packaging other than PET bottles will increase to 55% (Ministry of Climate and Economy 2022). These recycling targets can be met through either recycling route, which favours the less costly process of grinding mixed waste plastics together and putting a low-quality product on the market. Hence, the Swedish EPR creates an incentive for downcycling over the high-quality recycling of plastics. This is also the most common route for consumer plastic recycling (Schwarz et al. 2021a).

Because of the incentives for downcycling given by the current recycling targets, we see a need for assessing and highlighting the climate impacts and potential benefits of advanced sorting and high-quality recycling. Such an assessment needs to have a broad systems perspective to account for potentially important systems impacts. Life cycle assessment (LCA) and carbon footprint (CF) calculations are well established tools for environmental assessment with a wide systems perspective. An LCA is a compilation and evaluation of the inputs, outputs, and potential environmental impacts of a product from cradle to grave, i.e., from raw material acquisition or generation from natural resources to final disposal (ISO 2006). CF calculations are based on LCA but are limited to the impact of climate change. Neither LCA nor CF are uniform methods; instead, they are families of methods that can generate widely diverging results. Specifically, more than a dozen different approaches have been proposed for how to model recycling in LCA (Rydberg 1995; Ekvall & Tillman 1997; Allacker et al. 2017; Ekvall et al. 2020).

Many LCAs and CFs of plastics recycling have been published previously. Early studies were presented by Mølgaard (1995) and Rydberg (1995). Davidsson et al. (2021) and Kousemaker et al. (2021) reviewed a total of 28 later studies. Additional recent studies have been presented by Schwarz et al. (2021a), Andreasi Bassi et al. (2021 & 2022), and Hermansson et al. (2022). Plastics recycling is also included in assessments of recycling of multiple materials. For example, the LCA by Tallentire & Steubing (2020) covers the recycling of several packaging materials: paper, plastics, metals, glass, and composite materials. However, few studies compare high-quality recycling to the downcycling of mixed plastics. The LCA by Tallentire & Steubing (2020) includes both the closed and open-loop recycling of plastics, because of the saturation on the market for recycled plastics in packaging production, but do not distinguish between recycled plastics of different qualities. Mølgaard (1995), on the other hand, compares recycling of plastics with and without sorting. Recycled unsorted plastics is assumed to substitute timber in this study. Schwarz et al. (2021a) compare mechanical recycling and various forms of chemical recycling of plastics to downcycling and energy recovery. They acknowledge that downcycled plastics cannot substitute the same primary materials. However, neither Mølgaard (1995) nor Schwarz et al. (2021a) account for impacts on the waste management of subsequent products. Hence, we see a need for a comparison between the climate impact of high-quality recycling and downcycling with an even broader systems perspective.

1.2 Aim and scope

This report was written within the project *Differentiated recycling criteria for plastics for a more sustainable recycling of plastics in Sweden (Differentierade plaståtervinningskriterier för en mer hållbar svensk plaståtervinning).*

The report aims to shed light on the climate impact and potential benefit of high-quality plastics recycling from a systems perspective, accounting for all key factors discussed in Section 1.1. We explain and justify our broad systems approach and apply it to the carbon-footprint calculations of three cases for the management of plastic packaging waste (excluding PET bottles) from Swedish households:

1. No recycling. All packaging waste is incinerated with energy recovery.
2. Downcycling. The current Swedish 50% recycling target is met through the downcycling of mixed plastic packaging waste to railway sleepers, which substitute wood sleepers.
3. Advanced sorting. The Swedish 50% recycling target is met through the advanced sorting of the mixed plastic packaging waste into separate polymer fractions, corresponding to the performance of the new sorting plant of Swedish Plastics Recycling. This sorting enables high-quality mechanical recycling of much of the material, while a smaller share is downcycled for plastic sleepers.

1.3 Limitations of study

Swedish Plastics Recycling provides detailed and specific input data related to the collection and sorting of plastic packaging waste from Swedish households. For other parts of the system, the assessment relies on secondary data from literature and databases. These are in part assumptions of future impacts based on expert knowledge of, for example, the development of the European waste-management and energy system. Much of the data is not available as inventory data but only as impact assessment results expressed as global warming potential with a 100-year perspective (GWP-100). This means that the results of the carbon-footprint calculations are rough estimates only. We account for this uncertainty in the discussion of the results and the conclusions, which do not focus on numerical results but on the broad systems impacts. This also means that we do not aim to adhere to the international standard for carbon footprints (ISO 2018). The assessment is a carbon-footprint study in the sense that it applies LCA methodology but is limited to greenhouse-gas emissions only.

Our assessment focusses on mechanical recycling after varying degrees of sorting. The complementary role of chemical recycling is discussed towards the end of the report.

All calculations are made in a Swedish context. The primary target audience for the results includes policymakers and other actors involved in plastics recycling in Sweden. However, the qualitative conclusions on the comparison between high-quality recycling and downcycling should also be valid in other countries, as discussed at the end of the report. The target group for the methodological findings in the report includes LCA researchers, practitioners, and commissioners around the world.

2 Approaches to modelling recycling in LCA

This chapter systematically explains the need for a broader systems perspective than normally applied in LCA and CF calculations (Sections 2.1-2.5) and presents the approach we use (Section 2.6). In contrast to, for example, Ekvall & Tillman (1997), Allacker et al. (2017) and Ekvall et al. (2020), we do not aim for a comprehensive review of approaches to model recycling. Instead, we expand the method and the related system boundaries directly from the two approaches that Frischknecht (2010) identified as the most common for modelling recycling in LCA: the cut-off and avoided-burden approaches.

2.1 The cut-off approach

The cut-off approach – also known as the recycled content approach or the 100/0 method – is one of the most common approaches to modelling recycling (Frischknecht 2010). Each product is assigned the environmental burdens of the processes in the cradle-to-grave system of that product (cf. Figure 1a). The challenge is to define where the grave of one product's life cycle ends and the cradle of the subsequent life cycle begins; should this boundary be before, within, or after the recycling process?

Guinée et al. (2004) recommend that the boundary between the two life cycles should be defined as the point where the waste with a negative economic value turns into a material or resource with a positive economic value. This often occurs within a process, which it gains revenues from treating waste with a negative economic value but also revenues from selling the products of the process. For such processes, Guinée et al. (2004) recommend that the environmental burdens are allocated to its functions in proportion to their share of the total revenues.

The cut-off approach is recommended in the guidelines for LCA-based Environmental Product Declarations (EPD International 2021). The British Standard for carbon footprint (PAS 2050; BSI 2011, p.31) recommends the method when recycling reduces the quality of the material. The Greenhouse Gas Protocol also recommends cut-off in some cases (WRI & WBCSD 2011, p.74). However, by excluding processes beyond the product life cycle investigated, the cut-off approach fails to account for the most important reason to send materials to recycling: reducing the need for primary material production.

2.2 Simple avoided-burden approach

The avoided-burden approach – known under various names, including 0/100 method, the end-of-life approach (Ekvall et al. 2020), etc. – is also common and has a broader systems perspective. When the investigated product life cycle generates material that is recycled and used in new products, this approach gives the product investigated a credit for the primary production avoided in other product systems. To avoid double-counting of the environmental benefits of recycling, the avoided-burden approach assigns the corresponding environmental burdens to the use of recycled material.

The avoided-burden approach is recommended by the international standard for LCA as a way to avoid allocation by expanding the system boundary (ISO 2020, Annex D.2). The approach is stipulated by the WorldSteel Association (2017) and the International Stainless Steel Forum (Fuji et al. 2005). The international CF standard (ISO 2018), the British Standard for carbon footprint (PAS 2050; BSI 2011, p. 31), and the Greenhouse Gas Protocol (WRI & WBCSD 2011, p. 74) also recommend the avoided-burden approach in some cases.

The avoided-burden approach exists in several versions. In its simplest form (e.g., ISO 2018, Annex D.3), the recycling is modelled as a closed loop into the same product. This reflects an assumption that each tonne of recycled material substitutes 1 tonne of primary material of the same type produced in the same way as the original material (cf. Figure 1b). Just like the Swedish EPR, this approach does not distinguish between high-quality recycling and downcycling but only accounts for the volume of material recycled. Hence, the simple avoided-burden approach is not adequate when comparing high-quality recycling to downcycling.

2.3 Quality-adjusted avoided-burden approach

The quality-adjusted avoided-burden approach is slightly more advanced than the simple avoided-burden approach because it takes quality losses into account (Figure 1c). Werner & Richter (2000), for example, suggest that the credit given for avoided primary production should be multiplied by a quality factor based on the economic value of the recycled material relative to the primary material. Schwarz et al. (2021b) simply assign a quality factor of 0.5 to downcycled plastics. The resulting LCA model reflects the case when 1 tonne of downcycled plastics replaces 0.5 tonnes of primary plastics.

Other researchers have presented frameworks for quantifying the quality of plastics and other materials based on physical properties (Vadenbo et al. 2017; Demets et al. 2021; Golkaram et al. 2022; Roosen et al. 2023; Schulte et al. 2023). This is challenging because the quality of a material has many aspects. The requirements on a recycled material also depend on where it is used and can vary between actors.

The quality-adjusted avoided-burden approach is accurate when the downcycling of plastics means it substitutes a smaller quantity of primary plastics. However, it is inaccurate when downcycled plastics substitutes another kind of material, such as wood.

2.4 Avoided-burden approach with foreseeable substitution

The avoided-burden approach can also be designed to account for primary production that is actually foreseen to be substituted with recycled material, for example when downcycled plastics substitute primary wood (Figure 1d). This is the substitution approach used by Mølgaard (1995). It is also how the current international standard on LCA explains system expansion (ISO 2020, Annex D.2). The foreseeable substitution is also accounted for in the methodology for Product Environmental Footprints developed within the European Union (EU; European Commission 2021, Annex 1, pp. 49-50). However, neither of these documents account for consequences beyond this substitution. For example, they do not account for the impact on the recyclability and waste management of the subsequent product. This is a significant limitation when downcycled plastics substitute wood or concrete that have completely different impacts for waste management.

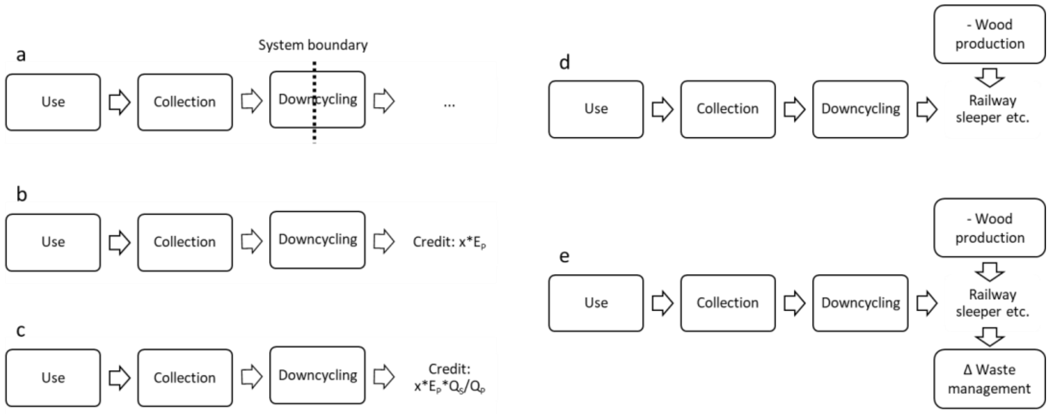

Figure 1: *Five approaches to modelling recycling in LCA illustrated in the case where x tonnes of plastic packaging is downcycled to replace wood: a) a cut-off approach; b) the simple avoided-burden approach (E_P is the environmental impact of producing 1 tonne of primary plastics); c) a quality-adjusted avoided-burden approach (Q_S and Q_P are the quality of the recycled and primary material, respectively); d) the avoided-burden approach with the foreseeable substitution; and e) system expansion that accounts for the impact on the waste management of the subsequent product (Δ Waste management).*

2.5 System expansion to include downstream impacts

The substitution approach can be expanded beyond just the substituted material production to also account for impacts on the emissions of subsequent products. Ekvall & Finnveden (2001, pp. 206-207) recommend such expansion when there are important differences in the environmental impacts of the use and/or waste management between generated and substituted products. If applied in our assessment of downcycling, the system would include not only avoided primary wood production and the difference in emissions from the waste incineration of plastic sleepers compared to wooden sleepers, as indicated by Figure 1e. It would also account for the higher energy content of plastic sleepers, which reduces the imports of waste from other countries, the induced increase in treatment of such waste in other countries, and the substitution of electricity, heat, and gas with electricity, heat, and gas generated in this waste treatment (cf. Figure 2).

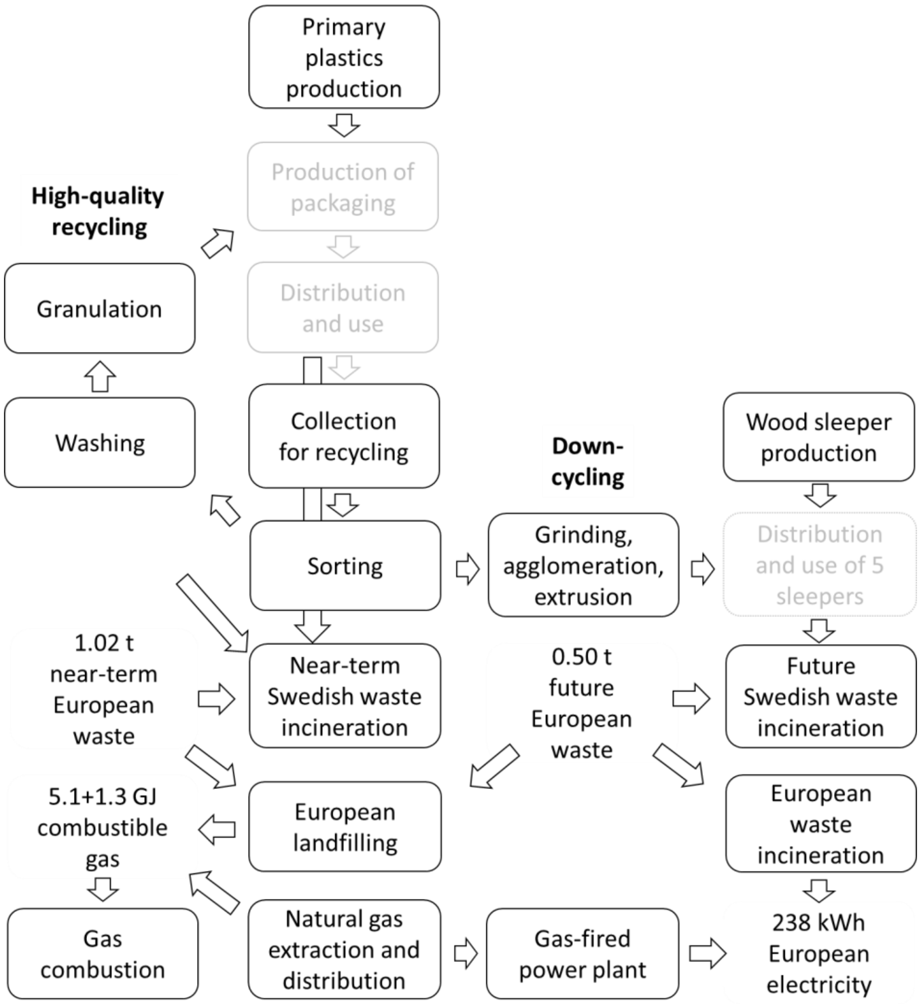

Figure 2: *Schematic illustration of the system affected by the recycling rate and the choice between advanced sorting and the downcycling of Swedish plastic packaging waste. The numbers indicate the quantities affected by the waste management. Light grey text describes parts of the system that we assume to be unaffected and exclude from the comparative assessment.*

The expansion to include affected downstream processes makes the assessment more comprehensive. However, it also introduces large uncertainty because it requires assumptions on the waste management of subsequent products and on the consequences of this waste management on energy and material-supply systems (cf. Section 3.2). The final results from this approach will be a rough net total, accounting for a variety of increases and reductions in emissions in different parts of the expanded system. This multitude of burdens and credits can be difficult to communicate.

2.6 The basket-of-functions approach

An alternative interpretation of system expansion, also suggested by ISO (2012, Section 6.4), is to expand the assessment into a multifunctional system that generates all functions provided by either of the options compared. Expansion of the assessment to a multifunctional system has been part of LCA methodology for a long time (Tillman et al. 1994). Such a study does not include any substitution and related credits, but only the foreseeable emissions associated with the generation of the basket of functions (Majeau-Bettez et al. 2014). This, we believe, makes the results easier to communicate. The assessment will be as comprehensive as the approach discussed in the previous section, i.e., more comprehensive than with the other approaches discussed in this report. For these reasons, we chose to apply the basket-of-functions approach in our study.

The starting point of our calculation is the function provided by 1 tonne of packaging produced from primary plastics. Alongside this initial functional unit, our assessment has a multi-dimensional supplementary functional unit, reflecting the volumes of each function that is at stake depending on if and how the plastic packaging is recycled (cf. Figure 2).

3 Methodology

This case study aims to illustrate how the climate impact of material production, waste management and energy supply depend on if and how Swedish household packaging plastics (excluding PET bottles) are mechanically recycled. It also serves the purpose of illustrating the basket-of-functions approach discussed in the previous chapter. The results from the study will be a rough estimate of the potential climate impacts of different ways of generating the basket of functions – in our case the provision of plastic packaging and railway sleepers, the treatment of waste that could be imported to Sweden, and the generation of heat, electricity, and combustible gas (Figure 2).

3.1 Plastic flows in the cases investigated

Results from hand sorting at Swedish Plastic Recycling are consistent with the polymer mix in European packaging production (Plastics Europe 2022, p.37) and indicate that Swedish plastic packaging collected for recycling is a mix containing:

- 30% low-density polyethylene (LDPE)
- 25% polypropylene
- 20% high-density polyethylene
- 12.5% polyethylene terephthalate (PET)
- 10% polypropylene film (in this study modelled as LDPE)
- 2.5% polystyrene and expanded polystyrene (modelled as PET)

As stated in Section 1.2, we assessed three management cases for the waste plastics: 50% recycling after advanced sorting, 50% downcycling, and no recycling.

Reaching the current 50% recycling target and the future 55% target with advanced recycling requires a high collection rate, as well as a high yield in the sorting and recycling processes. A recycling rate of 55% can be obtained with an 82% collection rate, 82% sorting yield, and 82% yield at washing and recycling. We estimate, based on test runs, that the sorting yield at Site Zero can reach an 82% sorting yield based on the current composition of collected mixed plastic packaging. The advanced sorting process at Site Zero is fully automatic and incorporates 60 near-infrared readers, where the sensors are configured depending on the specific sorting task. To enhance sorting quality, the plant also uses lasers, deep learning camera systems, and electromagnetic sensors. In addition, it is equipped with screening drums, ballistic separators, and digital process monitoring. The sorting retains much of the quality of the plastics through the separation of the mixed plastic packaging waste into the ten most common polymers with little contamination from non-targeted polymers and other waste. Multilayer packaging, consisting of more than one polymer, is recovered from the waste stream and sorted in two fractions of mixed polyolefins. This enables a high sorting efficiency coupled with sorted fractions of high purity, which allows for high-quality recycling of much of the collected material. In our model, 80% of the output from advanced sorting is recycled at a high quality.

For comparison with our 82% estimate, Tallentire & Steubing (2020) assume, in their best-practice scenario, 71% sorting yield for the common plastics high- and low-density polyethylene (HDPE and LDPE), polypropylene (PP), and polystyrene (PS). After a survey of existing sorting plants and a literature study, Antonopoulos et al. (2021) present a material flow analysis where the future sorting yield varies between 65% and 91% for different plastics fractions. Note that the sorting yield based on the total amount of collected mixed plastic packaging will be lower because this mix will contain plastic packaging (e.g., multilayer packaging) that is not sorted at all.

Our model assumes that the yield also reaches 82% in the washing and recycling of the sorted material. This is consistent with Tallentire & Steubing (2020), who assume 81-87% recycling yield for the common plastics in the best-case scenario. Antonopoulos et al. (2021) set the future recycling yield at 71-93%, depending on the plastics fraction.

With 82% sorting yield and 82% recycling yield, the future 55% recycling target requires a collection rate of 82% (because 0.82*0.82*0.82=0.55). This seems difficult to reach through source separation: Tallentire & Steubing (2020) state that the collection rate through source separation is 69% in the best existing practice. However, additional plastics can be collected for mechanical recycling through the sorting of mixed residual waste (Höglund 2024). To reach the current 50% recycling target, the collection rate needs to be 71.2% (because 0.712*0.82*0.82=0.50). This is the collection rate we assume in our calculations.

The resulting high-quality recyclate, which is equivalent to 38.3% of the plastic packaging volume put on the market, can substitute primary plastics. In our model, it is recycled in a closed loop as input material in packaging production. The material degradation in this loop is assumed to be minor, because the recycled material is mixed with 61.7% primary material. The average number of times the plastics is used in the closed loop is just 1/(1-0.383) = 1.6.

However, the degradation that occurs in mechanical recycling still means that the substitution rate is less than 1:1 (Demets et al. 2021). We assume that each kg of high-quality recycled plastics substitutes 0.9 kg primary plastics. This is consistent with the default quality ratio of secondary to primary plastics in Product Environmental Footprints (European Commission 2020). As a result, the packaging and the packaging waste is 4% heavier when it contains 38.3% recycled material.

The plastic flows in a system with advanced sorting are calculated based on these numbers: 4% greater weight, 71.2% collection, 82% yield in sorting and in recycling, and 80% high-quality recyclate. The results are presented in Table 1.

Material losses also occur at downcycling. We have unpublished data indicating a yield range between 50% to 100%. We assume a yield of 82%, which is the same as the sorting yield in the case of advanced sorting. The material losses affect the collection rate required to reach 50% recycling through downcycling. With a yield of 82%, the required collection rate is 61% (because 0.61*0.82=0.50; cf. Table 1).

In our model, downcycled material substitutes wood in sleepers for low-speed railways within the country. These have a service life of several decades and are incinerated with energy recovery after use. This means that the incineration of the downcycled waste plastics is postponed by several decades compared to when no recycling occurs. The same holds if the downcycled plastics substitute wood in other products with a long service life, for example pallets and planks.

Table 1: *The plastic waste flows in the three cases investigated for the waste management of plastic packaging providing the functionality of a tonne of packaging produced from primary plastics. The numbers in the table are rounded to kg. More details are available in the Supplementary material, which can be shared upon request.*

Case	No recycling	Downcycling	Advanced sorting
Packaging waste (kg)	1000	1000	1040
Collection (kg)	0	610	741
Losses at sorting (kg)	0	110	133
Downcycling (kg)	0	500	121
Losses at washing & recycling (kg)	0	0	87
High-quality recycling (kg)	0	0	398
Total recycling (kg)	0	500	520
Incineration with energy recovery (kg)	1000	500	520

Plastic waste that is not recycled is incinerated with energy recovery, since landfilling of combustible waste is prohibited in Sweden (Ministry of Climate and Economy 2001). This includes both plastic waste that is not collected for recycling and residues from sorting and recycling.

3.2 Affected system

As stated in Section 1.1 and illustrated in Figure 2, the choice between high-quality recycling and downcycling has foreseeable impacts far beyond the recycling process. If recycled plastics of a high quality replace primary material in the production of plastic packaging, this reduces the primary production in the packaging life cycle. The production, distribution and use of the packaging are excluded from the study. The climate impact of these activities is affected by high-quality recycling, but by less than the 4% that the weight increases in the case with advanced sorting. Hence, we consider the effect to be negligible.

Downcycled plastics that replace wood as raw material, for example in sleepers, affect the primary production of timber and its associated land-use change. We assume that the distribution and use of the wooden and plastic sleepers have a negligible or similar climate impact and exclude these activities from the assessment.

The recycling rate affects waste incineration with energy recovery from waste plastics in Sweden. Incinerators have high investment costs but generate energy at a low or even negative net running cost, because they charge a gate fee for accepting and treating the combustible waste (Knutsson et al. 2006). For this reason, waste incinerators tend to be used at full capacity. Hence, a change in the flow of plastic packaging to Swedish incinerators will affect the quantity of other waste flows, which will need to be treated elsewhere. This induced waste treatment will not include the disposal of waste generated in Sweden at landfills, because a national ban on landfilling of combustible waste prohibits such landfilling (Ministry of Climate and Economy 2001). Instead, a change in the incineration of Swedish plastic waste is likely to affect the import of waste for incineration. The current incinerator capacity in Sweden is greater than the quantity of domestic residual waste, and 2 of the 7 Mtonnes of waste incinerated in 2021 originated in other European countries (Waste Sweden 2022, pp. 30-32). Waste for incineration is imported from, for example, Norway, the UK, and Ireland (Fråne et al. 2016).

A change in the waste imports will, in turn, have a complex and uncertain impact on the waste treatment in other European countries. Hagberg et al. (2017) suggest managing this uncertainty through scenario analysis. Ekvall et al. (2021) applied two scenarios based on data from Hagberg et al. (2017), where the waste export to Sweden affects landfill disposal and waste incineration, respectively, in other European countries. In the landfill scenario, combustible mixed waste that is not exported to Sweden is disposed of in a modern, well-designed landfill, and the extracted landfill gas is assumed to compete with natural gas. In the incineration scenario from Ekvall et al. (2021), combustible mixed waste that is not exported to Sweden is instead incinerated for electricity production in another European country. The electricity competes in this scenario with electricity produced from natural gas in modern combined cycle power plants.

Our model of the affected European waste management and energy supply is based on the scenarios from Ekvall et al. (2021), but we distinguish between the impacts of near-term and future waste imports to Sweden. Near-term imports are affected by the near-term incineration of uncollected plastic waste and residues from the sorting, washing and recycling of collected waste plastics. Near-term waste imports will, we argue, mainly affect landfilling in the country where the waste is generated. This is because waste incinerators tend to be used at full capacity, and because existing incineration capacity is much smaller than the available combustible waste; almost a third of the household waste in Europe is still deposited at landfills (Eurostat 2023a).

Future waste imports will be affected by the incineration of wooden and plastic sleepers as plastic sleepers generate much more energy than a wooden sleeper. The systems impact of a change in the waste imports several decades from now are highly uncertain. The EU Landfill Directive (European Commission 1999) restricts landfilling, which is declining while energy recovery from waste incineration expands (Eurostat 2023a). Assuming these trends will continue, there might be competition over combustible waste in Europe a few decades from now, and all incinerators may no longer be used at full capacity. This means downcycling to energy-rich plastic sleepers can, when incinerated, in part increase the total incineration with energy recovery in Europe. We assume that the marginal treatment of future combustible waste will be 50% landfilling with the systems impact described in previous paragraph, and 50% incineration. Swedish incinerators are highly competitive because they recover energy as both electricity and district heat (Olofsson et al. 2005). Hence, the affected incinerators are likely to be in other countries.

Based on the previous three paragraphs, our model applies a combination of the two scenarios from Ekvall et al. (2021): the landfill scenario for imports of near-term European waste, and a 50/50 combination of the landfill and incineration scenarios for imports of future European waste (cf. Figure 2). We consider this model to be plausible, but it relies on several assumptions and, hence, includes great uncertainties.

3.3 Functions and functional unit

As illustrated in Figure 2, the affected system generates several different functions in terms of products, waste treatment services, fuels, and electricity. The functional quantities generated are determined by the rates of high-quality recycling and downcycling in the investigated cases. Alongside our initial functional unit, the function provided by 1 tonne of packaging produced from primary plastics, our assessment has a multi-dimensional supplementary functional unit, reflecting the volumes of each function that is at stake depending on if and how the plastic packaging is recycled (cf. Figure 2). Quantifying the flows at stake requires the cases with extreme values to be identified. The quantity at stake and, hence, the functional output, is the difference between the extreme values (see Table 2).

Table 2: *The supplementary functions (besides the function of 1 tonne of primary-plastics packaging) of the multifunctional system. The multidimensional supplementary functional unit is given as the volumes at stake for each supplementary function. These are calculated in the Supplementary material, which can be shared upon request.*

Function	Conventional option	Alternative option	Case with maximum conventional	Case with minimum conventional	Functional output
Railway sleepers	Wood	Downcycled plastics	No recycling, Advanced sorting	Downcycling	5.0 pieces
Treatment of near-term imported waste	Incineration in Sweden	Landfilling abroad	Downcycling	No recycling	1.02 tonne
Near-term combustible gas	Natural gas	Landfill gas	Downcycling	No recycling	5.1 GJ
Treatment of future imported waste	Incineration in Sweden	50/50 landfilling/ incineration abroad	No recycling	Downcycling	0.50 tonne
Future combustible gas	Natural gas	Landfill gas	No recycling	Downcycling	1.3 GJ
Future European electricity	Natural gas	Waste incineration	No recycling	Downcycling	238 kWh

3.4 Energy and climate input data

The most important input data and data sources are presented in Tables 3 and 4. Additional details can be found in the Supplementary material (Sheet "Unit processes"), which also includes the calculations of the climate impact of the multifunctional system in the three cases (Sheet "Calculations and results").

Table 3: The specific climate impact of activities in our study. CO_2e = carbon dioxide equivalent.

	Climate impact (kg CO_2e per reference unit)	Reference unit	Data sources
Primary production of the polymer mix	1.92	kg plastics	Zheng & Suh (2019); USEPA (2015); Ecoprofiles from Plastics Europe
Collection of source-separated packaging	0.06	kg collected material	SPR (2023)
Baling	0.003	kg collected material	SPR (2023)
Transport to sorting plant	0.012	kg transported material	SPR (2023)
Sorting	0.0007	kg collected material	SPR (2023)
Transport to recycling plant	0.05	kg transported material	SPR (2023)
Washing	0.11	kg material sorted for high-quality recycling	SPR (2023)

Granulation	0.14	kg material from high-quality recycling	SPR (2023)
Agglomeration	0.002	kg downcycled material	WiPa (2024)
Extrusion	0.004	kg downcycled material	Kent (2009)
Wood-sleeper production	0.14	kg wood sleeper	Swedish Wood (2021)
Swedish waste incineration, mixed plastics	2.7	kg incinerated mixed plastics	Lätt et al. (2020)
Swedish waste incineration, wood	0.02	kg incinerated wood	WAMPS model
Swedish waste incineration, imported waste	0.54	kg imported waste	WAMPS model
Waste incineration in other countries	0.58	kg not imported waste	WAMPS model
Competing electricity supply	0.35	kg waste not incinerated	WAMPS model
Landfill emissions	1.01	kg deposited waste	WAMPS model
Combustion of extracted landfill gas	0.09	kg deposited waste	WAMPS model
Production and use of competing natural gas	0.34	kg waste not deposited at landfills	WAMPS model

Table 4: Heating values of different waste flows in the study.

Heating value	Data source
22 MJ per kg waste plastics	SEA (2021)
14 MJ per kg waste wood	SEA (2021)
11 MJ per kg imported waste	Ekvall et al. (2021), based on the WAMPS model

The electricity supply for baling, sorting, washing, and granulation is modelled using technology-specific data when contracts with the power supplier specify the electricity source. In other cases, we use electricity data reflecting the residual Nordic mix. The climate impact of downcycling (agglomeration and extrusion) is an estimation based on the assumption – in this context, a conservative one – that only hydropower electricity is used.

A 100 kg railway sleeper produced from downcycled plastics competes in our model with an 80 kg sleeper produced from Swedish wood. The climate impact of wood production in the model accounts for land use and land-use change, although this contribution is negligible.

Much of the emission data for waste incineration and landfills is based on the model Waste Management Planning System (WAMPS), which was originally described by Moora et al. (2006) and was updated in 2016. Based on the heating values in Table 4, we estimate that each tonne of waste plastics competes with 2 tonnes of imported waste in the incinerators, as we assume the capacity of the incinerators to be limited by the energy content of the waste flow.

As stated in the previous section, our models of affected waste-management and energy-supply processes in other European countries are the same as those presented by Ekvall et al. (2021). The modern landfills in these countries are modelled with a capture of 70% of the methane (CH_4) formed during a hundred-year period. Of the unextracted CH_4, 10% (i.e., 3% of the generated CH_4) is assumed to oxidize in the landfill cover and not affect the climate. The extracted landfill gas competes in our model with natural gas.

Waste incineration in other European countries is modelled at 33% electricity efficiency. The generated electricity competes in the model with combined cycle power plants at 58% efficiency.

As stated in Section 1.3, data on the emissions of individual substances is not available for all activities; however, the climate impact of most activities is completely dominated by emissions of fossil CO_2, with other greenhouse gases, such as CH_4 and nitrous oxide (N_2O), contributing little to the total climate impact of the process The most important exception is landfill emissions, which are dominated by 37 g of CH_4 per kg of deposited waste. The combustion of biogenic material (wood sleepers and landfill gas) is also an exception; here, the relatively small climate impact is dominated by N_2O emissions. Our characterisation factors (27 for CH_4 and 273 for N_2O, based on Forster et al. 2021) are important for these processes but not in the rest of the system.

4 Results and discussion

The calculated climate impacts of the three multifunctional systems are presented in Figure 3. A few parts of the multifunctional system in Figure 2 have been aggregated to increase the visibility of Figure 3:

- “Recycling/downcycling” in Figure 3 includes the Collection, Sorting, Washing, Granulation, and Grinding, agglomeration, and extrusion in Figure 2.
- “Near-term emissions, Europe” includes emissions from European landfilling, Natural gas extraction and distribution, and Gas combustion.
- “Future emissions, Europe” includes emissions from European landfilling, European waste incineration, Natural gas extraction and distribution, Gas combustion, and Gas-fired power plants.

A few key parts of the system dominate the results: the production of primary plastics (teal in the bar graph); the incineration of plastic packaging in residual waste (i.e., not source-separated waste) and in the reject from the recycling system (orange); the incineration of plastic railway sleepers after use (brown); and the near-term European emissions (light grey). The latter is dominated by CH_4 emissions from landfills in European countries outside Sweden.

The climate impacts of collection and recycling processes (light blue) and the production of wooden railway sleepers (blue) are nearly negligible in comparison to other parts of the system.

The total results indicate that the climate impact is slightly lower for downcycling compared to no recycling. The impact of primary plastics production (teal) is the same in these cases, as are the total emissions from plastic waste incineration (orange and brown), although downcycling involves that half of the plastics being incinerated after a second use in railway sleepers. The production of wood sleepers in the case with no recycling contributes marginally to the difference that downcycling makes.

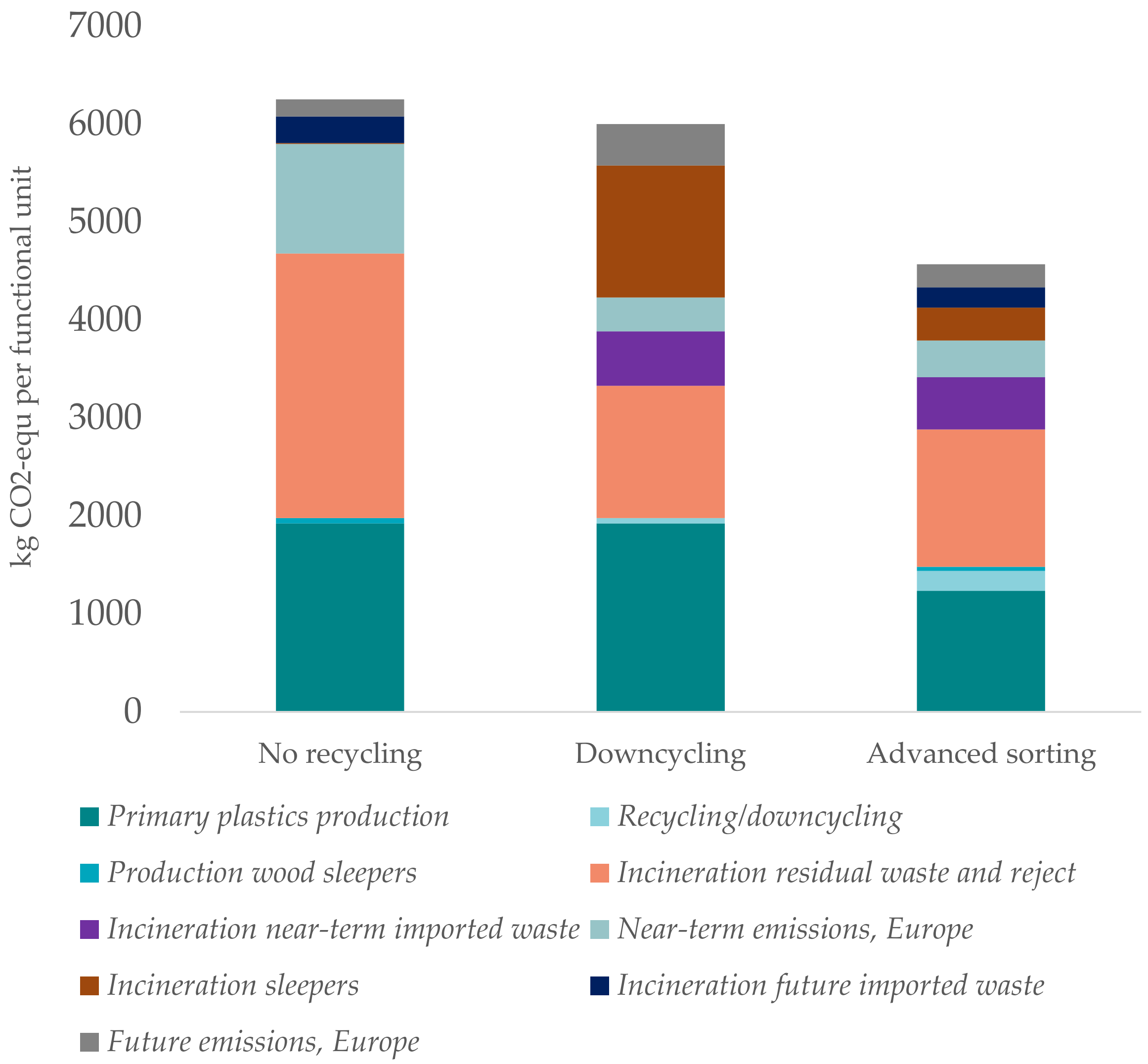

Figure 3: *The calculated climate impact in different scenarios of the system generating the basket of functions investigated.*

More important for the climate benefits of downcycling is the delay in the incineration of downcycled plastics. Since less plastic is incinerated in the near-term, there is greater capacity to treat more near-term waste from other European countries in Swedish waste incineration plants. Consequently, less waste is deposited at European landfills, which reduces the near-term European emissions from waste management. This reduction is greater than the increase in emissions from incinerated imported waste (purple).

When plastic railway sleepers are incinerated in the future, this affects the future imports of waste and, hence, future European waste management. However, our model assumes that future waste imports affect not only landfills but also incineration in other European countries (see Section 3.2). Hence, the climate impact of reduced waste imports in the future (dark grey) is less important than reduced near-term waste imports (light grey).

Our results indicate that high-quality recycling brings a much greater climate benefit than downcycling, compared to no recycling. The climate benefit from high-quality recycling stems primarily from a reduced need for primary plastics production, and the reduced incineration of plastics. The latter makes room for the incineration of imported waste with a lower fossil content than pure plastics. As a further climate benefit, the imported waste is not deposited at landfills, which means near-term European emissions are significantly reduced in the model. These benefits increase with increased collection rates and improved sorting. Key to this development is that high-quality recycled plastic material in the model replaces primary plastics in products that, after use, are also sent to high-quality recycling.

Our results should be regarded merely as indications of actual systems effects, since the model includes substantial uncertainties on the causal relationships, efficiency data, and emissions data. The near-term European emissions (light grey) are highly uncertain, as they rely on assumptions on, for example, what fuel imports are affected and the capture rate of CH_4 in affected European landfills. The future European emissions (dark grey), albeit less important for the total results, are even more uncertain. This is because we cannot know what European waste treatment is affected by future waste imports to Sweden. Note, though, that the climate benefit of high-quality recycling is evident from Figure 3 even without accounting for the uncertain impacts on European emissions; the climate benefits of reduced primary production (teal) and reduced plastics incineration (orange and brown) are much greater than the climate impact of recycling (light blue).

Our study excludes chemical recycling, which may potentially complement mechanical recycling and, hence, increase the circularity of plastics. Most processes for chemical recycling require a degree of sorting, and some also require washing (Lange et al. 2024). This means that they, at least in part, compete with mechanical recycling over the same sorted waste flows. The exception is gasification, which can chemically recycle mixed plastic waste, such as sorting residues. However, the recycling yield of gasification is low (approx. 50%; Lange et al. 2024) because part of the input material is used as an energy source in the gasification process, releasing half of the carbon to the atmosphere. The yield and climate impact may improve in the future if the process is electrified.

The results for downcycling and advanced sorting in Figure 3 would both indicate a lower climate impact if sorting residues, recycling residues, and used railway sleepers are chemically recycled through gasification rather than being incinerated with energy recovery. This is because: 1) part of the carbon in the plastic waste will be retained in the material, and 2) a larger share of Swedish incinerator capacity can be used to treat imported waste. The results for downcycling would be improved the most, particularly if gasification is electrified and the fossil-fuel share of the electricity supply is small, when the sleepers reach their end of life. In this case, downcycling will have significantly less impact on the climate, compared to no recycling. The climate advantage of high-quality recycling, compared to downcycling, will mainly be that mechanical recycling substitutes primary polymers, while downcycling followed by chemical recycling displaces the feedstock to polymer production.

The downcycled plastic in our model is used in railway sleepers that substitute wooden sleepers in low-speed railways. We would obtain similar results if the downcycled material substituted wood in other products with a long service life, such as planks. However, with future technological developments, plastic sleepers might compete with concrete sleepers in high-speed railways (Sustainability Victoria 2024). This would significantly affect the results. The production of concrete sleepers has a greater impact on the climate, and concrete sleepers cannot be incinerated at their end of life. If downcycled plastics are used to produce aggregates in concrete, our results would also be affected; they would depend on how the competing aggregates are produced and on the waste management of the concrete product.

The study accounted for the climate impact of extracting wood for sleeper production, including the associated land-use change. Another option would be to expand the system further to account for the alternative use of the wood. If the wood in this alternative use substitutes fossil-based materials or energy, this expansion of the system would probably indicate a greater climate impact of the wooden sleepers and, hence, increase the benefit of downcycling in our comparison. However, the alternative use of the wood, if any, and its climate implications are highly uncertain.

When applying the basket-of-functions approach, we aimed to account for all key factors in the assessment of high-quality plastics recycling. It is clear from the discussion in this chapter that the study could have been further expanded to account for additional causal relationships and for alternative assumptions. There might not be a definite end to such expansions. However, when the system is expanded, highly uncertain elements are introduced, for example the unknown impacts on future waste management and energy supply, and the unknown alternative use of wood. Scenario analysis can be an effective tool for managing and communicating such uncertainties.

5 Conclusions and Outlook

The recycling of plastic packaging affects not only the packaging life cycle but also a broader system of material production, waste management and energy supply. Recycling can bring a climate benefit in this broader system that is comparable to the climate impact of producing the primary plastics from the start. This level of climate benefit requires that the recycled plastics replace primary plastics in products that, after use, are recycled again. High-quality recycling makes this development more likely than downcycling does, implying that high-quality recycling is the preferrable recycling route from a climate impact perspective. Hence, a Swedish policy instrument that only stipulates a recycling rate does not adequately address the climate benefits of recycling. The climate would benefit from a policy that also accounts for the quality of the recycled plastics.

These qualitative conclusions are likely to also hold true in other countries. A large share of the climate benefit of recycling in our model is reduced emissions from incineration. Hence, the climate benefit of plastics recycling is likely to be lower in countries where the alternative to recycling is not incineration but landfill disposal of plastics material. However, much of the climate benefit also arises due to reduced emissions from the production of primary plastics. This part of the benefit is likely to be similar in all countries, since primary plastics are traded globally.

With a basket-of-functions approach, the carbon footprint results indicate the climate impact per multidimensional functional unit. Our study demonstrates that this approach can be used for estimating the system-wide climate impacts of changing a single flow (in our case the treatment of used plastic packaging in Sweden). The multidimensional functional unit can in such cases be defined by an initial functional unit (in our case the function of 1 tonne primary-plastic packaging) and the functional flows that are at stake in the comparison.

As expected, the results from the basket-of-functions approach include no avoided emissions or credits, but only the emissions associated with generating the many functions of the system (see Figure 3). However, even without credits complicating the results, a thorough discussion of the system impacts is challenging. Our results on near-term and future emissions from European waste management include emissions from several interconnected subsystems: landfills, incineration, electricity production, and gas supply. This aggregation facilitated our discussion, as we could add the observation that, in our case, landfill emissions dominated European emissions.

Further research can be initiated to expand the assessment and/or to reduce the uncertainties in the input data and, hence, the results. Complementary studies can be made where downcycled plastic substitutes aggregates or concrete products. Our study could be expanded to account for chemical recycling, and for the alternative use or fate of the wood in sleepers. The existing model and can also be improved through the collection and use of more detailed, specific, and complete input data on, for example, primary plastics production, sleeper production, and transport. An investigation on how the European waste management is affected by Swedish waste imports might significantly reduce the large uncertainty in the near-term impacts on European emissions. The impacts on future European emissions are inherently uncertain, but this uncertainty could be managed by scenarios based on projections on the future European waste management and energy supply.

Acknowledgements

The research presented in this report was funded by Swedish Plastic Recycling and by Formas and the Swedish Environmental Protection Agency through the Foundation for IVL Swedish Environmental Research Institute (SIVL). Additional funding for completing the report was provided by the Finnish Environment Institute (SYKE) within the project LIFE21-IPE-FI-PlastLIFE. This project is co-funded by the EU. Views and opinions presented in the report are, however, those of the authors only and do not necessarily reflect those of the EU, CINEA, or SYKE. Neither the EU nor the granting authority can be held responsible for them.

The calculations are developed based on previous calculations made by 2050 Consulting on behalf of Swedish Plastic Recycling.

6 Reference list

Allacker K, Mathieux F, Pennington D, Pant R (2017) The search for an appropriate end-of-life formula for the purpose of the European Commission Environmental Footprint initiative. *Int. J. Life Cycle Assess.* 22(9):1441–1458. https://doi.org/10.1007/s11367-016-1244-0

Andreasi Bassi S, Tonini D, Ekvall T, Fruergaard Astrup T (2021) A life cycle assessment framework for large-scale changes in material circularity. *Waste management 135:360-371*. https://doi.org/10.1016/j.wasman.2021.09.018

Andreasi Bassi S, Tonini D, Saveyn H, Fruergaard Astrup T (2022) Environmental and Socioeconomic Impacts of Poly(ethylene terephthalate) (PET) Packaging Management Strategies in the EU. *Environ. Sci. & Technol.* 56(1):501-511. https://doi.org/10.1021/acs.est.1c00761

Antonopoulos I, Faraca G, Tonini D (2021) Recycling of post-consumer plastic packaging waste in the EU: Recovery rates, material flows, and barriers. *Waste Management* 126:694-705. https://doi.org/10.1016/j.wasman.2021.04.002

Björklund A, Finnveden G (2005) Recycling revisited—life cycle comparisons of global warming impact and total energy use of waste management strategies. *Resour., Conserv. and Recycl.* 2005;44:309–17. https://doi.org/10.1016/j.resconrec.2004.12.002

BSI (2011) PAS 2050:2011 - Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution, London, UK. url: https://shop.bsigroup.com/forms/PASs/PAS-2050/

Caro D, Albizzati PF, Cristóbal Garcia J, Saputra Lase I, Garcia-Gutierrez P, Juchtmans R, Garbarino E, Blengini G, Manfredi S, De Meester S, Tonini D (2023) Towards a better definition and calculation of recycling. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/636900

Civancik-Uslu D, Nhu TT, Van Gorp B, Kresovic U, Larrain M, Billen P, Ragaert K, De Meester S, Dewulf J, Huysveld S (2021) Moving from linear to circular household plastic packaging in Belgium: Prospective life cycle assessment of mechanical and thermochemical recycling. *Resour., Conserv. and Recycl.* 171:105633. https://doi.org/10.1016/j.resconrec.2021.105633

Davidsson MG, Furlong RA, McManus MC (2021) Developments in the life cycle assessment of chemical recycling of plastic waste – A review. *J. Cleaner Prod.* 293:126163. https://doi.org/10.1016/j.jclepro.2021.126163

Demets R, Huysveld S, Dewulf J, De Meester S (2021) Addressing the complex challenge of understanding and quantifying substitutability for recycled plastics. *Resour., Conserv. and Recycl.* 174:105826. https://doi.org/10.1016/j.resconrec.2021.105826

Dias BZ, de Alvarez CE (2017) Mechanical properties: wood lumber versus plastic lumber and thermoplastic composite (Propriedades mecânicas: madeira versus madeira plástica e compósitos termoplásticos) Ambiente Construído 17(2):201-219 (in Portuguese with English abstract). https://doi.org/10.1590/s1678-86212017000200153

Ekvall T, Finnveden G (2001) Allocation in ISO 14041 – A Critical Review. *J. Cleaner Prod.* 9(3):197-208. https://doi.org/10.1016/S0959-6526(00)00052-4

Ekvall T, Tillman AM (1997) Open-loop recycling: Criteria for allocation procedures. *Int. J. Life Cycle Assess.* 2(3):155–162. https://doi.org/10.1007/BF02978810

Ekvall T, Björklund A, Sandin G, Jelse K, Lagerström J, Rydberg M (2020) Modeling recycling in life cycle assessment. Report 2020:05. Swedish Life Cycle Center, Gothenburg, Sweden.

Ekvall T, Gottfridsson M, Nellström M, Nilsson J, Rydberg M, Rydberg T (2021) Modelling incineration for more accurate comparisons to recycling in PEF and LCA. *Waste Manag.* 136:153-161. https://doi.org/10.1016/j.wasman.2021.09.036

EPD International (2021) General Programme Instructions for the International EPD® System. Version 4.0. url: https://www.environdec.com/home

European Commission (1999) COUNCIL DIRECTIVE 1999/31/EC of 26 April 1999 on the landfill of waste. Official Journal of the European Communities L 182:1-19. url: Landfill waste - European Commission (europa.eu)

European Commission (2020) Annex_C_V2.1_May2020. url: https://view.officeapps.live.com/op/view.aspx?src=https%3A%2F%2Feplca.jrc.ec.europa.eu%2Fpermalink%2FAnnex_C_V2.1_May2020.xlsx&wdOrigin=BROWSELINK

European Commission (2021) COMMISSION RECOMMENDATION of 16.12.2021 on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organization. C(2021) 9332 final. European Commission, Brussels. url: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=PI_COM:C(2021)9332&from=EN

Eurostat (2023a) Data browser. Accessed February 2nd 2023. url: https://ec.europa.eu/eurostat/databrowser/view/ENV_WASTRT__custom_4809212/default/table?lang=en

Eurostat (2023b) Guidance for the compilation and reporting of data on packaging and packaging waste according to Decision 2005/270/EC. Version of 30 March 2023. 297d0cda-e5ff-41e5-855b-5d0abe425673 (europa.eu)

Forster P, Storelvmo T, Armour K, Collins W, Dufresne J-L, Frame D, Lunt DJ, Mauritsen T, Palmer MD, Watanabe M, Wild M, Zhang H (2021) The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity. In Masson-Delmotte V, Zhai P, Pirani A, Connors SL, Péan C, Berger S, Caud N, Chen Y, Goldfarb L, Gomis MI, Huang M, Leitzell K, Lonnoy E, Matthews JBR, Maycock TK, Waterfield T, Yelekçi O, Yu R, Zhou B (eds.) Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the 6th Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA, pp. 923–1054. https://doi.org/10.1017/9781009157896.009

Fråne A, Youhanan L, Ekvall T, Jensen C (2016) Avfallsimport och materialåtervinning. Report B2266. IVL Swedish Environmental Research Institute, Stockholm, Sweden (in Swedish). url: FULLTEXT01.pdf (diva-portal.org)

Frischknecht R (2010) LCI modelling approaches applied on recycling of materials in view of environmental sustainability, risk perception and eco-efficiency. *Int. J. Life. Cycle. Assess.* 15(7):666-671. https://doi.org/10.1007/s11367-010-0201-6

Fuji H, Nagaiwa T, Kusuno H, Malm S (2005) How to quantify the environmental profile of stainless steel. Paper presented at the SETAC North America 26th Annual Meeting, Baltimore, USA, November 13-17 2005. url: http://www.worldstainless.org/Files/issf/non-image-files/PDF/Howtoquantifytheenvironmentalprofileofstainlesssste.pdf

Golkaram M, Mehta R, Taveau M, Schwarz A, Gankema H, Urbanus JH, De Simon L, Cakir-Benthem S, van Harmelen T (2022) Quality model for recycled plastics (QMRP): An indicator for holistic and consistent quality assessment of recycled plastics using product functionality and material properties. *J. Cleaner Prod.* 362:132311. https://doi.org/10.1016/j.jclepro.2022.132311

Guinée JB, Heijungs R, Huppes G (2004) Economic allocation: Examples and Derived Decision Tree. *Int. J. Life Cycle Assess.* 9(1):23-33. https://doi.org/10.1007/BF02978533

Gupt Y, Sahay S (2015) Review of extended producer responsibility: A case study approach. *Waste Management & Research* 33(7):595-611. https://doi.org/10.1177/0734242X15592275

Hagberg M, Gode J, Lätt A, Ekvall T, Adolfsson I, Martinsson F (2017) Miljövärdering av energilösningar i byggnader – Etapp 2. Fjärrsynsrapport 2017:409. Energiforsk, Stockholm, Sweden. (in Swedish). url: https://www.ivl.se/publikationer/publikationer/miljovardering-av-energilosningar-i-byggnader-etapp-2.html

Hermanns R, Kraft A, Hartmann P, Meys R (2023) Comparative Life Cycle Assessment of Pyrolysis – Recycling Germany's Sorted Mixed Plastic Waste. *Chemie Ingenieur Technik* 95(8):1259-1267 https://doi.org/10.1002/cite.202300041

Hermansson F, Ekvall T, Janssen M, Svanström M (2022) Allocation in recycling of composites - the case of life cycle assessment of products from carbon fiber composites. *Int. J. Life Cycle Assess.* 27(3):419–432. https://doi.org/10.1007/s11367-022-02039-8

Höglund S (2024) Case 'Sweden' Presentation of a sorting facility and the outlook how it complements the national MSW management system. Oral presentation at the webinar Raising recycling by sorting plastics from mixed MSW. March 6th 2024. URL: Raising recycling by sorting plastics from mixed municipal solid waste - European Commission (europa.eu)

ISO (2006) Environmental management – Life cycle assessment – Principles and framework (ISO 14040:2006). International Organization for Standardization. Geneva, Switzerland.

ISO (2012) Environmental management – Life cycle assessment – Illustrative examples on how to apply ISO 14044 to goal and scope definition and inventory analysis (ISO/TR 14049:2013, IDT). International Organization for Standardization. Geneva, Switzerland.

ISO (2018) Greenhouse gases – Carbon footprint of products – Requirements and guidelines for quantification. ISO 14067:2018. International Organization for Standardization. Geneva, Switzerland.

ISO (2020) Environmental management – Life cycle assessment – Requirements and guidelines – Amendment 2 (ISO 14044:2006/Amd 2:2020). International Organization for Standardization. Geneva, Switzerland.

Jubinville D, Esmizadeh E, Saikrishnan S, Tzoganakis C, Mekonnen T (2020) A comprehensive review of global production and recycling methods of polyolefin (PO) based products and their post-recycling applications. *Sustainable Materials and Technologies* 25:e00188. https://doi.org/10.1016/j.susmat.2020.e00188

Karaagac E, Jones MP, Koch T, Archodoulaki V-M (2021) Polypropylene contamination in post-consumer polyolefin waste: characterisation, consequences and compatibilisation. *Polymers* 13:2618. https://doi.org/10.3390/polym13162618

Kent R (2009) THERMOFORMING - What's Your Process Energy Fingerprint? Published 3/1/2009. url: https://www.ptonline.com/articles/whats-your-process-energy-fingerprint. Accessed September 25th 2023.

Knutsson D, Sahlin J, Werner S, Ekvall T, Ahlgren EO (2006) HEATSPOT – a simulation tool for national district heating analyses. *Energy* 31(2-3):278-293. https://doi.org/10.1016/j.energy.2005.02.005

Kousemaker TM, Jonker GH, Vakis AI (2021) LCA Practices of Plastics and Their Recycling: A Critical Review. *Appl. Sci.* 11(8):3305. https://doi.org/10.3390/app11083305

Lange J-P, Kersten SRA, De Meester S, van Eijk CP, Ragaert K (2024) Plastic recycling stripped naked – from circular product to circular industry with recycling cascade. *ChemSusChem*: e202301320. https://doi.org/10.1002/cssc.202301320

Lätt A, Johannesson C, Nellström M, Hallberg L, Guban P, Josefsson Ortiz C, Gunnarsson J, Mawdsley I (2020) Hållbar plastanvändning. SMED report No. 05 2020. Swedish Meteorological and Hydrological Institute, Norrköping, Sweden (in Swedish). url: http://naturvardsverket.diva-portal.org/smash/get/diva2:1459067/FULLTEXT02.pdf

Majeau-Bettez G, Wood R, Strømman AH (2014) Unified Theory of Allocations and Constructs in Life Cycle Assessment and Input-Output Analysis. *J. Ind. Ecol.* 18(5):747-770. https://doi.org/10.1111/jiec.12142

Ministry of Climate and Economy (2001) Förordning (2001:512) om deponering av avfall. Governmental Regulation 2001:512. Government Offices, Stockholm, Sweden (in Swedish).

Ministry of Climate and Economy (2022) Förordning (2022:1274) om producentansvar för förpackningar. Governmental Regulation 2022:1274. Government Offices, Stockholm, Sweden (in Swedish).

Ministry of Environment (2018) Förordning (2018:1462) om producentansvar för förpackningar. Governmental Regulation 2018:1462. Government Offices, Stockholm, Sweden (in Swedish).

Mølgaard C (1995) Environmental impacts by disposal of plastic from municipal solid waste. *Resour., Conserv. and Recycl.* 15(1): 51-63. https://doi.org/10.1016/0921-3449(95)00013-9

Moora H, Stenmarck Å, Sundqvist J-O (2006) Use of life cycle assessment as decision-support tool in waste management – Optimal waste management scenarios for the Baltic states. *Environ. Eng. & Manag. J.* 5(3):445-455.

Najafi SK (2013) Use of recycled plastics in wood plastic composites – A review. *Waste Manag.* 33(9):1898-1905. https://doi.org/10.1016/j.wasman.2013.05.017

Oblak P, Gonzalez-Gutierrez J, Zupancic B, Aulova A, Emri I (2015) Processability and mechanical properties of extensively recycled high density polyethylene. *Polym. Degrad. Stab.* 114:133–145. https://doi.org/10.1016/j.polymdegradstab.2015.01.012

OECD (2014) The State of Play on Extended Producer Responsibility (EPR): Opportunities and Challenges. Issues paper at Global Forum on Environment: Promoting Sustainable Materials Management through Extended Producer Responsibility (EPR), 17-19 June 2014, Tokyo, Japan. url: https://www.oecd.org/environment/waste/Global%20Forum%20Tokyo%20Issues%20Paper%2030-5-2014.pdf

Olofsson M, Sahlin J, Ekvall T, Sundberg J (2005) Driving forces for import of waste for energy recovery in Sweden. *Waste Manag. & Res.* 23(1):3-12. https://doi.org/10.1177/0734242X0505146

Plastics Europe (2022) Plastics – the Facts 2022. url: PE-PLASTICS-THE-FACTS_V7-Tue_19-10-1.pdf (plasticseurope.org). Accessed February 27th 2024.

Ragaert K, Delva L, Van Geem K (2017) Mechanical and chemical recycling of solid plastic waste. *Waste Management* 69:24-58. https://doi.org/10.1016/j.wasman.2017.07.044

Raghuram H, Roitner J, Jones MP, Archodoulaki V-M (2023) Recycling of polyethylene: Tribology assessment. *Resour., Conserv. and Recycl.* 192:106925. https://doi.org/10.1016/j.resconrec.2023.106925

Roosen M, Tonini D, Albizzati PF, Caro D, Cristóbal J, Lase IS, Ragaert K, Dumoulin A, De Meester S (2023) Operational Framework to Quantify "Quality of Recycling" across Different Material Types. *Env. Sci. Tech.* 57(36):13669-13680. https://doi.org/10.1021/acs.est.3c03023

Rydberg T (1995) Cascade accounting in life cycle assessment of polymer recycling. *Polym. Recycl.* 1(4):233-241.

Schulte A, Salinas Velarde PA, Marbach L, Mörbitz P (2023) Measuring the circularity potential of recycled LDPE based on quantity and quality conservation - a functional requirement matrix approach. *Resour., Conserv. and Recycl. Adv.* 17:200127. https://doi.org/10.1016/j.rcradv.2022.200127.

Schwarz AE, Ligthart TN, Godoi Bizarro D, De Wild P, Vreugdenhil B, van Harmelen T (2021a) Plastic recycling in a circular economy; determining environmental performance through an LCA matrix model approach. *Waste Manag.* 121:331–342. https://doi.org/10.1016/j.wasman.2020.12.020

Schwarz AE, Ligthart TN, Godoi Bizarro D, De Wild P, Vreugdenhil B, van Harmelen T (2021b) Supplementary material to Plastic recycling in a circular economy; determining environmental performance through an LCA matrix model approach. *Waste Manag.* 121:331–342. https://doi.org/10.1016/j.wasman.2020.12.020

Schweighuber A, Felgel-Farnholz A, Bögl T, Fischer J, Buchberger W (2021) Investigations on the influence of multiple extrusion on the degradation of polyolefins. *Polym. Degrad. Stab.* 192:109689. https://doi.org/10.1016/j.polymdegradstab.2021.109689

SEA (2021) Värmevärden från Energimyndighetens datalager (DW). Swedish Energy Administration, Eskilstuna, Sweden. Excel file accessible through url: https://www.energimyndigheten.se/statistik/den-officiella-statistiken/statistikprodukter/varmevarden-och-densiteter/

SPR (2023) Unpublished, company-internal data. Swedish Plastic Recycling, Motala, Sweden.

Sustainability Victoria (2024) Recycled plastic composite railway sleepers replace timber sleepers for low speed rail applications. url: Recycled plastic composite railway sleepers replace timber sleepers for low speed rail applications | Sustainability Victoria. Accessed July 3rd 2024.

Swedish Wood (2021) Swedish sawn and planed wood product. Environmental Product Declaration S-P-02657. Environdec. url: https://environdec.com/library/epd2657

Tallentire CW, Steubing B (2020) The environmental benefits of improving packaging waste collection in Europe. Waste management 103:426-436. https://doi.org/10.1016/j.wasman.2019.12.045

Tillman A-M, Ekvall T, Baumann H, Rydberg T (1994) Choice of system boundaries in life cycle assessment. *J. Cleaner Prod.* 2(1):21-29. https://doi.org/10.1016/0959-6526(94)90021-3

USEPA (2015) WARM Version 13 – Plastics. US Environmental Protection Agency. url: https://archive.epa.gov/epawaste/conserve/tools/warm/pdfs/Plastics.pdf

Vadenbo C, Hellweg S, Astrup TF (2017) Let's Be Clear(er) about Substitution: A Reporting Framework to Account for Product Displacement in Life Cycle Assessment. *J. Ind. Ecol.* 21:1078-1089. https://doi.org/10.1111/jiec.12519

Villanueva A, Wenzel H (2007) Paper waste – recycling, incineration or landfilling? A review of existing life cycle assessments. *Waste Management* 27:S29–S46. https://doi.org/10.1016/j.wasman.2007.02.019

Waste Sweden (2022) 2021 Swedish Waste Management. Waste Sweden, Malmö, Swedish. url: https://www.avfallsverige.se/in-english/

Werner F, Richter K (2000) Economic allocation in LCA: A case study about aluminium window frames. *Int. J. Life Cycle Assess.* 5(2):79–83. https://doi.org/10.1007/BF02979727

WiPa (2024) Technical specifications for WiPa Plast Compactor Type PC. WiPa Werkzeug und Maschinenbau GmbH, Stadtlohn, Germany. url: https://www.interempresas.net/FeriaVirtual/Catalogos_y_documentos/3655/SR-WiPa-Plast-Compactor-2016_eng.pdf. Accessed July 3rd 2024.

Worldsteel Association (2017) Life Cycle Inventory Methodology Report. World Steel Association, Brussels, Belgium. url: https://www.worldsteel.org/en/dam/jcr:6eefabf4-f562-4868-b919-f232280fd8b9/LCI+methodology+report_2017_vfinal.pdf

WRI/WBCSD (2011) Product Life Cycle Accounting and Reporting Standard. World Resources Institute and World Business Council for Sustainable Development. url: https://ghgprotocol.org/sites/default/files/standards/Product-Life-Cycle-Accounting-Reporting-Standard_041613.pdf

Zheng J, Suh S (2019) Supplementary material to Strategies to reduce the global carbon footprint of plastics. *Nat. Clim. Chang.* 9:374–378. https://doi.org/10.1038/s41558-019-0459-z

STOCKHOLM
Box 21060, 100 31 Stockholm

GOTHENBURG
Box 53021, 400 14 Gothenburg

MALMÖ
Nordenskiöldsgatan 24
211 19 Malmö

KRISTINEBERG
(Center for Marine Research and Innovation)
Kristineberg 566
451 78 Fiskebäckskil

SKELLEFTEÅ
Kanalgatan 59
931 32 Skellefteå

BEJING, CHINA
Room 612A
InterChina Commercial Building No.33
Dengshikou Dajie
Dongcheng District
Beijing 100006
China