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Environmental effects of plastic waste recycling

Focus on Climate Change effects

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Executive summary

The New Circular Economy Action Plan is expected to play a determining role to fulfil the objectives of the European Green Deal. In that, material recycling is anticipated to contribute with important environmental benefits, especially in respect to Greenhouse Gas (GHG) emission reduction and related climate change mitigation effects. Among the different waste materials generated by our society, plastics represent considerable challenges, due to high generation and currently low recycling rates. A significant portion of the generated plastic waste is currently disposed of in landfills or incinerated, thus incurring loss of valuable resources alongside environmental emissions, notably CO_2 via incineration. Plastics life cycle gives rise to approximately 400 million tonnes CO_2 emissions per year globally (year 2012). If current trends continue, by 2050 it could rise to 20% of global oil consumption and contribute to 15% of the global annual carbon emissions[1]. Facing these threats, the Plastic Strategy is naturally a milestone of the New Circular Economy Package and reuse and recycling of plastic waste is at the very heart of such strategy[2].

To support the EU Plastic Strategy with quantitative figures, this study estimates the potential environmental effects (savings or burdens) achieved with recycling of a number of relevant polymers at EU level. These include Polyethylene Terephthalate (both amorphous and bottle-grade), High-density Polyethylene, Low-density Polyethylene, Polypropylene, Polystyrene, Expanded Polystyrene, Polyurethane, and Polyvinylchloride. The study applies the Life Cycle Assessment (LCA) method, and builds upon previous research conducted by the Joint Research Centre in the context of the EU Plastics Strategy. The primary focus is on the impact category Climate Change, reflecting the effects of GHGs emission from the investigated recycling and alternative systems. Two different perspectives are considered in the analysis: i) the total system-wide effects that can be achieved when recycling is implemented in place of alternative treatment routes currently applied in the EU, i.e. incineration and/or landfilling (waste management or system perspective) and ii) the savings attributable to the user of recycled polymer, i.e. secondary raw material, in place of an equivalent amount of virgin material (product perspective). The two sets of results therefore provide different but complementary information that can be used in different contexts and for different accounting and reporting purposes. While the system perspective can be used in the context of impact assessments (e.g. to estimate the overall savings due to the transition from a reference situation to a given target), scenario analyses, or for the calculation of expected savings from projects or changes applied to the waste management systems, the product perspective can be used in the context of product comparison (e.g. recycled-based versus virgin materials) to estimate savings associated with recycled content regardless of (or without including) the current fate of the waste to be recycled. At a system level, GHG emission savings, expressed as Climate Change benefits, in the order of about 1 140-3 573 kg CO_2-eq./t polymer waste can be achieved when one additional tonne of plastic waste is recycled in place of being sent to the alternative treatments applied today, including a mix of incineration and landfilling. Such system-wide level savings account for both the replacement of virgin production and the avoidance of current incineration and landfilling practices. An average figure of 1 852 kg CO_2-eq./t polymer waste was quantified based on the market shares of these polymers in the EU. Savings are highest for recycling of Polyurethane and bottle-grade Polyethylene Terephthalate, and when recycling displaces incineration, due to the avoided CO_2 emissions from combustion of fossil carbon in polymers. When focusing on the use of recycled polymers by manufacturers, GHG savings, expressed as Climate Change benefits, in the order of 147-1 493 kg CO_2-eq./t recycled polymer were quantified relative to using virgin material. Similarly to the former approach, savings were maximum for secondary Polyurethane and bottle-grade Polyethylene Terephthalate. While these savings are lower than those obtained with the former approach, they cannot be directly compared as the two approaches address two different questions and hence involve different functional units, system boundaries and calculation methods. Nevertheless, while applicable to different contexts and/or accounting purposes, the two approaches and associated results should be seen as complementary.

The results for Low-Density Polyethylene, Polystyrene, Expanded Polystyrene, Polyurethane, and Polyvinylchloride should be used carefully as, at the time of this research, poor information existed in respect to their recycling and for some polymers also sorting processes, as well as on the quality of the output-recyclate. This was especially the case for Polyurethane and Expanded Polystyrene. In a broader perspective, considering that the annual generation of plastic waste in the EU is estimated to about 29.1 Mt, a total (additional to what achieved today with current recycling) annual GHG saving potential of nearly 17.6 Mt CO_2-eq. can be estimated under the assumption that 70% of the investigated polymer waste currently non-

[1]Ellen MacArthur Foundation, The new plastics economy, 2016 (https://www.ellenmacarthurfoundation.org/assets/downloads/EllenMacArthurFoundation_TheNewPlasticsEconomy_Pages.pdf).

[2]EU Commission: A European strategy for plastics in a circular economy (https://ec.europa.eu/environment/circular-economy/pdf/plastics-strategy-brochure).

collected (i.e. landfilled or incinerated) is instead collected and sent for recycling, and assuming constant conditions for technology efficiencies and energy systems (as of today). This corresponds to the average impact of 2.3 million of EU citizens. Under the hypothetical scenario that 100% of the investigated polymer waste were collected for recycling, the annual GHG savings would reach about 25 Mt CO_2-eq. The results of this study are highly relevant for circular economy policies related to plastics and for informing how the circular economy can contribute to the objectives of the EU Green Deal, especially in respect to decarbonisation.

Abstract

To support the EU Plastic Strategy with quantitative figures, this study estimates the potential environmental effects achieved with recycling of selected polymers that are relevant at EU market level, applying Life Cycle Assessment (LCA) and building upon previous research conducted by the Joint Research Centre. The polymers investigated include Polyethylene Terephthalate (both amorphous and bottle-grade), High-density Polyethylene, Low-density Polyethylene, Polypropylene, Polystyrene, Expanded Polystyrene, Polyurethane, and Polyvinylchloride. The primary focus is on the impact category Climate Change, reflecting the effects of Greenhouse Gas emissions. Two different perspectives are considered in the analysis: i) the total system-wide effects that can be achieved when recycling is implemented in place of alternative treatment routes currently applied in the EU (waste management or system perspective) and ii) the savings attributable to the user of recycled polymer in place of an equivalent amount of virgin material (product perspective). Using recycled polymers in plastic product manufacture, GHG savings, expressed as Climate Change benefits, in the order of about 147-1 493 kg CO_2-eq./t recycled polymer were quantified relative to using virgin material. At a system-wide level, GHG emission savings, expressed as Climate Change benefits, in the order of about 1 140-3 573 kg CO_2-eq./t polymer waste can be achieved when one additional tonne of plastic waste is recycled in place of being sent to the alternative treatments applied today, which include a mix of incineration and landfilling. Such system-wide level savings account for both the replacement of virgin production (as in the product perspective) and the avoidance of current incineration and landfilling practices. The results of this study are highly relevant for circular economy policies related to plastics and for informing, through quantitative figures, how the circular economy can contribute to the objectives of the EU Green Deal, especially in respect to decarbonisation.

1 Introduction

Within the Green Deal, the New Circular Economy Action Plan plays a central role (European Commission 2020). At a European level, material recycling is anticipated to contribute with important environmental benefits, especially in respect to the reduction of Greenhouse Gas (GHG) emissions and related mitigation effects on climate change. Among the different waste materials generated by society, plastics present considerable challenges, due to high generation and currently low recycling rates. The most up-to-date estimates regarding plastic waste collected and sent for recycling are as low as 32.5% for post-consumer plastic waste as a whole, and 42% for plastic packaging waste (PlasticsEurope 2020; data for EU28+NO+CH). However, such figures become even lower when the entire value chain is considered through detailed material flow analysis, including the losses occurring at sorting and recycling stages. For example, for post-consumer plastic packaging waste Antonopoulos et al. (2021) estimated a final recycling rate[3] of about 14% within EU27 (25% when assuming that all plastic waste collected, sorted and exported outside EU27, e.g. to Asia, is also fully recycled) using a dedicated material flow analysis. A significant portion of the generated plastic waste is therefore currently disposed of in landfills, incinerated, or exported outside EU, thus incurring loss of valuable resources alongside environmental emissions, primarily (but not limited to) CO_2 from the oxidation of the fossil carbon in the incinerated plastic material[4]. In addition, littering is another remarkable effect related not only to improper consumer behaviour, but also to inappropriate plastic waste management, especially in developing countries (e.g. after landfilling or open dumping), leading to detrimental consequences on ecosystem and biodiversity, which still have to be completely understood. In view of this situation, the European Commission has set ambitious targets on the recycling of plastic packaging waste, i.e. 50%, 55%, and 60% of the amount generated to be achieved by 2025, 2030, and 2035, respectively (European Parliament and the Council, 2018; amendment to Directive 94/62/EC on Packaging and Packaging Waste).

In consideration of the importance of the expected GHG reduction and other environmental benefits associated with plastic waste recycling, this study aims to quantify the potential environmental effects achieved through recycling for a set of relevant polymers. Investigated polymers include Polyethylene Terephthalate (both amorphous – PETa, and bottle grade – PETbg), High-density Polyethylene (HDPE), Low-density Polyethylene (LDPE), Polypropylene (PP), Polystyrene (PS), Expanded Polystyrene (EPS), Polyurethane (PUR), and Polyvinylchloride (PVC). These polymers cover together 81% of the EU plastic market demand (PET 8%, HDPE 12%, LDPE 18%, PP 19%, PS 3%, EPS 3%, PUR 8%, PVC 10%; PlasticsEurope 2020). The study applies the Life Cycle Assessment (LCA) method, and builds upon previous research conducted by the Joint Research Centre (JRC) in the context of the EU Plastics Strategy, mainly Nessi et al. (2020) and Antonopoulos et al. (2021). The primary focus is on the impact category Climate Change, herein represented with the traditional midpoint indicator aggregating airborne emissions of GHGs based on the respective Global Warming Potential over a 100-year time horizon (GWP_{100}; IPCC, 2013). The latter thus reflects the potential of the assessed recycling chains to reduce (or increase) overall GHG emissions compared to alternative plastic waste management options currently applied in EU or virgin material production activities (depending on the adopted perspective, as described in the following). Yet, the effects are quantified also for other 15 impact categories (e.g. Ozone Depletion, Resource Use, and Water Use). Two different perspectives are considered in the analysis: i) a "waste management (or system) perspective", quantifying the total system-wide effects that can be achieved when recycling is implemented in place of the currently applied alternative treatment routes in EU, i.e. incineration and/or landfilling; and ii) a "product perspective", quantifying the effects attributable to the user of recycled polymer, i.e. secondary raw material, when used in place of an equivalent amount of virgin polymer for the intended application[5]. The two sets of results respond to two different policy/research questions, therefore providing complementary information that can be used in different contexts and for different purposes.

[3] Calculated based on the amount of recycled material ultimately obtained, considering the whole recycling chain (collection, sorting, and reprocessing into pellets, flakes and granules).

[4] Most polymers currently used in the EU market are fossil-based (as it can be inferred from e.g. Spekreijse et al., 2019).

[5] The focus is on the use of recycled material in unspecified plastic applications, except for PETbg, which is used in the manufacturing of new bottles.

2 Goal and scope of the study

To support the EU Plastic Strategy with quantitative figures, this study quantifies the potential environmental effects (savings or burdens) from plastic waste recycling at the EU level, using Life Cycle Assessment (LCA) method. While the potential savings/burdens are herein quantified for sixteen different environmental and resource-related impact categories[6], the primary focus of the study is on the Climate Change impact category, reflecting the potential of plastic waste recycling to reduce (or possibly increase) GHG emissions.

The polymers investigated are Polyethylene Terephthalate amorphous (PETa), Polyethylene Terephthalate bottle-grade (PETbg), High-density Polyethylene (HDPE), Low-density Polyethylene (LDPE), Polypropylene (PP), Polystyrene (PS), Expanded Polystyrene (EPS), Polyurethane (PUR), and Polyvinylchloride (PVC). These polymers cover together 81% of the EU plastic market demand from converters (PET 8%, HDPE 12%, LDPE 18%, PP 19%, PS 3%, EPS 3%, PUR 8%, PVC 10%; PlasticsEurope 2020); the complement to 100% consists of other plastics (e.g. hub caps made of Acrylonitrile Butadiene Styrene, optical fibres, roofing sheets made of polycarbonate, coatings, touch screens, medical implants) that are not considered in this study. For most of the polymers investigated (PET, HDPE, LDPE, PP, and PS) the assessment specifically focuses on recycling of material (separately) collected as municipal waste, being such polymers mostly used in packaging applications (PlasticsEurope, 2020) which are typically discarded by final consumers as part of the municipal waste stream. For the remaining polymers (PUR, PVC, and partially EPS), other types of source are expected to be relevant (e.g. construction and demolition waste), but the same origin as above was assumed in the absence of representative data on collection pathways and sorting operations for plastic products collected as part of such waste streams (see section 4.3 for details). While this assumption is not expected to significantly affect the results, the estimates provided for these polymers have nevertheless to be interpreted with caution owing to the poor information available on the recycling processes.

The results are calculated as: i) the system-wise overall environmental savings or burdens associated with recycling of plastic waste, rather than sending it to alternative treatment pathways applied in the EU (i.e. incineration and/or landfilling), regardless of the specific actors/stages involved in the waste management chain; and ii) environmental savings or burdens associated with using secondary raw material (polymer), i.e. recycled feedstock, in place of virgin material, for an unspecified application of the relevant polymer (except for PETbg, assumed to be used in the manufacturing of new bottles).

[6] Including all the environmental impact categories covered in the Product Environmental Footprint (PEF) method and the related impact category indicators, calculated through the prescribed impact assessment models and factors (Zampori and Pant, 2019), as listed in Annex 1.

3 Waste management (system) and product perspective

The waste management (or system) perspective reflects the overall system-wide savings achieved through recycling plastic waste by diverting it from its alternative fate. The waste management (system) perspective can be used in the context of impact assessments (e.g. to estimate the overall savings due to the transition from a reference situation to a given target), scenario analyses, or for the calculation of expected savings from projects or changes applied to the waste management systems (e.g. a project of a municipality willing to move away from incineration by implementing a separate collection system followed by recycling).

The product perspective, instead, reflects the savings attributable to the user of the recycled feedstock. The product perspective can be used in the context of product comparison (e.g. recycled-based versus virgin materials to be used for a product) to estimate average savings associated with the recycled content regardless of (without including) the current fate of the waste to be recycled. The two sets of results therefore respond to two different research questions: the waste management approach looks at the waste management side and expected savings associated with the management or with the technologies involved in the treatment of the waste (and from changing a specific management/technology to implement a different one). The product perspective focuses on the comparison between products providing the same service or function (e.g. using recycled PET versus virgin PET to produce a bottle).

Note that the study focuses on recycling occurring in the EU, thus excluding the effects of plastic waste possibly exported for recycling in other countries outside the EU. Further details on the applied system boundaries and functional units are separately provided in sections 4.1 and 4.2 for the waste management system and product perspective, respectively.

4 Materials and methods

This section details the method used in the study to quantify the environmental effects (savings or burdens) of recycling. Section 4.1 describes the method used to quantify the effects of recycling plastic waste relative to the current End-of-Life, based on the most up-to-date information available. Section 4.2 describes the method used to quantify the effects associated with using secondary material (i.e. recycled feedstock) rather than virgin material, and attributable to the final users of recycled polymers.

4.1 Environmental effects of recycling (system perspective)

4.1.1 Goal and functional unit

The overarching goal of this assessment is to quantify the net environmental effects (savings or burdens) associated with recycling of plastic waste relative to the alternative treatment and disposal options currently applied, i.e. incineration and/or landfilling. The question we intend to answer is: "what are the system-wide net environmental savings (or burdens) from recycling one additional tonne of plastic waste relative to sending it to the alternative treatment (incineration and/or landfilling) currently applied in the EU?" In other words: "which are the net environmental effects that can be achieved by diverting one additional tonne of plastic waste from its current fate to recycling?" To respond to this question it is necessary to take a two-step approach:

First, it is necessary to individually assess the impact of recycling, incineration, and landfilling of one tonne of plastic waste (considering each individual polymer separately). The functional unit (FU) we use is "*the management of one tonne of polymer waste in the EU*". The system boundary includes all the operations involved in the collection and treatment of the waste. Any co-service generated during the management of the waste (e.g. electricity and heat) is credited to the waste management scenario via system expansion (see section 4.1.2) following common practice in LCA of waste management (ISO 2006a,b). The impacts are quantified for sixteen different environmental and resource-related impact categories conforming to the PEF method (Annex 1).

Second, it is necessary to quantify the net effect (impact change or delta) from diverting one additional tonne of plastic waste to recycling. This implies to choose a baseline treatment route (from now onwards called 'alternative treatment scenario') for the plastic waste that is currently not subject to recycling. This alternative treatment scenario should be subtracted to recycling in order to obtain the net effect (impact change or delta) of the transition. The impact change (savings/burdens) from additional recycling are quantified for the abovementioned sixteen environmental and resource-related impact categories conforming to the PEF method (Annex 1).

4.1.2 System boundary and main assumptions

This section defines the system boundary for recycling, incinerating, and landfilling one tonne of plastic waste. For the recycling route, the system boundary includes the operations of plastic waste collection and transport, sorting and separation, mechanical recycling, as well as treatment of process losses from both sorting and recycling through incineration. In case of incineration and/or landfilling, only collection and transport are included before treatment and/or disposal of the waste, as well as handling of combustion and flue gas treatment residues. The co-products generated along with the management of the waste (i.e. recycled polymers, electricity and heat) are credited to the waste management system by assuming the displacement of corresponding products obtained from virgin material (i.e. virgin polymers) or conventional energy sources (Figure 1). The electricity mix considered to be replaced is the residual EU electricity mix as of 2012, while for heat an average EU heat mix is credited (based on IEA statistics and EF-compliant datasets representing the supply and combustion of the different fuels). Regarding the virgin polymers assumed to be substituted by recycled polymers, a displacement based on the default substitution factors prescribed in the PEF method was assumed, to account for differences in quality of the recycled material compared to the replaced virgin material. Such factors are based on technical or economic considerations, and are equal to 0.9 for PETa, PP and HDPE, and 0.75 for LDPE (i.e. 1 kg of recycled LDPE substitutes 0.75 kg of virgin LDPE). For PETbg a 1:1 substitution is assumed, considering the quality of the recycled polymer identical to that of the corresponding virgin polymer. For EPS, PUR and PVC a value of 0.9 is considered, in the absence of specific provisions from the PEF method. An overview of the applied substitution factors is available in Table 2, which also summarises the sorting and recycling rates assumed in the study.

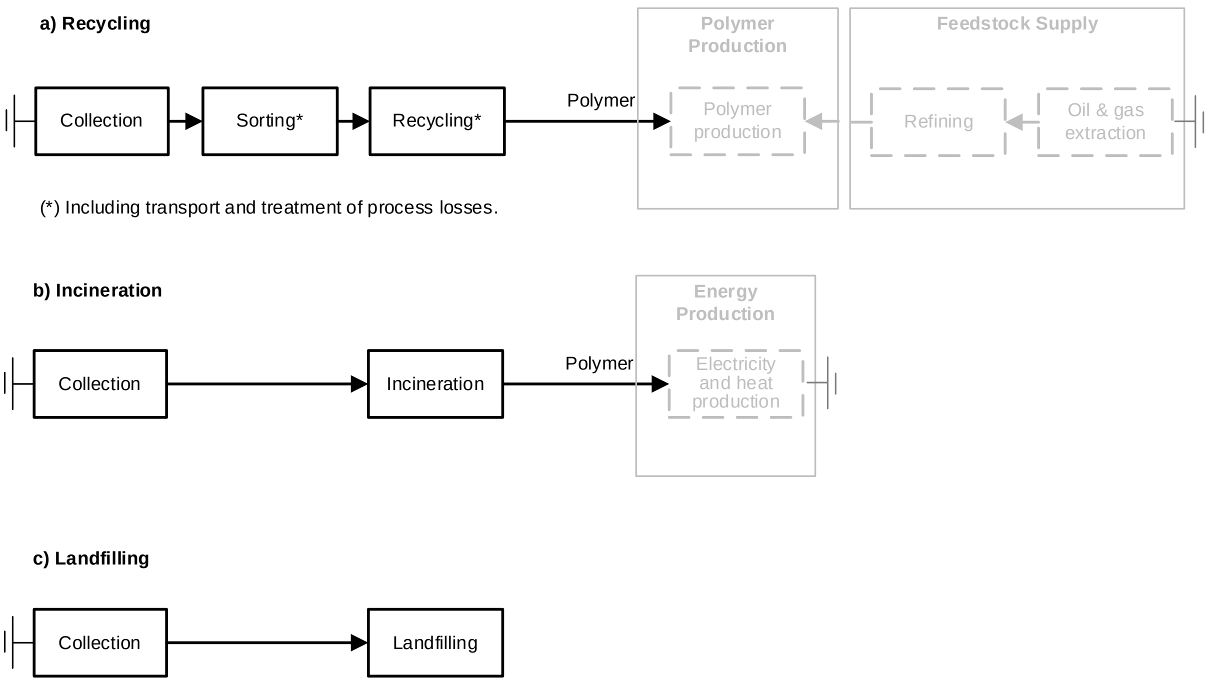

Figure 1. System boundary for a) recycling, b) incineration, and c) landfilling of plastic waste. For the default scenario analysis, a mix of incineration (b) and landfilling (c) has been calculated for each polymer, to define an EU average alternative treatment scenario displaced thanks to additional recycling (Equation 2). Black-continuous boxes indicate induced processes, while grey-dashed boxes indicates avoided processes (substitution of energy and virgin material). Process losses from sorting and recycling are sent to incineration. For the mathematical formulation applied to calculate the net environmental impacts from recycling relative to the currently applied alternative treatment routes, refer to Equation 1-Equation 3.

4.1.3 Alternative treatment scenario

As default, an average mix of incineration and landfilling is considered as the EU alternative treatment scenario. The mix is specific to each individual polymer assessed and is defined based on the latest statistics on the management of the waste category with the largest abundance of the specific polymer (i.e. packaging waste for PETa, PETbg, HDPE, LDPE, PP, and PS; packaging and construction & demolition waste for EPS; construction & demolition waste for PUR; see Table 1) or, in the absence of this information, based on more generic data available from the recent literature (for PVC; see Table 1). Such average treatment scenario aims at reflecting the current average mix of incineration and landfilling applied in the EU. However, in the result section, the savings or burdens are also quantified relative to 100% incineration or 100% landfilling taken individually, to provide a more exhaustive picture. Notice that the share of plastic waste, for each individual polymer considered, currently collected, prepared and sent for recycling is taken from Antonopoulos et al. (2021). Exceptions are EPS, PUR, and PVC, for which other sources were considered (see Table 1).

Table 1. End-of-Life treatment and disposal routes assumed for the investigated polymers at the EU level. Values in parenthesis represent the relative incineration and landfilling rates applied in the EU average alternative treatment scenario displaced thanks to recycling (expressed as %).

Polymer	Collected and sent for recycling (% of generated)(1)	Incineration and Landfilling (% of generated)
PETa	63%	37% (of which 68% Incineration, 32% Landfilling)(2)
PETbg	63%	37% (of which 68% Incineration, 32% Landfilling)(2)
HDPE	63%	37% (of which 68% Incineration, 32% Landfilling)(2)
LDPE	50%	50% (of which 68% Incineration, 32% Landfilling)(2)

PP	54%	56% (of which 68% Incineration, 32% Landfilling)(2)
PS	45%	55% (of which 68% Incineration, 32% Landfilling)(2)
EPS	27%(3)	73% (of which 56% Incineration, 44% Landfilling)(3)
PUR	0%(4)	100% (of which 45% Incineration, 55% Landfilling)(4)
PVC	32.5%(5)	67.5% (of which 63% Incineration, 37% Landfilling)(5)

(1) Based on the figures for post-consumer plastic packaging waste reported in Antonopoulos et al. (2021), unless otherwise stated.
(2) Based on the figures for post-consumer plastic packaging waste reported in PlasticsEurope (2020), unless otherwise stated.
(3) Based on the figures for post-consumer EPS waste reported in EUMEPS (2018).
(4) Conforming to PEF Category Rules for thermal insulation products (Ferreira and Adibi, 2019).
(5) Based on the figures for total post-consumer plastic waste reported in PlasticsEurope (2020).

4.1.4 Calculation method

Once the impacts associated with recycling, incineration, landfilling are calculated individually, it is necessary to quantify the net effect (impact change or delta) from diverting one additional tonne of plastic waste to recycling. As mentioned earlier, for the default calculation in this study we choose an average mix of incineration and landfilling as the alternative treatment scenario. However, the net effect (impact change or delta) is also quantified relative to incineration or landfilling taken individually (i.e. 100% incineration or 100% landfilling as alternative treatment route), to provide a more exhaustive picture. Equation 1-Equation 3 are applied to calculate the potential impacts from the recycling route (Equation 1), the average EU alternative treatment route (Eq. 2), and the net impact from diverting the specific polymer from the alternative treatment route to recycling (Eq. 3). The net effect (impact change *ΔI*; Equation 3) thus reflects the fact that the environmental impact change is quantified relative to the alternative treatment scenario, i.e. the result represents the net savings/burdens due to the *change* in waste management (from incineration and/or landfilling to recycling). From a mathematical perspective, a positive result indicates a saving while a negative one represents a burden. Note that such calculation approach for the quantification of the net savings/burdens is widely used in LCA of waste management systems when a project/decision incurs a *change* in management, and it is also aligned with the proposal of methodology for calculation of GHG emission avoidance developed in the context of the Innovation Fund by JRC and DG CLIMA (Edwards et al., 2020).

$$I_{Recycling} = I_{Collection} + I_{Sorting} + I_{Recycling} - I_{Virgin\ Production} \cdot \frac{Q_S}{Q_P} \qquad \left[\frac{impact}{t}\right]$$

Equation 1

$$I_{Alternative\ Treatment} = I_{collection} + \left(\alpha \cdot I_{Incineration} + \beta \cdot I_{Landfilling}\right) \qquad \left[\frac{impact}{t}\right]$$

Equation 2

$$\Delta I = I_{Recycling} - I_{Alternative\ Treatment} \qquad \left[\frac{impact}{t}\right]$$

Equation 3

$I_{Collection}$: impact of plastic waste collection and transport operations[7] (including possible transport between different downstream facilities) (impact t^{-1})

[7] Calculated differently according to the applied treatment or disposal route.

$I_{Sorting}$: impact of sorting operations (including incineration of process losses) (impact t^{-1})

$I_{Recycling}$: impact of mechanical recycling operations (processing of sorted plastic waste into recycled material; including incineration of process losses) (impact t^{-1})

$I_{Virgin\ Production}$: impact of virgin polymer production (impact t^{-1})

$I_{Alternative\ treatment}$: impact of the current average mix of incineration and landfilling (alternative treatment route to recycling) (impact t^{-1})

$I_{Incineration}$: impact of incineration operations (including avoided impact from replaced electricity and heat) (impact t^{-1})

$I_{Landfilling}$: impact of landfilling operations (impact t^{-1})

α, β: share of incineration and landfilling based on current treatment practices in EU (alternative to recycling; polymer specific - Table 1)

Q_S/Q_P: ratio between the quality of the secondary material and the replaced primary material

4.2 Environmental effects of using recycled material (product perspective)

4.2.1 Goal and functional unit

The goal of this assessment is to quantify the environmental effects (savings or burdens) associated with the use of recycled relative to virgin polymer. The question we intend to answer is: "what are the environmental savings (or burdens) associated with the use of one tonne of secondary material (recycled polymer) relative to the use of an equivalent amount of virgin material?" The FU is "*the use of 1 tonne of recycled polymer (secondary raw material) in place of virgin polymer*". The focus is on the use of recycled material in unspecified plastic applications, except for PETbg, which is used in the manufacturing of new bottles.

Average differences in quality of recycled and virgin materials are taken into account to calculate the potential substitution of virgin polymers with recycled ones, as described in section 4.2.3. It must be noted, however, that ultimate substitutability depends on the actual application where the recycled polymer will be used (which defines the technical performance requirements for the material to be used) and on the final quality of the recycled polymer itself (which is affected, among others, by the applied sorting and recycling technologies). The potential savings/burdens are quantified for sixteen different environmental and resource-related impact categories.

4.2.2 System boundary and main assumptions

For recycled polymer production, the system boundary includes the operations associated with the supply of plastic waste used as a feedstock (i.e. plastic waste collection, transport and sorting), as well as the mechanical recycling process, and treatment of process losses from both sorting and recycling through incineration. According to the applied allocation approach (see section 4.2.2), the system also includes a portion of the original primary production processes for the recycled material itself (from raw material extraction to polymer production). The co-products generated along with the management of process losses (electricity and heat) are credited to the system by assuming the displacement of corresponding products obtained from conventional energy sources. Avoided production of these co-products is hence also included via system boundary expansion (though this is not illustrated in Figure 2). As for the virgin polymers assumed to be substituted by recycled ones, the system boundary includes all "cradle-to-gate" supply chain processes from crude oil and/or natural gas extraction and transport, through crude oil refining into naphtha, monomer/intermediate production (e.g. naphtha cracking), to polymer production itself. Note that the stages of product manufacturing (e.g. extrusion or blow moulding), use and End-of-Life are assumed to be identical for both the recycled and the corresponding virgin material, hence are excluded from the system boundary.

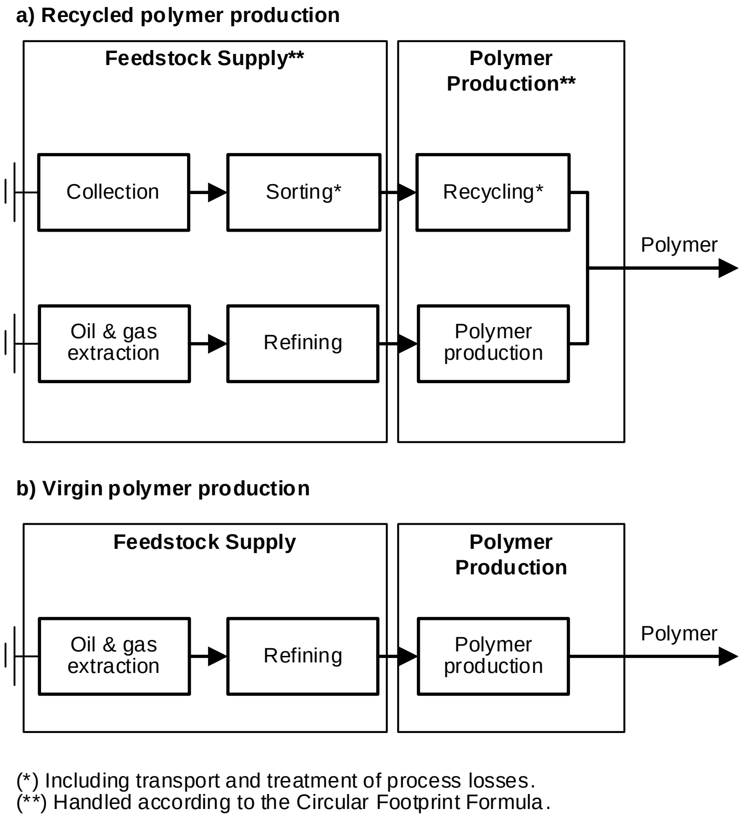

Figure 2. System boundary for a) production of recycled polymer and b) virgin polymer production. Note that the stages of product manufacturing (e.g. extrusion or blow moulding), use and End-of-Life are assumed to be identical for both the recycled and the corresponding virgin material, hence are excluded. For the mathematical formulation applied to calculate the net environmental impacts from recycled material use in place of virgin material, refer to Equation 4-Equation 5.

4.2.3 Calculation method

The potential impacts of recycled material production (Equation 4) are calculated by means of the Circular Footprint Formula (CFF), which is adopted in the PEF method to allocate the burdens from recycling activities and of original primary material production between two subsequent product life cycles (i.e. the one supplying and the one using the recycled material) (Zampori and Pant, 2019). The default allocation factor ("A" factor in Equation 4) prescribed in the PEF method for the investigated polymers and for generic plastic materials is equal to 0.5, which means assuming a 50-50 allocation of the burdens (and benefits) of recycling and virgin material production activities between two subsequent product life cycles.

The net impact (savings or burdens) from replacing virgin with recycled material use is calculated through Equation 5, i.e. as the difference between the impact associated with the supply of recycled material and that of the production of the replaced virgin material. Differences in quality of recycled and virgin material are accounted when discounting virgin production impacts, by means of the substitution factor *Qs/Qp*. The delta (*Δ*) calculated through Equation 5 hence represents the savings (or burdens) relative to the amount of virgin material potentially replaced in an unspecified application for the given polymer (except for PETbg, which is intended for the manufacturing of new bottles). From a mathematical perspective, a positive result reflects a saving while a negative one represents a burden.

The substitution factors applied to calculate displacement of virgin polymers by recycled ones are those prescribed as default values in the PEF method. They are defined based on technical or economic considerations (e.g. the relative technical performance or market value of virgin and recycled polymers) and are equal to 0.9 for PETa, PP and HDPE, and to 0.75 for LDPE (i.e. 1 kg of recycled LDPE substitutes 0.75 kg of virgin LDPE). For PETbg a 1:1 substitution is assumed, considering the quality of the recycled polymer identical to that of the corresponding virgin polymer, in line with the substitution factor prescribed in the PEF method. For EPS, PUR and PVC a value of 0.9 is considered, in the absence of specific indications from the PEF method. An overview of the applied substitution factors is available in Table 2, which also summarises the sorting and recycling rates assumed in the study.

$$I_{Recycled\ Material\ Production} = \left(I_{Collection} + I_{Sorting} + I_{Recycling}\right) \cdot A + I_{Virgin\ Production} \cdot (1-A) \cdot {}^{Q_s}/_{Q_P} \qquad \left[\frac{impact}{t}\right]$$

Equation 4

$$\Delta I = I_{Recycled\ Material\ Production} - I_{Virgin\ Production} \cdot {}^{Q_s}/_{Q_P} \qquad \left[\frac{impact}{t}\right]$$

Equation 5

$I_{Collection}$: impact of plastic waste collection and transport operations (including downstream transport of sorted plastic waste between sorting and recycling facilities) (impact t^{-1})

$I_{Sorting}$: impact of sorting operations (including incineration of process losses) (impact t^{-1})

$I_{Recycling}$: impact of recycling operations (processing of sorted plastic waste into recycled material for subsequent use; including incineration of process losses) (impact t^{-1})

$I_{Virgin\ Production}$: impact of virgin material production (impact t^{-1})

A: market factor allocating impacts between different life cycles (0.5 for recycled plastics; based on PEF)

Q_S/Q_P: ratio between the quality of the secondary material and the replaced primary material

4.3 Life Cycle Inventory

This section describes the inventory data and the main assumptions used to model the stages of waste collection and transport, sorting, recycling, incineration, landfilling and virgin polymer production. Table 2 summarises the inventory datasets applied to model the recycling process of each polymer, along with the respective recycling and sorting efficiencies assumed in the study. An overview of the virgin polymer production datasets applied in the study is instead available in Table 3. Further inventory details may be found in Annex 2-Annex 4.

4.3.1 Collection

Collection and transport of source-separated plastic waste for recycling was modelled according to the information and data reported in Rigamonti et al. (2013) for separate (mono-material) collection of municipal plastic waste through kerbside and drop-off collection systems. These data refer to a region with a well-developed waste management scheme in northern Italy, and can be considered representative of several regions in Europe where good levels of separate collection are achieved with the implementation of kerbside collection systems. While this may not still be a common practice across all Europe, the approximation is considered reasonable in view of the generally marginal role played by waste collection and transport compared to other processes in the system boundary (e.g. Rigamonti et al., 2014). Similarly, the applied data are likely poorly representative of EPS, PUR and PVC waste collection, being these polymers mostly used in the building sector, which normally do not contribute to municipal waste generation. However, this limitation is expected to only marginally affect the results, for the same reason as above. Annex 2 summarises the main assumptions and modelling details of post-consumer plastic waste collection and transport for recycling (including assumed collection pathways, vehicle types and distances). Inventories related to the use of vehicles for collection and transport were derived from the pool of EF-compliant datasets.

Relevant data for the modelling of collection of plastic waste prior to incineration and landfilling (as mixed residual waste) were also based on Rigamonti et al. (2013), and implemented in the inventory as detailed in Annex 3. Transport of collected waste to incineration facilities and landfilling sites is already accounted for in the EF-compliant incineration and landfilling dataset applied, hence no specific transport was modelled separately.

4.3.2 Sorting

After collection, separately collected polymer waste containing post-consumer plastic products is first sorted in specific material recovery facilities (MRFs). The aim of sorting is to separate plastic materials from any other co-collected materials (in case of multi-material collection), remove impurities (i.e. materials and products not intended for recycling), and to further separate mixed plastics into individual polymer streams (e.g. PET, HDPE and PP). Additional sorting of homogeneous polymer streams by colour may be performed, directly at sorting facilities or also before recovery at recycling plants. An average life cycle inventory of mixed plastic waste sorting has been developed in Franklin Associates (2018) based on input/output data from different dual-stream and single-stream sorting facilities in the United States. This inventory has been considered as a reference for the modelling of the sorting process of all the investigated polymers, complementing the reported input and output data with background EF-compliant datasets representative of EU-average conditions. The applied data are mostly representative of sorting of plastic polymers collected at as part of municipal waste (and hence of PET, HDPE, LDPE, PP and PS, mostly used in packaging or household applications). However, they can also be considered a reasonable approximation of the burdens of any sorting operations applied to EPS, PUR and PVC waste streams mostly coming from other sectors than those contributing to municipal waste (e.g. building and construction rather than packaging). Limitations due to the absence of specific data are however acknowledged. The sorting (material recovery) rates were based upon the findings of a recent study (Antonopoulos et al. 2021) and equalled 91% for PET (both amorphous and bottle grade) and HDPE, 73% for LDPE, 79% for PP, 65% for PS, and 73% for PVC. For EPS and PUR, the same sorting efficiency as PS was assumed (65%), in the absence of specific data. An overview of the sorting process inventory is presented in Annex 4, while the sorting (and recycling) rates applied are summarised in Table 2. For additional insights the reader is referred to Antonopoulos et al. (2021).

4.3.3 Recycling

A mechanical recycling process was modelled for all the polymers investigated. Mechanical Recycling of sorted PET waste into new amorphous PET resin (suitable for e.g. textile fibre manufacturing) was modelled through an aggregated, EF-compliant dataset representing the burdens of secondary PET granulate production from sorted, post-consumer plastic waste via grinding, metal separation, washing, and extrusion to pellets. The dataset, developed based on literature data for these unit operations, accounts for an overall recycling rate equal to 85.5% (on the sorted input material), with process waste and scrap being sent to incineration. This assumption is in line with the typical fate of plastic recycling residues, which due to their high calorific value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

Secondary bottle-grade PET production out of sorted post-consumer PET waste was modelled based on the *ecoinvent* dataset *"[CH] polyethylene terephthalate production, granulate, bottle grade, recycled"* in the absence of representative EF-compliant datasets. The dataset has been developed based on data from two Swiss recycling facilities and represents the burdens associated with the processing of sorted, pure-coloured waste PET bales into bottle-grade PET flakes, with an overall recycling rate equal to 82%. Main process steps include bale opening, metal separation, shredding, air separation of light-weighting label residues (sent to incineration), flotation (separating HDPE cap fragments from PET flakes), and a further step where PET flakes are treated with a Sodium Hydroxide solution and heated to nearly 200°C for decontamination purposes. Purified PET flakes are finally washed with water and then dried. The inventory was assumed to include also the Solid State Polymerisation (SSP) process, typically required to increase the intrinsic viscosity of recycled PET flakes to a level comparable with that of virgin PET resin, as well as final extrusion of flakes into new polymer granulate. However, it is not totally clear from the dataset documentation if these steps are actually considered. For calculation purposes, the original dataset was adjusted from Swiss to EU background conditions, and background datasets related to energy supply were replaced with EF-compliant datasets. The small amount of recovered HDPE from bottle caps (0.122 kg/kg R-PET) was neglected (i.e. no virgin material substitution was considered) to avoid a distortion of the results of the PET recycling chain under investigation. Finally, a few additional adjustments were performed to improve reliability of LCIA results for the Ozone Depletion impact category (as detailed in Nessi et al., 2020).

As for HDPE recycling, the *ecoinvent* dataset "*[Europe without Switzerland] Polyethylene production, high density, granulate, recycled*" was used as a basis for modelling purposes, in combination with EF-compliant background datasets for energy and material supply under EU-average conditions. Moreover, the dataset was adjusted based on a most recent and expanded version of the original inventory data source (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011). The inventory is calculated as mass-weighted average of data collected from several recycling facilities in the United States. The overall recycling rate is equal to 84%, with removed contaminants and process waste being sent to incineration (in line with the typical fate of plastic waste recycling residues, as discussed above).

For PP recycling, no specific dataset is available in the EF context nor in existing life cycle inventory databases. A new dataset was thus developed based on foreground inventory data available from the literature (Franklin Associates, 2018), combined with EF-compliant background datasets for energy and material supply under EU-average conditions. Similarly to HDPE recycling, the inventory is based on mass-weighted average values of data collected from several recycling facilities in the United States. The overall recycling rate is equal to 85.5%, with removed contaminants being sent to incineration along with process waste (consistently with the previously described recycling processes).

Similarly to PP, for LDPE, PS, EPS, PUR and PVC recycling no specific recycling datasets are available in the EF context nor in existing life cycle inventory databases. In addition, no literature data were found for LDPE, PS and PVC, while data available for EPS and PUR are only partial or outdated. Therefore, the recycling process was approximated by means of an aggregated EF-compliant dataset representing the production of a generic secondary plastic granulate out of sorted post-consumer plastics waste, namely "*[EU-28] Plastic granulate secondary (low metal contamination)*". The modelled recycling process includes the operations commonly applied to other thermoplastics polymers (grinding, metal separation, washing, and extrusion to pellets), based on operation-specific inventory data form the literature. The overall recycling rate is equal to 84% on the input material, with process waste and scrap being sent to incineration. While this modelling approximation is considered acceptable for LDPE, PS and PVC (all being thermoplastic polymers that can be recycled through traditional operations such as shredding, water-based separation, cleaning, and regranulation), there are limitations in extending these data also to EPS and (especially) PUR, which are recycled through at least partially different processes. For instance, PUR recycling requires, beyond traditional shredding, also pressure bonding and addition of specific substances (e.g. diphenyl methane diisocyanate) as a binder, while flame retardants may also need to be removed (Zevenhoven, 2004).

4.3.4 Incineration

For all the investigated polymers, aggregated, material-specific waste incineration datasets are available from the EF database and were applied to model the treatment of such polymers in a municipal waste incineration plant. The datasets are based on a waste incineration model considering combustion in a grate furnace, a steam generator to recover heat in flue gases, and subsequent cleaning of these in a dry treatment line. Bottom ash is used as construction material after metal separation and ageing, while air pollution control residues (including fly ash, boiler ash and slag) are disposed of in underground exhausted salt mines. The model applies specific transfer coefficients (based on data from real plants, stoichiometry, or expert estimate) to calculate the distribution of each element in the input waste composition between flue gases (air emissions) and the different treatment residues (bottom ash and air pollution control residues). However, air emissions of a number of substances are modelled irrespectively of the waste composition, as they are rather considered a function of the concentration in cleaned flue gas that can be achieved thanks to the applied cleaning technologies. For these substances (including HCl, HF, NOx, VOC, N_2O, CO, NH_3, SO_2, particulate matter, dioxins, and the heavy metals As, Cd, Co, Cr, Ni and Pb), emissions are calculated based on average concentrations in cleaned flue gas reported in the Reference Document on Best Available Technologies for waste incineration (adjusted with measured concentrations from real plants) and the waste-specific flue gas production (m^3/kg waste). The energy content (net calorific value) of the input waste is taken into account to calculate the amount of recovered energy (electricity and heat), based on EU-average energy efficiencies and recovery rates. EU-average values are also considered for the share of catalytic and non-catalytic systems for NO_x reduction, affecting reagent consumption for removal of such pollutant and its final emission with flue gas.

In line with the approach to handle energy recovery situations adopted in the PEF method, the incinerated polymer waste takes the full burdens from the incineration process. Similarly, it is credited for 100% of the benefits associated with avoiding production of conventional energy (electricity and heat) replaced by energy recovered from waste. These credits are already accounted in the aggregated incineration datasets applied in this study, considering the EU residual electricity grid mix for displaced electricity (2012 data), while for thermal energy an EU-average production mix is modelled, based on statistics from the International Energy Agency (IEA) and using EF-compliant datasets to represent supply and combustion of the fuel mix.

Table 2. Overview of the datasets applied in the study to represent polymer recycling, of the sorting and recycling rates assumed, and of the virgin material substitution factors assumed. EI: Ecoinvent; EF: Environmental Footprint.

Polymer	**Dataset recycling**	**Sorting rate(1)**	**Recycling rate(2)**	**Substitution factor(3)**
PETa	EF(4)	91%	85%	0.9
PETbg	EI(5) – adjusted to EU conditions and as further described in section 4.3.3	91%	82%	1
HDPE	EI(6) – updated based on Franklin Associates (2018) and EF background datasets	91%	84%	0.9
LDPE	EF(7)	73%	84%	0.75
PP	Developed on purpose based on Franklin Associates (2018) and EF background datasets	79%	85%	0.9
PS	EF(7)	65%	84%	0.9
EPS	EF(7)	65%(8)	84%	0.9
PUR	EF(7)	65%(8)	84%	0.9
PVC	EF(7)	73%	84%	0.9

(1) Also called "sorting rate or efficiency": represents the efficiency of the sorting plant defined as amount of target material out divided by amount of target material in (from collection; including impurities). Based on Antonopoulos et al. (2021).

(2) Also called "recycling process efficiency rate or reprocessing rate": represents the efficiency of the recycling plant defined as amount of target material out divided by amount of target material in (including impurities coming in). The rate is here consistent with the polymer-specific recycling life cycle inventory dataset applied.

(3) According to the default values reported in the PEF method for PET, PE and PP, and assumptions for PS, EPS, PUR and PVC.

([4]) [EU-28] Polyethylene terephthalate (PET) granulate secondary; no metal fraction, from post-consumer plastic waste, via grinding, metal separation, washing, pelletization | single route, at consumer.
([5]) [CH] Polyethylene terephthalate production, granulate, bottle grade, recycled.
([6]) [Europe without Switzerland] Polyethylene production, high density, granulate, recycled.
([7]) [EU-28] Plastic granulate secondary (low metal contamination), from post-consumer plastic waste, via grinding, metal separation, washing, pelletization | production mix, at plant.
([8]) Assumed equal to the sorting rate of PS, in the absence of specific data in Antonopoulos et al. (2021) and other literature.

4.3.5 Landfilling

Landfilling of all the polymers was modelled based on a common, aggregated, EF-compliant dataset representing disposal of plastic waste in a managed municipal waste landfill (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*). The underlying inventory is material-specific, but refers to the average chemical composition and degradability of generic (average) plastic waste, rather than to those of the specific polymers. This is considered an acceptable approximation for this study, since the degradation rate in the landfill body (one of the most relevant parameters for landfilling modelling) is similar for all non-biodegradable polymers as the ones investigated in this study, and set to approximately 1% from the time of deposition. The inventory is developed based on a landfill model applying specific transfer coefficients to calculate the distribution of elements in the waste composition to landfill gas and leachate, and their ultimate emissions to the environment over a 100 year time horizon. Emissions occurring beyond 100 years are not accounted for in the model. Landfill gas generation is calculated based on the amount of carbon in the waste degraded over such a timeframe, while considering an average volumetric landfill gas composition (in terms of CO_2 and CH_4) for the stable methane phase. The model also adapts relevant site-specific and technology-specific parameters to the geography and technology of reference (e.g. precipitation, type of sealing and cap layers, collection and use rate of landfill gas, energy efficiencies of engines, as well as collection rate of leachate and respective treatment efficiencies). In the selected dataset, these parameters reflect the EU-average situation as follows, for a landfill with a height of 30 m, and an area of 40 000 m^2. The landfill is equipped with a surface and a basic sealing consisting of gravel and sand (filtering layers), a polyethylene waterproofing sealing, and clay as mineral coverage. Any generated landfill gas is collected at a rate of 50%, with the rest being directly released to air. The utilisation rate of (possibly collected) gas for energy generation in gas engines is 56% (corresponding to an overall utilisation rate of 28%), while the remaining 44% is flared (22% of the overall gas production). Energy conversion efficiencies of engines are not available. As for parameters relevant to leachate generation, a mean precipitation of 660 mm per year is assumed, with an overall transpiration and run-off rate of 60%. Leachate is captured with a 70% efficiency and is treated in a dedicated plant via active carbon filtration and flocculation/precipitation processes. Sludge generated from leachate treatment is assumed to be dried and disposed of in an underground deposit.

4.3.6 Virgin polymers production

The virgin polymer production includes the stage of feedstock supply, polymer production and associated transport. The stage of feedstock supply includes the processes of crude-oil and natural gas extraction and transport, naphtha production in crude oil in refineries, and its subsequent transport to downstream conversion processes (typically naphtha cracking, or also catalytic reforming). Polymer production covers all the supply-chain activities for the conversion of naphtha and/or natural gas into relevant intermediates and monomers (or directly into monomers), the final polymerisation process into plastic granulate or a semi-finished product (e.g. rigid PUR and EPS foams), and any transport between these activities.

4.3.6.1 Feedstock supply

The burdens associated with crude oil supply to petroleum refineries in the EU were modelled through the aggregated, EF-compliant dataset "*[EU-27] Crude oil mix; technology mix of conventional (primary, secondary and tertiary production) and unconventional production (oil sands, in-situ) | consumption mix, to consumer*". The dataset represents the average crude-oil supply mix to the EU in terms of country of origin and respective oil sources and extraction/processing technologies (according to IEA statistics). Both conventional and unconventional oil sources (e.g. oil sands) are taken into account, as far as relevant. The considered crude-oil mix refers to the year 2014, and was found to properly reflect the situation as of today (see Nessi et al., 2020 for more detail). All relevant activities related to crude oil supply are covered in the underlying inventory, including exploration activities, well drilling, crude oil extraction and processing, long-distance transport via pipeline and (when required) tanker vessels, as well as regional distribution to the final consumer via pipeline. Structural oil losses occurring during transportation via pipeline or vessels are also taken into account in the datasets, while accidental losses (due to e.g. spills from pipelines or accidents to vessels) and oil fires are excluded since LCA typically looks at normal production conditions, disregarding the effects from accidents. Land-transformation and occupation burdens are accounted for land-based sources (e.g. oil sands), and where relevant also for activities related to the other considered oil sources. In the case of combined crude oil and natural gas production, allocation by energy (net calorific value) is performed. Activity data applied to model exploration, extraction and processing are taken from industry or the literature. Capital goods, including infrastructure, are not included, according to the applied 95% cut-off rule, based on material or energy flows, or the level of environmental significance.

Similarly to crude oil, an aggregated dataset from the EF database was applied also to the modelling of natural gas supply: "*[EU-27] Natural gas mix; technology mix | consumption mix, to consumer | medium pressure level (< 1 bar)*". The EU-average natural gas supply mix is represented in the dataset, covering both domestic production and imports from external countries according to IEA statistics for the year 2011. For each country contributing to the mix, the respective gas sources and extraction/processing technologies are considered, including both conventional and unconventional sources (e.g. shale gas, tight gas, coal bed methane). The dataset covers all relevant activities in the supply chain of natural gas, including exploration, well drilling, extraction, processing (e.g. desulphurisation), possible liquefaction followed by regasification (for imports of liquefied natural gas via vessels), long-distance transport via pipeline and vessels, and final regional distribution to the final consumer via pipeline. Natural gas losses occurring during transport are accounted for, as well, in the dataset, for both pipeline and vessel transport. Consistently with the approach adopted for crude oil (see above), in the case of combined natural gas and crude oil production, allocation by energy (net calorific value) is performed. Activity data applied to model exploration, extraction and processing are taken from industry or the literature. Capital goods, including infrastructure, are not included, according to the applied 95% cut-off rule, based on material or energy flows, or the level of environmental significance.

Naphtha production via crude-oil refining was modelled based on an aggregated gate-to-gate dataset provided by Thinkstep. The dataset represents a (mass-weighted) average refining process for Europe in terms of refining technologies and product outputs, and is based on a dedicated oil refinery model. This is built by largely relying on data and measurements from more than 100 refineries, for a total processing capacity of over 2 billion litres of crude oil per day. Industry data are complemented, where necessary, by literature data. Allocation of refinery inputs and outputs to individual products (final or intermediate) is performed based on different criteria, depending on the input or output. Crude oil demand of a specific unit process is allocated to the respective products and/or intermediate products based on energy (i.e. net calorific value of the product), assigning larger shares of overall upstream burdens to products with higher calorific value. Energy inputs (thermal energy, steam, external electricity) are allocated based on the mass share of the product or intermediate product, out of the total mass of products from the unit process. This approach allows the higher energy demand from processing "heavier" products to be better captured compared with allocation based on net calorific value (Schuller, 2020). Direct emissions to the environment are allocated based on mass, as well.

Transport of naphtha from refineries to downstream users was assumed to entirely take place via pipeline, and was modelled based on transport-related burdens included in the *ecoinvent* dataset *[RER] market for naphtha*. Compared to the original dataset, which reflects transport to different end-users – including petrol stations, a number of adjustments were made. In particular, default transport via road, rail and barge (likely associated to non-industrial users) was entirely converted to pipeline transport, which is considered more appropriate for naphtha used for industrial purposes. Since the distance associated with the different types of transport is not known, the overall original quantity (in kg*km) associated with road, rail and barge transport was converted to pipeline. This is considered a reasonable approximation, being the overall quantity of such transport modes only a small share of the overall transport demand (around 19%).

4.3.6.2 Polymer production

For all the investigated polymers, the whole process chain from naphtha and/or natural gas processing to polymerisation and/or foaming, through the production of any intermediates and monomers, was modelled by means of aggregated, gate-to-gate datasets provided by Thinkstep (Table 3)[8]. Inputs include combinations of crude oil, natural gas and naphtha, depending on the polymer. All conversion processes are assumed to take place in Europe, so that the datasets reflect EU-average background conditions in terms of e.g. energy generation, material supply and transport. For all polymers, the main conversion process involved in the

[8] An exception is PETa, for which an aggregated EF-compliant dataset was used. However, it is based on the same modelling principles and data sources as the datasets applied for the other polymers, described in this paragraph.

supply chain is steam cracking of naphtha and natural gas, delivering Ethylene and Propylene (used directly as monomers or as intermediates in PS and PUR production), along with Butadiene and other relevant intermediate flows (e.g. pyrolysis gas including mixtures of Benzene, Toluene and Xylenes). Other relevant conversion processes are catalytic reforming of naphtha and steam reforming of natural gas. In catalytic reforming, naphtha is processed, to produce Benzene, Toluene and Xylenes, with the latter (para-Xylene) being an intermediate in the production of Purified Terephthalic Acid (PTA, a co-monomer of PET), and Benzene having the same role in the synthesis of Styrene and PUR. Steam reforming of natural gas generates synthesis gas consisting of Carbon Monoxide and Hydrogen, both used in the production of Methanol (a precursor of Acetic Acid used as a solvent in PTA production), and in the synthesis of MDI (Methylene Diphenyl Diisocyanate) applied in PUR production along with Polyether Polyols. Further details on data sources and main modelling choices applied in the datasets may be found in Nessi et al. (2020).

Table 3. Overview of the datasets used in the study to represent virgin polymer production and respective data source. EF: Environmental Footprint; TS: Thinkstep.

Polymer	Dataset virgin polymer production	Data source
PETa	[EU-28+EFTA] PET granulates, amorphous; Polymerisation of ethylene | production mix, at plant | 0.91- 0.96 g/cm3, 28 g/mol per repeating unit	EF
PETbg	[EU-28] Polyethylene terephthalate bottle grade granulate (PET) via PTA - open flows naphtha, ng and crude oil; via purified terephthalic acid (PTA) and ethylene glycol | single route, at plant | 1.38 g/cm3, 192.17 g/mol per repeating unit	TS
HDPE	[EU-28] Polyethylene high density granulate (HDPE/PE-HD) - open flows naphtha, natural gas; polymerisation of ethylene | production mix, at plant | 0.91- 0.96 g/cm3, 28 g/mol per repeating unit	TS
LDPE	[EU-28] Polyethylene Low Density Granulate (LDPE/PE-LD) - open flows naphtha, natural gas; polymerisation of ethylene | production mix, at plant | 0.91- 0.96 g/cm3, 28 g/mol per repeating unit	TS
PP	[EU-28] Polypropylene Granulate (PP) - open flows naphtha, natural gas; polymerisation of propene | production mix, at plant | 0.91 g/cm3, 42.08 g/mol per repeating unit	TS
PS	[EU-28] Polystyrene Granulate (PS) - open flows naphtha, natural gas, crude oil; polymerisation of styrene | production mix, at plant | 1.05 g/cm3, 104.15 g/mol per repeating unit	TS
EPS	[EU-28] Expanded Polystyrene (PS 30) - open flows naphtha, crude oil, natural gas; technology mix | production mix, at plant | PS 30	TS
PUR	[EU-28] Polyurethane rigid foam (PUR) - open flows naphtha, natural gas, crude oil; technology mix |production mix, at plant | hard foam, high-density foam	TS
PVC	[EU-28] Polyvinyl Chloride Granulate (S-PVC) - open flows naphtha, natural gas; polymerisation of vinyl chloride | production mix, at plant | 62 g/mol per repeating unit	TS

4.4 Total annual GHG savings from recycling – calculation approach

The total annual Climate Change effect (reflecting GHG savings) from plastic waste recycling are calculated based on the results from the waste management (or system) perspective, simply extrapolating the results per tonne to the total waste flow that can potentially be additionally recycled. To quantify the total potential savings, we herein consider an annual generation of plastic waste in the EU equal to 29.1 Mt, based on the figures from PlasticsEurope (2020) (EU28+NO+CH in year 2018; in the absence of specific information for EU27).

The amount of each polymer generated as waste (*Polymer waste i*) is calculated first (Equation 6), based on the current market shares (% of total plastic demand; *market share$_i$*) reported in PlasticsEurope (2020). The distribution of the total plastic demand by polymer type is as follows: 7.7% PET[9], 12.2% HDPE, 17.5% LDPE, 19.3% PP, 3.2% PS, 3.2% EPS[10], 7.9% PUR, 10% PVC, and 19% other polymers such as ABS, PBT, PC, PMMA and PTFE, which are here assumed as not contributing to recycling (neither currently nor in the future) and hence not accounted in the calculation of total effects/savings.

$$Polymer\ waste\ generated_i = Annual\ generation\ plastic\ waste\ EU \cdot market\ share_i$$

Equation 6

The amount of each polymer waste currently separately collected and sent for recycling is calculated using Equation 7, based on collection rates for recycling reported in Table 1 (column #2).

$$Polymer\ waste\ currently\ collected\ and\ sent\ for\ recycling_i = Polymer\ waste\ generated_i \cdot \%\ collected\ and\ sent\ for\ recycling_i$$

Equation 7

The total annual Climate Change effect (GHG savings as CO_2-eq.) from current recycling of plastic waste can be quantified according with Equation 8, assuming that this waste material would otherwise be landfilled and/or incinerated (following current practice) if recycling was not in place, and that the secondary material replaces virgin material. However, it may be argued that since there is no diversion of plastic waste to recycling (which is already occurring), the effects from avoided incineration and landfilling should not be accounted for. On the other hand, a similar reasoning may be applied by analogy to the effects from virgin material substitution: if recycling is already in place, no virgin material displacement would occur since the use of secondary material in product manufacturing should already be an established practice. For consistency with the calculation of the effect from additional (future) plastic waste recycling (see below), and to allow a consistent calculation of incremental benefits, the effects from avoided incineration and landfilling alongside virgin material substitution are here taken into account in the estimate of the overall effect (savings) from current recycling. The net effects are captured, for each polymer type, in the term *Climate Change effect$_i$*,. Notice that the term *Climate Change effect$_i$* of each polymer simply refers to the "Net effect" results presented in section 5.1.1 (Figure 3; "net effect"; see also Annex 5) for the waste management (or system) perspective. Note also that, in all equations, the index *i* refers to each individual plastic polymer waste investigated in this study.

$$Annual\ GHG\ savings\ (current\ recycling) = \sum_i^n Polymer\ waste\ currently\ collected\ and\ sent\ for\ recycling_i \cdot Climate\ Change\ effect_i$$

Equation 8

The amount of each polymer waste that can be additionally collected and sent for recycling compared to the current situation is calculated using Equation 9, while the related additional Climate Change effect (GHG savings) is calculated using Equation 10, representing the effect/savings from diverting currently incinerated or landfilled plastic waste to recycling. With respect to the *Expected Capture Rate* for plastic waste, we rely on the figures reported by Triconomics (2020) that are based on a research conducted by the Nordic Council of Ministers (2014), which suggest a maximum level of 70%.

[9] Assuming this is equally split between amorphous PET (PETa) and bottle grate PET (PETbg).

[10] Assuming that the overall share provided for PS and EPS altogether (6.4%) is equally split between the two polymers.

$$Polymer\ waste\ that\ can\ be\ additionally\ collected\ for\ recycling_i = (Polymer\ waste\ generated_i - Polymer\ waste\ currently\ collected\ and\ sent\ for\ recycling_i) \cdot Expected\ Capture\ Rate$$

Equation 9

$$Annual\ GHG\ savings\ (future\ recycling) = \sum_i^n Polymer\ waste\ that\ can\ be\ additionally\ collected\ for\ recycling_i \cdot Climate\ Change\ effect_i$$

Equation 10

5 Results

This section reports the results for both the system/waste management perspective (section 5.1) and product perspective (section 5.2). The results are presented according to two layers: i) focus on the Climate Change impact category with breakdown of the impact contributions; ii) overview of the net savings or burdens across the remaining environmental impact categories assessed.

5.1 Environmental effects of recycling (system perspective)

This section presents the results for the system (or waste management) perspective (i.e. diversion of polymer waste to recycling from current alternative treatment) focusing on the Climate Change impact category (section 5.1.1). An overview of the effects on the remaining environmental impact categories assessed is presented in section 5.1.2. Throughout Figure 3a,b,c the results are expressed per tonne (t) of polymer waste managed (e.g. recycling, landfilling, and incineration), but also quantified in terms of net effect of diverting plastic waste from the alternative treatment route to recycling. We call this "Net effect" at the system level. Notice that the results are reported not only for the single plastic polymers investigated in this study, but also for the respective EU average mix, calculated according to the market shares of the same (based on PlasticsEurope; 2020; see Table 6), and assuming these as a proxy of the shares of single polymers in total plastic waste ("EU mix"). Negative results indicate environmental savings while positive results indicate environmental burdens. Additional details on the results may be found in Annex 5-Annex 6.

5.1.1 Climate Change

For all the investigated polymers, recycling one additional tonne of plastic waste relative to the average alternative treatment scenario (i.e. a mix of incineration and landfilling following the current situation in EU) was found to be beneficial for Climate Change (Figure 3a; see "Net effect"). The savings observed ranged from 1 140 for LDPE to 3 573 for PUR kg CO_2-eq./t polymer waste (Figure 3a and Annex 6; see "Net effect"). In general, the benefits were larger for recycling of PETbg, PETa, and PUR, due to the higher impacts of the respective virgin production processes compared to the remaining polymers investigated. LDPE recycling showed the lowest savings, mostly due to the reduced virgin material substitution assumed (1:0.75) with respect to the other polymers (1:0.9 or 1:1 for PETbg), and to the relatively lower sorting rate (73% vs e.g. 91% for PET and HDPE). For HDPE, PP, PS, EPS and PVC, the observed benefits were comparable and ranged from 1 479 to 1 820 kg CO_2-eq./t polymer waste (see Figure 3a; "Net effect"). For the EU average mix of all investigated polymers, the savings amounted to 1 852 kg CO_2-eq./t polymer waste. Notice that the results for LDPE, PS, PVC and especially EPS and PUR should be interpreted and used with caution, due to the poor information available on the recycling process and hence to the lower specificity or representativeness of the data applied in calculations (see sections 4.3.2 and 4.3.3).

When incineration was assumed as the only alternative treatment of the waste (Figure 3b), the net Climate Change effect from recycling increased considerably thanks to avoided fossil CO_2 emissions associated with otherwise incinerating the polymers (here assumed to be entirely fossil-based). The net effect ranged from 1 512 (LDPE) to 4 219 (PUR) kg CO_2-eq./t polymer waste (Figure 3b; "Net effect"). When landfilling was instead assumed to be the sole alternative treatment route displaced by recycling (Figure 3c), the net effect on Climate Change showed substantially decreased savings (Figure 3c; "Net effect"). This is an expected result as landfilling, contrarily to incineration, does not incur significant fossil CO_2 emissions, as degradation of fossil carbon in the landfill body is almost negligible. Thus, the savings associated with the avoided alternative treatment (landfilling in this case) become much lower and the main overall system benefits are related to avoided virgin polymer production incurred through the recycling pathway. Overall, the net effect ranged from 346 (LDPE) to 3 043 (PUR) kg CO_2-eq./t polymer waste.

In all situations, the most important contribution to the overall burdens was associated with the sorting and recycling processes, while collection and transport were negligible (Figure 3a,b,c). The burdens from sorting were mainly due to the emission of CO_2 following incineration of the sorting residues (i.e. the share of polymer waste entering the plant that is not captured and becomes a 'system loss'), rather than to sorting operations as such. This was especially relevant for LDPE, PS, and PP due to the comparatively lower sorting rate assumed in the study (from 65% to 73%). In this respect, it should be noted that the sorting rates are based on the findings of Antonopoulos et al. (2021), where typical existing EU sorting plants have been investigated and polymer-specific sorting rates quantified. Hence, such study reflects the *status quo* in EU, without forward looking to best available techniques or technologies that may be implemented in the future.

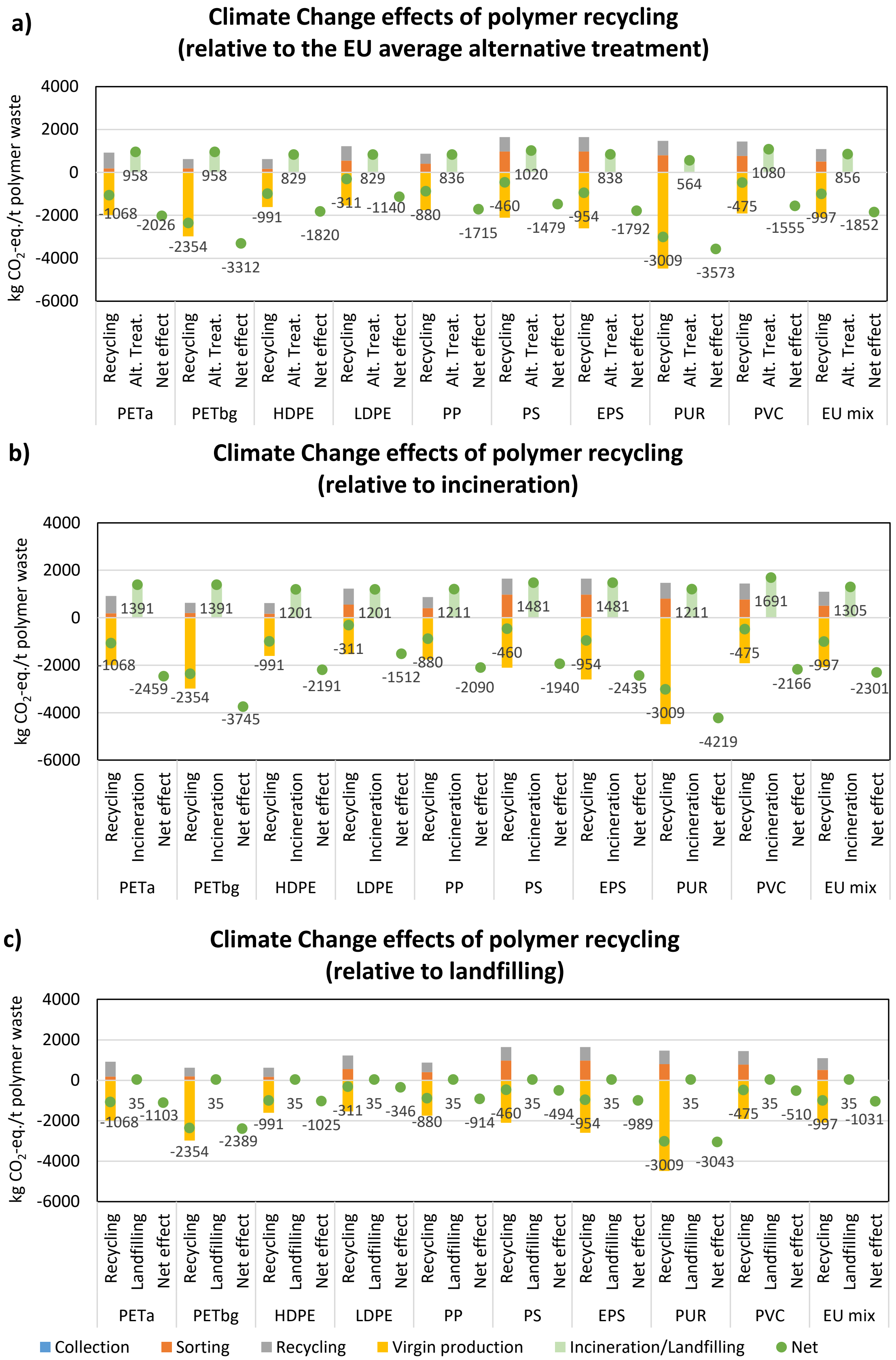

Figure 3. Effects on the Climate Change impact indicator following recycling of 1 additional t of post-consumer polymer waste relative to: a) the average alternative treatment scenario (i.e. the EU mix of incineration and landfilling), b) 100% incineration, c) 100% landfilling of the same polymer. Negative values represent savings, while positive ones represent burdens. Detailed results are available in Annex 5 (contributions breakdown).

The most important contribution to the overall savings was associated with the virgin material substitution (directly incurred by recycling), followed at a system level by the avoided treatment of the waste (Figure 3a,b,c; see Annex 5 for numerical breakdown). The latter was more relevant when incineration was the avoided treatment, owing to the fossil CO_2 emissions otherwise occurring from plastic combustion (Figure 3b; see Annex 5 for numerical breakdown). Instead, when landfilling was the avoided treatment, the associated GHG savings were negligible, for the reason discussed above (i.e. negligible CO_2 emissions from degradation of the investigated polymers in landfills; Figure 3c). Avoidance of the current EU average treatment scenario has a lower benefit compared with avoiding 100% incineration due to the presence of a certain share of landfilling in the mix (see Table 1).

5.1.2 Other environmental impact categories

The net effect of recycling one additional tonne of plastic waste showed environmental savings in the majority of the assessed environmental impact categories, when considering the EU average alternative treatment scenario (Table 4; Annex 6 shows the same results expressed as percentage of the virgin production impacts[11]). Yet, some exceptions were observed where recycling incurred an increased environmental impact compared with the average alternative treatment. This was the case for the categories Ozone Depletion (all polymers), Ionising Radiation (all except for PETbg, LDPE, EPS, PUR, and PVC), Eutrophication – freshwater (PETbg), Land Use (PETa, PETbg, HDPE, PP, and PS) and Resource Use - fossils (PETbg). In all cases, the responsible for such trade-off was the contribution of the recycling process, which outweighed the benefits from avoided virgin polymer production and displaced alternative treatment or disposal of the waste. As pinpointed in Nessi et al. (2020), this may also be a consequence of discrepancies and inconsistencies between the datasets used to represent recycling and virgin polymer production (which are partly based on different data sources). In view of this, all results should be interpreted and used carefully.

Table 4. Net environmental effects (savings/burdens at system level) of recycling 1 additional t of polymer waste in place of the EU average alternative treatment. Negative values (in green) represent savings, while positive ones (in red) represent burdens.

Indicator	**PETa**	**PETbg**	**HDPE**	**LDPE**	**PP**	**PS**	**EPS**	**PUR**	**PVC**	**EUmix**
Climate Change (kg $CO_{2\ eq}$)	-2026	-3312	-1820	-1140	-1715	-1479	-1792	-3573	-1555	-1852
Ozone Depletion (kg CFC-11 $_{eq}$)	1E-06	2E-05	3E-06	8E-07	2E-06	8E-07	8E-07	1E-06	9E-07	3E-06
Human Toxicity – cancer (CTUh)	-1E-05	-2E-05	-2E-05	-2E-05	-2E-05	-2E-05	-3E-05	-2E-05	-1E-05	-2E-05
Human Toxicity - non-cancer (CTUh)	-1E-04	-3E-05	-7E-05	-6E-05	-7E-05	-7E-05	-9E-05	-6E-04	-8E-05	-1E-04
Particulate Matter (Disease incidence)	-5E-05	-2E-05	-1E-05	-1E-05	-1E-05	-2E-05	-3E-05	-9E-05	-3E-05	-3E-05
Ionising radiation (kBq U^{235}-$_{eq}$)	70	-3	128	-8	58	27	-23	-473	-150	-30
Photochemical Ozone Formation kg NMVOC $_{eq}$	-2.7	-3.3	-3.0	-2.5	-3.0	-3.2	-27.2	-8.4	-5.4	-4.7
Acidification mol H+ $_{eq}$	-2.5	-3.0	-1.9	-2.5	-2.5	-3.4	-4.6	-8.7	-3.5	-3.3
Eutrophication – terrestrial (mol N $_{eq}$)	-7.7	-9.2	-4.7	-5.6	-5.9	-8.0	-10.4	-24.8	-10.0	-8.5
Eutrophication – freshwater (kg P $_{eq}$)	-8E-03	1E-02	-1E-02	-9E-03	-1E-02	-8E-03	-1E-02	-3E-02	-9E-03	-1E-02
Eutrophication – marine (kg N $_{eq}$)	-0.7	-0.6	-0.4	-0.5	-0.5	-0.7	-1.0	-2.4	-0.9	-0.8
Ecotoxicity Freshwater (CTUe)	-400	-150	-397	-320	-402	-382	-498	-447	-182	-352
Land Use (pt)	246	6094	2505	-1006	2,737	454	-388	-17534	-4586	-1160
Water Use (m^3 world $_{eq}$)	-609	-881	-317	-277	-286	-366	-396	-873	-290	-398
Resource Use - minerals and metals (kg Sb $_{eq}$)	-4E-01	3E-04	-2E-05	-1E-04	-7E-05	-1E-04	-2E-04	-6E-04	-2E-04	-2E-02
Resource Use – fossils (MJ)	-4E+04	-7E+04	-5E+04	-5E+04	-5E+04	-6E+04	-8E+04	-9E+04	-4E+04	- 5E+04

[11] For consistency with calculation of percentage variations for results related to the product perspective (section 5.2). However, notice that for the system perspective the savings are much higher because the benefits associated with the avoided alternative treatment are also accounted for in the calculation of the percentage.

5.2 Environmental effects of using recycled material (product perspective)

This section presents the results for the product perspective (i.e. use of recycled polymer instead of virgin material) focusing on the Climate Change impact category (section 5.2.1). An overview of the effects on the remaining environmental impact categories assessed is presented in section 5.2.2. Throughout figures and tables the results are expressed per tonne (t) of recycled polymer used. Notice that the results are also reported for the average EU mix of plastic polymers investigated in this study ("EU mix"), calculated according to the market shares of the same (based on PlasticsEurope, 2020; see Table 6). Negative results indicate environmental savings while positive results indicate environmental burdens. Additional details may be found in Annex 7-Annex 8.

5.2.1 Climate Change

For all the investigated polymers, replacement of virgin with recycled material use proved to be beneficial on Climate Change. The savings observed relative to the virgin alternative varied from 147 (LDPE) to 1 493 (PUR) kg CO_2-eq./t recycled polymer; this corresponded to a relative saving of 10% (LDPE), 33% (PUR) and up to 39% (PETbg) (Figure 4 and Annex 8). Overall, the benefits were larger for PETbg and PUR, due to the higher avoided impacts of the respective virgin production processes, compared with the remaining polymers, and to the higher substitution ratio between virgin and recycled material assumed for PETbg (1:1). The lowest saving was observed for LDPE, due to the lower virgin material substitution considered (1:0.75) with respect to the other polymers (1:0.9 or 1:1) and reduced sorting efficiency (73% vs e.g. 91% for PET and HDPE) implying higher incineration burdens from treatment of sorting losses. Savings were moderate for PS and PVC (219 - 229 kg CO_2-eq./t polymer) due to the combination of comparatively lower avoided virgin production impacts and lowest sorting efficiency (65%). Finally, PETa, HDPE, PP and EPS showed comparable savings in the range of 433 (PP) – 530 (PETa) kg CO_2-eq./t recycled polymer. For the average EU mix of these polymers, the savings amounted to 491 kg CO_2-eq./t recycled polymer. It should be borne in mind that the benefits calculated for LDPE, PS, PVC and especially EPS and PUR are more uncertain because of the data limitations, especially related to the respective recycling and sorting processes (see sections 4.3.2 and 4.3.3).

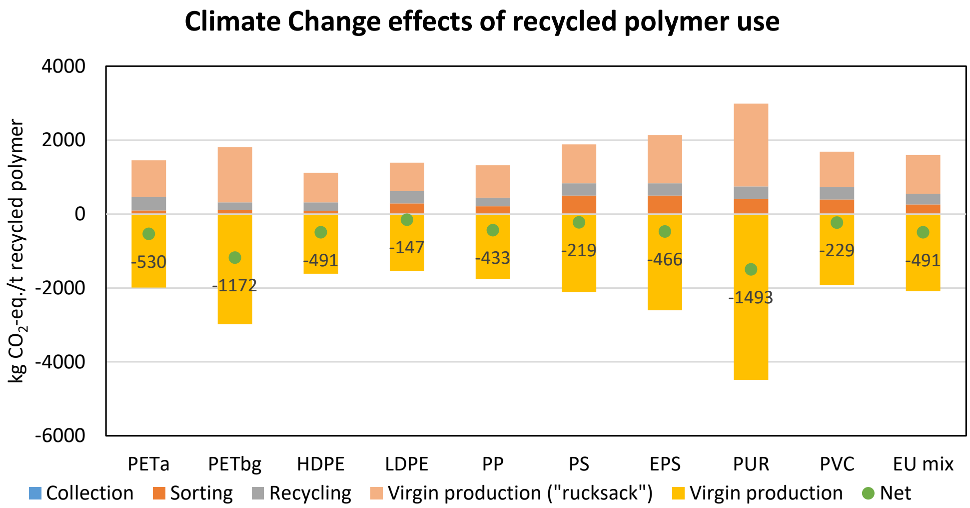

Figure 4. Effects on the Climate Change impact indicator following use of 1 t of recycled polymer in place of an equivalent amount of virgin polymer. Negative values represent savings, while positive ones represent burdens. Detailed results are available in Annex 7 (contributions breakdown).

The most important contribution to the overall burdens from recycled polymer use in place of virgin material was associated with the virgin production "rucksack", i.e. the share of virgin polymer production impact assigned to the user of the recycled material (see section 4.2.3). Sorting and recycling generally had a lower contribution to the overall burdens, while collection was negligible as it is the case of sorting for PETa, PETbg and HDPE, which have the highest sorting efficiencies, and hence lower burdens from incineration of losses. The burden of sorting was indeed mainly due to the emission of CO_2 following incineration of the sorting residues (i.e. non-captured polymer waste that ends up as a system loss). This was more relevant for PS, EPS, PUR PVC and LDPE, due to the relatively lower sorting efficiency assumed in the study for these polymers. It should be noticed, however, that the sorting rates are based on the findings of Antonopoulos et al. (2021), where typical existing EU sorting plants have been investigated under current operational conditions. The only contribution to the overall savings was associated with avoided virgin material production, as the current formulation of the CFF (applied to calculate the total impact of recycled polymer supply) does not assign any credits for the avoided disposal of the recycled plastic waste during its previous life cycle.

5.2.2 Other environmental impact categories

Similarly to what observed earlier for the waste management perspective (section 5.1.2), using recycled polymer instead of virgin material incurred savings in the majority of the environmental impact categories considered in the study (Table 5). The savings ranged from 7% (Land Use for PETa) to 202% (Acidification for PS) relative to virgin polymer production. Yet, some exceptions were observed where the use of recycled material incurred increased environmental burdens compared with the replaced virgin polymer. This was the case for the categories Ozone Depletion (all polymers), Eutrophication – freshwater (PETbg), Land Use (PETbg, HDPE, and PP) and Resource use – minerals and metals (PETbg). In all cases, the responsible for such trade-off was the contribution of the recycling process, which outweighed the benefits from avoided virgin polymer production. As pinpointed in Nessi et al. (2020), this may also be a consequence of discrepancies and inconsistencies between the datasets used to represent recycling and virgin polymer production (which are partly based on different data sources). Having this in mind, all results should be interpreted and used carefully.

Table 5. Environmental effects (savings/burdens) of using 1 t of recycled polymer in place of an equivalent amount of virgin polymer. Negative values (in green) represent savings, while positive ones (in red) represent burdens.

Indicator	PETa	PETbg	HDPE	LDPE	PP	PS	EPS	PUR	PVC	EU mix
Climate Change (kg $CO_{2\ eq}$)	-530	-1172	-491	-147	-433	-219	-466	-1493	-229	-491
Ozone Depletion (kg CFC-11 $_{eq}$)	8E-07	1E-05	4E-06	1E-06	2E-06	2E-06	2E-06	2E-06	2E-06	3E-06
Human Toxicity – cancer (CTUh)	-7E-06	-9E-06	-2E-05	-2E-05	-2E-05	-2E-05	-3E-05	-2E-05	-1E-05	-2E-05
Human Toxicity - non-cancer (CTUh)	-7E-05	-2E-05	-7E-05	-7E-05	-8E-05	-8E-05	-9E-05	-6E-04	-2E-05	-1E-04
Particulate Matter (Disease incidence)	-3E-05	-1E-05	-3E-05	-3E-05	-3E-05	-4E-05	-4E-05	-9E-05	-3E-05	-4E-05
Ionising radiation (kBq U^{235}-$_{eq}$)	-10	-46	-59	-194	-137	-139	-158	-552	-212	-178
Photochemical Ozone Formation kg NMVOC $_{eq}$	-1.6	-1.9	-4.2	-3.7	-4.3	-4.1	-27.8	-7.5	-5.4	-5.3
Acidification mol H+ $_{eq}$	-1.7	-1.9	-3.9	-4.5	-4.5	-4.9	-5.8	-8.5	-3.8	-4.5
Eutrophication – terrestrial (mol N $_{eq}$)	-4.5	-5.2	-8.4	-9.0	-9.4	-10.1	-11.9	-20.3	-9.3	-9.9
Eutrophication – freshwater (kg P $_{eq}$)	-1E-03	9E-03	-5E-03	-4E-03	-5E-03	-3E-03	-4E-03	-2E-02	-3E-03	-5E-03
Eutrophication – marine (kg N $_{eq}$)	-0.4	-0.4	-0.8	-0.9	-0.9	-1.0	-1.2	-2.0	-0.9	-0.9
Ecotoxicity Freshwater (CTUe)	-205	-80	-425	-348	-431	-405	-513	-447	-179	-357
Land Use (pt)	-221	2708	548	-2896	721	-1142	-1629	-18054	-4179	-2639
Water Use (m^3 world $_{eq}$)	-245	-381	-167	-126	-135	-221	-276	-784	-178	-232
Resource Use - minerals and metals (kg Sb $_{eq}$)	-2E-01	1E-04	-1E-04	-2E-04	-2E-04	-2E-04	-3E-04	-6E-04	-2E-04	-1E-02
Resource Use – fossils (MJ)	-2E+04	-4E+04	-7E+04	-6E+04	-7E+04	-8E+04	-9E+04	-1E+05	-5E+04	-7E+04

5.3 Total annual GHG savings from recycling

The total annual Climate Change effects (reflecting GHG savings) from recycling are calculated based on the results from the waste management perspective, i.e. accounting for the savings from avoiding current incineration or landfilling and for those associated with recycling the material (i.e. substituting virgin material production). The methodological details may be found in section 4.4. Knowing the current collection rates for recycling, as well as sorting and recycling rates for each of the polymers investigated (Table 1 for collection rates, Table 2 for sorting and recycling rates), the amount of plastic waste currently sent for recycling and of secondary raw material produced can be estimated for each polymer type and as a whole (Table 6; columns #4 and #5). This corresponds to a total annual GHG saving potential of ca. 18.5 Mt CO_2-eq./y under the assumption that the plastic waste today collected and sent for recycling would otherwise have been routed to landfilling and incineration following current practices and rates (Table 6; column #6). The amount of plastic waste that can be additionally collected, prepared and sent for recycling can be quantified by difference between the amount generated and that currently recycled (Table 6; column #7), further corrected by applying a capture rate of 70% as suggested in Triconomics (2020) on the basis of a research performed by the Nordic Council of Ministers (2014). Applying the sorting and recycling rates, it is then possible to calculate the amount of secondary raw material that can be additionally produced (Table 6; column #8). This corresponds to an annual GHG saving potential of ca. 17.6 Mt CO_2-eq./y (Table 6; column #9), which is to be considered additional to what already achieved today, estimated to ca. 18.5 Mt CO_2-eq./y, with the current performances, as reported above. It should be noticed that the following assumptions apply to the calculated additional saving:

- This result is true under the assumption that 70% of the investigated polymer waste[12] currently non-separately collected is instead separately collected, prepared and sent for recycling. The capture rate for plastic waste is based on the figure suggested in Triconomics (2020) (originally provided by the Nordic Council of Ministers, 2014). Notice such 70% figure includes both the limits due to collection and to the technically non-recyclable plastic items.
- A constant (non-dynamic) system is assumed, i.e. no changes in the energy systems (electricity and heat mix) or in waste generation, collection systems/infrastructure and waste treatment technologies (i.e. collection, sorting, and recycling rates) are considered.

Notice that the latter assumption incurs an underestimate of the savings from recycling because it does not take into account: i) the fact that the Climate Change effects of incineration would worsen because the EU electricity and heat mix are likely to become less CO_2-intensive with time (therefore the energy recovered at incineration facilities would offset less GHG emissions from the process itself, or none, and this would increase the overall system-wise GHG savings when diverting waste material from incineration to recycling) and ii) the improved performance of sorting and recycling technologies over time (mainly reduced system losses thanks to improved efficiencies). Further, considering a hypothetical 100% capture rate (instead of the 70% assumed following Triconomics, 2020), the full theoretical GHG saving potential would equal 25 Mt CO_2-eq./year (assuming a waste generation of 29.1 Mt for EU28+NO+CH as for year 2018; PlasticsEurope 2020).

[12] The investigated polymer waste constitute 81% of the total polymer waste generated. The remaining 19% is classified by PlasticEurope (2020) as "Others", i.e. a mix of different polymers that is today not recycled and sent to landfilling/incineration. In this study, we assume that this fraction will be sent to incineration or disposal also in the future.

Table 6. Estimate of the annual Climate Change mitigation potential from current and additional plastic waste recycling at the EU level; "Others" are other plastics not considered in this assessment (e.g. hub caps made of Acrylonitrile Butadiene Styrene, optical fibres, roofing sheets made of polycarbonate, coatings, touch screens, medical implants); nr: not relevant or not considered in this study.

#1 Polymer	#2 Share (% plastic demand)[a]	#3 Waste generated (Mt/y)[b]	#4 Waste currently collected and sent for recycling (Mt/y)[c]	#5 Secondary Raw Material currently produced (Mt/y)[d]	#6 Climate Change effect from current recycling (Mt CO_2-eq./y)[e]	#7 Waste that can be additionally collected and sent for recycling (Mt/y)[f]	#8 Secondary Raw Material additionally produced (Mt/y)[g]	#9 Climate Change effect from additional recycling (Mt CO_2-eq./y)[h]
PETa	4%	1.1	0.7	0.5	-1.4	0.3	0.2	-0.6
PETbg	4%	1.1	0.7	0.5	-2.3	0.3	0.2	-1.0
HDPE	12%	3.6	2.2	1.7	-4.1	0.9	0.7	-1.7
LDPE	18%	5.1	2.5	1.6	-2.9	1.8	1.1	-2.0
PP	19%	5.6	3.0	2.0	-5.2	1.8	1.2	-3.1
PS	3%	0.9	0.4	0.2	-0.6	0.4	0.2	-0.5
EPS	3%	0.9	0.2	0.1	-0.4	0.5	0.3	-0.9
PUR	8%	2.3	0.0	0.0	0.0	1.6	0.9	-5.7
PVC	10%	2.9	0.9	0.6	-1.5	1.4	0.8	-2.1
Others	19%	5.53	nr	nr	nr	nr	nr	nr
Total	100%	29.1	10.8	7.3	**-18.5**	8.9	5.6	**-17.6**

(a) The share of each polymer on the total plastic waste annually generated in EU27 was not available; therefore, the share relative to the total plastic demanded by converters in the broader area of EU28+NO+CH (PlasticsEurope 2020) was used as a proxy. This might not necessarily correspond to the actual annual EU plastic waste composition. As the individual shares of PETbg and PETa were not available, we assumed that 50% of the total PET demand consists of PETbg and 50% of PETa.

(b) Obtained multiplying the share of each polymer (column #2) by the total plastic waste generated annually in EU (assumed equal to 29.1 Mt for EU28+NO+CH in the year 2018; PlasticsEurope 2020).

(c) Obtained multiplying the amount of polymer generated as waste (column #3) by the collection rate for recycling (Table 1, column #2), which are based on a recent JRC study on post-consumer plastic packaging waste (Antonopoulos et al., 2021).

(d) Obtained multiplying the amount of polymer waste collected and sent for recycling (column #4) by the sorting and recycling rates (Table 2; column #3 and #4), which are based on a recent JRC study on post-consumer plastic packaging waste (Antonopoulos et al., 2021).

(e) Obtained multiplying the amount of polymer waste currently collected and sent for recycling (column #4) by the total Climate Change effect from recycling (Table 4; EU-average alternative treatment route). It is hence assumed that the plastic waste today collected and sent for recycling would otherwise be treated through landfilling and incineration following current practices.

(f) Obtained as difference between the amount of polymer waste generated and that currently collected and sent for recycling (column #3 minus column #4), corrected by applying a capture rate of 70% based on the figures reported in Triconomics (2020).

(g) Obtained multiplying the amount of polymer waste that can be additionally collected and sent for recycling (column #7) by the sorting and recycling rates as reported in Table 2 (columns #3 and #4), which are based on a recent JRC study on post-consumer plastic packaging waste (Antonopoulos et al., 2021).

(h) Obtained multiplying the amount of polymer waste that can be additionally collected and sent for recycling (column #7) by the total Climate Change effect from recycling (Table 4; EU-average alternative treatment route).

As the annual GHG emissions of EU27 was quantified to 3 893 Mt CO_2-eq. in year 2018 (including those from international aviation and excluding LULUCF; EEA 2020), the estimated potential saving from additional plastic waste recycling (17.6 Mt CO_2-eq./y) corresponds to a reduction potential in the order of 0.5% of the total annual GHG emissions of EU27, equivalent to the average annual impact of 2.3 million of EU citizens (assuming a normalisation factor of 7 760 kg CO_2-eq/person-eq./year). Considering a hypothetical 100% capture rate, the theoretical GHG saving potential of 25 Mt CO_2-eq./year corresponds to the average annual impact of 3.2 million of EU citizens or 0.65% of the total annual GHG emissions of EU27 using 2018 as reference year. Such figures should be used carefully as we did not take into account possible changes in the future EU energy system, and waste composition and treatment technology.

6 Limitations and perspectives

The results show that, overall, a net reduction of the potential Climate Change impact in the order of 1 140-3 573 kg CO_2-eq./t polymer waste can be achieved when additional polymer recycling is implemented in place of current alternative treatment routes including incineration and landfilling. Benefits are higher (1 512-4 219 kg CO_2-eq./t polymer waste) when recycling displaces incineration only (here assumed with energy recovery) thanks to the avoided CO_2 emissions from combustion of fossil carbon in polymers. While calculation approaches and background data applied may differ, these results are mostly aligned with the (recent) scientific literature. For example, Faraca et al. (2019) quantified GHG savings for advanced mechanical recycling of a mix of hard plastics (PP, PET, PE, and PS) in the order of 750 kg CO_2-eq./t waste, when accounting only for the benefits from avoided virgin material production (i.e. without considering the additional savings relative to the displaced alternative treatment pathways, e.g. incineration and/or landfilling). Under the same assumption, Astrup et al. (2009) estimated savings from recycling in the order of 838-1 574 kg CO_2-eq./t mix hard plastic waste. The range mainly depended on the assumptions regarding the fuel mix applied to represent production of electricity and heat used throughout the recycling chain and avoided virgin material production. When including the savings from avoided incineration (or landfilling or a mix) the results of these studies are largely aligned with the present one.

The additional estimates presented in this study for the savings associated with recycled polymer use in place of virgin material (product perspective) provides an indication of the savings attributable to the market users of recyclates. These figures may be used when dealing with product comparisons, where the alternative fate of the waste (i.e. the alternative treatment avoided) is beyond the scope of the analysis and thus not accounted for. In this case, the savings estimated ranged from 147 to 1 493 kg CO_2-eq./t recycled polymer used. However, it should be kept in mind that such figures are calculated following the so-called Circular Footprint Formula (CFF) adopted in the PEF context to partition the burdens/savings of recycling and the impacts of virgin production between two subsequent product life cycles (i.e. between the supplier and user of recycled material). Allocation factors (i.e. the so-called "A factor", equalling 0.5 for plastic polymers) are hence applied for the estimate (see section 4.2.3). At a system-wide (societal) level such allocation is not necessary, so that for the overall system-wide savings of recycling the reader should rather refer to the former approach.

The savings calculated in this study assume that the substitution ratio between recycled and virgin materials is 90% for the majority of the polymers investigated, except LDPE (75%) and bottle-grade PET (100%). However, this value is uncertain, as it clearly depends on the quality of the recycled polymer, which may vary according to the applied sorting and recycling technologies, and its ultimate performance relative to the material actually substituted in the appropriate market segment or, more often, in the specific application (Faraca et al., 2019; Rigamonti et al., 2020; Vadenbo et al., 2017). While approaches and frameworks to determine the technical substitutability have been proposed for different materials (notably Rigamonti et al., 2020; Vadenbo et al., 2017), a broad consensus on the specific values to be applied for recycled polymers still does not exist. For example, Gala et al. (2020) considered a technical substitution coefficient of 75% for HDPE. Faraca et al. (2019) applied 95% for PET, 91% for PE, 83% for PP and 66% for PS assuming that these polymers were obtained with advanced sorting and recycling technologies delivering high-quality outputs. Sevigné-Itoiz et al. (2015) considered a technical substitutability of 86% for a mix of plastic waste recycled in Spain. Astrup et al. (2009) assumed a generic substitutability of 80% for all the recycled plastics based on a Danish market report dated 2006. It is clear that, the lower the technical substitutability, the lower the potential environmental savings associated with the recycling pathway, since in principle more recycled material is needed to provide the same function as the replaced virgin material.

Another limitation of this study lies on the quality (representativeness) of some of the life cycle inventory data applied. In particular, no specific data were available for recycling of LDPE, PS, EPS, PUR, and PVC, and generic data representing production of unspecified secondary thermoplastic granulate from sorted post-consumer plastic waste were used as a proxy. While this approximation is considered acceptable for thermoplastic polymers such as LDPE, PS and PVC, which can be recycled through traditional operations represented by the applied data, this is not the case of EPS and especially PUR, which are recycled through different processes involving alternative and/or additional operations (also including possible removal of additives such as flame retardants). In addition, the data applied for plastic waste sorting operations also relies on generic literature data that are mostly representative of sorting of polymer streams traditionally collected as part of municipal waste (e.g. PET, HDPE, LDPE, PP and PS), while they poorly represent sorting of EPS, PUR and PVC waste, which is mainly generated in other sectors (e.g. building and construction). Similar considerations apply to the data used to represent the collection and transport for these polymers, which are mostly representative of municipal plastic waste collection, although this limitation only marginally affects the environmental results. With this in mind, the results presented for PUR, EPS, and to a lower extent, PVC, PS and LDPE, should be interpreted and used with caution. It shall also be noted that the sorting rates applied refer to existing material recovery facilities in EU and certainly do not reflect best available technologies that could be implemented in the future in the same region. However, while this should be kept in mind when using the results, it has to be noted that similar assumptions were consistently followed for incineration, landfilling and collection, where existing technologies were applied, representing therefore an average *status quo* of plastic waste management in the EU. On this basis, forward-looking studies should therefore pay attention to consistently model state-of-the-art techniques and efficiencies for all the waste treatments involved.

In a broader EU perspective, considering an annual generation of plastic waste in the EU of about 29.1 Mt (PlasticsEurope 2020)[13], a total annual GHG saving potential of nearly 17.6 Mt CO_2-eq. could be estimated by diverting to recycling the portion of the investigated polymers that is currently incinerated or landfilled. This saving is additional to what already achieved today with current recycling performances, and corresponds to a reduction potential in the order of 0.5% of the total annual GHG emissions of EU27 in 2018, equivalent to the average impact of 2.3 million of EU citizens. These results are true under the assumption that: i) 70% of the investigated polymer waste currently non-collected separately is instead captured and sent for recycling, and ii) the system does not change, for instance considering constant treatment efficiencies (i.e. sorting and recycling rates as of today), a stable mix of energy sources for electricity and heat, and a waste generation equal to that of year 2018. Having this in mind, these figures represent a preliminary estimate of the overall GHG saving potential and should be used carefully.

[13] Data for EU28+NO+CH in year 2018, in the absence of better information for EU27.

7 Conclusions

This study quantified the Climate Change and other environmental effects from recycling of a broad set of plastic polymers at the EU level. Investigated polymers include Polyethylene Terephthalate amorphous (PETa), Polyethylene Terephthalate bottle-grade (PEbg), High-density Polyethylene (HDPE), Low-density Polyethylene (LDPE), Polypropylene (PP), Polystyrene (PS), Expanded Polystyrene (EPS), Polyurethane (PUR), and Polyvinylchloride (PVC). The study largely builds on previous research from the Joint Research Centre. Two different perspectives were considered to provide a broader overview of the consequences from recycling, i.e. a system-wide (or waste management) perspective (overall effects from recycling additional plastic waste instead of incineration and/or landfilling), and at a product-level (effects attributable to the user of recycled polymer in place of virgin material). All in all, in respect to climate change mitigation potential, GHG savings in the order of about 1 140-3 573 kg CO_2-eq./t polymer waste can be achieved when additional recycling is implemented in place of current alternative treatments including a combination of incineration and landfilling. An average figure of 1 852 kg CO_2-eq./t plastic waste was quantified based on the market shares of these polymers in the EU. Savings are highest for recycling of PUR and bottle-grade PET, and when recycling displaces the sole incineration route, due to the avoided CO_2 emissions from combustion of fossil carbon in polymers. When focusing on recycled polymer use by manufacturers, GHG savings in the order of 147-1 493 kg CO_2-eq./t recycled polymer were quantified compared with the use of replaced virgin material. Similarly to the former approach, savings were highest for recycled PUR and bottle-grade PET. While it may be observed that the savings quantified with the second approach are lower than the former, the two sets of results should not be directly compared as they are achieved with the aim of answering different questions (hence based on different functional units, system boundaries and calculation approaches). Nevertheless, while applicable to different contexts, the two approaches and associated results should be seen as complementary.

The limitations of this study are mainly related to the substitution factors assumed between recycled and virgin material and to the data used to represent some of the recycling and sorting processes. For the first, while different frameworks have been recently proposed in the literature, no broad consensus exists on the values to be used for plastic waste materials and default values from the Product Environmental Footprint method were applied, considering a substitution of 90% for most of the investigated polymers (i.e. all except LDPE - 75% and bottle-grade PET - 100%). It should however be noticed that such figures lie in the higher end of the range proposed by scientific literature (ca. 66-95%). As for the data, no specific recycling datasets were available at the time of the study for LDPE, PS, EPS, PUR and PVC and reasonable proxies were used instead. However, such proxies are poorly representative of EPS and PUR recycling and the benefits calculated thereof are thus uncertain. Similarly, sorting of EPS, PUR and PVC may not be adequately represented by the applied data. It is therefore desirable that future studies focus on these aspects to improve the robustness and reliability of the results.

In a broader perspective, considering that the annual generation of plastic waste in the EU is estimated to about 29.1 Mt, a total (additional to what achieved today with current recycling) annual GHG saving potential of nearly 17.6 Mt CO_2-eq. can be estimated under the assumption that 70% of the investigated polymers waste currently non-collected (i.e. landfilled or incinerated) is instead collected and sent for recycling, and assuming constant conditions for technology efficiencies and energy systems (as of today). This corresponds to the average impact of 2.3 million of EU citizens. Under the hypothetical scenario that 100% of the investigated polymer waste were collected for recycling, the annual GHG savings would reach about 25 Mt CO_2-eq. The results of this study are highly relevant for circular economy policies related to plastics and for informing how the circular economy can contribute to the objectives of the EU Green Deal, especially in respect to decarbonisation.

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List of abbreviations and definitions

CFF	Circular Footprint Formula
CH	Switzerland
EC	European Commission
EF	Environmental Footprint
EI	Ecoinvent
EoL	End-of-Life
EPS	Expanded Polystyrene
EU	Europe
EU27	Europe 27 member states
EU28	Europe 28 member states
FU	Functional unit
GHG	Greenhouse Gas
GWP	Global Warming Potential
HDPE	High-Density Polyethylene
INC	Incineration
JRC	Joint Research Centre
LCA	Life Cycle Assessment
LDPE	Low-Density Polyethylene
LF	Landfilling
MDI	Methylene Diphenyl Diisocyanate
MRF	Material Recovery Facility
NO	Norway
PE	Polyethylene
PETa	Polyethylene terephthalate amorphous
PEbg	Polyethylene terephthalate bottle grade
PEF	Product Environmental Footprint
PET	Polyethylene Terephthalate
PP	Polypropylene
PS	Polystyrene
PUR	Polyurethane
PVC	Polyvinylchloride
REC	Recycling
TS	Thinkstep

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Annexes

Annex 1. List of environmental impact categories assessed in this study and related impact assessment models (from the PEF method).

Impact Category	Impact Assessment Model	Unit
Climate change	IPCC, 2013	kg $CO_{2\ eq}$
Ozone depletion	World Metereological Organisation (WMO), 2014 and integrations	kg CFC-11 $_{eq}$
Human toxicity, cancer	USEtox 1.01 (Rosenbaum et al., 2008)	CTUh
Human toxicity, non-cancer	USEtox 1.01 (Rosenbaum et al., 2008)	CTUh
Particulate matter	UNEP, 2016	Disease incidence
Ionising radiation, human health	Dreicer et al., 1995; Frischknecht et al., 2000	kBq $U^{235}{}_{-eq}$
Photochemical ozone formation, human health	LOTOS-EUROS (Van Zelm et al., 2008) as in ReCiPe 2008	kg NMVOC $_{eq}$
Acidification	Seppälä et al., 2006; Posch et al., 2008	mol H+ $_{eq}$
Eutrophication, terrestrial	Seppälä et al., 2006; Posch et al., 2008	mol N $_{eq}$
Eutrophication, freshwater	EUTREND (Struijs et al., 2009) as in ReCiPe	kg P $_{eq}$
Eutrophication, marine	EUTREND (Struijs et al., 2009) as in ReCiPe	kg N $_{eq}$
Ecotoxicity, freshwater	USEtox 1.01 (Rosenbaum et al., 2008)	CTUe
Land use	LANCA (based on; Beck et al., 2010; Bos et al., 2016)	pt
Water use	AWARE 100 (based on; UNEP, 2016)	m^3 world $_{eq}$
Resource use, minerals and metals	CML 2002 (Guinée et al., 2002) as updated in van Oers et al., 2002 (ultimate reserves)	kg Sb $_{eq}$
Resource use, fossils	CML 2002 (Guinée et al., 2002) as updated in van Oers et al., 2002	MJ

Annex 2. Modelling of source-separated plastic waste collection and transport for recycling (applies identically to all types of polymer).

Collection type	Share (%)	Distance (km/t)	Vehicle	Share (%)	Dataset (1)	Amount (km*t/t collected waste)
Kerbside	59%	49	Medium/large-sized truck	41%	[EU-28+3] Articulated lorry transport, Total weight 28-32 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | 28 - 32t gross weight / 22t payload capacity	11.9
			Small-sized truck	59%	[EU-28+3] Articulated lorry transport, Total weight <7.5 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | up to 7,5t gross weight / 3,3t payload capacity	17.1
Street collection	29%	48	Medium/large-sized truck	100%	[EU-28+3] Articulated lorry transport, Total weight 28-32 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | 28 - 32t gross weight / 22t payload capacity	13.9
Drop-off areas	12%	2.5	Van (2)	100%	[EU-28+3] Articulated lorry transport, Total weight <7.5 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | up to 7,5t gross weight / 3,3t payload capacity	0.3
Transport						
From collection centres to sorting facilities	100%	50	Large truck	100%	[EU-28+3] Articulated lorry transport, Total weight >32 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | more than 32t gross weight / 24,7t payload capacity	50
From sorting to recycling facilities	100%	50	Large truck	100%	[EU-28+3] Articulated lorry transport, Total weight >32 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | more than 32t gross weight / 24,7t payload capacity	50

(1) From the pool of EF-compliant datasets.
(2) Approximating delivery by car or small vans by citizens.

Annex 3. Modelling of plastic waste collection to incineration and landfilling (as mixed residual waste).

Collection type	Share (%)	Distance(km/t)	Vehicle	Share (%)	Dataset ([1])	Amount (km*t/t collected waste)
Kerbside	71%	15.5	Medium/large-sized truck	100%	[EU-28+3] Articulated lorry transport, Total weight 28-32 t, mix Euro 0-5, diesel driven, Euro 0 - 5 mix, cargo | consumption mix, to consumer | 28 - 32t gross weight / 22t payload capacity	11.0
Street collection	29%	7.5	Medium/large-sized truck	100%		2.2

([1]) From the pool of EF-compliant datasets.

Annex 4. Life cycle inventory of source-separated mixed plastic waste sorting at material recovery facilities (per kg of waste to be sorted). EF: Environmental Footprint; EI: ecoinvent.

Flow	**Amount**	**Unit**	**Dataset**	**Database**
Electricity	0.0458	MJ	[EU-28+3] Electricity grid mix 1kV-60kV; AC, technology mix | consumption mix, at consumer | 1kV - 60kV {34960d4d-af62-43a0-aa76-adc5fcf57246}	EF
Natural gas	1.09×10-4	MJ	[EU-28+3] Thermal energy from natural gas, technology mix regarding firing and flue gas cleaning | production mix, at heat plant | MJ, 100% efficiency {81675341-f1af-44b0-81d3-d108caef5c28}	EF
Diesel	0.00153	kg	[GLO] Diesel combustion in construction machine, diesel driven {dae81b4f-688f-44cd-906b-9435d3843e65}	EF
LPG	0.078	MJ	[GLO] propane, burned in building machine {4dd96eab-d6a2-48d2-a192-ac59e55e0d47}	EI

Annex 5. Climate Change effects (kg CO_2-eq.) of recycling 1 additional t of plastic waste relative to the alternative treatment scenario (system perspective): numerical breakdown of impact contributions.

(A) Alternative treatment scenario: EU average treatment (mix of incineration and landfilling – polymer specific)										
Contribution	**PETa**	**PETbg**	**HDPE**	**LDPE**	**PP**	**PS**	**EPS**	**PUR**	**PVC**	**EUmix**
Collection	9	9	9	9	9	9	9	9	9	9
Sorting	179	185	159	546	394	966	967	793	762	501
Recycling	733	432	452	671	472	671	671	671	671	582
Virgin production	-1989	-2980	-1611	-1538	-1755	-2106	-2601	-4482	-1917	-2089
Net (Recycling)	-1068	-2354	-991	-312	-880	-460	-954	-3009	-475	-997
Alternative treatment (EU average)	958	958	829	829	836	1020	838	564	1080	856
Net effect	-2026	-3312	-1820	-1140	-1715	-1479	-1792	-3573	-1555	-1852

(B) Alternative treatment scenario: incineration										
Contribution	**PETa**	**PETbg**	**HDPE**	**LDPE**	**PP**	**PS**	**EPS**	**PUR**	**PVC**	**EUmix**
Collection	9	9	9	9	9	9	9	9	9	9
Sorting	179	185	159	546	394	966	967	793	762	501
Recycling	733	432	452	671	472	671	671	671	671	582
Virgin production	-1989	-2980	-1611	-1538	-1755	-2106	-2601	-4482	-1917	-2089
Net (Recycling)	-1068	-2354	-991	-312	-880	-460	-954	-3009	-475	-997
Alternative treatment (100% INC*)	1391	1391	1201	1201	1211	1481	1481	1211	1691	1305
Net effect	-2459	-3745	-2191	-1512	-2090	-1940	-2435	-4219	-2166	-2301

(C) Alternative treatment scenario: landfilling										
Contribution	**PETa**	**PETbg**	**HDPE**	**LDPE**	**PP**	**PS**	**EPS**	**PUR**	**PVC**	**EUmix**
Collection	9	9	9	9	9	9	9	9	9	9
Sorting	179	185	159	546	394	966	967	793	762	501
Recycling	733	432	452	671	472	671	671	671	671	582
Virgin production	-1989	-2980	-1611	-1538	-1755	-2106	-2601	-4482	-1917	-2089
Net (Recycling)	-1068	-2354	-991	-312	-880	-460	-954	-3009	-475	-997
Alternative treatment (100% LF*)	35	35	-35	35	35	35	35	35	35	35
Net effect	-1103	-2389	-1025	-346	-914	-494	-989	-3043	-510	-1031

α INC: Incineration; LF: landfilling.
β Calculated as: "Net (Recycling)" minus "Alternative treatment". See also Equation 3.

Annex 6. Climate Change effects (kg CO_2-eq.) of using 1 t of recycled polymer in place of virgin material (product perspective): numerical breakdown of impact contributions.

	PETa	PETbg	HDPE	LDPE	PP	PS	EPS	PUR	PVC	EU mix
Collection	6	6	6	8	7	8	8	8	8	7
Sorting	92	95	82	279	201	490	491	404	386	255
Recycling	367	216	226	336	236	336	336	336	336	291
Virgin production ("rucksack")	995	1490	806	769	878	1053	1301	2241	959	1044
Virgin production	-1989	-2980	-1611	-1538	-1755	-2106	-2601	-4482	-1917	-2089
Net (Recycled material)	-530	-1172	-491	-147	-433	-219	-466	-1493	-229	-491

Annex 7. Environmental effects (savings/burdens) of recycling 1 additional t of plastic waste relative to the EU-average alternative treatment scenario (system perspective), expressed as % of virgin production.

Indicator	PETa	PETbg	HDPE	LDPE	PP	PS	EPS	PUR	PVC	EU mix
Climate Change [kg CO_2 eq.]	-211%	-346%	-220%	-138%	-205%	-145%	-214%	-633%	-144%	-232%
Ozone Depletion [kg CFC-11 eq.]	1850%	31576%	2159%	483%	928%	590%	669%	1485%	1554%	2626%
Human Toxicity - cancer [CTU_h]	-53619%	-66462%	-4583%	-3870%	-4671%	-5777%	-17910%	-6962%	-1700%	-10172%
Human Toxicity - non-cancer [CTU_h]	-7931%	-1954%	-1039%	-974%	-1069%	-1549%	-17303%	-2736%	-128%	-2119%
Particulate Matter [Disease incidence]	-490%	-179%	-72%	-73%	-57%	-122%	-221%	-1448%	-925%	-341%
Ionising radiation [kBq U235 eq.]	78%	-3%	68%	-4%	30%	16%	-17%	-590%	-236%	-67%
Photochemical Ozone Formation [kg NMVOC eq.]	-484%	-595%	-227%	-189%	-234%	-333%	-3546%	-1141%	-8771%	-1529%
Acidification [mol H^+ eq.]	-269%	-317%	-91%	-120%	-122%	-195%	-334%	-9731%	-1049%	-1196%
Eutrophication - terrestrial [mol N eq.]	-513%	-614%	-119%	-141%	-150%	-302%	-508%	-643%	-3608%	-678%
Eutrophication - freshwater [kg P eq.]	-149%	266%	-208%	-190%	-222%	-178%	-164%	-300%	-144%	-180%
Eutrophication - marine [kg N eq.]	-382%	-344%	-85%	-113%	-116%	-237%	-397%	-732%	-10659%	-1512%
Ecotoxicity - freshwater [CTU_e]	-3707%	-1392%	-1328%	-1072%	-1332%	-1476%	-2777%	-12759%	-65769%	-10524%
Land Use [Pt]	33%	815%	124%	-50%	129%	26%	-27%	-2532%	-1660%	-373%
Water Use [m^3 world eq.]	-516%	-746%	-212%	-185%	-190%	-254%	-336%	-999%	-262%	-330%
Resource Use - minerals and metals [kg Sb eq.]	-912281%	616%	-17%	-148%	-65%	-167%	-280%	-1518%	-1539%	-43738%
Resource Use - fossils [MJ]	-461%	-782%	-278%	-259%	-301%	-395%	-594%	-1260%	-949%	-508%

Annex 8. Environmental effects (savings/burdens) of using 1 t of recycled polymer in place of virgin material (product perspective), expressed as % of virgin production.

Indicator	PETa	PETbg	HDPE	LDPE	PP	PS	EPS	PUR	PVC	EU mix
Climate Change [kg CO_2 eq.]	-27%	-39%	-30%	-10%	-25%	-10%	-18%	-33%	-12%	-22%
Ozone Depletion [kg CFC-11 eq.]	3809%	7851%	483%	229%	298%	353%	323%	561%	346%	872%
Human Toxicity - cancer [CTU_h]	-48%	-41%	-98%	-98%	-99%	-99%	-99%	-97%	-91%	-92%
Human Toxicity - non-cancer [CTU_h]	-48%	-20%	-90%	-94%	-94%	-96%	-97%	-95%	-24%	-79%
Particulate Matter [Disease incidence]	-44%	-32%	-90%	-112%	-100%	-123%	-119%	-100%	-83%	-95%
Ionising radiation [kBq U235 eq.]	-9%	-30%	-69%	-121%	-122%	-202%	-181%	-105%	-83%	-103%
Photochemical Ozone Formation [kg NMVOC eq.]	-42%	-41%	-94%	-104%	-100%	-105%	-101%	-81%	-89%	-91%
Acidification [mol H^+ eq.]	-39%	-37%	-93%	-110%	-104%	-116%	-113%	-90%	-84%	-94%
Eutrophication - terrestrial [mol N eq.]	-38%	-38%	-85%	-96%	-92%	-95%	-96%	-71%	-75%	-83%
Eutrophication - freshwater [kg P eq.]	-15%	228%	-65%	-57%	-72%	-54%	-59%	-85%	-42%	-47%
Eutrophication - marine [kg N eq.]	-38%	-30%	-86%	-101%	-95%	-101%	-101%	-74%	-77%	-85%
Ecotoxicity - Freshwater [CTU_e]	-48%	-19%	-88%	-85%	-91%	-88%	-90%	-87%	-69%	-80%
Land Use [Pt]	-7%	71%	25%	-74%	25%	-62%	-70%	-93%	-62%	-25%
Water Use [m^3 world eq.]	-41%	-43%	-53%	-41%	-41%	-50%	-56%	-79%	-53%	-49%
Resource Use - minerals and metals [kg Sb eq.]	-50%	42%	-50%	-99%	-72%	-108%	-107%	-98%	-80%	-75%
Resource Use - fossils [MJ]	-45%	-47%	-103%	-110%	-109%	-113%	-111%	-105%	-95%	-100%

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