

JRC TECHNICAL REPORT

Life Cycle Assessment (LCA) of alternative feedstocks for plastics production

Part 2: illustrative case studies

Nessi S., Sinkko T., Bulgheroni C.,
Garbarino E., Garcia-Gutierrez P.,
Giuntoli J., Konti A., Orveillon G.,
Sanye-Mengual E., Tonini D., Pant R.,
Marelli L., Ardente F.

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Contact information

Name: Fulvio Ardente

Address: European Commission – Joint Research Centre – Via E. Fermi 2749, I-21027 Ispra, Italy.

Email: Fulvio.ARDENTE@ec.europa.eu

Tel.: –

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<https://ec.europa.eu/jrc>

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Foreword

This report is one outcome of the Administrative Agreement (AA) No. 34854-2017 / DG GROW No. SI2.762599 between the Directorate General for Internal Market, Industry, Entrepreneurship and SMEs (DG GROW) and the Joint Research Centre. It describes a number of illustrative Life Cycle Assessment (LCA) case studies on selected plastic products, aimed at demonstrating and evaluating the applicability of the "*Plastics LCA*" method elaborated by the JRC as part of the same AA. The *Plastics LCA* method provides detailed guidelines and methodological rules to conduct as much as possible consistent, reproducible, robust, transparent and verifiable LCA studies of plastic products from different feedstocks (fossil resources, plastic waste, biomass and CO₂ from gaseous effluents) at the EU level. It builds upon and conforms to the EU Product Environmental Footprint (PEF) method, and was developed following a participatory process involving interested stakeholders at different key steps through public consultations. A number of supporting case studies was also conducted to evaluate the applicability of draft methodological requirements and reliability of preliminary results. A selection of these studies was revised based on final requirements and described in this report along with the main lessons learnt and recommendations from the application exercise. The case studies have the main purpose to demonstrate the applicability of the procedural, methodological and modelling rules of the *Plastics LCA* method. Therefore, while the best data and information available to the JRC were applied in the studies, their results are purely illustrative and do not intend to provide conclusive estimates of the potential environmental impacts of the analysed product scenarios. They should also not be used to compare the different scenarios, as this can only be made in the presence of specific Product Environmental Footprint Category Rules (PEFCRs) for the relevant product category (as required by the *Plastics LCA* and PEF methods). The report is mainly addressed to companies and practitioners interested in applying the *Plastics LCA* method, to guide and facilitate them in the application of the main procedural, methodological and modelling rules through illustrative examples.

The information and views set out in this report are those of the author(s) and do not necessarily reflect the official opinion of the Commission.

Abstract

The *European Strategy for Plastics in a Circular Economy* includes actions aimed at investigating and better understanding the lifecycle impacts of using alternative feedstocks for plastics production. The Joint Research Centre has thus developed, upon request from DG GROW¹, a Life Cycle Assessment (LCA)-based method (the "*Plastics LCA*" method) to consistently evaluate the potential environmental impacts of plastic products relying on different feedstocks and derived materials. The method was built upon and fully conforming to the EU Product Environmental Footprint (PEF) method, with the aim of enabling as much as possible consistent, reproducible, robust, transparent and verifiable LCA studies of plastic products at the EU level. The development was supported by a number of illustrative case studies, conducted to evaluate and demonstrate the applicability of the method, and to assess reliability of results. A participatory process was followed, involving interested stakeholders at different key steps. This report describes a selection of the originally developed case studies, which were revised and updated according to the final version of the *Plastics LCA* method, while taking into account relevant stakeholder comments. The main lessons learnt and obstacles encountered are also discussed, and recommendations for improvement are provided. Overall, the case studies are expected to facilitate the application of the procedural, methodological and modelling rules of the *Plastics LCA* method by companies and other interested users.

¹ Directorate General for Internal Market, Industry, Entrepreneurship and SMEs.

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We also thank *Andrea Amadei* for helping with the development of material-specific end-of-life inventories for a number of products.

Finally, we acknowledge the assistance of *Alexandre Bouchet* and *Julien Boucher* from *Environmental Action* to estimate macro- and micro-plastics generation and release through the *PLP method*.

Authors

Simone Nessi

Taija Sinkko

Claudia Bulgheroni

Elena Garbarino

Pelayo Garcia-Gutierrez

Jacopo Giuntoli

Aikaterini Konti

Glenn Orveillon

Esther Sanye-Mengual

Davide Tonini

Rana Pant

Luisa Marelli

Fulvio Ardente

Contribution of authors

The contribution of single authors, as reported in the list above, is as follows.

Simone Nessi coordinated the development of the ten case studies originally conducted to illustrate the second draft of the *Plastics LCA* method, and the revision of the three studies described in this report, to ensure consistency in modelling and reporting. He developed and revised the inventories of the case study on beverage bottles, and of two case studies not included in this report (focusing on food packaging film and printer housing panels). With the contribution of other authors (as specified below), he also developed and revised the inventories for agricultural mulching film and for other products not addressed in this report (flowers/plants pots and stacking chairs). He drafted Sections 1 and 2, developed the description of the beverage bottles case study (Section 3), elaborated the final description of the mulching film case study (Section 4), and drafted the concluding section on the main lessons learnt and recommendations derived from carrying out the studies (Section 6). Finally, he revised the sections developed by other authors (as described below) and prepared interim and final versions of this report for consultation and publication.

Taija Sinkko developed and revised the life cycle models of the product scenarios investigated in the different case studies except those directly developed by other authors (as described below), based on product inventories prepared by authors responsible for each study. She also calculated and elaborated the results of the modelled scenarios, and took care of editing and improving the layout of interim and final versions of this report.

Claudia Bulgheroni contributed to the development of the original inventories and life cycle models for the case study on agricultural mulching film (via data collection and model refinement), and drafted an intermediate version of the respective description (Section 4). She equally contributed to the development of the case study on flowers/plants pots, which is not included in this report.

Elena Garbarino and *Glenn Orveillon* coordinated the initial and final selection of relevant plastic products to be investigated in the case studies (based on specific analyses and stakeholders inputs) and described the applied selection procedure, reported in Annex A.

Pelayo Garcia-Gutierrez developed the case studies on trays for food and automotive interior panels, which are not included in this report.

Jacopo Giuntoli elaborated and described the analysis on the representativeness of the crude oil mix applied for modelling, as reported in Annex C. He also contributed to early drafts of Section 2.6 (on common limitations of the case studies) and Section 2.7 (on value choices, assumptions and normative decisions).

Aikaterini Konti contributed to the development of the case studies on stacking chairs and cleansing wipes (not included in this report), through data collection, refinement of life cycle models, and drafting of the associated descriptions.

Esther Sanye-Mengual refined calculations and results related to the generation and release of macro- and micro-plastics, and drafted the respective sections in each case study.

Davide Tonini developed the case study on building insulation boards and drafted the respective description (Section 5).

Rana Pant, *Luisa Marelli* and *Fulvio Ardente* led, at different stages and to different extents, the project (Administrative Arrangement) where the case studies and this report have been developed.

Executive summary

The *European Strategy for Plastics in a Circular Economy* (COM(2018) 28 final) proposes a vision where innovative materials are developed and alternative feedstocks to fossil resources are used for plastics production, where evidence clearly shows that they are more sustainable compared to traditional, non-renewable alternatives. Alternative feedstocks to oil and gas include plastic waste, biomass and other bio-based resources, and CO₂ from gaseous effluents. The strategy also urges the identification of those applications where the use of plastics with biodegradability properties may provide clear environmental benefits. It hence calls for actions aimed at investigating and better understanding the potential lifecycle impacts of using alternative feedstocks for plastics production, as well as to develop Life Cycle Assessment (LCA) studies seeking to identify any common conditions under which the use of biodegradable or compostable plastics may be beneficial for the environment.

In this context, the Joint Research Centre (JRC) was requested by DG GROW² to elaborate an appropriate LCA-based method to consistently evaluate the potential environmental impacts of plastic products from the different applicable feedstocks and derived materials. A structured and comprehensive methodological framework was thus developed by the JRC (the *Plastics LCA* method³), building upon and fully conforming to the EU Product Environmental Footprint (PEF) method⁴. In line with the objectives of the Environmental Footprint initiative, the *Plastics LCA* method provides detailed guidelines and rules to enable as much as possible consistent, reproducible, robust, transparent and verifiable LCA studies of plastic products at the EU level, based on a common and harmonised framework. To support the development of the method, to test and demonstrate its applicability, and to evaluate reliability of delivered results, a number of illustrative LCA case studies were also conducted for selected plastic products. A participatory process was followed during the development of both the method and the case studies, involving interested stakeholders at different key steps through public technical consultations and calls for data.

Originally, ten illustrative case studies were carried out, based on an advanced draft of the *Plastics LCA* method accounting for the inputs of a first stakeholder consultation in December 2018. However, only a selection of these studies was subsequently updated and revised to reflect the provisions of the final version of the method, and the comments received during a second stakeholder consultation in June 2020. This report thus describes the illustrative case studies accompanying the final version of the *Plastics LCA* method, and the main lessons learnt from the application exercise. Specific obstacles encountered are also discussed, and recommendations to further improve and facilitate future applicability of the method are provided.

The specific products individually assessed in these case studies were beverage bottles, agricultural mulching film, and insulation boards for buildings. For every product, a number of alternative scenarios were analysed, each considering the use of a different feedstock or polymer for the product in scope. Both conventional and alternative feedstocks and materials were considered, mostly reflecting current market availability, in line with the focus of the *Plastics LCA* method on commercially available products. However, a few scenarios also investigated the use of materials or feedstocks not yet applied to real products on the market, to demonstrate the applicability of the rules and procedures of the *Plastics LCA* method also to non-commercially available products, as well as to ensure illustration of all the most relevant methodological and modelling rules of the method itself. The analysed scenarios are described for each case study, along

² Directorate General for Internal Market, Industry, Entrepreneurship and SMEs.

³ Nessi, S., Sinkko, T., Bulgheroni, C., Garcia-Gutierrez, P., Giuntoli, J., Konti, A., Sanye Mengual, E., Tonini, D., Pant, R., Marelli, L., Ardente, F. (2021). *Life Cycle Assessment (LCA) of alternative feedstocks for plastics production - Part 1: the Plastics LCA method*. EUR 30725 EN, Publications Office of the European Union, Luxembourg, ISBN 978-92-76-38145-7, doi:10.2760/271095 (online), JRC125046.

⁴ Commission Recommendation C(2021) 9332 final of 16.12.2021 on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations.

with other relevant case-specific methodological and modelling aspects, as well as major limitations and critical assumptions. These are followed by a presentation of the results and their interpretation, including a number of sensitivity analyses.

Due to the illustrative nature of the case studies, they do not intend to provide conclusive estimates of the potential environmental impacts of the investigated product scenarios, and their results should not be interpreted as such. Moreover, both common and case-specific limitations and critical assumptions described throughout the report have to be properly taken into account in the interpretation. Any direct comparison among the results related to different product scenarios/alternatives should be particularly avoided, as according to the *Plastics LCA* method comparisons can only be made in the presence of specific Product Environmental Footprint Category Rules (PEFCRs) for the relevant product category.

Overall, the *Plastics LCA* method was found to be properly applicable to products representing the most common end-use markets for plastics (packaging, building and construction, agriculture, etc.). The application requires a minimum background and experience in the field of LCA and environmental assessment in general, as well as the support of suitable LCA and spreadsheet software tools. However, the proposed procedural, methodological and modelling rules are considered to actually contribute towards more consistent, reproducible, robust, transparent and verifiable LCA studies of plastic products from different feedstocks at the EU level. On the other hand, a number of aspects that should be improved further were identified, and recommendations were provided to overcome specific obstacles, facilitate future applicability of the method, and to further increase consistency and reproducibility of derived studies.

The main aspects identified for possible improvement and the related recommendations include: (i) the development of Environmental Footprint (EF)-compliant datasets for polymer production from alternative feedstocks (especially bio-based and CO₂-based) and for specific End of Life options applying to products derived from them (especially composting, anaerobic digestion, incineration and landfilling); (ii) the development of EF-compliant datasets for the recycling of a number of conventional polymers, such as PE, PP, and plastics from WEEE⁵ and ELVs⁶ (e.g. ABS and PS); (iii) checking and ensuring full alignment between the modelling rules applied to develop EF-compliant datasets and those provided in the PEF and *Plastics LCA* methods, when releasing updated versions of the different EF data packages; (iv) expand and complement the lists of material-specific and application-specific default values of the A and R₂ factors of the Circular Footprint Formula (CFF)⁷, to cover the entire range of most relevant polymers, applications and/or product categories currently available on the market (to be especially pursued during the development of any specific PEFCRs); (v) methods to quantify macro- and micro-plastics generation and release (e.g. the recommended *PLP method*) should be implemented in existing LCA software tools, to reduce the risk of inconsistencies in calculations, the time needed for these, and to facilitate applicability; (vi) EF-compliant datasets and existing life cycle inventory databases should be complemented with harmonised data and information useful to estimate the associated generation and release of macro- and micro-plastics (and the resulting potential impacts on the environment and human health once suitable methods would be available); (vi) plastic release indicators should be refined to account for the effects of biodegradation, once sufficiently specific, consistent, complete and reliable data would be available for biodegradation rates of single polymers and products in different environmental compartments; (vii) procedures and calculations to identify most relevant impact categories, life cycle stages, processes and elementary flows should be implemented in available LCA software tools or in other dedicated tools to allow for a more controlled and faster application and reduce the risk of errors.

⁵ WEEE: Waste from Electrical and Electronic Equipment.

⁶ ELVs: End-of-Life Vehicles.

⁷ The Circular Footprint Formula is the approach prescribed in the *Plastics LCA* and PEF methods to model the End of Life of products, with a special focus on material and energy recovery from waste products.

1 Introduction

1.1 Context and objectives

In January 2018, the European Commission adopted the *European Strategy for Plastics in a Circular Economy* (COM(2018 28 final) (EC, 2018a), proposing a vision where innovative materials are developed and alternative feedstocks to fossil resources are used for plastics production, to reduce dependency on imported fossil fuels and decrease CO₂ emissions. Alternative feedstocks to oil and gas include plastic waste, biomass and other bio-based resources, and CO₂ from gaseous effluents, and should be used where evidence clearly shows that they are more sustainable compared to traditional, non-renewable alternatives. Moreover, the Strategy also urges the identification of those applications where the use of plastics with biodegradability properties (regardless of the feedstock used for production) may provide clear environmental benefits. Therefore, the Strategy called for actions aimed at investigating and better understanding the potential lifecycle impacts of using alternative feedstocks for plastics production, as well as to develop Life Cycle Assessment (LCA) studies seeking to identify any common conditions under which the use of biodegradable or compostable plastics may be beneficial for the environment.

In this framework, the Joint Research Centre (JRC) was tasked by DG GROW⁸ to elaborate an appropriate LCA-based method to consistently evaluate the potential environmental impacts of plastic products based on the different applicable feedstocks and derived materials. Relevant life cycle-based methods and approaches set out in the existing EU legislation should have been properly taken into account in the development, as well as methodological learnings from updating the Renewable Energy Directive⁹, and any relevant standards related to LCA of plastic products and respective feedstock sources (e.g. the European standard EN 16760¹⁰ and the Technical Report CEN TR 16957¹¹ on LCA of bio-based products and related End of Life inventories).

Following this request, a structured and comprehensive methodological framework, referred to as the *Plastics LCA* method (Nessi et al., 2021), was thus developed by the JRC, building upon and fully conforming to the EU Product Environmental Footprint (PEF) method (EC, 2021). The *Plastics LCA* method provides detailed procedural, methodological and modelling rules to conduct as much as possible consistent, reproducible, robust, transparent and verifiable LCA studies of plastic products from different feedstocks at the EU level, based on a common and harmonised framework. The development of the method was based on a participatory approach, as described in Section 1.2. A number of illustrative LCA case studies on selected plastic products was also conducted to support the development process, with the aim of testing and demonstrating the applicability of the different rules, and evaluating reliability of delivered results.

This report describes the illustrative case studies accompanying the final version of the *Plastics LCA* method, focusing on the most relevant methodological steps and modelling aspects. These include the description of the investigated product scenarios, the definition of the functional unit and the calculation of the corresponding reference flow, setting of the system boundary, the identification of major limitations and critical assumptions, the development of the life cycle inventory of each product scenario (modelling approach, assumptions and data), the presentation of the results (default LCIA results and additional environmental information), and the respective interpretation (including specific sensitivity analyses). The main lessons learnt from the application exercise are also discussed, including the specific obstacles encountered and the

⁸ Directorate General for Internal Market, Industry, Entrepreneurship and SMEs.

⁹ Directive (EU) 2018/2001 of the European Parliament and of the Council of 11 December 2018 on the promotion of the use of energy from renewable sources (recast).

¹⁰ EN 16760:2015 Bio-based products – Life Cycle Assessment.

¹¹ CEN TR 16957:2016 Bio-based products – Guidelines for Life Cycle Inventory (LCI) for the End-of-life phase.

resulting recommendations to improve and facilitate future applicability of the method, as well as to further increase consistency and reproducibility of derived studies.

The different case studies are described individually in Sections 3-5, while their goal and general scope are discussed in Section 2, along with common limitations and other relevant general aspects. The main lessons learnt from conducting the case studies are finally reported in Section 6.

It is important to note that, due to their illustrative nature, the case studies presented in this report do not intend to provide conclusive estimates of the potential environmental impacts of the investigated product scenarios. Moreover, their results are affected by the common limitations discussed in Section 2.6 and by the case-specific limitations and critical assumptions reported in Section #.4 of each case study. These limitations and assumptions have to be properly taken into account in the interpretation of the results of the single studies. Finally, it is reminded that according to the *Plastics LCA* method, comparisons among alternative product scenarios can only be made in the presence of specific PEFCRs (Product Environmental Footprint Category Rules) for the relevant product category. Any direct comparison among different product scenarios/alternatives based on the results presented in this report should thus be avoided.

1.2 Development process

The *Plastics LCA* method and the supporting illustrative case studies presented in this report were developed following a participatory approach, involving interested stakeholders at different stages of the development process. This was made through both dedicated technical consultations (written and/or oral) and calls for data and information. The main process steps are described below, focusing especially on the development of the case studies. For further details on the development of the *Plastics LCA* method, the reader is referred to Section 1.3 of the respective report (Nessi et al., 2021).

A first draft of the *Plastics LCA* method was initially elaborated in May-July 2018, building upon the PEF method (EC, 2013), the most recent version of the associated PEFCR guidance (EC, 2018b), relevant technical standards and legislation, as well as the outcome of a systematic review on selected LCA studies in the field of plastics. The applicability and reliability of this first draft method were subsequently tested in five screening case studies on different plastic products¹², selected by applying a range of criteria to an initial list of relevant candidates (see Section 1.3). Selected products included beverage bottles, flexible food packaging film, agricultural mulching film, insulation boards for buildings, and automotive interior panels. The screening studies were developed based on readily available data and information, while accounting for relevant inputs received during a specific call for data and information open to all interested stakeholders, held in summer 2018. The draft method and the screening case studies were finally submitted to a technical stakeholder consultation to collect feedback on both documents, as well as useful comments and suggestions for improvement. The consultation comprised both a written consultation process (21 November – 19 December 2018), and a stakeholder workshop (held in Brussels on 29-30 November 2018).

Following the consultation process, a revised version of the *Plastics LCA* method was developed during 2019, taking into account the inputs received from stakeholders and the lessons learnt from conducting the screening case studies. The main suggestions for updating the PEF method elaborated in the meantime by Zampori and Pant (2019) were also incorporated. Based on the revised *Plastics LCA* method, ten detailed case studies on specific plastic products were carried out between June 2019 and May 2020, to evaluate and illustrate the practical applicability of the method itself, and to assess reliability of

¹² The screening case studies were described in a separate, interim working document, which had the main purpose to inform the first stakeholder consultation of November-December 2018, and to collect feedback on the document itself. The report was hence not updated further for publication. However, as discussed below, the screening case studies were all amended and developed further into detailed illustrative studies evaluating and demonstrating the applicability of an advanced draft of the *Plastics LCA* method. A selection of these studies is reported in this document.

results. Investigated products included those already covered in the screening case studies (see above), and additional products selected from the initial list of candidates, taking also into account relevant suggestions and comments received from stakeholders during the preceding consultation process (as described in Section 1.3)¹³. A second call for data and information was also carried out in spring 2019, allowing interested stakeholders to comment on the proposed product scenarios and functional units, and to provide useful data and technical information to develop the case studies. Stakeholders were then again invited to provide feedback, comments and suggestions on both the updated *Plastics LCA* method and on the ten illustrative LCA case studies, during a written stakeholder consultation that took place from 3rd to 30th June 2020.

Inputs received from this second stakeholder consultation were analysed and taken into account, as far as appropriate, to elaborate a final version of the *Plastics LCA* method, while aiming at ensuring full compliance with the updated methodological and modelling rules suggested for the PEF method by Zampori and Pant (2019). A selection of the ten original case studies was also revised according to the provisions of the final *Plastics LCA* method, taking into account relevant comments received from stakeholders during the consultation process. The focus of the revision was on those case studies that were expected to be more complete, representative, solid, and consistent at the end of the process. However, particular care was taken in ensuring that the application of all the most relevant procedural, methodological and modelling rules of the *Plastics LCA* method was adequately demonstrated in the selected case studies.

1.3 Selected plastic products and analysed scenarios

The following plastic products were individually assessed in the ten illustrative LCA case studies accompanying the revised version of the *Plastics LCA* method submitted to the second stakeholder consultation of June 2020 (see Section 1.2):

1. Beverage bottles;
2. Food packaging film;
3. Trays for food;
4. Agricultural mulching film;
5. Flowers/plants pots;
6. Insulation boards for buildings;
7. Automotive interior panels;
8. Printer housing panels;
9. Monobloc stacking chairs;
10. Cleansing wipes.

As discussed in Section 1.2, only a selection of the ten original case studies investigating these products was revised according to the provisions of the final version of the *Plastics LCA* method, and relevant comments received from stakeholders during the second consultation. Such case studies are described in this report, and focus on beverage bottles, agricultural mulching film and insulation boards for buildings.

The original selection of specific plastic products to be investigated in the illustrative case studies was made by applying a number of selection criteria to an initial, wider list of candidate products, as described in Annex A. Moreover, relevant suggestions and comments received during the first stakeholder consultation of November-December 2018 were taken into account. The initial list of candidate products was defined based on an extended review of market studies and data, research projects and studies, and of a

¹³ Additionally selected products included trays for food, flowers/plants pots, printer housing panels, monobloc stacking chairs and cleansing wipes.

vast portion of the scientific and technical literature collected for the systematic review of selected LCA studies initially conducted to inform and support the development of a first draft of the *Plastics LCA* method (see Annex K in Nessi et al., 2021). The applied selection criteria included: policy priority; market potential; promise for deployment; availability of data to develop a LCA study on the specific product and quality of available data; applicable End of Life options and related peculiar aspects; relevant product features (durability, type of application¹⁴); and market coverage. A number of sub-criteria were also taken into account, as described in Annex A.

In each case study, a number of product scenarios considering the use of conventional and alternative feedstocks and/or polymers for the investigated plastic product were analysed. These scenarios were defined based on dedicated market and literature surveys, while taking into account the inputs received during the call for specific data and information held in March 2019. The resulting combinations of materials and feedstocks considered in the scenarios analysed in the revised case studies included in this report are summarised in Table 1.1, while further details are available in the descriptions of the single case studies (Sections 3-5). For completeness, Table 2.2 provides an overview of the scenarios originally investigated in the remaining case studies not addressed in this report.

In line with the general scope of the *Plastics LCA* method (which is especially addressed to commercially available products), feedstocks and polymers considered in the different scenarios were mostly selected to reflect the current market situation. However, in some cases, also materials or feedstocks not (yet) applied to real products on the market were considered, to demonstrate the applicability of the procedures and rules of the *Plastics LCA* method also to non-commercially available products (which is relevant wherever a company may intend to conduct an internal study based on company-specific data for the manufacturing process). This was, for instance, the case of Polyethylene Furanoate (PEF) beverage bottles, which were not yet available on the market at the time of the study, but pre-commercial demonstration plants were already in place to produce the polymer. In other cases, the use of a non-commercially applied feedstock or material was considered to ensure illustration of all the most relevant methodological and modelling rules specified in the *Plastics LCA* method, which would have otherwise remained unaddressed. These include, for instance, the rules related to the modelling of the use of captured CO₂ as a feedstock, which were illustrated in scenarios relying on CO₂-based Polyurethane (for insulation boards), and CO₂-based Polypropylene (for food packaging film; not include in this report).

¹⁴ Rigid or flexible application.

Table 1.1. Overview of the materials and related feedstock sources considered in the alternative product scenarios analysed in the LCA case studies described in this report.

Case study / Plastic product	Market Sector	Scenarios					
		Conventional feedstock (fossil-based)		Plastic waste or captured CO ₂ as feedstock		Bio-based or partially bio-based feedstock	
		Material	Feedstock	Material	Feedstock	Material	Feedstock
1 - Beverage bottles	Packaging (rigid)	PET	Crude oil/natural gas	R-PET (24%)	Waste PET (post-consumer)	Bio-PET (30%)	Sugarcane (BR; bio-MEG) Crude oil/natural gas (PTA)
		HDPE		R-HDPE (16%)	Waste HDPE (post-consumer)	Bio-HDPE	Sugarcane (BR)
2 - Agricultural mulching film	Agriculture	LDPE	Crude oil/natural gas	R-LDPE (35%)	Waste LDPE (post-consumer)	TPS/PBAT blend	EU mix of starch crops (TPS) Crude oil/natural gas (PBAT)
						PLA/PBAT blend	Maize (US; PLA) Crude oil/natural gas (PBAT)
3 - Insulation boards for buildings	Building & Construction	PUR	Crude oil/natural gas	CO ₂ -based PUR (6%)	CO ₂ (coal fired power plant) Crude oil/natural gas	Bio-PUR (39%)	Soybean (EU; PO) ⁽¹⁾ Crude oil/natural gas (EO; MDI) ⁽²⁾
		EPS		R-EPS (100%)	Waste EPS (post-consumer)	-	-
		-	-	R-PET	Waste PET (post-consumer)	-	-

⁽¹⁾ PO: Propylene Oxide.

⁽²⁾ EO: Ethylene Oxide; MDI: Methylene diphenyl di-isocyanate.

Table 1.2. Overview of the materials and related feedstock sources considered in the alternative product scenarios analysed in the non-revised LCA case studies originally accompanying the updated draft *Plastics LCA* method (not included in this report).

Case study / Plastic product	Market Sector	Scenarios					
		Conventional feedstock (fossil-based)		Plastic waste or captured CO ₂ as feedstock		Bio-based or partially bio-based feedstock	
		Material	Feedstock	Material	Feedstock	Material	Feedstock
1 - Food packaging film	Packaging (flexible)	PP	Crude oil/natural gas	CO ₂ -based PP	CO ₂ (coal fired power plant) H ₂ (EU-average production mix)	PLA	Maize (US)
						PLA/PBAT blend	Maize (US; PLA) Crude oil/natural gas (PBAT)
						TPS/PBAT blend	EU mix of starch crops (TPS) Crude oil/natural gas (PBAT)
		LDPE		-	-	Bio-LDPE	Sugarcane (BR)
2 - Trays for food	Packaging (flexible)	PET	Crude oil/natural gas	R-PET	Waste PET (post-consumer)	Bio-PET	Sugarcane (BR; bio-MEG) Crude oil/natural gas (PTA)
		PP		-	-	PLA	Maize (US)
						TPS/PBAT blend	EU mix of starch crops (TPS) Crude oil/natural gas (PBAT)
			Bio-PBS			Maize (US)	
3 - Flowers/plants pots	Agriculture	PP	Crude oil/natural gas	R-PP	Waste PP (post-consumer)	Bio-PP	Sugarcane (BR)
		HDPE		R-HDPE	Waste HDPE (post-consumer)	Bio-HDPE	Sugarcane (BR)

Case study / Plastic product	Market Sector	Scenarios					
		Conventional feedstock (fossil-based)		Plastic waste or captured CO ₂ as feedstock		Bio-based or partially bio-based feedstock	
		Material	Feedstock	Material	Feedstock	Material	Feedstock
4 - Automotive interior panels	Automotive	PP	Crude oil/ natural gas	R-PP	Waste PP (post-consumer)	PLA	Maize (US)
		ABS		-	-		
		PBS		-	-	Bio-PBS	Maize (US)
5 - Printer housing panels	Electrics & Electronics	ABS	Crude oil/ natural gas	R-ABS	Waste ABS (post-consumer)	PLA	Maize (US)
		PC/ABS		R-PC/ABS	Waste PC/ABS (post-consumer)	PLA/PC blend	Maize (US; PLA) Crude oil/natural gas (PC)
6 - Monobloc stacking Chairs	Other ⁽¹⁾	HDPE	Crude oil/ natural gas	R-HDPE	Waste HDPE (post-consumer)	Bio-HDPE	Sugarcane (BR)
		PP		R-PP	Waste PP (post-consumer)	Bio-PP	Sugarcane (BR)
7 - Cleansing wipes	Consumer goods	PP	Crude oil/ natural gas	-	-	Bio-PP	Sugarcane (BR)
		LDPE		-	-	Bio-LDPE	Sugarcane (BR)

⁽¹⁾ Furniture & furniture equipment.

2 Goal, general scope and common limitations

This section discusses the goal of the illustrative case studies presented in this report, common aspects of the scope (e.g. considered impact categories, applied impact assessment methods, and general data sources), as well as cross-cutting limitations. Case-specific aspects of the scope (e.g. the functional unit, the system boundary, as well as specific limitations and assumptions) are separately discussed in the single case studies (Sections 3-5).

2.1 Goal

The goal of the illustrative case studies described in this report is to test and demonstrate the practical applicability of the *Plastics LCA* method (Nessi et al., 2021) in quantifying the potential environmental impacts of plastic products relying on different feedstocks and derived polymers. Each case study thus investigates a set of LCA scenarios considering the use of different feedstocks and/or materials for a specific plastic product (as described in Section 1.3), including both conventional (fossil-based) feedstocks (crude oil and natural gas) and alternative ones (i.e. plastic waste, biomass and/or CO₂ from gaseous effluents, depending on the case study).

In line with the nature of the *Plastics LCA* method, the case studies include a product-level (or “micro-level”) assessment of the potential environmental impacts of selected plastic products with a defined and quantifiable function. A value-chain perspective is adopted, which focuses on relevant processes and activities within the value chain, and on directly connected effects (e.g. direct land use change). Therefore, the studies do not assess the effects of strategies or policy decisions/initiatives implying relevant, large-scale changes at societal and/or industrial level, such as a complete or partial substitution of the feedstock or material used to manufacture a specific plastic product or to produce a given polymer in the EU or other geographical regions. For instance, the studies do not assess the potential environmental effects of having all beverage bottles manufactured from a specific bio-based polymer in Europe as of 2030. Such type of “macro-level” assessments would require additional considerations and evaluations, other than a partially different approach, and much more practical experience than currently exists.

2.2 General scope

In line with the goal specified in Section 2.1, and with the nature of the *Plastics LCA* method, the case studies quantify the potential environmental impacts associated with all relevant processes and activities in the investigated product value chains, accounting for the burdens and effects directly connected with the throughput of such processes and activities. Moreover, a situation of equilibrium between demand and supply of the investigated feedstocks and polymers is considered, along with average market conditions for both foreground and background processes/activities (e.g. material and energy supply). This scope is suitable to fulfil the purpose of quantifying potential environmental impacts at the product level (as discussed above).

Since no relevant changes in feedstock or product demand and supply are considered (due to, e.g., the implementation of strategies or policy decisions), the studies exclude the burdens associated with possible large variations in scales of production compared to the current situation (requiring, for instance, infrastructure changes). The burdens from any indirect (market-mediated) effects due to relevant changes in the demand for a given feedstock or polymer are also excluded (such as from possible changes in production and availability of common refinery outputs due to a reduced demand of naphtha for fossil-based polymer production). The same applies to the burdens from any indirect macro-economic effects due to changes in petroleum usage and price. However, as recommended in the *Plastics LCA* method, and considering the results of earlier studies on bio-fuels conducted by the JRC (EC, 2016), the effects of GHG emissions estimated to occur as a consequence of indirect Land Use Change (iLUC) were evaluated

and reported as additional environmental information. This means that, due to the higher uncertainty associated with such estimates, and to the absence of an internationally agreed method for quantification, the potential impact of GHG emissions from iLUC was calculated, reported and interpreted separately from direct value-chain-related impacts (see Section 2.4 for a broader discussion).

Following the rules of the *Plastics LCA* method and the above discussion, the system boundary of the different product scenarios were set to include all relevant processes and activities of the investigated product value chain, considering the default life cycle stages specified in the *Plastics LCA* method itself. These stages include *Raw Material Acquisition and Pre-Processing* (which was here split into two different stages, using a more case-specific nomenclature, i.e. *Feedstock Supply* and *Polymer Production*), *Manufacturing* of the product in scope, *Distribution* and *End of Life*. The *Use* stage was generally excluded, as not involving any relevant burdens or not being affected by the considered product scenario. More details on the specific processes and activities included in the system boundary in the single case studies are provided in the respective descriptions (Sections 3-5).

As for the geographical and temporal scope, the case studies reflect the current (i.e. 2020) average situation at the EU level, in terms of supply, use and End of Life (treatment or disposal) of the investigated plastic products. This means, for instance, that currently applied feedstocks, polymers and End of Life options were considered in the analysed product scenarios, as reflected in recent past data and information. For non-commercially available products (e.g. as still under development), the average situation that would apply if the product was placed on the EU market today was considered (e.g. in terms of applied feedstock and End of Life options). Where relevant for the product in scope, extra-EU End of Life treatment or disposal was also considered, if sufficiently detailed and representative data were available to quantify the share of these End of Life routes and to model the respective burdens. This was the case, for instance, of printer housing panels, which are not addressed in the case studies included in this report. Applied foreground and background life cycle inventory data reflected the current or recent past average technology in the relevant geography for the specific process or activity, unless only data representative of other geographies and/or of specific (i.e. non-average) technologies were available. The evaluation of any future or prospective scenarios (in terms of, e.g., feedstocks or technology) was beyond the scope of the case studies.

2.3 Impact categories and impact assessment methods

The potential environmental and human health impacts of the investigated plastic products were assessed considering the whole set of default impact categories and related impact assessment (characterisation) models prescribed in the Product Environmental Footprint (PEF) method, and adopted consistently also in the *Plastics LCA* method (Section 3.2.4). For completeness, the full list of considered impact categories and applied impact assessment models is also reported in Table 2.1 below.

The set of characterisation factors applied for each impact category is the one reported in the EF 2.0 reference package (Fazio et al., 2018a). This is the most recent version of the package that, at the time of developing the case studies, could be consistently and properly used in combination with the available pool of EF-compliant datasets in the LCA software supporting the assessment (i.e. GaBi 9.2). Note, however, that a more recent version of both the EF reference package and of the pool of EF-compliant datasets was released in the meantime (i.e. EF 3.0; Fazio et al., 2018b), and shall be applied in any future LCA study conforming to the *Plastics LCA* method.

The most relevant difference between the EF 3.0 and 2.0 reference packages is related to the characterisation factors applying to Human Toxicity and Ecotoxicity impact categories. In the most recent package (EF 3.0), these factors were improved based on updated toxicological and physicochemical properties available, for relevant substances,

in the REACH, OpenFood Tox and Pesticide Properties databases (as described in Saouter et al., 2020) thereby reducing the uncertainty underlying each factor. Therefore, the case studies described in this report could not benefit of this improvement in the uncertainty level of life cycle impact assessment results calculated for Human Toxicity and Ecotoxicity impact categories.

Consistently with the applied characterisation factors, also normalisation and weighting factors were taken from the EF 2.0 reference package (as reported in Annex B). The set of weighting factors determined excluding Human Toxicity and Ecotoxicity impact categories was specifically applied, since the contribution of these categories was not considered in the calculation of the total normalised and weighted impact score (as not yet benefitting from the abovementioned reliability improvement). This is in line with the rules specified in the latest Product Environmental Footprint Category Rules Guidance (version 6.3; EC, 2018b), which specifically regulated the use of normalisation and weighting factors from the EF 2.0 package.

Table 2.1. Impact categories considered in the LCA case studies and corresponding impact assessment models (as prescribed in the EU Product Environmental Footprint method and consistently adopted in the *Plastics LCA* method).

Impact Category	Unit	Impact Assessment Model
Climate Change, total ⁽¹⁾	kg CO ₂ eq.	Baseline model of the IPCC over a 100-year time horizon (IPCC, 2013)
Ozone Depletion	kg CFC-11 eq.	Steady-state model of the WMO over an infinite time horizon (WMO, 2014 + integrations)
Human Toxicity – cancer	CTU _h	USEtox model 1.01 (Rosenbaum et al., 2008)
Human Toxicity – non-cancer	CTU _h	USEtox model 1.01 (Rosenbaum et al., 2008)
Particulate Matter	Disease incidence	PM method recommended by UNEP (UNEP, 2016)
Ionising Radiation – human health	kBq U ²³⁵ eq.	Human health effect model (Dreicer et al., 1995; Frischknecht et al., 2000)
Photochemical Ozone Formation – human health	kg NMVOC eq.	LOTOS-EUROS model (Van Zelm et al., 2008) as implemented in ReCiPe 2008
Acidification	mol H ⁺ eq.	Accumulated Exceedance model (Seppälä et al., 2006; Posch et al., 2008)
Eutrophication – terrestrial	mol N eq.	Accumulated Exceedance model (Seppälä et al., 2006; Posch et al., 2008)
Eutrophication – freshwater	kg P eq.	EUTREND model (Struijs et al., 2009) as implemented in ReCiPe
Eutrophication – marine	kg N eq.	EUTREND model (Struijs et al., 2009) as implemented in ReCiPe
Ecotoxicity – freshwater	CTU _e	USEtox model 1.01 (Rosenbaum et al., 2008)
Land Use	Pt	Soil quality index based on LANCA (Beck et al., 2010; Bos et al., 2016)
Water use	m ³ world eq.	Available WATER REmaining (AWARE) (UNEP, 2016)
Resource Use – minerals and metals	Kg Sb eq.	CML 2002 (Guinée et al., 2002) and van Oers et al. (2002)
Resource Use – fossils	MJ	CML 2002 (Guinée et al., 2002) and van Oers et al. (2002)

⁽¹⁾ The indicator “Climate Change, total” comprises three sub-indicators: Climate Change, fossil; Climate Change, biogenic; Climate Change, land use and land use change. The contribution of each sub-indicator was separately reported in the case studies, despite the *Plastics LCA* method requires this only when the contribution is larger than 5% of the total Climate Change impact.

2.4 Environmental effects and issues covered as “additional information”

The set of default impact categories prescribed in the *Plastics LCA* method and considered in the case studies (see Section 2.3) cover a broad range of potentially relevant environmental issues for (plastics) value chains. For these categories, an agreed and sufficiently reliable, robust and mature impact assessment method is considered to be available in the PEF framework, although with different levels of recommendations. However, a number of environmental issues going beyond those addressed in the default impact categories can be also considered relevant for plastics value chains. These issues include (but are not necessarily limited to):

- Potential impact on *biodiversity* (relevant, for different aspects, to plastic products relying on both conventional and alternative feedstocks);
- *Indirect Land Use Change (iLUC)* possibly induced by any type of land demand throughout the product life cycle (e.g. for the supply of bio-based feedstocks and materials), and resulting potential impact on Climate Change, Eutrophication, Acidification and any other relevant impact category; as well as
- *Generation and release of macro-plastics (including product litter) and micro-plastics* across the life cycle, and associated potential impacts on the environment and human health. Macro-plastics release at End of Life is especially relevant for specific product categories intended for end consumers, such as single-use packaging items used mostly for outdoor consumption (e.g. small-sized beverage bottles, take-away food packaging, etc.) which are expected to have a larger potential to be littered into the environment.

As recommended in the *Plastics LCA* method (Section 3.2.5.1), part of these emissions and impacts were separately quantified and reported in the case studies as complementary “additional environmental information”, as described below.

The potential impact on *biodiversity* was quantitatively estimated by means of an existing endpoint impact indicator expressing the potential loss of animal and vegetal species per year, due to a number of relevant impact drivers (categories), including Climate Change, Photochemical Ozone Formation, Terrestrial Acidification, Eutrophication (freshwater and marine), Ecotoxicity (terrestrial, freshwater and marine), Land Use and Water Use. The indicator was calculated using *ReCiPe 2016* impact assessment method, relying on endpoint impact assessment models and characterisation factors differing from the midpoint models and factors prescribed in the *Plastics LCA* and PEF methods for the same impact categories (Section 2.3). It is noted that none of the impact assessment methods proposed so far to quantify potential impacts on biodiversity (including the one described above) were deemed mature and robust enough to be recommended for use in the Product Environmental Footprint context (and consequently adopted in the *Plastics LCA* method).

As for *indirect Land Use Change (iLUC)*, the resulting potential impact on Climate Change was quantified, focusing especially on land demand associated with bio-based feedstock supply. The quantification was based on recalculated GHG emission factors proposed in the EU 2015/1513 Directive (EC, 2015), as recommended in the *Plastics LCA* method (Section 4.4.15.3). The use of an alternative iLUC model (Schmidt et al., 2015) and of the resulting GHG emission factors was also explored as a sensitivity analysis. The potential impacts of any other emissions possibly generated from iLUC (due to, e.g., the use of converted land and/or intensification on already used land), were not considered, in the absence of sufficiently tested quantification methods for these emissions. If quantified, such emissions would have contributed to an impact on Acidification, Eutrophication, and other relevant impact categories typically affected by agricultural production, such as Water Use and Land Use.

Finally, the potential generation and release of macro- and micro-plastics from the investigated product value chains was estimated. This was made by means of inventory-

level indicators quantifying both the initial loss of plastics from the techno-sphere (i.e. the amount of plastics directly generated from processes or consumers) and the ultimate plastics release to ocean and to the terrestrial environment. As suggested in the *Plastics LCA* method (Section 4.4.10.12), the *Plastics Leak Project (PLP) method* (Peano et al., 2020) was applied for quantification. Accounted contributions to macro-plastics generation and release were product littering from consumers and mismanagement of product waste at End of Life (due to, e.g., direct discharge to waterways or poor management in landfill sites or dumpsites). For micro-plastics generation and release, a number of sources throughout the entire product life cycle (excluding End of Life) were considered, as far as relevant to the specific supply chain. These included plastic pellets lost during conversion and handling, tyre abrasion during transport, and synthetic textiles micro-particles generated from washing activities (during textile production or use by consumers). Only the contribution of foreground conversion, transport and washing processes was considered, wherever they were not part of vertically aggregated datasets (see Section 3.5.5.6 for details). Any generation and release of secondary micro-plastics from plastic products or pieces released at End of Life after littering or waste mismanagement was not quantified, being excluded from the *PLP method* due to incomplete understanding and knowledge of fragmentation and (bio)-degradation pathways of plastic products once released into the environment. More in general, for the same reason, the applied loss and release indicators excluded the effects of biodegradation and of any other environmental mechanism affecting the fate of released plastic items beyond any initial (short-term) redistribution among different compartments. Therefore, the release values estimated for biodegradable products in the different case studies can be even significantly overestimated compared to the ultimate release after biodegradation has occurred (which was not quantified). This is acknowledged as a limitation of the case studies, albeit for transparency the release of biodegradable plastic products/materials was clearly differentiated from that associated with conventional, non-biodegradable ones.

At the time of developing the case studies, no sufficiently developed, complete and solid impact assessment methods were available to evaluate the potential impacts (e.g. biophysical, toxicological or aesthetic/landscape impacts) of macro- and micro-plastics released into the environment. Therefore, such impacts could not be quantitatively addressed in the studies. As above, this was mainly due to only partial understanding and knowledge of the mechanisms governing the fate, exposure, and subsequent (biophysical and toxicological) effects on ecosystems and humans of plastic products and particles released into the environment. This makes the development of a suitable (e.g. sufficiently robust and complete) impact assessment method challenging. However, this gap may be filled in the future, as long as better knowledge is gained on such mechanisms and suitable methods are developed.

2.5 Data sources

According to the *Plastics LCA* method, company-specific data shall be applied to model the product manufacturing process and to determine the associated material components (i.e. the “bill of materials”; Section 4.6.1 of the *Plastics LCA* method). However, having no direct access to any (measured) company-specific data¹⁵, all foreground processes in the investigated product value chains were modelled based on secondary life cycle inventory datasets or data. These were selected in general agreement with the rules prescribed in the *Plastics LCA* method (Section 4.6.3), which were applied with some deviations in order to:

- i) ensure the use of sufficiently representative data (in terms of process, reference technology and/or geography) when no suitable EF-compliant or ILCD-Entry Level (EL) compliant datasets were available (e.g. for the

¹⁵ The case studies were not developed by or on behalf of specific companies or organisations directly operating specific manufacturing facilities, or having access to relevant company-specific data.

production or recycling process of some polymers), rather than applying EF-compliant or ILCD-EL compliant proxies (as originally prescribed), as well as

- ii) to avoid excluding from the lifecycle model any of the foreground processes when no EF-compliant or ILCD-EL compliant proxy datasets were available (as prescribed in the original rule).

The following criteria were thus applied, in hierarchical order, to select both foreground and background inventory datasets and data:

- Use of a representative EF-compliant dataset;
- Use of a representative ILCD-EL compliant dataset (generally sourced from the GaBi database);
- Use of a representative dataset from other life cycle inventory databases (the *ecoinvent* 3.5 database was mostly used);
- Use of representative data from the literature to develop a new dataset;
- Use of an EF-compliant proxy dataset;
- Use of an ILCD-EL compliant proxy dataset;
- Use of a proxy dataset from other life cycle inventory databases (e.g. *ecoinvent*).

When activity data from the literature were used to develop a new dataset, these were combined with background inventory datasets selected according to the same criteria specified above, to model the different inputs and outputs.

Note that companies, organisations or any user of the *Plastics LCA* method shall follow the original data(set) selection hierarchy reported in Section 4.6.3 of the method itself.

2.6 Common limitations

As in any LCA study, a number of common limitations affect the illustrative case studies described in this report. These limitations are acknowledged and discussed in this section, while case-specific limitations are reported for each study in the respective description. Both common and case-specific limitations have to be properly taken into account in the interpretation of the results of the single case studies.

A first set of common limitations is related to the nature and scope of LCA in general and, consequently, of the *Plastics LCA* method. Other limitations relate to the absence of suitable impact assessment methods for use in LCA, due to difficult quantification of impacts and/or incomplete understanding and knowledge of the underlying environmental mechanisms. Similarly, some emissions into the environment cannot be (fully) quantified and inventoried at present, due to partial knowledge of the mechanisms governing the ultimate release of specific substances from a main source (e.g. littered plastic products). This also generates some limitations, while other ones are associated with data availability, as discussed below. Finally, the assessment of products relying on processes and/or technologies presenting different levels of development, maturity optimisation and/or different scales of production introduces some additional limitations, to an extent depending on the specific product scenarios investigated in a given case study.

1. *Limitations associated with the scope of LCA*

There are a number of environmental aspects and impacts (some of which relevant to plastics value chains) which are commonly not addressed in LCA, simply because they are beyond the nature and scope of the method itself.

For instance, by its inherent nature LCA focuses on normal (average) production conditions. This means that emissions and impacts from risks or exceptional circumstances, such as accidents, disasters, or conflicts are typically excluded. This is, for

instance, the case of emissions from accidental oil spills and fires occurring during extraction activities, natural disasters, and military operations to protect oil supply. Similarly, emissions and impacts of improper production practices (e.g. misuse of pesticides and fertilisers in agriculture) are also commonly excluded. However, the impacts of certain accidental emissions that can be considered a structural property of the supply chain are normally taken into account. For instance, emissions due to structural losses occurring during oil transport are generally included in life cycle inventories of crude oil supply, as well as the burdens from additional oil production to compensate for such losses. Similarly, emissions from venting of certain gases may be also included in such datasets. Details about the inclusion or exclusion of structural emissions in life cycle inventories of relevant processes and activities within the investigated value chains are provided in the single case studies, or in the documentation of the applied datasets.

Normally, LCA neither accounts for emissions and impacts due to incorrect citizen's behaviour, such as those associated with product littering into the environment (the prediction of which is also affected by more or less large uncertainty). However, being the release and accumulation of plastic products into the environment (especially oceans) an increasingly relevant issue at present, product littering was addressed in the case studies. This was made by estimating the potential generation and release of plastic products (macro-plastics) in different environmental compartments at the inventory level, while the resulting potential impacts could not be quantified, as discussed in Section 2.4.

2. Limitations associated with the absence of a suitable impact assessment method

Certain environmental issues are not covered in LCA because of the absence of a suitable impact assessment method to calculate one or more relevant impact indicators. For instance, potential impacts on landscape (e.g. aesthetic/visual impacts) are typically not considered, due to challenges associated with expressing such impacts in a quantitative manner, and to relate them to a specific process throughput. These impacts include, for example, landscape impacts due to oil sand extraction or land conversion to monoculture.

Similarly, as discussed in Section 2.4, no sufficiently developed, complete and solid impact assessment methods are currently available to quantitatively evaluate the potential environmental and human-health impacts from macro- and micro-plastics released into the environment (e.g. after product littering). This is due to only partial understanding and knowledge of the mechanisms governing the fate, exposure, and subsequent (bio-physical and toxicological) effects on ecosystems and humans of plastic products and particles released into the environment.

3. Limitations associated with the exclusion of specific emissions into the environment

In some cases, specific emissions from processes or activities within the product life cycle cannot be quantified, as direct measurement or indirect estimate/calculation is not possible. Such emissions are thus excluded from the product inventory, and the respective potential impacts are not captured in the impact assessment phase.

In these case studies, the most relevant exclusion is related to ultimate emissions generated from non-biodegradable plastic products or parts released into the environment as macro-plastics (due to, e.g., littering from consumers or incomplete removal from the environment after use). Therefore, any emissions of secondary micro-plastics from possible fragmentation, additives, metals, and/or of other possible degradation products were not inventoried, due to very partial knowledge and understanding of fragmentation and degradation pathways of plastic products into the environment, and scarcity of data on specific additives used in plastic products and polymers.

Similarly, emissions generated from biodegradable plastic products released or left into the environment after use were estimated assuming that ultimate biodegradation is achieved at the end of the 100-year time horizon applied for modelling, leading to the sole formation of CO₂, water, new soil biomass, and mineral salts of any other elements included in the product composition. However, relevant emissions might have been omitted with this approach, such as the release of any non-biodegradable additives and metals, of intermediate biodegradation products, and of any (micro)-plastic fragments/particles generated during the biodegradation process itself. Also in this case, the reason for such an exclusion is represented by incomplete product composition data (which apart few exceptions omitted any additives used), and by only partial understanding and knowledge of biodegradation pathways of plastic products into the environment.

4. *Limitations associated with data availability*

The lack of representative data for specific processes or activities is responsible for other limitations affecting the case studies. While most data-related limitations are case-study specific, and are hence reported in the respective description, others are in common to all or part of the studies, as discussed below.

A first common data-related limitation is the exclusion of the life cycle (i.e. production and any emission during use and/or End of Life) of additives used during polymer production and subsequent product manufacturing. Currently available data and information are not complete nor sufficiently specific and/or representative for a proper assessment at the product level, as not covering several polymers, and being unsuitable to properly differentiate among different products. While an important effort was made by ECHA to develop a comprehensive list of polymer additives and respective average concentrations or concentration ranges (ECHA, 2019), the data only cover most common conventional polymers, and do not differentiate by type of application or market sector (nor between food contact and non-food contact applications). Therefore, these data were not suitable, alone, to identify the specific substances used in the investigated products, nor those used in more recent alternative polymers such as Polylactic Acid or starch-based polymers. Information on additives used during polymer recycling is also very scarce. In light of this, the use of additives and the related potential impacts were only explored in one case study (not included in this report) for merely illustrative purposes. The case study focuses on printer housing panels, which were considered one of the most relevant products among those originally selected for assessment, due to the generally acknowledged use of specific additives (e.g. flame retardants) in electronics applications. For modelling purposes, a number of assumptions were made to identify representative substances to be considered based on available data, elementary flows, and related characterisation factors. A number of approximations were also performed. Due to the even large uncertainty, this evaluation was thus conducted as a sensitivity analysis, and the respective results were presented and discussed separately.

Another common limitation is associated with the use of data entirely reflecting EU technology and conditions to model production of conventional fossil-based polymers from relevant feedstocks (e.g. naphtha), despite between 2% and 25% of the conventional polymers investigated in the case studies were estimated to be produced outside EU. Specific data representing production in most of the exporting countries were indeed not available, while any existing data (e.g. for US production) were not consistent with those used for EU production, and could not be applied. Therefore, while the total share of imports was generally moderate, production impacts attributed to fossil-based polymers might have been underestimated where imports relied on less efficient and more outdated technologies, and/or on more polluting energy sources.

As discussed in detail in the single case studies, representative life cycle inventory data for a number of foreground processes were also lacking, or were only partially available. Therefore, suitable proxies or partial inventories had to be applied. This was, for instance, the case of the following processes: (i) production of some (emerging) monomers or intermediates such as FDCA (Furandicarboxylic Acid) and soy-based

polyols; (ii) polymerisation of specific polymers (e.g. PBAT and PEF); (iii) compounding of PLA-based polymer blends; and (iii) mechanical recycling of specific polymers or products such as HDPE for use in food contact applications, flexible LDPE, rigid EPS, and plastics in waste electrical and electronic equipment (including ABS and PC/ABS). In a few cases, theoretical data from process simulation had to be applied, which are generally not representative of large-scale industrial production, and do not account for possible (efficiency) improvement due to upscaling and process optimisation. This was mostly the case of processes involved in the production of specific intermediates to emerging polymers (such as Hydroxymethylfurfural, an intermediate to PEF production), or of precursors used in innovative production routes (such as CO₂-based methanol and derived olefins, used for CO₂-based PP production). These limitations are clearly acknowledged in the affected case studies.

Finally, in several scenarios, a combination of datasets and data from different sources had to be applied to build the overall product inventory, as representative EF-compliant or ILCD-EL compliant datasets were available only for part of the foreground and background processes to be modelled. This was especially the case of scenarios relying on alternative polymers and/or feedstocks, where datasets from the *ecoinvent* database were mostly used to fill data gaps, followed, to a lower extent, by literature data. While, to increase consistency, such datasets and data were applied as far as possible in combination with EF-compliant or ILCD-EL compliant datasets to model the different process inputs and outputs, this could not be done consistently for all of such inputs and outputs. Impact assessment results calculated for scenarios relying more largely on datasets and data from these alternative sources may thus be even significantly affected by discrepancies in the modelling of individual processes, compared to scenarios entirely or mostly relying on EF-compliant or ILCD-EL compliant datasets¹⁶. A careful interpretation is hence needed for the results of such scenarios, and any direct comparison between the two sets of scenarios should be avoided, especially for those impact categories that across the different case studies were found to be mostly affected by the use of datasets from alternative sources and related potential discrepancies (i.e. where such datasets were significantly contributing to the total impact). These categories include Ozone Depletion, Resource Use – minerals and metals, Human Toxicity (cancer and non-cancer), Ecotoxicity, and, to a lower extent, Water Use and Land Use, depending on the product scenario.

5. Limitations associated with the application of processes/technologies at different levels of development and maturity

Due to their objectives and scope, the case studies described in this report investigate alternative product scenarios partly relying on processes and/or technologies that nowadays do not necessarily present the same level of development and/or scale of production (maturing processes/technologies). Particularly, products totally or partially derived from alternative feedstock sources or materials often rely at least in part on upstream conversion processes that run at smaller scales, and/or are less optimised and improved compared to products based on more established conventional (fossil-based) polymers. Production of fossil-based polymers has indeed generally benefitted of several years of upscaling and process optimisation (in terms of, e.g., conversion efficiency and energy/process integration), typically leading to improved yields and energy efficiency. These are in turn generally reflected in an improved environmental performance over time. Conversely, while most polymers based on alternative feedstock sources and/or materials have been already available on the market for 20 years or more (e.g. PLA and starch-based polymers), the respective production processes may still undergo even substantial optimisation, development and upscaling (with related scale effects) in the future, potentially leading to an improved environmental profile. On the other hand, production of conventional polymers is likely to have more limited options for

¹⁶ Discrepancies between datasets from alternative sources and EF-compliant/ILCD-EL compliant datasets may be associated with the applied modelling approach, the calculation of process emissions, and/or the elementary flows used to represent such emissions.

improvement compared to what already achieved so far. Reflecting current or recent-past industrial practice, the data applied in the case studies to model polymer production are inevitably affected by any differences in the level of maturity and/or scale of the represented processes. These possible differences are clearly acknowledged, where relevant, in the different case studies (as part of the case-specific limitations) and shall be properly taken into account by the reader in the interpretation of the results.

Products relying on emerging technologies not (yet) applied at the industrial scale were also investigated in some case studies. This was made to demonstrate the applicability of the different methodological steps and rules of the *Plastics LCA* method also to non-commercially available products¹⁷ (e.g. Polyethylene Furanoate -PEF- beverage bottles), and to illustrate specific modelling rules that would have otherwise remained unaddressed (e.g. on the use of CO₂ captured from gaseous effluents as a feedstock). In the absence of (publicly available) industrial-scale data from real production facilities, such processes were modelled based on theoretical data from process simulation, generally referring to pilot-scale configurations. While possible process optimisation strategies are taken into account in the generation of such data (e.g. energy integration), they are not representative of industrial-scale production and do not account for potential (efficiency) improvements due to upscaling and further possible process optimisation/integration. The potential impacts of product scenarios relying on these processes may thus be even largely overestimated and shall be interpreted carefully, especially avoiding any comparison with other investigated scenarios. On the other hand, it should be considered that data from simulation may neglect relevant process exchanges, such as waste flows or certain direct emissions (being such data mostly focused on major energy and material flows), thus leading to underestimating total process and scenario impacts (at least in some categories). Limitations associated with the use of such data and their implications are clearly reported in the case studies, as part of the case-specific limitations.

Note that, according to the temporal scope of the case studies (which in line with the *Plastics LCA* method focuses on current conditions; Section 2.2), no prospective scenarios attempting to account for potential improvements from possible upscaling and optimisation of processes relying on maturing or emerging technologies were investigated. While broad ranges of “learning rates” can be identified from the literature for different types of industries or technologies, these are subject to wide variability and uncertainty, depending on several and not easily predictable factors (as better discussed in Annex H of the *Plastics LCA* method). This makes it often difficult to predict future learning rates (Daugaard et al., 2015), or to extrapolate existing rates to other products or industrial sectors than those they refer to. Similarly, possible “scaling factors” reflecting any future process improvement may be ideally estimated based, for instance, on historical improvements achieved in comparable but more established conversion processes (e.g. those involved in conventional polymer production). Such factors could be then applied to the burdens of the less advanced process (provided that the affected inputs and/or outputs are known), to estimate their magnitude after possible upscaling and/or optimisation. However, representative data on improvements experienced by specific industrial processes are hardly available, so that such an estimate cannot be easily conducted.

Applying any learning rates from the literature or any estimated “scaling factors” to specific process burdens would thus likely introduce even substantial additional uncertainty in the results of any forward-looking LCA scenario. Moreover, it would have not been possible to uniformly apply any learning rates or scaling factors across the different product scenarios investigated in the case studies, since production inventories of several polymers relying on alternative feedstocks or materials are only available in an

¹⁷ It is reminded that the methodological and modelling rules of the *Plastics LCA* method are applicable also to non-commercially available products, wherever (measured) company-specific data from real facilities are used to model the product manufacturing process. However, the results of the study shall not be communicated externally.

aggregated form. Therefore, the quantity of relevant process inputs (e.g. energy), outputs, or parameters (e.g. yield) potentially affected by the expected improvement could not be changed (provided that these are known for available rates and factors).

2.7 Value choices, assumptions and normative decisions

As many other analytical tools, LCA involves the creation of a conceptual model of reality. This artificial reduction of complex (product) systems to a simplified model inevitably leads to a number of value choices and more or less critical, representative and appropriate assumptions. In the case of a product life cycle model, value choices mainly relate to some methodological or modelling choices (e.g. approach used to handle multi-functionality situations, applied time horizons, etc.). Assumptions are instead introduced when no representative or sufficiently specific data or information is available, or when these are affected by large variability. Generally, assumptions are mainly related to: (a) the considered supply-chain configuration (e.g. type and origin of feedstock, transport routes, process location, applied End of Life scenarios and options for the product and other waste streams, etc.); (b) specific technological or spatial-dependent parameters/characteristics of processes (e.g. applied cultivation practices, process efficiencies, process configuration and technology, substitution ratios, etc.); and/or (c) technical properties of products and materials (e.g. thickness, lifespan, recycled content, mechanical and physical properties, etc.). While representativeness and appropriateness of assumptions can be improved when better information and data are made available, and better knowledge is gained on the investigated system, value choices cannot be deemed per se 'right' or 'wrong', and additional research would not bring additional clarity on which is the best choice to make. Rather, additional efforts may be made to ensure that the largest consensus is achieved among interested parties on the best choice to be made according to the goal and scope of the study, while keeping scientific integrity.

Complying with existing normative provisions of the Product Environmental Footprint (PEF) method (to which the *Plastics LCA* method aligns), the case studies implement several methodological choices and modelling rules that were established ex-ante to support more consistent, reproducible, robust, transparent and verifiable LCA studies of products. Such rules were defined based on a participatory process involving multiple and diverse stakeholders, and extending over several years. Therefore, they were considered sufficiently agreed and tested to be implemented also in the *Plastics LCA* method and in the illustrative case studies described in this report. However, specific sensitivity analyses were conducted on those aspects for which a full consensus has not yet been achieved, such as the handling of non-released biogenic carbon at End of Life ("carbon storage"). Also the additional or more specific choices and rules provided in the *Plastics LCA* method to cover methodological aspects relevant to plastic products were defined taking into account the comments and feedback received from many stakeholders during two public consultations (written and oral) held at different stages of the development process (see Section 1.2).

Case-specific assumptions, mainly linked to the different aspects reported above, are transparently reported throughout the description of each case study. Most relevant assumptions are also extensively summarised, along with case-specific limitations, in Section #.4 of each study. Wherever possible, these assumptions were submitted to specific sensitivity analyses, to evaluate the relevance of their effects on the results.

3 Case study 1: Beverage bottles

This case study focuses on single-use beverage bottles for juices and non-carbonated soft drinks. Bottles are the most common application for plastics used in the packaging sector, which is the largest in Europe, contributing to nearly 40% of the total plastic demand by converters in the Country (PlasticsEurope, 2019). While a variety of sizes is used for juice and drink bottles (ranging mostly from 300 to 1000 ml; Markwardt et al., 2017), a small size of 0.5 litres (500 ml) was specifically considered in this study. This is because small-sized bottles were expected to have a higher potential to be leaked into the environment (due to their typically wider use for outdoor consumption)¹⁸, thus allowing for better testing of the suggested method to estimate macro-plastics generation and release at End of Life, including the contribution of product littering. It is important to notice that the assessment focuses exclusively on the bottles life cycle, to evaluate the applicability of the *Plastics LCA* method to bottles relying on different plastic materials and alternative feedstock sources for such materials. Therefore, the study does not investigate the life cycle of any specific beverage, and was developed with this perspective in mind (e.g. in terms of selected scenarios, system boundary setting and modelling assumptions) as detailed in the rest of this chapter.

3.1 Assessed scenarios

A number of scenarios were analysed to evaluate and demonstrate the applicability of the *Plastics LCA* method in separately quantifying the potential impacts associated with the use of different existing or developing plastic materials for the manufacturing of plastic beverage bottles for juices and non-carbonated soft drinks, and of using alternative feedstocks for such materials, when available (Table 3.1). In this perspective, the assessed scenarios were not defined to reflect the combined, actual or potential market shares of the investigated materials and feedstocks in the sector of juices and non-carbonated soft drinks. Rather, they were aimed at allowing the *Plastics LCA* method to be individually tested on the range of materials and feedstocks that are currently used, or that may be potentially used (today or in the near future), in that sector.

Polymers traditionally used for the manufacturing of juices and non-carbonated soft drinks bottles are PET and HDPE from (virgin) fossil-based feedstock sources, which were considered as reference materials for the assessment in Scenarios 1 and 2 (S1 and S2). While no data were publicly available on the relative market shares of these materials at the EU level at the time of the study, PET bottles appeared to be more widely used in the considered sector compared to HDPE bottles.

The use of separately collected, post-consumer plastic waste as a feedstock for PET and HDPE bottles was investigated in Scenarios 3 and 4 (S3 and S4), considering estimates of the current average recycled content in bottles made of such materials at the EU level, i.e. 24% for PET bottles and 16% for HDPE bottles¹⁹. These shares refer to bottles used in all types of application, as no specific EU-average data for beverage bottles were available²⁰. However, they were considered sufficiently representative for the illustrative

¹⁸ Generally, bottles having larger sizes are mostly used for indoor consumption, where proper waste collection is more likely to take place (at least at the European level).

¹⁹ These values represent estimates of the penetration rate of recycled material in PET and HDPE packaging applications (typically dominated by bottles) at 2020. They were calculated assuming a linear variation between the penetration rates estimated for 2014 and 2025 in the study carried out by Hestin et al. (2017). The 2014 rate (9.5% for PET and 3.7% for HDPE) is based on an analysis of plastic packaging waste flows in Europe during the same year. The 2025 forecast (47% for PET and 35.3% for HDPE) is the estimated penetration required to meet the 55% recycling target originally set by the Packaging and Packaging Waste Directive (94/62/EC) for that year. However, for calculation purposes, 2030 was considered as the reference year to achieve such target, according to the requirement from the revised Packaging and Packaging Waste Directive (EU 2018/852).

²⁰ An estimate of the average recycled content in PET bottles in Europe for the year 2017 (i.e. 11%) is reported by EPBP (2018), based on the survey conducted by ICIS and Petcore Europe on the European PET Recycling Industry for the same year. However, this estimate was not applied in the study, to keep consistency with the procedure followed to estimate the average recycled content in HDPE bottles (for which no official estimates of the current average recycled content are available).

purpose of this assessment. On the other hand, the EU Directive on single-use plastics (EU 2019/904)²¹ requires all single-use beverage bottles with a capacity of up to three litres to incorporate at least 30% recycled material from 2030. This scenario was thus evaluated in a sensitivity analysis (Section 3.8.5.1), which also explored the effects of a complete substitution of virgin material with recycled material (i.e. a 100% recycled content). In this respect, it is noted that while some initial examples of PET bottles incorporating 100% recycled material appears (or are anticipated) to be available on the market (especially for small-sized bottles), this is not the case of HDPE bottles, where a 100% recycled content is apparently used only in non-food contact applications (e.g. detergent packaging). The explored 100% recycled content scenario has thus to be considered as a hypothetical scenario for HDPE bottles, as of today.

A partially bio-based, drop-in alternative to fossil-based PET is currently available, where fossil-based Mono Ethylene Glycol (MEG), constituting nearly 30% of the polymer by mass, is replaced with bio-based MEG derived from bioethanol. The use of this polymer (referred to as bio-based PET or Bio-PET) was investigated in Scenario 5 (S5), considering Brazilian sugarcane as a feedstock for bio-based MEG. At present, the only commercial-scale producer of bio-based MEG is located in India (De Jong et al., 2020), and can be reasonably assumed to rely on sugarcane grown in the same country as a feedstock. However, available life cycle inventory data for bio-based PET production from sugarcane consider that processing of this feedstock into bioethanol occurs in Brazil, so that Brazilian sugarcane was assumed as a feedstock for bio-based PET in this study. The use of Indian sugarcane was explored in a sensitivity analysis (Section 3.8.5.3), although production data for bioethanol and bio-based MEG, and the related transport routes, could not be adapted to reflect Indian conditions. The same sensitivity analysis also investigated the use of alternative feedstocks entirely sourced in Europe (i.e. maize, wheat and sugar beet).

In contrast to PET, it is possible to produce fully bio-based HDPE (Bio-HDPE) by totally replacing fossil-based Ethylene with its bio-based counterpart derived from bioethanol. This alternative was assessed in Scenario 6 (S6), considering again Brazilian sugarcane as a feedstock for bioethanol and bio-Ethylene production, being the main bio-ethylene supplier currently located in Brazil (De Jong et al., 2020), and sugarcane the most widely used feedstock for bioethanol production in such country (OECD-FAO, 2020). However, similarly to Bio-PET, the use of alternative feedstocks entirely sourced in Europe was also explored in a sensitivity analysis (Section 3.8.5.3).

Finally, the use of an emerging, fully bio-based polymer not yet available at the commercial scale, i.e. Polyethylene Furanoate (PEF) was investigated in Scenario 7 (S7). This was done to evaluate and demonstrate the applicability of the methodological and modelling rules provided in the *Plastics LCA method* also to non-commercially available products (as it is the case of PEF bottles), wherever a company intends to perform an internal study based on (measured) company-specific data for the product manufacturing process. As for the other investigated scenarios, the aim was hence not to provide conclusive estimates of the potential environmental impacts of 0.5 litre PEF bottles for juices and non-carbonated soft drinks.

Polyethylene Furanoate is reported to provide better mechanical and barrier properties compared to PET (e.g. barrier to oxygen and carbon dioxide) (Avantium, 2021). Therefore, relevant stakeholders have indicated that PEF is expected to be at least initially targeted especially at bottles with small capacities (even lower than 500 ml) and/or at those applications requiring additional specific functionality (e.g. defined permeability properties) such as in packaging of carbonated soft drinks or oxygen sensitive products like juices. In these situations, PEF may represent a potential lighter-weight alternative to the use of multilayer PET, metals or glass, which are traditionally used to ensure suitable mechanical and/or barrier performances. On the other hand, a number of technical challenges would need to be addressed in case PEF beverage bottles

²¹ Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment.

would be introduced as an alternative to PET bottles in applications with large market shares, especially regarding mechanical recycling at End of Life. This is because, within the current recycling infrastructure, the presence of an additional material like PEF increases the risk to contaminate PET waste streams separated for recycling. As long as penetration of PEF products on the market would be limited to quantities that would not make their recycling economically viable, they should hence be designed to enable detection and separation as residue by existing sorting equipment. Otherwise, suitable infrastructure enabling sorting of PEF into a separate material stream for recycling should be developed. At present, a maximum market penetration of 2% (in sectors traditionally relying on PET bottles) has been temporarily granted by the European PET Bottle Platform (EPBP), to ensure proper separation of PEF products within existing recycling facilities, and to avoid incompatible contamination levels of sorted PET bottle streams (EPBP, 2017).

Compared to partially bio-based PET (where only fossil-based MEG is replaced with its bio-based counterpart), in the case of PEF also fossil-based Purified Terephthalic Acid (PTA) is replaced with a bio-based alternative, i.e. 2,5-Furan Dicarboxylic Acid (FDCA). In this study, bio-based MEG was assumed to derive from bioethanol produced from Brazilian sugarcane, similarly to bio-MEG used in partially bio-based PET. The use of Indian sugarcane was not explored as an alternative, but the results of the sensitivity analysis performed for bio-based PET bottles (Section 3.8.5.3) may be reasonably extended also to PEF bottles, which equally incorporate 30% bio-based MEG. FDCA was derived from starch-based glucose from the EU-average mix of relevant starch crops (one of the main pilot plants for FDCA production is located in Europe). The mix was estimated to include maize (54% on starch basis) and wheat (46%), based on data from Starch Europe (2019). While potato starch is also reported in Starch Europe data, potatoes were not included in the considered mix of crops since according to stakeholders potato starch is not used as a feedstock for glucose production (being economically unfavourable).

Due to the absence of (publicly available) industrial scale data on the production process of the main FDCA precursor (i.e. Hydroxymethylfurfural – HMF), only theoretical data from process simulation at the pilot scale could be applied to develop a life cycle inventory of this key process step (see Section 3.5.2.3.3). While possible process optimisation strategies are taken into account in the generation of such data (e.g. energy integration), they are certainly not representative of commercial-scale production (in contrast to data available for the other polymers investigated in this case study) and do not account for potential (efficiency) improvement due to upscaling and further process optimisation. This likely implies an even large overestimate of the environmental burdens associated with HMF production, and of the potential impacts of PEF beverage bottles quantified in this study. Moreover, for further downstream conversion of HMF into FDCA, and for the polymerisation process, data for similar processes in the PET supply chain were applied as an approximation. Overall, impact assessment results related to PEF bottles thus need to be interpreted with extreme caution, taking into account the mentioned limitations and those additionally reported in Section 3.4.

It is noted that another fully bio-based polymer, i.e. Polylactic Acid (PLA), could also be used for beverage bottles production. Different examples of bottles made of this material can be found on the market, although it is not clear whether these items would be suitable as general packaging solutions for juices and non-carbonated drinks, or if they could only cover specific applications/sectors. Other than being of bio-based origin, PLA is also biodegradable under controlled aerobic conditions, such as in industrial composting facilities (and, to a much lower extent, also under anaerobic conditions) (UBA, 2018). However, the use of this material was not considered in this study, since according to a relevant stakeholder, PLA is currently unsuitable for (0.5 litre) beverage bottles, and it

can even severely interfere with PET recycling if not properly sorted out²². Even a small contamination in the range of 0.1% can indeed compromise the quality of recycled PET with respect to optical properties (e.g. transparency; Alaerts et al., 2018).

Table 3.1. LCA scenarios assessed for the beverage bottles case study and respective End of Life options and scenarios.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options /scenario (1)
1 - Conventional polymer 1	PET	MEG (2) PTA (3)	Fossil-based (crude oil/natural gas)	Recycling (60%) Incineration (21%) Landfilling (19%)
2- Conventional polymer 2	HDPE	Ethylene	Fossil-based (crude oil/natural gas)	Recycling (64%) Incineration (19%) Landfilling (17%)
3 - Alternative polymer 1	R-PET (24% recycled content)	MEG (2) PTA (3)	Waste PET (post-consumer)	Recycling (60%) Incineration (21%) Landfilling (19%)
4 - Alternative polymer 2	R-HDPE (16% recycled content)	Ethylene	Waste HDPE (post-consumer)	Recycling (64%) Incineration (19%) Landfilling (17%)
5 - Alternative polymer 3	Bio-PET (30% bio-based)	Bio-MEG (2) PTA (3)	Sugarcane (BR) Crude oil/natural gas	Recycling (60%) Incineration (21%) Landfilling (19%)
6 - Alternative polymer 4	Bio-HDPE (100% bio-based)	Bio- Ethylene	Sugarcane (BR)	Recycling (64%) Incineration (19%) Landfilling (17%)
7 - Alternative polymer 5	PEF	Bio-MEG (2) FDCA (4)	Sugarcane (BR) EU mix of starch crops (5)	Recycling (60%) Incineration (21%) Landfilling (19%)

(1) The impacts of each scenario were calculated considering an EU-average End of Life scenario combining the listed End of Life options according with the reported shares. A sensitivity analysis individually considering the application of each listed option was also conducted.

(2) MEG: Mono Ethylene Glycol.

(3) PTA: Purified Terephthalic Acid.

(4) FDCA: Furan Dicarboxylic Acid.

(5) The mix includes Maize (54%) and Wheat (46%), in terms of starch product equivalents (Starch Europe, 2019).

Regarding the product End of Life, all treatment and disposal options currently applied at the EU level to beverage bottles made of each specific material were considered, including mechanical recycling, incineration and landfilling. For bottles relying on non-commercially available polymers (i.e. PEF bottles), those End of Life options that would be potentially applied at present once the product was introduced into the market were considered, taking into account relevant material properties (e.g. biodegradability), and the options currently applied to bottles made of the other investigated materials. As a base case, the impacts of each beverage bottles scenario were thus assessed with reference to an EU-average End of Life scenario including all the considered End of Life options, which were combined as described in Section 3.5.5.1 and as summarised in

²² The received comment states that: "The industry decided years ago that PLA should not be used in 0.5 L beverage bottles, since PLA is not suitable for this application and, even more important, PLA also can interfere PET recycling, if sorting facilities are outdated."

Table 3.1. However, in a sensitivity analysis, scenario impacts were also recalculated by individually applying each considered End of Life option (Section 3.8.5.8). Note that, despite bio-based PET, bio-based HDPE and PEF entirely or partially incorporate bio-based material, they are not biodegradable, as the final polymers have the same characteristics as their fossil-based counterpart, i.e. they all are "drop-in" solutions (except PEF, which does not have any fossil-based equivalent). Therefore, biological treatment options such as composting and anaerobic digestion are not viable for these materials and were not considered in this study. It is also noted that the hypothetical EU-average End of Life scenario for PEF bottles was defined assuming that the product was introduced to the market in a quantity justifying separate collection and sorting into a dedicated material stream for recycling (and that the same collection rate for recycling as PET bottles is achieved). While this ensures consistency with the other investigated scenarios, it does not fully reflect the current average situation, where in the absence of specific adjustments to the recycling infrastructure PEF bottles would be likely separated as residue during pre-treatment (sorting) of separately collected plastic waste they are part of. The assumed End of Life scenario should hence be considered optimistic for the present situation.

3.2 Functional Unit and reference flow

The main function of the studied product (0.5 litre bottles) is the delivery of beverage (juice or non-carbonated soft drinks) from producers to final users. The functional unit was thus defined as: *"delivering 1000 litres of beverage by means of 0.5 litre single-use bottles in the EU, without breaking during transport and ensuring a minimum defined shelf life of the product"* (Table 3.2).

Table 3.2. Functional unit defined for beverage bottles LCA scenarios.

Aspect	Description
"What" (function or service provided)	Delivering of beverage (juice or non-carbonated soft drinks) by means of 0.5 litre single-use bottles
"How much" (extent of the function or service)	1000 litres of beverage
"How well" (expected level of quality of the function or service)	Without breaking during transport, and ensuring a minimum defined product shelf life
"How long" (duration/lifetime of the function or service)	One time
"Where" (location/geography of the function or service)	In the entire EU-28 market
"For whom" (beneficiary of the function or service)	To an EU-28 average consumer

The reference flow of each scenario (i.e. the amount of polymer required to fulfil the functional unit), was calculated based on estimated average masses of bottles made of the relevant material, when used for the delivery of juices and non-carbonated soft drinks (Table 3.3).

Regardless of the type of feedstock used for polymer production, the average mass of PET and HDPE bottles was estimated based on linear regression on measured mass

values reported by Markwardt et al. (2017)²³ for bottles with a size ranging from 330 ml to 1000 ml²⁴ and used as packaging for juices, non carbonated soft drinks and milk. The resulting average mass of 0.5 litre PET bottles was equal to 24 g, which is in line with the range reported by a relevant stakeholder (20-22 g), although not supported by specific evidence and hence not considered in this study. For HDPE bottles, an average mass of 26.5 g was instead estimated. It is reasonable to consider that these estimates inherently take into account the different barrier and mechanical properties of the two investigated materials, although such properties are not explicitly considered in the calculation of the bottle mass. Indeed, real packaging masses are normally a result of a more or less long (empirical) optimisation process intended to meet different design constraints and functional requirements, based on a given material and its underlying properties.

For PEF bottles, no measured mass values were available, due to the absence of commercial applications of this material at the time of the study. An average bottle mass was thus estimated based on potential material savings compared to bottles made of one of the materials that PEF may replace in specific applications, i.e. PET. According to the suggestions of a relevant stakeholder, the improved barrier and mechanical properties of PEF should approximately allow for an overall 20% reduction in material usage with respect to PET when used in 0.5 litre bottles, corresponding to a final bottle mass of 19.2 g. Since this estimate cannot be currently validated through measurements on real products on the market, and it is not possible to judge whether it is conservative or optimistic, results calculated for PEF beverage bottles have to be interpreted with caution, also taking into account the additional limitations reported in Section 3.4. On the other hand, it is noted that potential material savings compared to PET bottles are expected to be higher in case smaller serving sizes were considered, according to the comments of another stakeholder.

Table 3.3. Calculation of the reference flow for beverage bottles LCA scenarios.

Material	Bottle mass (g)	Reference flow (kg/FU)
PET (all types of feedstock)	24.0	48.0
HDPE (all types of feedstock)	26.5	53.0
PEF	19.2 ⁽¹⁾	38.4

⁽¹⁾ Based on an estimated 20% potential material saving compared to PET, due to improved mechanical and barrier properties (see the main text in this section for details).

²³ No other references reporting specific mass values for bottles used for juices and non-carbonated soft drinks packaging were found at the time of the study. While a number of EPDs have been published for bottled beverages, these either refer to other market sectors (e.g. bottled water) or do not explicitly report the mass of the packaging (bottles).

²⁴ The following size-mass data pairs were considered for PET bottles: 330 ml-21.30 g (average of two measured values for the "grab & go" sector); and 1000 ml-32.72 g (average of four measured values for the "juice and natural soft drinks" sector and one measured value for the "dairy" sector). For HDPE bottles, the following data pairs were considered: 350 ml-22.51 g (average of two measured values for the "grab & go" sector); 380 ml-22.07 g (single value for the "grab & go" sector); 900 ml-37.87 g (average of two measured values for the "dairy" sector); and 1000 ml-44.73 g (single value for the "juice and natural soft drinks" sector).

3.3 System boundary

In all the analysed scenarios, the system boundary was set to cover the default life cycle stages specified in the *Plastics LCA* method for cradle-to-grave LCAs of final products²⁵, and the associated most relevant processes of the beverage bottles life cycle. The considered life cycle stages and processes are described below, and are also schematically represented in the system boundary diagrams depicted in Figures 3.1 to 3.7:

- **Feedstock Supply**²⁶ – covering extraction, processing, transport and possible refining of crude oil and natural gas (fossil-based polymers); collection, transport and sorting of post-consumer plastic waste (recycled polymers); crop cultivation (bio-based polymers), as well as transport of these feedstock materials to downstream conversion processes (e.g. naphtha cracking, polymer recycling, sugarcane fermentation, wet milling of starch crops).
- **Polymer Production**²⁷ – covering all the activities associated with the conversion or recycling of relevant feedstock materials into the specific polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes.
- **Manufacturing** – including bottle manufacturing (directly at filling plants)²⁸ through stretch-blow moulding of injection-moulded preforms (PET and PEF bottles), or extrusion-blow moulding of polymer granulate (HDPE bottles).
- **Distribution** – including transport of bottles from the manufacturing and filling site to retailers, and from these to final consumers²⁹.
- **End of Life** – covering collection, transport, recycling, incineration, and disposal of bottles after use, including any avoided processes from virgin material or energy substitution by recovered material/energy.

The default life cycle stage “Raw Material Acquisition and Pre-processing” was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating and separately quantifying the impacts of feedstock supply and those associated with downstream conversion processes into final polymer granulate. Moreover, a different nomenclature was applied to such stages compared to the default nomenclature specified in the *Plastics LCA* method, to better reflect the investigated supply chains and the scope of the study. The stage of “Raw Material Acquisition” was thus identified with that of “Feedstock Supply”, while “Pre-processing” corresponds to “Polymer Production”.

Filling of bottles (including all operations occurring at filling plants beyond bottle production) and the Use Stage were excluded from the assessment. This is because the study focuses on bottles for beverage delivery (and not on the life cycle of a specific packaged beverage), and no relevant burdens are associated with the sole use of bottles. Moreover, it can be reasonably assumed that the type of bottle material does not affect the burdens of filling operations and of activities related to the Use Stage of beverages

²⁵ Note that, as permitted by the *Plastics LCA* method, and as described below, some of the default life cycle stages have been split into different sub-stages, and their naming was adjusted to better reflect the scope of this study.

²⁶ Corresponding to the default life cycle stage “Raw Material Acquisition and Pre-Processing” specified in the *Plastics LCA* method.

²⁷ Corresponding to the default life cycle stage “Raw Material Acquisition and Pre-Processing” specified in the *Plastics LCA* method.

²⁸ This is normally the case of PET bottles used in large filling plants, where the conversion of polymer resin into preforms and subsequently into bottles entirely takes place on-site. In smaller facilities, only the final formation of bottles is generally carried out, relying on preforms sourced from external suppliers. For simplification purposes, PET bottles were assumed to be totally manufactured directly at filling plants, and the same assumption was consistently applied to HDPE and PEF bottles (although HDPE bottles are normally manufactured in external facilities).

²⁹ Transport of the bottle content was excluded in this case study, due to the focus on the bottle life cycle, and not on the life cycle of the packaged product (i.e. juice or non-carbonated soft drinks).

delivered by means of such bottles (e.g. any chilling activity or the use of drinking cups)³⁰, as far as bottles are designed to ensure equivalent mechanical properties (e.g. stiffness, which affects ability and speed of filling) and permeability (which affects the shelf life of the product). Under these conditions (which are at least partially ensured by the requirement incorporated in the “how well” aspect of the functional unit defined in Section 3.2), such activities would thus generate identical impacts in all the investigated scenarios and may be excluded. However, if this is not the case, they should be also included in the system boundary.

Similar considerations apply to the additional packaging items that are normally used, along with bottles, for beverage delivery to consumers (e.g. caps, labels, and secondary and transport packaging). The life cycle of these components was excluded from the assessment, considering the specific focus on bottles, and to avoid unnecessary complication of the model. Also in this case, it is reasonable to assume that the same additional packaging items would be used, regardless of the material or feedstock applied for bottle manufacturing, so that the potential impacts of the different scenarios are not affected. On the other hand, this exclusion implies underestimating the overall impacts associated with beverage delivery.

In general, the mentioned exclusions may weaken the assessment if they are not adequately justified (e.g. if relevant differences that are not taken into account exist among alternatives) and a comparison has to be made. Therefore, the Use Stage, filling activities (and the life cycle of additional packaging components) shall always be considered for possible inclusion in any comparative LCA study on beverage bottles, paying particular attention to possible differences among the compared alternatives. Moreover, such stages and processes shall always be included in LCA studies of specific packaged beverages (e.g. bottled water), in line with the system boundary requirements specified in the *Plastics LCA* method.

Finally, it has to be noted that additives were not included in the assessment, due to the lack of complete and consistent data on the use of additives in the production of beverage bottles, of the respective polymers, and of plastics in general. This lack is even more relevant if a proper differentiation among different materials needs to be made, as in this case study. Data and knowledge gaps also exist on the possible release and fate of additives over the product life cycle. Exclusion of additives is acknowledged as a limitation of this study, as additive production can account for a non-negligible portion of the cradle-to-gate Climate Change impact and energy demand of polymers, which is up to 46% for starch-based polymer grades including larger shares of additives in the range of 30% (Broeren et al., 2017). Moreover, additives can also be relevant at the End of Life stage, where they can be released, as such or after degradation/conversion into different compound(s), in the environment (e.g. the soil in case of biodegradable products sent to biological treatments, or subject to in-situ degradation).

³⁰ An exception may be represented by CO₂ emissions to air throughout the life cycle if the study would focus on bottles for carbonated beverages with relevant differences in permeability properties. However, the requirement of ensuring similar shelf lives incorporated in the functional unit implies that bottles with similar permeability properties would be used.

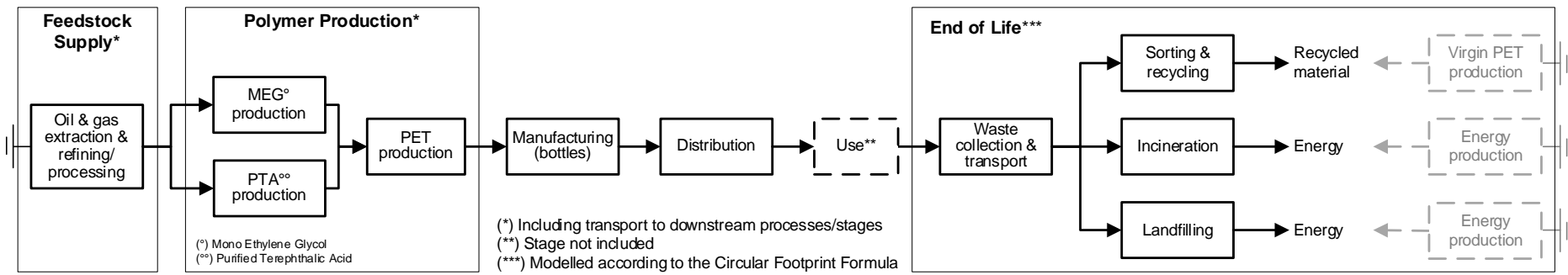


Figure 3.1. System boundary for fossil-based PET beverage bottles (Scenario 1).

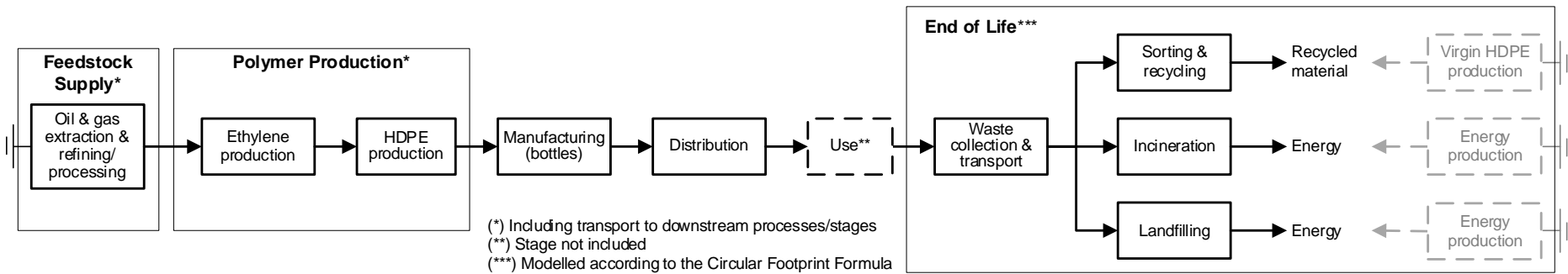


Figure 3.2. System boundary for fossil-based HDPE beverage bottles (Scenario 2).

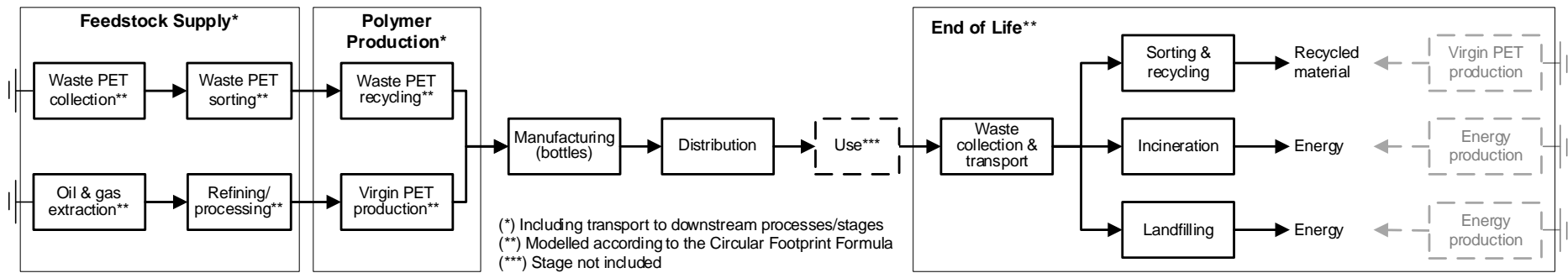


Figure 3.3. System boundary for 24% recycled PET beverage bottles (Scenario 3).

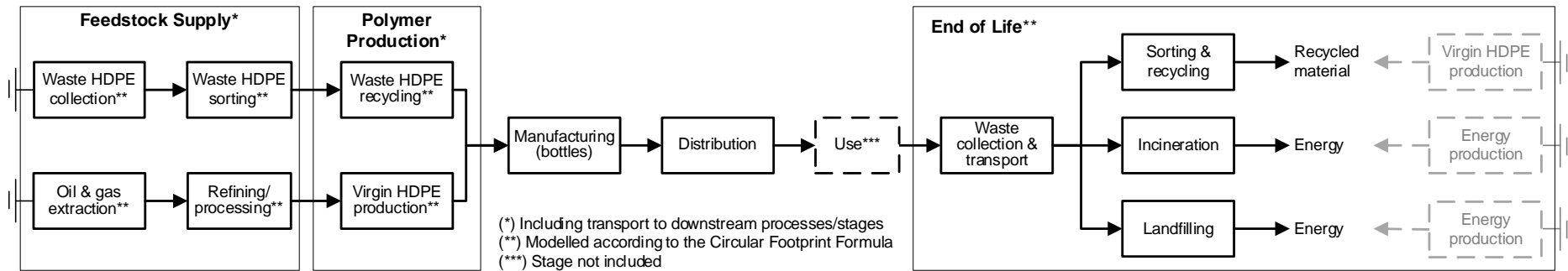


Figure 3.4. System boundary for 16% recycled HDPE beverage bottles (Scenario 4).

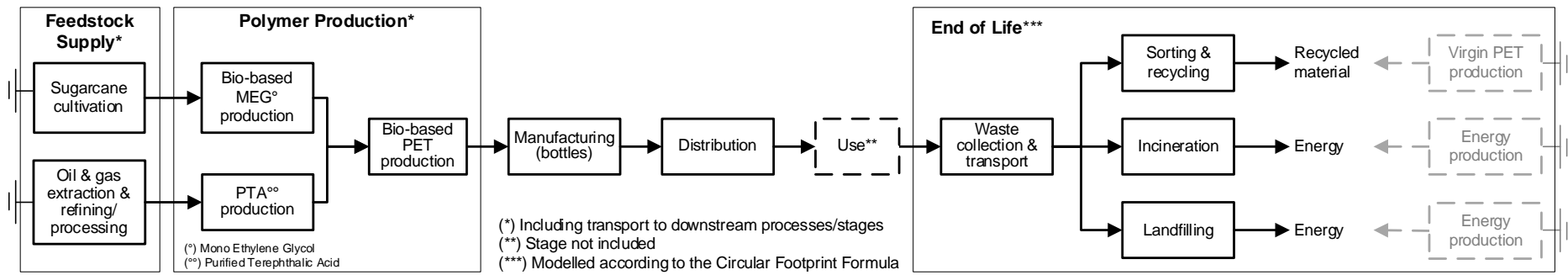


Figure 3.5. System boundary for partially bio-based PET beverage bottles (Scenario 5).

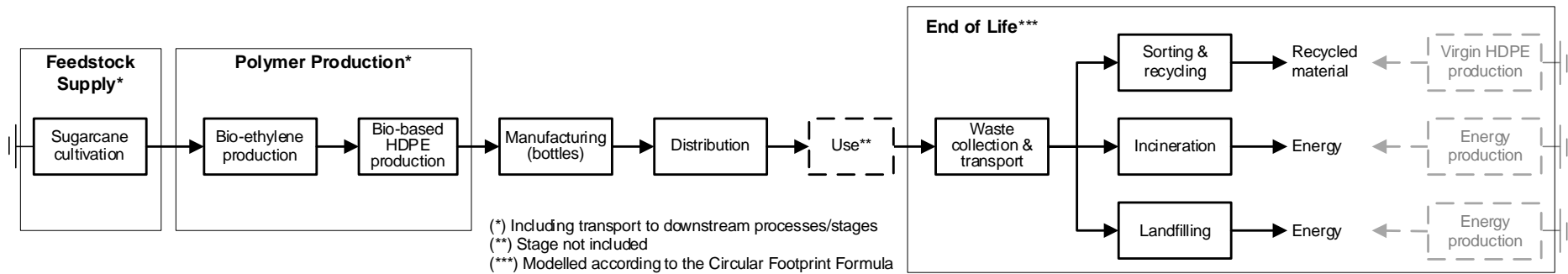


Figure 3.6. System boundary for bio-based HDPE beverage bottles (Scenario 6).

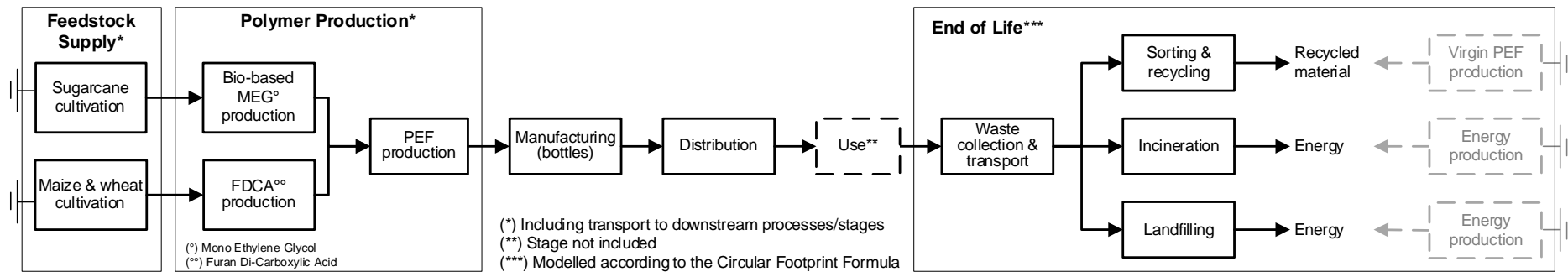


Figure 3.7. System boundary for PEF beverage bottles (Scenario 7).

3.4 Limitations and critical assumptions

The following key limitations and critical assumptions apply to the LCA scenarios investigated in this case study, and have to be properly taken into account, where relevant, in the interpretation of the respective results.

- While already commercially available and produced in industrial-scale facilities, bio-based PET and bio-based HDPE partly rely on upstream conversion processes that are more recent compared to their fossil-based counterparts, and which have likely experienced less optimisation and improvement (e.g. in terms of conversion efficiency and process integration). Therefore, the results related to these bio-based alternatives shall be interpreted taking into account differences in the level of maturity of part of the underlying technology compared to their fossil-based counterparts.
- The average mass of PET and HDPE bottles was estimated based on measured mass values reported in the literature for bottles with a size ranging from 330 to 1000 ml, and used in the Nordic market for the delivery of juices, non-carbonated soft drinks and milk (see Section 3.2). While these values are considered sufficiently representative for this illustrative study, they may not fully reflect the average mass of bottles currently available in the EU market as a whole.
- The estimated current EU-average recycled content for partially recycled PET and HDPE bottles (i.e. 24% and 16%, respectively) is not specifically representative of beverage bottles, but refer to bottles used in all market sectors (e.g. including also detergent packaging) (see Section 3.1). For food-contact applications like beverage bottles, the actual current recycled content may be lower, especially for HDPE bottles, where the use of recycled material appear to be less established.
- Bio-based MEG (used as a co-monomer in both Bio-PET and PEF) was assumed to derive from Brazilian sugarcane as a feedstock (for reasons of data availability), while at present it is likely produced relying on sugarcane grown in India (where the only commercial-scale supplier of bio-based MEG currently available on the market is located; De Jong et al., 2020) (see Section 3.1). For the same reason, production of sugarcane bioethanol out of sugarcane was modelled as taking place in Brazil, with raw bioethanol being subsequently transported to Europe for further conversion into bio-based MEG and final polymerisation (Section 3.5.2.3.1). However, both conversion processes to bioethanol and bio-MEG would be more likely carried out in India, with bio-based MEG being ultimately transported to Europe for polymerisation. The potential impacts quantified for bio-based PET and PEF bottles might be even significantly affected by this assumption, although it has to be reminded that bio-based MEG constitutes only about 30% of both polymers by mass. Note that the use of Indian sugarcane as a feedstock was investigated in a sensitivity analysis, while still considering subsequent processing into bioethanol occurring in Brazil and conversion into bio-based MEG taking place in Europe (production data and transport routes could not be adapted to reflect Indian conditions).
- The life cycle (i.e. production and resulting emissions at End of Life or throughout the product life cycle) of any additives used during polymer production or bottles manufacturing (e.g. antioxidants or heat stabilisers) was not included in the assessment, due to the lack of complete and sufficiently specific and/or representative data on additive use and release (see Section 3.3). While additives are generally reported to be mostly used in small shares, their potential impacts may be proportionally (much) higher compared to used quantities, once they are released into the environment. Impact assessment results presented in this study may thus be even significantly underestimated in specific categories, and do not capture any differences in additive use among the investigated product alternatives.
- A combination of datasets from different sources was applied to build the life cycle inventory of the investigated beverage bottles scenarios, since representative EF-

compliant or ILCD-EL compliant datasets were only available for part of the foreground and background processes to be modelled. This was especially the case of bio-based HDPE and PEF bottles, and, to a lower extent, of partially recycled PET and HDPE bottles, where datasets from the *ecoinvent* database were mostly used to fill data gaps. Impact assessment results calculated for these scenarios may thus be even largely affected by discrepancies in the modelling³¹ of individual processes compared to scenarios entirely or mostly relying on EF-compliant or ILCD-EL compliant datasets (e.g. fossil-based PET and HDPE bottles). A careful interpretation is hence needed for the results of such scenarios, and any direct comparison between these two sets of scenarios should be avoided, especially for those impact categories that across the different case studies were generally affected to a larger extent by the use of datasets from different sources and the associated potential modelling discrepancies (i.e. where such datasets were significantly contributing to the total impact). These categories include Ozone Depletion, Resource Use – minerals and metals, Human Toxicity (cancer and non-cancer), Ecotoxicity, and, to a lower extent, Water Use and Land Use, depending on the product scenario.

- Production of virgin, fossil-based PET and HDPE was modelled based entirely on data reflecting EU technology and conditions, despite approximately 20% of these polymers (21% for PET and 22% for HDPE) were estimated to be imported (see Section 3.5.2.1). This is due to the lack of specific data representing production in exporting countries such as Korea, India, Turkey, Saudi Arabia and Qatar. While the share of imports is moderate, virgin production impacts of PET and HDPE may thus be underestimated if imports rely, for instance, on less efficient/outdated technologies and/or on more polluting energy sources.
- In the absence of specific data, production of secondary bottle-grade HDPE granulate (used as recycled content in partially recycled HDPE bottles) was approximated with data related to secondary bottle-grade PET production (see Section 3.5.2.2). While the approximation is considered reasonable (both processes are expected to be similar, albeit not including exactly the same steps), the applied data do not refer to the real process and introduce a source of uncertainty. The use of any additives needed to achieve a suitable quality of the recycled polymer to be used as a replacement for virgin material was also excluded. However, this exclusion equally applies to recycled bottle-grade PET.
- Collection rates for recycling estimated and assumed for PET and HDPE bottles (from any feedstock) do not specifically refer to beverage bottles but to bottles used in all kinds of applications, due to the absence of sufficiently granular data (see Section 3.5.5.1). However, while this is acknowledged as a limitation, the applied data were considered suitable to build sufficiently representative average End of Life scenarios for the scope of this study.
- For reasons of data availability and reliability of results, non-bottle grade PET granulate obtained from End of Life recycling of PET bottles (from any feedstock) was modelled as replacing bottle-grade virgin resin (see Section 3.5.5.3). This likely results in a partial overestimate of the benefits associated with PET bottles recycling at End of Life.
- To model landfilling of PET and HDPE bottles from any feedstock, a dataset related to the disposal of a generic, average, conventional plastic material was applied as a proxy, due to the lack of a material-specific EF-compliant dataset for landfilling of such polymers³² (Section 3.5.5.5). The approximation was considered acceptable, but it is reported for completeness, as prescribed in the *Plastics LCA* method (Section 4.4.10.11).

³¹ Discrepancies may be associated with the applied modelling approach, calculation of process emissions, and/or the elementary flows used to represent such emissions.

³² Note that for bio-based PET and bio-based HDPE bottles, CO₂ and CH₄ emissions inventoried in the dataset were changed from fossil to biogenic, in line with the bio-based origin of (part of) the carbon contained in such polymers (see Section 3.5.5.5 for details).

- Being PEF bottles not yet available on the market, and due to the subsequent lack of representative (industrial-scale) data for many of the respective upstream conversion processes and of specific statistics for product waste management at End of Life, a number of limitations and relevant assumptions apply to the PEF beverage bottles scenario, as discussed below. The results presented in this case study for PEF beverage bottles thus have to be interpreted with utmost caution, and should not be compared with the results obtained for the other investigated scenarios.
- Due to the current absence of commercial applications for PEF, the average mass of bottles made of this material was estimated based on theoretical calculations of potential material savings with respect to PET bottles (taking into account differences in relevant mechanical and barrier properties of the two materials) (see Section 3.2). The mass of bottles directly affects the reference flow and the final impact assessment results calculated for PEF bottles, which could even significantly change if the mass of any real product that may be introduced on the market in the future will substantially differ from the value estimated in this study.
- The life cycle inventory of a key process step in the PEF supply chain, i.e. production of Hydroxymethylfurfural (HMF; the main precursor of the FDCA comonomer) was based on theoretical data from process simulation at the pilot scale, due to the absence of (publicly available) industrial-scale data from real facilities (see Section 3.5.2.3.3). These data are not representative of commercial-scale production and do not account for potential (efficiency) improvement due to upscaling and further process optimisation. Moreover, interpretation and elaboration of raw data from simulation to develop the process inventory may have led to the quantification or modelling of contributions that do not fully reflect real process inputs. Therefore, the environmental burdens associated with HMF and PEF production calculated in this study are likely to be even largely overestimated.
- The burdens associated with production of glucose used in the HMF synthesis process may be overestimated since dried starch was considered as an input to the glucose production process, according to available data, while non-dried starch slurry is more likely to be used (see Section 3.5.2.3.3).
- Conversion of HMF into FDCA and subsequent polymerisation into PEF were modelled based on data for similar processes in the PET supply chain (i.e. PTA production and PET polymerisation) (see Section 3.5.2.3.3). While this is considered a reasonable approximation (based on consulted literature), it adds further uncertainty to the potential impacts quantified in this study for PEF beverage bottles.
- The End of Life scenario considered for PEF bottles was defined assuming that their level of market penetration is suitable to make separate collection and sorting into a dedicated material stream for recycling economically viable (see Section 3.5.5.1). While this assumption ensures consistency with the other investigated scenarios, it does not fully reflect the current average situation, where in the absence of specific adjustments to the recycling infrastructure PEF bottles would be likely separated as residue during sorting operations. Moreover, the same separate collection rate for recycling as PET bottles was assumed (in the absence of specific data for non-commercially available PEF bottles), considering that such rate may be reasonably achieved once a similar collection and recycling scheme would be implemented also for PEF bottles. However, the assumed separate collection rate may not be necessarily achieved in reality. The modelled End of Life scenario should hence be considered optimistic for the present situation.
- Mechanical recycling of PEF bottles was approximated with data related to mechanical recycling of post-consumer PET into secondary, non-food grade

granulate, in the absence of specific data for PEF (see Section 3.5.5.3). This approximation was considered reasonable (as a similar process would be likely applied to PEF recycling), but introduces additional uncertainty.

3.5 Life Cycle Inventory

This section describes the main details of the Life Cycle Inventory of the analysed scenarios, including the related assumptions and data sources. The description is separately reported for each life cycle stage in the following sub-sections (3.5.1 – 3.5.6).

3.5.1 Feedstock Supply Stage

3.5.1.1 Fossil-based polymers

For virgin fossil-based polymers (i.e. PET and HDPE), the stage of Feedstock Supply includes the activities of crude-oil and natural gas exploration, drilling, extraction, processing and transport to downstream users, as well as naphtha production in crude oil refineries and its transport to subsequent conversion processes (i.e. naphtha cracking, or also catalytic reforming).

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. In line with the reference year of the dataset, the considered crude-oil mix refers to the year 2014, and it was demonstrated to properly reflect also the current situation, since no relevant changes were estimated to take place (see the analysis reported in Annex C, which also provides more detail on crude oil origins considered in the applied mix). All relevant activities related to crude oil supply are covered in the dataset, including exploration activities, well drilling, crude oil extraction and processing, long-distance transport via pipeline and (where relevant) tanker vessels, as well as regional distribution to the final consumer via pipeline. Oil losses regularly occurring during transport via pipelines or vessels are also taken into account in the dataset, as these can be considered a structural property of the supply chain under normal average supply conditions. Handling of oil losses is not explicitly reported, but they are likely accounted in terms of increased crude oil input to provide the intended unit output, according to common LCA practice. In addition, direct emissions of oil (unspecified) to seawater, freshwater and soil are inventoried, which may be associated to oil leakage. The magnitude of these emissions is quite limited, equalling 0.0115 g per kg of crude oil in the case of seawater emissions, and 0.0539 g/kg in the case of freshwater ones³³. Oil spills and fires due to accidents are not mentioned in the dataset, and likely not accounted, according to the typical focus of LCA on normal (average) production and supply conditions, excluding burdens and potential impacts from accidents. Land transformation and occupation burdens are accounted for land-based oil sources (e.g. oil sands), while any landscape impacts (e.g. from oil sand mining) are not captured in LCA (nor in the applied dataset), and no flows are inventoried in this respect. 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

³³ Note that ecotoxicity impacts of these oil emissions are not quantified, due to the absence of suitable characterisation factors in the version of the default Environmental Footprint (EF) life cycle impact assessment (LCIA) method applied in this study (i.e. EF 2.0). Characterisation factors for unspecified oil emissions are available in the most recent version of the LCIA method (EF 3.0) and can be applied in future evaluations based on the *Plastics LCA* method. Note, however, that characterisation factors for oil (calculated as the average of factors related to several refinery products) are lower compared to those of a large portion of substances covered in the LCIA method itself (i.e. 67% of them).

goods, including infrastructure, are not included, according to the 95% cut-off rule applied in the dataset³⁴, based on material or energy flows, or the level of environmental significance.

Similarly to crude oil, an aggregated, EF-compliant dataset 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)" (reference year 2014). 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 and regasification (for imports of liquefied natural gas via vessels), as well as long-distance transport via pipeline and vessels, and final regional distribution to the end consumer via pipeline. Regular natural gas losses occurring during transport are also accounted in the dataset, for both pipeline and vessel transport. Consistently with the approach adopted for crude oil, 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 95% cut-off rule applied in the dataset³⁵, based on material or energy flows, or the level of environmental significance.

Naphtha production from crude oil was modelled based on an aggregated, gate-to-gate, ILCD-EL compliant dataset provided by Thinkstep (reference year 2016). 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. The model is built by largely relying on statistical data and measurements from more than one-hundred (i.e. 103) refineries, for a total processing capacity of more than 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 considered input or output. The crude oil demand of a specific unit process is allocated to the respective output products and/or intermediate products based on energy (i.e. the net calorific value of the product), thus assigning larger shares of upstream supply burdens to product with higher calorific values. Energy inputs (i.e. thermal energy, steam and externally sourced electricity) are allocated based on the mass share of the product or intermediate product, out of the total mass of products obtained from the same unit process. With this approach, products requiring more processing steps for production are assigned higher burdens from energy consumption. Direct emissions to the environment are allocated based on mass, as well.

Transport of naphtha from refineries to downstream users (e.g. cracking facilities) 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" (reference year for most information and data is 2000). Compared to the original dataset, which reflects transport to different end-users (including petrol stations), a number of adjustments were made. First of all, default transport via road, rail and barge (likely associated to non-industrial users) was entirely converted to pipeline transport, which was considered more appropriate for naphtha used for industrial purposes. Since the distance associated to the different replaced transport routes is not known, the overall original quantity (in kg*km) associated with road, rail and barge transport was converted to pipeline

³⁴ Note that the cut-off applied in this specific dataset would be additional to any cut-off applied to processes and activities included in the main product life cycle inventory (no cut-off was explicitly applied in this study), which would follow the 3% cut-off rule specified in the *Plastics LCA* method (Section 4.6.4).

³⁵ Note that the cut-off applied in this specific dataset would be additional to any cut-off applied to processes and activities included in the main product life cycle inventory (no cut-off was explicitly applied in this study), which would follow the 3% cut-off rule specified in the *Plastics LCA* method (Section 4.6.4).

transport. This is considered a reasonable approximation, being the overall inventoried quantity of such transport routes only a small share of the overall transport amount (around 19%). Secondly, energy inputs to the process were modelled by means of suitable EF-compliant datasets reflecting EU-average conditions, which replaced the originally linked *ecoinvent* datasets. Finally, the treatment of fly ash and scrubber sludge, polluted rainwater and wastewater was entirely modelled through datasets referring to the geography “Europe without Switzerland”, including also the amount of waste originally modelled as being treated under Swiss conditions.

3.5.1.2 Recycled polymers

For recycled polymers (i.e. R-PET and R-HDPE), Feedstock Supply consists of collection of post-consumer plastic waste of the relevant polymer, and its subsequent transport and sorting in specific facilities for recycling. These activities and processes were modelled as described in Sections 3.5.5.2 and 3.5.5.3 on End of Life modelling). Applied data refer to collection and transport of separately collected plastic waste at the municipal level (most of the plastic waste feedstock for the production of recycled PET and HDPE used in bottle manufacturing is expected to be post-consumer packaging waste from municipal collection), as well as to sorting of mixed plastic waste in dedicated facilities.

The collection, transport and sorting processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the approach prescribed in the *Plastics LCA* method to handle recycling situations. For recycled PET, the formula was applied considering the default application-specific value of the A factor reported in Annex C of the *Plastics LCA* method for bottle-grade recycled PET, which is equal to 0.5. The same value was considered also for recycled HDPE, according to the material-specific value reported in Annex C for recycled PE used in unspecified applications (in the absence of an application-specific value). Therefore, only 50% of the burdens from waste collection, transport, sorting (and subsequent recycling; see Section 3.5.2.2) were assigned to the recycled material content in beverage bottles, the rest being assigned to the system providing waste material for recycling. However, the recycled polymer content was assigned an equal share $((1-A) \times Q_{Si}/Q_p)$ of the burdens associated with the supply of the virgin fossil-based feedstock used in the production of the replaced virgin polymer. Such burdens were modelled as described in Section 3.5.1.1. Further details and considerations on the implementation of the CFF are provided in Section 3.5.2.2, addressing the modelling of the Polymer Production Stage for recycled polymers.

3.5.1.3 Bio-based polymers

For bio-based polymers (Bio-PET, Bio-HDPE and PEF), the stage of Feedstock Supply includes cultivation of the relevant crop or mix of crops, and their subsequent transport to further processing in the same country.

Cultivation of Brazilian sugarcane (used as a feedstock for bio-MEG and bio-Ethylene) was modelled through aggregated, ILCD-EL compliant datasets from the GaBi database³⁶, which consider different pre-harvest burning rates for the crop (i.e. 0% and 100%). A pre-harvest burning rate of 0% means that all sugarcane is modelled as harvested mechanically (via agricultural machinery). When a rate equal to 100% is considered, all sugarcane is modelled as harvested manually, i.e. sugarcane residues (tops and leaves) are burned on standing plants before harvesting. The two datasets were combined to reflect a situation where 10% of sugarcane is manually harvested via the pre-harvest burning practice, which was estimated to be representative of the current situation. The estimate was based on the share of sugarcane-cultivated areas in the south-central region of Brazil that still applied pre-harvest burning in 2016 (i.e. 6% according to Bordonal et al., 2018), and on the contribution of such region to the total sugarcane-

³⁶ The two applied sugarcane cultivation datasets are: [BR] *Sugar cane perennial (0% slash and burn); technology mix | production mix, to consumer | 74% H2O, 0% pre-harvest burning*; and [BR] *Sugar cane perennial (0% slash and burn); technology mix | production mix, to consumer | 74% H2O, 0% pre-harvest burning*.

cultivated land of Brazil, which was 90% on average during the period 2016-2018 (based on data from UNICA, 2021). For the remaining 10% of sugarcane-cultivated areas in the north-northeast region of Brazil, a 50% pre-harvest burning rate was instead assumed, in the absence of specific data. However, the pre-harvest burning practice will be legally phased out by 2031 (State Law n. 11241/02) and was expected to be phased out by 2017 according to industry association protocol of intention (Tsiropoulos et al., 2014). Therefore, a sensitivity analysis has been performed on this parameter, considering a complete abandonment of this practice (see Section 3.8.5.2).

In general, the applied sugarcane production datasets represent cultivation in the state of Sao Paulo (South-Central Region) with no irrigation (only rainwater is used), and refer to the year 2018. They are mainly based on industry data, completed, where necessary, by secondary data (Brandstetter 2011; Seabra et al. 2011). All relevant activities associated with field and culture management are covered in the datasets, including agricultural operations (in terms of diesel and/or electricity consumption from tractors, machinery and pumps and resulting emissions), production of synthetic N-P-Ca fertilisers and pesticides (for weed, fungi and insects control), use of vinasse and press cake as organic fertilisers, as well as field emissions from fertilisers and pesticides application. The life cycle (production, maintenance and End of Life) of agricultural machinery and other infrastructures is excluded. The uptake of biogenic CO₂ from the atmosphere during sugarcane growth is considered (1.26 kg CO₂/kg_{sugarcane}), as well as GHG (CO₂) emissions from direct land use change (0.127 kg CO₂/kg_{sugarcane}). These emissions are quantified based on the approach from PAS 2050-1:2021 (in line with the requirements of the *Plastics LCA* method), considering the situation where the country is known (Brazil) while the previous land use is unknown, and distributing estimated emissions from changes in carbon stocks (soil and vegetation) between subsequent land uses over a period of 20 years. Changes in carbon stocks are quantified based on the calculation rules provided in IPCC (2006; Volume 4). No allocation nor substitution are directly applied in the datasets (the cultivation process is not multifunctional), while vinasse used as fertiliser input is likely modelled as a by-product from sugarcane processing (information provided is not completely clear) with a share of upstream burdens allocated to it according to a suitable criteria (likely economic value). Transport of harvested sugarcane to further processing in sugarcane mills for bioethanol production was assumed to take place along an overall distance of 25 km, by means of large lorries (> 32 t, fuelled with the Brazilian diesel mix).

Cultivation of the European starch crops used as a feedstock for FDCA production (maize, and wheat) was modelled through available EF-compliant datasets³⁷ referring to the year 2016. The latter are based on the approach used to develop agricultural inventories available in the Agri-footprint database (Agri-footprint methodology; Blonk Consultants, 2015a,b), relying on 5-year average yield data from FAOSTAT (2010-2014), and on country-, crop-, and process-specific data for the other relevant parameters, as appropriate. Included activities are seeding and seeds production, fertilizers and pesticides production (including packaging), their application and resulting field emissions, water use for possible irrigation, energy use and transport associated with field management practices, as well as capital goods for cultivation and transport (production, maintenance, and End of Life). Any other consumables used during cultivation are excluded, as well as any activities related to living at the farm or to other businesses (e.g. wind energy generation), consumables not used as a raw or auxiliary material for possible processing of the crop, and any packaging used during processing. The burdens from these activities and inputs are quantified based mostly on a combination of several literature sources and dedicated modelling activity (e.g. for fertiliser and pesticide emissions) or simulation tools (e.g. for energy use), considering crop- and country-specific parameters, where relevant. In particular, specific fertilizer input is defined based on crop-specific nutrient requirements and country-specific

³⁷ The dataset considered for maize is "[EU+28] Maize (corn grain) production; technology mix, production mix | at farm", while for wheat the applied dataset is "[EU+28] Wheat grain; technology mix, production mix | at farm".

fertilizer mix derived from International Fertilizer Association (IFA) statistics (IFA, 2015). Heavy metals emissions due to manure and synthetic fertilizers application are calculated based on an adapted methodology from Nemecek and Schnetzer (2012), taking into account the heavy metal balance as a function of deposition, use of fertilizer and crop uptake. Pesticide application rates were derived from different literature sources, while water use has been based on the "blue water" footprint (Mekonnen and Hoekstra, 2010). GHG (CO₂) emissions from direct land use change are estimated through the "*Direct Land Use Change Assessment Tool (2015)*" (Blonk Consultants, 2015c), which is based on the approach outlined in PAS 2050-1:2021 and prescribed in the *Plastics LCA* method. Estimated emissions are allocated to the crop considering a period of 20 years after the direct land use change has occurred, and are equal to 0.01075 kg CO₂/kg_{maize} and 0.01897 kg CO₂/kg_{wheat}. The uptake of biogenic CO₂ by the crop during growth is not inventoried (as considered part of the short-term carbon cycle) with no effects on final impact assessment results (default characterisation factors for biogenic CO₂ uptake and release are set to zero). Allocation of cultivation burdens between any co-products such as wheat grain and wheat straw is based on economic criteria (five-year average price). All crops were assumed to be transported to downstream processing along an overall distance of 100 km, covered by large lorries (> 32 t, fuelled with the EU diesel mix).

3.5.2 Polymer Production Stage

The Polymer Production stage covers the activities of feedstock processing into any relevant intermediate(s) and monomer(s), the polymerisation or recycling process, as well as any transport among these activities and final transport of polymer granulate to the beverage bottles manufacturing and filling site. The following subsections (3.5.2.1 – 3.5.2.4) describe how these activities were modelled in this case study, distinguishing between fossil-based, recycled and bio-based polymers.

3.5.2.1 Fossil-based polymers

For conventional, fossil-based polymers (PET and HDPE), the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of partially aggregated, cradle-to-gate, ILCD-EL compliant datasets provided by Thinkstep³⁸. These datasets disaggregate upstream feedstock inputs (crude oil, natural gas and naphtha), reflect the main technologies adopted in EU-28, and refer to the year 2018. They are mainly based on industry data from internationally adopted production processes, integrated, where needed, with literature data from several sources. The number of industry data sources considered for individual process steps is not specified. Disaggregated upstream inputs include combinations of crude oil, natural gas and naphtha, depending on the polymer. For PET, crude oil accounts for 92% of the total feedstock input (including the contribution of naphtha), with the remaining 8% being covered by natural gas. For HDPE, the shares are equal to 75% for crude oil, and 25% for natural gas. All conversion processes are assumed to take place in Europe, so that the datasets not only reflect the main technology applied in the region, but also EU-average background conditions in terms of e.g. energy generation, material supply and transport. Note, however, that approximately 20% of PET (21%) and HDPE (22%) used in the EU were estimated to be imported, based on average annual import shares calculated from *Prodcom* data for the years 2016-2018 (Eurostat, 2019d). The use of data reflecting European technology and conditions also for imports (due to the absence of data for polymer production in exporting countries) thus represents a limitation of this study, although the estimated import shares are modest.

³⁸ The dataset considered for PET is "[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", while for HDPE the applied dataset is "[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".

For both PET and HDPE, the main conversion process involved in the modelled supply chain is steam cracking of naphtha and natural gas, delivering the monomer ethylene, along with propylene, butadiene, and other co-products such as pyrolysis gas (a mixture of benzene, toluene and xylenes), refinery gas, and hydrogen. Other relevant conversion processes are catalytic reforming of naphtha and pyrolysis gas, and steam reforming of natural gas. In catalytic reforming, naphtha and pyrolysis gas are processed to produce benzene, toluene and xylenes, with one isomer of the latter (para-xylene) being an intermediate in the production of Purified Terephthalic Acid (PTA, a co-monomer of PET). 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).

As for the process of steam cracking of naphtha, allocation among the different co-products (Ethylene, Propylene, Butadiene, refinery gas, pyrolysis gas and Hydrogen) is based on energy, considering the net calorific value of each co-product. For catalytic reforming of naphtha and the resulting reformat output, the same allocation rules and data sources considered for refinery operations are applied (see Section 3.5.1.1), as the process typically takes place at refining facilities. In the subsequent separation process of reformat gas into its components (including para-Xylene used for PTA production), allocation based on energy content (net calorific value) is applied, instead. The same criteria is applied as well to Carbon Monoxide and Hydrogen separated (via cryogenic separation) from synthesis gas generated through natural gas reforming. In the final polymerisation step no allocation is performed, being PET and HDPE the only outputs of the respective production process.

3.5.2.2 Recycled polymers

The production of recycled, bottle-grade PET granulate out of sorted, post-consumer PET waste was modelled based on the *ecoinvent* dataset "[CH] polyethylene terephthalate production, granulate, bottle grade, recycled", referring to the year 2014. 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 granulate, through a number of steps. These 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 reasonably assumed to correctly include also the Solid State Polymerisation (SSP) process (required to increase the intrinsic viscosity of recycled PET flakes to a level comparable with primary PET), and extrusion of flakes into final polymer granulate. However, while both process steps are mentioned in the dataset documentation, it is not clear whether they are actually considered. In the implementation in the model, the original dataset was adjusted from Swiss to EU background conditions, and background datasets related to energy supply were replaced with relevant EF datasets. The small amount (0.122 kg/kg R-PET) of secondary HDPE recovered from reprocessing of caps normally discarded with bottles was assumed to directly replace virgin HDPE granulate (1:1 substitution) according to the hierarchy for the handling of multifunctional processes specified in the *Plastics LCA* method³⁹. Finally, a

³⁹ Note that this substitution was proven to have no relevant effects on the results (of the PET recycling process and as a whole). Therefore, while HDPE recovered from the PET recycling process might not directly replace virgin HDPE granulate on a 1:1 basis, the effects of this uncertainty are minimal.

few adjustments had to be performed to improve reliability of LCIA results for the Ozone Depletion impact category⁴⁰.

The same dataset described above was also applied to approximate the burdens associated with the production of recycled, bottle-grade HDPE, in the absence of more specific data for such a process (only data for generic recycled HDPE granulate are available). This is acknowledged as a limitation of this study. The same adjustments reported above for recycled bottle-grade PET were performed, along with the replacement of the input flow of sorted PET waste with that of sorted HDPE waste.

According to the approach prescribed in the *Plastics LCA* method to model recycling situations (Circular Footprint Formula), only a share of the total burdens of the described recycling processes⁴¹ were allocated to the recycled content in PET and HDPE bottles, based on the values of the A and Q_{sin}/Q_p factors used in the formula. Annex C of the *Plastics LCA* method specifies a default value of the A factor equal to 0.5 for both recycled bottle-grade PET (application-specific value) and recycled HDPE used in unspecified applications (material-specific value, selected in the absence of an application-specific alternative). Therefore, only 50% of the total burdens of the recycling process (per functional unit) were allocated to the recycled content in bottles. On the other hand, the recycled content carried a share of the primary production burdens of the replaced virgin material (i.e. the same burdens that would have been credited to End of Life recycling in the previous product life cycle providing the recycled material). Since the Q_{sin}/Q_p factor is equal to 1 for (bottle-grade) PET granules from the SSP process (being their quality comparable to that of virgin granules) the total allocated share of virgin PET production burdens was again equal to 50% (i.e. $(1-A) \times Q_{sin}/Q_p = (1-0.5) \times 1 = 0.5$). The same value was applied also to recycled HDPE, being its quality necessarily suitable for use in bottle-grade applications, and hence similar to that of replaced virgin HDPE (although the modelled recycling process only approximates the real process and excludes the use of any additives used to achieve comparable technical properties). Virgin polymer production burdens were modelled as described in Section 3.5.2.1 for conventional fossil-based PET and HDPE, and in Section 3.5.1.1 for the respective Feedstock Supply.

3.5.2.3 Bio-based polymers

3.5.2.3.1 Bio-based PET

The production of partially bio-based PET out of sugarcane and fossil feedstock sources (oil and natural gas) was modelled based on an aggregated, ILCD-EL compliant dataset provided by Thinkstep referring to the year 2018 (*[EU-28] Polyethylene terephthalate granulate (PET) via PTA+EG (part. biobased- sugar cane) - open sugar cane | partially biobased via terephthalic acid and ethylene glycol from ethylene based on sugar cane | single route, at plant*). The dataset covers the steps of sugarcane processing into bioethanol in Brazil (fermentation and distillation), bioethanol transport to Europe via transoceanic ship, conversion to bio-Ethylene (via dehydration) and Ethylene Glycol (via oxidation and hydration), and its subsequent polymerisation to PET along with fossil-based Terephthalic Acid. Gate-to-gate production burdens of this monomer based on crude oil and natural gas are accounted as well in the dataset, similarly to fossil-based PET (Section 3.5.2.1). Combustion of bagasse from sugarcane processing generates surplus energy (electricity and heat) which were assumed to directly replace average electricity from the Brazilian grid, and thermal energy produced in the same country from

⁴⁰ Default *ecoinvent* datasets for the supply of soap (*[GLO] Market for soap*), and Sodium Hydroxide (*[GLO] market for sodium hydroxide, without water, in 50% solution state*) were replaced with EF-compliant datasets related to the production of the same materials. In addition, a number of chemicals were removed from the proxy dataset for unspecified organic chemicals (*[GLO] chemical production, organic*), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde. The respective shares were then equally subdivided among the remaining chemicals.

⁴¹ Including both mechanical recycling and the subsequent solid state polymerisation process in the case of recycled PET.

natural gas. Allocation between the different outputs from Ethylene oxidation (i.e. mono-Ethylene and di-Ethylene Glycol) is based on the respective economic value. Inventories of the processes related to feedstock conversion into the two co-monomers (bio-MEG and fossil-based PTA) are mainly based on industry data from internationally adopted production processes, completed, where necessary, by literature data. Polymerisation is instead based on literature data and know-how of the data provider. Background inventories from the GaBi database are used to model the burdens of the inventoried inputs and outputs.

3.5.2.3.2 Bio-based HDPE

For bio-based HDPE production, no datasets were available from the pool of EF-compliant datasets nor other databases. All processes involved in the conversion of Brazilian sugarcane into the final polymer were hence modelled individually based on different data sources, as described below.

Sugarcane fermentation to bioethanol was based on the *ecoinvent* dataset "[BR] Ethanol production from sugarcane | Ethanol, without water, in 95% solution state, from fermentation". The dataset mainly relies on literature data and resulting calculations (e.g. for emissions from bagasse burning), and considers Brazil as a relevant geography, as well as 2000-2006 as the reference period. Compared to the original dataset, allocation between the co-products Ethanol and surplus electricity from bagasse combustion was removed, and replaced with direct substitution of average electricity from the Brazilian grid. This was made for consistency with the approach adopted in the aggregated gate-to-gate datasets used to model the production of the other bio-based polymers relying on sugarcane bioethanol investigated in this and other case studies (i.e. Bio-PET, Bio-LDPE and Bio-PP), and aligns with the hierarchy for the handling of multifunctional processes prescribed in the *Plastics LCA* method. Background datasets related to energy generation (i.e. only electricity substitution in this case) were also replaced with background EF-compliant datasets. Finally, a number of other adjustments had to be performed, to improve reliability of LCIA results for the impact categories of Ozone Depletion and Resource Use – minerals and metals⁴². The specific amount of sugarcane required for bioethanol production is equal to 15 kg per kg of bioethanol.

Bioethanol produced in Brazil was assumed to be transported to Europe for further conversion and polymerisation, consistently with the supply-chain configuration considered in the aggregated, gate-to-gate datasets used to model the production of the other bio-based polymers derived from sugarcane bioethanol considered in this and other case studies (Bio-PET, Bio-LDPE and Bio-PP). Transport is modelled according to the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers located outside Europe to factories/users in Europe. This includes transport by lorry (> 32 t, Euro 4) from the bioethanol factory to a Brazilian harbour along a default distance of 1000 km, transoceanic ship transport to Europe, and final transport by lorry (> 32 t, Euro 4) to the conversion plant in Europe, again along a default distance of 1000 km. The oversea distance for ship transport was estimated based on the calculation tool available on SeaRates.com⁴³, and set equal to 11,300 km (from Porto Alegre to Rotterdam). This is in line with the distance assumed in the abovementioned aggregated gate-to-gate datasets used for the modelling of the other sugarcane bioethanol-based polymers considered in this project, which is around 8500 km. It is noted that the assumption of raw bioethanol being transferred to Europe for further conversion may differ from the current average situation, where one of the main

⁴² Infrastructure processes related to the bioethanol fermentation plant and the heat and power co-generation unit were removed. Moreover, default *ecoinvent* datasets for the supply of lime ([RoW] Market for lime, hydrated, packed), lubricating oil ([RoW] Market for lubricating oil), and Sulphuric Acid ([RoW] Market for sulfuric acid) were replaced with EF-compliant datasets related to the production of the same materials. Finally, End of Life treatment of wood ash mixture ([RoW] Market for wood ash mixture, pure) was removed, to improve reliability of results related to the impact categories of Human Toxicity – non-cancer, and Ecotoxicity - freshwater.

⁴³ Available at: <https://www.searates.com/services/distances-time/>

producers of bio-based Ethylene is located in Brazil (De Jong et al., 2020). In this specific case, the whole process chain of conversion and polymerisation would likely take place in Brazil, with the final HDPE resin being eventually transported to Europe rather than raw bioethanol. The assumption performed as a base case in this study is thus slightly in disfavour of the Bio-HDPE supply chain, since nearly 2 kg of bioethanol are required per kg of Bio-HDPE (see below in this section), and hence a higher mass of material is transported per functional unit compared to transporting the final HDPE resin (i.e. 1 kg per each kg of polymer required per functional unit). Production of bio-based HDPE entirely taking place in Brazil was explored in a sensitivity analysis, still considering sugarcane cultivation and subsequent conversion to bioethanol occurring in the same country (see Section 3.8.5.4).

The inventory of Ethanol dehydration to Ethylene (in Europe) was developed based on literature data related to a real industrial process, and available in a life cycle assessment study on bio-based HDPE conducted for the company Braskem (ACV Brasil, 2017). Input and output data from the reported inventory were combined with background EF datasets for energy generation, and *ecoinvent* background datasets for material production⁴⁴. Moreover, the different transport activities reported in the original inventory were not considered, as transport of bioethanol (the main raw material) was accounted for separately in the foreground product inventory, while transport of the other input materials to the process is already included in the datasets applied to model the supply of such inputs. Beyond Ethylene, a small amount of naphtha is also obtained as a co-product from the process, which is handled via direct substitution of naphtha from crude oil refinery (modelled as described for the "Feedstock Supply" stage in Section 3.5.1.1). The considered data source does not report the specific bioethanol requirement of the process, which was thus determined as the average of the values reported in IEA-ETSAP and IRENA (2013; 1.74 kg bioethanol/kg Ethylene) and IfBB (2018; 2.08 kg bioethanol/kg Ethylene), corresponding to an estimated specific consumption of 1.91 kg bioethanol per kg of Ethylene.

The final polymerisation step of Ethylene to HDPE was modelled based on data from the most recent PlasticsEurope ecoprofile available at the time of this study (PlasticsEurope, 2016), as implemented in the *ecoinvent* database. Indeed, inventory data related to the sole polymerisation step could not be extracted from the partially aggregated Thinkstep dataset used to model fossil-based HDPE production. Therefore, it was not possible to perform a consistent modelling of the polymerisation process across all the investigated HDPE-based scenarios (i.e. fossil, partially recycled, and bio-based HDPE bottles). The applied data represent average values of data collected from several European production units operated by PlasticsEurope member companies, covering 68% of the total production capacity in Europe. The mix of commercial HDPE production technologies is considered, including slurry suspension polymerisation, gas phase polymerisation and solution polymerisation (using Ziegler-Natta, Philips, and Metallocene catalysts). The final process inventory was built by combining input/output activity data from the mentioned source, with background EF datasets for energy generation and *ecoinvent* background datasets for material production⁴⁵. No allocation nor substitution was performed, since the process is mono-functional, delivering only HDPE as a product. The specific requirement of Ethylene reported in the ecoprofile is equal to 1.0018 kg per kg of HDPE, which is lower than the consumption reported in the abovementioned study by ACV Brasil

⁴⁴ An exception is liquid Nitrogen supply, for which a dataset from the EF database was used, to overcome issues of reliability of LCIA results in the Ozone Depletion impact category.

⁴⁵ Exceptions are represented by Chromium Oxide supply ([GLO] *Market for chromium oxide, flakes*) and liquid Nitrogen supply ([RER] *Market for nitrogen, liquid*), for which EF-compliant datasets were used to overcome issues of reliability of LCIA results in the impact categories of Resource Use - minerals and metals and Ozone Depletion. For similar reasons, but limited to the Ozone Depletion category, infrastructure processes related to the polymerisation plant ([RER] *Chemical factory construction, organics*) were also removed. Finally a number of chemicals were removed from the proxy dataset for unspecified organic chemicals ([GLO] *chemical production, organic*), i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde). The respective shares were then equally subdivided among the remaining chemicals.

(2017), i.e. 1.07 kg Ethylene/kg of HDPE, representative of suspension polymerisation only.

3.5.2.3.3 Polyethylene Furanoate (PEF)

Being not yet commercially produced, no life cycle inventory datasets are available for PEF production, neither as EF- or ILCD-EL compliant dataset, nor in any other existing database. The process chain to convert the different feedstock sources into the final polymer was hence modelled based on a combination of existing datasets used as proxies, as well as new datasets developed based on elaborations of literature data.

Conversion of Brazilian sugarcane to bio-based MEG was based on an aggregated, ILCD-EL compliant dataset provided by Thinkstep, consistently with the dataset used to model partially bio-based PET production from sugarcane (*[EU-28] Ethylene glycol from ethane (biobased - sugar cane) and oxygen via EO - open input sugar cane; oxidation of ethene with oxygen and water | production mix, at plant | 1.11 g/cm³, 62 g/mol*). The dataset refers to the year 2018 and covers the steps of sugarcane processing into bioethanol in Brazil (fermentation and distillation), bioethanol transport to Europe via transoceanic ship, conversion to bio-Ethylene (via dehydration), and its subsequent oxidation and hydration to Ethylene Glycol. Combustion of bagasse from sugarcane processing generates surplus energy (electricity and heat) which were assumed to directly replace average electricity from the Brazilian grid and thermal energy produced in the same country from natural gas. Allocation between the different outputs from Ethylene oxidation (i.e. mono-Ethylene and di-Ethylene Glycol) is based on the respective economic value. The dataset is developed based on industry data from internationally adopted production processes, completed, where necessary, by literature data. Background inventories from the GaBi database are used to model the burdens of the inventoried inputs and outputs.

Transport of bio-MEG to the PEF polymerisation plant was modelled according to the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers to factories/users both located in Europe. This includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

The main process steps involved in the conversion of European starch crops (maize and wheat) into Furandicarboxylic acid (FDCA) were modelled individually, based on different data sources (as described below). The process chain includes starch production via wet milling of the different starch crops, starch conversion into glucose, production of Hydroxymethylfurfural (HMF) from glucose, and its final conversion to the monomer Furandicarboxylic acid. HMF can be technically produced from both glucose and fructose. In the short term, HMF production from fructose was suggested by some authors as a technically and economically preferable alternative, thanks to the possible integration into existing high fructose corn syrup production facilities, and subsequent lower initial capital investment (Motagamwala et al., 2019). On the other hand, according to the same authors, the glucose-based route was estimated to generate more revenues (as far as the price of fructose is nearly 400 \$/t higher than glucose). Such route was thus proposed as a possible upgrading option for any initially developed fructose-based HMF plants, once production of this precursor would have become more established. While production from fructose may be currently economically preferable, in this study the glucose-based route was considered, due to the lack of data on pure fructose production from high fructose syrups, and of suitable (e.g. sufficiently disaggregated) data for the production of high fructose syrups out of starch. Starch and glucose production were assumed to take place in the same facility, and no transport was thus modelled between these process steps. The same assumption was applied to the conversion of glucose into HMF, its subsequent oxidation to FDCA, and to the final polymerisation of FDCA with bio-MEG into PEF.

Starch production via wet milling of maize and wheat was modelled based on life cycle inventory data for relevant process steps available in different datasets from the Agri-

footprint database (v 4.0), referring to the year 2014⁴⁶. These data were then combined with background EF-compliant datasets to model the burdens of individual inputs and outputs from the specific process, reflecting EU conditions. The allocation of process burdens to the different co-products (e.g. from maize wet milling) is based on the respective economic value (based on five-year average prices), consistently with the fully vertically aggregated EF-compliant datasets available for starch production via wet milling (which are developed by the same data provider). Maize starch production data are derived from the literature, while for wheat starch a combination of data from literature and industry/industry experts is used.

The inventory of glucose production was based on the *ecoinvent* dataset "[RER] Glucose production", which represents pure glucose obtained via enzymatic hydrolysis of dried (maize) starch, and relies on literature data for the year 2014. In the implementation of the dataset in the lifecycle model, background datasets related to energy generation (i.e. electricity, thermal energy and steam) were replaced with background EF-compliant datasets, or datasets developed based on EF-compliant datasets (i.e. for steam production). In addition, the default input of maize starch was replaced with the EU-average mix of maize and wheat starch considered in this case study (i.e. 54% maize starch and 46% wheat starch; see Section 3.1). Note that the inventories applied to model starch production (described above in this section) refer to dried starch (i.e. starch with a low water content), although starch slurry with a higher water content seems to be actually used from industry as an input to the production of glucose and glucose syrups (Starch Europe, 2019). However, dried starch (@14% water content) is considered as process input in the applied *ecoinvent* data for glucose production, and datasets covering the sole production of starch slurry are only available for maize starch (but not for wheat starch). Therefore, for consistency reasons, the original input of dried starch was maintained in the inventory (and adjusted to properly reflect the water content of the starch output from the applied starch production datasets described above). This may have led to overestimating the burdens from glucose production, due to the inclusion of an additional process step that is not applied in reality.

Glucose was assumed to be transported to a separate facility where HMF is produced, subsequently oxidised to FDCA, and eventually polymerised to PEF. This transport was again modelled according to the default transport scenario described above for transferring of goods from suppliers to factories/users both located in Europe. This includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

For HMF production, no life cycle inventories are available in existing databases or in the pool of EF-compliant datasets. A new inventory was thus developed based on the theoretical results of the process simulation exercise conducted by Motagamwala et al. (2019). The resulting input and output flows were then combined with background EF-compliant datasets reflecting EU conditions to model the respective burdens. Simulation results by Motagamwala and colleagues are reported for process configurations using either fructose or glucose as a feedstock, for an output of nearly 1300 kg of HMF (at 99% purity) per hour (i.e. simulating pilot-scale conditions). As discussed above, the glucose-based route was considered in this study, where glucose is firstly isomerised to fructose, and then dehydrated to HMF. To develop the final inventory (summarised in Table 3.4), data reported in the mentioned source were complemented with additional information and data acquired via personal communications with the authors. As already discussed in Sections 3.1 and 3.4, being based on process simulation results at the pilot-scale, the developed inventory is not representative of full-scale industrial production that may be implemented in the future, and do not account for any additional (efficiency) improvement that may take place thanks to further scale-up and process optimisation

⁴⁶ For maize starch production, the following set of datasets was considered: (i) *Maize, steeped, from wet milling (receiving and steeping), at plant*; (ii) *Maize degermed, from wet milling (degermination), at plant*; (iii) *Maize starch and gluten slurry, from wet milling (grinding and screening), at plant*; (iv) *Maize starch, wet, from wet milling (gluten recovery), at plant*; and (v) *Maize starch, from wet milling (starch drying), at plant*. For wheat starch, the considered dataset is "*Wheat starch, from wet milling, at plant*".

once commercial production would be established. The applied inventory may thus have even largely overestimated the inputs and outputs (e.g. energy consumption) and the resulting environmental burdens associated with HMF production with respect to any future commercial-scale process. However, it is noted that the underlying simulation data already take into account the implementation of common initial process optimisation strategies (i.e. energy integration). On the other hand, the interpretation of simulation results and their elaboration and modelling to develop the process inventory was not always straightforward, and implied a number of assumptions and approximations, potentially leading to the modelling of inputs and outputs not fully reflecting the simulated process. In particular, the calculated amounts of energy used for refrigeration and cooling water (36.6 and 35 MJ per kg HMF, respectively) may be overestimated, as being almost comparable with the heat demand of the process (47 MJ per kg HMF) (see Table 3.4). Moreover, such energy inputs were modelled based on a proxy dataset related to cooling energy supply from natural gas (Table 3.4), which may not be sufficiently representative of the real supply/production process, especially in the case of refrigeration energy. Since these energy inputs were expected to substantially contribute to the overall process impacts, the potential impacts of PEF beverage bottles were recalculated by omitting such contributions (see the sensitivity analysis presented in Section 3.8.5.5). While this analysis considers an optimistic theoretical situation (the energy demand for refrigeration and cooling would never be equal to zero), it is useful to show how total scenario impacts would be affected if such energy demand was reduced by half or minimised (see Section 3.8.5.5 for details).

In the absence of specific data on the conversion of HMF to FDCA, the environmental burdens of this activity were approximated with those associated to PTA⁴⁷ production from para-Xylene, which is based on a similar process (i.e. oxidation in the presence of a catalyst and a solvent, typically Acetic Acid). In this respect, it should be noted that oxidation of HMF to FDCA is expected to involve lower energy and solvent requirements compared to the conversion of p-Xylene into PTA, as the process operates at lower temperatures (180 vs 210 °C) and pressures (7 bar vs 10 bar), and less Acetic Acid is lost via oxidation to CO₂ (Eerhart et al., 2012). Moreover, less air is required in the oxidation reaction to FDCA, as HMF already contains oxygen within its chemical structure. However, considering the moderate temperature and pressure differential, and the many decades of optimisation in the PTA production process, the latter was considered a reasonable (albeit potentially conservative) approximation of the FDCA production process (a similar approximation was also performed by Eerhart et al., 2012). The gate-to-gate inventory of PTA production was derived from the most recent *PlasticsEurope* ecoprofile (CPME, 2016), as implemented in the *ecoinvent* database. The inventory is based on data collected from five European PTA producers, covering 79% of the total installed production capacity in Europe, and reflecting the current average technology used in this country. No allocation nor substitution is performed, since the process delivers only PTA as an output. In the implementation of the inventory, the original para-Xylene input was replaced with HMF, considering a specific consumption equal to 1.04 kg HMF per kg of FDCA (based on average mass balance results from Eerhart et al., 2012). Moreover, input/output activity data from the ecoprofile were combined with background EF-compliant datasets for energy generation under EU conditions, and *ecoinvent* background datasets for material production.

A similar approximation to that described above for FDCA production was also applied to the final polymerisation step of FDCA and bio-MEG to produce PEF, with this process step being assimilated to that of PET polymerisation out of fossil-based PTA and MEG. Similarly to the FDCA synthesis process, polymerisation of PEF out of its co-monomers operates at lower temperatures than polymerisation of PET (nearly 30°C less), and twice as fast as the latter (Eerhart et al., 2012). However, considering also in this case the moderate temperature differential and the many decades of process optimisation, the PET polymerisation process was considered a reasonable (albeit potentially conservative)

⁴⁷ With PTA being the fossil-based alternative of FDCA, and used as a co-monomer in PET production along with fossil-based or partially bio-based MEG.

approximation of FDCA and bio-MEG polymerisation into PEF. This approximation is also in line with the assumption performed by Eerhart et al. (2012). The gate-to-gate inventory for PET polymerisation was derived from the most recent PlasticsEurope ecoprofile (CPME, 2017), as implemented in the *ecoinvent* database. The inventory is based on data collected from seven European PET producers (for a total of 12 plants), covering 85% of the total installed production capacity in Europe, and reflecting the average technology currently applied in this country. In the implementation of the inventory, the original inputs of fossil-based MEG and PTA were replaced with their bio-based alternatives (i.e. bio-MEG and FDCA), considering a specific consumption equal to 0.341 kg bio-MEG/kg PEF and 0.857 kg FDCA/kg PEF (based on average mass balance results from Eerhart et al., 2012). Moreover, input/output activity data from the ecoprofile were combined with background EF-compliant datasets for energy generation under EU conditions and *ecoinvent* background datasets for material production.

Table 3.4. Life cycle inventory of the production of 1 kg of Hydroxymethylfurfural (HMF) at 99% purity (based on Motagamwala et al., 2019).

Process section	Input / output	Dataset / Elementary flow (1)	Source (2)	Quantity	Unit	Comments
Inputs						
HMF production (glucose isomerisation + fructose dehydration)	Glucose (100% purity)	[EU-28] Glucose production, 100% purity, at plant	EI + EF	1.49	kg	Developed based on the <i>ecoinvent</i> dataset "[RER] glucose production" replacing background energy datasets with EF datasets
	Acetone (pure)	[RER] Acetone (dimethylcheton), technology mix production mix, at producer	Plastics Europe (TS)	0.0313	kg	Share of the different production technologies derived from the <i>ecoinvent</i> dataset representing acetone supply to the European market "[RER] Market for acetone, liquid"
		[RER] Acetone from isopropanol production, technology mix production mix, at plant 100% active substance	EF	0.00704	kg	
		[RER] Acetone from cumene production, technology mix production mix, at plant 100% active substance	EF	0.00211	kg	
HMF purification	MIBK	[GLO] Methyl isobutyl ketone (4-methylpentan-2-one), Technology mix Production mix, at plant	EF	0.0518	kg	-
Energy (overall process)	Electricity	[EU-28+3] Electricity grid mix 1kV-60kV AC, technology mix consumption mix, at consumer 1kV - 60kV	EF	0.0976	MJ	
	Heat	[EU-28] Steam production, as energy carrier, in chemical industry Based on foreground data from the corresponding <i>ecoinvent</i> dataset and background EF datasets, where available	EI + EF background	47.2	MJ	Developed based on the <i>ecoinvent</i> dataset "[RER] steam production, as energy carrier, in chemical industry" replacing background datasets with EF datasets (where available)
	Cooling water	[EU-28] Cooling energy, from natural gas, at cogen unit with absorption chiller 100kW	EI + EF background	35.0 (3)	MJ	Based on the <i>ecoinvent</i> dataset "[RoW] Cooling energy, from natural gas, at cogen unit with absorption chiller 100kW", adjusted to reflect EU-average conditions and replacing background datasets with EF datasets
	Refrigeration	Based on foreground data from the corresponding <i>ecoinvent</i> dataset and background EF datasets, where available		36.6 (3)	MJ	
Other inputs	Activated carbon	[RER] activated carbon production, granular from hard coal	EI	1.56E-04	kg	Assumed lifespan of the material is 3 months, after which it is sent to regeneration

Process section	Input / output	Dataset / Elementary flow (1)	Source (2)	Quantity	Unit	Comments
	Sn-Beta catalyst	[GLO] Zeolite, from aluminium hydrate, sodium silicate and sodium hydroxide single route, at plant 2- 2.5 g/cm3	EF	4.79E-03	kg	<i>Sn-containing (Beta) zeolite.</i> Approximated with zeolite production. The total amount is assumed to be replaced every 3 months and then disposed of
	Amberlyst catalyst	[RER] naphthalene sulfonic acid production, technology mix production mix, at plant 100% active substance	EF	4.22E-04	kg	<i>Benzenesulfonic acid, ethenyl-, polymer with diethenylbenzene.</i> Approximated with Naphthalene Sulfonic Acid. The total amount is assumed to be replaced every 3 months and then disposed of
Outputs						
Acetone/H ₂ O recovery	Excess H ₂ O (H ₂ O, acetone) to wastewater treatment	[EU-28] Waste water treatment, chemical reduction/oxidation process, municipal waste water production mix (region specific plants), at waste water treatment plant	TS	0.449	kg	Process-specific burdens for excess wastewater treatment. Emissions were modelled separately as reported below
	H ₂ O to water (91%)	Processed water to river	TS	0.408	kg	Only processed water
	Acetone to water (9%)	Acetone (dimethylcheton), Organic emissions to fresh water	TS	6.46E-04	kg	Based on average BOD removal efficiency (98.4%)
	CO ₂ to air	Carbon dioxide - Inorganic emissions to air	TS	0.0904	kg	CO ₂ from biodegradation of acetone (based on 98.4% removal efficiency)
	H ₂ O to water	Water - Other emissions to fresh water	EI	0.0370	kg	H ₂ O from biodegradation of acetone (based on 98.4% removal efficiency)
	Humins (to disposal)	[EU-28] Treatment of humins, municipal incineration Developed based on the Doka (2009a) tool for the modelling of material incineration in MSW incineration plants, and background EF and ecoinvent datasets (for energy and material inputs and outputs, respectively)	-	0.0862	kg	Assumed Humins composition: 56.7% C; 5.4% H; 37.9% O (Agarwal et al., 2017)
HMF purification	Purge (MIBK, H ₂ O) to wastewater treatment	[EU-28] Waste water treatment, chemical reduction/oxidation process, municipal waste water production mix (region specific plants), at waste water treatment plant	TS	0.0661	kg	Process-specific burdens for purge wastewater treatment. Emissions were modelled separately as reported below

Process section	Input / output	Dataset / Elementary flow ⁽¹⁾	Source ⁽²⁾	Quantity	Unit	Comments
	<i>MIBK to water (70.1%)</i>	<i>Methyl isobutyl ketone - Organic emissions to fresh water</i>	TS	7.41E-04	kg	Based on average BOD removal efficiency (98.4%)
	<i>CO₂ to air</i>	<i>Carbon dioxide - Inorganic emissions to air</i>	TS	0.120	kg	CO ₂ from biodegradation of MIBK (based on 98.4% removal efficiency)
	<i>H₂O to water</i>	<i>Water - Other emissions to fresh water</i>	EI	0.0492	kg	H ₂ O from biodegradation of MIBK (based on 98.4% removal efficiency)
	<i>H₂O to water (27.8%)</i>	<i>Processed water to river - Other emissions to fresh water</i>	TS	0.0184	kg	Only processed water
	<i>Acetone to water (1.6%)</i>	<i>Acetone (dimethylcheton) - Organic emissions to fresh water</i>	TS	0.0225	kg	Based on average BOD removal efficiency (98.4%)
	<i>CO₂ to air</i>	<i>Carbon dioxide - Inorganic emissions to air</i>	TS	2.37E-03	kg	CO ₂ from biodegradation of Acetone (based on 98.4% removal efficiency)
	<i>H₂O to water</i>	<i>Water - Other emissions to fresh water</i>	EI	9.69E-04	kg	H ₂ O from biodegradation of Acetone (based on 98.4% removal efficiency)
	<i>HMF (0.5%)</i>	-	-	3.30E-04	kg	No suitable elementary flows nor CFs available for HMF (negligible amount)
Other outputs	Reactivation of activated carbon	[RER] treatment of spent activated carbon, granular from hard coal, reactivation	EI	1.56E-04		Reactivation process with 10% losses according to the dataset (Bayer et al., 2005)
	Avoided virgin activated carbon production	[RER] activated carbon production, granular from hard coal	EI	1.41E-04		Net amount of activated carbon from regeneration replacing virgin production (resulting in a total virgin production of 1.56 x 10 ⁻⁵ kg)
	Disposal of Sn-Beta catalyst	[RoW] treatment of waste zeolite, inert material landfill	EI	4.79E-03		Assumed to be disposed of in landfill
	Disposal of Amberlyst catalyst	[RoW] treatment of spent anion exchange resin from potable water production, municipal incineration	EI	4.22E-04		Avoided electricity and heat generation were additionally included in the dataset: 0.5 MJ/kg electricity (net production) 1.18 MJ/kg heat (net heat)

⁽¹⁾ Dataset names reported in italic blue text refer to newly created datasets, with the rest being existing datasets from the pool of EF-compliant datasets or existing databases. Elementary flows are reported in italic black text.

⁽²⁾ Acronyms: EF = Pool of Environmental Footprint-compliant datasets; EI = *ecoinvent* database; TS = Thinkstep dataset from the GaBi database (ILCD-EL compliant).

⁽³⁾ Corresponding to the energy demand for "cooling water" and "refrigeration" reported by Motagamwala et al. (2019).

3.5.2.4 Transport of polymer granulate to the product manufacturing site

Modelling of transport of polymer granulate from the polymerisation or recycling plant (inside or outside the EU), to the bottles manufacturing and filling site in Europe, was based on the default transport scenarios (distances and vehicle types) specified in the *Plastics LCA* method for the route “supplier-to-factory”. In the case of polymers produced in Europe (i.e. all the polymers investigated in this case study except for the imported share of fossil-based PET and HDPE), the following routes were thus considered:

1. 130 km by lorry (total weight >32 t; Euro 4);
2. 240 km by train (average freight); and
3. 270 km by ship (barge).

For the imported share of fossil-based PET and HDPE (21% and 22%, respectively), a transoceanic ship transport was considered as the main transport route. The corresponding overall sea distance was determined as weighted average of the harbour-to-harbour distances between each exporting country and the EU (defined based on the calculation tool available on SeaRates.com)⁴⁸. Countries contributing to at least 90% of the overall imported quantity were considered in the calculation, leading to an overall distance equal to 10,796 km for fossil-based PET, and to 7,982 km for fossil-based HDPE (see Tables 3.5 and 3.6). Oceanic ship transport was complemented with road transport to the harbour in the single exporting countries, and from the harbour to the manufacturing site in the EU. Road transport was made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

LCIs for transport through all types of considered vehicles were available as EF-compliant datasets, which were used in the modelling.

Table 3.5. Calculation of the overall average sea distance for imports of virgin fossil-based PET to Europe.

Exporting country	Import (1) (%)	Import (% cum.)	Distance (2) (km)	Weighted distance (km)
KOREA, REPUBLIC OF (SOUTH KOREA)	29.1	29.1	16,702.31	4,852
INDIA	19.0	48.0	10,267.28	1,949
TURKEY	16.0	64.0	3,015.48	481
INDONESIA (ID+TP from 77,excl. TP -> 2001)	9.45	73.4	13,967.99	1,320
CHINA (PEOPLE'S REPUBLIC OF)	8.54	82.0	16,092.8	1,374
MEXICO	4.24	86.2	9,508.19	403
PAKISTAN	2.74	89.0	8,126.21	222
OMAN	2.46	91.4	7,877.75	194
Other countries	8.58	100	-	-
<i>Overall weighted distance</i>				<i>10,796</i>

(1) Based on *Comext* data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). The shares reported were determined as 3-year averages of import shares calculated, based on raw *Comext* data, for the years 2016-2018.

(2) From harbour to harbour, based on the calculation tool available on SeaRates.com (<https://www.searates.com/services/distances-time/>). Distances for imports from countries in the Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of PET only Mexico), Rotterdam was considered as destination port.

⁴⁸ Available at: <https://www.searates.com/services/distances-time/>

Table 3.6. Calculation of the overall average sea distance for imports of virgin fossil-based HDPE to Europe.

Exporting country	Import (1) (%)	Import (% cum.)	Distance (2) (km)	Weighted distance (km)
SAUDI ARABIA	40.4	40.4	8,767.28	3543
QATAR	11.0	51.4	8,597.23	942
KOREA, REPUBLIC OF (SOUTH KOREA)	8.63	60.0	16,702.31	1442
UNITED STATES	8.49	68.5	6,061.9	514
EGYPT	7.87	76.4	3,212.14	253
BRAZIL	5.78	82.1	10,107.97	584
MEXICO	2.35	84.5	9,508.19	224
UNITED ARAB EMIRATES	2.25	86.7	8,439.46	190
UZBEKISTAN	2.10	88.8	5,645.74	119
IRAN, ISLAMIC REPUBLIC OF	1.93	90.8	8,864.27	171
Other countries	9.23	100	-	-
<i>Overall weighted distance</i>				<i>7,982</i>

(1) Based on *Comext* data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). The shares reported were determined as 3-year averages of import shares calculated, based on raw *Comext* data, for the years 2016-2018.

(2) From harbour to harbour, based on the calculation tool available on SeaRates.com (<https://www.searates.com/services/distances-time/>). Distances for imports from countries in the Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of HDPE United States, Brazil, and Mexico), Rotterdam was considered as destination port.

3.5.3 Manufacturing Stage

Regardless of the feedstock used, manufacturing of PET beverage bottles typically takes place in two steps. First, preforms are produced via injection moulding of melted plastic granules (directly at bottling plants or, frequently, in separate facilities). Preforms are then converted into bottles through stretch-blow moulding. The same process would also apply to PEF bottles, once introduced into the market. As discussed in Section 3.3, in this case study the conversion of PET or PEF polymer granules into beverage bottles was assumed to entirely take place directly at bottling plants.

The burdens of the overall conversion process of PET or PEF granulate into bottles were modelled through the aggregated, EF-compliant dataset "[EU-28+EFTA] Stretch blow moulding; stretch blow moulding | production mix, at plant | 3% loss, 5MJ electricity consumption", which accounts for a 96.2% conversion efficiency (despite the value specified in the dataset name). Process losses (e.g. bottles with flaws) were assumed to be entirely recycled in external facilities via re-granulation into new polymer pellets, ultimately replacing virgin granules of the same material. Hence, recycled PET granules were assumed to replace virgin, fossil-based PET granules (being the estimated share of bio-based PET very low, i.e. 4%)⁴⁹, while recycled PEF granules replaced virgin PEF granules. The recycling process and the resulting virgin material substitution were modelled based on the same data as End of Life recycling of sorted, post-consumer PET or PEF bottles, in the absence of more specific data for recycling of pre-consumer,

⁴⁹ The share of bio-based PET in the EU market was estimated based on the total apparent PET consumption calculated from *Prodcum* data for the year 2015 (Eurostat, 2019d; 5323 kt), and the total bio-based PET consumption estimated for the same year in Spekrijse et al. (2019; 214.2 kt). The latter was calculated assuming that the share of the EU Bio-PET consumption amounts to 27.1% of the global market for Bio-PET in the same year (790.4 kt).

industrial scraps. The Circular Footprint Formula (CFF) was applied to calculate the actual quantities of both recycling and avoided virgin production processes to be modelled in the overall product inventory. For further detail on the modelling, including the implementation of the CFF, the reader is referred to Section 3.5.5.3 on End of Life modelling.

HDPE bottles are manufactured through a partially different process compared to PET bottles, i.e. via extrusion-blow moulding of melted HDPE granules. The process consists of two consecutive stages, including a first extrusion of melted polymer into a hollow tube (called parison), which is then transferred into a metal mould to be converted into a bottle by air inflation. The burdens of this process were modelled through the aggregated, EF-compliant dataset "[EU-28+EFTA] Blow moulding; blow moulding | production mix, at plant | PET, HDPE and PP", which was deemed a good approximation of the real process. In this case, the reported conversion efficiency of the process is equal to 99.9%. Similarly to manufacturing of PET bottles, process losses were assumed to be entirely recycled (re-granulated), with secondary HDPE granules replacing virgin, fossil-based HDPE granules (the estimated share of bio-based HDPE currently available on the market is negligible, equalling 0.2% only)⁵⁰. Further details on the modelling of the recycling process and avoided virgin material production (including the implementation of the Circular Footprint Formula) are available in Section 3.5.5.3 on End of Life modelling.

3.5.4 Distribution Stage

The transport of bottles from the manufacturing and filling site to the final user was modelled based on the default transport scenario specified in the *Plastics LCA* method for the pathway *factory* → *retail* → *final client*. The following routes were thus considered:

1. 1200 km by lorry (total weight >32 t; Euro 4) from factory to retailers;
2. 5 km by passenger car for 62% of the roundtrips from retailers to final users;
3. 5 km by van for 5% of the roundtrips from retailers to final users; and
4. no burdens assigned to 33% of the roundtrips from retailers to final users (assumed to take place with no motorised vehicles).

LCIs for transport through all types of vehicles were available as EF-compliant datasets, which were used in the modelling.

3.5.5 End of Life Stage

This section addresses the modelling of the End of Life stage of the investigated beverage bottles scenarios. In particular, Section 3.5.5.1 describes the EU-average End of Life scenario considered as a base case for the calculation of the potential impacts of the different LCA scenarios. The remaining sections (3.5.5.2 – 3.5.5.5) address the modelling of waste collection and transport, and of the different End of Life options applied. Finally, Section 3.5.5.6 provides case study-specific details on the estimate of the potential generation and release of macro-plastics at End of Life (including product litter) and of micro-plastics throughout the supply chain.

3.5.5.1 End of Life scenario

The same EU-average End of Life scenario was considered for PET bottles, regardless of the feedstock used for polymer production (fossil resources, plastic waste or biomass), as this does not affect the viable End of Life options for bottles made of this material, nor those currently applied to them. The scenario was estimated to include 60% collection for mechanical recycling, 21% incineration, and 19% landfilling. The EU-average collection

⁵⁰ The share of bio-based HDPE in the EU market was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastics, 2019; 200 kt) and the production capacity of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

rate for recycling was based on the results of the latest Annual Survey on the European PET Recycling Industry (ICIS and Petcore Europe, 2018). According to this source, out of the nearly 3.31 million tonnes (i.e. 3.308.300 t) of PET bottles placed on the European market in 2017, about 1.92 million tonnes (1.923.100 t) were collected for recycling during the same year in the Country. These figures correspond to a collection rate for recycling ("recycling input rate") of 58.2%, which was rounded to 60% for modelling purposes. Being based on most recent statistics, this estimate was applied as a replacement of the default, application-specific recycling rate (R_2) prescribed for PET bottles in Annex C of the *Plastics LCA* method (i.e. 54%, if measured at the input of the recycling process)⁵¹, which shall be applied in the absence of company-specific data. The R_2 parameter of the CFF (recycling output rate) was thus set to 46.7%, based on the sorting and recycling efficiencies assumed in this study, i.e. 91% and 85.5%, respectively (see Section 3.5.5.3 for details). Note that this value refers to PET bottles in general, and not specifically to beverage bottles. Moreover, bottles were assumed to be recycled into non-food grade PET granulate for unspecified applications such as textiles or other products than beverage bottles (no closed-loop, bottle-to-bottle recycling).

No data on the amount or share of PET bottles incinerated or landfilled in the EU were available at the time of the study. Therefore, average incineration and landfilling rates of total plastic packaging waste were estimated based on statistics on plastic packaging waste management in the EU for the years 2014-2016 (Eurostat, 2019c). Assuming that all the packaging waste sent to other recovery operations than recycling is incinerated, it was possible to estimate an incineration rate equal to 31%. Similarly, assuming that all the generated packaging waste that is not recovered is landfilled, a landfilling rate of 28% could be estimated. In relative terms, this means that on average 53% of plastic packaging waste that is not recycled was incinerated, while 47% was landfilled. The share of non-recycled PET bottles (40%) was thus assumed to be routed to incineration and landfilling according to these proportions (i.e. 21% to incineration and 19% to landfilling), with the R_3 parameter of the CFF being consequently set to 21%.

A similar approach as the one described above for PET bottles was followed also to define an EU-average End of Life scenario for HDPE bottles (from all types of feedstock), which was estimated to include 64% collection for mechanical recycling, 19% incineration, and 17% landfilling. The collection rate for recycling was defined based on the results of the analysis of plastic packaging waste flows in Europe for the year 2014 reported in Hestin et al. (2017), which is based on extrapolations at the EU level of plastic waste flows in Germany, France, UK, Spain and Italy. The analysis estimated a collection rate for recycling of the flow "HDPE bottles/flasks" equal to 76% in the case of household waste (66% of total waste), and to 40% in the case of commercial & industrial waste (34%). This corresponds to an overall collection rate for recycling of HDPE bottles equal to 64%⁵², which has been considered in this study for modelling purposes (as being more specific and representative than the material-specific R_2 value specified in Annex C of the *Plastics LCA* method). As for PET bottles, this estimate refers to HDPE bottles in general and not specifically to beverage bottles. Moreover, bottles were assumed to be recycled into non-food grade HDPE granulate for unspecified applications other than beverage bottles (such as non-food liquid containers, pots, etc.). Incineration and landfilling rates were estimated based on statistics for plastic packaging waste management at the EU-level, as described above for PET bottles. The resulting total shares are equal to 19% for incineration ($R_3 = 19\%$), and 17% for landfilling.

As for PEF bottles, the End of Life scenario was assumed to include those End of Life options that would be realistically applied today if the product was introduced in the

⁵¹ The "recycling output rate" reported in Annex C (42%), which is measured at the output of the recycling process, was converted into the corresponding "recycling input rate" considering a recycling efficiency of 85.5% and a sorting efficiency of 91%. These values are consistent with those assumed in the modelling of End of Life recycling and sorting of PET bottles, as reported in Section 3.5.5.3.

⁵² This value corresponds to a "recycling output rate" (R_2 parameter of the CFF) equalling 49%, based on the sorting and recycling efficiencies assumed in this study, i.e. 91% and 84%, respectively see Section 3.5.5.3 for details.

market in a quantity justifying separate collection for sorting and recycling into a dedicated material stream. This assumption ensures consistency with the End of Life scenarios considered for PET and HDPE bottles, but likely requires an at least partial adaptation of the existing recycling infrastructure to enable proper separation of such additional polymer stream for recycling. Therefore, it does not fully reflect the current situation, where PEF bottles would be likely sorted out as residues during separation of other relevant material streams for recycling. In the absence of specific data for PEF bottles (which are not yet on the market), the same collection rate for recycling as PET bottles (60%) was assumed to be reasonably achieved once a similar collection and recycling scheme would be established. Incineration and landfilling rates were again determined based on the relative shares estimated for these options for plastic packaging waste in the EU, as described above for PET and HDPE bottles. Overall, the assumed End of Life scenario thus included 60% collection for mechanical recycling, 21% incineration, and 10% landfilling.

3.5.5.2 Modelling of waste bottle collection and transport

In all the investigated scenarios, collection and transport of separately collected waste bottles for recycling was modelled according to the pathways, vehicle types and distances reported in Rigamonti et al. (2013) for separately collected plastic waste at the municipal level (Table 3.7). These pathways and related characteristics 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 the EU, the approximation is considered reasonable, due to the typically moderate contribution of waste collection and transport to the overall End of Life (and lifecycle) impacts (e.g. Rigamonti et al., 2014). Inventories related to the use of vehicles for collection and transport were derived from EF-compliant datasets, which were applied to each collection pathway as described in Table 3.7.

Relevant data and assumptions for the modelling of collection and transport of non-separately collected waste bottles to incineration and landfilling (as residual waste) were derived as well from Rigamonti et al. (2013), and implemented in the model as detailed in Table 3.8. Note that when EF-compliant incineration or landfilling datasets were used, no transport of collected waste bottles was separately modelled, since transport burdens are already accounted for in such aggregated datasets. In this case, only collection was modelled, following the approach described above.

Table 3.7. Modelling of source-separated plastic waste collection and transport for recycling ⁽¹⁾.

Collection type	Share (%)	Distance (km/t)	Vehicle	Share (%)	Dataset	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 containers	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 to sorting facilities						
Transport	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⁾ Based on Rigamonti et al. (2013).

⁽²⁾ Approximating delivery by car or small vans by citizens.

Table 3.8. Modelling of residual plastic waste collection and transport to incineration and landfilling ⁽¹⁾.

Collection type	Share (%)	Distance (km/t)	Vehicle	Share (%)	Dataset	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 containers	29%	7.5	Medium/large-sized truck	100%		2.2
<i>Transport to incineration and landfilling</i>						
Transport to incineration	100%	18	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	18
Transport to landfilling	100%	26.5	Large truck	100%		26.5

⁽¹⁾ Based on Rigamonti et al. (2013).

3.5.5.3 Modelling of sorting and recycling

Before recycling, bales of separately collected plastic waste containing post-consumer bottles are sorted in specific facilities. The aim of sorting is to separate plastic materials from any other co-collected materials, 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.

No EF-compliant nor ILCD-EL compliant datasets or proxy datasets were available for sorting of separately collected plastic waste. An average life cycle inventory of mixed plastic waste sorting has been developed in Franklin Associates (2018), based on input/output data collected from different dual-stream and single-stream sorting facilities in the United States. A new dataset for plastic waste sorting was thus created based on this inventory, complementing the reported input and output data with background EF datasets representative of EU-average conditions (Table 3.9). In the implementation, a 91% sorting efficiency was assumed for waste PET and HDPE bottles, according to the values reported in Antonopoulos et al. (2021). The same value was also applied to PEF bottles, in the absence of specific data. Discarded bottles were assumed to be incinerated, as together with co-combustion in cement kilns, incineration is one of the two most common fates of plastic residues from sorting and recycling operations (Rigamonti et al., 2014). No burdens from the treatment of any impurities sorted out as rejects were assigned to the bottle waste stream, in the absence of specific data on the presence of impurities and to avoid falsely “punishing” the product in scope (beverage bottles) with burdens from unrelated product waste (and mostly coming from incorrect citizen behaviour).

Table 3.9. Life cycle inventory of source-separated mixed plastic waste sorting (per kg of waste to be sorted; incineration of bottles discarded as residue is not reported, but accounted in the model).

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}	<i>ecoinvent</i>

Mechanical Recycling of PET bottles into non-food grade polymer granulate was modelled through an aggregated EF-compliant dataset representing the burdens of secondary PET granulate production out of sorted, post-consumer plastic waste via grinding, metal separation, washing, and extrusion to pellets⁵³. The dataset, developed based on literature data for these unit operations, refers to the year 2016, reflects EU background conditions, and accounts for an overall recycling efficiency 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

⁵³ The applied dataset is the following: [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 | plastic waste without metal fraction.

value are normally sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

In the absence of specific data for mechanical recycling of PEF (bottles), the same process dataset described above for PET bottles was applied as an approximation. It can indeed be reasonably expected that a recycling process relying on a similar combination of the same unit operations reported above for PET bottles recycling (i.e. grinding or shredding, metal separation, washing/flotation, and granulation) would be implemented also for PEF bottles, as far as non-food grade polymer granulate needs to be obtained.

As for mechanical recycling of HDPE bottles into non-food grade polymer granulate, no specific EF-compliant or ILCD-EL compliant dataset was available. Therefore, the *ecoinvent* dataset “[*Europe without Switzerland*] Polyethylene production, high density, granulate, recycled” was used as a basis for modelling, in combination with EF background datasets for energy and material supply under EU-average conditions⁵⁴. Since a most recent and expanded version of the original inventory data source used to develop the dataset is available (i.e. Franklin Associates, 2018, updating Franklin Associates, 2011), the inventory was adjusted according to the updated exchange values and, if needed, exchange types, reported in the latest source. These exchanges are determined as mass-weighted averages of data collected from several recycling facilities in the United States. The overall recycling efficiency is equal to 84%, with removed contaminants and process waste being sent to incineration (consistently with the assumption made in the PET bottle recycling dataset described above, and with the typical fate of recycling residues).

Recycled polymer granulate of all materials (PET, HDPE, PEF) was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system following the Circular Footprint Formula. For polymers having both a fossil-based and a bio-based alternative available on the market as of today (i.e. PET and HDPE) the current average mix between the two production routes was considered for crediting. However, the estimated share of the bio-based pathway is currently marginal, being equal to 4% in the case of PET (i.e. 96% of PET is still of fossil origin), and only to 0.2% for HDPE (which is for 99.8% fossil-based)⁵⁵. To account for the lower overall average quality of recycled polymers compared to the replaced virgin polymers, a substitution ratio equal to 0.9 was considered for both recycled PET and HDPE, according to the default values specified in Annex C of the *Plastics LCA* method for such materials when used in packaging applications. In the absence of specific values in Annex C, and for consistency reasons, the same substitution ratio was also assumed for recycled PEF when replacing virgin PEF. To model the burdens of avoided virgin polymer production and of the related feedstock supply, the same datasets (or combination of datasets) used for the modelling of upstream production of the relevant polymer and of its feedstock were applied (as described in Sections 3.5.2 and 3.5.1, respectively). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources. However, in the case of recycled PET, this implied that avoided primary

⁵⁴ Note that a generic EF-compliant dataset representing the production of a generic secondary plastic granulate from sorted post-consumer plastic waste was available, and might have been considered a reasonable approximation for the HDPE recycling process. However, the development of a material-specific inventory based on available data from the literature and other databases was considered more appropriate for increased representativeness.

⁵⁵ The share of bio-based PET in the EU market was estimated based on the total apparent PET consumption calculated from *Prodcom* data for the year 2015 (Eurostat, 2019d; 5323 kt), and the total bio-based PET consumption estimated for the same year in Spekrijse et al. (2019; 214.2 kt). The latter was calculated assuming that the share of the EU Bio-PET consumption amounts to 27.1% of the global market for Bio-PET in the same year (790.4 kt). The share of bio-based HDPE in the EU market was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastics, 2019; 200 kt) and the production capacity of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

production of bottle-grade virgin resin was modelled⁵⁶, despite the output from the considered recycling process is not intended for bottles production (i.e. no upgrading to bottle-grade quality is modelled in the applied PET recycling dataset, as PET bottles were assumed to be recycled into non-food grade resin for use in other unspecified applications than beverage bottles). Therefore, the benefits associated with PET recycling are partially overestimated in this study.

According to the Circular Footprint Formula and the related default values of the A factor specified in Annex C of the *Plastics LCA* method, only 50% of the burdens of the sorting and recycling processes, and of the benefits from avoided virgin material production, were allocated to the system. The default value of A reported in Annex C is indeed equal to 0.5 for recycled bottle-grade PET (application-specific value) and recycled PE used in unspecified applications (material-specific value). For consistency reasons, and in the absence of an existing market, the A factor was set to 0.5 also for recycled PEF, reflecting a hypothetical situation of equilibrium between supply and demand of recycled material. This is in line with the rule specified in the *Plastics in LCA* method for those situations where no default application- nor material-specific A values are available in Annex C. Note, however, that the actual market situation should be considered in any future evaluation conducted after possible commercialisation of this material.

3.5.5.4 Modelling of incineration

For conventional fossil-based polymers (i.e. PET and HDPE), aggregated material-specific incineration datasets (referring to the year 2012) are available from the pool of EF-compliant datasets. They were hence applied to model the fate of virgin and partially recycled PET and HDPE bottles in a municipal waste incineration plant⁵⁷. Similarly, for bio-based PET and bio-based HDPE bottles, partially aggregated, material-specific, ILCD-EL compliant inventories from the GaBi database (referring to the year 2018) were applied⁵⁸ (no EF-compliant datasets were available for these polymers).

All the selected datasets are developed based on a waste-specific 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 element-specific transfer coefficients (based on data from real plants, stoichiometry, or expert estimates) 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 disregarding 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 treatment technologies. For these substances

⁵⁶ Note that no (ILCD-EL compliant) datasets for non-bottle-grade PET production were available in the GaBi database. An EF-compliant dataset was available for amorphous PET, but it might have not necessarily been a suitable alternative (amorphous PET is an intermediate in the manufacture of –bottle-grade- PET resin), and its preliminary application provided distorted results (especially in the Resource Use – minerals and metals impact category). It was thus not considered a suitable alternative to the bottle-grade PET datasets applied for upstream polymer production. The application of datasets from other sources (e.g. the *ecoinvent* database) was also not considered, for consistency reasons and to avoid potentially larger distortions compared to applying EF-compliant datasets or ILCD-EL compliant datasets from the GaBi database.

⁵⁷ The dataset applied for PET incineration is "[EU-28+EFTA] Waste incineration of PET; waste-to-energy plant with dry flue gas treatment, including transport and pre-treatment | production mix, at consumer | polyethylene terephthalate waste", while for HDPE incineration the applied dataset is "[EU-28+EFTA] Waste incineration of PE; waste-to-energy plant with dry flue gas treatment, including transport and pre-treatment | production mix, at consumer | polyethylene waste".

⁵⁸ The dataset applied for Bio-PET incineration is "[EU-28] Polyethylene terephthalate (PET) (biobased) in waste incineration plant; waste-to-energy plant with dry flue gas treatment, without collection, transport and pre-treatment | production mix, at plant | Net calorific value 22.2 MJ/kg", while for Bio-HDPE incineration the applied dataset is "[EU-28] Polyethylene (PE) (biobased) in waste incineration plant; waste-to-energy plant with dry flue gas treatment, without collection, transport and pre-treatment | production mix, at plant | Net calorific value 43.5 MJ/kg".

(including HCl, HF, NO_x, VOC, N₂O, CO, NH₃, SO₂, 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 waste incineration BREF (i.e. earlier versions of Neuwahl et al., 2019; adjusted with measured concentrations from real plants), and the waste-specific flue gas production (m³/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 (SCR) and non-catalytic (SNCR) systems for NO_x reduction, affecting reagent consumption for removal of such substance and its final emission with flue gas.

In line with the approach specified in the *Plastics LCA* method to model energy recovery from waste products (i.e. the Circular Footprint Formula), the product system generating the waste material sent to incineration (i.e. the beverage bottles life cycle, in this case) was allocated the full burdens from the incineration process. However, the system was credited with 100% of the benefits from avoided production of conventional energy (electricity and heat) assumed to be replaced by energy recovered from waste. In the applied EF-compliant incineration datasets, these credits are already accounted for in the aggregated inventory, while for the selected GaBi datasets they were added to the main process inventory. In this case, the EU residual electricity grid mix (as modelled in the EF-compliant dataset "[EU-28+3] Residual grid mix; AC, technology mix | consumption mix, to consumer | 1kV - 60kV") was credited to the amount of recovered electricity. For recovered heat, a new dataset representing the current EU-average heat supply mix was created, based on background EF-compliant datasets for each specific heat source included in the mix. The EU-average mix was defined based on most recent statistics for heat generation in Europe from the International Energy Agency (IEA, 2019), and included 42.4% natural gas, 30.8% hard coal, 21.8% biomass, and 5% heavy fuel oil. In the calculation of these figures, small shares of heat generated from geothermal, nuclear, and solar thermal sources (less than 1% overall) were excluded, in the absence of specific datasets for the modelling of the respective burdens. Thermal energy from waste (11%) was also excluded, as according to the Circular Footprint Formula, the use of energy from waste in a product system shall be modelled as 100% primary energy (being the benefits of its avoided primary production entirely allocated to the system generating such energy).

For Polyethylene Furanoate (PEF), no EF-compliant or ILCD-EL compliant incineration datasets or suitable proxies were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) to model material and product incineration within municipal solid waste incineration plants. The model operates similarly to the one used to develop the EF-compliant incineration datasets described above, allowing the practitioner to account for the specific composition and energy content of the incinerated waste to develop a material-specific incineration inventory based on transfer coefficients (see Table 3.10 for an overview of the considered PEF composition and energy content). The tool also allows to adjust other technological parameters to the relevant geography or scope, including energy efficiencies, the share of alternative NO_x control technologies applied, and a few other specific parameters. In this study, energy efficiencies were adjusted to better reflect the current EU-average situation, while default values were kept for other parameters, which are representative of modern incineration plants in central and Western Europe. A gross electricity efficiency equal to 13.7% and a gross thermal efficiency of 31.8% were estimated for Europe (based on data from CEWEP, 2012), and applied in the modelling. These efficiencies inherently account for the share of waste routed to incineration plants operating without any energy recovery (estimated to be 9% for municipal waste)⁵⁹, while considering that plants with energy recovery operates with an average gross electricity efficiency equal to 15.1%, and a gross thermal efficiency

⁵⁹ Calculated as the average share of municipal waste incinerated without energy recovery over the years 2015-2017 (Eurostat, 2019b).

equal to 35% (CEWEP, 2012)⁶⁰. Benefits associated with recovered energy were modelled as described above for existing GaBi datasets (i.e. electricity from the EU residual grid mix, and thermal energy from the current EU-average mix of heat sources). In the final dataset, the inventory flows generated by applying the Doka (2009a) tool were combined with the background *ecoinvent* datasets typically applied within incineration inventories available in such database. However, for energy-related flows (including avoided energy generation), background EF-compliant datasets were applied.

Table 3.10 Elemental composition and lower heating value of waste PEF considered to model incineration and landfilling of this material ⁽¹⁾.

Element	Share (%)	Element	Share (%)
TS	100	Cd	1.54×10^{-4}
Water	0	Co	4.62×10^{-4}
VS (%TS)	98.9	Cr	1.50×10^{-3}
Ash (%TS)	1.10	Cu	9.74×10^{-3}
C fossil	-	Hg	8.43×10^{-6}
C biogenic	52.0	Mn	3.95×10^{-3}
H	3.25	Mo	4.92×10^{-4}
O	43.9	Ni	1.03×10^{-3}
Cl	1.98×10^{-1}	Pb	1.14×10^{-2}
F	1.56×10^{-3}	Sb	9.82×10^{-4}
N	9.89×10^{-2}	Si	5.13×10^{-1}
S	3.12×10^{-2}	Tl	2.46×10^{-5}
As	1.07×10^{-3}	V	4.91×10^{-3}
Br	9.84×10^{-5}	Zn	1.26×10^{-2}
LHV = 16.8 MJ/kg ⁽²⁾			

⁽¹⁾ The elemental composition of PEF was defined based on the stoichiometric content of C, H and O in the polymer, while relying for the remaining elements (essentially metals) on composition data assumed to develop the EF dataset applied in this study to model incineration of waste PET. Values in kg per tonne of material from stoichiometry and from the PET incineration dataset were initially combined, and an updated percentage composition was then calculated accordingly. This approach ensured a consistent modelling for all the investigated materials, even without the availability of a full composition analysis for post-consumer PEF (bottles) in municipal waste.

⁽²⁾ Theoretical LHV calculated based on the formula by Michel (1938) and the considered content of C, H, O, N and S in the polymer.

3.5.5.5 Modelling of landfilling

Landfilling of bottles made of conventional non-biodegradable polymers (i.e. virgin and partially recycled fossil-based PET and HDPE) was modelled based on a common aggregated EF-compliant dataset representing disposal of non-biodegradable (fossil-based) plastic waste in a managed municipal solid waste landfill, referring to the year 2012 (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific*

⁶⁰ Based on the results of energy balances conducted on more than 300 waste-to-energy plants in Europe over the period 2007-2010 (CEWEP, 2012).

sites))⁶¹. The underlying inventory is material-specific, but refers to the average chemical composition and degradability of generic plastic waste, rather than to those of the specific polymers being landfilled. This is considered an acceptable approximation for the scope of 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 (conventional) polymers, including PET and HDPE (i.e. degradation in the range of 1% over 100 years; Doka 2009b). The inventory is developed based on a landfill model applying element-specific transfer coefficients to calculate the distribution of elements in the waste composition to landfill gas and leachate, and their ultimate emission to the environment over a 100-year time horizon. Emissions occurring beyond 100 years are not accounted in the model. Landfill gas generation is calculated based on the organic carbon content in the waste material and the respective degradation rate over 100 years assumed in the model (not reported). However, for simplification reasons, an average landfill gas composition for the stable methane phase is considered. 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 rates of landfill gas, energy efficiencies of gas engines, collection rate of leachate and respective treatment efficiencies). In the selected dataset, these parameters reflect the EU-average situation as follows, considering a landfill with a height of 30 m, and an area of 40.000 m². 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. Landfill gas is collected at a rate of 50%, with the rest being directly released to air. The utilisation rate of 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 reported. 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 dried and disposed of in an underground deposit.

The same EF-compliant dataset described above for landfilling of generic plastic waste was also applied as a proxy for landfilling of bottles made of “drop-in”, non-biodegradable, bio-based polymers, i.e. partially bio-based PET and bio-based HDPE. Compared to the original dataset, emissions of CO₂ and CH₄ to air have been converted from fossil to biogenic emissions, to reflect the bio-based origin of carbon in such polymers. Since in the completely aggregated dataset it is not possible to distinguish between direct emissions from polymer degradation and those coming from background activities, the entire inventoried amounts of fossil CO₂ and CH₄ released to air were converted to biogenic emissions (disregarding at the same time the only partial biogenic origin of carbon in bio-based PET bottles). This approximation is considered acceptable, as inventoried CO₂ and CH₄ emissions are modest, amounting to only 1.4% of the carbon content in the landfilled generic plastic material, and reasonably dominated by emissions due to polymer degradation. Indeed, since the degradation rate of carbon in conventional, non-biodegradable polymers after 100 years from landfilling is typically reported to be in the range of 1% (Doka, 2009b), the contribution of emissions from background processes can be considered marginal. On the other hand, it is acknowledged that this approach is partly in favour of the two bio-based polymers.

In line with the time horizon applied for landfill emission modelling in the selected landfilling dataset, (biogenic) carbon in the landfilled polymers that is not degraded after 100 years from deposition (i.e. approximately 99% of their total carbon content) was considered to be never released from the landfill body. However, the effects of biogenic

⁶¹ While material-specific landfilling datasets are available from other databases (i.e. *ecoinvent*) for fossil-based PET and HDPE, this EF-compliant dataset for landfilling of generic plastic waste was selected, as specifically referring to EU as the reference geography (in contrast to available polymer-specific datasets), and to comply with the dataset selection “hierarchy” specified in the *Plastics LCA* method (Section 4.4.10.11).

carbon taken up during biomass growth and not released during the first 100 years of landfilling are not captured in the Climate Change impact indicator calculated for Bio-PET and Bio-HDPE bottles, since characterisation factors for biogenic CO₂ emissions and removals are set to zero in the *Plastics LCA* method (fully conforming to the PEF method). To better understand the implications of this methodological choice on the overall results, in a sensitivity analysis the Climate Change impact indicator of the two mentioned bio-based alternatives was thus recalculated accounting for the effects of non-released biogenic carbon (Section 3.8.5.7).

For bottles made of non-“drop-in”, bio-based polymers (i.e. PEF), no EF-compliant or ILCD-EL compliant datasets or suitable proxies were available. A material-specific landfilling inventory was thus developed, based on the calculation tool created by Doka (2009b) to model waste disposal into sanitary landfills. Similarly to waste incineration, the tool allows to develop material-specific landfilling inventories accounting for the specific chemical composition and other relevant chemo-physical properties of the landfilled waste (Table 3.10), as well to adjust a number of relevant site-specific and technology-specific parameters to the reference geography and to the corresponding average landfilling technology. The model hence applies element-specific transfer coefficients to define the distribution of decomposition products originating from elements in the waste composition between landfill gas and leachate, and to calculate their ultimate emission to the environment (air, surface water or groundwater). Emissions are distinguished between those taking place within the first 100 years from deposition, and delayed (“long-term”) emissions of decomposition products generated over the same timeframe, but released afterwards due to temporary storage in the landfill body (e.g. metals liberated from the waste matrix and then re-precipitated in solid form). Delayed emissions only include waterborne emissions with non-collected leachate (which are inventoried separately), while air emissions with landfill gas entirely take place over the first 100 years from deposition.

Beyond the chemical composition of the landfilled material, one of the most relevant parameters to be defined in the model is the degradability of the waste within 100 years from deposition. This parameter represents the portion of waste that is decomposed during such a timeframe, and the share of its constituents that is liberated (e.g. metals) or converted to decomposition products (e.g. to CH₄ and CO₂ in the case of carbon) within the landfill. For non-biodegradable, bio-based polymers (i.e. PEF in this case study), the degradability over 100 years from deposition was set to 1%, consistently with the value considered in the model for conventional, non-biodegradable polymers such as PET and PE. As in the datasets described above for Bio-PET and Bio-HDPE bottles, (biogenic) carbon in the landfilled polymers that is not degraded after 100 years from deposition was considered to be never released from the landfill body. Therefore, also for PEF bottles the Climate Change impact indicator was recalculated accounting for the effects of non-released biogenic carbon, to evaluate the implications of setting default characterisation factors for biogenic carbon emissions and removals to zero in the *Plastics LCA* method (see Section 3.8.5.7).

Site-specific and technology-specific parameters of the applied tool were set to reflect as much as possible the average situation at the EU level. According to the values reported in Couturier et al. (2010) for the year 2008, 49% of the generated landfill gas was assumed to be captured, the rest being directly emitted to the environment during the first 100 years from disposal. Captured landfill gas used for energy generation in stationary engines was estimated at 45% of total collected gas, while the remaining 55% was flared without any kind of energy recovery (Couturier et al., 2010). Engines were assumed to operate with a net electricity efficiency of 27.8%, and a net heat efficiency of 13.5% (according to the default values assumed in the tool). The mean annual precipitation was adjusted to 652 mm/year, while the mean annual temperature was changed to 10.6 °C. For the mean actual evapotranspiration, the default value of 500 mm/year was kept, as it is the case for other parameters not explicitly reported here (e.g. a landfill height of 20 m and a duration of the filling phase of 30 years was considered).

In the final dataset, the inventory flows generated by applying the Doka (2009b) tool were combined with the background *ecoinvent* datasets typically applied within landfilling inventories available in such database⁶². However, for energy-related flows, background EF-compliant datasets were applied.

3.5.5.6 Generation and release of macro- and micro-plastics (including product litter at End of Life)

The generation (loss) and release of macro- and micro-plastics associated with the analysed beverage bottles scenarios were estimated based on the *Plastic Leak Project (PLP) method* (Peano et al., 2020). This method distinguishes between: (a) macro- and micro-plastics "loss" ("L"), i.e. the amount of plastic material directly generated from processes and/or consumers and initially lost from the technosphere, and (b) the subsequent "release" ("R"), i.e. the amount of macro- or micro- plastics ultimately emitted to the environment. Three different compartments are considered for the final release to the environment, including ocean, freshwater sediments and the terrestrial environment (consisting of soil and other terrestrial environments)⁶³. Accounted contributions to macro-plastics loss and release are direct product littering from consumers and mismanagement of product waste at End of Life (due to, e.g., illegal dumping of uncollected waste and/or poor management of collected waste via uncontrolled landfilling or open dumping). For micro-plastics, a number of sources throughout the entire product life cycle are considered, as far as relevant to the specific supply chain. These include plastic pellets lost during conversion and handling, tyre abrasion during transport, and synthetic textiles micro-particles generated during washing activities (during textile production or use by consumers). The *PLP method* was applied according to the operational description reported Section I.3 of the *Plastics LCA method*, as briefly detailed in the rest of this section, especially focusing on the considered product- and case-specific parameters. Calculations were conducted based on a spreadsheet version of the inventories of the investigated product scenarios, rather than relying on the respective life cycle models developed within the applied LCA software (which as any other available software does not implement the *PLP method*). The focus has thus mainly been on quantifying the contribution of foreground processes and activities, while that of background processes was in most cases not quantified (i.e. wherever an aggregated dataset was used for a given foreground process, and for background processes linked to most articulated foreground disaggregated datasets). This means that the contribution of single process steps and transports modelled within vertically or horizontally aggregated foreground datasets applied throughout the inventories of the different scenarios could not be quantified. Similarly, the contribution of most background processes connected with disaggregated foreground datasets was also not accounted. In particular, the contribution to micro-plastics generation of any intermediate transport activities among the different process steps covered by vertically aggregated datasets could not be calculated, and the same applies to transport activities occurring within horizontally aggregated dataset and to any background transport included within disaggregated foreground datasets (e.g. to provide relevant material inputs). Since the life cycle models of the investigated beverage bottles scenarios rely on datasets with different levels of (vertical) aggregation (especially for the Polymer Production stage)⁶⁴, the contribution of intermediate and background (transport)

⁶² Exceptions are the inputs of diesel (burned in building machine) and pitch, which were replaced with suitable EF-compliant datasets, to improve reliability of LCIA results in the Ozone Depletion impact category.

⁶³ Note, however, that an aggregated estimate of the final release to both freshwater sediments and terrestrial environment was performed here, to reduce the number of indicators to be presented, while still distinguishing between aquatic (ocean) and non-aquatic end compartments (with the latter including both the terrestrial environment and freshwater sediments).

⁶⁴ For instance, a higher level of vertical disaggregation could be applied for bio-based HDPE and PEF bottles, where the main process steps involved in the conversion of the feedstock into the polymer, and the related intermediate transport activities, were modelled individually through specific datasets. Conversely, for fossil-based PET and HDPE, and for bio-based PET, vertically aggregated datasets had to be applied in the modelling of such processes and activities, leading to a lower level of vertical disaggregation.

processes could not be accounted in a completely consistent manner across the different scenarios. This issue is acknowledged as a limitation of the study and, partly, of the applied quantification method, as far as its direct and straightforward applicability to product inventories relying on datasets with different levels of (vertical) aggregation is concerned.

To estimate the total loss and release of macro-plastics at the End of Life stage (due to product littering and waste mismanagement), Equations I.1 and I.2 reported in the *Plastics LCA* method were applied, respectively. Beyond the default, case-unspecific parameters specified in Table I.2 of the method itself, the product-specific parameters reported in Table 3.11 were considered to apply these equations. Such parameters were defined based on the approach described in Peano et al. (2020, pp. 74-80), taking into account the size and location of use of the product, and its residual economic value after it becomes (mismanaged) waste. Considering a medium size (5-25 cm) and a use “on-the-go” for 0.5 litres bottles, a littering rate equal to 2% was selected. Moreover, considering a high residual value of littered or mismanaged waste bottles of any material, the final release rates to ocean and to the terrestrial environment were set to 10% and 5%, respectively. These rates hence do not depend on the type of feedstock used for bottles manufacturing. However, the application of identical littering and release rates does not mean that all beverage bottles scenarios contribute to macro-plastics loss and release to the same extent, as these are calculated also based on the mass of bottles used per functional unit, thus reflecting any differences in the reference flow among alternative product scenarios (this is also further discussed in Section 3.8.4).

Table 3.11. Product-specific parameters considered to apply the *PLP method* to quantify the macro-plastics loss and release of the investigated beverage bottles LCA scenarios.

Parameter ⁽¹⁾	Value
Littering rate (LR _{lit}) (%)	2
Release rate to ocean (Rel _{ocean}) (%)	10
Release rate to the terrestrial environment (Rel _{terenv}) (%) ⁽²⁾	5

⁽¹⁾ For details on the meaning of each parameter, the reader is referred to Section I.3 of the *Plastics LCA* method.

⁽²⁾ Including release to freshwater sediments, as discussed above.

As for micro-plastics, relevant sources considered in this case study include pellet losses from product manufacturing and micro-particles from tire abrasion during foreground road transport (no textiles are used in the foreground system). The contribution of these sources to the total value-chain loss and release of micro-plastics to ocean and to the terrestrial environment was estimated according to Equations I.3-I.6 of the *Plastics LCA* method, considering the default source- and pathway-specific parameters specified in Tables I.3-I.5 of the method itself. No product-specific parameters had to be determined, as the only case-specific parameter linking the different equations to the specific product inventory (and hence to the functional unit of each scenario) is either the amount of plastic pellets entering the product manufacturing process, or the mass of product/material transferred along each foreground road transport route and the related distance (all expressed per functional unit)⁶⁵. Apart from these parameters, the quantification was thus made by means of default parameters that are not affected by the type of product, polymer or feedstock source.

3.5.6 Calculation of the Climate Change impact from iLUC

As a base case, the potential Climate Change impact from indirect Land Use Change (iLUC) associated with the investigated bio-based or partially bio-based beverage bottles

⁶⁵ An exception is the Average Vehicle Load (kg), which depending on the situation may be considered a value-chain specific parameter. However, a unique average default value was considered in this case study, as specified in Table I.3 of the *Plastics LCA* method.

scenarios was calculated according to the approach outlined in Section 4.4.15.3 of the *Plastics LCA* method. A sensitivity analysis applying an alternative method and resulting emission factors was also performed, as described in Section 3.8.5.6.

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), as recommended in the *Plastics LCA* method, the specific land demand of the crop(s) used as feedstock for each bio-based polymer ($\text{m}^2 \cdot \text{year} / \text{kg crop}$) was calculated first. The calculation was based on the total aggregated amount of arable and agriculture land occupation flows reported in the dataset used to model the production of the specific crop, considering only those flows referring to the country where the crop is grown (e.g. Brazil for sugarcane). If the geography of such flows was not specified, all arable and agricultural land occupation flows reported in the dataset were aggregated. The obtained estimates were checked against the values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), and both calculation routes were found to deliver generally aligned results (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows reported in the datasets were ultimately considered, to keep consistency with the actual data applied in the modelling of the investigated scenarios.

The specific land demand for crop production was then converted into a demand per functional unit (FU) ($\text{m}^2 \cdot \text{year} / \text{FU}$), based on the specific crop consumption for polymer production ($\text{kg crop} / \text{kg polymer}$)⁶⁶ and the amount of polymer needed to fulfil the functional unit (reference flow) in the specific scenario ($\text{kg polymer} / \text{FU}$). The potential Climate Change impact from iLUC was finally calculated by applying the recalculated GHG emission factors from the EU 2015/1513 Directive ($\text{kg CO}_2 \text{ eq.} / \text{m}^2 \cdot \text{y}$) to the estimated land demand per functional unit. All the described calculation steps to estimate the potential Climate Change impact due to iLUC are summarised in Table 3.12.

⁶⁶ Defined or calculated consistently with the data applied in the modelling of the Polymer Production stage, as described in Section 3.5.2.3.

Table 3.12. Calculation of the potential Climate Change impact due to GHG emissions from iLUC associated with beverage bottles LCA scenarios relying on bio-based polymers.

Scenario / Polymer	Feedstock	Land demand for crop production ⁽¹⁾ [m²·y/kg_{crop}]	Crop demand for polymer production [kg_{crop}/kg_{polymer}]	Polymer demand per functional unit (FU) [kg_{polymer}/FU]	iLUC GHG emission factor [kg CO₂ eq./ (m²·y)]	iLUC Climate Change impact [kg CO₂ eq./FU]
S5 – 30% bio-based PET bottles	Sugarcane (BR)	0.117 (0.135)	4.27	49.9	0.176	4.38
S6 – Bio-based HDPE bottles	Sugarcane (BR)	0.117 (0.135)	28.6	53.1	0.176	31.3
S7 – PEF bottles	Sugarcane (BR)	0.117 (0.135)	4.3	39.9	0.176	3.53
	Maize (EU)	1.34 (1.37)	0.577	39.9	0.0612	1.89
	Wheat (EU)	1.52 (1.75)	0.548	39.9	0.0612	2.04
	<i>Total</i>					

⁽¹⁾ Calculated based on arable and agriculture land occupation exchanges reported in the dataset applied to model production of the specific crop, considering only those flows referring to the country of cultivation (or all reported flows, if the country was not specified). Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

3.6 Life Cycle Impact Assessment results

The characterised, normalised and weighted impact assessment results of the investigated product scenarios are reported in Tables 3.13–3.19. For characterised results, the contribution of the main life cycle stages is also reported, and further illustrated in Figures D.1.1–D.1.3 in Annex D.1. Consistently with the applied system boundary, the considered contributions include:

- Feedstock Supply, i.e. depending on the feedstock/scenario: (i) oil/natural gas extraction, processing, transport and possible refining, as well as transport of naphtha from refinery to downstream users (fossil-based polymers); (ii) collection, transport and sorting of post-consumer plastic waste (recycled polymers); or (iii) crop cultivation and transport to further processing (bio-based polymers);
- Polymer Production, i.e. all gate-to-gate activities carried out to convert or recycle relevant feedstock materials into the specific polymer, including any transport among these activities and transport of polymer granulate to the beverage bottles manufacturing and filling site;
- Manufacturing, i.e. conversion of the polymer into beverage bottles by injection-stretch-blow moulding (PET and PEF bottles) or extrusion-blow moulding (HDPE bottles);
- Distribution, i.e. transport of beverage bottles from the manufacturing and filling site to the final user; and
- End of Life, i.e. waste bottles collection, transport and treatment or disposal, as well as any avoided processes from downstream displacement of virgin materials and energy. This contribution hence represents the net impact from the End of Life stage, resulting from the balance between real burdens of the applied waste management activities and resulting benefits (if any).

The last row of Tables 3.13–3.19 also reports the total weighted impact score (single score) of individual scenarios, calculated by aggregating normalised and weighted impact assessment results across all impact categories. Single impact scores provide a more immediate and synthetic representation of the overall (relative) environmental performance of the analysed product scenarios. However, they are affected by greater uncertainty (due to the application of additional normalisation and weighting factors), and by value choices necessarily applied to define weighting factors establishing an order of relevance of the different impact categories in a European decision context. Note that all the results presented in this section are affected by the limitations and critical assumptions discussed in Section 3.4, and shall be interpreted taking them carefully into account.

Table 3.13. Characterised, normalised and weighted impact assessment results of fossil-based PET beverage bottles (per functional unit).

Impact category ⁽¹⁾	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.62E+01	1.36E+02	1.32E+01	3.09E+01	-5.55E+00	1.91E+02	2.46E-02	5.46E-01
<i>Climate Change (fossil) [kg CO₂ eq.] ⁽²⁾</i>						<i>1.91E+02</i>	<i>2.46E-02</i>	<i>5.46E-01</i>
<i>Climate Change (biogenic) [kg CO₂ eq.] ⁽²⁾</i>						<i>1.51E-01</i>	<i>1.95E-05</i>	<i>4.32E-04</i>
<i>Climate Change (land use and land use change) [kg CO₂ eq.] ⁽²⁾</i>						<i>-3.91E-02</i>	<i>-5.04E-06</i>	<i>-1.12E-04</i>
Ozone Depletion [kg CFC-11 eq.] (I)	7.64E-09	4.87E-11	5.16E-09	7.35E-11	1.66E-08	2.95E-08	1.26E-06	8.53E-06
Human Toxicity - cancer [CTUh] (III)	8.19E-07	2.61E-07	6.02E-11	3.43E-07	-2.04E-07	1.22E-06	3.17E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	2.77E-06	1.34E-06	2.68E-07	1.41E-06	-7.05E-07	5.09E-06	1.07E-02	0.00E+00
Particulate matter [Disease incidence] (I)	5.27E-07	2.08E-06	3.96E-07	8.30E-07	-7.90E-07	3.04E-06	4.77E-03	4.56E-02
Ionising Radiation [kBq U ²³⁵ eq.] (II)	3.09E-01	7.23E+00	5.83E+00	6.49E-02	-1.85E+00	1.16E+01	2.75E-03	1.48E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	6.17E-02	2.26E-01	2.46E-02	8.70E-02	-4.94E-02	3.50E-01	8.62E-03	4.40E-02
Acidification [mol of H ⁺ eq.] (II)	8.09E-02	2.52E-01	3.66E-02	1.07E-01	-5.73E-02	4.19E-01	7.54E-03	5.01E-02
Eutrophication - terrestrial [mol N eq.] (II)	1.45E-01	7.79E-01	1.01E-01	4.64E-01	-1.37E-01	1.35E+00	7.63E-03	2.98E-02
Eutrophication - freshwater [kg P eq.] (II)	6.70E-05	1.51E-04	3.79E-05	1.63E-04	1.65E-04	5.83E-04	2.28E-04	6.74E-04
Eutrophication - marine [kg N eq.] (II)	1.32E-02	7.13E-02	9.45E-03	3.75E-02	-1.53E-02	1.16E-01	4.10E-03	1.28E-02
Ecotoxicity - freshwater [CTUe] (III)	1.64E+01	5.65E+00	3.28E-01	6.58E+00	-4.21E+00	2.48E+01	2.10E-03	0.00E+00
Land Use [Pt] (III)	9.50E+00	1.95E+02	1.27E+02	1.75E+02	-6.91E+01	4.37E+02	3.27E-04	2.76E-03
Water Use [m ³ world eq.] (III)	2.43E-01	4.38E+01	1.19E+00	2.64E+00	-5.93E+00	4.20E+01	3.66E-03	3.31E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	4.40E-06	8.98E-06	4.56E-06	2.25E-06	-2.31E-06	1.79E-05	3.09E-04	2.50E-03
Resource Use - fossils [MJ] (III)	1.92E+03	2.08E+03	2.02E+02	4.02E+02	-8.64E+02	3.74E+03	5.73E-02	5.11E-01
<i>Total weighted impact (single score)</i>								<i>1.29E+00</i>

⁽¹⁾ Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

⁽²⁾ For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 3.14. Characterised, normalised and weighted impact assessment results of fossil-based HDPE beverage bottles (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	3.49E+01	6.35E+01	6.93E+00	3.41E+01	-3.71E-02	1.39E+02	1.79E-02	3.98E-01
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						<i>1.39E+02</i>	<i>1.79E-02</i>	<i>3.98E-01</i>
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						<i>8.76E-02</i>	<i>1.13E-05</i>	<i>2.51E-04</i>
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						<i>8.87E-02</i>	<i>1.14E-05</i>	<i>2.54E-04</i>
Ozone Depletion [kg CFC-11 eq.] (I)	4.48E-08	5.05E-11	2.47E-09	8.12E-11	4.51E-08	9.25E-08	3.96E-06	2.67E-05
Human Toxicity - cancer [CTUh] (III)	1.02E-06	3.04E-07	7.46E-09	3.79E-07	-2.80E-07	1.43E-06	3.71E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	3.66E-06	1.39E-06	1.81E-07	1.56E-06	-9.63E-07	5.82E-06	1.23E-02	0.00E+00
Particulate matter [Disease incidence] (I)	9.09E-07	1.85E-06	2.37E-07	9.16E-07	-7.68E-07	3.15E-06	4.95E-03	4.72E-02
Ionising Radiation [kBq U ²³⁵ eq.] (II)	1.09E+00	4.01E+00	2.71E+00	7.17E-02	-3.52E+00	4.37E+00	1.04E-03	5.56E-03
Photochemical Ozone Formation [kg NMVOC eq.] (II)	1.05E-01	2.08E-01	1.37E-02	9.61E-02	-7.18E-02	3.51E-01	8.64E-03	4.41E-02
Acidification [mol of H ⁺ eq.] (II)	1.40E-01	1.73E-01	2.25E-02	1.18E-01	-7.91E-02	3.74E-01	6.73E-03	4.47E-02
Eutrophication - terrestrial [mol N eq.] (II)	2.42E-01	5.52E-01	5.47E-02	5.12E-01	-1.52E-01	1.21E+00	6.84E-03	2.67E-02
Eutrophication - freshwater [kg P eq.] (II)	3.86E-04	6.35E-05	1.62E-05	1.80E-04	7.66E-05	7.22E-04	2.83E-04	8.35E-04
Eutrophication - marine [kg N eq.] (II)	2.19E-02	5.16E-02	5.11E-03	4.14E-02	-1.63E-02	1.04E-01	3.68E-03	1.15E-02
Ecotoxicity - freshwater [CTUe] (III)	2.13E+01	7.79E+00	3.25E-01	7.26E+00	-5.83E+00	3.08E+01	2.61E-03	0.00E+00
Land Use [Pt] (III)	3.18E+01	1.14E+02	7.10E+01	1.93E+02	-4.95E+01	3.60E+02	2.70E-04	2.27E-03
Water Use [m ³ world eq.] (III)	7.78E-01	1.79E+01	9.21E-01	2.92E+00	5.83E-02	2.25E+01	1.96E-03	1.77E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	8.85E-06	4.03E-06	2.04E-06	2.48E-06	-2.76E-06	1.46E-05	2.52E-04	2.04E-03
Resource Use - fossils [MJ] (III)	2.93E+03	8.93E+02	1.16E+02	4.43E+02	-1.10E+03	3.28E+03	5.03E-02	4.48E-01
<i>Total weighted impact (single score)</i>								<i>1.05E+00</i>

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 3.15. Characterised, normalised and weighted impact assessment results of 24% recycled PET beverage bottles (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.54E+01	1.21E+02	1.32E+01	3.09E+01	-5.55E+00	1.75E+02	2.26E-02	5.01E-01
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						1.72E+02	2.22E-02	4.92E-01
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						4.07E-01	5.25E-05	1.16E-03
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						-6.53E-02	-8.42E-06	-1.87E-04
Ozone Depletion [kg CFC-11 eq.] (I)	1.73E-08	1.33E-07	5.16E-09	7.35E-11	1.66E-08	1.72E-07	7.36E-06	4.97E-05
Human Toxicity - cancer [CTUh] (III)	7.21E-07	2.42E-07	6.02E-11	3.43E-07	-2.04E-07	1.10E-06	2.86E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	2.45E-06	1.46E-06	2.68E-07	1.41E-06	-7.05E-07	4.88E-06	1.03E-02	0.00E+00
Particulate matter [Disease incidence] (I)	4.70E-07	1.86E-06	3.96E-07	8.30E-07	-7.90E-07	2.77E-06	4.35E-03	4.15E-02
Ionising Radiation [kBq U ²³⁵ eq.] (II)	1.99E-01	6.79E+00	5.83E+00	6.49E-02	-1.85E+00	1.10E+01	2.61E-03	1.40E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	5.50E-02	1.97E-01	2.46E-02	8.70E-02	-4.94E-02	3.15E-01	7.76E-03	3.96E-02
Acidification [mol of H ⁺ eq.] (II)	7.13E-02	2.25E-01	3.66E-02	1.07E-01	-5.73E-02	3.82E-01	6.88E-03	4.57E-02
Eutrophication - terrestrial [mol N eq.] (II)	1.31E-01	6.95E-01	1.01E-01	4.64E-01	-1.37E-01	1.25E+00	7.07E-03	2.76E-02
Eutrophication - freshwater [kg P eq.] (II)	6.23E-05	2.48E-04	3.79E-05	1.63E-04	1.65E-04	6.76E-04	2.65E-04	7.81E-04
Eutrophication - marine [kg N eq.] (II)	1.19E-02	6.48E-02	9.45E-03	3.75E-02	-1.53E-02	1.08E-01	3.82E-03	1.19E-02
Ecotoxicity - freshwater [CTUe] (III)	1.45E+01	5.95E+00	3.28E-01	6.58E+00	-4.21E+00	2.31E+01	1.96E-03	0.00E+00
Land Use [Pt] (III)	8.95E+00	2.27E+02	1.27E+02	1.75E+02	-6.91E+01	4.69E+02	3.51E-04	2.96E-03
Water Use [m ³ world eq.] (III)	3.52E-01	3.87E+01	1.19E+00	2.64E+00	-5.93E+00	3.70E+01	3.23E-03	2.91E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	3.86E-06	1.08E-05	4.56E-06	2.25E-06	-2.31E-06	1.92E-05	3.32E-04	2.68E-03
Resource Use - fossils [MJ] (III)	1.68E+03	1.81E+03	2.02E+02	4.02E+02	-8.64E+02	3.23E+03	4.95E-02	4.42E-01
<i>Total weighted impact (single score)</i>								1.16E+00

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 3.16. Characterised, normalised and weighted impact assessment results of 16% recycled HDPE beverage bottles (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	3.28E+01	6.03E+01	6.93E+00	3.41E+01	-3.71E-02	1.34E+02	1.73E-02	3.83E-01
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						<i>1.33E+02</i>	<i>1.71E-02</i>	<i>3.80E-01</i>
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						<i>2.70E-01</i>	<i>3.48E-05</i>	<i>7.72E-04</i>
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						<i>8.30E-02</i>	<i>1.07E-05</i>	<i>2.37E-04</i>
Ozone Depletion [kg CFC-11 eq.] (I)	4.86E-08	9.50E-08	2.47E-09	8.12E-11	4.51E-08	1.91E-07	8.18E-06	5.52E-05
Human Toxicity - cancer [CTUh] (III)	9.41E-07	2.96E-07	7.46E-09	3.79E-07	-2.80E-07	1.34E-06	3.48E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	3.36E-06	1.48E-06	1.81E-07	1.56E-06	-9.63E-07	5.62E-06	1.18E-02	0.00E+00
Particulate matter [Disease incidence] (I)	8.32E-07	1.77E-06	2.37E-07	9.16E-07	-7.68E-07	2.99E-06	4.70E-03	4.48E-02
Ionising Radiation [kBq U ²³⁵ eq.] (II)	8.73E-01	4.00E+00	2.71E+00	7.17E-02	-3.52E+00	4.14E+00	9.81E-04	5.27E-03
Photochemical Ozone Formation [kg NMVOC eq.] (II)	9.69E-02	1.95E-01	1.37E-02	9.61E-02	-7.18E-02	3.30E-01	8.13E-03	4.14E-02
Acidification [mol of H ⁺ eq.] (II)	1.28E-01	1.65E-01	2.25E-02	1.18E-01	-7.91E-02	3.54E-01	6.37E-03	4.23E-02
Eutrophication - terrestrial [mol N eq.] (II)	2.23E-01	5.22E-01	5.47E-02	5.12E-01	-1.52E-01	1.16E+00	6.56E-03	2.56E-02
Eutrophication - freshwater [kg P eq.] (II)	3.57E-04	1.55E-04	1.62E-05	1.80E-04	7.66E-05	7.85E-04	3.08E-04	9.07E-04
Eutrophication - marine [kg N eq.] (II)	2.01E-02	4.96E-02	5.11E-03	4.14E-02	-1.63E-02	9.99E-02	3.53E-03	1.10E-02
Ecotoxicity - freshwater [CTUe] (III)	1.96E+01	8.31E+00	3.25E-01	7.26E+00	-5.83E+00	2.97E+01	2.51E-03	0.00E+00
Land Use [Pt] (III)	2.88E+01	1.44E+02	7.10E+01	1.93E+02	-4.95E+01	3.88E+02	2.91E-04	2.45E-03
Water Use [m ³ world eq.] (III)	8.37E-01	1.68E+01	9.21E-01	2.92E+00	5.83E-02	2.16E+01	1.88E-03	1.70E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	8.10E-06	5.78E-06	2.04E-06	2.48E-06	-2.76E-06	1.56E-05	2.70E-04	2.18E-03
Resource Use - fossils [MJ] (III)	2.69E+03	8.45E+02	1.16E+02	4.43E+02	-1.10E+03	2.99E+03	4.58E-02	4.09E-01
<i>Total weighted impact (single score)</i>								<i>9.85E-01</i>

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 3.17. Characterised, normalised and weighted impact assessment results of 30% bio-based PET beverage bottles (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	4.55E+01	1.17E+02	1.32E+01	3.09E+01	-1.04E+01	1.96E+02	2.53E-02	5.61E-01
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						1.72E+02	2.22E-02	4.92E-01
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						3.22E+00	4.15E-04	9.21E-03
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						2.78E+01	3.58E-03	7.95E-02
Ozone Depletion [kg CFC-11 eq.] (I)	9.95E-11	-4.43E-11	5.16E-09	7.35E-11	1.61E-08	2.14E-08	9.16E-07	6.19E-06
Human Toxicity - cancer [CTUh] (III)	7.89E-07	1.69E-07	6.02E-11	3.43E-07	-2.03E-07	1.10E-06	2.86E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	4.15E-06	1.79E-06	2.68E-07	1.41E-06	-8.25E-07	6.79E-06	1.43E-02	0.00E+00
Particulate matter [Disease incidence] (I)	1.02E-05	2.15E-06	3.96E-07	8.30E-07	-6.04E-07	1.30E-05	2.04E-02	1.95E-01
Ionising Radiation [kBq U ²³⁵ eq.] (II)	1.47E-01	7.12E+00	5.83E+00	6.49E-02	-2.40E+00	1.08E+01	2.56E-03	1.37E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	1.40E-01	2.33E-01	2.46E-02	8.70E-02	-5.20E-02	4.33E-01	1.07E-02	5.44E-02
Acidification [mol of H ⁺ eq.] (II)	1.46E-01	2.94E-01	3.66E-02	1.07E-01	-6.39E-02	5.19E-01	9.34E-03	6.20E-02
Eutrophication - terrestrial [mol N eq.] (II)	4.96E-01	8.18E-01	1.01E-01	4.64E-01	-1.50E-01	1.73E+00	9.78E-03	3.82E-02
Eutrophication - freshwater [kg P eq.] (II)	1.37E-03	1.48E-04	3.79E-05	1.63E-04	1.54E-04	1.87E-03	7.33E-04	2.16E-03
Eutrophication - marine [kg N eq.] (II)	3.45E-01	7.47E-02	9.45E-03	3.75E-02	-1.67E-02	4.50E-01	1.59E-02	4.97E-02
Ecotoxicity - freshwater [CTUe] (III)	1.65E+01	5.16E+00	3.28E-01	6.58E+00	-4.28E+00	2.42E+01	2.05E-03	0.00E+00
Land Use [Pt] (III)	6.42E+03	1.85E+02	1.27E+02	1.75E+02	-2.38E+02	6.67E+03	5.00E-03	4.21E-02
Water Use [m ³ world eq.] (III)	4.08E-02	3.99E+01	1.19E+00	2.64E+00	-6.21E+00	3.75E+01	3.27E-03	2.95E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	5.69E-06	7.91E-06	4.56E-06	2.25E-06	-2.54E-06	1.79E-05	3.09E-04	2.50E-03
Resource Use - fossils [MJ] (III)	1.45E+03	1.75E+03	2.02E+02	4.02E+02	-8.63E+02	2.94E+03	4.51E-02	4.02E-01
<i>Total weighted impact (single score)</i>								1.45E+00

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 3.18. Characterised, normalised and weighted impact assessment results of bio-based HDPE beverage bottles (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	2.52E+02	8.38E+01	6.93E+00	3.41E+01	-3.39E+01	3.43E+02	4.42E-02	9.81E-01
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						1.61E+02	2.08E-02	4.60E-01
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						2.17E+01	2.80E-03	6.21E-02
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						1.97E+02	2.54E-02	5.63E-01
Ozone Depletion [kg CFC-11 eq.] (I)	6.98E-13	2.57E-07	2.47E-09	8.12E-11	4.43E-08	3.04E-07	1.30E-05	8.79E-05
Human Toxicity - cancer [CTUh] (III)	1.04E-06	8.91E-07	7.46E-09	3.79E-07	-2.81E-07	2.04E-06	5.30E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	1.43E-05	1.75E-05	1.81E-07	1.56E-06	-1.23E-06	3.23E-05	6.80E-02	0.00E+00
Particulate matter [Disease incidence] (I)	7.01E-05	3.68E-05	2.37E-07	9.16E-07	-9.20E-07	1.07E-04	1.68E-01	1.60E+00
Ionising Radiation [kBq U ²³⁵ eq.] (II)	1.54E-01	9.85E+00	2.71E+00	7.17E-02	-4.60E+00	8.18E+00	1.94E-03	1.04E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	6.88E-01	8.88E-01	1.37E-02	9.61E-02	-7.84E-02	1.61E+00	3.97E-02	2.02E-01
Acidification [mol of H ⁺ eq.] (II)	6.33E-01	1.05E+00	2.25E-02	1.18E-01	-9.38E-02	1.73E+00	3.11E-02	2.07E-01
Eutrophication - terrestrial [mol N eq.] (II)	2.80E+00	3.47E+00	5.47E-02	5.12E-01	-1.81E-01	6.65E+00	3.76E-02	1.47E-01
Eutrophication - freshwater [kg P eq.] (II)	9.73E-03	2.02E-03	1.62E-05	1.80E-04	5.64E-05	1.20E-02	4.70E-03	1.39E-02
Eutrophication - marine [kg N eq.] (II)	2.39E+00	3.13E-01	5.11E-03	4.14E-02	-1.89E-02	2.73E+00	9.65E-02	3.01E-01
Ecotoxicity - freshwater [CTUe] (III)	2.63E+01	3.08E+01	3.25E-01	7.26E+00	-5.97E+00	5.87E+01	4.97E-03	0.00E+00
Land Use [Pt] (III)	4.58E+04	9.75E+02	7.10E+01	1.93E+02	-3.91E+02	4.66E+04	3.49E-02	2.94E-01
Water Use [m ³ world eq.] (III)	-5.02E-01	6.99E+01	9.21E-01	2.92E+00	-4.48E-01	7.28E+01	6.35E-03	5.73E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	2.00E-05	3.68E-05	2.04E-06	2.48E-06	-3.19E-06	5.81E-05	1.00E-03	8.11E-03
Resource Use - fossils [MJ] (III)	2.12E+02	1.20E+03	1.16E+02	4.43E+02	-1.10E+03	8.63E+02	1.32E-02	1.18E-01
<i>Total weighted impact (single score)</i>								3.94E+00

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 3.19. Characterised, normalised and weighted impact assessment results of PEF beverage bottles (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	5.11E+01	5.85E+02	2.86E+00	2.47E+01	-1.27E+02	5.37E+02	6.92E-02	1.54E+00
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						<i>5.18E+02</i>	<i>6.68E-02</i>	<i>1.48E+00</i>
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						<i>1.37E+00</i>	<i>1.77E-04</i>	<i>3.92E-03</i>
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						<i>1.78E+01</i>	<i>2.29E-03</i>	<i>5.09E-02</i>
Ozone Depletion [kg CFC-11 eq.] (I)	1.43E-08	4.28E-06	-5.95E-08	5.88E-11	-8.38E-07	3.39E-06	1.45E-04	9.80E-04
Human Toxicity - cancer [CTUh] (III)	1.27E-06	2.22E-06	-3.93E-08	2.74E-07	-6.17E-07	3.11E-06	8.08E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	5.15E-05	2.23E-05	-8.33E-07	1.13E-06	-1.13E-05	6.28E-05	1.32E-01	0.00E+00
Particulate matter [Disease incidence] (I)	1.15E-05	9.77E-06	2.63E-08	6.64E-07	-4.31E-06	1.76E-05	2.76E-02	2.64E-01
Ionising Radiation [kBq U ²³⁵ eq.] (II)	1.02E+00	3.05E+01	4.28E+00	5.20E-02	-6.40E+00	2.95E+01	6.99E-03	3.75E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	1.45E-01	8.76E-01	7.39E-03	6.96E-02	-2.05E-01	8.93E-01	2.20E-02	1.12E-01
Acidification [mol of H ⁺ eq.] (II)	5.31E-01	1.16E+00	7.36E-03	8.54E-02	-3.44E-01	1.44E+00	2.59E-02	1.72E-01
Eutrophication - terrestrial [mol N eq.] (II)	2.34E+00	2.90E+00	1.14E-02	3.71E-01	-1.05E+00	4.57E+00	2.58E-02	1.01E-01
Eutrophication - freshwater [kg P eq.] (II)	6.92E-03	4.56E-02	-7.47E-04	1.31E-04	-1.06E-02	4.13E-02	1.62E-02	4.77E-02
Eutrophication - marine [kg N eq.] (II)	6.41E-01	3.93E-01	-7.02E-03	3.00E-02	-2.06E-01	8.51E-01	3.01E-02	9.39E-02
Ecotoxicity - freshwater [CTUe] (III)	5.59E+02	2.18E+02	-1.10E+01	5.26E+00	-1.48E+02	6.23E+02	5.27E-02	0.00E+00
Land Use [Pt] (III)	1.30E+04	1.13E+03	-1.06E+02	1.40E+02	-2.94E+03	1.12E+04	8.39E-03	7.07E-02
Water Use [m ³ world eq.] (III)	1.13E+02	1.47E+02	-2.40E+00	2.11E+00	-5.21E+01	2.08E+02	1.81E-02	1.64E-01
Resource Use - mineral and metals [kg Sb eq.] (III)	1.01E-05	1.12E-03	-1.29E-05	1.80E-06	-2.27E-04	8.87E-04	1.53E-02	1.24E-01
Resource Use - fossils [MJ] (III)	2.16E+02	8.97E+03	7.25E+01	3.21E+02	-1.88E+03	7.70E+03	1.18E-01	1.05E+00
<i>Total weighted impact (single score)</i>								<i>3.78E+00</i>

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

3.7 Additional Environmental Information

This section presents the results related to additional environmental impacts or aspects going beyond the default set of impact categories considered in the *Plastics LCA* method, but that are considered relevant for the investigated product category. Additional environmental impacts and aspects addressed in this study include: (i) the potential impact on Climate Change due to GHG emissions from indirect Land Use Change (iLUC); (ii) potential Biodiversity impacts occurring at the endpoint level due to a number of relevant midpoint impact categories; (iii) the generation and release of macro-plastics at End of Life (including product litter); as well as (iv) the generation and release of micro-plastics throughout the product life cycle.

3.7.1 iLUC impact on Climate Change

Table 3.20 presents the estimated potential impact on Climate Change due to GHG emissions from iLUC expected to occur as a consequence of bio-based feedstock supply in the investigated beverage bottles scenarios. The total Climate Change impact accounting for such additional contribution is also reported where relevant, for each product scenario.

Table 3.20. Potential Climate Change impact of GHG emissions from iLUC and resulting total Climate Change impact of beverage bottles LCA scenarios. Results are not intended to compare the different scenarios.

Scenario	iLUC Climate Change impact [kg CO ₂ eq./FU]	Total Climate Change impact (incl. iLUC) ⁽¹⁾ [kg CO ₂ eq./FU]
S1 – Fossil-based PET bottles	-	(191)
S2 – Fossil-based HDPE bottles	-	(139)
S3 – 24% R-PET bottles	-	(175)
S4 – 16% R-HDPE bottles	-	(134)
S5 – 30% bio-based PET bottles	4.38	201 (196)
S6 – Bio-based HDPE bottles	31.3	374 (343)
S7 – PEF bottles	7.46	544 (537)

⁽¹⁾ Values in parenthesis refer to the total Climate Change impact of scenarios, without the iLUC contribution.

3.7.2 Biodiversity impacts

Potential Biodiversity impacts estimated for the investigated beverage bottles scenarios, expressed as potential loss of animal and vegetal species per year, are presented in Table 3.21. The impact is quantified through an endpoint-level impact indicator accounting for a number of determining midpoint impact categories, including Climate Change, Photochemical Ozone Formation, Terrestrial Acidification, Eutrophication (freshwater and marine), Ecotoxicity (terrestrial, freshwater and marine), Land Use and Water Use. However, it is important to note that the impact assessment methods applied to some of these underlying midpoint impact categories differ from those prescribed in the *Plastics LCA* method (where impacts are assessed at the midpoint level). Moreover, direct potential biodiversity impacts from oil leakage are not quantified (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 3.5.1.1).

Table 3.21. Potential biodiversity impact of beverage bottles LCA scenarios, expressed as potential loss of animal and vegetal species per year (species*year) per functional unit. Results are not intended to compare the different scenarios.

Scenario	Total	Feedstock supply	Polymer production	Manu-facturing	Distribution	End of Life
S1 – Fossil-based PET bottles	7.28E-07	1.37E-07	4.60E-07	4.99E-08	1.23E-07	-4.22E-08
S2 – Fossil-based HDPE bottles	8.55E-07	5.71E-07	2.30E-07	2.72E-08	1.36E-07	-1.08E-07
S3 – 24% R-PET bottles	6.98E-07	1.26E-07	4.42E-07	4.99E-08	1.23E-07	-4.22E-08
S4 – 16% R-HDPE bottles	8.26E-07	5.28E-07	2.42E-07	2.72E-08	1.36E-07	-1.08E-07
S5 – 30% bio-based PET bottles	8.30E-07	3.09E-07	4.11E-07	4.99E-08	1.23E-07	-6.20E-08
S6 – Bio-based HDPE bottles	2.80E-06	1.91E-06	9.39E-07	2.72E-08	1.36E-07	-2.15E-07
S7 – PEF bottles	3.60E-04	4.52E-04	7.58E-06	-6.78E-06	9.84E-08	-9.31E-05

3.7.3 Macro- and micro-plastics generation and release

Table 3.22 shows the total plastic loss, final release to ocean and final release to the terrestrial environment estimated for the assessed beverage bottles scenarios, specifying the contribution of both macro- and micro-plastics. The contributions of the different macro- and micro- plastics sources to the respective total releases to ocean and to the terrestrial environment are shown in Figure 3.8. As reported in Section 3.5.5.6, the considered macro-plastics sources include product littering directly from consumers and mismanagement of product waste at End of Life, while micro-plastics are generated from pellet losses during product manufacturing and tyre abrasion during foreground road transport. Note that, while the results for all the investigated scenarios are presented together, they are not intended to compare the different scenarios, and should not be used for this purpose by the reader.

Table 3.22. Total plastic loss, final release to ocean and final release to the terrestrial environment⁽¹⁾ estimated for beverage bottles LCA scenarios, including the contribution of both macro- and micro-plastics. Results are not intended to compare the different scenarios.

Indicator	Total		Macro-plastics		Micro-plastics	
	kg/FU	%	kg/FU	%	kg/FU	%
S1 – Fossil-based PET bottles						
Loss	5.37	100.0%	5.31	99.0%	5.23E-02	1.0%
Release to ocean	0.537	100.0%	0.531	98.9%	5.91E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.305	100.0%	0.266	87.2%	3.91E-02	12.8%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-

S2 – Fossil-based HDPE bottles						
Loss	5.92	100.0%	5.87	99.1%	5.56E-02	0.9%
Release to ocean	0.593	100.0%	0.587	98.9%	6.27E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.335	100.0%	0.293	87.6%	4.16E-02	12.4%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-
S3 – 24% R-PET bottles						
Loss	5.37	100.0%	5.31	99.0%	5.21E-02	1.0%
Release to ocean	0.537	100.0%	0.531	98.9%	5.91E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.305	100.0%	0.266	87.2%	3.90E-02	12.8%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-
S4 – 16% R-HDPE bottles						
Loss	5.92	100.0%	5.87	99.1%	5.55E-02	0.9%
Release to ocean	0.593	100.0%	0.587	98.9%	6.27E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.335	100.0%	0.293	87.6%	4.15E-02	12.4%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-
S5 – 30% bio-based PET bottles						
Loss	5.37	100.0%	5.31	99.0%	5.19E-02	1.0%
Release to ocean	0.537	100.0%	0.531	98.9%	5.90E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.304	100.0%	0.266	87.3%	3.88E-02	12.7%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-
S6 – Bio-based HDPE bottles						
Loss	5.93	100.0%	5.87	99.0%	6.12E-02	1.0%
Release to ocean	0.593	100.0%	0.587	98.9%	6.37E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.340	100.0%	0.293	86.4%	4.63E-02	13.6%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-
S7 – PEF bottles						
Loss	4.29	100.0%	4.25	99.0%	4.19E-02	1.0%

Release to ocean	0.430	100.0%	0.425	98.9%	4.73E-03	1.1%
Release to the terrestrial environment⁽¹⁾	0.244	100.0%	0.213	87.2%	3.13E-02	12.8%
Ratio total release / loss	15.7%	-	15.0%	-	86.1%	-

(¹) Including release to freshwater sediments, as discussed in Section 3.5.5.6.

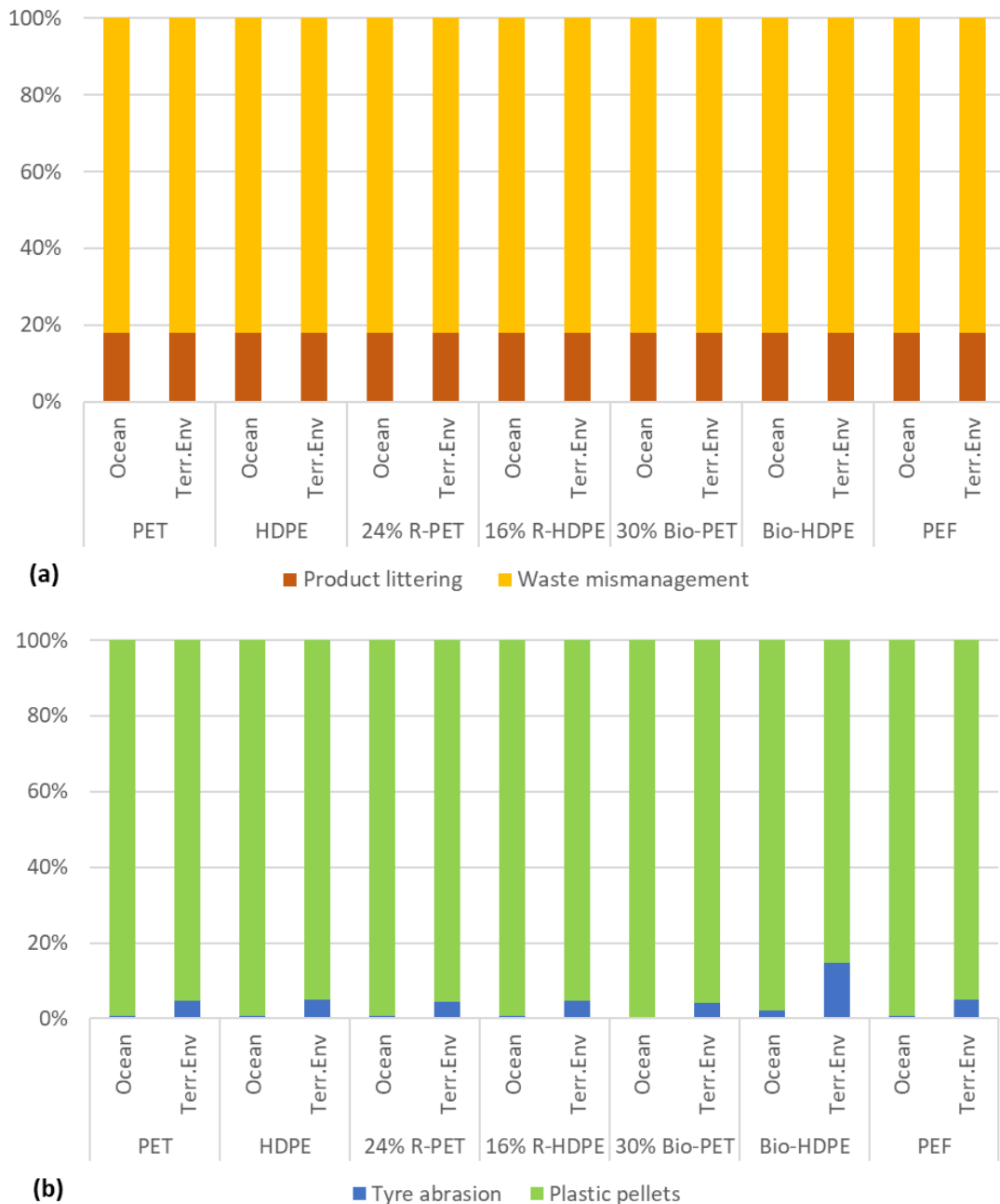


Figure 3.8. Contribution of single emission sources to the total release of macro-plastics (a) and micro-plastics (b) estimated for beverage bottles LCA scenarios to both ocean (Ocean) and the terrestrial environment (Terr.Env; including freshwater sediments). Results are not intended to compare the different scenarios.

3.8 Interpretation

In the interpretation of results, the most relevant impact categories of the analysed beverage bottles scenarios are firstly identified (Section 3.8.1). The contribution of individual life cycle stages to each most relevant impact category is then calculated, and the most relevant life cycle stages are identified (Section 3.8.2). The effects of GHG emissions from iLUC are also discussed (Section 3.8.3), while the results related to macro- and micro-plastics generation and release are addressed in Section 3.8.4. Finally, the results of a sensitivity analysis on a number of parameters, assumptions and methodological choices are presented (Section 3.8.5), including characterised scenario impacts calculated by individually applying each End of Life option reported in Table 3.1.

It is noted that most relevant processes were not identified, since the life cycle inventories of the analysed product scenarios present different levels of vertical disaggregation of included foreground processes (e.g. inventories of bio-based HDPE and PEF bottles are more disaggregated than the other scenarios). Therefore, the identification of most relevant processes would have not been carried out consistently across all the scenarios, and a more detailed investigation for specific scenarios would have not been meaningful. The identification of most relevant elementary flows was also not undertaken, as this would have required prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the *Plastics LCA* method shall proceed with the identification of both most relevant processes and elementary flows in each most relevant impact category.

3.8.1 Identification of most relevant impact categories

Table 3.23 shows the most relevant impact categories identified for each beverage bottles scenario, based on normalised and weighted impacts, according to the procedure described in Section 6.2.1 of the *Plastics LCA* method. Most relevant impact categories were hence identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact (single score) of each scenario. The contribution of toxicity-related impact categories was excluded from the calculation of total normalised and weighted impact scores, as being still based on the characterisation factors implemented in the EF 2.0 impact assessment methods applied in this study (and hence not yet updated based on REACH data)⁶⁷. Where needed, additional impact categories from the obtained ranking were added to the list of most relevant categories, to fulfil the requirement of having a minimum of three categories identified as most relevant.

For both virgin and partially recycled PET bottles, only three impact categories are identified as most relevant, i.e. Climate Change, Resource Use – fossils and, with a much lower contribution of nearly 4%, Acidification. Similarly, for virgin and partially recycled HDPE bottles, Resource Use – fossils is the most relevant category, followed by Climate Change and Particulate Matter (which only contributes with 4.5% to the total impact).

Climate Change and Resource Use – fossils are the two most relevant categories also for partially bio-based PET and PEF bottles, along with the lower contribution of Particulate Matter (13.4% and 7%, respectively), and Acidification (4-5%).

For bio-based HDPE bottles, Particulate Matter is the most relevant impact category, followed by Climate Change and, with a more limited contribution of 7.5%, Eutrophication – marine and Land Use.

⁶⁷ According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018b), toxicity-related impact indicators calculated based on EF 2.0 characterisation factors (applied in this study) shall be excluded from the procedure to identify the most relevant impact categories. However, any user of the *Plastics LCA* method shall apply the latest characterisation factors available at the time of the study (currently those provided in the 3.0 EF reference package) and include also toxicity-related impact categories in the calculation of the total normalised and weighted impact score (and hence in the identification of most relevant impact categories).

Table 3.23. Most relevant impact categories identified for beverage bottles LCA scenarios and related contribution to the total normalised and weighted impact score of each scenario ⁽¹⁾.

S1 – Fossil-based PET bottles		S2 – Fossil-based HDPE bottles		S3 – 24% R-PET bottles	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	42.2%	Resource Use - fossils	42.8%	Climate Change	43.2%
Resource Use – fossils	39.5%	Climate Change	37.9%	Resource Use - fossils	38.1%
Acidification	3.9%	Particulate Matter	4.5%	Acidification	3.9
Total	85.6%	Total	85.2%	Total	85.3%
S4 – 16% R-HDPE bottles		S5 – 30 % bio-based PET bottles		S6 – Bio-based HDPE bottles	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Resource Use – fossils	41.5%	Climate Change	38.6%	Particulate Matter	40.7%
Climate Change	38.9%	Resource Use - fossils	27.7%	Climate Change	24.9%
Particulate Matter	4.5%	–Particulate Matter	13.4%	Eutrophication - marine	7.6%
		Acidification	4.3%	Land Use	7.5%
Total	84.9%	Total	84.0%	Total	80.6%
S7 – PEF bottles					
<i>Impact category</i>	<i>Contrib.</i>				
Climate Change	40.7%				
Resource Use - fossils	27.9%				
Particulate Matter	7.0%				
Acidification	4.6%				
Total	80.1%				

⁽¹⁾ Note that decimals are reported to avoid rounding errors, but do not reflect an accuracy corresponding to the number of digits shown for the specific contribution.

3.8.2 Identification of most relevant life-cycle stages

Table 3.24 shows the contribution of individual life cycle stages to the most relevant impact categories identified, for each beverage bottles scenario, in Section 3.8.1. The contribution was quantified according to the rules described in Sections 6.2.2 and 6.2.5 of the *Plastics LCA* method. Most relevant life cycle stages are also identified (in yellow) for each impact category, and include those that together contribute to at least 80% of the total impact in the specific category. Note that the net total impact, resulting from the algebraic sum of both positive and negative impact contributions of single life cycle stages, was considered to calculate the percentage contribution of each stage. Therefore, the contribution of specific life cycle stages may be larger than 100% if the respective impact is higher than the net total impact in the specific category, and proportionally higher than the impact of the other life cycle stages. Moreover, the sum of all positive

contributions is necessarily larger than 100%, and is balanced by the negative contribution of specific life cycle stages (typically End of Life), leading to the sum of all positive and negative contributions correctly adding up to 100%. The possible negative impact and contribution from the End of Life stage is a result of the inclusion, along with the burdens of the applied waste management activities, of any benefits from secondary material production or energy recovery.

For both virgin and partially recycled PET bottles, Polymer Production is identified as the most relevant stage, followed by Distribution (in Climate Change and Acidification) or Feedstock Supply (in Resource Use – fossils). For virgin and partially recycled HDPE bottles, Polymer Production is still the most relevant stage in Climate Change, followed by Feedstock Supply and Distribution (virgin HDPE) or by Distribution and Feedstock Supply (partially recycled HDPE). In Resource Use – fossils, Feedstock Supply is the only relevant stage for these two alternatives, as it contributes, alone, to nearly 90% of the overall impact. In Particulate Matter, the most relevant stages are Polymer Production and Distribution.

For bio-based PET bottles, Polymer Production and Feedstock Supply are the most relevant stages in Climate Change, Resource Use – fossils, and Acidification. Conversely, in Particulate Matter, Feedstock Supply is the most relevant stage (contributing, alone, to almost 80% of the overall impact), followed by Polymer Production. Feedstock Supply is also the most relevant stage in all the four impact categories identified as most relevant for bio-based HDPE bottles. In Eutrophication – marine and Land Use, such stage is alone responsible for 87.5-98% of the total impact, thus being the only most relevant stage. In Particulate Matter and Climate Change, the contribution of Feedstock Supply is lower (65-73.5%), and Polymer Production is also identified as an additional most relevant stage (with a contribution in the range of 34-24%).

In the case of PEF bottles, Polymer Production dominates the overall impact in the majority of the identified most relevant categories, where it is the only most relevant stage. In Particulate Matter, the most relevant stage is instead Feedstock Supply, followed in this case also by Polymer Production. Note, however, that the dominant role of Polymer Production in the majority of the identified most relevant categories is affected by the substantial impact associated with HMF production, which is likely to be even largely overestimated compared to any future industrial-scale production process (see Section 3.5.2.3.3).

Table 3.24. Contribution of individual life cycle stages to the most relevant impact categories identified for each beverage bottles LCA scenario ⁽¹⁾. Most relevant stages (i.e. those contributing to at least 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based PET bottles					
Climate Change		Resource Use - fossils		Acidification	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	71.3%	Polymer Production	55.6%	Polymer Production	60.1%
Distribution	16.2%	Feedstock Supply	51.3%	Distribution	25.5%
Feedstock Supply	8.5%	Distribution	10.7%	Feedstock Supply	19.3%
Manufacturing	6.9%	Manufacturing	5.4%	Manufacturing	8.7%
End of Life	-2.9%	End of Life	-23.1%	End of Life	-13.7%
S2 – Fossil-based HDPE bottles					
Resource Use - fossils		Climate Change		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Feedstock Supply	89.3%	Polymer Production	45.6%	Polymer Production	58.8%
Polymer Production	27.2%	Feedstock Supply	25.0%	Distribution	29.1%
Distribution	13.5%	Distribution	24.5%	Feedstock Supply	28.9%
Manufacturing	3.5%	Manufacturing	5.0%	Manufacturing	7.5%
End of Life	-33.5%	End of Life	-0.03%	End of Life	-24.4%
S3 – 24% R-PET bottles					
Climate Change		Resource Use - fossils		Acidification	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	69.2%	Polymer Production	56.0%	Polymer Production	58.8%
Distribution	17.7%	Feedstock Supply	52.0%	Distribution	28.0%
Feedstock Supply	8.8%	Distribution	12.4%	Feedstock Supply	18.6%
Manufacturing	7.5%	Manufacturing	6.3%	Manufacturing	9.6%
End of Life	-3.2%	End of Life	-26.7%	End of Life	-15.0%
S4 – 16% R-HDPE bottles					
Resource Use - fossils		Climate Change		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Feedstock Supply	89.8%	Polymer Production	45.0%	Polymer Production	59.3%
Polymer Production	28.2%	Distribution	25.4%	Distribution	30.7%
Distribution	14.8%	Feedstock Supply	24.5%	Feedstock Supply	27.9%
Manufacturing	3.9%	Manufacturing	5.2%	Manufacturing	7.9%
End of Life	-36.7%	End of Life	-0.03%	End of Life	-25.7%

S5 – 30% bio-based PET bottles					
Climate Change		Resource Use - fossils		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	59.6%	Polymer Production	59.5%	Feedstock Supply	78.6%
Feedstock Supply	23.2%	Feedstock Supply	49.3%	Polymer Production	16.6%
Distribution	15.7%	Distribution	13.7%	Distribution	6.4%
Manufacturing	6.7%	Manufacturing	6.9%	Manufacturing	3.1%
End of Life	-5.3%	End of Life	-29.3%	End of Life	-4.7%
Acidification					
<i>Life cycle stage</i>	<i>Contrib.</i>				
Polymer Production	56.6%				
Feedstock Supply	28.1%				
Distribution	20.6%				
Manufacturing	7.0%				
End of Life	-12.3%				
S6 – Bio-based HDPE bottles					
Particulate Matter		Climate Change		Eutrophication - marine	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Feedstock Supply	65.4%	Feedstock Supply	73.5%	Feedstock Supply	87.5%
Polymer Production	34.3%	Polymer Production	24.4%	Polymer Production	11.5%
Distribution	0.9%	Distribution	9.9%	Distribution	1.5%
Manufacturing	0.2%	Manufacturing	2.0%	Manufacturing	0.2%
End of Life	-0.9%	End of Life	-9.9%	End of Life	-0.7%
Land Use					
<i>Life cycle stage</i>	<i>Contrib.</i>				
Feedstock Supply	98.2%				
Polymer Production	2.1%				
Distribution	0.4%				
Manufacturing	0.2%				
End of Life	-0.8%				
S7 – PEF bottles					
Climate Change		Resource Use - fossils		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	109.0%	Polymer Production	116.5%	Feedstock Supply	65.2%
Feedstock Supply	9.5%	Distribution	4.2%	Polymer Production	55.4%
Distribution	4.6%	Feedstock Supply	2.8%	Distribution	3.8%
Manufacturing	0.5%	Manufacturing	0.9%	Manufacturing	0.1%
End of Life	-23.7%	End of Life	-24.4%	End of Life	-24.4%

Acidification	
<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	80.6%
Feedstock Supply	36.9%
Distribution	5.9%
Manufacturing	0.5%
End of Life	-23.9%

(¹) Note that decimals are reported to avoid rounding errors, but do not reflect an accuracy corresponding to the number of digits shown for the specific contribution.

3.8.3 Effects of indirect land use change (iLUC) on Climate Change

The additional contribution of GHG emissions from iLUC to the total Climate Change impact of the investigated bio-based alternatives (Bio-PET, Bio-HDPE and PEF bottles) is generally limited (Table 3.20). A maximum impact increase by 9.1% is observed for bio-based HDPE bottles, while for bio-based PET bottles the increase is equal to only 2.2%, reflecting the partial bio-based content of this polymer (i.e. 30%). For PEF bottles, the variation is even smaller (1.4%) despite it is fully bio-based. This is at least partly a consequence of the proportionally higher Climate Change impact associated with the use of this material even without considering iLUC. However, it must be reminded that such impact is affected by the contribution from the production of HMF (one of the main PEF precursors), which is likely to be even largely overestimated compared to any future industrial-scale production process of this chemical (see Section 3.5.2.3.3).

It must be noted that these results need to be read also in light of the GHG emission factors applied for iLUC, which appeared to fall in the lower end of the range of values available in the recent literature (see Section J.2 of the *Plastics LCA* method). The use of an alternative iLUC model and of the resulting GHG emission factors may thus result in an increased Climate Change contribution from iLUC, which was explored in a sensitivity analysis (see Section 3.8.5.6). Moreover, only the effects of GHG emissions from iLUC were addressed in this study, while nutrient-related emissions and other relevant emissions possibly associated with the additional use of converted land and/or with intensification in land use were not considered. However, accounting for such additional emissions may affect the impact of bio-based alternatives also in other impact categories than Climate Change, such as Acidification and Eutrophication.

3.8.4 Macro- and micro-plastics generation and release

This section discusses the results presented in Section 3.7.3 on the estimated potential generation (loss) and release of macro- and micro-plastics of the assessed beverage bottles scenarios. Focusing initially on the relation between the total loss of macro- and micro-plastics and the resulting total release to ocean and to the terrestrial environment (Table 3.22), it is observed that the total release represents only a modest share of the initial loss (i.e. 16% on average). This is because of the reduced overall release rate to the environment applied to bottles initially lost as macro-plastics (i.e. 15%)⁶⁸ and to the dominant role of macro-plastics within the total plastic loss (as discussed below). The transfer of the plastics loss to the different release compartments is thus almost entirely determined by the release rates applied to macro-plastics, while it is only marginally affected by the higher final release rates assumed for lost micro-plastics (around 86%, overall)⁶⁹, as their contribution to the total loss is negligible (as discussed below). In relative terms, the total release to ocean accounts for a larger share of the total initial

⁶⁸ Based on the high residual economic value of the different bottle materials, on the non-negligible size of lost bottles (medium; 5-25 cm), and hence on the resulting substantial take-back to the technosphere expected through further collection (Section 3.5.5.6).

⁶⁹ See Tables I.4 and I.5 in the *Plastics LCA* method.

plastic loss (around 10% on average, corresponding to two thirds of the total final release, i.e. 64%), while the total release to the terrestrial environment is lower (6% of the initial loss and one third -36%- of the final release). The shares of the total plastic loss released to ocean and to the terrestrial environment reflect the final release rates of lost macro-plastics to such compartments (10% to ocean and 5% to the terrestrial environment; Section 3.5.5.6), due to the prevailing role of macro-plastics within the total plastic loss.

As shown in Table 3.22, also the total plastic release to both ocean and the terrestrial environment is dominated by macro-plastics released at End of Life, which account for 99% of the total release to ocean and for 86.5-87.5% of that to the terrestrial environment. Micro-plastics released throughout the upstream life cycle (via pellet loss and tyre abrasion) have only a minor role, instead. This is a result of the prevailing mass of generated and released macro-plastics compared to micro-plastics, since the macro-plastics loss directly depends on the total mass of product (bottles) used per functional unit (i.e. the reference flow), and is calculated based on a higher total "loss rate" compared to the considered micro-plastics sources⁷⁰. Moreover, the final release rate to ocean of bottles lost as macro-plastics is higher compared to the final release rate to the same compartment of micro-plastics generated from tire abrasion⁷¹. Therefore, the mass of released macro-plastics is at least one order of magnitude higher than the mass of released micro-plastics, which instead only indirectly depends on the reference flow (through parameters related to the quantity of relevant lifecycle process)⁷², and is calculated based on (much) lower loss and (depending on the compartment and source) release rates. Note, however, that the release of micro-plastics from tire abrasion was underestimated, due to the exclusion of the contribution of (as discussed in Section 3.5.5.6): (a) most background transport activities; (b) intermediate transports between different process steps included within the vertically aggregated datasets used to model production of some polymers (i.e. virgin PET and HDPE –affecting also partially recycled PET and HDPE–, and bio-based PET); and (c) any transport activities occurring within horizontally aggregated datasets.

Focusing on the sources of macro-plastics released to ocean and to the terrestrial environment (Figure 3.8a), for all scenarios the contribution of product littering directly from consumers represents only a modest share of the total release, equalling 18% on average. The most relevant contribution is instead provided by product waste mismanagement, which is responsible for the remaining share of the estimated macro-plastics release, due to the higher loss rate (waste mismanagement index) assumed for this source (i.e. 9.25%; Table I.2 of the *Plastics LCA* method) compared to product littering. The littering rate estimated for bottles is indeed equal to 2%, as specified in Table 3.11.

As for the micro-plastics release to ocean and to the terrestrial environment, the relative contribution of the different sources (loss of plastic pellets during product manufacturing and tire abrasion during road transport) generally depends on the configuration of the modelled supply-chain. The origin of the feedstock and the location of subsequent conversion processes of feedstock materials into final polymers are especially relevant as they affect the contribution from transport activities across the life cycle. Moreover, the level of vertical disaggregation applied in the modelling of such upstream conversion

⁷⁰ The total "loss rate" of post-consumer beverage bottles as macro-plastics at End of Life (calculated as a combination of the littering rate and of the mismanaged waste index) is around 11% (see Table 3.11 in this report, and Equation I.1 plus Table I.2 in the *Plastics LCA* method). Conversely, the total "loss rate" of micro-plastics from tire abrasion is in the range of 1.5-2.6% (depending on the vehicle; see Equation I.3 and Table I.3 in the *Plastics LCA* method). For plastic pellets from product manufacturing, the loss rate is even lower, equalling 0.1% (see Equation I.5 and Table I.5 in the *Plastics LCA* method).

⁷¹ The final release rate to ocean is equal to 10% for beverage bottles lost as macro-plastics (Table 3.11), while it is equal to 2% for micro-plastics generated from tire abrasion (Table I.4 in the *Plastics LCA* method).

⁷² Including the mass of plastic pellets used for product manufacturing (which depends on the quantity of this process per functional unit), and the amount of product or material transported during each modelled road transport activity (per functional unit) with the associated distance.

processes plays an important role, as it affects the number of intermediate transport activities between different process steps which are modelled separately in the foreground inventory, and for which the contribution to micro-plastics generation can be quantified⁷³. Keeping this in mind, and considering the marginal role of micro-plastics within the total plastic release to ocean and to the terrestrial environment (see above), the most relevant contribution is generally provided by pellet losses during product manufacturing, while micro-plastics generated from tire abrasion during road transport generally have a more limited role (Figure 3.8b). An exception is represented by bio-based HDPE bottles, where micro-plastics from tire abrasion are more important in terms of release to the terrestrial environment (Figure 3.8b), due to increased transport activities individually modelled throughout the upstream supply chain (following a more disaggregated development of the foreground inventory, as discussed above). Note, however, that the same considerations made above regarding the likely underestimate of the contribution of micro-plastics generated and released from transport activities apply also in this case.

From a methodological perspective, it is noted how scenarios having a lower reference flow (i.e. requiring a lower amount of plastic material per functional unit) generate a lower loss and release of macro-plastics (Table 3.22). Therefore, scenarios relying on lighter bottles show a lower macro-plastics loss and release to ocean and to the terrestrial environment. Due to the prevailing role of released macro-plastics compared to micro-plastics (as discussed above), the same considerations apply also to the total macro- and micro-plastics loss and release to both ocean and the terrestrial environment (Table 3.22). However, these results are determined by the use of mass-based loss and release indicators, which for macro-plastics are directly and largely affected by differences in the reference flow, and hence by variations in the bottle mass. Moreover, they are valid as long as the same loss and release rates are applied regardless of the bottle material (as considered in this case study). If indicators quantifying the total number of items lost or released to the environment were applied, all scenarios would instead result in the same macro-plastics loss and release, as long as the number of items (i.e. bottles) required to fulfil the functional unit would not change across the different scenarios⁷⁴. Using such item-based indicators, the contribution of macro-plastics generated from product littering and waste mismanagement would indeed be the same even for different reference flows of plastic material required per functional unit (due to, e.g., different bottle masses). In turn, an identical macro-plastics generation and release would result in comparable total loss and releases to ocean and to the terrestrial environment, these being mostly determined by macro-plastics rather than micro-plastics (as discussed above), and provided that micro-plastics would also be quantified in terms of lost or released items.

3.8.5 Sensitivity analysis

A sensitivity analysis was performed on a number of parameters, assumptions, and methodological choices, to evaluate their influence and the effects of their variation on the characterised impact assessment results of the affected LCA scenario(s). The following aspects were considered:

1. Recycled content in R-PET and R-HDPE bottles;
2. Pre-harvest burning rate of sugarcane used as a feedstock for bio-based PET and bio-based HDPE bottles;
3. Feedstock source for bio-based PET and bio-based HDPE bottles (origin and type);

⁷³ When vertically aggregated datasets were applied to model the process chain involved in the conversion of feedstock materials into the final polymer (i.e. for fossil-based PET and HDPE, bio-based PET, and partly for recycled PET and HDPE bottles), the contribution of intermediate transports between different process steps could not be quantified. Conversely, when such process steps were modelled individually (i.e. for bio-based HDPE and PEF bottles), the contribution of intermediate transports was accounted.

⁷⁴ Which may happen, for instance, if the size of bottles would change among the different scenarios.

4. Origin (production location) of bio-based HDPE;
5. Use of refrigeration and cooling energy in the production of HMF, precursor of PEF;
6. Applied iLUC model and resulting GHG emission factors;
7. Handling of non-released biogenic carbon at End of Life;
8. Alternative End of Life scenarios.

The following sections present the results of the sensitivity analysis for each of the aspects above. This is done by comparing recalculated impacts of the affected scenario(s) with those of the base case assessment.

3.8.5.1 Recycled content in R-PET and R-HDPE bottles

This sensitivity analysis explores the effects of an increased recycled material content in partially recycled PET and HDPE beverage bottles. As a base case, this parameter was estimated to reflect the current average recycled content at the EU level, equalling 24% for PET bottles and 16% for HDPE bottles (Section 3.1). However, the EU Directive 2019/904⁷⁵ on single-use plastics requires all single-use beverage bottles with a capacity of up to three litres to incorporate at least 30% recycled material from 2030. Moreover, a few initial examples of PET beverage bottles claiming a 100% recycled content appear to start being available on the market, and the same applies to HDPE bottles used in non-food contact applications. A sensitivity analysis was thus performed to assess the effects of separately increasing the recycled content of PET and HDPE beverage bottles to both 30% and 100%, while recognising that the use of 100% recycled HDPE beverage bottles is a hypothetical optimistic scenario that is currently unlikely to happen in reality.

The results of the analysis are presented in Table 3.25 for recycled PET bottles and in Table 3.26 for recycled HDPE bottles. In both tables, recalculated potential impacts of the affected scenario are reported along with those of the respective base case, and the resulting percentage impact variation with respect to the base case scenario is calculated.

Increasing the recycled content of partially recycled PET and HDPE beverage bottles to 30% only marginally affects the potential impacts of these alternatives, which in most impact categories (i.e. all except one) show a non-significant variation lower than 5% (PET bottles) and 10% (HDPE bottles), compared to the base case⁷⁶. This reflects the (very) limited increase initially considered for the recycled content, equalling 6% (from 24% to 30%) for PET bottles and 14% (from 16% to 30%) for HDPE ones. Only the Ozone Depletion impact category shows a non-negligible increase of 21% and 46%, respectively, for PET and HDPE bottles.

On the other hand, considering a 100% recycled material content significantly reduces the impact of recycled PET bottles in the vast majority of the assessed impact categories (i.e. all except four), with a decrease ranging from 15% (Ionising Radiation) to 50% (Resource Use – fossils) compared to the base case. However, with a 100% recycled content, an impact increase is observed in terms of Ozone Depletion (+263% compared to 24% R-PET bottles), Eutrophication – freshwater (+43%), Land Use (+20%), and Resource Use – minerals and metals (+21%). A similar situation is observed also for HDPE bottles, with 100% recycled HDPE bottles presenting a reduced impact compared to 16% recycled HDPE bottles within the same range of impact categories as recycled PET bottles, while an impact increase is observed in the same four impact categories (Ozone Depletion, Eutrophication – freshwater, Land Use and Resource Use – minerals and metals). Impact reductions range, in this case, between 20% and 52%.

⁷⁵ Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment.

⁷⁶ Variations below 10% can be considered as not significant, in light of the uncertainty that generally affect a LCA study both at the inventory and at the impact assessment level.

Table 3.25. Results of the sensitivity analysis on the recycled material content in partially recycled PET beverage bottles, increased from 24% to 30% and 100%. Results are expressed per functional unit.

Impact category	24% R-PET bottles (base case)	30% R-PET bottles (SA-1) ⁽¹⁾	Variation (%)	100% R-PET bottles (SA-2) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	1.75E+02	1.71E+02	-2%	1.23E+02	-30%
Ozone Depletion [kg CFC-11 eq.]	1.72E-07	2.08E-07	21%	6.24E-07	263%
Human Toxicity - cancer [CTUh]	1.10E-06	1.07E-06	-3%	7.21E-07	-34%
Human toxicity - non-cancer [CTUh]	4.88E-06	4.83E-06	-1%	4.17E-06	-15%
Particulate matter [Disease incidence]	2.77E-06	2.70E-06	-3%	1.65E-06	-40%
Ionising Radiation [kBq U ²³⁵ eq.]	1.10E+01	1.09E+01	-1%	9.31E+00	-15%
Photochemical Ozone Formation [kg NMVOC eq.]	3.15E-01	3.06E-01	-3%	1.79E-01	-43%
Acidification [mol of H ⁺ eq.]	3.82E-01	3.73E-01	-2%	2.29E-01	-40%
Eutrophication - terrestrial [mol N eq.]	1.25E+00	1.23E+00	-2%	8.38E-01	-33%
Eutrophication - freshwater [kg P eq.]	6.76E-04	7.00E-04	4%	9.68E-04	43%
Eutrophication - marine [kg N eq.]	1.08E-01	1.06E-01	-2%	7.42E-02	-31%
Ecotoxicity - freshwater [CTUe]	2.31E+01	2.27E+01	-2%	1.76E+01	-24%
Land Use [Pt]	4.69E+02	4.77E+02	2%	5.65E+02	20%
Water Use [m ³ world eq.]	3.70E+01	3.57E+01	-4%	2.11E+01	-43%
Resource Use - mineral and metals [kg Sb eq.]	1.92E-05	1.95E-05	2%	2.32E-05	21%
Resource Use - fossils [MJ]	3.23E+03	3.11E+03	-4%	1.62E+03	-50%

⁽¹⁾ SA: Sensitivity Analysis.

Table 3.26. Results of the sensitivity analysis on the recycled material content in partially recycled HDPE beverage bottles, increased from 16% to 30% and 100%. Results are expressed per functional unit.

Impact category	16% R-HDPE bottles (base case)	30% R-HDPE bottles (SA-1) ⁽¹⁾	Variation (%)	100% R-HDPE bottles (SA-1) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	1.34E+02	1.30E+02	-3%	1.06E+02	-21%
Ozone Depletion [kg CFC-11 eq.]	1.91E-07	2.78E-07	46%	7.10E-07	272%
Human Toxicity - cancer [CTUh]	1.34E-06	1.27E-06	-5%	8.62E-07	-36%
Human toxicity - non-cancer [CTUh]	5.62E-06	5.44E-06	-3%	4.50E-06	-20%
Particulate matter [Disease incidence]	2.99E-06	2.85E-06	-5%	1.93E-06	-35%
Ionising Radiation [kBq U ²³⁵ eq.]	4.14E+00	3.94E+00	-5%	2.95E+00	-29%
Photochemical Ozone Formation [kg NMVOC eq.]	3.30E-01	3.11E-01	-6%	1.95E-01	-41%
Acidification [mol of H ⁺ eq.]	3.54E-01	3.37E-01	-5%	2.20E-01	-38%
Eutrophication - terrestrial [mol N eq.]	1.16E+00	1.12E+00	-3%	8.08E-01	-30%
Eutrophication - freshwater [kg P eq.]	7.85E-04	8.39E-04	7%	1.11E-03	41%
Eutrophication - marine [kg N eq.]	9.99E-02	9.66E-02	-3%	7.18E-02	-28%
Ecotoxicity - freshwater [CTUe]	2.97E+01	2.86E+01	-4%	2.32E+01	-22%
Land Use [Pt]	3.88E+02	4.11E+02	6%	5.26E+02	36%
Water Use [m ³ world eq.]	2.16E+01	2.07E+01	-4%	1.65E+01	-24%
Resource Use - mineral and metals [kg Sb eq.]	1.56E-05	1.65E-05	6%	2.08E-05	33%
Resource Use - fossils [MJ]	2.99E+03	2.73E+03	-9%	1.43E+03	-52%

⁽¹⁾ SA: Sensitivity Analysis.

3.8.5.2 Pre-harvest burning rate of sugarcane used as feedstock for bio-based PET and bio-based HDPE

This sensitivity analysis explores the effects of completely removing the pre-harvest burning practice in the cultivation of Brazilian sugarcane used as a feedstock to produce bioethanol, a precursor of both partially bio-based PET and bio-based HDPE. In Brazil, this practice will be legally phased out by 2031 (State Law n. 11241/02) and was expected to be phased out by 2017 according to industry association protocol of intention (Tsiropoulos et al., 2014). Therefore, in this analysis the pre-harvest burning rate was decreased from 10% (assumed as a base case; see Section 3.5.1.3) to 0%, to evaluate the effects of this expected complete phasing out. The results are presented in Tables 3.27 and 3.28 for partially bio-based PET and bio-based HDPE bottles, respectively. In both tables, recalculated potential impacts of the considered scenario are reported along with those of the respective base case, and the resulting percentage impact variation with respect to the base case scenario is calculated.

Table 3.27. Results of the sensitivity analysis on the pre-harvest burning rate of Brazilian sugarcane used as a feedstock for 30% bio-based PET beverage bottles (decreased from 10% to 0%). Results are expressed per functional unit.

Impact category	10% pre-harvest burning (base case)	0% pre-harvest burning (SA) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	1.96E+02	1.95E+02	-1%
Ozone Depletion [kg CFC-11 eq.]	2.14E-08	2.14E-08	0%
Human Toxicity - cancer [CTUh]	1.10E-06	1.01E-06	-8%
Human toxicity - non-cancer [CTUh]	6.79E-06	5.36E-06	-21%
Particulate matter [Disease incidence]	1.30E-05	3.91E-06	-70%
Ionising Radiation [kBq U ²³⁵ eq.]	1.08E+01	1.08E+01	0%
Photochemical Ozone Formation [kg NMVOC eq.]	4.33E-01	4.05E-01	-6%
Acidification [mol of H ⁺ eq.]	5.19E-01	5.02E-01	-3%
Eutrophication - terrestrial [mol N eq.]	1.73E+00	1.65E+00	-5%
Eutrophication - freshwater [kg P eq.]	1.87E-03	1.87E-03	0%
Eutrophication - marine [kg N eq.]	4.50E-01	4.50E-01	0%
Ecotoxicity - freshwater [CTUe]	2.42E+01	2.39E+01	-1%
Land Use [Pt]	6.67E+03	6.67E+03	0%
Water Use [m ³ world eq.]	3.75E+01	3.75E+01	0%
Resource Use - mineral and metals [kg Sb eq.]	1.79E-05	1.79E-05	0%
Resource Use - fossils [MJ]	2.94E+03	2.94E+03	0%

⁽¹⁾ SA: Sensitivity Analysis.

Table 3.28. Results of the sensitivity analysis on the pre-harvest burning rate of Brazilian sugarcane used as a feedstock for bio-based HDPE beverage bottles (decreased from 10% to 0%). Results are expressed per functional unit.

Impact category	10% pre-harvest burning (base case)	0% pre-harvest burning (SA) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	3.43E+02	3.35E+02	-2%
Ozone Depletion [kg CFC-11 eq.]	3.04E-07	3.04E-07	0%
Human Toxicity - cancer [CTUh]	2.04E-06	1.38E-06	-32%
Human toxicity - non-cancer [CTUh]	3.23E-05	2.20E-05	-32%
Particulate matter [Disease incidence]	1.07E-04	4.25E-05	-60%
Ionising Radiation [kBq U ²³⁵ eq.]	8.18E+00	8.18E+00	0%
Photochemical Ozone Formation [kg NMVOC eq.]	1.61E+00	1.40E+00	-13%
Acidification [mol of H ⁺ eq.]	1.73E+00	1.60E+00	-8%
Eutrophication - terrestrial [mol N eq.]	6.65E+00	6.12E+00	-8%
Eutrophication - freshwater [kg P eq.]	1.20E-02	1.20E-02	0%
Eutrophication - marine [kg N eq.]	2.73E+00	2.74E+00	0%
Ecotoxicity - freshwater [CTUe]	5.87E+01	5.63E+01	-4%
Land Use [Pt]	4.66E+04	4.66E+04	0%
Water Use [m ³ world eq.]	7.28E+01	7.28E+01	0%
Resource Use - mineral and metals [kg Sb eq.]	5.81E-05	5.80E-05	0%
Resource Use - fossils [MJ]	8.63E+02	8.72E+02	1.0%

(1) SA: Sensitivity Analysis.

Completely phasing out the sugarcane pre-harvest burning practice decreases the impact of partially bio-based PET bottles in two impact categories, including Particulate Matter (-70%) and Human Toxicity – non-cancer (-21%). Conversely, no or non-significant impact reductions lower than 10% take place in the remaining categories. For bio-based HDPE bottles, the effects of eliminating the pre-harvest burning practice are similar to those observed for partially bio-based PET ones, with an impact reduction between 13% and 60% in four impact categories. These still include Particulate Matter (-60%) and Human Toxicity – non-cancer (-32%), but also Human Toxicity – cancer (32%) and Photochemical Ozone Formation (13%). In the remaining categories, no or non-significant impact variations below 10% occur. This sensitivity analysis has not specifically focused on PEF, which also partially relies on bio-based MEG derived from Brazilian sugarcane (30% of the polymer). However, similar results as bio-based PET bottles (which equally incorporate 30% bio-based MEG) can be reasonably expected, with most impact categories showing no or irrelevant impact variations compared to the base case.

3.8.5.3 Feedstock source for bio-based PET and bio-based HDPE

This sensitivity analysis evaluates the effects of using alternative feedstock sources to produce bioethanol, the main precursor of bio-based MEG (the bio-based co-monomer of partially bio-based PET) and of bio-based HDPE. For partially bio-based PET, Brazilian sugarcane was initially replaced with the same crop cultivated in India, which as reported in Section 3.1 could be reasonably assumed to be the mostly used feedstock at present for bio-based MEG production (the only commercial-scale producer of this monomer is

currently located in India; De Jong et al., 2020). However, for consistency with available life cycle inventory data for partially bio-based PET (considering bioethanol production in Brazil and further conversion to bio-MEG in Europe), Brazilian sugarcane was considered as a feedstock in the base case scenario. In a second sensitivity analysis, Brazilian sugarcane was replaced with the current EU-average mix of bioethanol crops, to assess the effects of sourcing the feedstock locally. This substitution was applied to both partially bio-based PET and bio-based HDPE.

To model the substitution of Brazilian with Indian sugarcane as a feedstock for partially bio-based PET, the sugarcane cultivation datasets applied as a base case (Section 3.5.1.3) were replaced with an EF-compliant dataset representing cultivation in India (*[IN] Sugar cane; technology mix | at farm*). However, the aggregated dataset available to model partially bio-based PET production from sugarcane could not be adapted to reflect sugarcane processing to bioethanol and bio-MEG in India, and subsequent transport to the EU for polymerisation. The results of this first sensitivity analysis thus need to be interpreted taking into account this limitation.

As for the substitution of Brazilian sugarcane with the current EU-average mix of bioethanol crops, European bioethanol was estimated to be mainly derived from maize (45%), wheat (34%) and sugar beet (21%), based on data reported by ePURE (2019). These shares thus refer to the three different bioethanol production pathways (not to the corresponding crops) and identically apply also at the polymer level, since the conversion efficiencies of bioethanol to bio-based PET and bio-based HDPE are the same regardless of the feedstock used for bioethanol production.

For partially bio-based PET, the feedstock substitution was modelled by means of vertically aggregated, ILCD-EL compliant inventory datasets available in the GaBi database, representing partially bio-based PET production from the three different crops (and that replaced the partially aggregated gate-to-gate dataset for bio-based PET production from sugarcane used in the base case, and the associated sugarcane cultivation and transport datasets). However, since the original dataset for maize-derived Bio-PET referred to maize grown in the US, it was adjusted by removing the burdens associated with maize cultivation in that Country, to subsequently add those of maize cultivation in Europe. For this purpose, a specific maize consumption of 0.806 kg maize/kg Bio-PET was considered⁷⁷, since the actual amount applied in the dataset is not disclosed. To model the burdens of both US and European maize cultivation, two ILCD-EL compliant datasets from the GaBi database were used, in the attempt to keep consistency with the originally applied cultivation dataset (which belongs to the same database, despite being unknown). Transoceanic transport of US maize by ship was also initially removed, based on an estimated distance of 6000 km (the original distance assumed in the dataset is also unknown). However, since the results were not significantly different compared to keeping transport included, this change was ultimately not implemented to avoid introducing an additional source of uncertainty.

For bio-based HDPE, the feedstock substitution was modelled by replacing the original disaggregated *ecoinvent* dataset for sugarcane bioethanol production with three different datasets from the same database, representing production from maize, rye and sugar beet⁷⁸. In the absence of specific data, wheat-based bioethanol production was thus approximated with rye-based production, which is reported in the corresponding dataset documentation to be similar to the wheat-to-ethanol process route. Compared to the original datasets, energy inputs (electricity and heat) were modelled by means of EF-compliant datasets reflecting EU-average conditions. Moreover, a few adjustments had to be performed to improve reliability of LCIA results for the impact categories of Ozone

⁷⁷ This estimate is based on a specific consumption of starch for Bio-PET production equal to 0.85 kg starch/kg Bio-PET (IfBB, 2018), and a maize input in starch production of 0.948 kg maize grain/kg starch (according to maize starch production datasets available in the Agri-footprint database).

⁷⁸ The datasets are based on economic allocation to assign the burdens of common process steps to the different co-products, i.e. ethanol and Distiller's Dried Grains with Solubles (DDGS; maize- and rye-based production), or ethanol, beet chips and vinasse (sugar beet-based production).

Depletion and Resource Use – minerals and metals⁷⁹. Upstream production of the three respective feedstock crops was modelled by means of aggregated EF-compliant datasets (wheat and sugar beet), or ILCD-EL compliant datasets from the GaBi database (maize). All crops were assumed to be transported to downstream conversion into bioethanol along an overall distance of 100 km, covered by large lorries (> 32 t, fuelled with the EU diesel mix). Finally, the originally modelled intercontinental transport of raw bioethanol from Brazil to EU for further conversion was replaced with intra-EU transport, according to the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers to factories/users both located in Europe. This includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

The results of the two analyses conducted for partially bio-based PET are presented in Table 3.29, while Table 3.30 show the results for the feedstock substitution considered for bio-based HDPE. In both tables, recalculated potential impacts of the affected beverage bottles scenario are reported along with those of the respective base case, and the resulting percentage impact variation with respect to the base case scenario is calculated.

The substitution of Brazilian sugarcane with Indian sugarcane increases the impacts of partially bio-based PET bottles in eight impact categories, with most of them showing a variation between 20% and 125%, while Ecotoxicity – freshwater and Water Use are increased by nearly 30 times. The specific reason for this huge increase could not be investigated, due to the aggregated nature of the applied sugarcane cultivation datasets. However, for Ecotoxicity, it may be reasonably attributed to increased pesticide emissions inventoried in the dataset (which dominate the impact of Indian sugarcane cultivation), or to the modelled use of different and more harmful substances compared to cultivation of Brazilian sugarcane. The increased Water Use impact may instead be associated to the 14 times higher characterisation factor (and hence impact contribution) of (irrigation) water use in India compared to Brazil. An impact reduction occurs only in three impact categories, including Particulate Matter (-64%), Photochemical Ozone Formation (-18%) and Land Use (-36%), while the five remaining categories show a non-significant impact variation below 10%. The existence of potential discrepancies in the development of the applied sugarcane cultivation datasets (derived from different sources and data providers)⁸⁰ should also be taken into account in the interpretation of these results, which may be even largely affected by such discrepancies.

Replacing Brazilian sugarcane with the EU-average mix of bioethanol crops increases the impacts of partially bio-based PET bottles in eight impact categories, with five of them showing an even important increase ranging from 118% (Water Use) to 977% (Particulate Matter). Only in one category the impact is decreased (Eutrophication – marine), while for the seven remaining categories the observed variations are not significant (or no changes occur). The reasons for this generally worsened picture could not be explored further, due to the use of aggregated datasets. However, one not necessarily major contribution may be associated with the higher net energy demand of the processes involved in the conversion of starch-based crops (maize, wheat) into bioethanol compared to the sugarcane route, and the higher share of such crops in the assumed mix of feedstocks with respect to sugar beet (which requires less energy intensive processing). This assumption is partly confirmed by impact assessment results of bioethanol production from these crops (calculated based on the disaggregated *ecoinvent* datasets used to model the feedstock substitution for bio-based HDPE, as discussed above), which showed a larger impact of maize-based bioethanol compared to sugarcane-based bioethanol in many impact categories. On the other hand, bioethanol

⁷⁹ The infrastructure process related to the bioethanol fermentation plant was removed, and the default *ecoinvent* dataset representing the supply of Sulphuric Acid (*[RoW] Market for sulfuric acid*) was replaced with an equivalent EF-compliant dataset.

⁸⁰ Cultivation datasets for Brazilian sugarcane are ILCD-EL compliant datasets from the GaBi database developed by Thinkstep, while Indian sugarcane cultivation was modelled based on an EF-compliant dataset developed by Blonk Consultants.

from sugar beet involved lower impacts in many categories compared to sugarcane-based bioethanol, but its share in the assumed bioethanol mix is lower (21%). Increased cultivation impacts of the alternative European bioethanol feedstocks compared to Brazilian sugarcane may also play an even major role, as discussed below for bio-based HDPE bottles.

In the case of bio-based HDPE bottles, the substitution of bioethanol derived from Brazilian sugarcane with the EU-average mix of bioethanol production pathways (relying on maize, wheat and sugar beet) results in a generally worsened environmental profile compared to the base case. The impact is increased in twelve out of sixteen impact categories, and in most of them (eight) the variation is substantial, ranging between 127% (Ionising radiation) and 736% (Human Toxicity – non-cancer). In Ecotoxicity – freshwater the increase is dramatic (28 times), and reasonably attributable to higher inventoried pesticide emissions, or to the modelled use of more harmful substances for pest control compared to Brazilian sugarcane. Only in four impact categories the impact is decreased, with a reduction ranging from 15% (Climate Change) to 81 % (Particulate Matter). It shall be noted that the results of this sensitivity analysis may be even largely affected by discrepancies in the development of the applied cultivation datasets (which are derived from different sources and data providers)⁸¹. It should thus be further investigated whether the observed increases properly reflect real differences in cultivation practices between the EU-mix of bioethanol crops and sugarcane, or are a results of differences in the applied modelling.

⁸¹ Cultivation datasets for Brazilian sugarcane and EU maize are ILCD-EL compliant datasets from the GaBi database developed by Thinkstep, while wheat and sugar beet cultivation was modelled based on EF-compliant datasets developed by Blonk Consultants.

Table 3.29. Results of the sensitivity analysis on the bio-based feedstock source for 30% bio-based PET production (i.e. Indian sugarcane or the EU-average mix of bioethanol crops instead of Brazilian sugarcane). Results refer to the total potential impacts of partially bio-based PET beverage bottles and are expressed per functional unit.

Impact category	Bio-based PET from Brazilian sugarcane (base case)	Bio-based PET from Indian sugarcane (SA-1) ⁽¹⁾	Variation (%)	Bio-based PET from the EU-average mix of bioethanol crops (SA-2) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	1.96E+02	1.77E+02	-10%	1.90E+02	-3%
Ozone Depletion [kg CFC-11 eq.]	2.14E-08	3.09E-08	44%	2.14E-08	0%
Human Toxicity - cancer [CTUh]	1.10E-06	1.32E-06	20%	1.08E-06	-2%
Human toxicity - non-cancer [CTUh]	6.79E-06	6.74E-06	-1%	3.49E-05	414%
Particulate matter [Disease incidence]	1.30E-05	4.74E-06	-64%	1.40E-04	977%
Ionising Radiation [kBq U ²³⁵ eq.]	1.08E+01	1.12E+01	4%	1.10E+01	2%
Photochemical Ozone Formation [kg NMVOC eq.]	4.33E-01	3.56E-01	-18%	4.11E-01	-5%
Acidification [mol of H ⁺ eq.]	5.19E-01	6.28E-01	21%	0.539	4%
Eutrophication - terrestrial [mol N eq.]	1.73E+00	2.21E+00	28%	1.89E+00	9%
Eutrophication - freshwater [kg P eq.]	1.87E-03	4.21E-03	125%	5.34E-03	186%
Eutrophication - marine [kg N eq.]	4.50E-01	6.14E-01	36%	1.91E-01	-58%
Ecotoxicity - freshwater [CTUe]	2.42E+01	7.12E+02	2842%	4.71E+01	95%
Land Use [Pt]	6.67E+03	4.26E+03	-36%	8.20E+03	23%
Water Use [m ³ world eq.]	3.75E+01	9.66E+02	2476%	8.18E+01	118%
Resource Use - mineral and metals [kg Sb eq.]	1.79E-05	1.77E-05	-1%	5.16E-05	188%
Resource Use - fossils [MJ]	2.94E+03	2.99E+03	2%	3.31E+03	13%

⁽¹⁾ SA: Sensitivity Analysis.

Table 3.30. Results of the sensitivity analysis on the feedstock source for bio-based HDPE production (i.e. the EU-average mix of bioethanol crops instead of Brazilian sugarcane). Results refer to the total potential impacts of bio-based HDPE beverage bottles and are expressed per functional unit.

Impact category	Bio-based HDPE from Brazilian sugarcane (base case)	Bio-based HDPE from the EU-average mix of bioethanol crops (SA) (1)	Variation (%)
Climate Change [kg CO ₂ eq.]	3.43E+02	2.90E+02	-15%
Ozone Depletion [kg CFC-11 eq.]	3.04E-07	1.05E-06	245%
Human Toxicity - cancer [CTUh]	2.04E-06	4.68E-06	129%
Human toxicity - non-cancer [CTUh]	3.23E-05	2.70E-04	736%
Particulate matter [Disease incidence]	1.07E-04	2.01E-05	-81%
Ionising Radiation [kBq U ²³⁵ eq.]	8.18E+00	1.86E+01	127%
Photochemical Ozone Formation [kg NMVOC eq.]	1.61E+00	8.08E-01	-50%
Acidification [mol of H ⁺ eq.]	1.73E+00	2.35E+00	36%
Eutrophication - terrestrial [mol N eq.]	6.65E+00	9.45E+00	42%
Eutrophication - freshwater [kg P eq.]	1.20E-02	4.53E-02	278%
Eutrophication - marine [kg N eq.]	2.73E+00	1.56E+00	-43%
Ecotoxicity - freshwater [CTUe]	5.87E+01	1.62E+03	2660%
Land Use [Pt]	4.66E+04	6.86E+04	47%
Water Use [m ³ world eq.]	7.28E+01	3.58E+02	392%
Resource Use - mineral and metals [kg Sb eq.]	5.81E-05	3.85E-04	563%
Resource Use - fossils [MJ]	8.63E+02	2.77E+03	221%

(1) SA: Sensitivity Analysis.

3.8.5.4 Origin (production location) of bio-based HDPE

As a base case, the production of bio-based HDPE and of the respective monomer (ethylene) was assumed to take place in Europe, based on sugarcane-derived bioethanol produced in Brazil and subsequently imported in Europe for conversion. This choice was made to keep consistency with the supply-chain configuration assumed in the aggregated, gate-to-gate datasets applied to model the other bio-based polymers derived from sugarcane bioethanol that are considered in this and other case studies (e.g. bio-based LDPE and PP) (see Section 3.5.2.3.2). However, such configuration may not fully reflect the origin of bio-based HDPE currently supplied to the average EU market, as one major producer of this polymer is actually located in Brazil (albeit specific market shares are unknown). A sensitivity analysis was thus conducted considering that the entire process chain from bioethanol conversion to bio-based HDPE production takes place in Brazil, and the finished polymer is then exported to Europe for conversion into specific end applications. This means that, in this analysis, bio-based HDPE used in the EU market was assumed to be entirely produced and sourced in Brazil.

The processes of production of bio-ethylene (via bioethanol dehydration) and subsequent polymerisation into bio-based HDPE in Brazil were modelled based on the same foreground data considered in the base case for production in the EU (as described in Section 3.5.2.3.2). However, background datasets representing the burdens associated

with the supply of the different inputs and handling of outputs were selected or developed to reflect, as far as possible, Brazilian rather than EU or European average conditions. In particular, for thermal energy supply, a specific dataset was created based on energy consumption statistics for the chemical industry in Brazil (EPE, 2017), and background EF-compliant or GaBi datasets for heat generation through the different sources included in the estimated average mix⁸² under Brazilian or south-American conditions. For the other inputs and outputs, datasets reflecting "Rest of the World" conditions were used to replace those applied in the base case to represent the EU/European average situation, when no specific datasets for Brazilian or south-American conditions were available. Original transport of raw bioethanol from Brazil to Europe for further conversion into bio-based HDPE was replaced with internal transport, considering, for consistency, the default transport scenario specified in the *Plastics LCA* method for transferring of goods within Europe from suppliers to factories/users. This scenario includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km. Conversely, original transport of final bio-based HDPE granulate within Europe for conversion into bottles was replaced with intercontinental transport from Brazil to Europe, consistently applying the same routes considered in the base case for raw bioethanol transport (i.e. the default scenario specified in the *Plastics LCA* method for transferring of goods from suppliers located outside Europe to factories in Europe). These routes include 1000 km by lorry (>32 t, Euro 4) within both Brazil and Europe (from factory to harbour or vice versa), and 11,300 km by freight ship between the two countries.

The characterised potential impacts of bio-based HDPE beverage bottles, recalculated after the changes described above, are reported in Table 3.31. This also provides percentage impact variations with respect to the base case scenario, assuming bio-based HDPE production in Europe based on sugarcane-derived ethanol imported from Brazil. When bio-based HDPE production entirely takes place in Brazil, an overall impact decrease is observed in five impact categories. For most of them (i.e. Photochemical Ozone Formation, Acidification and Eutrophication – terrestrial), the reduction is modest (12%), while it is more relevant for Resource Use – fossils (40%), and Ionising Radiation (108%). On the other hand, the impact is higher in five categories, with an increase mostly ranging from 11% (Water Use) and 19.5% (Ecotoxicity - freshwater), although for Ozone Depletion it is as high as 81%. In the six remaining categories, no relevant changes are observed (i.e. positive or negative impact variations lower than 10%). It must be noted that these results may also be affected by discrepancies in the background datasets applied to model the different inputs and outputs of bio-based ethylene and HDPE production under Brazilian and EU conditions, which could not be all consistently sourced from the same database.

⁸² The mix includes the following heat sources: natural gas (44%); light fuel oil (39.5%); heavy fuel oil (6%); liquefied petroleum gas (4%); coal (3%); and biomass (3%).

Table 3.31. Results of the sensitivity analysis on the origin of bio-based HDPE (considering production of bio-based ethylene and HDPE occurring in Brazil rather than in Europe). Results refer to the total potential impacts of bio-based HDPE beverage bottles and are expressed per functional unit.

Impact category	Bio-based HDPE production in EU (base case)	Bio-based HDPE production in Brazil (SA) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	3.43E+02	3.26E+02	-5%
Ozone Depletion [kg CFC-11 eq.]	3.04E-07	5.50E-07	81%
Human Toxicity - cancer [CTUh]	2.04E-06	2.43E-06	19%
Human toxicity - non-cancer [CTUh]	3.23E-05	3.25E-05	1%
Particulate matter [Disease incidence]	1.07E-04	1.06E-04	-1%
Ionising Radiation [kBq U ²³⁵ eq.]	8.18E+00	-6.42E-01	-108%
Photochemical Ozone Formation [kg NMVOC eq.]	1.61E+00	1.42E+00	-12%
Acidification [mol of H ⁺ eq.]	1.73E+00	1.52E+00	-12%
Eutrophication - terrestrial [mol N eq.]	6.65E+00	5.87E+00	-12%
Eutrophication - freshwater [kg P eq.]	1.20E-02	1.37E-02	14%
Eutrophication - marine [kg N eq.]	2.73E+00	2.66E+00	-3%
Ecotoxicity - freshwater [CTUe]	5.87E+01	7.01E+01	19%
Land Use [Pt]	4.66E+04	4.59E+04	-2%
Water Use [m ³ world eq.]	7.28E+01	8.05E+01	11%
Resource Use - mineral and metals [kg Sb eq.]	5.81E-05	5.99E-05	3%
Resource Use - fossils [MJ]	8.63E+02	5.18E+02	-40%

⁽¹⁾ SA: Sensitivity Analysis.

3.8.5.5 Use of refrigeration and cooling energy in HMF production

As described in Section 3.5.2.3.3, the production process of HMF (the main precursor of the FDCA co-monomer used in the synthesis of PEF) was modelled based on process simulation results from the literature (Motagamwala et al., 2019). However, the interpretation of simulation results and their elaboration and modelling to develop the process inventory was not always straightforward, and implied a number of assumptions and approximations, potentially leading to the modelling of inputs and outputs not fully reflecting the simulated process. In particular, the quantification of one of the most relevant contributions to the overall process impacts (i.e. energy used for refrigeration and cooling water) was associated with larger uncertainty, and the calculated amounts (36.6 and 35 MJ per kg HMF, respectively) may be overestimated, as being almost comparable with the heat demand of the process (47 MJ per kg HMF) (see Table 3.4). Moreover, the modelling of such energy inputs was made based on a proxy dataset related to cooling energy supply from natural gas⁸³, which may not be sufficiently representative of the real supply/production process, especially in the case of refrigeration energy. The impacts of PEF beverage bottles were thus recalculated by first excluding the contribution of refrigeration energy, and afterwards the combined contribution of both refrigeration energy and energy used for cooling water. It must be noted, however, that this is an optimistic theoretical situation, since it totally omits a part of the energy demand of the process, which would be hardly equal to zero. The results of the analysis are presented in Table 3.32.

Excluding the contribution of the sole refrigeration energy decreases the potential impacts of PEF beverage bottles by 13-42% in ten impact categories. In the six remaining categories, the observed impact reduction is not significant, being lower than 10%. When the contribution of energy used for both refrigeration and cooling water is excluded, only two impact categories show a non-significant impact reduction of 3-4% (i.e. Ecotoxicity – freshwater and Land Use), while an even substantial decrease between 12% and 82% is observed in the remaining categories. As mentioned above, this impact reduction is only theoretical and has to be interpreted with caution, since the use of energy for refrigeration and cooling cannot be completely eliminated from a real process.

⁸³ [EU-28] Cooling energy, from natural gas, at cogen unit with absorption chiller 100kW.

Table 3.32. Results of the sensitivity analysis on the use of energy for refrigeration and cooling water in the production of HMF (main precursor of the FDCA co-monomer used in the synthesis of PEF). Results refer to the total potential impacts of PEF beverage bottles and are expressed per functional unit.

Impact category	HMF production including energy for refrigeration and cooling water (base case)	HMF production excluding energy for refrigeration (SA-1) ⁽¹⁾	Variation (%)	HMF production excluding energy for refrigeration and cooling water (SA-2) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	5.37E+02	3.67E+02	-32%	2.04E+02	-62%
Ozone Depletion [kg CFC-11 eq.]	3.39E-06	3.10E-06	-9%	2.83E-06	-17%
Human Toxicity - cancer [CTUh]	3.11E-06	2.53E-06	-19%	1.98E-06	-36%
Human toxicity - non-cancer [CTUh]	6.28E-05	5.85E-05	-7%	5.44E-05	-13%
Particulate matter [Disease incidence]	1.76E-05	1.60E-05	-9%	1.44E-05	-18%
Ionising Radiation [kBq U ²³⁵ eq.]	2.95E+01	2.29E+01	-22%	1.66E+01	-44%
Photochemical Ozone Formation [kg NMVOC eq.]	8.93E-01	7.16E-01	-20%	5.47E-01	-39%
Acidification [mol of H ⁺ eq.]	1.44E+00	1.22E+00	-15%	1.01E+00	-30%
Eutrophication - terrestrial [mol N eq.]	4.57E+00	3.99E+00	-13%	3.44E+00	-25%
Eutrophication - freshwater [kg P eq.]	4.13E-02	3.07E-02	-26%	2.05E-02	-50%
Eutrophication - marine [kg N eq.]	8.51E-01	7.99E-01	-6%	7.50E-01	-12%
Ecotoxicity - freshwater [CTUe]	6.23E+02	6.09E+02	-2%	5.95E+02	-4%
Land Use [Pt]	1.12E+04	1.10E+04	-2%	1.09E+04	-3%
Water Use [m ³ world eq.]	2.08E+02	1.73E+02	-17%	1.39E+02	-33%
Resource Use - mineral and metals [kg Sb eq.]	8.87E-04	5.16E-04	-42%	1.61E-04	-82%
Resource Use - fossils [MJ]	7.70E+03	4.96E+03	-36%	2.35E+03	-69%

⁽¹⁾ SA: Sensitivity Analysis.

3.8.5.6 Applied iLUC model and resulting GHG emission factors

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the contribution of GHG emissions from iLUC to the Climate Change impact indicator. The iLUC GHG emission factors calculated with the model of Schmidt et al. (2015) were applied in place of the emission factors based on EC (2015) applied in the base case (Table 3.12, Section 3.5.6). The results are displayed in Table 3.33, limited to the Climate Change impact category, which is the only one affected by this sensitivity analysis.

Overall, the application of alternative iLUC GHG emission factors only marginally affects the results. The highest increase in the total Climate Change impact observed in the base case when accounting for the iLUC contribution was equal to 9.1% (bio-based HDPE bottles from Brazilian sugarcane). Applying the factors derived from the model of Schmidt et al. (2015) leads to a comparable increase of 9.9% (corresponding to a nearly 8% increase of the iLUC contribution compared to the base case). An equal increase of the iLUC contribution by 8% is also observed for bio-based PET bottles, these being still based on sugarcane as a feedstock (although limited to 30% of the polymer). This corresponds to a negligible variation in the increase of the total Climate Change impact when the iLUC contribution is accounted (from 2.2% to 2.4%). For PEF bottles, the impact increase due to iLUC was the lowest observed in the base case results (1.4%), but it is the one showing the highest relative increase in the iLUC contribution when the model of Schmidt et al. (2015) is applied, i.e. 64%. However, the increase in the total Climate Change impact is again comparable (2.3% instead of 1.4%). In conclusion, the results of this case study can be considered reasonably robust with respect to the estimated contribution of GHG emissions from iLUC to Climate Change (which is modest or negligible).

Table 3.33. Results of the sensitivity analysis on the emission factors applied to quantify the contribution of GHG emissions from iLUC to the Climate Change impact indicator, and resulting total Climate Change impact of affected scenarios (in brackets). Results are not intended to compare the different beverage bottles scenarios.

Scenario	Emission factors based on EC (2015) (base case) [kg CO ₂ eq./FU]	Emission factors from the model of Schmidt et al. (2015) (SA) ⁽¹⁾ [kg CO ₂ eq./FU]	Variation (%)
Bio-based PET bottles	4.38 (201)	4.75 (201)	+8.32 (+0.18)
Bio-based HDPE bottles	31.3 (374)	33.9 (377)	+8.32 (+0.70)
PEF bottles	7.46 (544)	12.2 (549)	+63.9 (+0.88)

⁽¹⁾ SA: Sensitivity Analysis.

3.8.5.7 Handling of non-released biogenic carbon at End of Life

This sensitivity analysis explores the effects of accounting for the impact of biogenic carbon not released after 100 years from landfilling of bio-based beverage bottles (i.e. Bio-PET, Bio-HDPE and PEF bottles) on the Climate Change impact indicator. In this study, landfill emissions were modelled considering a time horizon of 100 years from waste deposition, and (biogenic) carbon in landfilled products that is not degraded or mineralised during such time horizon was considered to be never released from the landfill body. However, the effects of non-released biogenic carbon are not captured in the Climate Change impact results calculated as a base case, since characterisation factors for biogenic CO₂ emissions and removals are set to zero in the *Plastics LCA* method (fully conforming to the PEF method). To better understand the implications of this methodological choice, the Climate Change impact indicator of the three mentioned

bio-based alternatives was thus recalculated accounting for the effects of non-released biogenic carbon.

For this alternative calculation, a specific CO₂ uptake was modelled in the landfilling inventories of Bio-PET, Bio-HDPE and PEF bottles, expressing the net amount of biogenic CO₂ taken up from biomass embodied in the product, and ultimately not released (as CO₂ or CH₄) during bottles degradation in landfill. This uptake was then characterised applying a characterisation factor of -1 kg CO₂ eq. per kg CO₂ not released. The uptake was calculated based on the biogenic carbon content in the product (i.e. in the respective polymer), and assuming a degradation rate over 100 years equal to 1% for both bio-based PET and bio-based HDPE (in line with the values considered for their fossil-based counterparts in Doka, 2009b). Considering a biogenic carbon content equal to 12.5% for bio-based PET (i.e. 20% of a total carbon content of 62.5%), and to 81.9% for bio-based HDPE, a specific carbon uptake equal to 0.454 kg CO₂/kg Bio-PET landfilled and 2.97 kg CO₂/kg Bio-HDPE landfilled was calculated, respectively. For PEF, which is a bio-based but non-biodegradable polymer, a degradation rate of 1% was again assumed, consistently with the values reported in Doka (2009b) for conventional non-biodegradable polymers such as PET and HDPE. Considering that the biogenic carbon content in PEF is equal to 52% (Table 3.10), a specific CO₂ uptake equal to 1.89 kg CO₂/kg PEF sent to landfill was calculated.

The results of this sensitivity analysis are presented in Table 3.34, considering that the affected beverage bottles scenarios apply the EU-average End of Life scenarios assumed as a base case, and described in Section 3.5.5.1. For completeness, the results were also calculated considering the application of landfilling as an individual “100%” End of Life option (as investigated in the sensitivity analysis described in Section 3.8.5.8), and are presented in Table 3.35. This is, however, an extreme situation that is unlikely to happen in reality (at least in a European context).

Table 3.34. Results of the sensitivity analysis on the handling of non-released biogenic carbon at End of Life (landfilling) for bio-based PET, bio-based HDPE and PEF beverage bottles, considering that the EU-average End of Life scenario assumed as a base case is applied. Only the Climate Change indicator is affected, and presented in the table. Results are not intended to compare the different beverage bottles scenarios.

Scenarios	Effects of non-released biogenic C not accounted (base case) [kg CO₂ eq./FU]	Effects of non-released biogenic C accounted (SA) ⁽¹⁾ [kg CO₂ eq./FU]	Variation (%)
Bio-based PET bottles	196	192	-2.1%
Bio-based HDPE bottles	343	316	-7.9%
PEF bottles	537	523	-2.5%

⁽¹⁾ SA: Sensitivity Analysis.

When considering the application of the assumed EU-average End of Life scenarios, the Climate Change impact of Bio-PET, Bio-HDPE and PEF bottles is only slightly reduced by accounting for the contribution of non-released biogenic carbon after 100 years from landfilling. Indeed, a decrease between 2% (Bio-PET bottles) and 8% (Bio-HDPE bottles) is observed, while for PEF bottles the reduction equals 2.5%. This can be explained by the modest share of landfilling in the applied EU-average End of Life scenario (19%) and, in the case of bio-based PET bottles, by the only partial biogenic carbon content in the polymer (i.e. 12.5%). When landfilling is applied as an independent End of Life option, observed reductions in the Climate Change impact are more relevant, especially for bio-based HDPE bottles (42%), which have the highest biogenic carbon content (approximately 82%). For bio-based PET and PEF bottles, the decrease is more reduced (around 11%), due to the lower biogenic carbon content (especially in Bio-PET, which is

only 30% bio-based), and to the proportionally higher impact of PEF bottles compared to the achieved reduction when the effects of non-released biogenic carbon are accounted.

Table 3.35. Results of the sensitivity analysis on the handling of non-released biogenic carbon at End of Life (landfilling) for bio-based PET, bio-based HDPE and PEF beverage bottles, considering that landfilling is applied as a stand-alone End of Life option. Only the Climate Change indicator is affected, and presented in the table. Results are not intended to compare the different beverage bottles scenarios.

Scenarios	Effects of non-released biogenic C not accounted (base case) [kg CO₂ eq./FU]	Effects of non-released biogenic C accounted (SA) ⁽¹⁾ [kg CO₂ eq./FU]	Variation (%)
Bio-based PET bottles	206	184	-10.7%
Bio-based HDPE bottles	377	219	-41.9%
PEF bottles	667	593	-11.1%

3.8.5.8 Alternative End of Life scenarios

This sensitivity analysis evaluates the effects of individually applying each End of Life option specified in Table 3.1 as an independent (100%) End of Life scenario replacing the EU-average scenario considered as a base case. Investigated options include mechanical recycling, incineration and landfilling. The main purpose is to evaluate how the potential impacts of each beverage bottle alternative are affected by changes in the applied End of Life scenario, although in reality the different considered End of Life options would be hardly implemented individually, but in combination (e.g. as reflected in the assumed EU-average End of Life scenario). The detailed numerical results are separately presented for each beverage bottle alternative in Tables 3.36–3.42, while a more synthetic overview is provided in Figures E.1.1–E.1.3 in Annex E.1.

Note that these results should not be interpreted as a direct comparison among alternative End of Life options for beverage bottles, since the evaluation applies a “product perspective”, and burdens/benefits of some End of Life options (e.g. recycling) are partitioned between different (consecutive) product life cycles. This prevents from capturing the full environmental implications of having a given waste stream routed to such End of Life options (especially in case of recycling). Conversely, in a waste management LCA of alternative End of Life options for the product (e.g. based on a functional unit of 1 tonne of product waste to be managed), each pathway would be assigned the full burdens and benefits it involves. In such case, there is indeed no need to break mass and energy flows between the waste management system providing recovered material/energy, and the product system using them (i.e. a “system perspective” is applied, and no allocation is needed). In this perspective, the total (system-wise) benefits associated with the End of Life pathway “100% recycling” would hence be higher than those considered to calculate the results presented in this section, based on a product perspective.

None of the product scenarios individually applying the three considered End of Life options (i.e. mechanical recycling, incineration and landfilling) can be identified as preferable across all the assessed impact categories and beverage bottles scenarios. Moreover, in several cases, no relevant changes in the overall impacts of such scenarios occur by implementing alternative End of Life options, with impact differences lower than 10% (which are considered not significant). Inside this framework, the following considerations can be made.

The scenario applying 100% mechanical recycling is preferable across the whole set of impact categories only in the case of PEF bottles, likely due to the higher Polymer

Production impacts and the consequent larger benefits from avoided virgin material production at End of Life recycling, compared to the other beverage bottles alternatives. For all other alternatives than PEF bottles, and except bio-based HDPE bottles, the 100% mechanical recycling scenario is preferable in four impact categories, including Climate Change, Human Toxicity – cancer, Ecotoxicity – freshwater and Water Use. For the three PET bottles scenarios (virgin, partially recycled and partially bio-based) this is also the case of Resource Use – fossils and Human Toxicity – non-cancer (except for bio-based PET bottles). Conversely, for bio-based HDPE bottles, the 100% recycling scenario is preferable only in Human Toxicity – cancer, while it is comparable with the 100% incineration scenario in most of the remaining categories (i.e. all except for Climate Change, Ozone Depletion, Human Toxicity – cancer and Ionising Radiation). Also for the three PET bottles alternatives the recycling and incineration scenarios are comparable in several (eight) impact categories beyond those where the recycling scenario is preferable (i.e. the six categories mentioned above), while for virgin and partially recycled HDPE bottles this happens in four categories. In Ozone Depletion, the recycling scenario shows the worst performance for all beverage bottles alternatives except for PEF bottles, where, as mentioned above, it is the preferable one. Similarly, in Ionising Radiation, the 100% recycling scenario is (except for PEF bottles) almost comparable with that relying on landfilling, which is the least preferable in that category. For bio-based HDPE bottles, this is also the case of Land Use.

If PEF bottles are excluded, the product scenario considering 100% incineration is the most favourable for all beverage bottles alternatives only in the Ionising Radiation impact category. However, in the case of virgin and partially recycled HDPE bottles, the incineration scenario is preferable also in the categories of Ozone Depletion (only virgin HDPE bottles), Particulate Matter, Acidification, Eutrophication – terrestrial, Eutrophication – marine, Land Use and Resource Use – minerals and metals. In Ozone Depletion, this is also the case of PET and partially bio-based PET bottles. On the other hand, the incineration scenario shows the highest impact in terms of Climate Change (all beverage bottles alternatives except for bio-based HDPE and PEF bottles) and Water Use (except for PEF bottles, where incineration is comparable with landfilling). For PEF bottles, 100% incineration is the worst scenario also in Human Toxicity – non-cancer.

In many impact categories, the product scenario applying 100% landfilling is the worst for all or most beverage bottles alternatives, although in some cases impact differences with respect to scenarios relying on one or more of the other End of Life options are not significant (lower than 10%). In general, this is in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option (EC, 2008). Exceptions to the above are Climate Change, Ozone Depletion, and Water Use, where, for most beverage bottles alternatives, scenarios relying on other End of Life options show the highest impact, as discussed above. Overall, in none of the assessed impact categories and beverage bottles scenarios the 100% landfilling scenario is preferable, although in some cases it is comparable with scenarios showing the lowest impact.

Table 3.36. Characterised potential impacts of fossil-based PET beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.91E+02	1.63E+02	2.63E+02	1.98E+02
Ozone Depletion [kg CFC-11 eq.]	2.95E-08	4.24E-08	7.64E-09	1.29E-08
Human Toxicity - cancer [CTUh]	1.22E-06	1.08E-06	1.40E-06	1.47E-06
Human toxicity - non-cancer [CTUh]	5.13E-06	4.52E-06	5.51E-06	6.66E-06
Particulate matter [Disease incidence]	3.30E-06	3.16E-06	3.12E-06	3.93E-06
Ionising Radiation [kBq U ²³⁵ eq.]	1.16E+01	1.26E+01	7.12E+00	1.35E+01
Photochemical Ozone Formation [kg NMVOC eq.]	3.51E-01	3.31E-01	3.56E-01	4.07E-01
Acidification [mol of H ⁺ eq.]	4.20E-01	4.04E-01	4.05E-01	4.86E-01
Eutrophication - terrestrial [mol N eq.]	1.35E+00	1.30E+00	1.37E+00	1.52E+00
Eutrophication - freshwater [kg P eq.]	5.83E-04	4.39E-04	3.90E-04	1.25E-03
Eutrophication - marine [kg N eq.]	1.16E-01	1.10E-01	1.18E-01	1.34E-01
Ecotoxicity - freshwater [CTUe]	2.48E+01	2.21E+01	2.78E+01	2.98E+01
Land Use [Pt]	4.37E+02	4.09E+02	4.50E+02	5.14E+02
Water Use [m ³ world eq.]	4.20E+01	3.51E+01	5.62E+01	4.80E+01
Resource Use - mineral and metals [kg Sb eq.]	1.79E-05	1.74E-05	1.71E-05	2.03E-05
Resource Use - fossils [MJ]	3.74E+03	3.37E+03	3.99E+03	4.62E+03

Table 3.37. Characterised potential impacts of fossil-based HDPE beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.39E+02	1.20E+02	2.03E+02	1.41E+02
Ozone Depletion [kg CFC-11 eq.]	9.25E-08	1.22E-07	3.51E-08	4.74E-08
Human Toxicity - cancer [CTUh]	1.43E-06	1.28E-06	1.65E-06	1.76E-06
Human toxicity - non-cancer [CTUh]	5.82E-06	5.31E-06	5.82E-06	7.74E-06
Particulate matter [Disease incidence]	3.15E-06	3.17E-06	2.30E-06	4.03E-06
Ionising Radiation [kBq U ²³⁵ eq.]	4.37E+00	6.72E+00	-6.74E+00	7.91E+00
Photochemical Ozone Formation [kg NMVOC eq.]	3.51E-01	3.40E-01	3.17E-01	4.31E-01
Acidification [mol of H ⁺ eq.]	3.74E-01	3.77E-01	2.86E-01	4.64E-01
Eutrophication - terrestrial [mol N eq.]	1.21E+00	1.21E+00	1.04E+00	1.39E+00
Eutrophication - freshwater [kg P eq.]	7.22E-04	5.42E-04	5.74E-04	1.57E-03
Eutrophication - marine [kg N eq.]	1.04E-01	1.04E-01	8.46E-02	1.23E-01
Ecotoxicity - freshwater [CTUe]	3.08E+01	2.81E+01	3.39E+01	3.76E+01
Land Use [Pt]	3.60E+02	3.78E+02	2.48E+02	4.18E+02
Water Use [m ³ world eq.]	2.25E+01	1.91E+01	3.41E+01	2.26E+01
Resource Use - mineral and metals [kg Sb eq.]	1.46E-05	1.52E-05	1.00E-05	1.76E-05
Resource Use - fossils [MJ]	3.28E+03	3.06E+03	3.01E+03	4.41E+03

Table 3.38. Characterised potential impacts of 24% recycled PET beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.75E+02	1.47E+02	2.47E+02	1.82E+02
Ozone Depletion [kg CFC-11 eq.]	1.72E-07	1.85E-07	1.50E-07	1.56E-07
Human Toxicity - cancer [CTUh]	1.11E-06	9.65E-07	1.28E-06	1.35E-06
Human toxicity - non-cancer [CTUh]	4.92E-06	4.31E-06	5.30E-06	6.45E-06
Particulate matter [Disease incidence]	3.02E-06	2.89E-06	2.84E-06	3.66E-06
Ionising Radiation [kBq U ²³⁵ eq.]	1.10E+01	1.20E+01	6.57E+00	1.29E+01
Photochemical Ozone Formation [kg NMVOC eq.]	3.15E-01	2.96E-01	3.21E-01	3.72E-01
Acidification [mol of H ⁺ eq.]	3.83E-01	3.66E-01	3.68E-01	4.49E-01
Eutrophication - terrestrial [mol N eq.]	1.26E+00	1.20E+00	1.27E+00	1.42E+00
Eutrophication - freshwater [kg P eq.]	6.76E-04	5.32E-04	4.83E-04	1.35E-03
Eutrophication - marine [kg N eq.]	1.08E-01	1.02E-01	1.10E-01	1.26E-01
Ecotoxicity - freshwater [CTUe]	2.31E+01	2.05E+01	2.62E+01	2.81E+01
Land Use [Pt]	4.69E+02	4.40E+02	4.81E+02	5.46E+02
Water Use [m ³ world eq.]	3.70E+01	3.01E+01	5.12E+01	4.30E+01
Resource Use - mineral and metals [kg Sb eq.]	1.92E-05	1.87E-05	1.84E-05	2.16E-05
Resource Use - fossils [MJ]	3.23E+03	2.86E+03	3.48E+03	4.12E+03

Table 3.39. Characterised potential impacts of 16% recycled HDPE beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.34E+02	1.15E+02	1.98E+02	1.36E+02
Ozone Depletion [kg CFC-11 eq.]	1.91E-07	2.20E-07	1.34E-07	1.46E-07
Human Toxicity - cancer [CTUh]	1.34E-06	1.19E-06	1.57E-06	1.67E-06
Human toxicity - non-cancer [CTUh]	5.62E-06	5.11E-06	5.62E-06	7.54E-06
Particulate matter [Disease incidence]	2.99E-06	3.01E-06	2.14E-06	3.87E-06
Ionising Radiation [kBq U ²³⁵ eq.]	4.14E+00	6.50E+00	-6.97E+00	7.68E+00
Photochemical Ozone Formation [kg NMVOC eq.]	3.30E-01	3.19E-01	2.95E-01	4.10E-01
Acidification [mol of H ⁺ eq.]	3.54E-01	3.57E-01	2.66E-01	4.44E-01
Eutrophication - terrestrial [mol N eq.]	1.16E+00	1.16E+00	9.88E-01	1.34E+00
Eutrophication - freshwater [kg P eq.]	7.85E-04	6.05E-04	6.36E-04	1.63E-03
Eutrophication - marine [kg N eq.]	9.99E-02	1.00E-01	8.08E-02	1.19E-01
Ecotoxicity - freshwater [CTUe]	2.97E+01	2.70E+01	3.27E+01	3.64E+01
Land Use [Pt]	3.88E+02	4.05E+02	2.75E+02	4.45E+02
Water Use [m ³ world eq.]	2.16E+01	1.81E+01	3.31E+01	2.17E+01
Resource Use - mineral and metals [kg Sb eq.]	1.56E-05	1.62E-05	1.10E-05	1.86E-05
Resource Use - fossils [MJ]	2.99E+03	2.77E+03	2.72E+03	4.11E+03

Table 3.40. Characterised potential impacts of 30% bio-based PET beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.96E+02	1.72E+02	2.54E+02	2.06E+02
Ozone Depletion [kg CFC-11 eq.]	2.14E-08	3.47E-08	-1.90E-09	5.30E-09
Human Toxicity - cancer [CTUh]	1.10E-06	9.60E-07	1.27E-06	1.34E-06
Human toxicity - non-cancer [CTUh]	6.79E-06	6.31E-06	6.66E-06	8.49E-06
Particulate matter [Disease incidence]	1.30E-05	1.29E-05	1.26E-05	1.37E-05
Ionising Radiation [kBq U ²³⁵ eq.]	1.08E+01	1.22E+01	4.53E+00	1.32E+01
Photochemical Ozone Formation [kg NMVOC eq.]	4.33E-01	4.16E-01	4.28E-01	4.93E-01
Acidification [mol of H ⁺ eq.]	5.19E-01	5.09E-01	4.82E-01	5.93E-01
Eutrophication - terrestrial [mol N eq.]	1.73E+00	1.68E+00	1.69E+00	1.91E+00
Eutrophication - freshwater [kg P eq.]	1.87E-03	1.73E-03	1.64E-03	2.55E-03
Eutrophication - marine [kg N eq.]	4.50E-01	4.45E-01	4.47E-01	4.69E-01
Ecotoxicity - freshwater [CTUe]	2.42E+01	2.17E+01	2.70E+01	2.93E+01
Land Use [Pt]	6.67E+03	6.78E+03	6.14E+03	6.91E+03
Water Use [m ³ world eq.]	3.75E+01	3.09E+01	5.08E+01	4.39E+01
Resource Use - mineral and metals [kg Sb eq.]	1.79E-05	1.75E-05	1.63E-05	2.06E-05
Resource Use - fossils [MJ]	2.94E+03	2.57E+03	3.20E+03	3.82E+03

Table 3.41. Characterised potential impacts of bio-based HDPE beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	3.43E+02	3.50E+02	2.86E+02	3.77E+02
Ozone Depletion [kg CFC-11 eq.]	3.04E-07	3.34E-07	2.44E-07	2.60E-07
Human Toxicity - cancer [CTUh]	2.04E-06	1.89E-06	2.25E-06	2.36E-06
Human toxicity - non-cancer [CTUh]	3.23E-05	3.20E-05	3.13E-05	3.45E-05
Particulate matter [Disease incidence]	1.07E-04	1.07E-04	1.06E-04	1.08E-04
Ionising Radiation [kBq U ²³⁵ eq.]	8.18E+00	1.14E+01	-6.80E+00	1.28E+01
Photochemical Ozone Formation [kg NMVOC eq.]	1.61E+00	1.60E+00	1.55E+00	1.69E+00
Acidification [mol of H ⁺ eq.]	1.73E+00	1.74E+00	1.58E+00	1.83E+00
Eutrophication - terrestrial [mol N eq.]	6.65E+00	6.67E+00	6.38E+00	6.86E+00
Eutrophication - freshwater [kg P eq.]	1.20E-02	1.18E-02	1.18E-02	1.29E-02
Eutrophication - marine [kg N eq.]	2.73E+00	2.74E+00	2.71E+00	2.76E+00
Ecotoxicity - freshwater [CTUe]	5.87E+01	5.61E+01	6.12E+01	6.56E+01
Land Use [Pt]	4.66E+04	4.69E+04	4.53E+04	4.70E+04
Water Use [m ³ world eq.]	7.28E+01	6.97E+01	8.25E+01	7.34E+01
Resource Use - mineral and metals [kg Sb eq.]	5.81E-05	5.90E-05	5.19E-05	6.15E-05
Resource Use - fossils [MJ]	8.63E+02	6.45E+02	5.88E+02	1.99E+03

Table 3.42. Characterised potential impacts of PEF beverage bottles for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	5.37E+02	4.60E+02	6.42E+02	6.67E+02
Ozone Depletion [kg CFC-11 eq.]	3.39E-06	2.81E-06	4.29E-06	4.25E-06
Human Toxicity - cancer [CTUh]	3.11E-06	2.58E-06	4.08E-06	3.73E-06
Human toxicity - non-cancer [CTUh]	6.28E-05	4.99E-05	8.92E-05	7.41E-05
Particulate matter [Disease incidence]	1.76E-05	1.49E-05	2.14E-05	2.20E-05
Ionising Radiation [kBq U ²³⁵ eq.]	2.95E+01	2.66E+01	3.19E+01	3.58E+01
Photochemical Ozone Formation [kg NMVOC eq.]	8.93E-01	7.64E-01	1.07E+00	1.10E+00
Acidification [mol of H ⁺ eq.]	1.44E+00	1.23E+00	1.73E+00	1.79E+00
Eutrophication - terrestrial [mol N eq.]	4.57E+00	3.89E+00	5.52E+00	5.63E+00
Eutrophication - freshwater [kg P eq.]	4.13E-02	3.43E-02	5.20E-02	5.19E-02
Eutrophication - marine [kg N eq.]	8.51E-01	7.12E-01	1.05E+00	1.07E+00
Ecotoxicity - freshwater [CTUe]	6.23E+02	5.11E+02	8.11E+02	7.71E+02
Land Use [Pt]	1.12E+04	9.39E+03	1.36E+04	1.42E+04
Water Use [m ³ world eq.]	2.08E+02	1.74E+02	2.60E+02	2.61E+02
Resource Use - mineral and metals [kg Sb eq.]	8.87E-04	7.36E-04	1.11E-03	1.12E-03
Resource Use - fossils [MJ]	7.70E+03	6.56E+03	9.25E+03	9.59E+03

4 Case study 2: Agricultural mulching film

This case study investigates plastic mulching film used in large-scale agriculture. Plastic mulching is an agricultural practice consisting in the mechanical application of a thin plastic film (black, white or transparent, depending on needs) on the soil. Seedlings and young plants are then planted by punching holes in the film. As reported by Steinmetz et al. (2016), the primary function of plastic mulching is seedlings and shoots protection, by thermally insulating soil and preventing water evaporation (Tarara, 2000), thus maintaining or slightly increasing soil temperature and humidity. However, it is also used to reduce exposure to pests and for weed control (McKenzie et al., 2001), while preventing soil erosion. A number of short-term agronomic benefits are hence often recognised to plastic mulching as a result of these beneficial functions, such as yield increase, earlier development of seeds and fruits, improved fruit quality, increased water-use efficiency, and a consequent reduction in the use of fertilisers and pesticides (Scarascia-Mugnozza, 2011; Chalker-Scott, 2007; Espí et al., 2006; Lamont, 1993). These beneficial effects typically lead to instant economic advantages for farmers, so that plastic mulching has become an increasingly used technique in the global agriculture (Steinmetz et al., 2016). However, as reported by the same authors, in the medium-long term using plastic mulches may also generate adverse effects on soil quality and the environment, which in turn may negatively affect crop growth and quality, thereby potentially defeating any short-term agronomic and economic benefits. Plastic mulching can indeed increase degradation of soil organic matter (SOM) (due to higher temperatures generating below the film during use), potentially promoting depletion of soil nutrients and carbon stocks, and subsequently reducing the water retention capacity of soil (strictly connected with its SOM content) and increasing the release of greenhouse gas emissions from SOM degradation (Steinmetz et al., 2016). Moreover, continued use of plastic film for mulching facilitates the accumulation of plastic fragments in soil over time (especially where proper and complete collection of film after use is not ensured), and in the long term this may further deteriorate soil quality and crop growth (Steinmetz et al., 2016). Plastic residues may then also disintegrate into micro-plastics, which can adsorb agro-chemicals (Steinmetz et al., 2016) and negatively affect soil and/or other ecosystems (although these effects are not yet proven nor understood). Finally, plastic additives with toxic effects may be released (both during and after use of mulching film), and pesticide runoff may be promoted (while their leaching is potentially reduced) (Steinmetz et al., 2016).

In Europe, plastic mulching is used mainly in horticulture. The most common material currently used for the production of mulching film is Polyethylene (PE), with Low Density Poly Ethylene (LDPE) being the most widely applied polymer (Scarascia-Mugnozza, 2011). This is because of its high flexibility and durability, low cost, and large availability, which enable easy mechanical application on a commercial scale (Tofanelli and Wortman, 2020). Films made of biodegradable plastic materials in soil have also been made available on the market during the last 15 years (European Bioplastics, 2017), mainly based on (combinations of) biodegradable polyesters such as PLA (Polylactic Acid) and PBAT (Polybutylene Adipate co-Terephthalate), or starch-polyester blends, while other materials are still under research (e.g. PHA and PHB)⁸⁴ (Tofanelli and Wortman, 2020). Biodegradable films do not require collection from the field after use and their subsequent treatment or disposal (Kasirajan and Ngouajio, 2012), thus avoiding the costs and environmental impacts associated with these operations. If well designed, they may also help to prevent accumulation of persistent plastic fragments and particles in soil over time, and the resulting potential detrimental effects on soil quality and crops. However, as for traditional PE film remaining in soil after use, the fate of biodegradable films still have to be completely investigated and understood, as well as their effects on plastic accumulation and on soil/crops quality. Moreover, on-farm adoption of biodegradable mulching film has been reported to be limited in the past by premature deterioration of the film during the growing season, unpredictably slow biodegradation

⁸⁴ PHA: Poly-Hydroxy-Alkanoates; PHB: Poly-Hydroxy-Butyrate.

rates in soil, and concerns about the environmental fate of polymers (Tofanelli and Wortman, 2020). This has hindered acceptance of biodegradable films as a functional alternative to PE films by farmers (Steinmetz et al., 2016).

The focus of this study is on generic mulching film used in large-scale agriculture for unspecified horticultural crops, and hence with no defined crop-specific functions to be provided. The colour of the film is also not specified, and a typical average thickness is considered for each type of film investigated in the study (see Sections 4.1 and 4.2). This choice allows to keep a broader scope, extend the validity of results beyond very specific situations, and to overcome the scarcity of information on the characteristics of mulching film applied to specific (horticultural) crops. On the other hand, this implies that the technical characteristics assumed for the different types of film do not necessarily ensure the achievement of an equivalent product (agronomic) performance across all possible applications (i.e. crop types).

4.1 Assessed scenarios

In this case study, the potential impacts of conventional and biodegradable mulching film alternatives relying on different polymers and/or feedstock sources were quantified by modelling four separate LCA scenarios (Table 4.1). The most common alternative currently used for large-scale agriculture is non-biodegradable LDPE mulching film (Tofanelli and Wortman, 2020; Scarascia-Mugnozza, 2011), which was considered as a first scenario, assuming that polymer production is entirely based on virgin material from fossil-based feedstock (Scenario 1 – S1). The use of partially recycled LDPE from separately collected post-consumer plastic waste was then assessed in Scenario 2 (S2), considering a 35% recycled content as representative of the current situation, and assuming that no changes in thickness are needed. The recycled content was calculated as the rounded average of the share estimated by Eunomia (2021) specifically for mulching film (i.e. 52% in 2018)⁸⁵, and of that estimated by Plastics Recyclers Europe (PRE, 2020) for agricultural films in general (i.e. 22% in 2018). Due to the non-negligible difference between the two estimates, the application of an average value was preferred over a single estimate, albeit more specific for mulching film. However, the specific estimate available for mulching film (rounded to 50%) was applied in a sensitivity analysis, which also explored the effects of a hypothetical complete substitution of virgin with recycled material (i.e. a 100% recycled content) (Section 4.8.5.2).

Two biodegradable mulching film alternatives were investigated in Scenarios 3 (S3) and 4 (S4), separately considering the use of the two most common partially bio-based polymer blends currently available on the market for biodegradable mulching film manufacturing. In S3, a starch-based polymer blend of Thermoplastic Starch (TPS; 40%) and Polybutylene Adipate co-Terephthalate (PBAT, 60%) was considered⁸⁶, while in S4a blend of Polylactic Acid (PLA) and PBAT (in a 45%-55% proportion)⁸⁷ was used as film material (referred to as PLA-based polymer).

PLA was assumed to be entirely produced in the US from maize grown in the same country, since nearly 90% of PLA currently consumed in Europe comes from the US (Eurostat, 2019a), and maize is used as a source of starch-derived sugar (dextrose) by the largest PLA producer in that country (Vink and Davies, 2015). However, the use of PLA entirely produced in the EU from locally cultivated maize was investigated in a

⁸⁵ The estimate is based on the total recycled material use for mulching film manufacturing in Europe quantified by Plastics Recyclers Europe for 2018 (i.e. 43 kt), and on a total plastic consumption for agricultural mulching film equal to 83 kt in 2019 (according to APE Europe data).

⁸⁶ Note that the shares assumed for the two copolymers do not specifically reflect any particular product currently on the market, as they were defined based on stakeholder inputs reporting that "renewability" (i.e. the bio-based content) of starch-based polymers used in packaging applications can reach 40%. In this case study, the reported share was reasonably extended also to polymer blends used for mulching film manufacturing, in the absence of specific data for these.

⁸⁷ The shares of the two copolymers were defined based on the technical specifications of one of the major PLA-based polymer blends currently available on the market for biodegradable mulching film manufacturing.

sensitivity analysis (Section 4.8.5.4). Starch was assumed to be produced in the EU from the current EU-average mix of starch crops, being one of the major producers of starch-based polymers for mulching film manufacturing located in Europe. The mix of crops was estimated to include maize (47% on starch basis), wheat (40%) and potatoes (13%) (Starch Europe, 2019). PBAT used as a copolymer in both polymer blends is based on fossil resources, and no agricultural feedstock are involved in this case.

Table 4.1. LCA scenarios assessed for the mulching film case study and respective End of Life options and scenarios.

Scenario	Polymer	Monomer or Co-polymer	Feedstock	End of Life options/ scenario ⁽¹⁾
1 - Conventional polymer (S1)	LDPE ⁽²⁾	Ethylene	Fossil-based (crude oil/ natural gas)	Incineration (44%) Landfilling (46%) Burial in soil (10%) (Recycling – 0%)
2 - Conventional polymer 2 (S2)	R-LDPE ⁽²⁾ (35% recycled content)	Ethylene	Waste LDPE (post-consumer)	Incineration (44%) Landfilling (46%) Burial in soil (10%) (Recycling – 0%)
3 - Alternative polymer 2 (S3)	TPS/PBAT blend ^(3,4)	TPS PBAT ⁽⁵⁾	EU mix of starch crops ⁽⁶⁾ Crude oil/natural gas	In-situ biodegradation (100%)
4 - Alternative polymer 3 (S4)	PLA/PBAT blend ^(7,8)	PLA ⁽⁹⁾ PBAT ⁽⁵⁾	Maize (US) Crude oil/natural gas	In-situ biodegradation (100%)

⁽¹⁾ The impacts of each scenario were calculated considering an EU-average End of Life scenario combining the listed End of Life options according with the reported shares. A sensitivity analysis individually considering the application of each listed option (plus recycling of non-biodegradable films) was also conducted.

⁽²⁾ LDPE: Low Density Poly Ethylene.

⁽³⁾ TPS: Thermoplastic Starch; PBAT: Polybutylene Adipate co-Terephthalate.

⁽⁴⁾ A polymer blend consisting of 40% TPS and 60% PBAT was considered, according to the information received from stakeholders consultation, reporting that “renewability” (i.e. the bio-based content) of starch-based polymers used in packaging applications can reach 40% (and reasonably extending this share also to polymer blends used for mulching film manufacturing, in the absence of specific data).

⁽⁵⁾ PBAT is a copolymer of 1,4-Butanediol, Adipic acid and Purified Terephthalic Acid (PTA).

⁽⁶⁾ The mix includes Maize (47%), Wheat (40%), and Potatoes (13%), in terms of starch product equivalents (Starch Europe, 2019).

⁽⁷⁾ PLA: PolyLactic Acid; PBAT: Polybutylene Adipate co-Terephthalate.

⁽⁸⁾ Consisting of 45% PLA and 55% PBAT, according to the technical specification of one of the major PLA-based polymer blends currently available on the market for biodegradable mulching film manufacturing.

⁽⁹⁾ PLA is a polymer of Lactic Acid.

Regarding the product End of Life, all treatment and disposal options currently applied at the EU level were considered for each mulching film alternative. For films made of conventional, non-biodegradable polymers (i.e. LDPE and partially recycled LDPE) applied options include incineration and landfilling, while (mechanical) recycling is reported to be currently not applied (Eunomia, 2021). Burial in soil of mulching film residues not collected from the field after use was also considered, albeit it is not an intended End of Life option. For biodegradable alternatives (i.e. starch-based and PLA-based film), only in-situ biodegradation was considered, since they are primarily intended to be left on the field at the end of the growing season and then ploughed into soil for biodegradation. As a base case, the impacts of each scenario were thus assessed with reference to an EU-average End of Life scenario including all the options currently applied to the specific mulching film alternative, which for non-biodegradable films were combined as described in Section 4.5.5.1 and as summarised in Table 4.1. However, in a sensitivity analysis,

scenario impacts were also recalculated by individually applying each considered End of Life option, including recycling as a potentially applicable option for non-biodegradable films (Section 4.8.5.7).

4.2 Functional Unit and reference flow

As discussed in the foreword, the main function of plastic mulching film is to protect crops, facilitate their growth, and improve the respective quality by controlling weeds, retaining the moisture content in soil and roots, and by maintaining or increasing soil temperature (Razza and Cerutti, 2017; Steinmetz et al., 2016). The functional unit was thus defined as: *"providing mulching and the associated agronomic functions to 1 hectare (10,000 m²) of agricultural land cultivated with horticultural crops (e.g. melon, strawberry, zucchini) in the EU, for a period corresponding to an average growing season (i.e. four to five months)"* (Table 4.2). A specific, quantified agronomic performance to be provided (beyond mulching the specified area of land) could not be identified.

Table 4.2. Functional unit defined for mulching film LCA scenarios.

Aspect	Description
"What" (function or service provided)	Providing mulching and the associated agronomic functions to agricultural land cultivated with horticultural crops (e.g. melon, strawberry, zucchini) through plastic film
"How much" (extent of the function or service)	1 hectare (10,000 m ²) of cultivated land (i.e. 0.6 ha -6000 m ² - of mulched land)
"How well" (expected level of quality of the function or service)	Adequately protecting the crop, facilitating its growth, and improving its final quality by controlling weeds, by retaining the moisture content in soil and roots, and by maintaining or increasing soil temperature
"How long" (duration/lifetime of the function or service)	For one entire growing season (i.e. four to five months on average)
"Where" (location/geography of the function or service)	In the EU-28
"For whom" (beneficiary of the function or service)	An average EU-28 large-scale farmer

The reference flow of each scenario (i.e. the amount of polymer required to fulfil the functional unit) was calculated based on the thickness of the film in the specific scenario, the net area of land to be mulched, and the density of the relevant film material (Table 4.3). For thickness, an average value was considered for each mulching film alternative, calculated based on typical ranges of thickness reported in the literature or in technical specifications of specific products available on the market⁸⁸ (see Table 4.3). It was not possible to identify a thickness (nor to calculate a corresponding reference flow) ensuring that a specific, quantified agronomic performance is achieved, and that the different mulching film alternatives provide an equivalent performance. This is acknowledged as a limitation of this study, albeit also previous studies have applied a similar approach (e.g. Razza and Cerutti, 2017 and EC, 2019). Note also that, considering the wide range of thickness reported for conventional LDPE mulching film (15-20 µm) a sensitivity analysis was conducted on this parameter for such alternative (see Section 4.8.5.1). As for the extension, it was assumed that mulching 1 ha of cultivated land requires, on average,

⁸⁸ Note that the thickness assumed for conventional LDPE mulching films (i.e. 35 µm) fulfil the requirement specified by EN 13655 for thermoplastic mulching film intended to be removed from soil after use for recovery, which is the case of non-biodegradable films investigated in this case study. For this type of films, thickness shall be higher than at least 20 µm (CEN, 2018a).

0.6 ha of film (Razza and Cerutti, 2017), considering the typical line-spacing between two contiguous seedlings lines applied in horticulture, which is usually not mulched. Finally, typical average material densities were considered, as reported in the literature or in technical specifications of commercially available mulching film materials (see Table 4.3).

Table 4.3. Calculation of the reference flow for mulching film LCA scenarios.

Material	Thickness (μm)⁽¹⁾	Density (kg/m^3)⁽²⁾	Reference flow⁽³⁾ (kg/FU)
LDPE; R-LDPE	35 (15-50) ⁽⁴⁾	0.925 (0.92-0.93) ⁽⁵⁾	194
Starch-based polymer	15 (12-18) ⁽⁶⁾	1.26 (1.23-1.29) ⁽⁶⁾	113
PLA-based polymer	10 (8-12) ⁽⁷⁾	1.385 (1.37-1.40) ⁽⁷⁾	83

⁽¹⁾ Values in parenthesis represent the typical range of thickness for mulching film of the specific material (as reported in the literature or in technical specifications of specific products on the market).

⁽²⁾ Values in parenthesis represent the typical range of density of the specific mulching film material (as reported in the literature or in technical specifications of specific products on the market).

⁽³⁾ Calculated considering that, on average, a net area of 0.6 ha needs to be mulched for 1 ha of land cultivated with horticultural crops (Razza and Cerutti, 2017).

⁽⁴⁾ According to OWS (2017), LDPE mulching films for vegetable cultivation have a thickness between 15 and 50 μm . The resulting average thickness (32.5 μm) was rounded up to the most proximate commercial thickness.

⁽⁵⁾ Source: Scarascia-Mugnozza et al. (2011).

⁽⁶⁾ Source: technical specifications of one of the major starch-based mulching film materials available on the market.

⁽⁷⁾ Source: technical specifications of one of the major PLA-based mulching film materials available on the market.

4.3 System boundary

In all the analysed scenarios, the system boundary was set to cover the default life cycle stages specified in the *Plastics LCA* method for cradle-to-grave LCAs of final products⁸⁹, and the associated most relevant processes within the mulching film life cycle. The considered life cycle stages and processes are described below, and are also schematically represented in the system boundary diagrams shown in Figures 4.1 to 4.4:

- **Feedstock Supply**⁹⁰ – covering extraction, processing, transport and possible refining of crude oil and natural gas (fossil-based polymers); collection, transport and sorting of post-consumer plastic waste (recycled polymers); crop cultivation (bio-based polymers), as well as transport of these feedstock materials to downstream conversion processes (e.g. naphtha cracking, polymer recycling, wet milling of maize and other starch crops, etc.).
- **Polymer Production**⁹¹ – covering all the activities associated with the conversion or recycling of relevant feedstock materials into the specific polymer, including any transport among these activities and final transport of polymer granulate to downstream manufacturing processes.

⁸⁹ Note that, as permitted by the *Plastics LCA* method, and as described below, some of the default life cycle stages have been split into different sub-stages, and their naming was adjusted to better reflect the scope of this study.

⁹⁰ Corresponding to the default life cycle stage "Raw Material Acquisition and Pre-Processing" specified in the *Plastics LCA* method.

⁹¹ Corresponding to the default life cycle stage "Raw Material Acquisition and Pre-Processing" specified in the *Plastics LCA* method.

- **Manufacturing** – including mulching film manufacturing through blown film extrusion of polymer granulate.
- **Distribution** – including transport of mulching film from the manufacturing site to distribution centres, and from these to final users.
- **End of Life** – covering removal, transport, incineration and disposal of non-biodegradable mulching film remaining on soil after use, as well as avoided generation of conventional energy replaced by any recovered energy. Transport, treatment and disposal of soil contamination collected together with conventional mulching film during removal from the field were also included. For biodegradable mulching film, in-situ biodegradation was considered.

The default life cycle stage “Raw Material Acquisition and Pre-processing” was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating and separately quantifying the impacts of feedstock supply and those associated with downstream conversion processes into final polymer granulate. Moreover, a different nomenclature was applied to such stages compared to the default nomenclature specified in the *Plastics LCA* method, to better reflect the investigated supply chains and the scope of the study. The stage of “Raw Material Acquisition” was thus identified with that of “Feedstock Supply”, while “Pre-processing” corresponds to “Polymer Production”.

The Use stage, including mulching film application and any emissions generated during its permanence on soil (e.g. release of additives), was excluded from the system boundary, due to the lack of suitable data or proxies to model the application process, and of quantitative evidence on the possible release of specific substances during use. However, the burdens of the application process can be reasonably considered identical for all the investigated mulching film alternatives, since the burdens of agricultural operations such as soil treatment, harvesting and fertiliser/pesticide application normally depend on the area to be covered, and this reasonably applies also to mulching film application. Therefore, despite different masses of film material need to be applied in the different scenarios, they are all expected to be equally affected by the exclusion of the film application process. On the other hand, it is recognised that the overall potential lifecycle impacts associated with mulching film use quantified in this study are underestimated.

Additives used in mulching film manufacturing or polymer production were also not included in the assessment, neither in terms of associated production burdens, nor of their possible release and fate during use and at End of Life. No complete and consistent data and information were indeed available in this respect, and especially to differentiate among the different investigated material alternatives. Exclusion of additives is acknowledged as a limitation of this study, as the properties of conventional mulching film materials are reported to be normally optimised or modified by means of specific additives (e.g. plasticisers, ultraviolet stabilisers, pigments) during film manufacturing (Steinmetz et al., 2016). However, additives used for Thermoplastic Starch production (mainly plasticisers) were taken into account, as they represent a non-negligible and essential portion of the final polymer (i.e. 25%; IfBB, 2018), are required to ensure suitable material properties and processability, and relevant data are partially available. Moreover, production of additives used in starch-based polymers can account for an important share of the cradle-to-gate Climate Change impact and energy demand of such polymers, reaching 46% for polymer grades including a share of additives in the range of 30% (Broeren et al., 2017).

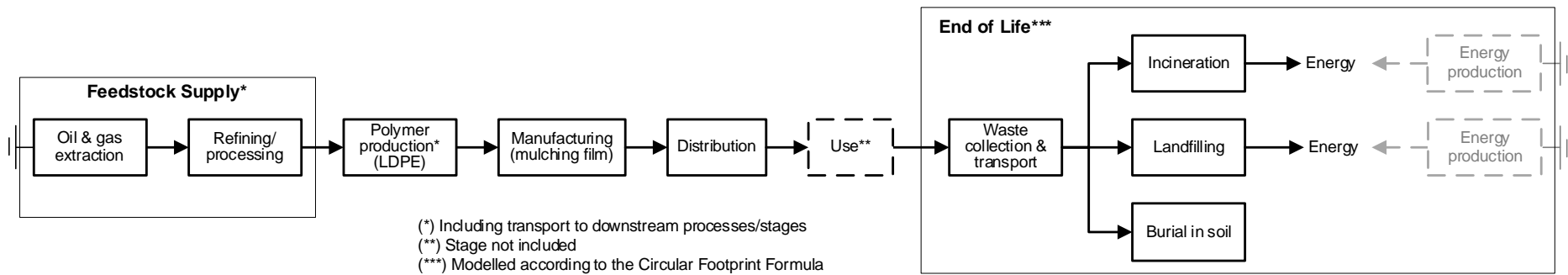


Figure 4.1. System boundary for fossil-based LDPE mulching film (Scenario 1).

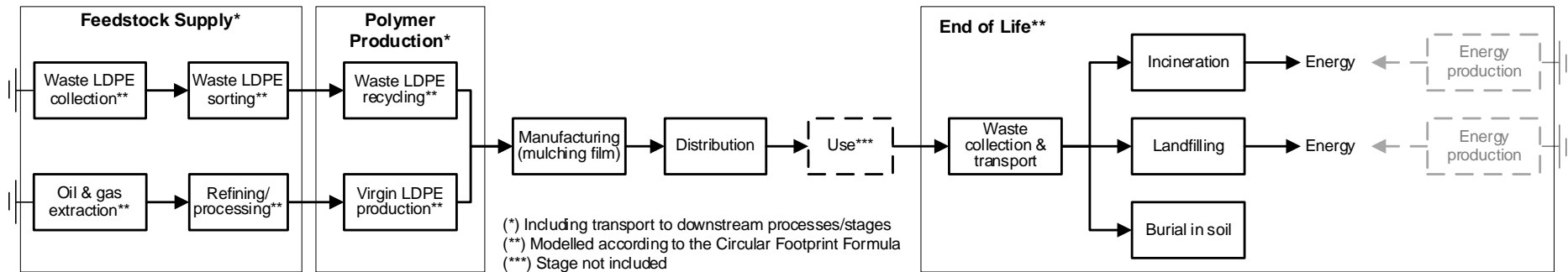


Figure 4.2. System boundary for 35% recycled LDPE mulching film (Scenario 2).

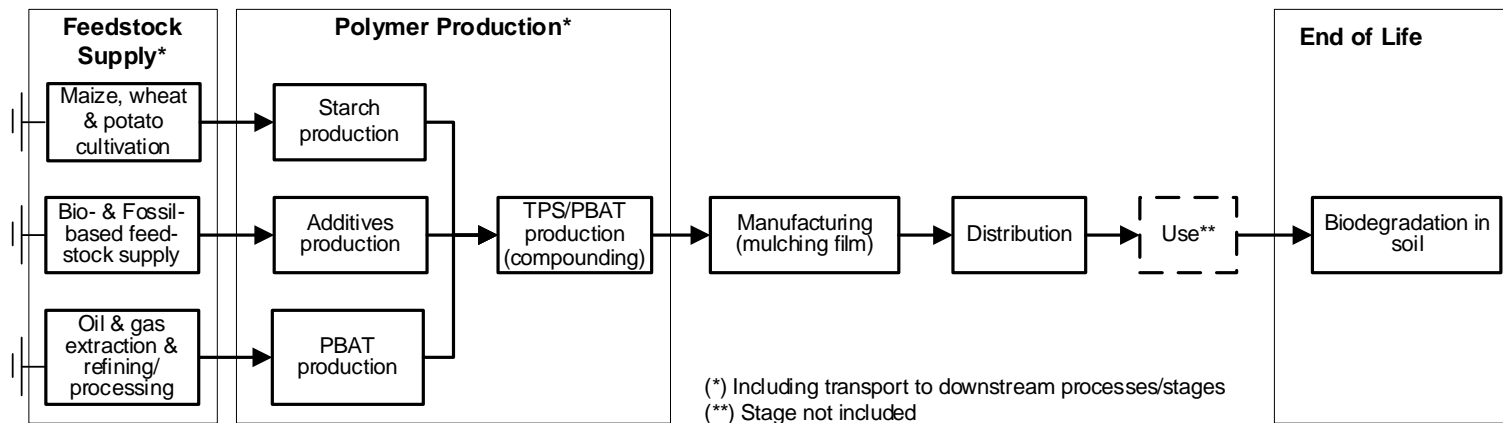


Figure 4.3. System boundary for starch-based mulching film (Scenario 3).

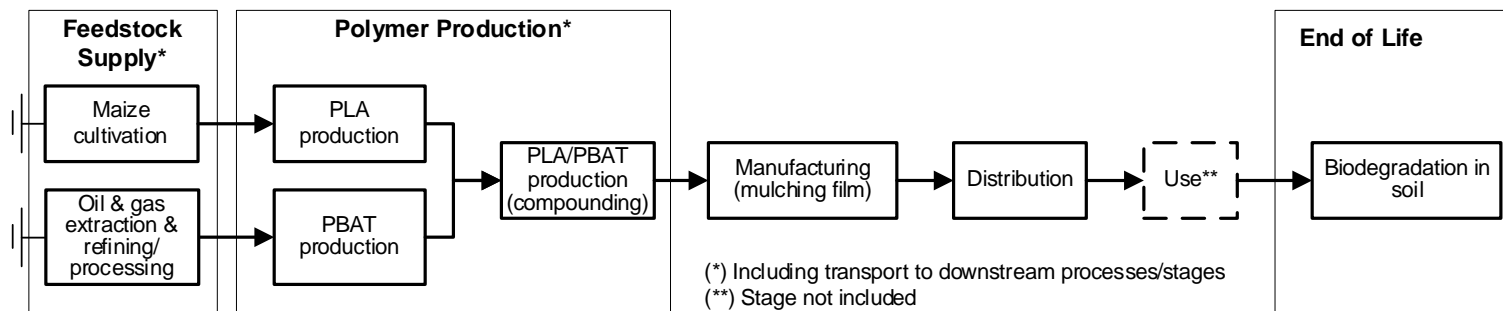


Figure 4.4. System boundary for PLA-based mulching film (Scenario 4).

4.4 Limitations and critical assumptions

The following key limitations and critical assumptions apply to the LCA scenarios investigated in this case study, and have to be properly taken into account, where relevant, in the interpretation of the respective results.

- While already commercially available and produced in industrial-scale facilities, biodegradable polymer blends used for starch-based and PLA-based mulching films partly rely on upstream conversion processes (e.g. PLA production from dextrose) that are more recent compared to processes involved in the production of fossil-based LDPE used for conventional mulching film. Such processes have thus likely experienced less optimisation and improvement, for instance in terms of conversion efficiency and process integration. Therefore, the results related to biodegradable mulching film alternatives shall be interpreted taking into account differences in the level of maturity of part of the underlying technology compared to conventional mulching films.
- For each mulching film alternative, an average thickness was estimated based on market or literature data, and used along with other relevant parameters to calculate the reference flow (i.e. the amount of film material required to fulfil the functional unit) (Section 4.2). This approach does not automatically ensure that the qualitative agronomic performances specified in the functional unit are achieved, nor that equivalent performances are provided by the different investigated mulching film alternatives. Note also that it was not possible to identify a specific quantitative agronomic performance to be included in the functional unit, due to the variety of functions typically attributed to mulching film (see the introductory section of this case study) and to their generally difficult quantification.
- The Use stage, including mulching film application and any emissions generated during its permanence on soil (e.g. release of additives), was excluded from the system boundary (Section 4.3). This was because of the lack of suitable data to model the application process, of quantitative evidence on the possible release of specific substances after application, and of complete and sufficiently specific data on additives used in mulching film (see also the next point). While all mulching film alternatives are expected to be equally affected by the exclusion of the film application process (as long as the associated burdens do not depend on the quantity of film material to be applied), the overall potential impacts quantified in this case study for the mulching film life cycle are underestimated.
- The assessment did not include the life cycle (i.e. production and any resulting emissions during use or End of Life) of additives used in polymer production or mulching film manufacturing (e.g. plasticisers, compatibilisers, UV stabilisers and pigments), due to the lack of complete and sufficiently specific and/or representative data on additive use and release. The only exception was related to additives (mainly plasticisers) used in the production of Thermoplastic Starch, which accounted for a substantial portion of the final polymer (i.e. 25%) and for which data were partially available (Section 4.3). While additives are generally reported to be mostly used in small shares, their potential impacts may be proportionally (much) higher compared to used quantities, once they are released into the environment. Impact assessment results presented in this study may thus be even significantly underestimated in specific categories, and only partially capture any differences in additive use among the investigated product alternatives.
- A combination of datasets from different sources was applied to build the life cycle inventory of the investigated mulching film scenarios, since representative EF-compliant or ILCD-EL compliant datasets were only available for part of the foreground and background processes to be modelled. This was especially the case of biodegradable mulching film alternatives (starch-based and PLA-based films), where datasets from the *ecoinvent* database were mostly used to fill data gaps (e.g. for

starch additives and PBAT production). While, to increase consistency, the originally associated background datasets were replaced with EF-compliant or ILCD-EL compliant datasets to the largest extent possible, this could not be done for all relevant process inputs and outputs. Impact assessment results calculated for biodegradable mulching films may thus be affected by discrepancies in the modelling⁹² of individual background processes compared to conventional mulching film scenarios (which entirely or mostly rely on EF-compliant or ILCD-EL compliant datasets). A careful interpretation is hence needed for such results, and any direct comparison between these two sets of scenarios should be avoided, especially for those impact categories that across the different case studies were generally found to be mostly affected by the use of datasets from different sources and related potential discrepancies (i.e. where such datasets were significantly contributing to the total impact). These categories include Ozone Depletion, Resource Use – minerals and metals, Human Toxicity (cancer and non-cancer), Ecotoxicity, and, to a lower extent, Water Use and Land Use, depending on the product scenario.

- Production of virgin, fossil-based LDPE was modelled based entirely on data reflecting EU technology and conditions, despite approximately 14% of this polymer was estimated to be imported from foreign countries (Section 4.5.2.1). This is due to the lack of specific data representing production in exporting countries such as Iran, Qatar, Korea, Saudi Arabia and Turkey. While the share of imports is moderate, virgin production impacts of LDPE may thus be underestimated if imports rely, for instance, on less efficient/outdated technologies and/or on more polluting energy sources.
- In the absence of specific data, production of secondary LDPE granulate (used as recycled content in partially recycled LDPE film) was approximated with data related to the production of a generic secondary plastic granulate out of sorted, post-consumer plastic waste (Section 4.5.2.2). While the approximation is considered reasonable (both processes are expected to rely on similar unit operations), the applied data do not refer to the real process and introduce a source of uncertainty. The use of any additives needed to achieve a suitable quality of the recycled polymer to be used as a replacement for virgin material was also excluded.
- A number of approximations and assumptions were made in the modelling of the upstream, cradle-to-gate life cycle of starch-based mulching film (as discussed in Section 4.5.2.3.1), and have to be specifically taken into account in the interpretation of the overall impact assessment results calculated for this product alternative:
 - In the absence of specific data, the total share of additives used in Thermoplastic Starch (TPS) was defined based on illustrative mass balance results from the literature, and the proportions of individual substances was based on assumptions. The resulting polymer composition may hence not be representative of TPS grades currently available on the market. Moreover, the production of some specific substances used as additives was modelled by means of more or less large approximations, introducing uncertainty.
 - PBAT production from its copolymers (1,4-Butanediol, Adipic Acid, and Terephthalic Acid) was modelled based on data related to PET polymerisation from the respective monomers, as both processes rely on esterification as the same synthesis route. While the approximation is considered reasonable, the applied data do not refer to the real process and add further uncertainty to the estimated (cradle-to-gate) results.
 - The compounding process of TPS and PBAT into the final polymer blend was modelled based on data derived from elaborations and extrapolations to the year 2018 of highly aggregated data from the environmental product declaration of a starch-based polymer dating back to 2004. The applied data may hence be no longer representative of current processes and technology.

⁹² Discrepancies may be associated with the applied modelling approach, calculation of process emissions, and/or the elementary flows used to represent such emissions.

- Most of the limitations and implications discussed in the point above for starch-based mulching film equally or similarly apply also to PLA-based film, particularly the use of PET polymerisation data to approximate PBAT production from its precursors. Moreover, the final compounding step was modelled based on data related to compounding of starch-based polymers (i.e. the same data used for starch-based mulching film), which may not adequately represent the real process.
- Regarding the End of Life of conventional, non-biodegradable mulching films (i.e. virgin and partially recycled LDPE films), the following approximations and relevant assumptions have been performed, affecting the potential impacts estimated for these product alternatives and their End of Life stage:
 - The assumed collection rate of mulching film from the field after use (90%) was defined based on literature data referring to LDPE films with a thickness that is 10 μm lower than the one considered in this study for conventional mulching films (i.e. 35 μm ; Section 4.5.5.1). The assumption is thus potentially conservative (the collection rate generally increases with thickness), although it could be considered a reasonable balance between the actual collection rate for a 35 μm film (unknown) and the overall collection rate estimated for mulching film in general at the EU level, regardless of thickness (i.e. 80%).
 - The contamination rate of collected mulching film by soil and vegetable residues was estimated as a function of thickness based on elaborations (linear regression) on data pairs defined considering that the contamination rate increases when thickness decreases. However, the values of contamination rate considered in the calculations were not originally associated to specific thicknesses (Section 4.5.5.1). The obtained estimate (i.e. a contamination rate of 60%) is thus affected by uncertainty, although it is in line with the overall average contamination rate that can be estimated, based on other available data, for mulching film in general at the EU level (i.e. 67%).
 - The burdens of the collection process of mulching film from land after use were approximated with data related to straw removal, in the absence of specific data for mulching film collection (Section 4.5.5.2). While the approximation is considered reasonable, it represents a source of uncertainty.
 - To model landfilling of conventional LDPE mulching films, a dataset related to the disposal of a generic, average, conventional plastic material was applied as a proxy, due to the lack of a material-specific EF-compliant dataset for landfilling of LDPE (Section 4.5.5.5). The approximation was considered acceptable, but it is reported for completeness, as prescribed in the *Plastics LCA* method (Section 4.4.10.11).
 - Incineration and landfilling of soil collected along with used mulching film during removal from the field were modelled based on (EF-compliant) datasets approximating incineration and landfilling of the main soil material components (including minerals and organic matter), in the absence of any material-specific datasets for soil treatment or disposal. Particularly, incineration and landfilling of the mineral part were approximated with treatment or disposal of inert material, while biodegradable waste treatment/disposal was considered as a proxy for incineration and landfilling of soil organic matter (Sections 4.5.5.4 and 4.5.5.5). These approximations introduce further uncertainty in the (End of Life) results.
 - Burial in soil of uncollected conventional mulching film residues was modelled as mere release of macro-plastic to soil (Section 4.5.5.6). This was due to incomplete understanding and knowledge of the fate of plastic products or parts once released in soil (and in the environment in general), and to the lack of data on the use of additives during polymer production and film manufacturing (Section 4.3). Additional relevant burdens might have thus been neglected in the modelling, such as those related to any release of micro-plastics from possible further

fragmentation of film residues buried in soil, and to any emission of additives, metals or possible degradation products.

- The potential impacts on ecosystems, particularly soil and its quality, and on human health associated with the release as macro-plastic of uncollected conventional mulching film residues could not be quantified, due to the current absence of a suitable impact assessment method (as more generally discussed in Sections 2.4 and 2.6 of this report). However, this gap may be filled in the future, as long as better knowledge is gained on the actual fate, exposure and effects of macro-plastics released in soil and into the environment in general.
- In-situ biodegradation of biodegradable mulching film was modelled assuming that ultimate biodegradation is achieved for the film material, i.e. that the latter is ultimately converted into CO₂, water, new soil biomass, and mineral salts of any other elements included in the product composition (none in this case study). No residual (micro)-plastic particles were assumed to remain in soil after biodegradation, while any plastic fragments and intermediate degradation products generated during biodegradation were assumed not to be transferred to other environmental compartments than soil, thus not contributing to any emission into the environment (Section 4.5.5.7). However, relevant emissions might have been omitted from the modelling (e.g. any non-biodegradable additives, any metals, intermediate biodegradation products and plastic fragments/particles), due to incomplete composition data (which exclude additives used during polymer compounding and film manufacturing), and to only partial understanding and knowledge of fragmentation and biodegradation pathways of plastics in soil.
- A 90% mineralisation rate was assumed when modelling biodegradation of biodegradable mulching film in soil, according to the minimum requirement from the standard EN 17033 (CEN, 2018b), which has to be achieved over a maximum period of 2 years when testing the film material according to the specified standard method (Section 4.5.5.7). While a higher rate may be achieved in reality over the 100-year timeframe considered for modelling, no adaptations were made, in the absence of consistent and sufficiently representative evidence, and following a conservative approach.
- The method and indicators applied to quantify the total loss and release of macro-plastics to the environment (i.e. the *PLP method*) focuses on the initial loss from the technosphere, and on the subsequent immediate release to the environment after any short-term redistribution of released items among different environmental compartments. However, the effects of biodegradation, and of any other environmental mechanism further affecting the fate of released items, are not taken into account, due to incomplete knowledge of fragmentation and (bio-)degradation pathways of (biodegradable) plastic products in soil and in the environment in general (Section 4.5.5.8). The values of release calculated in this study for biodegradable mulching films may thus be even significantly overestimated compared to the ultimate release after biodegradation has occurred (which is not quantified), albeit the results clearly differentiated the estimated release of biodegradable plastic material from the release of conventional, non-biodegradable plastic.

4.5 Life Cycle Inventory

This section describes the main details of the Life Cycle Inventory of the analysed scenarios, including the related assumptions and data sources. The description is separately reported for each life cycle stage in the following sub-sections (4.5.1 – 4.5.6).

4.5.1 Feedstock Supply Stage

4.5.1.1 Fossil-based polymers

For virgin, fossil-based polymers and copolymers (i.e. LDPE and PBAT), the stage of Feedstock Supply includes the activities of crude-oil and natural gas exploration, drilling, extraction, processing and transport to downstream users, as well as naphtha production in crude oil refineries, and its transport to subsequent conversion processes (i.e. naphtha cracking).

For fossil-based LDPE, the modelling of these activities was carried out as described in Section 3.5.1.1 of the Beverage Bottles case study, where the reader is referred to for further details on the applied data in terms of source, reference technology, geography and time period, covered sub-processes and related environmental burdens, main modelling choices (e.g. allocation), other relevant aspects and modelling details, as well as of any adjustments performed in the implementation of the applied datasets in the overall product inventories.

For PBAT used as a copolymer in starch-based and PLA-based polymers, the stage of Feedstock Supply could not be separately modelled from downstream conversion and polymerisation processes, due to the absence of disaggregated datasets covering the burdens associated with the production of most PBAT co-monomers (disaggregation would have only been possible for PTA). The overall, cradle-to-gate inventory modelling of PBAT production and supply of the associated feedstock is hence entirely described in Section 4.5.2.3, focusing on Polymer Production.

4.5.1.2 Recycled LDPE

For recycled LDPE, Feedstock Supply consists of collection of post-consumer LDPE waste, and its subsequent transport and sorting in specific facilities for recycling. LDPE waste from different sectors contributing to municipal collection (e.g. household and commercial packaging waste) was assumed to be used as a feedstock, while agricultural film waste is reported to be currently not recycled into new film material (Eunomia, 2021). The collection, transport and sorting processes were modelled as described in Sections 3.5.5.2 and 3.5.5.3 for the Beverage Bottles case study, based on data referring to collection and transport of separately collected plastic waste at the municipal level, as well as to sorting of mixed plastic waste in dedicated facilities. However, a sorting efficiency of 73% was considered for collected LDPE waste, according to Antonopoulos et al. (2021).

These activities and processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the approach prescribed in the *Plastics LCA* method to handle recycling situations. The formula was applied considering the default material-specific value of the A factor reported in Annex C of the *Plastics LCA* method for recycled PE used in unspecified applications (i.e. 0.5), in the absence of an application-specific value. Therefore, only 50% of the burdens from LDPE waste collection, transport, sorting (and subsequent recycling; see Section 4.5.2.2) were assigned to the recycled material content in mulching film, the rest being assigned to the system providing waste material for recycling. However, the recycled material content was assigned an equal share $((1-A) \times Q_{s_{in}}/Q_p)$ of the burdens associated with the supply of the virgin fossil-based feedstock used in the production of the replaced virgin polymer. Such burdens were modelled as described in Section 4.5.1.1 for virgin, fossil-based LDPE. Further details and considerations on the implementation of the CFF are provided in Section 4.5.2.2, addressing the modelling of the Polymer Production Stage for recycled LDPE.

4.5.1.3 Bio-based polymers

For bio-based polymers (i.e. Thermoplastic Starch and PLA used as blending co-polymers), the stage of Feedstock Supply includes cultivation of the relevant starch crop(s) and their subsequent transport to further processing in the same country.

Cultivation of US Maize as a feedstock for PLA was not modelled separately, being already covered in the fully aggregated, cradle-to-gate dataset applied to model PLA production⁹³ (see Section 4.5.2.3.2). This refers to the year 2014, and is based on the life cycle inventory of Ingeo™ polylactides produced by NatureWorks in its facility located in Nebraska (Vink and Davies, 2015). According to the information provided in the dataset documentation, the modelled maize production inventory represents cultivation in the twenty-six counties on the borderline between Nebraska and Iowa (which provide the feedstock used for PLA production), and is based on average data collected from farmers operating in such region. Covered activities include production of all the relevant agricultural inputs, such as seeds (accounting for the associated energy demand), fertilisers, lime stone, herbicides and insecticides, electricity and fuels (natural gas, diesel, propane and gasoline) used on farm, as well as supply of irrigation water. Emissions of dinitrogen oxide, nitrogen oxides, nitrates and phosphates (due to fertiliser application) are also taken into account. Other emissions (e.g. from pesticides) are not explicitly mentioned, although a number of pesticide emissions (likely associated with maize cultivation) are reported in the overall aggregated inventory. Production of relevant farm equipment (tractors and harvest combines) was investigated during the development of the inventory, but its contribution was found to be negligible (and hence it was likely excluded). Uptake of atmospheric CO₂ during maize growth through photosynthesis is included, and quantified based on the biogenic carbon content of PLA at 1.83 kg CO₂/kg_{PLA} (Vink and Davies, 2015), which is in line with the total inventoried amount of 1.88 kg CO₂/kg_{PLA} (accounting also for other potential contributions from the covered process chain). CO₂ emissions from direct land use change are also inventoried, although the quantification method is not reported and the contribution of the maize cultivation process alone is not specified. The total aggregated emission is equal to 8.37x10⁻⁵ kg CO₂/kg_{PLA} (i.e. approximately 5.32x10⁻⁵ kg CO₂/kg_{maize})⁹⁴, which is one order of magnitude lower than the emission inventoried in the ILCD-EL compliant dataset for US maize available in the GaBi database (1.1x10⁻⁴ kg CO₂/kg_{maize}). Road transport of dried maize grains to downstream processing (wet milling) takes place along a relatively short distance, which is not specified in the dataset nor in Vink and Davies (2015).

Production of the European starch crops (maize, wheat, potatoes) used as a feedstock for Thermoplastic Starch, was modelled through available EF-compliant datasets⁹⁵ referring to the year 2016. These are based on the approach used to develop agricultural inventories available in the Agri-footprint database (Agri-footprint methodology; Blonk Consultants, 2015a,b), relying on 5-year average yield data from FAOSTAT (2010-2014), and on country-, crop-, and process-specific data for the other relevant parameters, as appropriate. Additional details on the specific activities, inputs and emissions covered in the datasets, on their modelling or quantification, and on other relevant applied methodological choices (e.g. allocation of cultivation burdens between any coproducts) are reported in Section 3.5.1.3 of the Beverage Bottles case study. While such description refers only to maize and wheat cultivation datasets, it equally applies to potato cultivation, with GHG (CO₂) emissions from direct Land Use Change (dLUC) being quantified for this crop at 0.000367 kg CO₂/kg_{potatoes}. All crops were assumed to be transported to downstream processing along an overall distance of 100 km, covered by large lorries (> 32 t, fuelled with the EU diesel mix).

⁹³ The following dataset was applied: *[US] Ingeo Polylactide (PLA) biopolymer production; corn production, dextrose wet milling process, fermentation to lactic acid, polymerisation | single route, at plant | 1.210 - 1.430 kg/m³.*

⁹⁴ Calculated considering a specific maize consumption for PLA production equal to 1.572 kg maize/kg PLA, as reported in the underlying process inventory by Vink and Davies (2015).

⁹⁵ The following datasets were applied:

- Maize: *[EU+28] Maize (corn grain) production; technology mix, production mix | at farm;*
- Wheat: *[EU+28] Wheat grain; technology mix, production mix | at farm;*
- Potatoes: *[EU+28] Starch potato; technology mix, production mix | at farm.*

4.5.2 Polymer Production Stage

The Polymer Production stage covers the activities of feedstock processing into any relevant intermediate(s) and monomer(s), the polymerisation or recycling process, compounding of polymer blends, as well as any transport among these activities and final transport of polymer granulate to the mulching film manufacturing site. The following subsections (4.5.2.1 – 4.5.2.4) describe how these activities were modelled in this case study, distinguishing between fossil-based, recycled and bio-based polymers.

4.5.2.1 Fossil-based LDPE

For conventional, fossil-based LDPE, the whole process chain from feedstock processing to polymerisation, through the production of intermediates and monomers, was modelled by means of a partially aggregated, cradle-to-gate, ILCD-EL compliant dataset provided by Thinkstep⁹⁶. The dataset disaggregates upstream feedstock inputs (naphtha and natural gas), reflects the main technologies adopted in EU-28, and refers to the year 2018. It is mainly based on industry data from internationally adopted production processes, integrated, where needed, with literature data from several sources. The number of industry data sources considered for individual process steps is not specified. Disaggregated upstream inputs include a combination of crude oil-derived naphtha and natural gas, with crude oil accounting for 75% of the total feedstock input, while the remaining 25% is covered by natural gas. All conversion processes are assumed to take place in Europe, so that the dataset not only reflects the main technology applied in the region, but also EU-average background conditions in terms of e.g. energy generation, material supply and transport. However, approximately 14% of LDPE used in the EU was estimated to be imported, based on average annual import shares calculated from *Prodcom* data for the years 2016-2018 (Eurostat, 2019d). The use of data reflecting European technology and conditions also for imports (due to the absence of data for polymer production in exporting countries) thus represents a limitation of this study, although the estimated import share is modest.

The main conversion process involved in the modelled supply chain is steam cracking of naphtha and natural gas, delivering the monomer ethylene, along with propylene, butadiene, and other co-products such as pyrolysis gas (a mixture of benzene, toluene and xylenes), refinery gas and hydrogen. Allocation among the different co-products is based on energy, considering the net calorific value of each co-product. In the final polymerisation step no allocation is performed, being LDPE the only output from the process.

4.5.2.2 Recycled LDPE

The production of recycled LDPE granulate out of sorted, post-consumer LDPE waste was approximated with the aggregated, EF-compliant dataset "*[EU-28] Plastic granulate secondary (low metal contamination); from post-consumer plastic waste, via grinding, metal separation, washing, pelletization; production mix, at plant; plastic waste with low metal fraction*", referring to the year 2018. As stated in the name, the dataset represents the burdens associated with the production of a generic secondary plastic granulate out of sorted, post-consumer plastic waste, and was applied in this study in the absence of more specific data on LDPE waste recycling. The modelled process is likely more representative of the recycling of rigid or semi-rigid plastic products (e.g. trays or bottles). However, it was considered a suitable approximation also for the recycling of flexible products (which are expected to represent most of LDPE waste sent for recycling), since both processes can be reasonably assumed to rely on similar unit operations (i.e. grinding/shredding, washing/flotation and granulation). The dataset is developed based on literature data for each of the underlying unit operations, and accounts for an overall recycling efficiency equal to 84% (on the sorted input material).

⁹⁶ The following dataset was applied: *[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.*

Process waste and scrap are sent to incineration, consistently with the typical fate of residues from plastic recycling, which due to their high calorific value are normally routed to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

According to the approach prescribed in the *Plastics LCA* method to model recycling situations (Circular Footprint Formula), only a share of the burdens of the recycling process were allocated to the recycled content in LDPE mulching film, based on the values of the A and $Q_{S_{in}}/Q_p$ factors used in the formula. The default, material-specific value reported in Annex C of the *Plastics LCA* method for LDPE used in unspecified applications was selected (i.e. 0.5), in the absence of an application-specific value. Therefore, only 50% of the burdens of the recycling process (per functional unit) were allocated to the recycled content in mulching film. However, the recycled content carried a share ($1-A = 0.5$) of the primary production burdens of the replaced virgin material (i.e. the same burdens that would have been credited to End of Life recycling in the previous product life cycle providing the recycled material). Assuming a value of the $Q_{S_{in}}/Q_p$ factor equal to 1 (since the quality of the recycled material needs to be necessarily suitable for mulching film manufacturing, and hence similar to that of the replaced virgin material), the total allocated share of virgin LDPE production burdens was again equal to 50% (i.e. $(1-A) \times Q_{S_{in}}/Q_p = (1-0.5) \times 1 = 0.5$). Note, however, that the modelled recycling process only approximates the real process, and excludes the use of any additives used to achieve comparable technical properties. Virgin polymer production burdens were modelled as described in Section 4.5.2.1 for conventional, fossil-based LDPE, and in Section 4.5.1.1 for the respective Feedstock Supply.

4.5.2.3 Bio-based polymers

4.5.2.3.1 Starch (TPS)/PBAT blend

Thermoplastic starch (TPS) used as a copolymer in starch-based mulching film consists of native starch and additives (mainly plasticisers) required to allow its processability in conventional plastic conversion processes (e.g. extrusion or injection moulding; Broeren et al., 2017). According to mass balance data reported in IfBB (2018), the share of additives was assumed to be 25% of the total polymer, including Glycerol, Sorbitol (both used as plasticisers), as well as Glycidyl methacrylate⁹⁷. Moreover, in the absence of specific data on the used amount of each additive, the total share of additives was equally split among the three mentioned substances (i.e. 33% each).

Starch production via wet milling of the different starch crops (maize, wheat and potatoes) was modelled based on life cycle inventory data for relevant process steps available in different datasets from the Agri-footprint database (v 4.0), referring to the year 2014⁹⁸. These data were then combined with EF-compliant background datasets to model the burdens of individual inputs and outputs from the specific process, reflecting EU conditions. The allocation of process burdens to the different co-products (e.g. from maize wet milling) is based on the respective economic value (based on five-year average prices), consistently with the fully vertically aggregated EF-compliant datasets available for starch production via wet milling (which are developed by the same data provider). Maize starch production data are derived from the literature, while for wheat starch a combination of data from literature and industry/industry experts is used. Data for potato starch are retrieved from an industry expert only.

Regarding additives, Glycerol production was modelled through a fully aggregated, cradle-to-gate EF-compliant dataset representing production from vegetable oils under

⁹⁷ Based on information provided in the documentation of the aggregated, cradle-to-gate, Thermoplastic Starch production dataset available in the GaBi database.

⁹⁸ For maize starch production, the following set of datasets was considered: (i) *Maize, steeped, from wet milling (receiving and steeping), at plant*; (ii) *Maize degermed, from wet milling (degermination), at plant*; (iii) *Maize starch and gluten slurry, from wet milling (grinding and screening), at plant*; (iv) *Maize starch, wet, from wet milling (gluten recovery), at plant*; and (v) *Maize starch, from wet milling (starch drying), at plant*. For wheat starch, the considered dataset is "*Wheat starch, from wet milling, at plant*", while that considered for potato starch is "*Potato starch dried, from wet milling, at plant*".

global conditions, and referring to the year 2017⁹⁹. Conversely, in the absence of specific data, sorbitol production was approximated with data related to glucose production from starch, being sorbitol synthesised through the reduction of glucose. The inventory of Glucose production was based on the *ecoinvent* dataset "[RER] Glucose production", which represents pure Glucose obtained via enzymatic hydrolysis of dried (maize) starch, and relies on literature data for the year 2014. However, in the implementation of the dataset in the lifecycle model, the default maize starch input was replaced with the same EU-average mix of starches from relevant European starch crops as considered for the main polymer (including 47% maize starch, 40% wheat starch, and 13% potato starch; see Section 4.1). Moreover, background datasets related to energy generation (i.e. electricity, thermal energy and steam) were replaced with background EF-compliant datasets, or datasets developed based on EF-compliant datasets (i.e. for steam production). Finally, the burdens associated with the life cycle of plant infrastructures¹⁰⁰ were removed, to improve reliability of LCIA results in specific impact categories (including Resource Use – minerals and metals and Ozone Depletion). For Glycidyl Methacrylate, an average inventory representing production of an unspecified organic chemical under global conditions was applied (i.e. the *ecoinvent* dataset [GLO] chemical production, organic)¹⁰¹, in the absence of representative data for the specific substance. The dataset represents an unweighted average of the first 20 most used organic chemicals included in the *ecoinvent* database, defined based on data for the year 2000 extrapolated to the year of calculation (2018). Due to the uncertainties associated with the assumed shares of additives (see above) and with the modelling approximations performed for some of them, the potential impacts of starch-based mulching film were also recalculated by excluding the contribution of additives, and assuming that TPS is entirely made of starch (see the sensitivity analysis presented in Section 4.8.5.3).

Compounding of starch and additives to produce TPS was assumed to take place in the same facility where the final starch-based polymer is produced by blending TPS and PBAT granulate. Such preliminary compounding activity was not modelled separately, since the respective burdens were considered to be already accounted in the dataset used to model the final compounding step (which is described below).

PBAT is a copolymer of 1,4-Butanediol, Adipic Acid, and Purified Terephthalic Acid (PTA). The respective production (polymerisation) process is similar to the synthesis of PET from Ethylene Glycol and PTA via esterification (Schrijvers et al., 2014). Therefore, this activity was modelled based on inventory data related to PET polymerisation from the most recent PlasticsEurope ecoprofile (CPME, 2017), as implemented in the *ecoinvent* 3.6 database. The two PET precursors used in the process were replaced with the three PBAT co-monomers, considering the specific consumption reported in Schrijvers et al. (2014). This is based on a 90% synthesis efficiency, and is equal to 0.41 kg for 1,4-Butanediol, 0.37 kg for Adipic Acid, and 0.33 kg for PTA (with all values expressed per kg of PBAT). Cradle-to-gate production inventories for these co-monomers were derived from aggregated ILCD-EL compliant datasets of the GaBi database (1,4-Butanediol and PTA) or from the pool of EF-compliant datasets (Adipic Acid). The datasets refer to the years 2017-2018 (depending on the product), and in the case of 1,4-Butanediol and Adipic Acid refer to German background conditions, in the absence of representative data for EU conditions. Due to the aggregated nature of the datasets, the contribution of Feedstock Supply to the total LCIA results could not be separately quantified for PBAT. Background energy inputs to the overall PBAT polymerisation dataset were combined with relevant

⁹⁹ The applied dataset is the following: "[GLO] Glycerine, from vegetable oil production, technology mix | production mix, at plant | 100% active substance".

¹⁰⁰ As modelled in the process "[RER] chemical factory construction, organics".

¹⁰¹ To improve reliability of LCIA results in the Ozone Depletion impact category, a number of chemicals were removed from the original dataset, i.e. Acetic Acid, Methanol, Urea, Vinyl Acetate, Ethylene Dichloride, and Formaldehyde. The respective shares were then equally subdivided among the remaining chemicals. Moreover, default background *ecoinvent* datasets modelling supply of some other chemicals were replaced with EF-compliant datasets for the same substances. These include Benzene ([GLO] market for benzene), Ethylene Glycol ([GLO: market for ethylene glycol]), Phenol ([GLO] market for phenol), Styrene ([GLO] market for styrene), and Formaldehyde ([RER] market for formaldehyde).

EF-compliant datasets, while for material inputs and waste outputs the original *ecoinvent* datasets were kept¹⁰². The input process modelling the life cycle of plant infrastructures (*[RER] chemical factory construction, organics*) was removed, to improve reliability of LCIA results in the Ozone Depletion and Resource Use – minerals and metals impact categories. Transport of the different PBAT co-monomers to the polymerisation plant was also included in the dataset, considering the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers to factories/users within Europe. The scenario includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

The inventory of the final compounding step of TPS and PBAT (including blending of raw materials, cutting, drying and cooling) was developed based on the *ecoinvent* dataset "*[RER] Polyester-complexed starch biopolymer production*", which was assumed to also cover the burdens of TPS production out of starch and related additives (as discussed above). The dataset relies on calculations and extrapolations from highly aggregated background data from the environmental product declaration of a starch-based polymer (MaterBi) for the year 2004 (data have been extrapolated to 2018). Compared to the original dataset, inputs of naphtha and natural gas were removed, being associated with the supply of raw materials and process energy required for the production of the unspecified fossil-based co-polyester originally considered in the inventory, which is here represented by PBAT and was modelled separately (as described above). Similarly, the original input of maize starch was replaced with the EU-average mix of starches from relevant European starch crops considered in this study (including 47% maize starch, 40% wheat starch, and 13% potato starch; see Section 4.1). Inventoried quantities of starch, PBAT, and starch additives were set or adjusted to reflect the respective shares in the final polymer blend investigated in this study (i.e. 40% starch -of which 25% of additives- and 60% PBAT). A 100% conversion efficiency was assumed, as material losses during compounding were reported to be negligible (Broeren et al., 2017). The remaining energy inputs (electricity and heat) and waste flows reported in the original inventory were combined with EF-compliant background datasets reflecting EU conditions for the modelling of the respective burdens. Finally, the life cycle of plant infrastructures (as modelled in the input process "*[RER] chemical factory construction, organics*") was removed, to improve reliability of LCIA results in the Ozone Depletion and Resource Use – minerals and metals impact categories.

Transport of copolymers (starch and PBAT granulate) and of starch additives to the final compounding facility was also accounted in the Polymer Production stage. The modelling was based on the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers to factories/users within Europe, which assumes covering 130 km by lorry (> 32 t, Euro 4), 240 km by train (technology mix), and 270 km by barge.

4.5.2.3.2 PLA/PBAT blend

PLA production from US maize was modelled based on a vertically aggregated, cradle-to-gate dataset available in the GaBi database¹⁰³, and developed based on the life cycle inventory of Ingeo™ polylactides produced by NatureWorks (the largest US supplier) in its facility located in Nebraska (Vink and Davies, 2015). The inventory refers to the year 2014, reflects US background conditions, and includes the processes of dextrose production through maize wet milling followed by enzymatic hydrolysis of starch, lactic acid production via starch fermentation, conversion of lactic acid into lactide monomer,

¹⁰² Except for inputs of Nitrogen (*[RER] market for nitrogen, liquid*) and compressed air (*[GLO] market for compressed air, 600 kPa gauge*), which were modelled through EF-compliant datasets to improve reliability of LCIA results for the Ozone Depletion impact category. For the same reason, the dataset related to the supply of unspecified organic chemicals (*[GLO] market for chemical, organic*), was replaced with the underlying organic chemical production dataset (*[GLO] chemical production, organic*) adjusted as described above for TPS additives.

¹⁰³ The following dataset was applied: *[US] Ingeo Polylactide (PLA) biopolymer production; corn production, dextrose wet milling process, fermentation to lactic acid, polymerisation | single route, at plant | 1.210 - 1.430 kg/m3*.

and final polymerisation of lactide into PLA. Data for the wet milling step are representative of the plant supplying dextrose to the subsequent conversion processes into PLA carried out by NatureWorks, which are also modelled based on actual production data developed by the company. The inputs and outputs of the maize wet milling process are allocated to the different co-products (starch, gluten feed, gluten meal, heavy steep water and germ) based on the dry mass of each final and/or intermediate product, after subdividing the overall wet milling process into 11 different sub-processes. The lactic acid production process also generates small quantities of gypsum as a coproduct, which is used as soil conditioner in place of mined gypsum. This co-product is handled through direct substitution, by crediting the process with the burdens from avoided gypsum mining. No additional co-products are obtained in the other downstream conversion processes, and no allocation nor substitution is required.

Modelling of PLA transport to Europe for compounding was made according to the default scenario specified in the *Plastics LCA* method for import of goods from suppliers located outside Europe. A transoceanic ship transport was assumed as the main route, considering a case-specific distance of 6,000 km (for the route New York – Rotterdam)¹⁰⁴. In addition, a default distance of 1,000 km was assumed to be covered by lorry (total weight >32 t; Euro 4) for road transport of the polymer granulate to and from the harbour in the US or Europe.

As for the starch-blend, PBAT production in Europe was modelled based on inventory data related to PET polymerisation from its precursors (Ethylene Glycol and PTA), being both processes based on a similar synthesis route (i.e. esterification; Schrijvers et al., 2014). Polymerisation data were derived from the most recent *PlasticsEurope* ecoprofile (CPME, 2017), as implemented in the *ecoinvent* 3.6 database, while production of the three PBAT co-monomers used in the synthesis process was modelled based on ILCD-EL compliant datasets from the GaBi database (1,4-Butanediol and Terephthalic Acid) or EF-compliant datasets (Adipic Acid), referring to the years 2017-2018. The remaining modelling was carried out as already described in Section 4.5.2.3.1. PBAT granulate was then assumed to be transported to the compounding facility based on the default transport scenario specified in the *Plastics LCA* method for transport of goods from suppliers to factories/users within Europe. This includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

The final compounding step of PLA and PBAT (including blending of raw materials, cutting, drying and cooling) was approximated with relevant process data related to the production of starch-based polymer blends, reported in the *ecoinvent* dataset “[RER] Polyester-complexed starch biopolymer production” (already described in Section 4.5.2.3.1). Compared to the original inventory, only energy inputs (electricity and heat), waste outputs and direct emissions to the environment were maintained. These were combined, where relevant, with EF-compliant background datasets reflecting EU conditions, to model the associated burdens. Moreover, inputs related to the supply of the original copolymers (i.e. starch, naphtha and natural gas) were replaced with relevant copolymer inputs (i.e. PLA and PBAT), modelled as described above and reflecting the assumed shares in the final polymer blend (i.e. 45% PLA and 55% PBAT; see Section 4.1). A 100% conversion efficiency of these material inputs was assumed for the process, considering that material losses during production of starch-based polymer blends were reported to be negligible (Broeren et al., 2017). Finally, the life cycle of plant infrastructures (as modelled in the input process [RER] *chemical factory construction, organics*) was removed, to improve reliability of LCIA results in the Ozone Depletion and Resource Use – minerals and metals impact category.

¹⁰⁴ Defined based on the calculation tool provided by SeaRates.com, and available at <https://www.searates.com/services/distances-time>

4.5.2.4 Transport of polymer granulate to the product manufacturing site

Modelling of transport of polymer granulate from the polymerisation, compounding or recycling plant (inside or outside the EU), to the mulching film manufacturing site in Europe, was based on the default transport scenarios (distances and vehicle types) specified in the *Plastics LCA* method for the route "supplier-to-factory". In the case of polymers produced in Europe (i.e. all the final polymers and copolymers investigated in this case study, except for the imported share of fossil-based LDPE), the following routes were thus considered:

1. 130 km by lorry (total weight >32 t; Euro 4);
2. 240 km by train (average freight); and
3. 270 km by ship (barge).

For the imported share of fossil-based LDPE (i.e. 14%), a transoceanic ship transport was considered as the main transport route. The corresponding overall sea distance was determined as weighted average of the harbour-to-harbour distances between each exporting country and the EU (defined based on the calculation tool available on SeaRates.com)¹⁰⁵. Countries contributing to at least 90% of the overall imported quantity were considered in the calculation, leading to an overall distance equal to 6,528 km for fossil-based LDPE (see Table 4.4). Oceanic ship transport was complemented with road transport to the harbour in the single exporting countries, and from the harbour to the manufacturing site in the EU. Road transport was made by lorry (total weight >32 t; Euro 4) along an overall default distance of 1000 km.

LCIs for transport through all types of considered vehicles were available as EF-compliant datasets, which were used in the modelling.

4.5.3 Manufacturing Stage

Regardless of the material or feedstock used, mulching film was assumed to be manufactured via blown film extrusion, which is the most common process for large-scale manufacturing of plastic films (Crawford and Martin, 2019). In blown film extrusion, melted plastic granules are passed through an annular die to form a thin tube, which during formation is inflated with air from inside to prevent collapsing. The film "bubble" is then cooled down, collapsed (through collapsing guides and nip rolls) and finally rolled up on storage drums or, for instance, gusseted and cut to length for bags production (Crawford and Martin, 2019).

The inventory of the film extrusion process was derived from the aggregated, EF-compliant dataset "[EU-28+EFTA] Film Extrusion (blowing); plastic extrusion | production mix, at plant | for PP, PE, PVC, PET and PS". The dataset is based on primary industry data from internationally adopted production processes, completed, where necessary, by secondary data, and refers to the year 2012. The underlying process inventory accounts for a 99% conversion efficiency, with the fate of process losses to be modelled based on the case-specific situation.

For LDPE and R-LDPE mulching film, the small amount of process losses (e.g. flawed film) was assumed to be entirely recycled in external facilities via re-granulation into new polymer pellets, ultimately replacing virgin granules of the same material. Hence, recycled LDPE granules were assumed to replace virgin, fossil-based LDPE granules, being the estimated share of bio-based LDPE on the market very low (i.e. 0.2%)¹⁰⁶. The recycling process was modelled based on the same data applied to model recycled LDPE production out of sorted, post-consumer LDPE waste (see Section 4.5.2.2), in the absence of more specific data for recycling of pre-consumer, industrial scraps. Avoided

¹⁰⁵ Available at: <https://www.searates.com/services/distances-time/>

¹⁰⁶ The share of bio-based LDPE in the EU market was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastics, 2019; 200 kt) and the production capacity of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

virgin polymer production and the related feedstock supply were instead modelled as described in Sections 4.5.2.1 and 4.5.1.1, respectively. The Circular Footprint Formula (CFF) was applied to calculate the actual quantities of both recycling and avoided virgin production processes to be modelled in the overall product inventory, considering the material-specific default value of the A factor reported in Annex C of the *Plastics LCA* method (i.e. 0.5). The quality ratio $Q_{s_{out}}/Q_p$ was set equal to 1, assuming that recycled material obtained from recycling of clean pre-consumer scrap has a quality similar to the replaced virgin material.

For starch-based and PLA-based mulching film, process losses were assumed to be incinerated, as recycling of these polymer blends is currently not established. The incineration process was modelled as described in Section 4.5.5.4 on End of Life modelling.

Table 4.4. Calculation of the overall average sea distance for imports of virgin fossil-based LDPE to Europe.

Exporting country	Import (¹) (%)	Import (% cum.)	Distance (²) (km)	Weighted distance (km)
NORWAY (incl.SJ excl.1995,1996)	13.5	13.5	1249.05	169
IRAN, ISLAMIC REPUBLIC OF	13.2	26.7	8864.27	1169
QATAR	12.0	38.7	8597.23	1028
KOREA, REPUBLIC OF (SOUTH KOREA)	9.77	48.4	16702.31	1633
SAUDI ARABIA	9.23	57.7	8767.28	809
TURKEY	8.79	66.4	3015.48	265
ISRAEL (GAZA and JERICHO->1994)	4.78	71.2	2987.48	143
RUSSIAN FEDERATION (RUSSIA)	4.52	75.7	5574.44	252
UNITED STATES	4.36	80.1	6061.9	264
BRAZIL	4.32	84.4	10107.97	436
UNITED ARAB EMIRATES	3.35	87.8	8439.46	283
EGYPT	2.43	90.2	3212.14	78
Other countries	9.80	100	-	-
<i>Overall weighted distance</i>				<i>6,528</i>

(¹) Based on *Comext* data on imported polymer quantities from extra-EU countries (Eurostat, 2019a). The shares reported were determined as 3-year averages of import shares calculated, based on raw *Comext* data, for the years 2016-2018.

(²) From harbour to harbour, based on the calculation tool available on SeaRates.com (<https://www.searates.com/services/distances-time/>). Distances for imports from countries in the Middle-East and Asia were determined considering Marseille as destination port in Europe. For imports from other countries (in the case of LDPE Norway, Russia, United States, and Brazil), Rotterdam was considered as destination port.

4.5.4 Distribution Stage

The transport of mulching film from the manufacturing site to the final LCA user was modelled based on the default transport scenario specified in the *Plastics LCA* method for the pathway *factory* → *distribution centres* → *final client*, assuming a 100% local supply chain. The following routes were thus considered:

1. 1200 km by lorry (total weight >32 t; Euro 4) from factory to distribution centres; and
2. 250 km by van (lorry <7.5t, Euro 3; utilisation ratio of 20%) for the roundtrip from distribution centres to final users.

LCIs for transport through all types of vehicles were available as EF-compliant datasets, which were used in the modelling.

4.5.5 End of Life Stage

This section addresses the modelling of the End of Life stage of mulching film in the different investigated scenarios. In particular, Section 4.5.5.1 describes the EU-average End of Life scenario considered as a base case for the calculation of the potential impacts. The remaining sections (4.5.5.2 – 4.5.5.6) address the modelling of non-biodegradable mulching film collection and transport, and of the different End of Life options applied to it (including burial in soil of non-collected film residues). In-situ biodegradation of biodegradable film is then addressed in Section 4.5.5.7. Finally, Section 4.5.5.8 provides case study-specific details on the estimate of the potential generation and release of macro-plastics at End of Life (including product litter) and of micro-plastics throughout the supply chain.

4.5.5.1 End of Life scenario

The End of Life options applicable to agricultural mulching film depend on different factors including the type of material and its biodegradability properties, the level of contamination with soil, agricultural residues and agrochemicals (affecting costs of collection and treatment, including potential recycling), and on local or national conditions. These include the existence of a specific legislation, the implementation of a dedicated collection scheme, and the available waste management infrastructure (Eunomia, 2021).

For conventional, non-biodegradable mulching film, mechanical recycling is reported to be currently not (or very scarcely) applied, due to the generally high level of contamination with soil (Eunomia, 2021; based on APE Europe data for 2019). The presence of contamination increases the mass of collected plastic material by 3 to 5 times, making recovery operations economically unfeasible at present. Similarly, Steinmetz et al. (2016) reported that mechanical recycling of used mulching film is only feasible for low levels of (soil) contamination (i.e. less than 5% by weight), which is unlikely the case of mulching film applied in large-scale agriculture. According to Eunomia (2021), incineration and landfilling are applied to agricultural plastic waste that cannot be sustainably recycled, including collected mulching film contaminated by soil and any pesticide residues (depending on applied agricultural practices). Moreover, a share of applied mulching film generally breaks up during use (due to exposure to sunlight and other weathering and/or mechanical agents) or during collection (due to, e.g., the presence of soil and crop residues). Such portion of film is thus ultimately not removed from land, but it is left on the field and then buried in soil during subsequent tillage. Based on this information and data, the EU-average End of Life scenario of both virgin and partially recycled LDPE mulching films was assumed to include either incineration or landfilling of the portion of film removed from the field after use, as well as burial into the soil of non-collected film.

The collection rate (efficiency) of used conventional mulching film generally depends on the thickness of the film itself: the thicker the film, the higher the collection rate, being thinner films more easily damaged during use or removal, leading to a only partial collection and to an increased amount of film residues left on the soil (Steinmetz et al., 2016). However, the relation between collection rate and thickness is not deterministic. Organic Waste Systems (OWS) estimated that the collection rate is as low as 35% for 10 µm mulching films, while it increases to 75% for a film thickness of 20 µm, and to 90% for 25 µm mulching films (OWS, 2017). On a more general level, an overall average

collection rate of nearly 80% can be estimated for mulching film in the EU, based on APE Europe data for the year 2019 reported by Eunomia (2021) (i.e. 66 kt of collected mulching film out of 83 kt placed on the market -and generated as waste- during the same year). Similarly, the Circular Plastics Alliance (CPA) estimated that the collection rate of mulching film for vegetable cultivation is over 80% (CPA, 2020). In this study, an average thickness of 35 μm was considered for LDPE mulching films, which is higher than the minimum thickness required by the European standard EN 13655 for mulching film intended to be removed from soil after use (i.e. 20 μm) (CEN, 2018a). Therefore, a 90% collection rate was assumed for both virgin and partially recycled LDPE mulching films, according to the thickness-specific estimate reported above for films with the most proximate (but still lower) thickness (i.e. 25 μm) (OWS, 2017). This assumption may thus be conservative, since a higher collection rate may be expected for a thickness of 35 μm as considered in this study, but in the absence of specific evidence it was not increased further. Note that in the sensitivity analysis performed on the thickness of virgin LDPE mulching film (as discussed in Section 4.2), the collection rate was adjusted accordingly (Section 4.8.5.1).

The incineration and landfilling rates of collected mulching film were defined based on the rates estimated for non-packaging agricultural plastic waste in a 2014 PlasticsEurope study reported by Eunomia (2021) (i.e. 30% incineration and 42% landfilling, which in relative terms are equal to 42% and 58%, respectively). These values were extrapolated to 2018 (the most recent year for which calculation was possible), assuming they have undergone the same percentage variation as the incineration and landfilling rates of total plastic waste between 2014 and 2018. For consistency, the rates reported by PlasticsEurope (2015 and 2020) were considered, which showed a variation equal to +7.8% for incineration and to -19.2% for landfilling. Applying these variations to the 2014 data reported above for non-packaging agricultural plastic waste, an incineration rate of 49% was estimated for collected mulching film, while the landfilling rate was equal to 51%. Since only 90% of the film applied on land was assumed to be collected after use (as discussed above), the actual incineration and landfilling rates were equal to 44% and 46%, respectively. The overall, EU-average End of Life scenario considered for non-biodegradable mulching film thus included 44% incineration, 46% landfilling, and 10% burial in soil during subsequent tillage applied to the same piece of land.

Handling and treatment of soil collected with removed mulching film was also accounted in the End of Life scenario, considering the burdens associated with its incineration, landfilling, and transport to these treatments after collection. While the burdens of collection from the field were considered, they are not affected by the amount of material removed from land, and hence by the presence of soil contamination (see Section 4.5.5.2). Similarly to the collection rate, the level of contamination of collected mulching film with soil generally depends on the film thickness: the lower the thickness the higher the level of contamination, and hence the more difficult (and expensive) the removal and subsequent treatment (OWS, 2017). However, available data do not specifically relate the estimated contamination level to a defined thickness. According to the European standard EN 13655, the observed level of contamination of mulching films with soil and vegetal residues can vary between 70% and 90%, with thinnest films (e.g. with a thickness lower than 25 mm) being the mostly contaminated (CEN, 2018a). Conversely, Steinmetz et al. (2016) reported that the weight of contaminants is in general up to 40-50%. Moreover, an overall average contamination rate with soil of nearly 67% can be estimated based on the 2019 APE Europe data reported by Eunomia (2021) (i.e. 166 kt of soil collected along with 83 kt of mulching film generated as waste during the same year). The same value can also be calculated based on the contamination ("soilage") factor reported by CPA (2020), which is equal to 3 (i.e. the mass of total collected material, including contamination, is 3 times the mass of plastic film removed from land). In order to apply consistent values of the contamination rate both in the base case scenario and in the sensitivity analysis considering different film thicknesses for virgin LDPE film (see Section 4.8.5.1), a relationship between film thickness and contamination rate was established, based on linear regression of available data. In the calculation, the

lowest contamination rate reported (40%; Steinmetz et al, 2016) was associated to the highest end of the range of commercial thickness available for conventional mulching film (i.e. 50 μm , according to OWS, 2017), while the highest contamination rate (90%; CEN, 2018a) was associated to the lowest reported thickness (i.e. 15 μm). Applying the obtained relationship, a contamination rate of 60% was calculated for a 35 μm film (as considered in this study), i.e. 1.5 kg of soil are collected per kg of mulching film removed from land. This value is in line with the overall average contamination rate estimated based on the values reported by Eunomia (2021) and CPA (2020), i.e. 67%. As mentioned in Section 4.2 and above, a sensitivity analysis was conducted on the thickness of virgin LDPE mulching film (Section 4.8.5.1), and the contamination rate was adjusted according to the assumed thickness.

For biodegradable mulching film alternatives (i.e. starch-based and PLA-based films), in-situ biodegradation was applied as the unique End of Life option, being these alternatives intended to be left on the field after use and then ploughed into soil for biodegradation. All the components of each film material (e.g. polymer and additives) were assumed to be suitable for biodegradation, and to completely exclude hazardous (e.g. toxic) substances or elements (e.g. metals). However, in a real study, this will ultimately depend on the actual material composition of the specific product investigated (and all constituents shall be properly taken into account to calculate intermediate and final emissions from biodegradation, as reported in Section 4.4.10.9 of the *Plastics LCA* method).

4.5.5.2 Modelling of mulching film collection and transport to treatment

As reported in Section 4.5.5.1, non-biodegradable mulching films (i.e. virgin and partially recycled LDPE films) are removed from the field at the end of the growing season, to be then routed to incineration or landfilling. Removal was assumed to take place mechanically, considering a collection rate equal to 90% of the film originally applied on land, based on the thickness considered in this study for conventional LDPE films (35 μm), as described in Section 4.5.5.1.

The removal process was modelled based on the ILCD-EL compliant dataset “[GLO] Soil cultivation; stubble cleaning (medium, 67 kW)” available in the GaBi database, which was applied as an approximation of the real process in the absence of more specific (EF-compliant) datasets for mulching film removal. In this dataset, the burdens of the removal process are calculated as a function of the area of land to be covered with relevant agricultural machinery, and not based on the mass of material to be collected. Therefore, they are not affected by the presence of soil removed as a contaminant during mulching film collection (estimated to be 60% of total collected material; see Section 4.5.5.1).

Transport of contaminated mulching film to incineration and landfilling was assumed to take place along an overall average distance of 100 km. However, the burdens associated with road transport for 50 km were considered to be already covered in the applied EF-compliant incineration and landfilling datasets (which account for waste transport to the relevant treatment or disposal site along an unspecified distance). An additional transport for a distance of 50 km was thus separately modelled, based on the EF-compliant dataset “[EU-28+3] Articulated lorry, Total weight 28-32 t, mix Euro 0-5”, and considering the current EU-average diesel mix as an input.

4.5.5.3 Modelling of sorting and recycling

This section is relevant only to the sensitivity analysis described in Section 4.8.5.7, where the impacts of non-biodegradable mulching films are recalculated individually applying each End of Life option included in the EU-average End of Life scenario considered as a base case¹⁰⁷ (Section 4.5.5.1). While such scenario does not include

¹⁰⁷ Apart from burial in soil of uncollected mulching film residues, which is not an intended End of Life option for non-biodegradable mulching film, in contrast to incineration and landfilling.

mechanical recycling (as being reported to be currently applied to a very limited extent or not to be applied at all), this option was nevertheless independently investigated in the sensitivity analysis, to at least preliminarily evaluate the effects of its application on the potential impacts of non-biodegradable mulching film alternatives. However, the assessment could be made only in an approximate manner, as no specific information and data on the sorting and recycling of contaminated mulching film were available (nor for agricultural plastic films in general), and a number of approximations had to be performed to model these processes, as discussed below.

Before recycling, contaminated mulching film collected from the field was assumed to be sorted in specific facilities, where soil is (mechanically) removed and the plastic material is sorted and prepared for subsequent recycling. The sorting process was modelled based on the average inventory developed by Franklin Associates (2018) for mixed plastic waste sorting in the United States (relying on input/output data collected from different dual-stream and single-stream sorting facilities in the Country). While these data are mostly representative of the sorting of cleaner, small- or medium-sized plastic products from municipal collection (such as packaging items or other consumer goods), they were considered a reasonable approximation for the burdens of any sorting operation mulching film may undergo before recycling. However, they do not account for any cleaning operation, nor for any operation specifically aimed at removing soil residues attached to the film, as assumed to occur during the sorting process (see above). The presence of contaminating soil was thus handled by modelling the sorting process (whose burdens are proportional to the mass of material to be treated) in a quantity that considers the contribution of both film and soil. In other words, the overall burdens of the sorting process were increased proportionally to the amount of soil residues entering the treatment. In the modelled process, the burdens from sorting are those associated with the supply of the material and energy inputs required to carry out this waste treatment activity. Such inputs were modelled through background EF-compliant datasets representative of EU-average conditions, as better detailed in Section 3.5.5.3 of the Beverage Bottles case study (Table 3.9). A 73% sorting efficiency was then considered for cleaned mulching film, according to the average value estimated by Antonopoulos et al. (2021) for waste LDPE. Discarded film was assumed to be incinerated, as this is one of the two most common fates of plastic rejects from sorting and recycling operations, along with co-combustion in cement kilns (Rigamonti et al., 2014). The same fate was applied, for consistency, to removed soil particles. The incineration processes were modelled based on the actual quantity of material (soil particles or discarded plastic rejects) to be treated, in contrast to the other inputs of the sorting process (which were increased proportionally to the mass of soil, as discussed above).

For mechanical recycling of (cleaned) LDPE film, no material-specific, EF-compliant or ILC-D-EL compliant datasets were available. Therefore, this process was modelled through an aggregated, EF-compliant proxy dataset representing the burdens associated with the production of a generic, secondary plastic granulate, out of sorted, post-consumer plastic waste (via grinding, metal separation, washing, and extrusion to pellets)¹⁰⁸. While the dataset is likely mostly representative of the recycling of rigid or semi-rigid plastic products, it was considered a suitable approximation also for flexible products (i.e. mulching film in this case), since their recycling is expected to rely on similar unit operations to those considered in the modelled process. The underlying inventory is developed based on literature data for each of these operations, refers to the year 2018, and accounts for an overall recycling efficiency equal to 84% (on the sorted input material). Process waste and scrap are sent to incineration, consistently with the typical fate of plastic recycling residues, which due to their high calorific value are normally routed to incineration or co-combustion in cement kilns (Rigamonti et al., 2014), as discussed above.

¹⁰⁸ [EU-28] Plastic granulate secondary (low metal contamination); from post-consumer plastic waste, via grinding, metal separation, washing, pelletization; production mix, at plant; plastic waste with low metal fraction.

Recycled LDPE granulate was assumed to replace virgin granulate of the same material, whose primary production burdens were credited to the system following the Circular Footprint Formula. Both fossil-based and bio-based LDPE were assumed to be replaced by recycled LDPE, according with the current EU-average mix of the two production routes. However, the estimated share of the bio-based pathway is currently marginal (0.2%), being LDPE currently available on the EU market mostly fossil-based (99.8%)¹⁰⁹. To account for the lower overall average quality of the recycled polymer compared to the replaced virgin polymer, a substitution ratio equal to 0.75 was considered, according to the default value specified in Annex C of the *Plastics LCA* method for LDPE used in (packaging) film applications. To model the burdens of avoided virgin polymer production and of the related feedstock supply, the same datasets (or combination of datasets) used for the modelling of upstream production of the two replaced polymers and of their feedstock were applied (as described in Sections 4.5.2 and 4.5.1, respectively¹¹⁰). This was made for consistency reasons, and to avoid possible distortions by applying different datasets from other sources.

According to the Circular Footprint Formula and the related default values of the A factor specified in Annex C to the *Plastics LCA* method, only 50% of the burdens of the sorting and recycling processes, and of the benefits from avoided virgin material production, were allocated to the system. The default value of A reported in Annex C is indeed equal to 0.5 for PE used in unspecified applications (considering the material-specific value, as no application specific value is provided).

4.5.5.4 Modelling of incineration

For conventional fossil-based polymers, including LDPE, aggregated material-specific incineration datasets (referring to the year 2012) are available from the pool of EF-compliant datasets. One of such dataset was hence applied to model the fate of both virgin and partially recycled LDPE mulching film in a municipal waste incineration plant¹¹¹. Similarly, for Thermoplastic Starch (TPS) and PLA used as copolymers in the respective polymer blends, partially aggregated, material-specific, ILCD-EL compliant inventories from the GaBi database (referring to the year 2018) were applied¹¹² (no EF-compliant datasets were available for these polymers).

All the selected datasets are developed based on a waste-specific incineration model, which is described in detail in Section 3.5.5.4 of the Beverage Bottles case study. The model applies element-specific transfer coefficients (based on data from real plants, stoichiometry, or expert estimates) 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). The energy content (net calorific value) of the input waste is also taken into account to calculate the amount of recovered energy (electricity and heat), based on EU-average energy efficiencies and recovery rates.

In line with the approach specified in the *Plastics LCA* method to model energy recovery from waste products (i.e. the Circular Footprint Formula), the product system generating the waste material sent to incineration (i.e. the mulching film life cycle, in this case) was allocated the full burdens from the incineration process. However, the system was

¹⁰⁹ The share of bio-based LDPE in the EU market was estimated based on the global production capacity of bio-based PE in 2018 (European Bioplastic, 2019; 200 kt) and the production capacity of PE as a whole in 2016 (PlasticsInsight, 2019; 103 Mt).

¹¹⁰ For Bio-based LDPE, please refer to the Packaging Film case study.

¹¹¹ The following incineration dataset was applied: "[EU-28+EFTA] Waste incineration of PE; waste-to-energy plant with dry flue gas treatment, including transport and pre-treatment | production mix, at consumer | polyethylene waste".

¹¹² The dataset applied for TPS incineration is "[EU-28] Thermoplastic starch polymer (TPS), unblended in waste incineration plant, waste-to-energy plant with dry flue gas treatment, without collection, transport and pre-treatment | production mix, at plant | Net calorific value 16 MJ/kg", while for PLA incineration the applied dataset is "[EU-28] Polylactic acid (PLA) in waste incineration plant, waste-to-energy plant with dry flue gas treatment, without collection, transport and pre-treatment | production mix, at plant | Net calorific value 17.9 MJ/kg".

credited with 100% of the benefits from avoided production of conventional energy (electricity and heat) assumed to be replaced by energy recovered from waste. While in the applied EF-compliant incineration datasets these credits are already accounted for in the aggregated inventory, for the selected GaBi datasets they were added to the main process inventory. In this case, the EU residual electricity grid mix (as modelled in the EF-compliant dataset “[EU-28+3] Residual grid mix; AC, technology mix | consumption mix, to consumer | 1kV - 60kV”) was credited to the amount of recovered electricity. For recovered heat, a new dataset representing the current EU-average heat supply mix was created, based on background EF-compliant datasets for each specific heat source included in the mix. The EU-average mix was defined based on most recent statistics for heat generation in Europe from the International Energy Agency (IEA, 2019), and included 42.4% natural gas, 30.8% hard coal, 21.8% biomass, and 5% heavy fuel oil. In the calculation of these figures, small shares of heat generated from geothermal, nuclear, and solar thermal sources (less than 1% overall) were excluded, in the absence of specific datasets for the modelling of the respective burdens. Thermal energy from waste (11%) was also excluded, as according to the Circular Footprint Formula, the use of energy from waste in a product system shall be modelled as 100% primary energy (being the benefits of its avoided primary production entirely allocated to the system generating such energy).

For PBAT used as a copolymer in both starch-based and PLA-based mulching film, no EF-compliant or ILCD-EL compliant incineration datasets or suitable proxies were available. A disaggregated, material-specific inventory was thus developed, based on the most recent version of the calculation tool developed by Doka (2009a) to model material and product incineration within municipal solid waste incineration plants. The model operates similarly to the one used to develop the EF-compliant incineration datasets described above, allowing the practitioner to account for the specific composition and energy content of the incinerated waste to develop a material-specific incineration inventory based on transfer coefficients. The elemental composition considered for PBAT is reported in Table 4.5, while an energy content (theoretical lower heating value) of 21.1 MJ/kg was calculated (based on the formula by Michel (1938) and the assumed polymer composition) and applied in the model. The tool also allows to adjust other technological parameters to the relevant geography or scope, including energy efficiencies, the share of alternative NO_x control technologies applied, and a few other specific parameters. For more details on how these parameters were set the reader is referred to Section 3.5.5.4 of the Beverage Bottles case study. Benefits associated with recovered energy were modelled as described above for existing GaBi datasets (i.e. electricity from the EU residual grid mix, and thermal energy from the current EU-average mix of heat sources). In the final dataset, the inventory flows generated by applying the Doka (2009a) tool were combined with the background *ecoinvent* datasets typically applied within incineration inventories available in such database. However, for energy-related flows (including avoided energy generation), background EF-compliant datasets were applied.

No material-specific datasets were available for incineration of soil collected with used mulching film removed from the field. This process was thus modelled by means of a combination of datasets approximating incineration of the main soil material components (i.e. the mineral part and organic matter). According to Jones et al. (2012), a typical sample of mineral soil comprises 45% minerals, 25% water, 25% air and 5% organic matter. However, air can be neglected in the modelling (as not contributing to any specific burden from incineration), and the same can be reasonably applied to (interstitial) water, as removed soil is expected to have a much lower water content at the gate of the incineration plant compared to in-situ conditions, due to evaporation and/or drainage of water in pores during collection, handling and transport. Excluding the contribution of air and interstitial water, soil was thus assumed to consist of 90% minerals and 10% organic matter, with the residual content of adsorbed water being covered in the composition of waste materials considered to approximate such components (i.e. inert waste for minerals and biodegradable waste for organic matter). For inert waste incineration, an aggregated EF-compliant dataset was applied in the

modelling¹¹³. For biodegradable waste, an incineration dataset from the *ecoinvent* database was applied¹¹⁴, while additionally including the benefits from avoided generation of conventional energy (electricity and heat) replaced by energy recovered from waste. As described above for other applied incineration datasets, replaced energy included electricity from the EU residual grid mix, and thermal energy from the current EU-average mix of heat sources. Note that the water content of the incinerated waste materials considered in the development of the applied datasets is equal to 0% for inert waste and 65% for organic waste. This corresponds to an overall water content of the incinerated waste equal to 6.5%, which is line with the water content expected for drained soil at the gate of the incineration plant (i.e. approximately 5%).

4.5.5.5 Modelling of landfilling

Landfilling of mulching film made of conventional non-biodegradable polymers (i.e. virgin and partially recycled LDPE) was modelled based on a common aggregated EF-compliant dataset representing disposal of non-biodegradable (fossil-based) plastic waste in a managed municipal solid waste landfill, referring to the year 2012 (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*)¹¹⁵. The underlying inventory is material-specific, but refers to the average chemical composition and degradability of generic plastic waste, rather than to those of the specific polymer being landfilled. This is considered an acceptable approximation for the scope of 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 (conventional) polymers, including LDPE (i.e. degradation in the range of 1% over 100 years; Doka 2009b). The inventory is developed based on a landfill model applying element-specific transfer coefficients to calculate the distribution of elements in the waste composition to landfill gas and leachate, and their ultimate emission to the environment over a 100-year time horizon. Emissions occurring beyond 100 years are not accounted in the model. Landfill gas generation is calculated based on the organic carbon content in the waste material and the respective degradation rate over 100 years assumed in the model (not reported). However, for simplification reasons, an average landfill gas composition for the stable methane phase is considered. 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 rates of landfill gas, energy efficiencies of gas engines, collection rate of leachate and respective treatment efficiencies). Further details on how these parameters were set in the model to reflect EU-average conditions are provided in Section 3.5.5.5 of the Beverage Bottles case study.

Similarly to incineration, landfilling of soil collected with used mulching film removed from the field was modelled based on EF-compliant datasets approximating landfilling of its main material components (estimated to include 90% minerals and 10% organic matter), in the absence of material-specific landfilling datasets for soil. Landfilling of the mineral part was approximated with inert material landfilling¹¹⁶, while landfilling of biodegradable waste¹¹⁷ was applied as a proxy for landfilling of soil organic matter. Further explanations of the assumptions behind the calculation of the considered soil composition, and of the applied modelling approximations, are available in Section 4.5.5.4.

¹¹³ *[EU-28+EFTA] Waste incineration of inert material; waste-to-energy plant with dry flue gas treatment, including transport and pre-treatment | production mix, at consumer | inert material waste.*

¹¹⁴ *[GLO] treatment of biowaste, municipal incineration.*

¹¹⁵ While a material-specific landfilling dataset for LDPE is available from other databases (i.e. *ecoinvent*), this EF-compliant dataset for landfilling of generic plastic waste was selected, as specifically referring to EU as the reference geography (in contrast to the available polymer-specific dataset), and to comply with the dataset selection "hierarchy" specified in the *Plastics LCA* method (Section 4.4.10.11).

¹¹⁶ Modelled through the following EF-compliant dataset: *[EU-28+EFTA] Landfill of inert material (other materials).*

¹¹⁷ Modelled through the EF-compliant dataset "*[EU-28+EFTA] Landfill of biodegradable waste*".

4.5.5.6 Modelling of burial in soil

Burial in soil of non-collected mulching film residues was modelled as direct release to soil of macro-plastic, which was accounted in the corresponding inventory-level indicator calculated as additional environmental information (see Sections 4.5.5.8 and 4.7.3). No other burdens after this initial macro-plastic release were modelled, such as any additional release of micro-plastics from further possible fragmentation in soil, nor any emission of additives, metals or possible degradation products. This is mostly due to incomplete understanding and knowledge of the fate of (conventional) plastic products or pieces once released into the environment (including fragmentation and degradation pathways), and to the lack of data on the use of additives in mulching film production (see Section 4.3). The same approach was also consistently applied to the (consumer) products investigated in the other case studies presented in this report, once littered into the environment.

As generally discussed in Sections 2.4 and 2.6, the potential impacts on ecosystems (particularly soil and its quality) and human health associated with the release of uncollected mulching film residues could not be quantified, due to the current absence of a suitable impact assessment method. However, this gap may be filled in the future, as long as better knowledge is gained on the actual fate, exposure and effects of macro-plastics released in soil and into the environment in general.

Beyond any direct, short-term impact on soil quality, the release of non-biodegradable plastic film residues may also generate indirect impacts on the medium-long term, in case of continued mulching film use and accumulation of plastic pieces and fragments in soil. Plastic accumulation can indeed (further) deteriorate soil quality and resulting crop growth (e.g. by reducing yield and fruit quality), which in turn may generate additional indirect environmental burdens and impacts due to an increased demand of agricultural inputs (e.g. fertilisers and water), to compensate for production losses. For example, a yield reduction between 2% and 23% was observed for cotton, wheat and maize cultivated in China on land mulched with thin (10 μm) mulching film. Due to its low thickness, the film remained largely uncollected on land, contributing to the accumulation of considerable amounts of plastic in soil (up to 260 kg/ha after 10 years) (Müller & Müller, 2015). For horticultural crops cultivated in the EU, no quantitative evidence was available in the literature, although the use of thicker films (>20 μm to comply with EN 13655) and the application of more sound collection practices are expected to involve the accumulation of lower amounts of plastic residues in soil compared to other countries such as China. This is also reflected in the relatively high collection rate assumed for used mulching film in this study (i.e. 90%, as discussed in Section 4.5.5.1). Therefore, while the possible effects on soil quality and crop yield could not be quantified, they can be reasonably expected to be more limited compared to the situation reported for China. The same applies to any indirect potential burdens and impacts associated with a possibly increased use of agricultural inputs to compensate for production losses (which according to the rules on "indirect effects" specified in the *Plastics LCA* method were excluded from this study).

4.5.5.7 Modelling of in-situ biodegradation

According to the rules of the *Plastics LCA* method, in-situ biodegradation of starch-based and PLA-based mulching films was modelled considering a timeframe of 100 years after the end of the Use stage. An overall biodegradation (i.e. carbon mineralisation) rate in soil equal to 90% was assumed for both mulching film alternatives, in line with the minimum biodegradation percentage to be achieved over a maximum period of 2 years according to the European Standard EN 17033 (CEN, 2018b), when testing the film material through the method specified in ISO 17556 (ISO, 2012). While data from real testing are reported in the literature (UBA, 2018), they were not considered sufficiently representative for modelling, as mostly referring to different material compositions compared to those assumed in this case study (in terms of relative shares between copolymers), and/or to unspecified or non-comparable material thicknesses. Moreover, in

most cases such values were determined for relatively short testing periods (4-9 months), which may have not allowed to capture the full biodegradation potential of the material on the medium/long-term. Finally, testing may have not been necessarily carried out according to the standard method prescribed by EN 17033 (i.e. the one specified in ISO 17556). However, it is noted that the mineralisation rate observed over 6 months for a PLA/PBAT film with unspecified thickness and including 9% PLA was equal to 94% (which is higher than the 90% rate assumed in this study). Conversely, much lower mineralisation rates are reported for starch-based materials/films with unspecified thickness, i.e. 37% over 9 months for a TPS/PCL blend consisting of 16% TPS, 75% PBAT and 9% additives, and 18% over 4 months for a Mater-Bi film. This diversity of results between the PLA-based and starch-based polymer blends likely reflects differences in material thickness, test duration and applied testing method, rather than real differences in their biodegradability. Therefore, as reported above, the same minimum biodegradation rate required by EN 17033 (90% over maximum 2 years of testing) was consistently applied for both mulching film alternatives. Moreover, while an even higher biodegradation rate may be achieved over the 100-year timeframe considered for modelling, no adjustments have been made to the assumed 90% value, in the absence of consistent and sufficiently representative evidence, and according to a conservative approach.

The modelled inventory included emissions from carbon biodegradation, calculated consistently with the assumed biodegradation rate (which indeed expresses the share of organic carbon in the polymer mineralised to CO₂ under testing conditions). According to the default conversion rates specified in the *Plastics LCA* method, all mineralised carbon was assumed to be aerobically converted to CO₂, while no methane emissions were modelled. Fossil and biogenic CO₂ emissions were separately quantified and inventoried, based on the origin of carbon in the different copolymers (i.e. biogenic for TPS and PLA, fossil for PBAT) and the material compositions reported in Table 4.5. Carbon not mineralised within the 100-year timeframe (i.e. 10% of the carbon content in the specific film) was considered to be no longer released (back) to the atmosphere, and no emissions were modelled for non-converted carbon. However, the effects of non-released biogenic carbon are not captured in the Climate Change impact indicator calculated for starch-based and PLA-based mulching films, since characterisation factors for biogenic CO₂ emissions and removals are set to zero in the *Plastics LCA* method (fully conforming to the PEF method). To better understand the implications of this methodological choice on the overall results, in a sensitivity analysis the Climate Change impact indicator of the two biodegradable mulching film alternatives was thus recalculated accounting for the effects of non-released biogenic carbon (Section 4.8.5.6).

Since the composition assumed for biodegradable mulching film materials does not include any non-biodegradable elements such as metals or additives, no emissions to soil of these substances were modelled. However, information on additive use was rather incomplete (limited to Thermoplastic Starch), and emissions of relevant substances may have been omitted.

Any non-degraded plastic fragments (including micro-plastics) and intermediate biodegradation products generated during the 100-year timeframe were assumed not to be transferred to other environmental compartments than agricultural soil, and hence no release was modelled. Similarly, no residual (micro)-plastic particles nor other final or intermediate biodegradation products (beyond CO₂, water and biomass) were assumed to be generated and emitted to soil at the end of the considered timeframe. However, these assumptions reflect incomplete understanding and knowledge of biodegradation pathways of biodegradable plastic products and polymers in soil, and may need to be revised in the future, as long as better knowledge is gained in this area. On the other hand, if all polymer components (including any additives) are ultimately biodegradable by soil microorganisms, and no hazardous (e.g. toxic) substances are included in the respective composition, the biodegradation process should ideally lead to the ultimate formation and release of only CO₂, water, new soil biomass, and mineral salts of any other elements included in the polymer composition.

Table 4.5. Elemental composition of PLA, TPS and PBAT copolymers considered to model in-situ biodegradation of biodegradable starch-based and PLA-based mulching films ⁽¹⁾.

Element	TPS ⁽²⁾	PLA ⁽³⁾	PBAT ⁽⁴⁾
TS (%)	100	100	100
Water (%)	0	0	0
VS (%TS)	100	100	100
Ash (%TS)	0	0	0
C fossil (%TS)	-	-	53.7
C biogenic (%TS)	44.8	49.5	-
H (%TS)	6.58	5.60	6.47
O (%TS)	48.6	44.5	39.8
N (%TS)	-	0.1	-
S (%TS)	-	0.3	-

⁽¹⁾ Starch-based mulching film comprises 40% TPS and 60% PBAT, while PLA-based film consists of 45% PLA and 55% PBAT.

⁽²⁾ The composition of TPS was defined based on stoichiometry, while considering that this polymer consists of 75% starch and 25% plasticisers/additives (equally split among Glycerol, Sorbitol and Glycidyl Methacrylate; see Section 4.5.2.3.1).

⁽³⁾ The composition of PLA was defined based on the results from the composition analysis of Ingeo™ PLA polymer available at: <https://www.natureworksllc.com/What-is-Ingeo/Where-it-Goes/Incineration>

⁽⁴⁾ The composition of PBAT was defined based on stoichiometry.

4.5.5.8 Generation and release of macro- and micro-plastics (including product litter at End of Life)

The generation (loss) and release of macro- and micro-plastics associated with the analysed mulching film scenarios were estimated based on the *Plastic Leak Project (PLP) method* (Peano et al., 2020). The *PLP method* was applied according to the operational description reported in Section I.3 of the *Plastics LCA method*, and to the general approach specified in Section 3.5.5.6 of this report (Beverage Bottles case study). Therefore, this section only focuses on the case-specific details and the product-specific parameters considered to apply the *PLP method* to the assessed mulching film LCA scenarios.

To estimate the total loss and release of macro-plastics at the End of Life stage (due to product littering and waste mismanagement), Equations I.1 and I.2 reported in the *Plastics LCA method* were applied, respectively. Beyond the default, case-unspecific parameters specified in Table I.2 of the method itself, the product-specific parameters reported in Table 4.6 were considered to apply these equations. A distinction was made between non-biodegradable films (i.e. virgin and partially recycled LDPE films) and biodegradable ones (i.e. starch-based and PLA-based films).

For non-biodegradable films, the littering rate was determined based on the assumed collection rate from the field after use, which was set to 90% (Section 4.5.5.1), corresponding to a littering rate of 10%. Final release rates were then defined based on the size and the residual economic value of the littered product/material, following the approach described in Peano et al. (2020, p. 78-80). Assuming a large size (>25 cm) and a medium residual value for uncollected mulching film residues, the final release rates to ocean and to the terrestrial environment were set to 5% and 95%, respectively. This means that uncollected mulching film was considered to almost entirely remain in soil (i.e. 95%), with only a minor portion ultimately reaching ocean (5%). Note, however, that collected mulching film subject to waste mismanagement (i.e. 9.25% of removed film) also contributes to the final release to both ocean and the terrestrial environment.

For biodegradable mulching films, the littering rate and the final release rate to the terrestrial environment were both set to 100%, being such alternatives intentionally left on the field after use, to be then incorporated in soil for biodegradation. The entire amount of biodegradable mulching film applied on land (per functional unit) was thus considered to be ultimately released to the terrestrial environment (i.e. to soil). However, this assumption reflects the fact that the *PLP method* only quantifies the initial loss of plastic products or pieces into the environment, and the release occurring only after any immediate (short-term) redistribution of released items among different environmental compartments (e.g. from freshwater to ocean). The effects of biodegradation (and of any other environmental mechanisms further affecting the fate of such items) are instead not taken into account, due to incomplete knowledge of fragmentation and (bio)-degradation pathways of (biodegradable) plastic products in soil and in the environment in general. The macro-plastics and total release calculated in this study for biodegradable mulching films through the *PLP method* may thus be even significantly overestimated compared to the ultimate release after biodegradation has occurred (which is not quantified). For the sake of clarity, the release of biodegradable plastic material was clearly distinguished in the results, with respect to the release of conventional, non-biodegradable plastic.

Table 4.6. Product-specific parameters considered to apply the *PLP method* to quantify the macro-plastics loss and release of the investigated mulching film LCA scenarios.

Parameter (1)	Scenario	
	LDPE film (S1) 35% R-LDPE film (S2)	Starch-based film (S3) PLA-based film (S4)
Littering rate (LR _{lit}) (%)	10	100
Release rate to ocean (Re _{ocean}) (%)	5	0
Release rate to the terrestrial environment (Re _{terenv}) (%) (2)	95	100

(1) For details on the meaning of each parameter, the reader is referred to Section I.3 of the *Plastics LCA method*.

(2) Including release to freshwater sediments, as discussed in Section 3.5.5.6 (Beverage Bottles case study).

As for micro-plastics, relevant sources considered in this case study include pellet losses from product manufacturing and micro-particles from tire abrasion during foreground road transport (no textiles are used in the foreground system). The contribution of these sources to the total value-chain loss and release of micro-plastics to ocean and to the terrestrial environment was estimated according to Equations I.3-I.6 of the *Plastics LCA method*, considering the default source- and pathway-specific parameters specified in Tables I.3-I.5 of the method itself. No product-specific parameters had to be determined, as the only case-specific parameter linking the different equations to the specific product inventory (and hence to the functional unit of each scenario) is either the amount of plastic pellets entering the product manufacturing process, or the mass of product/material transferred along each foreground road transport route and the related distance (all expressed per functional unit)¹¹⁸. Apart from these parameters, the quantification was thus made by means of default parameters that are not affected by the type of product, polymer or feedstock source.

4.5.6 Calculation of the Climate Change impact from iLUC

As a base case, the potential Climate Change impact from indirect Land Use Change (iLUC) associated with the investigated partially bio-based mulching film scenarios was calculated according to the approach outlined in Section 4.4.15.3 of the *Plastics LCA*

¹¹⁸ An exception is the Average Vehicle Load (kg), which depending on the situation may be considered a value-chain specific parameter. However, a unique average default value was considered in this case study, as specified in Table I.3 of the *Plastics LCA method*.

method. A sensitivity analysis applying an alternative method and resulting emission factors was also performed, as described in Section 4.8.5.5.

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), as recommended in the *Plastics LCA* method, the specific land demand of the crop(s) used as a feedstock for each partially bio-based polymer ($\text{m}^2 \cdot \text{year} / \text{kg crop}$) was calculated first. The calculation was based on the total aggregated amount of arable and agriculture land occupation flows reported in the dataset used to model the production of the specific crop, considering only those flows referring to the country where the crop is grown. If the geography of such flows was not specified, all arable and agricultural land occupation flows reported in the dataset were aggregated. However, cultivation of US maize used in PLA production was not modelled individually, as already covered in the fully aggregated, cradle-to-gate dataset applied to model PLA production (see Section 4.5.1.3). The specific land demand of this crop was thus calculated based on the 10-year average yield reported for maize grown in Nebraska by Vink and Davies (2015), who describes the life cycle inventory used to develop the applied PLA production dataset. All the obtained estimates were checked against the values of land demand calculated based on 5-years average crop yields from FAOSTAT (FAO, 2019), and both calculation routes were found to deliver generally aligned results (absolute variation between 3% and 20%). Hence, the values estimated based on land occupation flows reported in the datasets were ultimately considered, to keep consistency with the actual data applied in the modelling of the investigated scenarios.

The specific land demand for crop production was then converted into a demand per functional unit (FU) ($\text{m}^2 \cdot \text{year} / \text{FU}$), based on the specific crop consumption for polymer production ($\text{kg crop} / \text{kg polymer}$)¹¹⁹ and the amount of polymer needed to fulfil the functional unit (reference flow) in the specific scenario ($\text{kg polymer} / \text{FU}$). The potential Climate Change impact from iLUC was finally calculated by applying the recalculated GHG emission factors from the EU 2015/1513 Directive ($\text{kg CO}_2 \text{ eq.} / \text{m}^2 \cdot \text{y}$) to the estimated land demand per functional unit. All the described calculation steps to estimate the potential Climate Change impact due to iLUC are summarised in Table 4.7.

¹¹⁹ Defined or calculated consistently with the data applied in the modelling of the Polymer Production stage, as described in Section 4.5.2.3.

Table 4.7. Calculation of the potential Climate Change impact due to GHG emissions from iLUC associated with mulching film LCA scenarios relying on partially bio-based polymers.

Scenario/ Polymer	Feedstock	Land demand for crop production ⁽¹⁾ [m²·y/kg_{crop}]	Crop demand for polymer production [kg_{crop}/kg_{polymer}]	Polymer demand per functional unit (FU) [kg_{polymer}/FU]	iLUC GHG emission factor [kg CO₂ eq./ (m²·y)]	iLUC Climate Change impact [kg CO₂ eq./FU]
S3 – Starch- based film (TPS/PBAT)	Maize (EU)	1.34 (1.37)	0.134	115	0.0612	1.25
	Wheat (EU)	1.52 (1.75)	0.121	115	0.0612	1.29
	Potatoes (EU)	0.376 (0.301)	0.140	115	0.0612	0.370
	<i>Total</i>					
S4 – PLA-based film (PLA/PBAT)	Maize (US)	0.973 (0.939)	0.707	83.9	0.0612	3.53

⁽¹⁾ Calculated based on arable and agriculture land occupation exchanges reported in the dataset applied to model production of the specific crop, considering only those flows referring to the country of cultivation (or all reported flows, if the country was not specified). However, for US Maize (used for PLA production), land demand was calculated based on the 10-year average yield reported for maize grown in Nebraska by Vink and Davies (2015; 1.028 kg maize / m²·year), who describes the LCI used to develop the fully aggregated cradle-to-gate dataset applied to model PLA production. Values in parenthesis refer to land demand calculated based on crop yield data from FAOSTAT (5-years average), and are reported as a reference.

4.6 Life Cycle Impact Assessment results

The characterised, normalised and weighted impact assessment results of the investigated product scenarios are reported in Tables 4.8–4.11. For characterised results, the contribution of the main life cycle stages is also reported, and further illustrated in Figures D.2.1–D.2.3 in Annex D.2. Consistently with the applied system boundary, the considered contributions include:

- Feedstock Supply, i.e. depending on the feedstock/scenario: (i) oil/natural gas extraction, processing, transport and possible refining, as well as transport of naphtha from refinery to downstream users (fossil-based polymers); (ii) collection, transport and sorting of post-consumer plastic waste (recycled polymers); or (iii) crop cultivation and transport to further processing (bio-based polymers);
- Polymer Production, i.e. all gate-to-gate activities carried out to convert or recycle relevant feedstock materials into the specific polymer, including any transport among these activities and transport of polymer granulate to the mulching film manufacturing site;
- Manufacturing, i.e. conversion of the polymer into mulching film by blown film extrusion;
- Distribution, i.e. transport of mulching film from the manufacturing site to the final user; and
- End of Life, i.e. collection, transport and treatment/disposal of non-biodegradable mulching films and attached soil residues, or in-situ biodegradation of biodegradable films. Any avoided processes from downstream displacement of virgin materials and energy are also included. This contribution hence represents the net impact from the End of Life stage, resulting from the balance between real burdens of the applied waste management activities and resulting benefits (if any).

The last row of Tables 4.8–4.11 also reports the total weighted impact score (single score) of individual scenarios, calculated by aggregating normalised and weighted impact assessment results across all impact categories. Single impact scores provide a more immediate and synthetic representation of the overall (relative) environmental performance of the analysed product scenarios. However, they are affected by greater uncertainty (due to the application of additional normalisation and weighting factors), and by value choices necessarily applied to define weighting factors establishing an order of relevance of the different impact categories in a European decision context. Note that all the results presented in this section are affected by the limitations and critical assumptions discussed in Section 4.4, and shall be interpreted taking them carefully into account.

Table 4.8. Characterised, normalised and weighted impact assessment results of virgin LDPE mulching film (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.29E+02	2.82E+02	4.97E+01	1.94E+01	1.62E+02	6.42E+02	8.28E-02	1.84E+00
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						6.32E+02	8.15E-02	1.81E+00
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						9.53E+00	1.23E-03	2.73E-02
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						5.41E-01	6.97E-05	1.55E-03
Ozone Depletion [kg CFC-11 eq.] (I)	1.65E-07	1.97E-10	1.74E-08	4.67E-11	1.35E-08	1.96E-07	8.39E-06	5.66E-05
Human Toxicity - cancer [CTUh] (III)	3.78E-06	1.14E-06	4.25E-08	1.25E-07	2.53E-07	5.33E-06	1.38E-01	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	1.35E-05	6.36E-06	1.22E-06	1.80E-06	5.25E-06	2.81E-05	5.92E-02	0.00E+00
Particulate matter [Disease incidence] (I)	3.36E-06	5.63E-06	1.77E-06	6.19E-07	4.24E-06	1.56E-05	2.45E-02	2.34E-01
Ionising Radiation [kBq U ²³⁵ eq.] (II)	4.01E+00	3.79E+01	1.99E+01	5.46E-02	-1.99E+01	4.19E+01	9.93E-03	5.33E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	3.89E-01	6.46E-01	9.82E-02	1.23E-01	2.15E-01	1.47E+00	3.62E-02	1.85E-01
Acidification [mol of H ⁺ eq.] (II)	5.17E-01	6.91E-01	1.60E-01	1.36E-01	5.25E-02	1.56E+00	2.81E-02	1.87E-01
Eutrophication - terrestrial [mol N eq.] (II)	8.94E-01	2.05E+00	3.86E-01	7.13E-01	1.03E+00	5.07E+00	2.87E-02	1.12E-01
Eutrophication - freshwater [kg P eq.] (II)	1.43E-03	3.69E-04	1.13E-04	1.22E-04	2.73E-03	4.76E-03	1.87E-03	5.50E-03
Eutrophication - marine [kg N eq.] (II)	8.08E-02	1.93E-01	3.61E-02	6.42E-02	8.81E-02	4.62E-01	1.63E-02	5.10E-02
Ecotoxicity - freshwater [CTUe] (III)	7.87E+01	3.02E+01	2.12E+00	3.29E+00	3.66E+00	1.18E+02	9.99E-03	0.00E+00
Land Use [Pt] (III)	1.18E+02	9.55E+02	4.43E+02	2.13E+02	2.06E+02	1.93E+03	1.45E-03	1.22E-02
Water Use [m ³ world eq.] (III)	2.87E+00	7.72E+01	6.56E+00	7.40E-01	3.46E+01	1.22E+02	1.06E-02	9.61E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	3.27E-05	2.82E-05	1.47E-05	1.12E-06	-6.07E-06	7.07E-05	1.22E-03	9.87E-03
Resource Use - fossils [MJ] (III)	1.08E+04	4.17E+03	8.15E+02	2.63E+02	-1.53E+03	1.45E+04	2.22E-01	1.98E+00
<i>Total weighted impact (single score)</i>								4.76E+00

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 4.9. Characterised, normalised and weighted impact assessment results of 35% recycled LDPE mulching film (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.12E+02	2.56E+02	4.97E+01	1.94E+01	1.62E+02	5.99E+02	7.72E-02	1.71E+00
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						5.89E+02	7.59E-02	1.68E+00
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						9.49E+00	1.22E-03	2.71E-02
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						5.26E-01	6.78E-05	1.50E-03
Ozone Depletion [kg CFC-11 eq.] (I)	1.95E-07	2.00E-10	1.74E-08	4.67E-11	1.35E-08	2.27E-07	9.72E-06	6.56E-05
Human Toxicity - cancer [CTUh] (III)	3.12E-06	9.61E-07	4.25E-08	1.25E-07	2.53E-07	4.50E-06	1.17E-01	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	1.11E-05	5.63E-06	1.22E-06	1.80E-06	5.25E-06	2.50E-05	5.26E-02	0.00E+00
Particulate matter [Disease incidence] (I)	2.74E-06	4.92E-06	1.77E-06	6.19E-07	4.24E-06	1.43E-05	2.25E-02	2.14E-01
Ionising Radiation [kBq U ²³⁵ eq.] (II)	2.31E+00	3.41E+01	1.99E+01	5.46E-02	-1.99E+01	3.65E+01	8.65E-03	4.64E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	3.20E-01	5.53E-01	9.82E-02	1.23E-01	2.15E-01	1.31E+00	3.23E-02	1.65E-01
Acidification [mol of H ⁺ eq.] (II)	4.20E-01	5.98E-01	1.60E-01	1.36E-01	5.25E-02	1.37E+00	2.47E-02	1.64E-01
Eutrophication - terrestrial [mol N eq.] (II)	7.43E-01	1.78E+00	3.86E-01	7.13E-01	1.03E+00	4.65E+00	2.63E-02	1.03E-01
Eutrophication - freshwater [kg P eq.] (II)	1.19E-03	4.00E-04	1.13E-04	1.22E-04	2.73E-03	4.56E-03	1.79E-03	5.27E-03
Eutrophication - marine [kg N eq.] (II)	6.67E-02	1.67E-01	3.61E-02	6.42E-02	8.81E-02	4.22E-01	1.49E-02	4.66E-02
Ecotoxicity - freshwater [CTUe] (III)	6.48E+01	2.78E+01	2.12E+00	3.29E+00	3.66E+00	1.02E+02	8.63E-03	0.00E+00
Land Use [Pt] (III)	9.27E+01	8.63E+02	4.43E+02	2.13E+02	2.06E+02	1.82E+03	1.36E-03	1.15E-02
Water Use [m ³ world eq.] (III)	3.33E+00	6.64E+01	6.56E+00	7.40E-01	3.46E+01	1.12E+02	9.77E-03	8.82E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	2.66E-05	2.52E-05	1.47E-05	1.12E-06	-6.07E-06	6.16E-05	1.06E-03	8.60E-03
Resource Use - fossils [MJ] (III)	8.84E+03	3.62E+03	8.15E+02	2.63E+02	-1.53E+03	1.20E+04	1.84E-01	1.64E+00
<i>Total weighted impact (single score)</i>								4.21E+00

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 4.10. Characterised, normalised and weighted impact assessment results of starch-based mulching film (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.66E+01	4.86E+02	3.00E+01	1.13E+01	1.21E+02	6.64E+02	8.56E-02	1.90E+00
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						6.61E+02	8.52E-02	1.89E+00
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						1.76E+00	2.27E-04	5.03E-03
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						9.88E-01	1.27E-04	2.83E-03
Ozone Depletion [kg CFC-11 eq.] (I)	1.10E-07	3.75E-07	1.08E-08	2.73E-11	0.00E+00	4.96E-07	2.12E-05	1.43E-04
Human Toxicity - cancer [CTUh] (III)	8.43E-07	2.37E-06	3.19E-08	7.28E-08	0.00E+00	3.32E-06	8.62E-02	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	4.50E-05	5.41E-05	7.21E-07	1.05E-06	0.00E+00	1.01E-04	2.13E-01	0.00E+00
Particulate matter [Disease incidence] (I)	2.61E-06	8.99E-06	1.03E-06	3.62E-07	0.00E+00	1.30E-05	2.04E-02	1.95E-01
Ionising Radiation [kBq U ²³⁵ eq.] (II)	7.09E-01	2.12E+01	1.15E+01	3.19E-02	0.00E+00	3.34E+01	7.91E-03	4.25E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	4.97E-02	9.41E-01	5.76E-02	7.16E-02	0.00E+00	1.12E+00	2.76E-02	1.41E-01
Acidification [mol of H ⁺ eq.] (II)	3.41E-01	1.04E+00	9.32E-02	7.96E-02	0.00E+00	1.56E+00	2.81E-02	1.87E-01
Eutrophication - terrestrial [mol N eq.] (II)	1.50E+00	3.01E+00	2.25E-01	4.17E-01	0.00E+00	5.15E+00	2.91E-02	1.14E-01
Eutrophication - freshwater [kg P eq.] (II)	4.62E-03	8.59E-03	6.74E-05	7.14E-05	0.00E+00	1.33E-02	5.21E-03	1.54E-02
Eutrophication - marine [kg N eq.] (II)	2.75E-01	3.58E-01	2.10E-02	3.75E-02	0.00E+00	6.92E-01	2.45E-02	7.64E-02
Ecotoxicity - freshwater [CTUe] (III)	4.49E+02	3.57E+02	1.35E+00	1.92E+00	0.00E+00	8.09E+02	6.85E-02	0.00E+00
Land Use [Pt] (III)	5.78E+03	6.51E+03	2.44E+02	1.25E+02	0.00E+00	1.27E+04	9.52E-03	8.01E-02
Water Use [m ³ world eq.] (III)	8.34E+01	4.67E+01	3.97E+00	4.33E-01	0.00E+00	1.35E+02	1.18E-02	1.06E-01
Resource Use - mineral and metals [kg Sb eq.] (III)	5.32E-06	8.36E-05	8.63E-06	6.56E-07	0.00E+00	9.82E-05	1.70E-03	1.37E-02
Resource Use - fossils [MJ] (III)	1.42E+02	9.86E+03	4.86E+02	1.54E+02	0.00E+00	1.06E+04	1.62E-01	1.45E+00
<i>Total weighted impact (single score)</i>								4.32E+00

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 4.11. Characterised, normalised and weighted impact assessment results of PLA-based mulching film (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply (2)	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	-	4.16E+02	2.17E+01	8.29E+00	8.10E+01	5.27E+02	6.79E-02	1.51E+00
<i>Climate Change (fossil) [kg CO₂ eq.] (3)</i>						<i>5.26E+02</i>	<i>6.78E-02</i>	<i>1.50E+00</i>
<i>Climate Change (biogenic) [kg CO₂ eq.] (3)</i>						<i>1.12E+00</i>	<i>1.44E-04</i>	<i>3.20E-03</i>
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (3)</i>						<i>2.46E-01</i>	<i>3.17E-05</i>	<i>7.04E-04</i>
Ozone Depletion [kg CFC-11 eq.] (I)	-	7.12E-08	7.82E-09	2.00E-11	0.00E+00	7.90E-08	3.38E-06	2.28E-05
Human Toxicity - cancer [CTUh] (III)	-	1.19E-05	2.33E-08	5.34E-08	0.00E+00	1.19E-05	3.09E-01	0.00E+00
Human toxicity - non-cancer [CTUh] (III)	-	2.44E-05	5.28E-07	7.70E-07	0.00E+00	2.57E-05	5.41E-02	0.00E+00
Particulate matter [Disease incidence] (I)	-	8.52E-06	7.53E-07	2.65E-07	0.00E+00	9.54E-06	1.50E-02	1.43E-01
Ionising Radiation [kBq U ²³⁵ eq.] (II)	-	2.03E+01	8.41E+00	2.34E-02	0.00E+00	2.87E+01	6.80E-03	3.65E-02
Photochemical Ozone Formation [kg NMVOC eq.] (II)	-	9.74E-01	4.22E-02	5.25E-02	0.00E+00	1.07E+00	2.64E-02	1.34E-01
Acidification [mol of H ⁺ eq.] (II)	-	1.08E+00	6.83E-02	5.83E-02	0.00E+00	1.21E+00	2.18E-02	1.45E-01
Eutrophication - terrestrial [mol N eq.] (II)	-	3.18E+00	1.65E-01	3.05E-01	0.00E+00	3.65E+00	2.06E-02	8.07E-02
Eutrophication - freshwater [kg P eq.] (II)	-	2.01E-03	4.92E-05	5.23E-05	0.00E+00	2.11E-03	8.27E-04	2.44E-03
Eutrophication - marine [kg N eq.] (II)	-	3.14E-01	1.54E-02	2.75E-02	0.00E+00	3.56E-01	1.26E-02	3.93E-02
Ecotoxicity - freshwater [CTUe] (III)	-	3.32E+02	9.89E-01	1.41E+00	0.00E+00	3.34E+02	2.83E-02	0.00E+00
Land Use [Pt] (III)	-	1.10E+04	1.78E+02	9.14E+01	0.00E+00	1.12E+04	8.39E-03	7.07E-02
Water Use [m ³ world eq.] (III)	-	7.48E+01	2.93E+00	3.17E-01	0.00E+00	7.80E+01	6.80E-03	6.14E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	-	6.44E-05	6.32E-06	4.81E-07	0.00E+00	7.12E-05	1.23E-03	9.94E-03
Resource Use - fossils [MJ] (III)	-	7.85E+03	3.56E+02	1.13E+02	0.00E+00	8.31E+03	1.27E-01	1.14E+00
<i>Total weighted impact (single score)</i>								<i>3.37E+00</i>

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) The contribution of the Feedstock Supply stage is accounted under "Polymer Production", since for reasons of data availability the two stages could not be modelled separately in the life cycle inventory (Section 4.5.1.3). The impact of the Feedstock Supply stage is thus reported to be zero in all categories.

(3) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

4.7 Additional Environmental Information

This section presents the results related to additional environmental impacts or aspects going beyond the default set of impact categories considered in the *Plastics LCA* method, but that are considered relevant for the investigated product category. Additional environmental impacts and aspects addressed in this study include: (i) the potential impact on Climate Change due to GHG emissions from indirect Land Use Change (iLUC); (ii) potential Biodiversity impacts occurring at the endpoint level due to a number of relevant midpoint impact categories; (iii) the generation and release of macro-plastics at End of Life (including product litter); as well as (iv) the generation and release of micro-plastics throughout the product life cycle.

4.7.1 iLUC impact on Climate Change

Table 4.12 presents the estimated potential impact on Climate Change due to GHG emissions from iLUC expected to occur as a consequence of bio-based feedstock supply in the investigated mulching film scenarios. The total Climate Change impact accounting for such additional contribution is also reported, where relevant, for each product scenario.

Table 4.12. Potential Climate Change impact of GHG emissions from iLUC and resulting total Climate Change impact of mulching film LCA scenarios. Results are not intended to compare the different scenarios.

Scenario	iLUC Climate Change impact [kg CO ₂ eq./FU]	Total Climate Change impact (incl. iLUC) ⁽¹⁾ [kg CO ₂ eq./FU]
S1 – Fossil-based LDPE film	-	(642)
S2 – 35% R-LDPE film	-	(599)
S3 – Starch-based (starch/PBAT) film	2.92	668 (665)
S4 – PLA-based (PLA/PBAT) film	3.53	531 (527)

⁽¹⁾ Values in parenthesis refer to the total Climate Change impact of scenarios, without the iLUC contribution.

4.7.2 Biodiversity impact

Potential Biodiversity impacts estimated for the investigated mulching film scenarios, expressed as potential loss of animal and vegetal species per year, are presented in Table 4.13. The impact is quantified through an endpoint-level impact indicator accounting for a number of determining midpoint impact categories, including Climate Change, Photochemical Ozone Formation, Terrestrial Acidification, Eutrophication (freshwater and marine), Ecotoxicity (terrestrial, freshwater and marine), Land Use and Water Use. However, it is important to note that the impact assessment methods applied to these underlying midpoint impact categories differ from those prescribed in the *Plastics LCA* method (where impacts are assessed at the midpoint level). Moreover, direct potential biodiversity impacts from oil leakage are not quantified (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 3.5.1.1 of the Beverage Bottles case study).

Table 4.13. Potential biodiversity impact of mulching film LCA scenarios, expressed as potential loss of animal and vegetal species per year (species*year) per functional unit. Results are not intended to compare the different scenarios.

Scenario	Total	Feedstock supply	Polymer production	Manu-facturing	Distribution	End of Life
S1 – Fossil-based LDPE film	3.93E-06	2.11E-06	1.01E-06	1.90E-07	1.04E-07	5.19E-07
S2 – 35% R-LDPE film	3.49E-06	1.77E-06	9.09E-07	1.90E-07	1.04E-07	5.19E-07
S3 – Starch-based (starch/PBAT) film	1.87E-03	1.70E-03	1.68E-04	1.16E-07	6.07E-08	3.38E-07
S4 – PLA-based (PLA/PBAT) film	2.55E-06	0.00E+00	2.20E-06	8.44E-08	4.45E-08	2.27E-07

4.7.3 Macro- and micro-plastics generation and release

Table 4.14 shows the total plastic loss, final release to ocean and final release to the terrestrial environment estimated for the assessed mulching film scenarios, specifying the contributions of both macro- and micro-plastics. The contribution of the different macro- and micro- plastics sources to the respective total releases to ocean and to the terrestrial environment are shown in Figure 4.5. As described in Section 3.5.5.6 of the Beverage Bottles case study, the considered macro-plastics sources include direct product littering (i.e. mulching film intentionally or unintentionally left on the field after use) and mismanagement of mulching film waste after possible collection. Micro-plastics are instead generated from pellet losses during product manufacturing and tyre abrasion during foreground road transport. Note that, while the results for all the investigated scenarios are presented together, they are not intended to compare the different scenarios, and should not be used for this purpose by the reader. It is also reminded that the macro-plastics release and the resulting total plastic release estimated for biodegradable mulching films (starch-based and PLA-based) do not account for the fate of the material left on the soil, including the effects of biodegradation (see Section 4.5.5.8). The ultimate release after biodegradation has occurred may thus be even significantly lower compared to the estimates presented in Table 4.14.

Table 4.14. Total plastic loss, final release to ocean and final release to the terrestrial environment⁽¹⁾ estimated for mulching film LCA scenarios, including the contribution of both macro- and micro-plastics. Values in light blue cells refer to, or include the contribution of, a loss/release of biodegradable material, the fate of which is not accounted for in the calculation. Results are not intended to compare the different scenarios.

Indicator	Total		Macro-plastics		Micro-plastics	
	kg/FU	%	kg/FU	%	kg/FU	%
S1 – Fossil-based LDPE film (non-biodegradable)						
Loss	35.8	100.0%	35.6	99.4%	2.06E-01	0.6%
Release to ocean	1.80	100.0%	1.78	98.7%	2.32E-02	1.3%
Release to the terrestrial environment ⁽¹⁾	34.0	100.0%	33.8	99.5%	1.54E-01	0.5%
Ratio total release / loss	99.9%	-	100.0%	-	86.1%	-
S2 – 35% R-LDPE film (non-biodegradable)						
Loss	35.8	100.0%	35.6	99.4%	2.06E-01	0.6%
Release to ocean	1.80	100.0%	1.78	98.7%	2.32E-02	1.3%
Release to the terrestrial environment ⁽¹⁾	34.0	100.0%	33.8	99.5%	1.54E-01	0.5%
Ratio total release / loss	99.9%	-	100.0%	-	86.1%	-
S3 – Starch-based film (biodegradable)						
Loss	114	100.0%	113	99.9%	1.19E-01	0.1%
Release to ocean	1.35E-02	100.0%	0	0.0%	1.35E-02	100.0%
Release to the terrestrial environment ⁽¹⁾	113	100.0%	113	99.9%	8.93E-02	0.1%
Ratio total release / loss	100.0%	-	100.0%	-	86.1%	-
S4 – PLA-based film (biodegradable)						
Loss	83.2	100.0%	83.1	99.9%	8.96E-02	0.1%
Release to ocean	9.95E-03	100.0%	0	0.0%	9.95E-03	100.0%
Release to the terrestrial environment ⁽¹⁾	83.2	100.0%	83.1	99.9%	6.72E-02	0.1%
Ratio total release / loss	100.0%	-	100.0%	-	86.1%	-

⁽¹⁾ Including release to freshwater sediments, as discussed in Section 3.5.5.6 of the Beverage Bottles case study.

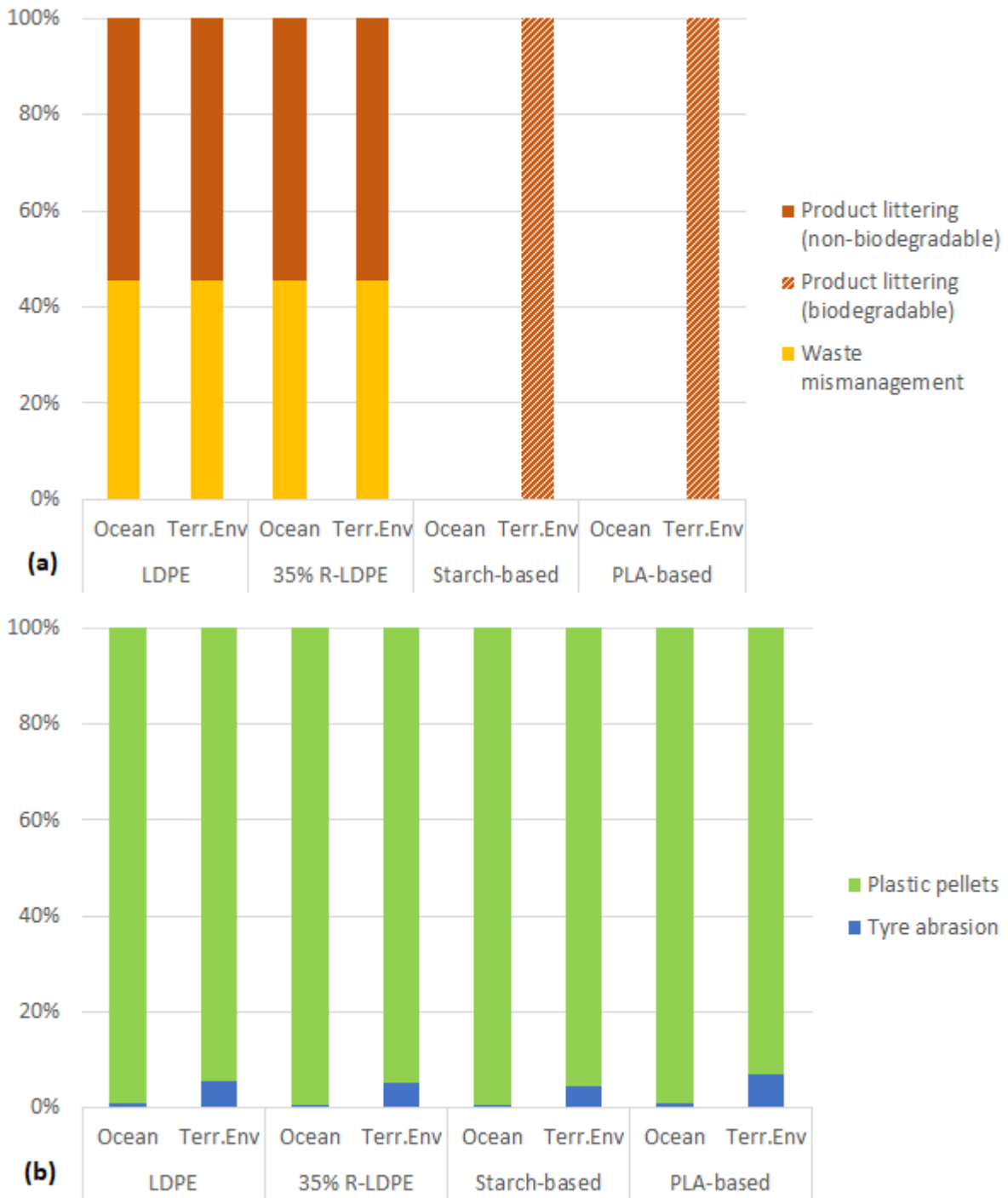


Figure 4.5. Contribution of single emission sources to the total release of macro-plastics (a) and micro-plastics (b) estimated for mulching film LCA scenarios to both ocean (Ocean) and the terrestrial environment (Terr.Env; including freshwater sediments). Results are not intended to compare the different scenarios.

4.8 Interpretation

In the interpretation of results, the most relevant impact categories of the analysed mulching film scenarios are firstly identified (Section 4.8.1). The contribution of individual life cycle stages to each most relevant impact category is then calculated, and the most relevant life cycle stages are identified (Section 4.8.2). The effects of GHG emissions from iLUC are also discussed (Section 4.8.3), while the results related to macro- and

micro-plastics generation and release are addressed in Section 4.8.4. Finally, the results of a sensitivity analysis on a number of parameters, assumptions and methodological choices are presented (Section 4.8.5), including characterised scenario impacts calculated by individually applying each End of Life option reported in Table 4.1.

It is noted that most relevant processes were not identified, since the life cycle inventories of the analysed product scenarios present different levels of vertical disaggregation of included foreground processes (e.g. the inventory of starch-based mulching film is more disaggregated than the ones developed for the other scenarios). Therefore the identification of most relevant processes would have not been carried out consistently across all the scenarios, and a more detailed investigation for specific scenarios would have not been meaningful. The identification of most relevant elementary flows was also not undertaken, as this would have required prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the *Plastics LCA* method shall proceed with the identification of both most relevant processes and elementary flows in each most relevant impact category.

4.8.1 Identification of most relevant impact categories

Table 4.15 shows the most relevant impact categories identified for each mulching film scenario, based on normalised and weighted impacts, according to the procedure described in Section 6.2.1 of the *Plastics LCA* method. Most relevant impact categories were hence identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact (single score) of each scenario. The contribution of toxicity-related impact categories was excluded from the calculation of total normalised and weighted impact scores, as being still based on the characterisation factors implemented in the EF 2.0 impact assessment methods applied in this study (and hence not yet updated based on REACH data)¹²⁰. Where needed, additional impact categories from the obtained ranking were added to the list of most relevant categories, to fulfil the requirement of having a minimum of three categories identified as most relevant.

Climate Change and Resource Use – fossils are identified as the two most relevant categories in all the assessed scenarios, being responsible for 77.5-80% of the total normalised and weighted impact. Particularly, Climate Change is the most relevant category for all alternatives except fossil-based LDPE film, where Resource Use – fossils is the most relevant one. Moreover, while the contribution of these categories is comparable for both LDPE and partially recycled LDPE films, for the two bio-based alternatives Climate Change is responsible for a larger share of the total impact.

For all scenarios, a third impact category is also identified as most relevant, albeit with a limited contribution in the range of 4-5%. Except for PLA-based film, where this additional contribution is provided by Acidification, for all the other alternatives the raking of most relevant categories is completed by Particulate Matter.

¹²⁰ According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018b), toxicity-related impact indicators calculated based on EF 2.0 characterization factors (applied in this study) shall be excluded from the procedure to identify the most relevant impact categories. However, any user of the *Plastics LCA* method shall apply the latest characterisation factors available at the time of the study (currently, those provided in the 3.0 EF reference package) and shall include also toxicity-related impact categories in the calculation of the total normalised and weighted impact score (and hence in the identification of most relevant impact categories).

Table 4.15. Most relevant impact categories identified for mulching film LCA scenarios and related contribution to the total normalised and weighted impact score of each scenario ⁽¹⁾.

S1 – Fossil-based LDPE film		S2 – 35% R-LDPE film	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Resource Use - fossils	41.6%	Climate Change	40.7%
Climate Change	38.6%	Resource Use - fossils	39.0%
Particulate Matter	4.9%	Particulate Matter	5.1%
Total	85.1%	Total	84.8%
S3 – Starch-based film		S5 – PLA-based film	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	44.0%	Climate Change	44.8%
Resource Use - fossils	33.6%	Resource Use - fossils	33.7%
Particulate Matter	4.5%	Acidification	4.3%
Total	82.0%	Total	82.8%

⁽¹⁾ Note that decimals are reported to avoid rounding errors, but do not reflect an accuracy corresponding to the number of digits shown for the specific contribution.

4.8.2 Identification of most relevant life-cycle stages

Table 4.16 shows the contribution of individual life cycle stages to the most relevant impact categories identified, for each mulching film scenario, in Section 4.8.1. The contribution was quantified according to the rules described in Sections 6.2.2 and 6.2.5 of the *Plastics LCA* method. Most relevant life cycle stages are also identified (in yellow) for each impact category, and include those that together contribute to at least 80% of the total impact in the specific category. Note that the net total impact, resulting from the algebraic sum of both positive and negative impact contributions of single life cycle stages, was considered to calculate the percentage contribution of each stage. Therefore, the sum of all positive contributions is necessarily larger than 100%, and is balanced by the negative contribution of specific life cycle stages (typically End of Life), leading to the sum of all positive and negative contributions correctly adding up to 100%. The possible negative impact and contribution from the End of Life stage is a result of the inclusion, along with the burdens of the applied waste management activities, of any benefits from secondary material production or energy recovery.

For both virgin and 35% recycled LDPE mulching films, Feedstock Supply and Polymer Production are identified as the most relevant life cycle stages in the Resource Use – fossils impact category. Particularly, Feedstock Supply is responsible for approximately 75% of the total impact in this category. Conversely, in Climate Change and Particulate Matter, Polymer Production is the most relevant stage (covering 34-44% of the total impact), followed by End of Life (25-30%) and Feedstock Supply (19-21%).

For both starch-based and PLA-based mulching films, Polymer Production is the most relevant life cycle stage across all the identified most relevant categories. This is especially the case of Resource Use – fossils, where it is responsible for over 90% of the total impact. Limited to PLA-based film, a similar contribution of the Polymer Production stage (approximately 90%) is also observed for Acidification. However, it should be reminded that, for PLA-based film, the Polymer Production stage also includes the impact from Feedstock Supply (which could not be modelled separately; Section 4.5.1.3). In Climate Change and Particulate Matter, the contribution of the Polymer Production stage is lower (69-79%), being followed also by End of Life (Climate Change) or Feedstock Supply (Particulate Matter) as additional most relevant stages.

Table 4.16. Contribution of individual life cycle stages to the most relevant impact categories identified for each mulching film LCA scenario ⁽¹⁾. Most relevant stages (i.e. those contributing to at least 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based LDPE film					
Resource use, fossils		Climate Change		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Feedstock Supply	74.4%	Polymer Production	43.9%	Polymer Production	36.0%
Polymer Production	28.7%	End of Life	25.2%	End of Life	27.1%
Manufacturing	5.6%	Feedstock Supply	20.1%	Feedstock Supply	21.5%
Distribution	1.8%	Manufacturing	7.7%	Manufacturing	11.3%
End of Life	-10.5%	Distribution	3.0%	Distribution	4.0%
S2 – 35% R-LDPE film					
Climate Change		Resource Use - fossils		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	42.7%	Feedstock Supply	73.6%	Polymer Production	34.4%
End of Life	27.0%	Polymer Production	30.1%	End of Life	29.7%
Feedstock Supply	18.7%	Manufacturing	6.8%	Feedstock Supply	19.2%
Manufacturing	8.3%	Distribution	2.2%	Manufacturing	12.4%
Distribution	3.2%	End of Life	-12.7%	Distribution	4.3%
S3 – Starch-based film					
Climate Change		Resource Use - fossils		Particulate Matter	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	73.1%	Polymer Production	92.7%	Polymer Production	69.2%
End of Life	18.2%	Manufacturing	4.6%	Feedstock Supply	20.1%
Manufacturing	4.5%	Distribution	1.4%	Manufacturing	7.9%
Feedstock Supply	2.5%	Feedstock Supply	1.3%	Distribution	2.8%
Distribution	1.7%	End of Life	0.0%	End of Life	0.0%
S4 – PLA-based film ⁽²⁾					
Climate Change		Resource Use - fossils		Acidification	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	78.9%	Polymer Production	94.4%	Polymer Production	89.5%
End of Life	15.4%	Manufacturing	4.3%	Manufacturing	5.7%
Manufacturing	4.1%	Distribution	1.4%	Distribution	4.8%
Distribution	1.6%	Feedstock Supply	0.0%	Feedstock Supply	0.0%
Feedstock Supply	0.0%	End of Life	0.0%	End of Life	0.0%

⁽¹⁾ Note that decimals are reported to avoid rounding errors, but do not reflect an accuracy corresponding to the number of digits shown for the specific contribution.

⁽²⁾ Note that the contribution of the Polymer Production stage also includes the one of the Feedstock Supply stage, as due to data availability the two stages could not be modelled separately in the life cycle inventory (see Section 4.5.1.3). The contribution of the Feedstock Supply stage is thus reported to be zero in all the identified most relevant impact categories.

4.8.3 Effects of indirect land use change (iLUC) on Climate Change

The additional contribution of GHG emissions from indirect land use change to the total Climate Change impact of the investigated bio-based alternatives (starch-based and PLA-based mulching films) is negligible. The impact increase observed when accounting for such contribution is indeed equal to only 0.4% for starch-based film, and to 0.7% for PLA-based film.

This result can be at least partly explained by the only partial bio-based content in the used polymers, which is lower than 50% (i.e. 40-45% depending on the polymer; Section 4.1). Moreover, the applied GHG emission factors may also play a role, as they appeared to fall in the lower end of the range of values available in the recent literature (see Section J.2 of the *Plastics LCA* method). The use of an alternative iLUC model and of the resulting GHG emission factors may thus result in an increased Climate Change contribution from iLUC, which was explored in a sensitivity analysis (see Section 4.8.5.5).

Finally, it must be noted that only the effects of GHG emissions from iLUC were addressed in this study, while nutrient-related emissions and other relevant emissions possibly associated with the additional use of converted land and/or with intensification in land use were not considered. However, accounting for such additional emissions may affect the impact of bio-based alternatives also in other impact categories than Climate Change, such as Acidification and Eutrophication.

4.8.4 Macro- and micro-plastics generation and release

This section discusses the results presented in Section 4.7.3 on the estimated potential generation (loss) and release of macro- and micro-plastics of the assessed mulching film scenarios.

Focusing initially on the relation between the total loss of macro- and micro-plastics and the resulting total release to ocean and to the terrestrial environment (Table 4.14), it is observed that almost 100% of the estimated plastic loss from the technosphere is ultimately released to the environment (ocean or terrestrial). For non-biodegradable mulching films, this is because the material lost as macro-plastics¹²¹ (which largely dominates the total macro- and micro-plastics loss, as discussed below) was considered to be entirely released to the terrestrial environment (95%) or to ocean (5%), according to the applied release rates (see Section 4.5.5.8). Similarly, for biodegradable films, 100% of the material applied on land, and lost as macro-plastics once left on the field after use, was assumed to be completely released to the terrestrial environment after incorporation in soil for biodegradation (Section 4.5.5.8). Since, also for biodegradable film alternatives, macro-plastics dominate the total plastic loss (see below) the same result as conventional non-biodegradable films is obtained in terms of relationship between total plastic loss and total release. In relative terms, for conventional mulching films the total release to ocean accounts for only a marginal share of the total initial plastic loss and of the total final release (5% on average), while the total release to the terrestrial environment is the most relevant (95% of both the initial loss and the final release). These shares reflect the final release rates specified above for lost macro-plastics, as the effects of the more reduced release rates applied to most micro-plastics losses (Section 4.5.5.8) is negligible, due to the irrelevant contribution of micro-plastics to the total loss. A similar situation is observed also for biodegradable films, where the total initial plastic loss is almost entirely released to the terrestrial environment, as lost macro-plastics (which dominate the total loss) are only released to this compartment (Section 4.5.5.8). The final release to the terrestrial environment thus also dominates the total final release to the environment (including both ocean and the terrestrial environment).

¹²¹ Including non-collected mulching film residues left on the field (10% of the film applied on land), as well as collected film subject to waste mismanagement (approximately 9% of the material collected, i.e. nearly 8% of that applied, if considering a collection rate of 90%) (see Section 4.5.5.8).

As shown in Table 4.14, the total plastic release to the terrestrial environment is largely dominated by macro-plastics released at End of Life, which account for more than 99% of the total release. Micro-plastics released throughout the upstream life cycle (via pellet loss and tyre abrasion) have only a minor role, instead. The same considerations almost equally applies also to the total plastic release to ocean associated with non-biodegradable mulching films (with a contribution of released macro-plastics of approximately 99%). Conversely, for biodegradable films, micro-plastics dominate the total plastic release to ocean, as 100% of the film applied on land, which is the only source of macro-plastics, was assumed to be released to the terrestrial environment after incorporation in soil (Section 4.5.5.8). Leaving this exception apart, the generally dominant role of the macro-plastics release is a consequence of the prevailing mass of generated and released macro-plastics compared to micro-plastics, since the macro-plastics loss directly depends on the total mass of product (film) used per functional unit (i.e. the reference flow), and is calculated based on a higher total "loss rate" compared to the considered micro-plastics sources¹²². Moreover, the final release rate to the terrestrial environment of mulching film lost as macro-plastic is higher compared to the final release rates of micro-plastics lost to the same compartment¹²³. Therefore, the mass of released macro-plastics is at least one order of magnitude higher than the mass of released micro-plastics, which instead only indirectly depends on the reference flow (through parameters related to the quantity of relevant lifecycle processes)¹²⁴, and is calculated based on (much) lower loss and (depending on the compartment and source) release rates (Section 4.5.5.8). Note, however, that the release of micro-plastics from tire abrasion was underestimated, due to the exclusion of the contribution of (as discussed in Section 3.5.5.6 of the Beverage Bottles case study): (a) most background transport activities; (b) intermediate transports between different process steps included within the vertically aggregated datasets used to model production of some polymers (i.e. virgin LDPE –affecting also partially recycled LDPE–, and PLA); and (c) any transport activities occurring within horizontally aggregated datasets.

Focusing on the sources of macro-plastics released to ocean and to the terrestrial environment (Figure 4.5a), for non-biodegradable film alternatives both direct product littering (i.e. residual uncollected film after use) and mismanagement of removed film waste provide a similar contribution to the estimated total release (i.e. 55% and 45%, respectively). This is because the loss rate via littering (littering rate; 10% of the applied film; Table 4.6) is comparable to the loss rate via waste mismanagement, equalling 8% of the applied film (i.e. 9.25% of the removed film, which is 90% of that applied; Table I.2 of the *Plastics LCA* method and Table 4.6). Overall, this leads to a comparable release from the two different sources. For biodegradable films, only direct product littering contributes to the total release to the terrestrial environment, as the only macro-plastics source is the film entirely incorporated in soil for biodegradation after use, while there is no collected film waste that may be mismanaged (Section 4.5.5.8). On the other hand, no macro-plastics are released to ocean for biodegradable alternatives, as the material left on the field as macro-plastics loss was assumed to be entirely released to the terrestrial environment after incorporation in soil, with no transfer to other compartments (i.e. ocean; Section 4.5.5.8).

¹²² The total "loss rate" of applied mulching film as macro-plastics at End of Life (calculated as a combination of the littering rate and of the mismanaged waste index) is equal to 18% for non-biodegradable films and to 100% for biodegradable ones (see Table 4.6 in this report, and Equation I.1 plus Table I.2 in the *Plastics LCA* method). Conversely, the total "loss rate" of micro-plastics from tire abrasion is in the range of 1.5-2.6% (depending on the vehicle; see Equation I.3 and Table I.3 in the *Plastics LCA* method). For plastic pellets from product manufacturing, the loss rate is even lower, equalling 0.1% (see Equation I.5 and Table I.5 in the *Plastics LCA* method).

¹²³ The final release rate to the terrestrial environment of mulching film lost as macro-plastic is equal to 95% for conventional films and to 100% for biodegradable ones (Table 4.6). The final release rate is instead equal to 74% for micro-plastics due to pellet losses, and to 84% for those generated from tire abrasion (Tables I.5 and I.4 in the *Plastics LCA* method, including the contribution of the release to freshwater sediments).

¹²⁴ Including the mass of plastic pellets used for product manufacturing (which depends on the quantity of this process per functional unit), and the amount of product or material transported during each modelled road transport activity (per functional unit) with the associated distance.

As for the micro-plastics release to ocean and to the terrestrial environment, the relative contribution of the different sources (loss of plastic pellets during product manufacturing and tire abrasion during road transport) generally depends on the configuration of the modelled supply-chain. The origin of the feedstock and the location of subsequent conversion processes of feedstock materials into final polymers are especially relevant, as they affect the contribution from transport activities across the life cycle. Moreover, the level of vertical disaggregation applied in the modelling of such upstream conversion processes plays an important role, as it affects the number of intermediate transport activities between different process steps which are modelled separately in the foreground inventory, and for which the contribution to micro-plastics generation can be quantified¹²⁵. Keeping this in mind, and considering the marginal role of micro-plastics within the total release to the environment (except for the release to ocean estimated for biodegradable film alternatives, as discussed above), it is observed that pellet losses during product manufacturing dominate the total release to both ocean (99% on average) and the terrestrial environment (93-95%) (Figure 4.5b). Micro-plastics generated from tire abrasion during road transport have a more limited role, instead (Figure 4.5b). However, the same considerations made above regarding the likely underestimate of this contribution still apply.

From a methodological perspective, it is noted how biodegradable mulching film alternatives (which are not removed from soil after use) generate a higher total loss and release to the terrestrial environment compared to non-biodegradable alternatives (which are in large part removed) (Table 4.14). The same applies to the macro-plastics loss and release to the terrestrial environment, which as discussed above dominate the total plastic loss and release, respectively (Table 4.14). These results are a consequence of the exclusion, from the applied quantification method (i.e. the *PLP method*) and indicators, of the fate of released macro-plastics, apart from any short-term redistribution among different environmental compartments (as discussed in Section 4.5.5.8). However, if the effects of biodegradation were accounted, the ultimate release to the terrestrial environment may be even significantly lower compared to the values estimated in this study for macro-plastics and for the resulting total plastic release. On the other hand, the macro-plastics release to ocean is equal to zero for biodegradable films (Table 4.14), as these were assumed not to be transferred to other compartments than soil (where they are incorporated for biodegradation) (Section 4.5.5.8). Conversely, non-biodegradable films lost as macro-plastics (residual film left on the field and collected film subject to waste mismanagement) were assumed to also reach ocean, albeit in a only minor share (i.e. 5%; Section 4.5.5.8). The release of macro-plastics to ocean and the resulting total plastic release to the same compartment are thus higher for non-biodegradable alternatives compared to biodegradable ones.

4.8.5 Sensitivity analysis

A sensitivity analysis was performed on a number of parameters, assumptions, and methodological choices, to evaluate their influence and the effects of their variation on the characterised impact assessment results of the affected LCA scenario(s). The following aspects were considered:

1. Thickness of conventional LDPE mulching film (and resulting collection and contamination rates);
2. Recycled content in R-LDPE mulching film;
3. Use of additives in starch-based mulching film;

¹²⁵ When vertically aggregated datasets were applied to model the process chain involved in the conversion of feedstock materials into the final polymer (i.e. for fossil-based LDPE and, partly, for recycled LDPE and PLA-based films), the contribution of intermediate transports between different process steps could not be quantified. Conversely, when such process steps were modelled individually (i.e. for starch-based film), the contribution of intermediate transports was accounted.

4. Feedstock origin for PLA used in PLA-based mulching film and resulting PLA production location;
5. Applied iLUC model and resulting GHG emission factors;
6. Handling of non-released biogenic carbon at End of Life;
7. Alternative End of Life scenarios.

The following sections present the results of the sensitivity analysis for each of the aspects above. This is done by comparing recalculated impacts of the affected scenario(s) with those of the base case assessment.

4.8.5.1 Thickness of conventional LDPE mulching film

As a base case, a thickness of 35 μm was assumed for conventional (non-biodegradable) mulching films. Such value was calculated as the rounded average of the typical thickness range reported by OWS (2017) for LDPE films used for vegetable cultivation, which varies between 15 and 50 μm . Considering this relatively high variability, and that the assumed film thickness directly affect the reference flow (i.e. the amount of film material required per functional unit), a sensitivity analysis was performed on this relevant parameter, to evaluate the effects on the results calculated for conventional mulching films. The analysis was limited to virgin LDPE film, considering both a possible decrease and increase in the film thickness, as detailed below and as summarised in Table 4.17. The effects of such changes in thickness on the collection and contamination rates of the film at End of Life were also taken into account (see Table 4.17), as these parameters depend (also) on thickness.

In the first instance, the thickness was reduced from 35 to 20 μm , according with the minimum nominal thickness required to comply with the European standard EN 13655 on thermoplastic mulching films recoverable after use. Therefore, a value of thickness very close to the lower end of the range reported by OWS (2017; i.e. 15 μm) was considered, corresponding to a new reference flow of 111 kg of LDPE film per functional unit (FU; instead of 194 kg/FU). Moreover, the collection rate of the film from soil after use was reduced from 90% to 75%, according with the thickness-specific estimate reported by OWS (2017) for a 20 μm film. On the other hand, the contamination rate of collected mulching film with soil was increased from 60% to 80%, based on the relationship between film thickness and the associated contamination rate determined following the approach described in Section 4.5.5.1 (i.e. linear regression on available literature values of contamination rate for specific film thicknesses).

A second case was then considered, where the thickness of virgin LDPE mulching film was increased from 35 to 50 μm , in line with the higher end of the range reported by OWS (2017). This corresponds to an increase in the reference flow from 194 to 277.5 kg of LDPE film per functional unit. The collection rate of the film from soil after use was maintained at 90%, in the absence of evidence justifying an increase with respect to the highest rate estimated in the literature for a 25 μm film (OWS, 2017). While this assumption may be conservative, it is in line with the approach followed in the base case, where a 90% collection rate was still assumed for film with a higher thickness of 35 μm (Section 4.5.5.1). Finally, the contamination rate of collected mulching film with soil was decreased from 60% to 40%, based on the lowest contamination rate reported in the literature (Steinmetz et al., 2016), which was attributed to a film with the highest thickness in the abovementioned range provided by OWS (2017; 10-50 μm), as assumed in this second sensitivity analysis.

The results of the analysis are presented in Table 4.18, which reports the characterised potential impacts of virgin LDPE mulching film as recalculated taking into account the changes described above, in comparison with those of the base case. The resulting percentage impacts variation with respect to the base case were also calculated and reported in the table.

Table 4.17. Assumed and calculated values of relevant parameters affected by the sensitivity analysis on the thickness of conventional non-biodegradable mulching film, as described in this section.

Parameter	Scenario		
	Base case	Sensitivity analysis 1	Sensitivity analysis 2
Thickness (μm)	35	20	50
Reference flow (kg/FU)	194	111	277.5
Collection rate from the field after use (%)	90	75	90
Contamination rate with soil residues (%)	60	80	40

Decreasing the thickness of virgin LDPE mulching film from 35 to 20 μm leads to an impact reduction in all the considered impact categories, with a decrease ranging from 29% (Eutrophication –freshwater) to 41% (Climate Change, Human Toxicity – cancer, and Ecotoxicity – freshwater) compared to the base case. The benefits from decreased material use for film manufacturing (-43% compared to the base case) are thus larger than the additional impacts due to reduced film collection from the field after use (from 90% to 75%) and to increased contamination with soil of collected film (which is more than doubled, increasing from 1.5 to 4 kg of soil per kg of film removed).

On the other hand, considering a thickness of 50 μm rather than 35 μm increases the impacts of virgin LDPE mulching film between 24% (Eutrophication – freshwater) and 42% (Ecotoxicity - freshwater), compared to the base case. The benefits of reduced contamination of collected film with soil (from 1.5 to 0.7 kg of soil per kg of film collected) are hence more than balanced by the additional impacts from increased material use for film manufacturing (+43% compared to the base case).

Table 4.18. Results of the sensitivity analysis on the thickness of virgin LDPE mulching film (decreased from 35 µm to 20 µm, and then increased to 50 µm) and on the resulting collection and contamination rates of the film at End of Life. Results are expressed per functional unit.

Impact category	35 µm thickness (base case)	20 µm thickness (SA-1) (¹)	Variation (%)	50 µm thickness (SA-2) (¹)	Variation (%)
Climate Change [kg CO ₂ eq.]	6.42E+02	3.80E+02	-41%	8.92E+02	39%
Ozone Depletion [kg CFC-11 eq.]	1.96E-07	1.38E-07	-30%	2.55E-07	30%
Human Toxicity - cancer [CTUh]	5.33E-06	3.13E-06	-41%	7.54E-06	41%
Human toxicity - non-cancer [CTUh]	2.81E-05	1.82E-05	-35%	3.79E-05	35%
Particulate matter [Disease incidence]	1.56E-05	1.01E-05	-35%	2.13E-05	37%
Ionising Radiation [kBq U ²³⁵ eq.]	4.19E+01	2.88E+01	-31%	5.71E+01	36%
Photochemical Ozone Formation [kg NMVOC eq.]	1.47E+00	9.12E-01	-38%	2.04E+00	39%
Acidification [mol of H ⁺ eq.]	1.56E+00	9.74E-01	-38%	2.16E+00	38%
Eutrophication - terrestrial [mol N eq.]	5.07E+00	3.17E+00	-37%	7.01E+00	38%
Eutrophication - freshwater [kg P eq.]	4.76E-03	3.39E-03	-29%	5.90E-03	24%
Eutrophication - marine [kg N eq.]	4.62E-01	2.93E-01	-37%	6.34E-01	37%
Ecotoxicity - freshwater [CTUe]	1.18E+02	6.95E+01	-41%	1.67E+02	42%
Land Use [Pt]	1.93E+03	1.26E+03	-35%	2.62E+03	36%
Water Use [m ³ world eq.]	1.22E+02	7.83E+01	-36%	1.63E+02	34%
Resource Use - mineral and metals [kg Sb eq.]	7.07E-05	4.45E-05	-37%	9.78E-05	38%
Resource Use - fossils [MJ]	1.45E+04	8.75E+03	-40%	2.05E+04	41%

(¹) SA: Sensitivity Analysis.

4.8.5.2 Recycled content in R-LDPE mulching film

This sensitivity analysis explores the effects of an increased recycled material content in partially recycled LDPE mulching film (S2). As a base case, a 35% recycled content was estimated to reflect the current average situation at the EU level, while assuming that no changes in thickness are needed compared to using 100% virgin material (Section 4.1). However, available estimates specifically conducted for mulching film provide a higher recycled content (i.e. 52% in 2018; Eunomia, 2021)¹²⁶, which was thus applied as a first alternative value for this parameter (rounding it to 50%) in this sensitivity analysis. Moreover, the effects of a potential complete substitution of virgin with recycled material (i.e. a 100% recycled content) were investigated. While it is acknowledged that this is a hypothetical and optimistic scenario for the current situation, it was considered relevant to evaluate the potential effects from pursuing such an increased recycled material content, under the assumption that this would be technically feasible within the existing recycling infrastructure and through currently available manufacturing technology.

The results of the analysis are presented in Table 4.19, where recalculated potential impacts of (partially) recycled LDPE mulching film are reported along with those of the base case scenario. The resulting percentage impact variation with respect to such base case scenario is also calculated and reported.

Increasing the recycled content from 35% to 50% only marginally affects the potential impacts of partially recycled LDPE mulching film, as all impact categories show a non-significant impact variation compared to the base case (i.e. lower than 10%, or even 5% in several categories)¹²⁷. This reflects the limited increase initially considered for the recycled content in the sensitivity analysis, which is equal to only 15% (i.e. from 35% to 50%).

On the other hand, considering a 100% recycled material content significantly reduces the impact of (partially) recycled LDPE mulching film in the vast majority of the assessed impact categories (i.e. all except Eutrophication – freshwater and Ozone Depletion), with a decrease ranging from 13% (Land Use) to 39% (Resource Use – fossils) compared to the base case. For Eutrophication – freshwater the decrease is not significant (-9%), while for Ozone Depletion an impact increase is observed (+25% compared to the base case).

¹²⁶ The estimate is based on the total recycled material use for mulching film manufacturing in Europe quantified by Plastics Recyclers Europe for 2018 (i.e. 43 kt), and on a total plastic consumption for agricultural mulching film equal to 83 kt in 2019 (according to APE Europe data).

¹²⁷ Variations below 10% can be considered as not significant, in light of the uncertainty that generally affect a LCA study both at the inventory and at the impact assessment level.

Table 4.19. Results of the sensitivity analysis on the recycled material content in partially recycled LDPE mulching film, increased from 35% to 50% and 100%. Results are expressed per functional unit.

Impact category	35% R-LDPE film (base case)	50% R-LDPE film (SA-1) (¹)	Variation (%)	100% R-LDPE film (SA-2) (¹)	Variation (%)
Climate Change [kg CO ₂ eq.]	5.99E+02	5.81E+02	-3%	5.18E+02	-14%
Ozone Depletion [kg CFC-11 eq.]	2.27E-07	2.39E-07	5%	2.83E-07	25%
Human Toxicity - cancer [CTUh]	4.50E-06	4.14E-06	-8%	2.92E-06	-35%
Human toxicity - non-cancer [CTUh]	2.50E-05	2.37E-05	-5%	1.91E-05	-24%
Particulate matter [Disease incidence]	1.43E-05	1.37E-05	-4%	1.14E-05	-20%
Ionising Radiation [kBq U ²³⁵ eq.]	3.65E+01	3.42E+01	-6%	2.64E+01	-28%
Photochemical Ozone Formation [kg NMVOC eq.]	1.31E+00	1.24E+00	-5%	9.66E-01	-26%
Acidification [mol of H ⁺ eq.]	1.37E+00	1.29E+00	-6%	9.58E-01	-30%
Eutrophication - terrestrial [mol N eq.]	4.65E+00	4.46E+00	-4%	3.67E+00	-21%
Eutrophication - freshwater [kg P eq.]	4.56E-03	4.47E-03	-2%	4.17E-03	-9%
Eutrophication - marine [kg N eq.]	4.22E-01	4.05E-01	-4%	3.30E-01	-22%
Ecotoxicity - freshwater [CTUe]	1.02E+02	9.47E+01	-7%	7.09E+01	-30%
Land Use [Pt]	1.82E+03	1.77E+03	-3%	1.59E+03	-13%
Water Use [m ³ world eq.]	1.12E+02	1.07E+02	-4%	9.26E+01	-17%
Resource Use - mineral and metals [kg Sb eq.]	6.16E-05	5.78E-05	-6%	4.46E-05	-28%
Resource Use - fossils [MJ]	1.20E+04	1.09E+04	-9%	7.29E+03	-39%

(¹) SA: Sensitivity Analysis.

4.8.5.3 Use of additives in starch-based mulching film

This sensitivity analysis evaluates the effects of excluding additives (mainly plasticisers) from the composition of Thermoplastic Starch (TPS) used as a copolymer, along with PBAT, in the manufacturing of starch-based mulching film. This choice was made considering the uncertainties and approximations performed in the modelling of additives¹²⁸, and their non-negligible contribution to the overall lifecycle impacts of this mulching film alternative in several impact categories (e.g. Ozone Depletion, Resource Use – minerals and metals, Land Use, Eutrophication – freshwater, Eutrophication – marine, and, to a lower extent, Human Toxicity – non-cancer, Ecotoxicity- freshwater and Water Use). For the purpose of simplicity, the exclusion was limited to the Polymer Production stage (assuming that TPS is entirely made of starch and that no additives are used), while for End of Life modelling the composition was not changed. This is not expected to have significant effects on the results, since the overall elemental composition of the starch-based polymer (which is the most relevant parameter for End of Life modelling) is only marginally affected by the considered change in the TPS composition (being the shares of single constituting elements almost comparable between additives and starch).

The characterised potential impacts of starch-based mulching film, recalculated after excluding additives as described above, are reported in Table 4.20 along with base-case impacts. The percentage impact variation with respect to the base case was also calculated and reported in the table.

When additives are excluded from the production of thermoplastic starch, no relevant changes are observed in most impact categories, which show negative or positive impact variations lower than 10% (and hence not significant in light of the uncertainty that generally affect a LCA study both at the inventory and at the impact assessment level). An impact reduction takes place only in three impact categories, including Ozone Depletion (-58%), Resource Use – mineral and metals (-25%), and Land Use (-11%), while in Ecotoxicity – freshwater the impact is moderately increased (+11%).

¹²⁸ Related to the assumed total share of additives, the specific substances considered, their relative proportions, and the modelling approximations performed for some of them (see Section 4.5.2.3.1).

Table 4.20. Results of the sensitivity analysis on the use of additives in starch-based mulching film (considering their exclusion from the production of thermoplastic starch). Results are expressed per functional unit.

Impact category	Production of TPS⁽¹⁾ including additives (base case)	Production of TPS⁽¹⁾ excluding additives (SA) ⁽²⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	6.64E+02	6.56E+02	-1%
Ozone Depletion [kg CFC-11 eq.]	4.96E-07	2.07E-07	-58%
Human Toxicity - cancer [CTUh]	3.32E-06	3.32E-06	0%
Human toxicity - non-cancer [CTUh]	1.01E-04	1.03E-04	2%
Particulate matter [Disease incidence]	1.30E-05	1.27E-05	-2%
Ionising Radiation [kBq U ²³⁵ eq.]	3.34E+01	3.33E+01	0%
Photochemical Ozone Formation [kg NMVOC eq.]	1.12E+00	1.07E+00	-4%
Acidification [mol of H ⁺ eq.]	1.56E+00	1.53E+00	-2%
Eutrophication - terrestrial [mol N eq.]	5.15E+00	5.12E+00	-1%
Eutrophication - freshwater [kg P eq.]	1.33E-02	1.25E-02	-6%
Eutrophication - marine [kg N eq.]	6.92E-01	6.62E-01	-4%
Ecotoxicity - freshwater [CTUe]	8.09E+02	9.01E+02	11%
Land Use [Pt]	1.27E+04	1.13E+04	-11%
Water Use [m ³ world eq.]	1.35E+02	1.45E+02	7%
Resource Use - mineral and metals [kg Sb eq.]	9.82E-05	7.32E-05	-25%
Resource Use - fossils [MJ]	1.06E+04	1.04E+04	-2%

⁽¹⁾ TPS: Thermoplastic Starch.

⁽²⁾ SA: Sensitivity Analysis.

4.8.5.4 Feedstock origin for PLA and resulting PLA production location

This sensitivity analysis evaluates the effects of relying on maize grown in Europe rather than in the US for the production of PLA used as a copolymer, along with PBAT, in PLA-based mulching film. As a consequence, also PLA production was assumed to take place in Europe and not in the US (as considered in the base case assessment). For modelling purposes, the vertically aggregated, cradle-to-gate dataset originally applied to represent the entire process chain from cultivation of maize to PLA production in the US (Sections 4.5.1.3 and 4.5.2.3.2) could not be adjusted to reflect European conditions. Therefore, it was replaced with a combination of two separate datasets referring to EU-28 practices and conditions. For maize grain production, an aggregated dataset from the GaBi database was applied¹²⁹, as it was considered to be more consistent with the data and modelling approach adopted for maize cultivation in the replaced vertically aggregated dataset, compared to the available EF-compliant dataset for maize cultivation in the EU. For subsequent PLA production from EU maize, an aggregated, gate-to-gate dataset provided by Thinkstep was applied¹³⁰. The dataset covers the processes of maize wet

¹²⁹ [EU-28] Corn grains, at farm (12% H₂O content); corn cultivation, transport, drying and storage; single producer, at plant; 12% H₂O.

¹³⁰ [EU-28] Polylactic acid (PLA) (Polylactide, continuous process) - open flow corn grains; lactide production from corn, continuous process | single route, at plant | 1210–1430 kg/m³, 72.06 g/mol per repeating unit.

milling for starch production, starch hydrolysis to glucose, its fermentation to lactic acid, oligomerisation of lactic acid to lactide monomer, and final polymerisation of the latter to PLA. For the maize wet milling process, economic allocation is applied in the dataset to handle the different co-products, while for the different downstream conversion processes no allocation is required. A distance of 100 km was then assumed to be covered by lorry (> 32t) to transport maize grains from the farm to the processing site. Finally, transoceanic transport of PLA granulate from US to the manufacturing site in Europe was replaced with intra-EU transport of granulate, according to the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers to factories/users within Europe. This includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

The results of the analysis are presented in Table 4.21, which reports the characterised potential impacts of PLA-based mulching film, as recalculated after the changes described above. The impacts of the base case of the same scenario, and the resulting percentage impact variations compared to the latter, are also reported.

Table 4.21. Results of the sensitivity analysis on feedstock origin and production location for PLA used in PLA-based mulching film (i.e. considering EU rather than US). Results are expressed per functional unit.

Impact category	Maize and PLA production in the US (base case)	Maize and PLA production in the EU (SA) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	5.27E+02	6.59E+02	25%
Ozone Depletion [kg CFC-11 eq.]	7.90E-08	7.10E-08	-10%
Human Toxicity - cancer [CTUh]	1.19E-05	2.11E-06	-82%
Human toxicity - non-cancer [CTUh]	2.57E-05	9.40E-05	266%
Particulate matter [Disease incidence]	9.54E-06	1.33E-05	39%
Ionising Radiation [kBq U ²³⁵ eq.]	2.87E+01	3.69E+01	29%
Photochemical Ozone Formation [kg NMVOC eq.]	1.07E+00	1.10E+00	3%
Acidification [mol of H ⁺ eq.]	1.21E+00	1.40E+00	16%
Eutrophication - terrestrial [mol N eq.]	3.65E+00	4.61E+00	26%
Eutrophication - freshwater [kg P eq.]	2.11E-03	2.03E-02	862%
Eutrophication - marine [kg N eq.]	3.56E-01	4.86E-01	37%
Ecotoxicity - freshwater [CTUe]	3.34E+02	1.33E+02	-60%
Land Use [Pt]	1.12E+04	3.52E+04	214%
Water Use [m ³ world eq.]	7.80E+01	2.47E+02	217%
Resource Use - mineral and metals [kg Sb eq.]	7.12E-05	2.45E-04	244%
Resource Use - fossils [MJ]	8.31E+03	1.05E+04	26%

⁽¹⁾ SA: Sensitivity Analysis.

Producing PLA in Europe from maize grown in same region raises the impact of PLA-based mulching film in most (twelve) impact categories, with more than half of them showing an increase between 16% (Acidification) and 39% (Particulate Matter). In the other five categories, the increase is even higher, being mostly in the range of 214-266%, although it is as high as 862% in Eutrophication – freshwater. An impact decrease

takes place in only three categories, including Human Toxicity – cancer (-82%), Ecotoxicity – freshwater (-60%) and, to a minor extent, in Ozone Depletion (-10%). In Photochemical Ozone Formation, no significant changes are observed, with an impact variation lower than 10%. It must be noted, however, that these results may be even largely affected by any discrepancies in the inventories (datasets) applied to model maize grain and PLA production in the EU rather than in the US (e.g. in terms of data generation and modelling approach). While being developed by the same data provider (i.e. Thinkstep), they rely on different data sources and aggregation levels, and even relevant discrepancies may thus exist.

4.8.5.5 Applied iLUC model and resulting GHG emission factors

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the contribution of GHG emissions from iLUC to the Climate Change impact indicator. The iLUC GHG emission factors calculated with the model of Schmidt et al. (2015) were applied in place of the emission factors based on EC (2015) applied in the base case (Table 4.7, Section 4.5.6). The results are displayed in Table 4.22, limited to the Climate Change impact category, which is the only one affected by this sensitivity analysis.

While the iLUC contribution (kg CO₂ eq./FU) estimated by applying the model of Schmidt et al. (2015) is more than doubled compared with the base case (+111-114%), the overall results are not significantly affected. The increase in the total Climate Change impact of bio-based mulching film alternatives, due to the additional contribution of GHG emissions from iLUC, is indeed still marginal, equalling 0.9% for starch-based film, and 1.4% for PLA-based film. In the base case, a comparable increase of 0.4% and 0.7% was observed, respectively, for the two alternatives. The results of this case study can thus be considered reasonably robust with respect to the estimated contribution of GHG emissions from iLUC to the Climate Change impact (which is, as discussed, negligible).

Table 4.22. Results of the sensitivity analysis on the emission factors applied to quantify the contribution of GHG emissions from iLUC to the Climate Change impact indicator, and resulting total Climate Change impact of the affected scenarios (in brackets). Results are not intended to compare the different mulching film scenarios.

Scenario	Emission factors based on EC (2015) (base case) [kg CO ₂ eq./FU]	Emission factors from the model of Schmidt et al. (2015) (SA) ⁽¹⁾ [kg CO ₂ eq./FU]	Variation (%)
Starch-based mulching film	2.92 (668)	6.24 (671)	+114 (+0.50)
PLA-based mulching film	3.53 (531)	7.44 (534)	+111 (+0.74)

⁽¹⁾ SA: Sensitivity Analysis.

4.8.5.6 Handling of non-released biogenic carbon at End of Life

This sensitivity analysis explores the effects of accounting for the impact of biogenic carbon not released from soil after 100 years of in-situ biodegradation of partially bio-based mulching film alternatives (i.e. starch-based and PLA-based films) on the Climate Change impact indicator. In this study, emissions from in-situ biodegradation were modelled considering a time horizon of 100 years from the beginning of the process (incorporation in soil), and (biogenic) carbon in the product that is not mineralised during such time horizon was considered to be never released from soil. However, the effects of non-released biogenic carbon are not captured in the Climate Change impact results calculated as a base case, since characterisation factors for biogenic CO₂ emissions and removals are set to zero in the *Plastics LCA* method (fully conforming to the PEF method). To better understand the implications of this methodological choice, the

Climate Change impact indicator of the two mentioned bio-based alternatives was thus recalculated accounting for the effects of non-released biogenic carbon.

For this alternative calculation, a specific CO₂ uptake was modelled in the in-situ biodegradation inventories of starch-based and PLA-based mulching films, expressing the net amount of biogenic CO₂ taken up from biomass embodied in the product, and ultimately not released (again as CO₂) during film biodegradation in soil. This uptake was then characterised applying a characterisation factor of -1 kg CO₂ eq. per kg CO₂ not released. The uptake was calculated based on the biogenic carbon content in the product (i.e. in the respective polymer), and assuming an overall biodegradation (i.e. carbon mineralisation) rate over 100 years equal to 90% for both starch-based and PLA-based film (see Section 4.5.5.7). Considering a biogenic carbon content equal to 17.9% for starch-based film (out of a total carbon content of 50.2%), and to 22.3% for PLA-based film (51.8% total carbon), a specific carbon uptake equal to 0.0657 kg CO₂/kg starch-based film and 0.0817 kg CO₂/kg PLA-based film was thus calculated, respectively.

The results of the analysis are presented in Table 4.23, considering that the EU-average End of Life scenario applied in the base case for biodegradable mulching film alternatives already includes only in-situ biodegradation as an End of Life option. Therefore, no alternative calculations considering the application of in-situ biodegradation as an individual "100%" End of Life option had to be conducted to evaluate this specific situation.

When accounting for the contribution of non-released biogenic carbon after 100 years of in-situ biodegradation, the Climate Change impact of both starch-based and PLA-based mulching film is only slightly decreased (-1.1% and -1.3%, respectively). This is mostly a consequence of the relatively minor portion of (biogenic) carbon not mineralised over 100 years of in-situ biodegradation (10%, according with the assumed mineralisation rate of 90%), and of the limited biogenic carbon content in the polymers (18%–22%, as discussed above). The results are thus clearly dependent on these parameters, and could even significantly change in case of large variations in the applied mineralisation rate and/or in the biogenic carbon content of the product.

Table 4.23. Results of the sensitivity analysis on the handling of non-released biogenic carbon at End of Life (in-situ biodegradation) for starch-based and PLA-based mulching films. Only the Climate Change indicator is affected, and presented in the table. Results are not intended to compare the different mulching film scenarios.

Scenarios	Effects of non-released biogenic C <u>not</u> accounted (base case) [kg CO ₂ eq./FU]	Effects of non-released biogenic C accounted (SA) ⁽¹⁾ [kg CO ₂ eq./FU]	Variation (%)
Starch-based mulching film	665	657	-1.1%
PLA-based mulching film	527	520	-1.3%

⁽¹⁾ SA: Sensitivity Analysis.

4.8.5.7 Alternative End of Life scenarios

This sensitivity analysis evaluates the effects of individually applying each End of Life option specified in Table 4.1 as an independent (100%) End of Life scenario replacing the EU-average scenario considered as a base case. The analysis is limited to conventional, non-biodegradable LDPE mulching films (virgin and partially recycled), as for biodegradable films the EU-average scenario already includes only in-situ biodegradation as a single End of Life option, and no additional specific investigation is needed. Conversely, for non-biodegradable films, incineration and landfilling were investigated individually, considering that they only apply to the amount of film collected from soil after use (i.e. to 90% of that applied, as in the base case). Albeit currently not (or very

scarcely) performed (see Section 4.5.5.1), mechanical recycling was also investigated as a specific, independent End of Life scenario for non-biodegradable alternatives, to better understand the effects of implementing this option on the resulting potential impacts, although in an approximate manner. No specific information and data on contaminated mulching film sorting and recycling were indeed available, and a number of approximations had to be made to model these processes (see Section 4.5.5.3) and to calculate the impacts of the recycling pathway.

The main purpose of this sensitivity analysis is to evaluate how the potential impacts of non-biodegradable mulching film alternatives are affected by changes in the applied End of Life scenario. However, in reality the different considered End of Life options would be hardly implemented individually, but in combination (e.g. as reflected in the assumed EU-average End of Life scenario). The detailed numerical results are separately presented for virgin LDPE mulching film and for 35% recycled LDPE film in Tables 4.24 and 4.25, respectively. A more synthetic overview is also provided in Figures E.2.1–E.2.3 in Annex E.2.

Note that these results should not be interpreted as a direct comparison among alternative End of Life options for non-biodegradable mulching films, since the evaluation applies a “product perspective”, and burdens/benefits of some End of Life options (e.g. recycling) are partitioned between different (consecutive) product life cycles. This prevents from capturing the full environmental implications of having a given waste stream routed to such End of Life options (especially in case of recycling). Conversely, in a waste management LCA of alternative End of Life options for the product (e.g. based on a functional unit of 1 tonne of product waste to be managed), each pathway would be assigned the full burdens and benefits it involves. In such case, there is indeed no need to break mass and energy flows between the waste management system providing recovered material/energy, and the product system using them (i.e. a “system perspective” is applied, and no allocation is needed). In this perspective, the total (system-wise) benefits associated with the End of Life pathway “100% recycling” would hence be higher than those considered to calculate the results presented in this section, based on a product perspective.

Focusing on the results, for both mulching film alternatives none of the product scenarios individually applying the three considered End of Life options (i.e. mechanical recycling, incineration and landfilling) can be identified as preferable across all the assessed impact categories. Moreover, in several categories, no relevant changes are observed in the overall impact of some of the alternative product scenarios implementing individual End of Life options, as impact differences between such scenarios are lower than 10% (which are considered not significant). Within this overall framework, the following considerations can be made.

For both virgin and partially recycled LDPE film, the scenario applying 100% mechanical recycling is preferable only in four impact categories, including Human Toxicity – cancer, Human Toxicity – non-cancer, Eutrophication – freshwater, and Ecotoxicity – freshwater. In most of the remaining categories (i.e. seven), the recycling scenario is comparable with that applying incineration, but it is still preferable to the landfilling one. Less frequently (i.e. in Climate Change and Water Use), the recycling scenario is instead comparable with that applying landfilling, while being preferable to the incineration one. Finally, in Ozone Depletion, the scenario applying 100% recycling is the least favourable one, while in Ionising Radiation and Acidification it has an intermediate impact between the incineration scenario (which is the preferable one) and that applying landfilling.

Similarly to recycling, the product scenario implementing 100% incineration is preferable only in two impact categories (Ionising Radiation and Acidification), while in most of the other ones (i.e. seven categories) it is comparable with the 100% recycling scenario and preferable to the 100% landfilling one. On the other hand, in Human Toxicity – cancer and Ecotoxicity – freshwater, the incineration scenario is comparable with that applying landfilling (which has the highest impact), while in Climate Change and Water Use the incineration scenario itself is the least preferable one. Finally, in Ozone Depletion, Human

Toxicity – non-cancer and Eutrophication – freshwater, the scenario applying 100% incineration has an intermediate performance between the recycling scenario (which has the lowest impact in all these categories except Ozone depletion) and the landfilling one.

In many impact categories (i.e. eleven), the product scenario applying 100% landfilling is the worst one for both mulching film alternatives. In Human Toxicity – cancer and Ecotoxicity – freshwater, the landfilling scenario still shows the highest impact, albeit it is comparable with that of the incineration scenario. Overall, this is in line with the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option to be pursued (EC, 2008). Few exceptions to the above are represented by Ozone Depletion (where the 100% landfilling scenario shows the lowest impact, albeit almost comparable with that of the incineration one), as well as by Climate Change and Water Use (where 100% landfilling is comparable with the recycling scenario, which has the lowest impact).

Table 4.24. Characterised potential impacts of fossil-based LDPE mulching film for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	6.42E+02	4.87E+02	7.51E+02	5.37E+02
Ozone Depletion [kg CFC-11 eq.]	1.96E-07	2.61E-07	2.11E-07	1.83E-07
Human Toxicity - cancer [CTUh]	5.33E-06	4.25E-06	5.19E-06	5.47E-06
Human toxicity - non-cancer [CTUh]	2.81E-05	2.12E-05	2.45E-05	3.16E-05
Particulate matter [Disease incidence]	1.56E-05	1.44E-05	1.36E-05	1.75E-05
Ionising Radiation [kBq U ²³⁵ eq.]	4.19E+01	5.15E+01	2.13E+01	6.17E+01
Photochemical Ozone Formation [kg NMVOC eq.]	1.47E+00	1.34E+00	1.31E+00	1.62E+00
Acidification [mol of H ⁺ eq.]	1.56E+00	1.48E+00	1.30E+00	1.81E+00
Eutrophication - terrestrial [mol N eq.]	5.07E+00	4.70E+00	4.64E+00	5.50E+00
Eutrophication - freshwater [kg P eq.]	4.76E-03	1.95E-03	3.04E-03	6.41E-03
Eutrophication - marine [kg N eq.]	4.62E-01	4.26E-01	4.09E-01	5.13E-01
Ecotoxicity - freshwater [CTUe]	1.18E+02	9.96E+01	1.13E+02	1.23E+02
Land Use [Pt]	1.93E+03	1.75E+03	1.81E+03	2.06E+03
Water Use [m ³ world eq.]	1.22E+02	8.15E+01	1.56E+02	8.92E+01
Resource Use - mineral and metals [kg Sb eq.]	7.07E-05	6.58E-05	6.17E-05	7.94E-05
Resource Use - fossils [MJ]	1.45E+04	1.30E+04	1.24E+04	1.66E+04

Table 4.25. Characterised potential impacts of 35% recycled LDPE mulching film for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	5.99E+02	4.44E+02	7.09E+02	4.94E+02
Ozone Depletion [kg CFC-11 eq.]	2.27E-07	2.91E-07	2.41E-07	2.13E-07
Human Toxicity - cancer [CTUh]	4.50E-06	3.42E-06	4.36E-06	4.64E-06
Human toxicity - non-cancer [CTUh]	2.50E-05	1.81E-05	2.14E-05	2.85E-05
Particulate matter [Disease incidence]	1.43E-05	1.30E-05	1.23E-05	1.62E-05
Ionising Radiation [kBq U ²³⁵ eq.]	3.65E+01	4.60E+01	1.59E+01	5.63E+01
Photochemical Ozone Formation [kg NMVOC eq.]	1.31E+00	1.18E+00	1.15E+00	1.46E+00
Acidification [mol of H ⁺ eq.]	1.37E+00	1.29E+00	1.11E+00	1.62E+00
Eutrophication - terrestrial [mol N eq.]	4.65E+00	4.28E+00	4.21E+00	5.07E+00
Eutrophication - freshwater [kg P eq.]	4.56E-03	1.75E-03	2.84E-03	6.21E-03
Eutrophication - marine [kg N eq.]	4.22E-01	3.86E-01	3.69E-01	4.73E-01
Ecotoxicity - freshwater [CTUe]	1.02E+02	8.33E+01	9.67E+01	1.06E+02
Land Use [Pt]	1.82E+03	1.63E+03	1.69E+03	1.94E+03
Water Use [m ³ world eq.]	1.12E+02	7.12E+01	1.46E+02	7.89E+01
Resource Use - mineral and metals [kg Sb eq.]	6.16E-05	5.68E-05	5.26E-05	7.03E-05
Resource Use - fossils [MJ]	1.20E+04	1.05E+04	9.90E+03	1.40E+04

5 Case study 3: Insulation boards for buildings

This case study focuses on insulation boards for buildings, as an example of long-lifetime product representing the second largest end-use market for plastic polymers in the EU (covering approximately 20% of the total plastic demand by converters in the region; PlasticsEurope, 2019). Currently, plastic foams account for nearly 40% of the European market for building insulation materials, with the remaining share being mostly covered by mineral materials such as glass and stone wool (see the figures elaborated in Pavel and Blagoeva, 2018). According with the same data, Expanded Polystyrene (EPS) is the most widely used plastic material (66%), followed by Polyurethane (PUR; 19.5%) and Extruded Polystyrene (XPS; 14.5%). Insulation boards made out of virgin EPS and PUR are thus considered as a basis for the assessment in this case study, reflecting current practice. Moreover, additional product scenarios were investigated, considering the use of alternative feedstock sources for these polymers (plastic waste, biomass and carbon dioxide), or of other applicable materials based on such feedstock (see Section 5.1 for details). It should be noted that, while some of these scenarios exemplify products that have no relevant market shares in the EU today, or pathways that are not yet available at the commercial scale, this is in line with the aim of the study to illustrate applicability of the *Plastics LCA* method to plastic products relying on different feedstock sources and materials (reflecting current conditions), and to ensure that modelling rules for all most relevant feedstock sources covered in the method are properly demonstrated.

5.1 Assessed scenarios

In this case study, the potential impacts of insulation boards made from different plastic materials and/or feedstock sources were individually quantified by assessing a number of specific product scenarios (Table 5.1). The two most commonly used materials at present, i.e. EPS and PUR (Pavel and Blagoeva, 2018), were initially considered (Scenarios 1 and 2; S1 and S2), assuming production entirely (PUR) or mostly (EPS) from virgin, fossil-based feedstock. The use of post-consumer plastic waste as a feedstock was then investigated for EPS (R-EPS; Scenario 3 – S3) and for PET (R-PET; Scenario 4 – S4), which for insulation boards was found to be only used in recycled form and hence does not have a virgin counterpart. While different shares of recycled and virgin material may be mixed for the manufacturing of these boards, a 100% recycled content was assumed, as this is already in practice in some applications (e.g. see 100% R-EPS boards described in Benchmark Foam, 2020a, and 100% R-PET boards illustrated in Armacell, 2020a). The modelled scenarios were thus based on the features of specific products on the market, and did not aim at reflecting the average recycled content and physical properties of R-EPS and R-PET boards at the EU level (which could not be determined, given the limited market of these applications and the scarce information available). As such, the scenarios should therefore be considered as an example, rather than a market-average representation of R-EPS and R-PET boards used in the EU.

A partly bio-based alternative to fossil-based insulation material, particularly PUR, was assessed in Scenario 5 (S5), where soy-based PUR was considered. This is obtained by replacing fossil-based polyols with soy-based polyols in the PUR production process. Soybean grown in the EU was considered as a feedstock for bio-based polyols, according with the geographical scope of the study and available life cycle inventory data for crude soybean oil production, while assuming that sourcing and conversion of the feedstock occur locally. However, the global market/production of soy-based polyols is still very small, and current EU-production is reported to be limited or even null (Spekreijse et al., 2019). For this reason, little information was available on feedstock origin, and data to compile a complete and representative life cycle inventory of the polyol production process out of refined soybean oil were also scarce. Only partial data referring to a specific producer in the US could be applied (see Section 5.5.2.3) and these are not representative of average (EU) industrial production of soy-based polyols. Therefore, overall, the assessed scenario only exemplified the use of bio-based feedstock for the manufacturing of PUR boards, did not necessarily reflect the current average market

situation (e.g. any applied mix of relevant feedstock sources), and the associated results should be interpreted and used carefully.

Table 5.1. LCA scenarios assessed for the insulation boards case study and respective End of Life options and scenarios.

Scenario	Polymer	Monomer(s)	Feedstock	End of Life options /scenario ⁽¹⁾
1 - Conventional polymer 1	EPS (2% recycled content)	Styrene	Fossil-based (crude oil/natural gas) Waste EPS	Incineration (45%) Landfilling (55%) (Recycling – 0%)
2 - Conventional polymer 2	PUR	Polyols ⁽²⁾ MDI ⁽³⁾	Fossil-based (crude oil/natural gas)	Incineration (45%) Landfilling (55%) (Recycling – 0%)
3 - Alternative polymer 1	R-EPS (100% recycled content)	Styrene	Waste EPS (post-consumer)	Incineration (45%) Landfilling (55%) (Recycling – 0%)
4 - Alternative polymer 2	R-PET (100% recycled content)	MEG ⁽⁴⁾ PTA ⁽⁵⁾	Waste PET (post-consumer)	Incineration (45%) Landfilling (55%) (Recycling – 0%)
5 - Alternative polymer 3	Bio-PUR ⁽⁶⁾ (39% bio-based)	Bio-based polyols ⁽²⁾ MDI ⁽³⁾	Soybean (EU) Fossil-based (crude oil/natural gas)	Incineration (45%) Landfilling (55%) (Recycling – 0%)
6 - Alternative polymer 4	CO ₂ -PUR ⁽⁶⁾ (6% CO ₂ -based)	CO ₂ -based polyols ⁽²⁾ MDI ⁽³⁾	Captured CO ₂ (fossil-based) Fossil-based (crude oil/natural gas)	Incineration (45%) Landfilling (55%) (Recycling – 0%)

⁽¹⁾ The impacts of each scenario were calculated considering an EU-average End of Life scenario combining the listed End of Life options according with the reported shares (conforming with the assumptions taken in the PEFCRs for thermal insulation products; Avnir, 2019). A sensitivity analysis individually considering the application of each listed option plus mechanical recycling was also conducted.

⁽²⁾ Polyols are intermediates in the production of PUR, along with methylene diphenyl di-isocyanate.

⁽³⁾ MDI: Methylene Diphenyl Di-isocyanate.

⁽⁴⁾ MEG: Mono Ethylene Glycol.

⁽⁵⁾ PTA: Purified Terephthalic Acid.

⁽⁶⁾ Assuming the same formulation as conventional (fossil-based) PUR in terms of polyols/MDI ratio, and that bio-based or CO₂-based polyols have the same characteristics (e.g. OH number) as aromatic polyols.

Finally, the use of CO₂ captured from point emission sources as a feedstock for polyols used in PUR production was investigated in Scenario 6 (S6). Capture from coal-fired power plants was considered as a base case, being the most abundant stationary source of CO₂ in Europe (Von der Assen et al., 2016) and worldwide (IPCC, 2005). The use of an alternative, more concentrate, but less abundant source was explored in a dedicated sensitivity analysis, where CO₂ was derived from ammonia producing plants (Section 5.8.5.2). In both cases, captured CO₂ partially replaced propylene oxide conventionally used for the production of polyols, following the alternative process route described in Fernandez-Dacosta et al. (2017). Due to the innovative nature of this route, the life cycle inventory data applied in this study were mostly derived from the results of process modelling available in the literature, rather than from real industrial-scale facilities. While certain process optimisation strategies might have been taken into account in the generation of such data (e.g. energy integration), they are not representative of commercial scale production (in contrast to data available for fossil-based polyols used in

virgin PUR boards), and do not account for possible (efficiency) improvement due to upscaling and further process optimisation. Therefore, impact assessment results calculated in this study for CO₂-based PUR boards need to be interpreted with caution, taking into account these limitations.

Regarding the End of Life of boards, all treatment and disposal options currently applied at the EU level to insulation products (as identified by Avnir, 2019) were considered, including incineration and landfilling. These options were combined as reported in Table 5.1, to define an EU-average End of Life scenario that was applied, as a base case, to quantify the potential impacts of each investigated scenario (see Section 5.5.5.1 for details). Moreover, in a sensitivity analysis, scenario impacts were recalculated by individually applying each considered End of Life option, including also mechanical recycling as a potentially applicable option for insulation boards (Section 5.8.5.5).

5.2 Functional Unit and reference flow

The main function of the studied product is providing thermal insulation in buildings. Particularly, in this case study, pitched roofs were considered as reference part of buildings to be insulated. The functional unit was thus defined as: “*providing thermal insulation for 1 m² of pitched roof in temperate EU regions, ensuring a thermal resistance (R) equal to 7.14 m²·K·W⁻¹ (or 0.14 W·m⁻²·K⁻¹), over a designed life span (service life) of 50 years*” (Table 5.2). This type of functional unit is typically used in LCAs of insulation solutions and allows taking into account the amount and volume of insulation material necessary to provide a given thermal resistance throughout the insulation life span (here assumed to be 50 years, according with the expected average design life of buildings) based on the insulating and other relevant physical properties of the specific material (see e.g. Avnir, 2019 and Pargana et al., 2014). Particularly, the thermal resistance value of 7.14 m²·K·W⁻¹ (or 0.14 W·m⁻²·K⁻¹) reflects the average performance normally required under temperate EU conditions, typically ranging from 3 to 10 m²·K·W⁻¹ (or 0.1 to 0.33 W·m⁻²·K⁻¹). For additional explanations on this value, the reader is referred to Avnir (2019).

Table 5.2. Functional unit defined for insulation boards LCA scenarios.

Aspect	Description
“ What ” (function or service provided)	Providing thermal insulation in buildings pitched roofs
“ How much ” (extent of the function or service)	To a surface equal to 1 m ² of pitched roof
“ How well ” (expected level of quality of the function or service)	Providing a thermal resistance (R) equal to 7.14 m ² ·K·W ⁻¹ (or 0.14 W·m ⁻² ·K ⁻¹)
“ How long ” (duration/lifetime of the function or service)	Throughout a life span (service life) of 50 years
“ Where ” (location/geography of the function or service)	In temperate EU regions ⁽¹⁾
“ For whom ” (beneficiary of the function or service)	For residential or commercial buildings

⁽¹⁾ Including the following Member States: AT, BE, BG, CZ, DE, DK, EE, northern ES, FI, northern and central FR, HU, IE, IS, LI, LT, LU, LV, LX, NL, PO, SE, SI, SK, RO.

The reference flow of each scenario, i.e. the amount of insulation material required to fulfil the functional unit, was calculated based on the function to be provided (Table 5.2) and relevant physical properties of the specific materials, including thermal conductivity

(λ) and density (ρ). The reference flow was thus quantified as $R \cdot \lambda \cdot \rho \cdot A$ (with A being the surface to be insulated), as summarised in Table 5.3. Data for relevant material properties were retrieved from previous studies (PEFCRs for thermal insulation products in the case of conventional EPS and PUR; Avnir 2019), actual producers (R-EPS and R-PET) or assumed (for bio-based and CO₂-based PUR, for which the physical properties were assumed identical to conventional PUR) in the absence of specific data.

Table 5.3. Calculation of the reference flow for insulation boards LCA scenarios.

Polymer	Thermal conductivity (λ) (W·m ⁻¹ ·K ⁻¹)	Density (ρ) (kg·m ⁻³)	Area (A) (m ²)	Reference flow (R· λ · ρ ·A) ⁽¹⁾ (kg)
EPS ⁽²⁾ (all types of feedstock)	0.031	15	1	3.3
PUR ⁽³⁾ (all types of feedstock)	0.022	31	1	4.9
R-PET (R ₁ =100%) ^(4,5)	0.028	48	1	9.6

⁽¹⁾ $R = 7.14 \text{ m}^2 \cdot \text{K} \cdot \text{W}^{-1}$.

⁽²⁾ Based on EUMEPS (2020; thermal conductivity) and PEFCRs for thermal insulation products (Avnir, 2019; density). The assumed density is mostly representative of boards for non-accessible flat roofs rather than pitched roofs, as assumed in this case study. Recycled EPS (R-EPS) was assumed to have the same characteristics as conventional EPS, based on the technical information available from providers (e.g., see Benchmark Foam, 2020b, and Iowa EPS Products, 2020).

⁽³⁾ From PEFCRs for thermal insulation products (Avnir, 2019), assuming that bio-based and CO₂-based PUR have the same characteristics as conventional PUR.

⁽⁴⁾ R₁: recycled content.

⁽⁵⁾ Based on technical data related to *ArmaPET Eco50* (Armacell, 2020b), taken as an example of R-PET board.

5.3 System boundary

In all the analysed scenarios, the system boundary was set to cover the default life cycle stages specified in the *Plastics LCA* method for cradle-to-grave LCAs of final products¹³¹, and the associated most relevant processes within the product life cycle. The considered life cycle stages and processes are described below, and are also schematically represented in the system boundary diagrams shown in Figures 5.1 to 5.6:

- **Feedstock Supply**¹³² – covering extraction, processing, transport and possible refining of crude oil and natural gas (fossil-based polymers); collection, transport and sorting of post-consumer plastic waste (recycled polymers); crop (i.e. soybean) cultivation and processing into crude soybean oil (bio-based PUR); CO₂ capture, purification and liquefaction (CO₂-based PUR), as well as transport of these feedstock materials to downstream conversion or utilisation processes (e.g. naphtha cracking, polymer recycling, soybean oil refining, and polyol production).
- **Polymer Production**¹³³ – covering all the activities associated with the conversion or recycling of relevant feedstock materials into the specific polymer, including any transport among these activities and final transport of polymer granulate, flakes or foam to downstream manufacturing processes.

¹³¹ Note that, as permitted by the *Plastics LCA* method, and as described below, some of the default life cycle stages have been split into different sub-stages, and their naming was adjusted to better reflect the scope of this study.

¹³² Corresponding to the default life cycle stage “Raw Material Acquisition and Pre-Processing” specified in the *Plastics LCA* method.

¹³³ Corresponding to the default life cycle stage “Raw Material Acquisition and Pre-Processing” specified in the *Plastics LCA* method.

- **Manufacturing** – including insulation board manufacturing through formation, cutting and packaging of the boards¹³⁴.
- **Distribution** – including transport of insulation boards from the manufacturing site to distribution centres, and from these to final users.
- **End of Life** – covering collection, transport, incineration and disposal of insulation boards after use, including any avoided processes from virgin material or energy substitution by recovered material/energy.

The default life cycle stage “Raw Material Acquisition and Pre-processing” was further split into two separate sub-stages (i.e. Feedstock Supply and Polymer Production), to allow disaggregating and separately quantifying the impacts of feedstock supply and those associated with downstream conversion processes into the final polymer. Moreover, a different nomenclature was applied to such stages compared to the default nomenclature specified in the *Plastics LCA* method, to better reflect the investigated supply chains and the scope of the study. The stage of “Raw Material Acquisition” was thus identified with that of “Feedstock Supply”, while “Pre-processing” corresponds to “Polymer Production”.

The Use stage was omitted, as it can be reasonably assumed that for the investigated building insulation materials it incurs negligible impacts. This is in line with the rules specified in PEFCRs for thermal insulation products (Avnir, 2019), which consider that no maintenance, repair, replacement or refurbishment is required during the 50-year service life of the product, when correctly applied and used in standard conditions.

Finally, it has to be noted that additives were not included in the assessment, due to the lack of complete and consistent data on the use of additives in the production of insulation boards, of the respective polymers, and of plastics in general. Data and knowledge are also lacking on the release and fate of additives throughout the product life cycle. Exclusion of additives is acknowledged as a limitation of this study, as additive production can account for a non-negligible portion of the cradle-to-gate Climate Change impact and energy demand of polymers, which is up to 46% for starch-based polymer grades including larger shares of additives in the range of 30% (Broeren et al., 2017). Moreover, additives can also be relevant at the End of Life stage, where they can be released, as such or after degradation/conversion into different compound(s), in the environment (e.g. the soil in case of biodegradable products sent to biological treatments, or subject to in-situ degradation).

¹³⁴ For R-PET boards, the manufacturing stage also includes production of PET fibres from recycled PET flakes.

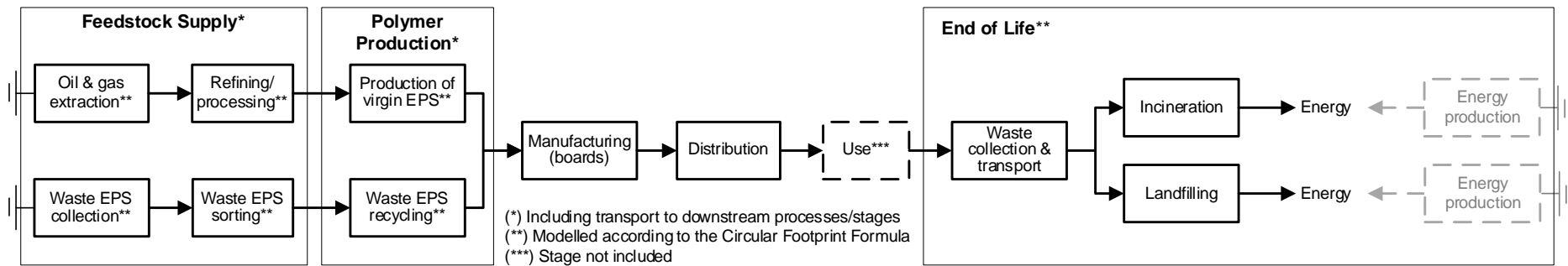


Figure 5.1. System boundary for fossil-based EPS insulation boards, including 98% virgin EPS and 2% recycled EPS (Scenario 1).

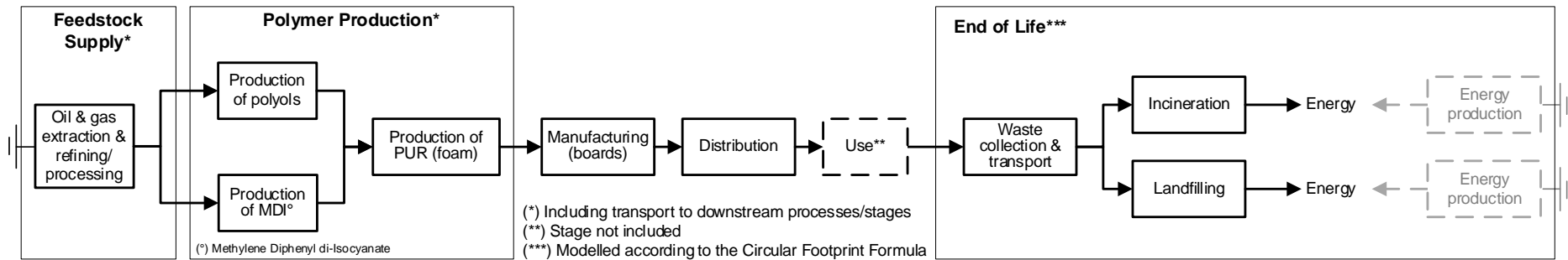


Figure 5.2. System boundary for fossil-based PUR insulation boards (Scenario 2).

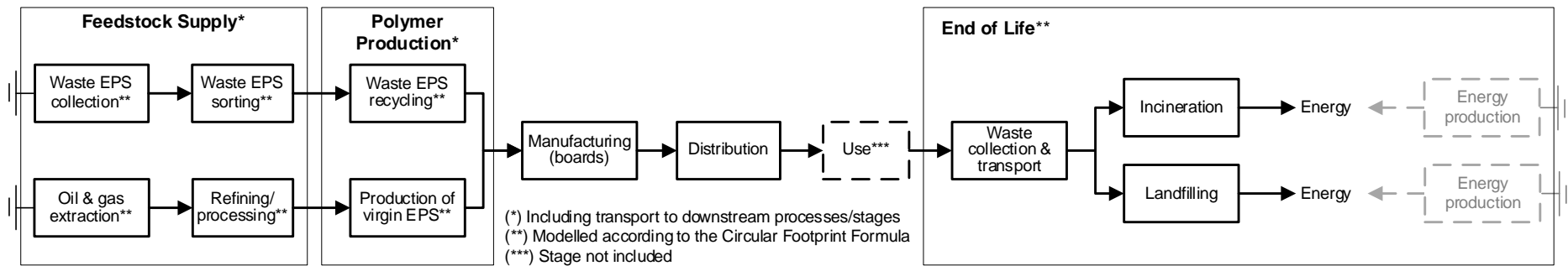


Figure 5.3. System boundary for 100% recycled EPS insulation boards (Scenario 3).

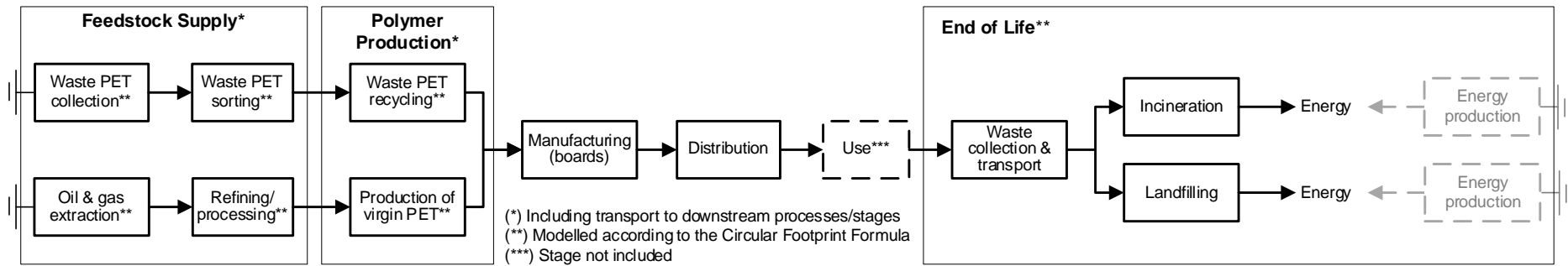


Figure 5.4. System boundary for 100% recycled PET insulation boards (Scenario 4).

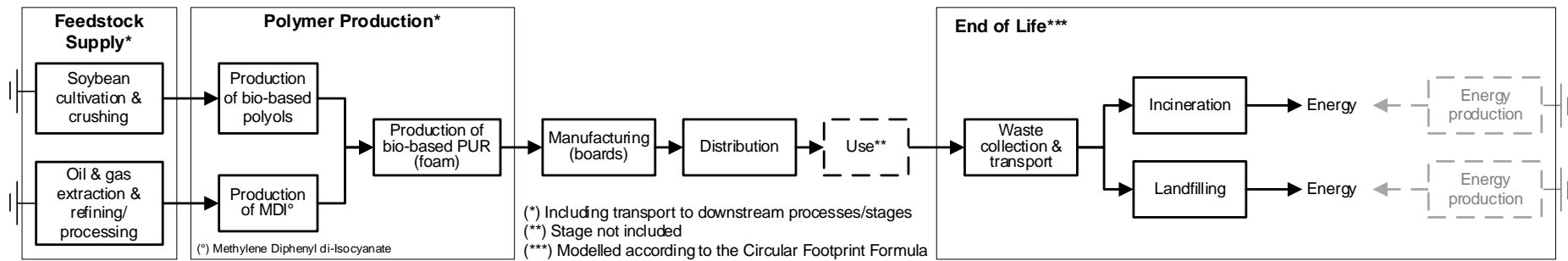


Figure 5.5. System boundary for partly bio-based PUR insulation boards (Scenario 5).

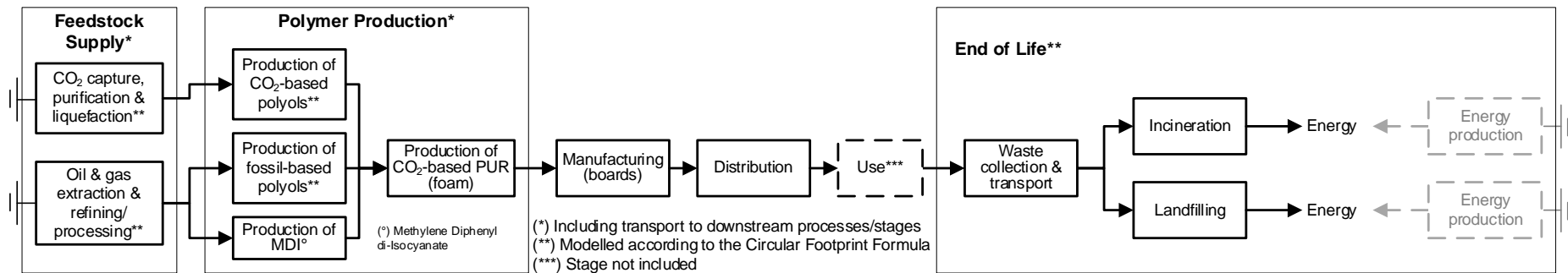


Figure 5.6. System boundary for partly CO₂-based PUR insulation boards (Scenario 6).

5.4 Limitations and critical assumptions

When interpreting the results of this case study, it is important to bear in mind the following key limitations and critical assumptions. These are mostly associated with the availability of suitable data and information for use in the study, or with the quality of the data available and ultimately applied.

A first point concerns the technical properties considered for the investigated insulation materials, i.e. their thermal conductivity (λ) and density. These are key parameters for the study, as they determine the reference flow, which is the quantity of insulation material required to fulfil the functional unit. In this respect, it is first noted that the thermal conductivity (λ) and density values of 100% recycled EPS (R-EPS) were assumed identical to those of conventional virgin EPS (see Section 5.2). This means that the assessment did not account for a possible decrease of the material quality after mechanical recycling of EPS, which was instead assumed to achieve a comparable quality between recycled and virgin material (i.e. a quality ratio $Q_{S_{in}}/Q_p$ equal to 1 was considered; Section 5.5.2.2). However, it is clear that if a larger amount of recycled material would be ultimately needed to obtain the required thermal resistance (e.g. because the material is more dense or because of a poorer λ), the environmental performance of 100% recycled EPS boards would worsen compared with that determined in this study. Similarly, the thermal conductivity and the density of partly bio-based PUR and partly CO₂-based PUR were assumed the same as those of conventional fossil-based PUR (Section 5.2). A perfect substitutability was thus assumed between the polyols obtained from refined soybean oil or captured CO₂ and the conventional aromatic polyols (i.e. a $Q_{S_{in}}/Q_p$ ratio of 1 was considered when replacing fossil-based polyols with partly CO₂-based ones; see Sections 5.5.1.4 and 5.5.2.4). Again, if a larger amount of material would instead be needed to obtain the required thermal resistance, the potential impacts of partly bio-based and partly CO₂-based PUR insulation boards would be higher than those estimated in this study.

Regarding the system boundary, an important limitation is related to the exclusion of additives used during polymer production or insulation boards manufacturing (e.g. flame retardants, UV stabilisers or pigments). This means that the burdens associated with the production of additives and with their possible release during use or End of Life were not quantified, due to the lack of complete and sufficiently specific and/or representative data on additive use and release (see Section 5.3). While additives are generally reported to be mostly used in small shares, their potential impacts may be proportionally (much) higher compared with used quantities, once they are released into the environment. Impact assessment results presented in this study may thus be even significantly underestimated in specific categories, and do not capture any differences in additive use among the investigated product alternatives.

Focusing on the data applied for modelling, a first set of limitations is related to the stages of Feedstock Supply and Polymer Production for 100% recycled EPS boards. Supply of waste EPS for recycling and subsequent use for insulation board manufacturing was modelled based on data related to the collection and transport of source separated plastic waste at the municipal level, and to the sorting of mixed plastic waste from municipal or industrial sources (see Section 5.5.1.2). However, most EPS is used in the building and construction sector, where different waste collection and sorting pathways may be applied due to, e.g., selective demolition practices and/or *ad hoc* management schemes for construction and demolition waste. The applied data are thus potentially representative of the supply of only a minor portion of EPS waste used as a feedstock for recycled EPS boards, as long as this feedstock is assumed to reflect the same market shares as virgin EPS used by converters. Moreover, a 100% sorting efficiency was assumed in the absence of specific information, and this unlikely reflects reality. Finally, the recycling of sorted EPS waste was modelled based on literature data collected from one recycling company, assuming the same sorting efficiency as PET recycling, in the absence of specific data (Section 5.5.2.2). The data were also assumed to cover all

recycling steps until granulate production, although they appeared to cover only the initial steps of compaction and shredding of EPS waste. The modelled process is thus not representative of current average recycling practice, and may include only part of the burdens of the actual recycling process. In light of these limitations and assumptions, the overall potential impacts estimated in this study for 100% recycled EPS boards may therefore not fully reflect actual supply-chain impacts, and are potentially underestimated.

For partly bio-based PUR boards, data-related limitations are mostly associated with the synthesis process of soy-based polyols used as a replacement for aromatic polyols in PUR production. For this process, the applied data were based on literature values referring to a single US company, and to the specific formulation of the respective product (see Section 5.5.2.3). The modelled process is thus only an example of soy-based polyol production, and is not representative of the average industrial production of this precursor or of bio-based polyols in general at the EU level. Moreover, the applied data only cover the energy demand of the production process, while excluding other potentially relevant burdens associated with, e.g., resource use, ancillary material inputs, or emissions into the environment (which are indeed mentioned but not quantified or specified in the applied data source). The developed inventory thus only partially covers the burdens of the polyol production process, underestimating the associated impacts. It should also be noted that soy-based polyol production is a more recent process compared with the synthesis of conventional aromatic polyols, thus being open to further potential optimisation and improvement (e.g. in terms of conversion efficiency and process integration). This lower level of maturity of the polyol production process should be taken into account in the interpretation of the results related to partly bio-based PUR boards. Finally, regarding the feedstock considered for this bio-based alternative (i.e. crude soybean oil), EU production was entirely assumed, in line with the applied dataset and the geographical scope of the study (see Sections 5.1 and 5.5.1.3). However, considering average production at the global scale (including also production in South America), would likely increase the impact of bio-based PUR boards on Climate Change, due to the additional contribution from direct Land Use Change (dLUC) associated with possible conversion of native forest to agriculture land.

Concerning partly CO₂-based PUR boards, theoretical data from process modelling available in the scientific literature were applied to model the synthesis of CO₂-based polyols, in the absence of real (publicly available) data for this innovative process (see Section 5.5.2.4). Such data are likely not representative of commercial scale production, and do not account for possible (efficiency) improvement due to upscaling and further process optimisation. The burdens associated with CO₂-based polyol production might thus have been even largely overestimated in this study, and impact assessment results calculated for CO₂-based PUR insulation boards should be interpreted carefully, taking also into account the emerging nature of part of the underlying technology.

As for the Manufacturing stage, for PUR-based insulation boards the modelling relied on data related to the manufacturing of EPS-based boards, introducing some assumptions to isolate the burdens of relevant operations (i.e. cutting and packing of the boards) from those of the previous thermoforming step (which is not relevant for PUR boards) (see Section 5.5.3). While this approximation was considered reasonable and to only moderately affect the results, it has to be noted that the applied data are not representative of the real manufacturing process, thus introducing an uncertainty. Similarly, manufacturing of R-PET boards was modelled based on literature data referring to a single full-scale facility, and are therefore not representative of average industrial production in the EU.

Regarding the End of Life stage, a number of data-related limitations apply, with more or less important implications on the results, as discussed in the following. First, collection and transport of waste insulation boards from demolition/dismantling to treatment or disposal were modelled based on data referring to municipal plastic waste collected separately for recycling, or as residual waste for incineration or landfilling (see Section

5.5.5.2). This approximation is not expected to significantly affect the results, as the contribution of waste collection and transport to overall lifecycle impacts is generally modest. Similarly, the use of data representing the sorting of mixed plastic waste from municipal or industrial collection was considered to reasonably approximate the burdens of any sorting operations applied to shredded insulation boards before recycling (Section 5.5.5.3). On the other hand, it is recognised that these operations and the assumed collection and transport pathways may differ from those actually applied to construction and demolition waste (of which insulation boards are part) due to the implementation of, e.g., selective demolition practices and/or *ad hoc* management schemes for this waste stream. Moreover, a 100% sorting efficiency was assumed for all insulation materials, in the absence of specific information, and this unlikely reflects reality.

For mechanical recycling of sorted EPS and PUR waste, limitations are more relevant than those discussed above for the collection and sorting of waste insulation boards. PUR recycling was modelled based on data directly available in the scientific literature or derived estimates (see Section 5.5.5.3), and these are not representative of any real full-scale facility nor reflect average industrial practice. Similarly, while EPS recycling data were company-specific, they are equally not representative of current average recycling practice, and may cover only part of the actual burdens of the recycling process (as discussed above for recycled EPS boards). Moreover, a process efficiency equal to that of PET recycling was assumed for the recycling of both EPS and PUR, in the absence of specific information, while a substitution ratio (Q_{Sout}/Q_p) between recycled and virgin material equal to 1 was assumed for all insulation materials (Section 5.5.5.3), which may not be the case in reality. The results of the sensitivity analysis considering the application of 100% mechanical recycling as an independent alternative End of Life scenario for insulation boards (Section 5.8.5.5) should thus be used carefully, especially for EPS-based and PUR-based boards, taking into account the limitations related to the entire recycling pathway, including collection and sorting.

Approximations were finally performed also to model incineration and landfilling of insulation boards, although only moderate effects are expected on total results, as discussed below. Regarding incineration, for EPS-based and PUR-based boards a dataset referring to the treatment of unspecified fossil-based plastic waste in a municipal incineration plant was applied as a proxy (with only minor changes in the case of bio-based PUR boards) (see Section 5.5.5.4). This approximation was considered reasonable and to only marginally affect the results, as the energy content assumed for the average plastic waste sent to incineration is similar to that of EPS and PUR, thus leading to similar benefits from energy recovery. Since these benefits generally determine overall incineration impacts, the latter are also expected to be similar for all the investigated insulation materials. As for landfilling, a similar approximation was performed, for all insulation board alternatives, modelling the disposal of generic non-biodegradable plastic waste rather than landfilling of the specific materials (see Section 5.5.5.5). Also in this case, the approximation was considered acceptable, as the degradation rate in the landfill body (one of the most relevant parameters for landfilling modelling) is similar for all non-biodegradable polymers, including those investigated in this study.

5.5 Life Cycle Inventory

This section describes the main details of the Life Cycle Inventory of the analysed scenarios, including the related assumptions and data sources. The description is separately reported for each life cycle stage in the following sub-sections (5.5.1 – 5.5.6).

5.5.1 Feedstock Supply Stage

5.5.1.1 Fossil-based polymers

For virgin fossil-based polymers (i.e. EPS, PUR and PET), the stage of Feedstock Supply includes the activities of crude-oil and natural gas exploration, drilling, extraction, processing and transport to downstream users, as well as naphtha production in crude oil

refineries, and its transport to subsequent conversion processes (i.e. naphtha cracking and catalytic reforming).

For EPS and PET, these activities were modelled as described in Section 3.5.1.1 of the Beverage Bottles case study, where the reader is referred to for details on the applied datasets and approach. For PUR, activities related to the Feedstock Supply stage were instead covered in the vertically aggregated datasets used to model production of PUR precursors (i.e. fossil-based polyols and methylene diphenyl di-isocyanate; see Section 5.5.2.1), so that they were not modelled nor accounted for separately. A separate modelling would have been possible by using a partially aggregated cradle-to-gate dataset for PUR production, where feedstock inputs are disaggregated. However, this dataset could not be applied due to the need to perform a consistent modelling with the production of partly bio-based and CO₂-based PUR, where fossil-based polyols are replaced by bio-based polyols or partly CO₂-based ones, thus requiring the use of the same datasets for both fossil-based polyol production (to be modelled when implementing the Circular Footprint Formula; see Sections 5.5.1.4 and 5.5.2.4) and for the subsequent synthesis of PUR from polyols.

5.5.1.2 Recycled polymers

For recycled polymers (i.e. R-EPS and R-PET), Feedstock Supply includes collection of post-consumer plastic waste of the relevant polymer, and its subsequent transport and sorting in specific facilities for recycling. These activities and processes were modelled as described in Sections 3.5.5.2 and 3.5.5.3 of the Beverage Bottles case study, based on data referring to collection and transport of separately collected plastic waste at the municipal level, as well as to sorting of mixed plastic waste in dedicated facilities. The applied data are thus likely to be only partially representative of EPS waste collection and sorting, as this material is mostly used in the building and construction sector (PlasticsEurope, 2019), where different pathways may be applied due to, e.g., selective demolition practices and/or *ad hoc* management schemes for construction and demolition waste. However, related technologies and processes are currently not widely established and mostly under research and development. Little or no information and data were hence available in their respect, and data referring to municipal plastic waste collection and sorting were applied to the entire recycled content. An efficiency equal to 100% was assumed for the sorting process, in the absence of specific information.

The collection, transport and sorting processes were implemented in the lifecycle model according to the Circular Footprint Formula (CFF), which is the approach prescribed in the *Plastics LCA* method to handle recycling situations. The formula was applied considering a value of the A factor equal to 0.5, based on the default application-specific value reported in Annex C of the *Plastics LCA* method for recycled EPS used in building and construction, and the material-specific value provided for recycled PET used in unspecified applications (in the absence of an application-specific value for PET used in insulation/construction products). Therefore, only 50% of the burdens from EPS and PET waste collection, transport, sorting (and subsequent recycling; see Section 5.5.2.2) were assigned to the recycled material content in insulation boards, the rest being assigned to the system providing waste material for recycling. However, the recycled material content was assigned an equal share $((1-A) \times Q_{S_{in}}/Q_p)$ of the burdens associated with the supply of the virgin fossil-based feedstock used in the production of the replaced virgin polymer. Such burdens were modelled as described in Section 5.5.1.1. Further details and considerations on the implementation of the CFF are provided in Section 5.5.2.2, addressing the modelling of the Polymer Production Stage for recycled polymers.

5.5.1.3 Bio-based polymers

For bio-based PUR, the stage of Feedstock Supply (crude soybean oil) includes cultivation of soybean in relevant EU countries, processing into soybean oil in the EU, and transport of soybean between these activities. The EF-compliant dataset “[EU+28] Crude soybean oil; from crushing (pressing and solvent extraction), production mix; at plant” was used

for modelling. This dataset represents the production of crude soybean oil from a soybean crushing process, where also soybean hulls and soybean meal are jointly produced as co-products. The data refer to the region EU-28 + EFTA, and the year 2016. Considered activities include soybean cultivation and associated transport, as well as direct inputs and outputs of the soybean crushing process. These include energy inputs (heat from natural gas and electricity), process water, auxiliary materials (e.g. hexane), direct emissions (e.g. hexane to air), as well as treatment of waste and wastewater outputs.

The origin of soybean used in the process (i.e. the crop mix) is determined based on crop production statistics from FAOSTAT, and import/export data from Eurostat. The soybean cultivation process in each country of the mix is modelled based on the Agri-footprint methodology (Blonk Consultants, 2015a,b), relying on 5-year average yield data from FAOSTat (2010-2014), and on country-, crop-, and process-specific data for the other relevant parameters, as appropriate. Specific activities, inputs and emissions covered in the dataset are those reported in Section 3.5.1.3 of the Beverage Bottles case study, which also provides additional detail on their modelling or quantification, and on other relevant applied methodological choices. While such description refers to maize and wheat cultivation datasets, it equally applies to soybean cultivation, with GHG (CO₂) emissions from direct Land Use Change (dLUC) being quantified for this crop at 5.44 kg CO₂/kg_{crude soybean oil} (cradle-to-gate emission, which are reasonably mostly associated to the soybean cultivation process). Transport of harvested soybean to the crushing process is modelled based on Eurostat data on average transport modes and distances for different commodities in the EU, but no specific information is provided on the assumptions performed to develop the dataset.

Crushing of soybeans normally occurs in a series of process steps, including de-hulling of soybeans, mechanical crushing of beans with partial extraction of oil, and further extraction of oil using hexane as a solvent, with subsequent hexane recovery steps. The entire crushing process is modelled based on average values of data retrieved from at least five different literature sources reporting the mass and energy balance and energy requirements of the process itself. Allocation between the different co-products from crushing (crude soybean oil, soybean hulls and soybean meal) is based on the respective market value (i.e. economic allocation).

Crude soybean oil was assumed to be transported to downstream processing (refining and polyol production) according with the default transport scenario specified in the *Plastics LCA* method for transferring of goods from suppliers to factories/users within Europe. This scenario includes transport by lorry (> 32 t, Euro 4) for 130 km, by freight train (technology mix) for 240 km, and by ship (barge) for 270 km.

5.5.1.4 CO₂-based polymers

Activities related to Feedstock Supply for CO₂-based PUR production include capture, purification and liquefaction of CO₂ in flue gas of coal-based power plants, and its transport to downstream conversion (utilisation) facilities via pipeline (assumed to be the most suitable transport form for industrial use at large scale). When considering the use of CO₂ sourced from ammonia production plants (as a sensitivity analysis; Section 5.8.5.2), only purification, liquefaction and transport of the almost pure CO₂ flow generated in the ammonia production process were included in the Feedstock Supply stage. Capture (i.e. separation/extraction) from the original mixture of CO₂ and hydrogen generated in the process itself was excluded, as not being purposefully carried out to enable downstream utilisation of CO₂, but rather to isolate hydrogen for subsequent use for ammonia synthesis. As such, capture of CO₂ is necessarily carried out as an integral part of the ammonia production process generating CO₂ as a waste, and it was hence considered to entirely belong to such an upstream activity.

For modelling purposes, raw gaseous CO₂ was considered to be a waste of the specific CO₂ source (coal-fired power plants or ammonia production plants, as discussed above), according with current requirements of the *Plastics LCA* method (Section 4.4.5.1). This

assumption especially takes into account that raw gaseous CO₂ generally has no economic value at the point of arising (i.e. before capturing or purification), and that for most existing CO₂ sources (including power plants) its availability is currently much higher compared to its demand for subsequent capture and utilisation (for any use, not only limited to polyols and PUR production). In this perspective, the subsequent processes of CO₂ capture¹³⁵, purification, liquefaction, transport, and (depending on the pathway) utilisation, were considered to be the components of a recycling chain aimed at converting waste CO₂ into a useful CO₂-based product (i.e., in this case, polyols and later polyurethane), ultimately replacing an equivalent product from primary resources (i.e. 100% fossil-based polyols and polyurethane). As in any recycling situation, the Circular Footprint Formula (CFF) was applied to model the supply of feedstock CO₂ and subsequent conversion/utilisation activities until the point of substitution between the CO₂-based product and its primary counterpart (i.e., in this case, between CO₂-based polyols and fossil-based ones, as discussed below). Alternative approaches were explored as a sensitivity analysis (Section 5.8.5.3), applying also a wider system perspective where the CO₂-based product (i.e. CO₂-based polyols) is considered, along with the main product of the CO₂ source (i.e. coal-based electricity) a co-product of the overall Carbon Capture and Utilisation (CCU) system.

Following the CFF, only a share of the burdens from capture, purification, liquefaction, transport and conversion activities applied to waste CO₂ until the point substitution were allocated to the CO₂-based product (i.e. CO₂-based polyols). Conversely, no burdens from any upstream activity occurring before CO₂ is generated as waste were attributed to it, such as activities related to the supply and conversion/use of the CO₂-providing fossil-based feedstock (e.g. coal extraction and combustion) and any subsequent processing not directly aimed at making CO₂ available for further use. On the other hand, the CO₂-based product was allocated a share of cradle-to-gate burdens associated with the primary production of the equivalent conventional product it replaces (i.e. fossil-based polyols), including supply of the respective primary feedstock. The point of substitution between CO₂-based product and primary product was assumed to occur at the polyol level, as CO₂-based polyols and fossil-based polyols are the first equivalent product entering the same identical process (i.e. PUR production) within the supply chain of both CO₂-based and conventional PUR¹³⁶. The gate of the PUR production process was thus identified as the most proximate point of substitution where the CFF has to be applied.

The allocation of burdens from the mentioned upstream activities to the CO₂-based product depends on the value of the A factor of the CFF and, for cradle-to-gate activities involved in the production of replaced fossil-based polyols, also on the quality ratio ($Q_{s_{in}}/Q_p$). According to the *Plastics LCA* method, the A factor has to be defined based on the relation between supply and demand of the recycled (in this case, CO₂-based) product. Since there is currently no established market for CO₂-based polyols, and no default values are available in Annex C of the *Plastics LCA* method, a situation of equilibrium between supply and demand of CO₂-based polyols was assumed for this study (i.e. $A=0.5$). However, any future study focusing on real products shall take into account the specific market situation at the time of the study itself. As for the quality ratio, CO₂-based polyols were assumed to have the same quality as the replaced conventional fossil-based polyols (i.e. $Q_{s_{in}}/Q_p=1$), in the absence of specific information. Following these assumptions, only 50% of the burdens from the processes of capture, purification, liquefaction, and transport of waste CO₂ were allocated to CO₂-based polyols in the Feedstock Supply stage. However, they were allocated 50% of the burdens

¹³⁵ Where purposefully carried out to enable downstream utilisation of CO₂.

¹³⁶ Upstream processes in the supply chain entirely or partly differ between CO₂-based and conventional PUR. Particularly, the CO₂-based polyol synthesis process directly using captured CO₂ as an input partly differs from the conventional polyol synthesis in terms of, e.g., chemical and energy inputs and waste outputs (and different data were indeed applied to the modelling of these processes, as reported in Sections 5.5.2.1 and 5.5.2.4). Therefore, the most proximate point of substitution could not be identified at the gate of the polyol synthesis process, i.e. at the level of captured CO₂ used in partial replacement of conventional propylene oxide. Conversely, as discussed above, substitution at the polyol level was assumed, with CO₂-based polyols replacing conventional fossil-based polyols.

associated with the supply of fossil-based feedstock used in the production of the replaced conventional polyols (and modelled as described in Section 5.5.1.1).

The CO₂ capture, purification and liquefaction processes were modelled based on the *ecoinvent* dataset “[RER] Carbon dioxide, liquid”, which represents the burdens associated with extraction of CO₂ from (waste) gaseous streams of industrial production processes by means of a 15-20% Monoethanolamide (MEA) solution, followed by purification and liquefaction. Compared to the original dataset (which relies on a combination of literature data from 1999 and industry data for the period 2011-2015), the electricity consumption was updated with a more representative value for extraction from flue gases of coal-based power plants (1.32 MJ/kg CO₂, including the consumption for purification and liquefaction). This figure was calculated as the average of several values reported in Von der Assen et al. (2016) for capture processes relying on MEA as a solvent. On the other hand, the original values applied in the dataset for the capture efficiency, MEA consumption, water flows, and air emissions were considered representative also for extraction from flue gases of power plants (except for methane emissions, which were removed). Original background datasets related to energy supply were also replaced with EF-compliant datasets, while datasets representing water supply and wastewater treatment for the geographies “CH” or “RER” were replaced with equivalent datasets referring to the geography “Europe without Switzerland”. Finally, infrastructure processes (representing chemical factory construction) were removed, to improve reliability of results in the Ozone Depletion impact category.

Transport of liquefied CO₂ to downstream industrial users via pipelines was assumed to take place along a distance of 300 km, in line with the “CO₂ deserts map” reported in Von der Assen et al. (2016) for CO₂ sources located in Europe. According to the latter, considering only CO₂ sources available at the time of the study, the current total European demand of 50 Mt CO₂/year can be met by CO₂ sources located at a distance not larger than 300 km for hypothetical industrial users in central Europe (and demanding up to 5 Mt CO₂/year). The burdens of onshore pipeline transport were approximated with those of long-distance transport of natural gas reported in the *ecoinvent* dataset “[DE] transport, pipeline, long distance, natural gas”, based on data referring to the years 2001 and 2006. However, for implementation in the model, the dataset was adjusted to reflect EU background conditions (in terms of electricity generation), by replacing the original electricity dataset with the relevant EF-compliant datasets for electricity generation in the EU (average grid mix). Moreover, leakage of natural gas and the related air emissions were replaced with leakage and emissions of CO₂.

5.5.2 Polymer Production Stage

The Polymer Production stage covers the activities of feedstock processing into any relevant intermediate(s) and monomer(s), the polymerisation or recycling process, as well as any transport among these activities and, where relevant, final transport of polymer granulate to the insulation boards manufacturing site. The following subsections (5.5.2.1 – 5.5.2.5) describe how these activities were modelled in this case study, distinguishing between fossil-based, recycled, bio-based, and CO₂-based polymers.

5.5.2.1 Fossil-based polymers

Production of expanded polystyrene (EPS) granulate was assumed to take place via expansion of pentane-containing polystyrene (PS) granulate. Polystyrene production from fossil-based feedstock was modelled using a partially aggregated, cradle-to-gate, ILCD-EL compliant dataset provided by Thinkstep¹³⁷. The dataset disaggregates upstream feedstock inputs (crude oil, natural gas and naphtha), reflects the main technologies adopted in EU-28, and refers to the year 2018. It is mainly based on industry data from internationally adopted production processes, integrated, where needed, with literature

¹³⁷ The following dataset was applied: [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.

data from several sources. The number of industry data sources considered for individual process steps is not specified. Crude oil accounts for 81% of the total feedstock input (including the contribution of naphtha), with the remaining 19% being covered by natural gas. All conversion processes are assumed to take place in Europe, so that the dataset not only reflects the main technology applied in the region, but also EU-average background conditions in terms of e.g. energy generation, material supply and transport. This was considered to adequately represent the current situation at the time of the study, as most PS used in the EU (i.e. 93%) was estimated to be produced domestically, while only 7% was imported¹³⁸. The main conversion process involved in the modelled supply chain is steam cracking of naphtha and natural gas, delivering ethylene and propylene (both used as intermediates for styrene production), along with other co-products such as butadiene, pyrolysis gas, refinery gas and hydrogen. Another relevant conversion processes is catalytic reforming of naphtha and pyrolysis gas to produce benzene, toluene and xylenes, with benzene being used as an intermediate in the production of styrene. Allocation among the different co-products from these processes is performed as described in Section 3.5.2.1 of the Beverage Bottles case study, while in the final polymerisation step no allocation is performed, being polystyrene the only output from such process.

The expansion process of PS to EPS granulate was modelled based on the *ecoinvent* dataset "[CH] polystyrene foam slab for perimeter insulation", taking into account only the burdens related to the expansion step. According with the information provided in the dataset, 1 kg of PS, 0.187 kWh of electricity, and 1.77 MJ of heat are needed to produce 1 kg of EPS. Pentane and carbon dioxide are also released to air during expansion (0.00107 kg and 0.0781 kg, respectively). The data are derived from industry (average of 2 installations in Switzerland), refer to the year 2009, and are complemented, where needed, by additional calculations (e.g. to estimate air emissions). To model the supply of energy inputs, EF-compliant background datasets representative of EU-average conditions were used. Processes related to infrastructures and equipment included in the original datasets were not taken into account.

Polyurethane (PUR) foam is produced by metering and mixing two or more streams of liquid components containing PUR precursors, mainly polyols and methylene diphenyl diisocyanate (MDI). Pentane is typically used as blowing agent to generate the foam. The majority (98%) of PUR used in the EU was estimated to be produced domestically (based on data from Eurostat, 2019d). This geography was thus considered as a reference for modelling, as described below.

Production of conventional, fossil-based, polyether polyols requires the following steps: (i) preparation of the initiator solution; (ii) addition of propylene oxide and ethylene oxide (polymerisation step); and (iii) filtration and finishing of the rigid polyether polyols. The process was modelled using a vertically aggregated, cradle-to-gate dataset developed for *PU Europe* and available in the GaBi database ([EU-28] *Aromatic Polyester Polyols (APP) production mix; polycondensation | production mix, at producer | Hydroxyl value: 150-360, aromatic content: 5-50%*). The dataset is based on primary gate-to-gate data and information on polyol production collected from five companies representing more than 75-85% of the total EU production of this material in 2014. Collected data refer to five specific production sites in Europe (Germany, Italy, Spain and The Netherlands), and are calculated as annual averages for the year 2013. Ancillary inputs to the process (e.g. fuel, energy) and respective outputs are modelled considering average European conditions and, whenever applicable, site-specific conditions, to reflect representative situations. Upstream processes in the supply chain, including feedstock supply and production of precursors, are modelled based on datasets available in the GaBi database, considering country-specific conditions.

MDI production is made via phosgenation of methylenedianiline (MDA). Similarly to polyol production, a vertically aggregated, cradle-to-gate dataset from the GaBi database

¹³⁸ The estimate was based on average annual import shares calculated from *Prodcom* data for the years 2016-2018 (Eurostat, 2019d).

was applied for modelling ([EU-28] *Methylenediphenyl diisocyanate ((p)MDI); phosgenation of methylenedianiline (MDA) | production mix, at plant | polymeric MDI ((p)MDI)*). The dataset was developed for ISOPA, relying on gate-to-gate primary data and information for MDI production collected from five European suppliers of this chemical, covering approximately 90% of its total EU production in 2010. Data were collected as annual averages for the same year (2010) from plants located in five different European countries (i.e. Germany, Belgium, Netherlands, Portugal and Hungary). As for the polyol dataset described above, ancillary inputs and outputs of the process were modelled considering average European conditions and, whenever applicable, site specific conditions, to reflect representative situations. Also, datasets from the GaBi database were used to model upstream activities associated with feedstock supply and production of the precursors, considering country-specific conditions. Allocation of process burdens between the two jointly produced co-products (i.e. MDI and hydrogen chloride) was based on the respective mass.

For the final production of PUR rigid foam slabs from its fossil-based precursors (polyols and MDI), a new dataset was developed based on the *ecoinvent* dataset "[RER] *polyurethane production, rigid foam*", using EF-compliant background datasets to model the respective ancillary inputs (pentane and electricity) and outputs (treatment of waste foam) under EU-average conditions. The dataset accurately implements the inventory reported in the latest PlasticsEurope eco-profile for PUR rigid foam (PlasticsEurope, 2005). Overall, per 1 kg of PUR, 0.62 kg of MDI, 0.39 kg of polyols, 0.42 kWh of electricity, and 0.054 kg of pentane are used, while 0.02 kg of PUR foam are generated as waste. 0.003 kg of pentane are also emitted to air. Infrastructure processes (chemical factory construction) included in the original *ecoinvent* dataset were not taken into account.

While virgin polyethylene terephthalate (PET) production is not directly used in any of the investigated insulation boards scenarios, virgin PET production also had to be modelled when applying the Circular Footprint Formula (CFF) to calculate the burdens associated with recycled PET supply (see Section 5.5.2.2). The process was modelled using a partially aggregated, cradle-to-gate, ILCD-EL compliant dataset provided by Thinkstep¹³⁹, representing the production of bottle-grade PET granulate (see Section 3.5.2.1 of the Beverage Bottles case study for a description). While an ILCD-EL compliant dataset was available for PET fibres (i.e. the equivalent virgin material to recycled PET fibres used for R-PET boards manufacturing; see Section 5.5.3), this was not applied because of its aggregated nature (i.e. including also the burdens from feedstock supply). Moreover, the application of datasets from other sources (e.g. the *ecoinvent* database) was not considered a suitable alternative, to avoid relevant inconsistencies and discrepancies with the datasets applied to model the production of the other investigated polymers and of recycled PET.

5.5.2.2 Recycled polymers

For recycled polymers, the Polymer Production stage includes mechanical recycling of sorted post-consumer plastic waste of the specific polymer (EPS or PET), to produce secondary material (plastic granulate) ready for use in a new product.

For recycled EPS production (R-EPS), a mechanical recycling process of EPS waste was modelled based on the information and data available in PWC (2011). According to this source, compaction and shredding of EPS waste to produce fragmented PS scrap (for further use as a feedstock in secondary PS or EPS production) overall requires 270 kWh/t waste treated. However, moderately lower energy consumptions could be calculated from available catalogues of plastic recycling machineries and equipment (shredders, densifier and pelletisers for EPS waste and similar low-density waste plastic materials), i.e. ca. 100-150 kWh/t waste treated, depending upon machineries capacity (see, for instance,

¹³⁹ The following dataset was applied: [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*.

the technical specifications available in Promeco, 2020). The total electricity demand reported in PWC (2011) was thus assumed to cover not only compaction and shredding of EPS waste, but also densification and pelletisation of fragmented EPS scrap into secondary PS pellets (ready for use in the manufacturing of new PS products or for expansion into secondary EPS granulate). An EF-compliant dataset was used in the inventory to represent electricity supply under EU-average conditions. Similarly to PET recycling (see below), an efficiency of the recycling process equal to 85% was assumed, in the absence of specific information on EPS recycling. The residues were assumed to be landfilled or incinerated, according to the same End of Life ratio applied in the foreground model for waste EPS boards (i.e. 55% landfilling and 45% incineration). Capital goods were excluded due to lack of data.

For recycled PET production (R-PET), an aggregated EF-compliant dataset was applied¹⁴⁰, which represents the burdens of the mechanical recycling process of sorted post-consumer PET waste via grinding, washing, metal separation and pelletising, with an overall efficiency of 85.5%. The dataset is developed based on literature data for each unit operation, refers to the year 2016, and reflects EU background conditions. Process waste and scrap are assumed to be incinerated (with incineration burdens being already included in the aggregated dataset). The recycled material output from the process is represented by non-food grade secondary polymer granulate, while PET flakes are actually used in the manufacturing of R-PET insulation boards after conversion to fibres (see Section 5.5.3). However, the applied dataset is not expected to significantly overestimate the burdens associated with the production of recycled PET flakes.

Following the approach prescribed in the *Plastics LCA* method to model recycling situations (Circular Footprint Formula, CFF), only a share of the burdens of the recycling process were allocated to the recycled content in R-EPS and R-PET boards, based on the values of the A and $Q_{S_{in}}/Q_p$ factors used in the formula. The A factor was set to 0.5, based on the default application-specific value reported in Annex C of the *Plastics LCA* method for recycled EPS used in building and construction, and on the material-specific value provided for recycled PET used in unspecified applications (in the absence of an application-specific value for PET used in insulation/construction products). Therefore, only 50% of the burdens of the recycling process (per functional unit) were allocated to the recycled content in boards. However, the recycled content carried a share ($1-A = 0.5$) of the primary production burdens associated with the replaced virgin material (i.e. the same burdens that would have been credited to End of Life recycling in the previous product life cycle providing the recycled material). Since the value of the $Q_{S_{in}}/Q_p$ factor was assumed equal to 1 (being the quality of the recycled material necessarily suitable for insulation board manufacturing, and hence expected to be similar to that of the replaced virgin material), the total share of virgin EPS and PET production burdens allocated to the recycled material was again equal to 50% (i.e. $(1-A) \times Q_{S_{in}}/Q_p = (1-0.5) \times 1 = 0.5$). Note, however, that the modelled recycling processes exclude the use of any additives used to achieve comparable technical properties between recycled and virgin material. Virgin polymer production burdens were modelled as described in Section 5.5.2.1 for conventional fossil-based EPS and PET, and in Section 5.5.1.1 for the respective Feedstock Supply.

5.5.2.3 Bio-based polymers

The production of partly bio-based PUR (Bio-PUR) foam slabs was modelled using the same process described in Section 5.5.2.1 for fossil-based PUR, replacing the input of fossil-based polyols with bio-based ones. In this, perfect substitutability of fossil-based polyols with soy-based polyols was assumed, in the absence of more specific information.

To produce bio-based polyols from crude soybean oil used as feedstock, the latter is first processed into refined soybean oil, which is then converted into soy-based polyols. For

¹⁴⁰ 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 | plastic waste without metal fraction.

both processes, a combined foreground inventory was developed based on the information and data provided in OmniTech International (2010), using EF-compliant background datasets to model the associated ancillary inputs (chemicals, energy, etc.) and outputs (e.g. wastewater) under EU-average conditions. Overall, the production of 1 kg of refined soybean oil requires 1.042 kg of crude soybean oil, 0.0023 kg of NaOH, 0.156 kg of water, 0.036 MJ electricity, and 0.133 MJ heat, while 0.123 kg of wastewater are generated in the process. The production of 1 kg of polyols requires 1 kg of refined soybean oil, 1.5 MJ of electricity, and 2.24 MJ of heat. Cooling water and nitrogen are also used, and air emissions are generated from water and material use. However, details are not provided and they were excluded from the inventory. Within the crude soybean oil refining process, a negligible amount of soap is generated as a co-product with refined soybean oil, which represents the vast majority of the useful process outputs in terms of both mass and economic value. Process burdens were thus entirely allocated to refined soybean oil. During polyol production, no co-products are obtained.

5.5.2.4 CO₂-based polymers

Similarly to bio-based PUR, the production of partly CO₂-based PUR (CO₂-PUR) foam slabs was modelled using the same process described in Section 5.5.2.1 for fossil-based PUR, replacing the input of fossil-based polyols with CO₂-based ones. Also in this case, perfect substitutability of fossil-based polyols with CO₂-based polyols was assumed, in the absence of more specific information.

The modelling of CO₂-based polyol production from feedstock CO₂ and other precursors was based on the process described in Fernández-Dacosta et al. (2017). In this, one of the main precursors, i.e. propylene oxide, is partly replaced by CO₂ (circa 16.5%). A new dataset was therefore developed departing from the information and data reported in Fernández-Dacosta et al. (2017), using EF-compliant background datasets whenever possible to model supply of the associated ancillary inputs and handling of outputs under EU-average conditions. Overall, the production of 1 kg of polyols requires 1.6E-1 kg of CO₂, 8.08E-1 kg of propylene oxide, 2.16E-2 kg of glycerine, 6.19E-2 kg of propylene glycol, 5.98E-2 MJ of steam, and 1.14E-4 kg of Double Metal Cyanide (DMC) catalyst. Mineral material (zeolite) is also generated as waste (1.14E-4 kg/kg polyol). For more details, the reader is referred to Supplementary Information of Fernández-Dacosta et al. (2017; Table S.15).

As discussed in Section 5.5.1.4, the Circular Footprint Formula with an A factor equal to 0.5 and a quality ratio ($Q_{s_{in}}/Q_p$) equal to 1 was applied to model the supply of feedstock CO₂, and subsequent conversion/utilisation processes until the point of substitution between CO₂-based polyols and fossil-based ones. Therefore, only 50% of the burdens of the CO₂-based polyol synthesis process described above were allocated to CO₂-based PUR in the Polymer Production stage. However, the latter was assigned 50% of the burdens associated with the production of the replaced conventional fossil-based polyols (which was modelled as described in Section 5.5.2.1, and in Section 5.5.1.1 for the respective Feedstock Supply).

5.5.2.5 Transport of polymer granulate/foam to the product manufacturing site

Modelling of transport of polymer granulate, flakes or formed foam slabs from the polymerisation or recycling plant to the insulation board manufacturing site in Europe was based on the default transport scenarios (distances and vehicle types) specified in the *Plastics LCA* method for the route "supplier-to-factory". All polymers considered in this case study for the product in scope were assumed to be produced in the EU, including virgin EPS and PUR (import shares to EU were estimated to be about 7% and 2%, respectively; see Section 5.5.2.1). For virgin PET (relevant when applying the CFF to model recycled PET supply) imports were higher (21%). However, domestic production was assumed to avoid unnecessarily complicating the model, with no expected relevant effects on the results. Overall, the following average transport routes were thus considered:

1. 130 km by lorry (total weight >32 t; Euro 4);
2. 240 km by train (average freight); and
3. 270 km by ship (barge).

LCIs for transport through these types of vehicles were available as EF-compliant datasets, which were applied in the modelling.

5.5.3 Manufacturing Stage

For EPS-based insulation boards, the Manufacturing stage includes thermoforming of EPS granulate into foam slabs, followed by cutting of slabs into boards and packing of the boards. These operations were modelled based on the *ecoinvent* dataset “[CH] polystyrene foam slab for perimeter insulation”, excluding the burdens related to the expansion of PS to EPS (which pertain to the Polymer Production stage), and using EF-compliant background datasets to model the supply of the associated energy and ancillary material inputs under EU-average conditions. Overall, manufacturing 1 kg of packaged EPS boards requires 1 kg of EPS, 0.373 kWh of electricity, 3.55 MJ of heat, and 0.02 kg of LDPE film. Processes related to infrastructures and equipment included in the original datasets were not taken into account.

For PUR-based insulation boards, the Manufacturing stage includes cutting of foam slabs directly formed during the polymerisation step, and final packing of the boards. These activities were modelled based on the same data applied to EPS board manufacturing (see above), assuming that similar operations apply for cutting and packing. To exclude the contribution of thermoforming (which is accounted in the original data but not relevant to PUR boards), the original heat input was assumed to be used for such operation only, while electricity use was entirely attributed to the cutting and packing steps, as it was the consumption of packaging film. Therefore, to manufacture 1 kg of packaged PUR boards, 0.373 kWh of electricity and 0.02 kg of LDPE film were required.

Manufacturing of R-PET insulation boards was modelled based on the technology described in Ingrao et al. (2014) and the associated input/output data, which refer to a full-scale production facility located in Italy. The process includes the production of recycled PET fibre from recycled PET flakes (2.53 MJ of heat for drying and 2.14 kWh of electricity for extrusion to produce 1 kg of R-PET fibre), and the following utilisation of the fibre for board manufacturing (1.025 kg of R-PET fibre, 1.94 MJ of heat, 0.4 kWh of electricity, and 0.03 kg of LDPE film to manufacture 1 kg of packaged R-PET boards). Capital goods were not included due to lack of data. EF-compliant datasets representative of EU-average conditions were used to model the different energy and ancillary material inputs to the process. For additional details on the applied technology and process, the reader is referred to the original publication from Ingrao et al. (2014).

5.5.4 Distribution Stage

The transport of insulation boards from the manufacturing site to the final user was modelled based on the default transport scenario specified in the *Plastics LCA* method for the pathway *factory* → *distribution centres* → *final client*, assuming a 100% local supply chain. Due to the low density of the investigated insulations materials, a volume-limited transport was considered for each route, which included:

1. 1200 km lorry transport (total weight >32 t; Euro 4) from factory to distribution centres, assuming a 20% utilisation ratio; and
2. 250 km van transport (lorry <7.5t; Euro 3; 20% utilisation ratio) for 100% of the roundtrips from distribution centres to final users.

LCIs for transport through these types of vehicles were available as EF-compliant datasets, which were applied in the modelling considering the reported utilisation ratios.

5.5.5 End of Life Stage

This section addresses the modelling of the End of Life stage of insulation boards. Particularly, Section 5.5.5.1 describes the EU-average End of Life scenario considered as a base case for the calculation of the potential impacts of the investigated scenarios. The remaining sections (5.5.5.2 – 5.5.5.5) address the modelling of collection and transport of waste insulation boards, and of the different End of Life options applied to them. Finally, Section 5.5.5.6 provides case study-specific details on the estimate of the potential generation and release of macro-plastics at End of Life (including product litter) and of micro-plastics throughout the supply chain.

5.5.5.1 End of Life scenario

The same EU-average End of Life scenario was considered for boards made of all the investigated materials, conforming with the current PEFCRs for thermal insulation products in buildings (Avnir, 2019), which prescribes the same End of Life rates for all the products in scope, including EPS and PUR insulation. The scenario consists of landfilling (55% of waste boards collected after demolition or dismantling) and incineration (for the remaining 45%), while no recycling is considered. In this respect, it is noted that very limited data exist on the current recycling rates of specific waste insulation materials, and on any applied recycling techniques. For EPS waste, including all market applications (not only insulation), EUMEPS (2018) reports a recycling rate in the order of 8.5%. For PUR waste, including again all applications and not only insulation, ISOPA (2005) reports a recycling rate in the order of 9.5% for the year 2004. After careful evaluation of the sources, these figures were nevertheless considered obsolete and/or not supported by robust background data and analyses. Moreover, no specific rates for building insulation waste were provided. Therefore, it was considered appropriate to exclude recycling from the assumed End of Life scenario, as prescribed in the mentioned PEFCRs. However, mechanical recycling was considered in the sensitivity analysis described in Section 5.8.5.5, which investigated the effects of individually applying viable or potentially viable End of Life options for insulation boards (i.e. mechanical recycling, incineration and landfilling) as independent (100%) End of Life scenarios.

5.5.5.2 Modelling of waste board collection and transport

Collection and transport of waste insulation boards from demolition/dismantling to further treatment or disposal were modelled based on data referring to municipal plastic waste collection and transport for recycling, incineration or landfilling, as described in Section 3.5.5.2 of the Beverage Bottles case study. These data thus only approximate the burdens from collection and transport of waste insulation boards, as different pathways may be followed for construction and demolition waste due to, e.g., the application of selective demolition practices and/or *ad hoc* management schemes. However, this approximation is not expected to significantly affect the results, as the contribution of waste collection and transport to overall End of Life impacts (and hence to those of the entire product life cycle) is generally moderate (see, e.g., Rigamonti et al., 2014). Note that when EF-compliant incineration or landfilling datasets were used, no transport of collected waste boards was separately modelled, since transport burdens are already accounted for in such aggregated datasets. In this case, only collection was modelled, based on the data described above.

5.5.5.3 Modelling of sorting and recycling

This section is relevant only to the sensitivity analysis described in Section 5.8.5.5, where the impacts of the different insulation board alternatives are recalculated considering the individual application of viable or potentially viable End of Life options for the product as independent (100%) End of Life scenarios, including also mechanical recycling. However, dedicated collection and recycling pathways for construction and demolition waste (including insulation products) are not yet widely established, and

related technologies and processes are mostly under research and development at present. Available data were thus limited or not up-to-date, and a number of approximations and estimates had to be performed in the modelling, or partially representative data had to be used, as described below.

Sorting of shredded insulation boards from demolition or dismantling was modelled as described in Section 3.5.5.3 of the Beverage Bottles case study, assuming that the insulation materials, once collected and shredded, require to undergo further sorting to obtain a number of individual plastic-polymer streams for recycling. The applied data refer to the sorting of mixed plastic waste from separate collection in dedicated facilities, and were considered a reasonable approximation of the overall burdens of any sorting operation applied to insulation materials before recycling. However, it is acknowledged that such operations may differ from those applied to the sorting of mixed plastic waste from municipal or industrial collection (as reflected in the applied data) due to, e.g., the implementation of selective demolition practices and/or *ad hoc* management schemes for construction and demolition waste, of which insulation materials are part. A sorting efficiency of 100% was assumed for all insulation materials, in the absence of specific information. Note also that the applied data exclude the burdens from shredding, which is considered as part of the subsequent recycling processes (see below), although in reality it likely occurs before sorting.

Mechanical recycling of EPS was modelled based on the information and data available in PWC (2011). According to this source, compaction and shredding of EPS waste to produce fragmented PS scrap (for further use as a feedstock in secondary PS or EPS production) overall requires 270 kWh/t waste treated. However, moderately lower energy consumptions could be calculated from available catalogues of plastic recycling machineries and equipment (shredders, densifier and pelletisers for EPS waste and similar low-density waste plastic materials), i.e. ca. 100-150 kWh/t waste treated, depending upon machineries capacity (see, for instance, the technical specifications available in Promeco, 2020). The total electricity demand reported in PWC (2011) was thus assumed to cover not only compaction and shredding of EPS waste, but also densification and pelletisation of fragmented EPS scrap into secondary PS pellets (ready for use in the manufacturing of new PS products or for expansion into secondary EPS granulate). An EF-compliant dataset was used in the inventory to represent electricity supply under EU-average conditions. Similarly to PET recycling (see below), an efficiency of the recycling process equal to 85% was assumed, in the absence of specific information for EPS recycling. The residues were assumed to be landfilled or incinerated, according to the same ratio applied in the foreground model to waste insulation boards (i.e. 55% landfilling and 45% incineration). Capital goods were excluded due to lack of data.

Mechanical recycling of PUR was modelled based on the information and data available from various literature sources. According to Zevenhoven (2004), PUR foam scrap (from End of Life refrigerators and freezers, for example) can be rebonded by mixing shredded material fragments (size ~1 cm) with methylene diphenyl di-isocyanate (MDI), followed by form-shaping at 100-200°C and 30-200 bar. With this process, new PUR construction boards with excellent water and moisture resistance are obtained, or insulation foam for use in new refrigerators or freezers. This is also illustrated as one of the likely recycling pathways in ISOPA (2005). Building upon the information and data reported in Zevenhoven (2004), the recycling process was assumed to require shredding (21 kWh/t waste treated; similarly to EPS recycling), pressure bonding (250 kWh/t waste treated; calculated assuming the need to achieve a mechanical pressure of 200 bar on a volume of 0.032 m³/kg PUR, considering a PUR-board density of 31.14 kg/m³ and a conversion factor of 0.7 between electricity and obtainable mechanical energy), and addition of methylene diphenyl di-isocyanate (MDI) as a binder (0.1 kg/kg recycled PUR, i.e. ca. 10% of the produced recycled material). EF-compliant datasets were used in the inventory to model electricity and MDI supply under EU-average conditions. Likewise PET recycling, an efficiency of the recycling process equal to 85% was assumed, in the absence of specific information for PUR recycling. The residues were assumed to be

disposed of via landfilling or incineration, according to the same ratio applied to waste insulation boards in the foreground model (i.e. 55% landfilling and 45% incineration).

To model mechanical recycling of PET, an aggregated EF-compliant dataset was applied¹⁴¹, representing the production of secondary non-food grade PET granulate from sorted post-consumer plastic waste, via grinding, washing, metal separation and pelletising. The dataset is developed based on literature data for each unit operation, refers to the year 2016, and reflects EU background conditions. An overall process efficiency of 85.5% is considered in the datasets, with process waste and scrap being sent to incineration (already covered in the aggregated inventory). This assumption is in line with the typical fate of plastic recycling residues, which due to their high calorific value are often sent to incineration or co-combustion in cement kilns (Rigamonti et al., 2014).

Following the approach prescribed in the *Plastics LCA* method to model recycling situations (Circular Footprint Formula, CFF), the recycled polymer granulate or foam was assumed to replace the corresponding virgin polymer in the same form, whose primary production burdens were credited to the system. A substitution ratio between recycled and virgin polymer (i.e. a quality ratio Q_{Sout}/Q_p) equal to 1 was assumed for all materials, in the absence of suitable default values recommended in Annex C of the *Plastics LCA* method, and of specific information on the relative quality and/or price of the different recycled and virgin materials.

In applying the CFF, the A factor was set to 0.5, based on the default value specified in Annex C of the *Plastics LCA* method for recycled EPS used in building and construction (application-specific value), and for recycled PET used in unspecified applications (material-specific value, in the absence of application-specific value for PET used in insulation/construction products). For recycled PUR, no application-specific nor material-specific values were provided in Annex C, so that also for this material the A factor was set, by rule, to 0.5 (see Section 4.4.10.2.2 of the *Plastics LCA* method). Only 50% of the burdens of the sorting and recycling processes (and from previous collection and transport) were thus allocated to the system (i.e. $(1-A) = 0.5$). Similarly, only 50% of the benefits from avoided primary production of the replaced virgin polymer were assigned to the system itself (i.e. $(1-A) \times Q_{Sout}/Q_p = (1-0.5) \times 1 = 0.5$). Virgin production burdens were modelled using the same datasets or combination of datasets described in Sections 5.5.2 and 5.5.1 for the Polymer Production and Feedstock Supply stages, respectively.

5.5.5.4 Modelling of incineration

Incineration of EPS-based and PUR-based insulation boards was modelled on the basis of a common aggregated EF-compliant dataset representing incineration of unspecified (fossil-based) plastic waste in an average EU municipal waste incineration plant (*[EU-28+EFTA] Waste incineration of plastics (unspecified); waste-to-energy plant with dry flue gas treatment, including transport and pre-treatment | production mix, at consumer | unspecified plastic waste*). While material-specific EF-compliant incineration datasets were available for both PS and PUR, they provided unreliable results for a few impact categories, so that the mentioned dataset for generic plastic waste was ultimately selected as a reasonable proxy. However, for partly bio-based PUR boards, the inventoried CO₂ emission was adjusted to reflect the biogenic origin of part of the product carbon content (i.e. the original emission of fossil-based CO₂ was partly converted to a biogenic CO₂ emission). For incineration of R-PET boards, an aggregated, material-specific, EF-compliant dataset was applied (*[EU-28+EFTA] Waste incineration of PET; waste-to-energy plant with dry flue gas treatment, including transport and pre-treatment | production mix, at consumer | polyethylene terephthalate waste*).

¹⁴¹ *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 | plastic waste without metal fraction.*

All the selected datasets refer to the year 2012, and are developed based on a waste incineration model applying element-specific transfer coefficients 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). Similarly, energy recovery from waste (electricity and heat) is correlated to the specific energy content (lower heating value) of the waste material in input, considering EU-average energy efficiencies and recovery rates. For further details on the model, the reader is referred to Section 3.5.5.4 of the Beverage Bottles case study.

In line with the approach specified in the *Plastics LCA* method to model energy recovery from waste products (i.e. the Circular Footprint Formula), the product system generating the waste material sent to incineration (i.e. the insulation board life cycle, in this case) was allocated the full burdens from the incineration process. However, the system was credited with 100% of the benefits from avoided production of conventional energy (electricity and heat) assumed to be replaced by energy recovered from waste. These benefits were already included in the applied aggregated incineration datasets, where crediting is made considering the EU residual electricity grid mix, and an average EU mix of thermal energy from different heat sources defined based on statistics from the International Energy Agency (IEA).

5.5.5.5 Modelling of landfilling

In all the investigated scenarios, landfilling of insulation boards was modelled based on a common aggregated EF-compliant dataset representing disposal of non-biodegradable (fossil-based) plastic waste in a managed municipal waste landfill, referring to the year 2012 (*[EU-28+EFTA] Landfill of plastic waste; landfill including leachate treatment and with transport without collection and pre-treatment | production mix (region specific sites)*)¹⁴². The dataset is material-specific, but refers to the average chemical composition and degradability of generic plastic waste, rather than to those of the specific polymers being landfilled. This is considered an acceptable approximation for the scope of 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, including EPS, PUR and PET (i.e. degradation in the range of 1% over 100 years; Doka 2009b). The process inventory is developed based on a landfill model applying element-specific transfer coefficients to calculate the distribution of elements in the waste composition to landfill gas and leachate, and their ultimate emission to the environment over a 100-year time horizon. Emissions occurring beyond 100 years from landfilling are not accounted in the model. For further details, the reader is referred to Section 3.5.5.5 of the Beverage Bottles case study.

When modelling landfilling of partly bio-based PUR boards, the emissions to air of fossil-based CO₂ and CH₄ originally included in the applied landfilling dataset were partly converted to biogenic CO₂ and CH₄ emissions, to reflect the bio-based origin of part of the carbon content in the product. For this purpose, an overall carbon degradation rate of 1% over 100 years was assumed, according with Doka (2009b) and in line with the degradation rate estimated to be applied in the aggregated dataset, i.e. 1.4% (considering that any contribution of background activities to the originally inventoried CO₂ and CH₄ emissions is likely marginal). Similarly to boards made of the other investigated materials, carbon that is not degraded after 100 years from deposition (i.e. approximately 99% of the product carbon content) was considered to be never released from the landfill body. However, the effects of non-released biogenic carbon are not captured in the Climate Change impact indicator calculated for bio-based PUR boards, since characterisation factors for biogenic CO₂ emissions and removals are set to zero in the *Plastics LCA* method (fully conforming to the PEF method). To better understand the

¹⁴² While material-specific landfilling datasets are available from other databases (i.e. *ecoinvent*) for fossil-based PS, PUR, and PET, this EF-compliant dataset for landfilling of generic plastic waste was selected, as specifically referring to EU as the reference geography (in contrast to the available polymer-specific datasets), and to comply with the dataset selection “hierarchy” specified in the *Plastics LCA* method (Section 4.4.10.11).

implications of this methodological choice on the results, in a sensitivity analysis the Climate Change impact indicator of such bio-based alternative was thus recalculated accounting for the effects of non-released biogenic carbon (Section 5.8.5.4).

5.5.5.6 Generation and release of macro- and micro-plastics (including product litter at End of Life)

The generation (loss) and release of macro- and micro-plastics associated with the analysed product scenarios were estimated based on the *Plastic Leak Project (PLP) method* (Peano et al., 2020). The *PLP method* was applied according to the operational description reported in Section I.3 of the *Plastics LCA method*, and to the general approach specified in Section 3.5.5.6 of this report (Beverage Bottles case study). Therefore, this section only focuses on the case-specific details and the product-specific parameters considered to apply the *PLP method* to the assessed LCA scenarios.

To estimate the total loss and release of macro-plastics at the End of Life stage (due to product littering and waste mismanagement), Equations I.1 and I.2 reported in the *Plastics LCA method* were applied, respectively. Beyond the default, case-unspecific parameters specified in Table I.2 of the method itself, the product-specific parameters reported in Table 5.4 were considered to apply these equations. Such parameters were defined based on the approach described in Peano et al. (2020, pp. 74-80), taking into account the size and location of use of the product, and its residual economic value after it becomes (mismanaged) waste. Particularly, considering the typically large size of insulation boards (>25 cm), that they are used within a same defined location throughout their whole life cycle (equivalent to “in-home” use), and that they normally follow specific collection and treatment pathways applied to construction and demolition waste, the littering rate was set to 0%. This means that no portions of boards were assumed to be intentionally littered to the environment at End of Life by the user/dismantler. Moreover, for panels made of all the investigated materials except recycled PET, a low residual economic value was assumed after they have become (mismanaged) waste, resulting in final release rates to ocean and to the terrestrial environment equal to 5% and 95%, respectively. For recycled PET boards, a high residual value was assumed, with final release rates being set to 1% for both the release to ocean and to the terrestrial environment.

Table 5.4. Product-specific parameters considered to apply the *PLP method* to quantify the macro-plastics loss and release of the investigated insulation boards LCA scenarios.

Parameter ⁽¹⁾	Scenario	
	All except R-PET boards (S4)	R-PET boards (S4)
Littering rate (LR _{lit}) (%)	0	0
Release rate to ocean (Re _{locean}) (%)	5	1
Release rate to the terrestrial environment (Re _{terenv}) (%) ⁽²⁾	95	1

⁽¹⁾ For details on the meaning of each parameter, the reader is referred to Section I.3 of the *Plastics LCA method*.

⁽²⁾ Including release to freshwater sediments, as discussed in Section 3.5.5.6 (Beverage Bottles case study).

As for micro-plastics, relevant sources considered in this case study include pellet losses from product manufacturing and micro-particles from tire abrasion during foreground road transport (no textiles are used in the foreground system). The contribution of these sources to the total value-chain loss and release of micro-plastics to ocean and to the terrestrial environment was estimated according to Equations I.3-I.6 of the *Plastics LCA method*, considering the default source- and pathway-specific parameters specified in Tables I.3-I.5 of the method itself. No product-specific parameters had to be determined,

as the only case-specific parameter linking the different equations to the specific product inventory (and hence to the functional unit of each scenario) is either the amount of plastic pellets entering the product manufacturing process, or the mass of product/material transferred along each foreground road transport route and the related distance (all expressed per functional unit)¹⁴³. Apart from these parameters, the quantification was thus made by means of default parameters that are not affected by the type of product, polymer or feedstock source.

5.5.6 Calculation of the Climate Change impact from iLUC

As a base case, the potential Climate Change impact from Indirect Land Use Change (iLUC) associated with the investigated bio-based insulation board alternatives (i.e. Bio-PUR boards) was calculated according to the approach outlined in Section 4.4.15.3 of the *Plastics LCA* method. A sensitivity analysis applying an alternative method and the resulting emission factors was also performed, as described in Section 5.8.5.1.

In order to apply (recalculated) iLUC GHG emission factors from the EU 2015/1513 Directive (EC, 2015), as recommended in the *Plastics LCA* method, the specific land demand of the used feedstock (i.e. crude soybean oil; $\text{m}^2 \cdot \text{year} / \text{kg}$ feedstock) was calculated first. The calculation was based on the total aggregated amount of agriculture and arable land occupation flows reported in the dataset applied to model the production of crude soybean oil (see Section 5.5.1.3). Since the reference geography of such flows was not specified, all of them were aggregated in the calculation.

The specific land demand for feedstock (crude soybean oil) production was then converted into a demand per functional unit (FU) ($\text{m}^2 \cdot \text{year} / \text{FU}$), based on the specific feedstock consumption for polymer production (kg feedstock / kg polymer)¹⁴⁴ and the amount of polymer needed to fulfil the functional unit (reference flow) in the specific scenario (kg polymer / FU; Section 5.2). The potential Climate Change impact from iLUC was finally calculated by applying the recalculated GHG emission factors from the EU 2015/1513 Directive (kg CO₂ eq. / $\text{m}^2 \cdot \text{y}$) to the estimated land demand per functional unit. All the described calculation steps to estimate the potential Climate Change impact due to iLUC are summarised in Table 5.5.

Note that an alternative estimate of the iLUC impact was also conducted, departing from a value of specific land demand for crop (soybean) production ($\text{m}^2 \cdot \text{year} / \text{kg}$ crop) calculated based on annual crop yield data available from FAOSTAT (FAO, 2019). Considering a 5-year average yield for the period 2013-2017, similar values of the iLUC impact were obtained from both calculation routes. The iLUC impact estimated based on inventoried land occupation flows (as described above) was thus ultimately considered in the study, to keep consistency with the data actually applied in the modelling.

¹⁴³ An exception is the Average Vehicle Load (kg), which depending on the situation may be considered a value-chain specific parameter. However, a unique average default value was considered in this case study, as specified in Table I.3 of the *Plastics LCA* method.

¹⁴⁴ Calculated based on the data applied in the modelling of the Polymer Production stage, as described in Sections 5.5.2.3 (refined soybean oil and soy-based polyol production) and 5.5.2.1 (PUR production).

Table 5.5. Calculation of the potential Climate Change impact due to GHG emissions from iLUC associated with bio-based PUR insulation boards.

Scenario / Polymer	Feedstock	Land demand for feedstock production (1) [m ² ·y/kg _{feedstock}]	Feedstock demand for polymer production (2) [kg _{feedstock} /kg _{polymer}]	Polymer demand per functional unit FU (3) [kg _{polymer} /FU]	iLUC GHG emission factor [kg CO ₂ eq./ (m ² ·y)]	iLUC Climate Change impact [kg CO ₂ eq./FU]
S6 – Bio-based PUR boards	Soybean oil (EU-28)	7.06	0.406	4.89	0.204	2.85

(1) Calculated based on the agriculture and arable land occupation exchanges reported in the vertically aggregated dataset applied to model the production of crude soybean oil (Section 5.5.1.3), considering all the inventoried flows (the country of reference is not specified).

(2) As reported in Section 5.5.2.3, 1.042 kg crude soybean oil are required per kg refined soybean oil, and 1 kg refined soybean oil is required per kg polyols. Also, 0.39 kg polyols are needed per kg PUR (Section 5.5.2.1).

(3) The reference flow is 4.89 kg PUR per FU (Section 5.2).

5.6 Life Cycle Impact Assessment results

The characterised, normalised and weighted impact assessment results of the investigated product scenarios are reported in Tables 5.6–5.11. For characterised results, the contribution of the main life cycle stages is also reported, and further illustrated in Figures D.3.1-D.3.3 in Annex D.3. Consistently with the applied system boundary, the considered contributions include:

- Feedstock Supply, i.e. depending on the feedstock/scenario: (i) oil/natural gas extraction, processing, transport and possible refining, as well as transport of naphtha from refinery to downstream users (fossil-based polymers); (ii) collection, transport and sorting of post-consumer plastic waste (recycled polymers); (iii) crop (soybean) cultivation, production of crude soybean oil and transport of this to further processing (bio-based PUR); or (iv) CO₂ capture, purification, liquefaction and transport via pipeline to subsequent utilisation;
- Polymer Production, i.e. all gate-to-gate activities carried out to convert or recycle relevant feedstock materials into the specific polymer, including any transport among these activities and transport of polymer granulate, flakes or foam to the insulation board manufacturing site;
- Manufacturing, i.e. formation, cutting and packaging of insulation boards¹⁴⁵;
- Distribution, i.e. transport of insulation boards from the manufacturing site until the final user; and
- End of Life, i.e. collection, transport and treatment or disposal of waste insulation boards, as well as any avoided processes from downstream displacement of virgin materials or energy. This contribution hence represents the net impact from the End of Life stage, resulting from the balance between real burdens of the applied waste management activities and resulting benefits (if any).

The last row of Tables 5.6–5.11 also reports the total weighted impact score (single score) of individual scenarios, calculated by aggregating normalised and weighted impact assessment results across all impact categories. Single impact scores provide a more immediate and synthetic representation of the overall (relative) environmental performance of the analysed product scenarios. However, they are affected by greater uncertainty (due to the application of additional normalisation and weighting factors), and by value choices necessarily applied to define weighting factors establishing an order of relevance of the different impact categories in a European decision context. Note that all the results presented in this section are affected by the limitations and critical assumptions discussed in Section 5.4, and shall be interpreted taking them carefully into account.

¹⁴⁵ For R-PET boards, the manufacturing stage also includes production of PET fibres from recycled PET flakes.

Table 5.6. Characterised, normalised and weighted impact assessment results for fossil-based EPS insulation boards (per functional unit).

Impact category ⁽¹⁾	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.8E+00	5.9E+00	2.4E+00	5.9E-01	1.7E+00	1.2E+01	1.6E-03	3.5E-04
<i>Climate Change (fossil) [kg CO₂ eq.] ⁽²⁾</i>						<i>1.2E+01</i>		
<i>Climate Change (biogenic) [kg CO₂ eq.] ⁽²⁾</i>						<i>6.4E-03</i>		
<i>Climate Change (land use and land use change) [kg CO₂ eq.] ⁽²⁾</i>						<i>1.1E-02</i>		
Ozone Depletion [kg CFC-11 eq.] (I)	2.0E-09	1.3E-11	5.2E-12	1.1E-16	3.2E-09	5.2E-09	2.2E-07	1.5E-08
Human Toxicity - cancer [CTUh] (III)	6.3E-08	1.7E-08	3.2E-09	3.9E-09	1.9E-10	8.7E-08	2.3E-03	0.0E+00
Human toxicity - non-cancer [CTUh] (III)	6.3E-08	1.7E-08	3.2E-09	3.9E-09	1.9E-10	8.7E-08	1.8E-04	0.0E+00
Particulate matter [Disease incidence] (I)	2.2E-07	7.2E-08	5.8E-08	4.2E-08	9.6E-09	4.0E-07	6.3E-04	6.0E-05
Ionising Radiation [kBq U ²³⁵ eq.] (II)	5.0E-08	6.6E-08	4.7E-08	2.7E-08	-3.6E-08	1.5E-07	3.6E-11	2.0E-12
Photochemical Ozone Formation [kg NMVOC eq.] (II)	5.1E-02	2.1E-01	3.6E-01	2.4E-03	-3.0E-01	3.2E-01	8.0E-03	4.1E-04
Acidification [mol of H ⁺ eq.] (II)	5.9E-03	8.7E-03	6.0E-03	4.3E-03	-2.1E-03	2.3E-02	4.1E-04	2.7E-05
Eutrophication - terrestrial [mol N eq.] (II)	7.7E-03	8.2E-03	5.6E-03	4.7E-03	-3.7E-03	2.3E-02	1.3E-04	5.0E-06
Eutrophication - freshwater [kg P eq.] (II)	1.4E-02	2.7E-02	1.3E-02	2.6E-02	-6.2E-03	7.3E-02	2.9E-02	8.4E-04
Eutrophication - marine [kg N eq.] (II)	1.6E-05	6.4E-06	6.4E-06	2.9E-06	2.6E-05	5.8E-05	2.0E-06	6.4E-08
Ecotoxicity - freshwater [CTUe] (III)	1.2E-03	2.5E-03	1.3E-03	2.3E-03	-7.1E-04	6.6E-03	5.6E-07	0.0E+00
Land Use [Pt] (III)	1.3E+00	3.9E-01	1.1E-01	9.4E-02	-3.0E-02	1.9E+00	1.4E-06	1.2E-07
Water Use [m ³ world eq.] (III)	1.5E+00	5.6E+00	5.5E+01	7.0E+00	-5.2E+01	1.7E+01	1.5E-03	1.3E-04
Resource Use - mineral and metals [kg Sb eq.] (III)	3.7E-02	1.5E+00	1.7E-01	1.3E-02	1.9E-01	2.0E+00	3.0E-05	2.7E-06
Resource Use - fossils [MJ] (III)	4.7E-07	2.8E-07	2.8E-07	4.8E-08	-3.3E-07	7.5E-07	1.3E-05	1.0E-06
<i>Total weighted impact (single score)</i>								<i>1.8E-03</i>

⁽¹⁾ Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

⁽²⁾ For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 5.7. Characterised, normalised and weighted impact assessment results for fossil-based PUR insulation boards (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply (2)	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	-	1.3E+01	1.4E+00	8.7E-01	2.4E+00	1.7E+01	2.2E-03	5.0E-04
<i>Climate Change (fossil) [kg CO₂ eq.] (3)</i>						<i>1.7E+01</i>		
<i>Climate Change (biogenic) [kg CO₂ eq.] (3)</i>						<i>5.8E-03</i>		
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (3)</i>						<i>2.2E-02</i>		
Ozone Depletion [kg CFC-11 eq.] (I)	-	2.3E-05	4.4E-10	1.6E-16	4.4E-09	2.3E-05	9.9E-04	6.7E-05
Human Toxicity - cancer [CTUh] (III)	-	5.4E-08	3.0E-09	5.7E-09	3.5E-10	6.3E-08	1.6E-03	0.0E+00
Human toxicity - non-cancer [CTUh] (III)	-	5.4E-08	3.0E-09	5.7E-09	3.5E-10	6.3E-08	1.3E-04	0.0E+00
Particulate matter [Disease incidence] (I)	-	8.2E-07	3.3E-08	6.2E-08	-5.7E-09	9.1E-07	1.4E-03	1.4E-04
Ionising Radiation [kBq U ²³⁵ eq.] (II)	-	2.9E-07	3.8E-08	3.9E-08	-5.9E-08	3.0E-07	7.2E-11	3.9E-12
Photochemical Ozone Formation [kg NMVOC eq.] (II)	-	9.2E-01	5.0E-01	3.6E-03	-4.0E-01	1.0E+00	2.5E-02	1.3E-03
Acidification [mol of H ⁺ eq.] (II)	-	3.8E-02	2.2E-03	6.3E-03	-3.1E-03	4.3E-02	7.8E-04	5.2E-05
Eutrophication - terrestrial [mol N eq.] (II)	-	3.4E-02	3.8E-03	6.9E-03	-5.6E-03	3.9E-02	2.2E-04	8.7E-06
Eutrophication - freshwater [kg P eq.] (II)	-	8.7E-02	7.9E-03	3.8E-02	-9.3E-03	1.2E-01	4.8E-02	1.4E-03
Eutrophication - marine [kg N eq.] (II)	-	1.1E-04	2.9E-06	4.2E-06	3.9E-05	1.5E-04	5.4E-06	1.7E-07
Ecotoxicity - freshwater [CTUe] (III)	-	8.3E-03	7.5E-04	3.4E-03	-1.1E-03	1.1E-02	9.7E-07	0.0E+00
Land Use [Pt] (III)	-	1.5E+00	9.1E-02	1.4E-01	-5.6E-02	1.7E+00	1.3E-06	1.1E-07
Water Use [m ³ world eq.] (III)	-	8.3E+00	9.5E+00	1.0E+01	-7.3E+01	-4.5E+01	-3.9E-03	-3.5E-04
Resource Use - mineral and metals [kg Sb eq.] (III)	-	4.6E+00	2.0E-01	2.0E-02	3.5E-01	5.1E+00	7.9E-05	7.0E-06
Resource Use - fossils [MJ] (III)	-	1.2E-06	4.1E-07	7.0E-08	-3.5E-07	1.3E-06	2.2E-05	1.8E-06
<i>Total weighted impact (single score)</i>								<i>3.1E-03</i>

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) The contribution of the Feedstock Supply stage is accounted under "Polymer Production", since for reasons of data availability it could not be modelled separately in the life cycle inventory (Sections 5.5.1.1 and 5.5.2.1). The impact of the Feedstock Supply stage is thus reported to be zero in all categories.

(3) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 5.8. Characterised, normalised and weighted impact assessment results for 100% recycled EPS insulation boards (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.0E+00	3.5E+00	2.4E+00	5.9E-01	1.7E+00	9.1E+00	1.2E-03	2.6E-04
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						9.1E+00		
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						5.2E-03		
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						1.1E-02		
Ozone Depletion [kg CFC-11 eq.] (I)	5.9E-09	5.3E-10	5.2E-12	1.1E-16	3.2E-09	9.7E-09	4.1E-07	2.8E-08
Human Toxicity - cancer [CTUh] (III)	3.3E-08	8.7E-09	3.2E-09	3.9E-09	1.8E-10	4.8E-08	1.3E-03	0.0E+00
Human toxicity - non-cancer [CTUh] (III)	3.3E-08	8.7E-09	3.2E-09	3.9E-09	1.8E-10	4.8E-08	1.0E-04	0.0E+00
Particulate matter [Disease incidence] (I)	1.2E-07	4.3E-08	5.8E-08	4.2E-08	9.5E-09	2.7E-07	4.2E-04	4.0E-05
Ionising Radiation [kBq U ²³⁵ eq.] (II)	3.3E-08	3.4E-08	4.7E-08	2.7E-08	-3.6E-08	1.0E-07	2.5E-11	1.3E-12
Photochemical Ozone Formation [kg NMVOC eq.] (II)	3.6E-02	1.6E-01	3.6E-01	2.4E-03	-3.0E-01	2.6E-01	6.4E-03	3.3E-04
Acidification [mol of H ⁺ eq.] (II)	3.6E-03	4.5E-03	6.0E-03	4.3E-03	-2.1E-03	1.6E-02	2.9E-04	1.9E-05
Eutrophication - terrestrial [mol N eq.] (II)	4.5E-03	4.3E-03	5.6E-03	4.7E-03	-3.7E-03	1.5E-02	8.7E-05	3.4E-06
Eutrophication - freshwater [kg P eq.] (II)	9.4E-03	1.4E-02	1.3E-02	2.6E-02	-6.3E-03	5.6E-02	2.2E-02	6.5E-04
Eutrophication - marine [kg N eq.] (II)	9.9E-06	8.5E-06	6.4E-06	2.9E-06	2.6E-05	5.4E-05	1.9E-06	5.9E-08
Ecotoxicity - freshwater [CTUe] (III)	8.5E-04	1.3E-03	1.3E-03	2.3E-03	-7.2E-04	5.0E-03	4.3E-07	0.0E+00
Land Use [Pt] (III)	6.7E-01	2.0E-01	1.1E-01	9.4E-02	-3.0E-02	1.0E+00	7.8E-07	6.6E-08
Water Use [m ³ world eq.] (III)	1.4E+00	-3.3E+00	5.5E+01	7.0E+00	-5.2E+01	8.0E+00	7.0E-04	6.3E-05
Resource Use - mineral and metals [kg Sb eq.] (III)	2.5E-02	1.3E+00	1.7E-01	1.3E-02	1.9E-01	1.7E+00	2.5E-05	2.3E-06
Resource Use - fossils [MJ] (III)	2.5E-07	1.6E-07	2.8E-07	4.8E-08	-3.3E-07	4.1E-07	7.1E-06	5.8E-07
<i>Total weighted impact (single score)</i>								1.4E-03

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 5.9. Characterised, normalised and weighted impact assessment results for 100% recycled PET insulation boards (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.4E+00	1.6E+01	3.5E+00	1.7E+00	4.8E+00	2.8E+01	3.6E-03	7.9E-04
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						2.8E+01		
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						1.9E-02		
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						3.3E-02		
Ozone Depletion [kg CFC-11 eq.] (I)	7.9E-09	3.5E-11	5.5E-12	3.2E-16	9.3E-09	1.7E-08	7.4E-07	5.0E-08
Human Toxicity - cancer [CTUh] (III)	6.3E-08	2.5E-08	8.7E-09	1.1E-08	5.0E-10	1.1E-07	2.8E-03	0.0E+00
Human toxicity - non-cancer [CTUh] (III)	6.3E-08	2.5E-08	8.7E-09	1.1E-08	5.0E-10	1.1E-07	2.3E-04	0.0E+00
Particulate matter [Disease incidence] (I)	2.2E-07	1.4E-07	1.0E-07	1.2E-07	2.7E-08	6.1E-07	9.5E-04	9.1E-05
Ionising Radiation [kBq U ²³⁵ eq.] (II)	5.2E-08	1.9E-07	7.4E-08	7.7E-08	-1.0E-07	2.9E-07	6.8E-11	3.7E-12
Photochemical Ozone Formation [kg NMVOC eq.] (II)	3.9E-02	1.2E+00	7.6E-01	7.0E-03	-8.6E-01	1.1E+00	2.8E-02	1.4E-03
Acidification [mol of H ⁺ eq.] (II)	5.6E-03	2.0E-02	5.7E-03	1.2E-02	-6.1E-03	3.8E-02	6.8E-04	4.5E-05
Eutrophication - terrestrial [mol N eq.] (II)	7.1E-03	2.2E-02	9.0E-03	1.4E-02	-1.1E-02	4.1E-02	2.3E-04	9.1E-06
Eutrophication - freshwater [kg P eq.] (II)	1.5E-02	6.7E-02	2.1E-02	7.4E-02	-1.8E-02	1.6E-01	6.2E-02	1.8E-03
Eutrophication - marine [kg N eq.] (II)	8.9E-06	3.9E-05	9.8E-06	8.3E-06	7.5E-05	1.4E-04	5.0E-06	1.6E-07
Ecotoxicity - freshwater [CTUe] (III)	1.4E-03	6.2E-03	2.0E-03	6.7E-03	-2.1E-03	1.4E-02	1.2E-06	0.0E+00
Land Use [Pt] (III)	1.3E+00	5.9E-01	2.5E-01	2.7E-01	-8.9E-02	2.3E+00	1.7E-06	1.5E-07
Water Use [m ³ world eq.] (III)	1.8E+00	3.2E+01	6.8E+01	2.0E+01	-1.5E+02	-2.7E+01	-2.4E-03	-2.2E-04
Resource Use - mineral and metals [kg Sb eq.] (III)	2.9E-02	4.5E+00	3.2E-01	3.8E-02	5.6E-01	5.5E+00	8.4E-05	7.5E-06
Resource Use - fossils [MJ] (III)	3.7E-07	1.3E-06	6.1E-07	1.4E-07	-9.6E-07	1.4E-06	2.5E-05	2.0E-06
<i>Total weighted impact (single score)</i>								4.0E-03

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 5.10. Characterised, normalised and weighted impact assessment results for partly bio-based PUR insulation boards (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	1.4E+01	9.4E+00	1.4E+00	8.7E-01	2.4E+00	2.8E+01	3.6E-03	8.0E-04
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						1.7E+01		
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						8.8E-03		
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						1.1E+01		
Ozone Depletion [kg CFC-11 eq.] (I)	3.9E-09	2.3E-05	4.4E-10	1.6E-16	4.4E-09	2.3E-05	9.9E-04	6.7E-05
Human Toxicity - cancer [CTUh] (III)	2.1E-07	1.4E-08	3.0E-09	5.7E-09	3.5E-10	2.4E-07	6.1E-03	0.0E+00
Human toxicity - non-cancer [CTUh] (III)	2.1E-07	1.4E-08	3.0E-09	5.7E-09	3.5E-10	2.4E-07	5.0E-04	0.0E+00
Particulate matter [Disease incidence] (I)	9.1E-06	2.9E-07	3.3E-08	6.2E-08	-5.7E-09	9.5E-06	1.5E-02	1.4E-03
Ionising Radiation [kBq U ²³⁵ eq.] (II)	2.7E-07	1.9E-07	3.8E-08	3.9E-08	-5.9E-08	4.8E-07	1.1E-10	6.1E-12
Photochemical Ozone Formation [kg NMVOC eq.] (II)	1.7E-01	8.3E-01	5.0E-01	3.6E-03	-4.0E-01	1.1E+00	2.7E-02	1.4E-03
Acidification [mol of H ⁺ eq.] (II)	1.5E-02	2.7E-02	2.2E-03	6.3E-03	-3.1E-03	4.7E-02	8.5E-04	5.6E-05
Eutrophication - terrestrial [mol N eq.] (II)	3.2E-02	2.2E-02	3.8E-03	6.9E-03	-5.6E-03	6.0E-02	3.4E-04	1.3E-05
Eutrophication - freshwater [kg P eq.] (II)	1.3E-01	5.6E-02	7.9E-03	3.8E-02	-9.3E-03	2.2E-01	8.7E-02	2.6E-03
Eutrophication - marine [kg N eq.] (II)	1.8E-03	1.6E-05	2.9E-06	4.2E-06	3.9E-05	1.9E-03	6.6E-05	2.0E-06
Ecotoxicity - freshwater [CTUe] (III)	3.1E-02	5.3E-03	7.5E-04	3.4E-03	-1.1E-03	3.9E-02	3.3E-06	0.0E+00
Land Use [Pt] (III)	4.5E+01	3.6E-01	9.1E-02	1.4E-01	-5.6E-02	4.5E+01	3.4E-05	2.9E-06
Water Use [m ³ world eq.] (III)	1.7E+03	2.2E+01	9.5E+00	1.0E+01	-7.3E+01	1.7E+03	1.5E-01	1.3E-02
Resource Use - mineral and metals [kg Sb eq.] (III)	2.2E+00	6.6E-01	2.0E-01	2.0E-02	3.5E-01	3.5E+00	5.3E-05	4.7E-06
Resource Use - fossils [MJ] (III)	9.8E-07	8.1E-07	4.1E-07	7.0E-08	-3.5E-07	1.9E-06	3.3E-05	2.7E-06
<i>Total weighted impact (single score)</i>								2.0E-02

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

Table 5.11. Characterised, normalised and weighted impact assessment results for partly CO₂-based PUR insulation boards (per functional unit).

Impact category (1)	Characterised impact						Normalised impact	Weighted impact
	Feedstock supply	Polymer production	Manufacturing	Distribution	End of Life	Total		
Climate Change [kg CO ₂ eq.] (I)	3.5E-01	1.5E+01	1.4E+00	8.7E-01	2.4E+00	2.0E+01	2.6E-03	5.7E-04
<i>Climate Change (fossil) [kg CO₂ eq.] (2)</i>						2.0E+01		
<i>Climate Change (biogenic) [kg CO₂ eq.] (2)</i>						2.4E-02		
<i>Climate Change (land use and land use change) [kg CO₂ eq.] (2)</i>						2.2E-02		
Ozone Depletion [kg CFC-11 eq.] (I)	3.1E-08	2.3E-05	4.4E-10	1.6E-16	4.4E-09	2.3E-05	1.0E-03	6.7E-05
Human Toxicity - cancer [CTUh] (III)	2.0E-09	2.7E-08	3.0E-09	5.7E-09	3.5E-10	3.8E-08	1.0E-03	0.0E+00
Human toxicity - non-cancer [CTUh] (III)	2.0E-09	2.7E-08	3.0E-09	5.7E-09	3.5E-10	3.8E-08	8.1E-05	0.0E+00
Particulate matter [Disease incidence] (I)	1.7E-08	5.1E-07	3.3E-08	6.2E-08	-5.7E-09	6.2E-07	9.7E-04	9.2E-05
Ionising Radiation [kBq U ²³⁵ eq.] (II)	6.6E-09	3.8E-07	3.8E-08	4.0E-08	-5.9E-08	4.1E-07	9.7E-11	5.2E-12
Photochemical Ozone Formation [kg NMVOC eq.] (II)	2.9E-02	1.7E+00	5.0E-01	3.6E-03	-4.0E-01	1.8E+00	4.4E-02	2.2E-03
Acidification [mol of H ⁺ eq.] (II)	4.9E-04	4.6E-02	2.2E-03	6.3E-03	-3.1E-03	5.2E-02	9.3E-04	6.2E-05
Eutrophication - terrestrial [mol N eq.] (II)	7.9E-04	3.8E-02	3.8E-03	6.9E-03	-5.6E-03	4.4E-02	2.5E-04	9.7E-06
Eutrophication - freshwater [kg P eq.] (II)	1.7E-03	1.0E-01	7.9E-03	3.8E-02	-9.3E-03	1.4E-01	5.6E-02	1.6E-03
Eutrophication - marine [kg N eq.] (II)	3.6E-05	9.1E-05	2.9E-06	4.2E-06	3.9E-05	1.7E-04	6.1E-06	1.9E-07
Ecotoxicity - freshwater [CTUe] (III)	2.0E-04	9.7E-03	7.5E-04	3.4E-03	-1.1E-03	1.3E-02	1.1E-06	0.0E+00
Land Use [Pt] (III)	7.0E-02	1.4E+00	9.1E-02	1.4E-01	-5.6E-02	1.7E+00	1.3E-06	1.1E-07
Water Use [m ³ world eq.] (III)	6.4E+00	5.2E+01	9.5E+00	1.0E+01	-7.3E+01	4.9E+00	4.3E-04	3.9E-05
Resource Use - mineral and metals [kg Sb eq.] (III)	4.7E-02	7.8E+00	1.6E-01	2.0E-02	3.5E-01	8.4E+00	1.3E-04	1.1E-05
Resource Use - fossils [MJ] (III)	1.4E-06	2.1E-06	4.1E-07	7.0E-08	-3.5E-07	3.7E-06	6.4E-05	5.2E-06
<i>Total weighted impact (single score)</i>								4.8E-03

(1) Values reported in parenthesis after the unit of each impact category indicator refer to the robustness level (I, II or III) of the underlying life cycle impact assessment model. Impact categories relying on less robust models require a more careful interpretation, as results are affected by larger uncertainty.

(2) For a description of the type of GHG emissions covered under this Climate Change sub-category, please refer to Section 4.4.15 of the *Plastics LCA* method.

5.7 Additional environmental information

This section presents the results related to additional environmental impacts or aspects going beyond the default set of impact categories considered in the *Plastics LCA* method, but that are considered relevant for the investigated product category. Additional environmental impacts and aspects addressed in this study include: (i) the potential impact on Climate Change due to GHG emissions from indirect Land Use Change (iLUC); (ii) potential Biodiversity impacts occurring at the endpoint level due to a number of relevant midpoint impact categories; (iii) the generation and release of macro-plastics at End of Life (including product litter); as well as (iv) the generation and release of micro-plastics throughout the product life cycle.

5.7.1 iLUC impact on Climate Change

Table 5.12 presents the estimated potential impact on Climate Change due to GHG emissions from iLUC expected to occur as a consequence of bio-based feedstock supply in the investigated insulation boards scenarios. The total Climate Change impact accounting for such additional contribution is also reported, where relevant, for each product scenario.

Table 5.12. Potential Climate Change impact of GHG emissions from iLUC and resulting total Climate Change impact of insulation boards LCA scenarios. Results are not intended to compare the different scenarios.

Scenario	iLUC Climate Change impact [kg CO ₂ eq./FU]	Total Climate Change impact (incl. iLUC) ⁽¹⁾ [kg CO ₂ eq./FU]
S1 – Fossil-based EPS boards	-	(12.4)
S2 – Fossil-based PUR boards	-	(17)
S3 – 100% R-EPS boards	-	(9.1)
S4 – 100% R-PET boards	-	(27.8)
S5 – Bio-based PUR boards	2.85	30.8 (27.9)
S6 – CO ₂ -based PUR boards	-	(20)

⁽¹⁾ Values in parenthesis refer to the total Climate Change impact of scenarios, without the iLUC contribution.

5.7.2 Biodiversity impacts

Potential Biodiversity impacts estimated for the investigated insulation boards scenarios, expressed as potential loss of animal and vegetal species per year, are presented in Table 5.13. The impact is quantified through an endpoint-level impact indicator accounting for a number of determining midpoint impact categories, including Climate Change, Photochemical Ozone Formation, Terrestrial Acidification, Eutrophication (freshwater and marine), Ecotoxicity (terrestrial, freshwater and marine), Land Use and Water Use. However, it is important to note that the impact assessment methods applied to these underlying midpoint impact categories differ from those prescribed in the *Plastics LCA* method (where impacts are assessed at the midpoint level). Moreover, direct potential biodiversity impacts from oil leakage are not quantified (although emissions from leakage per unit of oil supplied are reported to be quite small; see Section 3.5.1.1 of the Beverage Bottles case study).

Table 5.13. Potential biodiversity impact of insulation boards LCA scenarios, expressed as potential loss of animal and vegetal species per year (species*year) per functional unit. Results are not intended to compare the different scenarios.

Scenario	Total	Feedstock supply	Polymer production	Manu-facturing	Distribution	End of Life
S1 – Fossil-based EPS boards	6.0E-08	2.5E-08	1.9E-08	9.8E-09	3.3E-09	2.3E-09
S2 – Fossil-based PUR boards	5.9E-08	0.0E+00	4.5E-08	5.1E-09	4.8E-09	3.4E-09
S3 – 100% R-EPS boards	4.1E-08	1.4E-08	1.1E-08	9.8E-09	3.3E-09	2.3E-09
S4 – 100% R-PET boards	9.8E-08	1.4E-08	5.4E-08	1.4E-08	9.4E-09	6.7E-09
S5 – Bio-based PUR boards	1.9E-07	1.4E-07	3.3E-08	5.1E-09	4.8E-09	3.4E-09
S6 – CO ₂ -based PUR boards	7.9E-08	8.9E-09	5.7E-08	5.2E-09	4.8E-09	3.4E-09

5.7.3 Macro- and micro-plastics generation and release

Table 5.14 shows the total plastic loss, final release to ocean and final release to the terrestrial environment estimated for the assessed insulation boards scenarios, specifying the contribution of both macro- and micro-plastics. The contributions of the different macro- and micro- plastics sources to the respective total releases to ocean and to the terrestrial environment are shown in Figure 5.7. As reported in Section 5.5.5.6, the considered macro-plastics sources only include mismanagement of product waste (dismantled boards) at End of Life, as no direct product littering was assumed. Micro-plastics are instead generated from pellet losses during product manufacturing and tyre abrasion during foreground road transport. Note that, while the results for all the investigated scenarios are presented together, they are not intended to compare the different scenarios, and should not be used for this purpose by the reader.

Table 5.14. Total plastic loss, final release to ocean and final release to the terrestrial environment⁽¹⁾ estimated for insulation boards LCA scenarios, including the contribution of both macro- and micro-plastics. Results are not intended to compare the different scenarios.

Indicator	Total		Macro-plastics		Micro-plastics	
	kg/FU	%	kg/FU	%	kg/FU	%
S1 – Fossil-based EPS boards						
Loss	0.311	100.0%	0.307	98.9%	3.39E-03	1.1%
Release to ocean	1.58E-02	100.0%	1.54E-02	97.5%	3.91E-04	2.5%
Release to the terrestrial environment⁽¹⁾	0.294	100.0%	0.292	99.1%	2.53E-03	0.9%
Ratio total release / loss	99.8%	-	100.0%	-	86.1%	-

S2 – Fossil-based PUR boards						
Loss	0.458	100.0%	0.453	98.9%	5.00E-03	1.1%
Release to ocean	2.32E-02	100.0%	2.26E-02	97.5%	5.77E-04	2.5%
Release to the terrestrial environment⁽¹⁾	0.434	100.0%	0.430	99.1%	3.73E-03	0.9%
Ratio total release / loss	99.8%	-	100.0%	-	86.1%	-
S3 – 100% R-EPS boards						
Loss	0.311	100.0%	0.307	98.9%	3.40E-03	1.1%
Release to ocean	1.58E-02	100.0%	1.54E-02	97.5%	3.91E-04	2.5%
Release to the terrestrial environment⁽¹⁾	0.295	100.0%	0.292	99.1%	2.54E-03	0.9%
Ratio total release / loss	99.8%	-	100.0%	-	86.1%	-
S4 – 100% R-PET boards						
Loss	0.898	100.0%	0.888	98.9%	9.83E-03	1.1%
Release to ocean	1.00E-02	100.0%	8.88E-03	88.7%	1.13E-03	11.3%
Release to the terrestrial environment⁽¹⁾	1.62E-02	100.0%	8.88E-03	54.8%	7.33E-03	45.2%
Ratio total release / loss	2.9%	-	2.0%	-	86.1%	-
S5 – Bio-based PUR boards						
Loss	0.458	100.0%	0.453	98.9%	5.00E-03	1.1%
Release to ocean	2.32E-02	100.0%	2.26E-02	97.5%	5.77E-04	2.5%
Release to the terrestrial environment⁽¹⁾	0.434	100.0%	0.430	99.1%	3.73E-03	0.9%
Ratio total release / loss	99.8%	-	100.0%	-	86.1%	-
S6 – CO₂-based PUR boards						
Loss	0.458	100.0%	0.453	98.9%	5.00E-03	1.1%
Release to ocean	2.32E-02	100.0%	2.26E-02	97.5%	5.77E-04	2.5%
Release to the terrestrial environment⁽¹⁾	0.434	100.0%	0.430	99.1%	3.73E-03	0.9%
Ratio total release / loss	99.8%	-	100.0%	-	86.1%	-

⁽¹⁾ Including release to freshwater sediments, as discussed in Section 3.5.5.6 of the Beverage Bottles case study.

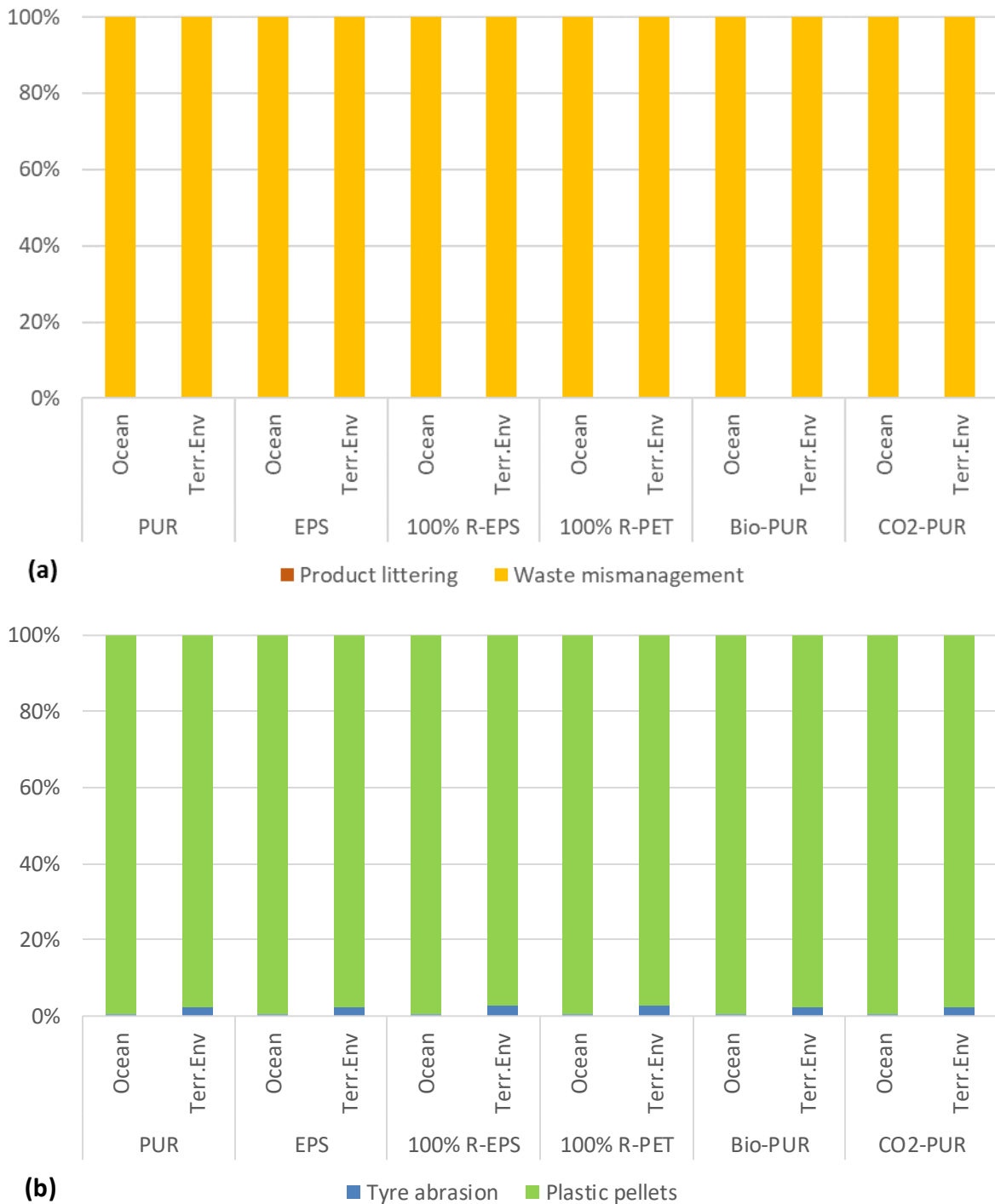


Figure 5.7. Contribution of single emission sources to the total release of macro-plastics (a) and micro-plastics (b) estimated for insulation boards LCA scenarios to both ocean (Ocean) and the terrestrial environment (Terr.Env; including freshwater sediments). Results are not intended to compare the different scenarios.

5.8 Interpretation

In the interpretation of results, the most relevant impact categories of the analysed insulation boards scenarios are firstly identified (Section 5.8.1). The contribution of individual life cycle stages to each most relevant impact category is then calculated, and the most relevant life cycle stages are identified (Section 5.8.2). The effects of GHG emissions from iLUC are also discussed (Section 5.8.3), while the results related to

macro- and micro-plastics generation and release are addressed in Section 5.8.4. Finally, the results of a sensitivity analysis on specific methodological choices and assumptions are presented (Section 5.8.5), including characterised scenario impacts calculated by individually applying each viable or potentially viable End of Life option for insulation boards (i.e. those reported in Table 5.1 plus mechanical recycling).

It is noted that most relevant processes were not identified, since the life cycle inventories of the analysed product scenarios present different levels of vertical disaggregation of included foreground processes. Therefore, the identification of most relevant processes would have not been carried out consistently across all the scenarios, and a more detailed investigation for specific scenarios would not been meaningful. The identification of most relevant elementary flows was also not undertaken, as this would have required prior identification of most relevant processes. Note, however, that any company, organisation or any other supply chain actor applying the *Plastics LCA* method shall proceed with the identification of both most relevant processes and elementary flows in each most relevant impact category.

5.8.1 Identification of most relevant impact categories

Table 5.15 shows the most relevant impact categories identified for each insulation board scenario, based on normalised and weighted impacts, according to the procedure described in Section 6.2.1 of the *Plastics LCA* method. Most relevant impact categories were hence identified as those that cumulatively contribute to at least 80% of the total normalised and weighted impact (single score) of each scenario. The contribution of toxicity-related impact categories was excluded from the calculation of total normalised and weighted impact scores, as being still based on the characterisation factors implemented in the EF 2.0 impact assessment methods applied in this study (and hence not yet updated based on REACH data)¹⁴⁶. Where needed, additional impact categories from the obtained ranking were added to the list of most relevant categories, to fulfil the requirement of having a minimum of three categories identified as most relevant.

For all the investigated scenarios, Climate Change and Resource Use – fossils were identified as the two most relevant impact categories. Following the rules of the *Plastics LCA* method, for fossil-based EPS and R-EPS boards, Photochemical Ozone Formation was also identified as a third most relevant impact category (with an average contribution of 3.5%), despite the cumulated contribution of Climate Change and Resource Use – fossils already exceeded 80%. For R-PET boards, this was the case of Acidification (showing a contribution of 3%). Conversely, for fossil-based PUR and CO₂-based PUR boards, Ozone Depletion and, limited to CO₂-based PUR boards, Water Use, were identified as additional most relevant impact categories to achieve a cumulated impact of at least 80%. For bio-based PUR boards, this was the case of Land Use, Particulate Matter and Acidification.

¹⁴⁶ According to the latest version of the Product Environmental Footprint Category Rules Guidance (EC, 2018), toxicity-related impact indicators calculated based on EF 2.0 characterization factors (applied in this study) shall be excluded from the procedure to identify the most relevant impact categories. However, any user of the *Plastics LCA* method shall apply the latest characterisation factors available at the time of the study (currently, those provided in the 3.0 EF reference package) and shall include also toxicity-related impact categories in the calculation of the total normalised and weighted impact score (and hence in the identification of the most relevant impact categories).

Table 5.15. Most relevant impact categories identified for insulation boards LCA scenarios and related contribution to the total normalised and weighted impact score of each scenario ⁽¹⁾.

S1 – Fossil-based EPS boards		S2 – Fossil-based PUR boards		S3 – 100% R-EPS boards	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Resource Use – fossils	43.3%	Climate Change	39.4%	Climate Change	47.6%
Climate Change	42%	Resource Use – fossils	36.2%	Resource Use – fossils	35.9%
Photochemical Ozone Formation	3.4%	Ozone Depletion	5.3%	Photochemical Ozone Formation	3.7%
Total	88.6%	Total	80.9%	Total	87.2%
S4 – 100% R-PET boards		S5 – Bio-based PUR boards		S6 – CO₂-based PUR boards	
<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>	<i>Impact category</i>	<i>Contrib.</i>
Climate Change	50.4%	Climate Change	47.5%	Climate Change	39.1%
Resource Use – fossils	33.8%	Resource Use – fossils	20.6%	Resource Use – fossils	34.3%
Acidification	3.1%	Land Use	6.4%	Ozone Depletion	4.6%
		Particulate Matter	4.3%	Water Use	4.5%
		Acidification	4.2%		
Total	87.3%	Total	83.1%	Total	82.6%

⁽¹⁾ Note that decimals are reported to avoid rounding errors, but do not reflect an accuracy corresponding to the number of digits shown for the specific contribution.

5.8.2 Identification of most relevant life-cycle stages

Table 5.16 shows the contribution of individual life cycle stages to the most relevant impact categories identified, for each insulation board scenario, in Section 5.8.1. The contribution was quantified according to the rules described in Sections 6.2.2 and 6.2.5 of the *Plastics LCA* method. Most relevant life cycle stages are also identified (in yellow) for each impact category, and include those that together contribute to at least 80% of the total impact in the specific category. Note that the net total impact, resulting from the algebraic sum of both positive and negative impact contributions of single life cycle stages, was considered to calculate the percentage contribution of each stage. Therefore, the contribution of specific life cycle stages may be larger than 100% if the respective impact is higher than the net total impact in the specific category, and proportionally higher than the impact of the other life cycle stages. Moreover, the sum of all positive contributions is necessarily larger than 100%, and is balanced by the negative contribution of specific life cycle stages (typically End of Life), leading to the sum of all positive and negative contributions correctly adding up to 100%. The possible negative impact and contribution from the End of Life stage is a result of the inclusion, along with the burdens of the applied waste management activities, of any benefits from secondary material production or energy recovery.

For fossil-based and recycled EPS boards, the most relevant life cycle stages in the Resource Use – fossils impact category are Feedstock Supply and Polymer Production. In the other identified most relevant categories (Climate Change and Photochemical Ozone Formation), Polymer Production and Manufacturing are the two most relevant stages, followed, in the case of fossil-based EPS boards, by Feedstock Supply. In the case of

recycled EPS boards, Polymer Production and Manufacturing are instead followed by End of Life (Climate Change) or Distribution (Photochemical Ozone Formation).

For fossil-based PUR and CO₂-based PUR boards, Polymer Production is the only most relevant life cycle stage in all the identified most relevant impact categories except Climate Change, where also End of Life is relevant. However, for conventional fossil-based PUR boards, the Polymer Production stage includes also the contribution from the stage of Feedstock Supply, as the latter could not be modelled separately in the life cycle inventory of this product alternative. Indeed, vertically aggregated cradle-to-gate datasets covering also the extraction, processing and transport of fossil-based feedstocks were used to model the production of PUR precursors, i.e. polyols and MDI (see Sections 5.5.1.1 and 5.5.2.1).

For bio-based PUR boards the picture is different than above, as in this case the Polymer Production stage is the unique most relevant life cycle stage only in Resource Use – fossils. In all the other identified most relevant impact categories, except Land Use, Feedstock Supply is the most relevant stage, followed by Polymer Production. In Land Use, Feedstock Supply is the only most relevant stage, instead.

For recycled PET boards, Polymer Production is the most relevant life cycle stage in all the identified most relevant impact categories, followed by End of Life and Manufacturing (Climate Change), Feedstock Supply (Resource Use – fossils), or Distribution (Acidification).

Table 5.16. Contribution of individual life cycle stages to the most relevant impact categories identified for each insulation board LCA scenario ⁽¹⁾. Most relevant stages (i.e. those contributing to at least 80% of the total impact) are highlighted in yellow.

S1 – Fossil-based EPS boards					
Resource Use - fossils		Climate Change		Photochemical Ozone Formation	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Feedstock Supply	63.6%	Polymer Production	47.9%	Polymer Production	38.3%
Polymer production	29.6%	Manufacturing	19.3%	Manufacturing	26.2%
Manufacturing	12.7%	Feedstock Supply	14.6%	Feedstock Supply	25.8%
Distribution	3.0%	End of Life	13.4%	Distribution	18.9%
End of Life	-9.1%	Distribution	4.8%	End of Life	-9.2%
S2 – Fossil-based PUR boards ⁽²⁾					
Climate Change		Resource Use - fossils		Ozone Depletion	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	73.2%	Polymer Production	99%	Polymer Production	100%
End of Life	13.7%	Manufacturing	8.2%	End of Life	-
Manufacturing	8%	Distribution	3.5%	Manufacturing	-
Distribution	5%	End of Life	-10.8%	Distribution	-
Feedstock Supply	-	Feedstock Supply	-	Feedstock Supply	-

S3 – 100% R-EPS boards					
Climate Change		Resource Use - fossils		Photochemical Ozone Formation	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	38.3%	Feedstock Supply	60.3%	Manufacturing	36.7%
Manufacturing	26.1%	Polymer Production	27.7%	Polymer Production	27.9%
End of Life	18.1%	Manufacturing	23.4%	Distribution	26.4%
Feedstock Supply	11%	Distribution	5.5%	Feedstock Supply	21.9%
Distribution	6.5%	End of Life	-16.9%	End of Life	-12.9%
S4 – 100% R-PET boards					
Climate Change		Resource Use - fossils		Acidification	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	59%	Polymer Production	57.7%	Polymer Production	54.0%
End of Life	17.2%	Feedstock Supply	37.8%	Distribution	33.1%
Manufacturing	12.6%	Manufacturing	16.7%	Manufacturing	21.9%
Distribution	6.2%	Distribution	5.9%	Feedstock Supply	17.2%
Feedstock Supply	5.0%	End of Life	-18.1%	End of Life	-26.2%
S5 – Bio-based PUR boards					
Climate Change		Resource Use - fossils		Land Use	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Feedstock Supply	49.5%	Polymer Production	87.1%	Feedstock Supply	101.8%
Polymer Production	33.8%	Feedstock Supply	11.7%	Polymer Production	1.3%
End of Life	8.6%	Article Production	10.8%	Distribution	0.6%
Article Production	5%	Distribution	4.6%	Article Production	0.6%
Distribution	3.1%	End of Life	-14.2%	End of Life	-4.3%
Particulate Matter		Acidification			
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>		
Feedstock Supply	56.8%	Feedstock Supply	54%		
Polymer Production	39.3%	Polymer Production	37.3%		
Distribution	8.2%	Distribution	11.6%		
Article Production	7.9%	Article Production	6.5%		
End of Life	-12.2%	End of Life	-9.3%		
S5 – CO₂-based PUR boards					
Climate Change		Resource Use - fossils		Ozone Depletion	
<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>	<i>Life cycle stage</i>	<i>Contrib.</i>
Polymer Production	75%	Polymer Production	98.3%	Polymer Production	99.8%
End of Life	12%	Manufacturing	7.4%	Feedstock Supply	0.2%
Manufacturing	6.9%	Distribution	3.2%	Manufacturing	-

Distribution	4.4%	Feedstock Supply	0.9%	Distribution	-
Feedstock Supply	1.7%	End of Life	-9.8%	End of Life	-
Water Use					
<i>Life cycle stage</i>	<i>Contrib.</i>				
Polymer Production	93.2%				
End of Life	4.1%				
Manufacturing	1.9%				
Feedstock Supply	0.6%				
Distribution	0.2%				

(¹) Note that decimals are reported to avoid rounding errors, but do not reflect an accuracy corresponding to the number of digits shown for the specific contribution.

(²) Note that the contribution of the Polymer Production stage also includes the one of the Feedstock Supply stage, as due to data availability it could not be modelled separately in the life cycle inventory (Sections 5.5.1.1 and 5.5.2.1). The contribution of the Feedstock Supply stage is thus reported to be null in all the identified most relevant impact categories.

5.8.3 Effects of indirect land use change (iLUC) on Climate Change

The additional contribution of GHG emissions from iLUC to the total Climate Change impact of the investigated bio-based alternatives (i.e. Bio-PUR insulation boards) is limited (Table 5.5). When such contribution is taken into account, the Climate Change impact of this product alternative is indeed increased by about 10% overall.

This limited increase is at least partly a consequence of the only partial bio-based content in the polymer (Bio-PUR), which is equal to 39% if considering the share of bio-based precursors (i.e. bio-based polyols) used in PUR production. Moreover, the applied GHG emission factors may also play a role, as they appeared to fall in the lower end of the range of values available in the recent literature (see Section J.2 of the *Plastics LCA* method). The use of an alternative iLUC model and of the resulting GHG emission factors may thus result in an increased Climate Change contribution from iLUC, which was explored in a sensitivity analysis (see Section 5.8.5.1).

Finally, it is noted that only the effects of GHG emissions from iLUC were addressed in this study, while nutrient-related emissions and other relevant emissions possibly associated with the additional use of converted land and/or with intensification in land use were not considered. However, accounting for such additional emissions may affect the impact of bio-based PUR boards also in other impact categories than Climate Change, such as Acidification and Eutrophication.

5.8.4 Macro- and micro-plastics generation and release

This section discusses the results presented in Section 5.7.3 on the estimated potential generation (loss) and release of macro- and micro-plastics of the assessed insulation boards scenarios.

Focusing initially on the relation between the total loss of macro- and micro-plastics and the resulting total release to ocean and to the terrestrial environment (Table 5.14), it is observed that, except for recycled PET boards, almost 100% of the estimated plastic loss from the technosphere is ultimately released to the environment (ocean or terrestrial). This is because insulation boards lost as macro-plastics at End of Life due to waste mismanagement (which is the only macro-plastics source in this case) were considered to be entirely released to the environment (ocean or terrestrial), due to the low residual economic value of the materials and to no subsequent expected take-back into the technosphere through further collection (Section 5.5.5.6). Moreover, most of generated micro-plastics are also released into the environment (about 86% overall), albeit their contribution to the total loss is negligible (as discussed below). The transfer of lost plastics to the different release compartments is thus almost entirely determined by the

release rates applied to macro-plastics (i.e. 5% to ocean and 95% to the terrestrial environment; Section 5.5.5.6). For recycled PET boards, the total release is only a minor share of the loss (3%), instead, because of the higher residual economic value of the material. As a consequence, only 2% of waste PET boards lost as macro-plastics were considered to be ultimately released to the environment (1% to ocean and 1% to the terrestrial environment), the rest being taken back into the technosphere through further collection (Section 5.5.5.6). Since, also in this case, boards lost as macro-plastics dominate the total plastic loss, the much higher overall release rates applied to the minor portion of lost micro-plastics (86%) only marginally affect the transfer of the total loss to the environment. In relative terms, if recycled PET boards are again excluded, the total plastic release to ocean accounts for only a marginal share of the total initial plastic loss and of the total final release (5% on average), while the total release to the terrestrial environment is the most relevant (95% of both the initial loss and the final release). These shares reflect the final release rates specified above for lost macro-plastics (5% to ocean and 95% to the terrestrial environment; Section 5.5.5.6) as, again, the effects of the release rates applied to micro-plastics are negligible, due to their minimal contribution to the total plastic loss. Similarly, for recycled PET boards, the shares of the total plastic loss released to ocean and to the terrestrial environment are aligned with the final release rates assumed for lost macro-plastics, being equal to 1% and 2%, respectively (in view of a release rate equalling 1% for both compartments; Section 5.5.5.6). In this case, the release to ocean accounts for 38% of the total final plastic release, with the remaining 62% being released to the terrestrial environment.

As shown in Table 5.14, the total plastic release to both ocean and the terrestrial environment is generally dominated by macro-plastics released at the End of Life, which account for 89-97.5% of the release to ocean, and for 99% of that to the terrestrial environment. Micro-plastics released throughout the upstream life cycle (via pellet loss and tyre abrasion) have only a minor role, instead. The only exception is represented by the total plastic release to the terrestrial environment associated with R-PET boards, where macro- and micro-plastics almost evenly contribute to the total final release (55% and 45%, respectively). This is due to the very low release rate of lost macro-plastics to such compartment (1%), to the higher loss of plastic pellets as micro-plastics compared to the other scenarios (which rely on lighter boards), and to the higher release rate of lost pellets to the terrestrial environment (74%) compared to ocean (12%) (Section 5.5.5.6 and Section I.3.4 of the *Plastics LCA* method). Leaving this exception apart, the generally dominant role of the macro-plastics release is a consequence of the prevailing mass of generated and released macro-plastics compared to micro-plastics, since the macro-plastics loss directly depends on the total mass of product (boards) used per functional unit (i.e. the reference flow), and is calculated based on a higher total "loss rate" compared to the considered micro-plastics sources¹⁴⁷. Moreover, the final release rate to the terrestrial environment of boards lost as macro-plastic is higher compared to the final release rates of micro-plastics lost to the same compartment¹⁴⁸. Therefore, the mass of released macro-plastics is at least one order of magnitude higher than the mass of released micro-plastics, which instead only indirectly depends on the reference flow (through parameters related to the quantity of relevant lifecycle processes)¹⁴⁹, and is calculated based on (much) lower loss and (depending on the compartment and source)

¹⁴⁷ The total "loss rate" of dismantled boards as macro-plastics at End of Life (calculated as a combination of the littering rate and of the mismanaged waste index) is equal to approximately 9% (see Table 5.4 in this report, and Equation I.1 plus Table I.2 in the *Plastics LCA* method). Conversely, the total "loss rate" of micro-plastics from tire abrasion is in the range of 1.5-2.6% (depending on the vehicle; see Equation I.3 and Table I.3 in the *Plastics LCA* method). For plastic pellets from product manufacturing, the loss rate is even lower, equalling 0.1% (see Equation I.5 and Table I.5 in the *Plastics LCA* method).

¹⁴⁸ The final release rate to the terrestrial environment is equal to 95% for insulation boards lost as macro-plastic (Table 5.4), while it is equal to 74% for micro-plastics due to pellet losses, and to 84% for those generated from tire abrasion (Tables I.5 and I.4 in the *Plastics LCA* method, including the contribution of the release to freshwater sediments).

¹⁴⁹ Including the mass of plastic pellets used for product manufacturing (which depends on the quantity of this process per functional unit), and the amount of product or material transported during each modelled road transport activity (per functional unit) with the associated distance.

release rates. Note, however, that the release of micro-plastics from tire abrasion was underestimated, due to the exclusion of the contribution of (as discussed in Section 3.5.5.6 of the Beverage Bottles case study): (a) most background transport activities; (b) intermediate transports between different process steps included within the vertically aggregated datasets used to model production of some polymers (e.g. virgin PS – affecting also partially recycled PS); and (c) any transport activities occurring within horizontally aggregated datasets.

Focusing on the sources of macro-plastics released to ocean and to the terrestrial environment (Figure 5.7a), the only contribution is provided by mismanagement of waste insulation boards after removal from buildings. No direct intentional littering of removed boards was indeed assumed to take place, based on the (large) size of the product, on its intended use, and considering that they normally end up as construction and demolition waste following specific collection and treatment pathways (Section 5.5.5.6). As for the micro-plastics release to ocean and to the terrestrial environment, the relative contribution of the different sources (loss of plastic pellets during product manufacturing and tire abrasion during road transport) generally depends on the configuration of the modelled supply-chain. The origin of the feedstock and the location of subsequent conversion processes of feedstock materials into final polymers are especially relevant, as they affect the contribution from transport activities across the life cycle. Moreover, the level of vertical disaggregation applied in the modelling of such upstream conversion processes plays an important role, as it affects the number of intermediate transport activities between different process steps which are modelled separately in the foreground inventory, and for which the contribution to micro-plastics generation can be quantified¹⁵⁰. Keeping this in mind, and considering the marginal role of micro-plastics within the total release to the environment (except for the release to the terrestrial environment of R-PET boards, as discussed above), it is observed that pellet losses during product manufacturing dominate the total release to both ocean (close to 100% on average) and terrestrial environment (97.5%) (Figure 5.7b). Micro-plastics generated from tire abrasion during road transport have a more limited role, instead (Figure 5.7b). However, the same considerations made above regarding the likely underestimate of this contribution still apply.

From a methodological perspective, it is noted how scenarios relying on lighter boards generate a lower macro-plastics loss, release to ocean and release to the terrestrial environment when the same loss and release rates are applied (i.e. for EPS-based and PUR-based boards) (Table 5.14). Due to the generally prevailing role of released macro-plastics compared to micro-plastics (as discussed above), the same apply also to the total plastic loss and release to both ocean and the terrestrial environment (Table 5.14). However, these results are a consequence of using mass-based loss and release indicators, which in the case of macro-plastics are directly and largely affected by the reference flow (mass of product used per functional unit), and hence by variations in the mass of single boards. If indicators quantifying the total number of items lost or released to the environment were applied, all scenarios would instead result in the same macro-plastics loss and release, as long as the number of items (i.e. boards) required to fulfil the functional unit would not change across the different scenarios¹⁵¹, and identical loss and release rates are applied. In such case, the contribution of macro-plastics generated from product littering and waste mismanagement would indeed be the same even for different reference flows of plastic material required per functional unit (due to e.g. different board masses). In turn, an identical macro-plastics generation and release would result in comparable total loss and releases to ocean and to the terrestrial

¹⁵⁰ When vertically aggregated datasets were applied to model the process chain involved in the conversion of feedstock materials into the final polymer (e.g. for virgin PS and, partly, for recycled PS), the contribution of intermediate transports between different process steps could not be quantified. Conversely, when such process steps were modelled individually (e.g. for fossil-based, bio-based and CO₂-based PUR), the contribution of any intermediate transports was accounted.

¹⁵¹ Which may happen, for instance, if in some scenarios the thickness required to achieve the defined thermal insulation performance would imply using two or more overlapped boards, leading to a different usage of panels per functional unit across the different scenarios.

environment, these being mostly determined by macro-plastics rather than micro-plastics (as discussed above), and provided that micro-plastics would also be quantified in terms of lost or released items.

5.8.5 Sensitivity analysis

A sensitivity analysis was performed on specific methodological choices and assumptions, to evaluate their influence and the effects of their variation on the characterised impact assessment results of the affected LCA scenario(s). The following aspects were considered:

1. Applied iLUC model and resulting GHG emission factors;
2. Source of CO₂ used as a feedstock for polyols and derived CO₂-based PUR boards;
3. Modelling of the use of CO₂ as a feedstock for polyol production;
4. Handling of non-released biogenic carbon at End of Life, temporary carbon storage and delayed carbon emissions;
5. Alternative End of Life scenarios.

This section presents the results for each of the aspects above.

5.8.5.1 Applied iLUC model and resulting GHG emission factors

This sensitivity analysis explores the use of an alternative model (i.e. Schmidt et al., 2015) to quantify the contribution of GHG emissions from iLUC to the Climate Change impact of the investigated bio-based alternatives, i.e. Bio-PUR insulation boards, in this case study. The iLUC GHG emission factor calculated with the model of Schmidt et al. (2015) was thus applied in place of the emission factor based on EC (2015) applied in the base case (see Table 5.5, Section 5.5.6 for calculation). The results are reported in Table 5.17, limited to the Climate Change impact category, which is the only one affected by this sensitivity analysis.

Applying the iLUC GHG emission factor estimated based on the model of Schmidt et al. (2015) decreases the iLUC contribution of bio-based PUR boards to Climate Change by 36%. This is because the specific emission factor for oil-based crops (including soybean) derived from such model (0.126 kg CO₂ eq./m²·y) is lower than the factor applied in the base case (0.204 kg CO₂ eq./m²·y). However, the increase in the total Climate Change impact of bio-based PUR boards due to the recalculated iLUC contribution (nearly 7%) is comparable with the increase observed in the base case (i.e. 10%). The results of this case study can thus be considered reasonably robust with respect to the estimated contribution of GHG emissions from iLUC to the Climate Change impact of bio-based PUR boards, which is limited. As discussed in Section 5.8.3, this limited contribution also reflects the only partial bio-based origin of the precursors used in PUR production (i.e. 39%, based on the share of bio-based polyols).

Table 5.17. Results of the sensitivity analysis on the emission factor applied to quantify the contribution of GHG emissions from iLUC to the Climate Change impact of bio-based PUR insulation boards, and resulting total Climate Change impact (in brackets).

Emission factors based on EC (2015) (base case) [kg CO₂ eq./FU]	Emission factors from the model of Schmidt et al. (2015) (SA) ⁽¹⁾ [kg CO₂ eq./FU]	Variation (%)
2.85 (30.8)	1.84 (29.7)	-36% (-3.31)

⁽¹⁾ SA: Sensitivity Analysis.

5.8.5.2 Source of CO₂ used in polyol production and derived CO₂-based PUR boards

This sensitivity analysis evaluates the effects of considering an alternative, more concentrated (but less abundant) source for CO₂ used as a feedstock in the production of partly CO₂-based polyols, one of the two main precursors of CO₂-based PUR boards investigated in Scenario 6. Particularly, CO₂ sourcing from ammonia synthesis plants was considered, in place of sourcing from coal-based power plants assumed in the base case. The use of this alternative source was expected to involve lower potential impacts, as the CO₂-containing gaseous stream generated during ammonia production has a higher CO₂ concentration compared with flue gases from (coal-based) power plants. Therefore, no additional capturing/extraction processes are purposefully carried out to make the (waste) CO₂ available for further use, while a less demanding purification step is generally performed before any subsequent processing (e.g. compression), transport and final use.

Based on this, CO₂ supply from ammonia production plants was modelled considering the activities of purification, liquefaction (compression) and transport to downstream users (polyol production). CO₂ extraction (capture) from the gaseous mixture of CO₂ and hydrogen generated in the ammonia production process was not included, since separation of CO₂ is regularly carried out during the process itself (to isolate hydrogen), even in the absence of any subsequent use. A concentrated CO₂ stream is thus the actual output from ammonia production, which in this study was assumed to be a waste from the process (see Section 5.5.1.4)¹⁵². Conversely, in the case of coal-based power plants, CO₂ is diluted in the generated flue gases, and capture/extraction is purposefully carried out to enable downstream utilisation, thus representing the first step of the waste-CO₂ recycling chain. Purification and liquefaction were modelled based on the same adjusted *ecoinvent* dataset applied in the base case for CO₂ sourced from coal-based power plants ("*RER* Carbon dioxide, liquid"; Section 5.5.1.4). However, consumptions of electricity and heat were further adjusted¹⁵³ to cover only the purification and liquefaction steps, while removing the input of monoethanolamine (used only for capture) and fugitive emissions of this substance to air.

The results of the analysis are shown in Table 5.18, which reports the characterised potential impacts of CO₂-based PUR boards recalculated considering the use of CO₂ sourced from ammonia production plants, and compares them with the impacts of the base case (considering CO₂ sourced from coal-based power plants). Percentage impact variations with respect to the base case scenario are also calculated and reported.

Despite the reduced energy consumption associated with CO₂ supply from ammonia production plants compared with coal-based power plants (especially heat: 0.01 vs 3.4 MJ/kg CO₂), using such an alternative CO₂ source only marginally decreases the impacts of CO₂-based PUR boards. In most impact categories, a reduction of the total impact between 0.1% and 1.2% is observed, which is negligible. This can be explained by the limited CO₂ content assumed for CO₂-based PUR in this case study, which amounts to approximately 6% by mass on dry basis (i.e. CO₂ constitutes ca. 16.5% of CO₂-based polyols, and these represent around 39% of PUR precursors).

¹⁵² Note, however, that the specific situation at the time of the assessment should be evaluated and considered when conducting any future study based on the *Plastics LCA* method. For instance, if the CO₂ separated during ammonia production would have a positive market value when leaving the production facility (i.e. after purification and possible compression/liquefaction), it should be rather considered a co-product of the ammonia production process and modelled accordingly (i.e. following the provisions in Section 4.5 of the *Plastics LCA* method).

¹⁵³ The electricity consumption was adjusted to 1.12 MJ/kg CO₂, including the demand for both purification (0.72 MJ/kg CO₂; Althaus et al., 2007) and liquefaction (0.4 MJ/kg CO₂; Von der Assen et al., 2016). The consumption of heat (for compression only), was set to 0.01 MJ/kg CO₂, based on Von der Assen et al. (2016).

Table 5.18. Results of the sensitivity analysis on the source of CO₂ used as a feedstock for the production of CO₂-based polyols (i.e. ammonia production plants instead of coal-based power plants). Results refer to the total potential impacts of CO₂-based PUR boards and are expressed per functional unit.

Impact category	CO ₂ from coal-based power plants (base case)	CO ₂ from ammonia production plants (SA) ⁽¹⁾	Variation (%)
Climate Change [kg CO ₂ eq.]	2.01E+01	2.01E+01	-0.1%
Ozone Depletion [kg CFC-11 eq.]	2.32E-05	2.32E-05	-0.002%
Human Toxicity - cancer [CTUh]	5.45E-08	5.44E-08	-0.2%
Human toxicity - non-cancer [CTUh]	8.19E-07	8.19E-07	-0.1%
Particulate matter [Disease incidence]	4.05E-07	4.04E-07	-0.1%
Ionising Radiation [kBq U ²³⁵ eq.]	1.65E+00	1.65E+00	-0.2%
Photochemical Ozone Formation [kg NMVOC eq.]	5.23E-02	5.23E-02	-0.05%
Acidification [mol of H ⁺ eq.]	4.58E-02	4.57E-02	-0.1%
Eutrophication - terrestrial [mol N eq.]	1.45E-01	1.45E-01	-0.1%
Eutrophication - freshwater [kg P eq.]	1.89E-04	1.87E-04	-1.2%
Eutrophication - marine [kg N eq.]	1.33E-02	1.33E-02	-0.3%
Ecotoxicity - freshwater [CTUe]	1.97E+00	1.97E+00	-0.2%
Land Use [Pt]	-1.12E+01	-1.12E+01	-0.5%
Water Use [m ³ world eq.]	8.19E+00	8.18E+00	-0.1%
Resource Use - mineral and metals [kg Sb eq.]	3.18E-06	3.15E-06	-1.0%
Resource Use - fossils [MJ]	3.88E+02	3.88E+02	-0.1%

(1) SA: Sensitivity Analysis.

5.8.5.3 Modelling of the use of CO₂ as a feedstock for polyol production

This sensitivity analysis explores a number of alternative methodological approaches to model the use of CO₂ as a feedstock for plastic production, considering the case of PUR insulation boards derived from CO₂-based polyols as an example. The following approaches were investigated:

1. "Cut-off" approach (raw gaseous CO₂ considered as a waste for recycling);
2. "50:50" waste allocation approach (raw gaseous CO₂ considered as a waste for recycling);
3. System Expansion via substitution (CO₂-based polyols considered as a co-product of the entire Carbon Capture and Utilisation system).

These approaches were applied in place of the Circular Footprint Formula (CFF) currently prescribed as default modelling approach in the *Plastics LCA* method, and implemented in the base case of the scenario relying on CO₂-based PUR insulation boards. A brief description of each alternative approach is provided below.

"Cut-off" (or "zero-burden") approach: according to this approach, no burdens from any upstream activity taking place before the CO₂ is generated as waste are assigned to the CO₂-based product (i.e. polyols and, later, PUR boards, in this case), since supply and conversion/use of the CO₂-providing feedstock (i.e. coal extraction, processing, transport

and combustion in power plants) are totally allocated to the CO₂ generating system (CO₂ source; i.e. electricity production). Similarly, the CO₂-based product is not assigned any cradle-to-gate burdens from primary production of the equivalent conventional product it is assumed to replace (i.e. fossil-based polyols), in contrast with the CFF. However, the CO₂-based product is allocated the full burdens from capture, purification, liquefaction of waste CO₂, and from subsequent conversion activities until the point of substitution (i.e. until CO₂-based polyol production). Downstream burdens from disposal (incineration and landfilling) of the final CO₂-based product (i.e. CO₂-based PUR insulation boards) are also totally allocated to it. Note that this approach is equivalent to applying the CFF with a value of the A factor equal to 1, reflecting a situation of much larger availability of the CO₂-based product compared to the respective demand.

"50-50" waste allocation approach: according to this approach, the burdens from upstream activities associated with the supply of the CO₂-providing feedstock (i.e. coal, in this case) are equally shared between the CO₂ generating system (i.e. coal-based electricity production) and the CO₂-based product (i.e. polyols and, later, PUR boards). The same criteria applies to the burdens from downstream disposal (incineration and landfilling) of the final CO₂-based product (i.e. CO₂-based PUR insulation boards). Therefore, 50% of the burdens from coal extraction, processing and transport to power plants were allocated to the life cycle of CO₂-based PUR boards, along with 50% of the burdens from their incineration and landfilling at End of Life. The burdens associated with capture, purification and liquefaction of waste CO₂, and with its subsequent conversion into polyols, were fully attributed to the insulation board life cycle, instead. This approach is proposed in Giegrich et al. (2018)¹⁵⁴ for a situation of equilibrium between the availability of CO₂ and its demand as a valuable raw material for CO₂-based products. However, while the approach may be considered similar to applying the CFF with an A factor equal to 0.5, different upstream and downstream activities are actually considered for allocation. For instance, while the CFF allocates the burdens of cradle-to-gate upstream activities associated with primary production of the conventional product assumed to be replaced by the CO₂-based product (i.e. fossil-based polyols, in this case), the 50-50 waste allocation approach allocates the burdens from cradle-to-gate activities associated with the supply of the CO₂-providing feedstock (i.e. coal, in this case).

System Expansion via substitution: in this approach, the Carbon Capture and Utilisation (CCU) system as a whole is considered as a starting point, rather than the CO₂ generating activity (CO₂ source) as a specific process of the investigated supply chain. The CCU system is represented by the combined production system delivering both the main product of the CO₂ source (i.e. coal-based electricity, in this case) and the CO₂-based product derived from CO₂ capture and utilisation (i.e. polyols). To calculate the burdens associated with delivering the CO₂-based product alone, the main product of the CO₂ source is assumed to replace the same product from an uncoupled production system where CO₂ capture is not performed. Cradle-to-gate burdens associated with uncoupled production of the replaced product are then credited to the CCU system as a whole. In this case study, electricity produced in the combined CCU system with CO₂-based polyols was assumed to replace electricity generated in a coal-based power plants without CO₂ capture, and the burdens from uncoupled electricity production were credited to the system itself. This ultimately led to model a system identical to that resulting from applying the "cut-off" approach, i.e. including the full burdens of the activities of CO₂ capture, purification, liquefaction, transport and conversion into CO₂-based polyols, and of all subsequent activities in the life cycle of CO₂-based PUR insulation boards, including their End of Life. However, the system was credited with an avoided release of fossil CO₂ corresponding to the amount taken up in the CO₂-based

¹⁵⁴ Note that in the approach originally proposed by Giegrich et al. (2018), only the burdens from disposal (incineration and landfilling) of the CO₂ incorporated in the CO₂-based product are shared between the CO₂-generating system (CO₂ source) and the product itself (and not the burdens from disposal of the CO₂-based product as a whole). However, this complicates the modelling, especially regarding the benefits from avoided energy generation (e.g. from incineration), as only the portion of energy recovered from CO₂ embodied in the disposed product should be considered for allocation, and this may not be easy to calculate. Therefore, disposal burdens of the CO₂-based product as a whole were allocated in this study.

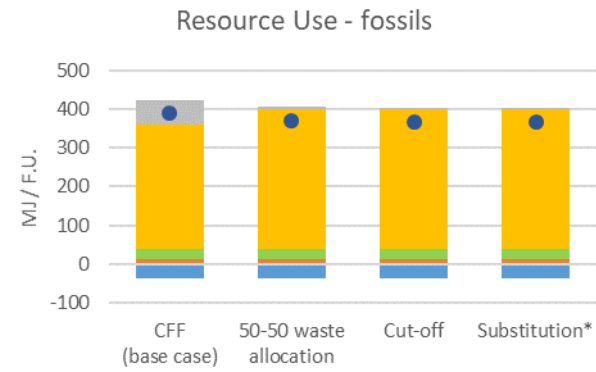
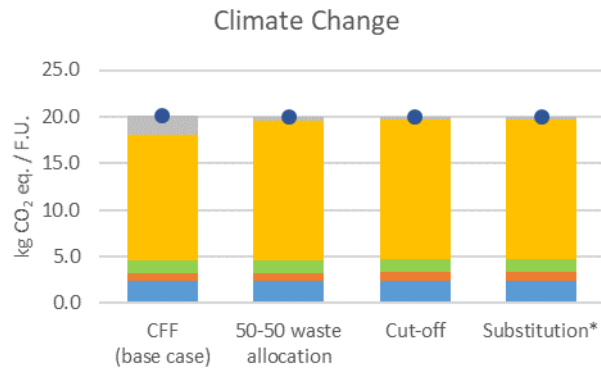
product (and physically representing the CO₂ that would otherwise be released in the uncoupled electricity production process without CO₂ capture). Note that applying the substitution approach to calculate product-specific impacts may lead to misleading results when it would generate negative overall emissions and impacts, thus being unsuitable for the purpose. The application of such approach is also problematic when an uncoupled mono-functional production process to be considered for substitution is not available for the main product of the CO₂ source, or where more processes exist (although more unlikely).

The results of the analysis are presented in Figure 5.8, which shows the impacts of CO₂-based PUR insulation boards calculated by applying the three alternative modelling approaches explored, and those calculated in the base case, where the CFF is applied. For clarity, only the two most relevant impact categories identified for CO₂-based PUR boards are addressed, i.e. Climate Change and Resource Use – fossils (see Section 5.8.1). However, similar results were obtained for the remaining impact categories when comparing the different approaches applied, as discussed below.

Overall, the difference between the individual approaches is negligible in this case study, because the amount of fossil carbon captured (as fossil CO₂-C) only constitutes a minor share of the total carbon in CO₂-based PUR boards (ca. 3%) and, therefore, of the total dry matter content of the product (ca. 1.8%). This largely explains the reduced variation across the different approaches. Moreover, by nature, cut-off and system expansion via substitution provide identical results in all impact categories except Climate Change (although this is not visible in Figure 5.8). This is because, as discussed above, the only difference between the two approaches is that system expansion via substitution credits the system for capturing the CO₂ incorporated in the product, and which would be otherwise released to air in the substituted uncoupled process of electricity production from coal. This means that, compared with the cut-off approach, the fossil CO₂ not emitted (i.e. the share of CO₂-based carbon incorporated in the product) contributes to the Climate Change impact with -1 kg CO₂-eq. per kg CO₂ taken up and not released¹⁵⁵.

Within the limited impact variation observed across the different approaches, the CFF and the 50-50 waste allocation approach generally led to slightly increased impacts compared to cut-off and system expansion via substitution, because the CO₂-based product was assigned a portion of the burdens from production of the replaced virgin polyols (CFF), or from upstream extraction and supply of the CO₂-providing fossil feedstock, i.e. coal (50-50 waste allocation approach). In particular, the CFF appeared to be the approach leading to highest impacts, albeit the magnitude was always comparable with the remaining approaches. The impacts calculated applying the 50-50 waste allocation approach were generally lower than the CFF, as in that case the CO₂-based product was discounted a share (50%) of the burdens (and benefits) from its disposal (incineration and landfilling) at End of Life, while these activities were fully allocated to the product in scope when applying the CFF. Moreover, only the burdens from upstream activities associated with coal supply were considered for allocation in the 50-50 waste allocation approach, while the burdens of all cradle-to-gate activities associated with the production of the replaced fossil-based polyols were considered according to the CFF.

¹⁵⁵ Like any other saved (i.e. negative) fossil CO₂ emission, which according to the applied impact assessment method (EF 2.0) is assigned a characterisation factor equal to 1 kg CO₂-eq./kg CO₂.



Life Cycle stage	CFF (base case)	50-50 waste allocation	Cut-off	Substitution*
Feedstock supply	2.1	0.4	0.3	0.3
Polymer production	13.4	15.0	15.0	15.0
Manufacturing	1.4	1.4	1.4	1.4
Distribution	0.9	0.9	0.9	0.9
End of Life	2.4	2.4	2.4	2.4
Total	20.1	20.0	20.0	20.0

Life cycle stage	CFF (base case)	50-50 waste allocation	Cut-off	Substitution*
Feedstock supply	63.0	5.3	3.4	3.4
Polymer production	322	361	361	361
Manufacturing	27.3	27.3	27.3	27.3
Distribution	11.7	11.7	11.7	11.7
End of Life	-35.9	-35.9	-36.1	-36.1
Total	388	370	367	367

(*) System expansion via substitution.

■ Feedstock Supply ■ Polymer Production ■ Manufacturing ■ Distribution ■ End of Life ● Total

Figure 5.8. Results of the sensitivity analysis on the approach applied to model the use of captured CO₂ as a feedstock for CO₂-based PUR insulation boards during polyol production, for Climate Change and Resource Use – fossils. The results are expressed per functional unit and include the base case, where the Circular Footprint Formula (CFF) is applied as default modelling approach.

5.8.5.4 Handling of non-released biogenic carbon at End of Life, temporary carbon storage and delayed carbon emissions

This sensitivity analysis evaluates the effects of the following methodological choices on the Climate Change impact indicator of affected product alternatives:

1. accounting for the impact of biogenic carbon not released at End of Life after 100 years from treatment/disposal of the product, i.e., in this case, from landfilling of bio-based PUR boards;
2. accounting for temporary storage of biogenic carbon in products and resulting delayed carbon emissions for the case of bio-based PUR boards; and
3. accounting for delayed emissions of fossil carbon for the case of CO₂-based PUR boards.

The effects of biogenic carbon not released at End of Life were not captured in the Climate Change impact indicator calculated in the base case, since characterisation factors for biogenic CO₂ emissions and removals are set to zero in the *Plastics LCA* method (fully conforming to the PEF method). Similarly, the effects of any temporary carbon storage in products and/or of any (resulting) delayed carbon emissions over a given time horizon were not taken into account, as according to such methods all emissions and removals throughout the product life cycle shall be accounted as occurring at the same point in time, and no “discounting” of emissions/removals over time shall be performed. To better understand the implications of these methodological choices, the Climate Change impact indicator of the affected product alternatives was thus recalculated considering the effects of both non-released biogenic carbon at End of Life and temporary carbon storage/delayed carbon emissions, as appropriate. Such effects were evaluated both individually and in combination.

To account for the effects of biogenic carbon not released after 100 years from landfilling of bio-based PUR boards, a specific CO₂ uptake was modelled in the applied landfilling dataset, corresponding to the amount of biogenic CO₂ taken up from biomass incorporated in the product, and ultimately not released (as CO₂ or CH₄) during its degradation in landfill. This uptake was then characterised applying a factor of -1 kg CO₂ eq. per kg CO₂ not released. The uptake was calculated based on the biogenic carbon content in the product (i.e. in bio-based PUR; ca. 16%), and assuming a degradation rate over 100 years equal to 1% (as reported for conventional non-biodegradable plastic materials in Doka, 2009b). This resulted in a specific CO₂ uptake equal to 0.581 kg CO₂/kg PUR sent to landfill, corresponding to 1.56 kg CO₂/FU, if considering the reference flow (4.89 kg PUR/FU) and the applied landfilling rate (55%; Section 5.5.5.1).

The effects of temporary biogenic carbon storage and delayed carbon emissions were evaluated considering a fixed time horizon of 100 years from product manufacturing, which was assumed to occur the same year of feedstock production/acquisition (i.e. biomass harvest or extraction of fossil resources; year 0). The potential impact of (delayed) emissions of biogenic carbon or CO₂-based fossil carbon in the product was then quantified taking into account the timing of the emission over such fixed time horizon, i.e. by quantifying the impact occurring only between the year of release and year 100. This calculation was made only for carbon emissions as CO₂, since carbon released as methane was negligible (less than 1% of the total carbon content in the product). For biogenic carbon, the quantification was made by applying the time-dependent characterisation factors (GWP_{bio}) provided by Guest et al. (2013) for different combinations of rotation period of the biomass used as a feedstock, and storage time of the carbon in the anthroposphere. Compared to other factors/approaches proposed to account for the effects of temporary carbon storage, the method suggested by Guest et al. (2013) is more generally applicable to all types of feedstock (forest-based and crop) and takes into account the effect of possible biomass regrowth after harvesting. The applied factors (GWP_{bio}) inherently account for the cumulated effect of both the initial CO₂ uptake during biomass growth, and of subsequent releases after a given time of storage

in the product. Therefore, the uptake of biogenic carbon (CO₂) from the biomass incorporated in the product was not separately modelled nor characterised. The impact of delayed emissions of CO₂-based fossil carbon was quantified by means of time-dependent characterisation factors calculated consistently with the principles and method used by Guest et al. (2013) to calculate the values of GWP_{bio}, while disregarding the effects from CO₂ uptake (which does not occur for fossil carbon). For clarity, the values of time-dependent GWPs applied in this analysis for delayed emissions of both biogenic and CO₂-based fossil carbon as CO₂ are reported in Table 5.19. For insulation boards, a lifetime of 50 years was assumed (Section 5.2), after which the product is either incinerated (and the carbon emitted) or landfilled (with 1% of the carbon being released). No relevant emissions were assumed to occur throughout the product life cycle.

Table 5.19. Time-dependent characterisation factors applied to quantify the effects on Climate Change from temporary storage of biogenic carbon (GWP_{bio-100}) and delayed emissions of CO₂-based fossil carbon (GWP-100). The factors applies to biogenic CO₂ emissions (GWP_{bio-100}) or to fossil CO₂-emissions originating from the CO₂-based carbon content in the product (GWP-100).

Time of emission/storage (year)	GWP _{bio-100} ^(1,2)	GWP-100
0	0	1
10	-0.07	0.921
20	-0.15	0.840
30	-0.23	0.757
40	-0.32	0.670
50	-0.40	0.580
60	-0.50	0.485
70	-0.60	0.383
80	-0.71	0.274
90	-0.84	0.154
100	-0.99	0

⁽¹⁾ From Table 1 in Guest et al. (2013).

⁽²⁾ The values of GWP_{bio} referring to a rotation period of 1 year were considered, being soybean used as a feedstock for bio-based PUR an annual crop.

Table 5.20 presents the results of the analysis, which were calculated considering the End of Life scenario applied in the base case, including 55% landfilling and 45% incineration (see Section 5.5.5.1). However, for completeness, the effects from accounting for non-released biogenic carbon were also calculated considering the application of landfilling as an individual (100%) End of Life option (as investigated in the sensitivity analysis described in Section 5.8.5.5), keeping in mind that this is an extreme situation unlikely to happen in reality (at least in a European context). The effects of temporary carbon storage and delayed carbon emissions were not explored for this situation, as not expected to be relevant. Only a negligible portion of the (biogenic or CO₂-based) carbon in the product (i.e. 1%) would indeed ultimately generate delayed emissions, the rest being stored in landfill at End of Life, and assumed to be never released (see below for further discussion on this).

Accounting for the effects of biogenic carbon not emitted at End of Life involved a certain reduction in the Climate Change impact of bio-based PUR boards, compared to the base case. The reduction was equal to 5.6% when considering the EU-average End of Life scenario, including 55% landfilling. However, it was almost doubled (-11%) when considering a theoretical 100% landfilling scenario. Overall, this limited reduction is a

consequence of the relatively low biogenic carbon content in bio-based PUR, which amounts to ca. 16% (i.e. 0.76 kg biogenic C per FU).

The effects from temporary carbon storage and/or delayed CO₂ emissions were less relevant than those from non-released biogenic carbon at End of Life, especially for CO₂-based PUR boards. For this alternative, the reduction in the Climate Change impact was relatively small, i.e. -0.5% compared to the base case. For bio-based PUR boards, the decrease was higher (-1.8%), but still marginal. Again, this result is due, on one hand, to the limited biogenic or CO₂-based fossil carbon content in the material, especially in CO₂-based PUR, where carbon derived from captured CO₂ amounted to only ca. 1.8% of PUR dry mass¹⁵⁶ (i.e. 0.09 kg CO₂-derived C per FU). Moreover, only a share (ca. 45%) of this carbon ultimately contributed to temporary storage and/or delayed emission effects, as more than half of the carbon (55%) was assumed to be landfilled at End of Life, where it was almost entirely stored and never emitted (99% of the landfilled carbon). In the case of bio-based PUR boards, this carbon was thus already accounted as non-released biogenic carbon (i.e. as "C_{bio} EoL"). In other words, this means that only a minor share of the total carbon in the product was actually eligible for (additional) credits for temporary carbon storage and/or delayed carbon emissions, which explains the relatively small savings achieved in terms of Climate Change impact.

Table 5.20. Effects on Climate Change from accounting for non-released biogenic carbon at End of Life (C_{bio} EoL) and for temporary biogenic carbon storage in products and/or delayed carbon (CO₂) emissions (C_{bio} temporary/delayed emissions), compared with the base case. Results are not intended to compare the different insulation boards scenarios. C_{bio}: Biogenic carbon; EoL: End of Life; n.r.: not relevant.

Scenario	Climate Change [kg CO ₂ eq.]			
	Base case ⁽¹⁾	C _{bio} EoL	C _{bio} temporary/ delayed emissions	C _{bio} EoL + C _{bio} temporary/ delayed emissions
Bio-based PUR boards (55% landfilling) ⁽²⁾	27.9	26.3 (-1.6; -5.6%)	27.4 (-0.5; -1.8%)	25.8 (-2.1; -7.4%)
Bio-based PUR boards (100% landfilling)	26.0	23.2 (-2.8; -11%)	n.r.	n.r.
CO ₂ -based PUR boards	20.0	n.r.	19.9 ⁽³⁾ (-0.1; -0.5%)	19.9 ⁽³⁾ (-0.1; -0.5%)

⁽¹⁾ Does not account for non-released biogenic carbon at End of Life nor for temporary biogenic carbon storage and/or delayed carbon emissions.

⁽²⁾ According with the End of Life scenario applied in the base case, including 55% landfilling and 45% incineration.

⁽³⁾ Accounts for delayed emissions only (carbon in the product is not biogenic).

5.8.5.5 Alternative End of Life scenarios

This sensitivity analysis evaluates the effects of individually applying each End of Life option considered in the EU-average End of Life scenario assumed in the base case (i.e. incineration and landfilling; Table 5.1). Albeit current application of mechanical recycling is uncertain (Section 5.5.5.1), and data availability is limited (Section 5.5.5.3), this option was also individually investigated as a potentially viable End of Life alternative for insulation boards, to better understand the implications on the respective potential impacts (although in an approximate manner). Mechanical recycling, incineration and

¹⁵⁶ Approximately 16.5% of the polyols used in CO₂-based PUR production are composed of captured CO₂, as this was assumed to only partially replace propylene oxide conventionally used in polyol production.

landfilling were thus ultimately implemented as independent (100%) End of Life scenarios replacing the EU-average scenario considered as a base case.

The main purpose of this sensitivity analysis is to evaluate how the potential impacts of each insulation board alternative are affected by changes in the applied End of Life scenario, although in reality the different considered End of Life options would be hardly implemented individually, but in combination (e.g. as reflected in the assumed EU-average End of Life scenario). The detailed numerical results are separately presented for each insulation board alternative in Tables 5.21–5.26, while a more synthetic overview is provided in Figures E.3.1–E.3.3 in Annex E.3.

Note that these results should not be interpreted as a direct comparison among alternative End of Life options for insulation boards, since the evaluation applies a “product perspective”, and burdens/benefits of some End of Life options (e.g. recycling) are partitioned between different (consecutive) product life cycles. This prevents from capturing the full environmental implications of having a given waste stream routed to such End of Life options (especially in case of recycling). Conversely, in a waste management LCA of alternative End of Life options for the product (e.g. based on a functional unit of 1 tonne of product waste to be managed), each pathway would be assigned the full burdens and benefits it involves. In such case, there is indeed no need to break mass and energy flows between the waste management system providing recovered material/energy, and the product system using them (i.e. a “system perspective” is applied, and no allocation is needed). In this perspective, the total (system-wise) benefits associated with the End of Life pathway “100% recycling” would hence be higher than those considered to calculate the results presented in this section, based on a product perspective.

For all insulation board alternatives, none of the product scenarios individually applying the three considered End of Life options (i.e. mechanical recycling, incineration and landfilling) could be identified as preferable across all the assessed impact categories. However, for all insulation alternatives, the scenario applying 100% mechanical recycling was the best one in Climate Change, Resource Use – fossils, Photochemical Ozone Formation, Human Toxicity – cancer, and Water Use. This was an expected results, because of the avoided usage of fossil fuels for primary production of the replaced virgin polymers (with savings on resource use), and due to the absence of typical air emissions from otherwise incinerating the plastic waste (e.g. CO₂ and NO_x). The 100% mechanical recycling scenario was also the best one in Ozone Depletion for all PUR-based insulation board alternatives, because of the avoided use of chemicals (e.g. MDI) for virgin polymer production.

The scenario implementing 100% incineration was preferable only in few impact categories for EPS, R-EPS, and R-PET insulation boards, including Particulate Matter, Ionising Radiation, Acidification, and Resource Use – minerals and metals. The favourable performance was mainly a result of the savings associated with the substitution of conventional electricity and heat (average EU mix), and of the reduced collection and sorting effort required compared with recycling (which involves the implementation of separate collection systems and specific sorting operations).

In most of the investigated impact categories, the product scenario applying 100% landfilling was the worst one across the different insulation board alternatives, confirming to a large extent the priority order outlined in the “Waste Hierarchy”, which sets disposal as the least preferable option to be pursued (EC, 2008). However, this was not the case in Climate Change and Water Use. For the first, this was because most (99%) of the landfilled carbon in the different (non-biodegradable) insulation materials is not degraded during the 100-year time horizon considered for modelling, thus remaining in the landfill as stored carbon and avoiding CO₂ release. For Water Use, less industrial processing is required, overall, in case of landfilling, thus involving a lower water consumption compared with mechanical recycling or incineration.

All product scenarios individually applying the three End of Life options achieved a comparable performance in the impact categories Human Toxicity – non-cancer, Ecotoxicity – freshwater, and Eutrophication (marine and freshwater), as generally these categories are mostly affected by emissions of metals (e.g. Zn, Cr, Cd, Cu and Ni) or nutrients (N and P) after the application of organic material on agricultural land, which does not apply to this case study (no compost or digestate is produced in the investigated scenarios). In the category Land Use, significant savings were observed with 100% incineration (e.g. for fossil-based PUR and R-PET insulation boards) due to the substitution effects of the average EU thermal energy mix assumed in this study.

Table 5.21. Characterised potential impacts of fossil-based EPS insulation boards for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.2E+01	7.8E+00	1.4E+01	1.1E+01
Ozone Depletion [kg CFC-11 eq.]	5.2E-09	4.3E-09	9.2E-09	2.0E-09
Human Toxicity - cancer [CTUh]	8.7E-08	5.4E-08	8.4E-08	9.0E-08
Human toxicity - non-cancer [CTUh]	4.0E-07	2.8E-07	3.4E-07	4.5E-07
Particulate matter [Disease incidence]	1.5E-07	1.5E-07	1.0E-07	2.0E-07
Ionising Radiation [kBq U ²³⁵ eq.]	3.2E-01	5.8E-01	-4.0E-02	6.2E-01
Photochemical Ozone Formation [kg NMVOC eq.]	2.3E-02	1.9E-02	2.0E-02	2.5E-02
Acidification [mol of H ⁺ eq.]	2.3E-02	2.0E-02	1.7E-02	2.7E-02
Eutrophication - terrestrial [mol N eq.]	7.3E-02	6.4E-02	6.3E-02	8.1E-02
Eutrophication - freshwater [kg P eq.]	5.8E-05	3.2E-05	1.9E-05	9.0E-05
Eutrophication - marine [kg N eq.]	6.6E-03	5.9E-03	5.5E-03	7.5E-03
Ecotoxicity - freshwater [CTUe]	1.9E+00	1.2E+00	1.8E+00	2.0E+00
Land Use [Pt]	1.7E+01	6.5E+01	-4.7E+01	6.9E+01
Water Use [m ³ world eq.]	2.0E+00	1.3E+00	2.2E+00	1.8E+00
Resource Use - mineral and metals [kg Sb eq.]	7.5E-07	8.4E-07	3.3E-07	1.1E-06
Resource Use - fossils [MJ]	2.7E+02	1.9E+02	2.3E+02	2.9E+02

Table 5.22. Characterised potential impacts of fossil-based PUR insulation boards for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	1.7E+01	9.7E+00	2.0E+01	1.5E+01
Ozone Depletion [kg CFC-11 eq.]	2.3E-05	1.5E-05	2.3E-05	2.3E-05
Human Toxicity - cancer [CTUh]	6.3E-08	5.4E-08	5.8E-08	6.7E-08
Human toxicity - non-cancer [CTUh]	9.1E-07	7.2E-07	7.9E-07	1.0E-06
Particulate matter [Disease incidence]	3.0E-07	2.1E-07	2.2E-07	3.7E-07
Ionising Radiation [kBq U ²³⁵ eq.]	1.0E+00	8.2E-01	5.3E-01	1.4E+00
Photochemical Ozone Formation [kg NMVOC eq.]	4.3E-02	2.8E-02	3.8E-02	4.7E-02
Acidification [mol of H ⁺ eq.]	3.9E-02	3.1E-02	3.1E-02	4.6E-02
Eutrophication - terrestrial [mol N eq.]	1.2E-01	9.3E-02	1.1E-01	1.4E-01
Eutrophication - freshwater [kg P eq.]	1.5E-04	7.8E-05	9.6E-05	2.0E-04
Eutrophication - marine [kg N eq.]	1.1E-02	8.9E-03	9.8E-03	1.3E-02
Ecotoxicity - freshwater [CTUe]	1.7E+00	1.1E+00	1.5E+00	1.8E+00
Land Use [Pt]	-4.5E+01 ⁽¹⁾	4.0E+00	-1.3E+02 ⁽¹⁾	2.9E+01
Water Use [m ³ world eq.]	5.1E+00	2.8E+00	1.3E+01	4.8E+00
Resource Use - mineral and metals [kg Sb eq.]	1.3E-06	8.7E-07	8.4E-07	1.7E-06
Resource Use - fossils [MJ]	3.3E+02	2.3E+02	2.9E+02	3.7E+02

⁽¹⁾ Negative impact due to the savings from heat recovery via incineration and substitution of the average EU thermal energy mix.

Table 5.23. Characterised potential impacts of 100% recycled EPS insulation boards for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	9.1E+00	4.6E+00	1.1E+01	7.6E+00
Ozone Depletion [kg CFC-11 eq.]	9.7E-09	8.8E-09	1.4E-08	6.5E-09
Human Toxicity - cancer [CTUh]	4.8E-08	1.5E-08	4.5E-08	5.1E-08
Human toxicity - non-cancer [CTUh]	2.7E-07	1.4E-07	2.1E-07	3.2E-07
Particulate matter [Disease incidence]	1.0E-07	9.9E-08	5.3E-08	1.5E-07
Ionising Radiation [kBq U ²³⁵ eq.]	2.6E-01	5.2E-01	-1.0E-01	5.6E-01
Photochemical Ozone Formation [kg NMVOC eq.]	1.6E-02	1.3E-02	1.3E-02	1.9E-02
Acidification [mol of H ⁺ eq.]	1.5E-02	1.3E-02	1.0E-02	2.0E-02
Eutrophication - terrestrial [mol N eq.]	5.6E-02	4.8E-02	4.6E-02	6.5E-02
Eutrophication - freshwater [kg P eq.]	5.4E-05	2.7E-05	1.5E-05	8.5E-05
Eutrophication - marine [kg N eq.]	5.0E-03	4.4E-03	3.9E-03	5.9E-03
Ecotoxicity - freshwater [CTUe]	1.0E+00	3.7E-01	9.4E-01	1.1E+00
Land Use [Pt]	8.0E+00	5.6E+01	-5.6E+01	6.0E+01
Water Use [m ³ world eq.]	1.7E+00	1.0E+00	1.9E+00	1.5E+00
Resource Use - mineral and metals [kg Sb eq.]	4.1E-07	5.1E-07	-6.7E-09	7.6E-07
Resource Use - fossils [MJ]	1.4E+02	6.4E+01	1.1E+02	1.7E+02

Table 5.24. Characterised potential impacts of 100% recycled PET insulation boards for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	2.8E+01	1.4E+01	3.3E+01	2.3E+01
Ozone Depletion [kg CFC-11 eq.]	1.7E-08	1.3E-08	2.9E-08	7.9E-09
Human Toxicity - cancer [CTUh]	1.1E-07	3.9E-08	9.9E-08	1.2E-07
Human toxicity - non-cancer [CTUh]	6.1E-07	3.2E-07	4.3E-07	7.5E-07
Particulate matter [Disease incidence]	2.9E-07	2.7E-07	1.4E-07	4.1E-07
Ionising Radiation [kBq U ²³⁵ eq.]	1.1E+00	1.8E+00	7.3E-02	2.0E+00
Photochemical Ozone Formation [kg NMVOC eq.]	3.8E-02	2.8E-02	2.8E-02	4.5E-02
Acidification [mol of H ⁺ eq.]	4.1E-02	3.6E-02	2.6E-02	5.4E-02
Eutrophication - terrestrial [mol N eq.]	1.6E-01	1.3E-01	1.3E-01	1.8E-01
Eutrophication - freshwater [kg P eq.]	1.4E-04	7.3E-05	2.9E-05	2.3E-04
Eutrophication - marine [kg N eq.]	1.4E-02	1.2E-02	1.1E-02	1.7E-02
Ecotoxicity - freshwater [CTUe]	2.3E+00	9.9E-01	2.0E+00	2.6E+00
Land Use [Pt]	-2.7E+01 ⁽¹⁾	1.2E+02	-2.1E+02 ⁽¹⁾	1.2E+02
Water Use [m ³ world eq.]	5.5E+00	1.6E+00	6.1E+00	4.9E+00
Resource Use - mineral and metals [kg Sb eq.]	1.4E-06	1.8E-06	2.1E-07	2.4E-06
Resource Use - fossils [MJ]	3.9E+02	1.9E+02	3.0E+02	4.7E+02

⁽¹⁾ Negative impact due to the savings from heat recovery via incineration and substitution of the average EU thermal energy mix.

Table 5.25. Characterised potential impacts of partly bio-based PUR insulation boards for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	2.8E+01	2.0E+01	3.1E+01	2.6E+01
Ozone Depletion [kg CFC-11 eq.]	2.3E-05	1.5E-05	2.3E-05	2.3E-05
Human Toxicity - cancer [CTUh]	2.4E-07	2.3E-07	2.3E-07	2.4E-07
Human toxicity - non-cancer [CTUh]	9.5E-06	9.3E-06	9.4E-06	9.6E-06
Particulate matter [Disease incidence]	4.8E-07	3.9E-07	4.0E-07	5.5E-07
Ionising Radiation [kBq U ²³⁵ eq.]	1.1E+00	9.1E-01	6.1E-01	1.5E+00
Photochemical Ozone Formation [kg NMVOC eq.]	4.7E-02	3.2E-02	4.2E-02	5.1E-02
Acidification [mol of H ⁺ eq.]	6.0E-02	5.1E-02	5.1E-02	6.6E-02
Eutrophication - terrestrial [mol N eq.]	2.2E-01	1.9E-01	2.1E-01	2.3E-01
Eutrophication - freshwater [kg P eq.]	1.9E-03	1.8E-03	1.8E-03	1.9E-03
Eutrophication - marine [kg N eq.]	3.9E-02	3.7E-02	3.8E-02	4.1E-02
Ecotoxicity - freshwater [CTUe]	4.5E+01	4.5E+01	4.5E+01	4.5E+01
Land Use [Pt]	1.7E+03	1.7E+03	1.6E+03	1.8E+03
Water Use [m ³ world eq.]	3.5E+00	1.1E+00	1.1E+01	3.1E+00
Resource Use - mineral and metals [kg Sb eq.]	1.9E-06	1.5E-06	1.5E-06	2.3E-06
Resource Use - fossils [MJ]	2.5E+02	1.5E+02	2.1E+02	2.9E+02

Table 5.26. Characterised potential impacts of partly CO₂-based PUR insulation boards for different End of Life scenarios (the EU-average scenario refers to the base case, others to the sensitivity analysis). Impacts are expressed per functional unit.

Impact category	EU-average	100% Recycling	100% Incineration	100% Landfilling
Climate Change [kg CO ₂ eq.]	2.0E+01	1.6E+01	2.6E+01	1.5E+01
Ozone Depletion [kg CFC-11 eq.]	2.3E-05	2.5E-05	2.3E-05	2.3E-05
Human Toxicity - cancer [CTUh]	3.8E-08	3.9E-08	3.6E-08	4.0E-08
Human toxicity - non-cancer [CTUh]	6.2E-07	6.0E-07	5.8E-07	6.5E-07
Particulate matter [Disease incidence]	4.1E-07	4.2E-07	4.1E-07	4.1E-07
Ionising Radiation [kBq U ²³⁵ eq.]	1.8E+00	2.0E+00	1.8E+00	1.8E+00
Photochemical Ozone Formation [kg NMVOC eq.]	5.2E-02	5.2E-02	5.3E-02	5.1E-02
Acidification [mol of H ⁺ eq.]	4.4E-02	4.5E-02	4.5E-02	4.3E-02
Eutrophication - terrestrial [mol N eq.]	1.4E-01	1.4E-01	1.5E-01	1.4E-01
Eutrophication - freshwater [kg P eq.]	1.7E-04	1.3E-04	1.3E-04	2.1E-04
Eutrophication - marine [kg N eq.]	1.3E-02	1.3E-02	1.3E-02	1.3E-02
Ecotoxicity - freshwater [CTUe]	1.7E+00	1.7E+00	1.6E+00	1.7E+00
Land Use [Pt]	4.9E+00	4.2E+00	4.5E+00	5.3E+00
Water Use [m ³ world eq.]	8.4E+00	8.4E+00	9.0E+00	7.9E+00
Resource Use - mineral and metals [kg Sb eq.]	3.7E-06	3.8E-06	3.7E-06	3.7E-06
Resource Use - fossils [MJ]	3.7E+02	3.9E+02	3.7E+02	3.7E+02

6 Main lessons learnt and recommendations from applying the *Plastics LCA* method

This section discusses the main lessons learnt from applying the *Plastics LCA* method to the illustrative case studies conducted to evaluate and demonstrate its applicability. Where relevant, the derived general recommendations are also discussed. The discussion mostly refers to the set of case studies included in this report, which were revised and updated to reflect the provisions in the final version of the *Plastics LCA* method, and relevant comments received from stakeholders during consultation. However, relevant aspects emerged during the development of the whole set of studies originally conducted to illustrate an earlier draft version of the method (see Sections 1.2 and 1.3) are also taken into account and discussed, as far as appropriate.

In general, the set of rules specified in the *Plastics LCA* method provided clear guidance on the key procedural steps, methodological choices and modelling approaches to be applied to conduct a compliant LCA study for plastic products, from setting its goal and scope, through developing appropriate lifecycle models of the product, to calculating, presenting and interpreting the results. While such rules are especially addressed to products already available on the market, they were also applicable to non-commercially available products, which can be investigated in company-internal studies provided that results are not used for external communication and company-specific data are applied to the manufacturing process (see Section 1.2 of the *Plastics LCA* method). However, a number of additional assumptions had to be inevitably made with respect to assessing commercially available products, especially regarding the End of Life stage (e.g. related to the specific options applied and the respective rates, as well as to determine relevant parameters of the Circular Footprint Formula). This may ultimately affect reproducibility of results and comparability with other studies investigating the same product.

The prescribed LCA report template was also helpful for harmonised reporting of the different case studies, based on a common general structure and specific aspects to be addressed (Annex E to the *Plastics LCA* method). However, it was not an aim of this work to strictly follow such template in every part, as the focus was rather on evaluating and demonstrating the applicability of the methodological and modelling rules of the *Plastics LCA* method. Moreover, some parts were not applicable to the illustrative case studies (e.g. those referring to data quality assessment/rating and validation). Any company or any other user of the *Plastics LCA* method shall anyway entirely conform to the template provided.

The support of a suitable LCA software tool implementing the pool of EF-compliant inventory datasets¹⁵⁷ and other relevant complementary databases¹⁵⁸ (especially including ILCD-EL compliant datasets) was required to develop specific case studies based on the *Plastics LCA* method. However, as better discussed below, additional tools (mostly spreadsheets) were also needed to calculate part of the input data to the

¹⁵⁷ EF-compliant inventory datasets are freely available to any user developing a LCA study in compliance with an existing Product Environmental Footprint Category Rule (PEFCR).

¹⁵⁸ In this respect, it is noted that, at the time of conducting the case studies, the pool of EF-compliant datasets was generally made available by LCA software providers in separate, stand-alone databases, which could not be combined with other existing life cycle inventory databases. This was made to minimise the risk of inconsistencies with respect to direct import of the datasets by the user (where possible), to avoid alteration of the same datasets from subsequent regular database updates, and, especially, to fulfil the request from the Commission of ensuring comparable impact assessment results regardless of the LCA software applied when running EF-compliant datasets. On the other hand, this prevents simultaneous use of (ILCD-EL compliant) datasets from any other database, when necessary to model the life cycle of the investigated product(s) (up to a maximum contribution to the respective total environmental impact that should not exceed 10%, as prescribed in the *Plastics LCA* method). While specific software tools allow the user to import single EF-compliant data packages within integrated databases including also datasets from other sources, this increases the risk of inconsistency (as discussed above) and of calculating wrong impact assessment results (due to, e.g., discrepancies between the applied life cycle inventory flows and characterisation factors used for impact assessment). However, the situation may be improved in the future, once the development of the latest (3.0) version of EF-compliant datasets has been completed, and their use has become more established to conduct EF-compliant studies.

lifecycle models, to quantify relevant environmental information for the product in scope (e.g. related to macro- and micro- plastics generation and release), and for the interpretation of results (e.g. to identify most relevant impact categories and life cycle stages).

Overall, the rules provided in the *Plastics LCA* method can be considered to enable an important step forward towards more robust, consistent, transparent, reproducible and verifiable LCA studies of plastic products from different feedstock sources at the EU level. On the other hand, it is recognised that a suitable background and experience in the field of LCA and related software tools is required to correctly apply the provided methodological and modelling rules, and to properly develop a compliant LCA study. Moreover, as discussed in the points below, a number of aspects that should be improved were identified, to overcome specific issues, facilitate future applicability of the method, and to further increase consistency and reproducibility of derived studies.

- Regarding the setting of the goal and scope of the study (Section 3 of the *Plastics LCA* method), in some cases it was not possible to obtain sufficiently specific, complete and/or representative data and information to define fully realistic product scenarios¹⁵⁹, to determine relevant characteristics and technical properties of the product¹⁶⁰, and/or to precisely identify actual supply-chain configuration for system boundary setting and modelling purposes¹⁶¹. Particularly, data and information available at the time of the studies regarding additives used in specific polymers and products were only partial and not specific enough for a proper assessment at the product level. Therefore, the life cycle of any additives used during polymer production and product manufacturing was excluded from the assessment (see Section 2.6). However, it is expected that most of these issues would be likely overcome when a study is directly conducted by (or on behalf of) real producers, manufacturers or other actors having more direct access to relevant data and information on the product in scope and the related supply chain, compared with accessibility achievable by the JRC.
- Defining the functional unit (FU) according to six specific aspects¹⁶² (as prescribed in the *Plastics LCA* method; Section 3.2.2.2) increases consistency, reproducibility and (wherever allowed by any specific PEFCR) comparability of studies and products. However, it is not always straightforward to quantitatively and/or sufficiently specifically define relevant aspects such as the “how well” one, which identifies the performance level to be achieved by the product. Therefore, in some cases, only a qualitative and/or generic definition could be made, and no specific, quantified performance to be provided by the product could be identified. This was the case, for instance, of agricultural mulching film, due to the variety of agronomic functions typically attributed to it, and to their generally difficult quantification (see Section 4.2 for more detail).
- Another important issue, closely linked with the one discussed in the previous point regarding the definition of the functional unit, is related to the calculation of the reference flow (i.e. the amount of material or product required to fulfil the functional unit itself; Section 3.2.2.3 of the *Plastics LCA* method). In some cases, the reference flow could not be quantified in a way ensuring that the product performance specified for the “how well” attribute of the functional unit (generally in a qualitative manner) was automatically achieved. This means that, for instance, average product masses or their typical dimensions combined with relevant inherent material properties (i.e. mostly density) were considered to calculate the reference flow. Conversely, functional properties (e.g. mechanical and barrier properties) expected to affect the

¹⁵⁹ For instance, in terms of actually applied materials or material combinations.

¹⁶⁰ Such as specific material compositions, including any additives used.

¹⁶¹ Regarding, for instance, feedstock and polymer origin, location of relevant processes, transport routes, and applied End of Life options.

¹⁶² Including the following ones: “what”, “how much”, “how well”, “how long”, “where”, “for whom” (see Section 3.2.2.2 of the *Plastics LCA* method for details).

(qualitative) technical performance specified in the functional unit could not be taken into account in such cases. Examples of this are represented by the case studies focusing on beverage bottles (where estimated average bottle masses were applied), and on mulching film (where typical average thickness and extension were considered, along with specific material densities, to calculate the reference flow). In other cases, relevant functional properties were instead (individually) taken into account (e.g. in the case studies on food packaging film and printer housing panels), although this was made by means of theoretical calculations based on the properties of a reference material and on the resulting estimated product mass. Therefore, the results might have not (fully) reflected the actual features of specific products for the real market. Overall, this was often due to the absence of a (known) explicit relationship between specific functional material properties and the resulting mass of material required for the final product to provide a defined technical performance (e.g. mechanical or agronomic). More in general, this was also a consequence of the lack of information on how the ultimate features of a product (e.g. its dimensions and/or mass) are ultimately determined to ensure a given technical performance. However, the calculation of an appropriate reference flow is expected to be relatively straightforward when a study is directly conducted by (or on behalf of) specific producers or manufacturers, involving relevant product specialists (e.g. packaging developers) with specific technical knowledge on how the investigated product is designed and ultimately developed and manufactured.

- The requirements on data collection (Section 4.6 of the *Plastics LCA* method) could not be completely fulfilled, while specific deviations had to be made when developing the case studies. These concerns both company-specific and secondary data, as detailed below.
 - While company-specific data shall be applied to model the product manufacturing process and to determine the associated material components (“bill of materials”; Section 4.6.1 of the *Plastics LCA* method), this could not be made in the case studies, as they have not investigated any real product from specific companies. Moreover, the JRC could not directly access any (measured) company-specific data, and no company-specific data on the manufacturing process of the investigated plastic products were received within the two calls for data made during the project (see Section 1.2). However, the possibility to use representative company-specific data is not expected to be an issue where the study is conducted by (or on behalf of) specific companies directly operating one or more production or manufacturing facilities, or being capable to access the required company-specific data from relevant actors downstream or upstream in the supply chain.
 - As for secondary datasets, the selection hierarchy specified in Section 4.6.3 of the *Plastics LCA* method had to be applied with some deviations^{163,164} (see Section 2.5), as representative EF-compliant or ILCD-EL compliant datasets (or suitable EF-compliant/ILCD-EL compliant proxies) were only available for part of the foreground and background processes to be modelled. Therefore, in several cases, following the original hierarchy as such would have resulted in the use of not (sufficiently) representative EF-compliant or ILCD-EL compliant proxy datasets, and/or in leaving many data gaps (after excluding from the model those processes/activities for which no suitable EF-compliant or ILCD-EL compliant proxies were available). However, while these deviations ensured the use of more representative datasets and avoided data gaps, in several scenarios they also

¹⁶³ Note that this hierarchy was applied to the selection of data for the whole set of foreground (and background) processes within the developed product inventories, including those processes that according to the rules of the *Plastics LCA* method should have been modelled based on company-specific data (which, for the reasons discussed in the point above, could not be applied).

¹⁶⁴ Note also that any user of the *Plastics LCA* method shall apply the originally prescribed data(set) selection hierarchy without any deviation, regardless of those specifically performed in the illustrative case studies and discussed further in this point.

resulted in the application of a more or less large combination of datasets from different sources (specific data providers, life cycle inventory databases and/or the literature) to develop the foreground and background product inventory. These inventories and the associated impact assessment results were thus affected, to different extents, by possible discrepancies among such datasets¹⁶⁵, and especially between EF-compliant/ILCD-EL compliant datasets and those derived from other databases or developed based on the literature (see Section 2.6 for more details). Particularly, a larger use of data(sets) from other databases and the literature was required for scenarios relying on alternative feedstock sources, as suitable EF-compliant or ILCD-EL compliant datasets or proxies were missing mostly for these scenarios (especially regarding polymer production from relevant feedstock materials and the End of Life stage; see also the last point below). The development of suitable EF-compliant datasets to properly cover these gaps is thus recommended, especially regarding the production of bio-based and CO₂-based polymers, as well as of specific recycled polymers (see also below).

- Despite the deviations described in the point above for the selection of secondary data, for a number of processes only proxy data(sets), partial inventories¹⁶⁶, or theoretical data from process simulation at the pilot scale could be applied (see Section 2.6). While these approximations were considered to be reasonable in most cases, any future application of the *Plastics LCA* method to real products relying on the same or similar processes should improve representativeness of applied data(sets). This is, for instance, the case of the recycling of plastics from waste electrical and electronic equipment, the production of relevant intermediates to Polyethylene Furanoate, as well as the production of CO₂-based polymers and intermediates.
- In the inventories of several product scenarios, the requirement of having a maximum of 10% of the total product environmental impact deriving from ILCD-EL compliant datasets (Section 4.6.3 of the *Plastics LCA* method) could not be fulfilled¹⁶⁷. As mentioned above, this was especially the case of scenarios relying on alternative feedstock sources, due to the current absence of EF-compliant datasets for the production of bio-based, CO₂-based and (to a lower extent) recycled polymers, as well as for other relevant foreground processes in the product life cycle. These include, for instance, recycling of specific polymers (e.g. PE and PP), incineration of bio-based polymers (e.g. bio-based PET, PE and PP), and biological treatment of biodegradable polymers (e.g. composting and digestion of PLA and starch-based polymers). EF-compliant datasets should thus be developed in the future for these processes, especially for polymer production from alternative feedstock sources, being the polymer production stage generally

¹⁶⁵ Discrepancies may be due, for instance, to the application of a different overall modelling approach, to the use of a different calculation method for process emissions (e.g. for agricultural processes), and/or to the use of different elementary flows to represent such emissions.

¹⁶⁶ Including only a part of the potentially relevant process burdens (inputs or emissions).

¹⁶⁷ Note that this is true regardless of the deviations discussed above with respect to the requirements on the use of company-specific data and the hierarchy to select (secondary) datasets. The use of secondary data for processes that should have been modelled based on company-specific data (i.e. product manufacturing) had indeed no or little influence on exceeding the 10% threshold for ILCD-EL compliant datasets, since EF-compliant datasets (which do not contribute to the threshold) were generally available for the manufacturing process. Moreover, this process generally provided only a minor contribution to the total lifecycle impacts of the investigated plastic products. On the other hand, even if the original dataset selection hierarchy was followed, it would have been anyway necessary to use ILCD-EL compliant datasets (or proxies) to model the production of most polymers relying on alternative feedstock sources, and this stage was generally responsible, alone, for more than 10% of the total lifecycle impact. Similarly, depending on the scenario, it might have been necessary to use ILCD-EL compliant datasets or proxies also for other foreground life cycle processes (e.g. at End of Life), thus providing additional contributions to exceeding the threshold. Finally, some product scenarios could have not been modelled if no deviations from the original hierarchy were applied, as EF-compliant or ILCD-EL compliant datasets or suitable proxies were not available for relevant life cycle processes (e.g. for the production of CO₂-based and some bio-based polymers). Therefore, checking any fulfilment of the 10% threshold on the use of ILCD-EL compliant datasets would have been automatically impossible if no scenarios had been modelled.

responsible for a significant share of the total impact. While this data gap may be filled relatively easily (by creating new EF-compliant datasets) when a study is conducted by (or on behalf of) a polymer producer having direct access to company-specific data, this may not be the case of product manufacturers or of any other downstream actor in the value chain. Indeed, the latter do not necessarily have direct access to company-specific data, and would have to rely on ILCD-EL compliant datasets (if available), with an increased probability to exceed the 10% threshold recommended for such type of datasets in the *Plastics LCA* method.

- The requirements specified in the *Plastics LCA* method (Section 4.4) for the modelling of specific processes and activities within the product life cycle aim at increased consistency, reproducibility and transparency of life cycle inventory datasets developed or applied according to such rules in a LCA study. However, while these requirements are in line with those provided in the PEF method (as suggested for update in Zampori and Pant, 2019), there may be some discrepancies with respect to the modelling approach actually applied to develop the EF-compliant datasets available at the time of conducting the case studies (i.e. EF 2.0 datasets). This may be the case of datasets developed based on older versions of the updated rules adopted in the latest version of the PEF method (and consistently implemented in the *Plastics LCA* method). For instance, the rules followed to model field emissions (especially from fertilisers) in agricultural production datasets may partially differ from those recommended in the latest PEF and *Plastics LCA* methods, and the same applies to the modelling of storage at distribution centres and retail in datasets including these activities. Moreover, EF-compliant datasets do not implement the additional or more specific modelling requirements provided in the *Plastics LCA* method, as it was finalised after the development of 2.0 EF-compliant datasets and when the update procedure for a new release (3.0) of the different data packages has already started. This means that, for instance, EF-compliant agricultural production datasets do not include the biogenic CO₂ uptake from crops, despite the specific requirement introduced in the *Plastics LCA* method to explicitly model all biogenic carbon removals and emissions occurring at such stage (albeit this exclusion has no effects on ultimate impact assessment results). Similarly, crude oil production datasets were likely developed without any prior evaluation of potentially relevant emissions to seawater from (off-shore) exploration, drilling, extraction and transport activities (e.g. drilling mud, metals from oxidation of sacrificial anodes, and particles of protective coatings), despite the additional recommendation to assess the relevance of such emissions and to include them accordingly in new or existing datasets. Full alignment of EF-compliant datasets with the modelling rules specified in the PEF and, as far as relevant, *Plastics LCA* methods should thus be checked and pursued, especially when releasing updated versions of the different EF-compliant data packages.
- The Circular Footprint Formula (CFF; Section 4.4.10.2 of the *Plastics LCA* method) could be properly applied to model the End of Life stage of all the investigated products and, where relevant, also recycled material use for product manufacturing. However, a few issues were encountered in the definition of the respective parameters, especially the A and R₂ factors, as discussed below.
 - In most cases, an application-specific default value of the A factor was not available in Annex C to the PEF and *Plastics LCA* methods¹⁶⁸, while a material specific-value had to be applied, despite being a less preferable option according with the prescribed selection hierarchy (Section 4.4.10.2.2 of the method). Moreover, for some recyclable or potentially recyclable polymers (e.g. PUR, ABS, PC/ABS blends and PLA), a material-specific value was also not provided, and a default value of 0.5 was applied, following the same rules. Additional

¹⁶⁸ Note that establishing additional or updated default values of the A factor (and of CFF parameters in general) to be included in Annex C was beyond the scope of this work.

investigations should thus be conducted in the future to expand the list of both application-specific and material-specific default values provided for the A factor, to increase consistency and reproducibility of LCA studies developed based on the *Plastics LCA* method.

- As for the R₂ factor (Section 4.4.10.2.9 of the method), a similar situation to the A factor was observed, as default application-specific values available in Annex C only covered a limited number of products or product categories¹⁶⁹, while default material-specific values were set to zero or were not provided for a number of polymers (such as those listed above for the A factor). To determine missing values, new estimates had to be generated based on available statistics on plastic waste management in the EU. However, while sufficiently representative estimates could be obtained for some products or product categories for which specific statistics or data are normally available (e.g. packaging products and plastic parts of EEE¹⁷⁰), generic estimates calculated for total plastic waste had to be applied for other products (e.g. chairs and pots), and these may not adequately represent reality for the specific application. Wherever possible, the list of default R₂ values provided in Annex C should thus be complemented to cover most common plastic products and product categories by means of application-specific estimates. Updating existing values based on most recent available statistics should also be periodically performed, to ensure application of the most representative values in compliant LCA studies (which aims at reflecting the situation occurring at the time of the assessment).

Any future development of PEFCRs for specific product categories, based on the *Plastics LCA* and PEF methods, should help to progressively overcome the issues reported above to determine the A and R₂ factors of the CFF. The provision of sufficiently specific and representative values for the product categories in scope should indeed be part of the development process.

- Focusing more specifically on the applicability of the method used to estimate the generation and release of macro- and micro- plastics from the product life cycle (i.e. the *Plastic Leak Project (PLP) method*; Section 4.4.10.12 and Annex H of the *Plastics LCA* method), the following considerations can be made.
 - Calculations had to be conducted outside the applied LCA software tool, based on separate spreadsheet versions of the life cycle inventories of the investigated product scenarios. The life cycle models developed in the software could indeed not be used as a basis for calculation, since the *PLP method* was not implemented in any of the existing LCA tools at the time of conducting the case studies. While the calculations to be performed were relatively straightforward and limited to foreground life cycle processes (as discussed in the point below), the use of an alternative and less structured version of the different product inventories increased the risk of inconsistencies compared with the developed life cycle models, and the risk of errors in calculations. Moreover, these risks would increase in case more complex and articulated product systems, relying on a larger number of both foreground and background datasets as multiple interconnected processes, had to be investigated. To reduce such risks, the time needed for calculation, and to facilitate overall applicability, it should thus be considered to implement the *PLP method* (and/or any other suitable method to quantify macro- and micro-plastics generation and release) within available LCA software tools. At the same time, life cycle inventory databases, including the pool of EF-compliant datasets, should be adapted to enable the application of such method(s) within specific tools (for instance, by including relevant elementary flows for macro- and micro- plastics based on a common and harmonised nomenclature).

¹⁶⁹ Including PET and PC (water) bottles, generic plastic packaging, as well as PE, PP, PS, EPS and PVC used in the building and construction sector.

¹⁷⁰ Electrical and Electronic Equipment.

- The possible contribution to macro- and micro-plastics generation and release of single processes and activities modelled within vertically aggregated datasets could not be quantified. Similarly, any contribution from background processes and activities associated with both vertically and horizontally aggregated datasets included in the different product inventories could not be accounted. Relevant parameters needed for calculation were indeed not available for those activities and processes, including essential data such as specific process quantities and the associated product or service flows. Overall, this led to underestimate more or less largely the total macro- and micro-plastics generation and release of the investigated product scenarios, depending on the level of disaggregation of the respective life cycle inventories. This was especially the case of micro-plastics from tire abrasion, due to the exclusion of the contribution of intermediate transports between the different process steps covered by vertically aggregated datasets, and to the generally larger presence of road transport activities among unaccounted background processes (compared with the other considered micro- and macro-plastic sources). Additionally, the estimate could not be conducted consistently across all the different scenarios analysed for a specific case study, as the respective life cycle inventories were characterised by different levels of vertical disaggregation (for reason of data availability), and the contribution of single processes could thus not be accounted uniformly. To overcome these issues, aggregated datasets should be adjusted to include relevant information and/or specific data useful to the estimate the associated generation and release of macro- and micro-plastics. More generally, in view of making the *PLP method* and/or any other suitable quantification method operational for use within existing LCA software tools (as discussed above), such information and data should be consistently included in all datasets provided with common life cycle inventory databases (including EF-compliant data packages), regardless of the aggregated or disaggregated nature of the datasets themselves.
 - The values of release to the environment calculated for biodegradable plastic products across the different case studies were even significantly overestimated, as the applied *PLP method* does not account for the effects of biodegradation, nor of any other environmental mechanism affecting the fate of released plastic items beyond any initial (short-term) redistribution among different environmental compartments. This is mostly due to only partial understanding and investigation of fragmentation and biodegradation pathways of plastic products into the environment, leading to a lack of sufficiently complete, representative and consistent estimates for biodegradation rates of relevant polymers and products in different compartments. For transparency, the release of biodegradable plastic products was clearly differentiated from that associated with non-biodegradable ones, and the potential overestimate compared with the actual release ultimately occurring after complete biodegradation was acknowledged, wherever relevant. Nevertheless, the *PLP method* and the associated release indicators should be adjusted to account for the effects of biodegradation, as soon as sufficiently reliable, consistent and complete data sets will be available for biodegradation rates of specific polymers and products in relevant environmental compartments.
- While the *Plastics LCA* method provides extensive and detailed requirements to conduct data quality assessment (Section 4.7 of the method), these were not tested in the case studies, as they have not investigated real products manufactured by specific companies. As such, the conditions to strictly follow the prescribed data collection requirements were missing (e.g. no company-specific data could be applied, as discussed in the points above, while only secondary data were used). Moreover, for the same reason, it was not possible to determine which data quality requirements and assessment criteria specified in the Data Needs Matrix (DNM; Section 4.7.5 of the method) should have been applied, as the level of influence of the company on each process in the supply chain could not be identified. Therefore, the application of the data quality assessment procedure would have been only partial

(i.e. limited to secondary datasets) and of little meaningfulness, so that time has been rather invested in properly illustrating the applicability of other relevant methodological and modelling rules. Note, however, that any company or other user of the *Plastics LCA* method shall conduct data quality assessment and include it in the LCA study report as prescribed in the related template.

- A key step of the interpretation phase of a LCA study following the *Plastics LCA* method is the identification of most relevant impact categories, life cycle stages, processes and elementary flows (Section 6.2 of the method). At present, conducting this step requires the user to perform additional elaborations and calculations (e.g. in separate spreadsheets) based on raw impact assessment results normally calculated with the support of a specific LCA software tool. However, to decrease the risk of errors in applying the prescribed procedures and calculations, and to reduce the time required for these, it would be preferable that relevant calculation and identification procedures are implemented in existing LCA software tools or other dedicated tools, where they can be conducted in a more controlled and faster manner.
- When investigating alternative product scenarios, a consistent identification of most relevant processes and elementary flows requires that the respective life cycle models are developed applying similar levels of vertical disaggregation for included processes and activities. Otherwise, most relevant processes and elementary flows are identified at different and potentially incomparable levels, which would not be meaningful (especially if a comparison among different product alternatives has to be made, wherever allowed by specific PEFCRs). Any relevant discrepancy may also prevent reproducibility of results, as well as comparability with the results from other studies investigating the same product(s). Life cycle models of alternative product scenarios investigated in a specific study should thus be developed relying as far as possible on datasets that enable comparable levels of vertical disaggregation (at least for foreground processes and activities), and this should also be pursued across different studies individually focusing on the same product. In turn, datasets to be applied in LCA studies following the *Plastics LCA* method should be developed or, where possible, adjusted, to ensure that this comparability is achieved during the modelling of specific product life cycles.
- Another fundamental step of the interpretation phase is conducting relevant sensitivity analyses (or “sensitivity checks”), as a tool to assess the robustness of the developed LCA models (Section 6.1 of the *Plastics LCA* method). This instrument was extensively applied throughout the case studies, evaluating the effects of varying a number of parameters, assumptions, and selected methodological choices, on the results. However, some sensitivity analyses were affected by potential modelling discrepancies in the applied alternative process datasets, compared with those originally used in the modelling of the base case of the affected scenarios. This was especially the case of sensitivity analyses considering different types and/or origin of the feedstock used for specific bio-based polymers, or a different origin of the bio-polymer itself, where datasets derived from different databases, providers and/or data sources had to be applied to model the affected processes and activities, compared with the relevant base case scenarios. Therefore, as partly discussed in previous points, it is recommended to develop consistent life cycle inventory datasets for the production of bio-based polymers already available on the market, covering all currently relevant feedstock sources in terms of type and origin.

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Annex A – Selection of plastic products for the case studies

A.1. Introduction

A set of criteria was defined for the selection of relevant plastic products to be investigated in the illustrative LCA case studies developed to demonstrate and evaluate applicability of the *Plastics LCA* method. Specifically, the definition and application of a selection methodology aimed at: (i) identifying five candidate products for the initial *screening LCA case studies* conducted to test applicability and reliability of the first draft of the *Plastics LCA* method (see Section 1.2); and (ii) proposing an extended list with ten additional candidate products to be considered for the selection of relevant products to be investigated in the ten *detailed LCA case studies* accompanying the revised version of the *Plastics LCA* method developed after the stakeholder consultation of November-December 2018 (Section 1.2).

In order to apply an appropriate methodology for the selection of relevant plastic products, different aspects were taken into account, along with the overarching aim of enabling demonstration of the *Plastics LCA* method on plastic products from different feedstocks, including both conventional fossil-based resources and alternative feedstocks such as recycled plastic waste, biomass, and CO₂ from gaseous effluents. Most relevant aspects considered to define appropriate selection criteria included:

- the market potential of plastic polymers or products relying on alternative feedstocks, including market trends and criticality;
- the different market sectors where such polymers or products are used;
- the promise for deployment, including technology readiness levels of underlying production and/or manufacturing processes for plastic polymers and products;
- the availability and quality of techno-scientific data needed for the LCA analysis;
- the relevance and peculiarity of applicable End of Life (EoL) options and/or scenarios;
- the intended lifetime and durability of the product;
- the type of application, especially differentiating between flexible and rigid plastic products; and
- the recyclability of the waste plastic product at End of Life.

The identification of potential candidates focused on simple plastic products from most common plastic conversion processes (e.g. injection moulding, blow moulding, extrusion), and not on polymers, building blocks, or complex products such as multi-components requiring different conversion steps. However, market data on plastic products based on alternative feedstocks were not publicly available. Therefore, statistics on bio-based polymers were used to understand the market of bio-based products, both at worldwide (European Bioplastics, 2017; Aeschelmann and Carus, 2017) and EU level (Spekreijse et al., 2018). Similarly, a French study on recycled polymers (Deloitte and Touche, 2015) was considered to understand the market of recycled plastic waste-based products. No data on CO₂-based polymers or plastic products were publicly available, instead.

According to such literature (European Bioplastics, 2017; Aeschelmann and Carus, 2017; Deloitte and Touche, 2015), plastic products and polymers derived from alternative feedstocks are used in the following market sectors:

- agriculture and horticulture;
- automotive and transport;
- building and construction;
- consumer goods;

- electrics and electronics;
- flexible packaging;
- rigid packaging; and
- textiles.

Plastic products and polymers from alternative feedstocks are mentioned and/or analysed in several scientific papers, studies, reports and projects (see Table A.4), and particularly in the following ones:

- BIO-SPRI study (EC, 2019b);
- Market studies (European Bioplastics, 2017; Aeschelmann and Carus, 2017; Molenveld et al., 2015; Van Den Oever et al., 2017; Deloitte and Touche, 2015; OECD, 2017; HCGA, 2009; Kaeb et al., 2016; Dommermuth and Raschka, 2015);
- Research projects and studies, including H2020 projects (Spekreijse et al., 2018; Open-Bio, 2018; STAR-ProBio, 2018; BioMotive, 2018; RefuCoat, 2018; BARBARA, 2018; Polybioskin, 2018; BioBarr, 2018; Embraced, 2018; PEference, 2018);
- The systematic review of selected LCA studies in the field of plastics conducted to inform and support the initial development of the *Plastics LCA* method (see Annex K in Nessi et al., 2021).

The developed selection methodology was partially based on the one applied in the *BioSpri* study, which was carried out for DG RTD (Task 1 of contract "Study on Support to R&I Policy in the Area of Bio-based Products and Services") and conducted a number of LCA case studies on innovative bio-based (plastic) products (EC, 2019b). In such study, a number of main criteria were defined and further broken down into sub-criteria, as summarised below, to select relevant products to be assessed:

- *Market potential*: types of jobs created, number of jobs created, market size/volume in specific sectors in terms of total amount of products and incomes, relative projected market growth.
- *Promise for deployment*: projected availability of feedstock (in terms of volume required, length of supply response and possible technical/environmental/economic supply restrictions), possibility of exchanging the feedstock with a residual feedstock, development status of technology, ensuring food security and safety.
- *Available data*: Life Cycle Impact Assessment (LCIA) data on potential environmental impacts, Life Cycle Inventory (LCI) data about feedstock, LCI data about conversion technology, LCI data about End of Life scenarios, product with innovative functionalities, new collaborations to enhancing synergies and coherence as compared to the production of the fossil-based reference product.
- *Innovation*: use of scarce resources.
- *Potential sustainability benefits*: direct and indirect land use change, relevance for key policies.

In the *BioSpri* study, the relevance and consequent prioritisation of the products to be investigated were determined using a scoring model, where the individual sub-criteria of each candidate product were assigned a score varying from 0 (worst case, no data) to 3 (best case, well documented data and best performances). According to the Authors of the *BioSpri* study, most of the scores were derived from relevant EU projects on bio-based products, EU frameworks and regulations, or from the authors' experience on the subject.

Unfortunately, references on specific data sources used for individual products and criteria were not provided for the entire set of candidate products and criteria considered in the *BioSpri* study. However, the latter provided useful information on a number of potentially relevant bio-based plastic products considered as candidates in this selection

procedure, namely: agricultural clips, binders and seeding pots; food rigid packaging / food containers; single-use carrier bags; beverage (PET) bottles; carpet; food packaging film; single-use cups for cold drinks; agricultural mulching film; lubricants; and single-use cutlery.

In this study, a scoring model similar to the one developed in the *BioSpri* study was applied for the selection of relevant plastic products to be investigated in the illustrative case studies (see Section A.2). However, it was also considered that the objectives are different between the two studies. For example, alternative feedstocks other than biomass, such as recycled plastic waste or captured CO₂, were additionally considered here. Moreover, documented thresholds were used in the scoring system.

Therefore, compared to the *BioSpri* study, the following actions were taken:

- the criteria were adapted and simplified, in order to build a reproducible selection methodology based on different aspects, such as policy priority, market potential (for polymers), promise for deployment, availability of LCA studies/data, and End of Life options/scenarios;
- the scoring system was also simplified by considering only two scores/options, i.e. 0= no priority, 1 = priority;
- some criteria were not retained, such as the ones more relevant for research and innovation studies (e.g. the relative projected market growth or policy opportunity), the ones that were already covered under other criteria (e.g. the availability of LCIA data on potential environmental impacts), or the ones that were already covered in other parts of the study (e.g. ensuring food security and safety).

Finally, additional criteria were applied to the entire set of identified candidate products, to achieve a balanced group of case studies, i.e. presenting a balanced coverage of relevant products and scenarios in terms of, e.g., durability of the products (short-lived vs long lived), their biodegradability (biodegradable under specific conditions vs non-biodegradable), and of considered materials and feedstocks for the products (see Section A.3). These additional criteria were not used in the scoring, but as a second-step checking exercise aiming at avoiding duplicates, preventing the selection of similar products, polymers and/or feedstocks, and at defining a more balanced group of case studies, as discussed above.

Based on the proposed criteria, a preselection of 15 candidate products was made, ranked in order of relevance for further possible investigation. Five products were then selected for the initial *screening LCA case studies*, whereas an extended list with 10 additional candidate products was proposed for discussion with stakeholders and collect possible feedback for the selection of relevant products to be investigated in the ten *detailed LCA case studies* (see Section A.4). A final list of 10 products to be considered in the *detailed case studies* was finally developed, taking into account the inputs received during the stakeholder consultation of November-December 2018 (Section A.4).

A.2. Criteria for the selection and scoring of candidate plastic products

This section describes the criteria and sub-criteria defined for the selection of relevant plastic products to be investigated in the case studies from a list of potential candidates, and the scoring system applied.

A.2.1. Selection criteria

The following selection criteria were applied:

a) Policy priority

- Priority was given to products other than the ones already covered by relevant existing EU legislation, such as lightweight plastic carrier bags addressed in Directive

2015/720 (EC, 2015), or by other initiatives, such as straws or disposable tableware prohibited to be placed on the market by the Single Use Plastic Directive (EC, 2019a).

b) Market potential

Including the following sub-criteria:

b.1) Market size of bio-based polymers

- Priority was given to the ten bio-based polymers with higher production capacities in the different market sectors. Since the production capacities of bio-based polymers at European level were not available for the entire set of candidate polymers, global production data were considered (European Bioplastics, 2017; Aeschelmann and Carus, 2017).

b.2) Market size of recycled plastic waste-based polymers

- Priority was given to the main recycled plastic waste-based polymers used in the different market sectors. Since data at European level could not be found, statistics on recycled plastic waste-based polymers provided by a French study were considered (Deloitte and Touche, 2015).
- The *market size of CO₂-based polymers* was not analysed, as statistics on these polymers were not publicly available at the time of this study.

b.3) Identifying market trend

- Priority was given to those bio-based polymers with an expected positive trend from 2016 to 2021. Since market trend data of bio-based polymers at European level were not available for the entire set of candidate polymers, global production data were considered (Aeschelmann and Carus, 2017). Market trend data on recycled plastic waste-based polymers or CO₂-based polymers were not available.

b.4) Market criticality

- Priority was given to critical alternative feedstock-based polymers that are mainly imported, i.e. a polymer was defined critical when its import was at least equal to 50% w/w of its consumption. Market criticality of bio-based polymers was evaluated based on an EU database (Spekreijse et al., 2018), which unfortunately did not cover the entire set of candidate bio-based polymers. Market criticality data on recycled plastic waste-based polymers or CO₂-based polymers were not available.

c) Promise for deployment

- Priority was given to alternative feedstock-based polymers and products relying on well-established production and/or manufacturing technologies, i.e. technologies with a technology readiness level (TRL) equal or superior to 8. In the case of bio-based products, the TRL values were found in an EU database (Spekreijse et al., 2018), while in the case of recycled-based plastic products, the TRL values were retrieved from a French study (Deloitte and Touche, 2015). Technologies for the production and/or manufacturing of CO₂-based polymers and products were considered to be at a research and innovation stage, with a TRL lower than 8.

d) Availability and quality of data needed for the LCA analysis

Including the following sub-criteria:

d.1) Quality of available LCA studies

- Priority was given to alternative feedstock-based plastic products for which high quality LCA papers were available, based on the systematic review of selected LCA studies conducted by JRC (see Annex K in Nessi et al., 2021).

d.2) LCA scenario number

- The availability of techno-scientific data needed for the LCA analysis was evaluated considering the number of LCA scenarios investigated for each alternative feedstock-based plastic product, based on the available literature data (see Table A.4). This criterion has a higher scoring than the other criteria, because it counted the total number of individual LCA scenarios available from literature. In other words, each scenario counted as 1, and the total number of scenarios available ranged from 1 to 12, depending on the product and the number of scenarios available.

e) End of Life (EoL) options/scenarios

- Priority was given to alternative feedstock-based plastic products which may remain in or onto soil after use for biodegradation.

A.2.2. Scoring criteria

The total score of each candidate product was calculated by applying the *weighting factors* reported in Table A.1 to the different criteria and sub-criteria described in Section A.2.1. Priority was given to market-related criteria (under “market potential”), which were assigned a weighting factor of 3. These were followed by the criteria focusing on the number of LCA scenarios investigated in the literature, and on applicable End of Life options/scenarios, with a weighting factor of 2. Criteria related to policy priority, promise for deployment, and quality of available LCA studies were assigned a weighting factor of 1, instead.

The results of the scoring exercise for the identified candidate plastic products are reported in Table A.5 (scoring matrix).

Table A.1: Weighting factors applied to each selection criteria specified in Section A.2.1 for the calculation of the total score of candidate products.

Selection criteria	Selection sub-criteria	Weighting factor
a) Policy priority	-	1
b) Market potential	b.1) Market size of bio-based polymers	3
	b.2) Market size of recycled plastic waste-based polymers	3
	b.3) Identifying market trend	3
	b.4) Market criticality	3
c) Promise for deployment (TRL)	-	1
d) Availability and quality of data needed for the LCA analysis	d.1) Quality of available LCA studies	1
	d.2) Number of LCA scenarios	2
e) End of Life options/scenarios	-	2

A.3. Additional criteria for the selection of relevant plastic products

In order to ensure a balanced coverage of the relevant products and scenarios, the criteria described in this Section were additionally applied to the entire set of identified candidate products. These additional criteria were not used in the scoring (Section A.2.2), but rather as discriminatory criteria, applied in a second step, and aiming at avoiding duplicates, i.e. selection of similar products (similar polymers, feedstocks, markets, uses and LCA scenarios), and at defining a more balanced group of case studies.

f) End of Life (EoL)-related aspects, including littering

Including the following sub-criteria:

f.1) Littering (marine)

- Priority was given to the ten single-use plastic products addressed in the Single Use Plastic Directive (EC, 2019a).
- The target was to have at least two, ideally three, selected products with a marine littering potential for the initial *screening LCA case studies*.

f.2) Expected lifetime/durability

- The objective was to have a balanced variety of expected product lifetimes, in order to include both single-use/disposable items (such as packaging), and durable products used, for example, in the building and construction or in the automotive sectors. To this aim, the lifetime of the different candidate products was categorised as follows: "short" if lower than one year, "long" if ranging from one to ten years, and "very long" if larger than ten years.
- The target was to have, for the initial *screening LCA case studies*, three selected products with short lifetime and two with long, or very long, lifetime.

f.3) Recyclability

- Priority was given to alternative feedstock-based plastic products or polymers that are recyclable at End of Life.
- The target was to have at least two, ideally three, selected products with high recyclability for the initial *screening LCA case studies*.

f.4) Bio-degradability

- Priority was given to alternative feedstock-based plastic products or polymers that are biodegradable under specific conditions (i.e. in technical/industrial systems or in/onto soil).
- The target was to have ideally two selected products with biodegradability properties for the initial *screening LCA case studies*. During the identification of candidate products, it was not always possible to differentiate between biodegradability in technical/industrial systems and biodegradability in nature (soil), due to lack of data.

g) Uses

Including the following sub-criteria:

g.1) Single-use vs multiple-use products

- The objective was to have a balanced variety of single-use and multiple-use products. To this aim, candidate products were classified as "single-use" products when meeting the definition provided in the Single Use Plastic Directive (EC, 2019a). Other products were classified as "multiple-use" ones.
- The target was to have at least three selected products categorised as single-use for the initial *screening LCA case studies*.

g.2) Rigid vs flexible applications

- The objective was to have a balanced variety of flexible and rigid applications (such as flexible packaging and rigid packaging).
- The target was to have at least one selected product categorised as flexible for the initial *screening LCA case studies*.

h) Market coverage

h.1) *Import dependency*

- The import dependency was already analysed under point *b.4) market criticality* (see Section A.2.1).
- The target of the additional market coverage criteria was to have at least two selected products with high import dependency for the initial *screening LCA case studies*.

A.4. Lists of selected products

By applying the selection and scoring criteria described in Section A.2, the entire set of identified candidate products was assigned a total score, as reported in the total scoring matrix shown in Table A.5. For clarity, the total scores of individual products, grouped in the different market sectors, are reported in Table A.2.

For the initial *screening LCA case studies*, the candidate products with the highest score in each market sector were selected. Considering that the main objective of the screening case studies was to apply and test the first draft of the *Plastics LCA* method, products with a good availability and quality of life cycle inventory data were selected. Moreover, in order to ensure a balanced coverage of the investigated products and scenarios, and more precisely to reach the lifetime/durability objectives considered as additional selection criteria (Section A.3), cleansing wipes, the first product in the consumer goods sector, were not selected for the initial *screening LCA case studies*, but were considered later for the ten *detailed LCA case studies* (see below). Textile fibres were also not selected, neither for the initial screening case studies nor for the ten detailed ones, as being an intermediate product, while the case studies were intended to address finished products with a defined functional unit (to ensure proper illustration of the *Plastics LCA* method). Nevertheless, fibres were proposed as a possible candidate product for the *detailed LCA case studies* during the stakeholder consultation of November-December 2018.

Eventually, the following products were selected for the five *screening LCA case studies*, based on their total score and after applying the additional criteria described in Section A.3:

1. Beverage bottles;
2. Flexible food packaging film;
3. Agricultural mulching film;
4. Insulation boards for buildings;
5. Automotive interior panels.

Therefore, for some of the additional criteria reported in Section A.3, such as expected product lifetime, biodegradability, and single-use vs multiple-use products, the requirements have been exactly fulfilled. For other additional criteria such as potential to contribute to marine littering, recyclability, rigid vs flexible applications, and import dependency, the minimum objectives were satisfied, instead (see Table A.3).

The products selected for the initial *screening LCA case studies* covered different LCA scenarios, making it possible to investigate the applicability of the draft *Plastics LCA* method to various conventional and alternative polymers and feedstocks, including fossil-based, bio-based, recycled plastic waste-based and CO₂-based ones. A range of different End of Life options and scenarios could also be evaluated in the framework of the *screening LCA case studies*.

In addition to the five products selected for the initial *screening LCA case studies*, ten additional candidates were then proposed for the identification of the entire set of relevant products to be investigated in the ten *detailed LCA case studies* carried out after the stakeholder consultation of November-December 2018 (based on the revised version of the *Plastics LCA* method). In practice, an extended list of 15 suggested candidate products was defined for discussion with stakeholders and collect possible feedback. The list was compiled by considering those products with higher scores in the different market sectors. In the case of similar products (such as, for example, cleansing wipes and sanitary towels), only one representative product was proposed for consideration. The following list of candidate products for the ten *detailed LCA case studies* was thus ultimately proposed for discussion and collection of feedback during the stakeholder consultation:

1. Beverage bottles (already considered for the *screening LCA case studies*);
2. Flexible food packaging film (already considered for the *screening LCA case studies*);
3. Agricultural mulching film (already considered for the *screening LCA case studies*);
4. Insulation boards for buildings (already considered for the *screening LCA case studies*);
5. Automotive interior panels (already considered for the *screening LCA case studies*);
6. Textile fibre (e.g. yarn);
7. Cleansing wipes;
8. Printer housing panels;
9. Trays for food;
10. Loose fill chips;
11. Other consumer goods (to be specified);
12. Cups;
13. Other agricultural products (to be specified; indicative examples are tree shelters, nets, twines, garden tools, plant clips, binders - see Table A.4);
14. Foam for consumer goods;
15. Other packaging (non-food) (to be specified; an indicative example being packaging for electric and electronic appliances - see Table A.4).

A balanced coverage of potential product scenarios to be investigated was reached also for the 10 additionally suggested candidate products for the *detailed LCA case studies* (i.e. beyond the five already considered in the screening LCAs). In particular, some objectives of the additional criteria from Section A.3, such as expected product lifetime and single-use vs multiple-use applications, were met (see Table A.3).

The final list of products ultimately selected for assessment in the ten *detailed LCA case studies* was defined taking into account the inputs received during the stakeholder consultation of November-December 2018. Based on the specific suggestions, the five products already investigated in the *screening LCA case studies* were confirmed to be eligible for further detailed assessment, and three additional candidates included in the preliminary list were selected (i.e. trays for food, printer housing panels and cleansing wipes). A specific product was then identified as representative of the category "Other agricultural products" (i.e. pots for plants and flowers) and selected as additional product for investigation. Finally, a product not included in the original preliminary list (i.e. stacking chairs) was selected based on specific stakeholder recommendations, as

representative of the furniture sector. Overall, the following ten products were thus selected for inclusion in the *detailed LCA case studies*:

1. Beverage bottles;
2. Food packaging film;
3. Trays for food;
4. Agricultural mulching film;
5. Flowers/plants pots;
6. Insulation boards for buildings;
7. Automotive interior panels;
8. Printer housing panels;
9. Monobloc stacking chairs;
10. Cleansing wipes.

Table A.2: Scoring results for candidate plastic products (grouped by market sector), selection of relevant products for the five *screening LCA case studies*, and identification of a restricted list of 15 candidates for the ten *detailed LCA case studies*.

Product	Relevant products for the screening LCA case studies and proposal of candidates for the detailed LCA case studies	Score
Rigid packaging		485
Beverage bottles	x (screening + detailed)	139
Trays for food	x (detailed)	99
Loose fill chips	x (detailed)	55
Cups	x (detailed)	38
Containers/boxes for food		36
Beverage cartons		29
Caps for non-beverage bottles		22
Crates		22
Containers/boxes for non-food		12
Caps for beverage bottles		12
Non-beverage bottles		12
Rigid-packaging		6
Pallets		3
Tableware		0
Straws		0
Flexible packaging		366
Food packaging	x (screening + detailed)	312
Other packaging (non-food)	x (detailed)	32
Packaging of packaging		22
Carrier bags		0
Garbage bags		0

Consumer goods		226
Wipes	x (detailed)	68
Sanitary towels		49
Other consumer goods	x (detailed)	48
Foam for consumer goods	x (detailed)	33
Toys/Houseware		12
Fibres		6
Electro-domestic parts		4
Injection moulding product		3
Cigarette buds		3
Automotive & transport		154
Interior panels	x (screening + detailed)	58
Other automotive components		24
Dashboard fascia		19
Automotive textiles		19
Door handles		19
Foam for automotive		9
Automotive flexible plastic part		6
Agriculture & horticulture		144
Mulching film	x (screening + detailed)	77
Other agricultural products	x (detailed)	34
Pots		33
Building & construction		87
Insulation material	x (screening + detailed)	48
Other construction products		30
Ropes		6
Frames		3
Textiles		80
Textile fibres	x (detailed)	70
Carpets		10
Electrics & electronics		70
Printers housing panels	x (detailed)	44
Laptop covers		9
Electro-domestic parts		8
Wire coatings		6
Printed Wiring Boards		3
Flexible & rigid packaging		28
Food packaging		28

Table A.3: Results of the application of the additional selection criteria in Section A.3 to the products selected for the five initial *screening LCA case studies*, and to the ten additional candidates proposed for the ten *detailed LCA case studies*.

Criteria*	Sub-criteria	5 products selected for the initial screening LCA case studies	Additional 10 candidate products proposed for the detailed LCA case studies
		Minimum number of products fulfilling the criteria	
f) End of Life (EoL) aspects including littering	f.1) littering (marine)	potentially contributing: 2	potentially contributing: 3
	f.2) expected lifetime/durability	short: 3 long: 1 very long: 1	short: 6 long: 4
	f.3) recyclability	low: 3 high: 2	low: 7 high: 3
	f.4) bio-degradability	not biodegradable: 2 biodegradable: 3	not biodegradable: 3 biodegradable: 7
g) Uses	g.1) single-use vs multiple-use products	multiple-use: 2 single-use: 3	multiple-use: 4 single-use: 6
	g.2) rigid vs flexible applications	flexible: 2 rigid: 3	flexible: 4 rigid: 6
h) Market coverage	h.1) import dependency	low: 1 high: 4	low: 2 high: 8

(*) Related to the entire set of case studies (not to the selection of each single case study).

Table A.4: Alternative feedstock-based plastic products available in different market sectors and considered as candidates for investigation in the illustrative LCA case studies. Additional examples and references are also included.

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
Agriculture & horticulture	Mulching film	PLA		(Open-Bio, 2018)
		Bio-PBAT		(Aeschelmann and Carus, 2017; Bilck et al., 2010)
		Bio-PBS		(Aeschelmann and Carus, 2017; Succinity, 2013)
		Kenaf-based polymer		(Wolfensberger and Dinkel, 1997)
		Starch blends		(BioSpri, 2018; Valpack Consulting Consortium, 2010; Open-Bio, 2018; STAR-ProBio, 2018)
	Other agricultural products	PLA	Tree shelters (tubes used to protect young seedlings)	(Arnold and Alston, 2012)
		R-PP	Nets, twines	(Molenveld et al., 2015)
		Starch blends	Garden tools, plant clips, binders, plant protecting structures, terratube	(BioSpri, 2018; Spekrijse et al., 2018; Biopolymers, 2018)
	Pots	PLA		(Molenveld et al., 2015)
		PLA-E		(Molenveld et al., 2015)
		Starch blends		(BioSpri, 2018; Spekrijse et al., 2018)
	Automotive & transport	Automotive flexible plastic parts	Bio-PET	Headliners, floor mats, sun visors
Bio-PUR			Regenerated fibres for textiles for covering vehicles seats	(BioMotive, 2018)
Automotive textiles		Bio-PE	Interior fabrics	(European Bioplastics, 2017)
Dashboard fascia		Bio-PUR		(BioMotive, 2018)
		Bio-PA		(BARBARA, 2018)
Door handles		Bio-PUR		(BioMotive, 2018)
		Bio-PA		(BARBARA, 2018)
Foam for automotive		Bio-PUR	Foam for seats	(BioMotive, 2018)
Interior panels		PLA	Panels for aircraft interiors	(Vidal et al., 2018)
		Hemp-based (66%v)+epoxy resin(34%v) + hardener	Interior side panels	(Wötzel et al., 1999)
		Linseed oil-based	Panels for aircraft interiors	(Vidal et al., 2018)
		Bio-PBS		(Aeschelmann and Carus, 2017; Succinity, 2013)
	Bio-PP (flax reinforced)	Under-floor panels	(Diener and Siehler, 1999)	

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
	Other automotive components	PHA		(Spekreijse et al., 2018; Bio-on, 2018)
		PHB		(Aeschelmann and Carus, 2017)
		R-PP		(Deloitte and Touche, 2015)
Building & construction	Frame	Starch blends		(STAR-ProBio, 2018)
		bio-PUR	Coating material	(Gonzalez-Garay et al., 2017)
	Insulation material	PLA		(Aeschelmann and Carus, 2017)
		Bio-fibres	Pre-manufactured components/insulation: bio-fibre based insulation mats	(Open-Bio, 2018)
		Miscanthus		(Uihlein et al., 2008)
	Other construction products	Bio-PE	Moulds for resin transfer moulding, truss joint prototype	(BARBARA, 2018)
		R-PE (HD-MD)		(Deloitte and Touche, 2015)
		R-PP		(Deloitte and Touche, 2015)
	Ropes	Bio-PA		(Aeschelmann and Carus, 2017)
	Cigarette buds	Starch blends		(Aeschelmann and Carus, 2017; HCGA, 2009)
Electro-domestic parts	CO ₂ -based PPC	Parts for a vacuum cleaner or a refrigerator	(Dommermuth and Raschka, 2015)	
Fibres	Bio-PA			
Consumer goods	Foam for consumer goods	PLA	Particleboards (traditional and ultralight with an expanded foam core)	(Ganne-Chdeville and Diederichs, 2015)
		Bio-PUR		(Aeschelmann and Carus, 2017)
		Starch blends	Foam for washing machine port-hole spacer, foam display boards	(Razza et al., 2015; Guo et al., 2013; Guo et al., 2011)
	Injection moulding products	Bio-PBS	Coffee capsules, tableware, cups	(Aeschelmann and Carus, 2017; MCPP, 2016)
	Other consumer goods	R-PE (HD-MD)		(Deloitte and Touche, 2015)
		R-PE (LD-LLD)		(Deloitte and Touche, 2015)
		R-PP		(Deloitte and Touche, 2015)
		R-PS		(Deloitte and Touche, 2015)
	Sanitary towels	PLA	Sanitary towels, nappies, dressings, Absorbent Hygiene Products (AHP)	(Aeschelmann and Carus, 2017; NatureWorks, 2018; Polybioskin, 2018; Hakala et al., 1997; Vink and Davies, 2015)
		PHA	Sanitary towels, nappies, dressings	(Polybioskin, 2018)
PHB		Absorbent Hygiene Products (AHP)	(BioMotive, 2018)	
Starch blends		Super Adsorbent Polymer (SAP) for nappies	(Aeschelmann and Carus, 2017; McIntyre, 2017)	

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
	Toys/ Houseware	Bio-PE		(Aeschelmann and Carus, 2017)
		PLA	Wet wipes, flushable and dispersible wet baby wipes, dressings	(Aeschelmann and Carus, 2017; Butschli, 2008; Total-Corbion, 2018; NatureWorks, 2018; Polybioskin, 2018; Vink and Davies, 2015)
	Wipes	PHA	Wipes, dressings	(Polybioskin, 2018)
		PHB	Absorbent Hygiene Products (AHP)	(Embraced, 2018)
		Chitosan	Wipes, dressings	(Polybioskin, 2018)
		Starch blends	Wipes, dressings	(Polybioskin, 2018)
Electrics & electronics	Electro-domestic parts	PHA	Electronic parts	(Spekreijse et al., 2018), (Bio-on, 2018)
	Laptop covers	PLA		(Meyer and Katz, 2016; Valpack Consulting Consortium, 2010)
	Printed Wiring Boards	Lignin-epoxy blend	Printed Wiring Boards	(Kosbar et al., 2000)
		PLA	Electronic parts	(Broeren et al., 2016)
		Bio-PTT	Electronic parts	(Broeren et al., 2016)
		Bio-PP/natural fibre	Electronic parts	(Broeren et al., 2016)
		R-PP		(Deloitte and Touche, 2015)
Wire coatings	Bio-PA		(Aeschelmann and Carus, 2017)	
Flexible & rigid packaging	Food packaging	PHA, PHA/PLA	Barrier film and rigid packaging for food with enhanced barrier properties	(BioBarr, 2018)
Flexible packaging	Carrier bags	PLA		(Molenveld et al., 2015; Mattila et al., 2011; Gironi and Piemonte, 2010; BASF, 2014; Mueller and Mueller, 2017; Chaffee and Yaros, 2007)
		Bio-PE (HD)		(Parker and Edwards, 2012)
		Bio-PE		(Molenveld et al., 2015; BioSpri, 2018)
		R-PET		(Bisinella et al., 2018)
		PHA		(Khoo et al., 2010a, 2010b)
		Starch blends		(Molenveld et al., 2015; BioSpri, 2018; Gironi and Piemonte, 2011a; Piemonte and Gironi, 2011; Bisinella et al., 2018)
		R-PE		(Mattila et al., 2011)
		R-PE (HD)		(Dilli, 2007)

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
Food packaging		R-PE (LD)		(Bisinella et al., 2018)
		R-PE (LD-LLD)		(Molenveld et al., 2015; Deloitte and Touche, 2015)
		Bio-PET		(Bisinella et al., 2018)
		PLA	Barrier film and covering film for fresh solid food (meat/fish/cheese), barrier film for non-perishable solid food (crisps, coffee), covering film and packaging for fresh solid food (fruit and vegetables, butter, bread, frozen food), packaging film for non-perishable solid food (biscuits, confectionery and chocolate), teabags, multilayer film	(Molenveld et al., 2015; BioMotive, 2018; Open-Bio, 2018; Hermann et al., 2010; Benetto et al., 2015; Petrucci et al., 2017; Vidal et al., 2007; Deng et al., 2013; Rossi et al., 2015; Piemonte, 2011; Valpack Consulting Consortium, 2010)
		Bio-PE	Packaging film for fresh solid food (bread), high water and medium oxygen barrier film, barrier film for non-perishable solid food (snacks)	(BioSpri, 2018; Molenveld et al., 2015; RefuCoat, 2018; Hermann et al., 2010; Detzel et al., 2013)
		Bio-PE laminate	Packaging film for non-perishable liquid (sauces)	(Molenveld et al., 2015)
		PHA		(Spekreijse et al., 2018; Bio-on, 2018)
		PHA, PGA		(RefuCoat, 2018)
		Starch blends	Coffee capsules, tea bags, covering film and packaging for fresh solid food (fruit and vegetables, meat/fish/cheese), barrier film for non-perishable solid food (snacks)	(Kaeb et al., 2016; Molenveld et al., 2015; Spekreijse et al., 2018; Biopolymers, 2018; Open-Bio, 2018; STAR-ProBio, 2018)
		Bio-PET	High water and medium oxygen barrier film	(RefuCoat, 2018)
		Cellulose		(Valpack Consulting Consortium, 2010)
		Chitosan		(Leceta et al., 2014; Leceta et al., 2013)
		Wheat gluten powder	Wheat-gluten-based (plasticised by glycerine)	(Deng et al., 2013)
		Bio-PP	Barrier film for non-perishable solid food (snacks)	(Hermann et al., 2010)
	Garbage bag		PLA	
		Bio-PE		(Saibuatrong et al., 2017)
		Bio-PBAT/Starch		(Saibuatrong et al., 2017)
		Starch blends		(Valpack Consulting Consortium, 2010; Estermann, 1998; Estermann and Schwarzwälder,

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
				1998)
	Other packaging (non-food)	Bio-PE	Packaging of electric and electronic appliances	(Molenveld et al., 2015)
		Starch blends	Packaging of electric and electronic appliances	(Molenveld et al., 2015)
		CO ₂ -based PPC		(Dommermuth and Raschka, 2015)
		Bio-PE	Shrink film, stretch film	(Molenveld et al., 2015)
	Packaging of packaging	Starch blends	Film bowlings	(Spekreijse et al., 2018; Biopolymers, 2018)
Rigid packaging		Bio-PE	Bottles for liquid fresh food (milk, juices)	(Molenveld et al., 2015)
		PLA	Bottles for liquid non-perishable food (water, beer), bottles for liquid fresh food (milk, juices)	(Molenveld et al., 2015; Gironi and Piemonte, 2010, 2011a, 2011b; Meyer and Katz, 2016; Papong et al., 2014; Shen et al., 2012)
		Bio-PET	Bottles for liquid non-perishable food (water, carbonated soft drinks, beer, sauces)	(Molenveld et al., 2015; BioSpri, 2018; Chen et al., 2016; Valpack Consulting Consortium, 2010; Shen et al., 2012)
	Beverage bottles	Rr-PET		(OECD, 2017; Deloitte and Touche, 2015; Grand and Roux, 2014; Shen et al., 2012; Thoden van Velzen et al., 2016)
		PHA	Bottles for liquid non-perishable food (water, carbonated soft drinks, beer)	(Spekreijse et al., 2018; Bio-on, 2018)
		Bio-PEF	Bottle for liquid non-perishable food (water, carbonated soft drinks, beer)	(Molenveld et al., 2015; BioSpri, 2018)
		R-(bio)PET		(Shen et al., 2012)
	Beverage cartons	R-PET		(Markwardt et al., 2017)
		Bio-PE		(Markwardt et al., 2017)
	Caps for beverage bottles	Bio-PE	Caps for bottles for liquid fresh food (milk, juices)	(Molenveld et al., 2015)
	Caps for non-beverage bottles	Bio-PE	Caps for non-food bottles (detergents)	(Molenveld et al., 2015)
		PHA	Caps for personal care bottles/cosmetics	(Molenveld et al., 2015)
Containers/boxes for food	PLA	Containers/boxes for solid non-perishable food (biscuits, confectionery and chocolate), containers/boxes for solid fresh food (fruit and vegetables, yogurt), containers/boxes for take	(Molenveld et al., 2015; Spekreijse et al., 2018; Synbra, 2018; Leejarkpai et al., 2016; Lorite et al., 2017; Madival et al., 2009; Bohlmann, 2004; Suwanmanee et al., 2013; Cheroennet et al., 2018; Kuczenski et al., 2012; Detzel et	

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
			away, bowls with a hinged lid	al., 2013)
		Bio-PBS		(Cheroennet et al., 2018)
		Starch blends	Containers/boxes for solid non-perishable food (confectionery and chocolate)	(Molenveld et al., 2015)
	Containers/boxes for non-food	PLA	Containers/boxes for electric and electronic appliances	(Molenveld et al., 2015)
		Bio-PE		(Molenveld et al., 2015)
	Crates	Starch blends	Crates, mussel crates, containers	(Molenveld et al., 2015; Spekrijse et al., 2018)
	Cups	PLA	Cups for liquid non-perishable food (coffee, cold drinks, hot drinks)	(Molenveld et al., 2015; BioSpri, 2018; Potting and Van der Harst, 2015; Uihlein et al., 2008; Van der Harst and Potting, 2013; Van der Harst et al., 2014; Vercalsteren et al., 2010; Binder and Woods, 2009; Pladerer et al., 2008; Pro.Mo/Unionplast, 2015; Vink and Davies, 2015)
		R-PET		(Van der Harst and Potting, 2013)
		Starch blends		(Fieschi and Pretato, 2017)
	Loose fill chips	PLA	BioFoam	(Spekrijse et al., 2018; Synbra, 2018)
		PLA-E	Foam for packaging electric and electronic appliances	(Molenveld et al., 2015)
		Starch blends		(Estermann et al., 2000; BIfA et al., 2002)
		Miscanthus		(Wolfensberger and Dinkel, 1997)
	Non-beverage bottles	Bio-PE	Bottles for detergents/personal care/cosmetics	(Molenveld et al., 2015)
	Pallets	R-PE (HD)		(Molenveld et al., 2015)
	Rigid-packaging	Bio-PBAT		(Aeschelmann and Carus, 2017)
	Straws	PLA		(Boonniteewanich et al., 2014)
		Bio-PBS		(Boonniteewanich et al., 2014)
	Tableware	PLA	Plates, cutlery, cutlery envelopes	(Molenveld et al., 2015; BioSpri, 2018; Fieschi and Pretato, 2017; Pro.Mo/Unionplast, 2015; Vink and Davies, 2015)
		PHA	Cutlery for catering	(Molenveld et al., 2015)
		PHB		(BioSpri, 2018)
		Starch blends	Plates, cutlery	(Molenveld et al., 2015; BioSpri, 2018; Postacchini et al., 2016; Razza et al., 2009; Fieschi and Pretato, 2017)
		CA (Biograde)	Cutlery for catering	(Molenveld et al., 2015)

Market Sector	Product	Alternative feedstock-based polymer	Additional examples	References
	Trays for food	PLA	Trays for fresh solid food (fruit and vegetables, meat/fish/cheese, eggs, frozen food), foamy expanded trays	(Molenveld et al., 2015; Ingrao et al., 2017)
		Bio-PBSC		(Yuki, 2012)
		PHA	Trays for frozen fresh solid food	(Molenveld et al., 2015)
		PHB		(Yuki, 2012)
		Starch blends		(BioSpri, 2018), (Yuki, 2012)
	Trays for food	Natural fibres		(BioSpri, 2018)
Textiles	Carpets	Bio-PA		(BioSpri, 2018)
		Bio-PTT		(BioSpri, 2018)
	Textile fibres	Bio-PET		(Aeschelmann and Carus, 2017)
		PLA		(Aeschelmann and Carus, 2017)
		R-PET		(Deloitte and Touche, 2015)
		Man-made cellulose fibre		(Shen et al., 2012)
		PHA		(Spekreijse et al., 2018; Bio-on, 2018)

Table A.5: Scoring matrix of the alternative feedstock-based plastic products and related material (polymer) and feedstock combinations identified as candidates for investigation as individual scenarios in the illustrative LCA case studies.

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)	
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2
Food packaging	Biomass	PLA	PE, PET, PP, PET/PE laminate, PP metallised	Flexible packaging	1	1	0	1	1	1	2	12	0	36	1	1	short	0	1	1	0
Food packaging	Biomass	Bio-PE	PE (LD), PP	Flexible packaging	1	1	0	1	1	1	0	12	0	34	1	1	short	1	0	1	0
Food packaging	Biomass	Bio-PE laminate	PE (HD)	Flexible packaging	1	1	0	1	1	1	0	12	0	34	1	1	short	1	0	1	0
Food packaging	Biomass	PHA, PHA/PLA	PE (HD), PE (LD), PP	Flexible & rigid packaging	1	0	0	1	0	1	0	12	0	28	1	1	short	0	1	1	0
Food packaging	Biomass	PHA	PE (HD), PE (LD), PP	Flexible packaging	1	0	0	1	0	1	0	12	0	28	1	1	short	0	1	1	0
Food packaging	Biomass	PHA, PGA	PP, PET (metallised)	Flexible packaging	1	0	0	1	0	1	0	12	0	28	1	1	short	0	1	1	0
Food packaging	Biomass	Starch blends	PE (LD), PET, PP, PE laminate	Flexible packaging	1	1	0	0	0	1	0	12	0	28	1	1	short	0	1	1	0
Food packaging	Biomass	Bio-PET	PET	Flexible packaging	1	0	0	0	0	1	0	12	0	25	1	1	short	1	0	1	0
Food packaging	Biomass	Cellulose	PP	Flexible packaging	1	0	0	0	0	1	0	12	0	25	1	1	short	1	0	1	0
Food packaging	Biomass	Chitosan	PP	Flexible packaging	1	0	0	0	0	1	0	12	0	25	1	1	short	0	1	1	0
Food packaging	Biomass	Wheat gluten powder	PE (LD)	Flexible packaging	1	0	0	0	0	0	1	12	0	25	1	1	short	0	1	1	0
Food packaging	Biomass	Bio-PP	PP	Flexible packaging	1	0	0	0	0	0	0	12	0	24	1	1	short	1	0	1	0
Beverage bottles	Biomass	Bio-PE	PE (HD)	Rigid packaging	1	1	0	1	1	1	0	7	0	24	2	1	short	1	0	1	1
Beverage bottles	Biomass	PLA	PET, PE (HD)	Rigid packaging	1	1	0	1	1	1	0	7	0	24	2	1	short	0	1	1	1
Beverage bottles	Biomass	Bio-PET	PET	Rigid packaging	1	1	0	1	0	1	1	7	0	22	2	1	short	1	0	1	1
Beverage bottles	R/Waste	R-PET	PET	Rigid packaging	1	0	1	0	0	1	1	7	0	19	2	1	short	1	0	1	1
Beverage bottles	Biomass	PHA	PET, PP	Rigid packaging	1	0	0	1	0	1	0	7	0	18	2	1	short	0	1	1	1
Beverage bottles	Biomass	Bio-PEF	PET	Rigid packaging	1	0	0	1	0	0	0	7	0	17	2	1	short	1	0	1	1
Beverage bottles	R/Waste	R-(bio)PET	PET	Rigid packaging	1	0	0	0	0	1	0	7	0	15	2	1	short	1	0	1	1
Mulching film	Biomass	PLA	PE (LLD)	Agriculture & horticulture	1	0	0	1	1	1	0	5	1	19	3	0	short	0	1	1	0
Mulching film	Biomass	Bio-PBAT	PBAT	Agriculture & horticulture	1	0	0	1	0	1	0	5	1	16	3	0	short	0	1	1	0
Mulching film	Biomass	Bio-PBS	PBS	Agriculture & horticulture	1	0	0	1	0	1	0	5	1	16	3	0	short	0	1	1	0
Mulching film	Biomass	Kenaf-based polymer	PE	Agriculture & horticulture	1	0	0	0	0	1	0	5	1	13	3	0	short	1	0	1	0
Mulching film	Biomass	Starch blends	PE (LD)	Agriculture & horticulture	1	0	0	0	0	1	0	5	1	13	3	0	short	0	1	1	0

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)	
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2
Textile fibres	Biomass	Bio-PET	PET	Textiles	1	1	0	1	0	1	0	5	0	17	4	0	long	1	0	0	0
Textile fibres	Biomass	PLA	PET, PP, PE (HD), PE (LD)	Textiles	1	0	0	1	1	1	0	5	0	17	4	0	long	0	1	0	0
Textile fibres	R/Waste	R-PET	PET	Textiles	1	0	1	0	0	1	0	5	0	14	4	0	long	1	0	0	0
Textile fibres	Biomass	Man-made cellulose fibre	PET	Textiles	1	0	0	0	0	1	0	5	0	11	4	0	long	0	1	0	0
Textile fibres	Biomass	PHA	PET, PP, PE (HD), PE (LD)	Textiles	1	0	0	0	0	1	0	5	0	11	4	0	long	0	1	0	0
Wipes	Biomass	PLA	PE, PP	Consumer goods	1	0	0	1	1	1	1	5	0	18	5	1	short	0	1	1	0
Wipes	Biomass	PHA	PE, PP	Consumer goods	1	0	0	1	0	1	0	5	0	14	5	1	short	0	1	1	0
Wipes	Biomass	PHB	PE, PP	Consumer goods	1	0	0	1	0	1	0	5	0	14	5	1	short	0	1	1	0
Wipes	Biomass	Chitosan	PE, PP	Consumer goods	1	0	0	0	0	1	0	5	0	11	5	1	short	0	1	1	0
Wipes	Biomass	Starch blends	PE, PP	Consumer goods	1	0	0	0	0	1	0	5	0	11	5	1	short	0	1	1	0
Interior panels	Biomass	PLA	PP	Automotive & transport	1	0	0	0	1	1	0	5	0	14	6	0	long	0	1	0	1
Interior panels	Biomass	Hemp-based (66%v)+epoxy resin(34%v) + hardener	ABS	Automotive & transport	1	0	0	0	0	1	1	5	0	12	6	0	long	1	0	0	1
Interior panels	Biomass	Linseed oil-based	PP	Automotive & transport	1	0	0	0	0	1	0	5	0	11	6	0	long	1	0	0	1
Interior panels	Biomass	Bio-PBS	PBS	Automotive & transport	1	0	0	0	0	1	0	5	0	11	6	0	long	0	1	0	1
Interior panels	Biomass	Bio-PP (flax reinforced)	PP (fibreglass reinforced)	Automotive & transport	1	0	0	0	0	0	0	5	0	10	6	0	long	1	0	0	1
Insulation material	Biomass	Bio-PUR	PUR, EPS	Building & construction	1	1	0	1	0	1	0	4	0	15	7	0	very long	1	0	0	1
Insulation material	Biomass	PLA	PUR, EPS	Building & construction	1	0	0	1	1	1	0	4	0	15	7	0	very long	0	1	0	1
Insulation material	Biomass	Bio-fibres	PUR, EPS	Building & construction	1	0	0	0	0	1	0	4	0	9	7	0	very long	0	1	0	1
Insulation material	Biomass	Miscanthus	PUR, EPS	Building & construction	1	0	0	0	0	1	0	4	0	9	7	0	very long	0	1	0	1
Printers housing panels	Biomass	PLA	ABS (pure), PC/ABS	Electrics & electronics	1	0	0	1	1	1	0	4	0	15	8	0	long	0	1	0	1
Printers housing panels	Biomass	Bio-PTT	ABS (pure), PC/ABS	Electrics & electronics	1	0	0	0	0	1	0	4	0	9	8	0	long	1	0	0	1

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)		
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2	
Printers housing panels	Biomass	Bio-PP/natural fibre	ABS (pure), PC/ABS	Electrics & electronics	1	0	0	0	0	0	4	0	8	8	0	long	1	0	0	0	1	
Printers housing panels	R/Waste	R-PP	PP	Electrics & electronics	1	0	1	0	0	1	0	4	0	12	8	0	long	1	0	0	0	1
Trays for food	Biomass	PLA	PS, PET	Rigid packaging	1	1	0	1	1	1	0	6	0	22	9	1	short	0	1	1	1	1
Trays for food	Biomass	Bio-PBSC	PS	Rigid packaging	1	0	0	1	0	1	0	6	0	16	9	1	short	0	1	1	1	1
Trays for food	Biomass	PHA	PP	Rigid packaging	1	0	0	1	0	1	0	6	0	16	9	1	short	0	1	1	1	1
Trays for food	Biomass	PHB	PS	Rigid packaging	1	0	0	1	0	1	0	6	0	16	9	1	short	0	1	1	1	1
Trays for food	Biomass	Starch blends	PS, EPS	Rigid packaging	1	0	0	1	0	1	0	6	0	16	9	1	short	0	1	1	1	1
Trays for food	Biomass	Natural fibres	EPS	Rigid packaging	1	0	0	0	0	1	0	6	0	13	9	1	short	0	1	1	1	1
Loose fill chips	Biomass	PLA	EPS	Rigid packaging	1	1	0	1	1	1	0	4	0	18	10	0	short	0	1	1	1	1
Loose fill chips	Biomass	PLA-E	EPS	Rigid packaging	1	0	0	1	1	1	0	4	0	15	10	0	short	0	1	1	1	1
Loose fill chips	Biomass	Starch blends	EPS	Rigid packaging	1	0	0	1	0	1	1	4	0	13	10	0	short	0	1	1	1	1
Loose fill chips	Biomass	Miscanthus	EPS	Rigid packaging	1	0	0	0	0	1	0	4	0	9	10	0	short	0	1	1	1	1
Other consumer goods	R/Waste	R-PE (HD-MD)	PE (HD-MD)	Consumer goods	1	0	1	0	0	1	0	4	0	12	11	0	long	1	0	0	0	1
Other consumer goods	R/Waste	R-PE (LD-LLD)	PE (LD-LLD)	Consumer goods	1	0	1	0	0	1	0	4	0	12	11	0	long	1	0	0	0	1
Other consumer goods	R/Waste	R-PP	PP	Consumer goods	1	0	1	0	0	1	0	4	0	12	11	0	long	1	0	0	0	1
Other consumer goods	R/Waste	R-PS	PS	Consumer goods	1	0	1	0	0	1	0	4	0	12	11	0	long	1	0	0	0	1
Cups	Biomass	PLA	PS, EPS, PP (also reusable), PET, PE, PE-coated cardboard	Rigid packaging	1	1	0	1	1	1	2	3	0	18	12	1	short	0	1	1	1	1
Cups	R/Waste	R-PET	PS, EPS, PP, PET, PE	Rigid packaging	1	0	1	0	0	1	0	3	0	10	12	1	short	1	0	1	1	1
Cups	Biomass	Starch blends	PS	Rigid packaging	1	0	0	1	0	1	0	3	0	10	12	1	short	0	1	1	1	1
Other agricultural products	Biomass	PLA	PP	Agriculture & horticulture	1	0	0	1	1	1	1	3	1	16	13	0	short	0	1	1	0	0
Other agricultural products	R/Waste	R-PP	PP	Agriculture & horticulture	1	0	0	0	0	1	0	3	1	9	13	0	short	1	0	1	0	0
Other agricultural products	Biomass	Starch blends	PE (HD), PP	Agriculture & horticulture	1	0	0	0	0	1	0	3	1	9	13	0	short	0	1	1	0	0

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)	
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2
Foam for consumer goods	Biomass	PLA	EPS core	Consumer goods	1	0	0	1	1	1	3	0	14	14	0	long	0	1	0	1	
Foam for consumer goods	Biomass	Bio-PUR	PUR	Consumer goods	1	0	0	1	0	1	3	0	10	14	0	long	1	0	0	1	
Foam for consumer goods	Biomass	Starch blends	EPS	Consumer goods	1	0	0	0	1	2	3	0	9	14	0	long	0	1	0	1	
Other packaging (non-food)	Biomass	Bio-PE	PE (LD)	Flexible packaging	1	1	0	1	1	0	3	0	16	15	0	short	1	0	1	0	
Other packaging (non-food)	Biomass	Starch blends	PE (LD)	Flexible packaging	1	1	0	0	1	0	3	0	10	15	0	short	0	1	1	0	
Other packaging (non-food)	CO ₂	CO ₂ -based PPC	PPC	Flexible packaging	1	0	0	0	0	0	3	0	6	15	0	short	1	0	1	0	
Fibres	Biomass	Bio-PA	PA/PTT	Consumer goods	1	0	0	1	0	1	1	0	6		0	long	1	0	0	0	
Sanitary towels	Biomass	PLA	PE, PP	Consumer goods	1	0	0	1	1	1	4	0	16		1	short	0	1	1	0	
Sanitary towels	Biomass	PHA	PE, PP	Consumer goods	1	0	0	1	0	1	4	0	12		1	short	0	1	1	0	
Sanitary towels	Biomass	PHB	PE, PP	Consumer goods	1	0	0	1	0	1	4	0	12		1	short	0	1	1	0	
Sanitary towels	Biomass	Starch blends	PE, PP	Consumer goods	1	0	0	0	1	0	4	0	9		1	short	0	1	1	0	
Other construction products	Biomass	Bio-PE	PE	Building & construction	1	0	0	0	1	0	3	0	10		0	very long	1	0	0	1	
Other construction products	R/Waste	R-PE (HD-MD)	PE (HD-MD)	Building & construction	1	0	1	0	0	1	0	3	10		0	very long	1	0	0	1	
Other construction products	R/Waste	R-PP	PP	Building & construction	1	0	1	0	0	1	0	3	10		0	very long	1	0	0	1	
Injection moulding products	Biomass	Bio-PBS	PBS	Consumer goods	1	0	0	0	1	0	1	0	3		0	long	0	1	0	1	
Automotive flexible plastic parts	Biomass	Bio-PET	PUR, PVC, PP	Automotive & transport	1	0	0	1	0	1	0	1	6		0	long	1	0	0	1	
Foam for automotive	Biomass	Bio-PUR	PUR	Automotive & transport	1	1	0	1	0	1	0	1	9		0	long	1	0	0	1	
Automotive textiles	Biomass	Bio-PUR	PUR	Automotive & transport	1	1	0	1	0	1	2	0	11		0	long	1	0	0	0	
Automotive textiles	Biomass	Bio-PE	PBT	Automotive & transport	1	0	0	0	1	1	2	0	8		0	long	1	0	0	0	
Beverage cartons	R/Waste	R-PET	Tetra Pak, PET, PE (HD)	Rigid packaging	1	0	1	0	0	1	7	0	19		1	short	1	0	1	1	
Beverage cartons	Biomass	Bio-PE	Tetra Pak, PET, PE (HD)	Rigid packaging	1	1	0	0	1	1	1	0	10		1	short	1	0	1	1	
Caps for beverage bottles	Biomass	Bio-PE	PE (HD)	Rigid packaging	1	1	0	1	1	0	1	0	12		1	short	1	0	1	1	

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)	
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2
Caps for non-beverage bottles	Biomass	Bio-PE	PE (HD)	Rigid packaging	1	1	0	1	1	1	0	2	0	14		0	short	1	0	1	1
Caps for non-beverage bottles	Biomass	PHA	PE (HD)	Rigid packaging	1	0	0	1	0	1	0	2	0	8		0	short	0	1	1	1
Carpets	Biomass	Bio-PA	PA, PTT	Textiles	1	0	0	0	0	1	0	2	0	5		0	long	1	0	0	1
Carpets	Biomass	Bio-PTT	PA, PTT	Textiles	1	0	0	0	0	1	0	2	0	5		0	long	1	0	0	1
Carrier bags	Biomass	PLA	PE, PET	Flexible packaging	0	1	0	1	1	1	2	11	0	0		1	short	0	1	1	0
Carrier bags	Biomass	Bio-PE(HD)	PE (HD)	Flexible packaging	0	1	0	1	1	1	1	11	0	0		1	short	1	0	1	0
Carrier bags	Biomass	Bio-PE	PE (LD)	Flexible packaging	0	1	0	1	1	1	0	11	0	0		1	short	1	0	1	0
Carrier bags	R/Waste	R-PET	PET	Flexible packaging	0	0	1	0	0	1	0	11	0	0		1	short	1	0	1	0
Carrier bags	Biomass	PHA	PP	Flexible packaging	0	0	0	1	0	1	0	11	0	0		1	short	0	1	1	0
Carrier bags	Biomass	Starch blends	PE (LD)	Flexible packaging	0	1	0	0	0	1	0	11	0	0		1	short	0	1	1	0
Carrier bags	R/Waste	R-PE	PE	Flexible packaging	0	0	0	0	0	1	0	11	0	0		1	short	1	0	1	0
Carrier bags	R/Waste	R-PE (HD)	PE (HD)	Flexible packaging	0	0	0	0	0	1	0	11	0	0		1	short	1	0	1	0
Carrier bags	R/Waste	R-PE (LD)	PE (LD)	Flexible packaging	0	0	0	0	0	1	0	11	0	0		1	short	1	0	1	0
Carrier bags	R/Waste	R-PE (LD-LLD)	PE (LD-LLD)	Flexible packaging	0	0	0	0	0	1	0	11	0	0		1	short	1	0	1	0
Carrier bags	Biomass	Bio-PET	PE (LD), PP, PET	Flexible packaging	0	0	0	0	0	1	0	11	0	0		1	short	1	0	1	0
Cigarette buds	Biomass	Starch blends	CA (?)	Consumer goods	1	0	0	0	0	1	0	1	0	3		1	short	0	1	1	0
Containers/boxes for food	Biomass	PLA	PET, PS, EPS, PP	Rigid packaging	1	1	0	1	1	1	0	3	0	16		1	short	0	1	1	1
Containers/boxes for food	Biomass	Bio-PBS	PS	Rigid packaging	1	0	0	1	0	1	0	3	0	10		1	short	0	1	1	1
Containers/boxes for food	Biomass	Starch blends	PS, PP	Rigid packaging	1	0	0	1	0	1	0	3	0	10		1	short	0	1	1	1
Containers/boxes for non-food	Biomass	PLA	PET, PS	Rigid packaging	1	1	0	1	1	1	0	1	0	12		0	short	0	1	1	1
Crates	Biomass	Bio-PE	PE (HD)	Rigid packaging	1	1	0	1	1	1	0	2	0	14		0	long	1	0	0	1
Crates	Biomass	Starch blends	PE (HD), PP	Rigid packaging	1	0	0	1	0	1	0	2	0	8		0	long	0	1	0	1
Dashboard fascia	Biomass	Bio-PUR	ABS, PUR	Automotive & transport	1	1	0	1	0	1	0	2	0	11		0	long	1	0	0	1
Dashboard fascia	Biomass	Bio-PA	ABS, PUR	Automotive & transport	1	0	0	1	0	1	0	2	0	8		0	long	1	0	0	1
Door handles	Biomass	Bio-PUR	PBT	Automotive & transport	1	1	0	1	0	1	0	2	0	11		0	long	1	0	0	1

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)	
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2
Door handles	Biomass	Bio-PA	PBT	Automotive & transport	1	0	0	1	0	1	0	2	0	8		0	long	1	0	0	1
Electro-domestic parts	Biomass	PHA	PET, PP, PE (HD-MD), PE (LD-LLD)	Electrics & electronics	1	0	0	1	0	1	0	2	0	8		0	long	0	1	0	1
Electro-domestic parts	CO ₂	CO ₂ -based PPC	PPC	Consumer goods	1	0	0	0	0	0	0	2	0	4		0	long	1	0	0	1
Frames	Biomass	Starch blends	PVC	Building & construction	1	0	0	0	0	1	0	1	0	3		0	very long	0	1	0	1
Garbage bags	Biomass	PLA	PE	Flexible packaging	0	1	0	1	1	1	1	4	0	0		1	short	0	1	1	0
Garbage bags	Biomass	Bio-PE	PE	Flexible packaging	0	1	0	1	1	1	0	4	0	0		1	short	1	0	1	0
Garbage bags	Biomass	Bio-PBAT/Starch	PE	Flexible packaging	0	0	0	1	0	1	0	4	0	0		1	short	0	1	1	0
Garbage bags	Biomass	Starch blends	PE (HD), PCL	Flexible packaging	0	1	0	0	0	1	0	4	0	0		1	short	0	1	1	0
Laptop covers	Biomass	PLA	PP, other ETP	Electrics & electronics	1	0	0	1	1	1	0	1	0	9		0	long	0	1	0	1
Non-beverage bottles	Biomass	Bio-PE	PE (HD)	Rigid packaging	1	1	0	1	1	1	0	1	0	12		0	short	1	0	1	1
Other automotive components	Biomass	PHA	PP	Automotive & transport	1	0	0	0	0	1	0	3	0	7		0	long	0	1	0	1
Other automotive components	Biomass	PHB	PP	Automotive & transport	1	0	0	0	0	1	0	3	0	7		0	long	0	1	0	1
Other automotive components	R/Waste	R-PP	PP	Automotive & transport	1	0	1	0	0	1	0	3	0	10		0	long	1	0	0	1
Packaging of packaging	Biomass	Bio-PE	PE (LD-LLD), PP	Flexible packaging	1	1	0	1	1	1	0	2	0	14		1	short	1	0	1	0
Packaging of packaging	Biomass	Starch blends	PE (LD)	Flexible packaging	1	1	0	0	0	1	0	2	0	8		1	short	0	1	1	0
Pallets	R/Waste	R-PE (HD)	PE (HD)	Rigid packaging	1	0	0	0	0	1	0	1	0	3		0	long	1	0	0	1
Pots	Biomass	PLA	PE (HD), PP	Agriculture & horticulture	1	0	0	1	1	1	0	3	0	13		0	short	0	1	1	1
Pots	Biomass	PLA-E	PE (HD), PP	Agriculture & horticulture	1	0	0	1	1	1	0	3	0	13		0	short	0	1	1	1
Pots	Biomass	Starch blends	PE (HD), PP	Agriculture & horticulture	1	0	0	0	0	1	0	3	0	7		0	short	0	1	1	1
Printed Wiring Boards	Biomass	Lignin-epoxy blend	PWB epoxy resin	Electrics & electronics	1	0	0	0	0	1	0	1	0	3		0	long	1	0	0	1
Rigid-packaging	Biomass	Bio-PBAT	PBAT	Rigid packaging	1	0	0	1	0	1	0	1	0	6		0	short	0	1	1	1
Ropes	Biomass	Bio-PA	PA	Building & construction	1	0	0	1	0	1	0	1	0	6		0	long	1	0	0	0
Straws	Biomass	PLA	PP	Rigid packaging	0	1	0	1	1	1	0	2	0	0		1	short	0	1	1	1

Product	Feed-stock	Alternative feedstock-based polymer	Conventional feedstock-based polymer	Market Sector	a)	b)				c)	d)		e)	Tot*	Rank	f)				g)	
						b.1	b.2	b.3	b.4/h.1		d.1	d.2				f.1	f.2	f.3	f.4	g.1	g.2
Straws	Biomass	Bio-PBS	PP	Rigid packaging	0	0	0	1	0	1	0	2	0	0		1	short	0	1	1	1
Tableware	Biomass	PLA	PP, PS, PE	Rigid packaging	0	1	0	1	1	1	0	5	0	0		1	short	0	1	1	1
Tableware	Biomass	PHA	PP, PS	Rigid packaging	0	0	0	1	0	1	0	5	0	0		1	short	0	1	1	1
Tableware	Biomass	PHB	PP, PS	Rigid packaging	0	0	0	1	0	1	0	5	0	0		1	short	0	1	1	1
Tableware	Biomass	Starch blends	PP, PS (GPPS)	Rigid packaging	0	0	0	1	0	1	0	5	0	0		1	short	0	1	1	1
Tableware	Biomass	CA (Biograde)	PP, PS	Rigid packaging	0	0	0	0	0	1	0	5	0	0		1	short	1	0	1	1
Toys/Houseware	Biomass	Bio-PE	PE (HD)	Consumer goods	1	0	1	1	1	1	0	1	0	12		0	long	1	0	0	1
Wire coatings	Biomass	Bio-PA	PA	Electrics & electronics	1	0	0	1	0	1	0	1	0	6		0	long	1	0	0	1
Selection criteria (Section A.2.1)				Additional criteria (Section A.3)																	
a) Policy options				f) End of Life (EoL)-related aspects, including littering																	
b) Market potential				f.1) littering (marine)																	
b.1) market size of bio-based polymers				f.2) expected lifetime/durability																	
b.2) market size of recycled plastic waste-based polymers				f.3) recyclability																	
b.3) identifying market trend				f.4) bio-degradability (bio-degradation in technical/industrial systems or in/onto soil)																	
b.4) market criticality				g) Uses																	
c) Promise for deployment				g.1) single-use vs multiple-use products																	
d) Availability and quality of data needed for the LCA analysis				g.2) rigid vs flexible applications																	
d.1) quality of available LCA studies				h) Market coverage																	
d.2) LCA scenario number				h.1) import dependency																	
e) End of Life (EoL) options/scenarios																					

(*) Total score calculated after weighting.

A.5. References

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Annex B – Normalisation and weighting factors

The sets of normalisation and weighting factors applied in the case studies to calculate normalised and weighted impact assessment results are reported in Table B.1. As discussed in Section 2.3, the factors specified in the EF 2.0 reference package were applied, albeit the EF 3.0 package has been released in the meantime. Particularly, per-capita normalisation factors calculated based on global values were considered, as prescribed in the *Plastics LCA* method (Section 5.2.1). For weighting, the set of factors determined excluding Human Toxicity and Ecotoxicity impact categories was applied, as these categories were not considered in the calculation of the total normalised and weighted impact score (see Section 2.3 for details).

Table B.1. Normalisation and weighting factors applied in the illustrative LCA case studies.

Impact category	Unit	Normalisation factor (1)	Weighting factor (2)
Climate change	kg CO ₂ eq.	7.76E+03	22.19%
Ozone depletion	kg CFC-11 eq.	2.34E-02	6.75%
Human toxicity – cancer	CTU _h	3.85E-05	-
Human toxicity – non-cancer	CTU _h	4.75E-04	-
Particulate matter	Disease incidence	6.37E-04	9.54%
Ionising radiation – human health	kBq U ²³⁵ eq.	4.22E+03	5.37%
Photochemical ozone formation – human health	kg NMVOC eq.	4.06E+01	5.1%
Acidification	mol H ⁺ eq.	5.55E+01	6.64%
Eutrophication – terrestrial	mol N eq.	1.77E+02	3.91%
Eutrophication – freshwater	kg P eq.	2.55E+00	2.95%
Eutrophication – marine	kg N eq.	2.83E+01	3.12%
Ecotoxicity freshwater	CTU _e	1.18E+04	-
Land use	Pt	1.33E+06	8.42%
Water use	m ³ world eq.	1.15E+04	9.03%
Resource use – fossils	Kg Sb eq.	6.53E+04	8.92%
Resource use – mineral and metals	MJ	5.79E-02	8.08%

(1) Expressed as Unit per person, based on global values.

(2) Excluding toxicity-related impact categories (Human toxicity – cancer, Human Toxicity – non-cancer, and Ecotoxicity – freshwater), as discussed in Section 2.3.

Annex C – Analysis of the representativeness of the applied crude oil mix for fossil-based polymers

This annex presents an analysis of the representativeness of the average crude oil mix assumed to be supplied to the EU market when modelling the life cycle of most fossil-based polymers investigated in the case studies. These polymers include PET, HDPE, LDPE and EPS, which could be modelled based on partially aggregated cradle-to-gate datasets disaggregating upstream feedstock inputs such as naphtha, crude oil and natural gas. However, similar considerations and results reasonably apply also to the remaining fossil-based polymers or co-polymers addressed (i.e. PUR and PBAT), as the fully vertically aggregated datasets used to model the respective fossil-based precursors (e.g. aromatic polyols and purified terephthalic acid) are developed by the same data provider as the applied crude oil supply dataset (described below).

C.1. Premises and objectives

When modelling the production of fossil-based polymers in the EU, a specific dataset had in most cases to be selected to represent the supply of crude oil to the average EU market (i.e. the EU-average crude oil mix). Following the selection hierarchy described in Section 2.5, the following Environmental Footprint (EF)-compliant dataset was specifically chosen for modelling (see Section 3.5.1.1 of the Beverage Bottles case study for further detail): "*[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*".

This dataset was used as an input to the partially aggregated cradle-to-gate datasets applied to model the entire downstream process chain associated with the conversion of relevant fossil-based feedstocks (i.e. crude oil and natural gas) into most of the investigated polymers (i.e. PET, HDPE, LDPE and PS). The refining and cracking steps, and/or any other conversion process until polymer production (polymerisation) were thus covered in these separate datasets, which are not addressed in this analysis.

The crude oil mix considered for EU-27 in the selected dataset refers to the year 2014, and may thus disregard any relevant more recent developments in the crude oil market, such as the expansion of unconventional oil sources. These include, for instance, light tight oil (shale oil) from US, and heavy crude from oil sand deposits in Canada. Considering the general relevance of the feedstock mix on the overall environmental impacts of fossil-based polymers, it was deemed important to evaluate whether the applied dataset was sufficiently representative of the global crude oil supply and trade at the time of the case studies, keeping in mind their illustrative nature.

C.2. Method

The following approach was followed in the analysis of the representativeness of the applied dataset:

- First, the characteristics of the EU-average crude oil mix considered in the dataset were described, and other relevant input data for the analysis were identified (Section C.3.1).
- Variations in the crude oil mix compared with the current situation were then evaluated (Section C.3.2).
- Potential environmental implications of the identified changes in the crude oil mix were subsequently investigated and, where possible, quantified (Sections C.3.3 and C.3.4).

- Finally, conclusions on the representativeness of the applied dataset were drawn, and general recommendations for future improvement were provided (Section C.4).

C.3. Analysis and results

This section describes the main steps and the results of the analysis of the representativeness of the EU crude oil mix applied for modelling in the case studies. Particularly, Section C.3.1 describes the data used as input to the analysis, while Section C.3.2 investigates the changes occurred in the EU-28 crude oil mix between 2014 and 2018. The potential environmental relevance of these changes is then quantitatively or qualitatively addressed in Section C.3.3 for Climate Change (quantitative assessment), and in Section C.3.4 for other relevant impact categories (qualitative discussion).

C.3.1. Input data

The average crude oil mix considered in the EF-compliant dataset applied in the case studies to model crude oil supply to the EU-27 market is reported in Figure C.1.

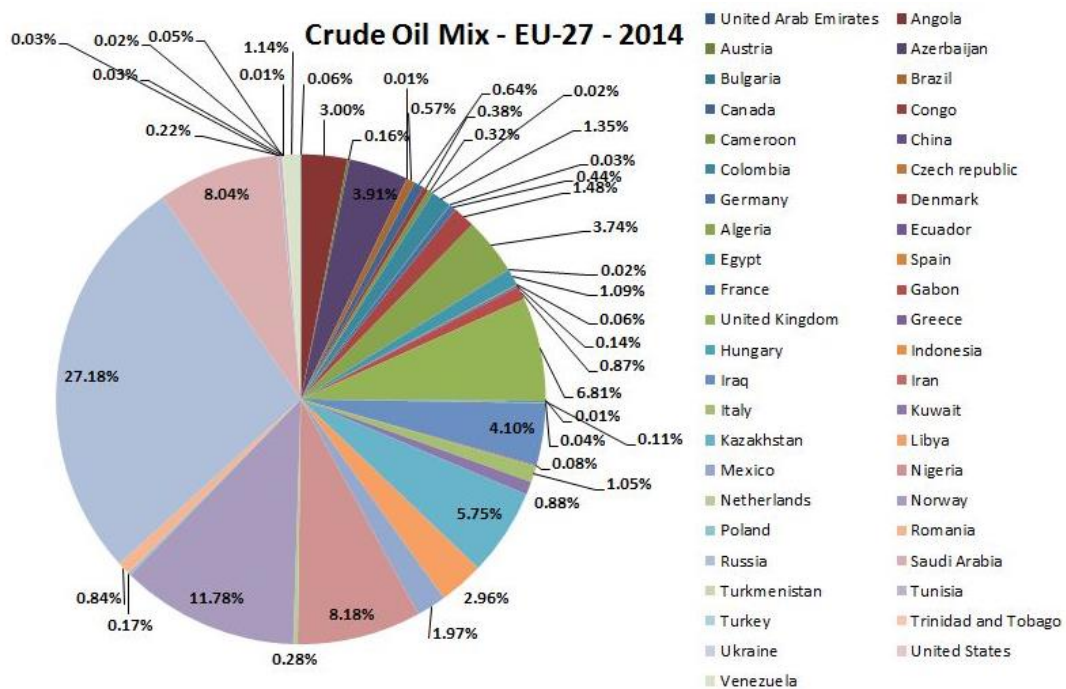


Figure C.1. Crude oil Mix for EU-27 in the year 2014, as considered in the EF-compliant dataset applied in the case studies to model crude oil supply to the EU market.

Since the mix considered in the applied crude oil dataset refers to the year 2014, the first issue to be addressed was to understand whether the mix had changed significantly compared to the current situation. For this purpose, official EU statistics on crude oil imports and deliveries to the EU-28 were considered (EC, 2020). Most recent data available at the time of this analysis referred to the year 2018, so that differences in the crude oil mix were evaluated between 2014 and 2018. The data series considering both intra and extra EU trade were chosen, to capture also deliveries among EU countries. Compared to the crude oil mix considered in the applied dataset (Figure C.1), these values presented the following differences:

- i) Data are reported for EU-28 rather than EU-27, thus including the contribution of Croatia, which is not considered in the oil mix represented in the applied dataset. However, Croatia accounted for only 0.5% of the total imports to the EU, and its contribution was considered negligible.
- ii) On the other hand, domestic supply and consumption of crude oil are not captured in EU statistics (as opposed to the applied dataset), and their contribution was more relevant (11% of the overall consumption of crude oil)¹⁷¹.

To exclude the effects of these differences, the analysis of changes in the EU crude oil mix between 2014 and 2018 was thus conducted relying entirely on official EU statistics (which are inherently consistent), disregarding the values considered in the applied crude oil dataset for 2014 (as shown in Figure C.1). The assessment of changes in the crude oil mix between 2014 and 2018 is described in Section C.3.2.

C.3.2. Changes in the EU crude oil mix between 2014 and 2018

Figure C.2 represents the crude oil mix of imports and deliveries to the EU-28 in 2014 and in 2018, according with official EU statistics from EC (2020). Building upon these data, Figure C.3 highlights the main changes between the two time steps, and helps to identify the most relevant differences between 2014 and 2018.

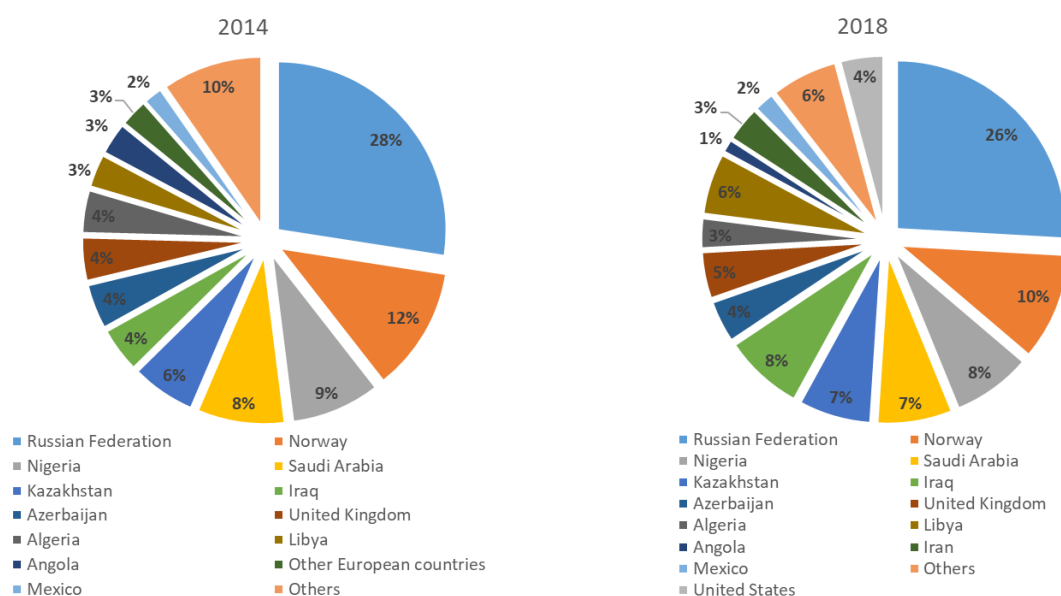


Figure C.2. Mix of import and deliveries of crude oil in EU-28 in 2014 and 2018. Source: EC (2020).

¹⁷¹ Calculated as: primary production / (primary production + imports – exports).

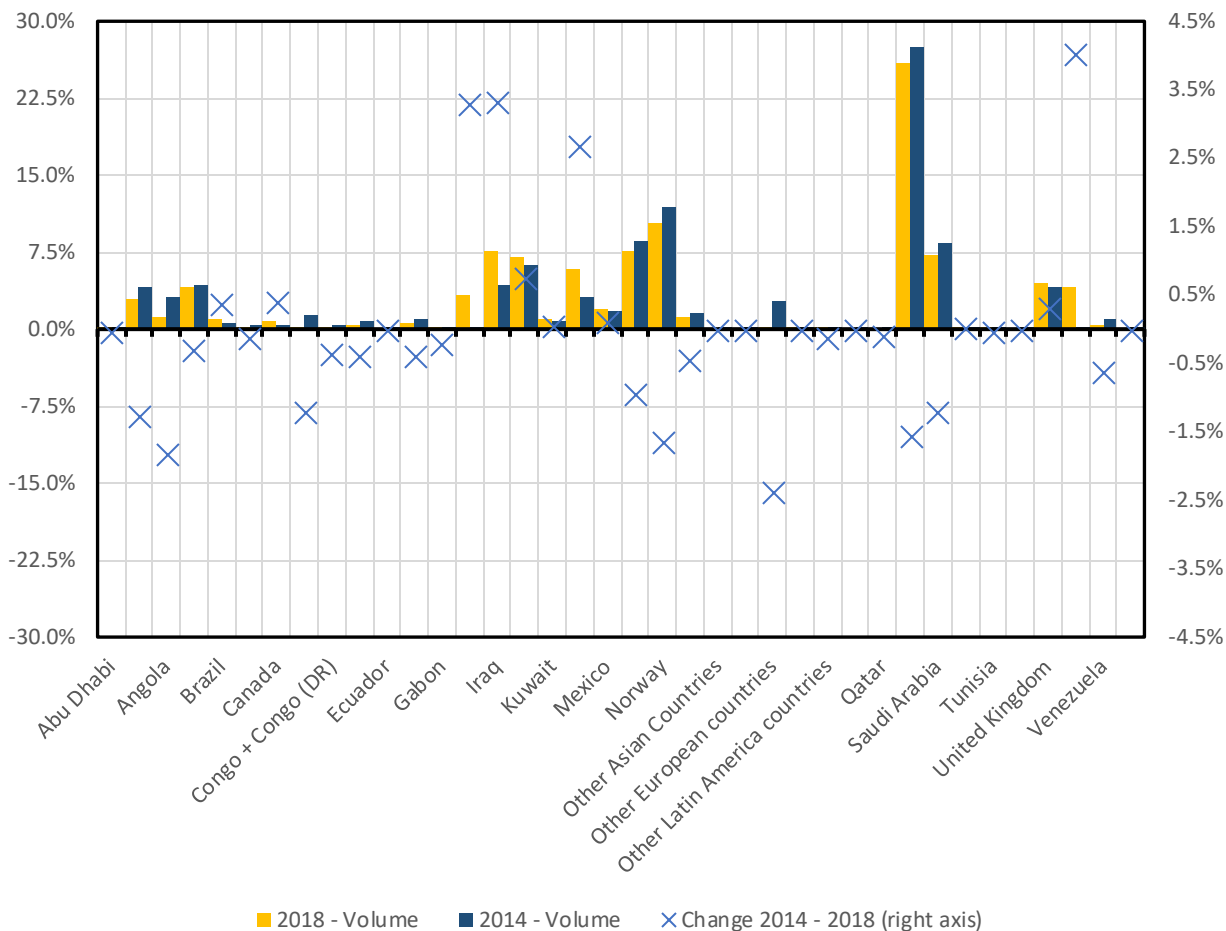


Figure C.3. Share of origin of crude oil imports and deliveries to EU-28 in 2014 and 2018 (left axis), and relative changes in share between 2014 and 2018 (right axis).

Overall, no relevant changes occur between 2014 and 2018 for the reported crude oil imports and deliveries to the EU-28, with the highest (absolute) variation equalling 4% (for US imports). However, for most of the contributions, the absolute variation is more limited, ranging between 0.1% and 1.8%. Within this general framework, the following specific considerations can be made:

- Most of the recorded variations can be ascribed to changing geopolitical conditions, such as the growth in imports from Iran linked to the signature of the Joint Comprehensive Plan of Action (JCPOA) in 2015 and subsequent lifting of oil trade restrictions, as well as imports from Iraq and Libya following improvements in political stability in those countries. Forecasting how this might change again, even in the near term, is close to impossible. For instance, it is likely that oil exports from Iran to EU may revert back to zero, following the failure of the JCPOA in 2019, and the subsequent reinstatement of US sanctions on Iran’s oil. Libya and Iraq, as well, have fallen back into civil war and internal unrest in 2019, which may affect future oil supply.
- One of the main changes between 2014 and 2018 is the growth of imports of US crude oil to the EU (Figure C.4). The share of US oil in the European mix has in fact grown from nearly 0% in 2014 to more than 4% in 2018 (Figure C.3). This development follows the continuous expansion of US tight oil (or shale oil)

production, which has caused US to become a net energy exporter in 2019 for the first time since the beginning of statistical records (Meyer, 2019). However, this does not automatically ensure that crude oil imported from US to EU is represented by light tight oil.

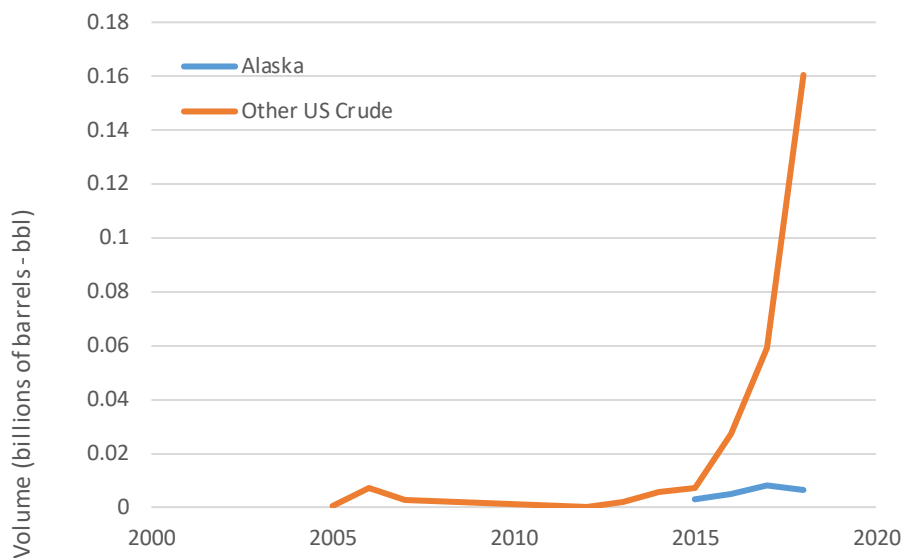


Figure C.4. Trend in imports of US crude oil to EU-28 (2005-2018).

- Another notable change is the decrease in intra-EU trade of crude oil (-2.4%; Figure C.3), which is to be expected as domestic production in the EU-28 has declined by 34% since 2008 (EUROSTAT, 2020) and it is forecasted to continue declining.

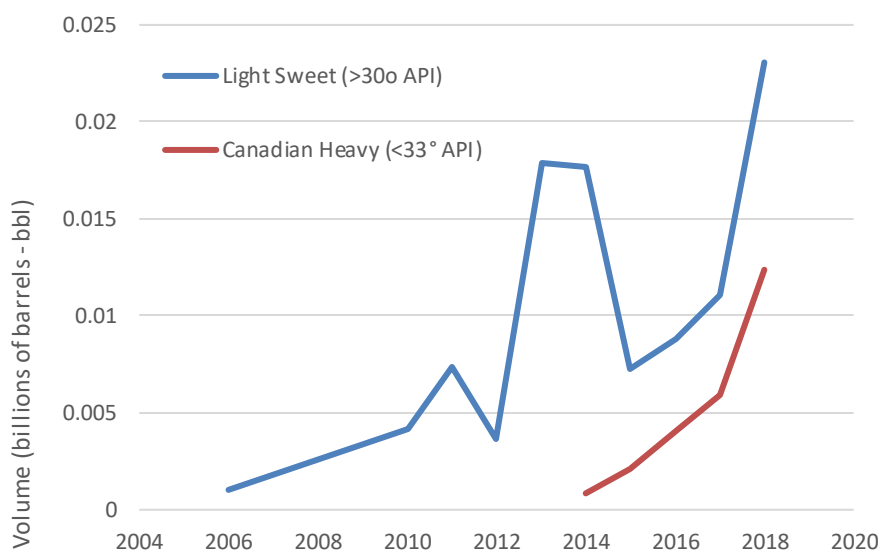


Figure C.5. Trend in imports of Canadian crude oil to EU-28 (2006-2018).

- A last important difference concerns the import of Canadian crudes, including both heavy crudes produced from oil sands, but also light crudes from conventional sources and tight oils (Government of Canada, 2019) (Figure C.5). While the quantities of Canadian heavy oils reaching the EU market were still limited in 2018 (ca. 0.3% of the EU crude oil mix import in that year), their import volume has ramped up between 2014 and 2018 (Figure C.5). Indeed, there are signs that the EU market for Canadian heavy oil may be expanding in the future due to the declining quantities of heavy crudes imports from other countries (Tuttle and Kassai, 2019) and to increasing investments to expand oil sands production and transport (Leahy, 2019; TC Energy, 2020). It has to be noted that only crude oil trade is relevant to this assessment, and not the trade of derived petroleum products from refined oil sands crude, which may be imported to EU after refining in the US (Swift and Droitsch, 2014).

C.3.3. Environmental significance of the changes in the EU crude oil mix: Climate Change impact (GHG emissions)

Based on the findings from Section C.3.2, it was considered meaningful to evaluate whether the modest changes in the EU crude oil mix estimated to occur between 2014 and 2018 might, on the other hand, lead to more relevant changes in the resulting potential environmental impacts. Any relevant change would have not been captured in the case studies, and should have been properly taken into account in the interpretation of their results (or alternative, more representative data(sets) should have been identified, if available).

For the Climate Change impact category (driven by GHG emissions), the evaluation was based on the exercise carried out by ICCT in 2014 (Malins et al., 2014) to calculate the upstream GHG intensity of the EU crude oil mix. In such exercise, the OPGEE model (Stanford University, 2018) was used to calculate a detailed GHG balance of most relevant upstream operations of many oil fields around the globe (including a series of activities such as exploration, drilling, extraction, processing, upgrading, maintenance, waste handling, flaring, etc.). Based on this balance, a final emission factor of 10 g CO₂-eq./MJ_{crude} was estimated for the EU oil mix as of 2011.

For this analysis, identical GHG emission factors as those provided by Malins et al. (2014) were applied (as reported in Table G of Swift and Droitsch, 2014), and subsequently weighed on the EU crude oil mix of 2014 and 2018. Notably, tight crude oil from US and heavy crude from Canadian oil sands were not addressed in the ICCT study. To cover this gap, the upstream GHG emission factors calculated by the Carnegie Endowment for International Peace (2019) in their Oil-Climate Index (OCI) were used. These values are quantified applying the same tool as the ICCT study (i.e. OPGEE), although full consistency between the two sets of factors could not be ensured. Specifically, the upstream GHG emission factors reported in Table C.1 were applied in this analysis for light tight oil from US and heavy crude from Canadian oil sands.

Table C.1. Upstream GHG emission factors considered for light tight oil from US and heavy crude from Canadian oil sands (not provided by Malins et al., 2014). Source: Carnegie Endowment for International Peace (2019).

Crude oil name (OCI)	Crude oil type	GHG emission factor [g CO₂-eq./MJ]
Canada		
Canada Athabasca SAGD Dilbit	Heavy (21-23 API)	21
Canada Cold Lake CSS Dilbit	Heavy (21-23 API)	24
<i>Canada Heavy (average of the above)</i>		<i>22.5</i>
Canada Athabasca FC-HC-SCO	Light (32-42 API)	33
Canada Athabasca DC-SCO	Light (32-42 API)	24
Canada Hibernia	Light (32-42 API)	5
<i>Canada Light Sweet (average of the above)</i>		<i>20.7</i>
United States		
U.S. Bakken Flare	Light (32-42 API)	16
U.S. Texas Spraberry	Light (32-42 API)	8
U.S. Texas Eagle Ford Black Oil Zone	Light (32-42 API)	8
U.S. Bakken No Flare	Light (32-42 API)	5
U.S. Wyoming WS	Light (32-42 API)	4
<i>Other US crude (average of the above)</i>		<i>8</i>

The calculation of the potential Climate Change impact (GHG intensity) of the EU crude oil mix as of 2014 and 2018 is reported in Tables C.2 and C.3, respectively. Only crude oil imports to EU were considered in the calculation, in the absence of specific emission factors for intra-EU deliveries and of information on single delivering countries. For similar reasons (i.e. no emission factors available), not all imports could be covered, although the overall coverage was in both cases larger than 90% (i.e. 91% in 2014 and 95% in 2018). A comparison between the total Climate Change impact estimated for the EU crude oil mix in 2014 and 2018 is shown in Figure C.6. For completeness, the comparison also accounts for the effects of the variation of the emission factors for US light tight oil and heavy crude from Canadian oil sands, considering the ranges reported in Table C.1.

Table C.2. Calculation of the potential Climate Change impact (GHG intensity) of the crude oil mix imported to EU-28 in 2014. Total coverage of crude oil imports considered for calculation is 91%.

Country	Crude oil type	Volume share	Volume share - cumulative	GHG emission factor [g CO ₂ -eq./MJ]
Russia	Other Russian Fed. Crude	12%	12%	9.8
Russia	Urals	16%	27%	12.5
Norway	Statfjord	1%	29%	4.5
Norway	Ekofisk	2%	31%	3.7
Norway	Other Norway Crude	6%	37%	4.8
Norway	Oseberg	1%	38%	4.8
Norway	Gullfaks	1%	39%	4
Nigeria	Medium (<33°)	3%	42%	18.3
Nigeria	Light (33-45°)	5%	48%	18.5
Nigeria	Condensate (>45°)	0%	48%	n.a.
Saudi Arabia	Arab Light	6%	54%	5.5
Saudi Arabia	Arab Medium	0%	55%	n.a.
Saudi Arabia	Other Saudi Arabia Crude	0%	55%	n.a.
Saudi Arabia	Arab Heavy	1%	55%	n.a.
Saudi Arabia	Berri (Extra Light)	1%	56%	5.5
Kazakhstan	Kazakhstan Crude	6%	63%	17.7
Iraq	Basrah Light	4%	66%	10.4
Iraq	Kirkuk	0%	67%	9
Iraq	Other Iraq Crude	0%	67%	11.5
Iraq	Azerbaijan Crude	4%	71%	5.4
UK	Flotta	0%	71%	10.4
UK	Forties	2%	73%	3.4
UK	Brent Blend	1%	74%	8.8
UK	Other UK Crude	2%	75%	6.7
Algeria	Saharan Blend	4%	79%	12.8
Algeria	Other Algeria Crude	0%	80%	15.4
Libya	Medium (30-40°)	1%	81%	13.6
Libya	Heavy (<30° API)	1%	82%	8.9
Libya	Light (>40°)	1%	83%	8.3
Angola	Cabinda	0%	83%	n.a.

Country	Crude oil type	Volume share	Volume share - cumulative	GHG emission factor [g CO ₂ -eq./MJ]
Angola	Other Angola Crude	3%	86%	9.2
Angola	Other Europe Crude	3%	89%	n.a.
Mexico	Olmecca	0%	89%	n.a.
Mexico	Isthmus	0%	89%	n.a.
Mexico	Maya	1%	90%	8.2
Egypt	Heavy (<30° API)	0%	91%	n.a.
Egypt	Medium/Light (30-40o)	1%	91%	8.9
Venezuela	Medium (22-30°)	0%	92%	n.a.
Venezuela	Extra Heavy (<17°)	1%	92%	8.4
Kuwait	Kuwait Blend	1%	93%	6
Denmark	Denmark Crude	1%	94%	3.2
Brazil	Brazil Crude	1%	95%	6.5
Canada	Canadian Heavy (<33° API)	0%	95%	22.5
Canada	Light Sweet (>30° API)	0%	95%	20.7
Congo	Congo Crude	0%	95%	13
Cameroon	Cameroon Crude	0%	96%	23.3
Weighted average GHG intensity of the crude oil mix [g CO₂-eq./MJ]				10.2

Table C.3. Calculation of the potential Climate Change impact (GHG intensity) of the crude oil mix imported to EU-28 in 2018. Total coverage of crude oil imports considered for calculation is 95%.

Country	Crude oil type	Volume share	Volume share - cumulative	GHG emission factor [g CO ₂ -eq./MJ]
Russia	Other Russian Fed. Crude	11%	11%	9.8
Russia	Urals	15%	26%	12.5
Norway	Statfjord	1%	27%	4.5
Norway	Ekofisk	2%	28%	3.7
Norway	Other Norway Crude	6%	34%	4.8
Norway	Oseberg	1%	35%	4.8
Norway	Gullfaks	1%	36%	4
Nigeria	Medium (<33°)	3%	39%	18.3

Country	Crude oil type	Volume share	Volume share - cumulative	GHG emission factor [g CO₂-eq./MJ]
Nigeria	Light (33-45°)	5%	44%	18.5
Nigeria	Condensate (>45°)	0%	44%	n.a.
Saudi Arabia	Arab Light	5%	49%	5.5
Saudi Arabia	Arab Medium	0%	49%	n.a.
Saudi Arabia	Other Saudi Arabia Crude	0%	50%	n.a.
Saudi Arabia	Arab Heavy	1%	50%	n.a.
Saudi Arabia	Berri (Extra Light)	1%	51%	5.5
Kazakhstan	Kazakhstan Crude	7%	58%	17.7
Iraq	Basrah Light	4%	62%	10.4
Iraq	Kirkuk	1%	63%	9
Iraq	Other Iraq Crude	3%	66%	11.5
Iraq	Azerbaijan Crude	4%	70%	5.4
UK	Flotta	0%	70%	10.4
UK	Forties	0%	71%	3.4
UK	Brent Blend	1%	71%	8.8
UK	Other UK Crude	3%	74%	6.7
Algeria	Saharan Blend	3%	77%	12.8
Algeria	Other Algeria Crude	0%	77%	15.4
Libya	Medium (30-40°)	3%	80%	13.6
Libya	Heavy (<30° API)	0%	81%	8.9
Libya	Light (>40°)	2%	83%	8.3
United States	Alaska	0%	83%	n.a.
United States	Other US Crude	4%	87%	8
Iran	Other Iran Crude	0%	87%	11.7
Iran	Iranian Heavy	2%	89%	11.5
Iran	Iranian Light	1%	90%	16.2
Mexico	Isthmus	0%	90%	n.a.
Mexico	Maya	2%	92%	8.2
Angola	Cabinda	0%	92%	n.a.
Angola	Other Angola Crude	1%	94%	9.2
Brazil	Brazil Crude	1%	95%	6.5

Country	Crude oil type	Volume share	Volume share - cumulative	GHG emission factor [g CO ₂ -eq./MJ]
Kuwait	Kuwait Blend	1%	95%	6
Canada	Canadian Heavy (<33° API)	0%	96%	22.5
Canada	Light Sweet (>30° API)	1%	96%	20.7
Egypt	Heavy (<30° API)	0%	96%	n.a.
Egypt	Medium/Light (30-40°)	1%	97%	8.9
Weighted average GHG intensity of the crude oil mix [g CO₂-eq./MJ]				10.6

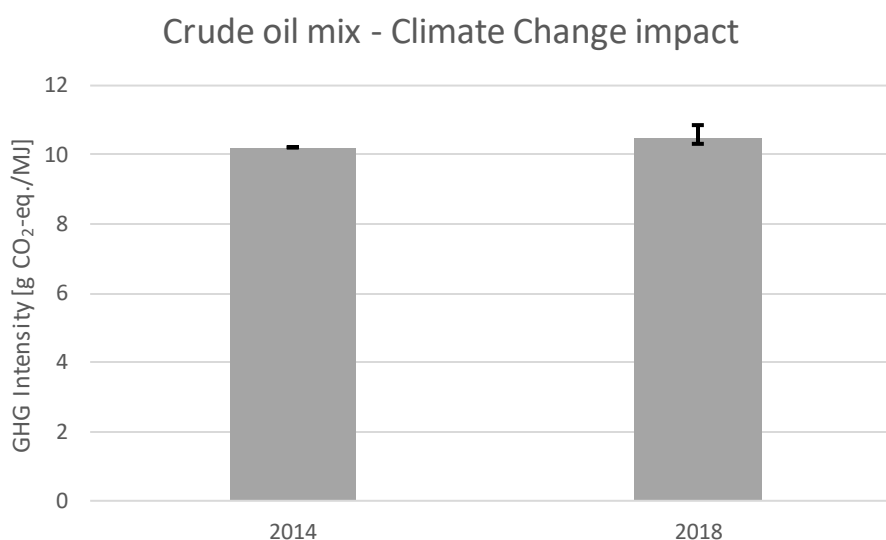


Figure C.6. Potential Climate Change impact (GHG intensity) of the crude oil mix imported to EU-28 in 2014 and 2018, considering the variation of the emission factors for US light tight oil and heavy crude from Canadian oil sands within the range reported in Table C.1.

From the values in Table C.1, it appears clear that while heavy crude from Canadian oil sands have higher upstream GHG emissions (25.5 g CO₂-eq./MJ on average) than most other crudes (3.2-18.5 g CO₂-eq./MJ; Tables C.2 and C.3), the same cannot be said for US tight oils, which mostly present values just below the average of all conventional crudes (i.e. 4-8 g CO₂-eq./MJ vs ca. 9 g CO₂-eq./MJ).

The overall variation in the Climate Change impact of the EU-28 crude oil mix, due to the changes estimated to occur between 2014 and 2018 (Section C.3.2), can be quantified based on the total impact values (GHG intensities) of the mix in those years, as reported in the last row of Table C.2 and Table C.3. Overall, a marginal impact increase of 0.4 g CO₂-eq./MJ_{crude} is registered in 2018 compared to 2014 (from 10.2 to 10.6 g CO₂-eq./MJ), corresponding to a relative increase of +3.9%.

Therefore, this simple exercise shows that an increased penetration of US light tight oil in the EU crude oil mix does not necessarily imply an increased Climate Change impact

(GHG intensity) of the mix, since the average GHG emission factor for US light crude (8 g CO₂-eq./MJ; Table C.1) is below the average value for conventional crudes (ca. 9 g CO₂-eq./MJ). On the other hand, a growing import of Canadian heavy oils could lead to an increased overall impact on Climate Change.

Placing these results in perspective, it can be noticed that calculations from Meili et al. (2018) indicate a rather higher value for GHG emissions from onshore production of crude oil in the US (16 g CO₂-eq./MJ; Table C.4), compared to the average emission factor estimated for US tight oil in Table C.1 (8 g CO₂-eq./MJ). They also show higher emissions for crude oil production in other countries, compared to the literature ranges they considered as a reference. Two tentative explanations are provided by the authors for these discrepancies:

1. venting emissions included in the underlying inventories might be higher than in the literature; and
2. emissions associated with infrastructures are accounted more consistently in such inventories compared to the literature. Particularly, for the case of US shale oil, these emissions may provide an important contribution to the overall emissions, due to the so-called "well treadmill" effect associated with shale oil production (Elliott, 2019). Following this effect, since productivity from wells decreases rather quickly in shale formations, new wells need to be drilled regularly in order to maintain or expand production rates. Therefore, it is possible that the values in Table C.1 underestimate the impact from infrastructures.

Also the emission factors reported in Table C.1 for Canadian oil sands may be underestimated, since Liggio et al. (2019) found that GHG emissions for oil sands-related operations may be underestimated by up to 30% compared to emissions reported by the industry. However, the values reported in the review presented in Malins et al. (2014) confirm the values considered in this analysis.

Table C.4. Total GHG emissions associated with the crude oil production datasets included in the *ecoinvent 3.6* database, and comparison with literature ranges. Source: Meili et al. (2018).

Tab. 14.4 Comparison of the greenhouse gas emissions and EROEI for crude oil with data from literature

technology	CO ₂ equivalents	CO ₂ -eq direct (IOGP2015)	Range EU study	EROEI (this study)	EROEI (direct model)	EROEI (in literature)	Comments
	gCO ₂ /MJ	gCO ₂ /MJ	gCO ₂ /MJ	-	-	-	
production onshore /RU	12.2	2.3	3-7.5	11	17	24	CO ₂ -eq higher mostly due to higher venting rate
production /NG	13.5	4.9	10-22	10	14	17	
production offshore /KZ	10.5	2.3	3-7.5	15	25	23	CO ₂ -eq higher mostly due to higher venting rate
production onshore /KZ	12.0	2.3	3-7.5	12	19	23	CO ₂ -eq higher mostly due to higher venting rate
production /KZ	11.7	2.3	3-7.5	12	20	23	CO ₂ -eq higher mostly due to higher venting rate
production offshore /NO	8.6	1.3	2.5-4	22	22	22	CO ₂ -eq higher mostly due to higher venting rate
production /IQ	15.1	1.3	3-10	9	11	11	CO ₂ -eq higher mostly due to higher venting and flaring rate
production offshore /MX	14.9	3.0	3-10	8		20	CO ₂ -eq higher mostly due to higher venting rate and high amount of wells needed
production onshore /MX	16.8	3.0	3-10	7		20	CO ₂ -eq higher mostly due to higher venting rate and high amount of wells needed
production /MX	15.4	3.0	3-10	7	20	20	CO ₂ -eq higher mostly due to higher venting rate and high amount of wells needed
production offshore /SA	7.3	1.3	1-5	35		49	CO ₂ -eq higher mostly due to higher venting rate
production onshore /SA	7.7	1.3	1-5	30		49	CO ₂ -eq higher mostly due to higher venting rate
production /SA	7.6	1.3	1-5	31	49	49	CO ₂ -eq higher mostly due to higher venting rate
production offshore /US	11.8	6.3	3-17	11		8	CO ₂ -eq higher mostly due to higher venting rate and high amount of wells needed
production onshore /US	16.0	6.3	3-17	7		8	CO ₂ -eq higher mostly due to higher venting rate and high amount of wells needed
production /US	15.2	6.3	3-17	7	7	8	CO ₂ -eq higher mostly due to higher venting rate and high amount of wells needed
production offshore /GB	3.4		3-4	20			Impacts seem to be heavily underestimated. Obsolete.
production offshore /NL	1.3		2-4	65			Impacts seem to be heavily underestimated. Obsolete.
production onshore /NL	0.8		2-4	102			Impacts seem to be heavily underestimated. Obsolete.
production onshore /RAF	8.8			12			Replaced now by NG
production onshore /RME	3.1			33			Replaced now by SA and IQ

C.3.4. Environmental significance of the changes in the EU crude oil mix: other impact categories

Concerning other potentially relevant environmental impacts associated with the use of oil sands for crude oil production (beyond those on Climate Change), no systematised datasets could be found. However, literature on the matter is abundant¹⁷². While, a full literature review was beyond the scope of this analysis, the following findings are highlighted here:

- Jordaan (2012) stressed the significant impact on land use from oil sand-based production of crude oil, especially during open pit mining. They also reported issues with water consumption (two times larger than the use associated with conventional oil resources), and the many unknowns concerning potential impacts on water quality, especially linked to tailings ponds. Rosa et al. (2016) found similar results.
- Leahy (2019) reported more anecdotal evidence of the impacts of oil sand-based crude production on deforestation, of the associated leakage of toxic elements from tailing waste ponds into the Athabasca River, and of the effects of local pollutants on acidification of precipitations.

Non-GHG related environmental impacts for tight oil are mainly associated with water consumption, water contamination, and waste disposal. Even in this case, while the need for more systematised datasets is identified, the following results are highlighted:

- Scanlon et al. (2014) found lower water consumption per unit of oil produced from unconventional sources, rather than in conventional production sites.
- On the other hand, Di Giulio and Jackson (2016) showed that fracking operations caused the contamination of underground drinking water by leakage of hydrocarbons and fracking fluids.
- Especially, the mix of chemicals in fracking fluids is the source of debate on potential toxicity-related impacts from tight oil production. For instance, Elliott et al. (2017) found that toxicity information was lacking for most of the chemicals used in fracking fluid, but where information was available, chemicals were associated to reproductive toxicity, developmental toxicity, or both.
- Finally, wastewater from fracking operations, contaminated by such chemicals, can also pose environmental risks if not stored, handled and disposed properly (Goñi, 2019).

C.4. Conclusions

The dataset used in the case studies to model crude oil supply to the EU market is representative of the crude oil mix used in the EU-27 in 2014. The current crude oil mix (estimated based on 2018 data from official EU statistics) did not present relevant changes compared to 2014, with the largest (absolute) variations in imports and deliveries in the range of 4%. However, the 2018 mix included a higher share of unconventional crude oils imports, especially light crude oil from hydraulic fracking operations in US, and a minor but growing portion of heavy crude from Canadian oil sands.

The difference in terms of potential Climate Change impact (GHG emissions) from using the estimated 2018 crude oil mix was quantified, and resulted in an increase of ca. 3.9%

¹⁷² A Scopus search query for 'oil sands environmental impact' returned 903 documents, and a Google Scholar search of the same query returned 18,000 results in papers published after 2015.

compared to 2014. Assuming that the impact of the Feedstock Supply stage of the fossil-based plastic products investigated in this report increases to the same extent (which is a conservative assumption as Feedstock Supply includes also the contribution of other activities), the total Climate Change impact of such products (i.e. PET and HDPE bottles, LDPE mulching film, and EPS insulation boards) would increase by between 0.2% and 1.3%, which is negligible.

The current lack of independent and/or harmonised Life Cycle Inventory (LCI) datasets for individual oil sources (including unconventional sources such as tight oil and oil sands), and the aggregated nature of the datasets available to model crude oil supply, make it quite difficult to account for the potential environmental impacts of such unconventional sources (and compare with conventional ones). However, for the purpose of this analysis, only differential impacts between the conventional sources (that may be displaced) and the unconventional sources (that may increase their share within the EU oil mix) are relevant. In this respect, light tight oil from US did not seem to involve differentially major higher impacts compared to conventional oil sources, at least in terms of Climate Change. On the other hand, heavy crude from Canadian oil sands may increase the overall impact of the EU crude oil mix, although the share of Canadian heavy crude in the total EU mix estimated for 2018 was still marginally small.

Overall, the dataset applied in the case studies to model crude oil supply to the EU market was considered to be sufficiently representative of the potential environmental impacts of the current crude oil mix (based on 2018 data), and to properly suit the illustrative purpose of the studies themselves. However, life cycle inventory data providers are recommended to take into account the contribution of any (unconventional) oil source that may become relevant in the future oil mix, when releasing updated version of (EF-compliant) datasets for crude oil supply.

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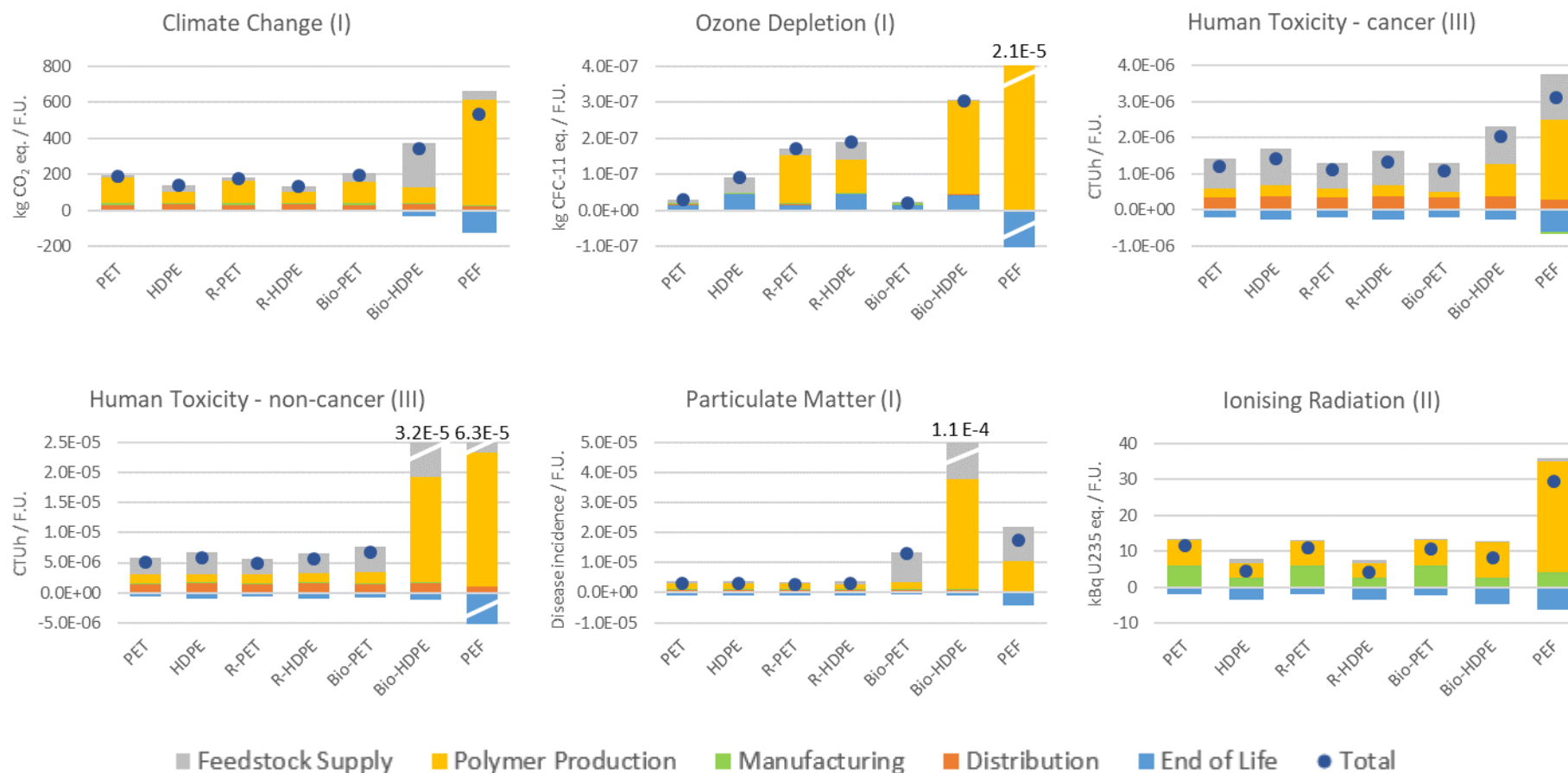
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Annex D – Contribution of Life Cycle Stages to characterised LCIA results: graphic representation

This annex provides graphical representations of the contribution of the different Life Cycle Stages to the total characterised Life Cycle Impact Assessment (LCIA) results of the alternative product scenarios investigated in each case study. According to the common general system boundary applied in the studies, the considered life cycle stages include *Feedstock Supply*, *Polymer Production*, *Manufacturing*, *Distribution* and *End of Life*. The results are separately presented for each case study in the single sub-sections of the annex (D.1–D.3).

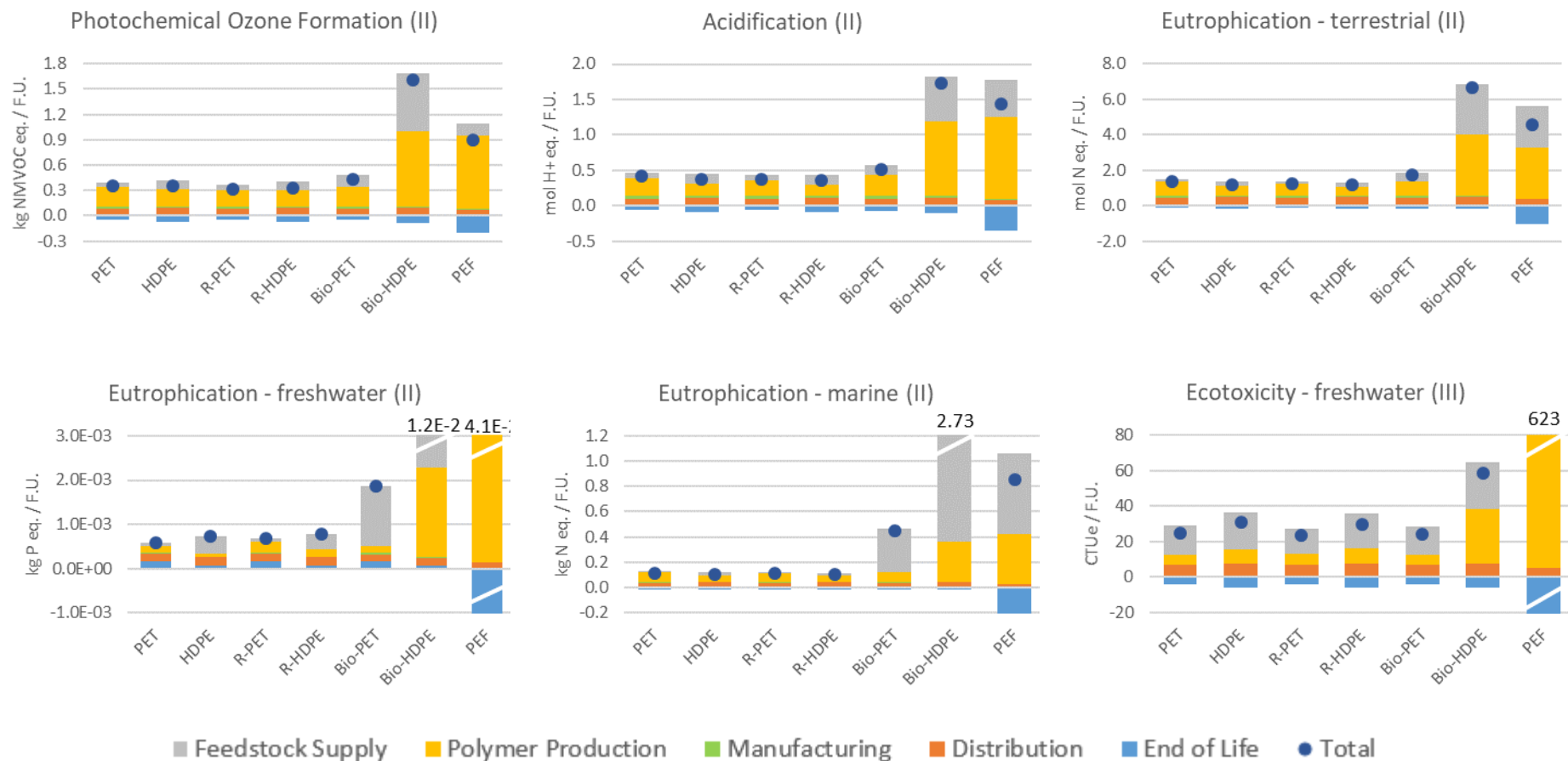
Note that the main aim of the reported illustrations is to facilitate understanding of the contribution of the different life cycle stages compared to the numerical results reported in the main text of the report (Section #.6 of each case study). Therefore, while results related to different scenarios are presented on a same figure (for graphical and space reasons), they are not intended to compare the investigated scenarios and should not be used for this purpose by the reader. Moreover, the limitations and critical assumptions reported in the main text of each case study (see Section #.4 of the relevant case study chapter) shall be properly taken into account in the interpretation of the presented results.

D.1. Beverage Bottles



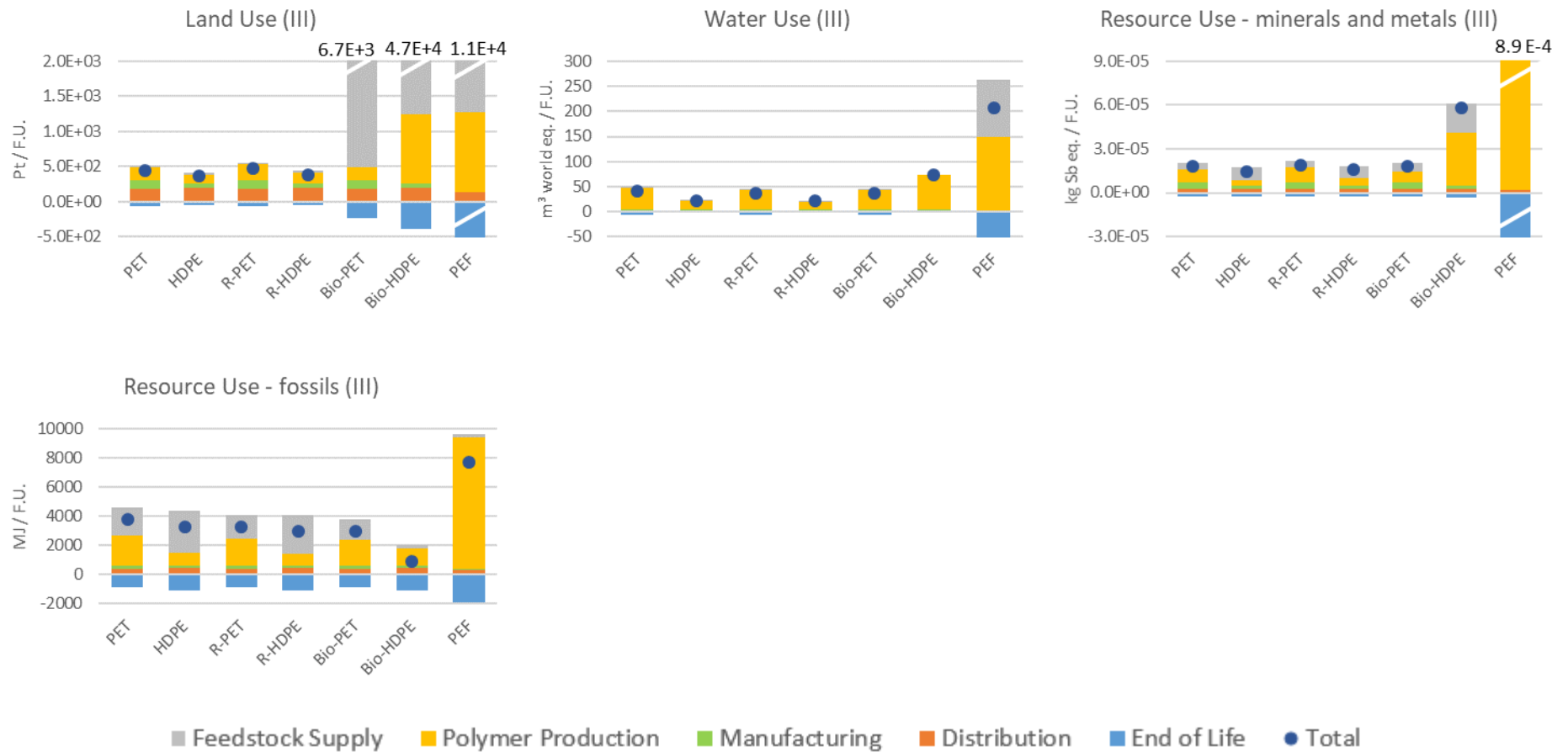
Notes: (a) Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category). (b) In some impact categories a part of the results is out of scale and is curtailed.

Figure D.1.1. Contribution of single life cycle stages to the characterised impact assessment results of beverage bottles LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity – cancer, Human Toxicity – non-cancer, Particulate Matter, and Ionising Radiation. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 3.4.



Notes: (a) Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level -I, II, III- is reported in brackets after the name of each category). (b) In some impact categories a part of the results is out of scale and is curtailed.

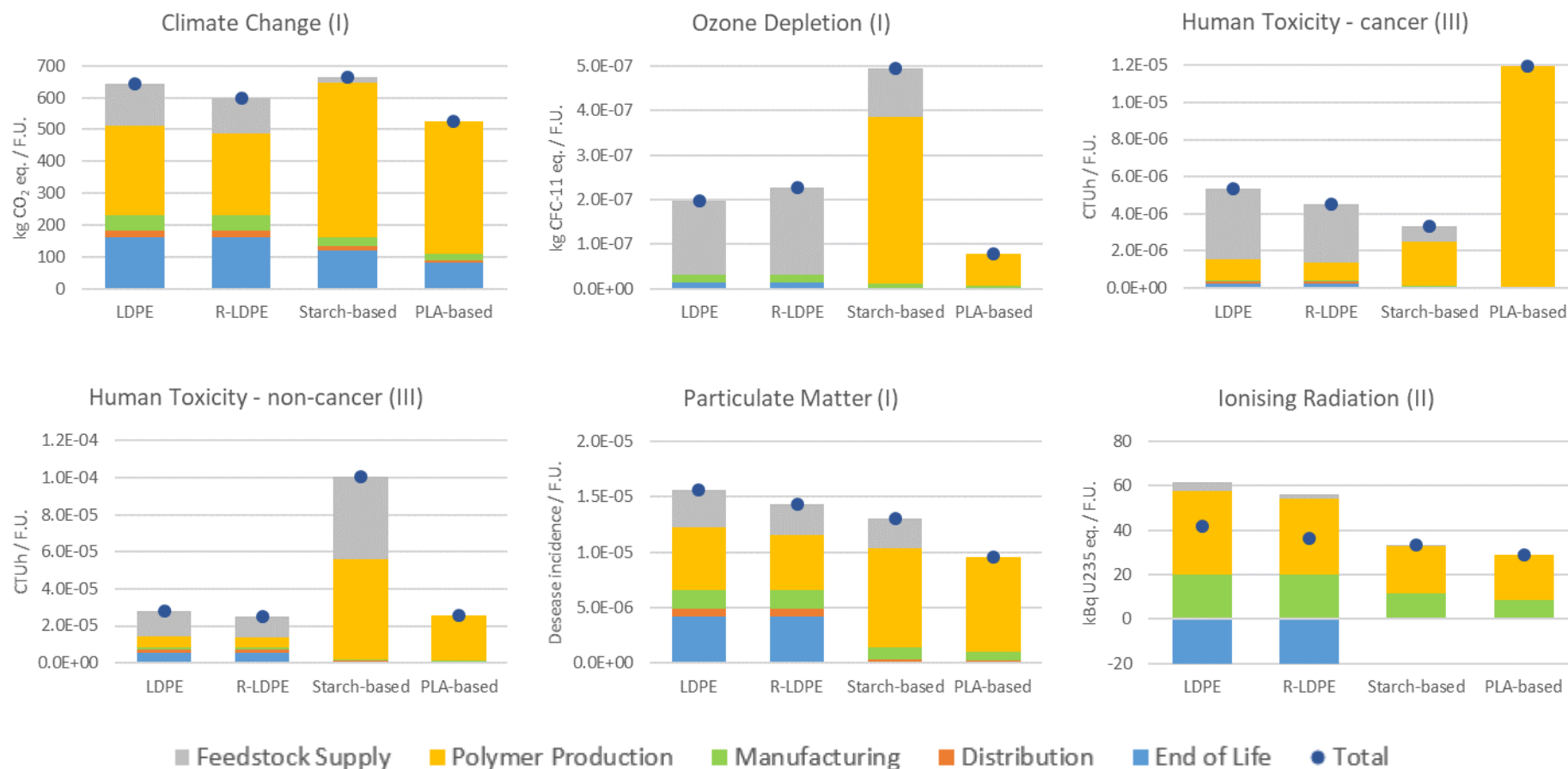
Figure D.1.2. Contribution of single life cycle stages to the characterised impact assessment results of beverage bottles LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, and Ecotoxicity – freshwater. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 3.4.



Notes: (a) Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category). (b) In some impact categories a part of the results is out of scale and is curtailed.

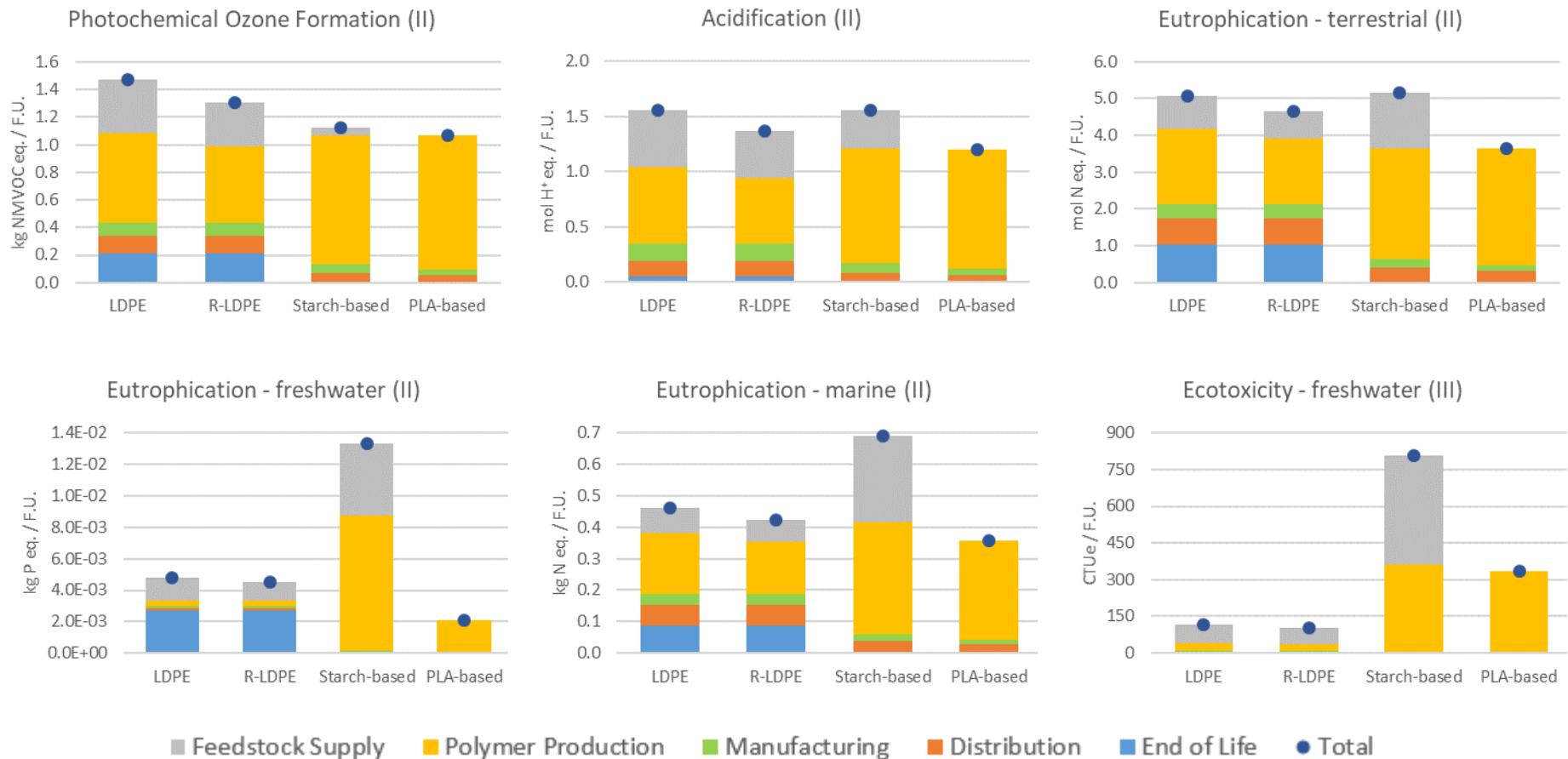
Figure D.1.3. Contribution of single life cycle stages to the characterised impact assessment results of beverage bottles LCA scenarios for the categories of Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 3.4.

D.2. Agricultural Mulching Film



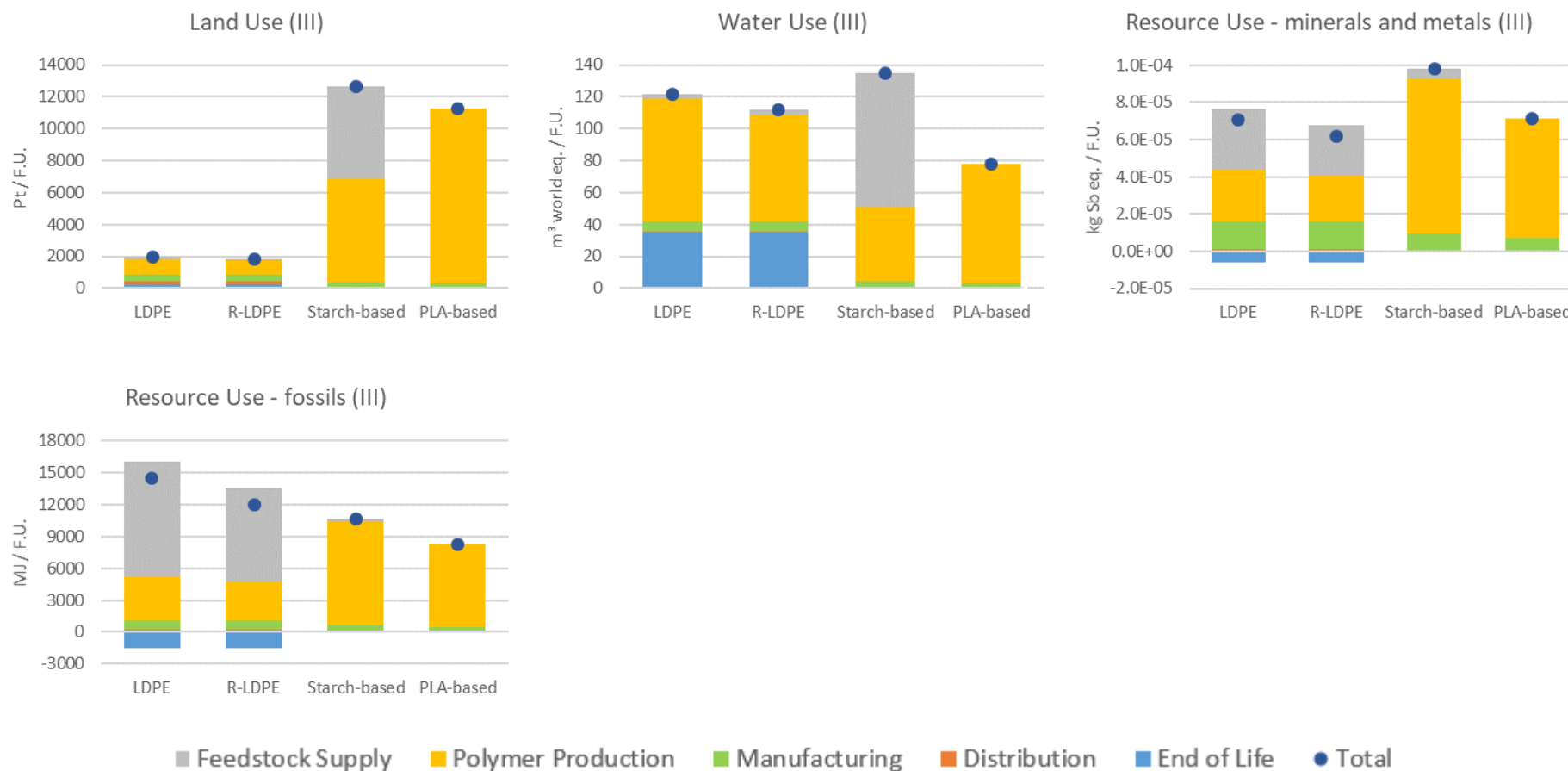
Note: Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category).

Figure D.2.1. Contribution of single life cycle stages to the characterised impact assessment results of agricultural mulching film LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity – cancer, Human Toxicity – non-cancer, Particulate Matter, and Ionising Radiation. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 4.4.



Note: Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level -I, II, III- is reported in brackets after the name of each category).

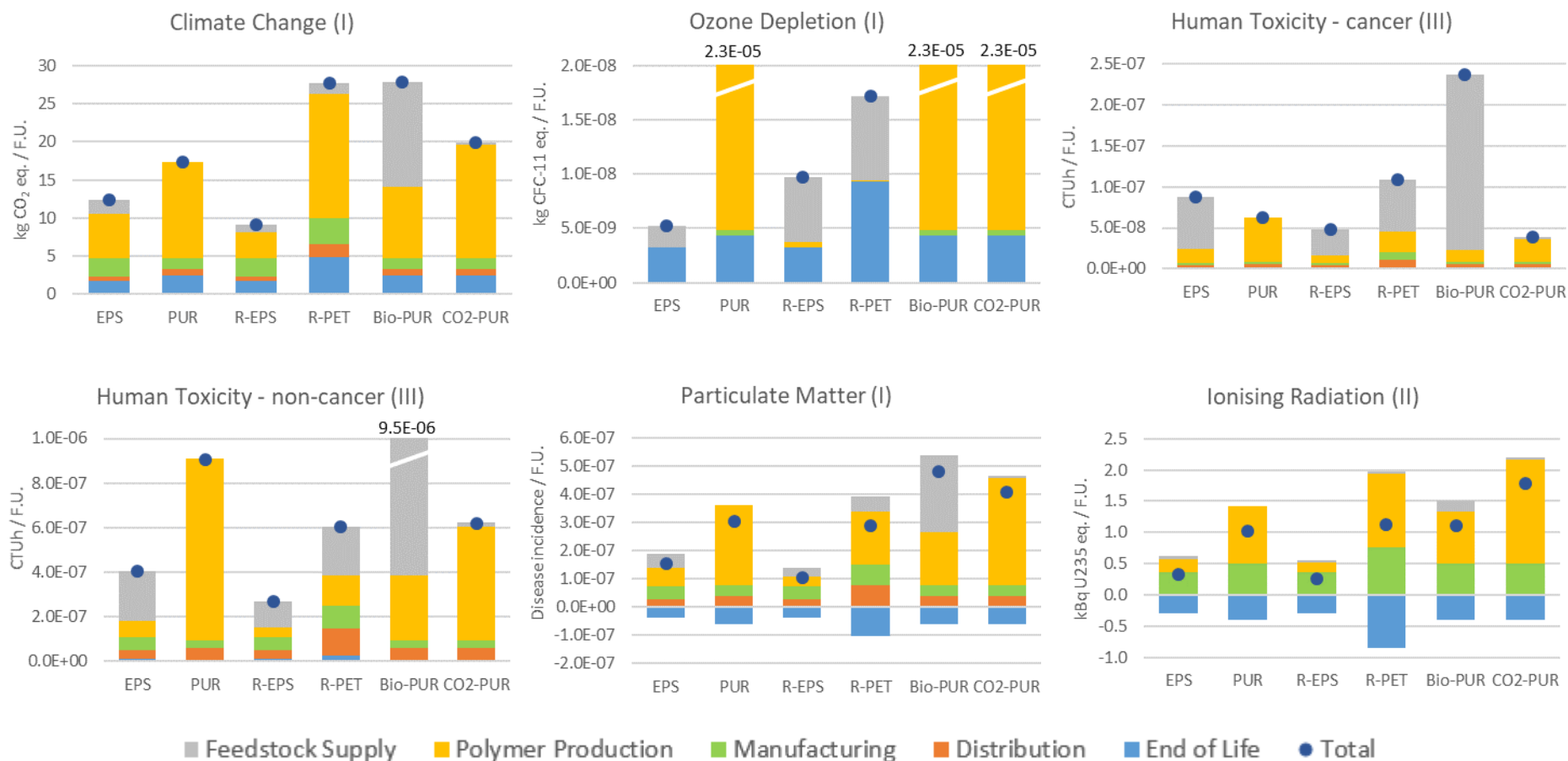
Figure D.2.2. Contribution of single life cycle stages to the characterised impact assessment results of agricultural mulching film LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, and Ecotoxicity – freshwater. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 4.4.



Note: Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category).

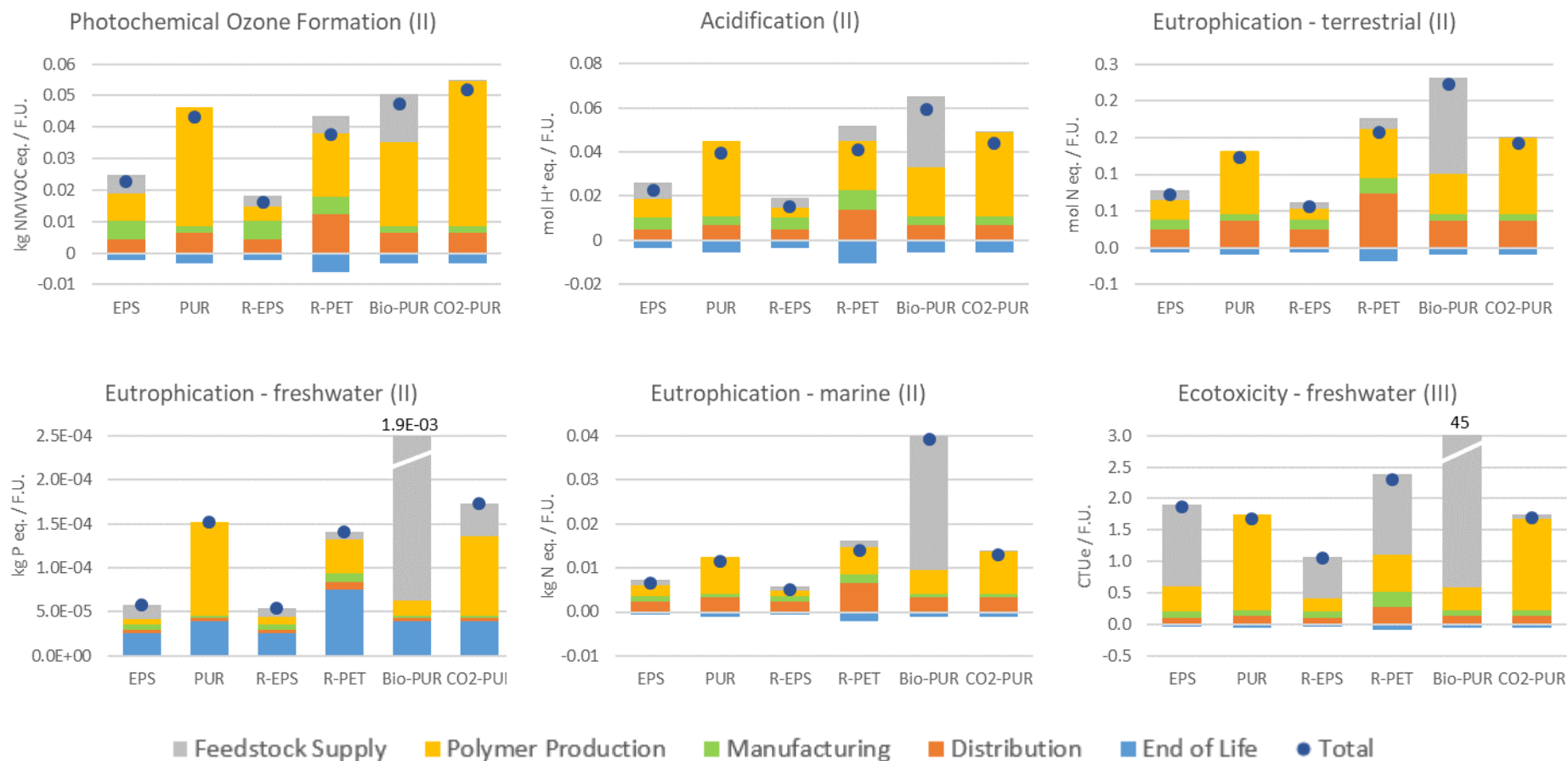
Figure D.2.3. Contribution of single life cycle stages to the characterised impact assessment results of agricultural mulching film LCA scenarios for the categories of Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 4.4.

D.3. Insulation Boards for Buildings



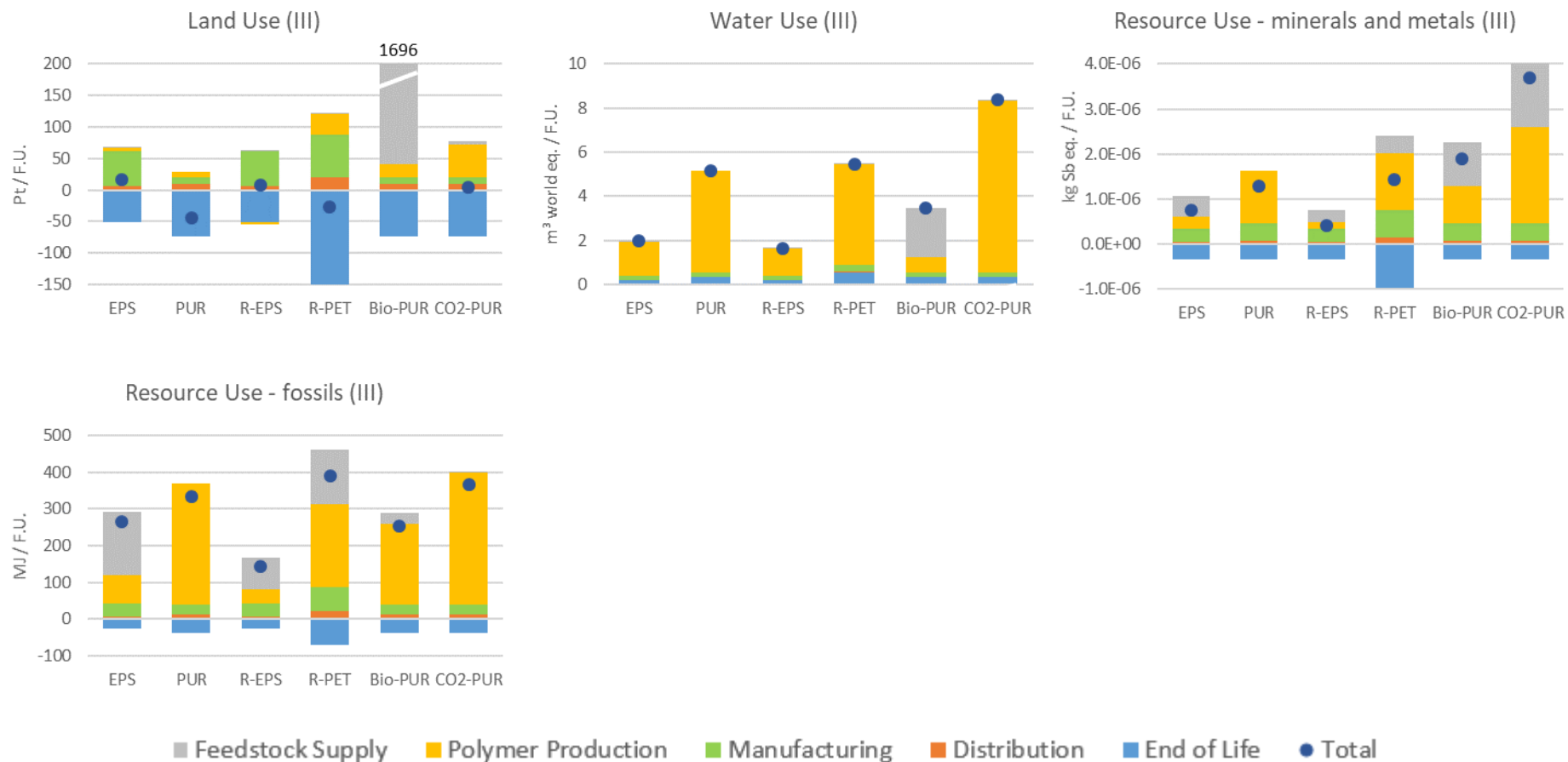
Notes: (a) Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category). (b) In some impact categories a part of the results is out of scale and is curtailed.

Figure D.3.1. Contribution of single life cycle stages to the characterised impact assessment results of building insulation boards LCA scenarios for the categories of Climate Change, Ozone Depletion, Human Toxicity – cancer, Human Toxicity – non-cancer, Particulate Matter, and Ionising Radiation. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 5.4.



Notes: (a) Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category). (b) In some impact categories a part of the results is out of scale and is curtailed.

Figure D.3.2. Contribution of single life cycle stages to the characterised impact assessment results of building insulation boards LCA scenarios for the categories of Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, and Ecotoxicity – freshwater. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 5.4.



Notes: (a) Impact categories relying on a less robust life cycle impact assessment model require a more careful interpretation, as results are affected by larger uncertainty (the robustness level –I, II, III– is reported in brackets after the name of each category). (b) In some impact categories a part of the results is out of scale and is curtailed.

Figure D.3.3. Contribution of single life cycle stages to the characterised impact assessment results of building insulation boards LCA scenarios for the categories of Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils. Results are not intended to compare the different scenarios and are affected by the limitations and critical assumptions discussed in Section 5.4.

Annex E – Alternative End of Life scenarios (sensitivity analysis): graphic results

This annex reports graphic representations of the results of the sensitivity analysis on the applied product End of Life scenario, conducted in each case study. As described in the main text, this analysis considered the application of the single End of Life options relevant to the product in scope as independent (100%) End of Life scenarios. The aim of the graphs is to provide a more synthetic and understandable overview of the numerical results presented in the main text, and to facilitate appreciation of impact variations occurring within individual product scenarios when changing the applied End of Life option/scenario. Therefore, while results related to different product scenarios are presented on a same figure (for graphical and space reasons), they are not intended to compare the investigated product scenarios and should not be used for this purpose by the reader. Each sub-section of this annex separately presents the results related to a specific case study.

E.1. Beverage Bottles

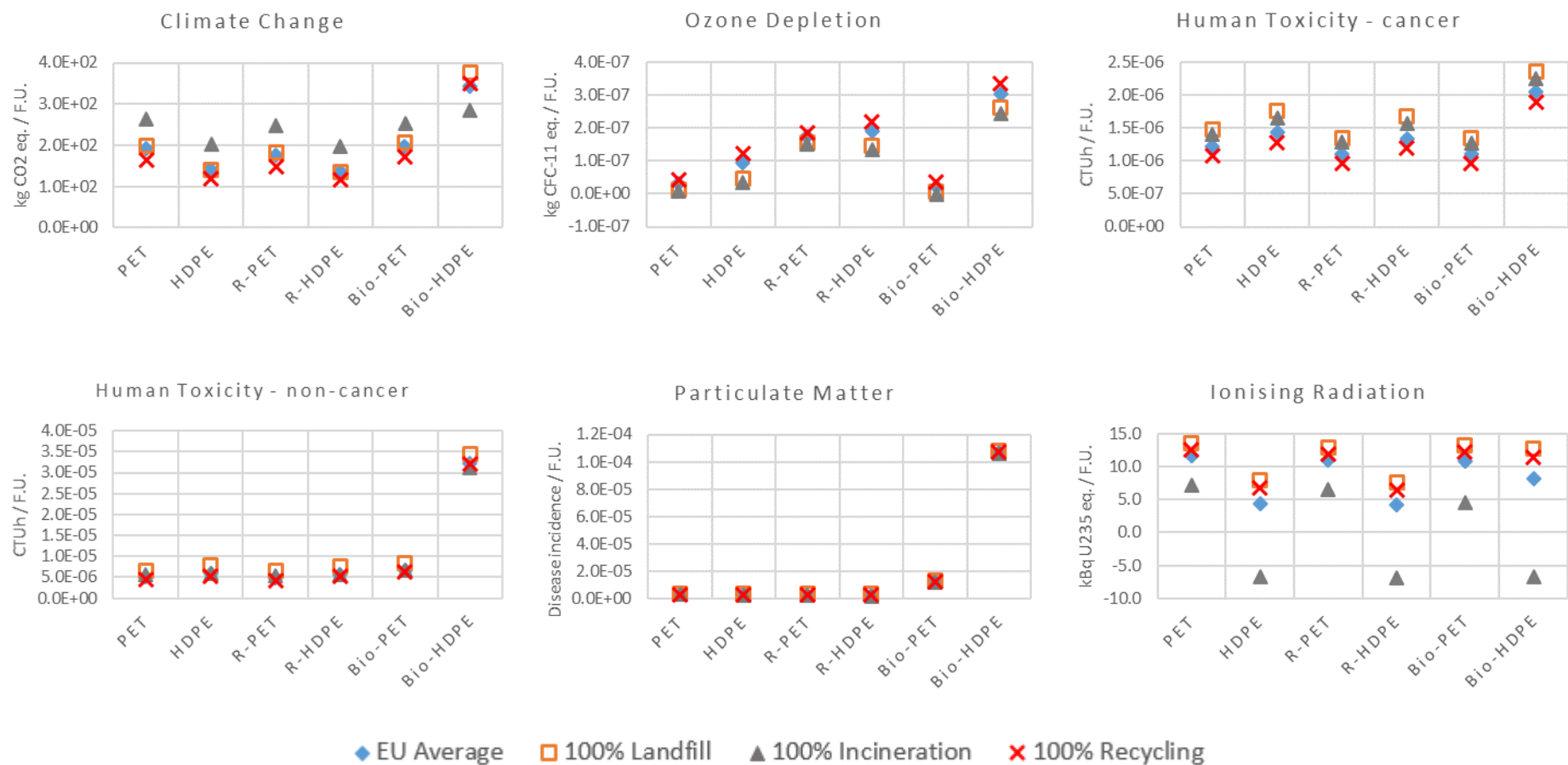


Figure E.1.1. Characterised potential impacts of beverage bottles for different End of Life scenarios, for the categories of Climate Change, Ozone Depletion, Human Toxicity – cancer, Human Toxicity – non-cancer, Particulate Matter, and Ionising Radiation. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different beverage bottles scenarios and are affected by the limitations and critical assumptions discussed in Section 3.4.

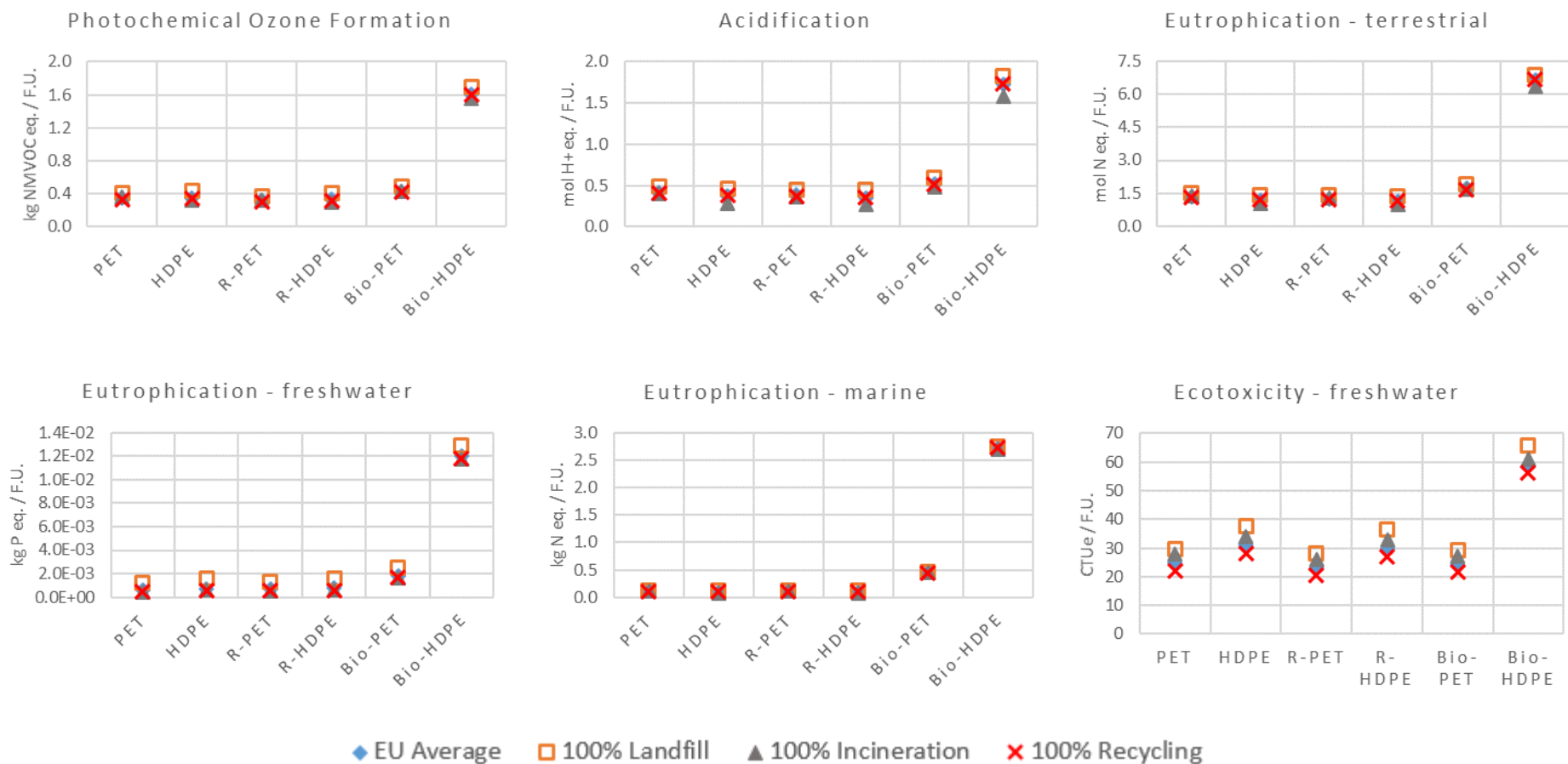


Figure E.1.2. Characterised potential impacts of beverage bottles for different End of Life scenarios, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, and Ecotoxicity – freshwater. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different beverage bottles scenarios and are affected by the limitations and critical assumptions discussed in Section 3.4.

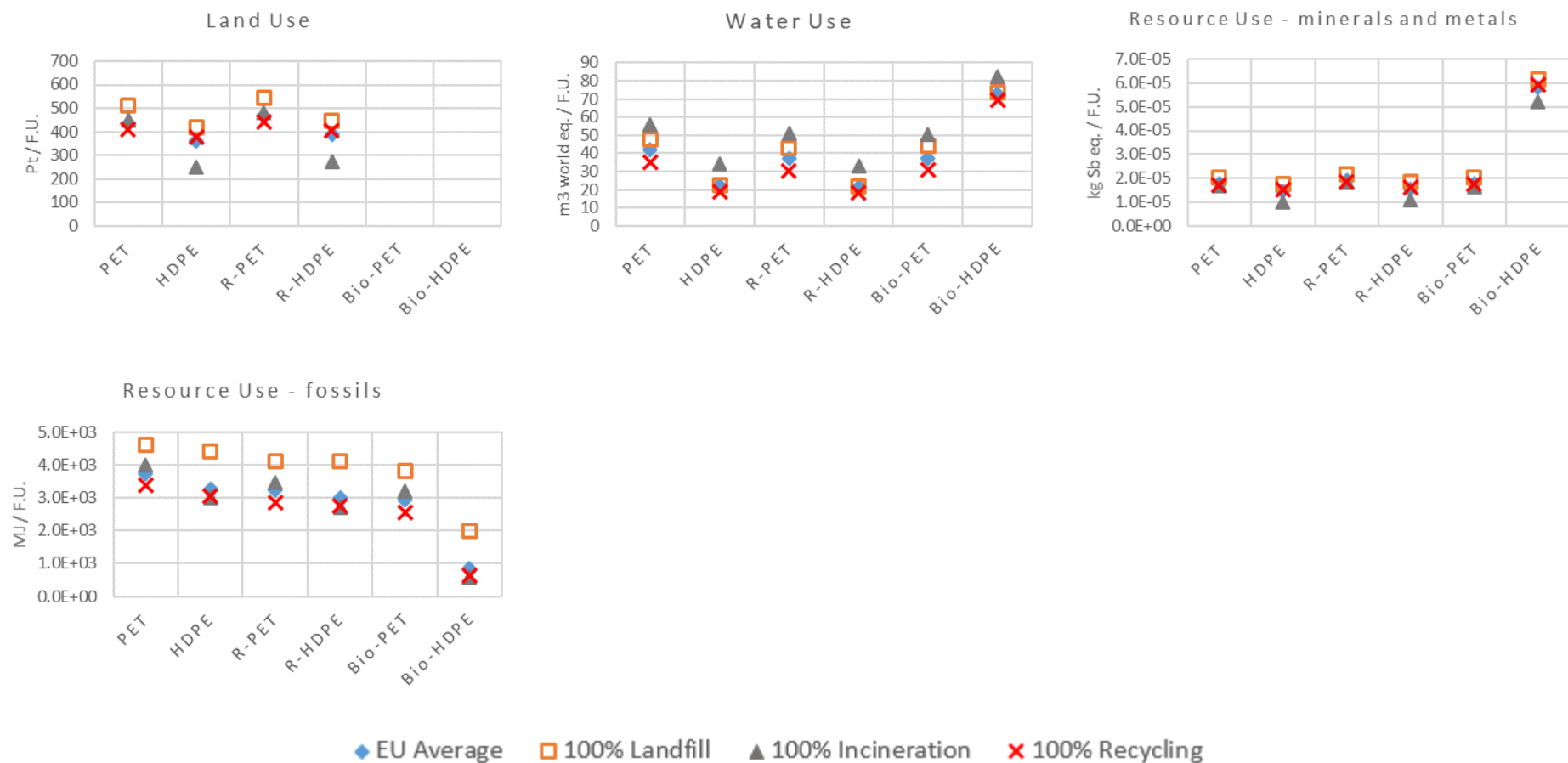


Figure E.1.3. Characterised potential impacts of beverage bottles for different End of Life scenarios, for the categories of Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different beverage bottles scenarios and are affected by the limitations and critical assumptions discussed in Section 3.4.

E.2. Agricultural Mulching Film

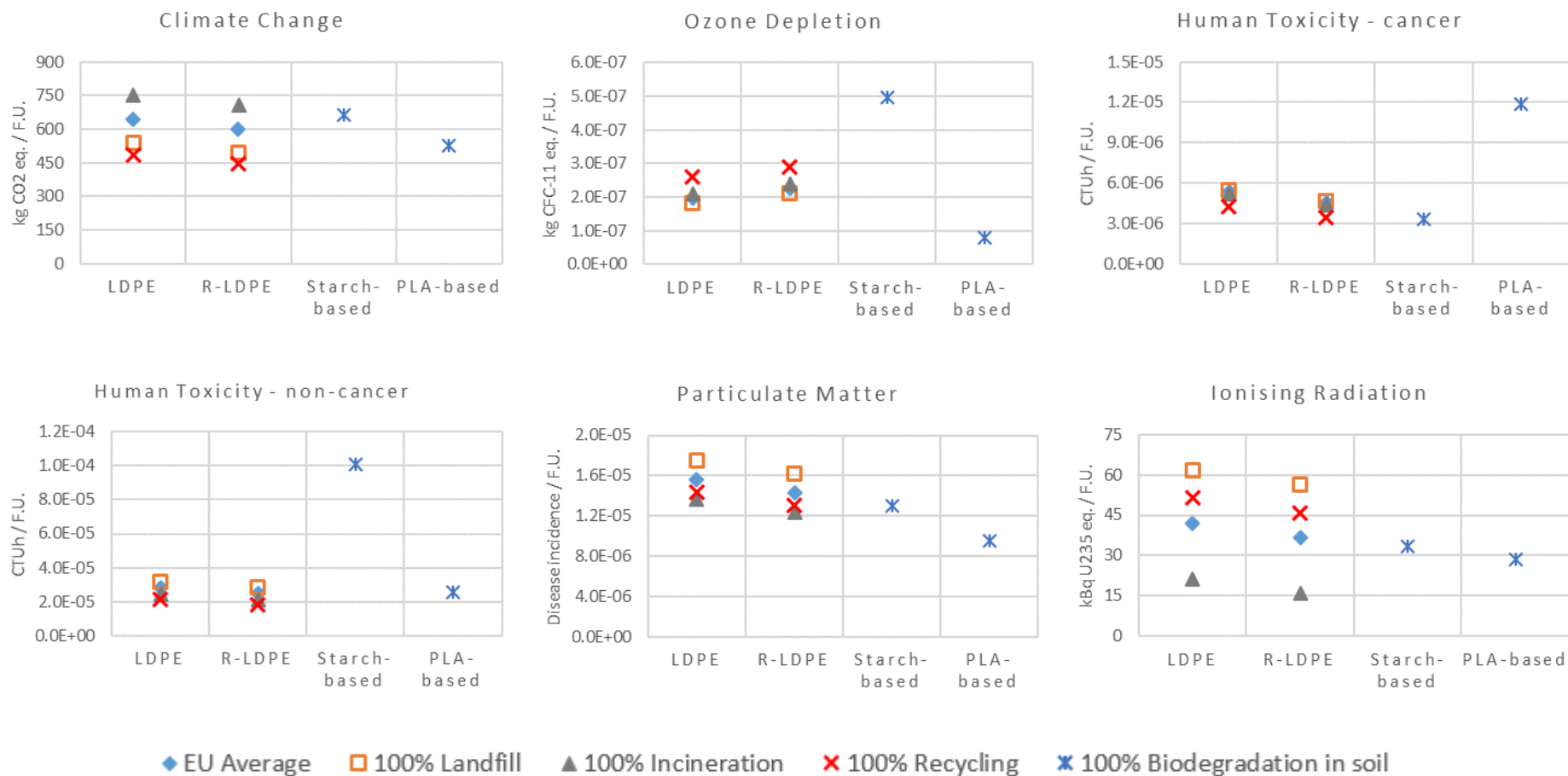


Figure E.2.1. Characterised potential impacts of agricultural mulching film for different End of Life scenarios (where applicable), for the categories of Climate Change, Ozone Depletion, Human Toxicity – cancer, Human Toxicity – non-cancer, Particulate Matter, and Ionising Radiation. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different mulching film scenarios and are affected by the limitations and critical assumptions discussed in Section 4.4.

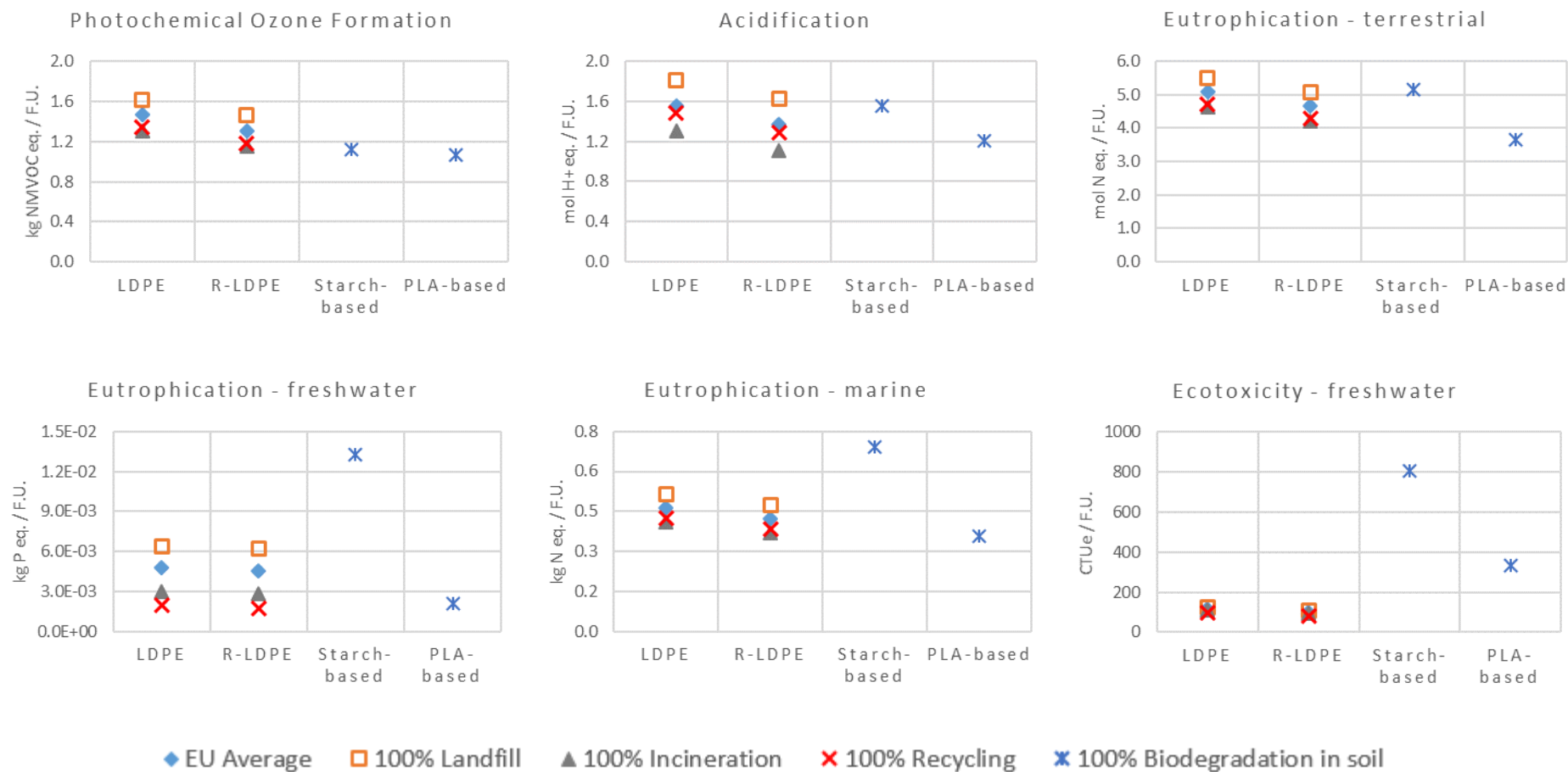


Figure E.2.2. Characterised potential impacts of agricultural mulching film for different End of Life scenarios (where applicable), for the categories of Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, and Ecotoxicity – freshwater. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different mulching film scenarios and are affected by the limitations and critical assumptions discussed in Section 4.4.

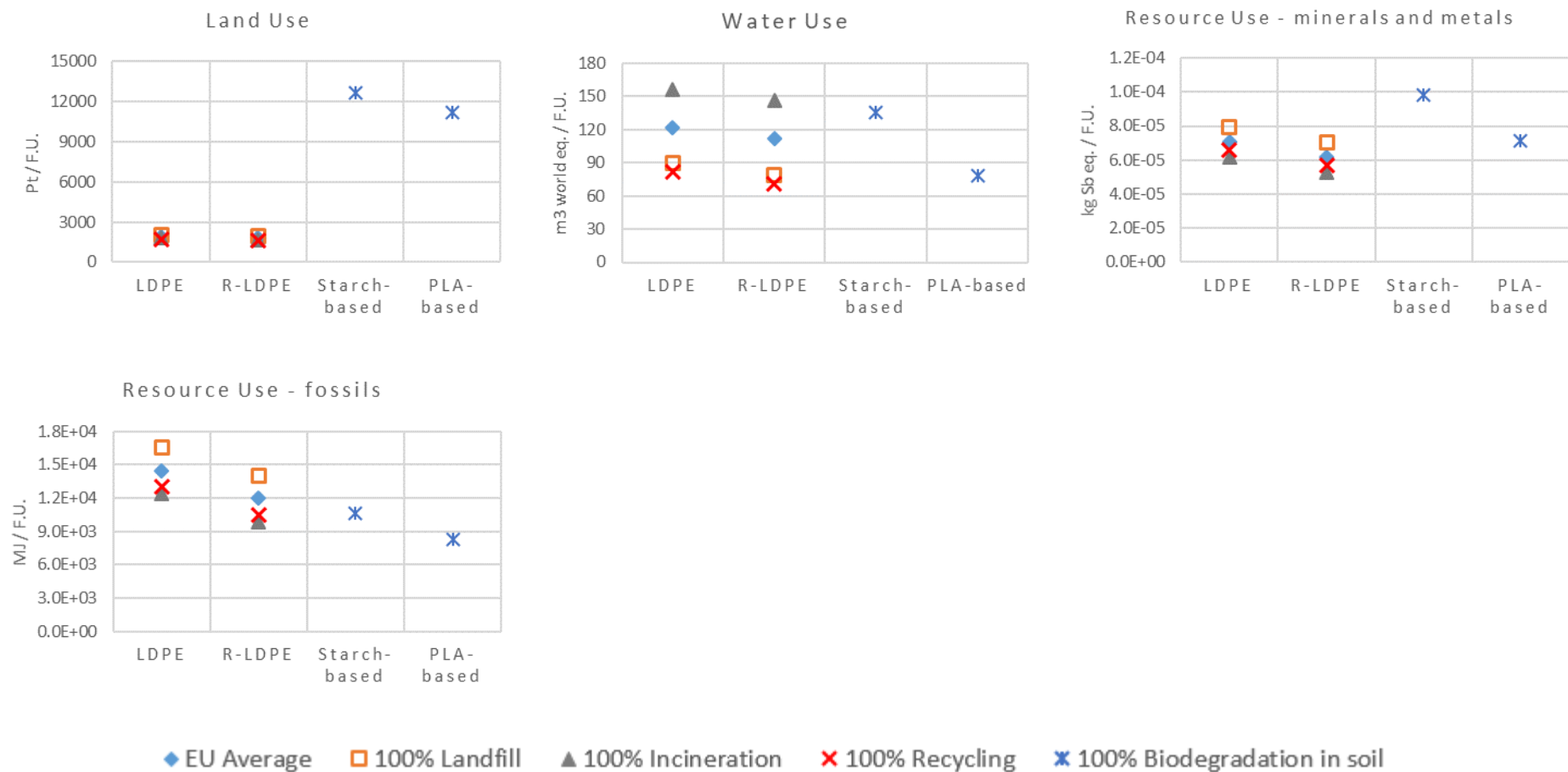


Figure E.2.3. Characterised potential impacts of agricultural mulching film for different End of Life scenarios (where applicable), for the categories of Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different mulching film scenarios and are affected by the limitations and critical assumptions discussed in Section 4.4.

E.3. Insulation Boards for Buildings

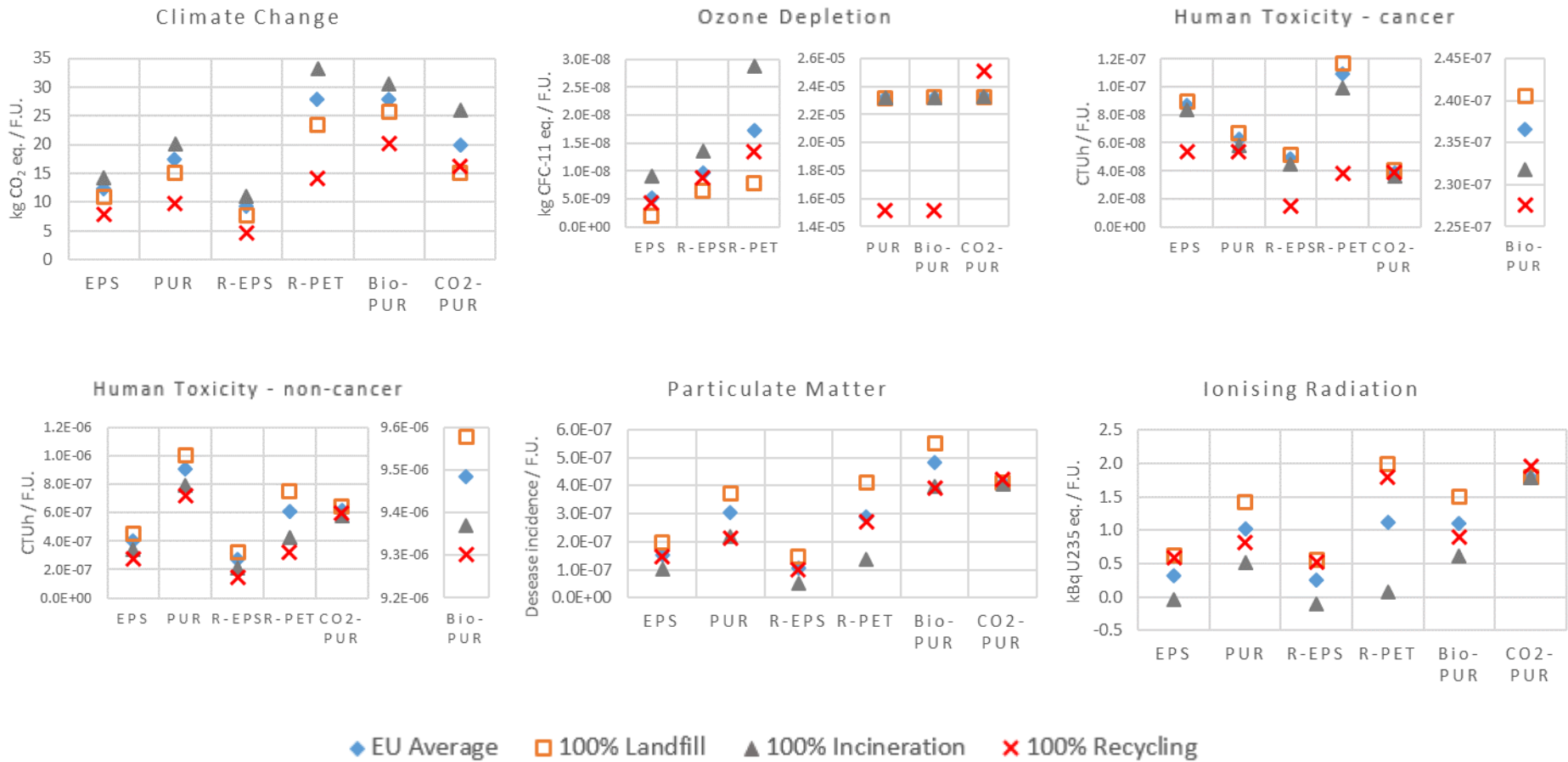


Figure E.3.1. Characterised potential impacts of building insulation boards for different End of Life scenarios, for the categories of Climate Change, Ozone Depletion, Human Toxicity – cancer, Human Toxicity – non-cancer, Particulate Matter, and Ionising Radiation. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different insulation boards scenarios and are affected by the limitations and critical assumptions discussed in Section 5.4.

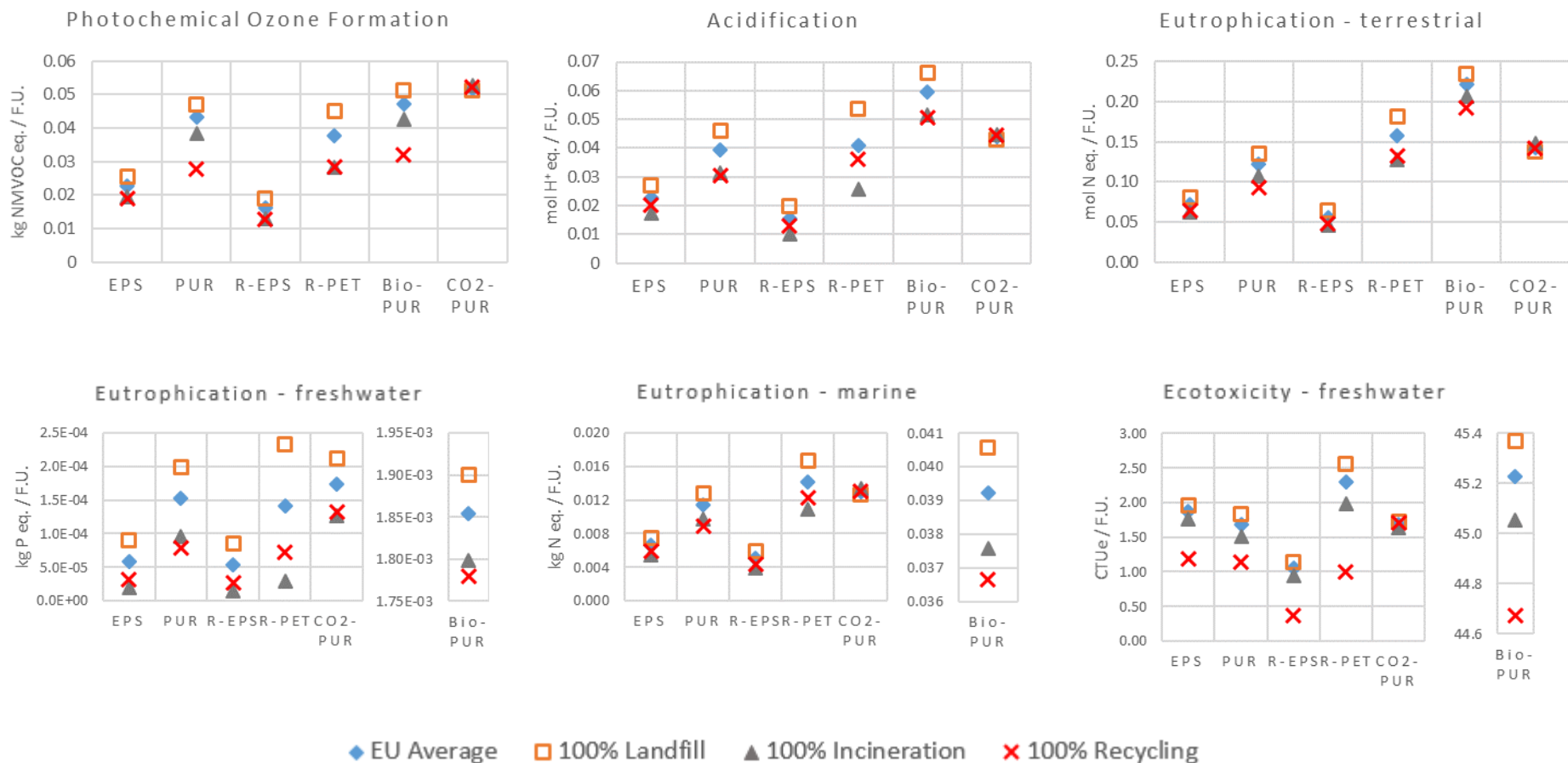


Figure E.3.2. Characterised potential impacts of building insulation boards for different End of Life scenarios, for the categories of Photochemical Ozone Formation, Acidification, Eutrophication – terrestrial, Eutrophication – freshwater, Eutrophication – marine, and Ecotoxicity – freshwater. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different insulation boards scenarios and are affected by the limitations and critical assumptions discussed in Section 5.4.

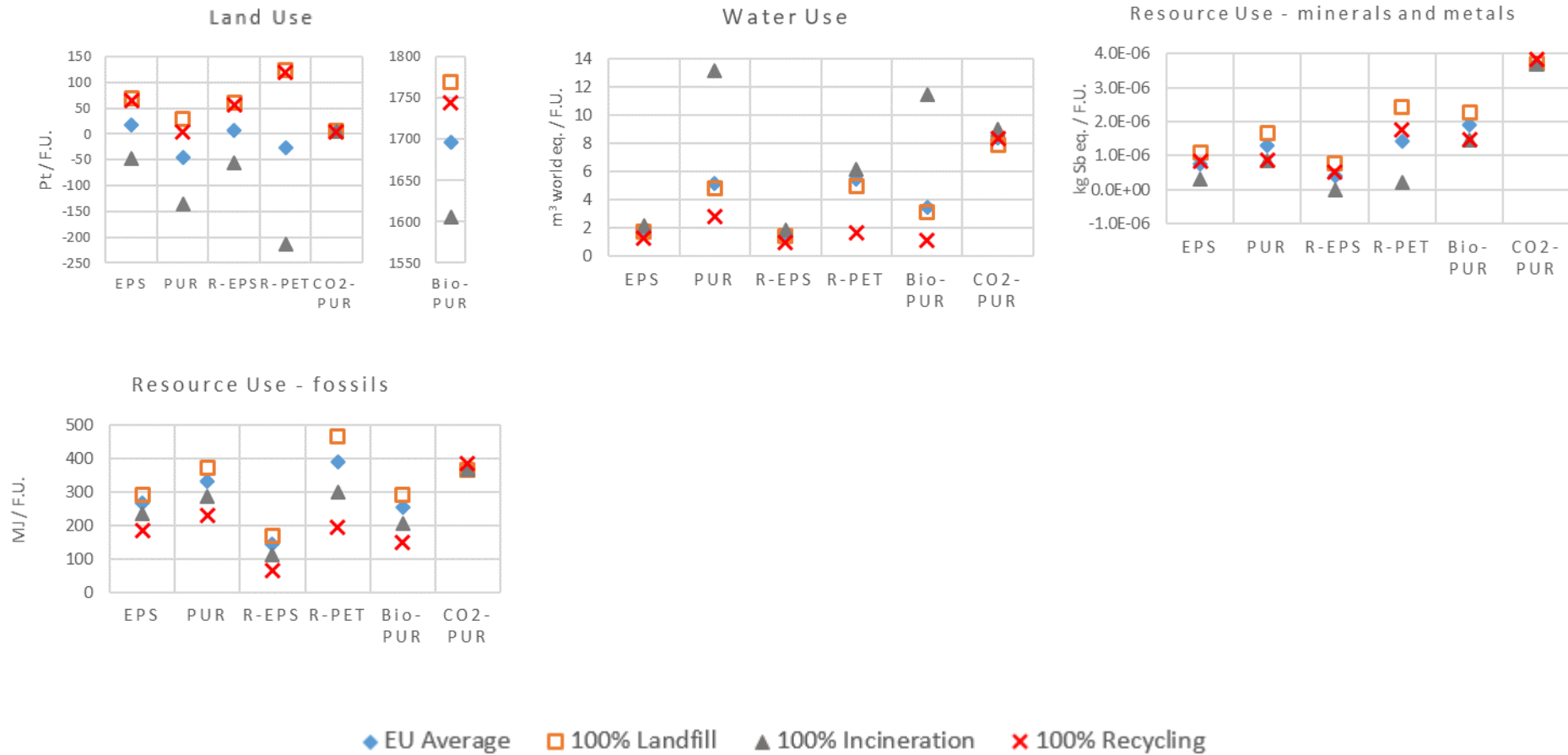


Figure E.3.3. Characterised potential impacts of building insulation boards for different End of Life scenarios, for the categories of Land Use, Water Use, Resource Use – minerals and metals, and Resource Use – fossils. The EU-average End of Life scenario refers to the base case, others to the sensitivity analysis. Results are not intended to compare the different insulation boards scenarios and are affected by the limitations and critical assumptions discussed in Section 5.4.

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